FISH SPECIES OF SPECIAL CONCERN IN CALIFORNIA

Third Edition

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Wild Klamath Mountain Province steelhead. Photo courtesy of Jeff Weaver.

The Resources Agency
Department of Fish and Wildlife
Sacramento, California

July, 2015

Suggested citation: Moyle, P.B., R. M. Quiñones, J. V. Katz and J. Weaver. 2015. Fish Species of Special Concern in California. Sacramento: California Department of Fish and Wildlife. www.wildlife.ca.gov

PREFACE

This is the third edition of the report on the status of California's Fish Species of Special Concern. The fishes addressed in this report all live and spawn in California's freshwater environments and face varying levels of threat. They are all species that could potentially become extinct by the end of this century, tracking trajectories set by seven species that are already extinct and 31 species that are formally listed as threatened or endangered within the state. The fact that 62 species are covered in this report, while 38 others are listed or extinct, means that 100 native fishes in California are in decline, headed toward extinction, or already extinct. *This represents 81% of California's highly distinctive inland fish fauna.* These species can be regarded as good indicators of the quality and quantity of freshwater habitats around the state which, as indicated by the high percentage of at-risk fishes, are apparently deteriorating.

This report differs from the previous two editions in that the reader does not have to take our word for the status of each of the fish species covered. We use a standardized system for evaluating status, so our assessments can be easily compared among species and can be repeated by others. Our goal is to create a baseline against which future assessments can be compared. Anyone reading this report, with some diligence, should be able to go through the scoring process for a given species and come up with a similar status rating. If the rating differs from ours, the reasons will be apparent from the scores of individual metrics. We assume that the accuracy of scores will improve with additional evaluations especially if you, the reader, have new and better information about a species. More accurate scores are particularly likely for species where we indicate that there is a relatively low amount of reliable information on their biology. Ideally, each account should be updated as new studies are completed.

We intend that these accounts will be useful first references for those engaged in management of California's fishes or will provide basic background for anyone interested in native fishes. We hope this report will stimulate better and more extensive conservation efforts for each of these declining species. All species treated here need our protection if they are going to survive through the coming decades.

For those interested in easily accessible accounts of species not covered in this report, as well as photographs of the species, we recommend the UC Davis California Fish Website: http://calfish.ucdavis.edu/.

ACKNOWLEDGEMENTS

Many people have contributed information to us on native fishes and their status over the years, for this and previous editions of the report. We apologize to those we inadvertently missed acknowledging for contributions to this edition of the report. We would like to thank the following persons for providing valuable input and/or for reviewing species accounts in this edition: Gar Abbas, Andres Aguilar, Don Alley, Randy Baxter, Brian Beal, Helen Benson, Roger Bloom, Jennifer Bull, Steve Cannata, Dave Cook, Alex Corum, Jim Deacon, Michael Dege, Paul Divine, Andrea Drauch, Bjorn Erickson, Ben Ewing, Michael Fawcett, Mandy Finger, Max Fish, Terry Foreman, Michelle Gilroy, Marty Gingras, Jon Grunbaum, Michael R. Harris, Josh Israel, Michael Kellett, Stafford Lehr, Dave Lentz, Rob Leidy, Alice Low, Rick Macedo, James

Malcolm, Dough Markle, Tony Metcalf, Curtis Milliron, Dan Mitchell, Jonathan Nelson, John O'Brien, Steve Parmenter, Joe Pisciotto, Stewart Reid, Jay Rowan, Don Sada, Kevin Shaffer, Jennifer Simon, Wade Sinnen, Jerry Smith, Stephen Swales, Camm Swift, Tom Taylor, Kevin Thomas, Andrew Thompson, Rob Titus, Cindy Williams, Jack Williams, Andy Van Scoyk, Jim Waldvogel, Desma Williams, and Glenn Yoshioka. The involvement of R. Quiñones in the final stages of this report was supported by the Stephen J Bechtel, Jr. Foundation and the California Landscape Conservation Cooperative. Jacob Katz's involvement was supported by California Trout and the California Water Foundation.

INTRODUCTION

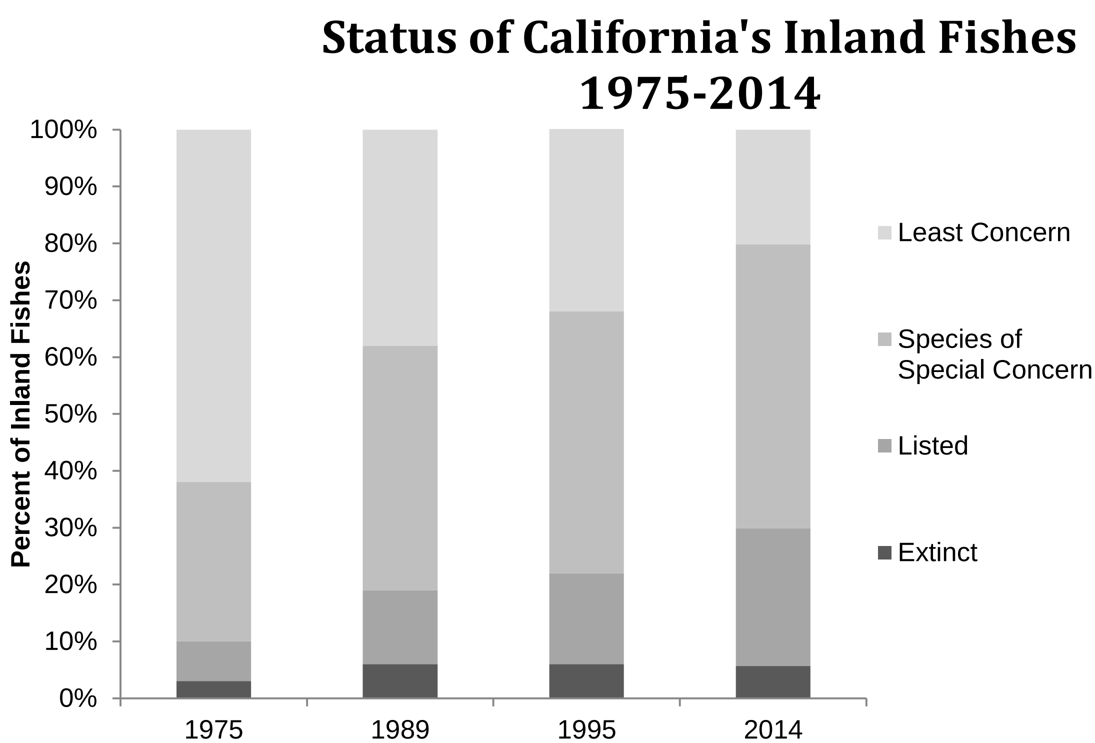
California has a rich fauna of native inland fishes. The state's large size (411,000 km^2), length (1,400 km and 10 degrees latitude) and complex topography result in diverse habitats from temperate rain forests to deserts, as well as 50 isolated, large watersheds in which fish evolution has occurred independently (Moyle 2002, Moyle and Marchetti 2006, Figure 1). For most of the state, the climate is Mediterranean; most precipitation falls in winter and spring, followed by long dry summers. This results in rivers that have high annual and seasonal variability in flows (Mount 1995) and native fishes adapted to hydrologic extremes. Of 124 native inland fishes (defined as those breeding in fresh water) evaluated for this report, 64% are endemic to the state, with an additional 19% also found in Nevada or Oregon. Thus, California has the high overlap between political and zoogeographic boundaries needed for this assessment to be considered bioregional (Moyle 2002).

The long coastline of California has produced a fish fauna containing an unusual proportion (23%) of anadromous (sea-run) taxa, while its dry interior watersheds have produced fishes that thrive in isolated environments such as desert springs, intermittent streams, and alkaline lakes. A majority of California's fishes live in rivers of the Central Valley and North Coast, areas with the most water and most diverse aquatic habitats. The Central Valley, in particular, has been a center of speciation, with 35 native taxa, many of them (16) endemic (found nowhere else) to the watershed, with some also giving rise to species now confined to adjacent watersheds. Recent genetic and taxonomic studies have increased appreciation of the distinctiveness of the California fish fauna, such that the total number of distinct taxa has risen from 113 recognized by Moyle and Williams (1990) to 124 analyzed for this report (Box 1, Table 8).

Figure 1. Map of California showing major watersheds. Each number represents a major zoogeographic region; each number + lowercase letter represents a distinct watershed that is physically separated from the other watersheds or is characterized by a distinct fish fauna, or both. Modified from Moyle 2002.

Unfortunately for the fishes, most of the rivers of California have been dammed and diverted to move water from places of abundance to places of scarcity, where most Californians live (Hundley 2001). Not surprisingly, native fishes have been in steady decline since the mid-19th century, although the first statewide evaluation was not done until 1975 (Moyle 1976) and an analysis of the formal conservation status was not published until 1989 (Figure 2). In 1975, 6 species were considered extinct but most species (64%) were considered stable. There has been only one recognized extinction in the intervening years but the numbers of listed and imperiled species have steadily increased so that, in 1989, 15 species (13%) were formally listed as threatened or endangered under state and federal endangered species acts and 50 (44%) were regarded as imperiled (Moyle et al. 1989). By 1995, the numbers were 18 (16%) listed and 53 (46%) imperiled (Moyle et al. 1995). Of the 124 species considered for this report, 7 are extinct, 31 (25%) are officially listed, and 62 (50%) are considered of critical, high or moderate concern, which means that at least 81% of California's native fishes are imperiled or extinct (Fig. 2). The purpose of this report is to synthesize the information available on these imperiled species, referred to herein as *Fish Species of Special Concern* (FSSC), to provide a basis for their conservation, as well to provide an objective means of evaluating their status in order to provide a baseline for future analyses.

Figure 2. Conservation status of fishes native to inland waters of California, 1975-2014. Data from reports in 1975 (N = 108), 1989 (N = 115), 1995 (N = 116) and this edition of the report (N = 124). ESA listed species are those listed as threatened or endangered under either state or federal endangered species acts. Species lists change between reports due to extinction, recognition of new taxa, and other reasons (See Box 1).



METHODS

This section describes the: (1) species accounts used for status determination, (2) sources of information used, (3) process used for evaluation, (4) determination of information quality, (5) incorporation of climate change into each evaluation, and (6) evaluation of diverse anthropogenic effects on each species.

*1. **Species accounts***

The status of native fishes of California was evaluated by Moyle et al. (2011) and scores from that study were used as the initial basis for choosing species for inclusion. For this report, eight species were omitted from the analysis for a variety of reasons (Box 1). A species account was created for each fish taxon known to spawn in California's inland waters that is not formally listed as threatened or endangered but is considered to be in decline or limited in distribution to the extent that they may be particularly susceptible to one or more stressors. The species accounts represent the synthesis of available information for each taxon, published and unpublished. Data that had become available since the last report (1995) augmented information from Moyle (2002), Moyle et al. (2008), Moyle et al. (2011) and the two previous editions of this report. For this report, the 62 species accounts are presented in a standard format (Table 1). Literature

Box 1. Species omitted from this report.

The flannelmouth sucker, *Catostomus latipinnis*, was included in the analysis of Moyle et al. (2011) but apparently the only population that now exists in California is in the Colorado River as the result of an introduction; its status is uncertain.

Summer steelhead, *Oncorhynchus mykiss,* is a distinctive life history form of anadromous rainbow trout covered in previous editions of this report. For this report, they are considered to be part of two distinct ESUs of mostly winter-run steelhead, the North California Coast ESU and the Klamath Mountains Province ESU, so are omitted. For an alternative view see Moyle et al. (2008, 2011) and Katz et al. (2012). The two populations were considered together as a distinct taxon (summer steelhead) in previous editions of this report.

Pink salmon (*O. gorbuscha*) and chum salmon (*O. keta)* were included in previous editions (chum salmon in 1995 version only) of this report but reviewers of the accounts thought more information on the status, distribution and stressors affecting their populations was needed before assigning a status score. However, given that California represents the extreme southern end of their range, it is likely that their naturally small populations within the state still merit their inclusion as species of special concern. They are included in Table 8 because they are reproducing members of the California fish fauna (Moyle 2002).

The Shay Creek stickleback, *Gasterosteus sp*., a distinctive fish with a highly restricted distribution in the San Bernadino Mountains, was included in previous editions. However, it is treated by state and federal agencies as part of the unarmored threespine stickleback (*G. aculeatus williamsoni*) complex, which is fully protected as an endangered species under state and federal ESAs.

Staghorn sculpin, *Leptocottus armatus,* and starry flounder, *Platichthys stellatus*, are marine fishes that frequent fresh or brackish water as juveniles, but do not breed in fresh water. They are abundant and were considered part of the total fish fauna in previous editions.

cited is provided as a separate section at the end of the report, rather than at the end of each account, in order to reduce redundancy.

Table 1. Standard format of fish species accounts

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I. Status summary
-Species status category (Table 2) with a brief description of current conservation threats

II. Description

III. Taxonomic relationships
-Summary of latest systematics

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

V. Habitat requirements
-Covers all life history stages and includes basic physiological tolerances (temperature ranges, etc.), where information is available

VI. Distribution
-Present and historic range of the species

VII. Trends in abundance
-An assessment of both long- and short-term trends, using quantitative data where available but, otherwise, assessments are based on whatever information is available

VIII. Nature and degree of threats
-A descriptive catalog of threats to the species, including a standardized table of anthropogenic factors limiting populations (Section 6, Table 7)

IX. Effects of climate change
-An evaluation of the likely effects of climate change on the species in the next 100 years (Section 5)

X. Status determination
-An evaluation of status based on seven metrics (Table 4), a certainty estimate (Table 5) and status ratings from other sources

XI. Management recommendations
-A discussion of what is being done, or proposed to be done, for management and conservation of the species, as well as possible management options

XII. California range map
-Maps included are general distributional maps, based on synthesis of all relevant information in the species accounts

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2. Sources of information

Taxa used are those that can be defined as "species" under the Federal Endangered Species Act of 1973, which include species, subspecies, Evolutionary Significant Units (ESU), and Distinct Population Segments (DPS). Information on the biology and status of each species was derived from detailed reviews in Moyle et al. (1995), Moyle (2002), Moyle et al. (2008), Moyle et al. (2010), Moyle et al. (2011), scientific literature and agency reports issued since the last FSSC report, and by personal communications with biologists working with each taxon. Non-salmonid species that have not yet been formally described in the taxonomic literature are treated as species if they clearly qualify as ESUs or DPSs, based on historic information, new genetic studies, or both. The rationale for inclusion is in the taxonomy section for each species. All species accounts underwent extensive peer-review by species experts. In a few cases, information was updated after field investigations by the authors. The status of each species is as of January 1, 2014. Note that species already listed under either federal or state endangered species acts (or both) are precluded from this report.

3. Evaluation of status

Status assessments were produced from information contained in each account with the use of a standardized protocol designed to quantify threat of extinction (Tables 2-7). Status was determined by averaging numeric scores given to seven metrics (Table 3). Each metric was standardized on a 1-5 scale, where '1' was low (negative effect on status) and '5' was high (no or positive effect) and '2' through '4' were intermediate. Threat level ratings are roughly equivalent across metrics. Collectively, the metrics were designed to cover all factors affecting freshwater fish status in California, with minimal redundancy between metrics. Scores for each metric were awarded according to a standardized rubric (Table 4) and then averaged to produce an overall numeric threat score for each species. A principal components analysis using scores for the entire native freshwater fish fauna of California indicated that no one metric dominated the final threat score (Moyle et al. 2011).

Fishes scoring between 1.0 and 1.9 were labeled Critical Concern and regarded as being in serious danger of extinction in their native range (Table 2). Species with scores between 2.0 and 2.9 were labeled High Concern and considered to be under severe threat but extinction was less imminent than for species with lower scores. However, these species could easily slip into the first category if current trends continue. Species scoring 3.0 - 3.9 were considered to be under no immediate threat of extinction but were in long-term decline or had naturally small, isolated populations which warrant frequent status re-assessment; thus, they were labeled Moderate Concern. Taxa scoring 4.0 to 5.0 were regarded as of Low Concern in California. The scores only apply to populations that spawn in California, so species with a wide distribution outside the state (e.g., western river lamprey) could receive low scores within the state, reflecting California's position at the edge of their range. Data compilation and status assessment methodology are more thoroughly described in Moyle et al. (2011), including evaluations of species not included in this report.

Table 2. Status categories, score ranges, and definitions of status categories for California fishes.

Status	Scores	Definition
Extinct	0	Globally extinct or extirpated from inland waters of California
Critical Concern	1.0 - 1.9	High risk of extinction in the wild; range seriously reduced or greatly restricted in California; population abundance critically low or declining; threats projected to reduce remaining California habitat and populations in the short-term (<10 generations)
High Concern	2.0 - 2.9	High risk of becoming a critical concern species; range and abundance significantly reduced; existing habitat and populations continue to be vulnerable in the short-term (<10 generations)
Moderate Concern	3.0 - 3.9	Declining, fragmented and/or small populations possibly subject to rapid status change; management actions needed to prevent increased conservation concern
Low Concern	4.0 - 5.0	California populations do not appear to be in overall decline; abundant and widespread

Table 3. Rubric used to assign scores to seven metrics developed to assess status of native freshwater fishes in California. Final status score is the average of all seven metric scores. Each metric is scored on a 1-5 scale, where 1 is a major negative factor contributing to status; 5 is a factor with no or positive effects on status; and 2-4 are intermediate values.

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1A. Area occupied: resident fish

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

1B. Area occupied: anadromous fish

1. 0-1 apparent self-sustaining populations
2. 2-4 apparent self-sustaining populations
3. 5-7 apparent self-sustaining populations
4. 8-10 apparent self-sustaining populations
5. More than 10 apparent self-sustaining populations

2. Estimated adult abundance

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

3. Dependence on human intervention for persistence

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

4. Environmental tolerance under natural conditions

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

5. Genetic risks

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

6. Vulnerability to climate change

1. Vulnerable to extinction 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, most habitats will remain within tolerance ranges

7. Anthropogenic threats analysis (see Section 6)

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

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Table 4. Example assessment table for determining status score for California golden trout. Each metric was scored on a 1-5 scale, where 1 is a major negative factor contributing to status; 5 is a factor with no or positive effects on status; and 2-4 are intermediate values. Scores are awarded according to the rubric in Table 3.

Metric	Score	Justification
Area occupied	1	"Pure" California golden trout are confined to a few small tributaries in one watershed
Estimated adult abundance	3	Volcano Creek populations may be <1,000 but, if other populations with conservation value within native range are counted, the numbers would be much higher, perhaps 50,000
Intervention dependence	3	Annual monitoring of barrier performance required; continued implementation of Conservation Strategy is critical
Tolerance	3	Generally tolerant of a wide range of conditions and habitats within their native range
Genetic risk	1	Hybridization with rainbow trout is a constant high risk
Climate change	2	Smaller streams may be negatively impacted by changing climate; improved watershed management may offset some impacts
Anthropogenic threats	2	See Table 1 (within species account)
Average	2.1	15/7
Certainty (1-4)	4	Well documented

4. Certainty of information

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

Table 5. Certainty of information for status evaluations

1.	Status is based on professional judgment, with little or no published information
2.	Status is based on professional judgment augmented by moderate amounts of published or gray literature
3.	Status is based on reports found mainly in the in gray literature with some information in peer-reviewed sources, but where gaps existed in some important areas (e.g., genetics)
4.	Status is based on highly reliable information, with numerous accounts in the peer reviewed and agency literature

5. Climate change

Climate change is already altering fish habitats in California and will continue to do so at an accelerating pace if trends do not change, so it was essential to incorporate ongoing and predicted impacts of climate change into each species evaluation. In general, conditions are worsening for native fishes and improving for many alien fishes. Moyle et al. (2012, 2013) developed a protocol, using 20 metrics, for rating the effects of climate change on each fish species in the state. These ratings are incorporated into this report. The ratings are based on climate change modeling from 2011, and likely underestimate the negative effects of climate change on aquatic ecosystems. For most species of fish in this report, the predicted outcomes of climate change are likely to accelerate current declines, potentially leading to extinction in the next 50-100 years if nothing is done to offset climatic impacts. This section is focused on three major aspects of climate change that affect fish distribution and abundance in California: temperature, precipitation, and sea level rise. This general discussion of expected changes to aquatic systems in California provides background for the individualized climate change sections in each species account.

Temperature. Temperatures have been rising in streams for some time and are continuing to rise (Kaushal et al. 2010). In California, there are diverse climate change models to predict future temperatures, but the more conservative models generally converge on scenarios that assume that within 50–100 years, if not sooner, winter and summer air temperatures will average between 1°C–4°C (1.8°F–7.2°F) and 1.5°C–6°C (2.7°F–10.8°F) warmer, respectively (Miller et al. 2003, Cayan et al. 2009). Further, annual snowpack in the Sierra Nevada and Cascade ranges is expected to diminish greatly, so stream flows will be increasingly driven by rainfall events. An increase in the ratio of rain to snow will result in more peak flows during winter, increased frequency of high flow events (floods), diminished spring pulses, and protracted periods of low (base) flow. In addition, there will be more extended droughts, as well as series of extremely wet years, albeit with dry summers. These conditions will translate into warmer water temperatures at most elevations, reflecting both increases in air temperatures and reduced summer flows.

The region of the state with the greatest uncertainty regarding the future effects of climate change is the North Coast, including the San Francisco Estuary (SFE), because of uncertainties in future changes in ocean temperature, coastal currents, and other factors. If summer fog does not diminish (Diffenbaugh et al. 2004), then many coastal streams may stay cool, if with reduced summer flows. However, observations of foggy day frequency indicate that fog is already decreasing on the coast (Johnstone and Dawson 2010), leading to increasing stream temperatures and decreasing summer flows.

From a fish perspective, the impacts of climate change are likely to be most severe on species requiring cold water (<18°C–20°C, or 64°F–68°F) for persistence, especially the iconic salmon and trout (Katz et al. 2012). The ability of waters of the United States to support cold-water fishes is projected to decrease by 4 to 20 percent by 2030 and by as much as 60 percent by 2100 (Eaton and Scheller 1996), with the greatest loss projected for California because of its naturally warm summer climate (O'Neal 2002, Preston 2006). Warming (more days with maximum temperatures >20°C or >68°F) of the more freshwater regions of the SFE is regarded as an additional threat to declining endemic species such as delta smelt (*Hypomesus transpacificus*) (Wagner et al. 2011).

California's rivers and streams have already been affected by increases in air temperature. Summer water temperatures have likely increased, on average, 0.5°C–1.0°C (0.9°F–1.8°F) in the past 20 years or so (e.g., Bartholow 2005). While such increases may seem small, they can push already marginal waters over thresholds for supporting cold-water fishes. In the Klamath River, where summer temperatures often exceed 22°C (72°F) (McCullough 1999, CDEC 2008), small temperature increases are making the mainstem increasingly inhospitable for Pacific salmon (*Oncorhynchus* spp.) and steelhead trout (*O. mykiss*) that use the river in summer and fall (Quiñones, in press). Likewise, Butte Creek, a salmonid stronghold tributary to the Sacramento River in Tehama County, will likely lose its salmonid fishes in the next 50–100 years as the result of temperature changes (Thompson et al. 2012). Similarly, streams tributary to the SFE are increasingly losing their capacity to support salmonid fishes as water temperatures warm, although the degree to which cold-water habitats will be lost depends on interactions among stream flow (including cold-water releases from dams), urbanization, and effectiveness of restoration efforts (Leidy 2007).

Precipitation. Models indicate that precipitation in California will become more variable, with more falling as rain and less as snow (Cayan et al. 2009). Generally, the total amount of precipitation by 2100 is projected to be less, although the extent of loss is highly uncertain (Cayan et al. 2009). From a fish perspective, present rain-dependent streams will respond somewhat differently than snowmelt-dependent streams, although, as temperatures rise, the hydrologic character of snowmelt streams will become more like those of rain-driven streams.

Snowmelt streams are mainly characteristic of the Sierra Nevada and Cascade mountain ranges. Historically, these mountains had extended spring flows to which local fishes were adapted. However, the hydrograph of many snowmelt streams has been greatly altered by the capture of spring recessional flows by dams. In general, streams will become more variable in flow, with warmer summer and fall temperatures as the result of lower flows and shallower depths (Allan and Castillo 2007). Reductions in flow and depth will result from reduced snowpack, increased frequency of rain storms, and reduced seasonal retention of water in soils and other natural reservoirs (Hayhoe et al. 2004, Stewart et al. 2004, 2005, Hamlet et al. 2005). Elevations below 3000 meters (m) will likely suffer the most (80 percent) loss of snowpack (Hayhoe et al. 2004), as well as reduction in water content of remaining snow (e.g., Van Kirk and Naman 2008). Earlier snowmelt has already moved the timing of high flows forward by 10 to 30 days, on average (Stewart et al. 2005), with annual peak discharges, in particular, occurring earlier

(Cayan et al. 2001, 2009). These changes dramatically affect flows in low-elevation rivers in the Central Valley and are leading to modified operation of reservoirs (dam releases), which further affect flows.

Streams that are already dependent on rain will become even more variable, with greater extremes in high and low flows, leading to drying of long stream reaches on occasion. In interior and south-coastal California, such streams already show highly variable flow regimes, with "flashy" flows in winter in response to rain events (e.g., Cosumnes River; Moyle et al. 2003). Winter rains created some of the most extreme flow events ever recorded for California such as the major floods of 1955 and 1964 in the Eel and other coastal rivers (e.g., Yoshiyama and Moyle 2010), as well as the 'New Year floods' of 1997 that had widespread impacts to riverine habitats.

Overall, the amount of water carried by streams in California (and the rest of the western United States), if present trends continue, will decrease by 10 to 50 percent during drier months (e.g., Cayan et al. 2001). More important, extreme high- and low-flow events are projected to increase by 15 to 20 percent (Leung et al. 2004), especially in the northern Sierra Nevada and southern Cascade Range (Kim 2005). This increased incidence of extreme events will test the adaptive ability of native stream fishes.

Sea level rise. Projections of the rate of sea level rise are changing, usually upwards, as better information becomes available. Cayan et al. (2009) project a rise in sea level of 35–50 centimeters (cm) in the next 50 years, while Knowles (2010) projects a rise of as much as 150 cm by 2100. Other scenarios range from optimistic projections of 45–70 cm by 2100 to pessimistic projections of 1500 to 3500 cm (Knowles 2010). Accompanying the mean rise of sea level will be an increase in major events that enhance effects of sea rise, such as high tides, storm surges, and coincidence of high tides with high outflows from rivers (Cayan et al. 2009). For fishes, a major consequence of sea level rise will be the reduction or loss of tidal marsh habitats (Moyle et al. 2012).

These predictions for climate change effects are consistent with other recent reports of large-scale climate change effects in California and how aquatic habitats and native flora and fauna will adapt to them (e.g., RLF 2012, Kadir et al. 2013).

6. Anthropogenic threats analysis

For each species, an analysis was conducted of 15 anthropogenic factors (listed below) which limit, or potentially limit, a taxon's viability (Table 7); the ratings of these factors were then combined to create a single evaluation variable. Factors were rated on a five-level ordinal scale (Table 6), where a factor rated "critical" could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated "high" could push a species to extinction in 10 generations or 50 years, whichever is less; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction is unlikely as a result; and a factor rated "n/a" has no known negative impact to the taxon under consideration. Descriptions of most of these factors, with access to literature on which they are based, can be found in Moyle (2002).

Table 6. Criteria for ratings assigned to anthropogenic threat factors with correlated time-lines.

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

Major dams. Dams were recorded as having a high impact on a species if they prevent access to a large amount of its range, if they caused major changes to habitats, or if they significantly changed downstream water quality and or quantity. The effects and impacts of reservoirs created by dams were also evaluated. Dams were regarded as having a low impact if they were present within the range of the species but their effects were either minimal or poorly known.

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

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

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

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

Instream mining. Widespread and often severe instream mining impacts occurred during the mid-19th and early 20th century in California, due largely to 'Gold Rush fever.' Many rivers were excavated, dredged and hydraulically mined for gold, causing dramatic stream degradation; these legacy effects are still evident in numerous watersheds (e.g., the so-called 'Gold Fields' on the lower Yuba River and the expansive tailing piles along the lower American and Trinity rivers). Locally severe impacts also occurred as a result of instream gravel mining operations, for which large pits were dug into streambeds and stream banks and riparian vegetation were highly degraded. Such mining is now largely banned (in favor of mining off-channel areas) but lasting habitat impacts remain in many areas. Instream mining was usually rated moderate when present, although severe legacy effects at a localized level resulted in high ratings for impacts to some species. The negative effects from contemporary recreational and professional suction dredge mining for gold (although currently under moratorium in California) led to high ratings in some instances, due to relatively recent (within the past 10 years) intensive suction dredging in some areas.

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

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

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

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

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

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

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

Hatcheries. Hatcheries and releases of hatchery-reared fish into the wild can negatively impact wild fish populations through competition, predation, potential introduction of disease, and loss of fitness and genetic diversity (Kostow 2008, Chilcote et al. 2011). Many California fish species of concern have no hatchery augmentation and or occur in waters that are not stocked; hatchery influences are largely relegated to anadromous fishes that occur in rivers blocked by major dams (e.g., the various races of salmon and steelhead trout) or those that occur in lake or reservoir habitats that are stocked for recreational purposes (e.g., Eagle Lake rainbow trout, Lahontan Lake tui chub). The severity of hatchery impacts were rated based, in part, on hatchery dependence to support a species of concern and or the threat of interbreeding between wild and hatchery populations.

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

Table 7. Major anthropogenic factors limiting, or potentially limiting, viability of native freshwater fishes of California, using California golden trout as an example.

	Rating	Explanation
Major dams	n/a	All major dams are outside the native range of California golden trout
Agriculture	n/a	
Grazing	Medium	Ongoing threat but greatly reduced from the past
Rural residential	n/a	
Urbanization	n/a	
Instream mining	n/a	
Mining	n/a	Historic mines are present but have no known impacts
Transportation	Low	Trails and off-road vehicle routes can be a source of sediment and pollution input into streams; direct habitat impacts from wet route crossings
Logging	Low	This is an important land use in the broader region but probably has no direct effect on golden trout streams
Fire	Low	Because of fire suppression, headwater areas could be impacted by hot fires, although this is unlikely given the sparse plant communities in region
Estuary alteration	n/a	
Recreation	Low	Pure populations within the Golden Trout Creek watershed are entirely within designated wilderness; South Fork populations with conservation value are also within designated wilderness
Harvest	Low	Potential impact but light pressure and most fishing is thought to be catch and release
Hatcheries	Low	Residual effects of hybridization with hatchery fish
Alien species	High	Major cause of limited distribution in South Fork Kern; however, very limited introgression with rainbow trout and no brown trout in waters within Golden Trout Creek watershed

Table 8. List of native freshwater fishes in California, showing status scores (from this report and Moyle et al. 2011) and status rating. See Box 1 for eight species not covered by this report. Species with names in bold are covered in this report. Species noted with an asterisk (*) are already listed under federal or state (or both) endangered species acts and, therefore, not included in this report. Species rated as Low Concern are not included, for intuitive reasons, with one exception. The following are roughly equivalent designations using criteria of the International Union for the Conservation of Nature (IUCN): Critical Concern = IUCN endangered; High Concern = IUCN vulnerable; Moderate Concern = IUCN near-threatened; Low Concern = IUCN least concern.

Species	**Score**	**Status (concern)**
Petromyzontidae		
Pacific lamprey, *Entosphenus tridentata*	3.3	Moderate
Goose Lake lamprey, *Entosphenus sp*.	2.9	High
Northern California brook lamprey, *E. folletti*	2.4	High
Klamath River lamprey, *E. similis*	3.9	Moderate
Western river lamprey, *Lampetra ayersi*	3.6	Moderate
Kern brook lamprey, *L. hubbsi*	2.3	High
Western brook lamprey, *L. richardsoni*	3.0	Moderate
Pit-Klamath brook lamprey, *L. lethophaga*	3.7	Moderate
Acipenseridae		
Northern green sturgeon, *Acipenser medirostris*	2.7	High
Southern green sturgeon, *A. medirostris**	1.6	Critical
White sturgeon, *A. transmontanus*	2.3	High
Cyprinidae		
Thicktail chub, *Siphatales crassicauda*	0.0	Extinct
Goose Lake tui chub, *S. t. thalassinus*	3.1	Moderate
Pit River tui chub, *S. thalassinus subsp.*	4.0	Low
Cow Head tui chub, *S. t. vaccaceps*	2.4	High
Klamath tui chub, *S. b. bicolor*	4.1	Low
High Rock Springs tui chub, *S. b. subsp.*	0.0	Extinct
Lahontan lake tui chub, *S. b. pectinifer*	2.4	High
Lahontan stream tui chub, *S. b. obesus*	4.7	Low
Eagle Lake tui chub, *S. b. subsp*.	3.3	Moderate
Owens tui chub, *S. b. snyderi**	1.4	Critical
Mojave tui chub, *S. mohavensis**	1.4	Critical
Bonytail, *Gila elegans*	0.0	Extinct

Blue chub, *Gila coerulea*	3.4	Moderate
Arroyo chub, *Gila orcutti*[1]	2.1	High
Lahontan redside, *Richardsonius egregius*	4.8	Low
Sacramento hitch, *Lavinia e. exilicauda*	3.1	Moderate
Clear Lake hitch, *L. e. chi**	1.7	Critical
Monterey hitch, *L. e. harengeus*	3.1	Moderate
Central California roach, *L. s. symmetricus*	3.3	Moderate
Red Hills roach, *L. s. subsp.*	2.1	High
Russian River roach, *L. s. subsp*	3.3	Moderate
Clear Lake roach, *L s. subsp.*	3.6	Moderate
Monterey roach, *L. s. subditus*	3.4	Moderate
Navarro roach, *L. s. navarroensis*	3.3	Moderate
Tomales roach, *L. s. subspecies*	3.1	Moderate
Gualala roach, *L. parvipinnus*	3.0	Moderate
Northern roach, *L. mitrulus*	2.9	High
Sacramento blackfish, *Orthodon microlepidotus*	4.4	Low
Sacramento splittail, *Pogonichthys macrolepidotus*	3.1	Moderate
Clear Lake splittail, *P. ciscoides*	0.0	Extinct
Hardhead, *Mylopharodon conocephalus*	3.1	Moderate
Sacramento pikeminow, *Ptychocheilus grandis*	4.7	Low
Colorado pikeminnow, *P. lucius*	0.0	Extinct
Sacramento speckled dace, *Rhinichthys osculus* subp.	4.1	Low
Lahontan speckled dace, *R. o. robustus*	4.8	Low
Klamath speckled dace, *R. o. klamathensis*	4.8	Low
Owens speckled dace, *R. o. subsp.*	2.6	High
Long Valley speckled dace, *R. o. subsp.*	1.0	Critical
Amargosa Canyon speckled dace, *R. o. nevadensis*	1.9	Critical
Santa Ana speckled dace, *R. o. subsp.*	1.6	Critical
<u>Catostomidae</u>		
Tahoe sucker, *Catostomus tahoensis*	5.0	Low
Owens sucker, *C. fumeiventris*[2]	4.0	Low
Lahontan mountain sucker, *C. platyrhynchus*	3.1	Moderate
Sacramento sucker, *C. o. occidentalis*	5.0	Low
Goose Lake sucker, *C. o. lacusanserinus*	2.3	High
Monterey sucker, *C. o. mniotiltus*	4.1	Low
Humboldt sucker, *C. o. humboldtianus*	4.3	Low
Modoc sucker, *C. microps**	1.6	Critical
Klamath smallscale sucker, *C. rimiculus*	4.1	Low
Klamath largescale sucker, *C. snyderi*	1.9	Critical

[1] Arroyo chub is rated 3.1 (Moderate Concern) if populations outside its native range are included in status assessment.

[2] The Owens sucker was a species of special concern in previous reports but our evaluation indicates it is secure; we leave it in this edition because of remaining uncertainties and its inclusion in previous reports.

Lost River sucker, *C. luxatus**	1.7	Critical
Santa Ana sucker, *C. santaanae**	1.7	Critical
Shortnose sucker, *Chasmistes brevirostris**	2.0	High
Razorback sucker, *Xyrauchen texanus**	1.3	Critical
~~Osmeridae~~		
Eulachon, *Thaleichthys pacificus**	1.6	Critical
Longfin smelt, *Spirinchus thaleichthys**	2.0	High
Delta smelt, *Hypomesus transpacificus**	1.4	Critical
~~Salmonidae~~		
Mountain whitefish, *Prosopium williamsoni*	3.9	Moderate
Bull trout, *Salvelinus confluentus*	0.0	Extinct
Upper Klamath-Trinity fall Chinook salmon, *Oncorhynchus tshawytscha*	3.0	Moderate
Upper Klamath-Trinity spring Chinook salmon, *O. tshawytscha*	1.7	Critical
Southern Oregon-Northern California coast fall Chinook salmon, *O. tshawytscha*	3.3	Moderate
California Coast fall Chinook salmon, *O. tshawytscha**	2.4	High
Central Valley winter Chinook salmon, *O. tshawytscha**	2.0	High
Central Valley spring Chinook salmon, *O. tshawytscha**	2.0	High
Central Valley fall Chinook salmon, *O. tshawytscha*	2.7	High
Central Valley late fall Chinook salmon, *O. tshawytscha*	2.6	High
Central coast coho salmon, *O. kisutch**	1.1	Critical
Southern Oregon Northern California coast coho salmon, *O. kisutch**	1.6	Critical
Pink salmon, *O. gorbuscha*[3]	?	Undecided
Chum salmon, *O. keta*[4]	?	Undecided
Northern California coast winter steelhead, *O. mykiss**	3.3	Moderate
Klamath Mountains Province steelhead, *O. mykiss*	2.9	High
Central California coast winter steelhead, *O. mykiss**	2.7	High
South Central California coast steelhead, *O. mykiss**	2.4	High
Southern California steelhead, *O. mykiss**	1.7	Critical
Central Valley steelhead, *O. mykiss**[5]	2.4	High
Coastal rainbow trout, *O. m. irideus*	4.7	Low
McCloud River redband trout, *O. m. stonei*	1.7	Critical
Goose Lake redband trout, *O. m. subsp*.	3.3	Moderate

[3] More information on the status, distribution and stressors affecting pink salmon populations in California is needed in order to score this species. However, given that California represents the extreme southern end of their range, it is likely that naturally small populations in relatively low numbers within the state would merit their inclusion as a species of special concern. See Box 1.
[4] Same comment as for pink salmon.
[5] Genetic evidence indicates that all CV steelhead as currently defined by NMFS are hybridized with north coast steelhead of hatchery origin.

Eagle Lake rainbow trout, *O. m. aquilarum*	2.1	High
Kern River rainbow trout, *O. m. gilberti*	1.7	Critical
California golden trout, *O. m. aguabonita*	2.1	High
Little Kern golden trout, *O. m. whitei**	2.0	High
Coastal cutthroat trout, *O. clarkii clarkii*	3.0	Moderate
Paiute cutthroat trout, *O. c. seleneris**	1.7	Critical
Lahontan cutthroat trout, *O. c. henshawi**	2.1	High
<u>Fundulidae</u>		
California killifish, *Fundulus parvipinnis*	4.1	Low
<u>Cyprinodontidae</u>		
Desert pupfish, *Cyprinodon macularius**	1.9	Critical
Owens pupfish, *C. radiosus**	1.4	Critical
Saratoga Springs pupfish, *C. n. nevadensis*	2.3	High
Amargosa River pupfish, *C. n. amargosae*	2.3	High
Tecopa pupfish, *C. n. calidae*	0.0	Extinct
Shoshone pupfish, *C. n. shoshone*	1.1	Critical
Salt Creek pupfish, *C. s. salinus*	2.7	High
Cottonball Marsh pupfish, *C. s. milleri**	2.4	High
<u>Cottidae</u>		
Rough sculpin, *Cottus asperrimus**	3.4	Moderate
Bigeye marbled sculpin, *C. klamathensis macrops*	3.0	Moderate
Lower Klamath marbled sculpin, *C. k. polyporus*	3.9	Moderate
Upper Klamath marbled sculpin, *C. k. klamathensis*	1.7	Critical
Coastal Prickly sculpin, *C. asper subsp.*	4.7	Low
Clear Lake prickly sculpin, *C. a. subsp.*	3.3	Moderate
Coastrange sculpin, *C. aleuticus*	4.4	Low
Riffle sculpin, *C. gulosus*	3.0	Moderate
Pit sculpin, C. pitensis	4.3	Low
Paiute sculpin, *C. beldingi*	4.4	Low
Reticulate sculpin, *C. perplexus*	4.0	Low
<u>Gasterosteidae</u>		
Coastal threespine stickleback, *Gasterosteus a. aculeatus*	4.6	Low
Inland threespine stickleback *G. a. microcephalus*	4.1	Low
Unarmored threespine stickleback, *G. a. williamsoni**	1.9	Critical
<u>Centrarchidae</u>		
Sacramento perch, *Archoplites interruptus*	1.9	Critical
<u>Embiotocidae</u>		
Sacramento tule perch, *Hysterocarpus traski traski*	4.0	Low
Russian River tule perch, *H.t. pomo*	3.7	Moderate
Clear Lake tule perch, *H. t. lagunae*	2.3	High
<u>Gobiidae</u>		
Tidewater goby, *Eucyclogobius newberryi**	2.9	High

PACIFIC LAMPREY
Entosphenus tridentatus

Status: Moderate Concern. Pacific lampreys are in decline throughout their range in California. However, they are still widespread so the species does not appear in immediate danger of extinction in the state. Some local or regional (e.g., southern California) populations may face considerably higher threat of extirpation in the near future.

Description: Pacific lampreys are the largest (> 40 cm TL) lampreys in California. However, landlocked Pacific lamprey populations may have dwarf (15-30 cm TL) morphs. The sucking disc is characterized by having sharp, horny plates (teeth) in all areas (Vladykov and Kott 1979). The crescent-shaped supraoral lamina is the most distinctive plate, with three sharp cusps, of which the middle cusp is smaller than the two lateral ones. There are four large lateral plates on both sides of the supraoral lamina. The outer two lateral plates are bicuspid, while the middle two are tricuspid (formula 2-3-3-2). The tip of the tongue has 14-21 small points (transverse lingual lamina), of which the middle one is slightly larger than the rest. The two dorsal fins are discontinuous but the second dorsal is continuous with the caudal fin. Adults generally have 62-71 body segments (myomeres), while juveniles have 68-70 body segments between the anus and last gill opening (Wang 1986). The diameter of the eye and oral disc, respectively, are 2-4 percent and 6-8 percent of the total length. Males tend to have higher dorsal fins than females, lack a conspicuous anal fin and possess genital papillae. Body color varies by developmental stage. For juveniles (ammocoetes), the body and lower half of the oral hood is dark or medium brown, with a pale area near the ridge of the caudal region. Newly metamorphosed juveniles (macropthalmia) are silvery with a slightly bronze cast. Spawning adults are usually dark greenish-black or dark brown in color.

Taxonomic Relationships: The use of the genus name *Entosphenus* reflects the phylogenetic study of Gill et al. (2003) that places this genus as a separate lineage from *Lampetra,* into which all western North American lampreys had been lumped. Genetic analysis of populations of from British Columbia to southern California have found little variation among populations, suggesting that gene flow occurs readily throughout their range (Goodman et al. 2008, Docker 2010). However, populations in the northern part of the range exhibit reduced genetic richness (Goodman et al. 2008), perhaps reflecting locally adapted population segments.

Pacific lampreys have given rise to landlocked populations throughout their range, including predatory species (e.g., *E. similis;* refer to separate species accounts). Populations have also become isolated upstream of reservoirs resulting from dam construction, including populations in Clair Engle Reservoir (Trinity River) and Clear Creek, upstream of Whiskeytown Reservoir (Brown and May 2007). Considerable overlap of morphometric characters exists between Pacific lamprey and its derivatives, as well as between predatory and nonpredatory forms, especially in the Klamath River basin (Bond and Kan 1973, Bailey 1980, Lorion et al. 2000), so careful examination is required for identification. Studies of mitochondrial DNA (Docker et al. 1999) and statistical analysis of morphometric characteristics (Meeuwig et al. 2006) show promise in resolving interrelationships among species.

Life History: Pacific lampreys have more diverse life histories than generally recognized. Within the same river system they may have more than one run (Anglin 1994) or individuals that do not migrate to sea. For example, two forms of Pacific lamprey exist in the Trinity River, one smaller and paler than the other, representing either separate runs or resident and anadromous individuals (T. Healey, CDFW, pers. comm. 1995). It is possible that lamprey in the Klamath and Eel rivers, as well as other large river systems, have a number of distinct runs, similar to salmon. One indication is that many adults migrate upstream and hide under logs and boulders for months until they mature, with a life history akin to that of summer steelhead or spring-run Chinook salmon (Beamish 1980, ENTRIX 1996). Two distinct runs may exist in the Klamath River: a spring-run of adults that spawn immediately after upstream migration and a fall-run of individuals that wait to spawn until the following spring (Anglin 1994). A large spring-run and smaller fall-run have been observed in the Russian River (Brown et al. 2010); the two runs were observed from 2000 to 2007 (S. Chase, Sonoma County Water Agency, unpubl. data) with the use of underwater video (at Mirable, 37 rkm), primarily from the beginning of August to the onset of heavy rains (November to December), as well as in the spring months. The general run trend is low numbers of migrants in October and November and higher numbers in the spring.

Adult Pacific lampreys are micropredators (i.e., they feed on prey larger than themselves) during their oceanic existence, consuming the body fluids of a variety of fishes, including salmon and flatfishes (Beamish 1980) and marine mammals (Close et al. 2002). Beamish (1980) found that 14-45 percent of the salmon returning to British Columbia had scars from lamprey predation. Similar data are not available for salmon in California. Adult lampreys themselves are prey for other fishes, including sharks, and are often found with parts of their tails missing. Sea lions, near the mouth of the Rogue River, Oregon, have been observed eating large numbers of migrating lampreys (Jameson and Kenyon 1977). Lamprey predation is largely confined to fishes that occupy estuaries and nearshore coastal areas. However, some individual lampreys have been caught in waters up to 70 m deep (Beamish 1980) and as far as 100 km from shore (Close et al. 2002). The oceanic phase lasts approximately 3-4 years in British Columbia, but is likely of shorter duration in southern waters. Pacific lamprey predation appears to have little effect on fish populations (Moyle 2002, Orr et al. 2004).

Adult (30-76 cm TL) spawning migrations usually take place between early March and late June, but migration has also been documented in January and February (ENTRIX 1996, Trihey and Associates 1996b), as well as in July in northern streams. Spawning migrations have been documented in August and September in the Trinity River (Moffett and Smith 1950). Most upstream movements occur in surges at night, with some individuals migrating fairly continuously over the course of two to four months. In the Santa Clara River (Ventura County), migration was initiated after the sand bar blocking the lagoon at the mouth was breached by winter rains in January, February, or March; adults reached a fish ladder 16.8 km upstream within 6-14 days of the breach (ENTRIX 1996). In the Santa Clara River, lampreys migrated mostly during high flows, but also moved in flows ranging from 25 to 1700 m^3/min (ENTRIX 1996).

Lampreys will migrate considerable distances and are stopped only by major barriers, such as dams. Lampreys were observed spawning in Deer Creek (Tehama County), about 440 km from the ocean (P. Moyle, unpublished observation). Presumably, migrations of more than 500 km were once common. In the Klamath River, Humboldt County, radio tagged lampreys migrated an average of 34 km over the course of 25 days at a travel rate of 2 km/day (McCovey et al. 2007). Adults do not feed during spawning migrations (Beamish 1980) but can survive extended periods (months to two years) without food, allowing them to migrate long distances (Whyte et al. 1993). Pacific lampreys seem to have poorly developed homing abilities (Hatch and Whiteaker 2009). If this is true, then lamprey populations are likely regulated by source-sink dynamics, where large river populations (such as those historically present in the Eel River) sustain populations in smaller adjacent rivers or tributaries, where localized extinctions can occur periodically due to stochastic events such as floods and droughts (e.g. a drying event, even short-term, could eliminate multiple age classes of ammocoetes). The source-sink model would also explain persistence of lampreys in habitats that are often unsuitable (e.g. in southern California rivers). The sink populations may disappear as source populations shrink and the number of potential recruits to the sink population becomes reduced or non-existent. This model is speculative but seems to fit with recent findings of lamprey behavior and population dynamics and is consistent with ecological theory (metapopulation dynamics).

Once at a spawning site, typically in a low-gradient riffle, both sexes build a nest depression 21-270 cm in diameter (Gunckel et al. 2009), with depths of 30-150 cm, at temperatures of 12-18 °C (Moyle 2002). Depths of nests range from 30-82 cm (mean of 59 cm) in the American River, while ranging from 36 to 73 cm (mean of 50 cm) in Putah Creek. Nest construction has been observed in water as deep as 1.5 m in Deer Creek, Tehama County (Moyle, unpublished observations). Water velocity at nests in the American River ranged from 24-84 cm/sec, in comparison to 17-45 cm/sec in Putah Creek. Although Pacific lampreys most commonly spawn in flowing water, spawning has also been observed in lentic systems (Russell et al. 1987). Lampreys attach themselves to the downstream end of rocks and swing vigorously in reverse to remove substrates during nest construction. More than one individual may pull at the same rock until the combination of pulling and pushing dislodges the rock (Stone 2006). Adults may test several nest sites ('false digs') before fully digging a nest (Stone 2006). Nests are shallow depressions, with piles of stones at either the downstream (Moyle 2002) or upstream (Susac and Jacobs 1999) end of the nest. In order to mate, the female attaches to a rock on the upstream end of the nest, while the male attaches himself to the head of the female and wraps his body around hers. Occasionally, both will attach to rocks while staying side by side (Wang 1986). Eggs and milt are released when both vibrate rapidly. Fertilized eggs float downstream, where most adhere to rocks at the downstream end of the nest.

After spawning, lampreys loosen sediment upstream of the nest to cover the embryos. Spawning is repeated in the same nest until the adults are spent. Males may mate with more than one female (Wang 1986). About 48 individuals were observed using the same nest in the Smith River, Oregon (Gunckel et al. 2006). The average time spent in spawning areas is less than seven days for both sexes (Brumo 2006). Adults may defend their nests; Stone (2006) observed a male using his oral disc to remove a sculpin

(*Cottus* spp.) from its nest in Cedar Creek, Washington. Both sexes usually die after spawning. However, some adults may live to spawn for one more year in Washington streams (Michael 1984). Repeat spawning may also occur in the Santa Clara River, as indicated by the fact that live adults have been caught in downstream migrant traps (ENTRIX 1996). The fecundity of females ranges from 20,000 to 238,000 eggs (Kan 1975).

At 15 °C, embryos hatch in 19 days. Upon hatching, ammocoetes stay in the nest for a short period of time and then swim into the water column where they are washed downstream to areas of sand or mud. Ammocoetes burrow into soft stream sediments tail first, at which point they begin filter feeding by sucking organic matter and algae from stream substrates. Survival to this stage may be related to stream discharge at time of spawning and density dependent effects (e.g., amount of rearing habitat and prey items) associated with ammocoete abundance (Brumo 2006). Ammocoetes leave their burrows and drift to other areas at night throughout their freshwater residency (White and Harvey 2003). Larger ammocoetes commonly drift in spring high flows, while smaller ammocoetes drift during the summer. Consequently, they can be trapped during much of the year (Moffett and Smith 1950, Long 1968). In the Trinity River, ammocoetes as small as 16 mm recolonized areas from which they had been removed by winter floods (Moffett and Smith 1950)

The ammocoete stage probably lasts 5-7 years, at the end of which ammocoetes measure 12-14 cm TL and metamorphosis to macropthalmia begins. Lampreys develop large eyes, a sucking disc, silver sides and dark blue backs during metamorphosis. Their physiology and internal anatomy (McPhail and Lindsey 1970) also change dramatically. Physiological changes allow adult lampreys to tolerate salt water, which is lethal to ammocoetes (Richards and Beamish 1981). Saltwater tolerance coincides with the opening of the foregut lumen (Richards and Beamish 1981). Downstream migration begins when metamorphosis is completed and is often associated with high flow events in the winter and spring, perhaps coincident with adult upstream migration. Most volitional movement of macropthalmia occurs at night (Dauble et al. 2006).

It is likely that Pacific lamprey life history has played a key role in their persistence. The extended freshwater residency of ammocoetes allows populations to withstand low flows or other conditions that might block adult spawning runs over the course of several years. This may explain, for example, why a small population of Pacific lamprey persists in the San Joaquin River near Fresno (D. Mitchell, CDFW, pers. comm. 2007).

An underappreciated aspect of Pacific lampreys is their importance in the food webs of stream ecosystems. Ammocoetes break down detritus and are sources of prey for other fishes (Cochran 2009). Adult carcasses may be an important source of marine derived nutrients (e.g. nitrogen) to oligotrophic streams (Wipfli et al. 1998, Close et al. 2002, Lewis 2009).

Habitat Requirements: Pacific lampreys share many habitat requirements with Pacific salmonids (*Oncorhynchus* spp; Close et al. 2002, Stone 2006), particularly cold, clear water (Moyle 2002) for spawning and incubation. They also require a wide range of habitats across life stages. In general, peak spawning appears to be closely tied to water temperatures that are suitable for early development (Close et al. 2003, Meeuwig et al.

2005) but can occur at temperatures above 22 °C (Luzier et al. 2006). Consequently, temperature may be important in determining ammocoete abundance (Young et al. 1990, Youson et al. 1993, Bayer et al. 2000). Juveniles can persist in flows of up to 40 cm/s but are generally most common at velocities of 20-30 cm/s (Close 2001).

Adults use gravel areas to build nests, while ammocoetes need soft sediments in which to burrow during rearing (Kostow 2002). Nests are generally associated with cover, including gravel and cobble substrates, vegetation and woody debris. Likewise, most nests observed in Cedar Creek, Washington, were observed in pool-tail outs, low gradient riffles and runs (Stone 2006). Pacific lamprey embryos hatch at a wide range of temperatures (10-22 °C). However, in the laboratory, time from fertilization to hatching was around 26 days at 10 °C and around 8 days at 22 °C (Meeuwig et al. 2005). Survival of embryos was highest at temperatures ranging from 10 to 18 °C. Survival declined sharply, with a significant increase in abnormalities, at 22 °C.

Ammocoetes burrow into larger substrates as they grow (Stone and Barndt 2005). Ammocoetes also need detritus that produces algae for food (Kostow 2002) and habitats with slow or moderately slow water velocities (0-10 cm/s; Stone and Barndt 2005), such as low gradient riffles, pool tailouts and lateral scour pools (Gunckel et al. 2009).

Adults can climb over waterfalls and other barriers, using their sucking disc, as long as there is a rough surface and some amount of flow. These features are rarely present on dams, so even small dams or fish ladders can be barriers if not designed with surfaces and features that allow climbing (as in CRBLTW 2004).

Distribution: Pacific lampreys occur along the Pacific coast from Hokkaido Island, Japan (Morrow 1980), through Alaska and south to Rio Santo Domingo in Baja California (Ruiz-Campos and Gonzalez-Guzman 1996). Anadromous forms of Pacific lamprey occur below impassible barriers throughout their range. In California, Pacific lampreys occur from Los Angeles to Del Norte counties and the rivers in the Central Valley. Although a few individuals have been recorded in the Santa Ana, Los Angeles, San Gabriel and Santa Margarita rivers, the occurrence of all forms is infrequent south of Malibu Creek, Los Angeles County. The southernmost record in California is a single ammocoete collected from the San Luis Rey River, San Diego County, in 1997 (Swift and Howard 2009). A sizable run was recorded in the 1990s in the Santa Clara River (Chase 2001). However, their numbers appear to have significantly declined in the last few years (Swift and Howard 2009). There are also records from the Rio Santo Domingo, Baja California (Ruiz-Campos and Gonzalez-Guzman 1996). In general, lamprey distribution in California becomes irregular and erratic south of San Luis Obispo County (Swift et al. 1993, Swift and Howard 2009). An unusual landlocked population has persisted in Clair Engle Reservoir (Trinity River, Trinity County) since 1963, when the dam was constructed.

In the Central Valley, their upstream range appears to be limited by impassable dams that exist on all large rivers. Ammocoetes and spawning individuals have been observed in the San Joaquin River below Friant Dam and in most major tributaries from the Merced River north to the Feather River, as well as in some smaller tributaries, such as Putah Creek, Yolo-Solano counties. Ammocoetes have been observed along the edges of channels in the Sacramento-San Joaquin Delta, primarily in the north Delta (e.g. around McCormick-Williamson Tract; P. Moyle unpublished data). Both downstream migrating juvenile lampreys and returning adults must pass through the entire San Francisco Estuary, but their requirements for passage are not known.

Trends in Abundance: Anadromous Pacific lamprey abundance has declined so that large runs have disappeared from rivers such as the Eel River (Moyle 2002, Yoshiyama and Moyle 2010), although small runs persist in some portions of their range. Runs have also largely disappeared from southern California streams (Swift and Howard 2009). Abundance estimates for Pacific lamprey populations in California are scarce, but rotary screw trap data from 1997 to 2004 in the Klamath River basin suggested a declining trend for all life stages (USFWS 2004). Native American fishermen in the Klamath basin have also observed that runs are much smaller than they once were in this system (Larson and Belchik 1998). Traps for salmonid smolts in Redwood Creek, Humboldt County, capture 5-91 lampreys per year, all post-spawners (M. Sparkman, CDFW, pers. comm. 2011). Lampreys in Oregon and Washington have also shown significant declines, similar to those in California. For example, counts at Winchester Dam on the lower Umpqua River, Oregon, have declined from a maximum of 46,785 in 1966 to 34 in 2001 (ODFW in Close et al. 2002). In the Columbia River basin, the number of Pacific lamprey passing Bonneville Dam has declined from an estimated 50,000 adults prior to 1970 to less than 25,000 with a progressively sharper decline in Pacific lamprey abundance further upstream (Kostow 2002). Despite obvious declines wherever lampreys are actually counted, declines in Pacific lamprey are largely unrecognized, in part because they still occupy much of their historic range and most streams appear to retain at least small runs. The latter may be due to a low degree of fidelity to spawning areas (Goodman et al. 2006, Docker 2010), so recolonization of altered streams may occur fairly quickly when conditions improve, provided there is a source population nearby. However, this pattern of rapid dispersal may actually mask an overall decline in numbers.

Thus, a population in Putah Creek (Yolo and Solano counties) reestablished itself following completion of the Solano Project, which dewatered lower portions of the stream, and, again, following an extended drought during which much of the stream was dry. The apparent lack of strong homing tendencies in Pacific lampreys suggests that they have the ability to temporarily colonize impaired habitats, even if they cannot sustain populations in these areas. However, the apparent loss of the largest known southern California population in the Santa Clara River (Swift and Howard 2009) indicates that their distribution and abundance is shrinking and certain portions of their range may no longer provide suitable habitats.

Nature and Degree of Threats: Threats to Pacific lampreys are diverse and usually multiple for any given population (Table 1). The nature and degree of these threats are poorly understood, given the general lack of information on factors affecting lamprey populations. The Pacific lamprey has such a wide geographic range that different factors likely influence its abundance in different areas. Hence, there are no ‘high’ or ‘critical’ scores for threats to all California populations, combined, but a remarkable nine ‘medium’ scores, which could actually be ‘critical’ or ‘high’ in different rivers (Table 1). It is likely that factors that have led to population declines of anadromous salmonids across California may also be the main causes for decline of Pacific lamprey, especially given these fishes share so many ecological and habitat requirements.

One universal factor, related to all others but not rated here, is reduction in prey abundance, especially salmonids, due to stressors such as dams, diversions, habitat degradation and over-exploitation. Adult Pacific lampreys depend on having large populations of large prey species, such as salmon, to maintain their own numbers. In British Columbia, salmon are among the most important prey of lampreys (Beamish 1980), as they may be elsewhere in their range. While the importance of different prey species is unknown for populations of lampreys in California, the fact that Chinook and coho salmon populations have severely declined in most California rivers suggests that lamprey declines may be closely tied to salmonid declines.

Dams and diversions. Large dams have reduced the range of Pacific lampreys in many streams, as they have for salmon and steelhead, by preventing upstream passage to spawning and rearing areas and reducing suitability of downstream habitats. Lampreys are capable of passing over some small dams and diversion structures, either by using fish ladders or by using their suction cup-like mouths to work their way over barriers, provided the surfaces are wet and rough. Large dams without passage structures, however, occur throughout their range and prohibit upstream migration to large portions of their former range.

Where documentation exists for regulated streams, lamprey populations have declined from historic numbers. Unsuitable flow regimes for migration, along with loss of spawning and backwater rearing habitats combine to make regulated streams unfavorable for lampreys. Flow regimes that limit emigration or immigration may have delayed effects and declines may be difficult to detect; the long lifespan of ammocoetes and the apparent lack of homing behavior in adults can give the impression of persisting populations in streams with only intermittent access. During unseasonably high-flow events, ammocoetes may be flushed to unsuitable habitats because they are poor swimmers (Dauble et al. 2006). Spawning habitat is lost when recruitment of sediments from upstream areas is blocked by dams; lack of sediment imbeds rocks in spawning areas, making them more difficult to move for nest creation. Reduction in sand and silt recruitment, combined with channelization, may also reduce suitable habitats available for ammocoetes below large dams (Close et al. 2002).

Agriculture. Lampreys are typically rare or absent from river reaches heavily influenced by agriculture. In particular, Pacific lampreys are usually eliminated from streams that are heavily polluted (Gunckel et al. 2006), such as the lower San Joaquin River.

Urbanization. The broad range of Pacific lampreys includes many areas that are now heavily urbanized. Typically, they are rare or absent in these areas, such as most of southern California, although the exact causes are poorly documented. Presumably, the disappearance of lampreys from urban areas has multiple causes related to habitat alteration (water diversion, channelization, concrete channels, etc.) and to pollution such as stormwater runoff and pesticides, although most urban streams are also dammed and diverted.

Instream mining. Gravel mining has been common in the lower reaches of streams favored by lampreys. While impacts have not been documented, gravel mining may disrupt spawning and displace ammocoetes, particularly through mobilization of fine sediment deposits, which are key rearing habitats, as well as removal of preferred substrates for spawning.

Mining. Hardrock mines are present in many lamprey watersheds but their effects (e.g., acid mine drainage) are largely unknown.

Logging. Coastal rivers, such as the Eel River (named for its lampreys), that have been heavily altered by logging and road building are generally less suitable for lampreys than they were historically because of excessive deposition of gravels in backwater areas needed for rearing, alteration of the annual hydrograph, increased sediment loads, increased solar input and corresponding higher water temperatures, or similar changes in habitats.

Estuary alteration. Estuaries have been significantly altered throughout the range of Pacific lamprey. Estuaries may be as important to lamprey as they are to anadromous salmonids, which rely on them for foraging, rearing and holding habitat, as well as transitional habitats that enable osmoregulation and migration orientation. Lamprey ammocoetes were commonly observed in the soft sediments of the Smith River estuary from 1997 to 2001 (R. Quiñones, pers. observations), an estuary that retains many of its natural characteristics because stream flows have not been altered significantly.

Harvest. Lampreys have long supported subsistence fisheries by coastal tribes, especially in the Klamath River, because their early arrival and high fat content made them highly desirable as food. This fishery continues today, although only small numbers are likely taken (Lewis 2009). Of greater concern is the fishery for spawning lampreys that has developed because of their value as bait for sturgeon. Adult lampreys are extremely vulnerable to capture when on their nests and the fishery is largely unregulated and unmonitored. Ammocoetes are also collected for bait on occasion and are called “worms” by striped bass fishermen.

Alien species. Alien species increasingly co-occur with Pacific lampreys but their impacts on lamprey populations are not well understood; however, localized impact may be considerable. Ammocoetes are documented prey of many predatory fishes. In the Eel River, for example, introduced Sacramento pikeminnows were observed feeding heavily on ammocoetes (P. Moyle, personal observations; Brown and Moyle 1997).

	Rating	Explanation
Major dams	Medium	Major dams present on many Pacific lamprey rivers; dams prevent access to spawning habitats and reduce habitat suitability downstream
Agriculture	Medium	Minor influence on lower Klamath and Eel rivers, major impact in Central Valley
Grazing	Low	Pervasive across Pacific lamprey range but probably minor impacts on large river habitats
Rural residential	Low	Can cause localized habitat loss or degradation
Urbanization	Medium	Large urban areas in southern part of range and Central Valley contribute to habitat degradation, stream channelization, input of pollutants and flashy flows associated with hardscapes
Instream mining	Medium	Gravel mining and gold dredging alter rearing habitats and increases mortality of ammocoetes; effects are highly localized
Mining	Low	Mines common in lamprey watersheds; direct effects unknown
Transportation	Medium	Roads line many rivers and streams, simplifying habitats (channelization, bank stabilization, etc.); sources of sediments and pollutants that may affect spawning and survivorship; culverts and other structures create barriers to migration
Logging	Medium	Major source of sediments via roads; greater historic impacts in most Pacific lamprey habitats than today
Fire	Low	Fire severity is increasing due to landscape changes, along with climate change, potentially increasing siltation and changing water quality
Estuary alteration	Medium	Most estuaries in California are highly altered through diking, draining, channelization and dredging
Recreation	Low	Possible disturbance to spawning and rearing
Harvest	Medium	Potential reduction of adult abundance in some streams, rivers and Delta; impacts not well understood
Hatcheries	n/a	
Alien species	Medium	Predation on ammocoetes may limit abundance in some areas

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

Effects of Climate Change: Predicted increases in river temperatures (to > 22 °C) brought about by climate change may increase incidence of deformities and mortalities of incubating eggs and of ammocoetes (Meeuwig et al. 2005). Summer water temperatures already frequently exceed 20°C in many California streams and temperatures are expected to increase under all climate change scenarios (Hayhoe et al. 2004, Cayan et al. 2008). Increases in summer temperatures may affect growth and metabolic costs of juveniles and stress adult Pacific lamprey holding in rivers throughout the summer (Clemens et al. 2009).

Climate change is also predicted to change the flow regime in rivers. For instance, flows in the Klamath River may peak earlier in the spring and continue tapering through the summer before pulsing again later in the fall (Quiñones 2011). Resulting changes in river flows and temperatures may alter the timing of adults and juveniles entering and exiting California rivers. Large flow events can disrupt incubation and rearing habitat due to increased bed mobility (Fahey 2006). However, flow-related impacts may be attenuated by dam operations in some systems or exacerbated by competing demands for water (e.g., agricultural irrigation) during low flow periods in others. The Pacific lamprey's migratory plasticity may facilitate movement into watersheds with more favorable habitat conditions (provided passage exists) so their populations may not be as threatened by climate change as are species with high migratory fidelity (e.g., salmon and steelhead). Nonetheless, the geographic range of Pacific lamprey may shift northward as temperatures and flows because unsuitable in more southern streams. Populations south of Monterey Bay may disappear, following those in southern California. Shifts upward in elevation toward remaining cold water refuges may be impeded by barriers or difficulties associated with passage through dams, as well as increased distance of migration and lack of suitable habitats in high-gradient reaches. Because of these concerns, Moyle et al. (2013) rated Pacific lamprey as "highly vulnerable" to extinction in California due to climate change impacts in the next 100 years.

Status Determination Score = 3.3 - Moderate Concern (See Methods section, Table 2). Pacific lampreys apparently still occupy much of their native range in California, but evidence suggests that large declines may have occurred in the past 50 years. Pacific lampreys no longer have access to numerous upstream habitats blocked by large dams or other impassable structures and they are no longer present in streams at the southern end of their range. The large runs that once occurred in coastal streams such as the Eel and Klamath have dwindled to a fraction of their former size.

Metric	Score	Justification
Area occupied	4	Present throughout much of their historic range; blocked from large portions of watersheds by dams
Estimated adult abundance	2	Population estimates lacking; large river populations presumably are >500 in most years
Intervention dependence	4	Improved flow management and habitat restoration efforts needed to prevent further declines, especially for more southern populations
Tolerance	3	Local populations are vulnerable to stochastic events and degraded habitats
Genetic risk	5	Gene flow apparently largely unimpaired between populations throughout range
Climate change	2	Limited spawning and rearing habitats suggests vulnerability to increased temperatures and altered flow regimes, especially in southern end of range
Anthropogenic threats	3	Nine factors rated as ‘medium’ (Table 1)
Average	3.3	23/7
Certainty (1-4)	2	Population size and environmental tolerances poorly understood

Table 2. Metrics for determining the status of Pacific lamprey, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: Pacific lamprey conservation and management is currently hindered by lack of information on their distribution, abundance, and life history. However, given their apparent decline throughout much of the historical range in California, additional conservation measures can and should be pursued in order to afford greater protection (Streif 2009). Management recommendations include the following:

1. Establish a Pacific lamprey research and monitoring program, with three primary goals: 1) determine the status of lampreys statewide and identify key conservation opportunities; 2) improve understanding of life history attributes and habitat requirements in California streams in order to enable a limiting factors analysis; and 3) determine if different genetic stocks of lampreys exist in California. Ideally, such a program would provide critical information about status, population dynamics and life history variability of the species throughout its range in order to inform management and conservation measures. Beneficial research should include studies to: (1) identify the presence or absence of multiple runs in large rivers; (2) document landlocked populations in large river systems; and (3) evaluate metapopulation dynamics to determine if a few large main-river populations sustain smaller tributary populations (source-sink dynamics).
2. Establish a lamprey data center, as part of the proposed research and monitoring program, which would standardize, collect and integrate *all* lamprey information collected in California. The many rotary screw traps used to monitor outmigration of juvenile salmonids, in particular, are a largely untapped source of

data. Many trap operators record captures of lamprey ‘smolts’ and ammocoetes. The lampreys are rarely identified to species, but most are likely Pacific lampreys.

3. Determine if conservation efforts for salmonids also benefit Pacific lampreys, especially in regulated streams. The following questions remain largely unanswered and should be the focus of additional research:
    a. Do passage structures constructed for salmonids also allow passage for lampreys?
    b. Do habitat restoration programs focused on salmonids also create backwater habitat for lampreys?
    c. Are populations of Pacific lamprey tied to those of salmon and steelhead (e.g., predator-prey interactions, migratory timing)?
4. Require that all instream alteration or diversion projects address lamprey habitat and life history requirements and provide appropriate mitigation measures. Strief (2009) documented that a single stream dewatering event, even of short duration, can inhibit up to seven years of lamprey production by eliminating all age classes of ammocoetes.
5. Address potential threats in order to reduce or reverse population declines. In many respects, addressing threats to lamprey requires restoring flows and habitats in most of California’s rivers. Possible actions include:
    a. Subsistence and bait fisheries for lamprey should be monitored to determine their effects on population structure and abundance.
    b. Where feasible, large dams should be retrofitted with fishways that are passable to all migratory stages of lamprey.
    c. Estuary and river restoration projects should consider establishing natural flow regimes, minimum base flows, and sediment budgets (to reestablish deposits of soft sediment in low velocity habitats and improve spawning gravel quality).

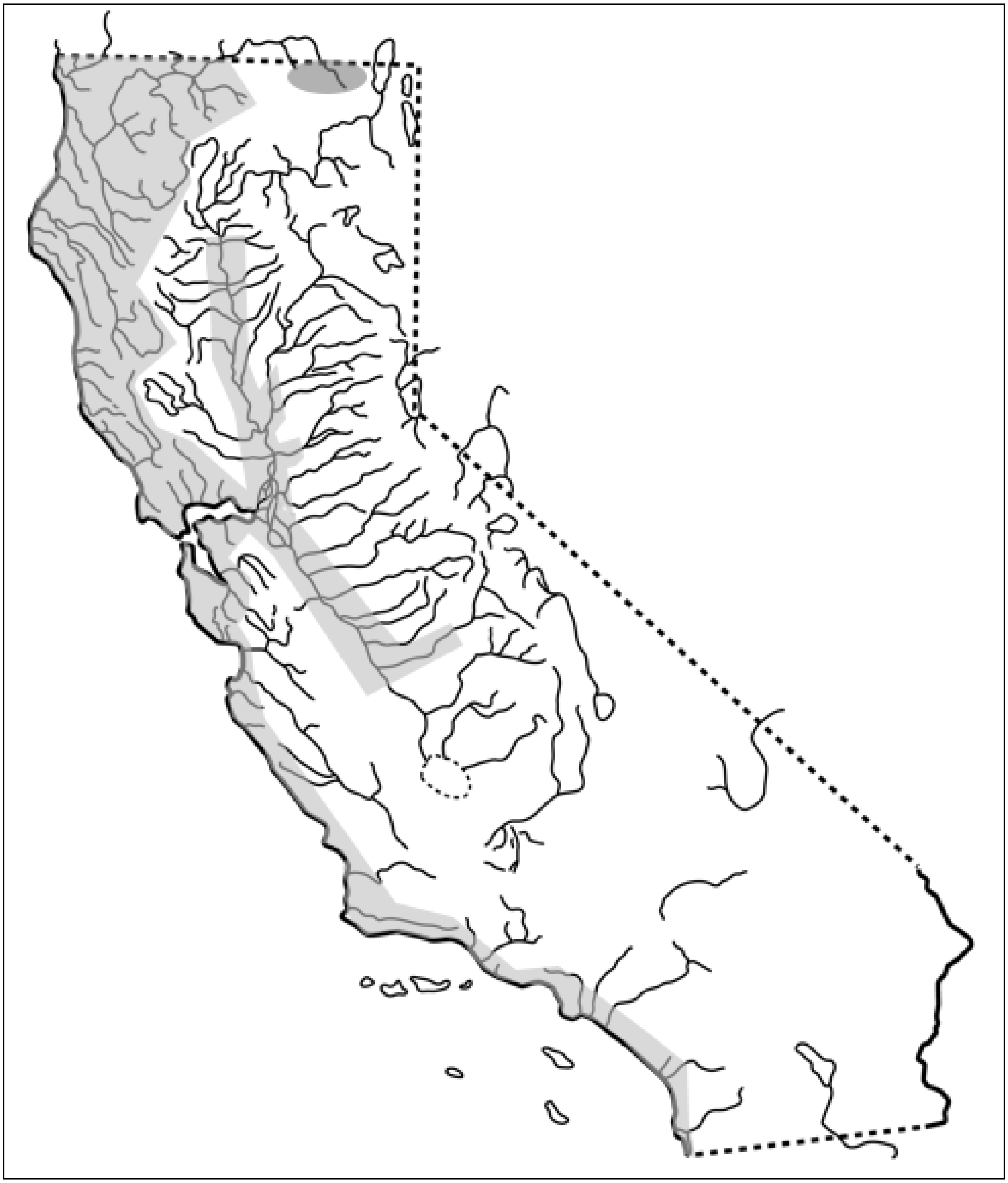


Figure 1. Generalized distribution of Pacific lamprey, *Entosphenus tridentatus***,** in California. Current distribution is reduced and fragmented, although recolonization of depleted areas may occur periodically.

GOOSE LAKE LAMPREY
Entosphenus sp.

Status: High Concern. The Goose Lake lamprey does not face immediate extinction risk but its restricted distribution makes it vulnerable to land and water use practices, climate change, and other factors which could compromise its status.

Description: This predatory lamprey is similar to the widespread Pacific lamprey, *E. tridentatus*, except that it is much smaller (adult TL 19-25 cm vs. 30-40 cm for Pacific lamprey) and not as dark in color. Both forms can be recognized by the sharp, horny plates in the sucking disc, the most distinctive being the crescent-shaped supraoral plate, which has three distinct cusps. The middle cusp is smaller than the two lateral cusps. Adult Goose Lake lampreys are shiny bronze in color. Ammocoetes can be distinguished from those of the sympatric Pit-Klamath brook lamprey (*E. lethophaga*) by the larger number of myomere segments (64-70 between the last gill opening and anus).

Taxonomic Relationships: The Goose Lake lamprey was first recognized as distinct by Carl Hubbs (1925) but he did not formally describe it as a species. It is presumably derived from Pacific lamprey or its derivatives from the Klamath River drainage. However, Goose Lake and the Pit River drainage, to which it connects, have been separated from the Klamath drainage since the early Pleistocene (1-3 million years). Some insights into evolution of the Goose Lake lamprey are provided by Lang et al. (2009); they used mitochondrial DNA (cytochrome B) to examine relationships among all lamprey species. While Goose Lake lamprey *per se* were not used in the analysis, the non-predatory Pit-Klamath brook lamprey was included, which is most likely the closest relative of the Goose Lake lamprey. Lang et al. (2009) found that it was part of the *Entosphenus* clade, which includes the various non-anadromous lampreys from the upper Klamath River as well as the Pacific lamprey. The relationship of Pit-Klamath brook lamprey to others within the clade is largely unresolved. Genetic differences, at least those based on mitochondrial DNA, indicate that the genome of lampreys is very conservative so that population structure, even in the widespread Pacific lamprey, has not been detected (Goodman et al. 2008). Regardless, the lampreys of the Goose Lake basin are likely a distinct evolutionary lineage, perhaps representing more than one.

Within the basin, there are two basic hypotheses about the relationship between the predatory Goose Lake lamprey and the non-predatory Pit-Klamath brook lamprey: (1) they represent different life history forms of the same species, or (2) they are separate species. These same hypotheses, often unresolved, exist for the pairs of predatory and non-predatory lampreys found throughout the world (Docker 2009). It is generally assumed that the non-predatory forms evolved from predatory forms. In the case of the Goose Lake basin, the issue is complicated by the fact the Pit-Klamath brook lamprey has been described as occurring in both the Goose Lake and Klamath River basins, despite their long separation (Hubbs 1971).

Nevertheless, because of its distinctive morphology and ecology and long isolation from other populations, it is most likely that the Goose Lake lamprey is a distinct species, separate from the Pit-Klamath brook lamprey (Kostow 2002) and from other lamprey species in the Klamath River (Docker et al. 1999). As a separate species, the Goose Lake lamprey may include both predatory and non-predatory life histories, assuming that the predatory form is only expressed when migrations to Goose Lake are feasible (Kostow 2002). Limited data on adult distribution, presented in Scheerer et al. (2010), suggest that the two lamprey species are at least partly segregated by elevation, with the Goose Lake lamprey found in stream reaches closest to the lake.

Life History: The life history of this taxon is largely unknown, but presumably the adults live for a year or two in Goose Lake, preying on Goose Lake tui chubs, suckers, and redband trout. In 1989, adult lampreys were observed attached to gill-netted tui chubs and lamprey wounds were common in larger chubs (P. Moyle and R. White, unpublished observations). They migrate up suitable tributary streams in spring for spawning, with a peak in May (Kostow 2002). They require clean gravels for spawning, combined with soft-bottomed habitat downstream of the spawning areas for rearing of ammocoetes. Thus, spawning areas may be as much as 20-30 km upstream from the lake. Ammocoetes probably spend 4-6 years in tributary streams before metamorphosing into adults (at about 8-13 cm TL) in the fall and moving into the lake in spring (Kostow 2002). During periods of drought, when access to the lake is not available, adult lampreys will feed on stream fishes although survival appears to be low (Kostow 2002).

Habitat Requirements: Adults live in shallow, alkaline Goose Lake where they prey on larger fishes. Like other lampreys, Goose Lake lampreys require gravel riffles in streams for spawning and ammocoetes require muddy backwater habitats downstream of spawning areas. Kostow (2002) characterizes the habitat of ammocoetes as "fine silt lenses along low gradient stream meanders, most often through meadows…(p. 18)." However, the habitat requirements of Goose Lake lamprey have not been well studied or distinguished from those required by Pit-Klamath brook lamprey. For further description of stream and lake habitats, see the Goose Lake redband trout account in this report.

Distribution: The Goose Lake lamprey is endemic to Goose Lake and its tributaries in Oregon and northeastern California. However, a comprehensive assessment of the distribution and habitat utilization of California tributary streams by lampreys has not been performed. Within California, they have been collected only from Lassen and Willow creeks, Modoc County, (G. Sato, BLM, pers. comm. 1994), both above and below potential migration barriers (Hendricks 1995). Ammocoetes were found to be common in Cold Creek, a tributary to Lassen Creek. No ammocoetes were found in Davis, Pine or Willow creeks. It is likely that dams and diversions now restrict distribution of lampreys by blocking adult migration and by drying up suitable habitats downstream. In Lake County, Oregon, they are common in Thomas Creek and a population apparently exists in Cottonwood Reservoir, on Cottonwood Creek (Oregon Dept. of Fish and Wildlife, unpubl. data, 1995). Scheerer et al. (2010) found lamprey ammocoetes to be widely distributed and often abundant in Oregon streams, but did not distinguish species.

Trends in Abundance: There are no trend data for Goose Lake lamprey but their populations likely decline during extended periods of drought and then increase rapidly when wet periods return and the lake fills again. Thus, Goose Lake lampreys were fairly common in Goose Lake, where they were readily collected from large tui chubs caught in gillnets, until the lake dried up in the summer of 1992 (R. White, USFWS, pers. comm. 1995). The Goose Lake lamprey has the potential of becoming extirpated, especially in California, if the lake and lower tributaries are dry for several years in a row. However, adults may survive by preying on stream fishes and the ammocoetes may persist for 3-4 years if there are adequate flows in the habitats they occupy. The Cottonwood Reservoir population is of unknown size but the reservoir may serve as a refuge, provided a minimum pool is maintained throughout extended drought periods. In Lassen and Willow creeks, ammocoetes were common at densities of 11-50 individuals per 150 ft of stream (Hendricks 1994). Abundance of spawners is not known but 50-100 spawners in most years in each stream may be a reasonable estimate, based on accessible habitat, number of ammocoetes, and abundance in the lake. The importance of Lassen and Willow creeks to persistence of the entire population in the Goose Lake basin is unknown but it is assumed that most spawning and rearing habitat occurs in Oregon streams (Scheerer et al. 2010).

Nature and Degree of Threats: The principal threat to the Goose Lake lamprey is desiccation of its habitats, Goose Lake and its tributaries, which is exacerbated by human activities, including diversions for agriculture and grazing. The combination of severe, extended drought, along with human demands for scarce water resources in the basin, may have resulted in accelerated desiccation of the lake during the 1986-1992 drought and, again, in 2010, resulting in a dry lakebed.

Agriculture. Farming occurs primarily on lands close to the lake, often adjacent to tributary streams, with the result that some streams reaches are channelized, down-cut, and silted from erosion. The diversion of water from streams for agriculture and other uses may reduce or completely dewater habitats required by ammocoetes and adults for survival during droughts, as well as accelerating desiccation of the lake itself. Diversions and dams may prevent adults from reaching spawning areas in tributary streams, although small reservoirs may also serve as refuges for adults. The loss of suitable habitat for ammocoetes is likely to be particularly severe in the lower reaches of streams near agricultural areas.

Grazing. Livestock grazing is one of the greater land uses in the Goose Lake basin. In-stream and riparian habitats can be degraded or eliminated through stream erosion and bank destabilization caused by livestock grazing in riparian areas, especially through the removal of woody riparian plants. In the past, many areas in the California portion of the Goose Lake basin were degraded by grazing, although restoration actions, especially on Lassen Creek, have reversed some of these impacts. While improved management of most grazed lands has reduced the threat of grazing in the short term, as the climate becomes warmer and more variable (see Effects of Climate Change section), there is considerable potential for negative impacts from grazing to increase without expanding the use of riparian protection measures such as exclusionary fencing.

Fire. The Goose Lake basin is semi-desert and wildfires are common. Impacts of fires on lampreys (and other fishes) are not known but are likely to be minimal, unless a major fire causes direct mortality through increased stream temperatures or indirect mortality associated with loss of canopy cover (in-stream shading), accelerated erosion, or landslides in upstream areas.

Alien species. Scheerer et al. (2010) found six species of alien fishes in Oregon streams tributary to Goose Lake, mostly in low elevation areas or areas associated with reservoirs and other altered habitats. Alien species appear to be scarce in Lassen and Willow creeks although predatory brown trout are common in Pine and Davis creeks. Illegal introductions of possible predators (catfish, bass) remain a concern.

	Rating	Explanation
Major dams	Low	Reservoirs may act as refuge during drought; diversion dams may block spawning and in-stream movement
Agriculture	Medium	Alfalfa fields along lower reaches of streams may negatively affect water quality
Grazing	Medium	Grazing is pervasive and is likely to have strong interactions during periods of reduced flow
Rural residential	Low	Few residences
Urbanization	n/a	
Instream mining	n/a	
Mining	Low	Uranium mines exist in the area but their impacts are unknown
Transportation	Medium	Roads and culverts can block migration; potential increased siltation
Logging	Low	Widespread in watersheds but impacts reduced from the past
Fire	Low	A continuous threat in this part of the state; impacts to lampreys unknown
Estuary alteration	n/a	
Recreation	n/a	
Harvest	n/a	
Hatcheries	n/a	
Alien species	Medium	Aliens present in certain portions of the basin; impacts to lampreys are unknown

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

Effects of Climate Change: The Goose Lake basin is located in an arid portion of California and this area has, in the recent past, suffered extended periods of drought. Climate change is likely to decrease summer stream flows in key streams, increasing competition for water and riparian habitats between humans (livestock, agriculture) and fishes. Goose Lake may dry more frequently and for longer periods of time due to increased frequency of drought. Increased stream temperatures of 2-4°C may affect lampreys, although similar species can tolerate fairly warm water. These conditions may also favor alien competitors and predators (Scheerer et al. 2010). An increase in fire frequency or intensity in this dry landscape may decrease riparian shading, add sediment, or otherwise make streams less suitable for lampreys and other fishes. Moyle et al. (2013) consider the Goose Lake lamprey to be "critically vulnerable" to extinction as the result of climate change because predicted reduction in snow pack will result in decreased flow in tributary streams with corresponding reduced lake levels.

Status Determination Score = 2.9 – High Concern (see Methods section Table 2). Goose Lake lamprey do not face immediate extinction risk but their California populations are small and isolated, making them vulnerable to climate change and other factors which could compromise their status. The American Fisheries Society regards Goose Lake lamprey as a threatened species, with declining populations (Jelks et al. 2008), while NatureServe ranks it as Critically Imperiled (T1) and the Forest Service regards it as Sensitive.

Metric	Score	Justification
Area occupied	2	Only known from Willow, Lassen, and Cold creeks in CA
Estimated adult abundance	1	California abundance not known but numbers of adult spawners is likely small in most years and zero in dry years
Intervention dependence	4	Persistence requires habitat improvement and maintenance
Tolerance	4	Not known but presumably fairly broad
Genetic risk	3	Potential for impacts from small population size and isolation
Climate change	2	Stream habitat likely to be reduced as is frequency of lake drying
Anthropogenic threats	4	See Table 1
Average	2.9	20/7
Certainty (1-4)	2	Very little is published on this lamprey

Table 2. Metrics for determining the status of Goose Lake lamprey in California, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: The Goose Lake lamprey and other Goose Lake fishes were little studied and largely unmanaged until 1991, which contributed to their increased likelihood of extinction. The Goose Lake Fishes Working Group was formed in 1991, with representatives from private landowners, federal and state agencies, and environmental groups to explore management measures for all fishes native to Goose Lake and its tributaries (Sato 1992a, see Goose Lake redband trout account in this report). As a result of this program, stream restoration projects have improved reaches of Lassen Creek, presumably providing better habitat for lamprey spawning and rearing. The biology and status of the population in Cottonwood Reservoir needs to be investigated, as well as the possibility of establishing similar refuge populations of the species elsewhere. An investigation of this unusual lamprey's life history and habitat requirements should be conducted in order to develop management and conservation strategies in both California and Oregon. In particular, stream flow models need to be developed under various climate scenarios in order to determine predicted base flows. At a minimum, flows in key tributary streams should provide adequate rearing and holding habitat during extended drought (>5 years) in order for the species to persist and recolonize the lake during wetter periods. Enhancing spawning access, as well as restoring rearing and holding habitats, in streams in California and Oregon (especially in Lassen, Willow, and Thomas creeks) would benefit all native Goose Lake fishes. In addition, studies should be developed to determine both the evolutionary and ecological relationships between the Goose Lake lamprey and the Pit-Klamath brook lamprey. See the Goose Lake sucker account in this report for further discussion of management actions that would encompass the entire Goose Lake basin and likely benefit Goose Lake lamprey.

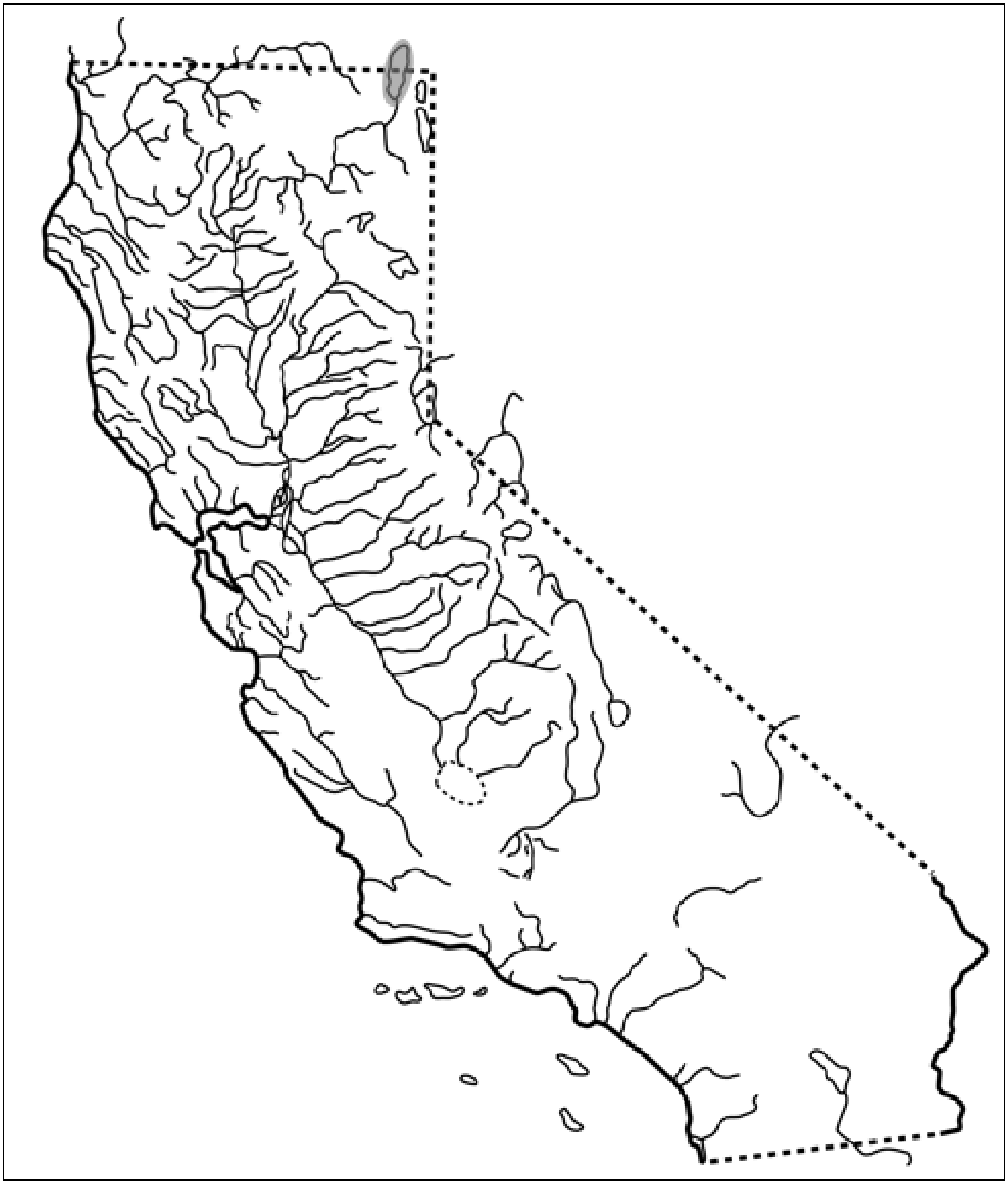


Figure 1. Distribution of Goose Lake lamprey, *Entosphenus sp.*, in Goose Lake, California and Oregon. The extent to which they are distributed upstream in the Thomas Creek drainage in Oregon is unknown.

NORTHERN CALIFORNIA BROOK LAMPREY
***Entosphenus folletti* (Valdykov and Kott)**

Status: High Concern. The northern California brook lamprey has a very limited known distribution and aquatic habitats within their range are heavily altered by agriculture and grazing. Their actual distribution and abundance is unknown.

Description: This lamprey is a non-predatory species that has an adult size of 17-23 cm in total length (Vladykov and Kott 1976b, Renaud 2011). Adult disc length is 6.6–7.8% of total length and the trunk myomere count is 61-65. The following description of dentition is from Renaud (2011, p. 27): "supraoral lamina, 3 unicuspid teeth, the median one smaller than the lateral ones; infraoral lamina, 5 unicuspid teeth; 4 endolaterals on each side; endolateral formula, typically 2–3–3–2, the fourth endolateral can also be unicuspid; 1–2 rows of anterials; first row of anterials, 2 unicuspid teeth; exolaterals absent; 1 row of posterials with 13–18 teeth, of which 0–4 are bicuspid and the rest unicuspid (some of these teeth may be embedded in the oral mucosa); transverse lingual lamina, 14-20 unicuspid teeth, the median one slightly enlarged; longitudinal lingual laminae teeth are too poorly developed to be counted. Velar tentacles, 8–9, with tubercles. The median tentacle is about the same size as the lateral ones immediately next to it…Oral papillae, 13." Ammocoetes are described in Renaud (2011).

The northern California brook lamprey is similar to the Pit-Klamath brook lamprey, with which it co-occurs, but is somewhat larger (most are >19 cm TL), has a larger oral disk (<6% of TL vs >6% of TL), and has elongate velar tentacles without tubercles. There are also minor differences in various tooth counts (Renaud 2011). According to Vladykov and Kott (1976b, p. 984): "The body and fins of *E. folletti* are more darkly pigmented than those of *E. lethophagus*. The entire caudal fin of the former is strongly pigmented, except for a narrow unpigmented margin, and it has a dark second dorsal fin. In the latter the caudal fin has broader unpigmented margin and its second dorsal is less pigmented." The region around the vent is darkly pigmented in *E. folletti* but pale in *E. lethophagus*, a potential distinguishing characteristic in the field.

Taxonomic Relationships: Non-predatory lampreys in the Klamath and upper Pit River systems are derived from Pacific lamprey (Renaud 2011). The northern California brook lamprey was described by Vladykov and Kott (1976b) based on specimens from Willow and Boles creeks, tributaries to the Lost River, Modoc County. However, the species was not recognized by the American Fisheries Society (AFS, Robins et al. 1991) because of unpublished doubts of its validity. Lang et al. (2009) listed it as a recognized species, as did Beamish (2010). The AFS then recognized it as a species based on Renaud's (2011) analysis of lamprey species worldwide (Page et al. 2013). Beamish (2010), using gill pore papillae as a diagnostic character, suggests that *E. folletti*, as currently recognized, may represent more than one species and included in his analysis both specimens from the Lost River and from Fall Creek above Copco Reservoir in California. While evidence increasingly supports the diversity of lamprey species in the upper Klamath and Pit River basins, including northern California brook lamprey, a thorough analysis is needed using additional specimens and additional genetic and morphological studies. Further studies are almost certain to find *E. folletti* in Oregon, given its presence in two distantly separated areas in California, so the common name "northern California brook lamprey" may not be appropriate for the species. Shapovalov et al. (1981) named it the Modoc brook lamprey, a name which reflects its likely distribution as being coincident with the Modoc Plateau region in California and Oregon, as well as with the territory of the Modoc people.

Life History: Nothing is known about the life history of this lamprey but it is presumably similar to other brook lampreys in the genus *Entosphenus.*

Habitat Requirements: Little specific information is available on its habitats, but the northern California brook lamprey is known only from a few, small, cool tributary streams that have areas with fine substrates and beds of aquatic plants.

Distribution: The northern California brook lamprey is known from only Willow and Boles creeks above Clear Lake Reservoir and from Fall Creek, a tributary to Copco Reservoir. It is almost certainly found in similar habitats in Oregon, as well as in the Lost and Klamath river basins.

Trends in Abundance: Abundance and population trend information are lacking. Their populations do not seem to be in danger of extinction at this time but face multiple threats as discussed below.

Nature and Degree of Threats: The northern California brook lamprey faces loss of suitable habitat via multiple factors affecting streams in this arid region, similar to those facing the Pit-Klamath brook lamprey.

Major dams. The only populations known are above large reservoirs, which suggests that they are isolated from other populations by dams. Dams and diversions on the upper Klamath and Lost River systems also alter downstream flows and habitats.

Agriculture. Water demands for irrigated agriculture and livestock are high in this region, leading to decreased stream flows. Flood-irrigated pastures introduce nutrients and pollutants from return waters into streams and raise water temperatures.

Grazing. Extensive grazing occurs throughout the known range of northern California brook lamprey. Grazing can degrade aquatic habitats through stream bank trampling, elimination of riparian vegetation, and pollutant inputs from animal wastes.

Alien species. Many alien fish species inhabit the Klamath and Lost river basins (Close et al. 2010). Species that can prey on lamprey include largemouth bass, brown bullhead, channel catfish, brook trout, brown trout, black crappie, and yellow perch (Close et al. 2010).

	Rating	Explanation
Major dams	High	Dams isolate populations and alter downstream habitats
Agriculture	Medium	Agriculture pervasive throughout range
Grazing	Medium	Grazing pervasive throughout range
Rural residential	n/a	
Urbanization	n/a	
Instream mining	n/a	
Mining	n/a	
Transportation	Low	Rural roads affect stream habitats
Logging	Low	Logging occurs in forested lands; impacts unknown
Fire	Low	Wildfires occur throughout range; impacts unknown
Estuary alteration	n/a	
Recreation	n/a	
Harvest	n/a	
Hatcheries	n/a	
Alien species	Low	Alien species uncommon in known stream habitats but are a potential threat

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

Effects of Climate Change: Climate change is expected to increase the frequency of both drought and floods in streams. Because ammocoetes likely rear for several years in soft substrates, large flooding events may disrupt rearing habitats (Fahey 2006) and displace ammocoetes. On the contrary, scouring events may clean sediments from gravels that would otherwise degrade spawning habitats (Stuart 2006 in Fahey 2006). It is likely that the northern California brook lamprey can tolerate, to some extent, shifts toward warmer water temperatures, which are expected to increase due to climate change. Moyle et al. (2013) did not rate climate change vulnerability for this species, but vulnerability should be similar to that of the Pit-Klamath brook lamprey.

Status Determination Score = 2.4 – High Concern (see Methods section, Table 2). Northern California brook lamprey apparently have limited distribution in small streams subject to degradation. However, their actual abundance and distribution is unknown.

Metric	Score	Justification
Area occupied	2	Known range limited to Lost River and parts of upper Klamath
Estimated adult abundance	2	Numbers unknown but likely small
Intervention dependence	4	Long-term management of grazing practices as well as alien species may be warranted
Tolerance	3	Not known but occurs in degraded streams
Genetic risk	2	Known populations isolated by dams
Climate change	2	Some habitats may dry more extensively or for longer periods; ammocoetes may be displaced by unusually high flows
Anthropogenic threats	2	See Table 1
Average	2.4	17/7
Certainty (1-4)	1	Species is largely unstudied

Table 2. Metrics for determining the status of Northern California brook lamprey in California, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: Habitat degradation, grazing practices and isolation by reservoirs pose the greatest threats to this brook lamprey, effects likely to be exacerbated by increasing temperatures and more frequent flood events predicted by climate change models. Watershed management strategies exist (e.g. Klamath Basin Restoration Agreement) that address these and other factors that may limit fish populations in the upper Klamath basins. Beyond implementation of these strategies, basic life history studies and population monitoring should occur in order to better understand the status of this species. The following questions should be addressed as part of a status evaluation:

What is the current distribution and abundance in California and Oregon?
Where are most important spawning and rearing grounds located in California?
What are the optimal and preferred environmental tolerances and habitat conditions for each life history stage?

Figure 1. Known distribution of northern California brook lamprey, *Entosphenus folletti*, in California.

KLAMATH RIVER LAMPREY
Entosphenus similis (Vladykov and Kott)

Status: Moderate Concern. Very little is known about this species; thus, the conservative course of action is to consider its numbers to be in decline until new information becomes available to indicate otherwise. However, Klamath River lamprey do not appear to be at immediate risk of extinction.

Description: The Klamath River lamprey is a small (14-27 cm TL, mean 21 cm), predatory lamprey that can be identified by strong, sharply hooked cusps on their oral plates. Three strong cusps on the supraoral plate ('tongue') are easily noticeable. The anterior field above the mouth has 10-15 teeth, 4 inner lateral plates on each side, resulting in the typical cusp formula of 2-3-3-2, 20- 29 cusps in line on the transverse lingual lamina (tongue plate), and 7-9 velar tentacles. The trunk usually has 60-63 myomeres (range of 58-65). The disc length is about 9 percent of the total body length, and is at least as wide as the head. The horizontal eye diameter is about 2 percent of the total body length. Although similar to Pacific lampreys, Klamath River lampreys tend to be more heavily pigmented. Ammocoete larvae have not been described.

Taxonomic Relationships: The Klamath River lamprey was described by Vladykov and Kott (1979), from specimens caught in the Klamath River, California. Four other lamprey species have also been described from the upper Klamath River basin: dwarf Pacific lamprey (*E. tridentata*), Pit-Klamath brook lamprey *(E. lethophaga)*, Miller Lake lamprey (*E. minimus)* and Modoc brook lamprey *(E. folletti).* The Pit-Klamath brook lamprey is the common nonpredatory lamprey of the upper Klamath and Pit river drainages, while the Miller Lake lamprey is an unusually small predatory form that is confined to the upper basin in Oregon (Lorion et al. 2000). The Modoc brook lamprey was also described by Vladykov and Kott (1976), from specimens collected from Willow Creek (Modoc County), a tributary to Clear Lake Reservoir on the Lost River. Although described as nonpredatory, it was later found to be predatory, providing little reason to separate it from Pacific lamprey (C. Bond, pers. comm. 1995). Consequently, Modoc brook lamprey has not been accepted as a separate species (Nelson et al. 2004). In contrast, the Klamath River lamprey is morphologically and biochemically distinct (Docker et al. 1999, Lorion et al. 2000, Lang et al. 2009).

Life History: No specific life history information is currently available, although Klamath River lamprey appear to be non-migratory and are resident in both rivers and lakes of the Klamath basin. Adults prey on adult coho and Chinook salmon and other large fishes in the basin. Wales (1951) thought that lamprey predation on migratory salmon was a major factor limiting salmon abundance in the Shasta River, because he observed such a high frequency of salmon with lamprey wounds (41%) and because "lampreys are abundant in the Shasta (p. 33)." However, salmon mortalities have not been attributed to lamprey predation in recent spawning ground (carcass) surveys or at weir operations (B. Chesney, CDFW, pers. comm. 2011).

Habitat Requirements: Little is known about the habitat requirements of Klamath River lamprey. Presumably, ammocoete larvae have the same basic requirements as those of Pacific lamprey, living in backwaters with soft substrates. The environmental tolerances of Klamath River lamprey have not been documented but they are likely similar to those of Pacific lamprey. If this is the case, then Klamath River lamprey need cold, clear water (Moyle 2002) for spawning and incubation. They also require a diverse range of habitats to complete their life cycle. Adults typically use spawning gravel to build nests, while ammocoetes burrow in soft sediments for rearing (Kostow 2002). Ammocoetes also need larger substrates as they grow (Stone and Barndt 2005) and algae for food (Kostow 2002) in habitats with slow or moderately slow water velocities (0-10 cm/s; Stone and Barndt 2005).

Distribution: Klamath River lamprey are found throughout the Klamath River basin in mainstem rivers, including the Trinity River in northern California and the Klamath River in southern Oregon (Boyce 2002). Their distribution in the lower Klamath and Trinity basins likely coincides with those of spawning Chinook and coho salmon, their main prey in the lower river, and with large suckers and cyprinids in the upper basin. However, detailed distribution of this species is not known.

Trends in Abundance: As with other upper Klamath basin lampreys, abundance estimates for Klamath River lamprey do not exist. However, they appear to be common throughout their range (S. Reid, pers. comm. 2008).

Nature and Degree of Threats: The declining quality of aquatic habitats throughout much of the Klamath-Trinity drainage, as well as the declining number of salmon (NRC 2004), make it likely that Klamath River lampreys are less abundant than they once were (Table 1). Generally, any factor that reduces abundance of large prey species is likely to also reduce Klamath River lamprey abundance (Moyle 2002).

Dams. Seven major dams are present in the Klamath-Trinity River basin. These dams change the physical and biological characteristics of the streams where they occur (Knighton 1998). In particular, they may limit or inhibit the longitudinal (upstream-downstream and vice-versa) movements of fishes, including both Klamath River lamprey and their prey, thereby limiting access to suitable spawning and rearing habitats. Dams have also degraded the quality of preferred habitat in the main stem Klamath River (Hamilton et al. 2011).

Agriculture. Alfalfa production and pasture in the Shasta and Scott basins may diminish flows, particularly in dry water years (NRC 2004). Diminished flows can reduce suitable habitats in streams, as well as create conditions (e.g., high water temperatures, low dissolved oxygen levels) that increase salmonid mortality, thereby reducing adult Klamath River lamprey prey availability. Diversion of water, warm polluted return water, and similar by-products of agriculture are also presumably limiting lamprey populations.

Grazing. Livestock grazing is pervasive in Klamath River watersheds, with disproportionate effects on smaller tributaries, reducing water and habitat quality (USFWS 1991). Grazing practices in some subbasins (e.g., Shasta River) have altered stream morphology and degraded habitat quality to the detriment of native fishes

(USFWS 1991, Gosnell and Kelly 2010). Grazing can lead to localized increases in water temperature when riparian vegetation is removed, as well as low oxygen concentrations from excess fecal nutrient loading.

	Rating	Explanation
Major dams	Medium	Seven major dams exist in the system and likely disrupt instream movement, gene flow, and opportunities for recolonization
Agriculture	Medium	Major influence on Scott and Shasta rivers by reducing salmon prey abundance (NRC 2004)
Grazing	Medium	Pervasive in Klamath River watersheds with disproportionate effects on smaller tributaries
Rural residential	Low	Widespread rural development throughout range but housing densities very low
Urbanization	n/a	
Instream mining	Low	Legacy effects have likely reduced the amount and quality of suitable habitats
Mining	Low	Impacts are unknown but assumed to be minor
Transportation	Medium	Roads are a source of sediment that may affect spawning and rearing
Logging	Medium	Widespread changes to watersheds; greater impact in past than today
Fire	Low	While wildfires are common throughout the Klamath basin, direct impacts to Klamath River lamprey are likely minimal
Estuary alteration	n/a	
Recreation	n/a	
Harvest	n/a	
Hatcheries	n/a	
Alien species	Low	No known impacts

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

Instream mining. Instream mining may alter larval rearing habitats through scour and deposition and through direct displacement of ammocoetes. When the Scott River and other areas were dredged for gold in the 19th and 20th centuries, large areas of potential habitat were destroyed; when combined with dewatering from diversions (often relicts of mining), past dredging may have had considerable legacy effects upon lamprey populations and their habitats.

Transportation. Roads, both paved and unsurfaced, have been built within the riparian corridor of many Klamath streams (USFWS 1991). Many miles of dirt roads have also been built in most of the Klamath-Trinity watersheds. Road building can decrease the quality of nearby aquatic environments to the extent of altering animal behavior and overall species composition (Trombulak and Frissell 2000). Road building can decrease the amount of canopy cover over streams and potentially increase water temperatures, limit the ability of streams to meander, impair the creation of slow water habitats, and increase sediment and pollutant input from surface run off. Increased fine sediment input into streams can decrease the quality of spawning gravels for adult lamprey and other fishes. However, it is possible that increased sedimentation may provide additional habitat for lamprey larvae.

Logging. The entire Klamath-Trinity basin has been heavily logged with attendant impacts on streams, especially increases in sedimentation from logging roads. Certain logging practices can alter the hydrology of streams (Wright et al. 1990), such that habitats become unsuitable for some fishes (Reeves et al. 1993). As with road building, logging can increase the amount of solar radiation reaching streams, decrease the amount of nutrients entering food webs, impair recruitment of large woody debris (habitat complexity, cover) and increase the amount of fine sediment eroding from hillslopes into streams. However, with current California timber harvesting rules, logging had a much more pronounced impact on stream habitats in the past than it does today (NRC 2004).

Effects of Climate Change: The potential impacts of predicted climate change to Klamath River lamprey are poorly understood because so little is known about their biology, life history, or environmental tolerances. Nevertheless, increased water temperatures (> 22 °C) brought about by climate change may increase incidence of deformities and mortalities of incubating eggs and larvae, as has been observed in Pacific lamprey populations (Meeuwig et al. 2005). Summer water temperatures already frequently exceed 20°C in many streams in the Klamath River basin and temperatures are expected to increase under all climate change scenarios (Hayhoe et al. 2004, Cayan et al. 2008). Increased summer temperatures may affect the growth and metabolic costs of juvenile and adult Klamath River lamprey that hold and rear in rivers throughout the summer. Climate change is also predicted to change the flow regimes in rivers. Klamath River flows may peak earlier in the spring and continue tapering through the summer before pulsing again later in the fall. The resulting changes in river flow and temperature may change the timing of adults and juveniles entering and exiting streams. High flows can disrupt incubation and rearing habitat due to increased bed mobility (Fahey 2006). However, flow alterations associated with climate change may be attenuated by dam operations. Shifts in distribution are expected to be upward in elevation and northward in latitude but may be impeded by passage through dams and culverts, along with increased metabolic costs associated with increased water temperatures. Moyle et al. (2013) rated Klamath river lamprey as “highly vulnerable” to extinction as the result of climate change in the next century, based on the largely speculative evidence presented above.

Status Determination Score = 3.9 - Moderate Concern (See Methods section, Table 2). The Klamath River lamprey does not appear to be at much risk, given its wide distribution within the Klamath and Trinity basins, although it may be negatively affected by climate change in the future (Table 2). The paucity of information available on this species, including present and past abundance and distribution, makes a conservation status determination difficult. Additional information is needed in order to better understand its status.

Metric	Score	Justification
Area occupied	5	Widely distributed in Klamath basin (Moyle 2002)
Estimated adult abundance	4	Unknown, but appears to be common throughout range (S. Reid, pers. comm. 2010)
Intervention dependence	5	Populations appear to be resilient and persistent
Tolerance	3	Environmental tolerances have not been identified, but are presumed similar to other lamprey species in the Klamath River basin
Genetic risk	5	No known genetic risk
Climate change	2	Potentially threatened by changes in hydrology and temperature
Anthropogenic threats	3	Five threats rated as intermediate (Table 1)
Average	3.9	27/7
Certainty (1-4)	1	Population size, distribution, and environmental tolerances largely unknown

Table 2. Metrics for determining the status of Klamath River lamprey, where 1 is a major negative factor contributing to status, 5 is factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: The principal impediment toward improved Klamath River lamprey management and conservation is the lack of empirical data and general knowledge of their abundance, distribution, environmental tolerances, and key aspects of life history. As such, the following management actions are recommended:

1. Establish a Klamath River lamprey research and monitoring program. Program goals should include: 1) a status assessment of all lampreys in the basin; 2) identification of key conservation opportunities; and 3) development of life history and habitat requirement studies, to inform a limiting factors analysis. Additionally, an identification key needs to be developed to distinguish ammocoetes of Klamath basin lamprey species.
2. Establish a lamprey data center, as part of the research and monitoring program, which would collect and integrate *all* lamprey information collected in California. The many rotary screw traps used to monitor outmigration of juvenile salmonids, in particular, are a largely untapped source of data, especially in the Klamath River system. Many trap operators record captures of lamprey ‘smolts’ and ammocoetes. The lampreys are rarely identified to species but most are likely Pacific lampreys in the lower river; however, Klamath River lampreys may also be represented in the catch.

3. Determine if conservation efforts for salmon and steelhead also benefit Klamath River lampreys, both in mainstem rivers and tributaries such as the Shasta and Scott rivers. Habitat restoration programs intended to benefit salmonids should be evaluated for their potential to create backwater habitat for lampreys. Studies should be performed to determine if populations of Klamath River lamprey are tied to those of salmon and steelhead (predator/prey relationships).

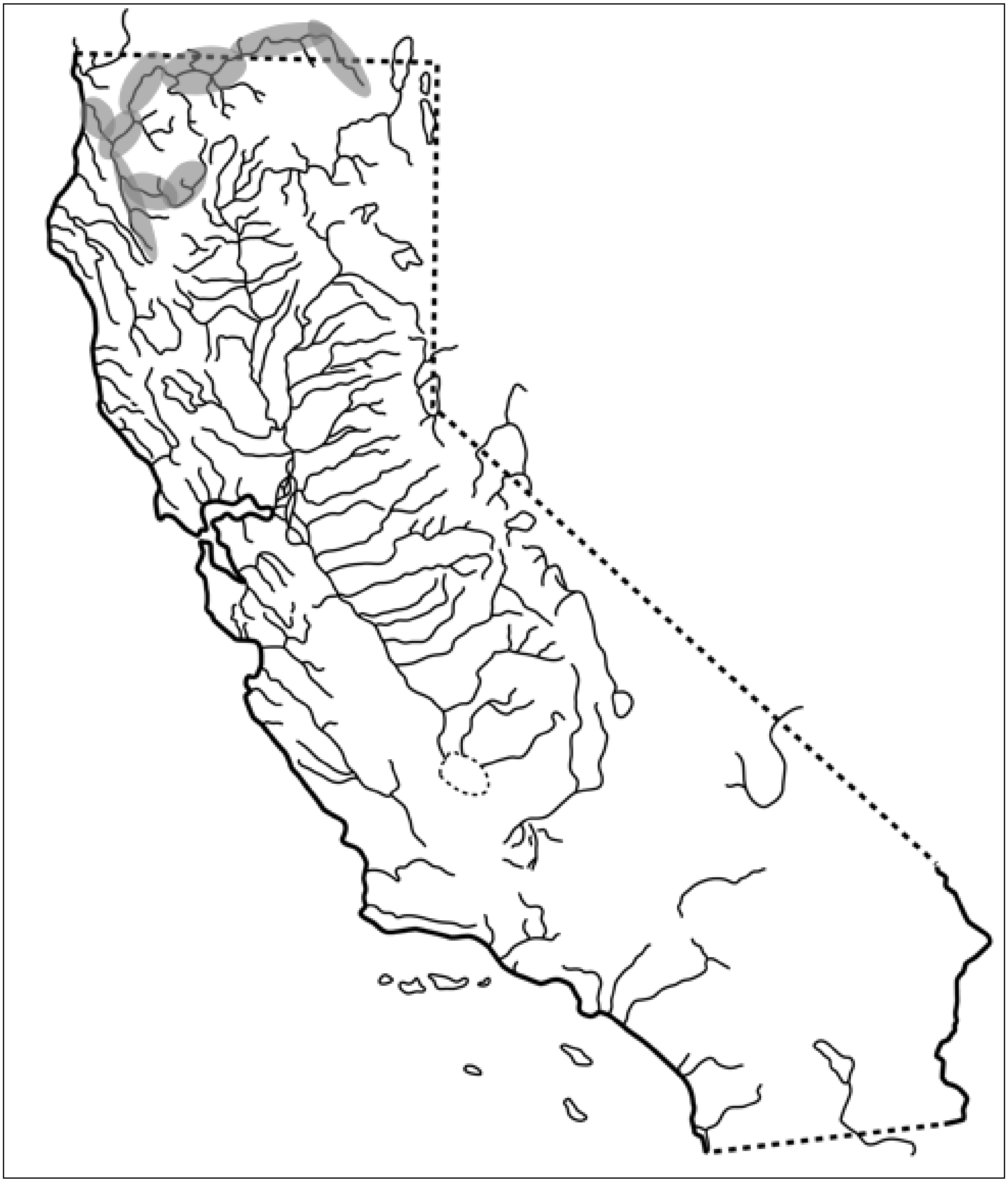


Figure 1. Distribution of Klamath River lamprey, *Entosphenus similis*, in the Klamath and Trinity rivers in California.

WESTERN RIVER LAMPREY
Lampetra ayresi

Status: Moderate Concern. Very little is known about the western river lamprey in California but it is uncommon in the state and potentially in decline.

Description: The western river lamprey is a small, predatory, species. Spawning adults reach a maximum size of about 17-18 cm TL. The oral disc is at least as wide as the head. The 'teeth' (horny plates) in the oral disc are conspicuous and pointed; however, they can be blunt in spawning individuals. The middle cusp of the transverse lingual lamina has three large lateral (circumoral) plates on each side; the outer two have two distinct cusps, while the middle one has three. The supraoral plate has only two cusps that often appear as separate teeth, while the infraoral plate has 7-10 cusps. The eye width is 1 to 1.5 times the distance from the posterior edge of the eye to the anterior edge of the first branchial opening. The number of trunk myomeres averages 68 in adults and 67 (65-70) in ammocoetes. Adult river lampreys are dark on the back and sides and silvery to yellow on the belly with a darkly pigmented tail. Ammocoetes have somewhat pale heads, a prominent line behind the eye spot, and a tail in which the center tends to be lightly pigmented (Richards et al. 1982).

Taxonomic Relationships: The western river lamprey was described in 1855 by William O. Ayres, from a single specimen collected in San Francisco Bay, as *Petromyzon plumbeus*. Because that name had already been given to a European lamprey, it was renamed *P. ayresi* in 1870. A careful redescription of the river lamprey by V.D. Vladykov and W.I. Follett (1958) demonstrated its distinctiveness. The Pacific brook lamprey (*L. richardsoni*) and Kern brook lamprey (*L. hubbsi*) apparently evolved independently from river lampreys. See the Kern brook lamprey account in this report for further discussion of taxonomic relationships.

Life History: Western river lampreys have not been studied in California (Moyle 2002); therefore, the information in this account is based on studies in British Columbia (Roos et al.1973, Beamish and Williams 1976, Beamish 1980, Beamish and Youson 1987).

Larval river lampreys (ammocoetes) begin transformation into adults when they are about 12 cm TL, during summer months. Metamorphosis may take 9-10 months, the longest known for any lamprey. Newly metamorphosed lampreys may aggregate immediately upriver from salt water and enter the ocean in late spring. Adults apparently only spend 3-4 months in salt water where they grow rapidly, reaching 25-31 cm TL.

River lampreys prey on fishes in the 10-30 cm TL size range; the most common prey appear to be herring and salmon. Unlike other species of lamprey in California, river lampreys typically attach to the back of the host fish, above the lateral line, where they feed on muscle tissue. Feeding continues even after death of the prey. River lamprey predation may negatively affect prey populations if both prey and predator are concentrated in small areas (Beamish and Neville 1995). River lampreys can apparently feed in either salt or fresh water.

Adults migrate back into fresh water in the fall and spawn during the winter or spring months in small tributary streams, although the timing and extent of migration in California is poorly known. While maturing, river lampreys can shrink in length by about 20 percent. Adults create saucer-shaped depressions in gravelly riffles for spawning by moving rocks with their mouths. Fecundity estimates for two females from Cache Creek, Yolo Co., were 37,300 eggs from one 17.5 cm TL and 11,400 eggs for one 23 cm TL (Vladykov and Follett 1958). It is assumed that adults die after spawning, although this life history attribute has not been carefully documented in California. Ammocoetes remain in silt-sand backwaters and eddies and feed on algae and microorganisms. River lampreys spend an unknown amount of time as ammocoetes (probably 3-5 years), so the total life span is likely 6-7 years.

Habitat Requirements: The habitat requirements and environmental tolerances of spawning adults and ammocoetes have not been studied in California. Presumably, like other lampreys, adults need clean, gravelly riffles in permanent streams for spawning, while ammocoetes require sandy to silty backwaters or stream edges in which to bury themselves, where water quality is continuously high and temperatures do not exceed 25°C.

Distribution: Western river lampreys occur in coastal streams from just north of Juneau, Alaska, south to San Francisco Bay. In California, they have been recorded from the Sacramento and San Joaquin Delta while migrating, tributaries to the San Francisco Estuary (Napa River, Sonoma Creek, Alameda Creek), and tributaries to the Sacramento and San Joaquin rivers (e.g. Tuolumne River, Stanislaus River, Cache Creek). A land-locked population may exist in upper Sonoma Creek (Wang 1986). There are no recent records of river lamprey in Oregon and most older records are for the Columbia River basin (Kostow 2002). Likewise, they are known only from two large river systems in British Columbia in the center of their range (Beamish and Neville 1992).

Trends in Abundance: Western river lamprey population trends are unknown in California but it is likely that they have declined, concomitant to degradation and fragmentation of suitable spawning and rearing habitat in rivers and tributaries throughout their range in the state, along with declines in prey species (e.g., Chinook and coho salmon, steelhead trout, etc.). River lamprey are abundant within a limited geographic area of British Columbia, at the center of their range, but there are relatively few records from California, which comprises the southern end of their range.

Nature and Degree of Threats: The western river lamprey has become uncommon in California; it is likely that populations are declining because the Sacramento, San Joaquin and Russian rivers, along with their tributaries, have been severely altered by dams, diversions, development, agriculture, pollution, and other factors. They spawn and rear in the lower reaches of rivers and are, thus, highly vulnerable to alteration from agriculture and urbanization, as well as pollution. Two tributary streams where spawning has been recorded in the past (Sonoma and Cache creeks) are both severely altered by channelization, urbanization, and other impacts. See the Pacific lamprey account in this report for more specific information on stressors that negatively affect anadromous lamprey abundance.

	Rating	Explanation
Major dams	Medium	Most rivers within range are regulated by major dams
Agriculture	Medium	Lower stream reaches are impacted by diversions and impaired water quality
Grazing	Low	Present along most rivers; impacts likely minimal in large river systems
Rural residential	Low	Rural development is increasing rapidly across species' range; direct effects unknown but habitat degradation and reduced instream flows likely contribute to declines
Urbanization	Medium	Known range in Central Valley mostly urbanized
Instream mining	Low	Gravel mining common in preferred spawning streams
Mining	Low	Impacts unknown
Transportation	Medium	Roads, bridges, and ship canals alter habitats and are sources of pollutants
Logging	Low	Impacts to lower portions of larger river systems likely minimal
Fire	n/a	
Estuary alteration	Medium	Extent of estuary utilization unknown; estuaries likely constitute important feeding habitats that have been heavily altered and degraded throughout the state
Recreation	n/a	
Harvest	n/a	
Hatcheries	n/a	
Alien species	Low	May be prey for some alien species; may also prey upon certain alien species (e.g., American shad)

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

Effects of Climate Change: With so little known about this species, climate change effects are hard to predict. Nevertheless, the fact that California marks the southern end of its range, combined with its presence in the lower reaches of just a few large, regulated rivers, suggests that altered flow regimes and temperatures could further reduce or eliminate populations. Moyle et al. (2013) considered river lamprey to be "highly vulnerable" to climate change mainly because of its limited distribution and likely small populations, coupled with lack of knowledge about its basic biology in California.

Status Determination Score = 3.6 – Moderate Concern (see Methods section Table 2). Very little is known about this species in California but, given its dependence on lower reaches of large, regulated rivers, the river lamprey may be vulnerable to altered flows, altered habitats through urbanization, urban and agricultural pollutants, and similar factors (Table 2). Jelks et al. (2008) list it as being 'vulnerable' to extinction due to habitat changes, while NatureServe calls it "apparently secure" over its entire range.

Metric	Score	Justification
Area occupied	4	Known from at least 5 watersheds
Effective population size	3	This rating is likely high based on limited catches in sampling programs
Intervention dependence	5	Populations appear self-sustaining; habitat improvements may benefit populations in some areas
Tolerance	3	Presumed similar to brook lamprey
Genetic risk	4	Gene flow among populations not known
Climate change	3	Poorly understood because distribution and environmental tolerances are largely unknown; score assumes reduced habitat suitability and higher water temperatures will negatively affect river lamprey populations
Anthropogenic threats	3	See Table 1
Average	3.6	25/7
Certainty (1-4)	1	Little information available

Table 2. Metrics for determining the status of western river lamprey in California, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: The western river lamprey cannot be properly managed until more is known about its biology. Studies and field surveys to assess the river lamprey's distribution, abundance, life history and habitat requirements in California should to be implemented. The lower portions of the Sacramento and San Joaquin rivers, along with portions of the Bay Delta, should be targeted for initial studies and surveys since migratory river lampreys are caught in the Delta on a regular basis in various sampling programs. Presumably, restoring natural flow regimes and reducing inputs of pollution and sediment to its spawning streams will benefit the river lamprey but, given that so little is known about its tolerances and requirements, specific restoration actions and management recommendations cannot be developed without further study.

Figure 1. Presumed distribution of western river lamprey, *Lampetra ayresi*, in California. Distribution along the north coast is based on available passage to suitable habitats, rather than actual collection records.

KERN BROOK LAMPREY
Lampetra hubbsi (Vladykov and Kott)

Status: High Concern. Only six populations of Kern brook lamprey exist and they are isolated from one another; five are in short reaches below dams, so their persistence depends on dam operations and maintenance of suitable habitats for ammocoetes. The possible discovery of a 7^{th} population in the Sacramento River watershed, however, suggests the species may be more widely distributed than is currently known.

Description: The Kern brook lamprey is a non-predatory lamprey, so the teeth in its oral disk are small and blunt (Brown and Moyle 1992). Its morphology is like that of other lampreys: eel-like body, no paired fins, and a sucking disc instead of jaws. Larvae, known as ammocoetes, are similar to adults in shape but lack eyes and a well-developed oral disc. The Kern brook lamprey is much smaller than predatory anadromous lampreys; adults range from 81 to 139 mm TL and ammocoetes from 117 to 142 mm TL. Ammocoetes are typically larger than adults because non-predatory lampreys shrink following metamorphosis (Vladykov and Kott 1976). The number of trunk myomeres (i.e. the "blocks" of muscle mass along the body) ranges from 51 to 57 in ammocoetes (Tables 1, 2). In adults, the supra-oral lamina (tooth) typically has two cusps, with four inner lateral teeth on each side of the disc. The typical cusp formula is 1-1-1-1 (Vladykov and Kott 1976). The sides and dorsum are a grey-brown and the ventral area is white. Dorsal fins are unpigmented, but there is some black pigmentation restricted to the area around the notochord in the caudal fin (Vladykov and Kott 1976).

Taxonomic Relationships: The Kern brook lamprey was first described by Vladykov and Kott (1976) as a dwarf, non-predatory species in the genus *Entosphenus*. Based on dentition, the describers indicated the Kern brook lamprey was derived from the predatory Pacific lamprey, *E. tridentatus*, as are some other brook lampreys (Docker 2009). However, molecular analysis demonstrated it was derived from the predatory river lamprey, *Lampetra ayersi*, as is the western brook lamprey, *L. richardsoni* (Docker et al. 1999, Lang et al. 2009). Boguski et al. (2012) examined the genetics of lampreys from many populations in Pacific coast drainages; a single ammocoete from Paynes Creek (Tehama County) proved to be closely related to *L. hubbsi*. There are three potential scenarios to explain this: (1) it is a single, highly isolated population of *L. hubbsi*; (2) it is a separate undescribed species, and (3) other *L. hubbsi* populations exist in watersheds in the Sacramento Valley but have been overlooked. Clearly, more work on lamprey distribution and systematics in California is needed. The Pacific brook lamprey is differentiated from Kern brook lamprey on the basis of anatomical features (Tables 1, 2), as well as by mitochondrial DNA. The two species do not appear to be sympatric.

Table 1. Comparative counts and measurements of lamprey ammocoetes. *L ayersi* is from Vladykov (1973), *L. tridentata* and *L. hubbsi A*, from Vladykov and Kott (1976, 1979), *L. ayersi* from Richards et al. (1982) and *L. hubbsi B* from Brown and Moyle (unpubl. data). Data from Brown and Moyle are given as mean $\pm$ S.D. (above) and range (below). Data from other studies are mean (above) and range (below).

	Lampetra ayresi	*L. richardsoni*	*L. tridentata*	*L. hubbsi* A	*L. hubbsi B*
Total length (mm)	-	117	128	130	106 $\pm$ 19
	69 - 119	75 - 143	117 - 144	66 - 140	
Trunk myomeres	65	54	68	55	54 $\pm$ 2
	63 - 67	52 - 57	66 - 70	53 - 57	51 – 5

Table 2. Diagnostic characteristics of recently transformed adult lampreys of four *Lampetra* species. Data are from Vladykov and Follett (1958, 1965), Vladykov (1973) and Vladykov and Kott (1976).

	L. ayresi	*L. richardsoni*	*L. tridentata*	*L. hubbsi*
Trunk myomeres	68	56	66	56
	(60 - 71)	(53 - 58)	(63 - 70)	(54 - 57)
Cusps on supraoral lamina	2	2	3	2 - 3
Inner lateral "teeth"	3	3	4	4
Cusps on infraoral lamina	8.9	7.7	5.1	5.0
	(7 - 10)	(7 - 10)	(5 - 6)	5
Row of posterial "teeth"	absent	absent	present	present[1]
Predatory?	yes	no	yes	no

[1]Absent from two of eleven specimens examined by Brown and Moyle (unpublished data)

Life History: No documentation of the life history of Kern brook lamprey exists. However, if their life history is comparable to that of other non-predatory brook lampreys, they should live for approximately 4-5 years as ammocoetes before metamorphosing into adults (Moyle 2002). Based on collections (P. Moyle and L. Brown, unpublished data), metamorphosis occurs during fall. The adults presumably over-winter and spawn the following spring after undergoing metamorphosis.

Habitat Requirements: Principal habitats of Kern brook lamprey are silty backwaters of large rivers in foothill regions (mean elevation= 135 m; range= 30-327 m). In summer, ammocoetes are usually found in shallow pools along edges of run areas with minimal flow (L.R. Brown, US Geological Survey, pers. comm.), at depths of 30-110 cm where water temperatures rarely exceed 25 degrees C. Common substrates occupied are sand, gravel, and rubble (average compositions are 40%, 22%, 23%, respectively). Ammocoetes seem to favor sand/mud substrate, where they remain buried with the head protruding above the substrate and feed by filtering diatoms and other microorganisms from the water. This type of habitat is apparently present in the siphons of the Friant-Kern Canal. Adults require coarser gravel-rubble substrate for spawning. Temperature requirements for Kern brook lamprey are not known but the fact they are present almost entirely in reaches where summer temperatures rarely exceed 24 degrees C suggests a cool-water requirement.

Distribution: The Kern brook lamprey was first discovered in the Friant-Kern Canal (hence the inaccurate name; it is not found in the Kern basin). It has since been found in six locales which, presumably, represent isolated populations: the lower reaches of the Merced River, Kaweah River, Kings River, and San Joaquin River, as well as in the Kings River above Pine Flat Reservoir and the San Joaquin River above Millerton Reservoir, but below Redinger Dam (Brown and Moyle 1987, 1992, 1993; Fig. 1). In 1988, ammocoetes and adult lampreys were found in several siphons of the Friant-Kern Canal, when they were poisoned during an effort to rid the canals of white bass (*Morone chrysops*). The "low-count" lampreys (i.e., low numbers of trunk myomeres) reported from the upper San Joaquin River between Millerton Reservoir and Kerckhoff Dam by Wang (1986) are also most likely *L. hubbsi*, as are similar ammocoetes from the Kings River above Pine Flat Reservoir. As indicated in the taxonomy section, presumed Kern brook lampreys have been identified from Paynes Creek, Tehama County, which may indicate other populations exist as well.

Trends in Abundance: Since this species was first discovered in 1976, attempts to fully document its range have been only partially successful. Little is known about its past or present abundance. However, data collected to date suggest that this species is a San Joaquin basin (including the Kings River) endemic (Brown and Moyle 1992, 1993). Isolated populations of Kern brook lamprey seem spottily distributed throughout the San Joaquin drainage in regulated rivers, so their distribution and abundance are probably much reduced from pre-dam times. Ammocoetes thrive in the dark siphons of the Friant-Kern Canal, but it is unlikely that there is suitable spawning habitat in the canal, so those individuals probably do not contribute to the persistence of the species.

Nature and Degree of Threats: Populations of this species are scattered throughout the middle San Joaquin-Kings drainage and are isolated from one another. Such a limited and fragmented distribution makes local extirpations increasingly probable, along with a high degree of genetic risks from small population sizes and isolation; without interconnected populations and the possibility of recolonizing degraded habitats, eventual extinction may occur.

Major dams. It is likely that the river reaches flooded by Millerton and Pine Flat reservoirs were once important habitats for Kern brook lamprey. Today, the probability of local extirpation is increased by the fact that all known populations, with one exception, are located below dams, where stream flows are regulated without regard to the habitat requirements or life history needs of lampreys. Fluctuations or sudden drops in flow may isolate ammocoetes or result in the drying of habitats. Gravels required for spawning may be eliminated (trapped by dams) or compacted so they cannot be used by adults, while silt required by ammocoetes may be flushed out of the cool-water reaches that appear to be preferred by larvae. Dams also isolate populations, eliminating gene flow and preventing recolonization from nearby populations. Management of flows in the lower reaches of the San Joaquin and Kings rivers, including the new restoration flows below Friant Dam, as well as flows to reduce impacts from agricultural return waters, will need to account for the needs of this species in order for populations to persist.

Agriculture. Channelization, road building, irrigation withdrawls, and other activities associated with farming eliminate backwater areas required by ammocoetes. Ammocoetes may also be carried by water being delivered to farms via the Kings River to "dead-end" habitats such as the Friant-Kern siphons. In addition, pollutants are of concern (including elevated temperatures) in agricultural return waters, which may reduce lamprey survival.

Urbanization. Fresno is rapidly expanding around the San Joaquin River with attendant stressors associated with urban development, including road building, bank stabilization, pollution, and recreation.

Instream mining. Large sections of the San Joaquin River have been mined for gravel, both destroying shallow-water habitats needed by ammoceotes and creating large pits that provide ideal habitats for predatory fishes. It is likely that lampreys were extirpated from gravel pit regions once mining began.

Alien species. Kern brook lamprey habitats typically support a mixture of native and non-native fishes (Moyle 2002). The impacts of alien fishes, especially predatory bass (*Micropterus* spp.), are not known, but are likely to be negative, given the vulnerability of migrating larvae and adults to predation.

	Rating	Explanation
Major dams	High	Most populations exist below dams, where habitat is degraded and flows are highly regulated
Agriculture	High	Most populations are susceptible to agricultural pollution, diversions and other factors
Grazing	Low	Present along some streams
Rural residential	Low	Effluent from waste water and bank protection to reduce flooding may affect habitats
Urbanization	Medium	Fresno and other urban areas are expanding; potential for increased impacts from pollution, habitat degradation and fragmentation
Instream mining	Medium	Gravel pits present in some areas; associated impacts may have eliminated lampreys from reaches of the San Joaquin River
Mining	n/a	
Transportation	Low	Roads and railroads along rivers may alter habitats and increase both sediment and pollutant input
Logging	n/a	
Fire	Low	
Estuary alteration	n/a	
Recreation	Low	Areas accessible to off-road vehicles and other uses may reduce ammocoetes habitats or disrupt spawning
Harvest	n/a	
Hatcheries	n/a	
Alien species	High	Alien predators present; effects unknown but potentially significant

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

Effects of Climate Change: The southern Central Valley of California is predicted to experience reduced stream flows and increased water temperatures, as a result of longer, more frequent, droughts and warmer air temperatures. Kern brook lampreys live in regulated rivers, so climate change effects are most likely to manifest from changes in dam and reservoir operations, including reduced dam releases (drying up rearing areas) or warmer temperatures of released water. Without consideration for lamprey needs, such operational changes can greatly increase extinction risk. Moyle et al. (2013) indicated the Kern brook lamprey is "critically vulnerable" to climate change, facing extinction because of changed dam operations, including reduced flows during droughts, and alteration/degradation of habitats to favor expansion of alien species.

Status Determination Score = 2.3 - High Concern (see Methods section, Table 2). The Kern brook lamprey does not appear to be at immediate risk of extinction but its status could change rapidly, given the limited number of isolated populations and their existing distribution either below or just above dams. Jelks et al. (2008) considered the species as threatened and declining, while NatureServe considers its status to be somewhere between Imperiled (G2) and Critically Imperiled (G1). The species was petitioned for federal listing in 2003 as threatened, but the petition was denied on Dec. 27, 2004 because “the petition did not provide sufficient information to warrant initiating a status review (USFWS 2004).”

Metric	Score	Justification
Area occupied	2	Six known populations occur in two watersheds but all are isolated from one another by dams and diversions; possible 7th population needs further investigation
Estimated adult abundance	3	Not known but probably <1000 adults in each population
Intervention dependence	3	Long-term persistence requires habitat improvements and flow regulation
Tolerance	3	Unstudied but probably moderate
Genetic risk	2	Populations fragmented; potential for bottlenecks or inbreeding depression
Climate change	1	Populations below dams could be threatened by changes in river management
Anthropogenic threats	2	See Table 3
Average	2.3	16/7
Certainty (1-4)	2	Little published information on abundance, distribution, or status, especially in the recent past

Table 4. Metrics for determining the status of Kern brook lamprey, where 1 is a major negative factor contributing to status, 5 is factor with no or positive effects on status, and 2-4 are intermediate values. This score does not take into account the apparent population in the Sacramento River watershed. See methods section for further explanation.

Management Recommendations: The Kern brook lamprey would most benefit from proactive management strategies and actions treating it as if it were already a listed species, in order to reduce the probability of actual listing. A thorough survey of the known habitats and populations of this species needs to be conducted to determine status and possible trends. Extensive surveys are needed to determine present distribution and to provide more exact information on habitat requirements within its known range, as well to determine if populations exist outsides the known range (e.g., in the Kaweah River, Sacramento Valley). A study needs to be conducted to determine if ammocoetes still use the silty bottoms of siphons in the Friant-Kern Canal and if rescue and transplantation of these larvae would be beneficial. Specialized surveys should focus on adults to determine population sizes and spawning habitat requirements. Known or probable populations should be monitored every two to five years, with trends determined by catch per effort or estimated densities of ammocoetes.

Once surveys are completed, several known areas of suitable habitat should be selected for special management or protection from incompatible uses, including some in the soon-to-be-restored San Joaquin River. These same areas should be the focus of life history studies and studies that determine habitat requirements.

Figure 1. Known (confirmed) distribution of Kern Brook lamprey, *Lampetra hubbsi* , in California.

WESTERN BROOK LAMPREY
Lampetra richardsoni (Vladykov and Follet)

Status: Moderate Concern. Western brook lampreys are still present in the least disturbed portions of many watersheds but all populations are likely small, isolated and declining.

Description: Western brook lampreys are small, usually less than 18 cm TL, and nonpredatory (Moyle 2002). They have poorly developed tooth plates in the oral disc and tooth plates in spawning adults may be missing from the anterior field. The supraoral plate is wide with one cusp at each end. The infraoral plate has 6-10 toothlike cusps and 3 circumoral plates on each side of the mouth. The middle circumoral plate has 2 or 3 cusps. Cusps on the transverse lingual lamina are inconspicuous. The oral disc is narrower than the head with a length that is less than 6 percent of the total length. Both adults and ammocoetes have trunks made up of 52-67 myomeres (52-58 in California populations). Body coloration is dark on the sides and back, and light (yellow or white) on the underside. Ammocoetes have dark tails and heads above the gill opening (Richards et al. 1982).

Taxonomic Relationships: The western brook lamprey was determined to be a species, *L. richardsoni*, distinct from the European brook lamprey, *L. planeri*, in 1965, but closely related to the predatory river lamprey, *L. ayersi* (Vladykov and Follett 1962). Later, populations in Oregon and California were described as belonging to *L. pacifica* by Vladykov (1973). C. Bond, in an unpublished study, determined that differences in myomere counts that were thought to distinguish *L. pacifica* from *L. richardsoni* did not do so when populations throughout their range were sampled, so the name was quashed without further review by the American Fisheries Society (Robins et al. 1991, Stewart et al. 2011). Stewart et al. (2011) determined it is, indeed, a valid species but confined to the Columbia River basin. Boguski et al. (2012) examined nominal river and brook lampreys from the entire Pacific Coast and found that, for the most part, the non-predatory brook lampreys conformed to *L. richardsoni*, on the basis of both morphology and genetics (mitochondrial DNA). However, there were some notable exceptions:

- The Kern brook lamprey was confirmed to be a distinct species, with a possible additional population in Paynes Creek, Tehama County (see the Kern brook lamprey account in this report for further information).
- A very distinctive population (based on mitochondrial DNA) was found isolated in Kelsey Creek, Lake County, a tributary to Clear Lake. Further investigation is needed to determine if this is another endemic species in the Clear Lake watershed.
- The population in Mark West Creek, a tributary to the lower Russian River, was found to be genetically distinct, perhaps indicating a distinct lineage in the Russian River.

The western brook lamprey is very similar to the river lamprey, based on mitochondrial DNA analysis (Docker et al. 1999). The nonpredatory brook lampreys in many coastal streams are, therefore, potentially derived from river lamprey through a series of independent evolutionary events, found in other “pair species” of lampreys as well (Docker 2009). Brook lamprey adults are not known to migrate although, in British Columbia, some streams contain both predatory and nonpredatory adults, with the predatory form able to migrate to salt water (Beamish 1987, Beamish et al. 2001). River and brook lampreys hybridize in the laboratory but hybridization in the wild has not been observed (Beamish and Neville 1992). Docker (2009) suggested that the distinctness of members of species pairs of lampreys depends on how recently the non-predatory form developed. Long isolation leads to speciation, as in the Kern brook lamprey. It is clear that further research on the systematics of the brook lamprey is required; however, mounting evidence indicates that California populations are distinct.

Life History: Most published studies relating to western brook lampreys were done outside of California (Schultz 1930, McIntyre 1969, Kostow 2002, Gunckel et al. 2009), with the exception of a study by Hubbs (1925). It is assumed, however, that differences in biology between California populations and those elsewhere are minor, based on unpublished observations (cited below).

Spawning adult brook lamprey build nests in gravel riffles that are slightly smaller in diameter than their body lengths. In Mark West Creek, during April, 1994, they were observed building nests 15-20 cm wide in gravel riffles at a depth of about 15 cm (M. Fawcett, pers. comm. 1998). In the Smith River, Oregon, most nests are about 12 cm (length) by 11 cm (width) by 3 cm (depth) and are located in low velocity (ca. 0.2 m/sec) water averaging 13 cm depth (Gunckel et al. 2006, 2009). Median gravel size in nests is 24 mm and most nests are associated with cover (boulder, wood, vegetation). Sixty-eight percent of nests in the Oregon study were found in either pool tail-outs or low gradient (<2% slope) riffles. Spawning begins when water temperatures exceed 10°C (Schultz 1930, Kostow 2002). However, in Cedar Creek, Washington, spawning occurred at temperatures ranging from 8.6 to 17.4°C (Stone et al. 2002). In California's North Fork Navarro River, spawning begins in early March, peaks between mid-April and mid-May, and may continue through the first week of June (S. Harris, pers. comm. 2011). In Outlet Creek (Eel River watershed), spawning begins slightly later (mid-March), peaks in late-April to late-May, and continues through mid-June (S. Harris, pers. comm. 2011).

Spawning behavior is similar to that of Pacific lamprey (Schultz 1930, Morrow 1980). In Cedar Creek, 3 to 12 lampreys were observed working together to move large rocks out of the nest prior to spawning (Stone et al. 2002). Upon completion of the nest, adhesive eggs are deposited and covered with sand and gravel (summarized in Kostow 2002). Adults die after spawning. Length of the spawning season varies from 6 months in Washington (Schultz 1930), where flow conditions are more constant, to 2 months (March-April) in Coyote Creek (Alameda County) (Hubbs 1925). Fecundity ranges from 1,100 to 5,500 eggs per female (Wydoski and Whitney 1979, Kostow 2002). Eggs hatch in about 30 days at 10°C, 17 days at 14°C, 12 days at 18°C and 9 days at 22°C (Meeuwig et al. 2005). Speckled dace (*Rhinichthys osculus*) and salmonids (*Oncorhynchus* spp.) have been observed to feed on eggs in and around lamprey nests (Brumo 2006).

After hatching, embryos and larvae (ammocoetes) may spend another week to a month in the nest (summarized in Kostow 2002). Once they reach about 10 mm, ammocoetes leave the nest and move downstream, usually at night, to burrow tail first into deposits of fine sediment; their mouths are located near the substrate surface so that they can filter feed. Movement of ammocoetes occurs year-round, mostly at night (Kostow 2002), but is primarily associated with increases in discharge (Stone et al. 2002). Ammocoetes move further downstream into deeper water as they grow (Kostow 2002). Ammocoetes are most common in sandy and silty areas of backwaters and pools, occurring in aggregations as dense as 170 per square meter (Schultz 1930). However, densities in two sites of the South Fork Walla Walla River, Washington and Oregon, were 5 and 37 individuals per square meter, respectively (Close et al. 1999). Western brook lampreys live as ammocoetes for 3-4 years in California and Oregon, and 4-6 years in British Columbia (Hubbs 1925, Schultz 1930, Pletcher 1963, Wydoski and Whitney 1979). California populations grow the fastest and largest (13-18 cm) by feeding on algae (especially diatoms) and organic matter (Wydoski and Whitney 1979). Ammocoetes begin transforming in the fall and mature by spring. Individuals develop eyes and an oral disc and undergo physiological changes in the gills and nasopineal gland (Kostow 2002). They become dormant in burrows during the transformation stage and do not feed as adults.

Where western brook and Pacific lamprey co-occur, there can be some degree of overlap in spawning habitat; in some cases western brook lamprey will spawn within Pacific lamprey nests (Stone et al. 2002, Luzier and Silver 2005, Brumo 2006, Gunckel et al. 2006, 2009). However, western brook lamprey generally spawn further upstream in smaller tributaries than Pacific lamprey. The bile acid, petromyzonol sulfate, may be used as a chemical cue between conspecifics (Yun et al. 2003), perhaps influencing in-river distribution.

Habitat Requirements: Western brook lampreys have habitat requirements similar to those of salmonid species, with which they co-occur. They need clear, cold, water in little disturbed watersheds, as well as clean gravel near cover (boulders, riparian vegetation, logs, etc.) for spawning. Additional habitat requirements include areas with low flow velocities and fine sediments for rearing that are not excessively scoured under high flows. Habitat utilization surveys of spawning western brook lamprey in Cedar Creek, Washington, found that adults avoided areas with deep, fast water and large substrates, suggesting specific habitat needs for spawning (Luzier and Silver 2005). Lamprey presence was positively correlated with temperature, percent fine substrate and dissolved oxygen and negatively correlated with stream gradient, velocity, percent bedrock and percent large gravel (Stone et al. 2002). In the Tualatin River basin, Oregon, western brook lampreys were most commonly found in shady glides or riffles with relatively fine substrates (soil or rock), in stream reaches without obvious signs of habitat degradation (Leader 2001). Optimum temperatures for embryo and larval development are 10-18°C (Meeuwig et al. 2005).

Distribution: Western brook lampreys occur in coastal streams from southeastern Alaska south to California and inland in the Columbia and Sacramento-San Joaquin River drainages (Vladykov 1973, Morrow 1980). California populations are primarily found in the Sacramento River watershed, including remote areas such as Kelsey Creek, upstream of Clear Lake (Lake County), and St. Helena Creek (Lake County), a tributary to upper Putah Creek. They are also found upstream of Pillsbury, Morris and Centennial reservoirs in the Eel River drainage (Mendocino County) (Brown and Moyle 1996, S. Harris, pers. comm. 2011) and in tributaries to the Russian River, such as Mark West Creek (Sonoma County) (M. Fawcett, pers. comm. 1998) and Austin Creek (J. Katz, pers. obs. 2009). Spawning adults have been collected from the Navarro River, Mendocino County (J.B. Feliciano, pers. comm. 1999). Ammocoetes were once collected from the Los Angeles River (Culver and Hubbs 1917) but they have been extirpated from this highly degraded system (Swift et al. 1993, Swift and Howard 2009). Hubbs (1925) also collected ammocoetes from Coyote Creek, Santa Clara County. They likely occur in other coastal rivers systems as well (Moyle 2002). Boguski et al. (2012) note that isolated populations they examined (e.g. from Kelsey Creek) are often genetically distinct and may deserve recognition as separate taxa.

Trends in Abundance: Western brook lampreys are probably more common than survey data indicate because they are difficult to observe and to distinguish from other species (Kostow 2002, Moyle 2002). In Oregon, they are assumed to occur in less than half of their historic habitats in the Columbia River and Willamette River subbasins (ODFW 2006). Consequently, they are considered to be "at risk" due to habitat loss, passage barriers and pollution. However, they are still common in other parts of Oregon such as the Smith River (tributary to the Umpqua River), where an estimated 4,692 (2004) and 4,265 (2005) western brook lamprey nests were observed (Gunckel et al. 2006). Abundance data for California populations are not available and there are no records of spawning numbers such as those observed in Oregon.

Nature and Degree of Threats: Little is known about the factors limiting abundance or distribution of western brook lamprey in California. Threats to western brook lamprey in Oregon include pollution, logging, degraded water quality, changes to natural hydrographs (including rapid reduction in flows, scouring), dredging and development in floodplains and low gradient stream reaches (Kostow 2002). It is likely that some, if not all, of these stressors also affect populations in California streams. In particular, brook lamprey populations are exceptionally vulnerable to single transitory events (pollution, dewatering) that can kill relatively immobile ammocoetes. Local extinctions caused by such events are likely to go unnoticed.

Major dams. Many streams occupied by western brook lampreys are dammed and/or diverted to some extent; small diversions are more prevalent than large dams in most portions of their range. Major dams on coastal and Central Valley rivers have likely fragmented habitats and isolated populations in upstream areas, as has been documented elsewhere (Close et al. 1999). Where altered flow regimes below dams have changed habitats (e.g. reduced backwaters, increased summer temperatures) brook lamprey are generally absent.

Agriculture. Western brook lamprey tend to occur in low gradient reaches of California streams that are impaired, to varying degrees, by local agriculture, both legal and illegal (e.g., marijuana cultivation). Such streams may be less suitable for all lamprey life stages as the result of diversions, pollution and poor water quality from agricultural return waters. For example, the rapid expansion of vineyards in coastal watersheds has likely reduced habitat quality and quantity for lampreys in many areas.

Grazing. Livestock grazing in headwater streams favored by brook lampreys alters channel morphology (stream bank degradation, widening and shallowing of stream channels), increases sedimentation (potentially degrading spawning habitats but also potentially increasing abundance of fine sediment deposition areas utilized by ammocoetes), reduces riparian vegetation (stream shading and water temperature moderation) and may cause localized impacts due to pollution input from animal wastes.

	Rating	Explanation
Major dams	Medium	Dams block passage, alter natural flow regimes and sediment budgets
Agriculture	Medium	Many populations affected by polluted water and reductions in flows from diversions
Grazing	Medium	Grazing occurs throughout species' range
Rural residential	Medium	Rural development increasing within species' range; may cause localized pollution and habitat degradation in many areas
Urbanization	Medium	Lampreys are absent from heavily urbanized areas
Instream mining	Low	Dredging formerly impacted many areas occupied by lampreys; dredging currently prohibited in CA
Mining	Low	Legacy toxic effects of mine drainage may still affect populations; may be particularly acute to ammocoetes, due to filter feeding in substrates where mercury accumulates
Transportation	Medium	Roads (particularly unsurfaced roads in headwater areas) can increase sediment delivery and fragment and degrade habitats
Logging	Medium	Most streams in species' range are affected by logging and logging roads
Fire	Medium	Forest fire frequency and intensity are increasing in species' range
Estuary alteration	n/a	
Recreation	n/a	Recreational impacts to lamprey populations are unknown
Harvest	n/a	
Hatcheries	n/a	
Alien species	Medium	Unknown impacts but co-occurrence likely throughout much of range

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

Rural residential. Rural communities are common throughout the species' range and rural development in many areas is increasing rapidly. Development (e.g., road building, building site preparation, water and power delivery), along with pollution from septic tanks and household wastes, can degrade aquatic habitats and water quality.

Urbanization . Urban development along streams (e.g., Mark West Creek in Santa Rosa) decreases the abundance of rearing habitats, while pollutants can kill adults and ammocoetes. Channelization simplifies stream morphology and often eliminates edge habitats needed by ammocoetes. Lampreys are usually absent from urban streams, such as the Los Angeles River and Coyote Creek, in which they were historically present, indicating that urban development adjacent to streams has a significant impact on their persistence.

Mining. Eggs, embryos and ammocoetes may have been negatively affected by suction dredging in the past; however, there is currently a moratorium on suction dredging in California. Nonetheless, dredging is still considered an important threat in Oregon (Kostow 2002) and could become so again in California if the moratorium is lifted. Legacy effects from widespread historic hard-rock mining (e.g., for mercury) may have eliminated or reduced populations in many areas. Toxins (e.g., heavy metals) from mostly historic mining operations may still persist in stream substrates, causing direct and prolonged exposure to ammocoetes with unknown effects on this life history stage. Instream gravel mining operations may contribute to removal of important spawning habitats or disruption of habitat utilization by all life stages.

Transportation. Culverts can create barriers and limit longitudinal movements within streams, especially for fishes with limited burst-speed swimming or jumping capabilities (e.g., lampreys). Roads along streams, especially unsurfaced roads in headwater areas (logging, recreational or other unimproved roads), often contribute to increased fine sediment or pollutant delivery to streams. Higher sediment loads are associated with degradation of spawning gravels and may contribute to excessive deposition in backwater or edgewater areas required for ammocoete rearing.

Logging. Timber harvest has been widespread and historically intensive throughout the range of western brook lamprey in California. Many areas have been logged multiple times, with resultant changes in forest vegetation composition, alteration to streams (e.g., geomorphology, annual hydrograph) and degradation of aquatic habitats (e.g., increased siltation, lack of canopy cover for shading and stream temperature moderation). Logging can reduce lamprey numbers after timber harvest occurs due to stream alteration (Moring and Lantz 1975), while extensive road networks created to facilitate logging continue to contribute sediments and increased surface run-off into streams.

Fire. Under predicted climate change scenarios, wildfires are expected to become more frequent and intense in many portions of the western brook lamprey's range, potentially leading to more extensive forest and aquatic habitat damage and longer recovery periods for these habitats. Fires can result in landslides that smother spawning gravels and removal of vegetation from riparian areas. Fire retardant reaching streams may cause localized areas of low dissolved oxygen, to which western brook lampreys are sensitive (Stone et al. 2002).

Alien species. Alien fishes (e.g., smallmouth bass) feed on ammocoetes and adults but the extent of impacts on lampreys from alien species predation and/or competition is not known. Alien fishes, however, are widespread throughout the western brook lamprey's range, so the potential for negative interactions is considerable.

Effects of Climate Change: The most noticeable and widespread impacts from climate change on lamprey habitats in California will be continued increases in water temperatures and changes to the frequency and timing of drought and flooding events. Water temperature increases may reduce the individual fitness of brook lampreys by decreasing growth, decreasing reproductive potential and increasing susceptibility to disease. The early life history stages (embryo to larva) are particularly sensitive to temperature increases. Both survival to hatch (~60%) and to the larval stage (~50%) significantly decreased at 22°C as compared to all other temperatures (10, 14 and 18°C; Bayer et al. 2001, Meeuwig et al. 2005). Survival to hatch and larva was about 90% from 10-18°C. Furthermore, physical deformities (e.g. deformed egg or yolk, fragmented yolk, bent or deformed prolarvae) occurred at all temperatures (<7%, Bayer et al. 2001) but was significantly higher at 22°C (~35%, Meeuwig et al. 2005). In general, most western brook lamprey populations are found in streams where temperatures are not likely to exceed 18°C during incubation or early rearing during spring months.

Elevated air temperatures associated with climate change will change the periodicity and magnitude of peak and base flows in streams, due to a reduction in snow pack levels and seasonal retention, particularly in watersheds at low elevations (< 3000 m) (Hayhoe et al. 2004). Predictions are that stream flow will increase in the winter and early spring and decrease in the fall and summer (Knox and Scheuring 1991, Field et al. 1999, CDWR 2006), perhaps changing the spawning ecology of fishes. If increased winter and spring flows make floodplain habitats accessible, western brook lamprey ammocoetes may benefit by rearing in highly productive habitats. Ammocoetes, however, can become stranded when flow decreases too quickly (Kostow 2002). If adults and ammocoetes spawn and rear in main channels, increased winter and spring flows may shift stream sediments to the detriment of nests and eggs. Because of their early life history stages' particular sensitivity to increased water temperatures, as well as their general immobility, Moyle et al. (2013) rated the species "highly vulnerable" to extinction within the next 100 years due to the added effects of climate change.

Status Determination Score = 3.0 - Moderate Concern (see Methods section Table 2). NatureServe lists western brook lamprey as globally secure (G4) but vulnerable in California (S3). In Oregon, they are considered a species "at risk." In 2003, a petition to list western brook lamprey in the Pacific Northwest and California under the Federal Endangered Species Act was received by the U.S. Fish and Wildlife Service (USFWS) (Nawa 2003). The petition cited habitat degradation and loss as major threats to the species. The USFWS determined the petition did not warrant further review based on insufficient scientific or commercial information (50 CFR Part 17). The high concern status in this report is driven by multiple interacting factors that have degraded many of the streams brook lampreys inhabit, combined with lack of information about their actual distribution or relative abundance within California (Table 2).

Metric	Score	Justification
Area occupied	5	Most historic watersheds are apparently still occupied
Estimated adult abundance	2	No population size information is available for California, but populations are assumed to be small
Intervention dependence	4	Persistence requires habitat improvements and stream protection
Tolerance	3	Moderately tolerant of warm temperatures; intolerant of low dissolved oxygen, pollution, low flows and disturbances to stream sediments
Genetic risk	2	Isolation and apparent small size of most populations increases vulnerability to genetic risks
Climate change	2	Populations are vulnerable to changes in natural flow regimes and increased temperatures
Anthropogenic threats	3	Multiple interacting threats exist (Table 1)
Average	3.0	21/7
Certainty (1-4)	2	Poorly known in California; better data available on populations in other states

Table 2. Metrics for determining the status of western brook lamprey, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: One of the greatest challenges to management of western brook lamprey is the lack of basic information on its status and biology in California; data are needed on distribution, abundance, genetics, environmental tolerances and population structure. In particular, research is needed to determine the status of isolated, distinctive populations such as those in Kelsey Creek and the Russian River; such forms may merit further taxonomic recognition (Moyle 2002, Boguski et al. 2102). Baseline surveys are needed to establish the relative abundance of this species within its range. Monitoring surveys (every 5 years) should be implemented in order to determine trends in distribution and abundance. Studies are also needed to establish the environmental tolerances of brook lampreys in California, especially to factors affected by land use and climate change, including temperature, turbidity, sedimentation, flows and water velocity.

Streams known to support brook lamprey populations, as well as those with the potential to do so, should be managed in ways that favor native fishes in general, including maintaining cool temperatures, spawning riffles and complex habitat structure using active management of water and land use practices or restoration actions, where necessary. For example, management of flow releases from hydroelectric projects should take into account the habitat requirements of native aquatic fauna, including western brook lamprey. Dam releases, in general, should mimic natural flow regimes in scale and periodicity. Grazing and logging activities should be buffered from riparian areas to protect riparian vegetation, limit nonpoint source pollution and minimize stream bank destabilization and excessive fine sediment inputs.

Figure 1. Assumed distribution of western brook lamprey, *Lampetra richardsoni*, in California. Actual distribution is largely unknown and distribution shown may include undescribed taxa.

PIT-KLAMATH BROOK LAMPREY
***Entosphenus lethophagus* Hubbs**

Status: Moderate Concern. While Pit-Klamath brook lamprey do not currently appear to be at risk of extinction, aquatic habitats within their range are heavily altered by agriculture and grazing and their actual abundance is unknown.

Description: Pit-Klamath brook lamprey are small and non-predatory (Hubbs 1971, Renaud 2011). Their oral disc resembles that of Pacific lamprey but have fewer and smaller teeth (plates). Lateral circumoral plates number 2-3-3-2 or 1-2-2-1, with cusps often missing. They have 9-15 posterior circumoral plates, often with just one cusp. The supraoral plate has 3 cusps, although the middle one may be smaller or absent. They usually have 5 infraoral teeth. Cusps on the transverse lingual lamina are difficult to see and are file-like. The small, puckered, mouth has a disc length less than 5 percent of body length. The disc is narrower than the head when stretched (Page and Burr 1991). Myomeres along the trunk number 60-70. Mature individuals exhibit gut atrophy. Coloration in adults is dark gray on the dorsum and brassy or bronze on the ventrum. See Renaud (2011) for a description of ammocoetes and comparisons with other lampreys in the Klamath region.

Taxonomic Relationships: Pit-Klamath brook lamprey were described from specimens collected from various locations in the Pit and Klamath basins by Hubbs (1971), as *Lampetra lethophaga*. This lamprey is closely related to Pacific lamprey (Docker et al. 1999, Lang et al. 2009). Recent phylogenetic analysis indicates that the species should be placed in the genus *Entosphenus,* and removed from the genus *Lampetra* (Lang et al. 2009). Analysis of characteristics of ammocoetes confirms this relationship (Goodman et al. 2009). Non-predatory lampreys in the two drainages may have been derived independently from Pacific lamprey and may ultimately be regarded as separate taxa (Kostow, 2002, Moyle 2002).

Life History: Spawning may begin in early spring and occur through summer (Moyle 2002). Fecundities may be similar to other lampreys with equivalent sizes at about 900 to 1,100 eggs per female (Kan 1975 in Kostow 2002). In some areas, adults may not develop nuptial features such as back and belly with dark, contrasting coloration; fused dorsal fins with frills; and enlarged anal fin (Moyle 2002). Larval lampreys (ammocoetes) usually burrow among aquatic vegetation into soft substrates (Moyle and Daniels 1982), where they likely feed on algae and detritus (Moyle 2002). Based on size classes, the ammocoete stage lasts for about four years, during which time they reach about 21 cm TL. Metamorphosis likely occurs in fall. Adults presumably only move short distances to spawning areas (Close et al. 2010). They commonly co-occur with trout, marbled and rough sculpins, and speckled dace (Moyle 2002).

Habitat Requirements: Pit-Klamath brook lampreys principally occupy habitats in clear, cool (summer temperatures < 25ºC) rivers and streams in areas with fine substrates and beds of aquatic plants (Moyle and Daniels 1982, Moyle 2002). Like other lampreys, Pit-Klamath brook lampreys require gravel riffles in streams for spawning, with muddy backwater habitats downstream of spawning areas for ammocoete burrows. In the Pit River system, they seem especially common in backwaters of the spring-fed Fall River and Hat Creek (Moyle and Daniels 1982). Pit-Klamath brook lamprey in the Oregon portion of the Goose Lake basin are most commonly found in high-elevation streams in forested lands (Scheerer et al. 2010).

Distribution: Pit-Klamath brook lampreys, as currently defined, are only found in the Pit River-Goose Lake basin in California and Oregon as well as in the upper Klamath basin, upstream of Klamath lakes in Oregon (Hubbs 1971, Moyle and Daniels 1982). If this species is broken into two entities, then only *E. lethophagus* occurs in California, where it is widely distributed throughout the Pit River basin and, presumably, the Goose Lake basin in both California and Oregon (Moyle and Daniels 1982, Kostow 2002, Moyle 2002).

Trends in Abundance: Abundance and population trend information are lacking. Their populations do not seem to be in danger of extinction at this time but face multiple threats (discussed below).

Nature and Degree of Threats: Pit-Klamath brook lamprey face degradation of suitable habitats by multiple factors affecting streams in this arid region. The main stem Pit River and some of its tributaries are currently listed as impaired due to high temperatures and nutrient loading, as well as low dissolved oxygen levels (Pit RCD 2006, DEQ 2010).

Major dams. The lower Pit River supports a chain of hydropower reservoirs and some tributaries also have small dams on them. The effects of these dams on lampreys are unknown but some habitats have been inundated and populations may be fragmented as a consequence.

Agriculture. Water demands for agriculture are high along the Pit and upper Klamath rivers, resulting in decreased instream flows. Water diversions in some areas may be reducing instream flows to the extent that certain reaches go dry (Pit RCD 2006). Flood-irrigated pastures introduce nutrients into streams and raise water temperatures, via return water, and fertilizers are thought to be increasing nutrient loadings in streams (Pit RCD 2006). Pit-Klamath brook lamprey may be well adapted for some altered habitats, especially in the larval stage. Ammocoetes were common in the mud substrates of an irrigation diversion from Rush Creek, Modoc County (Moyle 2002). They are also common in silt substrates of pools below channelized sections of streams.

Grazing. Extensive grazing occurs throughout the range of Pit-Klamath brook lamprey. Grazing can degrade aquatic habitats through streambank trampling, removal of riparian vegetation, or input of nutrients and other pollutants from animal wastes. Fecal matter is thought to be increasing the nutrient loading of streams in this region (Pit RCD 2006). Removal of vegetation increases erosion and entrenchment of stream channels (Pit RCD 2006) and contributes to increased solar input and corresponding water temperature increases in streams.

Rural residential. Several towns exist within the Pit-Klamath brook lamprey range (e.g. Alturas) in California. Residential areas can be sources of pollutants and increased water demands that may decrease water quantity and quality in streams.

Alien species. Many alien fish species inhabit the Klamath and Pit River basins (Close et al. 2010, Moyle and Daniels 1982). Species that can prey on lamprey include largemouth bass, brown bullhead, channel catfish, brook trout, brown trout, black crappie, and yellow perch (Close et al. 2010).

	Rating	Explanation
Major dams	Low	Dams present in range but impacts are unknown
Agriculture	Medium	Agriculture pervasive throughout range; direct effects unknown but likely contributes to substantial diversion and water quality degradation; effects may be severe at a localized level
Grazing	Medium	Grazing pervasive throughout range; direct effects unknown but likely contributes to aquatic and riparian habitat degradation, along with water quality impairment across much of range
Rural residential	Low	Small towns and residences common but widely dispersed within range; impacts likely minimal except for water withdrawls and potential pollutant inputs at a localized scale
Urbanization	n/a	
Instream mining	n/a	
Mining	n/a	
Transportation	Medium	Extensive network of unimproved roads across range; potential for increased sediment inputs and habitat fragmentation
Logging	Low	Logging occurs in forested lands; impacts unknown
Fire	Low	Wildfires occur throughout range; impacts unknown
Estuary alteration	n/a	
Recreation	n/a	
Harvest	n/a	
Hatcheries	n/a	
Alien species	Medium	Absent where alien species abundant

Table 1. Major anthropogenic factors limiting, or potentially limiting, viability of populations of Pit-Klamath brook lamprey in California. Factors were rated on a five-level ordinal scale where a factor rated "critical" could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated "high" could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction unlikely as a result; and a factor rated "no" has no known negative impact to the taxon under consideration. Certainty of these judgments is low. See methods section for descriptions of the factors and explanation of the rating protocol.

Effects of Climate Change: Climate change is expected to increase the frequency of both drought and floods in streams. Because Pit-Klamath lamprey rear for several years in stream substrates, large flooding events may disrupt rearing habitats (Fahey 2006) and displace ammocoetes from soft sediments. On the contrary, scouring events may clean sediments from gravels that would otherwise degrade spawning habitats (Stuart 2006 in Fahey 2006). This species may not be as vulnerable as other fishes to stream flow changes associated with climate change because a few populations occur in large, spring-fed river systems (e.g. Fall River). Changes to the natural hydrograph will likely be attenuated in streams that are spring-fed, as in the upper Klamath basin at the northern end of the Pit-Klamath brook lamprey range (Quiñones 2011). Pit-Klamath brook lamprey can tolerate high turbidities and persist in seasonally intermittent streams (S. Reid, in Close et al. 2010). They also appear tolerant of higher water temperatures, which are expected to increase due to climate change. Pit-Klamath brook lamprey can tolerate summer water temperatures >25ºC in the Pit River (S. Reid, in Close et al. 2010). Moyle et al. (2013) listed the Pit-Klamath brook lamprey as "highly vulnerable" to extinction as the result of climate change by 2100; however, little is understood both about the biology of this lamprey and the potential effects of climate change on aquatic systems in the arid Pit River basin, so this rating was applied with a low degree of certainty.

Status Determination Score = 3.7 - Moderate Concern (see Methods section, Table 2). Pit-Klamath brook lamprey appear to be common throughout their range in California. However, their actual abundance is unknown. Pit-Klamath brook lamprey are subject to multiple stressors (Table 1) that can create adverse habitat conditions. NatureServe classifies Pit-Klamath brook lamprey as secure to vulnerable throughout their range.

Metric	Score	Justification
Area occupied	5	Range limited to Pit River drainage in California, but includes several tributary systems (e.g. Fall River)
Estimated adult abundance	3	Species is thought to be abundant within range but actual numbers are unknown
Intervention dependence	4	Long-term management of agriculture and grazing practices, as well as alien species, may be warranted
Tolerance	3	Pit-Klamath brook lamprey apparently tolerate warmer temperatures than other lamprey species but still require cool, clean water
Genetic risk	5	Thought to be genetically diverse, although populations in Goose Lake and Klamath basin may constitute separate species
Climate change	2	Some habitats may dry more extensively or for longer durations; ammocoetes may be displaced by unusually high flows
Anthropogenic threats	4	See Table 1
Average	3.7	26/7
Certainty (1-4)	1	Species is largely unstudied

Table 2. Metrics for determining the status of Pit-Klamath brook lamprey in California, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: Habitat degradation from agricultural and grazing practices poses the greatest threat to Pit-Klamath brook lamprey, effects likely to be exacerbated by increasing temperatures and more frequent flood events predicted by climate change models. Watershed management strategies exist (e.g., Pit RCD 2006, Klamath Basin Restoration Agreement) that address these and other factors that may limit fish populations in the Pit and upper Klamath basins. Beyond implementation of these strategies, basic life history studies and population monitoring should occur in order to better understand the status of this species. The following questions should be addressed as part of a status evaluation:

1) Are brook lampreys in the Pit River-Goose Lake and Klamath basins separate taxa?
2) What is the current distribution and abundance of Pit-Klamath brook lamprey in California?
3) Where are most important spawning and rearing grounds located in California?
4) What are the optimal and preferred environmental tolerances and habitat conditions for each life history stage?

Figure 1. Distribution of Pit-Klamath brook lamprey, *Entosphenus lethophagus,* in California.

NORTHERN GREEN STURGEON
***Acipenser medirostris* (Ayres)**

Status: High Concern. Very little is known about the current size of the single northern green sturgeon population in California. However, habitat degradation and climate change continue to threaten their status.

Description: Sturgeons, with their large size, subterminal barbeled mouths, lines of bony plates (scutes), and heterocercal (shark-like) tail, are among the most distinctive of freshwater fishes. Green sturgeon have 8-11 scutes in the dorsal row, 23-30 in the lateral rows, and 7-10 in the bottom rows. The dorsal fin has 33-36 rays, and the anal fin 22-28. They are distinguished from white sturgeon, with which they co-occur, by: (1) having one large scute behind the dorsal and anal fins, (2) having scutes that are sharp and pointed, and (3) having barbels that are closer to the mouth than to the tip of the long, narrow snout (Moyle 2002). Their color is olive-green to pale brown, with an olivaceous stripe on each side and scutes that are paler than the body.

Taxonomic Relationships: Green sturgeon were described from San Francisco Bay in 1854 by W. O. Ayres as *Acipenser medirostris*, the only one of three species he described from the Bay that is still recognized. Green sturgeon are tetraploids and have lower fecundity and larger eggs than most other sturgeon (Gessner et al. 2007). The zoogeographic origin of green sturgeon is uncertain; evidence can be mounted for either an Asian or North American ancestry (Artyukhin et al. 2007). The closest relative is the Asian green sturgeon, *Acipenser mikadoi*, described from one poorly preserved specimen (Jordan and Snyder 1906). Schmidt (1950) designated the Asian form (the Sakhalin sturgeon in the Russian literature) as a distinct subspecies, *Acipenser medirostris mikadoi*. DNA measurements show that the Asian form has approximately twice the DNA content of the North American form (Birstein 1993), indicating that *A. mikadoi* is distinct from *A. medirostris*. Recent comparisons found considerable differences in the morphometrics (e.g., snout length measurements) of Asian and North American populations, although meristic counts overlapped one another (North et al. 2002). Birstein (1993) also suggested that there may be considerable genetic difference between California populations of *A. medirostris* and those north of California. Subsequent analysis of North American green sturgeon found genotypic differences between individuals in the Rogue and Klamath rivers from those in the Sacramento River (Israel et al. 2004). This has led to the split of green sturgeon into two Distinct Population Segments (DPS): southern (Sacramento) green sturgeon DPS and northern green sturgeon DPS (Adams et al. 2002, Adams et al. 2007). The National Marine Fisheries Service has designated populations from the Rogue (Oregon), Klamath-Trinity, Eel, and Umpqua (Oregon) rivers as constituting the northern DPS (Adams et al. 2002, Adams et al. 2007). The population in the Sacramento River has been designated as the southern DPS. In this report, the northern DPS of the green sturgeon is referred to as northern green sturgeon.

Life History: The recent recognition of green sturgeon as having two distinct populations (northern and southern DPS) is confounded by the fact that individuals from both populations likely interact in the ocean; therefore, most studies of ecology and behavior do not separate the two forms outside their native rivers. Until the listing of the southern green sturgeon DPS in 2006, the ecology and life history of green sturgeon had received little study because of their generally low abundance and their low commercial and sport-fishing value. Adults are more marine than white sturgeon but can spend up to six months in fresh water (Benson et al. 2007, Erickson et al. 2002).

Spawning populations of northern green sturgeon are confirmed only for the Rogue (Oregon) and Klamath rivers. Green sturgeon migrate up the Klamath River between late February and late July. The spawning period is March-July, with a peak from mid-April to mid-June (Emmett et al. 1991, Van Eenennaam et al. 2006, Benson et al. 2007). Although the spawning period is similar in the Rogue River, post-spawn adults are found in fresh water in both spring and fall (Webb and Erickson 2007). Spawning females are generally larger, heavier, older and in better condition than spawning males (Van Eenennaam et al. 2006, Benson et al. 2007, Erickson and Webb 2007). From 1999 to 2003, the length of spawning females in the Klamath River was 151-223 cm FL, while males measured 139-199 cm FL. In the Rogue River, male and female green sturgeon become sexually mature at 145 cm TL and 166 cm TL, respectively (Erickson and Webb 2007). Most females were 19-34 years old, while males were 15-28 years old. Males are slightly more abundant than females in spawning runs (female:male = 1:1.4). Adults in the Klamath River exhibit four distinct migration patterns characterized by varying lengths of freshwater residency of up to 199 days (Benson et al. 2007). Individuals migrate at rates of 1.18 to 2.15 km per day. Adults do not appear to spawn in successive years but, rather, at intervals of two or more year (Erickson and Webb 2007, Webb and Erickson 2007).

According to Moyle (2002, p. 110): "Spawning takes place in deep, fast water. In the Klamath River, a pool known as The Sturgeon Hole (Humboldt County) apparently is a major spawning site, because leaping and other behavior indicative of courtship and spawning are often observed there during spring and early summer." Female green sturgeon produce 51,000-224,000 eggs (Adams et al. 2002) which have an average diameter of 4.3 mm (Van Eenennaam et al. 2006). Based on their similarity to white sturgeon, green sturgeon eggs probably hatch around 196 hours (at 13°C) after spawning and the larvae should be 8-19 mm long (Gisbert and Doroshov 2006); juveniles likely range in size from 2.0 to 150 cm TL (Emmett et al. 1991). Morphological (large pectoral fins) and behavioral (rostral wedging) traits allow smaller green sturgeon to hold in rivers for extended periods of time (Allen et al. 2006). Juvenile green sturgeon appear to be largely nocturnal in their migratory, feeding and rearing behavior during the first 10 months of life (Kynard et al. 2005). Green sturgeon retinas are dominated by rods, supporting the idea that they are adapted to live in dim environments (Sillman et al. 2005).

Most juveniles migrate out to sea before two years of age, primarily during summer through fall (Emmett et al. 1991, Allen et al. 2009). Length-frequency analyses of northern green sturgeon caught in the Klamath Estuary by beach seine indicate that most green sturgeon leave the system at lengths of 30-60 cm, when they are 1 to 4 years old, although the majority apparently leave as yearlings (USFWS 1982). Although juvenile green sturgeon can withstand brackish (10 ppt) water at any age, their ability to osmoregulate in salt water develops around 1.5 years of age (Allen and Cech 2007). In the ocean, adults make annual migrations northward in the fall and southward in the spring (Lindley et al. 2008). Important overwintering habitats have been identified between Cape Spencer, Alaska and Vancouver Island. Adults can migrate more than 50 km per day during return spring migrations. Individuals from all spawning populations are known to congregate at Willapa Bay, Washington in the summer (Moser and Lindley 2007).

Northern green sturgeon grow approximately 7 cm per year until they reach maturity at 130-140 cm TL, around age 15-20 years. Thereafter, growth slows. The maximum size is presumed to be around 230 cm TL (USFWS 1982). The oldest fish known are 42 years, based on annuli of fin rays, but the largest fish are probably much older (T. Kisanuki, pers. comm., 1995). Juveniles and adults are benthic feeders on both invertebrates and fish. Adult sturgeon caught in Washington feed mainly on sand lances (*Ammodytes hexapterus*) and callianassid shrimp (P. Foley, pers. comm., 1992). In the Columbia River estuary, green sturgeon are known to feed on anchovies and, perhaps, on clams (C. Tracy, minutes to USFWS meeting). Adults may optimize growth in the summer by feeding on burrowing shrimp in the relatively warmer waters of Washington estuaries (Moser and Lindley 2007).

Habitat Requirements: The habitat requirements of northern green sturgeon are not well studied, but spawning and larval ecology are probably similar to that of white sturgeon. Preferred spawning substrate is likely large cobble, but can range from clean sand to bedrock (Nguyen and Crocker 2007). Eggs are broadcast-spawned and externally fertilized in relatively fast water at depths >3 m (Emmett et al. 1991). Excessive silt can prevent embryos from adhering to one another (Gisbert et al. 2001). Sand can impair the growth and survival of larval green sturgeon by decreasing feeding effectiveness (Nguyen and Crocker 2007).

Temperature appears to be closely linked to migration timing. In the Rogue River, adults enter freshwater from March through May, when water temperatures range from 9 to 16 °C (Erickson and Webb 2007). Adults may hold in deep (>5 m) pools with low velocities after spawning for up to six months (Erickson et al. 2002, Benson et al. 2007). Adult river outmigration initiates with low river temperatures (< 12 °C) and increases in flow (>100 cms). Juveniles appear to prefer dark, deep pools with large rock substrate during winter rearing (Kynard et al. 2005). Nocturnal downstream migration by juveniles continues until water temperatures decrease to about 8°C (Kynard et al. 2005).

Temperature has a major influence on green sturgeon physiology and survival. The upper thermal limit for developing embryos is 17- 18 °C (Van Eenennaam et al. 2005). Incubation temperatures above 22°C result in deformities (Mayfield and Cech 2004, Werner et al. 2007) and/or mortality (Van Eenennaam et al. 2005) of developing embryos. Although age 1 to 3 year old green sturgeon appear to tolerate moderate changes in water temperatures (Kaufman et al. 2007), optimal temperatures for age 1 juvenile sturgeon range from 11 to 19°C. In this same age group, temperatures between 19 and 24°C increase metabolic costs, while temperatures above 24 °C cause severe stress (Mayfield and Cech 2004). However, the metabolic costs associated with temperatures between 19 and 24 °C may be offset when food and oxygen are abundantly available, resulting in unimpaired growth (Allen et al. 2006). Kaufman et al. (2006) determined that juvenile green sturgeon are limited in their ability to handle increases in CO_2. Time of day, length of exposure to a given stressor, and temperature affect the ability of green sturgeon juveniles to respond to stress (Lankford et al. 2003, Werner et al. 2007).

Distribution: Green sturgeon have been caught in the Pacific Ocean from the Bering Sea to Ensenada, Mexico, a range which includes the entire coast of California. However, except for a few tagged fish, it is not known from which river(s), or DPS, ocean-caught sturgeon originate. Migrations generally follow northern routes along shallow waters within the 110 m contour, with individuals from all populations congregating in Willapa Bay, Washington (Moser and Lindley 2007). There are records of green sturgeon from rivers in British Columbia south to the Sacramento River. There is no evidence of green sturgeon spawning in Canada or Alaska, although small numbers have been caught in the Fraser, Nass, Stikine, Skeena and Taku rivers, British Columbia (COSEWIC 2004). Green sturgeon are common in the Columbia River estuary and were observed as far as 225 km inland in the Columbia River, prior to the construction of

Bonneville Dam (Wydoski and Whitney 1979). They apparently do not spawn in the Columbia River or other rivers in Washington, although Israel (2004) discussed genetic evidence for a distinct Columbia River population. In Oregon, juvenile green sturgeon have been found in several coastal rivers (Emmett et al. 1991) but spawning is confirmed only in the Rogue River (Erickson et al. 2002, Erickson and Webb 2007). For northern green sturgeon, spawning has been confirmed in recent years only in the Klamath and Rogue rivers (Moyle 2002, Adams et al. 2007). However, repeated observations of small numbers of adult and juvenile green sturgeon in the Eel River since 2002 suggest spawning may have resumed there after decades of spawning absence (Higgins 2013). There is some evidence of occasional spawning in the Umpqua River (Farr and Kern 2005). Overall, it is likely that northern green sturgeon once spawned in the larger coastal rivers from the Eel River in California north to the Columbia River in Oregon/Washington. Today, the Klamath River is presumed to be the principal spawning river, based on size, flow/temperature regime, and habitat availability.

The following distributional information on northern green sturgeon in California waters was compiled by Patrick Foley (University of California, Davis 1992) and updated with information in Adams et al. (2007).

North Coast. From the Eel River northward, it is likely that most records of sturgeon caught in rivers and estuaries refer to northern green sturgeon. However, most early references regarding sturgeon from the north coast did not identify the species and some reports indicated white sturgeon to be more abundant (Fry 1979). While white sturgeon do occur on occasion in the Klamath and other rivers, it is highly likely that most historic records are for northern green sturgeon. Nineteenth century newspapers (The Humboldt Times) report sturgeon from the mainstem Eel River, South Fork Eel River and Van Duzen River (Wainwright 1965). Length and weights given in these newspaper accounts are most consistent with those of adult green sturgeon.

In the 1950s, two young northern green sturgeon were collected in the mainstem Eel River and large sturgeon were observed jumping in tidewater (Murphy and DeWitt 1951). Two additional young green sturgeon (101 mm and 123 mm) were taken by CDFW from the Eel River in 1967 and are now in the fish collection at Humboldt State University. Substantial numbers of juveniles were caught by CDFW in the mainstem Eel River during trapping operations from 1967-1970 (O'Brien et al. 1976): 22 at Eel Rock in 1967, 53 at McCann in 1967 and 161 in 1969, 221 at Fort Seward in 1968, and smaller numbers at other localities. Green sturgeon have been included in lists of natural resources found in the Eel River delta (Monroe and Reynolds 1974, Blunt 1980). Adult green sturgeon are still occasionally seen in the Eel River (Adams et al. 2007). Higgins (2013) compiled seven records of green sturgeon, usually in groups, observed in the Eel River since 2002 and suggested they are now spawning in the river again. Adams et al. (2007) list the Eel River as a site of “suspected spawning.”

Records of sturgeon in the Humboldt Bay system, comprising Arcata Bay to the north and Humboldt Bay to the south, are almost exclusively green sturgeon. Ten years of trawl investigations in south Humboldt Bay produced three green sturgeon (Samuelson 1973). Records from Arcata Bay are more numerous. On August 6 and 7, 1956, 50 green sturgeon were tagged in Arcata Bay by CDFW biologist Ed Best (D. Kohlhorst, pers. comm.). Total length ranged from 57.2 cm to 148.6 cm with a mean TL of 87.0 cm (± 20.6 cm SD). In 1974, nine green sturgeon were collected over a two-month period in Arcata Bay (Sopher 1974). Total length of these fish ranged between 73-112 cm TL. The Coast Oyster Company, Eureka, pulls an annual series of trawls in Arcata Bay in order to decrease the abundance of bat rays, *Myliobatis californica*. Green sturgeon are incidentally taken in this operation. Eight green sturgeon collected for parasite evaluation in 1988 and 1989 had total lengths ranging between 78-114 cm. One large individual, 178 cm TL and 18.2 kg, was returned to the bay. In 2007, green sturgeon tagged with acoustic tags were detected moving in and out of Humboldt Bay by an array set up to study the movements of coho salmon (S. Lindley, USFWS, unpublished report). Both northern and southern green sturgeon use Humboldt Bay during spring and fall (S. Lindley, pers. comm. 2009) as summarized in Tables 1-3.

Northern green sturgeon have been reported from the Mad River (Fry 1979), but evidence of their recent presence is scant (Bruce Barngrover, pers. comm. 1992). One adult was trapped in the lower river near Mad River Hatchery and rescued by CDFW biologists in 2005 (M. Gilroy, pers. comm. 2011). A carcass was also found in July, 2010 (T. Moore, file report, CDFG, 2010). California Department of Fish and Wildlife biologists D. McLeod and L. Preston observed a 1+ m long sturgeon, most likely a green sturgeon, in a gravel extraction trench in the mainstem Mad upstream of the Blue Lake Bridge (river mile 16) on May 20, 1992.

An occasional green sturgeon is encountered in the coastal lagoons of Humboldt County (Terry Roelofs, pers. comm. 1992). Big Lagoon and Stone Lagoon are connected to the ocean during part of the year and migrating sturgeon may gain entry at this time. In June, 1991, a 120-cm TL green sturgeon was gillnetted in Stone Lagoon (Terry Roelofs, pers. comm. 1992).

Green Sturgeon Tag Code	*Tagging Origin*	*First Detection*	*Last Detection*	***Number of Detections***
0111	Rogue River	July	July	20
0907	**San Pablo Bay**	**June**	**August**	**1,391**
0918	**San Pablo Bay**	**September**	**October**	**5,995**
0933	San Pablo Bay	September	September	5
0989	**San Pablo Bay**	**June**	**September**	**6,660**
1004	**San Pablo Bay**	**September**	**September**	**4**
1008	**San Pablo Bay**	**September**	**September**	**15**
1072	Rogue River	August 6	October	10,218
1127	Willapa Bay	August	August	22
1138	**Willapa Bay**	**June**	**October**	**3,401**
1187	Grays Harbor	June	July	45

Table 1. Green sturgeon detections in 2006, Humboldt Bay, California, recorded on acoustic receiver network maintained by Arcata Fish and Wildlife Office, U.S. Fish and Wildlife Service. Tag codes in bold were detected both in 2006 and 2007. (Provided by W. Pinnix, USFWS, 2012). No fish were tagged in Humboldt Bay.

Green Sturgeon Tag Code	*Tagging Origin*	*First Detection*	*Last Detection*	***Number of Detections***
0151	Sacramento River	July	August	196
0182	Sacramento River	July	August	29,327
0223	Sacramento River	May	July	15,467
0897	San Pablo Bay	July	August	624
0903	San Pablo Bay	July	July	3
0906	San Pablo Bay	July	September	1,186
0907	**San Pablo Bay**	**May**	**August**	**9,033**
0918	**San Pablo Bay**	**July**	**September**	**19,077**
0982	San Pablo Bay	July	July	83
0989	**San Pablo Bay**	**April**	**July**	**625**
0990	San Pablo Bay	July	October	15,019
0995	San Pablo Bay	September	September	39
1004	**San Pablo Bay**	**July**	**July**	**3**
1008	**San Pablo Bay**	**July**	**July**	**73**
1138	**Willapa Bay**	**May**	**September**	**16,938**
1144	Willapa Bay	July	July	344
1147	Willapa Bay	July	July	3
1173	Grays Harbor	May	May	384
1180	Grays Harbor	June	June	241
1182	Grays Harbor	June	June	275
2203	San Pablo Bay	May	August	128
2216	San Pablo Bay	August	August	17
2220	San Pablo Bay	April	July	135
2222	San Pablo Bay	July	October	5,874
2225	San Pablo Bay	September	September	15

Table 2. Green sturgeon detections in 2007, Humboldt Bay, California, recorded on acoustic receiver network maintained by Arcata Fish and Wildlife Office, U.S. Fish and Wildlife Service. Tag codes in bold were detected both in 2006 and 2007. (Provided by W. Pinnix, USFWS, 2012). No fish were tagged in Humboldt Bay.

Green Sturgeon Tag Code	*Tagging Origin*	*First Detection*	*Last Detection*	***Number of Detections***
0219	Sacramento River	June	August	793
0223	**Sacramento River**	**September**	**September**	**12,302**
0238	Sacramento River	September	September	1
0438	Sac???	September	September	3
0906	**San Pablo Bay**	**June**	**June**	**1,637**
0907	**San Pablo Bay**	**May**	**August**	**7,415**
0913	San Pablo Bay	June	September	16,705
0918	**San Pablo Bay**	**September**	**September**	**2,971**
0979	San Pablo Bay	September	September	3
0984	San Pablo Bay	July	July	24
0985	San Pablo Bay	August	August	88
0989	**San Pablo Bay**	**March**	**March**	**3**
0990	**San Pablo Bay**	**August**	**September**	**9,763**
1005	San Pablo Bay	August	August	1
1138	**Willapa Bay**	**June**	**September**	**6,827**
1144	**Willapa Bay**	**August**	**August**	**165**
1153	Willapa??	July	July	1
2203	**San Pablo Bay**	**May**	**May**	**3**
2210	San Pablo Bay	August	August	174
2212	San Pablo Bay	August	September	425
2217	San Pablo Bay	June	August	415
2225	San Pablo Bay	September	September	15

Table 3. Green sturgeon detections in 2008, Humboldt Bay, California, recorded on acoustic receiver network maintained by Arcata Fish and Wildlife Office, U.S. Fish and Wildlife Service. Tag codes in bold were detected both in 2007 and 2008. (Provided by W. Pinnix, USFWS, 2012). No fish were tagged in Humboldt Bay.

<u>Klamath and Trinity rivers</u>. The largest spawning population of northern green sturgeon in California is in the Klamath River basin. Both green sturgeon and white sturgeon have been found in the Klamath River estuary (Snyder 1908b, USFWS 1980-91), but white sturgeon are taken infrequently in very low numbers and are presumed to be coastal migrants (USFWS 1982). Almost all sturgeon found above the estuary during systematic sampling have been green sturgeon (USFWS 1980-83). Green sturgeon primarily use the mainstem Klamath River and mainstem Trinity River but have also been seen in the lower portions of the Salmon River (Adams et al. 2007).

Both adult and juvenile northern green sturgeon have been identified in the mainstem Klamath River. Adults are taken annually from spring through summer by an in-river tribal gillnet fishery. The numbers taken are between 200 and 750 fish per year (Table 5). They have also been taken by sport fishermen as far inland as Happy Camp (river km 172; unpubl. CDFW tagging data 1969-73, Fry 1979, USFWS 1981). The apparent upstream limit for spawning migration is Ishi Pishi Falls, Siskiyou County, at approximately river km 113. A few juveniles have been taken as high up as Big Bar at river km 81 (Tom Kisanuki, pers. comm. 1995) but most have been recovered by seining operations directed at salmonids in the estuary (USFWS, CDFW). Sampling by the USFWS captured 7 juveniles in 1991 and 23 in 1992 (T. Kisanuki, pers. comm. 1995). Six outmigrant traps placed in the Klamath River caught juvenile green sturgeon every year (2000-2005) (Cunanan and Hines 2006, USFWS, unpublished data). The number of green sturgeon captured each year varied from one (2005) to 775 (2003). The total number of juvenile green sturgeon captured over the six years of operation was 1599, with sizes varying from 20 mm to 252 mm TL and averaging 68.5 mm TL. Green sturgeon captured by the traps were most likely juveniles ranging in age from a couple of weeks to less than two years old, based on growth curves developed by Nakamato et al. (1995) and Van Eenennamm et al. (2001). The average size (69 mm TL) was similar to the size of artificially reared Klamath River green sturgeon at 35 days old (66 mm; Van Eenennaam et al. 2001).

The Trinity River enters the Klamath River at Weitchpec (river km 70). The first green sturgeon described from the Klamath basin came from the Trinity River (Gilbert 1897). Both adults and juveniles have been identified; 211 green sturgeon, between 7-29 cm TL, were captured in screw traps near Willow Creek, Humboldt County, incidental to a salmonid migration study in July-September, 1968 (Healey 1970). The USFWS has collected small numbers of juvenile green sturgeon from the Trinity River, as far up as Big Bar (T. Kisanuki, pers. comm. 1992). Adults are caught yearly in a tribal gillnet fishery (USFWS 1980), a traditional fishery with a long history (Kroeber and Barrett 1960). Spawning adults migrate the mainstem Trinity River up to about Grays Falls, Burnt Ranch, Trinity County (river km 72).

Northern green sturgeon have also been reported to use the South Fork Trinity River, a third-order stream entering above Willow Creek (river km 51) (USFWS 1981), according to oral histories from long-time residents. However, a large flood in 1964 had devastating effects on anadromous fish habitat in this subbasin (U.S. Department of the Interior 1985). Millions of cubic yards of soil were moved into South Fork Trinity River and its tributaries, with resulting channel widening and loss of depth in many areas. This event, along with other changes in basin morphology, has apparently resulted in the loss of suitable sturgeon habitat. There are no recent records of green sturgeon from this watershed.

The Salmon River is a fourth-order stream entering the Klamath River at Somes Bar (river km 106). Adult green sturgeon have been observed upstream as far as the mouth of Wooley Creek (river km 8).

Del Norte County. Northern green sturgeon have been taken during gillnet sampling in Lake Earl (D. McLeod, pers. comm.). Lake Earl is located along the coast of Del Norte County, 8 km north of Crescent City and 11 km south of the mouth of Smith River. Lake Earl is connected to Lake Talawa, a smaller lake directly to the west. A sand spit separates Lake Talawa from the ocean and is occasionally breached by winter storms or mechanically per the Lake Earl Wildlife Area Management Plan. Coastal migrant green sturgeon may enter at this time and become trapped after the sand spit is reestablished (Monroe et al. 1975).

The Smith River is the northernmost river along the California coast, entering the ocean approximately 5 km south of the Oregon border. Blunt (1980) included green sturgeon in an inventory of anadromous species found in the Smith River. They occasionally enter the estuary and have been observed in Patrick's Creek, an upstream tributary 53 km from the ocean (Monroe et al. 1975). Juveniles have not been found in the Smith drainage.

Trends in Abundance. Although northern green sturgeon apparently occur in fewer streams than they did historically, trends in abundance are poorly understood (Adams et al. 2002). The only time series data available for adult green sturgeon abundance in the Klamath River comes from tribal catch data (see below). The number of females spawning in the Klamath River is estimated at 760-1500 per year. The population of subadults-adults is estimated at tens of

thousands, with no clear evidence of population decline (Adams et al. 2002).
However, northern green sturgeon abundance and population trends remain largely unknown and should be treated conservatively until information indicates otherwise because:

(1) Virtually all other sturgeon species are in decline. Rochard et al. (1990) state in their review of the status of sturgeons worldwide: "Those [species of sturgeon] which do not have particular interest to fishermen (*A. medirostris, Pseudoscaphirhynchus* spp.) are paradoxically most at risk, for we know so little about them" (p. 131). The southern green sturgeon is listed as a threatened species.

(2) The only confirmed spawning populations of northern green sturgeon are in the Klamath and Rogue (Oregon) rivers, both of which have flow and temperature regimes affected by water projects and, potentially, climate change. It is highly probable that these are now the only spawning populations in North America, although recent reports from the Eel River are promising.

(3) Green sturgeon are subject to legal, illegal, and by-catch fisheries. It is likely that these fisheries depend largely on sturgeon from the Klamath River. The various fisheries, including past sport fishing, have harvested at least 6,000 to 11,000 green sturgeon per year. Studies have shown that green sturgeon populations are sensitive to overharvest (Heppell 2007).

Nature and Degree of Threats: Green sturgeon depend on large rivers so their populations are subject to numerous anthropogenic stressors that occur across large geographic areas, as described below (see Table 4).

Major dams. The Klamath, Trinity and Rogue (Oregon) rivers all have flows regulated by major dams. Apparently, the impact of these dams upon green sturgeon has been minimal perhaps because spawners tend to be in the river when flows are highest and because all life stages mainly live in the lowermost reaches, where dam impacts are reduced. However, a single green sturgeon was part of a large fish kill in the lower Klamath River in September, 2002, which has been attributed partially to the operation of Iron Gate Dam (Belchick et al. 2004), suggesting at least some vulnerability.

Grazing, roads, logging. Land use practices, such as road building, logging and grazing have all changed the quality of spawning and rearing habitats in large mainstem rivers by increasing sediment loads, impairing water quality and otherwise reducing habitat suitability. Thus, it is likely that optimal conditions (especially temperature, flow, and stream substrate composition) for spawning and rearing of green sturgeon occur less frequently now than they once (pre-1940s) did, especially during or after periods of extended drought. Of particular concern is siltation of river portions used for spawning and incubation of embryos, although the timing and location of spawning tends to reduce the probability that this is a factor in survival. The huge 1964 floods may have severely degraded many areas of sturgeon spawning and rearing habitat, perhaps eliminating this species from rivers, or tributaries thereof, such as the Eel and South Fork Trinity.

Estuary alteration. While the Klamath River estuary is relatively unmodified, other California estuaries such as those of the Eel and Smith rivers have been diked and drained for pasture or other land uses. This degradation of key rearing areas may have contributed to reductions or loss of green sturgeon and other anadromous fishes from these rivers (Yoshiyama and Moyle 2010).

Harvest. Although California anglers were prohibited from taking or possessing green sturgeon beginning in 2007, the legacy of past fishing practices may still be impacting populations today due to the species' longevity and infrequency of spawning. Of particular concern is removal of adult females from the population, which have the highest fecundity and, therefore, the greatest potential for replenishing depleted populations. The following are accounts of the two principal fisheries that may have affected green sturgeon in the northern DPS:

	Rating	Explanation
Major dams	Medium	Major dams present on all spawning rivers; however, effects are largely unknown
Agriculture	Low	Minor influence on lower Klamath and Eel rivers; alfalfa pastures for grazing widespread in the Smith estuary
Grazing	Low	Pervasive in watersheds but probably little effect on large river habitats
Rural Residential	Low	Pervasive in watersheds but probably little effect on large river habitats
Urbanization	Low	No large urban areas within known distribution
Instream mining	Low	Gravel mining and gold dredging may increase fine sediment mobilization in rivers; greater historic impact
Mining	Low	No known impact but some dredging in range (currently suspended in California)
Transportation	Medium	Roads are a source of sediment that may affect spawning
Logging	Medium	Major source of sediment from extensive network of access roads; greater historic impact
Fire	Low	Wildfires are common within the range of northern green sturgeon but impacts are not well understood
Estuary alteration	Medium	Smith and Eel estuaries are altered and have reduced capacity for rearing juvenile sturgeon
Recreation	Low	No known impact but boating may disturb fish
Harvest	Medium	Adults taken in fisheries for many years but impacts not well understood
Hatcheries	n/a	
Alien species	n/a	

Table 4. Major anthropogenic factors limiting, or potentially limiting, viability of populations of northern green sturgeon. Factors were rated on a five-level ordinal scale where a factor rated "critical" could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated "high" could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction is unlikely as a result. A factor rated "n/a" has no known impact. Certainty of these judgments is moderate. See methods section for descriptions of the factors and explanation of the rating protocol.

Columbia River region. The majority of past northern green sturgeon harvest occurred in this region; they were caught by commercial fishermen, anglers, and Native American gillnetters. Sturgeon landings were recorded from the Columbia River estuary and from Grays

Harbor and Willapa Bay, Washington, to the immediate north of the estuary. There is little or no evidence of green sturgeon spawning in the rivers of this region, so it is likely that sturgeon harvested there migrated from California or Oregon, as indicated by limited recaptures of tagged individuals (Adams et al. 2007). Further evidence of the lack of local recruitment into the fishery is indicated by the fact that few juvenile sturgeon (<1.3 m) have been caught in this region (Emmett et al. 1991).

The commercial catch in the Columbia River region (Columbia River estuary, Grays Harbor, Willapa Bay) has fluctuated considerably over time, but catches appear to have increased in recent decades. Between 1941 and 1951, catches averaged about 200-500 fish per year, while between 1951 and 1971 the catch averaged about 1,400 fish per year (Houston 1988). In the late 1980s, an average of 4.7 tons of green sturgeon (ca. 500-1,000 fish) were harvested each year in Grays Harbor and 15.9 tons (ca. 2,000-4,000 fish) were harvested in Willapa Bay (Emmett et al. 1991). There have also been some notably high catches; in 1986, 6,000 green sturgeon were harvested in the Columbia River estuary (Oregon Dept. of Fish and Wildlife (ODFW) 1991) and 4,900 were taken in 1987 (ODFW, unpubl. data). From the 1960s-1980s, the commercial catch of green sturgeon in the Columbia River has averaged 1,440 fish (1960s), 1,610 (1970s) and 2,360 (1980s); the catch since 1990 has ranged from 3200 fish (1991) to 0 fish (2002) (Adams 2007). The Columbia River recreational catch has been consistently below 200 fish per year since 1988 (ODFW 1991, Adams 2007). For 1985-2003, Adams et al. (2007) estimated annual harvest of green sturgeon from all sources as ranging from 500 to over 9000 fish, with catches since 2001 being less than 1,000 fish per year, mostly taken in Washington. While fishing for green sturgeon is now prohibited in Washington, some mortality from fishing presumably continues as the result of by-catch from other fisheries (Adams et al. 2002). The commercial fishery took both northern and southern green sturgeon; only tagged fish were identified to the appropriate DPS.

<u>Klamath and Trinity rivers</u>. A small number of northern green sturgeon were probably taken in this sport fishery in the past but the main harvest is now by the Yurok, Karuk, and Hupa tribal gillnet fisheries (USFWS 1990, Adams et al. 2005). A small, but possibly significant, number are also taken in an illegal snag fishery. All fisheries target sturgeon as they move upriver to spawn during the spring and as they return seaward through the estuary during June-August (USFWS 1990). In the tribal fishery, mainly adult sturgeon (>130 cm FL) are captured (mean length 179 cm FL in 1988). The percent of the total (sport and tribal) harvest in the Pacific Northwest taken from the Klamath River increased from a low of 5% in 1987 to 59% in 2003 (Van Eenennaam et al. 2006, Table 5). This increase most likely reflected changes in regulations to limit green sturgeon harvest in Oregon and Washington (Adams et al. 2002).

	Klamath River				Total Harvest (CA, OR, WA)	Percent of Total Harvest from Klamath River
Year	Yurok	Hupa	Sport	Total		
1985	351	10	NA	361	5,156	7
1986	421	30	153	604	9,065	7
1987	171	20	170	361	7,669	5
1988	212	20	258	490	6,514	8
1989	268	30	202	500	4,067	12
1990	242	20	157	419	4,736	9
1991	312	13	366	691	6,788	10
1992	212	3	197	412	4,551	9
1993	417	10	293	720	4,267	17
1994	293	14	160	467	1,342	35
1995	131	2	78	211	1,286	16
1996	119	17	210	346	1,692	20
1997	306	7	158	471	3,199	15
1998	335	10	103	448	1,692	26
1999	204	27	73	304	1,491	20
2000	162	31	15	208	1,796	12
2001	268	10	NA	278	862	32
2002	273	5	NA	278	696	40
2003	287	16	NA	303	514	59
2004	222	12	NA	234	NA	NA

Table 5. Green sturgeon harvest numbers and percent of total harvest (California, Oregon and Washington combined) from the Klamath River, California (Source: Adams et al. 2002, Van Eenennaam et al. 2006).

The average total length of northern green sturgeon captured in the Yurok Tribal fishery increased slightly from 1980 to 2004 (Figure 1). Moreover, the proportion of green sturgeon greater than 190 cm increased from 30% in 1995 to approximately 40% in 2004 (D. Hillemeier, Yurok Tribal Fisheries Program, unpublished data). Because the length of captured individuals did not decrease, the Yurok Tribal fishery apparently does not adversely impact the size distribution of spawning adults. However, it is uncertain whether the increase in numbers of large adults signifies a change in population structure towards larger individuals or a loss of younger year classes.

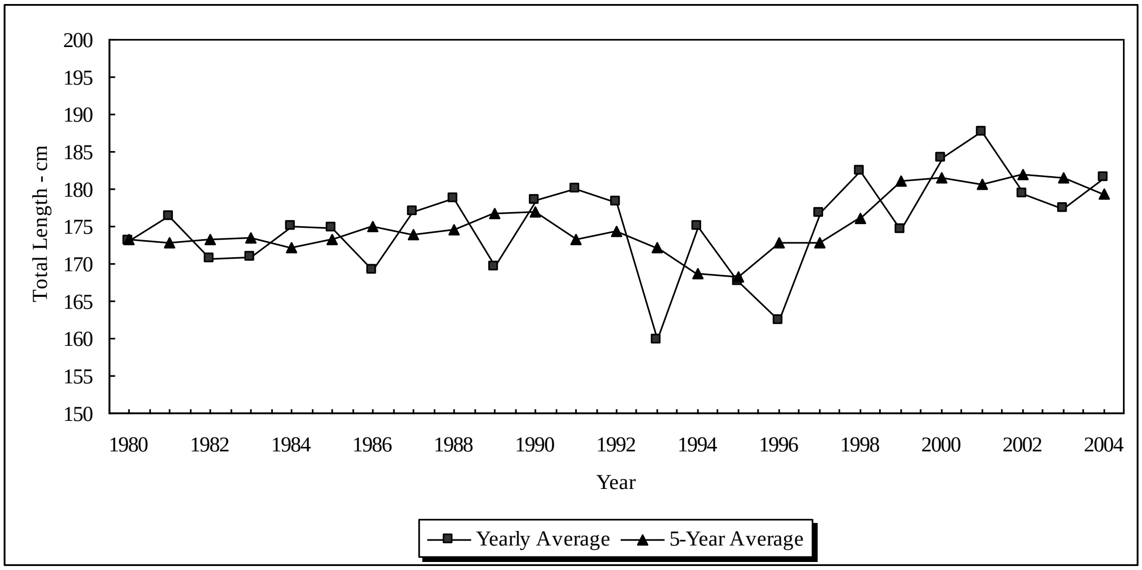


Figure 1. Average total length of northern green sturgeon sampled in the Yurok fishery, 1980-2004 (Source: D. Hillemeier, Yurok Tribal Fisheries Program, unpublished data).

Although present in low numbers, there is no indication that green sturgeon are in decline in the Klamath River basin (Adams et al. 2002, 2005; Beamesderfer and Webb 2002). However, given the status of other anadromous species in the Klamath River basin, the extended freshwater residency of at least some individuals, delayed maturity, and longevity of green sturgeon, there is concern that adverse impacts to the population may not be detected unless they are analyzed at the appropriate time scale (17 to 23 years; D. Hillemeier, unpublished data).

Effects of Climate Change: Increased water temperatures brought about by climate change may place northern green sturgeon under chronic stress that can result in metabolic costs that impair reproduction, growth and immune function (Lankford et al. 2005). Mayfield and Cech (2004) recommended that, in order to enhance growth, management plans should protect green sturgeon from prolonged exposure to temperatures above 19°C. Similarly, Van Eenennaam et al. (2005) concluded that temperatures above 20°C are detrimental to reproduction and most likely result in low hatching success, especially during dry water years. Summer water temperatures in the mainstem Klamath River already frequently exceed 20°C and temperatures in California are expected to increase under all climate change scenarios (Hayhoe et al. 2004, Cayan et al. 2008). Increases in summer temperatures may affect the growth and metabolic costs of juvenile and adult green sturgeon that hold in rivers throughout the summer. Climate change is also predicted to alter the flow regimes in rivers. In the Klamath and Trinity rivers, river flow may peak earlier in the spring and continue tapering through the summer before pulsing again later in the fall. The resulting changes in river flow and temperature may change the timing of adults and juveniles entering and exiting these systems. Quiñones and Moyle (2012) predicted these changes will cause increased declines in anadromous salmonids in the Klamath basin, so negative impacts to green sturgeon are likely as well. Moyle et al. (2013) rated northern green sturgeon as "highly vulnerable" to extinction in California as the result of climate change, largely as a result of increased temperatures and reduced flows in the Klamath River.

Status Determination Score = 2.7 - High Concern (see Methods section, Table 2)**.** Northern green sturgeon merit high concern status, even though they are not in immediate danger of extirpation from California. The Klamath-Trinity River population is the sole reproducing population in California and, apparently, is by far the largest population, giving it added significance. Green sturgeon are considered to be a threatened species in Canada. In 2006, the National Marine Fisheries Service determined that the northern green sturgeon DPS did not warrant listing under the Federal Endangered Species Act (50 CFR part 223); however, it was designated a species of concern (www.nmfs.noaa.gov). Green sturgeon (both DPS's combined) are given a near-threatened status by International Union for Conservation of Nature (IUCN) Red List (www.iucnredlist.org). The southern (Sacramento) DPS of green sturgeon was listed in 2006 as a threatened species under the Federal Endangered Species Act. After the southern green sturgeon was listed, both Oregon and Washington banned take by both commercial and sport fisheries.

Metric	Score	Justification
Area occupied	1	Only Klamath-Trinity population appears to be self-sustaining in California - this would score '2' if Oregon populations were included
Estimated adult abundance	2	Unknown, but 1,000-5,000 adults would be a conservative estimate
Intervention dependence	4	Long-term persistence depends on fisheries management and habitat restoration
Tolerance	3	Fairly tolerant of conditions in the Klamath River although susceptible to warm temperatures
Genetic risk	4	Presumably some genetic connections to Rogue population
Climate change	2	Limited spawning and rearing habitats suggests vulnerability to increased temperatures, reduced summers flows and other climate change-related stressors
Anthropogenic threats	3	Five threats scored 'medium' (see Table 4)
Average	2.7	19/7
Certainty	3	Abundance not well understood but many publications exist on distribution and behavior

Table 6. Metrics for determining the status of northern green sturgeon, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

In California, only one spawning population is recognized in the Klamath River, raising concerns about limited genetic diversity and gene flow. The possibly reproducing population in the Eel River is presumably derived from strays from the Klamath River. Conditions in the Klamath River for spawning and rearing have likely worsened due to the presence of major dams in both the main stem Klamath and Trinity rivers. Dams have dramatically altered the hydrology and geomorphology of these systems (NRC 2004). Degradation of habitats, combined with the predicted effects of climate change, make northern green sturgeon vulnerable to changing environmental conditions and potentially less suitable habitat conditions.

The closure of green sturgeon fishing, except for tribal fisheries, has reduced harvest rates in California. However, the legacy of harvest prior to 2007 may still be impairing the recovery of some populations. Green sturgeon population growth is particularly sensitive to adult and subadult mortality, especially if the effective spawning population size becomes low (Heppell 2007). Large increases in egg production and juvenile survival are required to counterbalance the impact from even relatively low levels of fishing mortality. In addition, recent work (Israel et al. 2004) suggests that not all spawning populations of green sturgeon have been identified, a necessary step for the adequate protection of green sturgeon genetic diversity.

Management Recommendations: The following conservation measures are needed to maintain or increase northern green sturgeon abundances:

1. Detailed studies on life history and ecological requirements are needed. Current population assessment and monitoring by the USFWS, Yurok Tribe, and others should be expanded, particularly for Klamath River populations. The current paucity of information and empirical data about the population status, structure and dynamics of northern green sturgeon means that population trends cannot be predicted, nor stocks properly managed. Females mature relatively late in life and may not spawn every year, so maintenance of sufficient reproductive potential (i.e., numbers of mature females) in populations is an important management consideration.

2. Nursery habitats for juveniles in river and estuarine habitats need to be identified and protected. One method for determining optimal habitats is to examine the digestive tracts of juvenile green sturgeon to evaluate the nutritional condition of fish rearing in different habitats (Gisbert and Doroshov 2003). Shortages of food supply can disrupt the organization and generation of juvenile digestive systems, directly affecting growth and survival.

3. Tribal fisheries that target northern green sturgeon should be limited until more is known about the biology and abundance of this species. At a minimum, special harvest regulations for green sturgeon are needed to reduce the catch of large females of peak reproductive ages of 25 to 40 years old (Heppell 2007). The effect of harvest on population productivity could be reduced by a slot limit to reduce the number of age classes harvested (Heppell 2007).

4. Populations can benefit from habitat restoration, especially of estuaries and lagoons. Measures should be adopted to keep summer water temperatures below 20°C, where possible, and to decrease the input of fine sediments into streams. Both of these measures can enhance the development and subsequent recruitment of juvenile green sturgeon.

5. The effects on northern green sturgeon of the proposed removal of four dams on the Klamath River need to be evaluated, especially in relation to low summer flows (e.g., lack of year-round tailwater flows from controlled dam releases) and with respect to potential for green sturgeon to use habitats made available by dam removal.

Figure 2. Freshwater distribution of northern green sturgeon, *Acipenser medirostris* (Ayres), in California. The only confirmed spawning population is in the Klamath-Trinity river system.

WHITE STURGEON
Acipenser transmontanus (Richardson)

Status: High Concern. Annual recruitment of white sturgeon in California appears to have decreased since the early 1980s but several strong year classes are evident. Continued close management is required to sustain white sturgeon populations into the future.

Description: White sturgeon adults have wide, rounded snouts, with four barbels in a row on the underside, closer to the tip of the snout than to the mouth (Moyle 2002). They feed with a toothless, highly protrusible mouth and process food with a palatal organ in the pharynx. Their bodies have 5 widely separated rows of bony plates (scutes). Scute counts per row are: 11-14 (dorsal row), 38-48 (two lateral rows) and 9-12 (bottom rows). Four to eight scutes are also found between the pelvic and anal fin. Although they lack the large scutes behind the dorsal and anal fins found in green sturgeon (*A. medirostris)*, small remnant scutes (fulcra) may be present. The dorsal fin has one spine followed by 44-48 rays. The anal fin has 28-31 rays. The first gill arch has 34-36 gill rakers. Body coloration is gray-brown on the dorsal surface above the lateral scutes, while the ventral surface is white and fins are gray. Their viscera are black. Dispersing juveniles tend to be darker than dispersing free embryos (Kynard and Parker 2005). Juveniles less than one year old have 42 dorsal fin rays, 35 lateral scutes, and 23 gill rakers on the first arch.

Taxonomic Relationships: Recent genetic analysis supports the close relationship between white sturgeon and Amur sturgeon (*A. schrenckii;* found only in Asia), which had a common ancestor approximately 45.8 million years ago (Peng et al. 2007, Krieger et al. 2008). In California, some genetic differentiation was thought to exist among white sturgeon populations from different river systems (Bartley et al. 1985) but a detailed genetic analysis using microsatellites failed to reveal any such population structure (Schreier et al. 2011). Recent DNA analysis using microsatellites has determined that genetic differentiation ($F_{ST} = 0.19$) is high enough among white sturgeon from the Columbia, Fraser and Sacramento River basins to be able to distinguish them (Rodzen et al. 2004), despite mixing in the ocean and high levels of genetic diversity (Schreier 2011). Schreier (2011) found that sturgeon captured in non-natal estuaries could be assigned by genetic techniques to their natal river, although the high level of genetic diversity found in the three major anadromous sturgeon populations indicates that some mixing of stocks takes place. Nevertheless, there is now sufficient evidence to treat the Sacramento-San Joaquin white sturgeon stock as a Distinct Population Segment (DPS).

Life History: White sturgeon primarily live in estuaries of large rivers but migrate to spawn in fresh water and often make long ocean movements between river systems. They commonly aggregate in deep, soft-bottomed areas of estuaries, where they move about in response to changes in salinity (Kohlhorst et al. 1991). In the lower Columbia River, white sturgeon make seasonal and diel movements (Parsley et al. 2008), moving upstream in the fall and downstream in the spring. They are most active at night, when they move into shallower waters to feed. Some individuals express site fidelity by returning to previously occupied sites (Parsley et al. 2008).

In the ocean, some individuals may migrate large distances. White sturgeon tagged in the San Francisco Estuary have been recaptured in the Columbia River estuary (L. Miller 1972a,b, Kohlhorst et al. 1991). One of these fish was then subsequently recaptured 1,000 km upstream in the Columbia River. Tagged individuals have routinely been detected 1,000 km from the tagging site (Chadwick 1959, Welch et al. 2006). Recently, one white sturgeon tagged in May, 2002, in the Klamath River, was tracked to the Fraser River, British Columbia, a distance far greater than 1000 km (Welch et al. 2006). Because this individual spent nearly equal amounts of time in both the Fraser and Klamath rivers, it was difficult to determine which was the natal river. However, genetic studies suggest that extensive movements are associated with feeding rather than spawning (Schrierer 2011).

In estuaries, white sturgeon move into intertidal areas during high tides to feed. Most prey are taken on or near the bottom. Young white sturgeon (~ 20 cm FL) prefer amphipods (*Corophium* spp.) and opossum shrimp (*Neomysis mercedis*) (Radtke 1966, Muir et al. 1988, McCabe et al. 1993). Diet becomes more varied as they grow but continues to be dominated by benthic invertebrates such as shrimp, crabs, and clams. Today, most benthic invertebrate prey species in the San Francisco Estuary are nonnative, demonstrating the opportunistic feeding nature of white sturgeon (Moyle 2002). One heavily used prey is the overbite clam, *Corbula amurensis*, which became very abundant after its invasion into Suisun Bay in the 1980s. However, foraging on the overbite clam may inhibit growth, because some clams pass through the gastrointestinal tract without being digested, possibly decreasing nutritional intake (Kogut 2008). Fish, especially herring, anchovy, striped bass, starry flounder, and smelt, are consumed by larger sturgeon. In the San Francisco Estuary, white sturgeon feed on Pacific herring eggs (McKechnie and Fenner 1971), much as their Columbia River counterparts do on eulachon eggs (McCabe et al. 1993). In California, stomach contents of large individuals have also included onions, wheat, Pacific lamprey, crayfish, frogs, salmon, trout, striped bass, carp, pikeminnow, suckers and, in one instance, a cat (Carlander 1969).

In the San Francisco Estuary, young sturgeon reach 18-30 cm by the end of their first year (Kohlhorst et al. 1991). Maximum growth is achieved by juvenile white sturgeon grown in captivity on artificial diets, consuming 1.5 to 2% of their body weight each day at 18°C (Hung et al. 1989). As white sturgeon age, growth rates slow so that they reach 102 cm TL by their seventh or eight year. They may ultimately reach 6 m FL. The largest white sturgeon on record weighed 630 kg and was likely more than 100 years old; fish of this size were probably the largest freshwater fish in North America (Moyle 2002). The largest white sturgeon caught in Oregon measured 3.2 m FL and was 82 years old (Carlander 1969). In California, the largest white sturgeon on record was from Shasta Reservoir in 1963; it was 2.9 m TL, 225 kg, and at least 67 years old (T. Healy, CDFW, pers. comm. 2001). Today, in California, white sturgeon larger than 2 m and older than 27 years are uncommon.

Male white sturgeon mature when10-12 years old (75-105 cm FL); females mature later at about 12-16 years old (95-135 cm FL) (Kohlhorst et al. 1991, Chapman et al. 1996). However, males mature at 3-4 years and females at 5 years while in captivity (Wang 1986). Photoperiod and temperature regulate maturation in adult white sturgeon (Doroshov and Moberg 1997). Prior to spawning, adults may move into the lower reaches of rivers during the winter months and later migrate upstream into spawning areas in response to increases in flow (Schaffter 1997a,b). Spawning initiates in response to high flows from late February to early June (McCabe and Tracy

1994). Only a small percentage of adults will spawn in any given year. In the Columbia River, males spawn every 1-2 years while females spawn every 3-5 years (McCabe and Tracy 1994).

Spawning in the Sacramento River occurs primarily between Knights Landing (233 rkm) and Colusa (372 rkm) (Schaffter 1997a,b). A few adults spawn in the Feather and San Joaquin rivers (Kohlhorst 1976, Kohlhorst et al. 1991), although recent activity in the Feather River is unconfirmed (A. Schierer, pers. comm. 2010). Genetic evidence suggests that there is little fidelity to spawning areas within the Sacramento River system (Schierer 2011). The fecundity of females from the Sacramento River averages 5,648 eggs/kilogram body weight, so an individual female (1.5 m TL) may contain 200,000 eggs (Chapman et al. 1996). White sturgeon typically spawn in deep water over gravel substrates or in rocky pools with swift currents. Eggs have been collected from the stream bed at depths of 10 m (Wang 1986). In the Columbia River, white sturgeon spawn over cobble and boulder at depths of 3-23 m and velocities of 0.6-2.4 m/sec (McCabe and Tracy 1994). Adults migrate back to the estuary after spawning.

Eggs (3.5-4.0 mm; in Billard and Lecointre 2001) become adhesive upon fertilization, allowing them to stick to stream substrates. Time to hatch is dependent on temperature but larvae generally hatch in 4-12 days (Wang 1986). Larvae are 11 mm at hatch and swim vertically while drifting towards the estuary. They switch to swimming horizontally and feed from the bottom once the yolk sac is absorbed, in about 7-10 days. Sacramento River white sturgeon larvae were found to be photonegative upon hatching, moving downstream short distances by swimming near the bottom, seeking cover (Kynard and Parker 2005). Larvae aggregated, swam, and foraged near the bottom and demonstrated an increasing trend to swim above the bottom. Strong dispersal occurred as early juveniles swam actively downstream. Consequently, Sacramento River white sturgeon are described as having a "two-step downstream dispersal" completed by larvae and early juveniles during both day and night, but peaking at night. Juvenile sturgeon use the less saline portions of estuaries, suggesting that the ability to osmoregulate increases with age and size (McEnroe and Cech 1987). Osmoregulation efficacy may also be size-dependent, even between individuals of the same age (Amiri et al. 2009). Consequently, size at time of estuary entry may be a limiting factor for juvenile survival. In the lower Fraser River, most juvenile white sturgeon use sloughs from June to August (Bennett et al. 2005); occupied sloughs were more than 5 m deep, turbid, and had multidirectional currents, soft sediments, and readily available prey (mysid shrimp, dipteran larvae, fish).

In the San Francisco Estuary, the white sturgeon population is dominated by a few strong year classes, reflecting variability of annual spawning success. Strong year classes result from years of high spring flows in the rivers (Kohlhorst et al. 1991, Schaffter and Kohlhorst 1999, Fish 2010). High spring flows may quickly move larval sturgeon downstream into suitable rearing areas (Stevens and Miller 1970) or induce more sturgeon adults to spawn (Kohlhorst et al. 1991). In the lower Columbia River, year class strength is correlated to the size and availability of prey at the onset of exogenous feeding (Muir et al. 2000). Amphipods (Corophiidae), copepods, and dipteran larvae and pupae are important prey to larval and young-of-year sturgeon. Predation on larvae, especially by prickly sculpin, may be another factor limiting recruitment in some areas (Gadomski and Parsley 2005, Gadomski and Parsley 2005b).

Habitat Requirements: White sturgeon adults respond to increases in flow to initiate spawning from late February to early June. Spawning takes place at temperatures ranging from 8 to 19°C, peaking at temperatures around 14°C (McCabe and Tracy 1994). Successful incubation requires stream substrates with minimum amounts of sand and silt because excessive siltation can smother embryos. Recruitment failure in the Nechako River, Canada, was attributed to siltation of main channel sediments after large scale (1,000,000 m^3) introduction of fine sediments by upstream stream avulsion (McAdam et al. 2005). The recruitment failure was attributed to egg suffocation and increased predation because larvae lacked interstitial spaces in the substrate in which to hide. Newly hatched embryos preferred substrates from 12 to 22 mm in laboratory tests (Bennett et al. 2007).

The first few months of life are considered to be critical for sustaining populations (Coutant 2004). Successful recruitment also appears to be associated with complex habitats, flooded riparian vegetation (floodplain habitat) and rocky substrates (Coutant 2004). Lack of cover in edge habitats downstream of spawning areas, along with low flows from the time of spawning until juvenile outmigration, decreases recruitment. Productive spawning areas in the Sacramento River are associated with areas where levees are set back, allowing access to floodplains and backwater habitats (e.g., Wilkins and Butte sloughs) during high spring flows.

Distribution: White sturgeon can be found in salt water from the Gulf of Alaska south to Ensenada, Mexico. However, spawning only occurs in a few large rivers from the Sacramento-San Joaquin system northward. Self-sustaining spawning populations are currently only known in the Fraser (British Columbia), Columbia (Washington), and Sacramento (California) rivers. Landlocked populations also occur above major dams in the Columbia River (McCabe and Tracy 1994). White sturgeon from California are caught in small numbers in the Columbia River and other estuaries (Schierer 2011). At least one white sturgeon tagged in the Klamath River spent extensive time in the Fraser River (Welch et al. 2006).

In California, white sturgeon spawn primarily in the Sacramento River (to Keswick Dam) but may also spawn in the San Joaquin River (Jackson and Van Eenennaam 2013) and in the Feather River (to Oroville Dam facilities), when water quality and flow conditions are favorable (Schaffter 1997a,b). The lower Pit River was likely an important spawning area, prior to construction of Shasta Dam in the 1940s (T. Healey, CDFW, pers. comm. 2001). Sturgeon became trapped behind Shasta Dam, establishing a landlocked population that became self-sustaining and supported a small fishery (Moyle 2002). However, subsequent dam construction on the Pit River blocked access to spawning areas and prevented ongoing reproduction of this population (T. Healey, CDFW, pers. comm. 2001). Long-lived individuals and fish from stocking attempts in the 1980s are still occasionally caught in Shasta Reservoir. Historically, small runs also occurred in the Russian, Klamath and Trinity rivers. White sturgeon have also been documented in the Eel River (M. Gilroy, CDFW, pers. comm. 2011). It is doubtful that any of these latter four rivers currently support populations of white sturgeon.

Aquaculture facilities now cultivate white sturgeon in California and juvenile sturgeon can be sold to aquarists. Presumably, aquarium releases have resulted in occasional white sturgeon being found in reservoirs in southern California (C. Swift, pers. comm. 1999) and the San Francisco region (e.g., a 21 kg individual caught in Lafayette Reservoir, Contra Costa County).

Trends in Abundance: The California Department of Fish and Wildlife has been monitoring trends in white sturgeon abundance for decades and information on trends for nearly 80 years is available. From that body of work, it is clear that large variations in recruitment, frequently including 5 or more consecutive years of low or no recruitment, have been routine since the 1930s and the proximate cause for this variation is low flows during winter and/or spring. Managing the population through predictable ebbs in abundance is the key to conservation of white sturgeon and protection of its fishery.

The CDFW's index of annual white sturgeon recruitment from age-0 and age-1 fish suggests that peak recruitment has decreased trend-wise since the early 1980s, recruitment in most years is a small fraction of peak recruitment, and the most recent notably-high recruitment was in 2006 (Figure 1). This trend is completely plausible and expected from the relationship between hydrology and recruitment, but the slope of the trend may be biased toward decline due to release of fingerlings by hatcheries from 1980-1988.

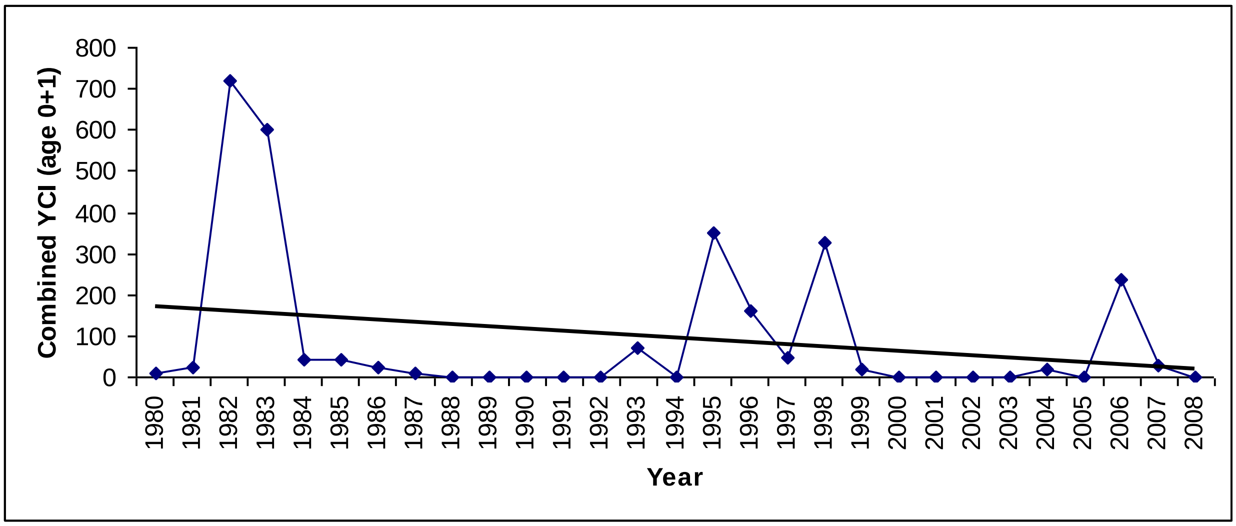


Figure 1. White sturgeon year class indices (age-0 and age-1 combined), San Francisco Bay, 1980-2012.

Trends since 1980 in the abundance of subadult and adult white sturgeon are as expected from variations in river hydrology and indices of recruitment, though abundance estimates are generally imprecise and sometimes lack confidence intervals. Interpretation of catch-per-unit-effort (CPUE) data from the fishery is confounded somewhat due to changes in regulations regarding size limits, daily bag limits, and annual bag limits. Length frequency distributions are a particularly important component when interpreting trends in abundance.

Estimated annual abundance of white sturgeon >= 102 cm Total Length (TL) has ranged from approximately 2,500-300,000 since 1980 (DuBois et al. 2011); the best estimates ranged from approximately 75,000-150,000 fish. The most recent and rigorous estimates are for fish 117-168 cm TL from the period 2007-2011, and those ranged from approximately 30,000-56,000 fish (DuBois and Gingras 2011). Extreme CPUE values should be discounted because they likely indicate unusual distributions of fish rather than rapid changes in the population's abundance. Using standardized fish capture and tagging techniques as part of a CDFW mark-recapture study, annual CPUE of white sturgeon 117-168 cm TL has varied from approximately

1-13 fish/100 net-fathom hours since 1980 and was less than 2 fish/100 net-fathom hours during the period 2005-2012 (DuBois and Gringas 2013).

Nearly all historical fishery-dependent data comes from Commercial Passenger Fishing Vessel (CPFV, a.k.a. party boat) logbooks. Annual white sturgeon CPUE in that fishery has varied between approximately 2-4 fish/100 hours of fishing effort since 1980 (DuBois and Gringas 2013). However, length data are not collected by the CPFV fleet and, since 1980, the size limit (TL) on white sturgeon changed from >=102 cm to 107-183 cm, 112-183 cm, 117-183 cm, and 117-168 cm in subsequent years, so it is only possible to describe coarse changes in white sturgeon demographics using CPFV data.

Annual length frequency distributions from CDFW's mark-recapture study and a pilot study using longlines clearly show the recruitment, growth, and subsequent decrease in abundance of strong year classes (Schaffter and Kohlhorst 1999, DuBois et al. 2011, DuBois and Gringas. 2013), as do length frequency distributions from CDFW Sturgeon Fishing Report Card data (CDFW Sturgeon Fishing Report Card reports, DuBois et al. 2011). Report cards have been in use since 2007. Because anglers commonly volunteer data on the lengths of fish too small to keep, the cards are helping bridge the long-standing gap in information on fish aged 2-8.

Trends of year-class indices (YCI), based on the number of age-0 and age-1 juveniles, suggest recruitment has decreased significantly, with low recruitment for 12 of the 29 years (1980-2008) on record (Figure 1). Although the present white sturgeon population appears to have been reduced over the last 30 years, some recent population trends are encouraging and stakeholder concerns about the white sturgeon population and fishery in California have resulted in highly restrictive angling regulations, new monitoring and research efforts, strong anti-poaching measures, and fish passage and habitat restoration efforts.

Nature and Degree of Threats: All sturgeon species worldwide are in serious decline and some are on the verge of extinction. Principal threats to sturgeon worldwide are similar to those in California (Table 1) and include: harvest (especially poaching), dam-related flow alteration and reduction, habitat degradation, and pollution (Billard and Lecointre 2001).

Major dams. Dams block access to important upstream spawning habitats and alter flows, which results in reduced habitat quantity and quality for early life stages (Coutant 2004). The major 'rim dams' in California largely lack fish passage facilities, so sturgeon are confined to downstream areas. In the Sacramento River, years of high spring outflow have been associated with strong year classes. The large dams on nearly all Central Valley rivers reduce the frequency, volume, and duration of these flows, reducing the frequency of successful sturgeon year classes (Moyle 2002). Dam operations can attenuate winter and spring flows required for the initiation of spawning and outmigration. Changes in the hydrograph can disconnect main channel habitats from floodplains, which may be especially important rearing habitats. Changes in sediment budgets and flow regime can decrease the quality and quantity of spawning and incubation habitats. For example, dam-attenuated winter flows can limit the amount of cover available in interstitial spaces in rocky substrates because the substrates are scoured less frequently. Changes to hydrographs can influence juvenile movements and predation rates. Lower turbidity levels and simplified channels as result of dam construction/impoundment may result in increased main channel predation of juveniles (Gadomski and Parsley 2005b). Lack of suitable habitats below dams may limit recruitment or lead to recruitment failure (Kynard and

Parker 2005).

Agriculture. Levees and land reclamation along rivers and estuaries have substantially eliminated large areas of floodplain habitats and their connectivity to main river channels, reducing access to important juvenile rearing areas. These historically abundant habitats once offered protection for sturgeon and many other native fishes from high flows, provided foraging habitats, and served as holding areas during migration. Diversion of water for agriculture can also reduce flows to the extent that sturgeon populations can no longer be supported in some areas (Moyle 2002). White sturgeon are particularly sensitive to agricultural pollutants. They readily bioaccumulate toxins from fertilizers and pesticides, which can cause deformities, decrease growth, and reduce reproductive potential. In the Columbia River, the incidence of physical deformities, such as misshapen fins, abnormal (short or forked) barbels and malformed or missing eyes increased with age, suggesting that they were a result of continued exposure to sediments contaminated with organic pollutants (Burner and Rien 2002). Exposure to organochlorine pesticides caused an overall decrease in the condition factor of juveniles, as well as decreasing the concentrations of sex hormones (testosterone and estradiol) in white sturgeon blood plasma (Gundersen et al. 2008). Electrophilic pesticides that can bond to DNA and other cellular macromolecules are common in the Sacramento River (Donham et al. 2006). Concentrations of mercury in white sturgeon livers also increased with age, suggesting that white sturgeon are prone to the bioaccumulation of heavy metals (Webb et al. 2006). Liver mercury content is negatively correlated with relative weight and gonadosomatic index. Consequently, exposure to mercury likely negatively affects white sturgeon reproductive potential and the potential for long-term mercury exposure in the Sacramento River basin is high.

Selenium entering the San Francisco Estuary from agricultural drainage (Presser and Luoma 2006) can decrease juvenile survival. Juveniles fed diets with high concentrations (> 41.7 ug Se/g) of selenium decreased swimming activity and grew less than other groups (Tashjian et al. 2006). Selenium accumulates in the kidney, muscle, liver, gill, and plasma tissues of these fish, contributing to decreased survival, particularly when exposed to brackish water (> 15 ppt) (Tashjian et al. 2007). Contaminated fish also had less energy reserves (whole body protein, lipids), perhaps limiting foraging activity and escape from predation. Although current regulatory thresholds for selenium toxicity (10-20 ug Se/g) may protect white sturgeon from adverse impacts, the concentration of selenium by the alien overbite clam, a major prey of sturgeon, may be resulting in increased levels in sturgeon as well.

Fertilizers entering the estuary cause algal blooms which may harm sturgeon both through release of toxins (*Microcystis*) and through depleting oxygen and increasing CO_2 in backwaters. Hypercapnia (elevated levels of CO_2) can cause mortality or morbidity in juvenile white sturgeon because energy normally used for growth, disease resistance and lipid storage is redirected toward maintaining homeostasis (Cech and Crocker 2002, Crocker and Cech 2002).

	Rating	Explanation
Major dams	High	All rivers occupied in CA are dammed, blocking access to spawning habitats and altering flows and habitat suitability
Agriculture	High	Water demands result in decreased flows in rivers during critical life history periods; pollution from agricultural return waters may acutely affect sturgeon
Grazing	Low	Effects mostly upstream of reaches occupied by sturgeon
Rural residential	Low	Rural residences occur along white sturgeon streams (e.g., Klamath River) but the effects from rural development are likely minor
Urbanization	High	Urban water demand, runoff and pollution inputs can create toxic environments; habitat alteration and simplification are severe in urban areas; multiple large urban areas within existing range
Instream mining	Low	Effects unknown but present in some coastal streams
Mining	Medium	Most toxic runoff is above dams, although Iron Mountain mine poses a major threat if controls of tailings and effluent fail
Transportation	Low	Roads, railroads, shipping lines and associated bridges and channelization modify rivers occupied by white sturgeon
Logging	Low	Impacts from sedimentation, etc. may affect rivers other than Sacramento River (e.g., Klamath River) but not likely to affect reproduction
Estuary alteration	High	California estuaries are severely altered; San Francisco Estuary and Delta habitats substantially altered and degraded from past
Recreation	Low	Boating and other activities can disturb sturgeon spawning and foraging; white sturgeon fatalities from vessel strikes are not uncommon
Fire	Low	Erosion from burned areas can increase fine sediment delivery to streams, but most impacts occur above major dams
Harvest	Medium	Legal and illegal harvest cause adult mortality, although legal harvest is now typically less than 10% of harvestable fish; illegal harvest for caviar and meat is a much greater threat
Hatcheries	Low	Aquaculture facilities exist for white sturgeon, but fish have not been released into the wild since approximately 1988
Alien species	Low	Alien species present throughout range; impacts largely unknown

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

Urbanization. The impacts from urbanization on white sturgeon are similar to those from agriculture, although perhaps not quite as widespread. Pollutants from sewage treatment plants, storm drains, and surface runoff have the potential to negatively affect sturgeon, as does often severe habitat simplification associated with urban development along river and stream corridors.

Mining. Iron Mountain Mine, an abandoned heavy metal mine above Keswick Reservoir (below Shasta Dam) on the Sacramento River, is an EPA Superfund site. While extensive measures have been taken to reduce the potential for toxic spills from the site, the impacts of a spill would be severe enough that even a low probability of failure rates concern. If the earthen retaining dam designed to impound mine effluents fails, an acidic slurry of toxic heavy metals could spill into the river, potentially resulting in massive fish kills; white sturgeon would likely be especially vulnerable to both acute (short-term) and subacute (long-term) exposure to these toxins, given their benthic foraging behavior and long life spans.

Logging. In the Sacramento River watershed, sturgeon are isolated from the effects of logging in headwaters by major dams, which minimizes their exposure to sedimentation and increased temperatures. However, white sturgeon may be negatively affected by logging in the Klamath and other river basins within their range. Introduced fine sediments (silt, sand, fine gravel) can fill substrate interstitial spaces and cause recruitment failure (McAdam et al. 2005). Laboratory experiments using Kootenai River white sturgeon found that fine sediment (5 mm) covering embryos resulted in 0-50% survival, delayed hatching and decreased larval length (Kock et al. 2006). Exposure of juvenile (3-78 days old) white sturgeon to didecyldimethylammonium chloride (DDAC), a highly soluble pesticide commonly used to protect lumber in Canada, resulted in mortality and sublethal effects (Teh et al. 2003). Didecyldimethylammonium chloride exposure resulted in 50% mortality of 78 day-old juveniles, the most resistant age group, within 18 and 36 hours of exposure. Sublethal effects to all age groups included decreased growth (weight and length) and decreased swimming activity. Juveniles that expressed sublethal effects had not recovered 21 days after exposure, perhaps increasing susceptibility to predation and disease and decreasing the probability of reaching sexual maturity. Although of particular concern in the Fraser River, Canada, DDAC may also impact sturgeon that migrate between rivers in California and Canada.

Estuary alteration. White sturgeon in California spend much of their life cycle in the heavily altered San Francisco Estuary or other smaller estuaries. The Delta's levees and rip-rapped channels restrict foraging habitat for sturgeon. At times, much of the freshwater inflow to the Delta is diverted into the pumps of the south Delta, altering or reducing river flow and entraining small sturgeon. Suisun Bay, Suisun Marsh and San Pablo Bay are primary rearing areas and are also subject to altered flows, contamination from many toxic compounds, invasions of alien species, and reduced water quality from urban runoff and effluent. Given the altered condition of the estuary and the fact that it is continuing to rapidly change, it is remarkable that white sturgeon have persisted in even moderately large populations (see Lund et al. 2007, 2008, Moyle 2008).

Harvest. White sturgeon populations were substantially reduced by commercial fishing in the San Francisco Estuary in the 19th century; consequently, commercial harvest has been prohibited since the mid-1900s (Moyle 2002). The sport fishery has become increasingly restrictive over time but, unlike in Oregon and Washington, California has not adopted a harvest quota.

White sturgeon fishing is currently closed in the north coast district (Humboldt, Del Norte, Trinity, Siskiyou counties), reaches of the Sacramento River in the Sierra and Valley districts (Shasta, Tehama, Glenn counties), in parts of San Francisco Bay, and at low-head dams (weirs) controlling flow into bypasses of the Sacramento River. The Sacramento River closure was implemented in 2009, closures at weirs were implemented in 2013, and other closures have been in effect for decades. Sport fishing regulations, established in 2007, allow individual anglers to harvest one fish per day and up to a total of three fish per year, whereas previous regulations did not limit the annual harvest. Also, in 2007, the size limit was changed from 46-72" TL to 46-66" TL. The size limits implemented are considered protective, yet were a compromise that still allows for potential harvest of female fish that have not yet spawned for the first time. In addition, Sturgeon Fishing Report Cards are required for all sturgeon anglers and are to be returned to the California Department of Fish and Wildlife upon completion; all harvested white sturgeon must be tagged. The Sturgeon Fishing Report Card and associated tags are the mechanisms whereby the daily and annual bag limits are enforced (see Management Recommendations section).

In anticipation of higher numbers of white sturgeon released in association with more restrictive angling regulations, several additional measures were taken in 2013 to improve the survival rates of fish that anglers are required to, or voluntarily, release. These protective regulations include: (a) only one single-point, single shank, barbless hook may be used on a line when taking white sturgeon, (b) snares may not be used to assist with landing a white sturgeon, (c) description of length limits in terms of fork length rather than total length, and (d) white sturgeon greater than 173 cm (68 in.) fork length may not be removed from the water and must be released immediately.

In general, harvest rates of fish 117-168 cm TL (e.g., the legally-harvestable size as of March, 2007, and a subset of all prior legal sizes) during 2000-2008 were lower than rates during the 1980s (DuBois et al. 2011) and the overall harvest rate trend is decreasing (M. Gingras, CDFW, pers. comm. 2013). Harvest rates have ranged from approximately 2-9%, but are likely biased low.

Illegal commercialization (poaching) of white sturgeon is common because of the high value of their caviar. As a consequence, the CDFW makes enforcement of sturgeon fishing regulations a high priority and, in 2007, a law was enacted that facilitated easier enforcement against those participating in illegal commercialization and drastically increased the severity of financial penalties associated with these activities.

White sturgeon contribute to a small Native American fishery in the Klamath River but only 186 juvenile and adult white sturgeon were caught by the Klamath River fishery from 1980 to 2002, about eight fish per year (Welch et al. 2006). Sacramento River white sturgeon may also be caught in fisheries in the Columbia River region but the potential effects on California populations are not known.

Hatcheries. In response to wide fluctuations in white sturgeon abundance and intermittent decreased catch rates over time in the sport fishery, outplanting of hatchery sturgeon stocks to augment natural populations has, although the subject of much debate, been proposed. White sturgeon have been raised in California aquaculture facilities for meat and caviar since 1980 and juvenile white sturgeon from those facilities were outplanted from 1980-1988; however, no hatchery stocks have been released into the wild since that time. The contribution of outplanted fish was not evaluated and records are sparse; nonetheless, it is estimated that a total of approximately 500,000 fry and fingerlings were released during the 1980s.

Hybridization of wild and hatchery stocks may have detrimental effects on the population structure of wild stocks, as studies of salmon populations have demonstrated (see Chinook salmon accounts in this report). Hatcheries may also facilitate the spread of disease such as iridovirus. Iridovirus infection of white sturgeon reduces the growth and survival of fry and fingerlings (Raverty et al. 2003).

Alien species. Alien fishes are abundant in the estuaries and rivers that white sturgeon inhabit. Alien fishes can reduce white sturgeon survival through predation on juveniles (Gadomski and Parsley 2005c), although this has not been demonstrated to be a problem in California. In the San Francisco Estuary, white sturgeon feed heavily on the overbite clam, which invaded in the 1980s. This clam (and other alien clams on which sturgeon feed) concentrate selenium and other heavy metals, which bioaccumulate in sturgeon and have the potential to negatively affect reproductive success.

Effects of Climate Change: Increases in water temperatures associated with climate change may decrease white sturgeon reproductive success. Successful spawning appears to be linked to cool water temperatures (< 18°C) and high spring flows. Females holding in 18-20°C water had inhibited ovulation and oocyte development (Webb et al. 1999, Linares-Casenave et al. 2002). Although based on laboratory results, these findings indicate that the pre-spawning temperature regime is important for normal ovarian development and should be considered in management of wild stocks. Bioenergetic modeling of white sturgeon in the Snake River also demonstrated that small increases in maximum water temperatures (19 to 24 °C) decreased growth and reproduction (spawning frequency, fecundity) because of decreases in caloric assimilation resultant from increases in energy costs (Bevelhimer 2002). Increased water temperatures may also hasten developmental times, perhaps resulting in a mismatch between the onsets of exogenous feeding and prey availability. Prey availability at onset of exogenous feeding was determined to be important in determining year class strength (Muir et al. 2000). Increased water temperatures may also make white sturgeon more susceptibility to disease. White sturgeon iridovirus is thought to be present in rivers throughout their range, and has been verified to occur in the anadromous waters of California's Central Valley (M. Gingras, CDFW, pers. comm. 2013). The virus is a slow wasting disease that primarily affects growth in fry and fingerlings by infecting the top layers of the skin, including the gills, barbels and nares (Drennan et al. 2007). Stressful conditions associated with poor water quality can induce the virus. Consequently, increased temperatures predicted from climate change models, in combination with pollution, may make young sturgeon more susceptible to the virus.

Climate change models predict seasonal shifts in precipitation, as well as increased frequency of floods and drought. Higher or more flashy winter flows may flush juvenile white sturgeon into estuarine areas before they are capable of adjusting to saline environments. The ability to osmoregulate is likely size dependent (Amiri et al. 2009), so younger and smaller juvenile sturgeon may be at risk, especially if floodplain and edge-habitat refuges are lacking, as is the case in much of the lower Sacramento River system. Coupled with predicted increases in estuary salinity levels due to sea level rise, earlier entry of juveniles into estuarine habitats may limit juvenile survival. In contrast, lower summer flows, exacerbated by increasing water demands, may decrease spawning and outmigration success.

Status Determination Score = 2.3 - High Concern (see Methods section Table 2). Despite a relatively robust population that presently includes tens of thousands of sub-adults and adults, white sturgeon must be managed carefully due to already demonstrated population cycles that may be exacerbated in the future by climate change, increasing human water demand, further degradation of habitats, overharvest, or some combination thereof. Management of white sturgeon is complicated by the combination of exposure to pollutants, freshwater and estuarine habitat alteration (particularly in the San Francisco Estuary), harvest, and because its long life span can mask the detection of poor reproductive success. NatureServe ranks white sturgeon as Globally secure (G4) but Imperiled (S2) in California due to anthropogenic impacts on their habitats. The American Fisheries Society considers the species to be Endangered (Jelks et al. 2008). Several populations in California are also considered "conservation dependent" (Musick et al. 2000).

Metric	Score	Justification
Area occupied	1	The only self-sustaining population in California appears to be in the Sacramento River
Estimated adult abundance	3	Based upon 2000-2009 estimates of age-15 fish and other demographic data
Intervention dependence	3	The population and fishery need to be monitored and managed closely, flows regulated, and pollution inputs and poaching reduced
Tolerance	2	Juvenile white sturgeon are intolerant of poor water quality, including high temperatures
Genetic risk	4	High genetic diversity
Climate change	2	Very sensitive to temperature increases, degraded water quality and flow changes predicted by climate change models
Anthropogenic threats	1	The combination of illegal harvest, pollution, and habitat alteration continue to threaten white sturgeon in the wild (see Table 1)
Average	2.3	16/7
Certainty (1-4)	4	

Table 2. Metrics for determining the status of white sturgeon, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: White sturgeon in the Sacramento River and the San Francisco Estuary have been regarded as well managed since the 1950s because they have sustained a fairly large fishery (Moyle 2002), though not as well managed as white sturgeon in Oregon and Washington. Unfortunately, increasing pollution, water diversion, habitat degradation, impacts from climate change, and poaching may limit recovery or contribute to further decline. The following are management recommendations to afford greater protection for white sturgeon in California:

Harvest management. Harvest regulations for white sturgeon have become increasingly restrictive, with severe limits placed on sport harvest in 2006 and, again, in 2009. However, California lags behind Oregon and Washington in regards to adaptive management of sturgeon harvest and has no white sturgeon management policy or plan.

Productivity of white sturgeon in California is lower than in Oregon and Washington, yet the white sturgeon fishery is very culturally and economically important; therefore, it is imperative to apply adaptive management to the recreational fishery and tight controls over harvest, both legal and illegal. The decline and subsequent listing of the southern green sturgeon DPS in California as threatened under the federal ESA may be indicative that white sturgeon are on the same trajectory and signals a need for greater conservation measures, monitoring, law enforcement and related resources to prevent further declines.

As a top priority, the California Fish and Game Commission should implement an annual quota on harvest of white sturgeon and should assure the continued availability of pertinent white sturgeon demographic and fishery statistics, implementation of a study on the effects of poaching, and the development of a white sturgeon management plan.

As noted, regulations established in 2007 require that sturgeon anglers record all fishing activity on Sturgeon Fishing Report Cards to be returned to the CDFW upon completion and that anglers tag all white sturgeon harvested. Data from Sturgeon Fishing Report Cards provide a much better description of the fishery than was available previously and complement the CDFW's on-going mark-recapture study. Prior to use of Sturgeon Fishing Report Cards, annual harvest could only be coarsely estimated from imprecise abundance estimates and annual harvest rate estimates. Data gathered from 2007-2012 Sturgeon Fishing Report Cards indicate that annual harvest was 1424-2048 fish and anglers released 4171-5802 fish. Accuracy of Sturgeon Fishing Report Card data is the subject of on-going investigation, but the trends and year-over-year numbers are generally consistent and reasonable.

Information on fishing effort for white sturgeon is incomplete and suggests a mixed picture. The only trend data available are from the CPFV fishery, where fishing effort from CPFVs that landed white sturgeon has declined trend-wise from a peak of nearly 25,000 hours in 1986 to a record low of barely 3,000 hours in 2012. Estimated annual fishing effort during daylight (i.e., biased low), in the Sacramento River watershed to Carquinez Strait, ranged from approximately 110,000-320,000 hours during 2006-2009.

Information on the number of sturgeon anglers in California is incomplete, but the number of issued Sturgeon Fishing Report Cards shows that interest in the recreational fishery is substantial. An annual average of roughly 55,000 Sturgeon Fishing Report Cards were issued for free, when issued by hand, an annual average of roughly 112,000 were issued for free, when issued by an automated system, and approximately 55,000 were issued 6 months into the first year they were sold ($7.50 plus up to 8% in fees), utilizing an automated system. One incongruity in the recent management of white sturgeon is that there are far fewer legal-sized white sturgeon than are authorized for harvest through issuance of Sturgeon Fishing Report Cards. In general, Sturgeon Fishing Report Cards provide valuable data and insights into the fishery and should be continued to be issued and their data analyzed into the future.

Illegal commercialization of white sturgeon remains a significant concern, given the high value of individual fish and the relative ease with which the largest and most fecund females are targeted. More intensive efforts are needed to identify, arrest and convict poachers and the dealers who buy illegal caviar and legislative action should be taken to increase the numbers of CDFW Wildlife Officers and ensure a dedicated number are assigned to white sturgeon-related enforcement throughout their range in the state.

Reducing pollution (especially from agriculture). White sturgeon are very sensitive to many pollutants (heavy metals, selenium, organic pollutants, pesticides), even when the pollutants are at low concentrations, in part because sturgeon are long-lived and bioaccumulate toxins over long periods of time in their bodies as well as in their eggs (passing them on to sensitive larvae). Improved monitoring and treatment of non-point source pollution is necessary to minimize impacts on white sturgeon. Restoration of tidal wetlands and floodplain habitats would likely enhance detoxification of water draining from agricultural fields and sewage.

Heavy metals, especially selenium, are of particular concern because of their effects on reproduction. Thus, both point and non-point sources of polluted effluents into Central Valley rivers and the San Francisco Estuary need to be identified and prioritized for treatment, containment, or other mitigation measures. Fortunately, selenium from oil refineries has been reduced to very low levels, while selenium inputs from farms on the west side of the San Joaquin Valley into the San Joaquin River have also been declining. These reductions have decreased selenium concentrations in overbite clams, a major sturgeon prey item (S. Luoma, pers. comm. 2009). This example demonstrates that pollution mitigation measures can be effective but efforts needs to be more comprehensive and systematic, focused on reducing inputs into waterways and eliminating point sources via treatment.

Habitat improvement. Freshwater and estuarine habitat alteration, especially from dam and levee construction, as well as elimination of most of the Central Valley's historic floodplain habitats, has limited spawning and rearing success in the Sacramento River (and possibly the Klamath River as well). Thus, restoring habitats required for juvenile rearing and spawning adults needs to be a priority in the Sacramento River basin. Access to rearing habitats with abundant prey may help mitigate effects of increased water temperatures resulting from climate change because larvae can better withstand increased temperatures when they feed at optimum (~ 15% body weight/day) or near-optimum feeding rates (Amiri et al. 2009). Restoration of tidal sloughs in California could also provide important rearing habitat.

Improving stream flows. The Sacramento River is a highly regulated river and white sturgeon depend on rare high water years - when dams spill or flood releases are high - for reproduction that leads to large year classes in the population. However, too little is known about specific flow requirements for spawning, instream rearing, downstream migration, growth rates, and mortality rates of young fish to evaluate the cost to benefit of alternative management of river flows. More research on white sturgeon life history and environmental tolerances (especially flow requirements at all life stages) may show that winter flow releases from dams would initiate additional spawning and alter substrate for improved survival of eggs and larvae, additional spring flows may improve downstream migration and survival of juveniles, and sustained high flows in the spring could also provide access to important floodplain habitats (e.g., Yolo bypass) for rearing and enhanced growth.

Potential use of hatcheries. White sturgeon aquaculture has been proven to be successful; therefore, there may be an inclination to use hatchery stocks to enhance the sturgeon fishery in California. However, dependence on hatcheries for either supplementing the sport fishery or meeting conservation and recovery objectives brings inherent risk and should not be prioritized over conservation and management measures intended to reverse declines of wild stocks. A long-term management and monitoring plan needs to be developed that includes management goals and genetic analyses to identify differences between wild and domesticated stocks. A principal goal should be to prevent domestication of wild stocks and to maintain maximum genetic and life history diversity. However, if populations become even more severely reduced, a conservation hatchery may be required. Proper use of wild broodstocks may serve to augment declining populations and allow time for conservation and restoration actions designed to improve spawning and rearing success, as well as adult and juvenile survivorship, to be implemented. In cases where hatchery- reared sturgeon have been used in conservation (e.g., Kootenai River, Idaho), a time lag of up to 3 years was necessary for hatchery-reared white sturgeon to adapt to natural conditions (Ireland et al. 2002). During that time, fish experienced decreased growth and populations exhibited 60-90% annual survival. If high survival rates to augment a population are important, hatchery-reared fish should be released after reaching 134 mm TL (~ 5 months old), because laboratory results suggest that fish of this size and larger are less vulnerable to predation (Gadomski and Parsley 2005c). All hatchery fish should be marked with coded wire tags so success of different management strategies can be evaluated.

Research. White sturgeon are well-studied but research is still needed to determine priorities for habitat restoration and best flow regimes to support successful reproduction and survivorship. There is also a continuing need for long-term monitoring of populations in order to develop population trends. Monitoring of tagged fish could help determine movement patterns, habitat utilization across life history stages, and potential interactions of Sacramento River white sturgeon with other populations. In particular, the role of the Klamath River in supporting the California white sturgeon population needs further study.

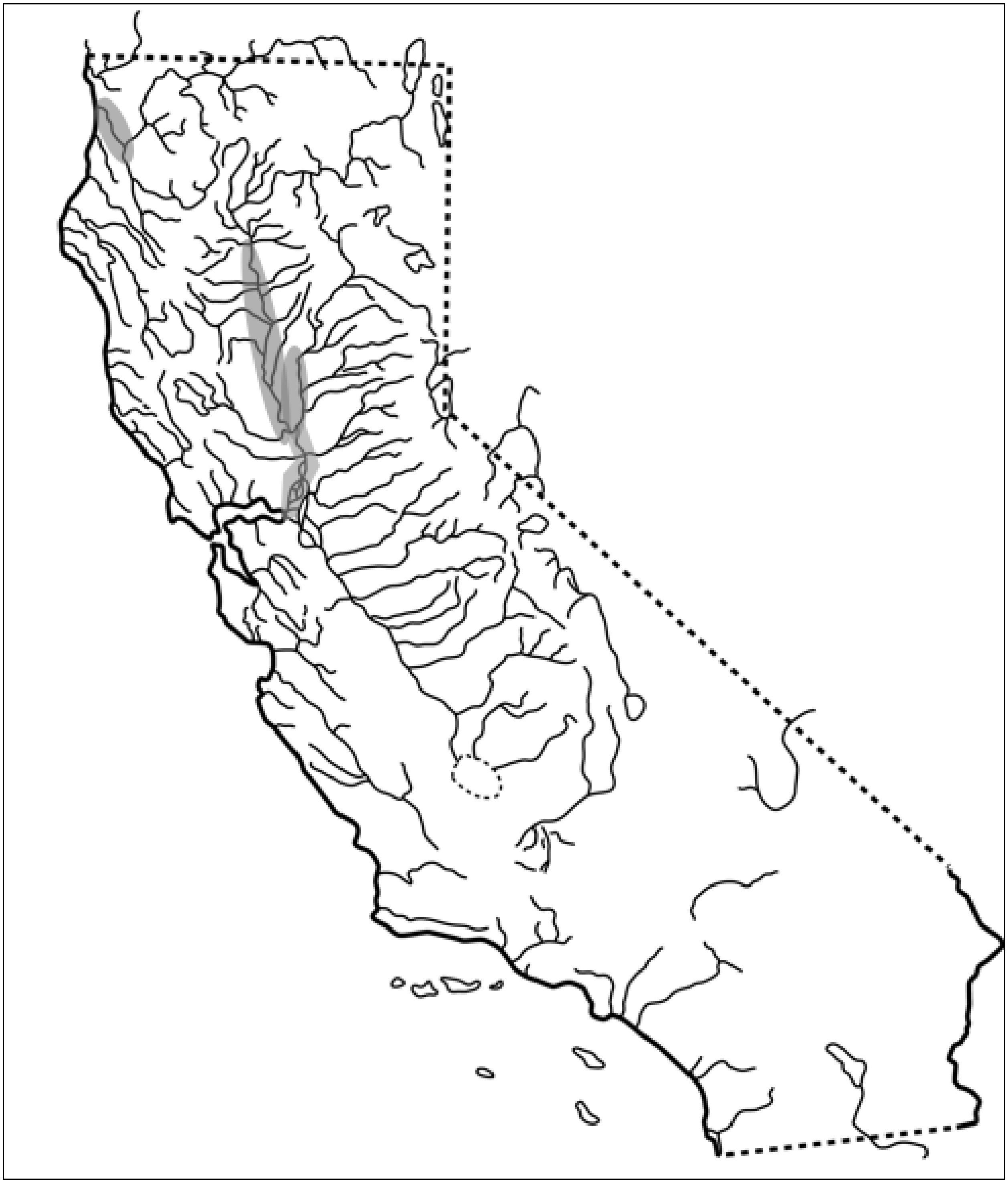


Figure 2. Distribution of white sturgeon, *Acipenser transmontanus* (Richardson), in California. Only freshwater distribution in the Sacramento and Klamath River basins is shown.

GOOSE LAKE TUI CHUB
Siphateles thalassinus thalassinus (Cope)

Status: Moderate Concern. Goose Lake tui chub remain numerous in Goose Lake and in the lower reaches of most large tributaries to the lake. However, Goose Lake dries out completely during periods of drought and the tui chub is particularly susceptible to periodic elimination of lake habitat, followed by great reductions in population size.

Description: The Goose Lake tui chub is differentiated from other *Siphateles* taxa by their longer fins, more posterior dorsal fin, longer head, and larger number of dorsal rays, usually nine (Snyder 1908b). Coloration is similar to Lahontan Lake tui chub, although larger specimens from Goose Lake (up to 30 cm FL) are uniformly silver except for a white belly. For a general description of tui chub see the Lahontan Lake tui chub account in this report.

Taxonomic Relationships: The Goose Lake tui chub was first described by E. D. Cope (1883) as *Myloleucus thalassinus*. He simultaneously described a second species of tui chub from the lake as well. Snyder (1908b) noted that Cope collected numerous dried chubs that had been dropped by fish-eating birds along the shoreline and hypothesized that the second species described by Cope was based on these poorly preserved specimens. However, there are apparently two morphological types of tui chub in Goose Lake: a "standard" heavy-bodied tui chub and another form with a less robust body and more pointed head (R. White and P. Moyle, unpubl. obs.). Snyder (1908b) placed *thalassinus* in the genus *Rutilus* because Jordan and Evermann (1896) synonymized *Myloleucus* with *Rutilus*. North American cyprinids placed in the European genus *Rutilus* eventually were referred to generic names of New World minnows, including *Gila*. Snyder (1908b) considered *thalassinus* to be native to Goose Lake and the upper Pit River from Big Valley upstream to Goose Lake. Hubbs et al. (1979), however, considered the form in the Pit River to be distinct from the Goose Lake form, although no evidence was provided. For reasons that are now obscure, Hubbs et al. (1979) used the specific name *thalassina* which was subsequently adopted by other workers; however, *thalassinus* (Cope 1883) has precedence and is used here.

In 2001, a genetic study using mitochondrial DNA found that tui chub in the Cow Head, Warner, and Goose Lake basins are closely related and are sufficiently genetically distinct from other tui chubs as to be recognized as a single species under the name *Siphateles thalassinus* (Harris 2000). Harris recognized two lineages within *S. thalassinus*, one in Goose Lake and the other in the Pluvial Lake Warner Basin, which includes both the Cow Head and Warner basins. Harris's findings supported Hubbs and Miller's (1948) postulation of a possible relationship between Cow Head tui chub and chubs from the lakes in Warner Valley, Oregon, because of the stream connection that existed between the Cow Head Basin and the Warner Valley drainage.

Chen et al. (2009) used microsatellite DNA to further resolve the taxonomy of tui chubs of the northwestern Great Basin. Chen's results supported Harris's systematics regarding the species status of *S. thalassinus.* Chen (2009) also found that tui chub populations of the upper Pit River drainage were genetically indistinguishable from those in Goose Lake and that these two populations, taken together, were sufficiently distinct to warrant subspecies status as *S. t. thalassinus.*

Rutter conducted the only known comparison of tui chub from above and below Pit River Falls and noted substantial differences in lateral line scale counts between the populations (Rutter 1908). However, both he and Snyder (1908b) considered tui chub populations in Goose Lake and the upper Pit River to be similar. Then, in 1979, without providing a rationale, Hubbs et al. listed Pit River and Hat Creek (tributary to the lower Pit River, below Pit River Falls) tui chub populations as discrete undescribed subspecies. No systematic work has been conducted on the lower Pit River tui chub populations since then, which means that, over a hundred years after Rutter published his findings, the relationship between upper and lower Pit River populations of tui chub remains unresolved.

For a general discussion of tui chub taxonomy, see the Lahontan Lake tui chub account in this report.

Life History: The life history of this subspecies has been little studied. Chubs commonly reach 250 mm FL in the lake and fish as large as 316 mm FL have been collected, indicating that this form may be very long-lived in lake habitats. In streams, however, they rarely exceed 120 mm FL. The size distribution of tui chubs sampled from Goose Lake in 1989 showed two modes. The great majority (>90%) of fish were less than 120 mm SL, while the remainder were 200-300 mm SL (R. White, USFWS, unpubl. data 1989). Most tui chubs are opportunistic omnivores and consume a wide variety of aquatic invertebrates (Moyle 2002). Tui chubs are a major prey base of Goose Lake lamprey; depending on the length class, 20-70% of the tui chubs >200 mm SL sampled in 1989 had lamprey scars (R. White, unpubl. data 1989).

Habitat Requirements: Goose Lake is a massive, natural alkaline lake covering approximately 39,000 surface hectares straddling the Oregon-California border. The lake is shallow, averaging 2.5 m deep, hyper-eutrophic and very turbid (Johnson et al. 1985). A thermocline (and hence temperature stratification and dissolved O_2) appear to be affected by wind conditions, as indicated by data from September, 2009 (R. White, unpubl. data 1989). On a calm September day, water temperature at one sampling locality was 17°C from the surface to 40 cm depth, with a sharp drop at 40-50 cm, and 14-15°C at 50-200 cm depths. At a second locality, temperature decreased from 23°C at the surface to 15°C at 35 cm, remaining at about 15°C between 35cm and 2.5 meter depths. At those two localities, dissolved oxygen concentration held at about 8-10 mg O_2 l^{-1} from the surface down through the water column, but dropped abruptly to <1 mg O_2 l^{-1} in deeper water, depending on locality. The drop in O_2 occurred at about 150 cm depth at one locality, and between 260-270 cm depths at the second locality. On a windy September day, the water temperature was 15˚C throughout the water column (surface to 185 cm depth) measured at one locality. Dissolved O_2 was constant (slightly <10 mg O_2 l^{-1}) from the surface to 170 cm depth, but dropped abruptly to <4 mg O_2 l^{-1} at about 175-180 cm.

The surface elevation of Goose Lake fluctuates seasonally, but averages 1,433 m. In California, no tui chubs have been found in streams above 1441 m in elevation, although tui chubs have been found above 1550 m in Oregon streams (J. Williams, unpubl. data). In streams, Goose Lake tui chub prefer pools and are generally not found in swift water, although they have been collected from runs in Battle Creek on the west shore of Goose Lake (J. Williams, unpubl. data). Goose Lake tui chubs have been collected in habitats with temperatures ranging from 9-29°C. In July, 1992, large numbers of chubs were observed in the lower reaches of Willow and Lassen creeks (G. Sato, pers. comm. 1993), where they may have been attempting to escape from the increasing alkalinity of the drying lake.

In Oregon streams, Scheerer et al. (2010) found tui chubs mainly in the lowermost reaches in low gradient, unforested stream channels and irrigation ditches, although a few tui chubs were also collected at higher elevation sites. The wide, silt-bottomed habitats were mainly associated with agricultural fields. The principal co-existing species in these agricultural reaches were alien species such as brown bullhead (*Ameiurus nebulosus*) and fathead minnow (*Pimephales promelas*).

Distribution: In addition to Goose Lake itself, *S. t. thalassinus* also occurs in low-elevation sections of streams tributary to the lake and in Everly Reservoir, Modoc County California, as well as in Cottonwood, Dog and Drews reservoirs in Oregon (Sato 1992a). In 2007, the Oregon Department of Fish and Wildlife collected relatively large numbers of tui chub from Dry, Drews, Dent, Thomas and Cox creeks on the Oregon side of the basin (Heck et al. 2008, Scheerer et al. 2010).

The Goose Lake basin is a disjunct subbasin of the upper Pit River. At extreme high water, Goose Lake spills into the North Fork Pit River as it did in 1868 and 1881. Since the late 19th century, storage and diversion for irrigation have substantially reduced the inflow to Goose Lake and future overflow of the lake into the Pit River is deemed unlikely (Phillips et al. 1971). However, because of this historical hydrologic connection, the fish faunas of Goose Lake and the upper Pit River share most taxa and tui chub populations from the two basins are genetically indistinguishable (Chen et al. 2009).

Reid et al. (2003) found tui chub in 7 of 12 sampling sites in the upper Pit River watershed, including the mainstem Pit River near Canby, the North Fork Pit River from the vicinity of Parker Creek down to the confluence with the South Fork Pit River, just below Alturas, and in the headwaters of the South Fork Pit River in Jess Valley.

Trends in Abundance: Goose Lake tui chub have been documented as extremely abundant in the lake. During 1966 gillnetting surveys of Goose Lake, tui chub comprised 88% of fishes collected (King and Hanson 1966). In 1984 it comprised nearly 96% of gillnet collections (J. Williams, unpubl. data) and, in 1989, it comprised 96% of fishes sampled by trawls, gillnets, and seines (R. White and P. Moyle, unpubl. data). Large numbers of chubs could be caught with relatively little sampling effort (e.g., 100+ in a 5-minute haul with a small trawl). In 1992, chubs were eliminated from the lake as it became progressively more shallow and alkaline and then dried. As lake levels dropped, fish crowded into the inflowing streams where they were extremely vulnerable to predation from white pelicans and other fish-eating birds. Apparently the tui chubs survived in greatly reduced numbers in stream pools and in some upstream reservoirs, but mainly in Oregon. Periodic drying of Goose Lake is a natural response to drought and the native fish assemblage evolved under these conditions. However, diversion of stream flows along with the effects of grazing, wetland reclamation and road construction have altered streams and riparian areas, reducing the extent of stream habitat that these fish rely on during periods of drought.

Nature and Degree of Threats: The principal threat to the Goose Lake tui chub is desiccation of its principal habitat, Goose Lake, accompanied by loss of refuge habitat in tributary streams and reservoirs in the drainage. This account does not include factors affecting poorly known Pit River populations, since the two populations are effectively disjunct; however, if the two regions are considered to have just one population, the Pit River may serve as a drought refuge, unless it is completely taken over by alien species. Tui chub populations may, however, persist in the presence of alien species: Big Sage Reservoir, on Rattlesnake Creek, a Pit River tributary, once supported a successful bass fishery, with a tui chub prey base (Kimsey and Bell 1955). See the Goose Lake sucker account in this report for further details.

Agriculture. Although the lake has dried historically, diversions for irrigation and loss of natural water storage areas (e.g., wet meadows) from agriculture and grazing presumably caused it to dry up more rapidly during the recent period of prolonged drought. Even in absence of complete drying of the lake, reduction of inflows increases the likelihood that the lake will periodically become too alkaline to support freshwater fishes such as tui chub. High alkalinity may be a particular problem for early life-history stages. The key to the survival of Goose Lake tui chubs, in the past, has likely been the presence of refuges in the springs and pools of the lower reaches of tributary streams. The same factors (agricultural diversions, road building, channel alterations) which affect lake inflow also negatively impact in-stream habitat, leaving tui chub few refuges during drought. It is likely that key refuge areas are mainly in Oregon, in the 'delta' marshy areas of Thomas Creek and other tributaries. Small reservoirs created for storage of irrigation water may also serve as refuges for tui chubs.

Grazing. Livestock grazing is, perhaps, the most pervasive land use in the Goose Lake basin. Lowland refuge habitats are degraded by stream erosion and bank destabilizations caused by livestock grazing in riparian areas, especially through the removal of woody riparian plants. While improved management of most grazed lands has reduced the threat of grazing in the short-run (e.g., in the Lassen Creek drainage), as the climate becomes warmer and more variable, there is considerable potential for negative impacts of grazing (and other land uses) to increase unless there is expanded use of riparian protection measures, such as exclusionary fencing.

Transportation. Virtually all streams used by Goose Lake tui chubs are crossed by roads, which often serve as sources of siltation or barriers to fish movement.

Alien species. Goose Lake tui chubs manage to coexist with a variety of alien species, mainly in highly disturbed habitats such as irrigation ditches and reservoirs (Scheerer et al. 2010). However, predation by alien fishes should be considered in management. Education and enforcement are important tools to prevent further illegal introductions of non-native species.

	Rating	Explanation
Major dams	n/a	Impacts may exist in Oregon
Agriculture	High	Diversion of water significantly impacts stream habitat and the frequency/duration of Goose Lake desiccation
Grazing	Medium	Grazing continues to impact stream and riparian habitats
Rural Residential	Low	Relatively little residential water use in comparison to agricultural use in native range
Urbanization	n/a	
Instream mining	n/a	
Mining	n/a	
Transportation	Low	Roads cross all major Goose Lake tributaries
Logging	Low	Widespread in watershed but not intense
Fire	Low	Entire watershed prone to forest and range fires
Estuary alteration	n/a	
Recreation	n/a	
Harvest	Low	Used as bait but practice has been made illegal (article 3, Section 4.30 of CA freshwater sport fishing regulations)
Hatcheries	n/a	
Alien species	Medium	Alien species present a potential threat in drought refuges, particularly in reservoirs

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

Effects of Climate Change: Goose Lake is located at the edge of the arid Great Basin, where relatively rare aquatic habitats are often tapped for human use. Any reduction in precipitation or increased frequency of drought is likely to further stress aquatic habitats in this basin. Snow melt and winter rains, the principle sources of water in the Goose Lake watershed, are likely to substantially decrease as the climate warms (Moyle et al. 2012). During low flow periods, lower streams reaches in the basin currently reach extreme temperatures (24-26°C). Thus an increase in air temperature, especially when combined with reductions in stream flow through diversions, could prove lethal to native fish populations. An increase in fire frequency or intensity in this dry area could also decrease riparian shading, add sediment, and otherwise alter the refuge stream habitats that tui chub depend on during drought. See the Goose Lake sucker account in this report for a more detailed description of climate change effects in the basin. Moyle et al. (2013) consider the Goose Lake tui chub to be "highly vulnerable" to extinction in California because of climate change, but considered the chub to be confined to the Goose Lake basin. If the limited populations in the upper Pit drainage are, indeed, part of this subspecies, the chub may have greater resistance to climate change.

Status Determination Score = 3.1 – Moderate Concern (see Methods section, Table 2). The limited distribution of Goose Lake tui chub in California and its vulnerability to extended drought merit its inclusion as a species of special concern. The Goose Lake tui chub is a US Forest Service and Oregon Department of Fish and Wildlife "Sensitive Species". The American Fisheries Society considers the Goose Lake tui chub to be "threatened" (Jelks et al. 2008), while NatureServe ranks it as "imperiled" (T2). Presumably, the tui chub develops large populations when Goose Lake is full but may drop to low numbers in isolated populations when the lake dries. These same factors make it particularly susceptible to climate change.

Metric	Score	Justification
Area occupied	2	Restricted to Goose Lake and, possibly, upper Pit River basins
Estimated adult abundance	5	Robust populations when lake is full but drought can cause substantial population reductions
Intervention dependence	4	Stream refuge habitats during times of drought are impacted by agricultural water use
Tolerance	4	Tolerant of extreme DO, temperature and alkalinity levels
Genetic risk	4	Little genetic risk
Climate change	1	Goose Lake is likely to be dry more often as climate becomes more arid
Anthropogenic threats	2	See Table 1
Average	3.1	22/7
Certainty (1-4)	3	

Table 2. Metrics for determining the status of Goose Lake tui chub in California, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: The Goose Lake Fishes Working Group was formed with representatives from federal and state agencies, as well as private individuals with interest in the lake, to explore management measures for all native fishes in the basin (Sato 1992a). The involvement of private landowners is particularly critical because many key refuge habitats occur on private land. The persistence of Goose Lake tui chub in the Goose Lake Basin will require active cooperation between Oregon and California because it is likely that most (if not all) natural drought refuges for tui chubs in the Goose Lake basin are in Oregon. Possible management actions include:

1. Determine the suitability of all reservoirs in the drainage as refuges for native fishes and negotiate, if necessary, for minimum pools during periods of drought. Special attention needs to be paid to potential refuges in California.
2. Identify and implement restoration projects to benefit native fishes in the lower reaches of Goose Lake tributaries in both Oregon and California.
3. Actively enforce the prohibition of use of live baitfish and introduction of nonnative fishes into Goose Lake basin, including Oregon. Where possible, eradicate existing populations of alien fishes in ponds and streams.
4. Establish instream flow protections for larger streams in the basin (Oregon: Thomas, Drews, and Dry creeks; California: Lassen and Willow creeks) to ensure adequate flows are present in lower stream reaches to maintain refuge areas and lake level during periods of drought.
5. Conduct a thorough study of the Goose Lake ecosystem, including a study of the distribution and habitat requirements of tui chubs and a systematic survey of the invertebrates present, expanding on studies in Oregon (Heck et al. 2008, Scheerer et al. 2010).
6. Investigate life history and habitat requirements of Goose Lake tui chub to determine what additional species-specific management measures are required.
7. Determine the systematic relationships among tui chubs in Goose Lake and the upper and lower Pit River.

Figure 1. Distribution of Goose Lake tui chub, *Siphateles thalassinus thalassinus* (Cope), in California. Distribution in the Pit River system is uncertain.

COW HEAD TUI CHUB
Siphateles thalassinus vaccaceps (Bills and Bond)

Status: High Concern. Because of its extremely small range and level of human alteration to habitats within that range, the Cow Head tui chub is vulnerable to both human-induced and natural perturbation, especially during periods of severe drought.

Description: The Cow Head tui chub (CHTC) is similar to the Klamath tui chub, *Siphateles bicolor bicolor,* but is differentiated primarily on the basis of more gill rakers (Bills and Bond 1980). The CHTC has 19-25 (mean = 22.5) short, "bluntly rounded" gill rakers, compared with 10-15 gill rakers in *S. b. bicolor*. Other morphological features that characterize this subspecies are: the head is not as deep as in other chubs, is relatively longer, and is convex in profile with a rounded interorbital; a nuchal hump is present, but low; the lower jaw is not overhung by the upper jaw; and the caudal peduncle is relatively deep. Predorsal scales number from 26-35 (mean = 31) and there are approximately 57 lateral line scales. The pectoral fin has 15-17 rays and the pelvic fin has 8-9 rays. Pharyngeal tooth counts are 0,5-4,0; 0,4-4,0; 0,5-5,0. Coloration is similar to other subspecies, except there is a dark lateral stripe with speckles on the head region, especially the cheek and operculum, and on the lower body. Reproductive males and females develop breeding tubercles, especially on the anterior rays of the pectoral fins. Smaller tubercles develop in rows on the edges of the breast scales. In males, tubercles also develop on the scales above the pectorals and across the nape. The largest CHTC on record is 235 mm (Scoppettone and Rissler 2003).

Taxonomic Relationships: The CHTC was first recognized as a distinct form by Hubbs and Miller (1948) and was formally described by Bills and Bond (1980). A genetic study using mitochondrial DNA found that tui chub populations in the Cow Head, Warner and Goose lake basins were closely related and were genetically distinct from other tui chubs, meriting recognition as a single species under the name *Siphateles thalassinus* (Harris 2000). Harris recognized two lineages within *S. thalassinus,* one in Goose Lake and the other in Pluvial Lake Warner, which includes both the Cow Head and Warner basins. Harris's findings supported Hubbs and Miller's (1948) postulation of a possible relationship between CHTC and chubs from the lakes in Warner Valley, Oregon, because of the connection that exists between the Cow Head Basin and the Warner Valley drainage (see Distribution section below). Bills and Bond (1980) had disputed this hypothesis on the basis of differences in gill-raker length and fin and head shapes between the two populations. In 2007, a study using microsatellite DNA allowed greater resolution of the taxonomy of the tui chub of the northwestern Great Basin (Chen et al. 2009). Chen's results supported Harris' systematics regarding *S. thalassinus* and also found that the CHTC was sufficiently distinct to warrant subspecies status as *S. t. vaccaceps.* For a more detailed discussion of tui chub taxonomy, see the Lahontan Lake tui chub, *S. b. pectinifer*, account in this report.

Moyle et al. (1995) and Moyle (2002) list the common name of the chub as "Cowhead Lake tui chub" but Reid (2007) indicated that Cow Head tui chub is more accurate (the chub mostly does not live in the lake) and more consistent with the geographic name.

Life History: Cow Head tui chubs grow to 40-50 mm SL during their first year and 60-80 mm SL during their second year (Moyle unpublished data). By five years of age they reach an average of 100 mm SL, with larger individuals uncommon. The largest individual captured was 235 mm SL and over ten years old (Scoppettone and Rissler 2003). Most tui chubs spawn from late April to early July, beginning in their second to fourth year (Moyle 2002). Although there is little specific information on the reproductive behavior of CHTC, it is believed that they first spawn at two or three years of age (Reid 2006). Fecundity is relatively high, and a female of 100 mm produces approximately 4,000 eggs, which she lays over a series of spawning events. Like other tui chubs, CHTC presumably spawn in groups over aquatic vegetation, algae covered rocks, or gravel with several males attending to each female. Eggs adhere to plants or to substrates. Embryos hatch in 3-6 days and larvae begin feeding soon after hatching (Moyle 2002).

Tui chubs are generally opportunistic omnivores and feed on invertebrates (i.e. snails, clams, insects, and crustaceans), algae and other plant material, and small fish associated with the benthos or aquatic plants (Moyle 2002). Scoppettone and Rissler (2003) examined the stomach contents of 64 CHTC from various sites. Aquatic insects accounted for 28% of the total food by volume, while terrestrial insects accounted for 20%, and algae formed 31%. A single stomach contained an unidentified fish. Unidentifiable animal remains (presumably invertebrates) formed the remaining 19 % of total volume.

Habitat Requirements: Having evolved in the arid Great Basin, tui chubs like CHTC are highly tolerant of high alkalinity, turbidity, high temperatures and low levels of dissolved oxygen (Castleberry and Cech 1986, Moyle 2002, Reid 2006). The most generalized characteristics of suitable CHTC habitat are quiet water with abundant aquatic plants and bottom substrates of sand or finer materials. Thus, CHTC typically occupy pool areas in streams and open water channels with dense beds of aquatic vegetation (Sato 1992b, 1993a, Homuth 2000, Scoppettone and Rissler 2003, 2006).

Distribution: The range of CHTC is limited to the Cow Head Basin in extreme northeastern California and northwestern Nevada (Reid 2006). The Cow Head Basin is relatively small (25,700 acres) and drains north into the Warner Basin of Oregon through Cow Head Slough and Twelve Mile Creek. Cow Head Slough is a small, muddy creek. Under summer water conditions, the creek consists of a series of pools (95%) and riffles (5%) and meanders through a lava canyon approximately 50 m wide. The pools are fairly large, approximately 50 m^2, and are interconnected by shallow trickles. Landownership in the Cow Head Basin is both private and Federal (U.S. Bureau of Land Management (BLM)), but most perennial CHTC habitat is on private land (Reid 2006).

Historically, the basin contained a shallow, marshy lake during wet climate periods. However, Cow Head Lake was altered in the 1930s to allow seasonal drainage of the lake to facilitate farming of the lakebed during spring, summer and fall. The lake still fills during winter in high precipitation years but is drained by active pumping in spring. Populations of CHTC occupy all principal low gradient streams in the basin (Cow Head Slough and Barrel, West Barrel and Keno creeks) and a relatively large population still exists in the permanent channels that drain the lake bed (Scoppettone and Rissler 2006).

Recent surveys have identified seven areas of occupied perennial habitat in five sub-drainages within the Cow Head Basin. Each area is seasonally isolated and is maintained by separate springs or creeks and each contains a population of 1,000-10,000 individuals of all age classes (Reid 2006). During wet periods, stream populations of chubs expand throughout most of the low gradient stream habitat in the basin. Connectivity between stream populations of chubs is generally unobstructed during springtime flows but, as summer progresses and streams dry, all populations become restricted to isolated perennial pools (Reid 2006). Recent genetic research indicates that the genetic variability of CHTC is appropriate for a stream resident population (Chen 2006).

Trends in Abundance: In 1998, when CHTC were proposed for listing as a federally threatened species (see Status), they were only known to occur in Cow Head Slough and Pump Canal (Reid 2006). The only population estimates available at the time were qualitative and based on limited sampling with minnow traps and dip nets (Sato 1992b, 1993a-b, Olson 1997). In 1999, a limited sampling program was conducted with minnow traps in the southern BLM portion of Cow Head Slough and estimated 108 CHTC (39-113 mm FL) were present in this reach (Richey 1999). However, this survey was limited to BLM land that composes only a small portion of the habitat available.

Population estimates conducted in August, 2002 found approximately 3 km of occupied habitat in Barrel Creek and 4 km in Cow head Slough, with a combined population of several thousand chubs over 40 mm (Scoppettone and Rissler 2003). The largest single population was found in the Pump Canal. Although no rigorous population analysis was conducted, four small seine hauls spaced at 200 m intervals produced 936 chubs (22-148 mm) in 2001. Even considering the sampling limitations, if these results were expanded out to the full kilometer of available perennial habitat, a very rough Pump Canal population estimate would exceed 20,000 fish (Reid 2006).

Nature and Degree of Threats: Cow Head tui chubs exist in a small, isolated basin where native aquatic habitats and stream and lake hydrology have been highly altered by human activities, especially agriculture and grazing.

Agriculture. The main threat to the continued existence of CHTC is water diversion from Cow Head Slough for pasture, especially during periods of drought. For example, in 1992, the chubs were largely confined to a short section of slough that was entirely on private land with a water supply that depended, in part, on inflow from an irrigation ditch. The Cow Head lakebed has been used for production crop agriculture in the past and may be utilized as such in the future. Such a transition from ranching to tilled agriculture could have direct impact on water allocation in the Cow Head Basin. Pest control programs that introduce pesticides into the drainage (e.g., USDA-APHIS Grasshopper Control Program) are also a potential threat, although this issue has not been studied in the Cow Head Basin.

Grazing. Grazing in the area has removed riparian vegetation, reducing cover available to fish, making them more vulnerable to predation. Natural predators include garter snakes and fish-eating birds, both of which prey on juveniles and adults, and aquatic insects which prey on eggs, larvae, and juveniles (Reid 2006).

Alien species. While no alien species apparently exist in the watershed at the present time, an illegal introduction of other fish species could easily happen (as has occurred in many other equally isolated parts of the state) and has the potential to threaten the subspecies with rapid extinction due to its limited range.

	Rating	Explanation
Major dams	n/a	
Agriculture	High	Agriculture has degraded habitats and can divert large amounts of water
Grazing	High	Almost all habitat is impacted by grazing
Rural residential	Low	Low population densities and relatively little residential pressure on water supplies
Urbanization	n/a	
Instream mining	n/a	
Mining	n/a	
Transportation	Low	No known impact, but roads run along or cross much of CHTC habitat
Logging	Low	No known impact but may accelerate sedimentation
Fire	Low	No known impact but fires common in desert regions
Estuary alteration	n/a	
Recreation	n/a	
Harvest	n/a	
Hatcheries	n/a	
Alien species	Medium	Although there are no alien species in the watershed at present, illegally introduced species could rapidly deplete populations

Table 1. Major anthropogenic factors limiting, or potentially limiting, viability of populations of Cow Head tui chub in California. Factors were rated on a five-level ordinal scale where a factor rated "critical" could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated "high" could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction unlikely as a result; and a factor rated "no" has no known negative impact to the taxon under consideration. Certainty of these judgments is high. See methods section for descriptions of the factors and explanation of the rating protocol.

Effects of Climate Change: Snow melt and spring recharge from winter rains, the principle sources of water for all CHTC habitat, are likely to substantially decrease as the climate warms, and standard climate models indicate water temperatures are likely to increase 2-4 degrees C by the end of the century. Increased human demand for water is also likely, given the limited supply in the basin and the increased likelihood of long-term drought. Moyle et al. (2013) rated the CHTC as highly vulnerable to extinction from climate change because of its limited habitat in an area that is already very dry and hot, conditions likely to exacerbated by climate change.

Status Determination Score = 2.4 - High Concern (see Methods section Table 2). The CHTC was proposed for federal listing as a threatened species in 1998 but the petition was withdrawn after a conservation action plan was established and new sampling revealed a much larger population than previously known. However, because of the extremely small range and level of human alteration within that range, the CHTC is still vulnerable to both human-induced and natural changes to its habitats. Its status should be re-evaluated every five years or annually during periods of severe drought. The CHTC is listed by the American Fisheries Society as "Endangered" (Jelks et al. 2008) and by NatureServe as "Imperiled".

Metric	Score	Justification
Area occupied	1	Limited to a single, small basin
Estimated adult abundance	4	Relatively large, but variable populations in pump canals with five other smaller populations in perennial habitats
Intervention dependence	3	The largest population lives in an artificial ditch, so management of this habitat is crucial for survival
Tolerance	3	Tolerant of wide range of environmental conditions but, during drought, tolerances may be exceeded
Genetic risk	2	Isolated population with little or no gene flow
Climate change	2	Snow melt and spring recharge for all habitats are likely to decrease
Anthropogenic threats	2	See Table 1
Average	2.4	17/7
Certainty (1-4)	4	Good recent data generated from ESA listing studies

Table 2. Metrics for determining the status of Cow Head tui chub in California, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: On October 22, 1999, stakeholders in the Cow Head watershed signed a conservation agreement (CA) and conservation strategy (CS), with the stated purpose of ensuring the long-term survival of the CHTC (USFWS 1999). Signatories included the US Fish and Wildlife Service, private landowners of Cow Head Lake, Cow Head Slough and the California reach of Barrel Creek (four owners, all CA signatories), principal permittees on BLM lands within the drainage, California and Modoc County Cattlemen's Associations, the California Farm Bureau Federation, the U.S. Bureau of Land Management (BLM - Surprise Field Office), and California Department of Fish and Wildlife (CDFW). The two owners on West Barrel and the single owner for perennial reaches of Barrel and Keno creeks (Nevada) were not original signatories to the CA, because these populations were not recognized at the time;

however, they have been collaborative in providing access to meet the needs of the Conservation Strategy (Reid 2006).

Management directives laid out under phase 2 of the Conservation Agreement and Strategy, which must be implemented, are as follows:

- Create more stable habitat for populations downstream of the Pump Canal.
- Provide greater stability for the chub population upstream of the pump canal by creating, to the extent feasible, additional habitat in the area of historic Cow Head Lake.
- Monitor, as appropriate, the status of chub populations and effectiveness of conservation actions.
- Establish a monitoring program, whereby chub populations are sampled at least once a year.

In addition:

- A study of the environmental requirements of CHTC is needed.
- Slough reaches on public lands should be designated as Areas of Critical Environmental Concern and methods and locations to establish a permanent refuge for the CHTC on public land should be identified.
- Cow Head slough should be fenced to reduce or eliminate cattle grazing in riparian areas.

Figure 1. Distribution of Cow Head tui chub, *Siphateles thalassinus vaccaceps*, in California.

LAHONTAN LAKE TUI CHUB
Siphateles bicolor pectinifer (Snyder)

Status: High Concern. The only verified population in California occurs in Lake Tahoe. This population is declining due to lake changes associated with intense human use of the Lake Tahoe Basin. If tui chub in nearby Prosser, Boca and Stampede reservoirs are confirmed to be *S. b. pectinifer,* then the threats facing this taxa are significantly diminished.

Description: Lahontan Lake tui chubs can reach lengths of 35 to 41 cm FL. The mouth is small, terminal, and oblique. Pharyngeal teeth occur in a single row (5-5, 5-4, or 4-4) and are hooked, with narrow grinding surfaces. This subspecies is characterized by numerous (29-40) long, slender gill rakers; this is the primary characteristic that serves to differentiate it from sympatric *S. b. obesa* (Miller 1951, Vigg 1985, Moyle 2002). The inter-gill raker distances are usually less than the width of the gill rakers themselves. Other morphological characteristics that differentiate *pectinifer* from *obesa* are the more oblique mouth, the slightly concave profile of the head, and uniform blackish or silvery body coloration (Miller 1951). Dorsal and anal fin rays usually number 8, but may range from 7-9; fins are short and rounded. Scales are large, with 44-60 along the lateral line. Spawning males have reddish fins and develop small, white breeding tubercles on their body surfaces; females have reddish fins, slightly enlarged anal regions, protruding genital papilla, and deeper bodies.

Taxonomic Relationships: The systematics of tui chubs are confounded by the fact that many populations appear morphologically similar but are genetically divergent. Distinctive populations occur in the many isolated drainages of the Great Basin, while large lake populations have two sympatric morphs – a pelagic form with many fine gill rakers and a benthic form with fewer, coarser gill rakers. Incomplete meristic and genetic studies add to the taxonomic confusion.

Prominent ichthyologists who have studied the native fishes of the Great Basin have had differing opinions about *S. b. pectinifer*'s taxonomy. Widely varying opinions range from "no valid standing as a taxonomic unit" (La Rivers 1962, p. 420) to assignment of its own genus by J. O. Snyder (1917). Consequently, La Rivers (1962) considered *S. b. pectinifer* to have the most complex taxonomic history of any member of the Great Basin fish fauna. It was first described as *Leucidius pectinifer* by Snyder (1917) who simultaneously described the sympatric 'stream' form as *Siphateles obesus;* the morphological differences between these two forms were great enough for Snyder to place *obesa* and *pectinifer* in different genera. Hubbs and Miller (1943) considered *L. pectinifer* to be a subspecies of *Siphateles obesus* and, thus, called it *Siphateles obesus pectinifer*. Shapovalov and Dill (1950) recognized that both forms were part of the *Siphateles bicolor* complex and renamed them *S. b. pectinifer* and *S. b. obesus*, respectively. Bailey and Uyeno (1964) designated *Siphateles* as a subgenus of *Gila* and designated the fine gill raker tui chub as *Gila bicolor pectinifer*. However, biochemical evidence suggests that tui chubs are more closely related to other Californian minnows than they are to other species of *Gila* (Simons and Mayden 1998). In light of this evidence, Moyle (2002) resurrected the generic name *Siphateles,* first used by Cope (1883) and then by Snyder (1918).

Presently, there are ten *Siphateles* taxa recognized in California (Moyle 2002), although three lack formal taxonomic descriptions: Lahontan Lake tui chub (*Siphateles bicolor pectinifer*), Eagle Lake tui chub, (*S. b. ssp*.), Goose Lake tui chub (*S. t. thalassinus*), Cow Head tui chub (*S. thalassinus vaccaceps*), High Rock Springs tui chub (*S. b. ssp*.), Owens tui chub (*S. b. snyderi*), Mohave tui chub (*S. mohavensis*), Lahontan Creek tui chub (*S. b. obesa*), Klamath tui chub (*S. b. bicolor*), and Pit River tui chub (S. *b. ssp.*). The first four subspecies are included in this report, while the Owens and Mohave tui chubs are already listed as endangered species by both state and federal governments. The Pit River tui chub was listed by Hubbs et al. (1979) as an undescribed subspecies. The tui chubs of the upper Pit River are now considered to be part of the Goose Lake population (Chen et al. 2009) but questions remain about taxonomic affinities of tui chubs distributed in the lower Pit River basin. The High Rock Springs tui chub is extinct.

Recent genetic studies have shown that considerable variation exists among populations of tui chubs, all of which were formerly classified as subspecies of *S. bicolor* (Harris 2000, Chen et al. 2007, Chen et al. 2009). Hence, the subspecific status of *S. b. pectinifer* remains controversial. Not only is the zoogeographic range of *S. b. pectinifer* contained within that of *S. b. obesa,* but Harris (2000) suggested that *S. b. obesa* should be elevated to species status and that *S. b. pectinifer* be submerged within it.

Conversely, studies in both Lake Tahoe and Pyramid Lake, Nevada, indicate that the two forms segregate ecologically (Miller 1951, Galat and Vucinich 1983) and do not interbreed. The existence of sympatric, morphologically distinct tui chub morphs has been repeatedly and consistently observed in large lakes throughout the range of *Siphateles*, most famously in Pyramid Lake and Lake Tahoe but also in Walker Lake, Goose Lake, Eagle Lake and Honey Lake, among others. The main character distinguishing the morphs is number and morphology of gill rakers, although only in Pyramid Lake and Lake Tahoe are the two morphs clearly separated.

It is possible that the distinctive fine gill raker form of tui chub has arisen multiple times in each of these large lake systems, although it may be just a single lineage in the Truckee basin. Similar situations of parallel evolution in California fish taxa may exist, such as the run timing of summer steelhead populations and bony plate development and migratory behavior of threespine stickleback in coastal California streams. A sizeable literature base has developed on trophic polymorphism; of particular relevance to lake dwelling tui chub are trophic polymorphisms among other fishes in lacustrine environments. Examples include char in arctic lakes, whitefish in Canadian and Idaho lakes, cichlids in African Rift lakes, threespine stickleback in British Columbia lakes and sunfishes in the eastern United States. References can be found compiled in reviews on the subject by Smith and Skülason (1996) and, more recently, by Dayan and Simberloff (2005). Until taxonomic studies are completed, all distinctive populations of tui chubs should be managed as separate taxa.

Life History: Lahontan Lake tui chub feed mostly on zooplankton, especially cladocerans and copepods, but also consume benthic insects such as chironomid larvae, annelid worms and winged insects such as ants and beetles (Miller 1951, Marrin and

Erman 1982). They are primarily mid-water feeders, with gill-raker structure adapted to feeding on plankton. In contrast, the co-occurring *obesus* form is primarily a benthic feeder (Miller 1951). A comparison of stomach contents of both subspecies captured together in bottom-set gill nets indicated *obesa* had fed on benthic insects such as chironomids and trichopterans, while *pectinifer* had fed on planktonic microcrustacea (Miller 1951). There is no significant ontogenetic niche shift in diet for *pectinifer*; it feeds on plankton throughout its life (Miller 1951). In Pyramid Lake, both types of tui chubs feed primarily on zooplankton (mostly microcrustaceans) when less than 25 mm FL, but the *obesa* subspecies feed increasingly on benthic and terrestrial macroinvertebrates as they become larger (Galat and Vucinich 1983). There is an ontogenetic change in gill-raker numbers in the two forms that accompanies the differentiation of diets. When less than 25 mm FL, the two morphs are indistinguishable, even based on gill-raker counts, but the gill-raker number increases in *pectinifer* with size until the two forms are readily distinguishable at ≥50 mm FL.

Tui chubs are preyed upon by large trout and, to a lesser extent, by birds and snakes. Examination of stomachs of rainbow trout and lake trout in Lake Tahoe revealed that 10% and 7%, respectively, of their stomach contents consisted of tui chubs (Miller 1951).

In Lake Tahoe, spawning apparently occurs at night during May and June and possibly later (Miller 1951). By early August, females do not have mature ova. Lahontan Lake tui chubs spawn by 11 cm SL (Miller 1951). They are probably serial spawners, capable of reproducing several times during a season (Moyle 2002). Snyder (1917) documented that reproductive adults spawned in near-shore shallow areas over beds of aquatic vegetation and found fertilized eggs adhering to the aquatic vegetation. He noted that young remained in the near-shore environment until winter when they were 1-2 cm in length and then migrated into deeper water offshore.

Growth (length increments) of tui chubs is linear until about age 4, when weight increases more rapidly and length increments decrease. The largest Lahontan Lake tui chub caught in Lake Tahoe was 13.7 cm SL (Miller 1951). These fish are considerably smaller than the tui chubs in Walker Lake, Nevada, where they grow to 21 cm SL (Miller 1951). It is likely that the largest Lahontan Lake tui chubs are in excess of 30 years old (Scoppetone 1988, Crain and Corcoran 2000).

Habitat Requirements: Lahontan Lake tui chub are schooling fish that inhabit large, deep lakes (Moyle 2002). They seem to be able to tolerate a wide range of physicochemical water conditions based on the fact that they are found in oligotrophic Lake Tahoe as well as in Pyramid Lake, a mesotrophic and highly alkaline lake. In Lake Tahoe, the larger fish (>16 cm TL) exhibit a diel horizontal migration by moving into deeper water (>50 m) during the day and back into shallower habitat at night (Miller 1951). However, they always remain high in the water column. Smaller individuals occupy shallower water. Additionally, there is a seasonal vertical migration, with fishes located deeper in the water column during winter and moving back into the upper water column during summer (Snyder 1917, Miller 1951). Algal beds in shallow, inshore, areas appear to be necessary for successful spawning, embryo hatching and larval survival.

Distribution: Lahontan Lake tui chubs are found in Lake Tahoe and Pyramid Lake, Nevada, which are connected to each other by the Truckee River, and in nearby Walker Lake, Nevada. Plankton-feeding populations of chubs in Stampede, Boca, and Prosser reservoirs on the Truckee and Little Truckee rivers may also be Lahontan Lake tui chubs because they have a superior oblique mouth and fine gill rakers and are never found in tributary streams (Marrin and Erman 1982, D. Erman, pers. comm.). Other tui chub populations in the Lahontan basin of uncertain taxonomic affinity also occur in Topaz Lake on the California-Nevada border and in Honey Lake, Lassen County.

Trends in Abundance: Actual abundance is not known, but is likely quite small compared to historic numbers. The Lake Tahoe population is the only confirmed population in California, but the chubs in Stampede, Boca, and Prosser reservoirs may also belong to this subspecies, although no sampling or analysis has been carried out to verify this assertion. Only small numbers have been collected from Lake Tahoe in recent years (P. Budry, Utah State University, unpubl. data) and the Lahontan Lake tui chub has not been studied in Lake Tahoe since the late 1940s (Miller 1951). In the intervening years, the zooplankton community in the lake has changed dramatically. *Daphnia*, which are an important prey of adult chubs, have been nearly eliminated (Richards et al. 1975) by introduced kokanee salmon (*Oncorhynchus nerka*) and opossum shrimp (*Mysis relicta*), both of which feed on zooplankton.

Putative *S. b. pectinifer* populations in the three California reservoirs mentioned above and verified *S. b. pectinifer* populations in Pyramid and Walker lakes in Nevada are large but abundance estimates are lacking.

Nature and Degree of Threats: Until the taxonomy of peripheral populations has been decided, the future of Lahontan Lake tui chubs in California essentially depends on their ability to persist in Lake Tahoe (Table 1).

Major dams. Dams on California tributaries to the Truckee River are apparently a mixed blessing for lake tui chubs. They allow for diversion of water, lowering the level of Pyramid Lake, Nevada and potentially negatively affecting tui chubs there, while creating potential habitat in their reservoirs (additional habitats within California). The reservoir populations are unstudied, however, and may not be *S. b. pectinifer*.

Urbanization and rural development. Water diversion, waste water treatment, wetlands destruction and increased sedimentation from ever increasing development in the Lake Tahoe Basin have altered the lake’s physical environment; however, it is unknown how these stressors affect tui chubs. Lake Tahoe has been undergoing physical and chemical change as the result of nutrients, sediments and pollutants entering the lake from surrounding development, as well as more distant sources. Shoreline development has presumably also negatively affected tui chubs because they spawn in shallow water and larvae may require warm habitats with adequate cover for the first few weeks of life (although this is not known). There is some indication that the marsh that is now the development called Tahoe Keys (a major source of alien species in the lake) was once an important rearing area for tui chubs (Miller 1951).

Logging. The Tahoe Basin has been heavily logged in the past and some logging continues, contributing to sediment delivery. Effects on tui chubs are likely minimal, especially when compared to other factors changing the lake.

Fire. The entire Tahoe Basin is increasingly prone to catastrophic fire which may, in turn, deliver huge sediment loads to the lake. This may affect tui chub spawning and feeding and generally change the nature of Lake Tahoe, especially as climate change effects are predicted to increase the frequency and intensity of fire in this region.

Recreation. The Lake Tahoe region is a year-round recreation destination and the increasing influx of permanent residents and visitors drives most of the changes that affect fishes and other organisms in the lake, from water chemistry (e.g. via air pollution) to sedimentation and increasing eutrophication (e.g., surface run off of nutrients and pollutants from ski resorts, casinos, golf courses, recreational parks and trail development).

Alien species. The greatest impacts to the aquatic ecosystem of Lake Tahoe have been the result of introductions of non-native fishes and invertebrates. Mysid shrimp and kokanee salmon have largely eliminated *Daphnia*, which were the major food source of tui chubs, while introduced lake trout (*Salvelinus namaycush*), rainbow trout (*O. mykiss*), and brown trout (*Salmo trutta*) prey on them. . In recent years, the invasions of predatory smallmouth bass (*Microterus dolomieui*) and largemouth bass (*M. salmoides*) into the lake constitute an additional threat to the tui chub population, especially since these predatory centrarchids occupy chub spawning and rearing habitats. As the lake becomes more eutrophic, it may actually be able to support more fish, including tui chubs, but the number and abundance of alien species will also likely increase. In contrast, the alkalinity of Pyramid Lake, Nevada, has largely prevented the establishment of non-native species, with the exception of Sacramento perch (*Archoplites interruptus*). Adult perch (<300 mm) feed largely on tui chubs (Galat et al. 1981).

	Rating	Explanation
Major dams	Medium	Reservoirs may have created habitat but they also reduce freshwater flow into Pyramid Lake
Agriculture	n/a	
Grazing	Low	Grazing occurs in the Tahoe Basin which may contribute to changes in water quality
Rural residential	Medium	Water diversion, waste water treatment, wetlands destruction and increased sedimentation in the Tahoe Basin have changed the lake's physical environment; direct impacts to tui chubs are unknown
Urbanization	Medium	Same as above
Instream mining	Low	No known effect
Mining	Low	Legacy effects are largely unstudied
Transportation	Low	A large portion of the suspended sediment in Lake Tahoe has its origins in sand applied to de-ice roads
Logging	Low	Logging contributes sediment delivery to the lake, with much greater impacts in the past
Fire	Low	The entire Tahoe basin is increasingly prone to catastrophic fire; direct impacts to tui chubs are likely to be minimal
Estuary alteration	n/a	
Recreation	Medium	Recreational use of the Tahoe Basin is the primary force driving the area's rapid development
Harvest	n/a	
Hatcheries	n/a	
Alien species	High	Long-term impacts from introduced predators and competitors may be reducing populations

Table 1. Major anthropogenic factors limiting, or potentially limiting, viability of populations of Lahontan Lake tui chub in California. Factors were rated on a five-level ordinal scale where a factor rated "critical" could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated "high" could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction unlikely as a result; and a factor rated "no" has no known negative impact to the taxon under consideration. Certainty of these judgments is moderate. See methods section for descriptions of the factors and explanation of the rating protocol.

Effects of Climate Change: The following list includes the predicted impacts and potential consequences of climate change to Lake Tahoe and the northern Sierra Nevada:

- A shift in winter precipitation from snow to rain. This shift in annual hydrologic timing could increase the transport of fine sediment and nutrients to the lake.
- A shift toward earlier snowmelt (Dettinger and Cayan, 1995; Cayan et al, 2001; Stewart et al., 2005). A change to the volume, temperature and timing of streamflow into the lake could increase the lake's thermal stability and could

possibly prolong the residence time of fine sediment near the lake surface, further decreasing water transparency.

- An increase in the average temperature of Lake Tahoe (Coats et al. 2006). An increase in temperature is likely to increase Lake Tahoe's resistance to mixing which could have profound effects on the lakes aquatic community. Thermally driven disruption to historic mixing conditions in Lake Tahoe would favor introduced species over native species.

The combination of these effects could change the water chemistry and temperatures in Tahoe and Pyramid lakes. These effects could also result in reservoirs becoming too low to support tui chub populations. While the Lahontan Lake tui chub is presumably quite physiologically tolerant, changes to its food supply may result in population declines. These predicted impacts are speculative; however, studies should be conducted to document changes and develop trend data in order to inform conservation strategies to address climate change. Moyle et al. (2013) rated this form as "less vulnerable" to extinction from the effects of climate change than most other native fish species because of its refuge in Lake Tahoe.

Status Determination Score = 2.4 - High Concern (see Methods section, Table 2)**.** The Lahontan Lake tui chub does not appear to be at risk of extinction; however, the status of the endemic population in California (Lake Tahoe) is largely unknown (Table 2). The Lake Tahoe population may have declined from its historic abundance, while the population in Pyramid Lake, Nevada continues to be large. The taxonomic identity and status of reservoir populations is not known.

Metric	Score	Justification
Area occupied	1	Found only in Lake Tahoe in CA
Estimated adult abundance	2	Population size in Lake Tahoe uncertain; no surveys conducted in over 60 years
Intervention dependence	5	No intervention required at this time
Tolerance	4	Relatively tolerant
Genetic risk	1	Genetics not well understood but the single confirmed population in California is isolated in one (albeit large and deep) lake
Climate change	2	Effects expected to be severe in the Lake Tahoe area
Anthropogenic threats	2	See Table 1
Average	2.4	17/7
Certainty (1-4)	2	Questions about taxonomy and lack of recent population surveys influence status evaluation

Table 2. Metrics for determining the status of Lahontan Lake tui chub in California, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: Surveys of Lake Tahoe and other Lahontan basin waters (Honey Lake, Topaz Lake, Stampede, Boca, and Prosser reservoirs) are needed to determine the distribution and abundance of Lahontan Lake tui chub in California. Equally important, a taxonomic study is needed of all potential populations of this subspecies in California and Nevada. A study comparing genetics to morphology, especially of sympatric morphs found in large lake systems, would be of particular interest. These studies are needed in order to develop a management plan to protect tui chub diversity. Currently, persistence of this form depends on the management of the water quality and biota of Lake Tahoe, including control of non-native, predatory fishes.

Figure 1. Distribution of verified Lahontan Lake tui chub, *Siphateles bicolor pectinifer* (Snyder), in Lake Tahoe, California.

EAGLE LAKE TUI CHUB
***Siphateles bicolor* ssp.**

Status: Moderate Concern. Although abundant, the Eagle Lake tui chub is endemic to a single, highly alkaline, terminal lake.

Description: No robust description of the Eagle Lake tui chub exists but they resemble other chubs in the *Siphateles pectinifer/obesa* complex. Eagle Lake tui chubs have a range of 12-28 gill-rakers on the first arch. Gill rakers are bimodally distributed, with peaks at 17-18 and 23-25, respectively (Kimsey 1954). Two body forms are present in the lake, one obese with a pronounced nuchal hump and the other slender. However, all other meristic characters are smoothly distributed across the entire population and Kimsey (1954) found no correlation between body form and gill raker number. Spawning individuals of both sexes develop reddish coloration on the fins. Males also develop small, white breeding tubercles on their body surfaces, while females develop slightly enlarged anal regions, protruding genital papilla, and deeper bodies (Kimsey 1954). Maximum size appears to be around 45 cm TL.

Taxonomic Relationships: This form was once regarded as a hybrid between *S. b. pectinifer* and *S. b. obesa* (Kimsey 1954, Hubbs and Miller 1943, Hubbs et al. 1974), based on gill raker counts. However, lack of other hybrid characters and the isolation of this lake from other parts of the Lahontan Basin indicate a long, separate evolutionary history. For a detailed discussion of tui chub taxonomy, see the Lahontan lake tui chub, *S. b. pectinifer,* account in this report.

Life History: Kimsey (1954) conducted the most comprehensive study of the natural history of this chub. Eagle Lake tui chub shoal in open waters of the lake, forming schools of fish of similar sizes. During the spawning season, schools break up and mature adults congregate in near-shore, shallow areas with dense beds of aquatic plants. At this time immature fish remain scattered throughout the lake.

Spawning occurs from mid-May through the beginning of July. Adults in spawning aggregations mill around dense macrophyte beds at about 1m depth and deposit adhesive eggs that stick to aquatic plants (*Myriophyllum spicatum, Ceratophyllum demersum, Potamogeton* sp.). The newly laid eggs are a pale orange-yellow, but color fades to a lighter straw-yellow after some time. Kimsey (1954) estimated the fecundity of a 27-cm female tui chub at 11,200 mature eggs, but he considered this a conservative estimate because not all eggs mature simultaneously. Thus, tui chubs are probably serial spawners, capable of reproducing several times during a season (Moyle 2002).

Newly hatched larvae are well developed and immediately begin to feed on rotifers, diatoms, desmids, and other microscopic material. Larval body plans of western cyprinids are extremely similar; however, larval tui chub develop a nuchal hump at just 5.5 mm (Remple and Markle 2005). Juveniles aggregate along the lakeshore in huge schools until about December, at which time they move into deeper waters. The young-of-year feed on zooplankton and on terrestrial insects blown into the lake from the surrounding forest (G. Grant, unpublished report; Eagles-Smith 2006). Adult Eagle Lake tui chubs appear to be opportunistic omnivores, although their diet shifts towards benthic organisms as they grow larger (Eagles-Smith 2006). Larger fish also show a shift into feeding at higher trophic levels, presumably because of their consumption of benthic invertebrates (Eagles-Smith 2006). The bulk of their stomach contents usually consist of detritus, with small quantities of algae, benthic and planktonic invertebrates, and aquatic macrophytes (Kimsey 1954). P. Moyle and students (unpublished data) found gut-contents of adult tui chub in Eagle Lake to consist of 83% detritus, 2% algae and 15% invertebrates. Eagle Lake tui chubs are also a key part of the lake ecosystem, as a major intermediary link between lower trophic levels (detritus, algae, invertebrates) and higher levels such as Eagle Lake rainbow trout and piscivorous birds (Eagles-Smith 2006). The lake supports exceptionally large breeding populations of osprey (*Pandion haliaetus*), western grebes (*Aechmorphorus occidentalis*), Clark's grebes (*A. clarkii*), eared grebes (*Podiceps nigricollis*) and other fish-eating birds; these abundant birds can be observed diving for and consuming large quantities of tui chub in most months of the year (J. Weaver, CDFW, unpublished observations).

Kimsey (1954) aged Eagle Lake tui chubs at 6-7 years using scales; however, Crain and Corcoran (2000) found that if opercular bones were used instead, the ages of adult tui chubs (30-35 cm SL) ranged from 12-33 years. Growth is rapid until age of 4 years, slows until age 7 and is very limited after 8 years (Crain and Corcoran 2000). Such ages and growth rates appear to be typical of tui chubs and suckers (Catostomidae) of the terminal lakes of the Great Basin (Scoppettone 1988).

Habitat Requirements: Eagle Lake is a large (22,000 ha) lake at an elevation of 1,557 m. It is estimated that 14% of the annual water budget for Eagle Lake is provided from stream flow, 38% from direct precipitation and 48% from sub-surface flow (Bureau of Land Management, Eagle Lake Water Budget 2010). Surface water enters the lake from Pine Creek and a number of smaller creeks, all of which are ephemeral, flowing only during winter and drying out by late spring. There is no outflow from Eagle Lake. Bly Tunnel (constructed in the 1920s), which was used to release small amounts of water into Willow Creek, a tributary to Honey Lake, is now closed off (P. Divine, CDFW, pers. comm. 2012). Most water loss is through evaporation.

Eagle Lake is highly alkaline (pH about 9 in most years), clear (secchi depth typically 4-6 m), and cool (summer temperatures rarely >20˚C at the surface). Average depth is 5-7 m, with a maximum depth of 30 m (in the southern basin). Eagle Lake tui chubs are found throughout the lake, but mature fish exhibit a seasonal migration from the deep southern basin of the lake in winter to the more shallow middle and northern basins, where spawning occurs, in spring. They require beds of aquatic vegetation in shallow, inshore areas for successful spawning, egg hatching, and larval survival (Kimsey 1954).

Distribution: This form is confined to Eagle Lake, Lassen County, California. Kimsey (1954) found no stream populations. However, tui chubs have been consistently found in three decades of fish surveys of upper Willow Creek (P. Moyle, unpublished data), which historically connected to Eagle Lake (outflow) via the Bly tunnel (BLM 2007).

Trends in Abundance: At present, tui chubs are the most abundant fish in Eagle Lake and support large populations of fish-eating birds and the piscivorous Eagle Lake rainbow trout. There is no indication that they are less abundant than they were formerly, but the population may suffer if lake levels continue to drop and alkalinity increases. Eagle Lake is currently (2011-13) at near-record low levels, so tui chub populations may decline with changing water chemistry and reduced habitat, particularly dense stands of tule beds they utilize for cover, many of which are now stranded on the dry shoreline.

Nature and Degree of Threats: Eagle Lake tui chubs and the entire unique Eagle Lake ecosystem face two major threats: alien fishes and extremely depressed lake levels. The greatest threat to Eagle Lake tui chub is reduced lake levels due to extended drought. Eagle Lake is a terminal lake, from which water leaves naturally by evaporation (90%) and subsurface flow (10%), resulting in its very alkaline waters (BLM 2010). Lesser threats include recreational development of the lakeshore and surrounding watershed as well as the continued effects of livestock grazing (Table 1). For a thorough discussion of all factors affecting the watershed, see the Eagle Lake rainbow trout account in this report.

Agriculture. The water diversion through Bly Tunnel has been completely closed. Other agriculture using ground water may influence lake levels; however, there are insufficient ground water data to assess potential impacts from ground water use outside the basin.

Alien species. With the complete closure of the Bly Tunnel, in combination with the unlikely event of a long wet period, lake levels could actually rise. Under such conditions, the lake would become considerably less alkaline and be able to support introduced fishes, as it did in the early 1900s, when largemouth bass and brown bullheads were common. These introduced fishes died out when lake levels dropped during the drought of the 1930s. The impact these fishes had on chub populations is not known. However, the effects of introduced diseases, predators, parasites, or competitors from future fish introductions could be disastrous to the lake ecosystem, including introductions of more alkalinity-tolerant species. Although illegal, introduction of bait or sport fishes by the public remains a possibility.

Effects of Climate Change: Climate change predictions indicate that snow melt and winter rain, the principle sources of recharge water for Eagle Lake, are likely to substantially decrease in the future. Temperature models indicate 2-4 degree rises in average air temperature by the end of the century, or higher, which will increase evaporation rates from the lake. Thus, the lake may recede to lower levels than experienced historically with alkalinities that may inhibit tui chub reproduction. Arguably, existing record low lake levels are already the result of climate change, at least in part. Moyle et al. (2013) rated Eagle Lake tui chub as “critically vulnerable” to climate change because of the potential for Eagle Lake levels to become so low and alkaline the lake can no longer support fish life.

	Rating	Explanation
Major dams	n/a	
Agriculture	Low	Agriculture using ground water may influence lake level; closure of Bly Tunnel (2012) a significant positive development
Grazing	Medium	Grazing affects most tributary streams and meadow systems by changing the timing and quality of surface water inflow to the lake and degrading riparian and instream habitats
Rural residential	Low	Residential population of the basin is limited but increasing
Urbanization	n/a	
Instream mining	n/a	
Mining	n/a	
Transportation	Low	Paved roads surround most of the lake and an extensive network of unpaved roads exists throughout the basin; impacts are unknown
Logging	Low	Watershed has been heavily logged; effects on Eagle Lake fishes, including tui chub, are unknown
Fire	Low	Entire watershed prone to fire; predicted climate change outcomes may increase frequency and intensity of fires; effects on lake ecology unknown
Estuary alteration	n/a	
Recreation	Low	Fishing and boating on the lake present little threat
Harvest	Low	Eagle Lake sustains a small sport fishery for tui chub, but no detrimental effects are known
Hatcheries	Low	The Eagle Lake rainbow trout, the principal (albeit native) fish predator of tui chub, is sustained by a large hatchery operation; however, it is unknown if hatchery stocking has created an artificially larger population than existed historically in the lake
Alien species	Low	Introduction of alien fishes could negatively affect the native fish community, provided lake levels increase and alkalinity decreases

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

Status Determination Score = 3.3 - Moderate Concern (see Methods section, Table 2). The population of Eagle Lake tui chub is large and the presence of many age classes in the population, including very old fish, suggests that they can outlast long periods of conditions unfavorable for reproduction. Nevertheless, their isolation in one location indicates a vulnerability that justifies continuing to recognize them as a Species of Special Concern and developing a monitoring program for them (Table 2).

Metric	Score	Justification
Area occupied	1	Restricted to Eagle Lake
Estimated adult abundance	5	Robust
Intervention dependence	5	No intervention needed at present
Tolerance	4	Broad tolerances but alkalinity of lake could become extremely high during sustained drought, inhibiting reproduction
Genetic risk	3	Single population
Climate change	1	Vulnerable in entire native range
Anthropogenic threats	4	See Table 1
Average	3.3	23/7
Certainty (1-4)	3	

Table 2. Metrics for determining the status of Eagle Lake tui chub, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: Eagle Lake should have special recognition as a one of the few lakes in the western United States that has a basically unaltered ecosystem, containing only native species and relatively low concentrations of contaminants. In particular, the lake should have recognition as habitat for its community of native fishes, including the endemic Eagle Lake rainbow trout, which feeds, in part, on tui chubs.

A management plan for the entire Eagle Lake basin (including tributary streams) should be developed, as discussed in the Eagle Lake rainbow trout account. One focus of this plan should be the establishment of a governance structure that can evaluate and regulate planned developments in the basin to ensure they are compatible with maintaining the integrity of the lake's ecosystem, including maintaining its large populations of fish-eating birds and the endemic fishes that support them.

In addition, a monitoring program for chubs should be established, as part of a broader program to monitor and manage Eagle Lake for its distinctive biota, as well as to ensure the continued absence of alien species. In particular, population age structure should be examined closely during periods when lake levels are low and alkalinities high and contingency plans should be developed in order to maintain the chub population if reproduction fails repeatedly.

Figure 1. Distribution of Eagle Lake tui chub, *Siphateles bicolor* ssp., in California.

PACIFIC LAMPREY
Entosphenus tridentatus

Status: Moderate Concern. Pacific lampreys are in decline throughout their range in California. However, they are still widespread so the species does not appear in immediate danger of extinction in the state. Some local or regional (e.g., southern California) populations may face considerably higher threat of extirpation in the near future.

Description: Pacific lampreys are the largest (> 40 cm TL) lampreys in California. However, landlocked Pacific lamprey populations may have dwarf (15-30 cm TL) morphs. The sucking disc is characterized by having sharp, horny plates (teeth) in all areas (Vladykov and Kott 1979). The crescent-shaped supraoral lamina is the most distinctive plate, with three sharp cusps, of which the middle cusp is smaller than the two lateral ones. There are four large lateral plates on both sides of the supraoral lamina. The outer two lateral plates are bicuspid, while the middle two are tricuspid (formula 2-3-3-2). The tip of the tongue has 14-21 small points (transverse lingual lamina), of which the middle one is slightly larger than the rest. The two dorsal fins are discontinuous but the second dorsal is continuous with the caudal fin. Adults generally have 62-71 body segments (myomeres), while juveniles have 68-70 body segments between the anus and last gill opening (Wang 1986). The diameter of the eye and oral disc, respectively, are 2-4 percent and 6-8 percent of the total length. Males tend to have higher dorsal fins than females, lack a conspicuous anal fin and possess genital papillae. Body color varies by developmental stage. For juveniles (ammocoetes), the body and lower half of the oral hood is dark or medium brown, with a pale area near the ridge of the caudal region. Newly metamorphosed juveniles (macropthalmia) are silvery with a slightly bronze cast. Spawning adults are usually dark greenish-black or dark brown in color.

Taxonomic Relationships: The use of the genus name *Entosphenus* reflects the phylogenetic study of Gill et al. (2003) that places this genus as a separate lineage from *Lampetra,* into which all western North American lampreys had been lumped. Genetic analysis of populations of from British Columbia to southern California have found little variation among populations, suggesting that gene flow occurs readily throughout their range (Goodman et al. 2008, Docker 2010). However, populations in the northern part of the range exhibit reduced genetic richness (Goodman et al. 2008), perhaps reflecting locally adapted population segments.

Pacific lampreys have given rise to landlocked populations throughout their range, including predatory species (e.g., *E. similis;* refer to separate species accounts). Populations have also become isolated upstream of reservoirs resulting from dam construction, including populations in Clair Engle Reservoir (Trinity River) and Clear Creek, upstream of Whiskeytown Reservoir (Brown and May 2007). Considerable overlap of morphometric characters exists between Pacific lamprey and its derivatives, as well as between predatory and nonpredatory forms, especially in the Klamath River basin (Bond and Kan 1973, Bailey 1980, Lorion et al. 2000), so careful examination is required for identification. Studies of mitochondrial DNA (Docker et al. 1999) and statistical analysis of morphometric characteristics (Meeuwig et al. 2006) show promise in resolving interrelationships among species.

Life History: Pacific lampreys have more diverse life histories than generally recognized. Within the same river system they may have more than one run (Anglin 1994) or individuals that do not migrate to sea. For example, two forms of Pacific lamprey exist in the Trinity River, one smaller and paler than the other, representing either separate runs or resident and anadromous individuals (T. Healey, CDFW, pers. comm. 1995). It is possible that lamprey in the Klamath and Eel rivers, as well as other large river systems, have a number of distinct runs, similar to salmon. One indication is that many adults migrate upstream and hide under logs and boulders for months until they mature, with a life history akin to that of summer steelhead or spring-run Chinook salmon (Beamish 1980, ENTRIX 1996). Two distinct runs may exist in the Klamath River: a spring-run of adults that spawn immediately after upstream migration and a fall-run of individuals that wait to spawn until the following spring (Anglin 1994). A large spring-run and smaller fall-run have been observed in the Russian River (Brown et al. 2010); the two runs were observed from 2000 to 2007 (S. Chase, Sonoma County Water Agency, unpubl. data) with the use of underwater video (at Mirable, 37 rkm), primarily from the beginning of August to the onset of heavy rains (November to December), as well as in the spring months. The general run trend is low numbers of migrants in October and November and higher numbers in the spring.

Adult Pacific lampreys are micropredators (i.e., they feed on prey larger than themselves) during their oceanic existence, consuming the body fluids of a variety of fishes, including salmon and flatfishes (Beamish 1980) and marine mammals (Close et al. 2002). Beamish (1980) found that 14-45 percent of the salmon returning to British Columbia had scars from lamprey predation. Similar data are not available for salmon in California. Adult lampreys themselves are prey for other fishes, including sharks, and are often found with parts of their tails missing. Sea lions, near the mouth of the Rogue River, Oregon, have been observed eating large numbers of migrating lampreys (Jameson and Kenyon 1977). Lamprey predation is largely confined to fishes that occupy estuaries and nearshore coastal areas. However, some individual lampreys have been caught in waters up to 70 m deep (Beamish 1980) and as far as 100 km from shore (Close et al. 2002). The oceanic phase lasts approximately 3-4 years in British Columbia, but is likely of shorter duration in southern waters. Pacific lamprey predation appears to have little effect on fish populations (Moyle 2002, Orr et al. 2004).

Adult (30-76 cm TL) spawning migrations usually take place between early March and late June, but migration has also been documented in January and February (ENTRIX 1996, Trihey and Associates 1996b), as well as in July in northern streams. Spawning migrations have been documented in August and September in the Trinity River (Moffett and Smith 1950). Most upstream movements occur in surges at night, with some individuals migrating fairly continuously over the course of two to four months. In the Santa Clara River (Ventura County), migration was initiated after the sand bar blocking the lagoon at the mouth was breached by winter rains in January, February, or March; adults reached a fish ladder 16.8 km upstream within 6-14 days of the breach (ENTRIX 1996). In the Santa Clara River, lampreys migrated mostly during high flows, but also moved in flows ranging from 25 to 1700 m^3/min (ENTRIX 1996).

Lampreys will migrate considerable distances and are stopped only by major barriers, such as dams. Lampreys were observed spawning in Deer Creek (Tehama County), about 440 km from the ocean (P. Moyle, unpublished observation). Presumably, migrations of more than 500 km were once common. In the Klamath River, Humboldt County, radio tagged lampreys migrated an average of 34 km over the course of 25 days at a travel rate of 2 km/day (McCovey et al. 2007). Adults do not feed during spawning migrations (Beamish 1980) but can survive extended periods (months to two years) without food, allowing them to migrate long distances (Whyte et al. 1993). Pacific lampreys seem to have poorly developed homing abilities (Hatch and Whiteaker 2009). If this is true, then lamprey populations are likely regulated by source-sink dynamics, where large river populations (such as those historically present in the Eel River) sustain populations in smaller adjacent rivers or tributaries, where localized extinctions can occur periodically due to stochastic events such as floods and droughts (e.g. a drying event, even short-term, could eliminate multiple age classes of ammocoetes). The source-sink model would also explain persistence of lampreys in habitats that are often unsuitable (e.g. in southern California rivers). The sink populations may disappear as source populations shrink and the number of potential recruits to the sink population becomes reduced or non-existent. This model is speculative but seems to fit with recent findings of lamprey behavior and population dynamics and is consistent with ecological theory (metapopulation dynamics).

Once at a spawning site, typically in a low-gradient riffle, both sexes build a nest depression 21-270 cm in diameter (Gunckel et al. 2009), with depths of 30-150 cm, at temperatures of 12-18 °C (Moyle 2002). Depths of nests range from 30-82 cm (mean of 59 cm) in the American River, while ranging from 36 to 73 cm (mean of 50 cm) in Putah Creek. Nest construction has been observed in water as deep as 1.5 m in Deer Creek, Tehama County (Moyle, unpublished observations). Water velocity at nests in the American River ranged from 24-84 cm/sec, in comparison to 17-45 cm/sec in Putah Creek. Although Pacific lampreys most commonly spawn in flowing water, spawning has also been observed in lentic systems (Russell et al. 1987). Lampreys attach themselves to the downstream end of rocks and swing vigorously in reverse to remove substrates during nest construction. More than one individual may pull at the same rock until the combination of pulling and pushing dislodges the rock (Stone 2006). Adults may test several nest sites ('false digs') before fully digging a nest (Stone 2006). Nests are shallow depressions, with piles of stones at either the downstream (Moyle 2002) or upstream (Susac and Jacobs 1999) end of the nest. In order to mate, the female attaches to a rock on the upstream end of the nest, while the male attaches himself to the head of the female and wraps his body around hers. Occasionally, both will attach to rocks while staying side by side (Wang 1986). Eggs and milt are released when both vibrate rapidly. Fertilized eggs float downstream, where most adhere to rocks at the downstream end of the nest.

After spawning, lampreys loosen sediment upstream of the nest to cover the embryos. Spawning is repeated in the same nest until the adults are spent. Males may mate with more than one female (Wang 1986). About 48 individuals were observed using the same nest in the Smith River, Oregon (Gunckel et al. 2006). The average time spent in spawning areas is less than seven days for both sexes (Brumo 2006). Adults may defend their nests; Stone (2006) observed a male using his oral disc to remove a sculpin

(*Cottus* spp.) from its nest in Cedar Creek, Washington. Both sexes usually die after spawning. However, some adults may live to spawn for one more year in Washington streams (Michael 1984). Repeat spawning may also occur in the Santa Clara River, as indicated by the fact that live adults have been caught in downstream migrant traps (ENTRIX 1996). The fecundity of females ranges from 20,000 to 238,000 eggs (Kan 1975).

At 15 °C, embryos hatch in 19 days. Upon hatching, ammocoetes stay in the nest for a short period of time and then swim into the water column where they are washed downstream to areas of sand or mud. Ammocoetes burrow into soft stream sediments tail first, at which point they begin filter feeding by sucking organic matter and algae from stream substrates. Survival to this stage may be related to stream discharge at time of spawning and density dependent effects (e.g., amount of rearing habitat and prey items) associated with ammocoete abundance (Brumo 2006). Ammocoetes leave their burrows and drift to other areas at night throughout their freshwater residency (White and Harvey 2003). Larger ammocoetes commonly drift in spring high flows, while smaller ammocoetes drift during the summer. Consequently, they can be trapped during much of the year (Moffett and Smith 1950, Long 1968). In the Trinity River, ammocoetes as small as 16 mm recolonized areas from which they had been removed by winter floods (Moffett and Smith 1950)

The ammocoete stage probably lasts 5-7 years, at the end of which ammocoetes measure 12-14 cm TL and metamorphosis to macropthalmia begins. Lampreys develop large eyes, a sucking disc, silver sides and dark blue backs during metamorphosis. Their physiology and internal anatomy (McPhail and Lindsey 1970) also change dramatically. Physiological changes allow adult lampreys to tolerate salt water, which is lethal to ammocoetes (Richards and Beamish 1981). Saltwater tolerance coincides with the opening of the foregut lumen (Richards and Beamish 1981). Downstream migration begins when metamorphosis is completed and is often associated with high flow events in the winter and spring, perhaps coincident with adult upstream migration. Most volitional movement of macropthalmia occurs at night (Dauble et al. 2006).

It is likely that Pacific lamprey life history has played a key role in their persistence. The extended freshwater residency of ammocoetes allows populations to withstand low flows or other conditions that might block adult spawning runs over the course of several years. This may explain, for example, why a small population of Pacific lamprey persists in the San Joaquin River near Fresno (D. Mitchell, CDFW, pers. comm. 2007).

An underappreciated aspect of Pacific lampreys is their importance in the food webs of stream ecosystems. Ammocoetes break down detritus and are sources of prey for other fishes (Cochran 2009). Adult carcasses may be an important source of marine derived nutrients (e.g. nitrogen) to oligotrophic streams (Wipfli et al. 1998, Close et al. 2002, Lewis 2009).

Habitat Requirements: Pacific lampreys share many habitat requirements with Pacific salmonids (*Oncorhynchus* spp; Close et al. 2002, Stone 2006), particularly cold, clear water (Moyle 2002) for spawning and incubation. They also require a wide range of habitats across life stages. In general, peak spawning appears to be closely tied to water temperatures that are suitable for early development (Close et al. 2003, Meeuwig et al.

2005) but can occur at temperatures above 22 °C (Luzier et al. 2006). Consequently, temperature may be important in determining ammocoete abundance (Young et al. 1990, Youson et al. 1993, Bayer et al. 2000). Juveniles can persist in flows of up to 40 cm/s but are generally most common at velocities of 20-30 cm/s (Close 2001).

Adults use gravel areas to build nests, while ammocoetes need soft sediments in which to burrow during rearing (Kostow 2002). Nests are generally associated with cover, including gravel and cobble substrates, vegetation and woody debris. Likewise, most nests observed in Cedar Creek, Washington, were observed in pool-tail outs, low gradient riffles and runs (Stone 2006). Pacific lamprey embryos hatch at a wide range of temperatures (10-22 °C). However, in the laboratory, time from fertilization to hatching was around 26 days at 10 °C and around 8 days at 22 °C (Meeuwig et al. 2005). Survival of embryos was highest at temperatures ranging from 10 to 18 °C. Survival declined sharply, with a significant increase in abnormalities, at 22 °C.

Ammocoetes burrow into larger substrates as they grow (Stone and Barndt 2005). Ammocoetes also need detritus that produces algae for food (Kostow 2002) and habitats with slow or moderately slow water velocities (0-10 cm/s; Stone and Barndt 2005), such as low gradient riffles, pool tailouts and lateral scour pools (Gunckel et al. 2009).

Adults can climb over waterfalls and other barriers, using their sucking disc, as long as there is a rough surface and some amount of flow. These features are rarely present on dams, so even small dams or fish ladders can be barriers if not designed with surfaces and features that allow climbing (as in CRBLTW 2004).

Distribution: Pacific lampreys occur along the Pacific coast from Hokkaido Island, Japan (Morrow 1980), through Alaska and south to Rio Santo Domingo in Baja California (Ruiz-Campos and Gonzalez-Guzman 1996). Anadromous forms of Pacific lamprey occur below impassible barriers throughout their range. In California, Pacific lampreys occur from Los Angeles to Del Norte counties and the rivers in the Central Valley. Although a few individuals have been recorded in the Santa Ana, Los Angeles, San Gabriel and Santa Margarita rivers, the occurrence of all forms is infrequent south of Malibu Creek, Los Angeles County. The southernmost record in California is a single ammocoete collected from the San Luis Rey River, San Diego County, in 1997 (Swift and Howard 2009). A sizable run was recorded in the 1990s in the Santa Clara River (Chase 2001). However, their numbers appear to have significantly declined in the last few years (Swift and Howard 2009). There are also records from the Rio Santo Domingo, Baja California (Ruiz-Campos and Gonzalez-Guzman 1996). In general, lamprey distribution in California becomes irregular and erratic south of San Luis Obispo County (Swift et al. 1993, Swift and Howard 2009). An unusual landlocked population has persisted in Clair Engle Reservoir (Trinity River, Trinity County) since 1963, when the dam was constructed.

In the Central Valley, their upstream range appears to be limited by impassable dams that exist on all large rivers. Ammocoetes and spawning individuals have been observed in the San Joaquin River below Friant Dam and in most major tributaries from the Merced River north to the Feather River, as well as in some smaller tributaries, such as Putah Creek, Yolo-Solano counties. Ammocoetes have been observed along the edges of channels in the Sacramento-San Joaquin Delta, primarily in the north Delta (e.g. around McCormick-Williamson Tract; P. Moyle unpublished data). Both downstream migrating juvenile lampreys and returning adults must pass through the entire San Francisco Estuary, but their requirements for passage are not known.

Trends in Abundance: Anadromous Pacific lamprey abundance has declined so that large runs have disappeared from rivers such as the Eel River (Moyle 2002, Yoshiyama and Moyle 2010), although small runs persist in some portions of their range. Runs have also largely disappeared from southern California streams (Swift and Howard 2009). Abundance estimates for Pacific lamprey populations in California are scarce, but rotary screw trap data from 1997 to 2004 in the Klamath River basin suggested a declining trend for all life stages (USFWS 2004). Native American fishermen in the Klamath basin have also observed that runs are much smaller than they once were in this system (Larson and Belchik 1998). Traps for salmonid smolts in Redwood Creek, Humboldt County, capture 5-91 lampreys per year, all post-spawners (M. Sparkman, CDFW, pers. comm. 2011). Lampreys in Oregon and Washington have also shown significant declines, similar to those in California. For example, counts at Winchester Dam on the lower Umpqua River, Oregon, have declined from a maximum of 46,785 in 1966 to 34 in 2001 (ODFW in Close et al. 2002). In the Columbia River basin, the number of Pacific lamprey passing Bonneville Dam has declined from an estimated 50,000 adults prior to 1970 to less than 25,000 with a progressively sharper decline in Pacific lamprey abundance further upstream (Kostow 2002). Despite obvious declines wherever lampreys are actually counted, declines in Pacific lamprey are largely unrecognized, in part because they still occupy much of their historic range and most streams appear to retain at least small runs. The latter may be due to a low degree of fidelity to spawning areas (Goodman et al. 2006, Docker 2010), so recolonization of altered streams may occur fairly quickly when conditions improve, provided there is a source population nearby. However, this pattern of rapid dispersal may actually mask an overall decline in numbers.

Thus, a population in Putah Creek (Yolo and Solano counties) reestablished itself following completion of the Solano Project, which dewatered lower portions of the stream, and, again, following an extended drought during which much of the stream was dry. The apparent lack of strong homing tendencies in Pacific lampreys suggests that they have the ability to temporarily colonize impaired habitats, even if they cannot sustain populations in these areas. However, the apparent loss of the largest known southern California population in the Santa Clara River (Swift and Howard 2009) indicates that their distribution and abundance is shrinking and certain portions of their range may no longer provide suitable habitats.

Nature and Degree of Threats: Threats to Pacific lampreys are diverse and usually multiple for any given population (Table 1). The nature and degree of these threats are poorly understood, given the general lack of information on factors affecting lamprey populations. The Pacific lamprey has such a wide geographic range that different factors likely influence its abundance in different areas. Hence, there are no ‘high’ or ‘critical’ scores for threats to all California populations, combined, but a remarkable nine ‘medium’ scores, which could actually be ‘critical’ or ‘high’ in different rivers (Table 1). It is likely that factors that have led to population declines of anadromous salmonids across California may also be the main causes for decline of Pacific lamprey, especially given these fishes share so many ecological and habitat requirements.

One universal factor, related to all others but not rated here, is reduction in prey abundance, especially salmonids, due to stressors such as dams, diversions, habitat degradation and over-exploitation. Adult Pacific lampreys depend on having large populations of large prey species, such as salmon, to maintain their own numbers. In British Columbia, salmon are among the most important prey of lampreys (Beamish 1980), as they may be elsewhere in their range. While the importance of different prey species is unknown for populations of lampreys in California, the fact that Chinook and coho salmon populations have severely declined in most California rivers suggests that lamprey declines may be closely tied to salmonid declines.

Dams and diversions. Large dams have reduced the range of Pacific lampreys in many streams, as they have for salmon and steelhead, by preventing upstream passage to spawning and rearing areas and reducing suitability of downstream habitats. Lampreys are capable of passing over some small dams and diversion structures, either by using fish ladders or by using their suction cup-like mouths to work their way over barriers, provided the surfaces are wet and rough. Large dams without passage structures, however, occur throughout their range and prohibit upstream migration to large portions of their former range.

Where documentation exists for regulated streams, lamprey populations have declined from historic numbers. Unsuitable flow regimes for migration, along with loss of spawning and backwater rearing habitats combine to make regulated streams unfavorable for lampreys. Flow regimes that limit emigration or immigration may have delayed effects and declines may be difficult to detect; the long lifespan of ammocoetes and the apparent lack of homing behavior in adults can give the impression of persisting populations in streams with only intermittent access. During unseasonably high-flow events, ammocoetes may be flushed to unsuitable habitats because they are poor swimmers (Dauble et al. 2006). Spawning habitat is lost when recruitment of sediments from upstream areas is blocked by dams; lack of sediment imbeds rocks in spawning areas, making them more difficult to move for nest creation. Reduction in sand and silt recruitment, combined with channelization, may also reduce suitable habitats available for ammocoetes below large dams (Close et al. 2002).

Agriculture. Lampreys are typically rare or absent from river reaches heavily influenced by agriculture. In particular, Pacific lampreys are usually eliminated from streams that are heavily polluted (Gunckel et al. 2006), such as the lower San Joaquin River.

Urbanization. The broad range of Pacific lampreys includes many areas that are now heavily urbanized. Typically, they are rare or absent in these areas, such as most of southern California, although the exact causes are poorly documented. Presumably, the disappearance of lampreys from urban areas has multiple causes related to habitat alteration (water diversion, channelization, concrete channels, etc.) and to pollution such as stormwater runoff and pesticides, although most urban streams are also dammed and diverted.

Instream mining. Gravel mining has been common in the lower reaches of streams favored by lampreys. While impacts have not been documented, gravel mining may disrupt spawning and displace ammocoetes, particularly through mobilization of fine sediment deposits, which are key rearing habitats, as well as removal of preferred substrates for spawning.

Mining. Hardrock mines are present in many lamprey watersheds but their effects (e.g., acid mine drainage) are largely unknown.

Logging. Coastal rivers, such as the Eel River (named for its lampreys), that have been heavily altered by logging and road building are generally less suitable for lampreys than they were historically because of excessive deposition of gravels in backwater areas needed for rearing, alteration of the annual hydrograph, increased sediment loads, increased solar input and corresponding higher water temperatures, or similar changes in habitats.

Estuary alteration. Estuaries have been significantly altered throughout the range of Pacific lamprey. Estuaries may be as important to lamprey as they are to anadromous salmonids, which rely on them for foraging, rearing and holding habitat, as well as transitional habitats that enable osmoregulation and migration orientation. Lamprey ammocoetes were commonly observed in the soft sediments of the Smith River estuary from 1997 to 2001 (R. Quiñones, pers. observations), an estuary that retains many of its natural characteristics because stream flows have not been altered significantly.

Harvest. Lampreys have long supported subsistence fisheries by coastal tribes, especially in the Klamath River, because their early arrival and high fat content made them highly desirable as food. This fishery continues today, although only small numbers are likely taken (Lewis 2009). Of greater concern is the fishery for spawning lampreys that has developed because of their value as bait for sturgeon. Adult lampreys are extremely vulnerable to capture when on their nests and the fishery is largely unregulated and unmonitored. Ammocoetes are also collected for bait on occasion and are called “worms” by striped bass fishermen.

Alien species. Alien species increasingly co-occur with Pacific lampreys but their impacts on lamprey populations are not well understood; however, localized impact may be considerable. Ammocoetes are documented prey of many predatory fishes. In the Eel River, for example, introduced Sacramento pikeminnows were observed feeding heavily on ammocoetes (P. Moyle, personal observations; Brown and Moyle 1997).

	Rating	Explanation
Major dams	Medium	Major dams present on many Pacific lamprey rivers; dams prevent access to spawning habitats and reduce habitat suitability downstream
Agriculture	Medium	Minor influence on lower Klamath and Eel rivers, major impact in Central Valley
Grazing	Low	Pervasive across Pacific lamprey range but probably minor impacts on large river habitats
Rural residential	Low	Can cause localized habitat loss or degradation
Urbanization	Medium	Large urban areas in southern part of range and Central Valley contribute to habitat degradation, stream channelization, input of pollutants and flashy flows associated with hardscapes
Instream mining	Medium	Gravel mining and gold dredging alter rearing habitats and increases mortality of ammocoetes; effects are highly localized
Mining	Low	Mines common in lamprey watersheds; direct effects unknown
Transportation	Medium	Roads line many rivers and streams, simplifying habitats (channelization, bank stabilization, etc.); sources of sediments and pollutants that may affect spawning and survivorship; culverts and other structures create barriers to migration
Logging	Medium	Major source of sediments via roads; greater historic impacts in most Pacific lamprey habitats than today
Fire	Low	Fire severity is increasing due to landscape changes, along with climate change, potentially increasing siltation and changing water quality
Estuary alteration	Medium	Most estuaries in California are highly altered through diking, draining, channelization and dredging
Recreation	Low	Possible disturbance to spawning and rearing
Harvest	Medium	Potential reduction of adult abundance in some streams, rivers and Delta; impacts not well understood
Hatcheries	n/a	
Alien species	Medium	Predation on ammocoetes may limit abundance in some areas

Table 1. Major anthropogenic factors limiting, or potentially limiting, viability of populations of Pacific lamprey in California. Factors were rated on a five-level ordinal scale where a factor rated “critical” could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated “high” could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated “intermediate” is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated “low” may reduce populations but extinction is unlikely as a result. A factor rated “n/a” has no known negative impact to the taxon under consideration. Certainty of these judgments is low. See methods section for descriptions of the factors and explanation of the rating protocol.

Effects of Climate Change: Predicted increases in river temperatures (to > 22 °C) brought about by climate change may increase incidence of deformities and mortalities of incubating eggs and of ammocoetes (Meeuwig et al. 2005). Summer water temperatures already frequently exceed 20°C in many California streams and temperatures are expected to increase under all climate change scenarios (Hayhoe et al. 2004, Cayan et al. 2008). Increases in summer temperatures may affect growth and metabolic costs of juveniles and stress adult Pacific lamprey holding in rivers throughout the summer (Clemens et al. 2009).

Climate change is also predicted to change the flow regime in rivers. For instance, flows in the Klamath River may peak earlier in the spring and continue tapering through the summer before pulsing again later in the fall (Quiñones 2011). Resulting changes in river flows and temperatures may alter the timing of adults and juveniles entering and exiting California rivers. Large flow events can disrupt incubation and rearing habitat due to increased bed mobility (Fahey 2006). However, flow-related impacts may be attenuated by dam operations in some systems or exacerbated by competing demands for water (e.g., agricultural irrigation) during low flow periods in others. The Pacific lamprey's migratory plasticity may facilitate movement into watersheds with more favorable habitat conditions (provided passage exists) so their populations may not be as threatened by climate change as are species with high migratory fidelity (e.g., salmon and steelhead). Nonetheless, the geographic range of Pacific lamprey may shift northward as temperatures and flows because unsuitable in more southern streams. Populations south of Monterey Bay may disappear, following those in southern California. Shifts upward in elevation toward remaining cold water refuges may be impeded by barriers or difficulties associated with passage through dams, as well as increased distance of migration and lack of suitable habitats in high-gradient reaches. Because of these concerns, Moyle et al. (2013) rated Pacific lamprey as "highly vulnerable" to extinction in California due to climate change impacts in the next 100 years.

Status Determination Score = 3.3 - Moderate Concern (See Methods section, Table 2). Pacific lampreys apparently still occupy much of their native range in California, but evidence suggests that large declines may have occurred in the past 50 years. Pacific lampreys no longer have access to numerous upstream habitats blocked by large dams or other impassable structures and they are no longer present in streams at the southern end of their range. The large runs that once occurred in coastal streams such as the Eel and Klamath have dwindled to a fraction of their former size.

Metric	Score	Justification
Area occupied	4	Present throughout much of their historic range; blocked from large portions of watersheds by dams
Estimated adult abundance	2	Population estimates lacking; large river populations presumably are >500 in most years
Intervention dependence	4	Improved flow management and habitat restoration efforts needed to prevent further declines, especially for more southern populations
Tolerance	3	Local populations are vulnerable to stochastic events and degraded habitats
Genetic risk	5	Gene flow apparently largely unimpaired between populations throughout range
Climate change	2	Limited spawning and rearing habitats suggests vulnerability to increased temperatures and altered flow regimes, especially in southern end of range
Anthropogenic threats	3	Nine factors rated as 'medium' (Table 1)
Average	3.3	23/7
Certainty (1-4)	2	Population size and environmental tolerances poorly understood

Table 2. Metrics for determining the status of Pacific lamprey, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: Pacific lamprey conservation and management is currently hindered by lack of information on their distribution, abundance, and life history. However, given their apparent decline throughout much of the historical range in California, additional conservation measures can and should be pursued in order to afford greater protection (Streif 2009). Management recommendations include the following:

1. Establish a Pacific lamprey research and monitoring program, with three primary goals: 1) determine the status of lampreys statewide and identify key conservation opportunities; 2) improve understanding of life history attributes and habitat requirements in California streams in order to enable a limiting factors analysis; and 3) determine if different genetic stocks of lampreys exist in California. Ideally, such a program would provide critical information about status, population dynamics and life history variability of the species throughout its range in order to inform management and conservation measures. Beneficial research should include studies to: (1) identify the presence or absence of multiple runs in large rivers; (2) document landlocked populations in large river systems; and (3) evaluate metapopulation dynamics to determine if a few large main-river populations sustain smaller tributary populations (source-sink dynamics).
2. Establish a lamprey data center, as part of the proposed research and monitoring program, which would standardize, collect and integrate *all* lamprey information collected in California. The many rotary screw traps used to monitor outmigration of juvenile salmonids, in particular, are a largely untapped source of

data. Many trap operators record captures of lamprey 'smolts' and ammocoetes. The lampreys are rarely identified to species, but most are likely Pacific lampreys.

3. Determine if conservation efforts for salmonids also benefit Pacific lampreys, especially in regulated streams. The following questions remain largely unanswered and should be the focus of additional research:
    a. Do passage structures constructed for salmonids also allow passage for lampreys?
    b. Do habitat restoration programs focused on salmonids also create backwater habitat for lampreys?
    c. Are populations of Pacific lamprey tied to those of salmon and steelhead (e.g., predator-prey interactions, migratory timing)?
4. Require that all instream alteration or diversion projects address lamprey habitat and life history requirements and provide appropriate mitigation measures. Strief (2009) documented that a single stream dewatering event, even of short duration, can inhibit up to seven years of lamprey production by eliminating all age classes of ammocoetes.
5. Address potential threats in order to reduce or reverse population declines. In many respects, addressing threats to lamprey requires restoring flows and habitats in most of California's rivers. Possible actions include:
    a. Subsistence and bait fisheries for lamprey should be monitored to determine their effects on population structure and abundance.
    b. Where feasible, large dams should be retrofitted with fishways that are passable to all migratory stages of lamprey.
    c. Estuary and river restoration projects should consider establishing natural flow regimes, minimum base flows, and sediment budgets (to reestablish deposits of soft sediment in low velocity habitats and improve spawning gravel quality).

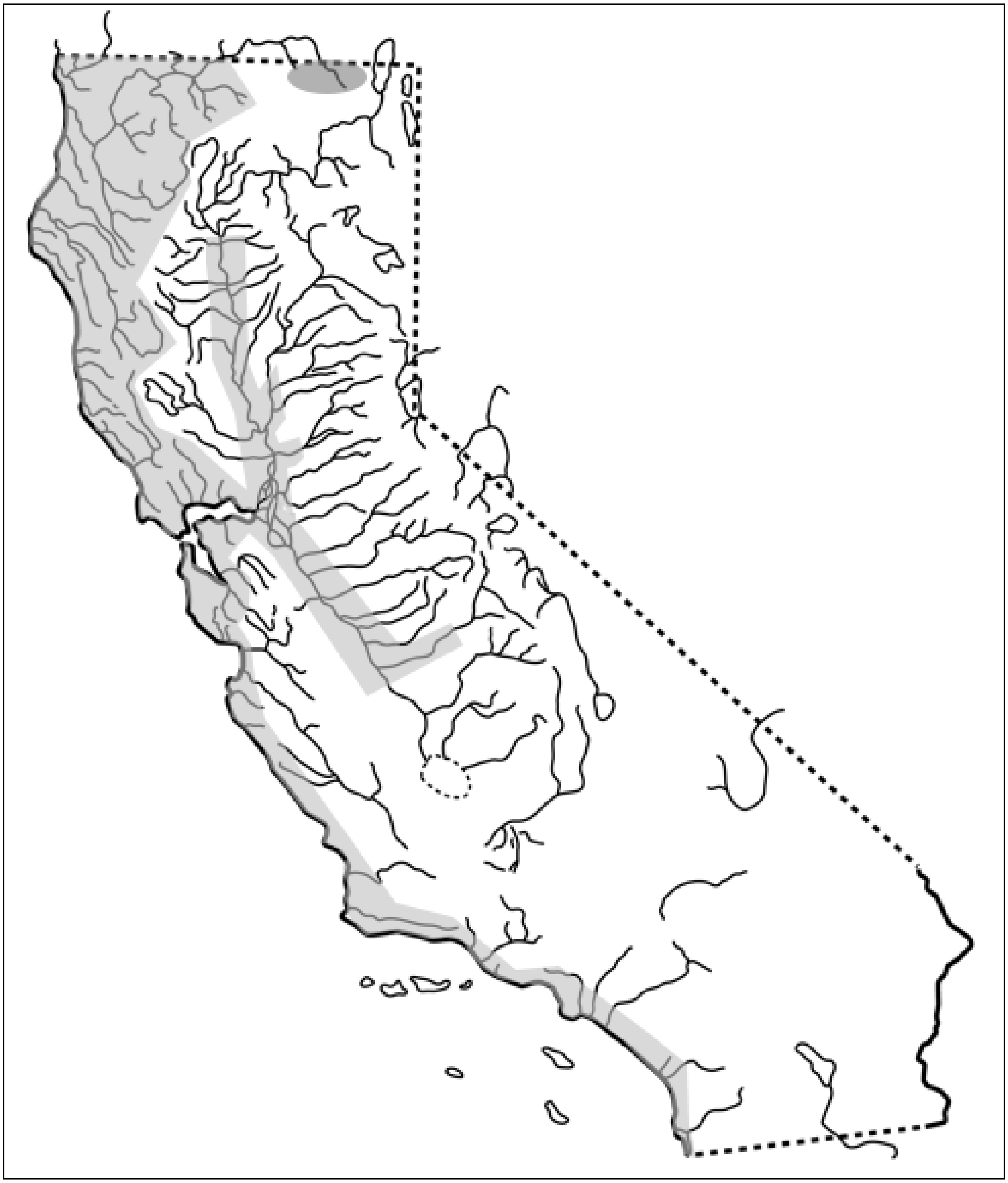


Figure 1. Generalized distribution of Pacific lamprey, *Entosphenus tridentatus***,** in California. Current distribution is reduced and fragmented, although recolonization of depleted areas may occur periodically.

GOOSE LAKE LAMPREY
Entosphenus sp.

Status: High Concern. The Goose Lake lamprey does not face immediate extinction risk but its restricted distribution makes it vulnerable to land and water use practices, climate change, and other factors which could compromise its status.

Description: This predatory lamprey is similar to the widespread Pacific lamprey, *E. tridentatus*, except that it is much smaller (adult TL 19-25 cm vs. 30-40 cm for Pacific lamprey) and not as dark in color. Both forms can be recognized by the sharp, horny plates in the sucking disc, the most distinctive being the crescent-shaped supraoral plate, which has three distinct cusps. The middle cusp is smaller than the two lateral cusps. Adult Goose Lake lampreys are shiny bronze in color. Ammocoetes can be distinguished from those of the sympatric Pit-Klamath brook lamprey (*E. lethophaga*) by the larger number of myomere segments (64-70 between the last gill opening and anus).

Taxonomic Relationships: The Goose Lake lamprey was first recognized as distinct by Carl Hubbs (1925) but he did not formally describe it as a species. It is presumably derived from Pacific lamprey or its derivatives from the Klamath River drainage. However, Goose Lake and the Pit River drainage, to which it connects, have been separated from the Klamath drainage since the early Pleistocene (1-3 million years). Some insights into evolution of the Goose Lake lamprey are provided by Lang et al. (2009); they used mitochondrial DNA (cytochrome B) to examine relationships among all lamprey species. While Goose Lake lamprey *per se* were not used in the analysis, the non-predatory Pit-Klamath brook lamprey was included, which is most likely the closest relative of the Goose Lake lamprey. Lang et al. (2009) found that it was part of the *Entosphenus* clade, which includes the various non-anadromous lampreys from the upper Klamath River as well as the Pacific lamprey. The relationship of Pit-Klamath brook lamprey to others within the clade is largely unresolved. Genetic differences, at least those based on mitochondrial DNA, indicate that the genome of lampreys is very conservative so that population structure, even in the widespread Pacific lamprey, has not been detected (Goodman et al. 2008). Regardless, the lampreys of the Goose Lake basin are likely a distinct evolutionary lineage, perhaps representing more than one.

Within the basin, there are two basic hypotheses about the relationship between the predatory Goose Lake lamprey and the non-predatory Pit-Klamath brook lamprey: (1) they represent different life history forms of the same species, or (2) they are separate species. These same hypotheses, often unresolved, exist for the pairs of predatory and non-predatory lampreys found throughout the world (Docker 2009). It is generally assumed that the non-predatory forms evolved from predatory forms. In the case of the Goose Lake basin, the issue is complicated by the fact the Pit-Klamath brook lamprey has been described as occurring in both the Goose Lake and Klamath River basins, despite their long separation (Hubbs 1971).

Nevertheless, because of its distinctive morphology and ecology and long isolation from other populations, it is most likely that the Goose Lake lamprey is a distinct species, separate from the Pit-Klamath brook lamprey (Kostow 2002) and from other lamprey species in the Klamath River (Docker et al. 1999). As a separate species, the Goose Lake lamprey may include both predatory and non-predatory life histories, assuming that the predatory form is only expressed when migrations to Goose Lake are feasible (Kostow 2002). Limited data on adult distribution, presented in Scheerer et al. (2010), suggest that the two lamprey species are at least partly segregated by elevation, with the Goose Lake lamprey found in stream reaches closest to the lake.

Life History: The life history of this taxon is largely unknown, but presumably the adults live for a year or two in Goose Lake, preying on Goose Lake tui chubs, suckers, and redband trout. In 1989, adult lampreys were observed attached to gill-netted tui chubs and lamprey wounds were common in larger chubs (P. Moyle and R. White, unpublished observations). They migrate up suitable tributary streams in spring for spawning, with a peak in May (Kostow 2002). They require clean gravels for spawning, combined with soft-bottomed habitat downstream of the spawning areas for rearing of ammocoetes. Thus, spawning areas may be as much as 20-30 km upstream from the lake. Ammocoetes probably spend 4-6 years in tributary streams before metamorphosing into adults (at about 8-13 cm TL) in the fall and moving into the lake in spring (Kostow 2002). During periods of drought, when access to the lake is not available, adult lampreys will feed on stream fishes although survival appears to be low (Kostow 2002).

Habitat Requirements: Adults live in shallow, alkaline Goose Lake where they prey on larger fishes. Like other lampreys, Goose Lake lampreys require gravel riffles in streams for spawning and ammocoetes require muddy backwater habitats downstream of spawning areas. Kostow (2002) characterizes the habitat of ammocoetes as "fine silt lenses along low gradient stream meanders, most often through meadows…(p. 18)." However, the habitat requirements of Goose Lake lamprey have not been well studied or distinguished from those required by Pit-Klamath brook lamprey. For further description of stream and lake habitats, see the Goose Lake redband trout account in this report.

Distribution: The Goose Lake lamprey is endemic to Goose Lake and its tributaries in Oregon and northeastern California. However, a comprehensive assessment of the distribution and habitat utilization of California tributary streams by lampreys has not been performed. Within California, they have been collected only from Lassen and Willow creeks, Modoc County, (G. Sato, BLM, pers. comm. 1994), both above and below potential migration barriers (Hendricks 1995). Ammocoetes were found to be common in Cold Creek, a tributary to Lassen Creek. No ammocoetes were found in Davis, Pine or Willow creeks. It is likely that dams and diversions now restrict distribution of lampreys by blocking adult migration and by drying up suitable habitats downstream. In Lake County, Oregon, they are common in Thomas Creek and a population apparently exists in Cottonwood Reservoir, on Cottonwood Creek (Oregon Dept. of Fish and Wildlife, unpubl. data, 1995). Scheerer et al. (2010) found lamprey ammocoetes to be widely distributed and often abundant in Oregon streams, but did not distinguish species.

Trends in Abundance: There are no trend data for Goose Lake lamprey but their populations likely decline during extended periods of drought and then increase rapidly when wet periods return and the lake fills again. Thus, Goose Lake lampreys were fairly common in Goose Lake, where they were readily collected from large tui chubs caught in gillnets, until the lake dried up in the summer of 1992 (R. White, USFWS, pers. comm. 1995). The Goose Lake lamprey has the potential of becoming extirpated, especially in California, if the lake and lower tributaries are dry for several years in a row. However, adults may survive by preying on stream fishes and the ammocoetes may persist for 3-4 years if there are adequate flows in the habitats they occupy. The Cottonwood Reservoir population is of unknown size but the reservoir may serve as a refuge, provided a minimum pool is maintained throughout extended drought periods. In Lassen and Willow creeks, ammocoetes were common at densities of 11-50 individuals per 150 ft of stream (Hendricks 1994). Abundance of spawners is not known but 50-100 spawners in most years in each stream may be a reasonable estimate, based on accessible habitat, number of ammocoetes, and abundance in the lake. The importance of Lassen and Willow creeks to persistence of the entire population in the Goose Lake basin is unknown but it is assumed that most spawning and rearing habitat occurs in Oregon streams (Scheerer et al. 2010).

Nature and Degree of Threats: The principal threat to the Goose Lake lamprey is desiccation of its habitats, Goose Lake and its tributaries, which is exacerbated by human activities, including diversions for agriculture and grazing. The combination of severe, extended drought, along with human demands for scarce water resources in the basin, may have resulted in accelerated desiccation of the lake during the 1986-1992 drought and, again, in 2010, resulting in a dry lakebed.

Agriculture. Farming occurs primarily on lands close to the lake, often adjacent to tributary streams, with the result that some streams reaches are channelized, down-cut, and silted from erosion. The diversion of water from streams for agriculture and other uses may reduce or completely dewater habitats required by ammocoetes and adults for survival during droughts, as well as accelerating desiccation of the lake itself. Diversions and dams may prevent adults from reaching spawning areas in tributary streams, although small reservoirs may also serve as refuges for adults. The loss of suitable habitat for ammocoetes is likely to be particularly severe in the lower reaches of streams near agricultural areas.

Grazing. Livestock grazing is one of the greater land uses in the Goose Lake basin. In-stream and riparian habitats can be degraded or eliminated through stream erosion and bank destabilization caused by livestock grazing in riparian areas, especially through the removal of woody riparian plants. In the past, many areas in the California portion of the Goose Lake basin were degraded by grazing, although restoration actions, especially on Lassen Creek, have reversed some of these impacts. While improved management of most grazed lands has reduced the threat of grazing in the short term, as the climate becomes warmer and more variable (see Effects of Climate Change section), there is considerable potential for negative impacts from grazing to increase without expanding the use of riparian protection measures such as exclusionary fencing.

Fire. The Goose Lake basin is semi-desert and wildfires are common. Impacts of fires on lampreys (and other fishes) are not known but are likely to be minimal, unless a major fire causes direct mortality through increased stream temperatures or indirect mortality associated with loss of canopy cover (in-stream shading), accelerated erosion, or landslides in upstream areas.

Alien species. Scheerer et al. (2010) found six species of alien fishes in Oregon streams tributary to Goose Lake, mostly in low elevation areas or areas associated with reservoirs and other altered habitats. Alien species appear to be scarce in Lassen and Willow creeks although predatory brown trout are common in Pine and Davis creeks. Illegal introductions of possible predators (catfish, bass) remain a concern.

	Rating	Explanation
Major dams	Low	Reservoirs may act as refuge during drought; diversion dams may block spawning and in-stream movement
Agriculture	Medium	Alfalfa fields along lower reaches of streams may negatively affect water quality
Grazing	Medium	Grazing is pervasive and is likely to have strong interactions during periods of reduced flow
Rural residential	Low	Few residences
Urbanization	n/a	
Instream mining	n/a	
Mining	Low	Uranium mines exist in the area but their impacts are unknown
Transportation	Medium	Roads and culverts can block migration; potential increased siltation
Logging	Low	Widespread in watersheds but impacts reduced from the past
Fire	Low	A continuous threat in this part of the state; impacts to lampreys unknown
Estuary alteration	n/a	
Recreation	n/a	
Harvest	n/a	
Hatcheries	n/a	
Alien species	Medium	Aliens present in certain portions of the basin; impacts to lampreys are unknown

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

Effects of Climate Change: The Goose Lake basin is located in an arid portion of California and this area has, in the recent past, suffered extended periods of drought. Climate change is likely to decrease summer stream flows in key streams, increasing competition for water and riparian habitats between humans (livestock, agriculture) and fishes. Goose Lake may dry more frequently and for longer periods of time due to increased frequency of drought. Increased stream temperatures of 2-4°C may affect lampreys, although similar species can tolerate fairly warm water. These conditions may also favor alien competitors and predators (Scheerer et al. 2010). An increase in fire frequency or intensity in this dry landscape may decrease riparian shading, add sediment, or otherwise make streams less suitable for lampreys and other fishes. Moyle et al. (2013) consider the Goose Lake lamprey to be "critically vulnerable" to extinction as the result of climate change because predicted reduction in snow pack will result in decreased flow in tributary streams with corresponding reduced lake levels.

Status Determination Score = 2.9 – High Concern (see Methods section Table 2). Goose Lake lamprey do not face immediate extinction risk but their California populations are small and isolated, making them vulnerable to climate change and other factors which could compromise their status. The American Fisheries Society regards Goose Lake lamprey as a threatened species, with declining populations (Jelks et al. 2008), while NatureServe ranks it as Critically Imperiled (T1) and the Forest Service regards it as Sensitive.

Metric	Score	Justification
Area occupied	2	Only known from Willow, Lassen, and Cold creeks in CA
Estimated adult abundance	1	California abundance not known but numbers of adult spawners is likely small in most years and zero in dry years
Intervention dependence	4	Persistence requires habitat improvement and maintenance
Tolerance	4	Not known but presumably fairly broad
Genetic risk	3	Potential for impacts from small population size and isolation
Climate change	2	Stream habitat likely to be reduced as is frequency of lake drying
Anthropogenic threats	4	See Table 1
Average	2.9	20/7
Certainty (1-4)	2	Very little is published on this lamprey

Table 2. Metrics for determining the status of Goose Lake lamprey in California, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: The Goose Lake lamprey and other Goose Lake fishes were little studied and largely unmanaged until 1991, which contributed to their increased likelihood of extinction. The Goose Lake Fishes Working Group was formed in 1991, with representatives from private landowners, federal and state agencies, and environmental groups to explore management measures for all fishes native to Goose Lake and its tributaries (Sato 1992a, see Goose Lake redband trout account in this report). As a result of this program, stream restoration projects have improved reaches of Lassen Creek, presumably providing better habitat for lamprey spawning and rearing. The biology and status of the population in Cottonwood Reservoir needs to be investigated, as well as the possibility of establishing similar refuge populations of the species elsewhere. An investigation of this unusual lamprey's life history and habitat requirements should be conducted in order to develop management and conservation strategies in both California and Oregon. In particular, stream flow models need to be developed under various climate scenarios in order to determine predicted base flows. At a minimum, flows in key tributary streams should provide adequate rearing and holding habitat during extended drought (>5 years) in order for the species to persist and recolonize the lake during wetter periods. Enhancing spawning access, as well as restoring rearing and holding habitats, in streams in California and Oregon (especially in Lassen, Willow, and Thomas creeks) would benefit all native Goose Lake fishes. In addition, studies should be developed to determine both the evolutionary and ecological relationships between the Goose Lake lamprey and the Pit-Klamath brook lamprey. See the Goose Lake sucker account in this report for further discussion of management actions that would encompass the entire Goose Lake basin and likely benefit Goose Lake lamprey.

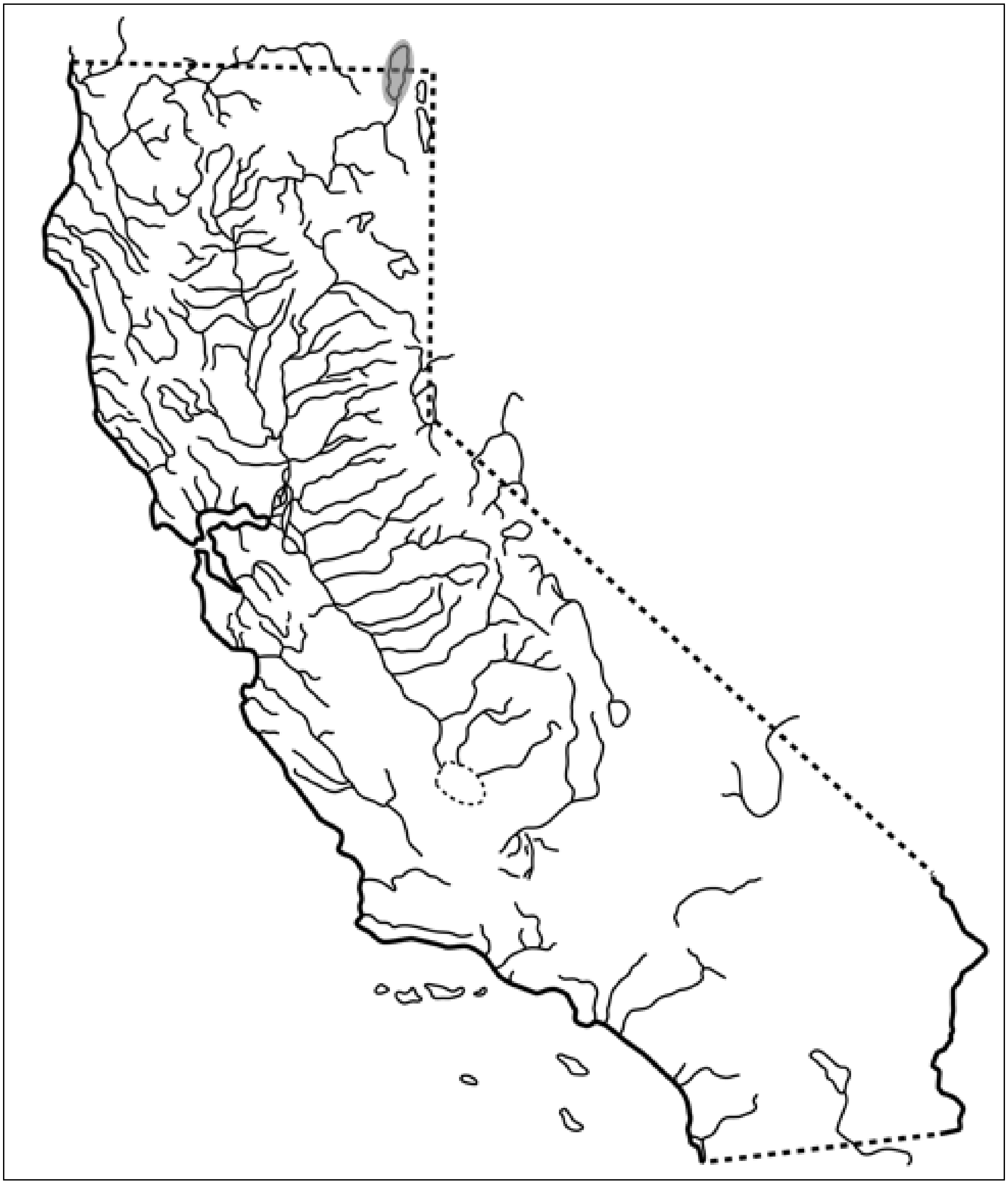


Figure 1. Distribution of Goose Lake lamprey, *Entosphenus sp.*, in Goose Lake, California and Oregon. The extent to which they are distributed upstream in the Thomas Creek drainage in Oregon is unknown.

NORTHERN CALIFORNIA BROOK LAMPREY
Entosphenus folletti (Valdykov and Kott)

Status: High Concern. The northern California brook lamprey has a very limited known distribution and aquatic habitats within their range are heavily altered by agriculture and grazing. Their actual distribution and abundance is unknown.

Description: This lamprey is a non-predatory species that has an adult size of 17-23 cm in total length (Vladykov and Kott 1976b, Renaud 2011). Adult disc length is 6.6–7.8% of total length and the trunk myomere count is 61-65. The following description of dentition is from Renaud (2011, p. 27): "supraoral lamina, 3 unicuspid teeth, the median one smaller than the lateral ones; infraoral lamina, 5 unicuspid teeth; 4 endolaterals on each side; endolateral formula, typically 2–3–3–2, the fourth endolateral can also be unicuspid; 1–2 rows of anterials; first row of anterials, 2 unicuspid teeth; exolaterals absent; 1 row of posterials with 13–18 teeth, of which 0–4 are bicuspid and the rest unicuspid (some of these teeth may be embedded in the oral mucosa); transverse lingual lamina, 14-20 unicuspid teeth, the median one slightly enlarged; longitudinal lingual laminae teeth are too poorly developed to be counted. Velar tentacles, 8–9, with tubercles. The median tentacle is about the same size as the lateral ones immediately next to it…Oral papillae, 13." Ammocoetes are described in Renaud (2011).

The northern California brook lamprey is similar to the Pit-Klamath brook lamprey, with which it co-occurs, but is somewhat larger (most are >19 cm TL), has a larger oral disk (<6% of TL vs >6% of TL), and has elongate velar tentacles without tubercles. There are also minor differences in various tooth counts (Renaud 2011). According to Vladykov and Kott (1976b, p. 984): "The body and fins of *E. folletti* are more darkly pigmented than those of *E. lethophagus*. The entire caudal fin of the former is strongly pigmented, except for a narrow unpigmented margin, and it has a dark second dorsal fin. In the latter the caudal fin has broader unpigmented margin and its second dorsal is less pigmented." The region around the vent is darkly pigmented in *E. folletti* but pale in *E. lethophagus*, a potential distinguishing characteristic in the field.

Taxonomic Relationships: Non-predatory lampreys in the Klamath and upper Pit River systems are derived from Pacific lamprey (Renaud 2011). The northern California brook lamprey was described by Vladykov and Kott (1976b) based on specimens from Willow and Boles creeks, tributaries to the Lost River, Modoc County. However, the species was not recognized by the American Fisheries Society (AFS, Robins et al. 1991) because of unpublished doubts of its validity. Lang et al. (2009) listed it as a recognized species, as did Beamish (2010). The AFS then recognized it as a species based on Renaud's (2011) analysis of lamprey species worldwide (Page et al. 2013). Beamish (2010), using gill pore papillae as a diagnostic character, suggests that *E. folletti*, as currently recognized, may represent more than one species and included in his analysis both specimens from the Lost River and from Fall Creek above Copco Reservoir in California. While evidence increasingly supports the diversity of lamprey species in the upper Klamath and Pit River basins, including northern California brook lamprey, a thorough analysis is needed using additional specimens and additional genetic and morphological studies. Further studies are almost certain to find *E. folletti* in Oregon, given its presence in two distantly separated areas in California, so the common name "northern California brook lamprey" may not be appropriate for the species. Shapovalov et al. (1981) named it the Modoc brook lamprey, a name which reflects its likely distribution as being coincident with the Modoc Plateau region in California and Oregon, as well as with the territory of the Modoc people.

Life History: Nothing is known about the life history of this lamprey but it is presumably similar to other brook lampreys in the genus *Entosphenus.*

Habitat Requirements: Little specific information is available on its habitats, but the northern California brook lamprey is known only from a few, small, cool tributary streams that have areas with fine substrates and beds of aquatic plants.

Distribution: The northern California brook lamprey is known from only Willow and Boles creeks above Clear Lake Reservoir and from Fall Creek, a tributary to Copco Reservoir. It is almost certainly found in similar habitats in Oregon, as well as in the Lost and Klamath river basins.

Trends in Abundance: Abundance and population trend information are lacking. Their populations do not seem to be in danger of extinction at this time but face multiple threats as discussed below.

Nature and Degree of Threats: The northern California brook lamprey faces loss of suitable habitat via multiple factors affecting streams in this arid region, similar to those facing the Pit-Klamath brook lamprey.

Major dams. The only populations known are above large reservoirs, which suggests that they are isolated from other populations by dams. Dams and diversions on the upper Klamath and Lost River systems also alter downstream flows and habitats.

Agriculture. Water demands for irrigated agriculture and livestock are high in this region, leading to decreased stream flows. Flood-irrigated pastures introduce nutrients and pollutants from return waters into streams and raise water temperatures.

Grazing. Extensive grazing occurs throughout the known range of northern California brook lamprey. Grazing can degrade aquatic habitats through stream bank trampling, elimination of riparian vegetation, and pollutant inputs from animal wastes.

Alien species. Many alien fish species inhabit the Klamath and Lost river basins (Close et al. 2010). Species that can prey on lamprey include largemouth bass, brown bullhead, channel catfish, brook trout, brown trout, black crappie, and yellow perch (Close et al. 2010).

	Rating	Explanation
Major dams	High	Dams isolate populations and alter downstream habitats
Agriculture	Medium	Agriculture pervasive throughout range
Grazing	Medium	Grazing pervasive throughout range
Rural residential	n/a	
Urbanization	n/a	
Instream mining	n/a	
Mining	n/a	
Transportation	Low	Rural roads affect stream habitats
Logging	Low	Logging occurs in forested lands; impacts unknown
Fire	Low	Wildfires occur throughout range; impacts unknown
Estuary alteration	n/a	
Recreation	n/a	
Harvest	n/a	
Hatcheries	n/a	
Alien species	Low	Alien species uncommon in known stream habitats but are a potential threat

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

Effects of Climate Change: Climate change is expected to increase the frequency of both drought and floods in streams. Because ammocoetes likely rear for several years in soft substrates, large flooding events may disrupt rearing habitats (Fahey 2006) and displace ammocoetes. On the contrary, scouring events may clean sediments from gravels that would otherwise degrade spawning habitats (Stuart 2006 in Fahey 2006). It is likely that the northern California brook lamprey can tolerate, to some extent, shifts toward warmer water temperatures, which are expected to increase due to climate change. Moyle et al. (2013) did not rate climate change vulnerability for this species, but vulnerability should be similar to that of the Pit-Klamath brook lamprey.

Status Determination Score = 2.4 – High Concern (see Methods section, Table 2). Northern California brook lamprey apparently have limited distribution in small streams subject to degradation. However, their actual abundance and distribution is unknown.

Metric	Score	Justification
Area occupied	2	Known range limited to Lost River and parts of upper Klamath
Estimated adult abundance	2	Numbers unknown but likely small
Intervention dependence	4	Long-term management of grazing practices as well as alien species may be warranted
Tolerance	3	Not known but occurs in degraded streams
Genetic risk	2	Known populations isolated by dams
Climate change	2	Some habitats may dry more extensively or for longer periods; ammocoetes may be displaced by unusually high flows
Anthropogenic threats	2	See Table 1
Average	2.4	17/7
Certainty (1-4)	1	Species is largely unstudied

Table 2. Metrics for determining the status of Northern California brook lamprey in California, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: Habitat degradation, grazing practices and isolation by reservoirs pose the greatest threats to this brook lamprey, effects likely to be exacerbated by increasing temperatures and more frequent flood events predicted by climate change models. Watershed management strategies exist (e.g. Klamath Basin Restoration Agreement) that address these and other factors that may limit fish populations in the upper Klamath basins. Beyond implementation of these strategies, basic life history studies and population monitoring should occur in order to better understand the status of this species. The following questions should be addressed as part of a status evaluation:

What is the current distribution and abundance in California and Oregon?
Where are most important spawning and rearing grounds located in California?
What are the optimal and preferred environmental tolerances and habitat conditions for each life history stage?

Figure 1. Known distribution of northern California brook lamprey, *Entosphenus folletti*, in California.

KLAMATH RIVER LAMPREY
Entosphenus similis (Vladykov and Kott)

Status: Moderate Concern. Very little is known about this species; thus, the conservative course of action is to consider its numbers to be in decline until new information becomes available to indicate otherwise. However, Klamath River lamprey do not appear to be at immediate risk of extinction.

Description: The Klamath River lamprey is a small (14-27 cm TL, mean 21 cm), predatory lamprey that can be identified by strong, sharply hooked cusps on their oral plates. Three strong cusps on the supraoral plate ('tongue') are easily noticeable. The anterior field above the mouth has 10-15 teeth, 4 inner lateral plates on each side, resulting in the typical cusp formula of 2-3-3-2, 20- 29 cusps in line on the transverse lingual lamina (tongue plate), and 7-9 velar tentacles. The trunk usually has 60-63 myomeres (range of 58-65). The disc length is about 9 percent of the total body length, and is at least as wide as the head. The horizontal eye diameter is about 2 percent of the total body length. Although similar to Pacific lampreys, Klamath River lampreys tend to be more heavily pigmented. Ammocoete larvae have not been described.

Taxonomic Relationships: The Klamath River lamprey was described by Vladykov and Kott (1979), from specimens caught in the Klamath River, California. Four other lamprey species have also been described from the upper Klamath River basin: dwarf Pacific lamprey (*E. tridentata*), Pit-Klamath brook lamprey *(E. lethophaga)*, Miller Lake lamprey (*E. minimus)* and Modoc brook lamprey *(E. folletti).* The Pit-Klamath brook lamprey is the common nonpredatory lamprey of the upper Klamath and Pit river drainages, while the Miller Lake lamprey is an unusually small predatory form that is confined to the upper basin in Oregon (Lorion et al. 2000). The Modoc brook lamprey was also described by Vladykov and Kott (1976), from specimens collected from Willow Creek (Modoc County), a tributary to Clear Lake Reservoir on the Lost River. Although described as nonpredatory, it was later found to be predatory, providing little reason to separate it from Pacific lamprey (C. Bond, pers. comm. 1995). Consequently, Modoc brook lamprey has not been accepted as a separate species (Nelson et al. 2004). In contrast, the Klamath River lamprey is morphologically and biochemically distinct (Docker et al. 1999, Lorion et al. 2000, Lang et al. 2009).

Life History: No specific life history information is currently available, although Klamath River lamprey appear to be non-migratory and are resident in both rivers and lakes of the Klamath basin. Adults prey on adult coho and Chinook salmon and other large fishes in the basin. Wales (1951) thought that lamprey predation on migratory salmon was a major factor limiting salmon abundance in the Shasta River, because he observed such a high frequency of salmon with lamprey wounds (41%) and because "lampreys are abundant in the Shasta (p. 33)." However, salmon mortalities have not been attributed to lamprey predation in recent spawning ground (carcass) surveys or at weir operations (B. Chesney, CDFW, pers. comm. 2011).

Habitat Requirements: Little is known about the habitat requirements of Klamath River lamprey. Presumably, ammocoete larvae have the same basic requirements as those of Pacific lamprey, living in backwaters with soft substrates. The environmental tolerances of Klamath River lamprey have not been documented but they are likely similar to those of Pacific lamprey. If this is the case, then Klamath River lamprey need cold, clear water (Moyle 2002) for spawning and incubation. They also require a diverse range of habitats to complete their life cycle. Adults typically use spawning gravel to build nests, while ammocoetes burrow in soft sediments for rearing (Kostow 2002). Ammocoetes also need larger substrates as they grow (Stone and Barndt 2005) and algae for food (Kostow 2002) in habitats with slow or moderately slow water velocities (0-10 cm/s; Stone and Barndt 2005).

Distribution: Klamath River lamprey are found throughout the Klamath River basin in mainstem rivers, including the Trinity River in northern California and the Klamath River in southern Oregon (Boyce 2002). Their distribution in the lower Klamath and Trinity basins likely coincides with those of spawning Chinook and coho salmon, their main prey in the lower river, and with large suckers and cyprinids in the upper basin. However, detailed distribution of this species is not known.

Trends in Abundance: As with other upper Klamath basin lampreys, abundance estimates for Klamath River lamprey do not exist. However, they appear to be common throughout their range (S. Reid, pers. comm. 2008).

Nature and Degree of Threats: The declining quality of aquatic habitats throughout much of the Klamath-Trinity drainage, as well as the declining number of salmon (NRC 2004), make it likely that Klamath River lampreys are less abundant than they once were (Table 1). Generally, any factor that reduces abundance of large prey species is likely to also reduce Klamath River lamprey abundance (Moyle 2002).

Dams. Seven major dams are present in the Klamath-Trinity River basin. These dams change the physical and biological characteristics of the streams where they occur (Knighton 1998). In particular, they may limit or inhibit the longitudinal (upstream-downstream and vice-versa) movements of fishes, including both Klamath River lamprey and their prey, thereby limiting access to suitable spawning and rearing habitats. Dams have also degraded the quality of preferred habitat in the main stem Klamath River (Hamilton et al. 2011).

Agriculture. Alfalfa production and pasture in the Shasta and Scott basins may diminish flows, particularly in dry water years (NRC 2004). Diminished flows can reduce suitable habitats in streams, as well as create conditions (e.g., high water temperatures, low dissolved oxygen levels) that increase salmonid mortality, thereby reducing adult Klamath River lamprey prey availability. Diversion of water, warm polluted return water, and similar by-products of agriculture are also presumably limiting lamprey populations.

Grazing. Livestock grazing is pervasive in Klamath River watersheds, with disproportionate effects on smaller tributaries, reducing water and habitat quality (USFWS 1991). Grazing practices in some subbasins (e.g., Shasta River) have altered stream morphology and degraded habitat quality to the detriment of native fishes

(USFWS 1991, Gosnell and Kelly 2010). Grazing can lead to localized increases in water temperature when riparian vegetation is removed, as well as low oxygen concentrations from excess fecal nutrient loading.

	Rating	Explanation
Major dams	Medium	Seven major dams exist in the system and likely disrupt instream movement, gene flow, and opportunities for recolonization
Agriculture	Medium	Major influence on Scott and Shasta rivers by reducing salmon prey abundance (NRC 2004)
Grazing	Medium	Pervasive in Klamath River watersheds with disproportionate effects on smaller tributaries
Rural residential	Low	Widespread rural development throughout range but housing densities very low
Urbanization	n/a	
Instream mining	Low	Legacy effects have likely reduced the amount and quality of suitable habitats
Mining	Low	Impacts are unknown but assumed to be minor
Transportation	Medium	Roads are a source of sediment that may affect spawning and rearing
Logging	Medium	Widespread changes to watersheds; greater impact in past than today
Fire	Low	While wildfires are common throughout the Klamath basin, direct impacts to Klamath River lamprey are likely minimal
Estuary alteration	n/a	
Recreation	n/a	
Harvest	n/a	
Hatcheries	n/a	
Alien species	Low	No known impacts

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

Instream mining. Instream mining may alter larval rearing habitats through scour and deposition and through direct displacement of ammocoetes. When the Scott River and other areas were dredged for gold in the 19th and 20th centuries, large areas of potential habitat were destroyed; when combined with dewatering from diversions (often relicts of mining), past dredging may have had considerable legacy effects upon lamprey populations and their habitats.

Transportation. Roads, both paved and unsurfaced, have been built within the riparian corridor of many Klamath streams (USFWS 1991). Many miles of dirt roads have also been built in most of the Klamath-Trinity watersheds. Road building can decrease the quality of nearby aquatic environments to the extent of altering animal behavior and overall species composition (Trombulak and Frissell 2000). Road building can decrease the amount of canopy cover over streams and potentially increase water temperatures, limit the ability of streams to meander, impair the creation of slow water habitats, and increase sediment and pollutant input from surface run off. Increased fine sediment input into streams can decrease the quality of spawning gravels for adult lamprey and other fishes. However, it is possible that increased sedimentation may provide additional habitat for lamprey larvae.

Logging. The entire Klamath-Trinity basin has been heavily logged with attendant impacts on streams, especially increases in sedimentation from logging roads. Certain logging practices can alter the hydrology of streams (Wright et al. 1990), such that habitats become unsuitable for some fishes (Reeves et al. 1993). As with road building, logging can increase the amount of solar radiation reaching streams, decrease the amount of nutrients entering food webs, impair recruitment of large woody debris (habitat complexity, cover) and increase the amount of fine sediment eroding from hillslopes into streams. However, with current California timber harvesting rules, logging had a much more pronounced impact on stream habitats in the past than it does today (NRC 2004).

Effects of Climate Change: The potential impacts of predicted climate change to Klamath River lamprey are poorly understood because so little is known about their biology, life history, or environmental tolerances. Nevertheless, increased water temperatures (> 22 °C) brought about by climate change may increase incidence of deformities and mortalities of incubating eggs and larvae, as has been observed in Pacific lamprey populations (Meeuwig et al. 2005). Summer water temperatures already frequently exceed 20°C in many streams in the Klamath River basin and temperatures are expected to increase under all climate change scenarios (Hayhoe et al. 2004, Cayan et al. 2008). Increased summer temperatures may affect the growth and metabolic costs of juvenile and adult Klamath River lamprey that hold and rear in rivers throughout the summer. Climate change is also predicted to change the flow regimes in rivers. Klamath River flows may peak earlier in the spring and continue tapering through the summer before pulsing again later in the fall. The resulting changes in river flow and temperature may change the timing of adults and juveniles entering and exiting streams. High flows can disrupt incubation and rearing habitat due to increased bed mobility (Fahey 2006). However, flow alterations associated with climate change may be attenuated by dam operations. Shifts in distribution are expected to be upward in elevation and northward in latitude but may be impeded by passage through dams and culverts, along with increased metabolic costs associated with increased water temperatures. Moyle et al. (2013) rated Klamath river lamprey as “highly vulnerable” to extinction as the result of climate change in the next century, based on the largely speculative evidence presented above.

Status Determination Score = 3.9 - Moderate Concern (See Methods section, Table 2). The Klamath River lamprey does not appear to be at much risk, given its wide distribution within the Klamath and Trinity basins, although it may be negatively affected by climate change in the future (Table 2). The paucity of information available on this species, including present and past abundance and distribution, makes a conservation status determination difficult. Additional information is needed in order to better understand its status.

Metric	Score	Justification
Area occupied	5	Widely distributed in Klamath basin (Moyle 2002)
Estimated adult abundance	4	Unknown, but appears to be common throughout range (S. Reid, pers. comm. 2010)
Intervention dependence	5	Populations appear to be resilient and persistent
Tolerance	3	Environmental tolerances have not been identified, but are presumed similar to other lamprey species in the Klamath River basin
Genetic risk	5	No known genetic risk
Climate change	2	Potentially threatened by changes in hydrology and temperature
Anthropogenic threats	3	Five threats rated as intermediate (Table 1)
Average	3.9	27/7
Certainty (1-4)	1	Population size, distribution, and environmental tolerances largely unknown

Table 2. Metrics for determining the status of Klamath River lamprey, where 1 is a major negative factor contributing to status, 5 is factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: The principal impediment toward improved Klamath River lamprey management and conservation is the lack of empirical data and general knowledge of their abundance, distribution, environmental tolerances, and key aspects of life history. As such, the following management actions are recommended:

1. Establish a Klamath River lamprey research and monitoring program. Program goals should include: 1) a status assessment of all lampreys in the basin; 2) identification of key conservation opportunities; and 3) development of life history and habitat requirement studies, to inform a limiting factors analysis. Additionally, an identification key needs to be developed to distinguish ammocoetes of Klamath basin lamprey species.
2. Establish a lamprey data center, as part of the research and monitoring program, which would collect and integrate *all* lamprey information collected in California. The many rotary screw traps used to monitor outmigration of juvenile salmonids, in particular, are a largely untapped source of data, especially in the Klamath River system. Many trap operators record captures of lamprey ‘smolts’ and ammocoetes. The lampreys are rarely identified to species but most are likely Pacific lampreys in the lower river; however, Klamath River lampreys may also be represented in the catch.

3. Determine if conservation efforts for salmon and steelhead also benefit Klamath River lampreys, both in mainstem rivers and tributaries such as the Shasta and Scott rivers. Habitat restoration programs intended to benefit salmonids should be evaluated for their potential to create backwater habitat for lampreys. Studies should be performed to determine if populations of Klamath River lamprey are tied to those of salmon and steelhead (predator/prey relationships).

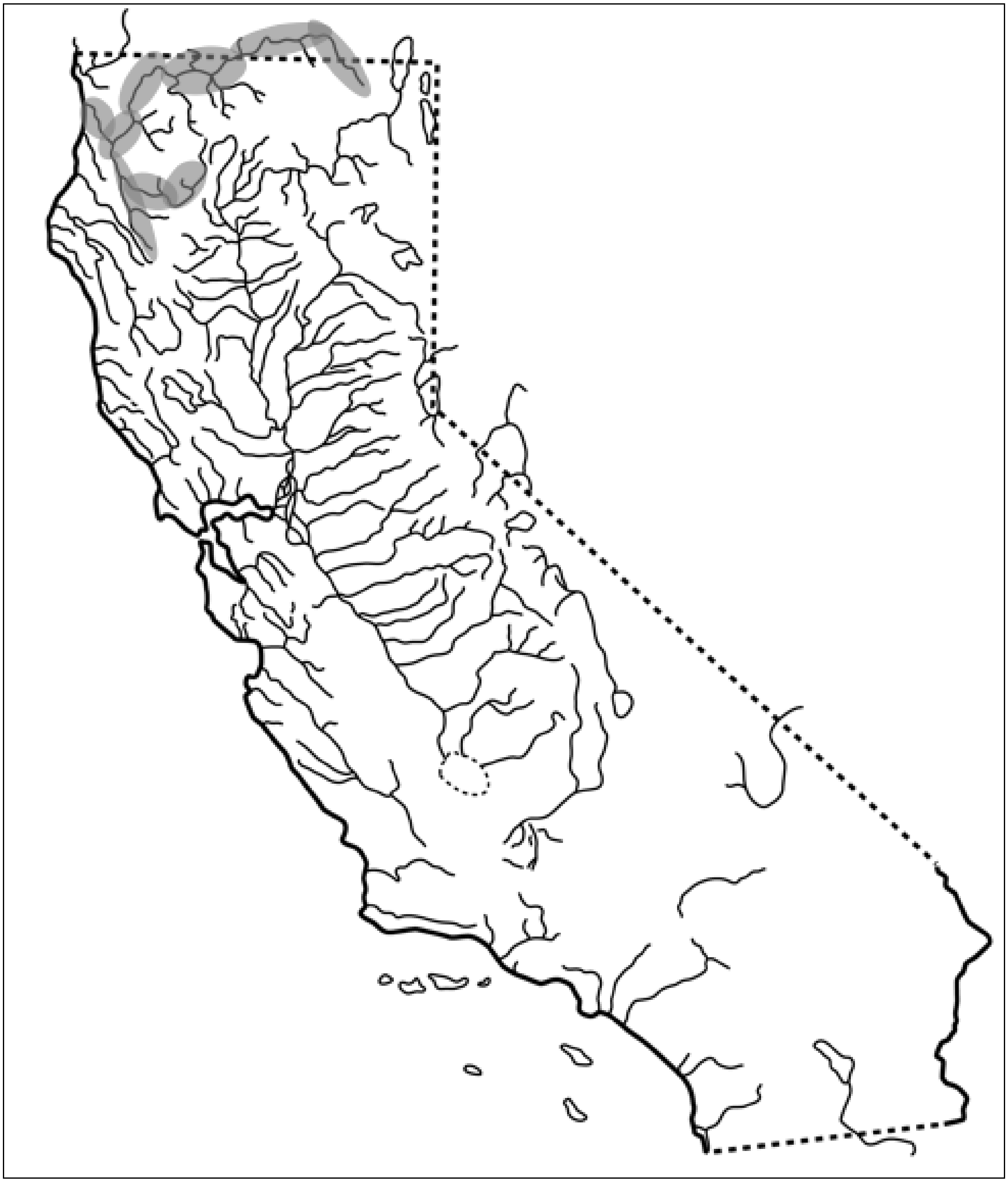


Figure 1. Distribution of Klamath River lamprey, *Entosphenus similis*, in the Klamath and Trinity rivers in California.

WESTERN RIVER LAMPREY
Lampetra ayresi

Status: Moderate Concern. Very little is known about the western river lamprey in California but it is uncommon in the state and potentially in decline.

Description: The western river lamprey is a small, predatory, species. Spawning adults reach a maximum size of about 17-18 cm TL. The oral disc is at least as wide as the head. The 'teeth' (horny plates) in the oral disc are conspicuous and pointed; however, they can be blunt in spawning individuals. The middle cusp of the transverse lingual lamina has three large lateral (circumoral) plates on each side; the outer two have two distinct cusps, while the middle one has three. The supraoral plate has only two cusps that often appear as separate teeth, while the infraoral plate has 7-10 cusps. The eye width is 1 to 1.5 times the distance from the posterior edge of the eye to the anterior edge of the first branchial opening. The number of trunk myomeres averages 68 in adults and 67 (65-70) in ammocoetes. Adult river lampreys are dark on the back and sides and silvery to yellow on the belly with a darkly pigmented tail. Ammocoetes have somewhat pale heads, a prominent line behind the eye spot, and a tail in which the center tends to be lightly pigmented (Richards et al. 1982).

Taxonomic Relationships: The western river lamprey was described in 1855 by William O. Ayres, from a single specimen collected in San Francisco Bay, as *Petromyzon plumbeus*. Because that name had already been given to a European lamprey, it was renamed *P. ayresi* in 1870. A careful redescription of the river lamprey by V.D. Vladykov and W.I. Follett (1958) demonstrated its distinctiveness. The Pacific brook lamprey (*L. richardsoni*) and Kern brook lamprey (*L. hubbsi*) apparently evolved independently from river lampreys. See the Kern brook lamprey account in this report for further discussion of taxonomic relationships.

Life History: Western river lampreys have not been studied in California (Moyle 2002); therefore, the information in this account is based on studies in British Columbia (Roos et al.1973, Beamish and Williams 1976, Beamish 1980, Beamish and Youson 1987).

Larval river lampreys (ammocoetes) begin transformation into adults when they are about 12 cm TL, during summer months. Metamorphosis may take 9-10 months, the longest known for any lamprey. Newly metamorphosed lampreys may aggregate immediately upriver from salt water and enter the ocean in late spring. Adults apparently only spend 3-4 months in salt water where they grow rapidly, reaching 25-31 cm TL.

River lampreys prey on fishes in the 10-30 cm TL size range; the most common prey appear to be herring and salmon. Unlike other species of lamprey in California, river lampreys typically attach to the back of the host fish, above the lateral line, where they feed on muscle tissue. Feeding continues even after death of the prey. River lamprey predation may negatively affect prey populations if both prey and predator are concentrated in small areas (Beamish and Neville 1995). River lampreys can apparently feed in either salt or fresh water.

Adults migrate back into fresh water in the fall and spawn during the winter or spring months in small tributary streams, although the timing and extent of migration in California is poorly known. While maturing, river lampreys can shrink in length by about 20 percent. Adults create saucer-shaped depressions in gravelly riffles for spawning by moving rocks with their mouths. Fecundity estimates for two females from Cache Creek, Yolo Co., were 37,300 eggs from one 17.5 cm TL and 11,400 eggs for one 23 cm TL (Vladykov and Follett 1958). It is assumed that adults die after spawning, although this life history attribute has not been carefully documented in California. Ammocoetes remain in silt-sand backwaters and eddies and feed on algae and microorganisms. River lampreys spend an unknown amount of time as ammocoetes (probably 3-5 years), so the total life span is likely 6-7 years.

Habitat Requirements: The habitat requirements and environmental tolerances of spawning adults and ammocoetes have not been studied in California. Presumably, like other lampreys, adults need clean, gravelly riffles in permanent streams for spawning, while ammocoetes require sandy to silty backwaters or stream edges in which to bury themselves, where water quality is continuously high and temperatures do not exceed 25°C.

Distribution: Western river lampreys occur in coastal streams from just north of Juneau, Alaska, south to San Francisco Bay. In California, they have been recorded from the Sacramento and San Joaquin Delta while migrating, tributaries to the San Francisco Estuary (Napa River, Sonoma Creek, Alameda Creek), and tributaries to the Sacramento and San Joaquin rivers (e.g. Tuolumne River, Stanislaus River, Cache Creek). A land-locked population may exist in upper Sonoma Creek (Wang 1986). There are no recent records of river lamprey in Oregon and most older records are for the Columbia River basin (Kostow 2002). Likewise, they are known only from two large river systems in British Columbia in the center of their range (Beamish and Neville 1992).

Trends in Abundance: Western river lamprey population trends are unknown in California but it is likely that they have declined, concomitant to degradation and fragmentation of suitable spawning and rearing habitat in rivers and tributaries throughout their range in the state, along with declines in prey species (e.g., Chinook and coho salmon, steelhead trout, etc.). River lamprey are abundant within a limited geographic area of British Columbia, at the center of their range, but there are relatively few records from California, which comprises the southern end of their range.

Nature and Degree of Threats: The western river lamprey has become uncommon in California; it is likely that populations are declining because the Sacramento, San Joaquin and Russian rivers, along with their tributaries, have been severely altered by dams, diversions, development, agriculture, pollution, and other factors. They spawn and rear in the lower reaches of rivers and are, thus, highly vulnerable to alteration from agriculture and urbanization, as well as pollution. Two tributary streams where spawning has been recorded in the past (Sonoma and Cache creeks) are both severely altered by channelization, urbanization, and other impacts. See the Pacific lamprey account in this report for more specific information on stressors that negatively affect anadromous lamprey abundance.

	Rating	Explanation
Major dams	Medium	Most rivers within range are regulated by major dams
Agriculture	Medium	Lower stream reaches are impacted by diversions and impaired water quality
Grazing	Low	Present along most rivers; impacts likely minimal in large river systems
Rural residential	Low	Rural development is increasing rapidly across species' range; direct effects unknown but habitat degradation and reduced instream flows likely contribute to declines
Urbanization	Medium	Known range in Central Valley mostly urbanized
Instream mining	Low	Gravel mining common in preferred spawning streams
Mining	Low	Impacts unknown
Transportation	Medium	Roads, bridges, and ship canals alter habitats and are sources of pollutants
Logging	Low	Impacts to lower portions of larger river systems likely minimal
Fire	n/a	
Estuary alteration	Medium	Extent of estuary utilization unknown; estuaries likely constitute important feeding habitats that have been heavily altered and degraded throughout the state
Recreation	n/a	
Harvest	n/a	
Hatcheries	n/a	
Alien species	Low	May be prey for some alien species; may also prey upon certain alien species (e.g., American shad)

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

Effects of Climate Change: With so little known about this species, climate change effects are hard to predict. Nevertheless, the fact that California marks the southern end of its range, combined with its presence in the lower reaches of just a few large, regulated rivers, suggests that altered flow regimes and temperatures could further reduce or eliminate populations. Moyle et al. (2013) considered river lamprey to be "highly vulnerable" to climate change mainly because of its limited distribution and likely small populations, coupled with lack of knowledge about its basic biology in California.

Status Determination Score = 3.6 – Moderate Concern (see Methods section Table 2). Very little is known about this species in California but, given its dependence on lower reaches of large, regulated rivers, the river lamprey may be vulnerable to altered flows, altered habitats through urbanization, urban and agricultural pollutants, and similar factors (Table 2). Jelks et al. (2008) list it as being 'vulnerable' to extinction due to habitat changes, while NatureServe calls it "apparently secure" over its entire range.

Metric	Score	Justification
Area occupied	4	Known from at least 5 watersheds
Effective population size	3	This rating is likely high based on limited catches in sampling programs
Intervention dependence	5	Populations appear self-sustaining; habitat improvements may benefit populations in some areas
Tolerance	3	Presumed similar to brook lamprey
Genetic risk	4	Gene flow among populations not known
Climate change	3	Poorly understood because distribution and environmental tolerances are largely unknown; score assumes reduced habitat suitability and higher water temperatures will negatively affect river lamprey populations
Anthropogenic threats	3	See Table 1
Average	3.6	25/7
Certainty (1-4)	1	Little information available

Table 2. Metrics for determining the status of western river lamprey in California, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: The western river lamprey cannot be properly managed until more is known about its biology. Studies and field surveys to assess the river lamprey's distribution, abundance, life history and habitat requirements in California should to be implemented. The lower portions of the Sacramento and San Joaquin rivers, along with portions of the Bay Delta, should be targeted for initial studies and surveys since migratory river lampreys are caught in the Delta on a regular basis in various sampling programs. Presumably, restoring natural flow regimes and reducing inputs of pollution and sediment to its spawning streams will benefit the river lamprey but, given that so little is known about its tolerances and requirements, specific restoration actions and management recommendations cannot be developed without further study.

Figure 1. Presumed distribution of western river lamprey, *Lampetra ayresi*, in California. Distribution along the north coast is based on available passage to suitable habitats, rather than actual collection records.

KERN BROOK LAMPREY
Lampetra hubbsi (Vladykov and Kott)

Status: High Concern. Only six populations of Kern brook lamprey exist and they are isolated from one another; five are in short reaches below dams, so their persistence depends on dam operations and maintenance of suitable habitats for ammocoetes. The possible discovery of a 7th population in the Sacramento River watershed, however, suggests the species may be more widely distributed than is currently known.

Description: The Kern brook lamprey is a non-predatory lamprey, so the teeth in its oral disk are small and blunt (Brown and Moyle 1992). Its morphology is like that of other lampreys: eel-like body, no paired fins, and a sucking disc instead of jaws. Larvae, known as ammocoetes, are similar to adults in shape but lack eyes and a well-developed oral disc. The Kern brook lamprey is much smaller than predatory anadromous lampreys; adults range from 81 to 139 mm TL and ammocoetes from 117 to 142 mm TL. Ammocoetes are typically larger than adults because non-predatory lampreys shrink following metamorphosis (Vladykov and Kott 1976). The number of trunk myomeres (i.e. the "blocks" of muscle mass along the body) ranges from 51 to 57 in ammocoetes (Tables 1, 2). In adults, the supra-oral lamina (tooth) typically has two cusps, with four inner lateral teeth on each side of the disc. The typical cusp formula is 1-1-1-1 (Vladykov and Kott 1976). The sides and dorsum are a grey-brown and the ventral area is white. Dorsal fins are unpigmented, but there is some black pigmentation restricted to the area around the notochord in the caudal fin (Vladykov and Kott 1976).

Taxonomic Relationships: The Kern brook lamprey was first described by Vladykov and Kott (1976) as a dwarf, non-predatory species in the genus *Entosphenus*. Based on dentition, the describers indicated the Kern brook lamprey was derived from the predatory Pacific lamprey, *E. tridentatus*, as are some other brook lampreys (Docker 2009). However, molecular analysis demonstrated it was derived from the predatory river lamprey, *Lampetra ayersi*, as is the western brook lamprey, *L. richardsoni* (Docker et al. 1999, Lang et al. 2009). Boguski et al. (2012) examined the genetics of lampreys from many populations in Pacific coast drainages; a single ammocoete from Paynes Creek (Tehama County) proved to be closely related to *L. hubbsi*. There are three potential scenarios to explain this: (1) it is a single, highly isolated population of *L. hubbsi*; (2) it is a separate undescribed species, and (3) other *L. hubbsi* populations exist in watersheds in the Sacramento Valley but have been overlooked. Clearly, more work on lamprey distribution and systematics in California is needed. The Pacific brook lamprey is differentiated from Kern brook lamprey on the basis of anatomical features (Tables 1, 2), as well as by mitochondrial DNA. The two species do not appear to be sympatric.

Table 1. Comparative counts and measurements of lamprey ammocoetes. *L ayersi* is from Vladykov (1973), *L. tridentata* and *L. hubbsi A*, from Vladykov and Kott (1976, 1979), *L. ayersi* from Richards et al. (1982) and *L. hubbsi B* from Brown and Moyle (unpubl. data). Data from Brown and Moyle are given as mean ± S.D. (above) and range (below). Data from other studies are mean (above) and range (below).

	Lampetra ayresi	*L. richardsoni*	*L. tridentata*	*L. hubbsi* A	*L. hubbsi B*
Total length (mm)	-	117	128	130	106 ± 19
	69 - 119	75 - 143	117 - 144	66 - 140	
Trunk myomeres	65	54	68	55	54 ± 2
	63 - 67	52 - 57	66 - 70	53 - 57	51 – 5

Table 2. Diagnostic characteristics of recently transformed adult lampreys of four *Lampetra* species. Data are from Vladykov and Follett (1958, 1965), Vladykov (1973) and Vladykov and Kott (1976).

	L. ayresi	*L. richardsoni*	*L. tridentata*	*L. hubbsi*
Trunk myomeres	68	56	66	56
	(60 - 71)	(53 - 58)	(63 - 70)	(54 - 57)
Cusps on supraoral lamina	2	2	3	2 - 3
Inner lateral "teeth"	3	3	4	4
Cusps on infraoral lamina	8.9	7.7	5.1	5.0
	(7 - 10)	(7 - 10)	(5 - 6)	5
Row of posterial "teeth"	absent	absent	present	present[1]
Predatory?	yes	no	yes	no

[1]Absent from two of eleven specimens examined by Brown and Moyle (unpublished data)

Life History: No documentation of the life history of Kern brook lamprey exists. However, if their life history is comparable to that of other non-predatory brook lampreys, they should live for approximately 4-5 years as ammocoetes before metamorphosing into adults (Moyle 2002). Based on collections (P. Moyle and L. Brown, unpublished data), metamorphosis occurs during fall. The adults presumably over-winter and spawn the following spring after undergoing metamorphosis.

Habitat Requirements: Principal habitats of Kern brook lamprey are silty backwaters of large rivers in foothill regions (mean elevation= 135 m; range= 30-327 m). In summer, ammocoetes are usually found in shallow pools along edges of run areas with minimal flow (L.R. Brown, US Geological Survey, pers. comm.), at depths of 30-110 cm where water temperatures rarely exceed 25 degrees C. Common substrates occupied are sand, gravel, and rubble (average compositions are 40%, 22%, 23%, respectively). Ammocoetes seem to favor sand/mud substrate, where they remain buried with the head protruding above the substrate and feed by filtering diatoms and other microorganisms from the water. This type of habitat is apparently present in the siphons of the Friant-Kern Canal. Adults require coarser gravel-rubble substrate for spawning. Temperature requirements for Kern brook lamprey are not known but the fact they are present almost entirely in reaches where summer temperatures rarely exceed 24 degrees C suggests a cool-water requirement.

Distribution: The Kern brook lamprey was first discovered in the Friant-Kern Canal (hence the inaccurate name; it is not found in the Kern basin). It has since been found in six locales which, presumably, represent isolated populations: the lower reaches of the Merced River, Kaweah River, Kings River, and San Joaquin River, as well as in the Kings River above Pine Flat Reservoir and the San Joaquin River above Millerton Reservoir, but below Redinger Dam (Brown and Moyle 1987, 1992, 1993; Fig. 1). In 1988, ammocoetes and adult lampreys were found in several siphons of the Friant-Kern Canal, when they were poisoned during an effort to rid the canals of white bass (*Morone chrysops*). The "low-count" lampreys (i.e., low numbers of trunk myomeres) reported from the upper San Joaquin River between Millerton Reservoir and Kerckhoff Dam by Wang (1986) are also most likely *L. hubbsi*, as are similar ammocoetes from the Kings River above Pine Flat Reservoir. As indicated in the taxonomy section, presumed Kern brook lampreys have been identified from Paynes Creek, Tehama County, which may indicate other populations exist as well.

Trends in Abundance: Since this species was first discovered in 1976, attempts to fully document its range have been only partially successful. Little is known about its past or present abundance. However, data collected to date suggest that this species is a San Joaquin basin (including the Kings River) endemic (Brown and Moyle 1992, 1993). Isolated populations of Kern brook lamprey seem spottily distributed throughout the San Joaquin drainage in regulated rivers, so their distribution and abundance are probably much reduced from pre-dam times. Ammocoetes thrive in the dark siphons of the Friant-Kern Canal, but it is unlikely that there is suitable spawning habitat in the canal, so those individuals probably do not contribute to the persistence of the species.

Nature and Degree of Threats: Populations of this species are scattered throughout the middle San Joaquin-Kings drainage and are isolated from one another. Such a limited and fragmented distribution makes local extirpations increasingly probable, along with a high degree of genetic risks from small population sizes and isolation; without interconnected populations and the possibility of recolonizing degraded habitats, eventual extinction may occur.

Major dams. It is likely that the river reaches flooded by Millerton and Pine Flat reservoirs were once important habitats for Kern brook lamprey. Today, the probability of local extirpation is increased by the fact that all known populations, with one exception, are located below dams, where stream flows are regulated without regard to the habitat requirements or life history needs of lampreys. Fluctuations or sudden drops in flow may isolate ammocoetes or result in the drying of habitats. Gravels required for spawning may be eliminated (trapped by dams) or compacted so they cannot be used by adults, while silt required by ammocoetes may be flushed out of the cool-water reaches that appear to be preferred by larvae. Dams also isolate populations, eliminating gene flow and preventing recolonization from nearby populations. Management of flows in the lower reaches of the San Joaquin and Kings rivers, including the new restoration flows below Friant Dam, as well as flows to reduce impacts from agricultural return waters, will need to account for the needs of this species in order for populations to persist.

Agriculture. Channelization, road building, irrigation withdrawls, and other activities associated with farming eliminate backwater areas required by ammocoetes. Ammocoetes may also be carried by water being delivered to farms via the Kings River to "dead-end" habitats such as the Friant-Kern siphons. In addition, pollutants are of concern (including elevated temperatures) in agricultural return waters, which may reduce lamprey survival.

Urbanization. Fresno is rapidly expanding around the San Joaquin River with attendant stressors associated with urban development, including road building, bank stabilization, pollution, and recreation.

Instream mining. Large sections of the San Joaquin River have been mined for gravel, both destroying shallow-water habitats needed by ammoceotes and creating large pits that provide ideal habitats for predatory fishes. It is likely that lampreys were extirpated from gravel pit regions once mining began.

Alien species. Kern brook lamprey habitats typically support a mixture of native and non-native fishes (Moyle 2002). The impacts of alien fishes, especially predatory bass (*Micropterus* spp.), are not known, but are likely to be negative, given the vulnerability of migrating larvae and adults to predation.

	Rating	Explanation
Major dams	High	Most populations exist below dams, where habitat is degraded and flows are highly regulated
Agriculture	High	Most populations are susceptible to agricultural pollution, diversions and other factors
Grazing	Low	Present along some streams
Rural residential	Low	Effluent from waste water and bank protection to reduce flooding may affect habitats
Urbanization	Medium	Fresno and other urban areas are expanding; potential for increased impacts from pollution, habitat degradation and fragmentation
Instream mining	Medium	Gravel pits present in some areas; associated impacts may have eliminated lampreys from reaches of the San Joaquin River
Mining	n/a	
Transportation	Low	Roads and railroads along rivers may alter habitats and increase both sediment and pollutant input
Logging	n/a	
Fire	Low	
Estuary alteration	n/a	
Recreation	Low	Areas accessible to off-road vehicles and other uses may reduce ammocoetes habitats or disrupt spawning
Harvest	n/a	
Hatcheries	n/a	
Alien species	High	Alien predators present; effects unknown but potentially significant

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

Effects of Climate Change: The southern Central Valley of California is predicted to experience reduced stream flows and increased water temperatures, as a result of longer, more frequent, droughts and warmer air temperatures. Kern brook lampreys live in regulated rivers, so climate change effects are most likely to manifest from changes in dam and reservoir operations, including reduced dam releases (drying up rearing areas) or warmer temperatures of released water. Without consideration for lamprey needs, such operational changes can greatly increase extinction risk. Moyle et al. (2013) indicated the Kern brook lamprey is "critically vulnerable" to climate change, facing extinction because of changed dam operations, including reduced flows during droughts, and alteration/degradation of habitats to favor expansion of alien species.

Status Determination Score = 2.3 - High Concern (see Methods section, Table 2). The Kern brook lamprey does not appear to be at immediate risk of extinction but its status could change rapidly, given the limited number of isolated populations and their existing distribution either below or just above dams. Jelks et al. (2008) considered the species as threatened and declining, while NatureServe considers its status to be somewhere between Imperiled (G2) and Critically Imperiled (G1). The species was petitioned for federal listing in 2003 as threatened, but the petition was denied on Dec. 27, 2004 because "the petition did not provide sufficient information to warrant initiating a status review (USFWS 2004)."

Metric	Score	Justification
Area occupied	2	Six known populations occur in two watersheds but all are isolated from one another by dams and diversions; possible 7^{th} population needs further investigation
Estimated adult abundance	3	Not known but probably <1000 adults in each population
Intervention dependence	3	Long-term persistence requires habitat improvements and flow regulation
Tolerance	3	Unstudied but probably moderate
Genetic risk	2	Populations fragmented; potential for bottlenecks or inbreeding depression
Climate change	1	Populations below dams could be threatened by changes in river management
Anthropogenic threats	2	See Table 3
Average	2.3	16/7
Certainty (1-4)	2	Little published information on abundance, distribution, or status, especially in the recent past

Table 4. Metrics for determining the status of Kern brook lamprey, where 1 is a major negative factor contributing to status, 5 is factor with no or positive effects on status, and 2-4 are intermediate values. This score does not take into account the apparent population in the Sacramento River watershed. See methods section for further explanation.

Management Recommendations: The Kern brook lamprey would most benefit from proactive management strategies and actions treating it as if it were already a listed species, in order to reduce the probability of actual listing. A thorough survey of the known habitats and populations of this species needs to be conducted to determine status and possible trends. Extensive surveys are needed to determine present distribution and to provide more exact information on habitat requirements within its known range, as well to determine if populations exist outsides the known range (e.g., in the Kaweah River, Sacramento Valley). A study needs to be conducted to determine if ammocoetes still use the silty bottoms of siphons in the Friant-Kern Canal and if rescue and transplantation of these larvae would be beneficial. Specialized surveys should focus on adults to determine population sizes and spawning habitat requirements. Known or probable populations should be monitored every two to five years, with trends determined by catch per effort or estimated densities of ammocoetes.

Once surveys are completed, several known areas of suitable habitat should be selected for special management or protection from incompatible uses, including some in the soon-to-be-restored San Joaquin River. These same areas should be the focus of life history studies and studies that determine habitat requirements.

Figure 1. Known (confirmed) distribution of Kern Brook lamprey, *Lampetra hubbsi* , in California.

WESTERN BROOK LAMPREY
***Lampetra richardsoni* (Vladykov and Follet)**

Status: Moderate Concern. Western brook lampreys are still present in the least disturbed portions of many watersheds but all populations are likely small, isolated and declining.

Description: Western brook lampreys are small, usually less than 18 cm TL, and nonpredatory (Moyle 2002). They have poorly developed tooth plates in the oral disc and tooth plates in spawning adults may be missing from the anterior field. The supraoral plate is wide with one cusp at each end. The infraoral plate has 6-10 toothlike cusps and 3 circumoral plates on each side of the mouth. The middle circumoral plate has 2 or 3 cusps. Cusps on the transverse lingual lamina are inconspicuous. The oral disc is narrower than the head with a length that is less than 6 percent of the total length. Both adults and ammocoetes have trunks made up of 52-67 myomeres (52-58 in California populations). Body coloration is dark on the sides and back, and light (yellow or white) on the underside. Ammocoetes have dark tails and heads above the gill opening (Richards et al. 1982).

Taxonomic Relationships: The western brook lamprey was determined to be a species, *L. richardsoni*, distinct from the European brook lamprey, *L. planeri*, in 1965, but closely related to the predatory river lamprey, *L. ayersi* (Vladykov and Follett 1962). Later, populations in Oregon and California were described as belonging to *L. pacifica* by Vladykov (1973). C. Bond, in an unpublished study, determined that differences in myomere counts that were thought to distinguish *L. pacifica* from *L. richardsoni* did not do so when populations throughout their range were sampled, so the name was quashed without further review by the American Fisheries Society (Robins et al. 1991, Stewart et al. 2011). Stewart et al. (2011) determined it is, indeed, a valid species but confined to the Columbia River basin. Boguski et al. (2012) examined nominal river and brook lampreys from the entire Pacific Coast and found that, for the most part, the non-predatory brook lampreys conformed to *L. richardsoni*, on the basis of both morphology and genetics (mitochondrial DNA). However, there were some notable exceptions:

- The Kern brook lamprey was confirmed to be a distinct species, with a possible additional population in Paynes Creek, Tehama County (see the Kern brook lamprey account in this report for further information).
- A very distinctive population (based on mitochondrial DNA) was found isolated in Kelsey Creek, Lake County, a tributary to Clear Lake. Further investigation is needed to determine if this is another endemic species in the Clear Lake watershed.
- The population in Mark West Creek, a tributary to the lower Russian River, was found to be genetically distinct, perhaps indicating a distinct lineage in the Russian River.

The western brook lamprey is very similar to the river lamprey, based on mitochondrial DNA analysis (Docker et al. 1999). The nonpredatory brook lampreys in many coastal streams are, therefore, potentially derived from river lamprey through a series of independent evolutionary events, found in other “pair species” of lampreys as well (Docker 2009). Brook lamprey adults are not known to migrate although, in British Columbia, some streams contain both predatory and nonpredatory adults, with the predatory form able to migrate to salt water (Beamish 1987, Beamish et al. 2001). River and brook lampreys hybridize in the laboratory but hybridization in the wild has not been observed (Beamish and Neville 1992). Docker (2009) suggested that the distinctness of members of species pairs of lampreys depends on how recently the non-predatory form developed. Long isolation leads to speciation, as in the Kern brook lamprey. It is clear that further research on the systematics of the brook lamprey is required; however, mounting evidence indicates that California populations are distinct.

Life History: Most published studies relating to western brook lampreys were done outside of California (Schultz 1930, McIntyre 1969, Kostow 2002, Gunckel et al. 2009), with the exception of a study by Hubbs (1925). It is assumed, however, that differences in biology between California populations and those elsewhere are minor, based on unpublished observations (cited below).

Spawning adult brook lamprey build nests in gravel riffles that are slightly smaller in diameter than their body lengths. In Mark West Creek, during April, 1994, they were observed building nests 15-20 cm wide in gravel riffles at a depth of about 15 cm (M. Fawcett, pers. comm. 1998). In the Smith River, Oregon, most nests are about 12 cm (length) by 11 cm (width) by 3 cm (depth) and are located in low velocity (ca. 0.2 m/sec) water averaging 13 cm depth (Gunckel et al. 2006, 2009). Median gravel size in nests is 24 mm and most nests are associated with cover (boulder, wood, vegetation). Sixty-eight percent of nests in the Oregon study were found in either pool tail-outs or low gradient (<2% slope) riffles. Spawning begins when water temperatures exceed 10°C (Schultz 1930, Kostow 2002). However, in Cedar Creek, Washington, spawning occurred at temperatures ranging from 8.6 to 17.4°C (Stone et al. 2002). In California's North Fork Navarro River, spawning begins in early March, peaks between mid-April and mid-May, and may continue through the first week of June (S. Harris, pers. comm. 2011). In Outlet Creek (Eel River watershed), spawning begins slightly later (mid-March), peaks in late-April to late-May, and continues through mid-June (S. Harris, pers. comm. 2011).

Spawning behavior is similar to that of Pacific lamprey (Schultz 1930, Morrow 1980). In Cedar Creek, 3 to 12 lampreys were observed working together to move large rocks out of the nest prior to spawning (Stone et al. 2002). Upon completion of the nest, adhesive eggs are deposited and covered with sand and gravel (summarized in Kostow 2002). Adults die after spawning. Length of the spawning season varies from 6 months in Washington (Schultz 1930), where flow conditions are more constant, to 2 months (March-April) in Coyote Creek (Alameda County) (Hubbs 1925). Fecundity ranges from 1,100 to 5,500 eggs per female (Wydoski and Whitney 1979, Kostow 2002). Eggs hatch in about 30 days at 10°C, 17 days at 14°C, 12 days at 18°C and 9 days at 22°C (Meeuwig et al. 2005). Speckled dace (*Rhinichthys osculus*) and salmonids (*Oncorhynchus* spp.) have been observed to feed on eggs in and around lamprey nests (Brumo 2006).

After hatching, embryos and larvae (ammocoetes) may spend another week to a month in the nest (summarized in Kostow 2002). Once they reach about 10 mm, ammocoetes leave the nest and move downstream, usually at night, to burrow tail first into deposits of fine sediment; their mouths are located near the substrate surface so that they can filter feed. Movement of ammocoetes occurs year-round, mostly at night (Kostow 2002), but is primarily associated with increases in discharge (Stone et al. 2002). Ammocoetes move further downstream into deeper water as they grow (Kostow 2002). Ammocoetes are most common in sandy and silty areas of backwaters and pools, occurring in aggregations as dense as 170 per square meter (Schultz 1930). However, densities in two sites of the South Fork Walla Walla River, Washington and Oregon, were 5 and 37 individuals per square meter, respectively (Close et al. 1999). Western brook lampreys live as ammocoetes for 3-4 years in California and Oregon, and 4-6 years in British Columbia (Hubbs 1925, Schultz 1930, Pletcher 1963, Wydoski and Whitney 1979). California populations grow the fastest and largest (13-18 cm) by feeding on algae (especially diatoms) and organic matter (Wydoski and Whitney 1979). Ammocoetes begin transforming in the fall and mature by spring. Individuals develop eyes and an oral disc and undergo physiological changes in the gills and nasopineal gland (Kostow 2002). They become dormant in burrows during the transformation stage and do not feed as adults.

Where western brook and Pacific lamprey co-occur, there can be some degree of overlap in spawning habitat; in some cases western brook lamprey will spawn within Pacific lamprey nests (Stone et al. 2002, Luzier and Silver 2005, Brumo 2006, Gunckel et al. 2006, 2009). However, western brook lamprey generally spawn further upstream in smaller tributaries than Pacific lamprey. The bile acid, petromyzonol sulfate, may be used as a chemical cue between conspecifics (Yun et al. 2003), perhaps influencing in-river distribution.

Habitat Requirements: Western brook lampreys have habitat requirements similar to those of salmonid species, with which they co-occur. They need clear, cold, water in little disturbed watersheds, as well as clean gravel near cover (boulders, riparian vegetation, logs, etc.) for spawning. Additional habitat requirements include areas with low flow velocities and fine sediments for rearing that are not excessively scoured under high flows. Habitat utilization surveys of spawning western brook lamprey in Cedar Creek, Washington, found that adults avoided areas with deep, fast water and large substrates, suggesting specific habitat needs for spawning (Luzier and Silver 2005). Lamprey presence was positively correlated with temperature, percent fine substrate and dissolved oxygen and negatively correlated with stream gradient, velocity, percent bedrock and percent large gravel (Stone et al. 2002). In the Tualatin River basin, Oregon, western brook lampreys were most commonly found in shady glides or riffles with relatively fine substrates (soil or rock), in stream reaches without obvious signs of habitat degradation (Leader 2001). Optimum temperatures for embryo and larval development are 10-18°C (Meeuwig et al. 2005).

Distribution: Western brook lampreys occur in coastal streams from southeastern Alaska south to California and inland in the Columbia and Sacramento-San Joaquin River drainages (Vladykov 1973, Morrow 1980). California populations are primarily found in the Sacramento River watershed, including remote areas such as Kelsey Creek, upstream of Clear Lake (Lake County), and St. Helena Creek (Lake County), a tributary to upper Putah Creek. They are also found upstream of Pillsbury, Morris and Centennial reservoirs in the Eel River drainage (Mendocino County) (Brown and Moyle 1996, S. Harris, pers. comm. 2011) and in tributaries to the Russian River, such as Mark West Creek (Sonoma County) (M. Fawcett, pers. comm. 1998) and Austin Creek (J. Katz, pers. obs. 2009). Spawning adults have been collected from the Navarro River, Mendocino County (J.B. Feliciano, pers. comm. 1999). Ammocoetes were once collected from the Los Angeles River (Culver and Hubbs 1917) but they have been extirpated from this highly degraded system (Swift et al. 1993, Swift and Howard 2009). Hubbs (1925) also collected ammocoetes from Coyote Creek, Santa Clara County. They likely occur in other coastal rivers systems as well (Moyle 2002). Boguski et al. (2012) note that isolated populations they examined (e.g. from Kelsey Creek) are often genetically distinct and may deserve recognition as separate taxa.

Trends in Abundance: Western brook lampreys are probably more common than survey data indicate because they are difficult to observe and to distinguish from other species (Kostow 2002, Moyle 2002). In Oregon, they are assumed to occur in less than half of their historic habitats in the Columbia River and Willamette River subbasins (ODFW 2006). Consequently, they are considered to be "at risk" due to habitat loss, passage barriers and pollution. However, they are still common in other parts of Oregon such as the Smith River (tributary to the Umpqua River), where an estimated 4,692 (2004) and 4,265 (2005) western brook lamprey nests were observed (Gunckel et al. 2006). Abundance data for California populations are not available and there are no records of spawning numbers such as those observed in Oregon.

Nature and Degree of Threats: Little is known about the factors limiting abundance or distribution of western brook lamprey in California. Threats to western brook lamprey in Oregon include pollution, logging, degraded water quality, changes to natural hydrographs (including rapid reduction in flows, scouring), dredging and development in floodplains and low gradient stream reaches (Kostow 2002). It is likely that some, if not all, of these stressors also affect populations in California streams. In particular, brook lamprey populations are exceptionally vulnerable to single transitory events (pollution, dewatering) that can kill relatively immobile ammocoetes. Local extinctions caused by such events are likely to go unnoticed.

Major dams. Many streams occupied by western brook lampreys are dammed and/or diverted to some extent; small diversions are more prevalent than large dams in most portions of their range. Major dams on coastal and Central Valley rivers have likely fragmented habitats and isolated populations in upstream areas, as has been documented elsewhere (Close et al. 1999). Where altered flow regimes below dams have changed habitats (e.g. reduced backwaters, increased summer temperatures) brook lamprey are generally absent.

Agriculture. Western brook lamprey tend to occur in low gradient reaches of California streams that are impaired, to varying degrees, by local agriculture, both legal and illegal (e.g., marijuana cultivation). Such streams may be less suitable for all lamprey life stages as the result of diversions, pollution and poor water quality from agricultural return waters. For example, the rapid expansion of vineyards in coastal watersheds has likely reduced habitat quality and quantity for lampreys in many areas.

Grazing. Livestock grazing in headwater streams favored by brook lampreys alters channel morphology (stream bank degradation, widening and shallowing of stream channels), increases sedimentation (potentially degrading spawning habitats but also potentially increasing abundance of fine sediment deposition areas utilized by ammocoetes), reduces riparian vegetation (stream shading and water temperature moderation) and may cause localized impacts due to pollution input from animal wastes.

	Rating	Explanation
Major dams	Medium	Dams block passage, alter natural flow regimes and sediment budgets
Agriculture	Medium	Many populations affected by polluted water and reductions in flows from diversions
Grazing	Medium	Grazing occurs throughout species' range
Rural residential	Medium	Rural development increasing within species' range; may cause localized pollution and habitat degradation in many areas
Urbanization	Medium	Lampreys are absent from heavily urbanized areas
Instream mining	Low	Dredging formerly impacted many areas occupied by lampreys; dredging currently prohibited in CA
Mining	Low	Legacy toxic effects of mine drainage may still affect populations; may be particularly acute to ammocoetes, due to filter feeding in substrates where mercury accumulates
Transportation	Medium	Roads (particularly unsurfaced roads in headwater areas) can increase sediment delivery and fragment and degrade habitats
Logging	Medium	Most streams in species' range are affected by logging and logging roads
Fire	Medium	Forest fire frequency and intensity are increasing in species' range
Estuary alteration	n/a	
Recreation	n/a	Recreational impacts to lamprey populations are unknown
Harvest	n/a	
Hatcheries	n/a	
Alien species	Medium	Unknown impacts but co-occurrence likely throughout much of range

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

Rural residential. Rural communities are common throughout the species' range and rural development in many areas is increasing rapidly. Development (e.g., road building, building site preparation, water and power delivery), along with pollution from septic tanks and household wastes, can degrade aquatic habitats and water quality.

Urbanization . Urban development along streams (e.g., Mark West Creek in Santa Rosa) decreases the abundance of rearing habitats, while pollutants can kill adults and ammocoetes. Channelization simplifies stream morphology and often eliminates edge habitats needed by ammocoetes. Lampreys are usually absent from urban streams, such as the Los Angeles River and Coyote Creek, in which they were historically present, indicating that urban development adjacent to streams has a significant impact on their persistence.

Mining. Eggs, embryos and ammocoetes may have been negatively affected by suction dredging in the past; however, there is currently a moratorium on suction dredging in California. Nonetheless, dredging is still considered an important threat in Oregon (Kostow 2002) and could become so again in California if the moratorium is lifted. Legacy effects from widespread historic hard-rock mining (e.g., for mercury) may have eliminated or reduced populations in many areas. Toxins (e.g., heavy metals) from mostly historic mining operations may still persist in stream substrates, causing direct and prolonged exposure to ammocoetes with unknown effects on this life history stage. Instream gravel mining operations may contribute to removal of important spawning habitats or disruption of habitat utilization by all life stages.

Transportation. Culverts can create barriers and limit longitudinal movements within streams, especially for fishes with limited burst-speed swimming or jumping capabilities (e.g., lampreys). Roads along streams, especially unsurfaced roads in headwater areas (logging, recreational or other unimproved roads), often contribute to increased fine sediment or pollutant delivery to streams. Higher sediment loads are associated with degradation of spawning gravels and may contribute to excessive deposition in backwater or edgewater areas required for ammocoete rearing.

Logging. Timber harvest has been widespread and historically intensive throughout the range of western brook lamprey in California. Many areas have been logged multiple times, with resultant changes in forest vegetation composition, alteration to streams (e.g., geomorphology, annual hydrograph) and degradation of aquatic habitats (e.g., increased siltation, lack of canopy cover for shading and stream temperature moderation). Logging can reduce lamprey numbers after timber harvest occurs due to stream alteration (Moring and Lantz 1975), while extensive road networks created to facilitate logging continue to contribute sediments and increased surface run-off into streams.

Fire. Under predicted climate change scenarios, wildfires are expected to become more frequent and intense in many portions of the western brook lamprey's range, potentially leading to more extensive forest and aquatic habitat damage and longer recovery periods for these habitats. Fires can result in landslides that smother spawning gravels and removal of vegetation from riparian areas. Fire retardant reaching streams may cause localized areas of low dissolved oxygen, to which western brook lampreys are sensitive (Stone et al. 2002).

Alien species. Alien fishes (e.g., smallmouth bass) feed on ammocoetes and adults but the extent of impacts on lampreys from alien species predation and/or competition is not known. Alien fishes, however, are widespread throughout the western brook lamprey's range, so the potential for negative interactions is considerable.

Effects of Climate Change: The most noticeable and widespread impacts from climate change on lamprey habitats in California will be continued increases in water temperatures and changes to the frequency and timing of drought and flooding events. Water temperature increases may reduce the individual fitness of brook lampreys by decreasing growth, decreasing reproductive potential and increasing susceptibility to disease. The early life history stages (embryo to larva) are particularly sensitive to temperature increases. Both survival to hatch (~60%) and to the larval stage (~50%) significantly decreased at 22°C as compared to all other temperatures (10, 14 and 18°C; Bayer et al. 2001, Meeuwig et al. 2005). Survival to hatch and larva was about 90% from 10-18°C. Furthermore, physical deformities (e.g. deformed egg or yolk, fragmented yolk, bent or deformed prolarvae) occurred at all temperatures (<7%, Bayer et al. 2001) but was significantly higher at 22°C (~35%, Meeuwig et al. 2005). In general, most western brook lamprey populations are found in streams where temperatures are not likely to exceed 18°C during incubation or early rearing during spring months.

Elevated air temperatures associated with climate change will change the periodicity and magnitude of peak and base flows in streams, due to a reduction in snow pack levels and seasonal retention, particularly in watersheds at low elevations (< 3000 m) (Hayhoe et al. 2004). Predictions are that stream flow will increase in the winter and early spring and decrease in the fall and summer (Knox and Scheuring 1991, Field et al. 1999, CDWR 2006), perhaps changing the spawning ecology of fishes. If increased winter and spring flows make floodplain habitats accessible, western brook lamprey ammocoetes may benefit by rearing in highly productive habitats. Ammocoetes, however, can become stranded when flow decreases too quickly (Kostow 2002). If adults and ammocoetes spawn and rear in main channels, increased winter and spring flows may shift stream sediments to the detriment of nests and eggs. Because of their early life history stages' particular sensitivity to increased water temperatures, as well as their general immobility, Moyle et al. (2013) rated the species "highly vulnerable" to extinction within the next 100 years due to the added effects of climate change.

Status Determination Score = 3.0 - Moderate Concern (see Methods section Table 2). NatureServe lists western brook lamprey as globally secure (G4) but vulnerable in California (S3). In Oregon, they are considered a species "at risk." In 2003, a petition to list western brook lamprey in the Pacific Northwest and California under the Federal Endangered Species Act was received by the U.S. Fish and Wildlife Service (USFWS) (Nawa 2003). The petition cited habitat degradation and loss as major threats to the species. The USFWS determined the petition did not warrant further review based on insufficient scientific or commercial information (50 CFR Part 17). The high concern status in this report is driven by multiple interacting factors that have degraded many of the streams brook lampreys inhabit, combined with lack of information about their actual distribution or relative abundance within California (Table 2).

Metric	Score	Justification
Area occupied	5	Most historic watersheds are apparently still occupied
Estimated adult abundance	2	No population size information is available for California, but populations are assumed to be small
Intervention dependence	4	Persistence requires habitat improvements and stream protection
Tolerance	3	Moderately tolerant of warm temperatures; intolerant of low dissolved oxygen, pollution, low flows and disturbances to stream sediments
Genetic risk	2	Isolation and apparent small size of most populations increases vulnerability to genetic risks
Climate change	2	Populations are vulnerable to changes in natural flow regimes and increased temperatures
Anthropogenic threats	3	Multiple interacting threats exist (Table 1)
Average	3.0	21/7
Certainty (1-4)	2	Poorly known in California; better data available on populations in other states

Table 2. Metrics for determining the status of western brook lamprey, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: One of the greatest challenges to management of western brook lamprey is the lack of basic information on its status and biology in California; data are needed on distribution, abundance, genetics, environmental tolerances and population structure. In particular, research is needed to determine the status of isolated, distinctive populations such as those in Kelsey Creek and the Russian River; such forms may merit further taxonomic recognition (Moyle 2002, Boguski et al. 2102). Baseline surveys are needed to establish the relative abundance of this species within its range. Monitoring surveys (every 5 years) should be implemented in order to determine trends in distribution and abundance. Studies are also needed to establish the environmental tolerances of brook lampreys in California, especially to factors affected by land use and climate change, including temperature, turbidity, sedimentation, flows and water velocity.

Streams known to support brook lamprey populations, as well as those with the potential to do so, should be managed in ways that favor native fishes in general, including maintaining cool temperatures, spawning riffles and complex habitat structure using active management of water and land use practices or restoration actions, where necessary. For example, management of flow releases from hydroelectric projects should take into account the habitat requirements of native aquatic fauna, including western brook lamprey. Dam releases, in general, should mimic natural flow regimes in scale and periodicity. Grazing and logging activities should be buffered from riparian areas to protect riparian vegetation, limit nonpoint source pollution and minimize stream bank destabilization and excessive fine sediment inputs.

Figure 1. Assumed distribution of western brook lamprey, *Lampetra richardsoni*, in California. Actual distribution is largely unknown and distribution shown may include undescribed taxa.

PIT-KLAMATH BROOK LAMPREY
***Entosphenus lethophagus* Hubbs**

Status: Moderate Concern. While Pit-Klamath brook lamprey do not currently appear to be at risk of extinction, aquatic habitats within their range are heavily altered by agriculture and grazing and their actual abundance is unknown.

Description: Pit-Klamath brook lamprey are small and non-predatory (Hubbs 1971, Renaud 2011). Their oral disc resembles that of Pacific lamprey but have fewer and smaller teeth (plates). Lateral circumoral plates number 2-3-3-2 or 1-2-2-1, with cusps often missing. They have 9-15 posterior circumoral plates, often with just one cusp. The supraoral plate has 3 cusps, although the middle one may be smaller or absent. They usually have 5 infraoral teeth. Cusps on the transverse lingual lamina are difficult to see and are file-like. The small, puckered, mouth has a disc length less than 5 percent of body length. The disc is narrower than the head when stretched (Page and Burr 1991). Myomeres along the trunk number 60-70. Mature individuals exhibit gut atrophy. Coloration in adults is dark gray on the dorsum and brassy or bronze on the ventrum. See Renaud (2011) for a description of ammocoetes and comparisons with other lampreys in the Klamath region.

Taxonomic Relationships: Pit-Klamath brook lamprey were described from specimens collected from various locations in the Pit and Klamath basins by Hubbs (1971), as *Lampetra lethophaga*. This lamprey is closely related to Pacific lamprey (Docker et al. 1999, Lang et al. 2009). Recent phylogenetic analysis indicates that the species should be placed in the genus *Entosphenus,* and removed from the genus *Lampetra* (Lang et al. 2009). Analysis of characteristics of ammocoetes confirms this relationship (Goodman et al. 2009). Non-predatory lampreys in the two drainages may have been derived independently from Pacific lamprey and may ultimately be regarded as separate taxa (Kostow, 2002, Moyle 2002).

Life History: Spawning may begin in early spring and occur through summer (Moyle 2002). Fecundities may be similar to other lampreys with equivalent sizes at about 900 to 1,100 eggs per female (Kan 1975 in Kostow 2002). In some areas, adults may not develop nuptial features such as back and belly with dark, contrasting coloration; fused dorsal fins with frills; and enlarged anal fin (Moyle 2002). Larval lampreys (ammocoetes) usually burrow among aquatic vegetation into soft substrates (Moyle and Daniels 1982), where they likely feed on algae and detritus (Moyle 2002). Based on size classes, the ammocoete stage lasts for about four years, during which time they reach about 21 cm TL. Metamorphosis likely occurs in fall. Adults presumably only move short distances to spawning areas (Close et al. 2010). They commonly co-occur with trout, marbled and rough sculpins, and speckled dace (Moyle 2002).

Habitat Requirements: Pit-Klamath brook lampreys principally occupy habitats in clear, cool (summer temperatures < 25ºC) rivers and streams in areas with fine substrates and beds of aquatic plants (Moyle and Daniels 1982, Moyle 2002). Like other lampreys, Pit-Klamath brook lampreys require gravel riffles in streams for spawning, with muddy backwater habitats downstream of spawning areas for ammocoete burrows. In the Pit River system, they seem especially common in backwaters of the spring-fed Fall River and Hat Creek (Moyle and Daniels 1982). Pit-Klamath brook lamprey in the Oregon portion of the Goose Lake basin are most commonly found in high-elevation streams in forested lands (Scheerer et al. 2010).

Distribution: Pit-Klamath brook lampreys, as currently defined, are only found in the Pit River-Goose Lake basin in California and Oregon as well as in the upper Klamath basin, upstream of Klamath lakes in Oregon (Hubbs 1971, Moyle and Daniels 1982). If this species is broken into two entities, then only *E. lethophagus* occurs in California, where it is widely distributed throughout the Pit River basin and, presumably, the Goose Lake basin in both California and Oregon (Moyle and Daniels 1982, Kostow 2002, Moyle 2002).

Trends in Abundance: Abundance and population trend information are lacking. Their populations do not seem to be in danger of extinction at this time but face multiple threats (discussed below).

Nature and Degree of Threats: Pit-Klamath brook lamprey face degradation of suitable habitats by multiple factors affecting streams in this arid region. The main stem Pit River and some of its tributaries are currently listed as impaired due to high temperatures and nutrient loading, as well as low dissolved oxygen levels (Pit RCD 2006, DEQ 2010).

Major dams. The lower Pit River supports a chain of hydropower reservoirs and some tributaries also have small dams on them. The effects of these dams on lampreys are unknown but some habitats have been inundated and populations may be fragmented as a consequence.

Agriculture. Water demands for agriculture are high along the Pit and upper Klamath rivers, resulting in decreased instream flows. Water diversions in some areas may be reducing instream flows to the extent that certain reaches go dry (Pit RCD 2006). Flood-irrigated pastures introduce nutrients into streams and raise water temperatures, via return water, and fertilizers are thought to be increasing nutrient loadings in streams (Pit RCD 2006). Pit-Klamath brook lamprey may be well adapted for some altered habitats, especially in the larval stage. Ammocoetes were common in the mud substrates of an irrigation diversion from Rush Creek, Modoc County (Moyle 2002). They are also common in silt substrates of pools below channelized sections of streams.

Grazing. Extensive grazing occurs throughout the range of Pit-Klamath brook lamprey. Grazing can degrade aquatic habitats through streambank trampling, removal of riparian vegetation, or input of nutrients and other pollutants from animal wastes. Fecal matter is thought to be increasing the nutrient loading of streams in this region (Pit RCD 2006). Removal of vegetation increases erosion and entrenchment of stream channels (Pit RCD 2006) and contributes to increased solar input and corresponding water temperature increases in streams.

Rural residential. Several towns exist within the Pit-Klamath brook lamprey range (e.g. Alturas) in California. Residential areas can be sources of pollutants and increased water demands that may decrease water quantity and quality in streams.

Alien species. Many alien fish species inhabit the Klamath and Pit River basins (Close et al. 2010, Moyle and Daniels 1982). Species that can prey on lamprey include largemouth bass, brown bullhead, channel catfish, brook trout, brown trout, black crappie, and yellow perch (Close et al. 2010).

	Rating	Explanation
Major dams	Low	Dams present in range but impacts are unknown
Agriculture	Medium	Agriculture pervasive throughout range; direct effects unknown but likely contributes to substantial diversion and water quality degradation; effects may be severe at a localized level
Grazing	Medium	Grazing pervasive throughout range; direct effects unknown but likely contributes to aquatic and riparian habitat degradation, along with water quality impairment across much of range
Rural residential	Low	Small towns and residences common but widely dispersed within range; impacts likely minimal except for water withdrawls and potential pollutant inputs at a localized scale
Urbanization	n/a	
Instream mining	n/a	
Mining	n/a	
Transportation	Medium	Extensive network of unimproved roads across range; potential for increased sediment inputs and habitat fragmentation
Logging	Low	Logging occurs in forested lands; impacts unknown
Fire	Low	Wildfires occur throughout range; impacts unknown
Estuary alteration	n/a	
Recreation	n/a	
Harvest	n/a	
Hatcheries	n/a	
Alien species	Medium	Absent where alien species abundant

Table 1. Major anthropogenic factors limiting, or potentially limiting, viability of populations of Pit-Klamath brook lamprey in California. Factors were rated on a five-level ordinal scale where a factor rated “critical” could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated “high” could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated “medium” is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated “low” may reduce populations but extinction unlikely as a result; and a factor rated “no” has no known negative impact to the taxon under consideration. Certainty of these judgments is low. See methods section for descriptions of the factors and explanation of the rating protocol.

Effects of Climate Change: Climate change is expected to increase the frequency of both drought and floods in streams. Because Pit-Klamath lamprey rear for several years in stream substrates, large flooding events may disrupt rearing habitats (Fahey 2006) and displace ammocoetes from soft sediments. On the contrary, scouring events may clean sediments from gravels that would otherwise degrade spawning habitats (Stuart 2006 in Fahey 2006). This species may not be as vulnerable as other fishes to stream flow changes associated with climate change because a few populations occur in large, spring-fed river systems (e.g. Fall River). Changes to the natural hydrograph will likely be attenuated in streams that are spring-fed, as in the upper Klamath basin at the northern end of the Pit-Klamath brook lamprey range (Quiñones 2011). Pit-Klamath brook lamprey can tolerate high turbidities and persist in seasonally intermittent streams (S. Reid, in Close et al. 2010). They also appear tolerant of higher water temperatures, which are expected to increase due to climate change. Pit-Klamath brook lamprey can tolerate summer water temperatures >25ºC in the Pit River (S. Reid, in Close et al. 2010). Moyle et al. (2013) listed the Pit-Klamath brook lamprey as "highly vulnerable" to extinction as the result of climate change by 2100; however, little is understood both about the biology of this lamprey and the potential effects of climate change on aquatic systems in the arid Pit River basin, so this rating was applied with a low degree of certainty.

Status Determination Score = 3.7 - Moderate Concern (see Methods section, Table 2). Pit-Klamath brook lamprey appear to be common throughout their range in California. However, their actual abundance is unknown. Pit-Klamath brook lamprey are subject to multiple stressors (Table 1) that can create adverse habitat conditions. NatureServe classifies Pit-Klamath brook lamprey as secure to vulnerable throughout their range.

Metric	Score	Justification
Area occupied	5	Range limited to Pit River drainage in California, but includes several tributary systems (e.g. Fall River)
Estimated adult abundance	3	Species is thought to be abundant within range but actual numbers are unknown
Intervention dependence	4	Long-term management of agriculture and grazing practices, as well as alien species, may be warranted
Tolerance	3	Pit-Klamath brook lamprey apparently tolerate warmer temperatures than other lamprey species but still require cool, clean water
Genetic risk	5	Thought to be genetically diverse, although populations in Goose Lake and Klamath basin may constitute separate species
Climate change	2	Some habitats may dry more extensively or for longer durations; ammocoetes may be displaced by unusually high flows
Anthropogenic threats	4	See Table 1
Average	3.7	26/7
Certainty (1-4)	1	Species is largely unstudied

Table 2. Metrics for determining the status of Pit-Klamath brook lamprey in California, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: Habitat degradation from agricultural and grazing practices poses the greatest threat to Pit-Klamath brook lamprey, effects likely to be exacerbated by increasing temperatures and more frequent flood events predicted by climate change models. Watershed management strategies exist (e.g., Pit RCD 2006, Klamath Basin Restoration Agreement) that address these and other factors that may limit fish populations in the Pit and upper Klamath basins. Beyond implementation of these strategies, basic life history studies and population monitoring should occur in order to better understand the status of this species. The following questions should be addressed as part of a status evaluation:

1) Are brook lampreys in the Pit River-Goose Lake and Klamath basins separate taxa?
2) What is the current distribution and abundance of Pit-Klamath brook lamprey in California?
3) Where are most important spawning and rearing grounds located in California?
4) What are the optimal and preferred environmental tolerances and habitat conditions for each life history stage?

Figure 1. Distribution of Pit-Klamath brook lamprey, *Entosphenus lethophagus,* in California.

NORTHERN GREEN STURGEON
***Acipenser medirostris* (Ayres)**

Status: High Concern. Very little is known about the current size of the single northern green sturgeon population in California. However, habitat degradation and climate change continue to threaten their status.

Description: Sturgeons, with their large size, subterminal barbeled mouths, lines of bony plates (scutes), and heterocercal (shark-like) tail, are among the most distinctive of freshwater fishes. Green sturgeon have 8-11 scutes in the dorsal row, 23-30 in the lateral rows, and 7-10 in the bottom rows. The dorsal fin has 33-36 rays, and the anal fin 22-28. They are distinguished from white sturgeon, with which they co-occur, by: (1) having one large scute behind the dorsal and anal fins, (2) having scutes that are sharp and pointed, and (3) having barbels that are closer to the mouth than to the tip of the long, narrow snout (Moyle 2002). Their color is olive-green to pale brown, with an olivaceous stripe on each side and scutes that are paler than the body.

Taxonomic Relationships: Green sturgeon were described from San Francisco Bay in 1854 by W. O. Ayres as *Acipenser medirostris*, the only one of three species he described from the Bay that is still recognized. Green sturgeon are tetraploids and have lower fecundity and larger eggs than most other sturgeon (Gessner et al. 2007). The zoogeographic origin of green sturgeon is uncertain; evidence can be mounted for either an Asian or North American ancestry (Artyukhin et al. 2007). The closest relative is the Asian green sturgeon, *Acipenser mikadoi*, described from one poorly preserved specimen (Jordan and Snyder 1906). Schmidt (1950) designated the Asian form (the Sakhalin sturgeon in the Russian literature) as a distinct subspecies, *Acipenser medirostris mikadoi*. DNA measurements show that the Asian form has approximately twice the DNA content of the North American form (Birstein 1993), indicating that *A. mikadoi* is distinct from *A. medirostris*. Recent comparisons found considerable differences in the morphometrics (e.g., snout length measurements) of Asian and North American populations, although meristic counts overlapped one another (North et al. 2002). Birstein (1993) also suggested that there may be considerable genetic difference between California populations of *A. medirostris* and those north of California. Subsequent analysis of North American green sturgeon found genotypic differences between individuals in the Rogue and Klamath rivers from those in the Sacramento River (Israel et al. 2004). This has led to the split of green sturgeon into two Distinct Population Segments (DPS): southern (Sacramento) green sturgeon DPS and northern green sturgeon DPS (Adams et al. 2002, Adams et al. 2007). The National Marine Fisheries Service has designated populations from the Rogue (Oregon), Klamath-Trinity, Eel, and Umpqua (Oregon) rivers as constituting the northern DPS (Adams et al. 2002, Adams et al. 2007). The population in the Sacramento River has been designated as the southern DPS. In this report, the northern DPS of the green sturgeon is referred to as northern green sturgeon.

Life History: The recent recognition of green sturgeon as having two distinct populations (northern and southern DPS) is confounded by the fact that individuals from both populations likely interact in the ocean; therefore, most studies of ecology and behavior do not separate the two forms outside their native rivers. Until the listing of the southern green sturgeon DPS in 2006, the ecology and life history of green sturgeon had received little study because of their generally low abundance and their low commercial and sport-fishing value. Adults are more marine than white sturgeon but can spend up to six months in fresh water (Benson et al. 2007, Erickson et al. 2002).

Spawning populations of northern green sturgeon are confirmed only for the Rogue (Oregon) and Klamath rivers. Green sturgeon migrate up the Klamath River between late February and late July. The spawning period is March-July, with a peak from mid-April to mid-June (Emmett et al. 1991, Van Eenennaam et al. 2006, Benson et al. 2007). Although the spawning period is similar in the Rogue River, post-spawn adults are found in fresh water in both spring and fall (Webb and Erickson 2007). Spawning females are generally larger, heavier, older and in better condition than spawning males (Van Eenennaam et al. 2006, Benson et al. 2007, Erickson and Webb 2007). From 1999 to 2003, the length of spawning females in the Klamath River was 151-223 cm FL, while males measured 139-199 cm FL. In the Rogue River, male and female green sturgeon become sexually mature at 145 cm TL and 166 cm TL, respectively (Erickson and Webb 2007). Most females were 19-34 years old, while males were 15-28 years old. Males are slightly more abundant than females in spawning runs (female:male = 1:1.4). Adults in the Klamath River exhibit four distinct migration patterns characterized by varying lengths of freshwater residency of up to 199 days (Benson et al. 2007). Individuals migrate at rates of 1.18 to 2.15 km per day. Adults do not appear to spawn in successive years but, rather, at intervals of two or more year (Erickson and Webb 2007, Webb and Erickson 2007).

According to Moyle (2002, p. 110): "Spawning takes place in deep, fast water. In the Klamath River, a pool known as The Sturgeon Hole (Humboldt County) apparently is a major spawning site, because leaping and other behavior indicative of courtship and spawning are often observed there during spring and early summer." Female green sturgeon produce 51,000-224,000 eggs (Adams et al. 2002) which have an average diameter of 4.3 mm (Van Eenennaam et al. 2006). Based on their similarity to white sturgeon, green sturgeon eggs probably hatch around 196 hours (at 13°C) after spawning and the larvae should be 8-19 mm long (Gisbert and Doroshov 2006); juveniles likely range in size from 2.0 to 150 cm TL (Emmett et al. 1991). Morphological (large pectoral fins) and behavioral (rostral wedging) traits allow smaller green sturgeon to hold in rivers for extended periods of time (Allen et al. 2006). Juvenile green sturgeon appear to be largely nocturnal in their migratory, feeding and rearing behavior during the first 10 months of life (Kynard et al. 2005). Green sturgeon retinas are dominated by rods, supporting the idea that they are adapted to live in dim environments (Sillman et al. 2005).

Most juveniles migrate out to sea before two years of age, primarily during summer through fall (Emmett et al. 1991, Allen et al. 2009). Length-frequency analyses of northern green sturgeon caught in the Klamath Estuary by beach seine indicate that most green sturgeon leave the system at lengths of 30-60 cm, when they are 1 to 4 years old, although the majority apparently leave as yearlings (USFWS 1982). Although juvenile green sturgeon can withstand brackish (10 ppt) water at any age, their ability to osmoregulate in salt water develops around 1.5 years of age (Allen and Cech 2007). In the ocean, adults make annual migrations northward in the fall and southward in the spring (Lindley et al. 2008). Important overwintering habitats have been identified between Cape Spencer, Alaska and Vancouver Island. Adults can migrate more than 50 km per day during return spring migrations. Individuals from all spawning populations are known to congregate at Willapa Bay, Washington in the summer (Moser and Lindley 2007).

Northern green sturgeon grow approximately 7 cm per year until they reach maturity at 130-140 cm TL, around age 15-20 years. Thereafter, growth slows. The maximum size is presumed to be around 230 cm TL (USFWS 1982). The oldest fish known are 42 years, based on annuli of fin rays, but the largest fish are probably much older (T. Kisanuki, pers. comm., 1995). Juveniles and adults are benthic feeders on both invertebrates and fish. Adult sturgeon caught in Washington feed mainly on sand lances (*Ammodytes hexapterus*) and callianassid shrimp (P. Foley, pers. comm., 1992). In the Columbia River estuary, green sturgeon are known to feed on anchovies and, perhaps, on clams (C. Tracy, minutes to USFWS meeting). Adults may optimize growth in the summer by feeding on burrowing shrimp in the relatively warmer waters of Washington estuaries (Moser and Lindley 2007).

Habitat Requirements: The habitat requirements of northern green sturgeon are not well studied, but spawning and larval ecology are probably similar to that of white sturgeon. Preferred spawning substrate is likely large cobble, but can range from clean sand to bedrock (Nguyen and Crocker 2007). Eggs are broadcast-spawned and externally fertilized in relatively fast water at depths >3 m (Emmett et al. 1991). Excessive silt can prevent embryos from adhering to one another (Gisbert et al. 2001). Sand can impair the growth and survival of larval green sturgeon by decreasing feeding effectiveness (Nguyen and Crocker 2007).

Temperature appears to be closely linked to migration timing. In the Rogue River, adults enter freshwater from March through May, when water temperatures range from 9 to 16 °C (Erickson and Webb 2007). Adults may hold in deep (>5 m) pools with low velocities after spawning for up to six months (Erickson et al. 2002, Benson et al. 2007). Adult river outmigration initiates with low river temperatures (< 12 °C) and increases in flow (>100 cms). Juveniles appear to prefer dark, deep pools with large rock substrate during winter rearing (Kynard et al. 2005). Nocturnal downstream migration by juveniles continues until water temperatures decrease to about 8°C (Kynard et al. 2005).

Temperature has a major influence on green sturgeon physiology and survival. The upper thermal limit for developing embryos is 17- 18 °C (Van Eenennaam et al. 2005). Incubation temperatures above 22°C result in deformities (Mayfield and Cech 2004, Werner et al. 2007) and/or mortality (Van Eenennaam et al. 2005) of developing embryos. Although age 1 to 3 year old green sturgeon appear to tolerate moderate changes in water temperatures (Kaufman et al. 2007), optimal temperatures for age 1 juvenile sturgeon range from 11 to 19°C. In this same age group, temperatures between 19 and 24°C increase metabolic costs, while temperatures above 24 °C cause severe stress (Mayfield and Cech 2004). However, the metabolic costs associated with temperatures between 19 and 24 °C may be offset when food and oxygen are abundantly available, resulting in unimpaired growth (Allen et al. 2006). Kaufman et al. (2006) determined that juvenile green sturgeon are limited in their ability to handle increases in CO_2. Time of day, length of exposure to a given stressor, and temperature affect the ability of green sturgeon juveniles to respond to stress (Lankford et al. 2003, Werner et al. 2007).

Distribution: Green sturgeon have been caught in the Pacific Ocean from the Bering Sea to Ensenada, Mexico, a range which includes the entire coast of California. However, except for a few tagged fish, it is not known from which river(s), or DPS, ocean-caught sturgeon originate. Migrations generally follow northern routes along shallow waters within the 110 m contour, with individuals from all populations congregating in Willapa Bay, Washington (Moser and Lindley 2007). There are records of green sturgeon from rivers in British Columbia south to the Sacramento River. There is no evidence of green sturgeon spawning in Canada or Alaska, although small numbers have been caught in the Fraser, Nass, Stikine, Skeena and Taku rivers, British Columbia (COSEWIC 2004). Green sturgeon are common in the Columbia River estuary and were observed as far as 225 km inland in the Columbia River, prior to the construction of

Bonneville Dam (Wydoski and Whitney 1979). They apparently do not spawn in the Columbia River or other rivers in Washington, although Israel (2004) discussed genetic evidence for a distinct Columbia River population. In Oregon, juvenile green sturgeon have been found in several coastal rivers (Emmett et al. 1991) but spawning is confirmed only in the Rogue River (Erickson et al. 2002, Erickson and Webb 2007). For northern green sturgeon, spawning has been confirmed in recent years only in the Klamath and Rogue rivers (Moyle 2002, Adams et al. 2007). However, repeated observations of small numbers of adult and juvenile green sturgeon in the Eel River since 2002 suggest spawning may have resumed there after decades of spawning absence (Higgins 2013). There is some evidence of occasional spawning in the Umpqua River (Farr and Kern 2005). Overall, it is likely that northern green sturgeon once spawned in the larger coastal rivers from the Eel River in California north to the Columbia River in Oregon/Washington. Today, the Klamath River is presumed to be the principal spawning river, based on size, flow/temperature regime, and habitat availability.

The following distributional information on northern green sturgeon in California waters was compiled by Patrick Foley (University of California, Davis 1992) and updated with information in Adams et al. (2007).

North Coast. From the Eel River northward, it is likely that most records of sturgeon caught in rivers and estuaries refer to northern green sturgeon. However, most early references regarding sturgeon from the north coast did not identify the species and some reports indicated white sturgeon to be more abundant (Fry 1979). While white sturgeon do occur on occasion in the Klamath and other rivers, it is highly likely that most historic records are for northern green sturgeon. Nineteenth century newspapers (The Humboldt Times) report sturgeon from the mainstem Eel River, South Fork Eel River and Van Duzen River (Wainwright 1965). Length and weights given in these newspaper accounts are most consistent with those of adult green sturgeon.

In the 1950s, two young northern green sturgeon were collected in the mainstem Eel River and large sturgeon were observed jumping in tidewater (Murphy and DeWitt 1951). Two additional young green sturgeon (101 mm and 123 mm) were taken by CDFW from the Eel River in 1967 and are now in the fish collection at Humboldt State University. Substantial numbers of juveniles were caught by CDFW in the mainstem Eel River during trapping operations from 1967-1970 (O'Brien et al. 1976): 22 at Eel Rock in 1967, 53 at McCann in 1967 and 161 in 1969, 221 at Fort Seward in 1968, and smaller numbers at other localities. Green sturgeon have been included in lists of natural resources found in the Eel River delta (Monroe and Reynolds 1974, Blunt 1980). Adult green sturgeon are still occasionally seen in the Eel River (Adams et al. 2007). Higgins (2013) compiled seven records of green sturgeon, usually in groups, observed in the Eel River since 2002 and suggested they are now spawning in the river again. Adams et al. (2007) list the Eel River as a site of "suspected spawning."

Records of sturgeon in the Humboldt Bay system, comprising Arcata Bay to the north and Humboldt Bay to the south, are almost exclusively green sturgeon. Ten years of trawl investigations in south Humboldt Bay produced three green sturgeon (Samuelson 1973). Records from Arcata Bay are more numerous. On August 6 and 7, 1956, 50 green sturgeon were tagged in Arcata Bay by CDFW biologist Ed Best (D. Kohlhorst, pers. comm.). Total length ranged from 57.2 cm to 148.6 cm with a mean TL of 87.0 cm (± 20.6 cm SD). In 1974, nine green sturgeon were collected over a two-month period in Arcata Bay (Sopher 1974). Total length of these fish ranged between 73-112 cm TL. The Coast Oyster Company, Eureka, pulls an annual series of trawls in Arcata Bay in order to decrease the abundance of bat rays, *Myliobatis californica*. Green sturgeon are incidentally taken in this operation. Eight green sturgeon collected for parasite evaluation in 1988 and 1989 had total lengths ranging between 78-114 cm. One large individual, 178 cm TL and 18.2 kg, was returned to the bay. In 2007, green sturgeon tagged with acoustic tags were detected moving in and out of Humboldt Bay by an array set up to study the movements of coho salmon (S. Lindley, USFWS, unpublished report). Both northern and southern green sturgeon use Humboldt Bay during spring and fall (S. Lindley, pers. comm. 2009) as summarized in Tables 1-3.

Northern green sturgeon have been reported from the Mad River (Fry 1979), but evidence of their recent presence is scant (Bruce Barngrover, pers. comm. 1992). One adult was trapped in the lower river near Mad River Hatchery and rescued by CDFW biologists in 2005 (M. Gilroy, pers. comm. 2011). A carcass was also found in July, 2010 (T. Moore, file report, CDFG, 2010). California Department of Fish and Wildlife biologists D. McLeod and L. Preston observed a 1+ m long sturgeon, most likely a green sturgeon, in a gravel extraction trench in the mainstem Mad upstream of the Blue Lake Bridge (river mile 16) on May 20, 1992.

An occasional green sturgeon is encountered in the coastal lagoons of Humboldt County (Terry Roelofs, pers. comm. 1992). Big Lagoon and Stone Lagoon are connected to the ocean during part of the year and migrating sturgeon may gain entry at this time. In June, 1991, a 120-cm TL green sturgeon was gillnetted in Stone Lagoon (Terry Roelofs, pers. comm. 1992).

Green Sturgeon Tag Code	*Tagging Origin*	*First Detection*	*Last Detection*	***Number of Detections***
0111	Rogue River	July	July	20
0907	**San Pablo Bay**	**June**	**August**	**1,391**
0918	**San Pablo Bay**	**September**	**October**	**5,995**
0933	San Pablo Bay	September	September	5
0989	**San Pablo Bay**	**June**	**September**	**6,660**
1004	**San Pablo Bay**	**September**	**September**	**4**
1008	**San Pablo Bay**	**September**	**September**	**15**
1072	Rogue River	August 6	October	10,218
1127	Willapa Bay	August	August	22
1138	**Willapa Bay**	**June**	**October**	**3,401**
1187	Grays Harbor	June	July	45

Table 1. Green sturgeon detections in 2006, Humboldt Bay, California, recorded on acoustic receiver network maintained by Arcata Fish and Wildlife Office, U.S. Fish and Wildlife Service. Tag codes in bold were detected both in 2006 and 2007. (Provided by W. Pinnix, USFWS, 2012). No fish were tagged in Humboldt Bay.

Green Sturgeon Tag Code	*Tagging Origin*	*First Detection*	*Last Detection*	***Number of Detections***
0151	Sacramento River	July	August	196
0182	Sacramento River	July	August	29,327
0223	Sacramento River	May	July	15,467
0897	San Pablo Bay	July	August	624
0903	San Pablo Bay	July	July	3
0906	San Pablo Bay	July	September	1,186
0907	**San Pablo Bay**	**May**	**August**	**9,033**
0918	**San Pablo Bay**	**July**	**September**	**19,077**
0982	San Pablo Bay	July	July	83
0989	**San Pablo Bay**	**April**	**July**	**625**
0990	San Pablo Bay	July	October	15,019
0995	San Pablo Bay	September	September	39
1004	**San Pablo Bay**	**July**	**July**	**3**
1008	**San Pablo Bay**	**July**	**July**	**73**
1138	**Willapa Bay**	**May**	**September**	**16,938**
1144	Willapa Bay	July	July	344
1147	Willapa Bay	July	July	3
1173	Grays Harbor	May	May	384
1180	Grays Harbor	June	June	241
1182	Grays Harbor	June	June	275
2203	San Pablo Bay	May	August	128
2216	San Pablo Bay	August	August	17
2220	San Pablo Bay	April	July	135
2222	San Pablo Bay	July	October	5,874
2225	San Pablo Bay	September	September	15

Table 2. Green sturgeon detections in 2007, Humboldt Bay, California, recorded on acoustic receiver network maintained by Arcata Fish and Wildlife Office, U.S. Fish and Wildlife Service. Tag codes in bold were detected both in 2006 and 2007. (Provided by W. Pinnix, USFWS, 2012). No fish were tagged in Humboldt Bay.

Green Sturgeon Tag Code	*Tagging Origin*	*First Detection*	*Last Detection*	***Number of Detections***
0219	Sacramento River	June	August	793
0223	**Sacramento River**	**September**	**September**	**12,302**
0238	Sacramento River	September	September	1
0438	Sac???	September	September	3
0906	**San Pablo Bay**	**June**	**June**	**1,637**
0907	**San Pablo Bay**	**May**	**August**	**7,415**
0913	San Pablo Bay	June	September	16,705
0918	**San Pablo Bay**	**September**	**September**	**2,971**
0979	San Pablo Bay	September	September	3
0984	San Pablo Bay	July	July	24
0985	San Pablo Bay	August	August	88
0989	**San Pablo Bay**	**March**	**March**	**3**
0990	**San Pablo Bay**	**August**	**September**	**9,763**
1005	San Pablo Bay	August	August	1
1138	**Willapa Bay**	**June**	**September**	**6,827**
1144	**Willapa Bay**	**August**	**August**	**165**
1153	Willapa??	July	July	1
2203	**San Pablo Bay**	**May**	**May**	**3**
2210	San Pablo Bay	August	August	174
2212	San Pablo Bay	August	September	425
2217	San Pablo Bay	June	August	415
2225	San Pablo Bay	September	September	15

Table 3. Green sturgeon detections in 2008, Humboldt Bay, California, recorded on acoustic receiver network maintained by Arcata Fish and Wildlife Office, U.S. Fish and Wildlife Service. Tag codes in bold were detected both in 2007 and 2008. (Provided by W. Pinnix, USFWS, 2012). No fish were tagged in Humboldt Bay.

Klamath and Trinity rivers. The largest spawning population of northern green sturgeon in California is in the Klamath River basin. Both green sturgeon and white sturgeon have been found in the Klamath River estuary (Snyder 1908b, USFWS 1980-91), but white sturgeon are taken infrequently in very low numbers and are presumed to be coastal migrants (USFWS 1982). Almost all sturgeon found above the estuary during systematic sampling have been green sturgeon (USFWS 1980-83). Green sturgeon primarily use the mainstem Klamath River and mainstem Trinity River but have also been seen in the lower portions of the Salmon River (Adams et al. 2007).

Both adult and juvenile northern green sturgeon have been identified in the mainstem Klamath River. Adults are taken annually from spring through summer by an in-river tribal gillnet fishery. The numbers taken are between 200 and 750 fish per year (Table 5). They have also been taken by sport fishermen as far inland as Happy Camp (river km 172; unpubl. CDFW tagging data 1969-73, Fry 1979, USFWS 1981). The apparent upstream limit for spawning migration is Ishi Pishi Falls, Siskiyou County, at approximately river km 113. A few juveniles have been taken as high up as Big Bar at river km 81 (Tom Kisanuki, pers. comm. 1995) but most have been recovered by seining operations directed at salmonids in the estuary (USFWS, CDFW). Sampling by the USFWS captured 7 juveniles in 1991 and 23 in 1992 (T. Kisanuki, pers. comm. 1995). Six outmigrant traps placed in the Klamath River caught juvenile green sturgeon every year (2000-2005) (Cunanan and Hines 2006, USFWS, unpublished data). The number of green sturgeon captured each year varied from one (2005) to 775 (2003). The total number of juvenile green sturgeon captured over the six years of operation was 1599, with sizes varying from 20 mm to 252 mm TL and averaging 68.5 mm TL. Green sturgeon captured by the traps were most likely juveniles ranging in age from a couple of weeks to less than two years old, based on growth curves developed by Nakamato et al. (1995) and Van Eenennamm et al. (2001). The average size (69 mm TL) was similar to the size of artificially reared Klamath River green sturgeon at 35 days old (66 mm; Van Eenennaam et al. 2001).

The Trinity River enters the Klamath River at Weitchpec (river km 70). The first green sturgeon described from the Klamath basin came from the Trinity River (Gilbert 1897). Both adults and juveniles have been identified; 211 green sturgeon, between 7-29 cm TL, were captured in screw traps near Willow Creek, Humboldt County, incidental to a salmonid migration study in July-September, 1968 (Healey 1970). The USFWS has collected small numbers of juvenile green sturgeon from the Trinity River, as far up as Big Bar (T. Kisanuki, pers. comm. 1992). Adults are caught yearly in a tribal gillnet fishery (USFWS 1980), a traditional fishery with a long history (Kroeber and Barrett 1960). Spawning adults migrate the mainstem Trinity River up to about Grays Falls, Burnt Ranch, Trinity County (river km 72).

Northern green sturgeon have also been reported to use the South Fork Trinity River, a third-order stream entering above Willow Creek (river km 51) (USFWS 1981), according to oral histories from long-time residents. However, a large flood in 1964 had devastating effects on anadromous fish habitat in this subbasin (U.S. Department of the Interior 1985). Millions of cubic yards of soil were moved into South Fork Trinity River and its tributaries, with resulting channel widening and loss of depth in many areas. This event, along with other changes in basin morphology, has apparently resulted in the loss of suitable sturgeon habitat. There are no recent records of green sturgeon from this watershed.

The Salmon River is a fourth-order stream entering the Klamath River at Somes Bar (river km 106). Adult green sturgeon have been observed upstream as far as the mouth of Wooley Creek (river km 8).

<u>Del Norte County</u>. Northern green sturgeon have been taken during gillnet sampling in Lake Earl (D. McLeod, pers. comm.). Lake Earl is located along the coast of Del Norte County, 8 km north of Crescent City and 11 km south of the mouth of Smith River. Lake Earl is connected to Lake Talawa, a smaller lake directly to the west. A sand spit separates Lake Talawa from the ocean and is occasionally breached by winter storms or mechanically per the Lake Earl Wildlife Area Management Plan. Coastal migrant green sturgeon may enter at this time and become trapped after the sand spit is reestablished (Monroe et al. 1975).

The Smith River is the northernmost river along the California coast, entering the ocean approximately 5 km south of the Oregon border. Blunt (1980) included green sturgeon in an inventory of anadromous species found in the Smith River. They occasionally enter the estuary and have been observed in Patrick's Creek, an upstream tributary 53 km from the ocean (Monroe et al. 1975). Juveniles have not been found in the Smith drainage.

Trends in Abundance. Although northern green sturgeon apparently occur in fewer streams than they did historically, trends in abundance are poorly understood (Adams et al. 2002). The only time series data available for adult green sturgeon abundance in the Klamath River comes from tribal catch data (see below). The number of females spawning in the Klamath River is estimated at 760-1500 per year. The population of subadults-adults is estimated at tens of

thousands, with no clear evidence of population decline (Adams et al. 2002).
However, northern green sturgeon abundance and population trends remain largely unknown and should be treated conservatively until information indicates otherwise because:

(1) Virtually all other sturgeon species are in decline. Rochard et al. (1990) state in their review of the status of sturgeons worldwide: "Those [species of sturgeon] which do not have particular interest to fishermen (*A. medirostris, Pseudoscaphirhynchus* spp.) are paradoxically most at risk, for we know so little about them" (p. 131). The southern green sturgeon is listed as a threatened species.

(2) The only confirmed spawning populations of northern green sturgeon are in the Klamath and Rogue (Oregon) rivers, both of which have flow and temperature regimes affected by water projects and, potentially, climate change. It is highly probable that these are now the only spawning populations in North America, although recent reports from the Eel River are promising.

(3) Green sturgeon are subject to legal, illegal, and by-catch fisheries. It is likely that these fisheries depend largely on sturgeon from the Klamath River. The various fisheries, including past sport fishing, have harvested at least 6,000 to 11,000 green sturgeon per year. Studies have shown that green sturgeon populations are sensitive to overharvest (Heppell 2007).

Nature and Degree of Threats: Green sturgeon depend on large rivers so their populations are subject to numerous anthropogenic stressors that occur across large geographic areas, as described below (see Table 4).

Major dams. The Klamath, Trinity and Rogue (Oregon) rivers all have flows regulated by major dams. Apparently, the impact of these dams upon green sturgeon has been minimal perhaps because spawners tend to be in the river when flows are highest and because all life stages mainly live in the lowermost reaches, where dam impacts are reduced. However, a single green sturgeon was part of a large fish kill in the lower Klamath River in September, 2002, which has been attributed partially to the operation of Iron Gate Dam (Belchick et al. 2004), suggesting at least some vulnerability.

Grazing, roads, logging. Land use practices, such as road building, logging and grazing have all changed the quality of spawning and rearing habitats in large mainstem rivers by increasing sediment loads, impairing water quality and otherwise reducing habitat suitability. Thus, it is likely that optimal conditions (especially temperature, flow, and stream substrate composition) for spawning and rearing of green sturgeon occur less frequently now than they once (pre-1940s) did, especially during or after periods of extended drought. Of particular concern is siltation of river portions used for spawning and incubation of embryos, although the timing and location of spawning tends to reduce the probability that this is a factor in survival. The huge 1964 floods may have severely degraded many areas of sturgeon spawning and rearing habitat, perhaps eliminating this species from rivers, or tributaries thereof, such as the Eel and South Fork Trinity.

Estuary alteration. While the Klamath River estuary is relatively unmodified, other California estuaries such as those of the Eel and Smith rivers have been diked and drained for pasture or other land uses. This degradation of key rearing areas may have contributed to reductions or loss of green sturgeon and other anadromous fishes from these rivers (Yoshiyama and Moyle 2010).

Harvest. Although California anglers were prohibited from taking or possessing green sturgeon beginning in 2007, the legacy of past fishing practices may still be impacting populations today due to the species' longevity and infrequency of spawning. Of particular concern is removal of adult females from the population, which have the highest fecundity and, therefore, the greatest potential for replenishing depleted populations. The following are accounts of the two principal fisheries that may have affected green sturgeon in the northern DPS:

	Rating	Explanation
Major dams	Medium	Major dams present on all spawning rivers; however, effects are largely unknown
Agriculture	Low	Minor influence on lower Klamath and Eel rivers; alfalfa pastures for grazing widespread in the Smith estuary
Grazing	Low	Pervasive in watersheds but probably little effect on large river habitats
Rural Residential	Low	Pervasive in watersheds but probably little effect on large river habitats
Urbanization	Low	No large urban areas within known distribution
Instream mining	Low	Gravel mining and gold dredging may increase fine sediment mobilization in rivers; greater historic impact
Mining	Low	No known impact but some dredging in range (currently suspended in California)
Transportation	Medium	Roads are a source of sediment that may affect spawning
Logging	Medium	Major source of sediment from extensive network of access roads; greater historic impact
Fire	Low	Wildfires are common within the range of northern green sturgeon but impacts are not well understood
Estuary alteration	Medium	Smith and Eel estuaries are altered and have reduced capacity for rearing juvenile sturgeon
Recreation	Low	No known impact but boating may disturb fish
Harvest	Medium	Adults taken in fisheries for many years but impacts not well understood
Hatcheries	n/a	
Alien species	n/a	

Table 4. Major anthropogenic factors limiting, or potentially limiting, viability of populations of northern green sturgeon. Factors were rated on a five-level ordinal scale where a factor rated "critical" could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated "high" could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction is unlikely as a result. A factor rated "n/a" has no known impact. Certainty of these judgments is moderate. See methods section for descriptions of the factors and explanation of the rating protocol.

Columbia River region. The majority of past northern green sturgeon harvest occurred in this region; they were caught by commercial fishermen, anglers, and Native American gillnetters. Sturgeon landings were recorded from the Columbia River estuary and from Grays

Harbor and Willapa Bay, Washington, to the immediate north of the estuary. There is little or no evidence of green sturgeon spawning in the rivers of this region, so it is likely that sturgeon harvested there migrated from California or Oregon, as indicated by limited recaptures of tagged individuals (Adams et al. 2007). Further evidence of the lack of local recruitment into the fishery is indicated by the fact that few juvenile sturgeon (<1.3 m) have been caught in this region (Emmett et al. 1991).

The commercial catch in the Columbia River region (Columbia River estuary, Grays Harbor, Willapa Bay) has fluctuated considerably over time, but catches appear to have increased in recent decades. Between 1941 and 1951, catches averaged about 200-500 fish per year, while between 1951 and 1971 the catch averaged about 1,400 fish per year (Houston 1988). In the late 1980s, an average of 4.7 tons of green sturgeon (ca. 500-1,000 fish) were harvested each year in Grays Harbor and 15.9 tons (ca. 2,000-4,000 fish) were harvested in Willapa Bay (Emmett et al. 1991). There have also been some notably high catches; in 1986, 6,000 green sturgeon were harvested in the Columbia River estuary (Oregon Dept. of Fish and Wildlife (ODFW) 1991) and 4,900 were taken in 1987 (ODFW, unpubl. data). From the 1960s-1980s, the commercial catch of green sturgeon in the Columbia River has averaged 1,440 fish (1960s), 1,610 (1970s) and 2,360 (1980s); the catch since 1990 has ranged from 3200 fish (1991) to 0 fish (2002) (Adams 2007). The Columbia River recreational catch has been consistently below 200 fish per year since 1988 (ODFW 1991, Adams 2007). For 1985-2003, Adams et al. (2007) estimated annual harvest of green sturgeon from all sources as ranging from 500 to over 9000 fish, with catches since 2001 being less than 1,000 fish per year, mostly taken in Washington. While fishing for green sturgeon is now prohibited in Washington, some mortality from fishing presumably continues as the result of by-catch from other fisheries (Adams et al. 2002). The commercial fishery took both northern and southern green sturgeon; only tagged fish were identified to the appropriate DPS.

<u>Klamath and Trinity rivers</u>. A small number of northern green sturgeon were probably taken in this sport fishery in the past but the main harvest is now by the Yurok, Karuk, and Hupa tribal gillnet fisheries (USFWS 1990, Adams et al. 2005). A small, but possibly significant, number are also taken in an illegal snag fishery. All fisheries target sturgeon as they move upriver to spawn during the spring and as they return seaward through the estuary during June-August (USFWS 1990). In the tribal fishery, mainly adult sturgeon (>130 cm FL) are captured (mean length 179 cm FL in 1988). The percent of the total (sport and tribal) harvest in the Pacific Northwest taken from the Klamath River increased from a low of 5% in 1987 to 59% in 2003 (Van Eenennaam et al. 2006, Table 5). This increase most likely reflected changes in regulations to limit green sturgeon harvest in Oregon and Washington (Adams et al. 2002).

	Klamath River				Total Harvest (CA, OR, WA)	Percent of Total Harvest from Klamath River
Year	Yurok	Hupa	Sport	Total		
1985	351	10	NA	361	5,156	7
1986	421	30	153	604	9,065	7
1987	171	20	170	361	7,669	5
1988	212	20	258	490	6,514	8
1989	268	30	202	500	4,067	12
1990	242	20	157	419	4,736	9
1991	312	13	366	691	6,788	10
1992	212	3	197	412	4,551	9
1993	417	10	293	720	4,267	17
1994	293	14	160	467	1,342	35
1995	131	2	78	211	1,286	16
1996	119	17	210	346	1,692	20
1997	306	7	158	471	3,199	15
1998	335	10	103	448	1,692	26
1999	204	27	73	304	1,491	20
2000	162	31	15	208	1,796	12
2001	268	10	NA	278	862	32
2002	273	5	NA	278	696	40
2003	287	16	NA	303	514	59
2004	222	12	NA	234	NA	NA

Table 5. Green sturgeon harvest numbers and percent of total harvest (California, Oregon and Washington combined) from the Klamath River, California (Source: Adams et al. 2002, Van Eenennaam et al. 2006).

The average total length of northern green sturgeon captured in the Yurok Tribal fishery increased slightly from 1980 to 2004 (Figure 1). Moreover, the proportion of green sturgeon greater than 190 cm increased from 30% in 1995 to approximately 40% in 2004 (D. Hillemeier, Yurok Tribal Fisheries Program, unpublished data). Because the length of captured individuals did not decrease, the Yurok Tribal fishery apparently does not adversely impact the size distribution of spawning adults. However, it is uncertain whether the increase in numbers of large adults signifies a change in population structure towards larger individuals or a loss of younger year classes.

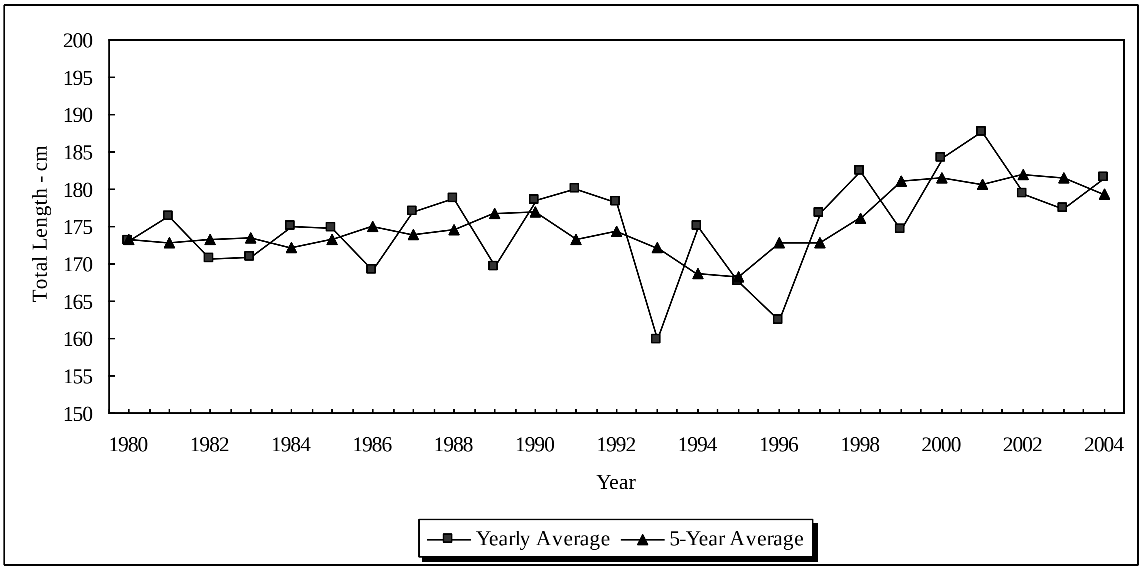


Figure 1. Average total length of northern green sturgeon sampled in the Yurok fishery, 1980-2004 (Source: D. Hillemeier, Yurok Tribal Fisheries Program, unpublished data).

Although present in low numbers, there is no indication that green sturgeon are in decline in the Klamath River basin (Adams et al. 2002, 2005; Beamesderfer and Webb 2002). However, given the status of other anadromous species in the Klamath River basin, the extended freshwater residency of at least some individuals, delayed maturity, and longevity of green sturgeon, there is concern that adverse impacts to the population may not be detected unless they are analyzed at the appropriate time scale (17 to 23 years; D. Hillemeier, unpublished data).

Effects of Climate Change: Increased water temperatures brought about by climate change may place northern green sturgeon under chronic stress that can result in metabolic costs that impair reproduction, growth and immune function (Lankford et al. 2005). Mayfield and Cech (2004) recommended that, in order to enhance growth, management plans should protect green sturgeon from prolonged exposure to temperatures above 19°C. Similarly, Van Eenennaam et al. (2005) concluded that temperatures above 20°C are detrimental to reproduction and most likely result in low hatching success, especially during dry water years. Summer water temperatures in the mainstem Klamath River already frequently exceed 20°C and temperatures in California are expected to increase under all climate change scenarios (Hayhoe et al. 2004, Cayan et al. 2008). Increases in summer temperatures may affect the growth and metabolic costs of juvenile and adult green sturgeon that hold in rivers throughout the summer. Climate change is also predicted to alter the flow regimes in rivers. In the Klamath and Trinity rivers, river flow may peak earlier in the spring and continue tapering through the summer before pulsing again later in the fall. The resulting changes in river flow and temperature may change the timing of adults and juveniles entering and exiting these systems. Quiñones and Moyle (2012) predicted these changes will cause increased declines in anadromous salmonids in the Klamath basin, so negative impacts to green sturgeon are likely as well. Moyle et al. (2013) rated northern green sturgeon as “highly vulnerable” to extinction in California as the result of climate change, largely as a result of increased temperatures and reduced flows in the Klamath River.

Status Determination Score = 2.7 - High Concern (see Methods section, Table 2)**.** Northern green sturgeon merit high concern status, even though they are not in immediate danger of extirpation from California. The Klamath-Trinity River population is the sole reproducing population in California and, apparently, is by far the largest population, giving it added significance. Green sturgeon are considered to be a threatened species in Canada. In 2006, the National Marine Fisheries Service determined that the northern green sturgeon DPS did not warrant listing under the Federal Endangered Species Act (50 CFR part 223); however, it was designated a species of concern (www.nmfs.noaa.gov). Green sturgeon (both DPS's combined) are given a near-threatened status by International Union for Conservation of Nature (IUCN) Red List (www.iucnredlist.org). The southern (Sacramento) DPS of green sturgeon was listed in 2006 as a threatened species under the Federal Endangered Species Act. After the southern green sturgeon was listed, both Oregon and Washington banned take by both commercial and sport fisheries.

Metric	Score	Justification
Area occupied	1	Only Klamath-Trinity population appears to be self-sustaining in California - this would score '2' if Oregon populations were included
Estimated adult abundance	2	Unknown, but 1,000-5,000 adults would be a conservative estimate
Intervention dependence	4	Long-term persistence depends on fisheries management and habitat restoration
Tolerance	3	Fairly tolerant of conditions in the Klamath River although susceptible to warm temperatures
Genetic risk	4	Presumably some genetic connections to Rogue population
Climate change	2	Limited spawning and rearing habitats suggests vulnerability to increased temperatures, reduced summers flows and other climate change-related stressors
Anthropogenic threats	3	Five threats scored 'medium' (see Table 4)
Average	2.7	19/7
Certainty	3	Abundance not well understood but many publications exist on distribution and behavior

Table 6. Metrics for determining the status of northern green sturgeon, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

In California, only one spawning population is recognized in the Klamath River, raising concerns about limited genetic diversity and gene flow. The possibly reproducing population in the Eel River is presumably derived from strays from the Klamath River. Conditions in the Klamath River for spawning and rearing have likely worsened due to the presence of major dams in both the main stem Klamath and Trinity rivers. Dams have dramatically altered the hydrology and geomorphology of these systems (NRC 2004). Degradation of habitats, combined with the predicted effects of climate change, make northern green sturgeon vulnerable to changing environmental conditions and potentially less suitable habitat conditions.

The closure of green sturgeon fishing, except for tribal fisheries, has reduced harvest rates in California. However, the legacy of harvest prior to 2007 may still be impairing the recovery of some populations. Green sturgeon population growth is particularly sensitive to adult and subadult mortality, especially if the effective spawning population size becomes low (Heppell 2007). Large increases in egg production and juvenile survival are required to counterbalance the impact from even relatively low levels of fishing mortality. In addition, recent work (Israel et al. 2004) suggests that not all spawning populations of green sturgeon have been identified, a necessary step for the adequate protection of green sturgeon genetic diversity.

Management Recommendations: The following conservation measures are needed to maintain or increase northern green sturgeon abundances:

1. Detailed studies on life history and ecological requirements are needed. Current population assessment and monitoring by the USFWS, Yurok Tribe, and others should be expanded, particularly for Klamath River populations. The current paucity of information and empirical data about the population status, structure and dynamics of northern green sturgeon means that population trends cannot be predicted, nor stocks properly managed. Females mature relatively late in life and may not spawn every year, so maintenance of sufficient reproductive potential (i.e., numbers of mature females) in populations is an important management consideration.

2. Nursery habitats for juveniles in river and estuarine habitats need to be identified and protected. One method for determining optimal habitats is to examine the digestive tracts of juvenile green sturgeon to evaluate the nutritional condition of fish rearing in different habitats (Gisbert and Doroshov 2003). Shortages of food supply can disrupt the organization and generation of juvenile digestive systems, directly affecting growth and survival.

3. Tribal fisheries that target northern green sturgeon should be limited until more is known about the biology and abundance of this species. At a minimum, special harvest regulations for green sturgeon are needed to reduce the catch of large females of peak reproductive ages of 25 to 40 years old (Heppell 2007). The effect of harvest on population productivity could be reduced by a slot limit to reduce the number of age classes harvested (Heppell 2007).

4. Populations can benefit from habitat restoration, especially of estuaries and lagoons. Measures should be adopted to keep summer water temperatures below 20°C, where possible, and to decrease the input of fine sediments into streams. Both of these measures can enhance the development and subsequent recruitment of juvenile green sturgeon.

5. The effects on northern green sturgeon of the proposed removal of four dams on the Klamath River need to be evaluated, especially in relation to low summer flows (e.g., lack of year-round tailwater flows from controlled dam releases) and with respect to potential for green sturgeon to use habitats made available by dam removal.

Figure 2. Freshwater distribution of northern green sturgeon, *Acipenser medirostris* (Ayres), in California. The only confirmed spawning population is in the Klamath-Trinity river system.

WHITE STURGEON
Acipenser transmontanus (Richardson)

Status: High Concern. Annual recruitment of white sturgeon in California appears to have decreased since the early 1980s but several strong year classes are evident. Continued close management is required to sustain white sturgeon populations into the future.

Description: White sturgeon adults have wide, rounded snouts, with four barbels in a row on the underside, closer to the tip of the snout than to the mouth (Moyle 2002). They feed with a toothless, highly protrusible mouth and process food with a palatal organ in the pharynx. Their bodies have 5 widely separated rows of bony plates (scutes). Scute counts per row are: 11-14 (dorsal row), 38-48 (two lateral rows) and 9-12 (bottom rows). Four to eight scutes are also found between the pelvic and anal fin. Although they lack the large scutes behind the dorsal and anal fins found in green sturgeon (*A. medirostris)*, small remnant scutes (fulcra) may be present. The dorsal fin has one spine followed by 44-48 rays. The anal fin has 28-31 rays. The first gill arch has 34-36 gill rakers. Body coloration is gray-brown on the dorsal surface above the lateral scutes, while the ventral surface is white and fins are gray. Their viscera are black. Dispersing juveniles tend to be darker than dispersing free embryos (Kynard and Parker 2005). Juveniles less than one year old have 42 dorsal fin rays, 35 lateral scutes, and 23 gill rakers on the first arch.

Taxonomic Relationships: Recent genetic analysis supports the close relationship between white sturgeon and Amur sturgeon (*A. schrenckii;* found only in Asia), which had a common ancestor approximately 45.8 million years ago (Peng et al. 2007, Krieger et al. 2008). In California, some genetic differentiation was thought to exist among white sturgeon populations from different river systems (Bartley et al. 1985) but a detailed genetic analysis using microsatellites failed to reveal any such population structure (Schreier et al. 2011). Recent DNA analysis using microsatellites has determined that genetic differentiation (F_{ST} = 0.19) is high enough among white sturgeon from the Columbia, Fraser and Sacramento River basins to be able to distinguish them (Rodzen et al. 2004), despite mixing in the ocean and high levels of genetic diversity (Schreier 2011). Schreier (2011) found that sturgeon captured in non-natal estuaries could be assigned by genetic techniques to their natal river, although the high level of genetic diversity found in the three major anadromous sturgeon populations indicates that some mixing of stocks takes place. Nevertheless, there is now sufficient evidence to treat the Sacramento-San Joaquin white sturgeon stock as a Distinct Population Segment (DPS).

Life History: White sturgeon primarily live in estuaries of large rivers but migrate to spawn in fresh water and often make long ocean movements between river systems. They commonly aggregate in deep, soft-bottomed areas of estuaries, where they move about in response to changes in salinity (Kohlhorst et al. 1991). In the lower Columbia River, white sturgeon make seasonal and diel movements (Parsley et al. 2008), moving upstream in the fall and downstream in the spring. They are most active at night, when they move into shallower waters to feed. Some individuals express site fidelity by returning to previously occupied sites (Parsley et al. 2008).

In the ocean, some individuals may migrate large distances. White sturgeon tagged in the San Francisco Estuary have been recaptured in the Columbia River estuary (L. Miller 1972a,b, Kohlhorst et al. 1991). One of these fish was then subsequently recaptured 1,000 km upstream in the Columbia River. Tagged individuals have routinely been detected 1,000 km from the tagging site (Chadwick 1959, Welch et al. 2006). Recently, one white sturgeon tagged in May, 2002, in the Klamath River, was tracked to the Fraser River, British Columbia, a distance far greater than 1000 km (Welch et al. 2006). Because this individual spent nearly equal amounts of time in both the Fraser and Klamath rivers, it was difficult to determine which was the natal river. However, genetic studies suggest that extensive movements are associated with feeding rather than spawning (Schrierer 2011).

In estuaries, white sturgeon move into intertidal areas during high tides to feed. Most prey are taken on or near the bottom. Young white sturgeon (~ 20 cm FL) prefer amphipods (*Corophium* spp.) and opossum shrimp (*Neomysis mercedis*) (Radtke 1966, Muir et al. 1988, McCabe et al. 1993). Diet becomes more varied as they grow but continues to be dominated by benthic invertebrates such as shrimp, crabs, and clams. Today, most benthic invertebrate prey species in the San Francisco Estuary are nonnative, demonstrating the opportunistic feeding nature of white sturgeon (Moyle 2002). One heavily used prey is the overbite clam, *Corbula amurensis*, which became very abundant after its invasion into Suisun Bay in the 1980s. However, foraging on the overbite clam may inhibit growth, because some clams pass through the gastrointestinal tract without being digested, possibly decreasing nutritional intake (Kogut 2008). Fish, especially herring, anchovy, striped bass, starry flounder, and smelt, are consumed by larger sturgeon. In the San Francisco Estuary, white sturgeon feed on Pacific herring eggs (McKechnie and Fenner 1971), much as their Columbia River counterparts do on eulachon eggs (McCabe et al. 1993). In California, stomach contents of large individuals have also included onions, wheat, Pacific lamprey, crayfish, frogs, salmon, trout, striped bass, carp, pikeminnow, suckers and, in one instance, a cat (Carlander 1969).

In the San Francisco Estuary, young sturgeon reach 18-30 cm by the end of their first year (Kohlhorst et al. 1991). Maximum growth is achieved by juvenile white sturgeon grown in captivity on artificial diets, consuming 1.5 to 2% of their body weight each day at 18°C (Hung et al. 1989). As white sturgeon age, growth rates slow so that they reach 102 cm TL by their seventh or eight year. They may ultimately reach 6 m FL. The largest white sturgeon on record weighed 630 kg and was likely more than 100 years old; fish of this size were probably the largest freshwater fish in North America (Moyle 2002). The largest white sturgeon caught in Oregon measured 3.2 m FL and was 82 years old (Carlander 1969). In California, the largest white sturgeon on record was from Shasta Reservoir in 1963; it was 2.9 m TL, 225 kg, and at least 67 years old (T. Healy, CDFW, pers. comm. 2001). Today, in California, white sturgeon larger than 2 m and older than 27 years are uncommon.

Male white sturgeon mature when10-12 years old (75-105 cm FL); females mature later at about 12-16 years old (95-135 cm FL) (Kohlhorst et al. 1991, Chapman et al. 1996). However, males mature at 3-4 years and females at 5 years while in captivity (Wang 1986). Photoperiod and temperature regulate maturation in adult white sturgeon (Doroshov and Moberg 1997). Prior to spawning, adults may move into the lower reaches of rivers during the winter months and later migrate upstream into spawning areas in response to increases in flow (Schaffter 1997a,b). Spawning initiates in response to high flows from late February to early June (McCabe and Tracy

1994). Only a small percentage of adults will spawn in any given year. In the Columbia River, males spawn every 1-2 years while females spawn every 3-5 years (McCabe and Tracy 1994).

Spawning in the Sacramento River occurs primarily between Knights Landing (233 rkm) and Colusa (372 rkm) (Schaffter 1997a,b). A few adults spawn in the Feather and San Joaquin rivers (Kohlhorst 1976, Kohlhorst et al. 1991), although recent activity in the Feather River is unconfirmed (A. Schierer, pers. comm. 2010). Genetic evidence suggests that there is little fidelity to spawning areas within the Sacramento River system (Schierer 2011). The fecundity of females from the Sacramento River averages 5,648 eggs/kilogram body weight, so an individual female (1.5 m TL) may contain 200,000 eggs (Chapman et al. 1996). White sturgeon typically spawn in deep water over gravel substrates or in rocky pools with swift currents. Eggs have been collected from the stream bed at depths of 10 m (Wang 1986). In the Columbia River, white sturgeon spawn over cobble and boulder at depths of 3-23 m and velocities of 0.6-2.4 m/sec (McCabe and Tracy 1994). Adults migrate back to the estuary after spawning.

Eggs (3.5-4.0 mm; in Billard and Lecointre 2001) become adhesive upon fertilization, allowing them to stick to stream substrates. Time to hatch is dependent on temperature but larvae generally hatch in 4-12 days (Wang 1986). Larvae are 11 mm at hatch and swim vertically while drifting towards the estuary. They switch to swimming horizontally and feed from the bottom once the yolk sac is absorbed, in about 7-10 days. Sacramento River white sturgeon larvae were found to be photonegative upon hatching, moving downstream short distances by swimming near the bottom, seeking cover (Kynard and Parker 2005). Larvae aggregated, swam, and foraged near the bottom and demonstrated an increasing trend to swim above the bottom. Strong dispersal occurred as early juveniles swam actively downstream. Consequently, Sacramento River white sturgeon are described as having a “two-step downstream dispersal” completed by larvae and early juveniles during both day and night, but peaking at night. Juvenile sturgeon use the less saline portions of estuaries, suggesting that the ability to osmoregulate increases with age and size (McEnroe and Cech 1987). Osmoregulation efficacy may also be size-dependent, even between individuals of the same age (Amiri et al. 2009). Consequently, size at time of estuary entry may be a limiting factor for juvenile survival. In the lower Fraser River, most juvenile white sturgeon use sloughs from June to August (Bennett et al. 2005); occupied sloughs were more than 5 m deep, turbid, and had multidirectional currents, soft sediments, and readily available prey (mysid shrimp, dipteran larvae, fish).

In the San Francisco Estuary, the white sturgeon population is dominated by a few strong year classes, reflecting variability of annual spawning success. Strong year classes result from years of high spring flows in the rivers (Kohlhorst et al. 1991, Schaffter and Kohlhorst 1999, Fish 2010). High spring flows may quickly move larval sturgeon downstream into suitable rearing areas (Stevens and Miller 1970) or induce more sturgeon adults to spawn (Kohlhorst et al. 1991). In the lower Columbia River, year class strength is correlated to the size and availability of prey at the onset of exogenous feeding (Muir et al. 2000). Amphipods (Corophiidae), copepods, and dipteran larvae and pupae are important prey to larval and young-of-year sturgeon. Predation on larvae, especially by prickly sculpin, may be another factor limiting recruitment in some areas (Gadomski and Parsley 2005, Gadomski and Parsley 2005b).

Habitat Requirements: White sturgeon adults respond to increases in flow to initiate spawning from late February to early June. Spawning takes place at temperatures ranging from 8 to 19°C, peaking at temperatures around 14°C (McCabe and Tracy 1994). Successful incubation requires stream substrates with minimum amounts of sand and silt because excessive siltation can smother embryos. Recruitment failure in the Nechako River, Canada, was attributed to siltation of main channel sediments after large scale (1,000,000 m^3) introduction of fine sediments by upstream stream avulsion (McAdam et al. 2005). The recruitment failure was attributed to egg suffocation and increased predation because larvae lacked interstitial spaces in the substrate in which to hide. Newly hatched embryos preferred substrates from 12 to 22 mm in laboratory tests (Bennett et al. 2007).

The first few months of life are considered to be critical for sustaining populations (Coutant 2004). Successful recruitment also appears to be associated with complex habitats, flooded riparian vegetation (floodplain habitat) and rocky substrates (Coutant 2004). Lack of cover in edge habitats downstream of spawning areas, along with low flows from the time of spawning until juvenile outmigration, decreases recruitment. Productive spawning areas in the Sacramento River are associated with areas where levees are set back, allowing access to floodplains and backwater habitats (e.g., Wilkins and Butte sloughs) during high spring flows.

Distribution: White sturgeon can be found in salt water from the Gulf of Alaska south to Ensenada, Mexico. However, spawning only occurs in a few large rivers from the Sacramento-San Joaquin system northward. Self-sustaining spawning populations are currently only known in the Fraser (British Columbia), Columbia (Washington), and Sacramento (California) rivers. Landlocked populations also occur above major dams in the Columbia River (McCabe and Tracy 1994). White sturgeon from California are caught in small numbers in the Columbia River and other estuaries (Schierer 2011). At least one white sturgeon tagged in the Klamath River spent extensive time in the Fraser River (Welch et al. 2006).

In California, white sturgeon spawn primarily in the Sacramento River (to Keswick Dam) but may also spawn in the San Joaquin River (Jackson and Van Eenennaam 2013) and in the Feather River (to Oroville Dam facilities), when water quality and flow conditions are favorable (Schaffter 1997a,b). The lower Pit River was likely an important spawning area, prior to construction of Shasta Dam in the 1940s (T. Healey, CDFW, pers. comm. 2001). Sturgeon became trapped behind Shasta Dam, establishing a landlocked population that became self-sustaining and supported a small fishery (Moyle 2002). However, subsequent dam construction on the Pit River blocked access to spawning areas and prevented ongoing reproduction of this population (T. Healey, CDFW, pers. comm. 2001). Long-lived individuals and fish from stocking attempts in the 1980s are still occasionally caught in Shasta Reservoir. Historically, small runs also occurred in the Russian, Klamath and Trinity rivers. White sturgeon have also been documented in the Eel River (M. Gilroy, CDFW, pers. comm. 2011). It is doubtful that any of these latter four rivers currently support populations of white sturgeon.

Aquaculture facilities now cultivate white sturgeon in California and juvenile sturgeon can be sold to aquarists. Presumably, aquarium releases have resulted in occasional white sturgeon being found in reservoirs in southern California (C. Swift, pers. comm. 1999) and the San Francisco region (e.g., a 21 kg individual caught in Lafayette Reservoir, Contra Costa County).

Trends in Abundance: The California Department of Fish and Wildlife has been monitoring trends in white sturgeon abundance for decades and information on trends for nearly 80 years is available. From that body of work, it is clear that large variations in recruitment, frequently including 5 or more consecutive years of low or no recruitment, have been routine since the 1930s and the proximate cause for this variation is low flows during winter and/or spring. Managing the population through predictable ebbs in abundance is the key to conservation of white sturgeon and protection of its fishery.

The CDFW's index of annual white sturgeon recruitment from age-0 and age-1 fish suggests that peak recruitment has decreased trend-wise since the early 1980s, recruitment in most years is a small fraction of peak recruitment, and the most recent notably-high recruitment was in 2006 (Figure 1). This trend is completely plausible and expected from the relationship between hydrology and recruitment, but the slope of the trend may be biased toward decline due to release of fingerlings by hatcheries from 1980-1988.

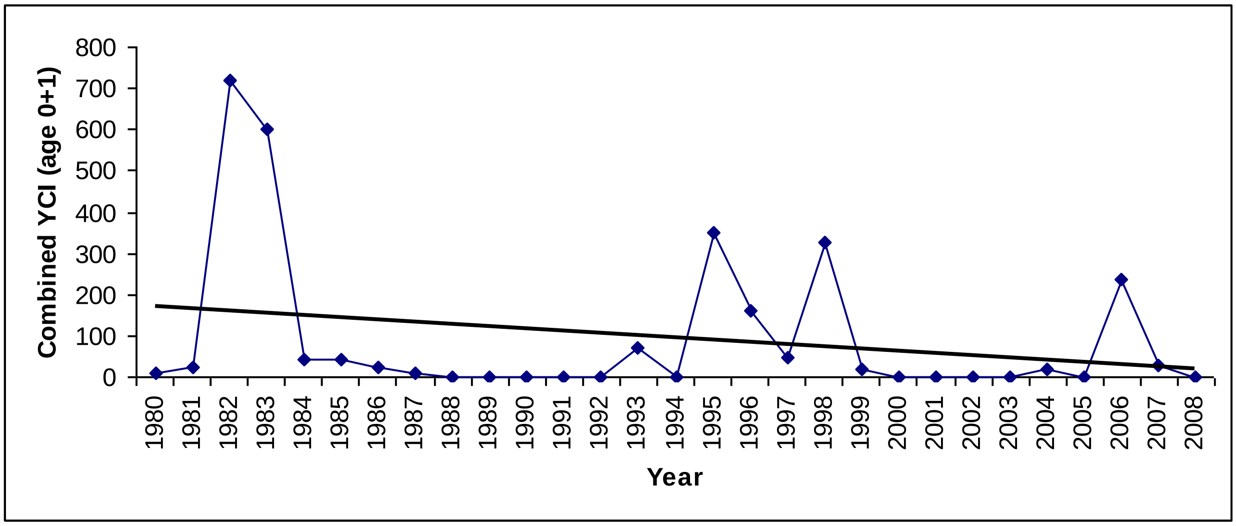


Figure 1. White sturgeon year class indices (age-0 and age-1 combined), San Francisco Bay, 1980-2012.

Trends since 1980 in the abundance of subadult and adult white sturgeon are as expected from variations in river hydrology and indices of recruitment, though abundance estimates are generally imprecise and sometimes lack confidence intervals. Interpretation of catch-per-unit-effort (CPUE) data from the fishery is confounded somewhat due to changes in regulations regarding size limits, daily bag limits, and annual bag limits. Length frequency distributions are a particularly important component when interpreting trends in abundance.

Estimated annual abundance of white sturgeon >= 102 cm Total Length (TL) has ranged from approximately 2,500-300,000 since 1980 (DuBois et al. 2011); the best estimates ranged from approximately 75,000-150,000 fish. The most recent and rigorous estimates are for fish 117-168 cm TL from the period 2007-2011, and those ranged from approximately 30,000-56,000 fish (DuBois and Gingras 2011). Extreme CPUE values should be discounted because they likely indicate unusual distributions of fish rather than rapid changes in the population's abundance. Using standardized fish capture and tagging techniques as part of a CDFW mark-recapture study, annual CPUE of white sturgeon 117-168 cm TL has varied from approximately

1-13 fish/100 net-fathom hours since 1980 and was less than 2 fish/100 net-fathom hours during the period 2005-2012 (DuBois and Gringas 2013).

Nearly all historical fishery-dependent data comes from Commercial Passenger Fishing Vessel (CPFV, a.k.a. party boat) logbooks. Annual white sturgeon CPUE in that fishery has varied between approximately 2-4 fish/100 hours of fishing effort since 1980 (DuBois and Gringas 2013). However, length data are not collected by the CPFV fleet and, since 1980, the size limit (TL) on white sturgeon changed from >=102 cm to 107-183 cm, 112-183 cm, 117-183 cm, and 117-168 cm in subsequent years, so it is only possible to describe coarse changes in white sturgeon demographics using CPFV data.

Annual length frequency distributions from CDFW's mark-recapture study and a pilot study using longlines clearly show the recruitment, growth, and subsequent decrease in abundance of strong year classes (Schaffter and Kohlhorst 1999, DuBois et al. 2011, DuBois and Gringas. 2013), as do length frequency distributions from CDFW Sturgeon Fishing Report Card data (CDFW Sturgeon Fishing Report Card reports, DuBois et al. 2011). Report cards have been in use since 2007. Because anglers commonly volunteer data on the lengths of fish too small to keep, the cards are helping bridge the long-standing gap in information on fish aged 2-8.

Trends of year-class indices (YCI), based on the number of age-0 and age-1 juveniles, suggest recruitment has decreased significantly, with low recruitment for 12 of the 29 years (1980-2008) on record (Figure 1). Although the present white sturgeon population appears to have been reduced over the last 30 years, some recent population trends are encouraging and stakeholder concerns about the white sturgeon population and fishery in California have resulted in highly restrictive angling regulations, new monitoring and research efforts, strong anti-poaching measures, and fish passage and habitat restoration efforts.

Nature and Degree of Threats: All sturgeon species worldwide are in serious decline and some are on the verge of extinction. Principal threats to sturgeon worldwide are similar to those in California (Table 1) and include: harvest (especially poaching), dam-related flow alteration and reduction, habitat degradation, and pollution (Billard and Lecointre 2001).

Major dams. Dams block access to important upstream spawning habitats and alter flows, which results in reduced habitat quantity and quality for early life stages (Coutant 2004). The major 'rim dams' in California largely lack fish passage facilities, so sturgeon are confined to downstream areas. In the Sacramento River, years of high spring outflow have been associated with strong year classes. The large dams on nearly all Central Valley rivers reduce the frequency, volume, and duration of these flows, reducing the frequency of successful sturgeon year classes (Moyle 2002). Dam operations can attenuate winter and spring flows required for the initiation of spawning and outmigration. Changes in the hydrograph can disconnect main channel habitats from floodplains, which may be especially important rearing habitats. Changes in sediment budgets and flow regime can decrease the quality and quantity of spawning and incubation habitats. For example, dam-attenuated winter flows can limit the amount of cover available in interstitial spaces in rocky substrates because the substrates are scoured less frequently. Changes to hydrographs can influence juvenile movements and predation rates. Lower turbidity levels and simplified channels as result of dam construction/impoundment may result in increased main channel predation of juveniles (Gadomski and Parsley 2005b). Lack of suitable habitats below dams may limit recruitment or lead to recruitment failure (Kynard and

Parker 2005).

Agriculture. Levees and land reclamation along rivers and estuaries have substantially eliminated large areas of floodplain habitats and their connectivity to main river channels, reducing access to important juvenile rearing areas. These historically abundant habitats once offered protection for sturgeon and many other native fishes from high flows, provided foraging habitats, and served as holding areas during migration. Diversion of water for agriculture can also reduce flows to the extent that sturgeon populations can no longer be supported in some areas (Moyle 2002). White sturgeon are particularly sensitive to agricultural pollutants. They readily bioaccumulate toxins from fertilizers and pesticides, which can cause deformities, decrease growth, and reduce reproductive potential. In the Columbia River, the incidence of physical deformities, such as misshapen fins, abnormal (short or forked) barbels and malformed or missing eyes increased with age, suggesting that they were a result of continued exposure to sediments contaminated with organic pollutants (Burner and Rien 2002). Exposure to organochlorine pesticides caused an overall decrease in the condition factor of juveniles, as well as decreasing the concentrations of sex hormones (testosterone and estradiol) in white sturgeon blood plasma (Gundersen et al. 2008). Electrophilic pesticides that can bond to DNA and other cellular macromolecules are common in the Sacramento River (Donham et al. 2006). Concentrations of mercury in white sturgeon livers also increased with age, suggesting that white sturgeon are prone to the bioaccumulation of heavy metals (Webb et al. 2006). Liver mercury content is negatively correlated with relative weight and gonadosomatic index. Consequently, exposure to mercury likely negatively affects white sturgeon reproductive potential and the potential for long-term mercury exposure in the Sacramento River basin is high.

Selenium entering the San Francisco Estuary from agricultural drainage (Presser and Luoma 2006) can decrease juvenile survival. Juveniles fed diets with high concentrations (> 41.7 ug Se/g) of selenium decreased swimming activity and grew less than other groups (Tashjian et al. 2006). Selenium accumulates in the kidney, muscle, liver, gill, and plasma tissues of these fish, contributing to decreased survival, particularly when exposed to brackish water (> 15 ppt) (Tashjian et al. 2007). Contaminated fish also had less energy reserves (whole body protein, lipids), perhaps limiting foraging activity and escape from predation. Although current regulatory thresholds for selenium toxicity (10-20 ug Se/g) may protect white sturgeon from adverse impacts, the concentration of selenium by the alien overbite clam, a major prey of sturgeon, may be resulting in increased levels in sturgeon as well.

Fertilizers entering the estuary cause algal blooms which may harm sturgeon both through release of toxins (*Microcystis*) and through depleting oxygen and increasing CO_2 in backwaters. Hypercapnia (elevated levels of CO_2) can cause mortality or morbidity in juvenile white sturgeon because energy normally used for growth, disease resistance and lipid storage is redirected toward maintaining homeostasis (Cech and Crocker 2002, Crocker and Cech 2002).

	Rating	Explanation
Major dams	High	All rivers occupied in CA are dammed, blocking access to spawning habitats and altering flows and habitat suitability
Agriculture	High	Water demands result in decreased flows in rivers during critical life history periods; pollution from agricultural return waters may acutely affect sturgeon
Grazing	Low	Effects mostly upstream of reaches occupied by sturgeon
Rural residential	Low	Rural residences occur along white sturgeon streams (e.g., Klamath River) but the effects from rural development are likely minor
Urbanization	High	Urban water demand, runoff and pollution inputs can create toxic environments; habitat alteration and simplification are severe in urban areas; multiple large urban areas within existing range
Instream mining	Low	Effects unknown but present in some coastal streams
Mining	Medium	Most toxic runoff is above dams, although Iron Mountain mine poses a major threat if controls of tailings and effluent fail
Transportation	Low	Roads, railroads, shipping lines and associated bridges and channelization modify rivers occupied by white sturgeon
Logging	Low	Impacts from sedimentation, etc. may affect rivers other than Sacramento River (e.g., Klamath River) but not likely to affect reproduction
Estuary alteration	High	California estuaries are severely altered; San Francisco Estuary and Delta habitats substantially altered and degraded from past
Recreation	Low	Boating and other activities can disturb sturgeon spawning and foraging; white sturgeon fatalities from vessel strikes are not uncommon
Fire	Low	Erosion from burned areas can increase fine sediment delivery to streams, but most impacts occur above major dams
Harvest	Medium	Legal and illegal harvest cause adult mortality, although legal harvest is now typically less than 10% of harvestable fish; illegal harvest for caviar and meat is a much greater threat
Hatcheries	Low	Aquaculture facilities exist for white sturgeon, but fish have not been released into the wild since approximately 1988
Alien species	Low	Alien species present throughout range; impacts largely unknown

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

Urbanization. The impacts from urbanization on white sturgeon are similar to those from agriculture, although perhaps not quite as widespread. Pollutants from sewage treatment plants, storm drains, and surface runoff have the potential to negatively affect sturgeon, as does often severe habitat simplification associated with urban development along river and stream corridors.

Mining. Iron Mountain Mine, an abandoned heavy metal mine above Keswick Reservoir (below Shasta Dam) on the Sacramento River, is an EPA Superfund site. While extensive measures have been taken to reduce the potential for toxic spills from the site, the impacts of a spill would be severe enough that even a low probability of failure rates concern. If the earthen retaining dam designed to impound mine effluents fails, an acidic slurry of toxic heavy metals could spill into the river, potentially resulting in massive fish kills; white sturgeon would likely be especially vulnerable to both acute (short-term) and subacute (long-term) exposure to these toxins, given their benthic foraging behavior and long life spans.

Logging. In the Sacramento River watershed, sturgeon are isolated from the effects of logging in headwaters by major dams, which minimizes their exposure to sedimentation and increased temperatures. However, white sturgeon may be negatively affected by logging in the Klamath and other river basins within their range. Introduced fine sediments (silt, sand, fine gravel) can fill substrate interstitial spaces and cause recruitment failure (McAdam et al. 2005). Laboratory experiments using Kootenai River white sturgeon found that fine sediment (5 mm) covering embryos resulted in 0-50% survival, delayed hatching and decreased larval length (Kock et al. 2006). Exposure of juvenile (3-78 days old) white sturgeon to didecyldimethylammonium chloride (DDAC), a highly soluble pesticide commonly used to protect lumber in Canada, resulted in mortality and sublethal effects (Teh et al. 2003). Didecyldimethylammonium chloride exposure resulted in 50% mortality of 78 day-old juveniles, the most resistant age group, within 18 and 36 hours of exposure. Sublethal effects to all age groups included decreased growth (weight and length) and decreased swimming activity. Juveniles that expressed sublethal effects had not recovered 21 days after exposure, perhaps increasing susceptibility to predation and disease and decreasing the probability of reaching sexual maturity. Although of particular concern in the Fraser River, Canada, DDAC may also impact sturgeon that migrate between rivers in California and Canada.

Estuary alteration. White sturgeon in California spend much of their life cycle in the heavily altered San Francisco Estuary or other smaller estuaries. The Delta's levees and rip-rapped channels restrict foraging habitat for sturgeon. At times, much of the freshwater inflow to the Delta is diverted into the pumps of the south Delta, altering or reducing river flow and entraining small sturgeon. Suisun Bay, Suisun Marsh and San Pablo Bay are primary rearing areas and are also subject to altered flows, contamination from many toxic compounds, invasions of alien species, and reduced water quality from urban runoff and effluent. Given the altered condition of the estuary and the fact that it is continuing to rapidly change, it is remarkable that white sturgeon have persisted in even moderately large populations (see Lund et al. 2007, 2008, Moyle 2008).

Harvest. White sturgeon populations were substantially reduced by commercial fishing in the San Francisco Estuary in the 19th century; consequently, commercial harvest has been prohibited since the mid-1900s (Moyle 2002). The sport fishery has become increasingly restrictive over time but, unlike in Oregon and Washington, California has not adopted a harvest quota.

White sturgeon fishing is currently closed in the north coast district (Humboldt, Del Norte, Trinity, Siskiyou counties), reaches of the Sacramento River in the Sierra and Valley districts (Shasta, Tehama, Glenn counties), in parts of San Francisco Bay, and at low-head dams (weirs) controlling flow into bypasses of the Sacramento River. The Sacramento River closure was implemented in 2009, closures at weirs were implemented in 2013, and other closures have been in effect for decades. Sport fishing regulations, established in 2007, allow individual anglers to harvest one fish per day and up to a total of three fish per year, whereas previous regulations did not limit the annual harvest. Also, in 2007, the size limit was changed from 46-72" TL to 46-66" TL. The size limits implemented are considered protective, yet were a compromise that still allows for potential harvest of female fish that have not yet spawned for the first time. In addition, Sturgeon Fishing Report Cards are required for all sturgeon anglers and are to be returned to the California Department of Fish and Wildlife upon completion; all harvested white sturgeon must be tagged. The Sturgeon Fishing Report Card and associated tags are the mechanisms whereby the daily and annual bag limits are enforced (see Management Recommendations section).

In anticipation of higher numbers of white sturgeon released in association with more restrictive angling regulations, several additional measures were taken in 2013 to improve the survival rates of fish that anglers are required to, or voluntarily, release. These protective regulations include: (a) only one single-point, single shank, barbless hook may be used on a line when taking white sturgeon, (b) snares may not be used to assist with landing a white sturgeon, (c) description of length limits in terms of fork length rather than total length, and (d) white sturgeon greater than 173 cm (68 in.) fork length may not be removed from the water and must be released immediately.

In general, harvest rates of fish 117-168 cm TL (e.g., the legally-harvestable size as of March, 2007, and a subset of all prior legal sizes) during 2000-2008 were lower than rates during the 1980s (DuBois et al. 2011) and the overall harvest rate trend is decreasing (M. Gingras, CDFW, pers. comm. 2013). Harvest rates have ranged from approximately 2-9%, but are likely biased low.

Illegal commercialization (poaching) of white sturgeon is common because of the high value of their caviar. As a consequence, the CDFW makes enforcement of sturgeon fishing regulations a high priority and, in 2007, a law was enacted that facilitated easier enforcement against those participating in illegal commercialization and drastically increased the severity of financial penalties associated with these activities.

White sturgeon contribute to a small Native American fishery in the Klamath River but only 186 juvenile and adult white sturgeon were caught by the Klamath River fishery from 1980 to 2002, about eight fish per year (Welch et al. 2006). Sacramento River white sturgeon may also be caught in fisheries in the Columbia River region but the potential effects on California populations are not known.

Hatcheries. In response to wide fluctuations in white sturgeon abundance and intermittent decreased catch rates over time in the sport fishery, outplanting of hatchery sturgeon stocks to augment natural populations has, although the subject of much debate, been proposed. White sturgeon have been raised in California aquaculture facilities for meat and caviar since 1980 and juvenile white sturgeon from those facilities were outplanted from 1980-1988; however, no hatchery stocks have been released into the wild since that time. The contribution of outplanted fish was not evaluated and records are sparse; nonetheless, it is estimated that a total of approximately 500,000 fry and fingerlings were released during the 1980s.

Hybridization of wild and hatchery stocks may have detrimental effects on the population structure of wild stocks, as studies of salmon populations have demonstrated (see Chinook salmon accounts in this report). Hatcheries may also facilitate the spread of disease such as iridovirus. Iridovirus infection of white sturgeon reduces the growth and survival of fry and fingerlings (Raverty et al. 2003).

Alien species. Alien fishes are abundant in the estuaries and rivers that white sturgeon inhabit. Alien fishes can reduce white sturgeon survival through predation on juveniles (Gadomski and Parsley 2005c), although this has not been demonstrated to be a problem in California. In the San Francisco Estuary, white sturgeon feed heavily on the overbite clam, which invaded in the 1980s. This clam (and other alien clams on which sturgeon feed) concentrate selenium and other heavy metals, which bioaccumulate in sturgeon and have the potential to negatively affect reproductive success.

Effects of Climate Change: Increases in water temperatures associated with climate change may decrease white sturgeon reproductive success. Successful spawning appears to be linked to cool water temperatures (< 18°C) and high spring flows. Females holding in 18-20°C water had inhibited ovulation and oocyte development (Webb et al. 1999, Linares-Casenave et al. 2002). Although based on laboratory results, these findings indicate that the pre-spawning temperature regime is important for normal ovarian development and should be considered in management of wild stocks. Bioenergetic modeling of white sturgeon in the Snake River also demonstrated that small increases in maximum water temperatures (19 to 24 °C) decreased growth and reproduction (spawning frequency, fecundity) because of decreases in caloric assimilation resultant from increases in energy costs (Bevelhimer 2002). Increased water temperatures may also hasten developmental times, perhaps resulting in a mismatch between the onsets of exogenous feeding and prey availability. Prey availability at onset of exogenous feeding was determined to be important in determining year class strength (Muir et al. 2000). Increased water temperatures may also make white sturgeon more susceptibility to disease. White sturgeon iridovirus is thought to be present in rivers throughout their range, and has been verified to occur in the anadromous waters of California's Central Valley (M. Gingras, CDFW, pers. comm. 2013). The virus is a slow wasting disease that primarily affects growth in fry and fingerlings by infecting the top layers of the skin, including the gills, barbels and nares (Drennan et al. 2007). Stressful conditions associated with poor water quality can induce the virus. Consequently, increased temperatures predicted from climate change models, in combination with pollution, may make young sturgeon more susceptible to the virus.

Climate change models predict seasonal shifts in precipitation, as well as increased frequency of floods and drought. Higher or more flashy winter flows may flush juvenile white sturgeon into estuarine areas before they are capable of adjusting to saline environments. The ability to osmoregulate is likely size dependent (Amiri et al. 2009), so younger and smaller juvenile sturgeon may be at risk, especially if floodplain and edge-habitat refuges are lacking, as is the case in much of the lower Sacramento River system. Coupled with predicted increases in estuary salinity levels due to sea level rise, earlier entry of juveniles into estuarine habitats may limit juvenile survival. In contrast, lower summer flows, exacerbated by increasing water demands, may decrease spawning and outmigration success.

Status Determination Score = 2.3 - High Concern (see Methods section Table 2). Despite a relatively robust population that presently includes tens of thousands of sub-adults and adults, white sturgeon must be managed carefully due to already demonstrated population cycles that may be exacerbated in the future by climate change, increasing human water demand, further degradation of habitats, overharvest, or some combination thereof. Management of white sturgeon is complicated by the combination of exposure to pollutants, freshwater and estuarine habitat alteration (particularly in the San Francisco Estuary), harvest, and because its long life span can mask the detection of poor reproductive success. NatureServe ranks white sturgeon as Globally secure (G4) but Imperiled (S2) in California due to anthropogenic impacts on their habitats. The American Fisheries Society considers the species to be Endangered (Jelks et al. 2008). Several populations in California are also considered "conservation dependent" (Musick et al. 2000).

Metric	Score	Justification
Area occupied	1	The only self-sustaining population in California appears to be in the Sacramento River
Estimated adult abundance	3	Based upon 2000-2009 estimates of age-15 fish and other demographic data
Intervention dependence	3	The population and fishery need to be monitored and managed closely, flows regulated, and pollution inputs and poaching reduced
Tolerance	2	Juvenile white sturgeon are intolerant of poor water quality, including high temperatures
Genetic risk	4	High genetic diversity
Climate change	2	Very sensitive to temperature increases, degraded water quality and flow changes predicted by climate change models
Anthropogenic threats	1	The combination of illegal harvest, pollution, and habitat alteration continue to threaten white sturgeon in the wild (see Table 1)
Average	2.3	16/7
Certainty (1-4)	4	

Table 2. Metrics for determining the status of white sturgeon, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: White sturgeon in the Sacramento River and the San Francisco Estuary have been regarded as well managed since the 1950s because they have sustained a fairly large fishery (Moyle 2002), though not as well managed as white sturgeon in Oregon and Washington. Unfortunately, increasing pollution, water diversion, habitat degradation, impacts from climate change, and poaching may limit recovery or contribute to further decline. The following are management recommendations to afford greater protection for white sturgeon in California:

Harvest management. Harvest regulations for white sturgeon have become increasingly restrictive, with severe limits placed on sport harvest in 2006 and, again, in 2009. However, California lags behind Oregon and Washington in regards to adaptive management of sturgeon harvest and has no white sturgeon management policy or plan.

Productivity of white sturgeon in California is lower than in Oregon and Washington, yet the white sturgeon fishery is very culturally and economically important; therefore, it is imperative to apply adaptive management to the recreational fishery and tight controls over harvest, both legal and illegal. The decline and subsequent listing of the southern green sturgeon DPS in California as threatened under the federal ESA may be indicative that white sturgeon are on the same trajectory and signals a need for greater conservation measures, monitoring, law enforcement and related resources to prevent further declines.

As a top priority, the California Fish and Game Commission should implement an annual quota on harvest of white sturgeon and should assure the continued availability of pertinent white sturgeon demographic and fishery statistics, implementation of a study on the effects of poaching, and the development of a white sturgeon management plan.

As noted, regulations established in 2007 require that sturgeon anglers record all fishing activity on Sturgeon Fishing Report Cards to be returned to the CDFW upon completion and that anglers tag all white sturgeon harvested. Data from Sturgeon Fishing Report Cards provide a much better description of the fishery than was available previously and complement the CDFW's on-going mark-recapture study. Prior to use of Sturgeon Fishing Report Cards, annual harvest could only be coarsely estimated from imprecise abundance estimates and annual harvest rate estimates. Data gathered from 2007-2012 Sturgeon Fishing Report Cards indicate that annual harvest was 1424-2048 fish and anglers released 4171-5802 fish. Accuracy of Sturgeon Fishing Report Card data is the subject of on-going investigation, but the trends and year-over-year numbers are generally consistent and reasonable.

Information on fishing effort for white sturgeon is incomplete and suggests a mixed picture. The only trend data available are from the CPFV fishery, where fishing effort from CPFVs that landed white sturgeon has declined trend-wise from a peak of nearly 25,000 hours in 1986 to a record low of barely 3,000 hours in 2012. Estimated annual fishing effort during daylight (i.e., biased low), in the Sacramento River watershed to Carquinez Strait, ranged from approximately 110,000-320,000 hours during 2006-2009.

Information on the number of sturgeon anglers in California is incomplete, but the number of issued Sturgeon Fishing Report Cards shows that interest in the recreational fishery is substantial. An annual average of roughly 55,000 Sturgeon Fishing Report Cards were issued for free, when issued by hand, an annual average of roughly 112,000 were issued for free, when issued by an automated system, and approximately 55,000 were issued 6 months into the first year they were sold ($7.50 plus up to 8% in fees), utilizing an automated system. One incongruity in the recent management of white sturgeon is that there are far fewer legal-sized white sturgeon than are authorized for harvest through issuance of Sturgeon Fishing Report Cards. In general, Sturgeon Fishing Report Cards provide valuable data and insights into the fishery and should be continued to be issued and their data analyzed into the future.

Illegal commercialization of white sturgeon remains a significant concern, given the high value of individual fish and the relative ease with which the largest and most fecund females are targeted. More intensive efforts are needed to identify, arrest and convict poachers and the dealers who buy illegal caviar and legislative action should be taken to increase the numbers of CDFW Wildlife Officers and ensure a dedicated number are assigned to white sturgeon-related enforcement throughout their range in the state.

Reducing pollution (especially from agriculture). White sturgeon are very sensitive to many pollutants (heavy metals, selenium, organic pollutants, pesticides), even when the pollutants are at low concentrations, in part because sturgeon are long-lived and bioaccumulate toxins over long periods of time in their bodies as well as in their eggs (passing them on to sensitive larvae). Improved monitoring and treatment of non-point source pollution is necessary to minimize impacts on white sturgeon. Restoration of tidal wetlands and floodplain habitats would likely enhance detoxification of water draining from agricultural fields and sewage.

Heavy metals, especially selenium, are of particular concern because of their effects on reproduction. Thus, both point and non-point sources of polluted effluents into Central Valley rivers and the San Francisco Estuary need to be identified and prioritized for treatment, containment, or other mitigation measures. Fortunately, selenium from oil refineries has been reduced to very low levels, while selenium inputs from farms on the west side of the San Joaquin Valley into the San Joaquin River have also been declining. These reductions have decreased selenium concentrations in overbite clams, a major sturgeon prey item (S. Luoma, pers. comm. 2009). This example demonstrates that pollution mitigation measures can be effective but efforts needs to be more comprehensive and systematic, focused on reducing inputs into waterways and eliminating point sources via treatment.

Habitat improvement. Freshwater and estuarine habitat alteration, especially from dam and levee construction, as well as elimination of most of the Central Valley's historic floodplain habitats, has limited spawning and rearing success in the Sacramento River (and possibly the Klamath River as well). Thus, restoring habitats required for juvenile rearing and spawning adults needs to be a priority in the Sacramento River basin. Access to rearing habitats with abundant prey may help mitigate effects of increased water temperatures resulting from climate change because larvae can better withstand increased temperatures when they feed at optimum (~ 15% body weight/day) or near-optimum feeding rates (Amiri et al. 2009). Restoration of tidal sloughs in California could also provide important rearing habitat.

Improving stream flows. The Sacramento River is a highly regulated river and white sturgeon depend on rare high water years - when dams spill or flood releases are high - for reproduction that leads to large year classes in the population. However, too little is known about specific flow requirements for spawning, instream rearing, downstream migration, growth rates, and mortality rates of young fish to evaluate the cost to benefit of alternative management of river flows. More research on white sturgeon life history and environmental tolerances (especially flow requirements at all life stages) may show that winter flow releases from dams would initiate additional spawning and alter substrate for improved survival of eggs and larvae, additional spring flows may improve downstream migration and survival of juveniles, and sustained high flows in the spring could also provide access to important floodplain habitats (e.g., Yolo bypass) for rearing and enhanced growth.

Potential use of hatcheries. White sturgeon aquaculture has been proven to be successful; therefore, there may be an inclination to use hatchery stocks to enhance the sturgeon fishery in California. However, dependence on hatcheries for either supplementing the sport fishery or meeting conservation and recovery objectives brings inherent risk and should not be prioritized over conservation and management measures intended to reverse declines of wild stocks. A long-term management and monitoring plan needs to be developed that includes management goals and genetic analyses to identify differences between wild and domesticated stocks. A principal goal should be to prevent domestication of wild stocks and to maintain maximum genetic and life history diversity. However, if populations become even more severely reduced, a conservation hatchery may be required. Proper use of wild broodstocks may serve to augment declining populations and allow time for conservation and restoration actions designed to improve spawning and rearing success, as well as adult and juvenile survivorship, to be implemented. In cases where hatchery- reared sturgeon have been used in conservation (e.g., Kootenai River, Idaho), a time lag of up to 3 years was necessary for hatchery-reared white sturgeon to adapt to natural conditions (Ireland et al. 2002). During that time, fish experienced decreased growth and populations exhibited 60-90% annual survival. If high survival rates to augment a population are important, hatchery-reared fish should be released after reaching 134 mm TL (~ 5 months old), because laboratory results suggest that fish of this size and larger are less vulnerable to predation (Gadomski and Parsley 2005c). All hatchery fish should be marked with coded wire tags so success of different management strategies can be evaluated.

Research. White sturgeon are well-studied but research is still needed to determine priorities for habitat restoration and best flow regimes to support successful reproduction and survivorship. There is also a continuing need for long-term monitoring of populations in order to develop population trends. Monitoring of tagged fish could help determine movement patterns, habitat utilization across life history stages, and potential interactions of Sacramento River white sturgeon with other populations. In particular, the role of the Klamath River in supporting the California white sturgeon population needs further study.

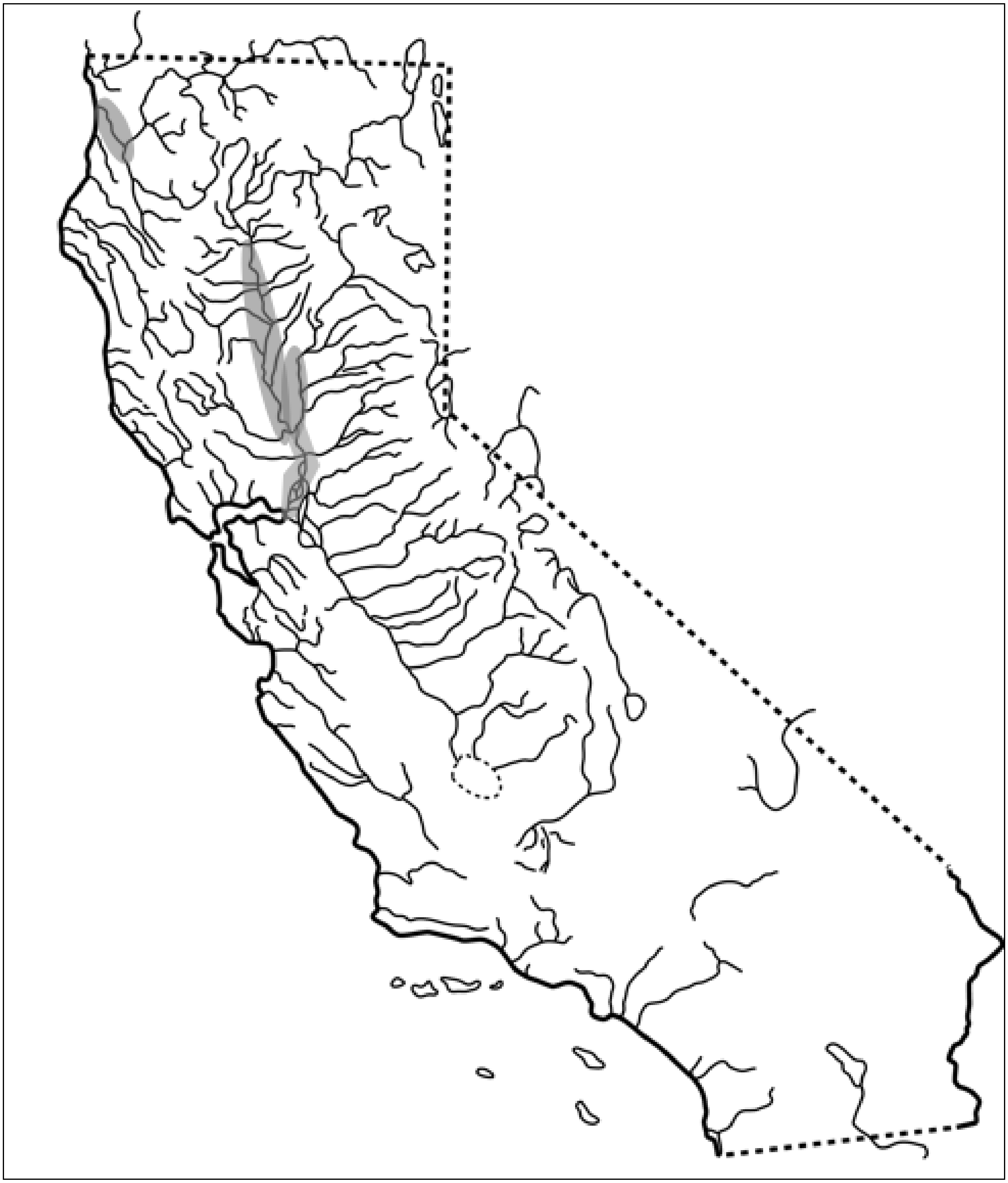


Figure 2. Distribution of white sturgeon, *Acipenser transmontanus* (Richardson), in California. Only freshwater distribution in the Sacramento and Klamath River basins is shown.

GOOSE LAKE TUI CHUB
Siphateles thalassinus thalassinus (Cope)

Status: Moderate Concern. Goose Lake tui chub remain numerous in Goose Lake and in the lower reaches of most large tributaries to the lake. However, Goose Lake dries out completely during periods of drought and the tui chub is particularly susceptible to periodic elimination of lake habitat, followed by great reductions in population size.

Description: The Goose Lake tui chub is differentiated from other *Siphateles* taxa by their longer fins, more posterior dorsal fin, longer head, and larger number of dorsal rays, usually nine (Snyder 1908b). Coloration is similar to Lahontan Lake tui chub, although larger specimens from Goose Lake (up to 30 cm FL) are uniformly silver except for a white belly. For a general description of tui chub see the Lahontan Lake tui chub account in this report.

Taxonomic Relationships: The Goose Lake tui chub was first described by E. D. Cope (1883) as *Myloleucus thalassinus*. He simultaneously described a second species of tui chub from the lake as well. Snyder (1908b) noted that Cope collected numerous dried chubs that had been dropped by fish-eating birds along the shoreline and hypothesized that the second species described by Cope was based on these poorly preserved specimens. However, there are apparently two morphological types of tui chub in Goose Lake: a "standard" heavy-bodied tui chub and another form with a less robust body and more pointed head (R. White and P. Moyle, unpubl. obs.). Snyder (1908b) placed *thalassinus* in the genus *Rutilus* because Jordan and Evermann (1896) synonymized *Myloleucus* with *Rutilus*. North American cyprinids placed in the European genus *Rutilus* eventually were referred to generic names of New World minnows, including *Gila*. Snyder (1908b) considered *thalassinus* to be native to Goose Lake and the upper Pit River from Big Valley upstream to Goose Lake. Hubbs et al. (1979), however, considered the form in the Pit River to be distinct from the Goose Lake form, although no evidence was provided. For reasons that are now obscure, Hubbs et al. (1979) used the specific name *thalassina* which was subsequently adopted by other workers; however, *thalassinus* (Cope 1883) has precedence and is used here.

In 2001, a genetic study using mitochondrial DNA found that tui chub in the Cow Head, Warner, and Goose Lake basins are closely related and are sufficiently genetically distinct from other tui chubs as to be recognized as a single species under the name *Siphateles thalassinus* (Harris 2000). Harris recognized two lineages within *S. thalassinus*, one in Goose Lake and the other in the Pluvial Lake Warner Basin, which includes both the Cow Head and Warner basins. Harris's findings supported Hubbs and Miller's (1948) postulation of a possible relationship between Cow Head tui chub and chubs from the lakes in Warner Valley, Oregon, because of the stream connection that existed between the Cow Head Basin and the Warner Valley drainage.

Chen et al. (2009) used microsatellite DNA to further resolve the taxonomy of tui chubs of the northwestern Great Basin. Chen's results supported Harris's systematics regarding the species status of *S. thalassinus.* Chen (2009) also found that tui chub populations of the upper Pit River drainage were genetically indistinguishable from those in Goose Lake and that these two populations, taken together, were sufficiently distinct to warrant subspecies status as *S. t. thalassinus.*

Rutter conducted the only known comparison of tui chub from above and below Pit River Falls and noted substantial differences in lateral line scale counts between the populations (Rutter 1908). However, both he and Snyder (1908b) considered tui chub populations in Goose Lake and the upper Pit River to be similar. Then, in 1979, without providing a rationale, Hubbs et al. listed Pit River and Hat Creek (tributary to the lower Pit River, below Pit River Falls) tui chub populations as discrete undescribed subspecies. No systematic work has been conducted on the lower Pit River tui chub populations since then, which means that, over a hundred years after Rutter published his findings, the relationship between upper and lower Pit River populations of tui chub remains unresolved.

For a general discussion of tui chub taxonomy, see the Lahontan Lake tui chub account in this report.

Life History: The life history of this subspecies has been little studied. Chubs commonly reach 250 mm FL in the lake and fish as large as 316 mm FL have been collected, indicating that this form may be very long-lived in lake habitats. In streams, however, they rarely exceed 120 mm FL. The size distribution of tui chubs sampled from Goose Lake in 1989 showed two modes. The great majority (>90%) of fish were less than 120 mm SL, while the remainder were 200-300 mm SL (R. White, USFWS, unpubl. data 1989). Most tui chubs are opportunistic omnivores and consume a wide variety of aquatic invertebrates (Moyle 2002). Tui chubs are a major prey base of Goose Lake lamprey; depending on the length class, 20-70% of the tui chubs >200 mm SL sampled in 1989 had lamprey scars (R. White, unpubl. data 1989).

Habitat Requirements: Goose Lake is a massive, natural alkaline lake covering approximately 39,000 surface hectares straddling the Oregon-California border. The lake is shallow, averaging 2.5 m deep, hyper-eutrophic and very turbid (Johnson et al. 1985). A thermocline (and hence temperature stratification and dissolved O_2) appear to be affected by wind conditions, as indicated by data from September, 2009 (R. White, unpubl. data 1989). On a calm September day, water temperature at one sampling locality was 17°C from the surface to 40 cm depth, with a sharp drop at 40-50 cm, and 14-15°C at 50-200 cm depths. At a second locality, temperature decreased from 23°C at the surface to 15°C at 35 cm, remaining at about 15°C between 35cm and 2.5 meter depths. At those two localities, dissolved oxygen concentration held at about 8-10 mg O_2 l^{-1} from the surface down through the water column, but dropped abruptly to <1 mg O_2 l^{-1} in deeper water, depending on locality. The drop in O_2 occurred at about 150 cm depth at one locality, and between 260-270 cm depths at the second locality. On a windy September day, the water temperature was 15˚C throughout the water column (surface to 185 cm depth) measured at one locality. Dissolved O_2 was constant (slightly <10 mg O_2 l^{-1}) from the surface to 170 cm depth, but dropped abruptly to <4 mg O_2 l^{-1} at about 175-180 cm.

The surface elevation of Goose Lake fluctuates seasonally, but averages 1,433 m. In California, no tui chubs have been found in streams above 1441 m in elevation, although tui chubs have been found above 1550 m in Oregon streams (J. Williams, unpubl. data). In streams, Goose Lake tui chub prefer pools and are generally not found in swift water, although they have been collected from runs in Battle Creek on the west shore of Goose Lake (J. Williams, unpubl. data). Goose Lake tui chubs have been collected in habitats with temperatures ranging from 9-29°C. In July, 1992, large numbers of chubs were observed in the lower reaches of Willow and Lassen creeks (G. Sato, pers. comm. 1993), where they may have been attempting to escape from the increasing alkalinity of the drying lake.

In Oregon streams, Scheerer et al. (2010) found tui chubs mainly in the lowermost reaches in low gradient, unforested stream channels and irrigation ditches, although a few tui chubs were also collected at higher elevation sites. The wide, silt-bottomed habitats were mainly associated with agricultural fields. The principal co-existing species in these agricultural reaches were alien species such as brown bullhead (*Ameiurus nebulosus*) and fathead minnow (*Pimephales promelas*).

Distribution: In addition to Goose Lake itself, *S. t. thalassinus* also occurs in low-elevation sections of streams tributary to the lake and in Everly Reservoir, Modoc County California, as well as in Cottonwood, Dog and Drews reservoirs in Oregon (Sato 1992a). In 2007, the Oregon Department of Fish and Wildlife collected relatively large numbers of tui chub from Dry, Drews, Dent, Thomas and Cox creeks on the Oregon side of the basin (Heck et al. 2008, Scheerer et al. 2010).

The Goose Lake basin is a disjunct subbasin of the upper Pit River. At extreme high water, Goose Lake spills into the North Fork Pit River as it did in 1868 and 1881. Since the late 19th century, storage and diversion for irrigation have substantially reduced the inflow to Goose Lake and future overflow of the lake into the Pit River is deemed unlikely (Phillips et al. 1971). However, because of this historical hydrologic connection, the fish faunas of Goose Lake and the upper Pit River share most taxa and tui chub populations from the two basins are genetically indistinguishable (Chen et al. 2009).

Reid et al. (2003) found tui chub in 7 of 12 sampling sites in the upper Pit River watershed, including the mainstem Pit River near Canby, the North Fork Pit River from the vicinity of Parker Creek down to the confluence with the South Fork Pit River, just below Alturas, and in the headwaters of the South Fork Pit River in Jess Valley.

Trends in Abundance: Goose Lake tui chub have been documented as extremely abundant in the lake. During 1966 gillnetting surveys of Goose Lake, tui chub comprised 88% of fishes collected (King and Hanson 1966). In 1984 it comprised nearly 96% of gillnet collections (J. Williams, unpubl. data) and, in 1989, it comprised 96% of fishes sampled by trawls, gillnets, and seines (R. White and P. Moyle, unpubl. data). Large numbers of chubs could be caught with relatively little sampling effort (e.g., 100+ in a 5-minute haul with a small trawl). In 1992, chubs were eliminated from the lake as it became progressively more shallow and alkaline and then dried. As lake levels dropped, fish crowded into the inflowing streams where they were extremely vulnerable to predation from white pelicans and other fish-eating birds. Apparently the tui chubs survived in greatly reduced numbers in stream pools and in some upstream reservoirs, but mainly in Oregon. Periodic drying of Goose Lake is a natural response to drought and the native fish assemblage evolved under these conditions. However, diversion of stream flows along with the effects of grazing, wetland reclamation and road construction have altered streams and riparian areas, reducing the extent of stream habitat that these fish rely on during periods of drought.

Nature and Degree of Threats: The principal threat to the Goose Lake tui chub is desiccation of its principal habitat, Goose Lake, accompanied by loss of refuge habitat in tributary streams and reservoirs in the drainage. This account does not include factors affecting poorly known Pit River populations, since the two populations are effectively disjunct; however, if the two regions are considered to have just one population, the Pit River may serve as a drought refuge, unless it is completely taken over by alien species. Tui chub populations may, however, persist in the presence of alien species: Big Sage Reservoir, on Rattlesnake Creek, a Pit River tributary, once supported a successful bass fishery, with a tui chub prey base (Kimsey and Bell 1955). See the Goose Lake sucker account in this report for further details.

Agriculture. Although the lake has dried historically, diversions for irrigation and loss of natural water storage areas (e.g., wet meadows) from agriculture and grazing presumably caused it to dry up more rapidly during the recent period of prolonged drought. Even in absence of complete drying of the lake, reduction of inflows increases the likelihood that the lake will periodically become too alkaline to support freshwater fishes such as tui chub. High alkalinity may be a particular problem for early life-history stages. The key to the survival of Goose Lake tui chubs, in the past, has likely been the presence of refuges in the springs and pools of the lower reaches of tributary streams. The same factors (agricultural diversions, road building, channel alterations) which affect lake inflow also negatively impact in-stream habitat, leaving tui chub few refuges during drought. It is likely that key refuge areas are mainly in Oregon, in the 'delta' marshy areas of Thomas Creek and other tributaries. Small reservoirs created for storage of irrigation water may also serve as refuges for tui chubs.

Grazing. Livestock grazing is, perhaps, the most pervasive land use in the Goose Lake basin. Lowland refuge habitats are degraded by stream erosion and bank destabilizations caused by livestock grazing in riparian areas, especially through the removal of woody riparian plants. While improved management of most grazed lands has reduced the threat of grazing in the short-run (e.g., in the Lassen Creek drainage), as the climate becomes warmer and more variable, there is considerable potential for negative impacts of grazing (and other land uses) to increase unless there is expanded use of riparian protection measures, such as exclusionary fencing.

Transportation. Virtually all streams used by Goose Lake tui chubs are crossed by roads, which often serve as sources of siltation or barriers to fish movement.

Alien species. Goose Lake tui chubs manage to coexist with a variety of alien species, mainly in highly disturbed habitats such as irrigation ditches and reservoirs (Scheerer et al. 2010). However, predation by alien fishes should be considered in management. Education and enforcement are important tools to prevent further illegal introductions of non-native species.

	Rating	Explanation
Major dams	n/a	Impacts may exist in Oregon
Agriculture	High	Diversion of water significantly impacts stream habitat and the frequency/duration of Goose Lake desiccation
Grazing	Medium	Grazing continues to impact stream and riparian habitats
Rural Residential	Low	Relatively little residential water use in comparison to agricultural use in native range
Urbanization	n/a	
Instream mining	n/a	
Mining	n/a	
Transportation	Low	Roads cross all major Goose Lake tributaries
Logging	Low	Widespread in watershed but not intense
Fire	Low	Entire watershed prone to forest and range fires
Estuary alteration	n/a	
Recreation	n/a	
Harvest	Low	Used as bait but practice has been made illegal (article 3, Section 4.30 of CA freshwater sport fishing regulations)
Hatcheries	n/a	
Alien species	Medium	Alien species present a potential threat in drought refuges, particularly in reservoirs

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

Effects of Climate Change: Goose Lake is located at the edge of the arid Great Basin, where relatively rare aquatic habitats are often tapped for human use. Any reduction in precipitation or increased frequency of drought is likely to further stress aquatic habitats in this basin. Snow melt and winter rains, the principle sources of water in the Goose Lake watershed, are likely to substantially decrease as the climate warms (Moyle et al. 2012). During low flow periods, lower streams reaches in the basin currently reach extreme temperatures (24-26°C). Thus an increase in air temperature, especially when combined with reductions in stream flow through diversions, could prove lethal to native fish populations. An increase in fire frequency or intensity in this dry area could also decrease riparian shading, add sediment, and otherwise alter the refuge stream habitats that tui chub depend on during drought. See the Goose Lake sucker account in this report for a more detailed description of climate change effects in the basin. Moyle et al. (2013) consider the Goose Lake tui chub to be "highly vulnerable" to extinction in California because of climate change, but considered the chub to be confined to the Goose Lake basin. If the limited populations in the upper Pit drainage are, indeed, part of this subspecies, the chub may have greater resistance to climate change.

Status Determination Score = 3.1 – Moderate Concern (see Methods section, Table 2). The limited distribution of Goose Lake tui chub in California and its vulnerability to extended drought merit its inclusion as a species of special concern. The Goose Lake tui chub is a US Forest Service and Oregon Department of Fish and Wildlife "Sensitive Species". The American Fisheries Society considers the Goose Lake tui chub to be "threatened" (Jelks et al. 2008), while NatureServe ranks it as "imperiled" (T2). Presumably, the tui chub develops large populations when Goose Lake is full but may drop to low numbers in isolated populations when the lake dries. These same factors make it particularly susceptible to climate change.

Metric	Score	Justification
Area occupied	2	Restricted to Goose Lake and, possibly, upper Pit River basins
Estimated adult abundance	5	Robust populations when lake is full but drought can cause substantial population reductions
Intervention dependence	4	Stream refuge habitats during times of drought are impacted by agricultural water use
Tolerance	4	Tolerant of extreme DO, temperature and alkalinity levels
Genetic risk	4	Little genetic risk
Climate change	1	Goose Lake is likely to be dry more often as climate becomes more arid
Anthropogenic threats	2	See Table 1
Average	3.1	22/7
Certainty (1-4)	3	

Table 2. Metrics for determining the status of Goose Lake tui chub in California, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: The Goose Lake Fishes Working Group was formed with representatives from federal and state agencies, as well as private individuals with interest in the lake, to explore management measures for all native fishes in the basin (Sato 1992a). The involvement of private landowners is particularly critical because many key refuge habitats occur on private land. The persistence of Goose Lake tui chub in the Goose Lake Basin will require active cooperation between Oregon and California because it is likely that most (if not all) natural drought refuges for tui chubs in the Goose Lake basin are in Oregon. Possible management actions include:

1. Determine the suitability of all reservoirs in the drainage as refuges for native fishes and negotiate, if necessary, for minimum pools during periods of drought. Special attention needs to be paid to potential refuges in California.
2. Identify and implement restoration projects to benefit native fishes in the lower reaches of Goose Lake tributaries in both Oregon and California.
3. Actively enforce the prohibition of use of live baitfish and introduction of nonnative fishes into Goose Lake basin, including Oregon. Where possible, eradicate existing populations of alien fishes in ponds and streams.
4. Establish instream flow protections for larger streams in the basin (Oregon: Thomas, Drews, and Dry creeks; California: Lassen and Willow creeks) to ensure adequate flows are present in lower stream reaches to maintain refuge areas and lake level during periods of drought.
5. Conduct a thorough study of the Goose Lake ecosystem, including a study of the distribution and habitat requirements of tui chubs and a systematic survey of the invertebrates present, expanding on studies in Oregon (Heck et al. 2008, Scheerer et al. 2010).
6. Investigate life history and habitat requirements of Goose Lake tui chub to determine what additional species-specific management measures are required.
7. Determine the systematic relationships among tui chubs in Goose Lake and the upper and lower Pit River.

Figure 1. Distribution of Goose Lake tui chub, *Siphateles thalassinus thalassinus* (Cope), in California. Distribution in the Pit River system is uncertain.

COW HEAD TUI CHUB
***Siphateles thalassinus vaccaceps* (Bills and Bond)**

Status: High Concern. Because of its extremely small range and level of human alteration to habitats within that range, the Cow Head tui chub is vulnerable to both human-induced and natural perturbation, especially during periods of severe drought.

Description: The Cow Head tui chub (CHTC) is similar to the Klamath tui chub, *Siphateles bicolor bicolor*, but is differentiated primarily on the basis of more gill rakers (Bills and Bond 1980). The CHTC has 19-25 (mean = 22.5) short, "bluntly rounded" gill rakers, compared with 10-15 gill rakers in *S. b. bicolor*. Other morphological features that characterize this subspecies are: the head is not as deep as in other chubs, is relatively longer, and is convex in profile with a rounded interorbital; a nuchal hump is present, but low; the lower jaw is not overhung by the upper jaw; and the caudal peduncle is relatively deep. Predorsal scales number from 26-35 (mean = 31) and there are approximately 57 lateral line scales. The pectoral fin has 15-17 rays and the pelvic fin has 8-9 rays. Pharyngeal tooth counts are 0,5-4,0; 0,4-4,0; 0,5-5,0. Coloration is similar to other subspecies, except there is a dark lateral stripe with speckles on the head region, especially the cheek and operculum, and on the lower body. Reproductive males and females develop breeding tubercles, especially on the anterior rays of the pectoral fins. Smaller tubercles develop in rows on the edges of the breast scales. In males, tubercles also develop on the scales above the pectorals and across the nape. The largest CHTC on record is 235 mm (Scoppettone and Rissler 2003).

Taxonomic Relationships: The CHTC was first recognized as a distinct form by Hubbs and Miller (1948) and was formally described by Bills and Bond (1980). A genetic study using mitochondrial DNA found that tui chub populations in the Cow Head, Warner and Goose lake basins were closely related and were genetically distinct from other tui chubs, meriting recognition as a single species under the name *Siphateles thalassinus* (Harris 2000). Harris recognized two lineages within *S. thalassinus*, one in Goose Lake and the other in Pluvial Lake Warner, which includes both the Cow Head and Warner basins. Harris's findings supported Hubbs and Miller's (1948) postulation of a possible relationship between CHTC and chubs from the lakes in Warner Valley, Oregon, because of the connection that exists between the Cow Head Basin and the Warner Valley drainage (see Distribution section below). Bills and Bond (1980) had disputed this hypothesis on the basis of differences in gill-raker length and fin and head shapes between the two populations. In 2007, a study using microsatellite DNA allowed greater resolution of the taxonomy of the tui chub of the northwestern Great Basin (Chen et al. 2009). Chen's results supported Harris' systematics regarding *S. thalassinus* and also found that the CHTC was sufficiently distinct to warrant subspecies status as *S. t. vaccaceps*. For a more detailed discussion of tui chub taxonomy, see the Lahontan Lake tui chub, *S. b. pectinifer*, account in this report.

Moyle et al. (1995) and Moyle (2002) list the common name of the chub as "Cowhead Lake tui chub" but Reid (2007) indicated that Cow Head tui chub is more accurate (the chub mostly does not live in the lake) and more consistent with the geographic name.

Life History: Cow Head tui chubs grow to 40-50 mm SL during their first year and 60-80 mm SL during their second year (Moyle unpublished data). By five years of age they reach an average of 100 mm SL, with larger individuals uncommon. The largest individual captured was 235 mm SL and over ten years old (Scoppettone and Rissler 2003). Most tui chubs spawn from late April to early July, beginning in their second to fourth year (Moyle 2002). Although there is little specific information on the reproductive behavior of CHTC, it is believed that they first spawn at two or three years of age (Reid 2006). Fecundity is relatively high, and a female of 100 mm produces approximately 4,000 eggs, which she lays over a series of spawning events. Like other tui chubs, CHTC presumably spawn in groups over aquatic vegetation, algae covered rocks, or gravel with several males attending to each female. Eggs adhere to plants or to substrates. Embryos hatch in 3-6 days and larvae begin feeding soon after hatching (Moyle 2002).

Tui chubs are generally opportunistic omnivores and feed on invertebrates (i.e. snails, clams, insects, and crustaceans), algae and other plant material, and small fish associated with the benthos or aquatic plants (Moyle 2002). Scoppettone and Rissler (2003) examined the stomach contents of 64 CHTC from various sites. Aquatic insects accounted for 28% of the total food by volume, while terrestrial insects accounted for 20%, and algae formed 31%. A single stomach contained an unidentified fish. Unidentifiable animal remains (presumably invertebrates) formed the remaining 19 % of total volume.

Habitat Requirements: Having evolved in the arid Great Basin, tui chubs like CHTC are highly tolerant of high alkalinity, turbidity, high temperatures and low levels of dissolved oxygen (Castleberry and Cech 1986, Moyle 2002, Reid 2006). The most generalized characteristics of suitable CHTC habitat are quiet water with abundant aquatic plants and bottom substrates of sand or finer materials. Thus, CHTC typically occupy pool areas in streams and open water channels with dense beds of aquatic vegetation (Sato 1992b, 1993a, Homuth 2000, Scoppettone and Rissler 2003, 2006).

Distribution: The range of CHTC is limited to the Cow Head Basin in extreme northeastern California and northwestern Nevada (Reid 2006). The Cow Head Basin is relatively small (25,700 acres) and drains north into the Warner Basin of Oregon through Cow Head Slough and Twelve Mile Creek. Cow Head Slough is a small, muddy creek. Under summer water conditions, the creek consists of a series of pools (95%) and riffles (5%) and meanders through a lava canyon approximately 50 m wide. The pools are fairly large, approximately 50 m^2, and are interconnected by shallow trickles. Landownership in the Cow Head Basin is both private and Federal (U.S. Bureau of Land Management (BLM)), but most perennial CHTC habitat is on private land (Reid 2006).

Historically, the basin contained a shallow, marshy lake during wet climate periods. However, Cow Head Lake was altered in the 1930s to allow seasonal drainage of the lake to facilitate farming of the lakebed during spring, summer and fall. The lake still fills during winter in high precipitation years but is drained by active pumping in spring. Populations of CHTC occupy all principal low gradient streams in the basin (Cow Head Slough and Barrel, West Barrel and Keno creeks) and a relatively large population still exists in the permanent channels that drain the lake bed (Scoppettone and Rissler 2006).

Recent surveys have identified seven areas of occupied perennial habitat in five sub-drainages within the Cow Head Basin. Each area is seasonally isolated and is maintained by separate springs or creeks and each contains a population of 1,000-10,000 individuals of all age classes (Reid 2006). During wet periods, stream populations of chubs expand throughout most of the low gradient stream habitat in the basin. Connectivity between stream populations of chubs is generally unobstructed during springtime flows but, as summer progresses and streams dry, all populations become restricted to isolated perennial pools (Reid 2006). Recent genetic research indicates that the genetic variability of CHTC is appropriate for a stream resident population (Chen 2006).

Trends in Abundance: In 1998, when CHTC were proposed for listing as a federally threatened species (see Status), they were only known to occur in Cow Head Slough and Pump Canal (Reid 2006). The only population estimates available at the time were qualitative and based on limited sampling with minnow traps and dip nets (Sato 1992b, 1993a-b, Olson 1997). In 1999, a limited sampling program was conducted with minnow traps in the southern BLM portion of Cow Head Slough and estimated 108 CHTC (39-113 mm FL) were present in this reach (Richey 1999). However, this survey was limited to BLM land that composes only a small portion of the habitat available.

Population estimates conducted in August, 2002 found approximately 3 km of occupied habitat in Barrel Creek and 4 km in Cow head Slough, with a combined population of several thousand chubs over 40 mm (Scoppettone and Rissler 2003). The largest single population was found in the Pump Canal. Although no rigorous population analysis was conducted, four small seine hauls spaced at 200 m intervals produced 936 chubs (22-148 mm) in 2001. Even considering the sampling limitations, if these results were expanded out to the full kilometer of available perennial habitat, a very rough Pump Canal population estimate would exceed 20,000 fish (Reid 2006).

Nature and Degree of Threats: Cow Head tui chubs exist in a small, isolated basin where native aquatic habitats and stream and lake hydrology have been highly altered by human activities, especially agriculture and grazing.

Agriculture. The main threat to the continued existence of CHTC is water diversion from Cow Head Slough for pasture, especially during periods of drought. For example, in 1992, the chubs were largely confined to a short section of slough that was entirely on private land with a water supply that depended, in part, on inflow from an irrigation ditch. The Cow Head lakebed has been used for production crop agriculture in the past and may be utilized as such in the future. Such a transition from ranching to tilled agriculture could have direct impact on water allocation in the Cow Head Basin. Pest control programs that introduce pesticides into the drainage (e.g., USDA-APHIS Grasshopper Control Program) are also a potential threat, although this issue has not been studied in the Cow Head Basin.

Grazing. Grazing in the area has removed riparian vegetation, reducing cover available to fish, making them more vulnerable to predation. Natural predators include garter snakes and fish-eating birds, both of which prey on juveniles and adults, and aquatic insects which prey on eggs, larvae, and juveniles (Reid 2006).

Alien species. While no alien species apparently exist in the watershed at the present time, an illegal introduction of other fish species could easily happen (as has occurred in many other equally isolated parts of the state) and has the potential to threaten the subspecies with rapid extinction due to its limited range.

	Rating	Explanation
Major dams	n/a	
Agriculture	High	Agriculture has degraded habitats and can divert large amounts of water
Grazing	High	Almost all habitat is impacted by grazing
Rural residential	Low	Low population densities and relatively little residential pressure on water supplies
Urbanization	n/a	
Instream mining	n/a	
Mining	n/a	
Transportation	Low	No known impact, but roads run along or cross much of CHTC habitat
Logging	Low	No known impact but may accelerate sedimentation
Fire	Low	No known impact but fires common in desert regions
Estuary alteration	n/a	
Recreation	n/a	
Harvest	n/a	
Hatcheries	n/a	
Alien species	Medium	Although there are no alien species in the watershed at present, illegally introduced species could rapidly deplete populations

Table 1. Major anthropogenic factors limiting, or potentially limiting, viability of populations of Cow Head tui chub in California. Factors were rated on a five-level ordinal scale where a factor rated "critical" could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated "high" could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction unlikely as a result; and a factor rated "no" has no known negative impact to the taxon under consideration. Certainty of these judgments is high. See methods section for descriptions of the factors and explanation of the rating protocol.

Effects of Climate Change: Snow melt and spring recharge from winter rains, the principle sources of water for all CHTC habitat, are likely to substantially decrease as the climate warms, and standard climate models indicate water temperatures are likely to increase 2-4 degrees C by the end of the century. Increased human demand for water is also likely, given the limited supply in the basin and the increased likelihood of long-term drought. Moyle et al. (2013) rated the CHTC as highly vulnerable to extinction from climate change because of its limited habitat in an area that is already very dry and hot, conditions likely to exacerbated by climate change.

Status Determination Score = 2.4 - High Concern (see Methods section Table 2). The CHTC was proposed for federal listing as a threatened species in 1998 but the petition was withdrawn after a conservation action plan was established and new sampling revealed a much larger population than previously known. However, because of the extremely small range and level of human alteration within that range, the CHTC is still vulnerable to both human-induced and natural changes to its habitats. Its status should be re-evaluated every five years or annually during periods of severe drought. The CHTC is listed by the American Fisheries Society as “Endangered” (Jelks et al. 2008) and by NatureServe as “Imperiled”.

Metric	Score	Justification
Area occupied	1	Limited to a single, small basin
Estimated adult abundance	4	Relatively large, but variable populations in pump canals with five other smaller populations in perennial habitats
Intervention dependence	3	The largest population lives in an artificial ditch, so management of this habitat is crucial for survival
Tolerance	3	Tolerant of wide range of environmental conditions but, during drought, tolerances may be exceeded
Genetic risk	2	Isolated population with little or no gene flow
Climate change	2	Snow melt and spring recharge for all habitats are likely to decrease
Anthropogenic threats	2	See Table 1
Average	2.4	17/7
Certainty (1-4)	4	Good recent data generated from ESA listing studies

Table 2. Metrics for determining the status of Cow Head tui chub in California, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: On October 22, 1999, stakeholders in the Cow Head watershed signed a conservation agreement (CA) and conservation strategy (CS), with the stated purpose of ensuring the long-term survival of the CHTC (USFWS 1999). Signatories included the US Fish and Wildlife Service, private landowners of Cow Head Lake, Cow Head Slough and the California reach of Barrel Creek (four owners, all CA signatories), principal permittees on BLM lands within the drainage, California and Modoc County Cattlemen’s Associations, the California Farm Bureau Federation, the U.S. Bureau of Land Management (BLM - Surprise Field Office), and California Department of Fish and Wildlife (CDFW). The two owners on West Barrel and the single owner for perennial reaches of Barrel and Keno creeks (Nevada) were not original signatories to the CA, because these populations were not recognized at the time;

however, they have been collaborative in providing access to meet the needs of the Conservation Strategy (Reid 2006).

Management directives laid out under phase 2 of the Conservation Agreement and Strategy, which must be implemented, are as follows:

- Create more stable habitat for populations downstream of the Pump Canal.
- Provide greater stability for the chub population upstream of the pump canal by creating, to the extent feasible, additional habitat in the area of historic Cow Head Lake.
- Monitor, as appropriate, the status of chub populations and effectiveness of conservation actions.
- Establish a monitoring program, whereby chub populations are sampled at least once a year.

In addition:

- A study of the environmental requirements of CHTC is needed.
- Slough reaches on public lands should be designated as Areas of Critical Environmental Concern and methods and locations to establish a permanent refuge for the CHTC on public land should be identified.
- Cow Head slough should be fenced to reduce or eliminate cattle grazing in riparian areas.

Figure 1. Distribution of Cow Head tui chub, *Siphateles thalassinus vaccaceps*, in California.

LAHONTAN LAKE TUI CHUB
Siphateles bicolor pectinifer (Snyder)

Status: High Concern. The only verified population in California occurs in Lake Tahoe. This population is declining due to lake changes associated with intense human use of the Lake Tahoe Basin. If tui chub in nearby Prosser, Boca and Stampede reservoirs are confirmed to be *S. b. pectinifer,* then the threats facing this taxa are significantly diminished.

Description: Lahontan Lake tui chubs can reach lengths of 35 to 41 cm FL. The mouth is small, terminal, and oblique. Pharyngeal teeth occur in a single row (5-5, 5-4, or 4-4) and are hooked, with narrow grinding surfaces. This subspecies is characterized by numerous (29-40) long, slender gill rakers; this is the primary characteristic that serves to differentiate it from sympatric *S. b. obesa* (Miller 1951, Vigg 1985, Moyle 2002). The inter-gill raker distances are usually less than the width of the gill rakers themselves. Other morphological characteristics that differentiate *pectinifer* from *obesa* are the more oblique mouth, the slightly concave profile of the head, and uniform blackish or silvery body coloration (Miller 1951). Dorsal and anal fin rays usually number 8, but may range from 7-9; fins are short and rounded. Scales are large, with 44-60 along the lateral line. Spawning males have reddish fins and develop small, white breeding tubercles on their body surfaces; females have reddish fins, slightly enlarged anal regions, protruding genital papilla, and deeper bodies.

Taxonomic Relationships: The systematics of tui chubs are confounded by the fact that many populations appear morphologically similar but are genetically divergent. Distinctive populations occur in the many isolated drainages of the Great Basin, while large lake populations have two sympatric morphs – a pelagic form with many fine gill rakers and a benthic form with fewer, coarser gill rakers. Incomplete meristic and genetic studies add to the taxonomic confusion.

Prominent ichthyologists who have studied the native fishes of the Great Basin have had differing opinions about *S. b. pectinifer*'s taxonomy. Widely varying opinions range from "no valid standing as a taxonomic unit" (La Rivers 1962, p. 420) to assignment of its own genus by J. O. Snyder (1917). Consequently, La Rivers (1962) considered *S. b. pectinifer* to have the most complex taxonomic history of any member of the Great Basin fish fauna. It was first described as *Leucidius pectinifer* by Snyder (1917) who simultaneously described the sympatric 'stream' form as *Siphateles obesus;* the morphological differences between these two forms were great enough for Snyder to place *obesa* and *pectinifer* in different genera. Hubbs and Miller (1943) considered *L. pectinifer* to be a subspecies of *Siphateles obesus* and, thus, called it *Siphateles obesus pectinifer*. Shapovalov and Dill (1950) recognized that both forms were part of the *Siphateles bicolor* complex and renamed them *S. b. pectinifer* and *S. b. obesus*, respectively. Bailey and Uyeno (1964) designated *Siphateles* as a subgenus of *Gila* and designated the fine gill raker tui chub as *Gila bicolor pectinifer*. However, biochemical evidence suggests that tui chubs are more closely related to other Californian minnows than they are to other species of *Gila* (Simons and Mayden 1998). In light of this evidence, Moyle (2002) resurrected the generic name *Siphateles,* first used by Cope (1883) and then by Snyder (1918).

Presently, there are ten *Siphateles* taxa recognized in California (Moyle 2002), although three lack formal taxonomic descriptions: Lahontan Lake tui chub (*Siphateles bicolor pectinifer*), Eagle Lake tui chub, (*S. b. ssp*.), Goose Lake tui chub (*S. t. thalassinus*), Cow Head tui chub (*S. thalassinus vaccaceps*), High Rock Springs tui chub (*S. b. ssp*.), Owens tui chub (*S. b. snyderi*), Mohave tui chub (*S. mohavensis*), Lahontan Creek tui chub (*S. b. obesa*), Klamath tui chub (*S. b. bicolor*), and Pit River tui chub (S. *b. ssp.*). The first four subspecies are included in this report, while the Owens and Mohave tui chubs are already listed as endangered species by both state and federal governments. The Pit River tui chub was listed by Hubbs et al. (1979) as an undescribed subspecies. The tui chubs of the upper Pit River are now considered to be part of the Goose Lake population (Chen et al. 2009) but questions remain about taxonomic affinities of tui chubs distributed in the lower Pit River basin. The High Rock Springs tui chub is extinct.

Recent genetic studies have shown that considerable variation exists among populations of tui chubs, all of which were formerly classified as subspecies of *S. bicolor* (Harris 2000, Chen et al. 2007, Chen et al. 2009). Hence, the subspecific status of *S. b. pectinifer* remains controversial. Not only is the zoogeographic range of *S. b. pectinifer* contained within that of *S. b. obesa,* but Harris (2000) suggested that *S. b. obesa* should be elevated to species status and that *S. b. pectinifer* be submerged within it.

Conversely, studies in both Lake Tahoe and Pyramid Lake, Nevada, indicate that the two forms segregate ecologically (Miller 1951, Galat and Vucinich 1983) and do not interbreed. The existence of sympatric, morphologically distinct tui chub morphs has been repeatedly and consistently observed in large lakes throughout the range of *Siphateles*, most famously in Pyramid Lake and Lake Tahoe but also in Walker Lake, Goose Lake, Eagle Lake and Honey Lake, among others. The main character distinguishing the morphs is number and morphology of gill rakers, although only in Pyramid Lake and Lake Tahoe are the two morphs clearly separated.

It is possible that the distinctive fine gill raker form of tui chub has arisen multiple times in each of these large lake systems, although it may be just a single lineage in the Truckee basin. Similar situations of parallel evolution in California fish taxa may exist, such as the run timing of summer steelhead populations and bony plate development and migratory behavior of threespine stickleback in coastal California streams. A sizeable literature base has developed on trophic polymorphism; of particular relevance to lake dwelling tui chub are trophic polymorphisms among other fishes in lacustrine environments. Examples include char in arctic lakes, whitefish in Canadian and Idaho lakes, cichlids in African Rift lakes, threespine stickleback in British Columbia lakes and sunfishes in the eastern United States. References can be found compiled in reviews on the subject by Smith and Skülason (1996) and, more recently, by Dayan and Simberloff (2005). Until taxonomic studies are completed, all distinctive populations of tui chubs should be managed as separate taxa.

Life History: Lahontan Lake tui chub feed mostly on zooplankton, especially cladocerans and copepods, but also consume benthic insects such as chironomid larvae, annelid worms and winged insects such as ants and beetles (Miller 1951, Marrin and

Erman 1982). They are primarily mid-water feeders, with gill-raker structure adapted to feeding on plankton. In contrast, the co-occurring *obesus* form is primarily a benthic feeder (Miller 1951). A comparison of stomach contents of both subspecies captured together in bottom-set gill nets indicated *obesa* had fed on benthic insects such as chironomids and trichopterans, while *pectinifer* had fed on planktonic microcrustacea (Miller 1951). There is no significant ontogenetic niche shift in diet for *pectinifer*; it feeds on plankton throughout its life (Miller 1951). In Pyramid Lake, both types of tui chubs feed primarily on zooplankton (mostly microcrustaceans) when less than 25 mm FL, but the *obesa* subspecies feed increasingly on benthic and terrestrial macroinvertebrates as they become larger (Galat and Vucinich 1983). There is an ontogenetic change in gill-raker numbers in the two forms that accompanies the differentiation of diets. When less than 25 mm FL, the two morphs are indistinguishable, even based on gill-raker counts, but the gill-raker number increases in *pectinifer* with size until the two forms are readily distinguishable at ≥50 mm FL.

Tui chubs are preyed upon by large trout and, to a lesser extent, by birds and snakes. Examination of stomachs of rainbow trout and lake trout in Lake Tahoe revealed that 10% and 7%, respectively, of their stomach contents consisted of tui chubs (Miller 1951).

In Lake Tahoe, spawning apparently occurs at night during May and June and possibly later (Miller 1951). By early August, females do not have mature ova. Lahontan Lake tui chubs spawn by 11 cm SL (Miller 1951). They are probably serial spawners, capable of reproducing several times during a season (Moyle 2002). Snyder (1917) documented that reproductive adults spawned in near-shore shallow areas over beds of aquatic vegetation and found fertilized eggs adhering to the aquatic vegetation. He noted that young remained in the near-shore environment until winter when they were 1-2 cm in length and then migrated into deeper water offshore.

Growth (length increments) of tui chubs is linear until about age 4, when weight increases more rapidly and length increments decrease. The largest Lahontan Lake tui chub caught in Lake Tahoe was 13.7 cm SL (Miller 1951). These fish are considerably smaller than the tui chubs in Walker Lake, Nevada, where they grow to 21 cm SL (Miller 1951). It is likely that the largest Lahontan Lake tui chubs are in excess of 30 years old (Scoppetone 1988, Crain and Corcoran 2000).

Habitat Requirements: Lahontan Lake tui chub are schooling fish that inhabit large, deep lakes (Moyle 2002). They seem to be able to tolerate a wide range of physicochemical water conditions based on the fact that they are found in oligotrophic Lake Tahoe as well as in Pyramid Lake, a mesotrophic and highly alkaline lake. In Lake Tahoe, the larger fish (>16 cm TL) exhibit a diel horizontal migration by moving into deeper water (>50 m) during the day and back into shallower habitat at night (Miller 1951). However, they always remain high in the water column. Smaller individuals occupy shallower water. Additionally, there is a seasonal vertical migration, with fishes located deeper in the water column during winter and moving back into the upper water column during summer (Snyder 1917, Miller 1951). Algal beds in shallow, inshore, areas appear to be necessary for successful spawning, embryo hatching and larval survival.

Distribution: Lahontan Lake tui chubs are found in Lake Tahoe and Pyramid Lake, Nevada, which are connected to each other by the Truckee River, and in nearby Walker Lake, Nevada. Plankton-feeding populations of chubs in Stampede, Boca, and Prosser reservoirs on the Truckee and Little Truckee rivers may also be Lahontan Lake tui chubs because they have a superior oblique mouth and fine gill rakers and are never found in tributary streams (Marrin and Erman 1982, D. Erman, pers. comm.). Other tui chub populations in the Lahontan basin of uncertain taxonomic affinity also occur in Topaz Lake on the California-Nevada border and in Honey Lake, Lassen County.

Trends in Abundance: Actual abundance is not known, but is likely quite small compared to historic numbers. The Lake Tahoe population is the only confirmed population in California, but the chubs in Stampede, Boca, and Prosser reservoirs may also belong to this subspecies, although no sampling or analysis has been carried out to verify this assertion. Only small numbers have been collected from Lake Tahoe in recent years (P. Budry, Utah State University, unpubl. data) and the Lahontan Lake tui chub has not been studied in Lake Tahoe since the late 1940s (Miller 1951). In the intervening years, the zooplankton community in the lake has changed dramatically. *Daphnia*, which are an important prey of adult chubs, have been nearly eliminated (Richards et al. 1975) by introduced kokanee salmon (*Oncorhynchus nerka*) and opossum shrimp (*Mysis relicta*), both of which feed on zooplankton.

Putative *S. b. pectinifer* populations in the three California reservoirs mentioned above and verified *S. b. pectinifer* populations in Pyramid and Walker lakes in Nevada are large but abundance estimates are lacking.

Nature and Degree of Threats: Until the taxonomy of peripheral populations has been decided, the future of Lahontan Lake tui chubs in California essentially depends on their ability to persist in Lake Tahoe (Table 1).

Major dams. Dams on California tributaries to the Truckee River are apparently a mixed blessing for lake tui chubs. They allow for diversion of water, lowering the level of Pyramid Lake, Nevada and potentially negatively affecting tui chubs there, while creating potential habitat in their reservoirs (additional habitats within California). The reservoir populations are unstudied, however, and may not be *S. b. pectinifer*.

Urbanization and rural development. Water diversion, waste water treatment, wetlands destruction and increased sedimentation from ever increasing development in the Lake Tahoe Basin have altered the lake's physical environment; however, it is unknown how these stressors affect tui chubs. Lake Tahoe has been undergoing physical and chemical change as the result of nutrients, sediments and pollutants entering the lake from surrounding development, as well as more distant sources. Shoreline development has presumably also negatively affected tui chubs because they spawn in shallow water and larvae may require warm habitats with adequate cover for the first few weeks of life (although this is not known). There is some indication that the marsh that is now the development called Tahoe Keys (a major source of alien species in the lake) was once an important rearing area for tui chubs (Miller 1951).

Logging. The Tahoe Basin has been heavily logged in the past and some logging continues, contributing to sediment delivery. Effects on tui chubs are likely minimal, especially when compared to other factors changing the lake.

Fire. The entire Tahoe Basin is increasingly prone to catastrophic fire which may, in turn, deliver huge sediment loads to the lake. This may affect tui chub spawning and feeding and generally change the nature of Lake Tahoe, especially as climate change effects are predicted to increase the frequency and intensity of fire in this region.

Recreation. The Lake Tahoe region is a year-round recreation destination and the increasing influx of permanent residents and visitors drives most of the changes that affect fishes and other organisms in the lake, from water chemistry (e.g. via air pollution) to sedimentation and increasing eutrophication (e.g., surface run off of nutrients and pollutants from ski resorts, casinos, golf courses, recreational parks and trail development).

Alien species. The greatest impacts to the aquatic ecosystem of Lake Tahoe have been the result of introductions of non-native fishes and invertebrates. Mysid shrimp and kokanee salmon have largely eliminated *Daphnia*, which were the major food source of tui chubs, while introduced lake trout (*Salvelinus namaycush*), rainbow trout (*O. mykiss*), and brown trout (*Salmo trutta*) prey on them. . In recent years, the invasions of predatory smallmouth bass (*Microterus dolomieui*) and largemouth bass (*M. salmoides*) into the lake constitute an additional threat to the tui chub population, especially since these predatory centrarchids occupy chub spawning and rearing habitats. As the lake becomes more eutrophic, it may actually be able to support more fish, including tui chubs, but the number and abundance of alien species will also likely increase. In contrast, the alkalinity of Pyramid Lake, Nevada, has largely prevented the establishment of non-native species, with the exception of Sacramento perch (*Archoplites interruptus*). Adult perch (<300 mm) feed largely on tui chubs (Galat et al. 1981).

	Rating	Explanation
Major dams	Medium	Reservoirs may have created habitat but they also reduce freshwater flow into Pyramid Lake
Agriculture	n/a	
Grazing	Low	Grazing occurs in the Tahoe Basin which may contribute to changes in water quality
Rural residential	Medium	Water diversion, waste water treatment, wetlands destruction and increased sedimentation in the Tahoe Basin have changed the lake's physical environment; direct impacts to tui chubs are unknown
Urbanization	Medium	Same as above
Instream mining	Low	No known effect
Mining	Low	Legacy effects are largely unstudied
Transportation	Low	A large portion of the suspended sediment in Lake Tahoe has its origins in sand applied to de-ice roads
Logging	Low	Logging contributes sediment delivery to the lake, with much greater impacts in the past
Fire	Low	The entire Tahoe basin is increasingly prone to catastrophic fire; direct impacts to tui chubs are likely to be minimal
Estuary alteration	n/a	
Recreation	Medium	Recreational use of the Tahoe Basin is the primary force driving the area's rapid development
Harvest	n/a	
Hatcheries	n/a	
Alien species	High	Long-term impacts from introduced predators and competitors may be reducing populations

Table 1. Major anthropogenic factors limiting, or potentially limiting, viability of populations of Lahontan Lake tui chub in California. Factors were rated on a five-level ordinal scale where a factor rated "critical" could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated "high" could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction unlikely as a result; and a factor rated "no" has no known negative impact to the taxon under consideration. Certainty of these judgments is moderate. See methods section for descriptions of the factors and explanation of the rating protocol.

Effects of Climate Change: The following list includes the predicted impacts and potential consequences of climate change to Lake Tahoe and the northern Sierra Nevada:

- A shift in winter precipitation from snow to rain. This shift in annual hydrologic timing could increase the transport of fine sediment and nutrients to the lake.
- A shift toward earlier snowmelt (Dettinger and Cayan, 1995; Cayan et al, 2001; Stewart et al., 2005). A change to the volume, temperature and timing of streamflow into the lake could increase the lake's thermal stability and could

possibly prolong the residence time of fine sediment near the lake surface, further decreasing water transparency.

- An increase in the average temperature of Lake Tahoe (Coats et al. 2006). An increase in temperature is likely to increase Lake Tahoe's resistance to mixing which could have profound effects on the lakes aquatic community. Thermally driven disruption to historic mixing conditions in Lake Tahoe would favor introduced species over native species.

The combination of these effects could change the water chemistry and temperatures in Tahoe and Pyramid lakes. These effects could also result in reservoirs becoming too low to support tui chub populations. While the Lahontan Lake tui chub is presumably quite physiologically tolerant, changes to its food supply may result in population declines. These predicted impacts are speculative; however, studies should be conducted to document changes and develop trend data in order to inform conservation strategies to address climate change. Moyle et al. (2013) rated this form as "less vulnerable" to extinction from the effects of climate change than most other native fish species because of its refuge in Lake Tahoe.

Status Determination Score = 2.4 - High Concern (see Methods section, Table 2**).** The Lahontan Lake tui chub does not appear to be at risk of extinction; however, the status of the endemic population in California (Lake Tahoe) is largely unknown (Table 2). The Lake Tahoe population may have declined from its historic abundance, while the population in Pyramid Lake, Nevada continues to be large. The taxonomic identity and status of reservoir populations is not known.

Metric	Score	Justification
Area occupied	1	Found only in Lake Tahoe in CA
Estimated adult abundance	2	Population size in Lake Tahoe uncertain; no surveys conducted in over 60 years
Intervention dependence	5	No intervention required at this time
Tolerance	4	Relatively tolerant
Genetic risk	1	Genetics not well understood but the single confirmed population in California is isolated in one (albeit large and deep) lake
Climate change	2	Effects expected to be severe in the Lake Tahoe area
Anthropogenic threats	2	See Table 1
Average	2.4	17/7
Certainty (1-4)	2	Questions about taxonomy and lack of recent population surveys influence status evaluation

Table 2. Metrics for determining the status of Lahontan Lake tui chub in California, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: Surveys of Lake Tahoe and other Lahontan basin waters (Honey Lake, Topaz Lake, Stampede, Boca, and Prosser reservoirs) are needed to determine the distribution and abundance of Lahontan Lake tui chub in California. Equally important, a taxonomic study is needed of all potential populations of this subspecies in California and Nevada. A study comparing genetics to morphology, especially of sympatric morphs found in large lake systems, would be of particular interest. These studies are needed in order to develop a management plan to protect tui chub diversity. Currently, persistence of this form depends on the management of the water quality and biota of Lake Tahoe, including control of non-native, predatory fishes.

Figure 1. Distribution of verified Lahontan Lake tui chub, *Siphateles bicolor pectinifer* (Snyder), in Lake Tahoe, California.

EAGLE LAKE TUI CHUB
***Siphateles bicolor* ssp.**

Status: Moderate Concern. Although abundant, the Eagle Lake tui chub is endemic to a single, highly alkaline, terminal lake.

Description: No robust description of the Eagle Lake tui chub exists but they resemble other chubs in the *Siphateles pectinifer/obesa* complex. Eagle Lake tui chubs have a range of 12-28 gill-rakers on the first arch. Gill rakers are bimodally distributed, with peaks at 17-18 and 23-25, respectively (Kimsey 1954). Two body forms are present in the lake, one obese with a pronounced nuchal hump and the other slender. However, all other meristic characters are smoothly distributed across the entire population and Kimsey (1954) found no correlation between body form and gill raker number. Spawning individuals of both sexes develop reddish coloration on the fins. Males also develop small, white breeding tubercles on their body surfaces, while females develop slightly enlarged anal regions, protruding genital papilla, and deeper bodies (Kimsey 1954). Maximum size appears to be around 45 cm TL.

Taxonomic Relationships: This form was once regarded as a hybrid between *S. b. pectinifer* and *S. b. obesa* (Kimsey 1954, Hubbs and Miller 1943, Hubbs et al. 1974), based on gill raker counts. However, lack of other hybrid characters and the isolation of this lake from other parts of the Lahontan Basin indicate a long, separate evolutionary history. For a detailed discussion of tui chub taxonomy, see the Lahontan lake tui chub, *S. b. pectinifer,* account in this report.

Life History: Kimsey (1954) conducted the most comprehensive study of the natural history of this chub. Eagle Lake tui chub shoal in open waters of the lake, forming schools of fish of similar sizes. During the spawning season, schools break up and mature adults congregate in near-shore, shallow areas with dense beds of aquatic plants. At this time immature fish remain scattered throughout the lake.

Spawning occurs from mid-May through the beginning of July. Adults in spawning aggregations mill around dense macrophyte beds at about 1m depth and deposit adhesive eggs that stick to aquatic plants (*Myriophyllum spicatum, Ceratophyllum demersum, Potamogeton* sp.). The newly laid eggs are a pale orange-yellow, but color fades to a lighter straw-yellow after some time. Kimsey (1954) estimated the fecundity of a 27-cm female tui chub at 11,200 mature eggs, but he considered this a conservative estimate because not all eggs mature simultaneously. Thus, tui chubs are probably serial spawners, capable of reproducing several times during a season (Moyle 2002).

Newly hatched larvae are well developed and immediately begin to feed on rotifers, diatoms, desmids, and other microscopic material. Larval body plans of western cyprinids are extremely similar; however, larval tui chub develop a nuchal hump at just 5.5 mm (Remple and Markle 2005). Juveniles aggregate along the lakeshore in huge schools until about December, at which time they move into deeper waters. The young-of-year feed on zooplankton and on terrestrial insects blown into the lake from the surrounding forest (G. Grant, unpublished report; Eagles-Smith 2006). Adult Eagle Lake tui chubs appear to be opportunistic omnivores, although their diet shifts towards benthic organisms as they grow larger (Eagles-Smith 2006). Larger fish also show a shift into feeding at higher trophic levels, presumably because of their consumption of benthic invertebrates (Eagles-Smith 2006). The bulk of their stomach contents usually consist of detritus, with small quantities of algae, benthic and planktonic invertebrates, and aquatic macrophytes (Kimsey 1954). P. Moyle and students (unpublished data) found gut-contents of adult tui chub in Eagle Lake to consist of 83% detritus, 2% algae and 15% invertebrates. Eagle Lake tui chubs are also a key part of the lake ecosystem, as a major intermediary link between lower trophic levels (detritus, algae, invertebrates) and higher levels such as Eagle Lake rainbow trout and piscivorous birds (Eagles-Smith 2006). The lake supports exceptionally large breeding populations of osprey (*Pandion haliaetus*), western grebes (*Aechmorphorus occidentalis*), Clark's grebes (*A. clarkii*), eared grebes (*Podiceps nigricollis*) and other fish-eating birds; these abundant birds can be observed diving for and consuming large quantities of tui chub in most months of the year (J. Weaver, CDFW, unpublished observations).

Kimsey (1954) aged Eagle Lake tui chubs at 6-7 years using scales; however, Crain and Corcoran (2000) found that if opercular bones were used instead, the ages of adult tui chubs (30-35 cm SL) ranged from 12-33 years. Growth is rapid until age of 4 years, slows until age 7 and is very limited after 8 years (Crain and Corcoran 2000). Such ages and growth rates appear to be typical of tui chubs and suckers (Catostomidae) of the terminal lakes of the Great Basin (Scoppettone 1988).

Habitat Requirements: Eagle Lake is a large (22,000 ha) lake at an elevation of 1,557 m. It is estimated that 14% of the annual water budget for Eagle Lake is provided from stream flow, 38% from direct precipitation and 48% from sub-surface flow (Bureau of Land Management, Eagle Lake Water Budget 2010). Surface water enters the lake from Pine Creek and a number of smaller creeks, all of which are ephemeral, flowing only during winter and drying out by late spring. There is no outflow from Eagle Lake. Bly Tunnel (constructed in the 1920s), which was used to release small amounts of water into Willow Creek, a tributary to Honey Lake, is now closed off (P. Divine, CDFW, pers. comm. 2012). Most water loss is through evaporation.

Eagle Lake is highly alkaline (pH about 9 in most years), clear (secchi depth typically 4-6 m), and cool (summer temperatures rarely >20˚C at the surface). Average depth is 5-7 m, with a maximum depth of 30 m (in the southern basin). Eagle Lake tui chubs are found throughout the lake, but mature fish exhibit a seasonal migration from the deep southern basin of the lake in winter to the more shallow middle and northern basins, where spawning occurs, in spring. They require beds of aquatic vegetation in shallow, inshore areas for successful spawning, egg hatching, and larval survival (Kimsey 1954).

Distribution: This form is confined to Eagle Lake, Lassen County, California. Kimsey (1954) found no stream populations. However, tui chubs have been consistently found in three decades of fish surveys of upper Willow Creek (P. Moyle, unpublished data), which historically connected to Eagle Lake (outflow) via the Bly tunnel (BLM 2007).

Trends in Abundance: At present, tui chubs are the most abundant fish in Eagle Lake and support large populations of fish-eating birds and the piscivorous Eagle Lake rainbow trout. There is no indication that they are less abundant than they were formerly, but the population may suffer if lake levels continue to drop and alkalinity increases. Eagle Lake is currently (2011-13) at near-record low levels, so tui chub populations may decline with changing water chemistry and reduced habitat, particularly dense stands of tule beds they utilize for cover, many of which are now stranded on the dry shoreline.

Nature and Degree of Threats: Eagle Lake tui chubs and the entire unique Eagle Lake ecosystem face two major threats: alien fishes and extremely depressed lake levels. The greatest threat to Eagle Lake tui chub is reduced lake levels due to extended drought. Eagle Lake is a terminal lake, from which water leaves naturally by evaporation (90%) and subsurface flow (10%), resulting in its very alkaline waters (BLM 2010). Lesser threats include recreational development of the lakeshore and surrounding watershed as well as the continued effects of livestock grazing (Table 1). For a thorough discussion of all factors affecting the watershed, see the Eagle Lake rainbow trout account in this report.

Agriculture. The water diversion through Bly Tunnel has been completely closed. Other agriculture using ground water may influence lake levels; however, there are insufficient ground water data to assess potential impacts from ground water use outside the basin.

Alien species. With the complete closure of the Bly Tunnel, in combination with the unlikely event of a long wet period, lake levels could actually rise. Under such conditions, the lake would become considerably less alkaline and be able to support introduced fishes, as it did in the early 1900s, when largemouth bass and brown bullheads were common. These introduced fishes died out when lake levels dropped during the drought of the 1930s. The impact these fishes had on chub populations is not known. However, the effects of introduced diseases, predators, parasites, or competitors from future fish introductions could be disastrous to the lake ecosystem, including introductions of more alkalinity-tolerant species. Although illegal, introduction of bait or sport fishes by the public remains a possibility.

Effects of Climate Change: Climate change predictions indicate that snow melt and winter rain, the principle sources of recharge water for Eagle Lake, are likely to substantially decrease in the future. Temperature models indicate 2-4 degree rises in average air temperature by the end of the century, or higher, which will increase evaporation rates from the lake. Thus, the lake may recede to lower levels than experienced historically with alkalinities that may inhibit tui chub reproduction. Arguably, existing record low lake levels are already the result of climate change, at least in part. Moyle et al. (2013) rated Eagle Lake tui chub as “critically vulnerable” to climate change because of the potential for Eagle Lake levels to become so low and alkaline the lake can no longer support fish life.

	Rating	Explanation
Major dams	n/a	
Agriculture	Low	Agriculture using ground water may influence lake level; closure of Bly Tunnel (2012) a significant positive development
Grazing	Medium	Grazing affects most tributary streams and meadow systems by changing the timing and quality of surface water inflow to the lake and degrading riparian and instream habitats
Rural residential	Low	Residential population of the basin is limited but increasing
Urbanization	n/a	
Instream mining	n/a	
Mining	n/a	
Transportation	Low	Paved roads surround most of the lake and an extensive network of unpaved roads exists throughout the basin; impacts are unknown
Logging	Low	Watershed has been heavily logged; effects on Eagle Lake fishes, including tui chub, are unknown
Fire	Low	Entire watershed prone to fire; predicted climate change outcomes may increase frequency and intensity of fires; effects on lake ecology unknown
Estuary alteration	n/a	
Recreation	Low	Fishing and boating on the lake present little threat
Harvest	Low	Eagle Lake sustains a small sport fishery for tui chub, but no detrimental effects are known
Hatcheries	Low	The Eagle Lake rainbow trout, the principal (albeit native) fish predator of tui chub, is sustained by a large hatchery operation; however, it is unknown if hatchery stocking has created an artificially larger population than existed historically in the lake
Alien species	Low	Introduction of alien fishes could negatively affect the native fish community, provided lake levels increase and alkalinity decreases

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

Status Determination Score = 3.3 - Moderate Concern (see Methods section, Table 2). The population of Eagle Lake tui chub is large and the presence of many age classes in the population, including very old fish, suggests that they can outlast long periods of conditions unfavorable for reproduction. Nevertheless, their isolation in one location indicates a vulnerability that justifies continuing to recognize them as a Species of Special Concern and developing a monitoring program for them (Table 2).

Metric	Score	Justification
Area occupied	1	Restricted to Eagle Lake
Estimated adult abundance	5	Robust
Intervention dependence	5	No intervention needed at present
Tolerance	4	Broad tolerances but alkalinity of lake could become extremely high during sustained drought, inhibiting reproduction
Genetic risk	3	Single population
Climate change	1	Vulnerable in entire native range
Anthropogenic threats	4	See Table 1
Average	3.3	23/7
Certainty (1-4)	3	

Table 2. Metrics for determining the status of Eagle Lake tui chub, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: Eagle Lake should have special recognition as a one of the few lakes in the western United States that has a basically unaltered ecosystem, containing only native species and relatively low concentrations of contaminants. In particular, the lake should have recognition as habitat for its community of native fishes, including the endemic Eagle Lake rainbow trout, which feeds, in part, on tui chubs.

A management plan for the entire Eagle Lake basin (including tributary streams) should be developed, as discussed in the Eagle Lake rainbow trout account. One focus of this plan should be the establishment of a governance structure that can evaluate and regulate planned developments in the basin to ensure they are compatible with maintaining the integrity of the lake's ecosystem, including maintaining its large populations of fish-eating birds and the endemic fishes that support them.

In addition, a monitoring program for chubs should be established, as part of a broader program to monitor and manage Eagle Lake for its distinctive biota, as well as to ensure the continued absence of alien species. In particular, population age structure should be examined closely during periods when lake levels are low and alkalinities high and contingency plans should be developed in order to maintain the chub population if reproduction fails repeatedly.

Figure 1. Distribution of Eagle Lake tui chub, *Siphateles bicolor* ssp., in California.

BLUE CHUB
***Gila coerulea* (Girard)**

Status: Moderate Concern. Blue chubs are still common within their limited range in California but they are affected by changing conditions, especially water quality, in the upper Klamath Basin.

Description: Blue chubs resemble Klamath tui chubs, with which they are usually associated, except that they have finer scales (58-71 in the lateral line), are not as deep bodied, have longer fins, and have pointed heads with larger mouths, with the maxillary reaching the eye. There are 9 dorsal rays, 8-9 anal rays, and 14-17 rays in each pectoral fin. The pharyngeal teeth (2, 5-5, 2) are sharp, slightly hooked and located in two rows. The lateral line is curved ventrally. Blue chubs seldom exceed 40 cm SL and often have dark backs and silvery blue sides. Spawning males have blue snouts and bright orange tinges on their sides and fins.

Taxonomic Relationships: This species has been taxonomically stable since Charles Girard described it in 1856. The confusion that once existed over its scientific name was cleared up by Bailey and Uyeno (1964). Biochemical studies have determined blue chub to be distinct from other *Gila* species (Simons and Mayden 1998) and Smith et al. (2002) place it in the monotypic genus *Klamathella.* Schönhuth et al. (2012), however, recommend refraining from using *Klamathella* until the full systematics of the genus *Gila* have been completed (a work in progress).

Life History: Blue chubs grow rapidly in their first four years of life and mature at about 12-15 cm SL. Growth slows upon maturity but can continue to reach 38 cm FL (Scoppettone 1988). Blue chubs are long-lived. One individual at 34 cm FL was aged at 17 years (Buettner and Scoppettone 1991). Little has been published on the growth or early life history of blue chubs. Blue chubs are omnivorous, as indicated by their generalized body shape and tooth structure. Blue chubs collected from Willow Creek, Modoc County, in August, 1972 (all one year of age, 29-59 mm SL) fed mostly (66% by volume) on aquatic insect larvae and flying insects, including chironomid midge larvae and pupae, water boatmen, and water fleas. In comparison, two-year-old chubs (61-109 mm SL) fed heavily on filamentous algae (68%) and larger aquatic and terrestrial insects (Moyle 2002). Blue chubs from Oregon were found to have a similar diet (Lee et al. 1980).

Spawning occurs in May through August over shallow, rocky areas at temperatures of 15-18°C (Lee et al. 1980). In 1966, C.R. Hazel documented their spawning behavior in Upper Klamath Lake, Oregon: "On the afternoon of May 4, 1966, I observed an estimated 200-300 blue chubs spawning at the shoreline on the northern end of Eagle Ridge. Spawning was taking place from near the surface to a depth of 0.3 to 0.5 m. The bottom was composed of large gravel and rubble of volcanic origin. The water was clear with a low concentration of blue-green algae . . . the water temperature was 17° C. Two to several males would approach a female and exhibit rapid and violent agitations of the water, making it impossible to see exactly what was taking place. In some instances the female was pushed from the water onto dry land and in a few situations, eggs were spawned outside the water. After these activities, egg masses were found attached to [submerged] rocks either on the sides or near the bottom edge. Many of the depositions were found along rocky edges at depths to 0.5 m." Blue chubs gather to broadcast spawn in large schools in the summer months (ODFW 1996). As many as 30,000 eggs may be released in one spawning event. Spawning usually occurs adjacent to shorelines over gravel substrates in shallow (< 3 inches) water. Embryos hatch in approximately 7-9 days and larvae are pelagic, with an ability to concentrate in favorable areas (e.g., to avoid being exported from a lake) (Markle et al. 2009). Juveniles rear in shallow water until they reach sexual maturity around 3 years of age. Blue chubs may be an important food source for waterfowl, such as red-necked grebe (*Podiceps grisegena*; Watkins 1988) and mammals, such as mink (ODFW 1996).

Habitat Requirements: Blue chubs are most abundant in habitats with warm (summer temperatures >20°C), low-velocity waters and mixed substrates (Bond et al. 1988). In the wild, they have been collected in waters as warm as 32°C (D. Markle, Oregon State University, pers. comm.). They are especially abundant in lakes but school conspicuously in a variety of habitats, including small streams, shallow reservoirs and deep lakes. Although found in perennial and intermittent sections of Boles Creek, a tributary to Clear Lake Reservoir (Modoc County), they are most common in the small, shallow, weedy reservoirs of larger perennial streams (Scoppettone et al. 1995). In Upper Klamath Lake, Oregon, they are (or were) most abundant in rocky shore and open water habitats, avoiding marshy shore areas (Vincent 1968). In the summer, they seem to be excluded from deeper parts of the lake by oxygen depletion but move back into them as oxygen levels rise (Vincent 1968).

Although wild blue chubs are often observed in warm water, laboratory studies have shown that they lose equilibrium at temperatures of 28–33°C (mean, 31.5°C) and oxygen levels of 0.6–1.5 mg/L at 20°C (Castleberry and Cech 1993). These tolerances suggest that increasingly degraded water quality can limit their distribution and cause a decline in abundance and viability (Castleberry and Cech 1993). Vincent (1968) also found that blue chub distribution was inversely related to dissolved oxygen concentrations.

Distribution: Blue chubs are widely distributed in the lower elevations of the upper Klamath and Lost rivers in Oregon and California. In California, they are also found in Clear Lake Reservoir, Lost River, Lower Klamath Lake, Tule Lake, as well as the canals and tributaries that feed them. Their distribution has expanded to include Iron Gate and Copco reservoirs on the Klamath River (CH2M Hill 2003). Their range has also expanded through introductions in Oregon (e.g., Paulina Lake; ODFW 1996).

Trends in Abundance: Blue chubs remain common in Upper Klamath and Agency lakes, Oregon (Markle and Simon 1997, Markle et al. 2009). They also are abundant in the Boles Creek watershed and Clear Lake Reservoir (Buettner and Scoppettone 1991, Scoppettone et al. 1995). Between 0.2 to 3.2% (n = 70 – 196) of the fishes collected from Iron Gate, Copco and J.C. Boyle reservoirs are blue chub (Desjardins and Markle 2000, CH2M Hill 2003). No systematic estimates of past or present abundances in California have been made. While the artificial habitats provided by reservoirs have expanded their range, pollution from agriculture, introduction and competition from alien fishes, and altered flows in the Lost River may have contributed to a reduction in overall abundance. Drought in the 1980s and 1990s further stressed the aquatic fauna of Upper Klamath basin, a system already strained by other factors, such as water diversion, pollution, introduced species, and entrainment in power plants.

Nature and Degree of Threats: Blue chubs are tolerant of a wide range of water quality conditions, but their populations in California should not be regarded as secure because the aquatic ecosystems in upper Klamath basin are heavily impacted by multiple stressors.

Major dams. The rivers and lakes of the upper Klamath Basin are largely regulated by dams which have, in some cases, created additional habitat for blue chubs (see distribution) but other factors, such as introductions of alien species, have made these habitats less secure.

Recently, focus has been placed on the impact of turbine entrainment on blue chub abundance. CH2M Hill (2003) estimated that median turbine entrainment at Copco and Iron Gate reservoirs was 115, 979 fish, while the median entrainment at J.C. Boyle was 75,655. If it is assumed that the 0.2– 3.2% of entrained fish are blue chub, as estimated by Desjardins and Markle (2002), then approximately 383 to 6,131 individuals are likely to be entrained by both dams each year and most of these are likely young-of-year (50 to 150 mm, CH2M Hill 2003). Entrainment in Klamath River dams peaks in spring and summer, between April and June, during the time that juvenile fish are moving into rearing habitats (CH2M Hill 2003). Therefore, yearly recruitment may be reduced due to disproportionate juvenile mortality associated with entrainment. In reports reviewed by CH2M Hill (2003), members of the Cyprinidae (minnows and chubs) were the third most likely group to be entrained. In Link River Dam, Oregon, blue chub made up 49% (214,204) of the entrained fish (CH2M Hill 2003). However, the high rate of entrainment at Link River Dam may reflect their relative high abundance in Upper Klamath Lake.

Agriculture. Agriculture affects blue chubs through a combination of water diversions and pollution via return water. Diversions from rivers and reservoirs have dried up low elevation habitats once preferred by blue chubs (e.g. Lower Klamath Lake). The widespread reclamation of land in the upper Klamath basin by the U.S. Bureau of Reclamation's Klamath Project has significantly altered the landscape, reducing the amount of habitat available to aquatic species. Only about 10% of the open water and marsh habitats once available in the Upper (Oregon) and Lower Klamath lakes exist today (National Research Council 2006). The maximum surface area of Lower Klamath Lake is currently about 4,700 acres, a substantial reduction from the historical maximum surface area of 94, 000 acres. The maximum surface area of Tule Lake has also decreased significantly from 110,000 to approximately 13,000 acres. Efforts to restore wetlands began in the 1980s and continue today. Water manipulation and diking have also changed the manner in which the lakes in the upper basin behave (NRC 2006). Water management in Clear Lake causes its area and depth to vary outside of its natural range. Similarly, the surface area of Tule Lake historically varied from 55,000 to 110,000 acres; however, its surface area currently fluctuates from 9,450 to 13,000 acres. These changes likely reduce the productivity of these systems (NRC 2006).

Organic pollutants from agriculture and grazing flow into Upper and Lower Klamath lakes and Tule Lake, making them more eutrophic and less suitable for native fishes, even though blue chub are tolerant of fairly extreme environmental conditions (Castleberry and Cech 1993). Increased temperature and lower dissolved oxygen levels may negatively affect blue chub populations (Castleberry and Cech 1993). Both Lower Klamath and Tule lakes have been listed by the California State Water Resources Control Board as impaired for high pH levels (www.swrcb.ca.gov/northcoast/). Tule Lake has also been listed as impaired because of high nutrient loads. The Lost River has been listed for both high nutrients and pH concentrations. Poor water quality from agricultural drainage in the Lost River and Tule Lake has presumably reduced habitat suitability for blue chubs and other fishes.

	Rating	Explanation
Major dams	Medium	Dams change lake and river dynamics and can result in turbine entrainment
Agriculture	High	Major influence on Upper Klamath Lake (Oregon), Lower Klamath Lake and Tule Lake, through diversions and pollution
Grazing	Medium	Pervasive in much of the upper Klamath basin resulting in sedimentation and erosion of nutrient-rich soils
Rural residential	Low	Low population densities throughout area
Urbanization	Low	Minor impact of urbanization throughout its range
Instream mining	Low	Minor impact on shallow stream habitats
Mining	n/a	
Transportation	Low	Road density is relatively low in its range, but may affect water quality
Logging	Medium	Major source of sediment with roads and surface runoff associated with forest vegetation removal; greater historical impact than today
Fire	n/a	
Estuary alteration	n/a	
Recreation	n/a	
Harvest	n/a	
Hatcheries	n/a	
Alien species	Medium	Introduced fathead minnow are likely replacing blue chub in parts of their range

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

Grazing. Grazing in the upper Lost River and lands surrounding Clear Lake Reservoir have degraded riparian habitat conditions and water quality (CRWQCBNCR 2004) with unknown impacts to blue chubs.

Rural residential. Rural communities with low population densities are common throughout the range of blue chub (NRC 2006). However, their impact on water quality is thought to be minor.

Instream mining. Past gold mining has impacted streams throughout California, leaving a legacy of degraded habitats (Moyle 2002). However, current impacts are assumed to be minor.

Transportation. Road density in the area is low but may have minor impacts to water quality when roads are located close to riparian areas.

Logging. Logging practices in the Lost River watershed and around Clear Lake Reservoir may have increased nonpoint source pollution (CRWQCBNCR 2004) and sediment input into streams and lakes.

Recreation. Translocations by anglers have increased the geographic range of blue chub in Oregon (Moyle 2002) but have had no known impact in California.

Alien species. The upper Klamath Basin has been invaded by a number of introduced aquatic species, which are dominant in some habitats. These invasive species likely impact native fishes, including blue chub, through indirect and direct competition and predation. For instance, fathead minnow (*Pimephales promelas*), ecologically similar to blue chub (Moyle 2002), have proliferated in the lakes and canals of the region in recent years. The effect of increasing fathead minnow numbers on blue chubs and other native fishes is unknown but is thought to decrease blue chub populations (Castleberry and Cech 1993), presumably because they are better able to survive in warmer waters with lower dissolved oxygen levels than blue chubs. In Clear Lake Reservoir, Lost River and Tule Lake, introduced Sacramento perch (*Archoplites interruptus*) have become abundant (Moyle 2002). Sacramento perch are piscivorous but their impact on blue chub populations is not known. Likewise, yellow perch (*Perca flavescens*) are abundant in reservoirs also inhabited by blue chubs and may reduce populations through predation.

Effects of Climate Change: Climate change may result in increased air temperatures in this largely arid, high desert region, which may lead to reductions in habitat suitable for blue chub and other fishes (Cahill et al. 2004, Cayan et al. 2008). Expected outcomes of increased air temperatures are: increased evaporation rates (further reducing already diminished lake and reservoir levels, as well as the amount of perennial stream habitat), increased water temperatures, and decreased dissolved oxygen concentrations. Blue chub are intolerant of temperatures above a mean of 31.5°C and dissolved oxygen concentrations below 1.5 mg/L (Castleberry and Cech 1993). Elevated temperatures may also exacerbate the incidence of parasitism and resulting infection, particularly associated with the ciliate parasite *Trichodina* sp., found in blue chubs collected from Upper Klamath Lake (Foote and Harmon 1999). However, the potential effects of infection upon individual fish health and overall fitness are unclear. Because their habitats have already become fragmented by dams and portions of river and lake systems that no longer provide suitable habitat, blue chub may be particularly susceptible to the effects of climate change. Moyle et al. (2013) rate blue chub as "moderately vulnerable" to extinction as the result of the added effects of climate change.

Status Determination Score = 3.4 - Moderate Concern (see Methods section Table 2). The blue chub is clearly not in danger of extinction in Oregon (Upper Klamath Lake) but it may be more at risk in California, which contains more peripheral and fragmented populations. Overall, it is a resilient species but limited distribution in waters subject to diversion, pollution, warming, and invasive species may make them vulnerable to future declines if these stressors are not ameliorated.

Metric	Score	Justification
Area occupied	3	Present throughout limited historic range
Estimated adult abundance	2	Not known but assumed to be greatly reduced in CA
Intervention dependence	5	Populations appear self-sustaining
Tolerance	4	Tolerant but vulnerable to increased water temperatures and low oxygen
Genetic risk	5	Risk assumed to be low because blue chub are common throughout their range
Climate change	3	Vulnerable to increased temperatures and low oxygen levels, exacerbated by increases in water demand and reduction in precipitation
Anthropogenic threats	2	See Table 1
Average	3.4	24/7
Certainty (1-4)	2	Very little is published on blue chubs in CA

Table 2. Metrics for determining the status of blue chub in California, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: Surveys of the distribution and abundance of blue chub are needed throughout its range, but especially in California. Basic life history and habitat requirement studies, particularly of early life history stages, are also needed in order to develop appropriate conservation and management strategies. CH2M Hill (2003) made several recommendations for reducing mortality due to turbine entrainment, including operating turbines at peak efficiency and elevating turbine intake depth. Presumably, the best management strategy would be to improve water quality and habitat in Clear Lake Reservoir and its outflow, Lost River, and to rewater Lower Klamath Lake, if possible, as part of renegotiations of the USBR's Klamath Project. Adherence to Total Maximum Daily Loads Action Plans should decrease nutrient loading and improve water quality (www.swrcb.ca.gov/northcoast/) for the benefit of blue chub populations.

Figure 1. Distribution of blue chub, *Gila coerulea* (Girard), in California and southern Oregon.

ARROYO CHUB
***Gila orcutti* (Eigenmann and Eigenmann)**

Status: High Concern. The arroyo chub is vulnerable to extinction in its native range in the next 100 years. However, populations exist outside the native range and are regarded as generally more secure (e.g., those in the Santa Clara and Ventura River basins) (J. O'Brien, CDFW, pers. comm. 2013).

Description: Arroyo chubs are relatively small fish. Adults can reach lengths of 120 mm SL but are typically 70-100 mm long. They are sexually dimorphic. Males have larger fins than females and develop tubercles on the upper surface of the pectoral fins during breeding (Tres 1992). Both males and females have thick bodies, large eyes, and small mouths. Pharyngeal teeth arrangement can vary but is generally closely spaced with a formula of 2,5-4,2. Fin ray counts are 7 and 8 for anal and dorsal fins, respectively. Gill rakers number from 5 to 9. The lateral line is straight and complete, with 48-62 scales extending to the caudal peduncle. Their body color varies from silver or grey to olive-green on the dorsum, white ventrum, and a dull grey lateral band (Moyle 2002). Larvae and juveniles from the Los Angeles and Santa Ana River drainages are described in Feeney and Swift (2008).

Taxonomic Relationships: Arroyo chub are morphologically and genetically very distinct, reflecting their long evolutionary isolation (Miller 1945a). Both *Gila orcutti* and Yaqui chub (*G. purpurea*) belong to the subgenus *Temeculina* (Miller 1945a). Both species are part of a group of related *Gila* species in the American southwest (Simons and Mayden 1998). Arroyo chub hybridize readily with two other cyprinids native to California: Mohave tui chub (*Siphatales mohavensis*) and California roach (*Lavinia symmetricus*) (Hubbs and Miller 1943, Greenfield and Greenfield 1972, Greenfield and Deckert 1973). The systematics of North American Cyprinidae are complex (La Rivers 1962, Simons and Mayden 1998) and still require further investigation and clarification.

Life History: Arroyo chubs spawn primarily in June and July, but can breed more or less continuously from February through August, as the eggs of females ripen in small batches (Tres 1992). During spawning, a group of males pursue a ripe female and rub their snouts against the area below the female's pelvic fins, stimulating egg release. More than one male may fertilize the eggs as they are being laid (Tres 1992). Embryos adhere to plants, rocks, and debris and hatch in 4 days at 24 °C. After hatching, fry remain attached to or in the substrate for several days and swim to the surface, presumably to fill the swimbladder, once the yolk sac is absorbed (Tres 1992).

Arroyo chubs in the Santa Clara River are about 60 mm SL after their first year and grow about 10 mm each year after, reaching 80-90 mm SL by their fourth year (Tres 1992). Females can begin reproducing after the age of one year. Females generally grow larger than males after their second year. The life expectancy of arroyo chubs is 1-4 years.

Arroyo chubs are true omnivores that feed on algae, insects, and small crustaceans, but they apparently prefer to feed on algae. In one study, algae made up most (60-80%) of the identified stomach contents (Greenfield and Deckert 1973). They also feed extensively on the roots of a floating water fern (*Azolla*), which is generally infested with nematodes (Greenfield and Greenfield 1972).

Habitat Requirements: Arroyo chub are physiologically adapted to survive in habitats with low oxygen concentrations and wide temperature fluctuations, conditions common in southern coastal streams (Castleberry and Cech 1986). They are found in habitats characterized by slow-moving water, mud or sand substrate, and depths greater than 40 cm (Wells and Diana 1975). However, they have also been found in pool habitats with gravel, cobble and boulder substrates (Feeney and Swift 2008, J. O'Brien, CDFW, unpublished data, 2006-2012). They are most common in streams with gradients of less than 2.5% slope (Feeney and Swift 2008), where water temperatures range from 10 to 28 °C (J. O'Brien, CDFW, unpublished data). Thus, Deinstadt et al. (1990) found them in only small numbers (compared to rainbow trout) in the West Fork San Gabriel River, above Cogswell Reservoir where water was cool in summer (maximum temperatures <22°C) and gradients were mostly >4%. Most spawning occurs in habitats with low velocity, such as pools or edge waters, at temperatures of 14- 22 °C. In Big Tujunga Creek, chub utilize multiple habitats and substrates and are found in pools, runs, riffles, and edge-water over substrate ranging from sand and silt to boulders. However, they are most abundant in low gradient pools and flat-water habitats with gravel and sand substrate that support at least some aquatic/emergent vegetation (J. O'Brien, CDFW, unpublished data, 2009). Juveniles spend their first 3-4 months in the water column, usually in habitats with still water and vegetation or other submerged cover (Tres 1992).

Distribution: Arroyo chubs were once found only in the Los Angeles, San Gabriel, San Luis Rey, Santa Ana, and Santa Margarita rivers and in Malibu and San Juan creeks (Wells and Diana 1975), in southern California. Introductions expanded their distribution into the Santa Ynez, Ventura, Santa Maria, Cuyama, Santa Clara, and Mojave River systems and other smaller streams (e.g., Arroyo Grande Creek) (Miller 1968, Moyle 2002). Arroyo chub were introduced into the Mojave River from the Los Angeles River basin (Hubbs and Miller 1943). The northern-most population was the result of an introduction into Chorro Creek, San Luis Obispo County (Moyle 2002). Other introductions were not successful (e.g., from San Luis Rey River to Rio San Tomas in Baja California; Miller 1968). Absent from much of their native range, arroyo chubs were abundant only in the upper Santa Margarita River and its tributary De Luz Creek, Trabuco Creek below O'Neill Park, and San Juan Creek (San Juan Creek drainage), Malibu Creek (Swift et al. 1993), and the West Fork of the upper San Gabriel River below Cogswell Reservoir in 1990 (J. Deinstadt, CDFW, pers. comm. 1990). Today they are also abundant in Big Tujunga Creek and middle Santa Ana River tributaries, between Riverside and the Orange County line (J. O'Brien, CDFW, pers. comm. 2012). They are apparently present in low numbers in Pacoima Creek above Pacoima Reservoir, Sepulveda Flood Control Basin, Los Angeles River drainage (Swift et al. 1993).

Several hundred arroyo chub were relocated from Big Tujunga Creek to a restored section of the Arroyo Seco below Devils Gate Dam in 2008 (J. O'Brien, CDFW, pers. comm. 2009). Since 2008, they have also been documented in the headwaters of the San Jacinto River, near the USFS Cranston Station on the mainstem, and Indian Creek on the Soboba Indian Reservation (S. Loe, pers. comm. 2009). They have been found in recent years up to the North Fork and South Fork confluence in the mainstem San Jacinto River and have been found up the South Fork to near the Lake Hemet Dam (G. Abbas, pers. comm. 2009). Arroyo chub also occur in Topanga Creek, Arroyo Simi, and Bear Creek (San Gabriel Drainage) (J. O'Brien, CDFW, stream survey reports and CNDDB, 2009). In 2009, they were abundant below and immediately above Big Tujunga Dam in Big Tujunga Creek (J. O'Brien, CDFW, unpublished data). Surveys in 2010 indicate a much lower abundance of chub in Big Tujunga Creek due to impacts from flooding and debris flows associated with the 2009 Station Fire (J. O'Brien, CDFW, pers. obs.). A small population of arroyo chub was salvaged from Big Tujunga Creek in October, 2009 and held at the Riverside-Corona Resource Conservation District in Riverside. These fish were returned to Big Tujunga Creek during the summer of 2010. Surveys in 2011 and 2012 detected an abundant chub population in Big Tujunga Creek, below Big Tujunga Dam, and in Malibu Creek, above and below Ringe Dam (J. O'Brien, CDFW, unpublished data).

Arroyo chub have been found in large numbers within Cogswell Reservoir and immediately above the reservoir in the West Fork San Gabriel River but are much less abundant below Cogswell Dam (J. O'Brien, CDFW, unpublished data). They also occur in the North Fork and East Fork of the San Gabriel rivers, where their distribution has changed little since the early 1990s (J. O'Brien, CDFW, pers. comm. 2011). Chub occur below Morris Dam on the San Gabriel River but are uncommon (J. O'Brien, CDFW, pers. obs.). Chub are the least abundant, and have the narrowest distribution, of the native fishes found in the upper San Gabriel River, which is primarily a high gradient system (O'Brien et al. 2011).

Trends in Abundance: Arroyo chubs are currently abundant in Malibu and Big Tujunga creeks (J. O'Brien, CDFW, unpublished data) and are thought to be abundant at only four other places within their native range: upper Santa Margarita River and its tributary, De Luz Creek; Trabuco Creek below O'Neill Park and portions of San Juan Creek; Malibu Creek (Swift et al. 1993); and West Fork San Gabriel River immediately above Cogswell Reservoir. The decline in arroyo chub abundance has been largely attributed to habitat degradation of low-gradient streams within their native range (Swift et al. 1993). Arroyo chub numbers appear to respond favorably to a decrease in flows in certain drainages (e.g. high gradient streams). From 1986-1990, arroyo chub numbers temporarily increased due to low-water conditions in the West Fork of the San Gabriel River. Numbers decreased again after rains in 1991-1992 but increased in 1993. Arroyo chubs are common and widely distributed in some of the streams into which they were introduced, particularly in the Ventura and Santa Clara rivers. Although a nearly 20 year data gap exists regarding species status, abundance, and distribution, a planned CDFW survey of all endemic populations, along with tissue collections for genetic analyses, is planned to be implemented beginning in 2013 (J. O'Brien, CDFW, pers. comm. 2012).

Nature and Degree of Threats: Although introductions have increased their distribution and abundance, arroyo chub face multiple stressors within and outside their native range from a combination of urbanization and alien species interactions.

Major dams. Most streams containing arroyo chub are dammed and diverted to a large degree. Dams are barriers to fish movement and can result in dewatering of downstream habitats, in both native and non-native streams. Minimum flow releases, however, may actually provide summer habitat for chubs where it was periodically scarce in the past (e.g., West Fork San Gabriel River). It can be expected, however, that as water becomes scarcer (e.g. during drought or due to climate change effects), the impacts from dams will become greater.

Urbanization. Their native range falls largely into the Los Angeles metropolitan area where most streams are channelized, dammed, diverted, and otherwise degraded, leading to a reduction in abundance and distribution and to the fragmentation of populations. Urbanization has especially degraded the low-gradient streams which formerly contained optimal habitat (Swift et al. 1993). Urbanization effects include land use changes as a result of residential and commercial development, stream alterations from bridges, freeways, and channelization, heavy recreational pressure including water 'play' (swimming, pool damming, recreational mining in the Angeles National Forest, as well as trash dumping and pollution from urban runoff.

Some streams within the arroyo chub's native range contain high levels of pollutants from urban run-off that may have adverse impacts as yet unknown. For example, levels of silver, arsenic, chromium, copper, nickel, lead and selenium in Malibu Creek were found to be above thresholds recommended by the State of California for human consumption (Moeller et al. 2003). However, potential impacts to chubs are unknown.

Mining. While hard rock mines in the region are largely a thing of the past, instream placer mining continues in some areas and may disrupt spawning and recruitment on a local scale (J. O'Brien, CDFW pers. comm, 2011).

Transportation. Stream crossings associated with roads have, in many areas, become barriers to upstream migration. Consequently, many populations have become isolated, preventing repopulation of upstream habitats, and some habitats have become inaccessible. Barriers to upstream migrations at stream crossings are common after fires and floods. The activities of various flood control agencies, including ongoing removal of riparian vegetation and diversion of flows, are a threat to the continued existence of remaining arroyo chub populations in the lower foothills (Rodriguez, pers. comm. 2011).

Fire. Hot brush fires are increasingly common within the range of arroyo chubs. While direct effects of fire on chubs are few, fires followed by heavy rain can create debris flows that can reduce chub populations and temporarily degrade habitats. While chubs are adapted to such conditions, increased frequency of severe fires that entirely eliminate large areas of decadent chaparral vegetation, leaving denuded steep slopes of highly friable soils, increases risk of harmful debris flows.

Alien species. Alien species are a continuous and immediate threat. Arroyo chubs in the Cuyama River have hybridized with California roach. Ironically, arroyo chubs introduced into the Mojave River have hybridized with the endangered Mojave chub and are largely responsible for its decline (Hubbs and Miller 1943, Castleberry and Cech 1986). Arroyo chub populations may also be threatened by competition from the alien red shiner (*Cyprinella lutrensis*) and fathead minnow (*Pimephales promelas*) that may exclude them or reduce their numbers from many areas (C. Swift, pers. comm. 1998, 1999, J. O'Brien, CDFW, pers. obs.). Chub numbers are generally inversely correlated to shiner abundance (T. R. Haglund, pers. comm. 1998). Bass (*Micropterus* spp.), green sunfish (*Lepomis cyanellus*) and other predators introduced into streams may also target chub as prey, as they also prefer slow moving habitats (Swift 2005). Declines in arroyo chub abundance in the Santa Ana River has been partly attributed to predation by centrarchids and western mosquitofish (Feeney and Swift 2008). The introduced African clawed frog (*Xenopus laevis*) has also been shown to prey on arroyo chub (Lafferty and Page 1997).

	Rating	Explanation
Major dams	High	Dams alter flows, impair sediment recruitment and create barriers that fragment chub populations (J. O'Brien, CDFW, pers. comm. 2011)
Agriculture	Low	Agriculture historically altered streams but has been largely replaced by urbanization
Grazing	Low	Historically altered streams but has been replaced by urbanization, except at higher elevations
Rural residential	High	Rural development is rapidly expanding in range; substantial habitat alteration and degradation
Urbanization	High	Urbanization and all its associated stressors (stream channelization, pollution, water diversion, etc.) alter habitat throughout its range
Instream mining	Low	Recreational mining alters habitat and likely disrupts spawning and recruitment
Mining	Low	No known impact, but present
Transportation	Medium	Road, railroads etc. are along most streams with chubs
Logging	n/a	
Fire	Medium	Native range extremely prone to catastrophic fire and debris flows
Estuary alteration	n/a	
Recreation	Medium	Recreational use of streams (dam and impoundment building, swimming, bathing) is heavy in some areas, potentially altering habitats but effects are localized
Harvest	n/a	
Hatcheries	n/a	
Alien species	High	Negative interactions with alien species are an immediate threat to most populations

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

Effects of Climate Change: Because arroyo chub are adapted to survive in low oxygen conditions and wide temperature fluctuations, increases in temperatures associated with global climate change may not harm them as much as species with narrower environmental tolerances (Castleberry and Cech 1986). However, arroyo chub appear to be sensitive to changes in hydrologic conditions, especially changes in flow. Predictions for flows in California are for higher flows in the winter and drier conditions in the summer and fall (Hayhoe et al. 2004, Stewart et al. 2005, Anderson et al. 2008). Arroyo chub abundance has been shown to decline in high flows (wintertime scenario) and increase in low flows (summer and fall scenario). Although arroyo chub appear to thrive under low water conditions and are adapted to "flashy" flow conditions, climate change may result in streams that go dry in low gradient reaches during the driest months. Therefore, arroyo chub populations may readily adapt to global climate change conditions (increases in temperatures) but only when surface flows are maintained. Fish assemblages in southern California appear to be more responsive to local hydrologic conditions than small changes in land use (Brown et al. 2005), yet another reason for climate change to be considered in restoration and management planning. Moyle et al. (2013) rated arroyo chub as less vulnerable to effects of climate change than many fishes but noted that impacts associated with climate change were likely to contribute to its overall decline.

Status Determination Score = 2.1– High Concern; 3.1 – Moderate Concern when populations outside native range are considered (see Methods section Table 2). The high concern score applies to the remaining populations within its native range. The score increases to 3.1 if introduced populations are considered (Table 2; numbers in parentheses), making it a species of moderate concern.

Despite being locally abundant in some streams, some populations of arroyo chub in their native range are in danger of local extirpation due to the increasing effects of urbanization in the Los Angeles, Orange, and San Diego metropolitan regions. Interactions with non-native species, exposure to pollutants, and continued habitat degradation result in arroyo chub populations that are not secure, despite being widely distributed. The many introduced arroyo chub populations provide some security from species extinction but most of those face threats as well, especially from other alien species. The fact that the range of the arroyo chub coincides with some of the most densely inhabited parts of California, with a rapidly growing human population, means its future may never really be secure.

The American Fisheries Society considers arroyo chub to be Vulnerable, because of habitat destruction and other factors (Jelks et al. 2008). NatureServe ranks arroyo chub as Globally Imperiled because of its limited range. It is managed by the U.S. Forest Service as a Sensitive Species.

Metric	Score	Justification
Area occupied	3 (4)	Arroyo chub are locally abundant but the area occupied within its native range is limited
Estimated adult abundance	2 (4)	Abundance is often low within native range but higher in streams to which they have been introduced
Intervention dependence	2 (3)	Populations within native range will need to be actively managed in order to ensure recovery
Tolerance	4 (4)	Tolerate low oxygen conditions and highly variable temperatures but are sensitive to changes in flows
Genetic risk	1 (3)	Hybridization with other species and low population sizes threaten genetic integrity
Anthropogenic threats	1 (2)	Alien species and urbanization are major threats (Table 1)
Climate change	2 (2)	Changes in flows threaten population stability
Average	2.1 (3.1)	15/7 (22/7)
Certainty (1-4)	3	Peer reviewed literature on biology is limited

Table 2. Metrics for determining the status of arroyo chub, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. Numbers in parentheses are for all chub populations, including those outside the native range. See methods section for further explanation of scoring procedures.

Management Recommendations: Arroyo chub population surveys should be conducted at least biannually in their native range and every five years at all known sites, in order to monitor the status of this species. Within its native range, streams should be managed in a manner that favors native fish survival and reproduction, including active removal of non-native species. Restoration of highly degraded streams can help provide arroyo chub with more favorable stream habitats. For example, channelized streams can be reconfigured so that slow water habitats can redevelop and fine sediment can be retained. Levees can be set back to allow reconnection to the floodplain and meanders to develop. “Daylighting” streams can redirect water to above ground surfaces so that stream function can be reestablished. An example of such restoration is Arroyo Seco, into which arroyo chubs were reintroduced in 2008 (http://www.arroyoseco.org/casrp.htm).

A number of streams and stream reaches should be designated as native fish streams/refuges and managed for their natural flows and fauna. Restoration of urbanized streams will favor populations of other native species such as the Santa Ana sucker, unarmored threespine stickleback, southern steelhead, and Santa Ana speckled dace (Swift et al. 1993). The best candidate for a “native fish management stream”, at present, is the upper San Gabriel River basin (J. O’Brien, CDFW, pers. comm. 2011).

Arroyo chub seem to be as efficient as the introduced western mosquitofish (*Gambusia affinis*) in controlling mosquitoes, so their use for mosquito management within its range should be encouraged where genetically appropriate (Van Dam and Walton 2007). Vector Control agencies are currently working with CDFW on a plan, beginning with pilot projects in Riverside and Orange counties, to study the use of arroyo chub in lieu of mosquito fish in appropriate habitats.

Much is still unknown about the arroyo chub. Future studies should focus on: abundance and distribution of populations within its native drainages, genetic population structure, age and growth and other basic life history parameters, describing taxonomic relationships with closely related genera, describing habitat requirements and environmental tolerances for specific developmental stages, and identifying areas with highest potential for restoration and reintroduction. The genetic and conservation relationships among populations inside and outside the native range should be investigated to determine the best overall conservation and genetic management strategies.

Figure 1. Distribution of arroyo chub, *Gila orcutti* (Eigenmann and Eigenmann),in California. Note: distribution in the Ventura River is not indicated on map.

SACRAMENTO HITCH
***Lavinia exilicauda exilicauda* (Baird and Girard)**

Status: Moderate Concern. Sacramento hitch exist mainly as scattered, small, populations over a fairly broad geographic area and appear to be in long-term decline. The status of remaining populations needs systematic investigation.

Description: Hitch are deep-bodied cyprinids with a terminal, slightly upturned mouth that can grow to over 350 mm SL. The body is moderately elongated and thick, almost oval shaped in cross section (Hopkirk 1973, Moyle 2002). The head is relatively small and conical. The caudal peduncle is narrow. Scales are fairly large, 54-62 along the complete, decurved lateral line. Sacramento hitch have 10-13 dorsal fin rays, 11-14 anal fin rays, and 17-26 gill rakers. The pharyngeal teeth are long, narrow and slightly hooked, while the surfaces are relatively broad and adapted for grinding food (Moyle 2002). Young fish are silver and have a dark, triangular blotch on the caudal peduncle. As fish age, they become duller in color with the dorsal area turning brownish-yellow (Moyle 2002).

Taxonomic Relationships: Hitch are most closely related to California roach (*Lavinia symmetricus*) and they interbreed in some areas (Avise et al. 1975). Hitch can also hybridize with Sacramento blackfish, although hybrids are apparently sterile (Moyle and Massingill 1981). Three subspecies of hitch exist in California: Clear Lake hitch, *L. e. chi*, Monterey hitch, *L. e. harengus,* from the Pajaro and Salinas rivers and the type subspecies, Sacramento hitch, *L. e. exilicauda*. For a more detailed review of hitch systematics, see the Clear Lake hitch account in this report.

Life History: Sacramento hitch are omnivorous and feed upon zooplankton and insects, usually in open waters or at the surface of streams (Moyle 2002). In streams, they feed on filamentous algae, aquatic insects and terrestrial insects. Small (5-7 cm SL) hitch will feed, like trout, on drift at the heads of pools during the summer. Hitch feed mostly during the day (Moyle 2002). In rivers, they tend to stay in fairly limited areas and have considerable capacity to find velocity refuge in side pools (Jeffres et al. 2006). Myrick and Cech (2000) found they had difficulty sustaining swimming at velocities greater than 0.3 m/sec.

Growth is not well studied but appears to be related to summer temperatures. In San Luis Reservoir, Merced County, hitch reach 11-15 cm by the end of their first year and 15–30 cm by the end of the second year, when they mature. Subsequent increases are 20–50 mm/year, with a maximum size of around 35-40 cm. Hitch in Beardsley Reservoir, on the middle fork Stanislaus River (Tuolumne County), in contrast, are only 40–50 mm FL by the end of the first year and 9–11 cm FL by the end of their second, with subsequent increments of 20–40 mm/year. In Putah Creek, they average about 65 mm FL at the end of their first year and reach 200–250 mm in 3–4 years. Females grow faster and larger than males. Scale analysis indicates that hitch live 4–6 years, but it is likely that analysis of the bony structures of larger fish would yield greater ages (Moyle 2002).

Females usually mature in their second or third year; males mature in their first, second, or third year. Hitch are rather prolific: females from Beardsley Reservoir contained 3,000–26,000 eggs, with a mean of 9,000, but much higher fecundities (50,000-60,000 eggs) are likely in warmer habitats which contain large fish.

Spawning takes place mainly in riffles of streams tributary to lakes, rivers, and sloughs after flows increase in response to spring rains, although spawning requirements are in need of further documentation. When they are present in ponds and reservoirs with Sacramento blackfish, the two species often hybridize, presumably because they are forced to share spawning areas.

Spawning occurs in groups, with vigorous splashing. A spawning female is closely followed by 1–5 males, which fertilize eggs immediately after their release. Fertilized eggs sink into gravel interstices before absorbing water and then swell to about 4 times their initial size; swelling lodges embryos in the gravel. Hatching takes place in 3–7 days at 15–22°C and larvae become free-swimming in another 3–4 days. Young-of-year hitch spend the next 2 months shoaling in shallow water or staying close to beds of aquatic plants, especially among emergent tules, before moving out into more open water at about 50 mm FL. In permanent streams and ponds, larval and postlarval hitch aggregate around aquatic plants or other complex cover in shallow water. They are most active during the day (Moyle 2002).

Habitat Requirements: Sacramento hitch inhabit warm, lowland, waters including clear streams, turbid sloughs, lakes and reservoirs. In streams they are generally found in pools or runs among aquatic vegetation, although small individuals will also use riffles. Sacramento hitch prefer shallow (< 1 m deep) stream habitats with smaller gravel to mud substrates. Hitch have high temperature tolerances: fish acclimated to 30˚ C can survive temperatures up to 38° C for short periods of time, although they are usually most abundant in the wild in waters cooler than 25°C in summer (Moyle 2002). However, they prefer temperatures between 27-29°C in the laboratory and May and Brown (2002) found small numbers in agricultural drainage canals at temperatures of 25-29°C. They can tolerate low salinities, up to 9 ppt (Moyle 2002, Leidy 2007).

Spawning takes place over gravel riffles, at temperatures ranging from 14 to 26°C, but spawning on vegetation can also take place (Moyle 2002). When floodplains are available, hitch will use them for rearing although juveniles can become stranded once floodwaters recede (Moyle et al. 2007).

Distribution: Hitch were once found throughout the Sacramento and San Joaquin valleys in low elevation streams and rivers, as well as in the Delta. Today they are absent from the San Joaquin River and the lower reaches of its tributaries from Friant Dam down to the Merced River (Brown 2000, CDFG 2007). Populations have become established through introductions in a few reservoirs, such as Beardsley Reservoir, San Luis Reservoir, and Bass Lake (Fresno County). Sacramento hitch have been carried by the California Aqueduct from San Luis Reservoir to several southern California reservoirs, although it is not known if these are reproducing populations (Moyle 2002).

In the Sacramento River, hitch appear to be spread across much of their native range, up to and including Shasta Reservoir. However, populations are scattered (Moyle 2002) so May and Brown (2002) found hitch only at a few localities, in relatively low numbers. Sacramento hitch are also present in some of the larger tributaries to the San

Francisco Estuary (Leidy 2007) and in a few sloughs in the Delta (see next section).

Trends in Abundance: The abundance and distribution of Sacramento hitch is poorly documented, although evidence suggests that they are much less abundant than they were historically. Their distribution is also fragmented, with largely isolated populations scattered among various streams, lakes, and reservoirs. May and Brown (2002), in a survey of Sacramento Valley streams, found hitch in small numbers at only a few valley floor locations. CDFG (2007) and Brown (2000) recorded no hitch in extensive sampling of the lower San Joaquin River. Leidy (2007) noted that hitch were present in 13 of 65 watersheds tributary to the lower San Francisco Estuary and "locally abundant" in only seven; all sites were heavily influenced by urbanization. In the Delta, once an area of great natural resource abundance (including a diversity of native fishes), Brown and May (2006) recorded only 24 hitch from an eight year seining program that captured over 43,000 fish of a variety of species. Moyle et al. (2007) captured only small numbers of hitch in a 5 year study of the fishes using the tidal sloughs and floodplain of the Cosumnes River and none in the river itself. Likewise, Nobriga et al. (2005) encountered only 174 hitch in a program that captured over 79,000 fish in the Delta. However, similar numbers were taken in extensive sampling of the Delta in 1961-62 (Turner 1966) suggesting little change in their minority status. Nevertheless, Brown and Michniuk (2007) compared electrofishing captures of native fishes in the Delta between 1980-83 and 2001-2003 and found a general decline in native fishes, including hitch. They also determined that hitch seem to be largely confined to the northern Delta. Feyrer and Healey (2002) concluded that hitch had been extirpated from the southern Delta by the time of their study (1993-94).

Nature and Degree of Threats: Sacramento hitch occur in the lowland reaches of rivers and streams most impacted by human use, as well as in some reservoirs. Given that they persist in some urban streams, it appears hitch are capable of surviving in highly altered habitats although their abundance in such extreme environments is likely limited. Best evidence indicates that their populations are localized and fragmented today which, in turn, suggests that they may be particularly susceptible to a combination of anthropogenic stressors (Table 1).

Dams. Many dams exist on California's Central Valley rivers; these dams fragment watersheds and often create tailwater conditions that are unfavorable to native fishes such as hitch. Dam releases often provide either too little water or too much cold water, as they are generally intended to benfit salmonids (Brown and Bauer 2009). Thus, hitch were common in the San Joaquin River at Friant until Friant Dam was built; they subsequently disappeared from the area (Moyle 2002). On the other hand, tailwater releases below dams can, at times, create improved habitat for hitch (e.g., Mokelumne River, Jeffres et al. 2006) and the reservoirs impounded by dams are often colonized by hitch. Unfortunately, it is unknown why some reservoirs support hitch populations and others do not, nor why some tailwater streams support hitch populations while others do not. Given the fragmented distribution of hitch populations, it does not appear that regulated rivers and their reservoirs can be relied upon to provide population interconnectivity and suitable habitats to support hitch indefinitely.

	Rating	Explanation
Major dams	High	Dams fragment populations and alter flows
Agriculture	High	Agricultural irrigation alters and reduces flows; agricultural return waters are often warm and polluted with fertilizers, pesticides and other compounds
Grazing	Low	Most grazing occurs at higher elevations than primary habitats occupied by hitch
Rural residential	Medium	Rural development increasing in lowland areas within hitch range
Urbanization	Medium	Numerous metropolitan areas within hitch range; alteration of urban streams reduces or eliminates populations
Instream mining	Low	Gravel mining may create beneficial pools for hitch; legacy impacts from gold mining and dredging widespread throughout range
Mining	n/a	
Transportation	Medium	Roads exist along or cross most habitats and contribute to pollution and sediment input along with potential habitat fragmentation (e.g, culverts or other barriers)
Logging	Low	Historic hitch range largely below forested regions of state
Fire	n/a	
Estuary alteration	High	Delta is now mostly unfavorable habitat with many stressors (altered flows, alien species, pollutants, etc.)
Recreation	n/a	
Harvest	n/a	
Hatcheries	n/a	
Alien species	High	Predation by centrarchid basses and other predatory species may be a threat; particularly acute in the Delta

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

Agriculture. Much of the Sacramento hitch’s historic habitat is now dominated by agricultural land uses. Along with urbanization, agriculture greatly reduces water quality. Sacramento hitch generally disappear from waters highly polluted with agricultural drainage water, such as the lower San Joaquin River, where temperature, nutrient, turbidity and pesticide levels are high.

Urbanization. A large portion of the Sacramento hitch’s historic habitat occurred in the lower reaches of streams now dominated by urban and suburban areas. If water quality and quantity is maintained in these areas, hitch can persist in some numbers as the study of Leidy (2007) demonstrates. However, these are also areas subject to rapid change, aquatic and riparian habitat alteration and simplification, increasing water demand, and chronic input of pollutants. This suggests that hitch persisting in urban streams may be susceptible to extirpation.

Estuary alteration. The Delta is thought to have once been ideal hitch habitat, with diverse deep water areas, abundant invertebrates for food and large areas of cover for juveniles. Today, the Delta supports only a few scattered populations, primarily in areas where there is higher water quality and the presence of cover along banks. In the lower San Francisco Estuary, loss of tidal marshes and decreases in freshwater outflow have largely precluded hitch from moving between rivers, limiting gene flow and recolonization potential.

Alien species. Because of their habitat requirements, hitch are generally associated with alien fishes. They have shown some ability to coexist with non-native fishes, including major predators such as the centrarchid basses. However, where aliens are abundant hitch populations are generally small or absent, especially where habitat and water quality conditions are also poor. It is likely that populations in favorable habitats, especially those affording adequate cover, can persist in the face of alien predators although they may disappear when stressed by other factors, such as high temperatures and pollution, which make them more vulnerable to predation. It is also likely that numbers have been reduced in the Delta because of competition from other plankton-feeding fishes, such as threadfin shad (*Dorsoma petenense*) and Mississippi silverside, (*Menidia audens*), along with a general reduction in plankton abundance associated with clam invasions (Nobriga et al. 2005).

Effects of Climate Change: Climate models for Central California provide scenarios that strongly indicate that waters in which hitch occur will become increasingly unsuitable for sustaining populations (Knowles and Cayan 2002, Miller et al. 2003, Carlisle et al. 2010, Null et al. 2012). Generally, the scenarios show streams and lakes becoming warmer by 2100 (2-6 degrees C), while flows will become lower by late summer. Multi-year droughts may become more frequent and major high flow events will occur earlier and be flashier, as less snow accumulates and the incidence of rain on snow events increases, potentially leading to reduced flows during spawning periods. In short, widely accepted scenarios indicate that streams and other habitats will become more variable, with warmer temperatures, especially in summer, and with extreme conditions reached more often. Hitch live in lowland areas that are already highly altered and predicted to experience additional degradation through increased temperatures and poorer water quality as more water is diverted and pollution inputs increase (Moyle et al. 2013). The effects of climate change may be mitigated in some areas if regulated streams are managed specifically for native fishes (e.g., by providing cool spring flows to increase reproductive success). Moyle et al. (2013) indicated that hitch were "less vulnerable" to eventual extinction than many other native fishes from the predicted impacts of climate change if present trends continue, in part, because of their wide distribution and occurrence in larger river systems. However, climate change effects may accelerate apparent ongoing trends of local extirpation.

Status Determination Score = 3.1 - Moderate Concern (see Methods section Table 2). Four primary conclusions can be drawn from this status review: (1) very little is known about the biology, distribution, and status of Sacramento hitch; (2) hitch populations are generally isolated from one another and are usually small, so localized extirpations are likely; (3) hitch have been largely extirpated from the San Joaquin Valley, which once offered an extensive geographic area with many potentially suitable habitats; and (4) what are thought to have been prime hitch habitats outside the San Joaquin Valley (Central Valley lower rivers, portions of Delta and San Francisco Estuary) have been highly altered (especially through dam-regulated flow alterations, urbanization, agriculture, and introduction of alien species) and many areas are now unsuitable for hitch and other native fishes. These factors indicate that Sacramento hitch are a declining species.

Metric	Score	Justification
Area occupied	5	Apparently still widespread across much of historic range; populations fragmented and generally small
Estimated adult abundance	4	Many small populations
Intervention dependence	3	Management needed to prevent declines (habitat restoration, possible reintroductions)
Tolerance	3	Requires fairly high quality water
Genetic risk	4	Limited mixing of populations
Climate change	2	Effects poorly understood but likely negative
Anthropogenic threats	1	See Table 1
Average	3.1	22/7
Certainty (1-4)	2	Overall status poorly understood; hitch not the focus of most studies in which it is mentioned

Table 2. Metrics determining the status of Sacramento hitch, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: Priority should be given toward design and implementation of a systematic survey of Sacramento hitch abundance and distribution in order to determine their current status. A literature search would provide insights into areas for focused field surveys; however, they are often captured in very small numbers and recorded as incidental species, so literature specific to hitch may be spotty. Once existing populations have been located, studies are needed to determine their population dynamics, life history attributes, genetic structure, and habitat requirements. A monitoring program should be implemented for a select group of hitch populations in order to develop population size and trend information. Refuge areas for hitch and other fishes native to the lowland areas of the Central Valley and Delta, such as Sacramento blackfish (*Orthodon microlepidotus*) and Sacramento tule perch (*Hysterocarpus traskii traskii*), should be identified, isolated (using barriers where possible to prevent invasions of alien species) and, where necessary, restored to provide an expanding network of protected areas. In particular, refuge areas should be established in the North Delta, including the Cache Slough region.

No map is provided because of uncertainties in current Sacramento hitch distribution.

MONTEREY HITCH
***Lavinia exilicauda harengus* (Girard)**

Status: Moderate Concern. Although Monterey hitch are in no apparent danger of extinction, the status of populations remains uncertain across major portions of the species' range.

Description: Hitch are deep-bodied cyprinids with a terminal, slightly upturned mouth that can grow to over 350 mm SL. The body is moderately elongated and thick, almost oval shaped in cross section (Hopkirk 1973, Moyle 2002). The head is relatively small and conical. The caudal peduncle is narrow. Scales are fairly large, 54-62 along the complete, decurved lateral line. Sacramento hitch (see next paragraph) have 10-13 dorsal fin rays, 11-14 anal fin rays, and 17-26 gill rakers. The pharyngeal teeth are long, narrow, and slightly hooked, but the surfaces are relatively broad and adapted for grinding food (Moyle 2002). Young fish are silver and have a dark, triangular blotch on the caudal peduncle. As fish age, they become duller in color, with the dorsal area turning brownish-yellow (Moyle 2002).

Monterey hitch, *Lavinia exilicauda harengus,* differs morphologically from Sacramento hitch, *Lavinia exilicauda exilicauda,* by being deeper-bodied and having smaller dorsal and anal fins (Miller 1945b).

Taxonomic Relationships: Hitch from the Monterey basin were first described by Girard (1856a) as *Lavinia harengus*. In 1913, Snyder, apparently unaware of Girard's description, described another *Lavinia* species from the Pajaro and Salinas rivers, which he called *Lavinia ardesiaca*. Miller (1945b) showed that not only was *L. ardesiaca* preoccupied by *L. harengus* but also that *harengus* did not differ sufficiently from the type species, *L. e. exilicauda*, from the Sacramento system to warrant full species designation. Miller (1945b pg. 198) concluded that "although *harengus* and *exilicauda* are very similar and have been synonymized (Jordan et al. 1930) it seems best to retain *harengus* as a subspecies." Miller also discovered that Snyder's collections contained many hitch/roach hybrids which were likely fertile and able back cross with either parent species. Avise et al. (1975) proved Miller correct when their allozyme analysis found that 8% of *Lavinia* examined from the Pajaro River were F_1 hybrids and 5% were backcrossed individuals. Analysis using microsatellites supports the subspecific classification of *L. e. harengus* (Aguilar et al. 2009).

Hitch have also been documented to hybridize with Sacramento blackfish (*Orthodon microlepidotus*), but the offspring are likely sterile (Moyle and Massingill 1981). In the past, they hybridized with the now extinct thicktail chub (*Gila crassicauda*) (Miller 1963).

Life History: Stream populations of Monterey hitch have a much shorter life cycle than the better studied lake and reservoir populations of Clear Lake, as well as Sacramento hitch. Smith (1982) found that Monterey hitch could mature in their second summer of life, as small as 49 mm SL for males and 54 mm for females. Spawning takes place after high flows have subsided, typically May-June, but can extend into early August. Early reproduction is clearly advantageous for fish living in rivers with highly variable flow regimes. Smith (1982) documented rapid (1-2 years) recolonization of stream reaches that had dried up during a drought by both juvenile and adult hitch from upstream refuges. With an extended spawning season (May-August) and no need to make long migrations to find suitable spawning habitats, Monterey hitch can quickly establish large local populations.

Habitat Requirements: Monterey hitch can occupy a wide variety of habitats, although they are most abundant in lowland areas with large pools or in small reservoirs that mimic such conditions. Smith (1977) found they were most abundant in low-gradient sites in the Pajaro River basin that had permanent water and large pools in summer. The water as these sites tended to be clear, warm in late summer (18-28°C), and moderately deep (ca. 1 m maximum depth on average). Bottom substrates were mostly a mixture of sand and gravel and the presence of cover (e.g. fallen trees, overhanging bushes) was an important factor. In other parts of California, hitch prefer water temperatures of 14-18°C for spawning. However, Smith (1982) witnessed Monterey hitch spawning at temperatures as high as 26°C during early summer months.

When the sandbar forms at its mouth in early summer, the Salinas River lagoon can substantially convert to fresh water with a lens of salt water near the bottom. Monterey hitch apparently tolerate such brackish conditions, as indicated by the fact that they have been captured in the lagoon from water with salt concentrations as high as 9 ppt (Habitat Restoration Group et al. 1992).

Distribution: Monterey hitch are widely distributed in the Pajaro and Salinas river systems, both tributary to Monterey Bay. Within the Pajaro watershed, Monterey hitch are found below reservoirs on lower Uvas, Llagas and Pacheco creeks. They also occur throughout the San Benito watershed and in the deeper pools of the Pajaro River, especially upstream of the San Benito River confluence (Smith 1998). Depending on conditions, hitch may seasonally inhabitant Salinas and Pajaro lagoons (Casagrande et al. 2003, Smith 2007). Hitch have been documented in highly altered habitats in the lower Salinas watershed, including the old Salinas River channel, lower Gabilan Creek, known as the Reclamation Ditch, and Temladero slough (J. Casagrande, pers. comm. 2009). These habitats all depend on agricultural return water to maintain summer flow. In a 2002 fisheries survey of 17 stream sections of the Salinas River and its major tributaries, hitch were found at only one site in the mainstem, near Ardo (Casagrande et al. 2003); however, sample sites were biased towards steelhead habitat (J. Casagrande, pers. comm. 2009). Hitch are thought to occur in both San Antonio and Nacimiento reservoirs and in the river stretches directly below them (J. Smith, J. Casagrande pers. comm. 2009); however, recent surveys have not been performed to validate their presence.

Trends in Abundance: Monterey hitch are locally abundant in the Pajaro River system but have been extirpated from some reaches, especially in the main river, due to habitat alteration and reduced water quality (Smith 1982, 2007). As noted, the most recent steelhead-oriented survey of the Salinas River found hitch in only a single location (Casagrande et al. 2003). Current status of the Salinas system populations is uncertain, although they would be expected to occur in habitats below dams (J. Smith, pers. comm. 2009). Long-term population trends in both systems are unknown; populations are likely fewer and more fragmented than they were historically, although hitch may have expanded their range upstream where large dams have tempered seasonal variation in flows (Smith 2007).

Nature and Degree of Threats: Monterey hitch exist in a rapidly changing environment where flows are often tenuous and intermittent as the result of intensive agricultural land use, an arid climate, and increasing human demand for water. This is compounded by the fact that the majority of Monterey hitch habitat occurs on private lands, where there is little formal protection for aquatic organisms (Table 1).

Major dams. In the Salinas drainage, Nacimiento and San Antonio reservoirs impound large amounts of water and change flow regimes below their dams. These reservoirs impound water for flood control and release it for groundwater recharge and diversion for irrigation, although the effects of artificial flow regimes on native fishes, such as hitch, is poorly understood. In the Pajaro River, reservoirs attenuate high winter flows and provide permanent summer flows. This altered hydrologic regime appears to benefit hitch, as they have expanded their range upstream into Pacheco, Uvas and Llagas creeks below the reservoirs, into what was roach habitat prior to reservoir construction (Smith 2007). The reservoirs themselves may also be utilized by hitch, although their use of such habitats may be limited by interactions with alien species. Water diversion reduces flows in some areas, potentially limiting habitat suitability for hitch.

Agriculture. The Salinas Valley is one of the most intensively farmed areas in California. The valley also experiences the worst non-point-source water quality problems in the state due to farm and urban drainage systems. Consequently, alteration of the natural hydrology and stream morphology in this region has been severe, especially in downstream portions of the valley. One of the consequences of large-scale habitat degradation has been the extirpation of three native fish species: the thicktail chub (*Gila crassicauda*), Sacramento perch (*Archoplites interruptus*) and tule perch (*Hysterocarpus traski*) (Moyle 2002). Recently, large fish kills (which included hitch) have been documented in what is referred to as the Reclamation Ditch system. This system is comprised of 13 miles of ditches, built in the early 20th century to drain marshland near Salinas (Casagrande et al. 2003). Pesticide applications to protect crops also impact aquatic systems in such intensively farmed agricultural landscapes. In 2001, Monterey County - which encompasses both the Salinas Valley and lower Pajaro Valley - ranked fourth in the state for the total pounds of pesticide applied (California Department of Pesticide Regulation 2001).

Sedimentation from agricultural fields also detrimentally affects hitch habitats. The California Wildlife Action Plan (CDFG 2005) states: "Runoff problems are particularly severe on steeply sloping, erosion-prone soils, where strawberries, artichokes, and vineyard grapes are commonly grown. On sloped agricultural fields near Elkhorn Slough, soil erosion after heavy rain is estimated to be from 30 to 140 times greater than from natural lands" (Caffrey et al. 2002). Agricultural water consumption also threatens aquatic and riparian habitats. Irrigated agriculture accounts for about 70 percent of the Central Coast's water use (DWR 2005a). Over the past century, increased production of water-intensive crops like strawberries and lettuce has contributed to further impairment of aquatic habitat quality and altered ecosystem function.

	Rating	Explanation
Major dams	Medium	Stream flow alterations from multiple dams; potential benefits from perennial flow releases below dams
Agriculture	Medium	Monterey streams have been highly altered and degraded by intensive agriculture
Grazing	Medium	Grazing contributes to habitat degradation, stream incision and to intermittent streams drying more quickly and completely
Rural residential	Medium	Residential water withdrawal is a principal cause of decreased summer streamflow
Urbanization	Medium	Urbanized areas reduce habitat through stream alteration, fragmentation, channelization, water removal and pollution
Instream mining	Medium	Gravel mining alters habitats
Mining	Low	Of little direct effect, although legacy effects of mercury mines make most fish unsafe to eat
Transportation	Low	Many streams are crossed by roads and culverts (passage barriers)
Logging	Low	Little contemporary logging in the Monterey basin
Fire	Low	May cause local extirpations in small watersheds
Estuary alteration	Medium	Pajaro and Salinas lagoons may provide seasonal hitch habitat but both are heavily impacted by agriculture
Recreation	n/a	
Harvest	n/a	
Hatcheries	n/a	
Alien species	Medium	Hitch face competition from introduced cyprinids and sunfish and predation from introduced predators

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

Water is supplied to agriculture by diversion of surface water, groundwater pumping and through import from other regions via the State Water Project. As of 1995, groundwater provided about 84 percent of the region's water supply and 20 percent of that was considered overdraft, exceeding the amount of incoming water replenishing regional aquifers (DWR 1993, 2003a). As groundwater levels are depleted, flows are also reduced in streams and rivers.

Rural residential. Historically, urban centers in the Monterey region were located along coastal lowlands, with agriculture concentrated in valley-floor areas and grazing lands occupying the surrounding foothills. In recent years, however, growth and development have expanded from urban centers into adjacent farmlands and rural areas (CDFG 2005). Increasing rural development has elevated human impacts on small streams through habitat alterations, including higher levels of water withdrawal, which are especially acute in summer months when flows are already low.

Urbanization. While Salinas is the principal town in the watershed, the region around Paso Robles is becoming increasingly urbanized. As the human population in the Monterey basin has grown, demand has outstripped water supply, despite the presence of large reservoirs in the Salinas River system. Groundwater is the primary source of water to meet agricultural and urban needs; consequently, salt water intrusion due to over-pumping from groundwater aquifers threatens all coastal water supplies for both municipal and agricultural use. Urbanization also results in stream channelization, pollution input and other impacts that reduce the quantity and quality of hitch habitats.

Fire. While fire is a natural part of the California landscape, wild fires are becoming more severe as consequence of fire suppression, human land use and increasing temperature and aridity. Because hitch populations are increasingly isolated from one another due to human alteration to stream systems (agriculture, dams, reservoirs, introduced fishes), populations affected by fires are more likely to be extirpated without the possibility of natural recolonization.

Alien fishes. Alien fishes, especially centrarchids, are widespread in the watersheds containing hitch, especially in ponds and reservoirs. They represent a threat through predation and competition, especially during periods of drought when hitch may be confined with alien species in small pools. Reservoir populations are also threatened by competition from introduced planktivores such as threadfin shad (*Dorosoma petenense*) and Mississippi silverside (*Menidia audens*), as well as by predators such as white bass (*Morone chrysops*).

Effects of Climate Change: Climate change models indicate that stream temperatures will substantially increase, summer flows will be reduced, and the effects of fire on already dry watersheds will increase (Hayhoe et al. 2004; Thompson et al. 2012). Monterey hitch are well adapted to the warm, arid conditions of the basin's Mediterranean summers, but their dependence on pools in intermittent streams suggests that are particularly susceptible to increasing aridity and stream flow variability associated with climate change, despite their tolerant physiology. They are likely to become extirpated from streams which now currently maintain isolated, disconnected, pools in summer. Under predicted climate change scenarios, these already intermittent streams may dry completely under the dual strains of reduced rainfall and increased human water use across the region. Moyle et al. (2013) found that hitch are "highly vulnerable" to extinction from the added effects of climate change to their already degraded environment.

Status Determination Score = 3.1 – Moderate Concern (see Methods section Table 2). Monterey hitch are apparently still present throughout much of their native range, although few supporting data exist. Existing populations are fragmented, threatened by severe habitat alteration, and are subject to localized extinctions. The status of Salinas River basin populations is particularly uncertain. The Monterey hitch is listed by NatureServe as Vulnerable.

Metric	Score	Justification
Area occupied	2	Only found in Pajaro and Salinas river systems
Estimated adult abundance	5	Population(s) large
Intervention dependence	3	Most stream flows are regulated, directly or indirectly, and require ongoing management
Tolerance	4	High environmental tolerances
Genetic risk	3	Human alteration to river courses has caused incidence of roach/hitch hybridization to increase
Climate change	2	Reduced flows, along with increased water demand, are likely to further dry streams
Anthropogenic threats	3	See Table 1
Average	3.1	22/7
Certainty (1-4)	2	Very little published information

Table 2. Metrics for determining the status of Monterey hitch in California, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: The greatest management need for Monterey hitch is to conduct comprehensive fisheries surveys of the Salinas River basin that focus on native fishes and include both stream and reservoir habitats. A similar basin-wide survey should also be conducted in the Pajaro basin. Survey goals should include determination of the status and distribution of hitch and other native species, as well as location of important refuge areas to provide suitable habitats and protection during periods of low flow.

Status should be monitored at least once every five years to determine if there is attrition in increasingly isolated hitch populations. If local extirpations are detected, a management plan should be developed to ensure flows in key streams and to restore extirpated populations. Re-regulation of flows below dams to favor native fishes should be part of the management strategy. Consideration should also be given to the reintroduction of hitch into watersheds with suitable habitats in which they were historically present, but have since been extirpated.

Figure 1. Generalized distribution of Monterey hitch, *Lavinia exilicauda harengus*, in California. Actual distribution is likely fragmented.

CENTRAL CALIFORNIA ROACH
Lavinia symmetricus symmetricus (Baird and Girard)

Status: Moderate Concern. Although Central California roach do not face extinction risk as a species, there remains a high degree of uncertainty regarding the status, abundance and taxonomy of many populations. Because roach systematics are poorly understood, it is possible that small, distinctive, populations may be lost before they can be provided the protection they deserve as distinct taxa.

Description: Central California roach are small, stout-bodied minnows (cyprinids) with a narrow caudal peduncle and a deeply forked tail. Fish rarely achieve lengths greater than 100 mm total length. The head is large and conical, eyes are large, and the mouth is subterminal and slants at a downward angle. Some populations, especially those in the streams of the Sierra Nevada, develop a cartilaginous plate on the lower jaw, often referred to as a "chisel lip". The dorsal fin is short (7-9 rays) and is positioned behind the insertion point of the pelvic fin. The anal fin has between 6-9 rays. Fish with more dorsal and anal fins rays are likely hybrids with hitch (*Lavinia exilcauda*) (Miller 1945b). The pharyngeal teeth (0,5-4,0) have curved tips which overhang grinding surfaces of moderate size. Roach are usually dark on the upper half of their bodies, ranging from a shadowy gray to a steel blue, while the lower half of the body is much lighter, usually a dull white/silver color. The scales are small, numbering 47-63 along the lateral line and 32-38 before the dorsal fin. Subspecies are distinguished by various distinctive subsets of characters, especially fin ray and scale counts.

Roach exhibit general (non-nuptial) sexual dimorphism (Snyder 1908a, Murphy 1943). In tributaries to San Francisco Bay, Snyder (1905, 1908a) demonstrated that the sexes could be differentiated by the ratio of pectoral fin length to body length. Males exhibited a ratio of $\geq$.21 while females bore pectoral fins between .16 and .20 the length of their body. Both sexes exhibit bright orange and red breeding coloration on the operculum, chin and the base of the paired fins. Males may also develop numerous small breeding tubercles (pearl organs) on the head (Murphy 1943).

Taxonomic Relationships: The Central California roach was first described as *Pogonichthys symmetricus* (Baird et al. 1854a) from specimens collected from the San Joaquin River at Fort Miller near the present-day location of Friant Dam. It was subsequently reassigned to the old world genus *Rutilus* until 1913, when John O. Snyder erected the genus *Hesperoluecus* and described the following six species based on locality, isolation and morphological differences:

1. *Hesperoleucus mitrulus* from the tributaries to Goose Lake, Lake County, Oregon. Dempster et al. (1979) referred roach from the Upper Pit River, Modoc County to this taxon.
2. *Hesperoleucus navarroensis* from the Navarro River, Mendocino County.
3. *Hesperoleucus parvipinnis* from the Gualala River, Sonoma County.
4. *Hesperoleucus symmetricus* from the Sacramento and San Joaquin rivers and their tributaries.

5. *Hesperoleucus venustus* from the Russian River and streams tributary to San Francisco Bay. Snyder (1913) included roach from Tomales Bay tributaries in this taxon.
6. *Hesperoleucus subditus* from the major streams flowing into Monterey Bay.

The generic name *Lavinia* (Baird et al. 1854a) has precedence over *Hesperoleucus* (Snyder 1913) and is preferred because roach and hitch (the only other species in the genus) are interfertile and closely related genetically (Avise et al. 1975, Avise and Ayala 1976, Massingill and Moyle 1981, Bernardi et al. 2002, Aguilar et al. 2009). Roach are known to hybridize with hitch in the Pajaro and Salinas rivers (Miller 1945b), in Coyote, Alameda, and Walnut creeks (Miller 1963, Leidy 1984, Leidy 2004, pers. comm.), in the Sacramento-San Joaquin drainages (Avise et al. 1975, Jones 2001), and with arroyo chub (*Gila orcutti*) in the Cuyama River (Greenfield and Greenfield 1972). Hybridization may occur as a result of low water conditions whereby hitch and roach, which normally occupy different habitats (roach use higher gradient stream sections than do hitch), become restricted to the same isolated pools as streams dry (Miller 1945b, Jones 2001). Hybrids are fertile in the Pajaro River; Avise et al. (1975) found 8% of the *Lavinia* examined to be F_1 hybrids and 5% to be backcrossed individuals.

Miller (1945b p. 197) in the same paper that described hitch/roach hybrids from the Pajaro River suggested that "preliminary analysis of the forms of *Hesperoleucus* shows that many if not all, of those described as species are geographic subspecies of *H. symmetricus*." In his unpublished M.S. thesis, Murphy (1948c) reanalyzed data from Snyder (1913), along with his own data from coastal California streams, and concluded that coastal forms should be relegated to subspecies of *H. symmetricus.* Murphy did not include *H. mitrulus* in his study. In his arguments for merging coastal forms into *Hesperoleucus symmetricus,* Murphy (1948c p. 49) did not dispute that Snyder's species were morphologically and genetically distinct, instead, he followed what appears to be a strict interpretation of the biological species concept as outlined by Mayr (1942, 1954). Murphy argued that the distinctiveness of isolated populations, such as those in the Gualala River, was due to "merely a chance genetic divergence" resulting from small numbers of colonizing individuals and, if physical barriers were removed from between forms isolated in separate basins, "a population would soon lose its identity". Although it was never published, Murphy's (1948c) diagnosis was adopted by subsequent workers (Hopkirk 1973, Moyle 1976) and by the California Academy of Sciences (Dempster et al. 1979).

Hopkirk (1973) pointed out that Murphy's principal argument in denying specific status to coastal roach populations, the concept of a "chance genetic divergence" during colonization, was actually an important mechanism in speciation; e.g., the "founder effect" (Mayr 1942, 1954, among others). Hopkirk also asserted that natural selection contributed to differences among roach populations and, therefore, the distinctiveness of populations was "not due solely to the chance combination of genetic factors," as Murphy had maintained. Regardless of his critique of Murphy's species concept, Hopkirk agreed that Murphy was correct in placing all roach in one species, but he differed in his conclusions as to which populations should be accepted as subspecies. Hopkirk (1973) recognized *H. s. symmetricus, H. s. parvipinnis, and H. s. subditus* as subspecies and suggested that roach from the Russian River should be grouped with *H. s. navarroensis* and those from the Tomales Bay region be given subspecific status. Hopkirk (1973) also concluded that *H. venustus* from San Francisco Bay tributaries and roach from the Clear Lake drainage were synonymous with *H. symmetricus* from the Central Valley. Hopkirk (1973) further cautioned that *H. s. symmetricus* likely consisted of several subspecies, citing as an example a population from the Cosumnes River that exhibited morphologically distinctive characters (Hopkirk 1973).

Some support for Hopkirk's assertion of variability in Sacramento Valley roach populations was provided by Loggins (1997) who, using DNA fingerprinting techniques, found evidence for fairly long isolation of four adjacent Sacramento populations. Similarly, in the San Joaquin drainage, Brown et al. (1992) found that populations from different drainages could be distinguished by multivariate analysis of 15 morphometric characters. Populations from the Kaweah River and from the Red Hills (i.e., Horton Creek and other small creeks near Sonora, Tuolumne County) were particularly distinctive because of the high frequency of a "chisel lip" feature. Bernardi et al. (2002, p. 261) found that the Red Hills roach population appeared "reciprocally monophyletic for assayed mitochondrial DNA markers" and this combination of morphological and genetic distinctiveness in the Red Hills roach led Moyle (2002) to assign it subspecific status. While acknowledging the need for a taxonomic reevaluation of *Lavinia,* Moyle (2002 p. 140) simultaneously recognized eight subspecies of roach and called for a thorough biochemical and morphological investigation into the roach "species complex."

In the most comprehensive genetic study of *Lavinia* to date, Aguilar and Jones (2009) used both nuclear microsatellite (nDNA) and mitochondrial DNA (mtDNA) markers. Employed in tandem, these two genetic markers supply insight into both the relationships between populations (phylogenetics) and the distinctiveness of individual populations (taxonomy). The microsatellite analysis of Aguilar and Jones (2009) clearly defined Gualala, Pit, Navarro and Red Hills populations as distinct genetic units and largely supports the subspecies proposed by Moyle (2002), with the notable exception that separate groupings were found for Russian River and Clear lake populations. Analysis of mtDNA identified roach from the Pit and Gualala rivers to be highly divergent from all other populations and reciprocally monophyletic for the haplotypes assayed, suggesting that these populations have been isolated for considerable time. In addition, mitochondrial results show Tomales, Red Hills and Russian River/Clear Lake roach populations to be highly supported clades.

In light of: (1) the recent genetic analysis (nuclear and mtDNA) that corroborates the distinctiveness of the species that Snyder (1913) originally described, and (2) the fact that Snyder's original species names were never properly submerged (i.e. through formal publication of an analysis in the peer-reviewed literature), the following taxonomic designations should be regarded as valid; however, a thorough analysis needs to be published in the peer-reviewed literature in order to solidify this taxonomy:

1. The Northern roach (Pit roach) is a valid full species. The subspecies name, *Lavinia s. mitrulus* (Murphy 1948c) is pre-occupied by *Lavinia mitrulus* (Snyder 1913).

2. The Gualala roach is a valid full species. *L. s. parvipinnis* (Murphy 1948c) is pre-occupied by *Lavinia parvipinnis* (Snyder 1913).

3. The subspecific designations for the Navarro, Tomales, and Red Hills roach subspecies should be retained. These taxa are probably sufficiently distinct to warrant full species status but genetic evidence from a sufficient sample size is necessary to allow

increased statistical support for such a conclusion. Based on all evidence gathered to date, the following is a list of the taxonomic units for roach in California. Full species are denoted in bold.

1. **Northern roach,** *L. mitrulus*
2. **Gualala roach***, L. parvipinnis*
3. **California roach,** *Lavinia symmetricus*
   3a. Central California roach, *L. s. symmetricus*
   3b. Navarro roach, *L. s. navarroensis*
   3c. Monterey roach, *L. s. subditus*
   3d. Clear Lake roach, *L. s. ssp.*
   3e. Russian River roach, *L. s. spp*.
   3f. Tomales roach, *L. s. ssp.*
   3g. Red Hills roach, *L. s. ssp.*

The central California roach consists of many populations that are isolated to varying degrees. This isolation was partially natural because roach populations seem to become easily isolated from one another and adapted to local conditions (Bennett et al. 1992). Many of these isolated populations are distinguishable, both morphologically (Brown and Moyle 1993) and genetically (Aguilar and Jones 2009), but the interrelationships are complex and poorly understood. Gaps in their distribution (e.g., Fresno River) suggest recent extirpations (Bennett et al. 1992). One population, found in small streams in the Red Hills near Sonora, Tuolumne County, is distinct both morphologically and genetically (see the Red Hills roach account in this report) and it is possible that a thorough analysis of central California roach systematics will identify other taxonomically distinct populations (e.g., Cosumnes River, Peoria Creek, Kaweah River, Los Gatos Creek) that would merit further recognition.

Life History: Roach are opportunistic omnivores whose diet varies greatly across watershed, habitat type and season. In small, warm, streams they primarily graze on filamentous algae, which is seasonally abundant, although they also ingest crustaceans and aquatic insects, which can account for nearly a third of stomach contents by volume (Fry 1936, Fite 1973, Greenfield and Deckert 1973). In larger streams, such as the North Fork Stanislaus River, roach have been observed to feed on drift and aquatic insects may dominate their diet year-round (Roscoe 1993). Juvenile roach consume large quantities of crustaceans and small chironomid midge larvae, while adult roach are more opportunistic feeders, feeding both off the substrate and from drifting insects in the water column. Although roach are primarily benthic feeders, Moyle (2002) observed roach feeding in the Tuolumne River in swift current on drift organisms, including terrestrial insects. Adult roach show little preference for food type and small midge, mayfly, caddisfly and stonefly larvae, along with elmid beetles, aquatic bugs and amphipods, are taken roughly in proportion to their availability in the benthos and drift (Fite 1973, Roscoe 1993, Feliciano 2004). Adult roach have also been observed to consume larger prey and one individual in the Navarro River contained three larval lampreys (Moyle 2002). As a result of their benthic feeding habits, stomach contents of adult roach are often found to contain considerable amounts of detritus and fine debris. It is thought that roach extract some nutritional value from this material because its retention is facilitated by the gill rakers and mucus secretions from epithelial cells (Cech et al. 1991).

Growth is highly seasonal, with most rapid growth typically occurring in early summer (Fry 1936, Barnes 1957). In perennial streams, roach frequently exceed 40 mm SL in their first summer, reach 50-75 mm by their second year and 80-95 mm SL by their third summer (Fry 1936, Roscoe 1993). Few individuals exceed 120 mm SL or live beyond 3 years, although a 6 year-old specimen was recorded in San Anselmo Creek, Marin County (Fry 1936).

Roach typically mature at 45-60 mm SL in their second or sometimes third year (Fry 1936). Fecundity is dependent on size and ranges from 250 – 2,000 eggs per female (Fry 1936, Roscoe 1993). Spawning activity is largely dependent on temperature and typically occurs in March through early July, when water temperatures exceed 16˚C. Spawning occurs in riffles over small rock substrates, 3-5 cm in diameter. Roach spawn in large groups over coarse substrates where each female repeatedly deposits eggs, a few at a time, into the interstices between rocks which are immediately fertilized by one or more attendant males. Spawning aggregations can be quite conspicuous and spawning fish can splash so vigorously that, at times, the splashing can be heard at some distance (Moyle 2002). This activity clears silt and sand from interstices of the gravel which improves adhesion for sticky fertilized eggs. Eggs hatch after 2-3 days, and larvae remain in the gravel until large enough to actively swim. Larval development is described by Fry (1936). The population studied in Bear Creek, Colusa County, apparently spawned in emergent vegetation and newly hatched larvae remained among the plants for some time (Barnes 1957). Once the yolk is absorbed, larval roach feed primarily on diatoms and small crustaceans (Fry 1936).

Larval drift may be a significant form of dispersal for roach during some years. Roach embryos and larvae in Eel River tributaries (introduced population) made up a significant portion of the nighttime planktonic drift from May through July (Harvey et al. 2002, White and Harvey 2003). White and Harvey (2003) suggest that the timing of roach spawn (in late spring as flows recede) and apparent short period of drift for individual larvae are adaptations that may reduce the risk of roach drifting downstream into unsuitable habitats types. In Central Valley streams, these attributes would largely prevent young roach from being passively transported to unsuitable valley-floor habitats.

Habitat Requirements: Central California roach are generally found in small streams and are particularly well adapted to life in intermittent watercourses; dense populations are frequently observed in isolated pools (Fry 1936, Moyle et al. 1982, Leidy 2007). Roach are most abundant in mid-elevation streams in the Sierra Nevada foothills and in lower reaches of some San Francisco Bay streams but they may also be found in the main channels of some rivers, such as the Stanislaus (Roehrig 1988) and Tuolumne (Moyle 2002). Roach tolerate a relatively wide range of temperatures and dissolved oxygen levels, as evidenced by the fact that they occupy habitats as varied as cold, clear well-aerated "trout" streams (Moyle et al. 1982, Roscoe 1993) and intermittent streams where they can survive extremely high temperatures (30 to 35˚ C) and low dissolved oxygen levels (1-2 ppm) (Moyle et al. 1982, Knight 1985, Castleberry et al. 1990).

In the tributary streams to the San Francisco Bay, roach occupy suitable habitats from headwaters to the mouth but are intolerant of saline waters (Moyle 2002). They have been recorded in salinities up to 3 ppt, but perish before salinities reach 9-10 ppt

(Moyle unpublished data). In headwater reaches of San Francisco Bay tributaries, roach typically co-occur with rainbow trout (*Oncorhynchus mykiss*), juvenile Sacramento sucker (*Catostomus occidentalis)* and prickly or riffle sculpin (*Cottus asper* and *gulosis*, respectively*)* (Leidy 2007). In small, warm, intermittent estuary streams, roach are most often found with juvenile Sacramento suckers and, occasionally, with green sunfish (*Lepomis cyanellus*) (Leidy 2007). In lower mainstem stream channels, roach occur as part of a predominately native fish assemblage which, depending on location, is characterized by combinations of Pacific lamprey (*Entosphenus tridentatus*), Sacramento pikeminnow (*Ptychocheilus grandis*), hardhead (*Mylopharodon conocephalus*), Sacramento sucker, riffle sculpin, prickly sculpin and tule perch (*Hysterocarpus traskii*) (Leidy 2007).

Although common in streams that support native fishes, roach are most abundant when found by themselves or with just one or two other species (Moyle and Nichols 1973, Leidy 1984, 2007, Brown and Moyle 1993). When found alone, roach will occupy open water in large pools; when found as part of complex fish assemblages, roach tend to congregate in low velocity (<40 cm/sec), shallow (<50cm) habitats (Baltz and Moyle 1985). Similarly, when collected with non-native fishes in the lower portions of Alameda, Coyote and Walnut creeks, roach were typically found in the shallow margins of pools (Leidy 2007). In the presence of native predators (e.g., pikeminnow) roach are also restricted to the edges of pools, riffles, and other shallow-water habitats or in dense cover, such as that provided by fallen trees (Brown and Moyle 1991, Brasher and Brown 1995). In Alameda Creek, juvenile roach and Sacramento sucker (<20 mm TL) are often found in great abundance in very shallow (< 10 cm) edgewater habitats of pools with sandy substrates (R. Leidy, pers. obs. 2009).

While roach rarely display aggressive behavior towards other fishes, they are important predators of lower trophic levels and may play a key role in regulating aquatic food webs, especially in watersheds where they are introduced. For instance, using net-pen mesocosm experiments, Marks et al. (1992) demonstrated that introduced roach suppressed benthic insects and affected persistence of algae in the South Fork Eel River.

Water temperatures in many Eel River tributaries have substantially warmed over the last 50 years (Harvey et al 2002). This change in thermal regime is attributed to a combination of human activities, primarily heavy logging, and to the large floods of the 1950s and 1960s which dramatically altered channel configurations (Moyle and Nichols 1973, Harvey and White 2003). Harvey et al. (2002) suggest that these changes in temperature regime enhanced the invasion of the drainage by California roach and Sacramento pikeminnow. Evidence from the Gualala and Navarro watersheds also suggests that human alteration of coastal watersheds creates thermal regimes favorable to roach.

Distribution: Central California roach are found in tributaries to the Sacramento and San Joaquin rivers and tributaries to San Francisco Bay. Their historic distribution in the upper Sacramento River basin is poorly understood but their upstream range limit is thought to have been Pit River Falls. Roach found above the falls are northern roach (*L. mitrulus*). They are absent today from the Fresno River and other tributaries to the San Joaquin River, where they might be expected, as the result of habitat change and invasions of alien predators (Moyle and Nichols 1973, Moyle 2002). They are also absent from most of their historic range in the Cosumnes River (Moyle et al. 2012).

The ability of roach to persist in small, high gradient, often intermittent tributaries has led, through erosional capture of interior headwater streams, to their colonization of adjacent drainages in a number of areas (Snyder 1908, 1913, Murphy 1948c, Moyle 2002). Because they are relatively intolerant of saline waters, dispersal to these coastal streams could not have occurred through ocean waters, although connections at low elevations may have been possible in some cases when sea levels were lower and freshwater estuaries existed that joined the mouths of rivers (Moyle 2002). Similarly, populations in the San Francisco Estuary are isolated from each other, to some extent, by the inability of roach to disperse through saline waters of the estuary. Exchange between populations may, nevertheless, occur during flood years when freshwater outflow is high enough to create freshwater lenses in the surface waters of the estuary. This process may, at times, allow fish intolerant of saline waters to exchange between watersheds around the estuary and provide inland fish swept downstream from the Central Valley access to Bay tributaries (Ayers 1862, Snyder 1905, Murphy 1948c, Leidy 2007). Historically, during high water periods, fish may also have been able to disperse through flooded marshes on the fringes of the estuary. Today, it thought that such dispersal happens only very rarely, if at all, because the marshes and floodplains that once fringed the estuary have been highly altered or narrowed to such a degree that movement between watersheds is very difficult for fish as small as roach (Moyle et al. 2012).

In a few instances, the range of central California roach has been expanded through introductions. Their small size makes roach an attractive bait fish and increases risk of illegal "bait bucket" transfer between watersheds by anglers. For example, Hetch-Hetchy Reservoir, on the upper Tuolumne River, supports a large pelagic population, high above a series of natural barriers (P. Moyle, unpublished observations). Soquel Creek and the Cuyama River, in southern California, support presumed introduced populations, although genetic investigations may reveal that both Southern California populations are actually native (Moyle 2002). Roach are widespread in the Eel River, apparently from an introduction in the 1970s; the origin of these roach is unknown but it is most likely from a Sacramento River tributary (Moyle 2002).

Trends in Abundance: In absolute terms, Central California roach are still abundant but growing evidence suggests that Central Valley populations are declining (Moyle and Nichols 1973, Daniels and Moyle 1982, Bennett et al. 1992, Brasher and Brown 1995). For example, surveys indicate that roach have been completely extirpated from the entire Fresno River watershed (Moyle 2002) and the South Fork Yuba River, except for one small population (Gard 2004). Between 1970 and 1990, roach were eliminated from numerous locations, such as the Cosumnes River (Moyle and Nichols 1973, Brown and Moyle 1991 & 1993, Moyle et al. 2003), in the San Joaquin Valley. In contrast, two comprehensive studies of San Francisco Bay tributaries (Leidy 1984, 2007) found roach to be abundant in both surveys. They were the most commonly collected native fish and populations appeared to be relatively stable.

Nature and Degree of Threats: The small streams that comprise the majority of roach habitat in their native range are acutely vulnerable to human alteration. Low elevation streams in the Sierra Nevada foothills are heavily altered by rural development, ranching and agriculture (Moyle and Nichols 1973), while all populations face some degree of threat from water diversion, urban and suburban development, and introduced fishes (Moyle 2002). These factors work in conjunction with the isolation of most roach populations, especially small populations in intermittent streams, because they collectively prevent recolonization following local extirpation. The factors which threaten roach persistence in their native range are multiple (Table 1) and differ at each locality.

Dams. Dams of all sizes have multiple effects on roach distribution and abundance: dams create impassible barriers for small fish moving upstream; impoundments generally support populations of predators that eliminate roach and other native fishes in upstream areas; dams alter downstream flows, which may or may not be beneficial to roach (although roach are rarely found below dams); and small dams divert water from streams, limiting and sometimes completely drying preferred roach habitats. Generally, where dams exist on Central Valley streams, Central California roach persist only in small tributaries above them. Since dams effectively isolate roach populations, when localized extinctions occur, otherwise suitable habitats cannot be recolonized naturally.

Agriculture. Roach are generally absent from streams flowing through intensively farmed lands. Such streams are usually: (1) diverted and may be dried by excessive pumping on occasion, (2) heavily polluted with irrigation return water containing fertilizers, pesticides and sediment, (3) channelized and rip-rapped with little complex habitat roach require, and (4) habitats favored by non-native fishes that prey on or compete with roach. Given the extensive conversion of lands to agricultural use across much of the roach's historic range, especially at lower elevations, their populations have likely been heavily impacted by agriculture and further isolated from one another due to the unsuitability of such streams.

Grazing. Central California roach are often found in streams flowing through pastures that have generally altered or degraded aquatic habitats; however, they may persist as long as stream banks are relatively intact and riparian trees and deep pools provide shading and cover. Heavy grazing may lead to stream bank collapse, increased sedimentation of pools, pollution input from animal wastes and reduced shading and cover. Despite their high tolerance of adverse conditions, roach populations can be extirpated from waters heavily impacted by grazing, especially in areas where cattle are allowed direct access to streams. Ponds used to provide water for cattle and other livestock divert water from streams and often support populations of alien predatory fishes. These fishes (e.g., green sunfish, largemouth bass) may escape stock ponds during wet periods if ponds spill and become connected to adjacent streams. Roach have disappeared from the south fork of Dye Creek, Tehama County, because of the invasion of green sunfish from stock ponds (Moyle, unpublished data).

Rural residential development. The Sierra Nevada foothills are undergoing dramatic change due to dispersed rural development. While roach populations can persist in the face of moderate development (and even increase when summer dams for swimming are built), they may be extirpated in areas of heavy development due to excessive water diversion or ground water pumping (e.g., wells) during low-flow periods, polluted inflow from septic tanks, siltation and instream passage impediments from roads, and loss of complex habitat through bank stabilization or flood control projects.

Urbanization. Roach tend to disappear from streams flowing through urban areas, presumably because of often dramatic habitat alteration and simplification, reduced and/or highly regulated flows, pollution inputs from surface runoff and wastewater effluents, and the presence of alien fishes that favor such altered habitats. However, Leidy (2007) found that roach were common in streams flowing into San Francisco Bay, many through urban areas. Nevertheless, roach distribution in San Francisco Bay tributaries, especially those in the south bay, which flow through a dense urban matrix, are limited by channelization and water pollution.

Mining. Roach are typically absent from streams heavily influenced by mining, especially instream mining. For example, they were apparently eliminated from the South Fork Yuba River because of hydraulic mining in the 19th century and have apparently been unable to recolonize (Gard 2004), in spite of 150+ years of generally favorable habitat recovery.

Estuarine alteration. Historically, exchange between San Francisco Estuary watersheds was facilitated by a band of seasonal freshwater wetlands at the fringe of the estuary that periodically connected the surface waters of tributaries, especially in South San Francisco Bay (Snyder 1905, Leidy 2007). These wetlands and dispersal pathways have been almost entirely eliminated; drained and paved over by urban development, leading to the isolation of roach populations.

Transportation. Many streams and rivers within the historic range of roach have one or more adjacent roads, often leading to the channelization of streambeds, simplification of aquatic habitats, and increased input of sediments and pollutants. Culverts and other road crossing may also form barriers to upstream fish movement, which can lead to the isolation of stream reaches and roach populations. If local populations are extirpated, such passage impediments may prevent natural recolonization of suitable habitats.

Fire. Central California roach are distributed in streams frequently affected by fires. Their continued presence in such areas indicates adaptation to stochastic events, including fire. However, fire intensity and frequency have increased because of human changes to the landscape, coupled with long-standing policies to suppress wildfires which have led to increased fuels in many areas. In conjunction with the predicted outcomes of climate change, if roach are eliminated due to the direct or indirect impacts from fire, recolonization may be inhibited by other changes to streams (reduced base flows, dams, diversions, passage barriers, etc.).

Alien species. Central California roach cannot coexist with large populations of alien fishes, especially centrarchids such as green sunfish (*Lepomis cyanellus*) and black basses (*Micropterus* spp.). As noted, green sunfish have almost entirely replaced roach in Dye Creek, Tehama County, in the intermittent pools of the south fork, while in the cooler, more permanent north fork, sunfish have not invaded and roach still dominate (Moyle 2002). In the mainstem below the union of the forks, the two species coexist but roach are largely absent from pools. In lower Deer Creek, tributary to the Cosumnes River, El Dorado County, Moyle et al. (2003) documented roach being displaced by invading green sunfish over the course of a summer. In the rest of the Cosumnes River watershed, roach were only found in clear, cool tributaries upstream of barriers that prevented invasion of redeye bass (*Micropterus coosae*) (Moyle et al. 2003). Gard (2004) indicated that predation pressure from smallmouth bass in the lower South Fork

Yuba River limited roach distribution to a single high gradient tributary. These examples highlight the importance of barriers in preventing upstream invasions of alien species into headwater areas that maintain enclaves of native fishes. However, the potential for transportation of fishes over barriers by humans and the escape of alien fishes (usually centrarchids) from numerous stock ponds in watershed above barriers suggests that even native fish refuges protected by barriers may succumb to invasion.

Roach display few aggressive behaviors towards other fishes and are typically displaced from prime feeding habitats by more aggressive fishes (e.g., green sunfish, Moyle et al. 2003; rainbow trout, Feliciano 2004 and smallmouth bass (*Micropterus dolomieu*), Gard 2004). Even where roach are not eliminated by other fishes, they may exhibit reduced growth and survival in their presence (Brown and Moyle 1991).

	Rating	Explanation
Major dams	High	Most central California populations are isolated by major dams and reservoirs which alter flows, reduce habitat suitability and prevent movement among and between populations
Agriculture	High	Roach are generally absent from streams in intensively farmed areas; agriculture is pervasive throughout their range
Grazing	Medium	Streams in heavily grazed pastures tend not to support roach
Rural residential	Medium	Development degrades roach habitats through diversion and pollution
Urbanization	High	Roach are absent from many streams flowing through dense urban areas; many urban areas exist across their range
Instream mining	Low	Gravel quarries create lentic habitats preferred by exotic predatory fishes
Mining	Low	Little direct effect known but contamination from mine effluent likely to have negative effects on roach
Transportation	Medium	Roads often border rivers and streams throughout the range of roach, leading to habitat degradation and simplification, sediment and pollutant input, and creating potential barriers (e.g., culverts)
Logging	Low	Roach are well adapted to the shallow, warm, exposed stream conditions often found in logged areas; intensively logged habitats may lead to increased sediment input and loss of riparian shading
Fire	Low	Increased isolation of roach populations, coupled with more frequent and intense fires, may lead to localized extirpation by fire
Estuary alteration	Medium	Key freshwater habitats that historically allowed interconnection of populations have been almost entirely eliminated; especially acute in San Francisco Estuary
Recreation	Low	Stream alterations for recreation can have both positive and negative effects
Harvest	n/a	
Hatcheries	n/a	
Alien species	High	Intolerant of predatory fishes, especially centrarchids

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

Effects of Climate Change: Climate models for Central California provide scenarios that strongly indicate that the streams in which central California roach occur will become increasingly unsuitable for sustaining populations (Knowles and Cayan 2002, Miller et al. 2003, Carlisle et al. 2010, Null et al. 2012). In general, these scenarios suggest that streams, especially small streams, will become warmer by 2100 (2-6 degrees C), while base flows will be reduced, presumably drying larger portions of many streams in which roach now occur on a seasonal basis. Multi-year droughts are likely to become more frequent, while major high flow events are predicted to occur earlier and become flashier, especially during 'rain on snow events.' In short, these widely accepted scenarios indicate that streams will become more variable in flows and temperatures, with extreme conditions reached more often.

The dependence of Central California roach on small, frequently intermittent streams suggests that they are particularly susceptible to increasingly harsh environmental conditions. This is despite the fact that roach are one of the few native fishes that are able to endure life in isolated summer pools where temperatures are high, dissolved oxygen levels are low, and most other fishes die. John O. Snyder (1905 p. 332) observed roach were able to persist when "nothing remains of the stream but a few small disconnected pools." While such tenacity bodes well for roach in a future of dwindling in-stream water supplies, it also suggests that they are likely to be extirpated from streams which may dry completely under the dual strains of altered precipitation patterns and increased human water use, especially where water is diverted for residential and agricultural use.

Despite its wide distribution and tolerance to adverse conditions, Moyle et al. (2013) rated the central California roach as "highly vulnerable" to extinction in the next 100 years if present trends continue.

Status Determination Score = 3.3 – Moderate Concern (see Methods section Table 2). If central California roach are assumed to be a single taxon, there appears to be little danger of extinction in the near future (Table 2). However, it is very likely that this form actually comprises multiple taxa, some of which may be under more immediate threat. It is of particular concern that the small, isolated populations which are the most likely to be extirpated also tend to be the most distinctive (Bennett et al. 1992). Emerging appreciation of the variation with the taxon (Moyle et al. 1989, Bennett et al. 1992, Jones 2001, Jones et al. 2002, Aguilar and Jones 2009) has highlighted the need to preserve distinctive populations endemic to specific areas.

Metric	Score	Justification
Area occupied	5	Sacramento, San Joaquin and San Francisco Bay drainages support many apparently independent populations
Estimated adult abundance	5	Many large populations
Intervention dependence	3	Increased isolation may limit recolonization after localized extirpation and necessitate deliberate reintroductions from adjacent watersheds
Tolerance	4	Broad environmental tolerances but vulnerable to exclusion by introduced fishes
Genetic risk	3	Distinct subpopulations are increasingly at genetic risk from small size and isolation
Climate change	2	Drying and warming of streams may eliminate many populations, particularly those isolated in small headwater tributaries
Anthropogenic threats	1	See Table 1
Average	3.3	23/7
Certainty (1-4)	3	Little information on abundance; taxonomy uncertain

Table 2. Metrics for determining the status of central California roach in California, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: The California roach species complex, in general, and the Central California roach, in particular, are in need of comprehensive study of their abundance, status, distribution and systematics. A thorough review of systematics is highlighted by the discovery of the Red Hills roach and indications that a number of undescribed forms likely exist around the state (Hopkirk 1973, Bennett et al. 1992, Jones et al. 2002, Aguilar and Jones 2009). One of the greatest threats facing Central California roach is our limited understanding of roach systematics, which may lead to the prospect of losing distinct taxa before they are described. A clear parallel exists in the lack of protection provided to distinct but undescribed populations of other widespread California fish species. For example, in Clear Lake, Lake County, formal description came too late to contribute to the conservation of the endemic Clear Lake splittail (*Pogonichthys ciscoides,* Hopkirk 1973); it became extinct almost a decade before being described.

Although roach populations remain geographically widespread, their status should be closely monitored in order to ensure that current population levels are maintained. Consideration should also be given to the possible reintroduction of roach into watersheds with suitable habitats in which they were historically present but have since been extirpated. Where possible, roach reintroductions should come from immediately adjacent watersheds, as proximate populations are more likely to have adaptations for local conditions. For example, roach could be reintroduced into the South Fork Yuba

River by using fish from Kentucky Ravine, which is an isolated tributary that supports the sole remaining population in the watershed.

Aquatic Diversity Management Areas, or protectively managed streams, should be established throughout the range of the Central California roach in order to protect roach genetic diversity and provide sanctuary for other California fishes, amphibians and aquatic invertebrates (Moyle et al. 1995).

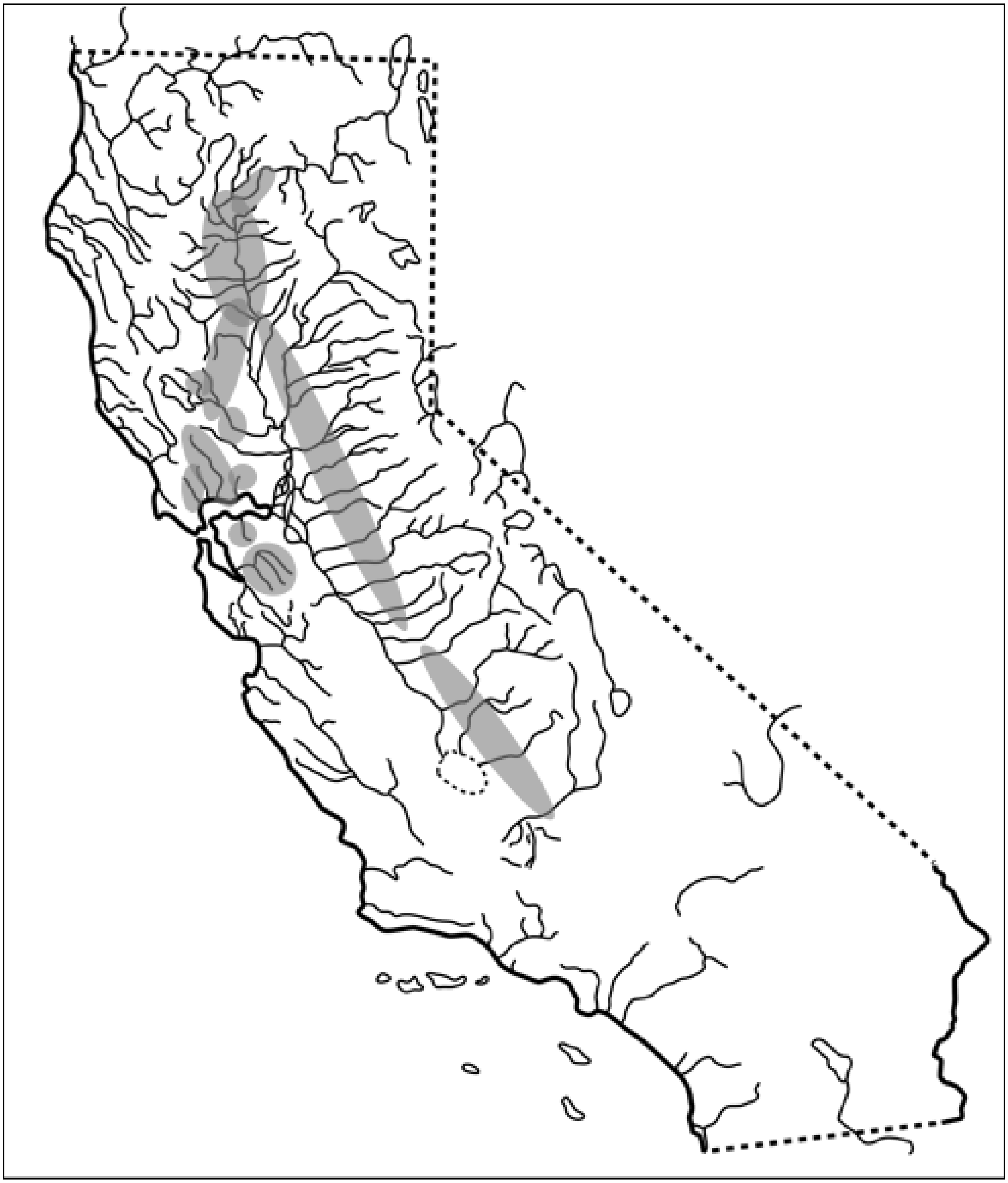


Figure 1. Historic distribution of Central California roach, *Lavinia symmetricus symmetricus* (Baird and Girard), in California. Current distribution is highly fragmented.

RED HILLS ROACH
Lavinia symmetricus ssp.

Status: High Concern. Red Hills roach have an extremely limited range in habitats that can easily become dewatered. Continued monitoring is needed to ensure their persistence, with a rescue plan in place if their streams are threatened with drying.

Description: The Red Hills roach is a small, bronzy minnow. In Horton Creek, the average length of adult specimens was 52 mm SL (Brown et al. 1992). They can be differentiated from neighboring populations of *Lavinia* by their shallower body profile, smaller interorbital distance, and fewer pectoral and pelvic fin rays (Jones et al. 2002). Red Hills roach exhibit a much higher frequency of a cartilaginous projection on the lower lip referred to as a "chisel lip" than any other roach population known (Brown et al. 1992). However, there appears to be substantial temporal variation in frequency of the chisel lip condition (Jones et al. 2002). Brown et al. (1992) suggested that the flattened body morphology of the Red Hills roach is reminiscent of that of speckled dace.

Spawning coloration is as follows: "The body is dark brown to brassy above, dark black lateral band, and brilliant white below, splashed with black blotches on the sides. Dorsal and caudal fins [are] dark olive-brown to reddish-brown, with the rays often deep-olive and with the nearly clear interradial membranes faintly flushed with brassy color; pectoral fins [are] yellowish with orange-red axils and very strong orange coloration at base; anal and pelvic fins [are] bright orange-red at the base with lessening coloration towards the rays. Cheeks and operculars with strong gilt reflections; strong orange coloration is found on the edges of the mouth (especially in males) with some blending into the upper mouth region. Lateral line [is] more strongly gilt than adjacent parts of body, thus often obscuring the lateral line. In females, the coloration is similar but less intense except for the orange coloration at the base of the paired fins that appears equally intense in both sexes. Males can be distinguished primarily by breeding tubercles on the top of the head." (W. J. Jones, pers. comm. 2009).

Taxonomic Relationships: A morphological analysis of *Lavinia symmetricus* (Brown et al. 1992) first suggested the existence of an unrecognized taxon of *Lavinia* in Horton Creek, a tributary to Don Pedro Reservoir on the Tuolumne River, Tuolumne County. A multivariate analysis of fifteen morphological characters found Red Hills roach to diverge significantly from populations in eight tributary drainages of the San Joaquin River, as well as from other populations in the Tuolumne watershed (Brown et al. 1992). Subsequent studies found Red Hills roach to be reciprocally monophyletic for the mitochondrial DNA (mtDNA) haplotypes assayed and distinct from adjacent populations in both the Stanislaus and Tuolumne river drainages (Jones et al. 2002). Additional morphometric analysis also found roach from all tributaries to Six Bit Gulch, including Horton Creek, to group together and to differ significantly from all adjacent populations (Jones et al. 2002). Mitochondrial evidence suggests past exchange between Red Hills roach populations and roach in Becca B, Hatch and Second creeks (now separated from Six Bit Gulch by Don Pedro Reservoir), but Red Hills roach have been isolated for, at minimum, 200 years (Jones et al. 2002). Moyle et al. (1995) and Moyle (2002) treat the Red Hills roach as an undescribed subspecies.

Life History: No life history studies of the taxon have been conducted. Their basic life history is presumably most similar to that of Central California roach, from which they likely differentiated. See the Central California roach account in this report for a generalized life history description.

Habitat Requirements: The Red Hills region is characterized by one of the largest outcropping of serpentine rock in the Sierra Nevada. Serpentine soils contain high concentrations of iron and magnesium and, as a result, are inhabited by predominantly endemic organisms which have evolved tolerances for such conditions. Red Hills roach occur in the spring-fed intermittent creeks of Six Bit Gulch, which is the primary drainage of the Red Hills (Jones et al. 2002). Red Hills roach are found in several pools and perennial stream reaches fed by springs (W. Jones pers. comm. 2009; field observations by authors, 2010). During summer, roach are confined to these few localities of perennial water but, during higher spring flows, they move upstream to spawn (W. J. Jones pers. comm. 2009).

Other fishes that can co-occur with Red Hills roach include native Sacramento sucker, *Catostomus occidentalis*, rainbow trout, *Oncorhynchus mykiss,* and introduced western mosquitofish, *Gambusia affinis.* W. Jones (pers. comm. 2009) has also documented large numbers of introduced green sunfish, *Lepomis cyanellus*, in lower Six Bit Gulch.

Distribution: The Red Hills roach is confined to Six Bit Gulch and its tributary streams; Amber Creek, Horton Creek, Minnow Creek and Poor Man's Gulch (Jones et al. 2002). Six Bit Gulch enters a western arm of Don Pedro Reservoir on the Tuolumne River, near Sonora, Tuolumne County. In July, 2010, roach were observed in three discontinuous wetted reaches of Horton Creek, which covered approximately 500 meters in total wetted length (P. Moyle, unpublished observations). However, only the lower reach, which extends about 200 meters upstream from the confluence with Six Bit Gulch, appeared to be perennial as indicated by lush growth of sedge and other riparian vegetation. A natural fish barrier approximately 1.2 km upstream from the confluence likely inhibits roach from accessing upper Horton Creek. Roach were also observed in Six Bit Gulch where it is forded by Six Bit Ranch Road and in a pool in Roach Creek.

Trends in Abundance: Jones et al. (2002) estimated total abundance at 200-500 individuals. More recent abundance estimates have not been performed.

Nature and Degree of Threats: The small, intermittent streams that comprise the entirety of Red Hills roach habitat are acutely vulnerable to human alteration. While some protection is offered by the 7,100 acre Red Hills Area of Critical Environmental Concern (ACEC, Bureau of Land Management), the protected area excludes most streams, with much of the Six Bit Gulch watershed on private land. Red Hills roach are threatened by a combination of land use practices and introduced fishes. These factors, in conjunction with the complete isolation of Six Bit Gulch from other roach populations and predicted outcomes of climate change, threaten Red Hills roach with extinction.

Dams. The construction of Don Pedro Dam in 1923 created Don Pedro Reservoir and flooded the lower portion of Six Bit Gulch. The reservoir is thought to be a barrier to long-distance roach dispersal and, therefore, effectively isolates Red Hills roach from all other roach populations. The reservoir also fragments Red Hills roach populations by isolating Poor Man's Gulch from Six Bit Gulch.

Grazing. The Red Hills are poor grazing lands because of the unique serpentine soils and sparse vegetation communities but even limited grazing can damage aquatic habitats because cattle concentrate around scarce water sources. Observations in July, 2010, indicated that grazing impacts were minimal but even a few head of cattle could have a major impact on the limited riparian and aquatic habitats available to Red Hills roach. Grazing may cause stream bank collapse, pool sedimentation, eutrophication from animal wastes, and reduction or elimination of already scarce cover and shading. Impacts from grazing would likely be particularly acute in summer and fall months, when perennial aquatic habitats are restricted to a few isolated pools.

Rural residential development. The foothills of the Sierra Nevada are being rapidly developed for dispersed rural residences and private lands in the Red Hills watershed are vulnerable to development, in spite of the unique landscape that features poor soils, few trees, and shortage of water. Tuolumne County is one of California's fastest growing counties and much of the Red Hills region is threatened by development. Residential development can threaten roach through water diversion during low-flow periods, pollutants (especially inflow from septic tanks), siltation from roads, and loss of complex habitat through bank stabilization projects.

Transportation. Most of the stream courses within the Red Hills roach range are lined by roads, which may contribute to increased sedimentation, channelization and pollution input.

Mining. Historic placer mining in the Red Hills region dramatically altered the hydrology and geomorphology of streams and introduced vast amounts of sediment into the Tuolumne River and its tributaries. However, the legacy effects of landscape-scale alteration to the watershed are unknown.

Recreation. Off-road vehicle use and other human recreational activities that damage banks and streambeds or reduce riparian vegetation around Red Hills roach summer habitat are particularly serious threats (B. Quelvog, CDFW, pers. comm. 1995). Off-road vehicle use is banned in the BLM ACEC but not on private lands, which surround much of the Red Hills roach's perennial habitats.

Fire. The Red Hills area is regarded as a region of high fire risk because of naturally high flammability of the vegetation and heavy recreational use. Large fires occurred in the area in 1982 and 1997. The effects of wildfire today tend to be more frequent and severe than in the past due to human alterations to the landscape and increasingly dry conditions associated with climate change. Red Hills roach may be particularly affected by catastrophic fire, due to their limited distribution in a fire-prone region.

Alien species. The presence of green sunfish in Six Bit and Poor Man's gulches is a potentially severe threat because roach populations in other locations have been extirpated by alien fishes such as green sunfish and black basses. See the Central California roach account in this report for more detailed coverage of roach interactions with alien fishes.

	Rating	Explanation
Major dams	High	Don Pedro Reservoir isolates Red Hills roach from other populations and blocks dispersal
Agriculture	n/a	
Grazing	Medium	Little current grazing but concentrated damage potential high
Rural residential	Medium	Rural development increasing rapidly in Tuolumne Co.
Urbanization	n/a	
Instream mining	Medium	Ponds created by past instream mining provide habitat for green sunfish and other invasive fishes
Mining	Low	Legacy effects (e.g. contaminants, stream bed alteration) from past large-scale mining may continue to negatively affect roach
Transportation	Medium	Roads and off-road vehicles are both potential contributors to habitat degradation
Logging	n/a	
Fire	Medium	Increased isolation of roach populations and more frequent and/or intense fires may lead to localized extirpation
Estuary alteration	n/a	
Recreation	Medium	Portions of range now protected (BLM ACEC) but off-road vehicles use and other activities on private lands may pose threats
Harvest	n/a	
Hatcheries	n/a	
Alien species	High	Intolerant of predatory fishes, especially centrarchids

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

Effects of Climate Change: Because they persist in isolated pools during low flow periods, Red Hills roach are particularly susceptible to increasing aridity associated with climate change predictions. Springs that provide pool habitats may be altered by human land use practices or naturally dry if the climate becomes more arid. While roach are one of the few native fishes in California that can persist in isolated pools in intermittent streams, they may become extirpated from Red Hills streams by predicted decreases in precipitation and increasing temperatures, along with increasing demand for human water use. Moyle et al. (2013) rated the Red Hills roach as "critically vulnerable" to extinction from the effects of climate change, in combination with other factors that threaten it.

Status Determination Score = 2.1 – High Concern (see Methods section Table 2). Red Hills roach have an extremely limited distribution and persist in isolated summer pools fed by springs of indeterminate source. Their persistence is threatened by fire, depleted stream flows, lack of protections on private lands, and, especially, invasive fishes. While some habitat is protected in the Red Hills ACEC, much is on private land and remains unprotected. The Red Hills roach is listed by the American Fisheries Society as "Vulnerable" (Jelks et al. 2008), by NatureServe as "G5T1, Critically Imperiled" and by the Bureau of Land Management as "Sensitive".

Metric	Score	Justification
Area occupied	1	Restricted to a single, small, fragmented, and intermittent drainage
Estimated adult abundance	1	Populations small and fragmented
Intervention dependence	3	Isolation limits recolonization after local extinctions occur and may necessitate deliberate reintroductions from nearby populations
Tolerance	5	Broad environmental tolerances
Genetic risk	2	Genetic risks from fragmentation, genetic drift, and isolation
Climate change	1	Increased aridity and decreased precipitation in the region could dry streams and standing pools completely
Anthropogenic threats	2	See Table 1
Average	2.1	15/7
Certainty (1-4)	3	Good documentation but little recent (since 2002) information

Table 2. Metrics for determining the status of Red Hills roach, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: The Bureau of Land Management recognized the ecological values of the area in which Red Hills roach occur and set aside 7,100 acres as the BLM ACEC to provide habitat for "the unique flora of the region, habitat for …the Red Hills roach and to protect Bald Eagle wintering habitat." http://www.blm.gov/pgdata/etc/medialib/blm/ca/pdf/folsom/gis_pdf_maps. Unfortunately, most roach habitat is not in the ACEC but, rather, is on private land along roads. Therefore, the main drainage of the Red Hills region, Six Bit Gulch, in addition to the majority of Horton Creek, has no formal protection. The most important conservation action to protect Red Hills roach is to expand the ACEC to include these drainages.

Other management recommendations include:

- Decommission roads that run along stream courses in the area.
- Develop and implement a plan to systematically remove alien fishes from streams and build barriers to prevent re-invasion from downstream reaches.
- Develop a monitoring program for fish populations (abundance, distribution, trends), stream flows, habitat quality and dry season habitat extent, in order to

develop recommendations to improve management of roach populations.

- Develop an emergency plan, including identification of refuge sites or captive rearing options, in the event population levels become critically low.
- Conduct studies of Red Hills roach life history, habitat requirements, and general ecology.
- Publish a formal description of the Red Hills roach as a distinct taxon so targeted conservation actions and associated funding can be identified and implemented.
- Engage stakeholders (especially private land owners) to develop collaborative conservation measures that will protect and enhance Red Hills roach habitats. Consider development of conservation agreements or identify funding for acquisition of lands from willing landowners utilizing conservation easements.
- Improve enforcement to reduce damage to streams, particularly on public lands within the ACEC.

Figure 1. Distribution of Red Hills roach, *Lavinia symmetricus* ssp., in California.

RUSSIAN RIVER ROACH
Lavinia symmetricus ssp.

Status: Moderate Concern. Although apparently in no danger of extinction, Russian River roach populations could decline or become extirpated from large portions of their range as result of alterations to streams, changes in climate, water withdrawal for urbanization and rural residential development, as well as water demands and pollutant runoff associated with rapidly expanding viticulture.

Description: Russian River roach are a small (adult size typically 50-120 mm), bronzy minnow (cyprinid), which are very similar to the Central California roach. However, they differ in having a trim, slender body, a somewhat pointed snout, a slender caudal peduncle and long fins. Russian River roach have a mean of 8.7 dorsal fin rays and 8.1 anal fin rays (Hopkirk 1973). Individuals rarely exceed 120 mm; the largest roach captured during a 2007 survey in Austin Creek, a tributary to the Russian River (Sonoma County), was 116 mm and weighed 20.5 g (Figure 1). The following account of roach morphology is based on information from roach populations outside the Russian River watershed.

Roach are small, stout-bodied, minnows with a narrow caudal peduncle and a deeply forked tail. Fish rarely achieve lengths greater than 100 mm total length. The head is large and conical. The eyes are large and the mouth is subterminal and slants at a downward angle. The dorsal fin is short (7-9 rays) and is positioned behind the insertion point of the pelvic fin. The anal fin has between 6-9 rays. The pharyngeal teeth (0,5-4,0) have curved tips with overhanging grinding surfaces of moderate size. Roach are usually dark on the upper half of their bodies, ranging from a shadowy gray to a steel blue, while the lower half of the body is much lighter, usually a dull white/silver color. The scales are small, numbering 47-63 along the lateral line and 32-38 before the dorsal fin. Roach exhibit general (non-nuptial) sexual dimorphism (Snyder 1908d, Murphy 1943). Snyder (1905, 1908d) demonstrated that the sexes could be differentiated by the ratio of pectoral fin length to body length. Males exhibit a ratio of $\geq .21$, while females have pectoral fins between .16 and .20 the length of their body. Both sexes exhibit bright orange and red breeding coloration on the operculum, chin and the base of the paired fins. Males may also develop numerous small breeding tubercles (pearl organs) on the head (Murphy 1943).

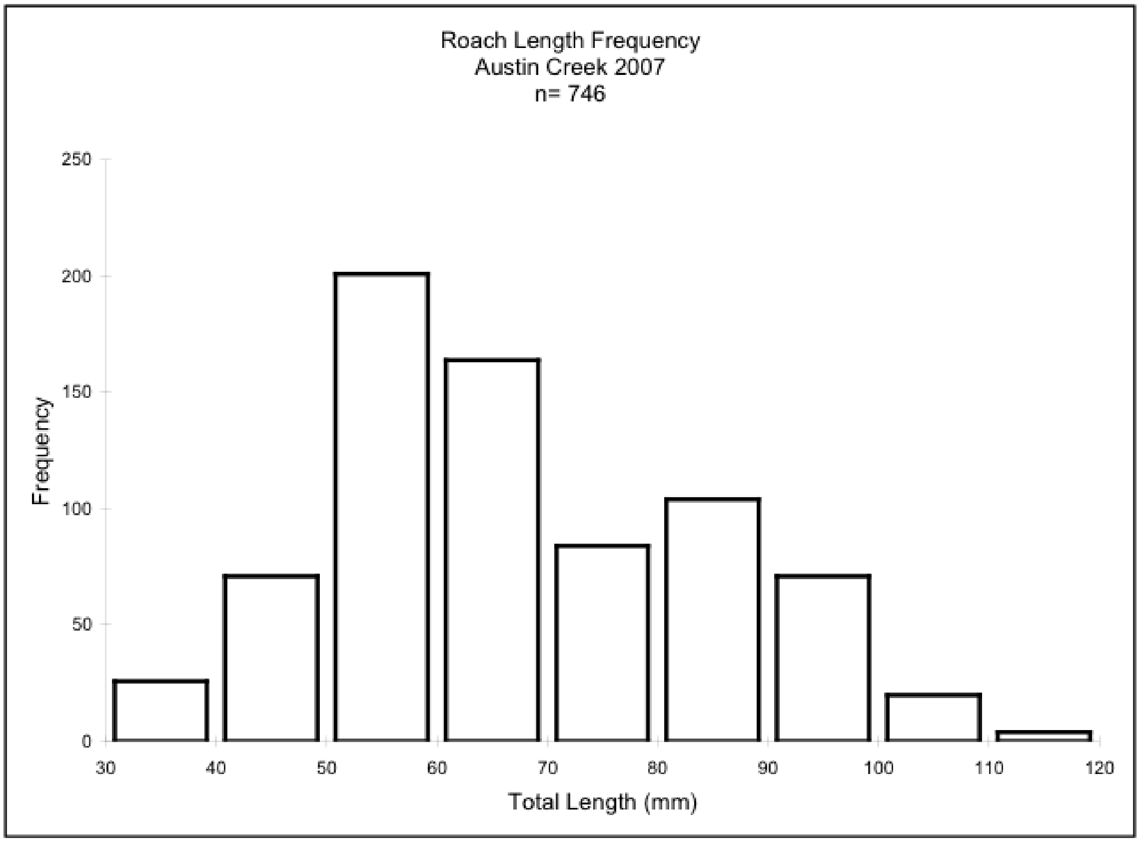


Figure 1. Length frequency of Russian River roach greater than 30mm, from Austin Creek, Sonoma County, February 15–June 15, 2007. J. Katz, unpublished data.

Taxonomic Relationships: Russian River roach were first collected by Snyder (1908d, p. 175) who recognized them as *Rutilus symmetricus* and found them to be "alike in all respects" to *R. symmetricus* from the Napa River and to "agree closely with representatives from the streams tributary to San Francisco Bay." In 1913, Snyder revised the systematics of roach, describing six species and erecting a new genus, *Hesperoleucus*, to house them. True to his initial assessment, Snyder placed Russian River roach into the species *H. venustus* along with roach from the streams entering San Pablo, Suisun, and San Francisco bays. In a footnote from a paper on hybridization between hitch (*Lavinia exilcauda*) and roach in the Monterey basin, Miller (1945) suggested that Snyder's roach species should be treated as geographic subspecies. In an unpublished M.S. thesis, Murphy (1948) agreed with Miller and concluded that all coastal species of roach should be reduced to subspecific status of *H. symmetricus*. Although critical of Murphy's reasoning, Hopkirk (1974) agreed with the diagnosis of placing all roach within one species. He reached different conclusions, however, as to which roach populations belonged to which subspecies. It is worth noting that Hopkirk (1974) asserted that roach from the Russian River were more closely related to those from the Navarro River than they were to populations from the San Francisco Bay region. Roach from the Russian River are presumed to be the parent stock of roach in the Gualala and Navarro rivers, to which they were likely transferred by headwater stream capture.

Based on the morphological evidence presented by Hopkirk (1973), Moyle et al. (1995) moved roach from the Russian River into *L. s. navarroensis*. However, subsequent genetic analysis (Jones 2001), using mitochondrial DNA (mtDNA), found that roach from the Russian River were more closely related to roach from Clear Lake than they were to those from the Navarro River, leading Moyle (2002) to propose grouping roach from the Russian River and Clear Lake roach as a new subspecies. The most recent genetic analysis (Aguilar and Jones 2009) used both mtDNA and nuclear DNA microsatellites (nDNA). The mtDNA analysis found that a number of mtDNA haplotypes were shared by fish from the Russian River and Clear Lake, adding support to their grouping as a common lineage. The microsatellite analysis, on the other hand, suggested that roach from the two basins should be treated as separate taxonomic entities.

As the mixed results suggest, there is considerable confusion regarding the interpretation of genetic information in the roach/ hitch species complex (Avise 1975, Avise and Ayala 1976, Jones 2001, Aguilar and Jones 2001). In light of remaining uncertainties, the precautionary approach is to treat the Russian River roach and Clear Lake roach as distinct and separately managed taxa until sufficient evidence is presented to determine otherwise.

Interestingly, Murphy (1948c) noted that roach from Austin Creek were the most morphologically divergent of all roach populations sampled from the watershed. No genetic studies have included roach from Austin Creek so the distinctiveness of this population has not been verified. However, Murphy's morphometric evidence suggests that the Russian River, like the Central Valley, may contain distinct roach populations endemic to tributary watersheds. An additional biogeographical consideration is the fact that Austin Creek shares a watershed boundary with the Gualala River, which contains a genetically divergent roach population. The geographic proximity of these two basins and the distinctive nature of their respective roach populations highlights the need for a thorough taxonomic study of all coast range roach populations.

See the Central California roach account in this report for a general description of roach systematics in California

Life History: The life history of the Russian River roach is largely unstudied. It is reasonable to assume that its life history is similar in most respects to that of the similar Central California roach, presented in this report.

Russian River roach spawn in spring and early summer, after water temperatures exceed 16°C, although spawning activity has been observed well into July (Moyle 2002). Length frequencies (Figure 1) suggest that they rarely live more than three years. They are presumably omnivorous, similar to other populations of California roach although, in Austin Creek, roach were observed taking mayflies at the surface, much like rising trout (J. Katz, pers. obsv.).

Habitat Requirements: Roach are found in a wide variety of habitats in the Russian River, including the main river where there is cover (e.g. fallen trees) to protect them from predators. They are most abundant, however, in tributaries. Pintler and Johnson (1958) found that roach accounted for between 45% and 60% (average 54%) of numeric fish abundance). Likewise, Price and Geary (1978, 1979) found that Russian River roach were frequently the dominant fish in small (0.025-0.10 m^2/sec summer flows) tributary streams with clear, well oxygenated, water, dominant substrates of cobble and boulder, and shallow depths (average 10-50 cm) with pools up to 1 m deep. Temperatures in these tributaries rarely exceed 25°C because they are well shaded. Roach and Sacramento suckers were the primary fishes documented in pools created by recreational summer dams on these tributaries (Cox 1984).

In the Russian River mainstem, roach are most common around the mouths of tributaries (Pintler and Johnson 1958). In beach seine surveys, Cook et al. (2003-2007) found that the fish assemblage at the confluence of Austin Creek was dominated by steelhead, tule perch, and roach, in order of abundance. Hopkirk et al. (1980) found that roach constituted a minor part of the large schools of juvenile cyprinids and Sacramento suckers (*Catostomus occidentalis*), which were numerous in the shallow side-channels of the mainstem in spring. It is possible that the distribution and abundance of roach in the Russian River is limited by the presence of alien predators, mainly smallmouth bass (*Micropterus dolomieui*) and green sunfish (*Lepomis cyanellus*).

Distribution: Russian River roach are restricted to the Russian River and its tributaries. They are among the dominant species in the middle sections of many tributary creeks, including Mark West and Santa Rosa creeks (Chase et al. 2005), Maacama Creek (Merrit Smith Consulting 1995, 2003), Austin Creek (Katz et al. 2006, 2007) and Big Sulphur and Pieta creeks (Price and Geary 1978, 1979).

In the mainstem Russian River, they account for only a small percentage of the fish assemblage in the middle and lower reaches (Pintler and Johnson 1958, Chase et al. 2001-2005, Cook et al. 2005). Roach become increasingly rare in the lower sections of the main river, where their downstream limit appears to be the upper portions of the estuary near Duncans Mills. However, Goodwin et al. (1993) found that roach tended to move down into the main body of the estuary and Willow Creek marsh during the summer and return to upstream habitats in the fall.

Trends in Abundance: Little is known about Russian River roach abundance trends since few survey data exist. As such, there is no indication that they are less abundant now than in the past; however, population monitoring is needed in order to establish baseline abundance and trend information.

Nature and Degree of Threats: Although resilient, Russian River roach may be negatively impacted in streams that are: (1) dewatered for residences, vineyards, pasture and other uses, (2) heavily altered by channelization, and (3) invaded by alien predators such as green sunfish (*Lepomis cyanellus*).

Agriculture. In the Russian River basin, the high rate of conversion of forestland to vineyards is a principal threat to native fishes. Forestland conversion to viticulture (hillside vineyards in particular) directly impacts flow in small streams. Deitch et al. (2009a,b) showed that vineyard irrigation and frost protection significantly reduce in-stream flow in Russian River tributaries and a regulatory process has been implemented to mitigate impacts. In spite of mitigation measures, widespread alteration to basin hydrology and aquatic ecosystems from vineyard conversion may remain an ongoing threat. Marijuana cultivation may also pose a threat to the fishes of the Russian River drainage, although no studies to demonstrate potential impacts have been performed in this area.

	Rating	Explanation
Major dams	Low	Dams control flows in the Russian River, altering the natural hydrograph and water quality
Agriculture	Medium	Water demands increasing for viticultural irrigation and frost protection
Grazing	Medium	Grazing is a major landscape use with negative effects on small streams
Rural Residential	Medium	Diversions or pumping from shallow wells can reduce flows or dewater streams
Urbanization	Medium	Santa Rosa and surrounding communities contribute to water withdrawal, pollution and altered aquatic habitats
Instream mining	Medium	Gravel mining in the main river simplifies habitat and increases turbidity
Mining	Low	Legacy effects from past mining plus hardrock mining for aggregate
Transportation	Medium	Watershed heavily roaded; associated impacts from siltation, channelization and habitat loss
Logging	Low	Mostly legacy effects; current timber harvest levels greatly reduced from past
Fire	Low	Fire frequency and intensity may increase with land use alterations and climate change
Estuary alteration	n/a	
Recreation	Low	Heavy recreational use; little direct threat
Harvest	n/a	
Hatcheries	n/a	
Alien species	High	Intolerant of introduced predatory fishes, especially centrarchids (e.g., smallmouth bass)

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

Grazing. Livestock grazing is a major land use of the Russian River watershed and heavy grazing by cattle can cause stream bank sloughing, stream incision, loss of riparian vegetation, and may contribute to reduced flows or earlier drying of tributary streams.

Rural residential. Residences are scattered throughout the watershed and diversions or pumping from shallow wells can dewater streams or reduce flows. Septic tank effluent may be a localized impact in some portions of the watershed.

Urbanization. Santa Rosa and surrounding communities are growing rapidly with increased water demand, associated alteration to streams and aquatic habitats, and pollution through urban run-off.

Instream mining. Gravel mining has long been a contentious practice in the Russian River, although gravel is increasingly mined outside of the stream channel. Gravel mining in the main river simplifies habitat and increases turbidity, which may reduce habitat suitability for roach and other fishes.

Transportation. Much of the Russian River and its tributaries are bordered by roads, potentially resulting in increased siltation, channelization, pollutant input, and habitat loss or degradation.

Alien species. Non-native fishes are increasingly common in the Russian River watershed, especially in ponds and reservoirs. Long-established smallmouth bass populations are likely restricting roach distribution in the main river. Invasions of green sunfish have the potential to eliminate roach from small streams, as has been documented elsewhere in California (Moyle 2002).

Effects of Climate Change: Russian River roach are well adapted to the warm, arid conditions of California's Mediterranean climate but their restriction to intermittent pools during drought periods suggests that they may be particularly susceptible to increasing aridity associated with climate change. Roach are one of the few native fishes that are able to endure life in isolated summer pools in intermittent streams, where temperatures increase, dissolved oxygen levels drop, and most other fishes die. John O. Snyder (1905) observed roach were able to persist when "nothing remains of the stream but a few small disconnected pools." While such tenacity bodes well for roach in a future of dwindling in-stream water supplies, it also suggests that they may be extirpated from streams which may dry completely under the dual strains of decreased precipitation and increased human water demand, including surface and ground water withdrawal for vineyard expansion and rural residential development. The increasingly stressful conditions likely to be found in the Russian River and its tributaries, as the result of climate change acting in concert with urban and agricultural development, led Moyle et al. (2013) to rate the Russian River roach as "highly vulnerable" to extinction as the result of climate change, if present trends continue.

Status Determination Score = 3.3 - Moderate Concern (see Methods section Table 2). Russian River roach do not appear to be in danger of extinction, although gradual loss of tributary populations, combined with changes to the main river itself that have largely eliminated connectivity between tributaries, may limit distribution, reduce abundance, and impede or prohibit natural recolonization. The Russian River roach is listed as "G5T2T3, Imperiled" by NatureServe, where it is included in a taxon described as the Clear Lake-Russian River roach subspecies.

Metric	Score	Justification
Area occupied	3	Widespread in the Russian River and its tributaries
Estimated adult abundance	5	Populations large and numerous
Intervention dependence	3	Protection of small streams (tributaries) needed
Tolerance	4	Remarkably resilient fish
Genetic risk	3	Little threat to genetic integrity if assumed to be a single taxon, although may be multiple taxa
Climate change	3	Complete drying of intermittent streams may extirpate roach from tributary watersheds
Anthropogenic threats	2	See Table 1
Average	3.3	23/7
Certainty (1-4)	2	Little published information

Table 2. Metrics for determining the status of Russian River roach, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: A comprehensive survey of the Russian River watershed is needed in order to determine Russian River roach distribution and abundance. In particular, streams that have been previously surveyed should be resurveyed to establish trend information. Further genetic studies are needed in order to clarify taxonomic relationships. Streams with intact habitats and minimal stressors should be selected as refuges for native fishes and amphibians and managed accordingly. Opperman and Merenlender (2004) studied Russian River tributaries and provided management recommendations, including maintaining live trees (live woody debris) in riparian zones and in stream channels to create pool habitats that roach prefer.

Understanding the relationship between groundwater withdrawal and stream flow in Russian River tributaries is of prime importance to the management of native fishes in the basin. Merenlender et al. (2008) developed GPS-based water resource analysis tools designed to quantify and balance water needs and water resources on a watershed scale. These tools were created to aid in sustaining instream flow while simultaneously enhancing water security for local landowners and vineyard operators. Applications include evaluation of various water-policy scenarios, estimation of the cumulative effects of water extraction methods on the natural hydrograph across a large spatial scale (including temporal variation), and deriving information for watershed-level planning required to recover environmental flows. If utilized in conjunction with quantitative surveys of streamflow and groundwater withdrawal, these analytical tools may be of great value in the Russian River watershed (and others) where human water demand may exceed supply and reduction in instream flows may be exacerbated by predicted climate change impacts.

Figure 2. Distribution of Russian River roach, *Lavinia symmetricus* ssp., in California.

CLEAR LAKE ROACH
Lavinia symmetricus ssp.

Status: Moderate Concern. Although apparently in no danger of extinction, isolated populations of Clear Lake roach could decline rapidly and disappear as the result of changes in climate, alterations to streams and water withdrawal for urbanization, rural residences, and agriculture (especially vineyards).

Description: Clear Lake roach are a small (adult size typically 50-100 mm), bronzy cyprinid very similar to the Central California roach in appearance. Clear Lake roach have 8-10 dorsal fin rays (mean 8.6) and 7-9 anal fin rays (mean 8.0) (Hopkirk 1973). The head is large (ca. 3.75 into standard length) and conical. The dorsal fin is positioned behind the insertion point of the pelvic fin. The eyes are small to moderate in size, the snout is short, and the mouth is subterminal, slanting at a downward angle. The pharyngeal teeth (0,5-4,0) have curved tips which overhang grinding surfaces. Roach are usually dark on the upper half of their bodies, ranging from a shadowy gray to a steel blue, while the lower half of the body is much lighter, usually a dull white/silver color. The scales are small, numbering 49-58 (mean 52.7) along the lateral line.

Roach exhibit general (non-nuptial) sexual dimorphism (Snyder 1908a, Murphy 1943). In tributaries to San Francisco Bay, Snyder (1905, 1908a) demonstrated that the sexes could be differentiated by the ratio of pectoral fin length to body length. Males exhibited a ratio of $\geq .21$ while females bore pectoral fins between .16 and .20 the length of their body (standard length). Both sexes exhibit bright orange and red breeding coloration on the operculum, chin and the base of the paired fins. Males may also develop numerous small breeding tubercles (pearl organs) on the head (Murphy 1943).

Taxonomic Relationships: Clear Lake roach were first mentioned by J. O. Snyder (1908d). He recognized them as *Rutilus symmetricus* and found them to be similar to roach from other Sacramento Valley tributaries. While he documented the considerable morphological differences between populations in the Sacramento and San Joaquin valleys, Snyder felt he did not have adequate collections from many inland populations to determine their relationships. Referring to differences among Central Valley roach populations Snyder (1908d, p. 175) stated: “whether any geographical significance can be attached to these can not be known until more extensive observations have been made.” By 1913, Snyder had acquired more roach samples but the collections were primarily from coastal basins. Consequently, when he revised the taxonomy of roach, he added four new species from coastal watersheds but only a single species from inland waters, the northern roach, from the upper Pit River and Goose Lake watershed (Snyder 1913). The Clear Lake population was not included in Snyder’s re-evaluation and was, therefore, by default, grouped with other *H. symmetricus* populations from the Central Valley.

In retrospect, it appears that Snyder’s focus on coastal populations steered the study of roach systematics away from inland populations, which received scant attention in the literature for the next half-century, even as considerable controversy embroiled the taxonomic status of coastal species. In a footnote in a paper on hybridization between hitch (*Lavinia exilicauda*) and roach in the Monterey basin, Miller (1945) suggested that Snyder’s roach species should be treated as “geographic subspecies.” In an unpublished

M.S. thesis, Murphy (1948) agreed with Miller and concluded that all coastal species of roach should be reduced to subspecific status of *H. symmetricus*. Murphy concluded Clear Lake roach were related to *H. symmetricus* from the Sacramento Valley, dismissing the phylogenetic significance of past hydrologic connection between the Russian River and the Clear Lake basin (Holway 1907, Snyder 1908a) by stating that any roach transferred from the Russian River to Clear Lake would have been "genetically swamped". Although critical of Murphy's reasoning, Hopkirk (1973) agreed with the diagnosis of placing all roach taxa within one species and that roach from Clear Lake were morphologically more similar to Central California roach than to Russian River roach.

While there remains considerable uncertainty regarding the interpretation of genetic information in the roach/ hitch species complex (Avise et al.1975, Avise and Ayala 1976, Jones 2001, Aguilar and Jones 2001), recent genetic evidence points to the close association of Clear Lake and Russian River roach populations. Mitochondrial DNA (mtDNA) analysis (Jones 2001) found that roach from the Russian River were closely related to roach from the Clear Lake basin, leading Moyle (2002) to propose the Russian River-Clear Lake roach as a new subspecies. The most recent genetic analysis used both mtDNA and nuclear DNA microsatellites (nDNA) (Aguilar and Jones 2009). The mtDNA analysis found that a number of mtDNA haplotypes were shared by fish from tributaries from the Russian River and Clear Lake, adding support to their grouping as a common lineage. The microsatellite analysis, however, provided greater resolution and suggested that roach from the Russian River and Clear Lake basins should be treated as separate taxonomic entities. Acknowledging that the systematics are still in flux, this account takes the precautionary approach of treating both the Clear Lake and Russian River roach as separate taxa.

Life History: Clear Lake roach presumably share much of their life history with the closely related Central California roach (Moyle 2002) but little information exists and their life history needs further research.

Habitat Requirements: Clear Lake roach occupy diverse stream habitats, from cool headwater reaches, where they are found with rainbow trout (*Oncorhynchus mykiss)* to warm, low-elevation mainstem reaches, where they associate with Sacramento pikeminnow (*Ptychocheilus grandis*) and Sacramento sucker (*Catostomus occidentalis*). They are most abundant in warm, exposed, mid to low-elevation stream reaches where they prefer quiet water, especially pools (Taylor et al. 1982). In the Clear Lake basin, roach abundance is positively correlated with stream temperature, conductivity, gradient, coarse substrates and bedrock, and negatively correlated with depth, cover, canopy (shade), and fast water (Taylor et al. 1982). It has been suggested that alteration of spawning and rearing habitats in the lower reaches of Clear Lake tributaries by agricultural land uses contributed to the decline or extinction of many of the lake's stream spawning native cyprinids, including the lake population of Clear Lake roach (Murphy 1948b, Hopkirk 1988). Agriculture has likely contributed to higher amounts of fines deposited over rocky riffle substrates where roach prefer to spawn.

At times, roach have been found at extraordinarily high densities (157 gram roach biomass/ cubic meter) in pools of intermittent streams where high temperatures, paired with low dissolved oxygen, tend to be lethal to other fish species (Taylor et al. 1982). Consequently, roach are often the first fish to recolonize stream reaches when surface flows resume in late fall.

Distribution: Clear Lake roach are restricted today to the tributaries of Clear Lake, where they are widely distributed in the basin's seven major drainages. They were presumably native to the lake as well (Stone 1873), using it mainly for dispersal, but there are no recent collections from the lake; roach are now unable to occupy the lake because of their vulnerability to alien predators (Moyle 2002).

Roach were found in 46% of 120 sites sampled by Taylor et al. (1982) in the seven major drainages of the Clear Lake Basin which include: (1) Seigler Creek, (2) Cole Creek, (3) Kelsey Creek, (4) Adobe Creek, (5) Highland Creek (tributary to Adobe), (6) Scotts Creek and (7) Middle Creek. All streams except Cole Creek become intermittent by late fall in their lower reaches. Roach were the dominant species in the middle sections of many streams, especially Seigler and Middle creeks. Roach were not found above waterfalls and other high gradient stream sections which form barriers to their upstream dispersal (Taylor et al. 1982).

Roach are common in the Cache Creek watershed; Cache Creek is the outlet of Clear Lake. However, the taxonomic relationship of these fish to Clear Lake roach is not known.

Trends in Abundance: Livingston Stone (1873) noted that roach were present in Clear Lake in "vast abundance" in shallow water. While Stone could have mistaken juvenile Clear Lake hitch or splittail for roach, he also noted the presence of both of these species as well. Today, Clear Lake roach presumably continue to maintain large populations in many tributary systems. However, systematic surveys have not been performed since the study of Taylor et al. (1982).

Nature and Degree of Threats: While roach are very resilient, they tend to disappear from streams that are heavily altered or dewatered for residences, vineyards, or pasture, as well as those invaded by alien predators such as green sunfish (*Lepomis cyanellus*) (Table 1).

Major dams. Cache Creek Dam was built on upper Cache Creek in 1914 to provide water for Yolo County agriculture. The dam raised maximum lake level and causes Clear Lake to fluctuate more than it did historically. It is unlikely that the dam itself was a substantial factor in the extirpation of roach from the lake; however, it does block any potential upstream dispersal of roach from Cache Creek. Smaller dams, such as Kelsey Dam on Kelsey Creek, also impede fish movement, leading to isolation of stream reaches and increasing the chances of extirpation because they often prevent recolonization from nearby populations.

Agriculture. The high rate of conversion of oak woodlands to vineyards is likely the largest threat facing stream fishes in the Clear Lake basin today, following decades of clearing lowland areas for orchards and other agriculture. Vineyard expansion on hillslopes has a direct impact on tributary flow if surface water is used for irrigation or if groundwater extraction affects headwater springs that feed tributaries. Alterations to basin hydrology resulting from new vineyard development are of equal concern. Deitch et al. (2009a,b) showed that vineyard water use for irrigation and frost protection is significantly affecting in-stream flow in tributaries to the Russian River, Sonoma County. Clear Lake, in adjacent Lake County, has similar land uses but receives less rain, possibly exacerbating this threat.

Grazing. Heavy grazing of Clear Lake watersheds has occurred since the 1870s and has likely contributed to sedimentation and nutrient loading of the lake and its tributaries (Suchanek et al. 2002). Heavy grazing can lead to stream bank collapse, sedimentation of pools and other instream habitats, pollution from animal wastes, and reduced cover and shading. Under these conditions roach tend to disappear from streams, despite their high tolerance of adverse conditions. Stock ponds for watering cattle may divert water from streams and support populations of alien predatory fishes. These fishes (e.g., green sunfish, largemouth bass) may colonize adjacent streams if ponds spill during wet periods, competing with or preying upon roach and other native populations. See the Central California roach account in this report for more on interactions between roach and predatory fishes.

Rural residential. As Clear Lake became popular as a resort area in the 19th century, the lakeshore became increasingly developed with vacation and permanent homes. This development removed tule beds (*Schoenoplectus acutus*), which provided important habitat for fish, and filled wetlands that filtered sediment and nutrient delivery to the lake. Widespread development also lead to increased discharge of septic tank effluent and, ultimately, large-scale application of pesticides to the lake to control the native but pestiferous Clear Lake gnat (*Chaoborus astictopus*). Such factors presumably contributed to the loss of roach from the lake itself, as well as the lower ends of tributaries to the lake.

Modern rural residential development of the basin is accelerating, along with increasing human demand for water, which may negatively affect instream flows. Roach can persist in intermittent pools but increasing water demand in summer and early fall may cause complete drying of long portions of streams, so roach may be more prone to localized extirpation in many stream reaches or even entire watersheds.

Urbanization. Roach tend to disappear from streams flowing through urban areas, presumably because of the combined effects of habitat alteration, reduced flows and pollution. However, the Clear Lake basin is predominantly rural with limited urban development centered on or near the lake's shore. Local residents were leading proponents of the application of pesticides to the lake in an attempt to control gnat populations. Three treatment of Dichloro-diphenyl-dichloroethane (DDD) were applied in 1949, 1954, and 1957, before the gnat became resistant. DDD built up in animal tissues and was implicated in the reproductive failure of western grebes on the lake as well as the decline of local raptor populations. DDD accumulates in the fatty tissues of fish and may affect survival and reproduction (Hunt and Bischoff 1960). The effects of these basin-wide treatments on roach are unknown.

	Rating	Explanation
Major dams	Low	Impacts, if any, of Cache Creek Dam are minimal other than potential fragmentation of roach populations
Agriculture	Medium	Water withdrawal from streams reduces and degrades habitats
Grazing	Medium	Grazing is pervasive along roach streams
Rural Residential	Medium	Residential water withdrawal may contribute to decreased summer flows throughout the basin
Urbanization	Low	Growth of towns surrounding the lake contributes to pollution, alters aquatic habitat, and increases water withdrawal from streams
Instream mining	Low	Gravel mining has simplified stream habitat in the lower reaches of some streams
Mining	Low	Mining for mercury has left Clear Lake with extremely high toxicity levels but there are no known effects on roach populations
Transportation	Medium	Roads channelize streams and contribute silt and other pollutants throughout the basin
Logging	Low	Logging impacts in the Clear Lake basin are largely a legacy issue
Fire	Medium	Combined with predicted climate change conditions, fires may cause local extirpation more frequently in the future than in the past
Estuary alteration	n/a	
Recreation	Low	Effects of OHVs and other activities can be substantial but are generally localized
Harvest	n/a	
Hatcheries	n/a	
Alien species	High	Most roach streams are under continual threat of invasions by green sunfish, fathead minnows, and other non-native fishes

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

Mining. Mining wastes from the Sulphur Bank Mercury Mine were dumped into the Oaks Arm of Clear Lake intermittently between the 1920s and 1950s. These wastes contaminated the lake ecosystem with mercury and arsenic (Suchanek et al. 2002), although the effects on roach are not known. Gravel mining has affected some potential roach streams (e.g., Scott Creek) by simplifying habitats.

Transportation. Clear Lake is entirely surrounded by roads, which cross all major streams entering and exiting the lake. Bridges and culverts are major gradient control structures, significantly altering the hydrology and geomorphology of the lower reaches of many of Clear Lake's tributaries. These channel modifications may have been a contributing factor in the extirpation or reduction of roach populations. Extensive road networks also exist in the upper portions of Clear Lake basin watersheds; these roads may further contribute to siltation, channelization and habitat loss.

Logging. Logging in the Clear Lake area began in the 1840s. By 1905, approximately 1.5×10^6 board feet of lumber were being processed locally (Suchanek et al. 2002). Erosion from timber harvest lands likely contributed to historic simplification and siltation of streams, but effects on roach populations today are likely substantially reduced because most streams in the basin have presumably recovered due to greatly reduced timber harvest activity.

Recreation. The Clear Lake basin is extensively used for recreation including fishing, motorized boating and off-road vehicle use. The effects of such recreational activities have not been quantified but may include increased localized sedimentation, input of pollutants into the lake, disruption of fish behavior or movement, potential introductions of alien fishes, and other impacts.

Fire. Natural and human-induced fires are common in the watersheds that drain into Clear Lake (Suchanek et al. 2002) and may, occasionally, alter stream habitats. However, future fire effects may become more severe and frequent due to human changes to the landscape, changes to land management practices, and the predicted outcomes of climate change. More intense fires, especially in upper watersheds, may particularly affect fishes like roach, which are found mainly in smaller tributary streams that may be disproportionately impacted by fires.

Alien species. Starting in 1872, with the unsuccessful introduction of 25,000 lake whitefish by the California Fish Commission, most game and forage fishes popular in the eastern United States were introduced to Clear Lake. Today, 16 alien fishes are present in the lake and only five (of 14) native species remain (Moyle 2002). Alien fishes occupy streams usually through stocking of adjacent ponds for angling, although some upstream movement (e.g., green sunfish) is also possible. Roach are particularly susceptible to displacement by predatory centrarchids such as green sunfish.

Alien fish species constitute a barrier to native fish dispersal through Clear Lake, effectively isolating roach populations in small tributary streams. Isolation of native fish populations increases the likelihood that stochastic events such as drought or fire will result in localized extirpation without opportunity for recolonization. See the Central California roach account in this report for detailed coverage of the threats of isolation and interactions between roach and predatory fishes.

Effects of Climate Change: Clear Lake roach are well adapted to the warm, intermittent nature of most of the basin's streams. However, they are susceptible to long reaches of stream going dry, a process which is likely to become more frequent and widespread with climate change. Roach are one of the few native fishes that are able to endure life in isolated pools in the intermittent reaches of creeks which flow into Clear Lake. By late summer, stream flow goes subsurface, temperatures increase, dissolved oxygen levels drop to low levels and most fish in these remnant pools die, except roach. While such tenacity bodes well for roach in a future of dwindling in-stream water, it also suggests that they are likely to be extirpated from streams that dry completely under the dual strains of decreased rainfall and increased human water use. The latter, in the Clear Lake basin, includes surface and ground water utilization for vineyard expansion, rural residential development and urbanization. In a separate analysis of 10 metrics, Moyle et al. (2013) found that the Clear Lake roach was 'highly vulnerable' to extinction as the result of climate change if present trends in land and water use continue.

Status Determination Score = 3.6 – Moderate Concern (see Methods section Table 2). Clear Lake roach do not appear to be in immediate danger of extinction; however, isolation of tributary populations and the inability of roach to use Clear Lake for recolonization or dispersal to available habitats may contribute to further population declines or extirpations. The Clear Lake roach is listed as "G5T2T3, Imperiled" by NatureServ, where it is included in a taxon described as the Clear Lake-Russian River roach subspecies.

Metric	Score	Justification
Area occupied	3	Confined to tributary watersheds of Clear Lake
Estimated adult abundance	5	Populations appear to be robust and widespread although locally confined to tributaries
Intervention dependence	4	Monitoring and possible reintroductions needed
Tolerance	4	Remarkably resilient species
Genetic risk	4	Possible threat to genetic integrity due to isolation in tributaries
Climate change	3	Drying of streams could result in local extirpation
Anthropogenic threats	2	See Table 1
Average	3.6	25/7
Certainty (1-4)	2	Little published information

Table 2. Metrics for determining the status of Clear Lake roach, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: The persistence of Clear Lake roach depends on maintaining its stream habitats. The following are recommendations based upon this fundamental requirement:

1. A thorough survey of all Clear Lake tributaries should be conducted in order to determine the distribution and status of roach populations. In particular, streams with past surveys should be resurveyed in order to establish trend information and surveys should be repeated on a regular basis. The life history and habitat requirements of Clear Lake roach need focused research.
2. Streams with intact habitats and minimal stressors should be selected as refuges for native fishes and amphibians and managed accordingly, including taking actions

to maintain summer and fall base flows. Opperman and Merenlender (2004) studied Russian River tributaries and provided management recommendations for such streams, including maintaining live trees (live woody debris), both in the riparian zone and within the stream channel in order to create pool habitats that roach prefer. These recommendations may benefit native fishes of the Clear Lake basin as well. In addition, Merenlender et al. (2008) developed GPS-based water resource analysis tools to quantify and balance water needs and water resources on a watershed scale. These tools were created to aid in sustaining stream flows, while simultaneously enhancing water security for local landowners and vineyard operators. The tools can be used to evaluate various water-policy scenarios, estimate cumulative effects of water extraction methods on the natural hydrograph across a large spatial scale (including temporal variation), and provide information for watershed-level planning required to recover/maintain environmental flows. Such tools would be of great value in the arid Clear Lake basin where water resources are increasingly in demand.

3. Protective measures for Clear Lake roach should be integrated into a general management plan for native fishes of Clear Lake basin streams, including local populations of low concern fishes such as rainbow trout, Sacramento sucker and Sacramento pikeminnow, as well as of poorly known species such as threespine stickleback (*Gasterosteus aculeatus*) and western brook lamprey (*Lampretra richardsoni*).

Figure 1. Distribution of Clear Lake roach, *Lavinia symmetricus* ssp., in California.

MONTEREY ROACH
Lavinia symmetricus subditus (Snyder)

Status: Moderate Concern. Although Monterey roach do not appear in danger of extinction in the near future, populations could decline rapidly and disappear in many areas as the result of alterations to streams and changes in climate.

Description: This subspecies differs from the 'type' subspecies, *L. s. symmetricus,* by having fewer dorsal (7-9, mean 8.0) and anal fins rays (6-8, mean 7.3), fewer scales in the lateral line, slightly shorter fins, a slightly more robust body and a thicker caudal peduncle (Snyder 1913, Murphy 1948c, Hopkirk 1973). Coloration is deep olive above, silvery to whitish beneath. See the Central California roach account in this report for a more complete description of general roach morphology.

Taxonomic Relationships: The Monterey roach was first described as *Hesperoleucus subditus* by Snyder (1913) from Uvas creek, tributary to the Pajaro River, Santa Clara County. While the type specimen appears to be a "pure" roach, some of Snyder's specimens later proved to be hitch/roach hybrids (Miller 1945b). Snyder had noted that a large portion of his atypical specimens (those having 8 instead of 7 anal fin rays) came from a single collection point in the mainstem Pajaro River, but it was not until 1945 that these individuals were recognized by Robert Rush Miller as hybrids with hitch (*L. exilicauda*). Miller (1945b) showed that hybrids had intermediate morphological characters to their parent species and insinuated that drought conditions may have played a role in hybridization, by bringing the two normally allopatric species together in remnant pools. Avise et al. (1975) found that, while hybridization was present between the two species in the Pajaro River, it was localized and introgression was unusual.

Recent genetic evidence suggests that Monterey roach are most closely related to roach from tributaries to south San Francisco Bay (Aguilar and Jones 2009). This relationship is supported by strong geologic evidence for past hydrologic connections between the Coyote Creek watershed (San Francisco Bay drainage) and the Pajaro River watershed (Branner 1907, Dupre 1990). Snyder (1913) proposed that colonization of the Monterey basin by freshwater fishes from the Sacramento River took place via this hydrologic connection in two distinct events. The first such event transferred roach, Sacramento sucker (*Catostomus occidentalis*) and speckled dace (*Rhinichthys osculus*). These three species then spread throughout the Monterey Bay drainage system which, because of lower sea levels, had a fluvial connection to the San Lorenzo River. Subsequent sea level rise resulting from melting continental ice sheets then cut off the San Lorenzo from the Pajaro/Salinas system before the second colonization event transferred the remainder of the fish assemblage from Coyote Creek to the Pajaro River.

Murphy (1948c) presented his own dual colonization theory to explain the depauperate San Lorenzo fish assemblage. He proposed that the first colonization came not from San Francisco Bay but from the west side of San Joaquin Valley, transferred into the headwaters of the San Benito River (tributary to the Pajaro) by stream capture. This purported connection was not supported by recent genetic analysis which, instead, showed a strong relationship between fish in the San Lorenzo and adjacent coastal creeks and roach from San Francisco Bay tributaries. Fish from Los Gatos Creek (San Joaquin tributary) and from the upper San Benito, however, were not included in the study, so their relationship to each other and to roach from other Monterey Bay sub-basins remains unclear. The genetic evidence does suggest that headwater capture in the geologically active Santa Cruz Mountains (headwaters of streams which flow both directions are bisected by the San Andreas fault) has facilitated fish transfer from San Francisco Bay drainages (Sacramento basin) to the San Lorenzo (Monterey basin) and adjacent coastal systems (Jones 2001, Aguilar and Jones 2009). Further genetic analysis is needed to clarify both the colonization history of the Monterey basin and the phylogenetic relationships of Monterey roach.

Life History: Few data specific to Monterey roach exist; however, the following generalized description of roach life history is based on data from other roach populations which are thought to be similar (Moyle 2002).

Roach are opportunistic omnivores whose diet varies greatly across watershed, habitat type and season. In small, warm, streams they primarily graze filamentous algae, which is seasonally abundant, although they also ingest crustaceans and aquatic insects (Fry 1936, Fite 1973, Greenfield and Decket 1973). Juvenile roach consume large quantities of crustaceans and small chironomid midge larvae, while adult roach are more opportunistic feeders, feeding both off the substrate and from drifting insects in the water column. As a result of their benthic feeding habits, the stomach contents of adult roach are often found to contain considerable amounts of detritus and fine debris. It is thought that roach extract some nutritional value from this material because it is retained by the gill rakers and by mucus secretions from epithelial cells (Sanderson et al. 1991).

Growth is highly seasonal, with most rapid growth typically occurring in early summer (Fry 1936, Barnes 1957). In perennial streams, roach frequently exceed 40 mm SL in their first summer, reach 50-75 mm by their second year and reach 80-95 mm SL by their third summer (Roscoe 1993, Fry 1936). Few individuals exceed 120 mm SL or live beyond 3 years.

Roach typically mature at 45-60 mm SL in their second or third year (Fry 1936). Fecundity is dependent on size and ranges from 250 – 2,000 eggs per female (Fry 1936, Roscoe 1993). Spawning activity is largely dependent on temperature and typically occurs in March through early July, when water temperatures exceed 16˚C. Spawning occurs in riffles over small rock substrates that are 3-5 cm in diameter. Roach spawn in large groups over coarse substrates. Each female repeatedly deposits eggs, a few at a time, into the interstices between rocks where the eggs are immediately fertilized by one or more males. Spawning aggregations can be quite conspicuous and spawning fish can splash so vigorously that, at times, the splashing can be heard at some distance (Moyle 2002). This activity clears silt and sand from gravel interstices and improves adhesion for sticky fertilized eggs. Eggs hatch after 2-3 days and larvae remain in the gravel until large enough to actively swim.

Habitat Requirements: California roach are generally found in small streams and are particularly well adapted to life in intermittent watercourses, where dense populations are frequently observed in isolated pools (Fry 1936, Moyle et al. 1982, Leidy 2007). Smith (1982) found that, in the Pajaro River, Monterey roach have similar requirements to California roach in other areas. They were generally associated with pools in unshaded and warm tributaries in relatively undisturbed areas. While most abundant in clear, well oxygenated streams, roach were also present in areas where dissolved oxygen levels were < 1ppt (Smith 1982).

They can tolerate a relatively wide range of temperatures and dissolved oxygen levels and are found in habitats ranging from cold, clear, well-aerated 'trout' streams to intermittent streams where they can survive extremely high temperatures (30 to 35˚ C) and low dissolved oxygen levels (1-2 ppm) (Taylor et al. 1982, Knight 1985, Cech et al. 1990). Smith (1982) found Monterey roach reached their highest densities in quiet, unshaded pools.

Although emblematic of streams that support native fishes, roach are most abundant when found by themselves or with just one or two other species (Moyle and Nichols 1973, Leidy 1984, 2007, Brown and Moyle 1993). When found alone, roach will occupy open water in large pools; when found as part of complex assemblages, roach tend to congregate in low velocity (<40 cm/sec), shallow (<50cm) habitats (Moyle and Baltz 1985). Smith (1982) noted that, in the presence of hitch, Monterey roach tended to favor shallower, faster water. In the presence of native predators, roach are also restricted to the edges of pools, riffles and other shallow water habitat or to dense cover, such as that provided by fallen trees.

Distribution: Monterey roach are confined to the Pajaro, Salinas and San Lorenzo river systems, all tributary to Monterey Bay. Within the Pajaro watershed, Monterey roach do not occur in the mainstem Pajaro River but are present in Uvas Creek, Llagas Creek upstream of Chesbro Reservoir, North Fork of Pacheco Creek upstream of Pacheco Reservoir, Arroyo Dos Picachos and in the San Benito River and it tributaries, including Tres Pinos, Laguna, and Clear creeks, among others (Smith 2007). In the Salinas River system, roach have been extirpated from the mainstem habitats they historically occupied and now occur primarily in tributaries such as Arroyo Seco (J.J. Smith, pers. comm. 2009) and Gabilan Creek (Hager 2001). Roach are native to and numerous in the San Lorenzo River and Pescadero Creek and are present in smaller numbers in Soquel Creek, to which they may have been introduced. Snyder (1913) did not collect roach in Soquel Creek; however, he sampled only one site and their presence may not have been detected.

Trends in Abundance: Monterey roach are numerous but have been extirpated from reaches of the Pajaro and Salinas river systems due to habitat alteration and degraded water quality and quantity (Smith 1982, 2007). Long-term trends are not known but populations are likely fewer and more fragmented than they were historically.

Nature and Degree of Threats: Monterey roach are an exceptionally hardy fish that can tolerate high temperatures and low dissolved oxygen levels lethal to most other California native fishes. Nevertheless, they exist in a rapidly changing environment which is threatened by climate change and increasing human demand for water (Table 1). This is compounded by the fact that the vast majority of Monterey roach habitat occurs on private lands, where there is little formal protection for aquatic organisms.

Major dams. As part of the Habitat Conservation Plan for Santa Clara County (2007), Jerry Smith of San Jose State University wrote:

"Monterey roach have been lost from some habitats in the Pajaro River system due to construction of reservoirs. Attenuated winter flows from the reservoirs have apparently allowed hitch (*L. exilicauda*), a native minnow of downstream habitats, to expand upstream into Pacheco, Uvas and Llagas creeks below the reservoirs. Abundant hitch can reduce the closely related roach by competition and hybridization (Smith 1982). Uvas Reservoir frequently spills during large floods, so hitch abundance fluctuates from scarce to common in Uvas Creek, but they are always less abundant than roach. Roach are now absent from Llagas Creek and Pacheco Creek downstream of the reservoirs. The loss of roach in Llagas Creek occurred in 1977 when drought dried the streambed downstream of the reservoir and eliminated roach; although present upstream of the reservoir, roach have not been able to recolonize through the reservoir in the almost 30 years since the drought (Smith 1982, 2006). Transplanting roach from above to below the reservoir would reestablish the species in lower Llagas Creek."

Agriculture. The Salinas Valley is one of the most intensively farmed areas in California. Consequently, hydrodynamics and stream morphology in the valley have been severely altered, creating inhospitable lowland habitats and leading to isolation of roach populations in headwater tributaries.

Grazing. Grazing takes place mainly in the lower elevation hills along streams which are the main habitat of Monterey roach; grazing contributes to streams incision and to intermittent streams drying more quickly and completely.

Rural residential. Increasing rural population density, particularly in Santa Cruz County, has dramatically increased human impacts on small streams through increased water withdrawal, especially in summer months when flows are reduced, and through pollution from faulty septic systems and surface runoff from roads and other hardscapes, as well as habitat simplification and fragmentation from road crossings, development adjacent to streams and other factors.

Urbanization. As the human population in the Monterey basin has grown, water demand has exceeded supply. Groundwater is the primary source to meet agricultural and urban needs and salt water intrusion due to over-pumping from groundwater aquifers currently threatens all coastal water supplies for both municipal and agricultural uses. Urbanization also leads to stream channelization, habitat simplification, pollution input and other impacts that degrade roach habitats.

Logging. Historically, logging in the Monterey Basin was primarily limited to the San Lorenzo River watershed. However, logging is currently of little consequence due to its diminished scope as the watershed is increasingly converted to urban and rural-residential uses.

Fire. While fire is a natural part of the California landscape, catastrophic wildfires are becoming more frequent and severe as a consequence of fire suppression, human land use and increasing temperatures and aridity (Thompson et al. 2012). Because roach populations are increasingly isolated from one another due to human alterations of stream systems (e.g., agriculture, dams, reservoirs, introduced fishes), populations affected by fires are more likely to be extirpated without the possibility of natural recolonization.

Alien fishes. Alien fishes, especially centrarchids, are widespread in the watersheds containing Monterey roach, especially in ponds and reservoirs. They represent a threat through predation and competition, especially during periods of drought when roach may be confined with alien fishes in small pools.

	Rating	Explanation
Major dams	Medium	Unfavorable stream flow alterations from multiple dams
Agriculture	Medium	Monterey basin streams have been diverted, channelized, polluted and otherwise altered by intensive agriculture
Grazing	Medium	Grazing is a major use of private lands and is often concentrated along streams
Rural residential	Medium	Residential water withdrawal may be a principal cause of decreased summer flows in small, high gradient streams, especially within the San Lorenzo watershed
Urbanization	Medium	Urbanized areas reduce habitat quantity and quality through stream alteration, fragmentation, channelization, water removal and pollution
Instream mining	Low	Gravel mining alters habitats; greater historic impacts
Mining	Low	Of little direct effect, although residue from mercury mines may have localized effects
Transportation	Low	Many streams have been altered by roads and culverts, possibly fragmenting habitats
Logging	Low	Little contemporary logging in the Monterey basin
Fire	Medium	Isolated populations may be extirpated by fire without opportunity for natural recolonization
Estuary Alteration	Low	Intolerant of salinity levels in some estuaries
Recreation	n/a	
Harvest	n/a	
Hatcheries	n/a	
Alien species	High	Intolerant of introduced predatory fish, especially centrarchids

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

Effects of Climate Change: Although Monterey roach are well adapted to the warm, arid conditions of the basin's Mediterranean summers, their dependence on pools in intermittent streams suggests that they are also particularly susceptible to increasing aridity associated with climate change, despite their tolerant physiology. Roach can be extirpated from streams which currently maintain isolated, disconnected, pools in summer if the streams dry completely under the dual strains of reduced rainfall and increased human water use, including groundwater withdrawal. Thompson et al. (2012) indicated that, under moderate climate change scenarios, streams in the Salinas River watershed could become less suitable for fish through increased temperatures, decreased flows and loss of woody debris, important as cover for roach and other species. Loss of woody debris recruitment into streams may potentially occur because of the likely increase in wildfires, which could also change the dominant vegetation from forest to grasslands. Riparian trees and other vegetation will be maintained only if diversions and other factors affecting instream flow do not contribute to further drying of streams. Because Monterey roach live mainly in small streams in watersheds that are especially prone to desiccation due to drought, Moyle et al. (2013) rated them as "highly susceptible" to the predicted impacts from climate change.

Status Determination Score = 3.4 - Moderate Concern (see Methods section Table 2). Monterey roach are still common throughout much of their native range, although populations are fragmented and subject to localized extinctions. The Monterey roach is listed by NatureServe as "G5T2T3, Imperiled".

Metric	Score	Justification
Area occupied	3	Confined to streams tributary to Monterey Bay, plus Pescadero Creek to the north
Estimated adult abundance	5	Numerous large populations
Intervention dependence	4	Little management of primary habitats occurs, yet populations persist
Tolerance	4	High environmental tolerances
Genetic risk	3	Human alteration to river courses has facilitated an increase in roach/hitch hybridization
Climate change	3	Decreased flows, along with increased water demand, is likely to reduce available habitat
Anthropogenic threats	2	See Table 1
Average	3.4	24/7
Certainty (1-4)	2	Very little published information available

Table 2. Metrics for determining the status of Monterey roach, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: Genetic research is needed in order to elucidate the phylogenetics of Monterey roach. Of particular interest are the relationships between fish in the upper San Benito River and those in Los Gatos Creek on the west side of the San Joaquin Valley. Findings may shed light on the radiation of California's native fish fauna and provide insights into the geologic formation of the central Coast Ranges.

Although Monterey roach populations remain geographically widespread, their status should be monitored at least once every five years to determine if there is attrition in their increasingly isolated populations. Particular attention should be paid to areas that have suffered from wildfires. If local extirpations occur, a management plan should be developed to maintain flows in key streams and restore extirpated populations, potentially through reintroduction, where necessary. Consideration should also be given to the reintroduction of roach into watersheds with suitable habitats in which they were historically present but have since been extirpated. Priority should be given to the reestablishment of roach in lower Llagas Creek via transplantation from above Chesbro Reservoir. Likewise, the opportunity exists to reestablish roach in Pacheco Creek via transplants from above Pacheco Reservoir.

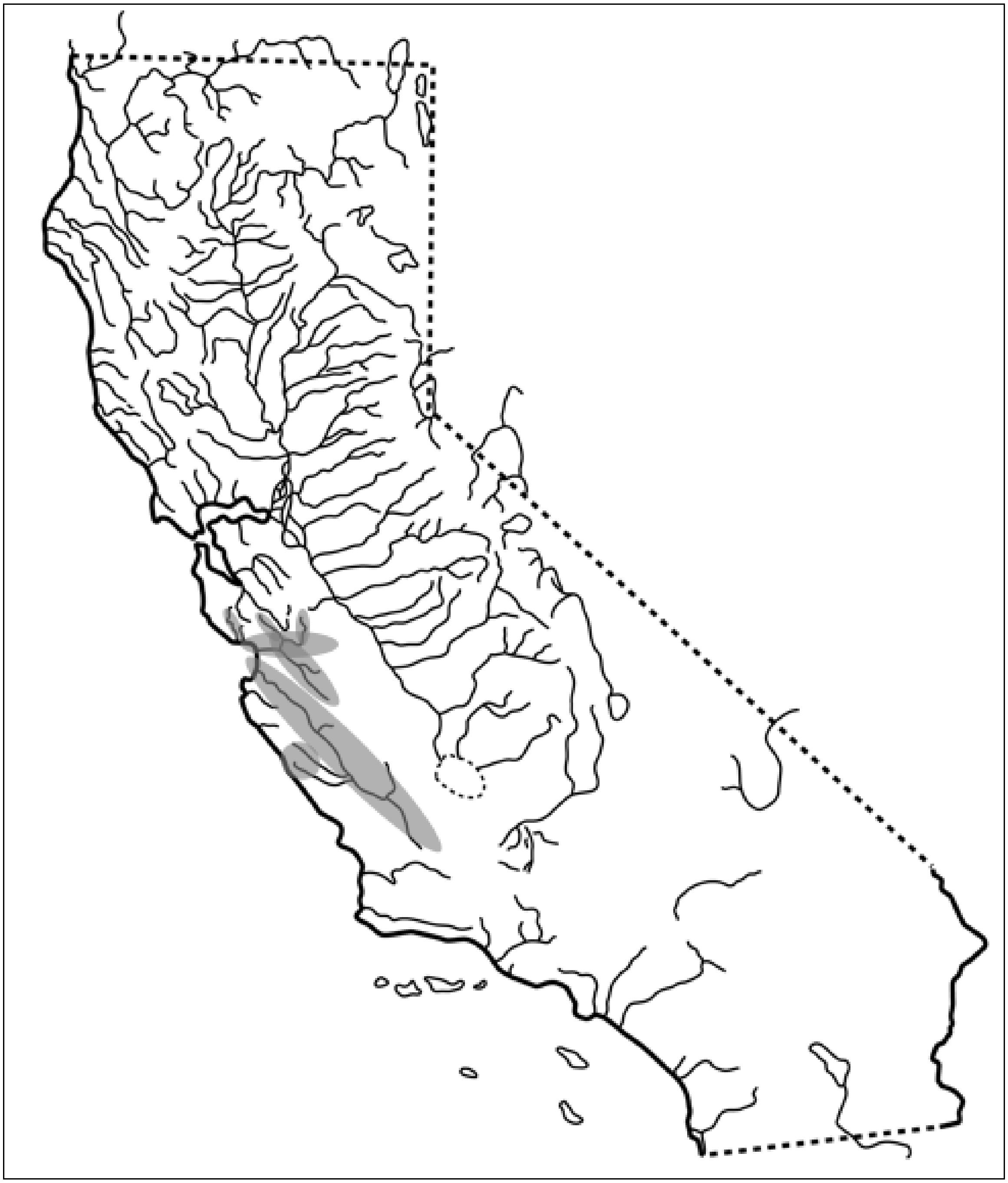


Figure 1. Generalized distribution of Monterey roach, *Lavinia symmetricus subditus* (Snyder), in California. Actual distribution is likely fragmented.

NAVARRO ROACH
Lavinia symmetricus navarroensis (Snyder)

Status: Moderate Concern. Although apparently in no immediate danger of extinction, populations of Navarro roach are subject to the dual strains of alterations to streams and associated habitat loss, along with predicted impacts from climate change.

Description: Navarro roach are small (adult size typically 50-100 mm), bronzy cyprinids. They have a robust body, deep caudal peduncle, short snout and short rounded fins. They are dark on the upper half of the body, light below and very similar in appearance to the Gualala roach; both fishes were described by Snyder (1913) as having a light lateral stripe approximately 2 scales wide extending from upper edge of the gill opening to the base of tail and entirely above the lateral line; below is a somewhat wider dark stripe, which, in turn, is followed by several narrower and very distinct dark stripes which grow lighter ventrally. Navarro roach have a mean of 8 dorsal fin rays (Hopkirk 1973). Navarro roach differ from Gualala roach in having fewer anal fin rays (usually 8, Hopkirk 1973) and one less, on average, row of scales above the lateral line (Snyder 1913). Roach captured in downstream migrant fyke nets in 1972 ranged between 51 –99 mm fork length, with an average of 51mm (Brown 1972).
Data specific to Navarro roach are limited; therefore, the following general description of roach morphology is based on studies from other CA roach populations. Roach are small, stout-bodied minnows (cyprinids) with a narrow caudal peduncle and a deeply forked tail. Fish rarely achieve lengths greater than 100 mm total length. The head is large and conical. The eyes are large and the mouth is subterminal and slants at a downward angle. Some populations, especially those in the streams of the Sierra Nevada, develop a cartilaginous plate on the lower jaw, often referred to as a "chisel lip." The dorsal fin is short (7-9 rays) and is positioned behind the insertion point of the pelvic fin. The anal fin has between 6-9 rays. The pharyngeal teeth (0,5-4,0) have curved tips which overhang grinding surfaces of moderate size. Roach are usually dark on the upper half of their bodies, ranging from a shadowy gray to a steel blue, while the lower half of the body is much lighter, usually a dull white/silver color. The scales are small, numbering 47-63 along the lateral line and 32-38 before the dorsal fin. Subspecies are distinguished by various distinctive subsets of characters, especially fin ray and scale counts.

Roach exhibit general (non-nuptial) sexual dimorphism (Snyder 1908b, Murphy 1943). In the tributaries to San Francisco Bay, Snyder (1905, 1908b) demonstrated that the sexes could be differentiated by the ratio of pectoral fin length to body length. Males exhibited a ratio of $\geq$.21 while females bore pectoral fins between .16 and .20 the length of their body. Both sexes exhibit bright orange and red breeding coloration on the operculum, chin and the base of the paired fins. Males may also develop numerous small breeding tubercles (pearl organs) on the head (Murphy 1943).

Taxonomic Relationships: Navarro roach were first collected by Snyder (1908d) who recognized them as *Rutilus symmetricus* but found that they (along with roach from the Gualala River, which are morphologically similar) were easily distinguished from other roach by their more robust body, deeper caudal peduncle, shorter rounded snout and shorter, less acute fins. While recognizing the close affinity between the Gualala and

Navarro roach, Snyder showed that the two taxa could be distinguished "without difficulty" by the greater number of anal fin rays and larger scales above the lateral line present in the Navarro roach (Snyder 1908d, p. 175).

In 1913, Snyder revised the systematics of roach, describing six full species (the Navarro and Gualala roaches among them) and erecting a new genus, *Hesperoleucus*, to house them. In a footnote from a paper on hybridization between hitch (*Lavinia exilicauda*) and roach in the Monterey basin, Miller (1945b) suggested that Snyder's roach species should be treated as geographic subspecies. In an unpublished M.S. thesis, Murphy (1948) agreed with Miller and concluded that all coastal species of roach should be reduced to subspecific status of *H. symmetricus*. In his arguments for merging *Hesperoleucus,* Murphy did not dispute that Snyder's species were morphologically and genetically distinct. Instead, he followed what appears to be a strict interpretation of the biological species concept as outlined by Mayr (1942). Murphy argued that the distinctiveness of isolated populations, such as those in the coastal rivers, resulted from "merely a chance genetic divergence" resultant from small numbers of colonizing individuals and, that if physical barriers were removed from between forms isolated in separate basins, "a population would soon lose its identity."

Twenty-five years later, Hopkirk (1973) pointed out that Murphy's principal argument in denying specific status to coastal roach populations (the concept of a "chance genetic divergence" during colonization) was an important mechanism in speciation, the "founder effect" (Mayr 1942, 1954). Hopkirk also asserted that natural selection contributed to differences among roach populations and, therefore, the distinctiveness of populations was "not due solely to the chance combination of genetic factors" as Murphy had asserted. However, despite his critique of Murphy's species concept, Hopkirk agreed that Murphy was correct in placing all roach in one species and Murphy's (1948c) diagnosis was adopted by subsequent workers (Hopkirk 1973, Moyle 1976, Hubbs et al. 1979), although it was never formally published.

However, in the subsequent four decades, much more has been learned about roach systematics. For example, genetic studies (Avise et al. 1975, Avise and Ayala 1976) demonstrated the close relationship of hitch and roach and led to the inclusion of *Hesperoleucus* within *Lavinia* (Moyle 2002); new subspecies have been discovered (Jones et al. 2002); and new groupings of lineages have been proposed (Moyle et al. 1995, Moyle 2002). While new genetic methods have allowed better resolution of *Lavinia* population boundaries, considerable confusion remains about the number and relationship of taxa (Aguilar et al. 2009).

Recently, Aguilar et al. (2009) used both nuclear microsatellite (nDNA) and mitochondrial DNA (mtDNA) markers in the most comprehensive genetic study of *Lavinia* to date. Employed in tandem, these two genetic markers supply insight into both the relationships between populations (phylogenetics) and the distinctiveness of individual populations (taxonomy). The microsatellite analysis of Aguilar et al. (2009) largely supports the distinct lineages that Snyder (1913) described as species and Moyle (2002) recognized as subspecies. In light of these recent genetic analyses and the fact that Snyder's original species names were never properly submerged (i.e., through formal publication of an analysis in the peer-reviewed literature), the subspecies designation for the Navarro roach should be retained. This and other roach taxa now listed as subspecies may be sufficiently distinct to warrant full species status, pending further analyses and publication of findings in the peer-reviewed literature. For additional information and a more comprehensive treatment of roach systematics in California, see the Central California roach account in this report.

Life History: No studies have specifically addressed Navarro roach life history but theirs is assumed to be similar to life histories of other roach subspecies. A general summary of California roach life history can be found in the Central California roach account.

Habitat Requirements: Compared to many other northern coastal watersheds in California, the Navarro has been the focus of extensive habitat and fisheries surveys. Roach are found throughout the system but are rare in the heavily forested North Fork and Mill Creek watersheds (CDFG 1945-1997, Entrix 1998, Feliciano 2004), where colder stream temperatures predominate (NCRWQCB 2000 Appendix A).

Navarro roach prefer pool habitats, with low water velocity, where they tend to be found throughout the water column. They are the dominant fish by number in the Navarro watershed and adults are found in large mixed-size schools that can number well into the hundreds of individuals (Feliciano 2004). Since roach are often found in open pool habitats, they are highly conspicuous for underwater observation. As such, single pass snorkel-surveys are a relatively accurate method for estimating roach abundance (Feliciano 2004). Larvae (less than 20-30mm) bunch in dense schools in low velocity habitats often associated with structural cover (Feliciano 2004). Navarro roach are freshwater obligate fish which can tolerate only very low levels of salinity. In the Navarro estuary, they have been collected at salinities of 3 ppt but perished as salinities reached 9-10 ppt due to the incoming tide (Moyle, unpublished observations). However, they apparently frequent the upper estuary in large shoals, usually around woody debris, and have been recorded in small numbers in the lower estuary (Cannata 1998). Roach use of the estuary is dependent on salinity, which fluctuates according to many variables including tide and the opening and closing of the sand bar at the river's mouth.

Roach tend to be most abundant in mid-elevation stream habitats associated with agricultural land use, rangeland and development. In the Navarro watershed, where the pre-European land cover was primarily redwood forest (Palmer 1967, Holmes 1996), roach are associated with the altered mixed deciduous/evergreen forest, a sign that the roach are capable of existing in heavily modified habitats. On a local stream-reach scale, roach abundance is also positively correlated to the level of disturbance. In a survey of 19 sites from throughout the basin, Navarro roach were closely associated with the most disturbed sites, including: (1) an active restoration site that had been dewatered before the restoration process, (2) a stream reach running through the center of a small town, and (3) a reach of stream immediately downstream from a seasonal gravel dam, used to create a pool for recreational use (Feliciano 2004). Overall, roach were found in the warmest and widest stream localities, where substrates were highly embedded and which had the least amount of shade and in-stream cover. Roach were also associated with riparian forest (buffer widths of 100 m) that had been highly disturbed (Feliciano 2004).

Feliciano (2004) observed interactions between Navarro roach and steelhead trout (*Oncorhynchus mykiss*) in experimental stream enclosures. Roach were never observed to initiate attacks on trout and, consequently, had little effect on trout habitat use or feeding behavior. The trout, on the other hand, aggressively displaced roach from prime feeding habitats and preyed upon both juvenile and adult roach (Fite 1973, Power 1990, Feliciano 2004). However, because the competition between roach and steelhead is likely moderated by temperature (i.e., roach can tolerate temperatures that cause extreme physiological stress to steelhead), roach may attain competitive advantage at higher temperatures than those under which the experiment was held. Feliciano (2004) asserts: "continuing anthropogenic modification of the stream system and surrounding watershed (e.g. surface and groundwater pumping, forest removal, suburbanization) results in streams that are shallower, warmer, less shaded, and thus more favorable for roach and more stressful to steelhead trout."

Navarro roach are also often found with three-spine stickleback (*Gasterosteus aculeatus*) and associated with distinct insect assemblages (Feliciano 2004). Both insect and fish assemblages in many areas shifted with the progression of summer, as cold water-dependent salmonids and insects were replaced by roach and other warm water-adapted species.

Distribution: Navarro roach are confined to the Navarro River and its tributaries.

Trends in Abundance: Although no population estimates have been conducted for roach in the Navarro watershed, stream surveys carried out by the California Department of Fish and Wildlife and the University of California, Davis over the past several decades show that roach have increased in abundance, while coho salmon are on the verge of localized extinction and steelhead abundance has declined dramatically. These population trends (increase of roach, decline of salmonids) are the direct result of warmer water associated with habitat degradation related to deforestation. Roach are a warm water-adapted species and can survive extremely warm water temperatures, while salmonids are cold water-dependent. Presumably, when the Navarro River watershed was more heavily forested, Navarro roach were less abundant and less widely distributed within the watershed.

Nature and Degree of Threats: Historic and contemporary land use practices in the Navarro watershed have resulted in severe alteration of the basin's hydrology, reduced the amount and quality of aquatic habitats, and have led to extreme simplification of the habitats that remains (Table 1). In 1996, habitat surveys of 11 streams from throughout the Navarro basin found excessive deposition of fine sediments in pools and riffles in all reaches surveyed (Entrix 1998). Aggradation (deposition of gravel and fine sediment in the stream channel) has led to higher water temperatures and significantly decreased aquatic habitat in summer as water retreats beneath aggraded gravel streambeds (North Coast Regional Water Quality Control Board 2000), while increasing human water demands for towns, rural residential development and, especially, for new vineyards, compound these legacy effects.

In 2000, the United States Environmental Protection Agency listed the Navarro River under 303(d) of the Clean Water Act as "impaired due to excessive sediment and high temperatures." In preparing the Total Maximum Daily Load (TMDL) report required for all 303(d) listed streams, the North Coast Water Quality Control Board (NCRWQCB 2000) found that in the Navarro:

> “Surface water diversions and groundwater extraction, from residential, commercial, and agricultural uses, can lower water tables and reduce baseflow contributions. Summer low-flow periods reduce the available pool habitat, increase stream temperatures, and may completely dry the channel. Streamflow monitoring performed by the Mendocino County Water Agency and the State Water Resources Control Board, Division of Water Rights indicate that segments of Anderson Creek can go dry for brief periods due to pumping (Entrix 1998).”

While this pattern of watershed use has probably increased roach populations in recent years, the potential for future overutilization of water resources in the basin may pose a threat. Stressors in the Navarro River watershed that impact roach and other native fishes: are (1) logging, (2) agriculture, (3) rural residential development, (4) urbanization, (5) transportation, (6) grazing, (7) fire, and (8) and alien species (Table 1). These impacts are not necessarily listed in order of importance and do not operate independently but, instead, must be viewed in aggregate as cumulative watershed impacts.

Agriculture. Vineyards are now being developed on a very large scale within the watershed (Anderson Valley) and their use of water for irrigation and frost protection is reducing summer flow in Navarro watershed streams. Vineyard expansion has a direct impact on tributary flow if either surface water or groundwater is used for irrigation. Pumping from wells affects groundwater inflow and flow from springs. Deitch et al. (2009b) showed that vineyard water use for irrigation and frost protection is significantly affecting in-stream flow in the Russian River tributaries in Sonoma County. These finding apply equally to the adjacent Navarro River basin where Entrix (1998) states:

> “Summer flows in the lower reaches of Anderson, Rancheria, and Indian Creek are at times significantly reduced by agricultural pumping. In aggraded stream reaches, summer flow may be entirely subsurface.”

Pumping for frost protection in spring is also an acute threat because the simultaneous withdrawal by vineyards on a regional scale can dry streams quickly and eliminate all life stages of fish present, including eggs incubating in streambed gravels. Fertilizer and agricultural chemicals are also of concern in that both are known to augment algal production in rivers. Increased eutrophication in the Navarro River would further degrade habitats that are already compromised by both excessive sediment and high temperatures (US EPA 2000).

	Rating	Explanation
Major dams	n/a	No major dams in watershed
Agriculture	Medium	Water withdrawal for irrigation and frost protection decreases flows; pollution inputs from return waters and runoff
Grazing	Medium	Grazing reduces shade and cover in riparian areas
Rural residential	Medium	Residential water withdrawal decreases summer base flows
Urbanization	Low	Urbanization is increasing but remains limited in the watershed
Instream mining	Low	Little or no mining takes place today
Mining	n/a	
Transportation	Medium	Much of the river is bordered by paved roads, while the watershed has a vast network of logging and ranch roads
Logging	Medium	Logging is the largest land use in the watershed; much greater historical impact but legacy effects of widespread deforestation remain
Fire	Low	Infrequent fires may cause localized extirpation, especially in smaller headwater tributaries
Estuary alteration	n/a	
Recreation	Low	Channel alterations from removal of dead trees and construction of summer dams
Harvest	n/a	
Hatcheries	n/a	
Alien species	Medium	Intolerant of introduced predatory fish, especially centrarchids such as green sunfish

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

Grazing. Sheep and cattle have been grazed in the Navarro River watershed since the 1870s (NCRWQCB 2000). Impacts from grazing in the Navarro River watershed are pervasive but are likely reduced from historic levels. Cattle grazing along streams may result in stream bank collapse, pools filled with sediment, riparian vegetation removal, pollution from animal wastes and reduction in cover and associated shading. In these situations, roach tend to disappear from streams despite their high tolerance of adverse conditions (Feliciano 2004). Stock ponds, which provide water sources for cattle, can divert water from streams and support populations of alien predatory fishes. These fishes (e.g., green sunfish, largemouth bass) may colonize adjacent streams during wet periods (when ponds spill), potentially eliminating roach populations. Capture of green sunfish in recent surveys may be the result of escapement from stock ponds (Feliciano 2004).

Rural residential development. The Franciscan geologic formation, which underlies much of the Navarro River watershed, is considered to be essentially non-water bearing. Only limited amounts of ground water can be found in the Franciscan formation's joints and fractures (NCRWQCB 2000). Ground water is present mainly in shallow surface gravel deposits and is easily depleted. The watershed is experiencing increased rural development and, while roach can coexist with humans in such environments (and even increase under certain conditions), populations may be negatively impacted by the combination of overutilization of water during low-flow periods, polluted inflow from septic tanks and agricultural runoff, siltation from roads, and loss of complex habitat through bank stabilization projects.

Urbanization. Although the Navarro basin is largely rural, urban development is increasing around Booneville and Philo, increasing water demand and further degrading water quality and channel habitats.

Instream mining. Gravel mining can simplify habitats, increase turbidity and contribute to drying of intermittent pools (NMFS 2008). Instream mining appears limited at present but legacy effects from past mining activities may still affect aquatic habitats.

Transportation. Small streambeds are disproportionately affected by roads and road crossings, which simplify and degrade riparian and instream habitats. When roads severely channelize small streams, roach tend to disappear from those streams (Feliciano 2004). Culverts and other road crossing may also form barriers to upstream fish movement, which can lead to isolation of populations and prevent recolonization of upstream habitats. Road building to facilitate logging, rural development and vineyard expansion changes the annual hydrograph by facilitating more runoff during storm events and reducing groundwater storage capacity, leading to reduced summer and fall base flows. Ranch and logging roads are also a leading source of sediment delivery to Navarro system streams (CDFG 1998), potentially limiting reproductive success of roach and other small fishes.

Logging. The Navarro River has a history of intensive logging that began in the mid-1850s, following the Gold Rush. A second logging boom occurred in the watershed from the late 1930s to the early 1950s, when large tracts of redwood-dominated forest were re-cut and the Douglas fir forests in the North Fork Navarro were cut for the first time (Adams 1971). Today almost all forestlands are second or third growth redwood or Douglas fir, intermixed with tanoak and other deciduous trees. The consequent reduced value of these timberlands is one reason that forestlands are being converted to vineyards, resulting in changes in stream flows and temperatures. The primary cause of high stream temperatures in the Navarro basin is the discontinuous canopy closure and consequent lack of shading. Aerial photography reveals that, in the early 1950s, many tributary streams were shaded by complete canopy closure; many of these same streams are now exposed to direct solar heating due to the loss of riparian forest to logging, development, and widening stream channels resultant from increased sediment delivery to streams. The NCRWQCB (2000) found that the Navarro River stream bed has been elevated by "over three to five feet" when compared to "the elevation that existed prior to Anglo-

American resource exploitation." More recent evidence of stream aggradation due to logging is also given:

> "The Greenwood Road Bridge cross-sections also illustrate the impacts of sedimentation. Comparison of the 1950 and 1999 cross-sections show that the maximum depth of the pool along the right bank of the channel has filled approximately five feet since 1950. The change in depth has been accompanied by an increase in width of approximately 20 feet. Entrix (1998) found that the width of unconfined stream channels increased substantially from 1952 to 1965 throughout the Navarro Watershed. Given the extent of logging activities observed in the 1952 aerial photos and the yarding methods employed at that time, it is reasonable to assume that the channel had been affected by increased sediment yields prior to 1950."

To a certain extent, logging has benefited Navarro roach by causing streams to warm and by eliminating cold-water requiring competitors and predators (albeit native ones), such as steelhead and coho salmon. In the long-run, however, the conversion of a diverse forested landscape to agricultural use is likely to eliminate large areas of roach habitat through reduced stream flows, impaired habitats (e.g., wider, shallower stream segments, lack of shading, filled pools, and lack of fallen trees in streams), and warm, polluted return waters that may exceed even the roach's wide thermal tolerances.

Fire. Fire is a natural, if historically infrequent, part of the Navarro River watershed. However, fires are now more frequent and their effects are more severe because of land management practices and associated changes to the landscape. Long-standing fire suppression policies have increased fuel loads, while historic logging has dramatically increased solar input in deforested areas and led to drier fuels. Thus, more severe and frequent wildfires may reduce roach habitat or eliminate localized populations, especially in smaller headwater tributaries.

Alien species. Roach cannot coexist with large populations of alien fishes, especially centrarchids such as green sunfish (*Lepomis cyanellus*) and black basses (*Micropterus* spp.). Centrarchids have been recorded in stream surveys in the Navarro system and could threaten roach populations in many stream reaches, as they have done in other areas of the state (see the Central California roach account in this report for examples). Thus, the transportation of alien fishes over natural barriers by humans and the escape of alien fishes (usually centrarchids) from stock ponds in the watershed can pose a serious threat to the persistence of roach in the Navarro watershed, although this may be mitigated by the winter flood hydrology of coastal rivers that may inhibit the establishment and persistence of alien fishes.

Effects of Climate Change: Navarro roach are well adapted to the warm, arid conditions of California's Mediterranean climate. However, their frequent dependence in late summer on intermittent pools suggests that they are also particularly susceptible to decreases in summer and fall base flows. Roach are one of the few native fish that are able to endure life in isolated, warm pools with low dissolved oxygen levels in intermittent streams. However, increasing water demands, coupled with predicted climate change impacts, may lead to complete drying of stream segments and elimination of roach populations. In the summer of 1992, the mainstem Navarro was pumped completely dry and, increasingly, flows of many aggraded stream reaches (e.g., lower Rancheria Creek, Little North Fork) go entirely subsurface even in “normal” water years. Already diminishing summer stream flows illustrate the possibility that Navarro roach could be extirpated from stream reaches or even entire tributary watersheds if annual precipitation decreases or becomes more variable. Because of its limited distribution in a highly altered watershed, Moyle et al. (2013) rated the Navarro roach as “highly vulnerable’ to extinction from climate change.

Status Determination Score = 3.3 – Moderate Concern (see Methods section Table 2). Although apparently in no immediate danger of extinction, populations could decline rapidly and disappear in some areas as the result of alterations to streams, changes in climate, water withdrawal for rural development and viticulture, and invasion of alien fishes. The Navarro roach is listed by NatureServe as Critically Imperiled.

Metric	Score	Justification
Area occupied	1	Confined to the Navarro River and its tributaries
Estimated adult abundance	5	Population large at present
Intervention dependence	3	The Navarro is a rapidly changing watershed so annual monitoring and management are needed
Tolerance	5	Remarkably resilient fish
Genetic risk	4	Little threat to genetic integrity at present
Climate change	2	Highly vulnerable in combination with watershed changes
Anthropogenic threats	3	See Table 1
Average	3.3	23/7
Certainty (1-4)	3	

Table 2. Metrics for determining the status of Navarro roach in California, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: The principal management need in the Navarro River watershed is a regular monitoring program, with basin-wide fish surveys every five years to determine population status and trends and to detect alien fishes and document their distribution. A secondary need is the development of an educational program for watershed residents, especially agricultural water users, to develop cooperative ventures to restore watershed function in ways that benefit fish. Additionally, reaches or tributaries that can be managed as native fish refuges should be identified and established, as insurance against long-term drought and changes in land and water use in the watershed.

Water quality standards recommended by state and federal agencies should be adopted and vigorously enforced, including restoration actions to reduce sediment loads (e.g., reducing the impact of roads of all types). Water rights in the watershed need to be adjudicated and a minimum flow established for all streams, including late summer and early fall low flow periods, to protect fishes and other aquatic organisms. Riparian vegetation buffers should be established and maintained throughout the watershed to increase shade and cover. In addition, Merenlender et al. (2008) developed GPS-based water resource analysis tools which quantify and balance water needs and water resources on a watershed scale. The tools were created to aid in sustaining instream flow while simultaneously enhancing water security for local landowners and vineyard operators. The tools can be used to evaluate various water-policy scenarios, estimate the cumulative effects of water extraction methods on the natural hydrograph across a large spatial scale (including temporal variation), and provide information for the watershed-level planning required to recover environmental flows. Such tools would be of great value in the Navarro River basin where water resources are increasingly over-allocated.

Figure 1. Distribution of Navarro roach, *Lavinia symmetricus navarroensis* (Snyder), in California.

TOMALES ROACH
Lavinia symmetricus ssp.

Status: Moderate Concern. Although apparently in no immediate danger of extinction, the Tomales roach has a limited range that is degraded by extensive habitat alteration, primarily from water diversion infrastructure and grazing.

Description: Tomales roach are a small (adult size typically 50-100 mm SL, up to 120 mm SL), bronzy cyprinid, very similar in appearance to the Russian River roach. Like roach from the Russian River, Tomales roach differ from Central California roach in having a trim, slender body, a somewhat pointed snout, a slender caudal peduncle and long fins. Tomales roach have a mean of 9 dorsal fin rays, 7 anal fin rays and 54 lateral line scales (Hopkirk 1973). For a more general description of roach morphology, see the Central California roach account in this report.

Taxonomic Relationships: The Tomales roach was first collected in Walker and Lagunitas creeks, Marin County, in 1910 but was not described until 1914, when Snyder assigned it to *Hesperoluecus venustus*. The Venus roach (*H. venustus*), as described by Snyder (1913, 1917), encompassed roach populations from the Russian River and the streams entering San Pablo, Suisun, San Francisco and Tomales bays. It is no longer considered a valid taxon because recent genetic analysis demonstrates that it consists of forms from different evolutionary lineages (Aguilar et al. 2009). Current systematic classification places roach from tributaries to San Pablo, Suisun, and San Francisco bays in *L. s. symmetricus*, the Central California roach, while the Russian River roach and the Tomales roach are considered to be undescribed lineages (Aguilar et al. 2009, Jones 2001). Tomales roach are probably descendants of roach that colonized Walker Creek through Lagoon Pass, the headwater divide that separates Walker Creek (Tomales Bay) and San Antonio Creek (San Pablo Bay).

Using morphological characters, Hopkirk (1973) found that roach from the Tomales Bay region should be given subspecific status. Moyle et al. (1989) agreed with this assessment and suggested that Tomales roach are an undescribed subspecies of *L. symmetricus*. Subsequently, a mitochondrial DNA genetic assay of the genus *Lavinia* supported the distinctiveness of the Tomales roach (Jones 2001). In the most comprehensive genetic study of *Lavinia* to date, Aguilar et al. (2009) used both nuclear microsatellite (nDNA) and mitochondrial DNA (mtDNA) markers to supply insight into both the relationships between populations (phylogenetics) and the distinctiveness of individual populations (taxonomy). The mtDNA analysis identified roach from Lagunitas and Walker creeks to be a highly supported clade. Microsatellites, however, were not as definitive, with one analysis finding "elevated" bootstrap support for grouping roach from the Tomales region into a distinct taxon, while another analysis (using the program STRUCTURE) found that, although distinguishable, roach from Lagunitas and Walker creeks clustered with Monterey roach.

Jones (2001) found that populations of roach from Lagunitas and Walker creeks share nuclear DNA allele frequencies but were reciprocally monophyletic for mitochondrial DNA haplotypes. Although the sample size was small (n=5), these results indicate that there is little genetic exchange between the two populations; however, a much larger sample would be required to validate this assumption. The genetics of roach from Pine Gulch Creek in the Bolinas Lagoon watershed have not been studied. Murphy found the Pine Gulch population to be morphologically "identical" to those from Tomales Bay streams and proposed that roach from Olema Creek (tributary to Lagunitas Creek) had colonized Pine Gulch Creek through the San Andreas Fault rift valley that the two watersheds share.

Roach are also found in Salmon Creek (CDFG 2001), which drains to the Pacific Ocean just north of Tomales Bay. The dynamic geologic history of the Coast Ranges has provided ample opportunity for transfer of roach from either the Tomales watershed to the south or from the Russian River watershed to the north. As in most coastal drainages, the possibility also exists that freshwater fishes may have had the opportunity to move between watersheds during times of lower sea levels via direct fluvial connections which were submerged as sea level rose. There has been no study of roach from Salmon Creek but, because of its proximity to the Tomales watershed, these fish are tentatively placed in the Tomales roach taxon until biochemical investigation resolves their identity.

Life History: No life history studies have been done specifically on Tomales roach but, presumably, their life history is similar to that of roach from adjacent watersheds studied by Fry (1936). For a general description of roach life history, see the Central California roach account in this report.

Habitat Requirements: No habitat requirement studies have been done specifically on Tomales roach, but their habitat requirements are assumed to be similar to roach from adjacent watersheds studied by Fry (1936) and from San Francisco Bay tributaries studied by Leidy (1987, 2004). The streams occupied by Tomales roach flow through watersheds that are heavily grazed, with flows regulated by dams, so they mostly live in highly altered habitats that include warm, aggraded, reaches with little riparian vegetation (e.g., Walker Creek). In Walker Creek, their most common associates are prickly sculpin (*Cottus asper*), threespine stickleback (*Gasterosteus aculeatus*), and rainbow trout (*Oncorhynchus mykiss*).

Distribution: Tomales roach are restricted to the western Marin County drainages of Lagunitas Creek and Walker Creek. Roach of uncertain taxonomic affinity have also been reported from Pine Gulch Creek, tributary to Bolinas Lagoon (Murphy 1948c) and Salmon Creek (CDFG 1996). However, a 1997 survey for freshwater shrimp (*Syncaris pacifica*) in Pine Gulch creek recorded no roach (Fong 1999).

Murphy (1948c) speculated that Tomales roach were descendents of roach from San Pablo Bay drainages. The headwater divide between Walker Creek (Tomales Bay tributary) and San Antonio Creek (San Pablo Bay tributary), known as Lagoon Pass, consists of a high, marshy valley. During heavy rain events, a surface water connection between the two drainages forms and provides a colonization route which could be used by fluvial fishes. The Sacramento sucker (*Catostomus occidentalis*), another native fish that frequents headwater habitats, is the only other fluvial fish in the Tomales system and is thought to have also gained access to the basin via this same intermittent connection (Murphy1948c).

Trends in Abundance: There is little indication that Tomales roach in Walker and Lagunitas creeks are less abundant than in the past, but no estimates of their abundance exist. No recent records of roach in Pine Gulch Creek could be found and its current status is uncertain.

Nature and Degree of Threats: While roach are very resilient fish, they tend to decline in or disappear from streams that are: (1) dewatered by diversion for residences, pasture, vineyards and other uses, (2) heavily altered by channelization (often in urban settings) and, (3) invaded by alien predators such as green sunfish (*Lepomis cyanellus*) (Table 1).

Roach are tolerant of the aggraded, shallow, open, and warm stream habitats which characterize much of Walker Creek, so they are the dominant species in the watershed. Conversely, steelhead numbers are much reduced from historic levels and coho salmon are nearly extirpated (Emig 1984). Current land use in the watershed is almost exclusively agricultural (pasture), with the exception of residential development in the town of Tomales. In contrast, Lagunitas Creek is a largely forested watershed with an extremely high density of rural residences and a higher proportion of salmonids in the fish assemblage. Due to considerable differences in land use and physical habitats between the Walker and Lagunitas watersheds, threats to roach populations in these two watersheds may also be very different. Genetic evidence (Jones 2001) suggests that there is very little movement (genetic exchange) between these two populations.

Dams. Dams of all sizes have multiple effects on roach: they create impassible barriers to upstream movement of small fishes (such as roach); impoundments generally support populations of predators that outcompete or eliminate roach and other native fishes; dams alter natural hydrographs and the tailwaters they create may or may not be beneficial to roach; small dams divert water from streams, increasing the likelihood of large portions of streams drying more quickly or completely, particularly during drought periods; and dams block dispersal routes, effectively isolating roach populations so that, when local extinctions occur, suitable habitats cannot be recolonized naturally.

In the Lagunitas watershed, Lake Lagunitas was built in 1872, followed by Alpine Lake in 1918, and then Bon Tempe in 1948. Peters Dam was built in 1953 to form Kent Lake and Nicasio Reservoir was formed by the construction of Seeger Dam (1960) on Nicasio Creek. In 1982, Peters Dam was raised 45 feet, doubling the volume of the reservoir behind it. Soulajule Reservoir, in the Walker Creek watershed, was created in 1978. Generally, environmental flows are required below these dams to support fishes, especially steelhead and coho salmon; however their impact(s) to warm water-tolerant fish such as roach remain unknown.

Agriculture. Current land use in the Walker Creek watershed is almost exclusively agricultural, with the exception of residential development in Tomales. Effluent from dairy operations had been a serious problem in the past (CDFG 1959); however, few dairies remain in the watershed and contemporary dairy practices employ much more stringent effluent treatment procedures. Currently, beef is the primary agricultural product (threat discussed under grazing), although at least one vineyard has been established in the watershed (Marin County Watershed Management Plan 2004).

Grazing. A long legacy of intensive grazing in the Walker Creek watershed has altered the hydrology and geomorphology of the basin. Overgrazing severely compacted soils and removed riparian vegetation, with subsequent stream bank failures and rapid streambed down-cutting in much of the watershed. Sediment delivered to streams resulted in lowering of the water table and a marked increase of complete drying of the streambed in summer months (Kelley 1976). Significant down-cutting of the streambed is common in the upper watershed, where some reaches have incised as much as five feet, while sections of the lower watershed have aggraded as much as four feet (Haible 1976). The CDFW listed severe erosion and siltation as a factor in the decline of salmonid populations in the creek (CDFG 1959) and Walker Creek is currently listed as impaired for sediment/siltation, pathogens, nutrients and mercury under Section 303(d) of the federal Clean Water Act (US EPA 2006). The filling of Lower Keys Creek, which was historically navigable by ships and barges, along with the growth of the depositional delta at the mouth of Walker Creek (UCCE 1995), provide additional evidence of significant sedimentation in the watershed.

Rural residential. The Lagunitas Creek watershed has high densities of rural residences which use significant amounts of water. Roach can persist in intermittent pools but, should increased water demand in summer and early fall cause more widespread or complete drying of streams (particularly in the context of predicted climate change impacts – see Effects of Climate Change section), roach are likely to be extirpated from many stream reaches or even entire watersheds.

Urbanization. Marin Municipal Water District (MMWD) maintains extensive water transfer infrastructure throughout the Lagunitas Creek and Walker Creek watersheds. It is believed that MMWD reservoirs now capture about 40% of the fresh water that historically flowed into Tomales Bay (TBWC, 2003). Much of the captured water is transported out of these watersheds to supply the population centers and residents of central and southern Marin County.

Mining. The legacy of past mercury mining in the Walker and Arroyo Sausal watersheds continues to contribute to persistent water quality problems in this region (Marshall 2008). High winter flows have repeatedly washed large amounts of mercury-rich sediments into streams from the former Gambonini Mine. The Gambonini Mine, which closed in 1970, was declared a superfund site in 1998 and remediation of the site was completed in 2000. However, as of 2001, sediment samples collected in Walker Creek and Tomales Bay contained high concentrations of mercury (Smelser and Whyte 2001). High levels of mercury are also found in Soulajule Reservoir on Arroyo Suasal, a tributary to Walker Creek; a Marin County Health Advisory (2009) warns against eating fish from the reservoir. The effects of mercury on roach populations are unknown.

Instream mining. Sand was mined from the streambed at the confluence of Lagunitas and Nicasio creeks until a short time after the construction of Nicasio Dam in 1960 (Marin County Watershed Management Plan 2004). Commercial gravel mining was never widespread although, in the past, ranchers regularly harvested small amounts of streambed gravel to maintain ranch roads. Such gravel extraction is now rare.

Alien species. Soulajule Reservoir on Arroyo Sausal, tributary to Walker Creek, contains largemouth bass, green sunfish, black crappie, bluegill and channel catfish (CDFG 1978). Escapees from the reservoir during high flow events or through intentional movement, especially centrarchids species, could displace roach in Walker Creek. Similar threats exist below the many dams in the Lagunitas Creek watershed.

	Rating	Explanation
Major dams	Medium	Dams fragment populations and alter flow regimes; multiple dams exist within Tomales roach range
Agriculture	Low	Agricultural diversions, landscape changes and dairy effluent have degraded habitats
Grazing	Medium	Heavy grazing has occurred in Tomales roach range; legacy effects from intensive past grazing and dairy operations remain
Rural residential	Medium	Residential water withdrawal is a potential cause of decreased summer and fall base flows in small streams
Urbanization	Low	Largely rural and agricultural land use
Instream mining	Low	Little instream mining occurs in western portions of Marin County
Mining	Low	Legacy effects from mercury mining in Tomales roach range result in high levels of contamination in fish tissues; impacts to roach are unknown
Transportation	Medium	Roads and road crossings result in increased siltation, channelization, habitat degradation and potential pollutant input
Logging	Low	Substantial legacy effects may still exist; much greater historical impact
Fire	Low	Fire may cause local extirpation, particularly if fire frequency and intensity increase under predicted climate change scenarios
Estuary alteration	Medium	Roach do not use estuarine habitats; however, estuarine marshes may provide freshwater connectivity between adjacent watersheds during flood events, increasing gene flow
Recreation	Low	Impacts likely minimal
Harvest	n/a	
Hatcheries	n/a	
Alien species	Medium	Intolerant of introduced predatory fishes, especially centrarchids such as green sunfish, which exist in upstream reservoirs

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

Effects of Climate Change: Tomales roach are well adapted to the warm, arid conditions of California's Mediterranean climate; however, their frequent dependence on intermittent pools suggests that they are also particularly susceptible to increasing aridity associated with climate change. Roach are one of the few native fish that are able to endure life in isolated summer pools in intermittent streams where temperatures are high, dissolved oxygen levels are low and most other fishes cannot survive. John O. Snyder (1905) observed roach were able to persist when "nothing remains of the stream but a few small disconnected pools." However, increasing water demands, coupled with predicted climate change impacts, may lead to more widespread drying of stream segments and elimination of roach populations. As a result, Moyle et al. (2013) rated Tomales roach as "highly vulnerable" to extinction by 2100 as the result of climate change.

Status Determination Score = 3.1 - Moderate Concern (see Methods section Table 2). Tomales roach do not appear to be in immediate danger of extinction, although fragmentation and isolation of existing populations, along with long-standing habitat alterations, may be limiting their distribution and abundance. Predicted outcomes of climate change may pose additional risks. The status of peripheral populations (e.g. Pine Gulch Creek, Salmon Creek) remains uncertain. The Tomales roach is listed by NatureServe as "G5T2T3, Critically Imperiled."

Metric	Score	Justification
Area occupied	2	Known populations confined to Walker and Lagunitas watersheds
Estimated adult abundance	4	Two large populations in the mainstems of Walker and Lagunitas creeks but isolated peripheral populations may be quite small
Intervention dependence	3	Survey of Pine Gulch Creek needed; monitoring of other populations required to establish trend information
Tolerance	4	Remarkably resilient fish
Genetic risk	3	Little threat to genetic integrity of large populations (e.g., Walker and Lagunitas mainstem populations); uncertainty about genetic integrity of peripheral populations
Climate change	3	Climate change, along with increasing human demand for water, may lead to more widespread drying of streams, possibly extirpating roach from stream reaches or entire watersheds
Anthropogenic threats	3	See Table 1
Average	3.1	22/7
Certainty (1-4)	2	Little published information

Table 2. Metrics for determining the status of Tomales roach, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: Studies are needed to address gaps in knowledge of the life history, taxonomy and habitat requirements of Tomales Roach, as well as the water budget in their limited stream habitats. Understanding the relationship between anthropogenic water use and stream flow is of utmost importance in developing effective management strategies for all native fishes; this is especially true in the intensively managed streams of western Marin County.

Opperman and Merenlender (2004) studied and provide management recommendations for nearby Russian River tributaries, including maintaining live trees (live woody debris) both in riparian areas and in-channel to create habitats that roach prefer. These recommendations would likely also benefit native fishes in Marin County watersheds. The following are regionally-specific management recommendations to ensure the persistence of Tomales roach:

Riparian fencing. Installation of exclusion fencing to prevent cattle from direct access to stream habitats has been a very successful restoration technique in Marin County watersheds and should be encouraged wherever cattle and other livestock have unimpeded access to streams. Off-site water sources (guzzlers) should be part of grazing mitigation efforts.

Support for local watershed groups. Citizens involved in the Lagunitas Creek watershed have provided much in the way of stream restoration and other watershed stewardship practices through nonprofit groups such as the Salmon Protection and Watershed Network (SPAWN) and Trout Unlimited. The Marin Municipal Water District has been an active partner with these organizations.

Balancing water needs. Merenlender et al. (2008) developed GPS-based water resource analysis tools that seek to quantify and balance water needs and water resources on a watershed scale. These tools were created to aid in sustaining instream flows, while simultaneously enhancing water security for local landowners and vineyard operators. This powerful software can be used to evaluate various water-policy scenarios, estimate the cumulative effects of water extraction methods on the natural hydrograph across a large spatial scale (including temporal variation) and provide information for watershed-level planning required to recover environmental flows. In order to ensure minimum base flows, especially in stretches of stream not fed by environmental releases from dams, use of such tools could be of great value.

Figure 1. Distribution of Tomales roach, *Lavinia symmetricus* ssp., in California.

GUALALA ROACH
Lavinia parvipinnis (Snyder)

Status: Moderate Concern. Populations of Gualala roach could decline rapidly or become extirpated as the result of stream alteration and water withdrawal associated with development, especially conversion of forest lands to vineyards and residences.

Description: Gualala roach are small (adult size typically 50-80 mm), bronzy cyprinids most similar to the Navarro roach. However, this species differs from other roaches by having smaller scales (54-65 along the lateral line), shorter, more rounded fins, a short snout and a more robust body. Gualala roach have a mean of 8.0 dorsal fin rays (7-8) and 7.2 (6-8) anal fin rays (Hopkirk 1973). Snyder (1913 p. 66) described Gualala roach as having "a light lateral stripe 2 scales wide extending from upper edge of gill opening to base of caudal and entirely above the lateral line; below is a somewhat wider dark stripe, which in turn is followed by several narrower and very distinct dark stripes which grow lighter ventrally."

Taxonomic Relationships: Gualala roach were first collected by Snyder (1908c, p. 175) who recognized them as *Rutilus symmetricus* but said that they bore "a distinctive local stamp by which they can be recognized without difficulty." In 1913, Snyder revised the systematics of roach, describing the Gualala roach as one of six roach species and erecting a new genus, *Hesperoleucus*, to house them. Murphy (1948c), in an unpublished MS thesis, relegated the Gualala roach to a subspecies of *Hesperoleucus* (*Lavinia*) *symmetricus*. Although his thesis was never published, Murphy's (1948c) diagnosis was adopted by subsequent workers (Hopkirk 1973, Moyle 1976, Hubbs et al. 1979). For a comprehensive review of the history of roach systematics, see the Central California roach account in this report.

Despite the century-long controversy surrounding roach taxonomy, all workers since 1913, whether they used morphometric or genetic methods, have agreed that the Gualala roach is among the most distinct of all roach taxa. Recently, Aguilar et al. (2009) used both nuclear microsatellite (nDNA) and mitochondrial DNA (mtDNA) markers in the most comprehensive genetic study of *Lavinia* to date. Employed in tandem, these two genetic markers supply insights into both the relationships between populations (phylogenetics) and the distinctiveness of individual populations (taxonomy). Analysis of mtDNA identified roach from the Pit and Gualala rivers to be highly divergent from all other populations and reciprocally monophyletic for the haplotypes assayed, suggesting that these populations have been isolated for considerable time. In addition, the microsatellite analysis showed Gualala roach to be a distinct genetic unit.

In light of: (1) the recent genetic analysis (nuclear and mtDNA) that corroborates the distinctiveness of the species that Snyder (1913) originally described; and (2) the fact that Snyder's original species names were never properly submerged (i.e. through formal publication of an analysis in the peer-reviewed literature), it remains that *Lavinia. s. parvipinnis* (Murphy 1948c) is pre-occupied by *Lavinia parvipinnis* (Snyder 1913) and so the Gualala roach merits recognition as a valid full species.

Life History: No studies have been done specifically on Gualala roach life history but, presumably, their life history is similar to that of the Navarro roach (Fry 1936), Russian River roach and other roach species and subspecies (see the Central California roach account in this report for a detailed description of roach life history).

Habitat Requirements: Data pertaining to Gualala roach habitat requirements are lacking but it is assumed they are similar to those of Navarro roach (Fry 1936) and Russian River roach, as their most proximate relatives occupying similar northern coastal stream habitats. Stream surveys carried out by the California Department of Fish and Wildlife (CDFW) and others over the past several decades show that roach have increased in abundance, while coho salmon have almost completely disappeared and steelhead abundance has declined dramatically from the Gualala River (Higgins 1997). These population trends (increase of roach, decline of salmonids) are the direct result of warmer water associated with habitat degradation related to deforestation and development. Roach are a warm water-adapted species and can survive extremely warm water temperatures, while salmonids are highly cold water-dependent.

Distribution: Gualala roach are confined to the Gualala River and its tributaries. They are the dominant fish taxon (both in biomass and number) in the South and Wheatfield forks and most headwater streams (Entrix 1992, EIP 1994, DeHaven 2006, 2007) but occur in lesser numbers in the colder North Fork (Parker 1964c, Parker et al. 1964b, CDFG 1991). They are present in reduced numbers in the mainstem below its confluence with the North Fork (Kimsey 1952, DeHaven 2006, 2007) and have been recorded only in small numbers in the estuary (Brown 1986).

Trends in Abundance: Historically, Gualala roach were present throughout the Gualala river basin, but were likely less abundant than they are today (Higgins 1997). Although no population estimates have been conducted for roach in the Gualala watershed, salmonid surveys carried out by the CDFW and others indicate that roach may have increased in abundance due to habitat alterations favorable to warm water-tolerant species (Higgins 1997).

Nature and Degree of Threats: The hydrology of the Gualala River basin has been dramatically altered by past and ongoing land use practices, especially logging, which was historically intensive in the region. Simplification of stream habitats resulting from logging practices, particularly increases in sediment delivery and solar input, have led to decreased aquatic habitat in summer as flows become subsurface beneath aggraded gravel streambeds. In 2008, many perennial pools in the Wheatfield Fork went dry. Pool elevation dropped quickly and reached levels of desiccation never before observed (Boccone and Rowser 1977, DeHaven 2008). NMFS (2008) stated:

> "Very low summer flow conditions were noted by DeHaven in the extreme drought condition years of 1976-77 in larger streams of the Gualala River watershed. Three decades later many reaches of the same streams were observed to be dry even in normal water years, resulting in the loss of summer rearing habitat, which is

> attributed to increased water diversions (both legal and illegal) and other anthropogenic activities…
>
> Intensive logging and roading, along with recently developed vineyards in the Gualala River watershed are likely responsible for reduced summer flow that have been noted by biologists during the summer months."

Thus, while Gualala roach may have benefitted from the degradation of stream habitats in the past, their future persistence in the system may be threatened if present trends continue. Stressors potentially limiting roach abundance and distribution in the Gualala River watershed are: (1) agriculture, (2) rural residential development, (3) urbanization, (4) logging, (5) transportation (roads), (6) grazing, (7) fire, and (8) and alien species (Table 1). These impacts are not necessarily listed in order of severity and do not operate independently but, instead, must be viewed in aggregate as cumulative watershed impacts.

Agriculture. Historically, agricultural water use in the Gualala River watershed was minimal; however, vineyards are now being developed at a significant scale in the watershed and water used for irrigation and frost protection is significantly affecting flows in Gualala basin streams (J. Katz, personal observations, 2009). Pumping for frost protection in spring is an acute threat, as simultaneous withdrawal from multiple sources across large geographic areas can dry streams completely. Vineyard expansion may have either direct or indirect impacts (or both) on tributary flow if surface water is used for irrigation or if groundwater extraction lowers the water table. Deitch et al. (2009a,b) showed that water use for vineyard irrigation and frost protection is significantly affecting in-stream flow in Russian River tributaries in Sonoma County. It is likely these same impacts are occurring in the nearby Gualala watershed.

Conversion of forestlands to vineyards is a principal threat to fishes and other aquatic organisms in the Gualala watershed. The National Marine Fisheries Service (2008) highlighted some impacts to aquatic species from such conversions of timberland: "conversion of timber lands to new vineyard development in the basin are of particular concern for both sediment runoff and water usage because agricultural water use is highest during summer, when sufficient flow is essential." Of particular concern is a proposal for the largest conversion of forestland to vineyards in California, which is slated to occur in the Gualala watershed. This proposal calls for cutting more than 1600-acres of forest and converting 200-acres of grassland to grape cultivation. In addition, 90 "vineyard estates" are proposed. This project has apparently been halted by the proposed purchase of the lands by a consortium of conservation organizations (Santa Rosa Democrat, February 27, 2013) but the fundamental threat of landscape conversion in other parts of the watershed remains.

Fertilizers and other agricultural pollutants are also of concern in that they are known to augment algal production in rivers with elevated temperatures; the Gualala River is listed as impaired by both excessive sediment and high temperature (US EPA 2002), increasing the risk of algal blooms and eutrophication of streams in the Gualala watershed.

Marijuana cultivation may also be an increasing threat, although no studies specific to the Gualala watershed have been performed to document impacts from water withdrawls or pollutant inputs from fertilizers.

Rural residential development. The northern coastal basins of California are increasingly developed for rural residences. While roach can apparently persist in degraded habitats, populations may decline or become extirpated due to a combination of increased water diversion during low-flow periods, polluted inflow from septic tanks or other non-point sources, siltation from roads, and loss of complex habitat through bank stabilization projects. In the mid-1990s, it was projected that rural residential development resulted in the use of up to 2.5 cubic-feet-per-second of surface water from the Gualala River, on a basin-wide scale (EIP 1994). Water withdrawls are now likely much higher, in light of ongoing rural and viticultural development over the past 20 years. The cumulative ecological impacts of development on such a large scale are of high concern, particularly in how they contribute to degradation of aquatic habitats in the Gualala basin.

Increasingly, residential water demand during low flow periods (late summer and early fall) is being supplemented by trucking in water pumped from other sources. In the face of climate change and possible reductions or temporal shifts in annual precipitation, the fact that demand, at times, already exceeds the Gualala basin's water supply is of great concern.

Urbanization. Although the Gualala basin is largely rural, the river supplies water to two municipal water districts that service the towns of Gualala, Mendocino County, and Sea Ranch, Sonoma County. Both areas continue to grow, along with demand for water, resulting in controversy surrounding the appropriative water rights of the North Gualala Water Company.

Logging. The Gualala River watershed was heavily logged beginning in the mid-1800s and has continued to support substantial timber harvest for over 150 years. Aerial photos from as late as 1952 "show mature stands of trees in the forested areas of the watershed, with very few roads." However, "…by 1965, aerial photos of the watershed show large areas denuded of trees and intensively scarred by roads and skid trails. The logging practices of the time had little consideration for water quality and fisheries, as evidenced by the common practice of using stream channels as roads and landings" (California Regional Water Quality Control Board 2001). By the 1980s, most Gualala basin forestlands contained second or third growth redwoods and Douglas fir, along with tanoak and other deciduous trees. The consequent reduced value of these timberlands is a principal reason for recent conversion of forestlands to vineyards, resulting in further reductions in stream flows and increasing stream temperatures. Ironically, timber harvest has likely benefited Gualala roach by contributing to increased stream temperatures and eliminating cold water-requiring competitors and predators (albeit native ones), such as steelhead and coho salmon. However, the large-scale conversion of a diverse forested landscape to one dominated by agricultural land use is likely to eliminate large areas of roach habitat through reduced stream flows, further degraded habitats and increased pollution input.

Transportation. Roads to facilitate logging, rural development and vineyard expansion are widespread throughout the Gualala basin; this extensive road network changes the annual hydrograph by facilitating more runoff during storm events and inhibiting groundwater (aquifer) storage, which is critical for maintaining stream base flows during low flow periods. Ranch and logging roads are also the largest source of sediment delivery to Gualala system streams (Klampt et al. 2002) and are a high priority for erosion control projects by CDFW. Culverts and other road crossing may create barriers to upstream fish movement which can lead to the isolation of populations or prevent recolonization of stream reaches.

Grazing. Impacts from grazing in the Gualala watershed are pervasive but are likely reduced from historic levels (J. Katz, unpublished observations, 2009). Impacts are likely similar to those described for the Navarro River basin (see the Navarro roach account in this report).

Instream mining. Past gravel mining in the vicinity of the confluence of the South and Wheatfield Forks simplified habitats, reduced water quality (increased turbidity) and impeded natural geomorphic processes such as pool scour and deposition (NMFS 2008). Legacy effects may continue to contribute to decreased habitat quality and quantity in this portion of the watershed.

Fire. Fire is a natural, if historically infrequent, process in the Gualala River watershed. However, fires are now more frequent and their effects are more severe because of land management practices and associated changes to the landscape. Long-standing fire suppression policies have increased fuel loads, while historic logging has dramatically increased solar input in deforested areas and led to drier fuels. Thus, more severe and frequent wildfires, coupled with predicted reduction in annual precipitation associated with climate change, may threaten roach habitats or eliminate localized populations, especially in smaller headwater tributaries in more arid portions of the basin.

Recreation. Little direct threat to roach exists from recreation, except when large woody debris is removed from streams to facilitate recreational boating or impoundments are created for 'summer swimming holes.'

Alien species. Roach populations decline and can be eliminated in the presence of alien fishes, especially centrarchids such as green sunfish (*Lepomis cyanellus*) and black basses (*Micropterus* spp.) (Moyle 2002). Centrarchids have been recorded in stream surveys in the Gualala drainage (Entrix 1992, EIP 1994) and may threaten roach populations in portions of the basin. Thus, expansion of existing alien populations, transportation of alien fishes over natural barriers by humans, or escape of non-native fishes from stock ponds during high flow periods when ponds spill and become interconnected with adjacent streams, all pose a serious threat to the persistence of roach in the Gualala watershed.

	Rating	Explanation
Major dams	n/a	No major dams in watershed
Agriculture	High	Water withdrawals associated with expanding viticulture and rural development have increased dramatically
Grazing	Medium	Grazing is common throughout the watershed and cattle often concentrate in riparian areas
Rural residential	Medium	Residential water withdrawal is increasing and contributes to decreased base flows in small streams throughout the watershed
Urbanization	Low	Sea Ranch and the North Gualala Water Company both draw from the Gualala River Aquifer
Instream mining	Low	Localized gravel mining has simplified habitats, increased turbidity and contributed to drying of intermittent pools; greater impact in the past
Mining	n/a	No known threats from hardrock mining
Transportation	Medium	Much of the Gualala River and its tributaries are bordered by paved roads, while a network of logging and ranch roads contributes to siltation, channelization, and habitat loss
Logging	Low	Logging continues in the watershed; much greater impact in the past but legacy effects persist due to intensive historic timber harvest in the region
Fire	Medium	More frequent and intense fires may cause local extirpations, especially in smaller headwater tributaries
Estuary alteration	Low	Relatively intolerant to salinity
Recreation	Low	Minor alterations occur in summer (e.g., impoundment building for swimming and water play)
Harvest	n/a	
Hatcheries	n/a	
Alien species	High	Intolerant of introduced predatory fishes, especially centrarchids (e.g., green sunfish)

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

Effects of Climate Change: Gualala roach are well adapted to the warm, arid conditions of California's Mediterranean climate. However, their frequent dependence on intermittent pools suggests that they are also particularly susceptible to increasing aridity associated with climate change. Roach are one of the few native fish that are able to endure life in isolated, warm pools with low dissolved oxygen levels in intermittent streams. However, increasing water demands, coupled with predicted climate change impacts, may lead to more widespread drying of stream segments and elimination of roach populations. The middle reaches of Wheatfield Fork dried completely in 2008, indicating that limiting flow conditions already exist and further reductions in precipitation and aquifer recharge may pose a substantial threat to roach and other native fishes. Moyle et al. (2013) rate Gualala roach as "highly vulnerable" to extinction as the result of climate change in conjunction with existing stressors.

Status Determination Score = 3.0 – **Moderate Concern** (see Methods section Table 2). Gualala roach should remain a Species of Special Concern, given increasing threats from agricultural development (e.g., viticulture), rural residential development, climate change, and legacy impacts from logging and other land uses which dramatically altered aquatic habitats in the Gualala watershed. The Gualala roach is listed by NatureServe as "G5T1T2, Critically Imperiled."

Metric	Score	Justification
Area occupied	1	Confined to the Gualala River and its tributaries
Estimated adult abundance	5	Populations assumed to be large but survey data are lacking
Intervention dependence	3	The Gualala River watershed is rapidly changing; frequent fish monitoring and management is needed; possible reintroductions required
Tolerance	5	Remarkably resilient fish
Genetic risk	4	No known threats to genetic integrity
Climate change	1	Highly vulnerable in combination with growing human water demands
Anthropogenic threats	2	See Table 1
Average	3.0	21/7
Certainty (1-4)	2	

Table 2. Metrics for determining the status of Gualala roach, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: Additional studies, particularly related to the life history of Gualala roach, should be performed to better inform our understanding of their needs. The Gualala River fish community has changed over time from one dominated by salmonids to one that favors warm water tolerant species, such as Gualala roach. If ongoing watershed restoration projects succeed and cold water flows are maintained year-round, the fish community structure should shift back to one dominated by salmonids (Higgins 1997). However, the Gualala watershed is being rapidly converted to open agricultural lands with surrounding patchy forests that are highly altered; as such, future stream flows are likely to continue to decrease. Thus, it is important to establish a monitoring program to document the distribution and status of Gualala roach, coho salmon, steelhead trout and other native fishes throughout the watershed. It is equally important to monitor the distribution and abundance of alien species (e.g., centrarchids) in order to prioritize management and conservation measures to protect native fishes.

The Gualala River Watershed Assessment Report (prepared by the California Resources Agency and California Environmental Protection Agency, for guidance on water demand and water supply in the Gualala River, 2003) states: "Any water extraction from surface or groundwater supplies, depending on the amount, location, and season, can affect streamflow, water quality, and consequently fish habitat."

With this in mind, pressure from rural residential development, along with forestland conversion and vineyard expansion, must be carefully weighed against the limited water resources in the Gualala basin. The establishment of minimum base flows in the Gualala River and its tributaries to support Gualala roach, coho salmon, and steelhead trout is of particular importance. Along with maintaining flows, restoration activities should focus on minimizing sediment delivery to streams, restoring healthy riparian zones and establishing refuge stream segments that are managed to benefit native aquatic species.

In addition, Merenlender et al. (2008) developed GPS-based water resource analysis tools which seek to quantify and balance water needs and water resources on a watershed scale. These tools were created to aid in sustaining instream flow while simultaneously enhancing water security for local landowners and vineyard operators. This powerful modeling program can be used to evaluate various water-policy scenarios, estimate the cumulative effects of water extraction methods on the natural hydrograph across a large spatial scale (including temporal variation), and provide information for watershed-level planning required to recover environmental flows. Such tools would be of great value in the Gualala basin, especially in light of the many stressors facing aquatic habitats and fishes in this highly altered landscape.

Figure 1. Distribution of Gualala roach, *Lavinia parvipinnis* (Snyder), in California.

NORTHERN ROACH
Lavinia mitrulus (Snyder)

Status: High Concern. This species is restricted to a few isolated populations in California which could decline rapidly and face extirpation as result of alterations to streams, invasion of alien fishes, water withdrawal for agriculture and predicted impacts from climate change.

Description: Northern roach are small (adult size typically 50-100 mm), bronzy cyprinids. They have a robust body, deep caudal peduncle, short snout and short rounded fins. They are dark on the upper half of the body, light below, and very similar in appearance to the Central California roach. Northern roach differ from Central California roach in having short rounded fins and "cup-like" scales (see Snyder 1913 for more detail on scale morphology). Snyder (1908a) published morphometric data on 20 fish from Drews Creek (Lake County, Oregon), among them the type specimen of the species; all individuals had 8 dorsal rays and 7 fin rays. Snyder found that male roach had longer, larger fins than did females, especially pectoral fins; he also found that the sexes could be differentiated by the ratio of pectoral fin length to body length. These differences in the relative fin length between the sexes led Snyder to publish one of the first accounts of general sexual dimorphism in cyprinid fishes.

See the Central California roach account in this report for a more in-depth description of general roach morphology.

Taxonomic Relationships: Northern roach were first collected in 1898 by C. Rutter (1908), who recognized them as *Rutilus symmetricus* (Baird and Girard 1854a). Speaking of the specimens collected on this trip, Rutter (1908 p. 139) said *"We have but few small specimens of this form, the longest being but 3 inches long. They were taken in North Fork Pitt (sic) River near Alturas and at the mouth of Joseph Creek, several hundred miles from where any other specimens of* symmetricus *have been taken. The form may prove to not to be* symmetricus*, but we can not identify it otherwise with the material at hand."*

In 1904, John O. Snyder surveyed broadly in northeastern California and southeastern Oregon, collecting in the upper Pit River, along with the Goose Lake, Summer, Abert, Harney and Warner basins of Oregon but found roach only in the tributaries to Goose Lake, Lake County, Oregon (Snyder 1908a). Snyder (1913) erected a new genus, *Hesperoluecus,* and described six new species based on locality, isolation and morphological differences. Among the new species was the northern roach, *Hesperoleucus mitrulus,* from Drews, Muddy, and Cottonwood creeks, Lake County, Oregon. Snyder also reported that the species had not been recorded from Goose Lake itself or from the high-gradient Californian streams that flow into the lake from the Warner Mountains to the east. There is no indication that he was aware of the previous collection of roach in the Pit River by Rutter.

Northern roach were classified as a distinct species of *Hesperoleucus* by subsequent workers (Evermann and Clark 1931, Shapovalov and Dill 1950, Shapovalov et al. 1959), but Miller (1945a p. 197) suggested the "Preliminary analysis of the forms of *Hesperoleucus* shows that many if not all, of those described as species are geographic subspecies of *H. symmetricus*." Murphy (1948c), in an unpublished master's thesis, proposed that all coastal forms be demoted to subspecific status and submerged into *H. symmetricus*. Murphy (1948c) did not study samples of the northern roach, nor did he suggest that his subspecific diagnosis should be applied to *H. mitrulus*. However, it appears that when Murphy's (1948c) subspecific diagnosis for *H. parvipinnis, H. navarroensis, H. venustus and H. subtitus* was adopted by subsequent workers (Hopkirk 1973, Moyle 1976, Hubbs et al. 1979), subspecies status was erroneously applied to *H. mitrulus* as well. For a thorough discussion of the debate over the specific status of all roach forms, see the Central California roach account in this report.

The first inclusion of roach from the Pit River in *mitrulus* was by Hubbs et al. (1979 p. 11), who used the common name "upper Pit roach" when referring to *H. mitrulus*. While no mention is made of a range extension for the taxon, it is assumed that this change was precipitated by the 1934 collection of 19 roach in the North Fork Pit River near Alturas, Modoc County (unpublished field notes and collections of Carl Hubbs at the University of Michigan, as reported in Reid et al. 2003). The California Department of Fish and Wildlife (Shapovalov et al. 1981) subsequently applied the common name "upper Pit" roach to *H. symmetricus* but, like Hubbs et al. (1979), did not publish distributional information. Moyle et al. (1995) and Moyle (2002), list the "Pit" roach (i.e. *mitrulus)* as being native to the upper Pit River system, as well as to Oregon tributaries of Goose Lake.

Northern roach are reciprocally monophyletic for mtDNA haplotypes and show strong differentiation from all other roach populations based on nuclear microsatellites (Aguilar et al. 2009). Based on mtDNA sequence diversion, Aguilar et al. (2009) estimate that the northern populations of roach have been isolated for 8 million years.

In light of: (1) the recent genetic analysis (nuclear and mtDNA) that corroborates the distinctiveness of northern roach as described by Snyder (1913); and (2) the fact that Snyder's original species were never properly submerged (i.e. through formal publication of an analysis in the peer-reviewed literature), the northern roach is a valid full species. The subspecies name, *Lavinia s. mitrulus* (Hopkirk 1973) is pre-occupied by *Lavinia mitrulus* (Snyder 1913). Many variations of the common name "upper Pit" or "Pit River" have been applied to *mitrulus*; however, because the range consists of multiple isolated basins and because the type locality is in Lake County, Oregon, Snyder's original name for the taxon, "northern roach," seems most fitting.

Life History: Northern roach presumably share much of their life history with Central California roach but the specific life history attributes of northern roach have not been studied so cannot be verified.

Habitat Requirements: Northern roach tend to be associated with spring pools and swampy stream reaches, habitats dissimilar from those occupied by roach in the rest of California (S. Reid, pers. comm. 2009). Thus, in Ash and Rush creeks, Lassen and Modoc counties, roach are found in small numbers inhabiting the weedy margins of streams and, in one case, an isolated spring pond (Moyle and Daniels 1982, S. Reid, pers. comm. 2009). They do not often occupy intermittent streams in the Pit system, as is usual with roach in the rest of their range. Instead, speckled dace (*Rhinichthys osculus*) dominate these habitats.

Moyle and Daniels (1982) found that 94% of the fish species that co-occurred with northern roach were also native. The most common associates were speckled dace, Sacramento sucker (*Catostomus occidentalis*) and Pit sculpin (*Cottus pitensis*). The fact that roach occur as part of a predominately native fish assemblage has been observed elsewhere (Moyle and Nichols 1973, Leidy 1984, Brown and Moyle 1993, Leidy 2007). Moyle (2002) attributes the uncommon co-occurrence of roach with alien species to the tendency for roach to be easily displaced by invasive fish species, especially centrarchids.

Distribution: In California, northern roach are restricted to several tributaries of the upper Pit River. It is likely that they once inhabited the meandering valley floor reaches of the Pit River in Big Valley, Modoc County, but this area is now completely dominated by alien species (Moyle and Daniels 1982). Roach have not been recorded from Goose Lake itself or from the high-gradient Californian streams that flow into the lake from the Warner Mountains to the east. However, roach found in the northern tributaries of Goose Lake in Lake County, Oregon are also included in *H. mitrulus*. In a recent comprehensive sampling of the Oregon portion of the Goose Lake watershed, the Oregon Department of Fish and Wildlife (ODFW) found northern roach to be widespread and relatively abundant (>80 fish/km) in Dry, Drews, Hay, Dent, Muddy and Augur creeks (Heck et al. 2008).

Roach populations in the terminal lake basins adjacent to Goose Lake, in the high desert of eastern Oregon, may also belong to this species but distributional records are spotty and taxonomic relationships among these populations remain uncertain.

Pit River Falls, located five miles downstream of the town of Fall River Mills, Shasta County, divides the Pit River basin into upper and lower drainages. The falls are, at least partially, a barrier to fish movement. Historically, they represented the northern range limit for some Sacramento River basin fishes, such as tule perch, *Hysterocarpus traski* (Moyle 2002). Only roach found above Pit River Falls are considered northern roach, *L. mitrulus*. Roach found below the falls would have historically had unimpeded access to Sacramento River system and are assumed to be *L. s. symmetricus*. However, genetic studies have not been conducted and relationships remain uncertain.

Historical collecting trips to the upper Pit River system captured only a few specimens (Rutter 1908, Hubbs et al. 1934, from field notes and collections at the University of Michigan, as reported in Reid et al. 2003) or none at all (Snyder 1908a). In the most comprehensive sampling of the Pit system to date, Moyle and Daniels (1982) found roach at only 8% of 261 collection sites. Above Pit River Falls, roach were found in three drainages: (1) Ash–Rush–Willow Creek drainage, Lassen and Modoc counties, (2) Bear Creek, tributary to the Fall River, Shasta County and (3) Beaver Creek, Lassen County.

Trends in Abundance: Historically, roach were probably much more widely distributed in the upper Pit River drainage (e.g., Big Valley) but modern surveys have found that they have disappeared from reaches in which they previously occurred (reviewed in Reid et al. 2003). Reid et al. (2003), in the only known survey of the Upper Pit drainage since 1978, surveyed 12 sites in the North Fork, South Fork and upper mainstem Pit River (between Alturas and Rose Canyon) without collecting roach. The following is a history of roach occurrence in the upper Pit River basin:

North Fork Pit River. Rutter (1908), collecting in 1898, captured "a few small specimens" of roach. Snyder (1908), collecting in 1904 near the same location, did not capture any roach, while Hubbs and others collecting in the North Fork near Alturas in 1934 captured only 19 (from field notes and collections at the University of Michigan, as reported in Reid et al. 2003). Subsequent collectors have found green sunfish but not roach (Moyle and Daniels 1982, Reid et al. 2003).

South Fork Pit River. Three historic sampling trips found roach in the South Fork. Modern collecting trips have failed to document roach in the South Fork (from information in Reid et al. 2003).

Mainstem Pit River, Alturas to Pit River Falls. The only known record of capture is a single specimen taken by R.R. Miller in 1961 (from University of Michigan field notes and collections, as reported in Reid et al. 2003). This is the reach flowing through Big Valley which has been highly altered and contains mainly alien species (Moyle and Daniels 1982). However, roach remain common in the Ash Creek drainage (S. Reid. pers. com. 2009).

Nature and Degree of Threats: Factors which limit the abundance and distribution of northern roach are: (1) agriculture, (2) grazing, (3) logging, (4) transportation, (5) fire, (6) and alien species. These impacts are not necessarily listed in order of importance and do not operate independently but, instead, must be viewed in aggregate, along with other less pressing threats (Table 1), as cumulative and synergistic watershed impacts.

Agriculture. Agricultural alteration of the Pit River basin has a long history. The earliest fish survey of the region (1898) already described the South Fork Pit as being "almost drained by irrigation ditches" (Rutter 1908, p. 110). The low gradient areas favored by roach are also areas in which extensive pasture, hay, and other types of farming occur. For example, much of Big Valley, through which the Pit River flows, is devoted to growing alfalfa, pasture, and potatoes. It is likely that the river in this region was once habitat for roach but agricultural alteration, combined with abundant alien species, has made it unsuitable habitat. Many tributary streams in this region are channelized to reduce spring flooding of pasture and agricultural lands, a practice which eliminates roach habitat (Moyle 1976). The relationship between water withdrawal for irrigation and stream flow is not documented in the region, but Pit River flows are low and polluted with agricultural return water between Alturas and Fall River Mills, as evidenced by the Pit River being listed as impaired by high temperature, nutrients and low dissolved oxygen content under The Clean Water Act section 303(d) (U.S. EPA 2006).

	Rating	Explanation
Major dams	n/a	No major dams in the upper Pit drainage; however, there are numerous small dams and diversions
Agriculture	Medium	Diversions and return water have altered hydrology and water quality; channels have been altered
Grazing	Medium	Most streams have been heavily grazed
Rural Residential	Low	Residential water withdrawal may cause decreased summer flows in many small streams
Urbanization	Low	Urban areas occupy only a small portion of the watershed
Instream mining	Low	Limited; effects unknown
Mining	n/a	No known threats from mining at present
Transportation	Medium	Much of the river is bordered by roads; logging and ranch roads contribute to siltation, channelization, and habitat loss
Logging	Medium	Logging is a major land use in higher elevation parts of the watershed
Fire	Medium	Fires may cause local extirpation, especially in upper watersheds occupied by isolated populations
Estuary alteration	n/a	
Recreation	Low	Recreation results in little direct threat except through off road vehicle use and similar activities
Harvest	n/a	
Hatcheries	n/a	
Alien species	High	Intolerant of introduced predatory fish, especially centrarchids such as green sunfish

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

Grazing. Livestock grazing is pervasive in the Pit River watershed. Grazing impacts to streams can include: removal of riparian vegetation, stream bank collapse, sedimentation of pools, impaired water quality from sedimentation and animal waste input, and reduction in the amount of cover and shading. If grazing impacts to streams are severe, roach tend to disappear despite their high tolerance of adverse conditions. Stock ponds, which are created to provide water for cattle, can divert water from streams and support populations of non-native predatory fishes in the upper portions of watersheds (e.g. Ash Creek). These fish (e.g. green sunfish, largemouth bass) may colonize adjacent streams during wet periods when ponds spill and become hydrologically connected to streams, potentially eliminating roach populations.

Transportation. Streambeds with adjacent roads and road crossings are subject to fragmentation (where road crossings create barriers to fish movement) and increased sediment and pollutant input, degrading aquatic habitat quality and quantity for roach and other fishes. Roach populations decline when severe channelization of small streams occurs.

Logging. Most of the Pit River watershed that is not devoted to agriculture is covered with dry forestland, which is logged and grazed. Logging in the arid Pit drainage likely contributes sediments to streams, especially considering the nature of the volcanic soils across the region and wide use of highly friable crushed cinders for road base.

Fire. Fire is a natural part of the high desert landscape in the Pit River watershed. However, fires are likely more frequent and severe than they were historically because of human land management practices and associated changes to the landscape (especially fire prevention and consequent shifts in forest vegetation composition and density). Coupled with predicted climate change effects, more severe wildfires may eliminate roach habitats or possibly extirpate small populations from tributary streams.

Alien species. Roach cannot coexist with large populations of alien fishes, especially centrarchids such as green sunfish (*Lepomis cyanellus*) and largemouth bass (*Micropterus* spp.). Green sunfish, largemouth bass and bluegill are found together and often dominate the fish biomass in warm, slow, turbid reaches of the mainstem Pit River (Moyle and Daniels 1982). These stretches of river are now dispersal barriers to roach, further isolating small populations in tributary steams. Roach populations in refuge tributary watersheds are also threatened by escape of alien fishes from stock ponds (treated above under grazing), located higher in these watersheds.

Effects of Climate Change: Northern roach are well adapted to the warm, arid conditions of northeastern California. However, their dependence upon spring pools in late summer and swampy headwaters suggests that they are also particularly susceptible to decreases in base flows. While their ability to persist in small bodies of water bodes well for roach in a future of dwindling in-stream water supplies, it also suggests that they are likely to be extirpated from watersheds with streams that dry completely under the dual strains of increasing aridity associated with climate change and increasing local surface water diversions and ground water withdrawal for rural residential homes and agricultural irrigation. Because of their dependence on small streams in an arid region and the isolation of populations from one another, Moyle et al. (2013) rated northern roach as “critically vulnerable” to climate change.

Status Determination Score = 2.9 - High Concern (see Methods section Table 2). Although northern roach do not appear in immediate danger of extinction, populations are likely to decline and become extirpated from many areas as the result of alterations to streams, introduction of alien fishes and water withdrawal for agriculture, in combination with changes in climate. Existing fragmentation of populations makes re-colonization of streams from which they have been extirpated unlikely. The northern roach (as Pit roach) is listed by the American Fisheries Society as “Vulnerable” (Jelks et al. 2008) and by NatureServe as “G5T2, Imperiled” and by the Oregon Department of Fish and Wildlife as “Sensitive- Peripheral”.

Metric	Score	Justification
Area occupied	2	California range confined to widely separated tributaries of the upper Pit River
Estimated adult abundance	4	Localized populations may be substantial but populations are isolated and survey data are lacking
Intervention dependence	3	Annual monitoring and protection of most populations is needed
Tolerance	5	Remarkably resilient fish but preferred habitat in system is greatly reduced and fragmented
Genetic risk	3	Uncertain genetic relationships between small populations; effects of isolation likely
Climate change	1	Highly vulnerable in combination with watershed changes
Anthropogenic threats	2	See Table 1
Average	2.9	20/7
Certainty (1-4)	2	Relatively little recent information

Table 2. Metrics for determining the status of northern roach, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: A thorough fish population and habitat survey of the Pit River watershed should be performed in order to determine abundance and distribution of native fish populations, including roach, and habitat attributes of both occupied and unoccupied streams (or reaches thereof). Once baseline data are collected, basin-wide monitoring every five years should be established to determine status and trends of native fish populations and their habitats, as well as to detect alien fish invasions. An educational program should be developed for watershed residents, especially agricultural water users, to encourage water conservation measures and cooperative ventures to restore watershed functions in ways that benefit native fish. Consideration should be given toward establishing one or more streams as protected areas for California roach and other native fishes (e.g. Ash Creek). Some protection for northern roach is provided by its co-occurrence in a few streams with Modoc sucker (*Catostomus microps*), which is listed as a threatened species.

The water quality standards recommended by state and federal agencies should be adopted and vigorously enforced, including finding ways to reduce sediment loads (e.g., reducing the impact of roads of all types). Water rights in the entire watershed need to be adjudicated and a minimum flow provided for all streams to provide suitable year-round habitat for native fishes.

A comprehensive genetic investigation of Pit River basin fishes should be implemented, including roach (*Lavinia* species), tui chub (*Siphateles* species) and suckers (*Catostomus* species), in order to clarify taxonomic confusions about the relationship(s) between populations from isolated portions of Oregon and California and to better inform future management and conservation actions.

Figure 1. Distribution of northern roach, *Lavinia mitrulus* (Snyder), in California and Oregon.

SACRAMENTO SPLITTAIL
Pogonichthys macrolepidotus (Ayres)

Status: Moderate Concern. The Sacramento splittail was delisted as a threatened species because of the demonstrated resiliency of its populations. Its abundance could be negatively impacted by ongoing changes to the San Francisco Estuary. In particular, the poorly studied but genetically distinct population in the lower estuary is of concern because of its small size.

Description: Splittail are large cyprinids, growing in excess of 40 cm SL, and are distinctive in having the upper lobe of the caudal fin larger than the lower lobe. The body shape is elongate with a blunt head. Small barbels may be present on either side of the subterminal mouth. Splittail possess 14-18 gill rakers, and their pharyngeal teeth are hooked and have narrow grinding surfaces. Dorsal rays number 9-10, pectoral rays 16-19, pelvic rays 8-9, and anal rays 7-9. The lateral line usually has 60-62 scales, but ranges from 57 to 64. Coloration is silver on the sides and dusky grey dorsally. Adults develop a slight hump behind the head (nuchal hump). During the breeding season, the caudal, pectoral, and pelvic fins take on a red-orange hue and males develop small white nuptial tubercles in the head region.

Taxonomic Relationships: The Sacramento splittail was described in 1854 by W. O. Ayres as *Leuciscus macrolepidotus* and by S. F. Baird and C. Girard as *Pogonichthys inaeqilobus.* Ayres' species description came out first so was accepted as the official description, although *Pogonichthys* became the genus name because of its striking difference from other cyprinids (Hopkirk 1973). The genus *Pogonichthys* comprises two species, *P. ciscoides* (Hopkirk 1973) and *P. macrolepidotus.* The former species was endemic to Clear Lake, Lake County, and became extinct in the early 1970s. Baerwald et al. (2008) showed that there are two genetically distinct populations of Sacramento splittail, one centered in San Pablo Bay around the Petaluma and Napa rivers in the lower San Francisco Estuary, and the other centered around the Delta and Suisun Marsh. Analysis of otolith microchemistry validates that the two populations segregate themselves in different habitats (Feyrer et al. 2007). These two populations would qualify as Distinct Population Segments (San Pablo DPS and Delta DPS), if listed under the Federal Endangered Species Act of 1973. The genetic relationships of splittail from San Francisco Bay tributaries (primarily Walnut Creek and Alameda Creek) are unknown.

Life History: The life history of splittail is reviewed in Moyle (2002), Moyle et al. (2004), and Sommer et al. (2007). Splittail depend both on brackish-water rearing habitats in the San Francisco Estuary and on floodplain and river-edge spawning habitats immediately above the estuary. Most migrate between these two habitat types on a near-annual basis. Historically, non-estuarine dependent populations existed, especially in the southern Central Valley (e.g. Lake Tulare), but these populations have been extirpated. The basic life history pattern for the remaining Delta/Suisun Marsh populations is: (1) from November through February adults migrate upstream in pulses in response to flow events; (2) adults spawn on floodplains or flooded edge habitats in March and April and then migrate back downstream; (3) embryos and larvae remain in flooded vegetation for 3-6 weeks during March and April (Crain et al. 2004); (4) in April and May, as flood waters recede, juveniles leave flooded areas and move downstream; and (5) juveniles rear in estuarine marshes for 1-2 years before spawning for the first time. The success of this life cycle depends on extended, large-scale floodplain inundation (e.g., in the Yolo Bypass or lower Cosumnes River), although some spawning is successful even in non-flood years (Moyle et al. 2004). This pattern may be somewhat different for the Petaluma/Napa populations because adults can spawn in brackish water (up to 5 ppt salinity) and juveniles can rear in water up to 14 ppt (Feyrer et al. 2010). Young-of-year splittail typically stay in water of the same salinity in which they were reared, but some individuals move readily between fresh and brackish water (Feyrer et al. 2010).

Splittail are relatively long-lived (7-10 years) and are highly fecund (up to 150,000 eggs per female). Their populations fluctuate on an annual basis, depending on spawning success and strength of the year class (Daniels and Moyle 1983, Sommer et al. 1997). Both male and female splittail mature by the end of their second year although, occasionally, males may mature by the end of their first year and females by the end of their third year. Splittail are about 18-20 cm SL when they attain sexual maturity (Daniels and Moyle 1983). There is some variability in reproductive period: older fish reproduce first, while younger fish tend to reproduce later in the season (Moyle et al. 2004). Generally, gonadal development is initiated by fall with a concomitant decrease in somatic growth (Daniels and Moyle 1983).

Splittail spawn on submerged annual vegetation in flooded areas or along the edges of rising rivers. The most important known spawning areas are the Yolo and Sutter bypasses and the Cosumnes River floodplain; however, ripe splittail have been found in areas as diverse as the Petaluma River, Suisun Marsh, Sacramento River and lower Tuolumne River. Fertilized embryos stick to plants and larvae remain in and among plants for the first few days of life. Sommer et al. (2008) observed that post-larval splittail (ca. 21 mm FL) sought out shallow areas with emergent and submerged vegetation during the day, but moved into deeper water at night among tules and submerged aquatic plants. Larger fish used deeper water more consistently, both day and night. Fish of all sizes exhibited schooling behavior.

Splittail are benthic foragers that feed mostly on aquatic invertebrates, although detrital material can make up a high percentage of their stomach contents by volume. Juvenile splittail feed on small benthic invertebrates, most consistently chironomid midge larvae; however, diet varies considerably with locality (Feyrer et al. 2007). Adults feed opportunistically on earthworms, clams, insect larvae, and other invertebrates. Historically, splittail fed extensively on opossum shrimp (*Neomysis mercedis*) but their diet shifted with the collapse of shrimp populations, following invasion of the overbite clam, *Potamocorbula amurensis* (Feyrer et al. 2003). Splittail feeding shifted to the clams themselves, as well as other benthic invertebrates.

Splittail are preyed upon by striped bass and other predatory fishes, as well as by aquatic birds. Striped bass preference for splittail as prey has long been recognized by anglers, who often fish for splittail in order to use them as bait.

Habitat Requirements: Splittail are adapted for estuarine life so they are tolerant of a wide range of salinities (0-29 ppt) and temperatures (5-33°C). This tolerance is demonstrated by year-round utilization of Suisun Marsh and the Petaluma River estuary, generally in sloughs < 4 m deep, where summer salinities are typically 6-10 ppt and temperatures range from 15 to 23°C. Feyrer et al. (2010) recorded young-of-year splittail in habitats with salinities as high as 14 ppt; although most young-of-year reared in fresh water, some reared in brackish water. Adults are more tolerant of high temperatures and salinities than juveniles, so optimal conditions are generally where salinities are low (<10 ppt) and temperatures cool (<20°C). They are remarkably tolerant of low dissolved oxygen and can be found, at least for short periods of time, in water where levels are around 1 mg O^2 L^{-1} (Young and Cech 1996, Moyle et al. 2004).

Splittail require a rising hydrograph for upstream migration and flooded vegetation for spawning and rearing areas for their early life history stages. Large flooded areas need to be at least 1 m deep with deeper, more open, areas as refuges from predation for adults and larger juveniles during the day (Sommer et al. 2008). On floodplains, small juveniles prefer to be among vegetation in shallow water during the day but move into deeper water at night.

Both adults and juveniles leave the floodplain as water levels drop and temperatures rise to 15-20°C (Moyle et al. 2007). Young-of-year and yearling splittail are common in beach seine sampling along the Sacramento River between Rio Vista and Chipps Island (R. Baxter, CDFW, pers. comm. 2001). Furthermore, in CDFW Bay Study samples, splittail are most common from stations <20 ft deep. Thus, juvenile splittail appear to concentrate in shallow edge habitats of the Sacramento River as they move downstream to rearing areas. In the lower Napa River watershed, splittail juveniles have been documented using shallow water habitats in recently restored tidal marsh (Stillwater Sciences 2006).

Distribution: The Sacramento splittail is endemic to California's Central Valley and was once distributed in lakes and rivers throughout the Central Valley. Historically, splittail were found as far north as Redding by Rutter (1908), who collected them below Battle Creek Fish Hatchery in Shasta County. Splittail are apparently no longer found in this area, although adults are occasionally observed passing over the fish ladder at Red Bluff Diversion Dam in Tehama County (R. Baxter, CDFW, pers. comm. 2013). They only rarely enter the lower reaches of the Feather River, although Rutter (1908) collected them as far upstream as Oroville. Splittail are observed, on occasion, in the American River and have been collected at the Highway 160 bridge in Sacramento; in the past, Rutter (1908) collected them as far upstream as Folsom. He also collected them from the Merced River at Livingston and from the San Joaquin River at Fort Miller (where Friant Dam exists today). In recent years, they have only been found as high as the confluence of the Tuolumne River and in the lower Tuolumne River. Spawning still occurs on a regular basis in the lower San Joaquin River and juveniles have been found in this region every year from 1995 to 2011 (Contreras et al. 2011, R. Baxter, CDFW, pers. comm. 2013). Occasionally, splittail are caught in San Luis Reservoir, which stores water that has been pumped from the Delta (Moyle et al. 2004).

Splittail were once abundant in Tulare Lake and in other waters of the San Joaquin Valley (Moyle et al. 2004). Snyder (1905) reported catches of splittail from southern San Francisco Bay and the mouth of Coyote Creek in Santa Clara County, but recent surveys are inconclusive as to whether a reproductive population of splittail continues to exist in the South Bay; Leidy (2007) notes records from only 1983 and 2000. Evidence of a persistent population is more convincing in lower Walnut Creek, Contra Costa County, where a 1998 CDFW gill-net survey of tidal reaches of the creek found splittail to be the most abundant fish (Leidy 2007). Sacramento splittail have also been recorded from the estuarine environments of Peyton and Hastings sloughs, near the mouth of Walnut Creek, and in Grayson Creek, just above its confluence with lower Walnut Creek (Leidy 2007).

Splittail are now largely confined to the Delta, Suisun Bay, Suisun Marsh, Napa River, Petaluma River, and other parts of the San Francisco Estuary, while spawning on upstream floodplains and channel edges (Moyle et al. 2004). The consistent presence of young-of-year splittail in the Sacramento River over 200 km upstream of the Delta may indicate that a small population persists there, although their presence may also indicate that some adults make long migrations to find suitable spawning areas, especially in dry years (Feyrer et al. 2005, R. Baxter, CDFW, pers. comm. 2013). In the Delta, they are most abundant in the north and west portions, although other areas may be used for spawning. Non-spawning fish are found in Suisun Bay and Suisun Marsh and the Petaluma and Napa marshes. Adults from both populations are found to forage in open waters of Suisun Bay; however, Suisun Marsh is almost exclusively used by juveniles and adults from the Delta population. Fish from the San Pablo population are found mostly in western Suisun Bay. This suggests that San Pablo fish preferentially live in Petaluma and Napa marshes, closer to their natal watersheds (Baerwald et al. 2008) and away from higher salinities in the Bay.

Trends in Abundance: Splittail were once harvested by Native Americans and by commercial fisheries in the 19th and early 20th centuries. Jordan (1884) recorded them as "very common in the Sacramento and … brought in considerable numbers to the San Francisco Market" (p. 617). Their overall abundance and range apparently shrunk with development of California's water system (e.g., the elimination of Lake Tulare) but they remained a common fish in the Delta (Turner and Kelly 1966).

There are currently seven sampling programs that monitor splittail abundance, along with other fishes in the estuary (Moyle et al. 2004). These programs include: (1) CDFW Summer Townet Survey (started in 1959), (2) CDFW Fall Midwater Trawl Survey (1967), (3) USFWS Chipps Island trawl survey (1975), (4) U. C. Davis Suisun Marsh trawling and seining surveys (1980), (5) USFWS Beach Seine Survey (1979), (6) CDFW San Francisco Bay trawling survey (1980), and (7) fish salvage operations at Central Valley Project (CVP) and State Water Project (SWP) pumps in the south Delta (1979). None of these surveys (or indices calculated from them) were designed specifically for splittail so results from each have to be interpreted with caution, although all have been used in analyses of splittail population trends (Moyle et al. 2004). According to Moyle et al. (2004): "Combined, the surveys indicate that (1) splittail populations have high natural variability, a reflection of their life history strategy, (2) some successful reproduction takes place every year, and (3) the largest numbers of young are produced only during years of relatively high outflow. These findings suggest that the majority of adult fish in the population result from spawning in wet years and lowest numbers are produced during drought years" (p. 13). Essentially, no consistent trends were detected from 1962 through 2002.

When numbers are compared from 1980-2008 between the two programs that catch the most splittail, results are similar. The fish salvage operations (Figure 1) capture migrating adults and juveniles during January through July, while the Suisun Marsh program captures resident fish year-round (Figure 2). Both show a peak in abundance in 1980, presumably the result of a series of wet years, followed by a decline through the 1990s, reflecting an eight-year drought. The USFWS (2003) indicated that there was some evidence of decline in splittail numbers but the species was not in danger of declining to extinction. Since then, numbers have fluctuated but have likely been well below the numbers present in the 1980s and earlier. The model of population dynamics presented in Moyle et al. (2004) indicates high resiliency in splittail populations, suggesting they can recover from very low levels quickly. However, the continued decline in abundance shown in the pumping plants suggests that the portion of the population that exists in the southern part of the Delta may be very small (Figure 1), while the northern part of the population, reflected in the Suisun Marsh data, appears to have declined less severely in recent years (Figure 2). Seining surveys at boat ramps around the Delta and lower rivers, which capture mainly young-of-year fish, show high variability in numbers and no real trends; the catches seem to primarily reflect annual spawning success near the seining areas.

Trends for the genetically distinct population in the lower San Francisco Estuary are not known, but it is likely they are much smaller than the Delta populations.

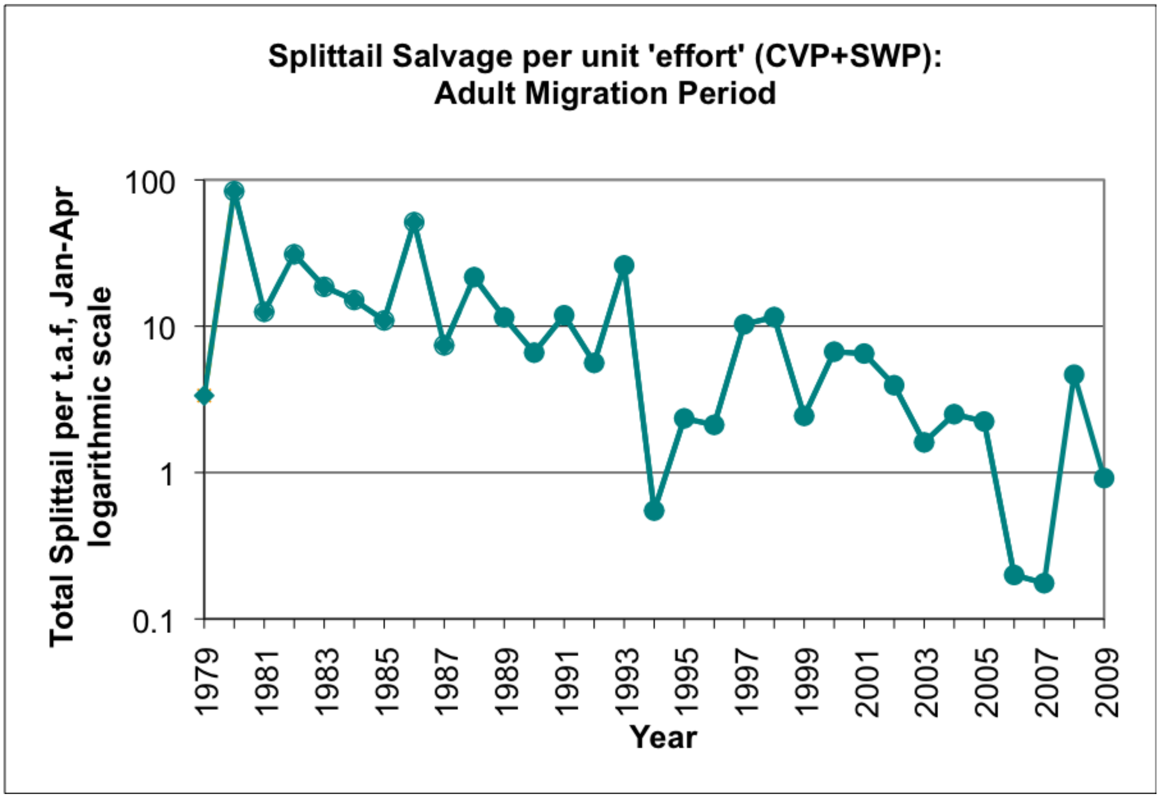


Figure 1. Number of splittail captured per thousand acre feet of water in the pumps of the South Delta, January-April, 1979-2009. Note the logarithmic scale on the Y vertical axis. Graph courtesy of David Wright, California Department of Fish and Wildlife.

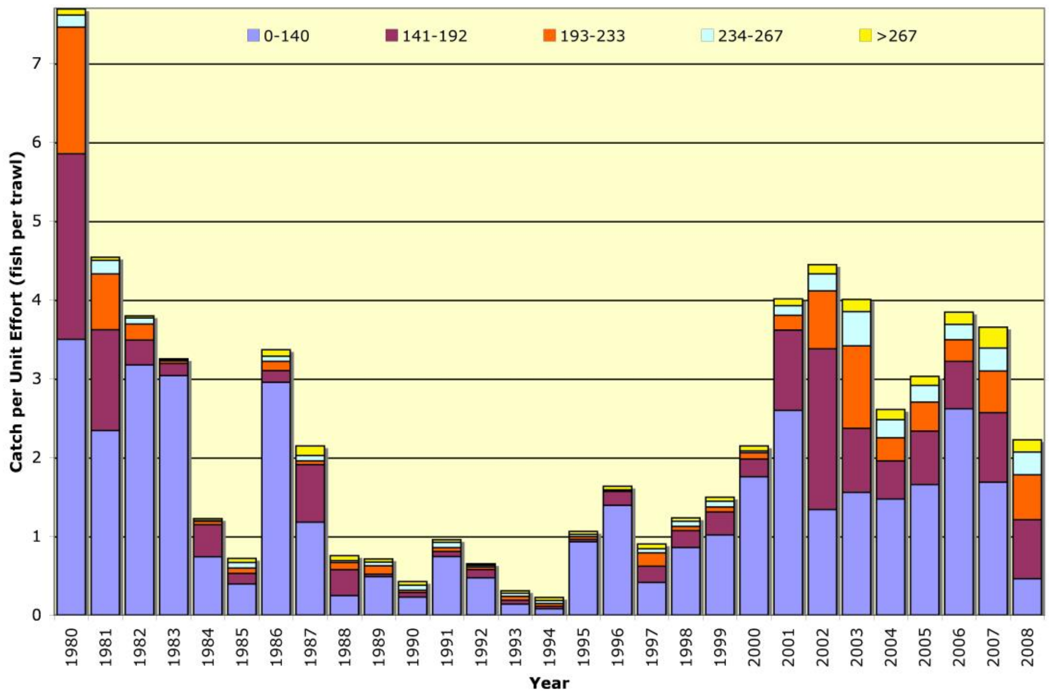


Figure 2. Annual catch per effort of splittail of different size classes (mm SL) in Suisun Marsh, 1980-2008. Each bar represents 150-200 trawls. The bottom-most section of each bar represents catches of young-of-year. Graph by Teejay Orear, UC Davis.

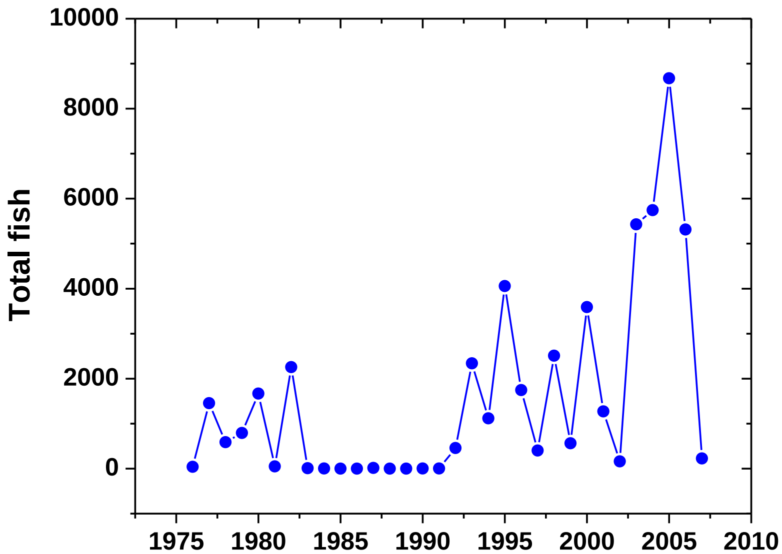


Figure 3. Splittail catch from USFWS Beach Seine Survey. Points are total annual catch over 20 stations consistently sampled each year from 1976-2007. Total catch is weighted by the number of months sampled each year, especially in the mid-80s to mid-90s. Values over the past decade are not influenced by the weighting (i.e., 11-12 months were sampled in each year). Graph by W. A Bennett.

Nature and Degree of Threats: The splittail is a resilient species (Sommer et al. 1997, Moyle et al. 2004) but it lives in a rapidly changing environment that is increasingly modified for human use (Table 1). Its present status may reflect short-term stability in an otherwise long-term decline. Between the period of massive influx of non-native peoples into California in the 1850s and the completion of Oroville Dam in 1962 (the final large dam constructed in the Central Valley of California), the range and total abundance of splittail declined. The splittail is now largely restricted to the Sacramento-San Joaquin Estuary, limited floodplain and riverine habitats upstream, and channels between Delta islands. Its populations are highly dependent on artificially maintained flows and floodplains, as well as on unusually wet years that create widespread flooding, such as occurred in 2011. Long-term persistence of splittail populations depends upon favorable estuary conditions, adequate spawning habitats, and access to those habitats.

Major dams. Splittail depend on outflows from the Sacramento and San Joaquin rivers and their major tributaries for successful life cycle completion. These rivers are all highly regulated by dams. Major dams have largely eliminated splittail habitat in the San Joaquin River above the mouth of the Merced River by shutting off flows completely and reducing suitable habitat downstream. Likewise, the Tulare Basin no longer supports splittail because dams and downstream diversions have dried large areas of former aquatic habitats. One of the few places where spawning can take place in most years is the re-created floodplain on the lower Cosumnes River; the river is unregulated so frequent flooding occurs (Moyle et al. 2007).

Spawning in the Sacramento River depends on releases from large rim dams (usually for flood control), high flow events from small tributaries (e.g., Cache and Putah creeks), and occasional passive spills from large dams. Strong year classes of splittail are created when the Yolo and Sutter bypasses and other floodplain areas are inundated for at least six weeks in late February through April, provided depths are adequate and temperatures are <20°C. These conditions need to occur, at a minimum, every 3-4 years to maintain large adult populations. Separation of floodplains from Central Valley rivers in the 19th and early 20th centuries must have been devastating to splittail populations, especially in the San Joaquin River. Fortunately, the creation of huge artificial floodways (Yolo and Sutter bypasses) in the 1920s and 1930s coincidentally created near-ideal conditions for splittail spawning and rearing and maintained their populations (Sommer et al. 2007). However, splittail numbers reached record lows in the estuary in 1994, following six years of drought, which greatly reduced the amount and frequency of flooding. Given the increasing human demand for water, and trends towards managing the Yolo and Sutter bypasses in ways that reduce or control flooding, it is uncertain whether these areas will continue to provide favorable splittail spawning habitat. The creation of a new floodplain area along the Cosumnes River (Moyle et al. 2007) and proposals to do so elsewhere are positive signs that additional spawning habitats may be created.

If freshwater outflows are further reduced in the future, the amount of brackish water habitat in the Estuary may decrease. Such habitat supports the principal rearing areas for splittail. The effect is likely to be especially strong in conjunction with predicted sea level rise.

	Rating	Explanation
Major dams	High	Splittail are dependent upon dam-regulated flows throughout their range
Agriculture	High	Agricultural return waters reduce habitat quality; channel modifications (e.g., levees) reduce habitat quantity and limit access to floodplains; entrainment occurs in major diversions
Grazing	n/a	Included as part of agriculture
Rural residential	Low	Residential areas on the edges of Petaluma, Napa, and Suisun marshes may create localized impacts (e.g., reclamation of tidal marshes, pollution input)
Urbanization	Medium	Numerous large metropolitan areas throughout range
Instream mining	n/a	
Mining	Low	Legacy effects of gold mining (e.g., mercury) may impact splittail; effects unknown
Transportation	Low	Roads and railroad may affect Suisun Marsh
Logging	n/a	
Fire	n/a	
Estuary alteration	High	San Francisco Estuary is highly modified and continues to change rapidly
Recreation	Low	Recreational boating, etc. may affect behavior and habitat utilization
Harvest	Low	Some harvest for bait; limited harvest of migrating adults for food
Hatcheries	n/a	
Alien species	High	Invasive species numerous and widespread across splittail range; alteration of food webs; new invasions are an ongoing potential threat

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

Agriculture. All habitats used by splittail are heavily influenced by agriculture. Suisun Marsh has been diked and drained to create a combination of cattle pasture and marsh for waterfowl habitat, as have the Napa and Petaluma marshes. River flows are regulated to provide water for agriculture. The bypasses and floodplains used by splittail for spawning are largely devoted to agriculture. Migrating fish are subject to entrainment in agricultural diversions, including the large pumps in the South Delta. Agricultural return waters generally increase temperatures and deliver pollutants; some rivers (e.g., the lower San Joaquin River) are consequently unsuitable for splittail and other native fishes. Some contaminants, such as selenium from San Joaquin Valley agricultural drainage, can accumulate in splittail and potentially interfere with reproduction and change behavior.

While Feyrer et al. (2007) showed that juvenile splittail are very flexible in their diet and habitat utilization characteristics, they also note that habitat quality varies widely across their range, as reflected in reduced growth rates in some areas. Currently, splittail are thriving in the highly altered Suisun and Petaluma marshes, although growth rates are somewhat reduced following food web alterations from the overbite clam invasion. In Suisun Marsh, periodic dissolved oxygen 'sags' occur from return water from duck clubs and other agricultural operations, which are laden with organic matter (R. E. Schroeter, unpublished data). While efforts are being made to reduce these impacts, poor water quality remains a stressor in Suisun Marsh and other portions of the Delta.

Urbanization. The San Francisco Estuary receives large amounts of urban runoff and both point and non-point sources of pollution inputs. The effects of pesticides and other toxic substances from urban runoff on splittail are not known, but there may be considerable potential for negative impacts (Sommer et al. 2007). Because splittail forage on the bottom, consuming large amounts of detritus, they may be particularly susceptible to contaminants that accumulate in substrates or in benthic organisms. Selenium, from urban and agricultural sources, is of particular concern because it accumulates in overbite clams, which splittail consume, and has been demonstrated to inhibit growth and reproduction in splittail and other fishes (Sommer et al. 2007). In addition, sewage outflows release micropollutants (hormones, etc.) that can affect splittail and other fishes. Thus, the proposed increase in tertiary treated sewage water flowing into the Suisun Marsh from Fairfield could affect splittail in a number of ways, both positive and negative. In addition, a largely undocumented effect of urbanization on splittail is the creation of large levees to protect urban areas from flooding, which have greatly diminished spawning and rearing habitat for splittail.

Mining. The mercury legacy from mining during the Gold Rush era has some potential to affect splittail reproduction and survival (e.g., Deng et al. 2008), but population-level effects are not known.

Transportation. Transportation corridors have contributed to alteration of splittail habitats, but the effects are minor compared to other habitat impacts. Transportation-related impacts may be particularly acute in Suisun Marsh, which is crossed by a railroad and restricted by highways. Ship channels may disrupt migration patterns or otherwise negatively affect splittail habitat utilization.

Estuary alteration. The upper San Francisco Estuary, a key portion of the splittail's range, is completely altered in structure and function; the Delta has become a collection of heavily farmed, subsided islands, protected by inadequate levees, which force water to flow through deep channels with little edge habitat (Lund et al. 2007, 2010). Freshwater is delivered into this changed system from altered rivers with controlled flow regimes. The San Joaquin River contributes minimal inflow that is highly polluted with agricultural drainage and urban wastewater. The Sacramento River contributes more water in the summer (and less in winter) than it did historically; this water is moved across the Delta to the giant pumps of the State Water Project and the federal Central Valley Project, dramatically changing the normal seaward gradients of flow, salinity, and other factors that fish evolved with and respond to. As Moyle and Bennett (2008) documented, the Delta has increasingly become, from a fish perspective, more like a freshwater lake than the upper part of an estuary. This lake-like habitat does not provide optimal conditions for splittail (Nobriga et al. 2005); however, splittail can still migrate through the Delta during times when flows are high and the Sacramento River outflow passes through floodplains. Likewise, net flow is usually downstream in spring when most juvenile splittail are out-migrating to rearing areas, although this does not prevent millions from being entrained in the CVP and SWP pumps in some years. In most years, splittail have only limited places to rear, mainly Suisun Marsh, Petaluma and Napa rivers, Suisun Bay, and, perhaps, the western Delta, with a limited number of suitable rearing areas elsewhere, such as the Sutter Bypass. In spite of these habitat limitations, splittail have sustained what appear to be large populations (Moyle et al. 2004), although there are no real estimates of numbers.

Continued changes to the Estuary may pose additional threats to splittail and other native fishes. Of note is the continuous increase in water diversion, both upstream of the Delta and from the Delta itself, although diversions from the south Delta have been reduced since 2007 to protect delta smelt and Chinook salmon. There are now immense pressures on the system to find ways to continue to remove more fresh water, including increasing upstream diversions, such as the recent push to develop bypass tunnels to divert water from the lower Sacramento River, transport it around the Delta, and deliver it directly into the SWP and CVP aqueducts. Substantial changes may occur in the Delta and Suisun Marsh as a consequence of predicted sea level rises and corresponding levee failures. In the Delta, subsided islands may fill with saline or fresh water, depending on location, creating large areas of open water (Lund et al. 2007, Moyle 2008). Breached islands will also increase the tidal prism, leading to more salt being drawn into the Delta. In Suisun Marsh, the patchwork of freshwater marsh maintained by duck hunting clubs and wildlife areas will likely be converted into tidal and subtidal habitat (Moyle et al. 2014). The outcomes and potential impacts of these systemic changes to splittail and other fishes in the Estuary remain uncertain; therefore, continued monitoring of populations and their habitats is imperative.

Harvest. Although splittail have been harvested as food and bait by recreational anglers, there is no evidence that this exploitation has contributed to their decline. However, an annual fishery concentrates on migrating spawners which could pose a threat if harvest of large females becomes excessive (Sommer et al. 2007). This potential impact was greatly alleviated by angling regulations implemented in 2010, which went from no daily bag limit for splittail to a daily limit of 2 splittail for inland waters.

Alien species. Introduced species are an ongoing threat to the San Francisco Estuary, especially those introduced from the ballast water of ships. The most recent introductions have been of several species of planktonic copepods, Brazilian waterweed (*Egeria densa*), and the overbite clam. The copepods are important food for larval and juvenile splittail and a shift in species, especially towards the tiny and apparently inedible *Limnoithona tetraspina*, is a cause for concern, although no effects have been detected. Changes in copepod abundance and composition are, at least in part, the result of the invasion by the overbite clam. This clam has become extremely abundant in Suisun Bay, from which it filters out much of the planktonic algae, the base of the food web that supports splittail (Feyrer et al. 2003). Increase in abundance of the clam led to collapse of the mysid shrimp population, a major food of splittail. The overbite clam has had much less of an impact on planktonic invertebrates in Suisun Marsh than it has elsewhere, perhaps accounting for high splittail abundance in this area. Adult splittail feed directly on the clams so, in this case, their presence may be beneficial to splittail, although selenium that accumulates in the clams may ultimately have a negative effect on splittail reproduction. However, any benefits from providing adult forage base may be offset by alterations to other components of the food web that support larvae and juveniles. Ultimately, alien species represent a threat to the entire estuary ecosystem because of their abundance and unpredictable effects.

Effects of Climate Change: Climate change will result in two major factors that will likely affect splittail and other native fishes: increased variability in flooding, and rise in sea level. An example of potential effects of increased variability in flooding, along with drought, can be seen in the period from 1980-1995, a period with some of the most extreme conditions the estuary has experienced since the arrival of Europeans. Within this 15-year period, an extended drought persisted from 1986-1992. Ironically, there were exceptionally high outflows at the beginning of the drought because of severe rain in February, 1986. The prolonged drought had two major interacting effects: a natural decrease in freshwater outflow to the Estuary, and an increase in the proportion of inflowing water being diverted from the Estuary. A natural decline in splittail numbers would be expected from greatly reduced outflow over an extended time because of the reduced availability of spawning and larval rearing habitat. The strong year class produced from the 1986 flooding was presumably an important factor in allowing splittail to persist through six years of drought, although some young-of-year were produced every year (Figure 2). It is important to recognize that extreme floods and droughts have occurred in the past, yet splittail populations have endured. However, climate change is predicted to greatly increase flow variability, with bigger floods (probably good for splittail if the timing is right) but longer droughts, with more extended periods of low flows. These extreme conditions will be made more severe by increased diversion of water both upstream and in the Delta. The net effects are likely to be more extreme fluctuations in splittail populations, with the potential to reach critically low numbers during extended drought.

As sea level rises, large parts of Suisun and Petaluma marshes will become inundated and saltier, unless action is taken to reduce the effects (Moyle et al. in press). As these key rearing areas change, food organism and predator populations will change in response. It is possible that flooded islands in the Delta or new tidal marsh habitat (e.g., Cache Slough region) will replace habitat lost elsewhere. Overall, rise in sea level has the potential to dramatically change splittail habitat; however, the effects of these changes cannot be predicted and may be negative, neutral, positive, or some combination thereof. Moyle et al. (2013) rated the splittail “highly vulnerable” to extinction in the next 100 years as the result of the added impacts of climate change to the estuary and its spawning habitats.

Status Determination Score = 3.1 - Moderate Concern (see Methods section; Table 2). Splittail have highly variable populations but do not appear to be threatened with extinction in the immediate future. Because the San Francisco Estuary and Central Valley rivers may become dramatically altered as a result of climate change and associated sea level rise, increases in flow variability, and more frequent and extended drought, splittail habitats cannot be regarded as secure (Sommer et al. 2007). The splittail was listed as a federally threatened species in 1999 by the USFWS, but was delisted in 2003 as the result of new information on its biology and status (Moyle et al. 2004, Sommer et al. 2007). The American Fisheries Society lists splittail as "Vulnerable," while NatureServe lists it as "Imperiled" (G2) (Jelks et al. 2008). Not recognized in these status evaluations is recent information indicating that the splittail consists of two Distinct Population Segments, one in the lower estuary (San Pablo DPS) and one centered in the Delta (Delta DPS). The San Pablo DPS is apparently much smaller than the Delta DPS and may, consequently, be in greater likelihood of severe decline or extinction.

Metric	Score	Justification
Area occupied	2	Delta, portions of Sacramento/San Joaquin rivers, Suisun Bay and Marsh, and Petaluma/Napa marshes and lower river watersheds
Estimated adult abundance	5	Large in Delta DPS, unknown in San Pablo DPS
Intervention dependence	4	All habitats heavily managed for things other than splittail production
Tolerance	5	One of the physiologically most tolerant native fish species
Genetic risk	3	Two distinct populations; San Pablo DPS may be at more risk of extinction, lowering diversity
Climate change	2	Highly vulnerable to droughts and sea level rise
Anthropogenic threats	1	See Table 1
Average	3.1	22/7
Certainty (1-4)	4	Well studied in many areas

Table 2. Metrics for determining the status of Sacramento splittail, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: Sacramento splittail are a relatively easy species to manage due to their high physiological tolerances, high fecundity, relatively long life span, and because of our good understanding of splittail life history requirements. Conversely, their restricted distribution and migration between two very different habitat types, both of which are already highly modified and likely to further change in response to climate change and sea level rise, may create significant management challenges. Management recommendations include:

1. Continue to monitor splittail abundance and age class structure on an annual basis in Suisun Marsh, in order to document changes in year class success.

2. Institute a monitoring program for the Petaluma and Napa river populations to determine population dynamics, habitat requirements, and migratory patterns, especially under various climate change models (see #5 below). The ecology and phylogenetic relationships of splittail in San Francisco Bay also need to be determined.

3. Create more suitable floodplain habitats that will support splittail spawning and rearing on a regular basis (at least every 2-3 years). Recent studies suggest that even small amounts of regularly flooded habitat can be used successfully by splittail to supplement populations in dry years (Feyrer et al. 2006, Moyle et al. 2007, Sommer et al. 2002, 2008).

4. Actively manage floodplain areas of known importance for splittail spawning (e.g., Yolo and Sutter bypasses, Cosumnes River floodplain) to maximize splittail spawning success. This could be done by annual flooding of relatively small areas.

5. Perform modeling and other studies to determine predicted changes to, and corresponding suitability of, existing estuarine splittail habitats under various sea level rise and climate change scenarios. Among questions to address is how varying levels of inundation will affect portions of Suisun Marsh and the Delta. Modeling could also focus on splittail 'salvage' in the South Delta pumps and how existing and/or modified operations may affect populations.

6. Restore habitats for splittail in the San Joaquin River and associated parts of the Delta so a large population, independent of the Sacramento River-Suisun Marsh population, can be reestablished.

7. Continue to assess the sport fishery for splittail during their spawning migration on the Sacramento River to determine impacts (if any) on populations. Similar monitoring should also occur on the Petaluma and Napa river populations.

Figure 4. Distribution of Sacramento splittail, *Pogonichthys macrolepidotus* (Ayres), in California. Historic distribution included most of the Central Valley, including the San Joaquin River, lower portions of its tributaries, and Tulare Lake (now dry).

HARDHEAD
***Mylopharodon conocephalus* (Baird and Girard)**

Status: Moderate Concern. Hardhead are still widespread in California but populations are likely declining and most are small and isolated, with exceptional vulnerability to climate change.

Description: Hardhead are large cyprinids, reaching lengths in excess of 60 cm SL. The body shape is similar to that of Sacramento pikeminnow (*Ptychocheilus grandis*), with which they co-occur, but the body is deeper and heavier and the head is less pointed. Hardhead also differ from pikeminnow in that their maxilla does not extend beyond the anterior margin of the eye and they possess a bridge of skin (frenum) connecting the premaxilla to the head. Hardhead have 8 dorsal rays, 8-9 anal rays, and 69-81 lateral line scales. Adults have large molariform pharyngeal teeth but juvenile teeth are hook-like. Juveniles are silver; adults are brown-bronze dorsally. During spawning, adult males develop small white nuptial tubercles on the head and along a band that extends from the head to the base of the caudal fin (Moyle 2002). Prolarvae and early postlarvae have scattered caudal pigmentation and two distinct dark spots, one above the flexion and one at the ventral base of the caudal peduncle (Wang and Reyes 2007). Some midventral pigmentation may also occur. Early juveniles have small mouths (maxilla ends in front of eye), high myomere counts (46-50) and enlarged nostril flaps.

Taxonomic Relationships: Hardhead were first described as *Gila conocephala* by Baird and Girard (Girard 1854) from one specimen collected from the San Joaquin River. Ayres (1854, 1855) later redescribed the species as *Mylopharodon robustus*. Girard (1856a) recognized the generic designation, reclassifying *G. conocephala* as *Mylopharodon conocephalus* and *M. robustus* as a closely allied, but separate, species. Jordan (1879), however, considered the genus monotypic and united both forms as *Mylopharodon conocephalus* (Jordan and Gilbert 1882). Electrophoretic studies by Avise and Ayala (1976) and morphometric analysis by Mayden et al. (1991) indicated that hardhead, although related to Sacramento pikeminnow, are sufficiently different from the pikeminnow to be retained in a separate genus.

Life History: Stream-dwelling juvenile (<150 mm SL) hardhead are often found in small aggregations in pools and runs during the day, actively feeding at the water's surface, holding in moving water to feed on drifting material, or browsing from the benthos (Alley 1977). Adults tend to school in the deepest part of pools, cruising about slowly during the day. They are most active when feeding, in early morning and evening (Moyle 2002). In small streams, they seldom move more than one kilometer away from home pools, except when spawning; the average summer home range of hardhead in a Sierra Nevada foothill stream was measured as 289 m (Grant and Maslin 1999). In Britton Reservoir (Shasta County), large hardhead will remain motionless near the water's surface (< 1 m depth) during warm summer days (Vondracek et al. 1988), making them readily accessible as prey to bald eagles, osprey and other fish eating birds (Hunt et al. 1988).

Hardhead are primarily bottom feeders that forage on invertebrates and aquatic plant material from stream substrates but they will also consume drifting insects and algae from the water column (Alley 1977). Occasionally, they will feed on plankton and surface insects and, in Shasta Reservoir (Shasta County), they were known to feed on cladocerans (Wales 1946). Smaller fish (<20 cm SL) feed primarily on benthic invertebrates, especially mayfly larvae, caddisfly larvae, and small snails (Reeves 1964). Larger fish feed on filamentous algae, as well as on crayfish and other large invertebrates (Moyle, unpubl. data). Ontogenetic changes in tooth structure reflect this shift in diet; juveniles have hooked teeth for capturing insects, while adults have molariform teeth that facilitate grinding of plants and large prey (Moyle 2002). Reeves (1964) did not find fish remains in the stomachs of large hardhead.

Hardhead reach 7-8 cm SL by their first year, 10-12 cm by the end of their second year and 16-17 cm by the end of their third year (Reeves 1964, Moyle et al. 1983, PG&E 1985, Grant 1992). They can reach 30 cm SL by age four (Reeves 1964) in the American River but only reach this size at age 5 or 6 in the Pit and Feather rivers (Moyle et al. 1983, PG&E 1985). Large (44-46 cm SL) hardhead from the Feather River were found to be 9-10 years old, but older and larger fish probably exist in the Sacramento River. In smaller streams, hardhead rarely grow beyond 28 cm SL (Grant 1992). Historic records suggest that hardhead reach up to 1 m TL (Jordan and Evermann 1896).

Hardhead mature following their second year and spawn in the spring, mainly in April and May (Reeves 1964, Grant and Maslin 1999), judging by the upstream migrations of adults into smaller tributary streams during this time of the year (Wales 1946, Murphy 1947, Bell and Kimsey 1955, Rowley 1955). Shapovalov (1932) reported the presence of mature eggs in females during March, but gonads of males and females caught in July and August were spent (Reeves 1964). Estimates based on juvenile recruitment suggest that hardhead spawn by April-June in Central Valley streams, although the spawning season may occasionally extend into August in the foothill streams of the Sacramento-San Joaquin drainage (Wang 1986). Spawning adults from larger rivers and reservoirs may migrate more than 75 km in April and May to spawn in tributary streams (Wales 1946, Moyle et al. 1995). In contrast, hardhead in small streams only migrate a short distance upstream or downstream of their home pool for spawning (Grant and Maslin 1999). In Pine Creek (Tehama County), spawning adults aggregate in nearby pools and return to home pools after spawning (Grant and Maslin 1999). Hardhead spawning has not been directly observed; however, it is likely similar to that of hitch and pikeminnow, which deposit their fertilized eggs in sand or gravel in riffles, runs, or heads of pools (Wang 1986; Moyle 2002). Spawning success of hardhead in the lower Tuolumne River is highest when there are higher flows during in April and May (Brown and Ford 2002).

Females are highly fecund, producing over 20,000 eggs (Burns 1966). Fecundity ranged from 7,100 to 23,900 eggs in females from Pine Creek (Tehama County) and the American River (Reeves 1964, Grant and Maslin 1999). The ovaries contain both developed and undeveloped eggs, suggesting that eggs mature after a full year (Grant and Maslin 1999). Fertilized eggs presumably develop in the interstices of the gravel until hatching. Larvae and postlarvae most likely move into stream margins with abundant cover (Wang 1986). They move into deeper habitats as they grow larger. Young from intermittent streams are swept downstream into areas of low velocity near the mouths of main rivers (Moyle 2002). In Deer Creek (Tehama County), small juveniles (2-5 cm SL) congregate in large schools in shallow backwaters. Small juveniles in the Kern River congregate among large substrates (cobble and boulders) along the stream margin (L. Brown, USGS, pers. comm. 1999).

Hardhead host a variety of parasites. Hardhead from the North Fork Feather River (Plumas and Butte counties) were infected with an average of three parasite species, including nematodes and trematodes (Alvarez 2008).

Habitat Requirements: Hardhead are often found at low to mid-elevations in relatively undisturbed habitats of larger streams (Moyle and Daniels 1982, Mayden et al. 1991) with high water quality (clear, cool). In the Sacramento River, however, they are common in both the mainstem and tributaries up to 1500 m in elevation (Reeves 1964). Summer temperatures in rivers where they are common reach 20°C, below optimal temperatures (24-28°) determined by laboratory experiments (Knight 1985). In a thermal plume in the Pit River, hardhead preferred the warmest temperatures available (17-21°C; Baltz et al. 1987). Similarly, hardhead acclimated to 12, 15, and 18 °C water temperatures, preferred water temperatures of 19.6 to 20 °C, and avoided water temperatures less than 17 °C in a laboratory setting (Cocherell et al. 2007). However, somewhat lower temperatures appear to increase swimming performance. Hardhead swimming performance was higher at 15 °C than at 10 or 20 °C (Myrick and Cech, Jr. 2000). Their distribution may be limited to well-oxygenated streams and reservoir surface waters by low oxygen levels at warm temperatures (Cech et al. 1990). They prefer pools and runs with deep (>80 cm), clear water, slow (20-40 cm/sec) velocities and sand-gravel-boulder substrates (Alley 1977, Cooper 1983, Knight 1985, Moyle and Baltz 1985, Mayden et al. 1991). May and Brown (2002) described summer water quality and habitat variables associated with a foothill group of mostly native fishes, including hardhead (Table 1).

pH	7.9
Specific conductivity (µS/cm)	144
Dissolved oxygen (mg/l)	8.8
Discharge (m/s)	3.2
Water temperature (°C)	19.7
Mean depth (m)	0.89
Mean velocity (m/s)	0.38
Mean dominant substrate size (mm)	2-64 (gravel)
Mean width (m)	18.9
Canopy cover (%)	25
Stream gradient (%)	0.71
Stream sinuosity	2.2
Elevation (m)	106
Agricultural + urban land (%)	2
Basin area (km^2)	519

Table 1. Mean summer water quality and habitat variables for a foothill fish assemblage, including hardhead (source: May and Brown 2002).

Adults mostly occupy the lower half of the water column in streams (Knight 1985, Moyle and Baltz 1985) but may stay close to the surface in reservoirs (Hunt et al. 1988, Vondracek et al. 1988). They are often sympatric with Sacramento pikeminnow and Sacramento sucker. Hardhead are usually absent from streams occupied by alien species, especially centrarchids

(Moyle and Daniels 1982, Mayden et al. 1991, Moyle et al. 2002) and streams that have been heavily altered (Baltz and Moyle 1993). Because they are poor swimmers, hardhead may also be absent from stream reaches above barriers, even if ladders are in place to allow salmonid passage (Myrick 1996, Myrick and Cech, Jr. 2000).

Hardhead populations are well established in mid-elevation reservoirs used exclusively for hydroelectric power generation, such as Redinger and Kerkhoff reservoirs on the San Joaquin River (Fresno County), and Britton Reservoir on the Pit River. In the Pit River, hardhead are most abundant in the upper portion of Britton Reservoir where habitat is more riverine; they are less abundant in the lacustrine habitat of the lower reservoir, where alien centrarchids, particularly predatory basses, are more abundant (PG&E 1985, Vondracek et al. 1988).

Distribution: Hardhead are widely distributed in streams at low to mid-elevations in the Sacramento-San Joaquin and Russian River drainages (Leidy 1984, Moyle 2002). Their range extends from the Pit River (south of the Goose Lake drainage), Modoc County, in the north to the Kern River, Kern County, in the south (Moyle and Daniels 1982, Cooper 1983). In the San Joaquin drainage, scattered hardhead populations are found in tributary streams, but only rarely in the valley reaches of the San Joaquin River (Moyle and Nichols 1973, Saiki 1984, Brown and Moyle 1987). Jones and Stokes (1987) found a very small number of hardhead during an extensive sampling program of the lower Kings and San Joaquin rivers, indicating that hardhead have opportunities to recolonize historic habitats but fail to do so, due to dewatering and other factors. They are absent from the Cosumnes River. In the Sacramento River drainage, hardhead are found in most large tributaries, as well as in the Sacramento River itself (Moyle 2002). In the South Fork Yuba River, they make up 55% of the fish caught in the lower 15 km (Gard 2002). They are present in the Russian and Napa rivers, although the Napa River population is very restricted in its distribution (R. Leidy, USEPA, pers. comm.). They are widely, if spottily, distributed in the Pit River drainage (Cooper 1983, Moyle and Daniels 1982), including the main stem Pit River and its series of hydroelectric reservoirs. Although their current status is uncertain, hardhead apparently also once occurred in Alameda and Coyote creeks, tributaries to the San Francisco Bay (Leidy 2007). They are present in the northern Coast Ranges, in the larger tributaries to the Sacramento River, such as Cache Creek, Putah Creek and Clear Creek, mainly in canyon reaches with deep pools.

Trends in Abundance: Historically, hardhead were regarded as widespread and locally abundant (Ayres 1854,1855, Jordan and Evermann 1896, Evermann 1905, Rutter 1908, Follett 1937, Murphy 1947, Soule 1951, Reeves 1964). Hardhead are still fairly widespread in foothill streams (May and Brown 2002) but their specialized habitat requirements, combined with widespread alteration of downstream habitats, has resulted in most populations being isolated from one another (Moyle 2002), making them vulnerable to localized extinctions. Consequently, hardhead are much less abundant than they were historically, especially in the southern half of their range (Moyle 2002). Historical records noted their presence in most foothill streams in the San Joaquin drainage (Reeves 1964), but Moyle and Nichols (1973) found them in only 9% of the streams sampled. Brown and Moyle (1987, 1993) subsequently resampled most of the same sites and found that a number of populations had disappeared during this 15-year period. Ford and Brown (2001) found they were uncommon in the lower Tuolumne River and largely confined to a cool-water reach about 30 km long, associating mainly with other native fishes. In the Cosumnes River, hardhead are absent despite a fairly natural flow regime, apparently because of the invasion of redeye bass (*Micropterus coosae*). They are still common in the mainstem Sacramento River, lower American and Feather rivers, some smaller streams (e.g., Deer, Pine, Clear creeks), and reaches upstream of foothill reservoirs (Moyle 2002). They are very rare in the Napa River (Leidy 1984 and pers. comm.) and uncommon in the Russian River (Moyle 2002). In the Pit River, they have a discontinuous distribution and are limited to canyon reaches and hydroelectric reservoirs (Moyle and Daniels 1982, Herbold and Moyle 1986).

Hardhead were once abundant enough in reservoirs to be regarded as a problem species, under the assumption they competed for food with game fishes such as trout (Moyle 2002). Most populations likely resulted from colonization by juveniles before introduced predators became abundant and largely extirpated hardhead from reservoirs. Populations declined dramatically within two years in Shasta Reservoir (Reeves 1964), leaving only a small number to persist (J. M. Hayes, CDFW, pers. comm.). Crashes of large populations in reservoirs were also reported from: Pardee Reservoir on the Mokelumne River, Amador/Calaveras County (Kimsey et al. 1956); Millerton Reservoir on the San Joaquin River, Fresno County (Bell and Kimsey 1955); Berryessa Reservoir, Napa County (Moyle 1976); Don Pedro Reservoir, Tuolumne County; and Folsom Reservoir, El Dorado County (Kimsey et al. 1956). Currently, they are largely absent from reservoirs that undergo strong annual variations in water level, although they can survive in hydroelectric reservoirs where water level fluctuations are less, such as Britton Reservoir on the Pit River and Redinger Reservoir on the San Joaquin River (Moyle 2002).

Nature and Degree of Threats: The apparent ongoing declines in hardhead distribution and abundance are a result of synergistic impacts from habitat loss, decline in water quality, and invasions of alien species (Moyle 2002, May and Brown 2002, Brown and Moyle 2005). The principal threats to hardhead include: (1) dams and diversions, (2) agriculture, (3) urbanization, (4) instream mining, (5) stream modification for transportation, (6) fisheries management (‘harvest’ associated with past eradication of ‘rough fishes’ to benefit recreational fisheries), and (7) alien species.

Dams and diversions. The large dams built on most California rivers have three principal effects: they greatly reduce flows and alter flow regimes downstream of dams; they alter water quality, usually making the downstream reaches warmer (but sometimes colder) with less dilution of pollutants; and they fragment watersheds, isolating fish populations. Dams also create conditions that favor alien fish species, especially in reservoirs. Generally, when flow regimes are altered so that elevated spring flows are uncommon, hardhead and other native fishes disappear from rivers (Brown and Moyle 2005). Pulsed flows also make hardhead juveniles susceptible to displacement and stranding (Chun et al. 2005). In addition, hardhead (71-91 mm SL) seem to exceptionally susceptible to entrainment by hydroelectric powerhouse turbines (ENTRIX Inc. 2001). Hardhead will persist in reaches below dams where habitat conditions are complex and high spring flows allow successful spawning. Where summer flows are low from dam releases or diversions, hardhead are absent from warm water reaches, which are often dominated by alien fishes (e.g., lower Tuolumne River, Brown and Ford 2002). Reservoirs associated with most dams harbor alien predator species that limit hardhead populations (see *Alien species* subsection below).

Agriculture. Hardhead are now largely absent from waters directly influenced by agricultural practices, such as streams polluted with irrigation return water and other waste water, or those bound by levees to reduce flooding, affected by silt-laden run-off, or with reduced flows due to irrigation diversions. Historically, hardhead were abundant throughout the Sacramento and San Joaquin watersheds, as indicated by their common presence in middens of native peoples at low-elevation sites (Gobalet 1989, Broughton 1994). While they disappeared from most of these waters before being documented, their absence is presumably the result of impaired water quality (high temperatures, low dissolved oxygen, high turbidity, high pollutant levels); these same conditions often favor alien species, further contributing to hardhead declines. Their persistence in the lower Sacramento River is presumably the result of increased summer flows to deliver water for agricultural and urban uses, which moderates water temperatures and dilutes pollutants to survivable levels.

Urbanization. The effects of urbanization include severe alteration to habitats, diversion of water and influx of pollutants. Hardhead are generally absent from urban streams; other native fishes are also scarce in such environments (Brown and Moyle 1993). Urban development near Alameda Creek was associated with declines of hardhead in this stream (Leidy 2007).

Instream mining. Ford and Brown (2001) noted that the downstream limits of hardhead in the lower Tuolumne River coincided with the presence of pits left over from gravel mining, which were 'captured' by the river. Such pits are common in the San Joaquin River basin and elsewhere. Mining pits create warm lake-like habitats that support centrarchid basses (*Micropterus* spp.) and the combination of poor habitat quality and presence of alien species appears to be lethal to hardhead. Similarly, there are legacy effects of placer, dredge and hydraulic mining for gold, which dramatically altered many hardhead streams, although hardhead populations have recovered somewhat as these streams have recovered (e.g., South Fork Yuba River). However, hydraulic mining has had lasting legacy effects that compromised much of the suitable fish habitat in streams such as the mainstem Yuba River, changing habitats from shaded pool-riffles to long, unshaded runs (True 2004). High seasonal sediment loads are also a legacy of past mining. Increased turbidities in the South Fork Yuba River decreased hardhead growth rates and increased physiological stress (Gard 2002). A poorly understood legacy effect of Gold Rush-era mining is the influx of mercury into many streams. Mercury has concentrated in the tissues of hardhead from Cache Creek (OEHHA 2005). Mercury can be toxic in high concentrations via disruption of the central nervous system. Hardhead appear to be relatively intolerant of pollutants, but whether or not mercury has affected their populations is not known.

Transportation. The best habitats for hardhead are at intermediate elevations in the larger streams of the Sierra Nevada foothills and Coast Ranges. These are also areas with extensive networks of highways and railroads, which often follow river courses. These transportation corridors can lead to partially channelized streambeds with fewer pools, coupled with increased siltation and pollution from road and railroad beds. Moyle and Randall (1998) found that native fishes, including hardhead, had a negative association with stream reaches that had high road densities.

	Rating	Explanation
Major dams	High	Many of the streams and rivers occupied by hardhead have altered flow regimes; dams isolate populations
Agriculture	High	Hardhead are largely absent from streams heavily influenced by agriculture
Grazing	Low	The impact of grazing on hardhead is likely minimal, although grazing may cause increased siltation or other habitat degradation in some streams
Urbanization	Medium	Hardhead populations decline and disappear where development alters their habitats
Instream mining	Medium	Instream mining has altered many of the stream reaches within hardhead range
Mining	Low	Most of the streams within hardhead range had hard rock mines adjacent to them which feed acidic, heavy metal pollutants into streams; direct effects on hardhead are unknown
Transportation	Medium	Proximity of roads and railroads to streams can lead to increased pollution, sedimentation and impaired habitats
Logging	Low	Increased sedimentation and stream temperatures resulting from logging practices may affect hardhead in some areas; greater impacts in the past
Fire	Low	Fire is a natural process within their range; impacts on hardhead are unknown, although probably low because hardhead occur mainly in larger rivers
Estuary alteration	n/a	
Recreation	Low	Stream-based recreation occurs throughout much of their range but impacts on hardhead are unknown
Harvest	Low	Past 'harvest' from fish eradication projects may have affected some populations
Hatcheries	n/a	
Alien species	High	Predation by alien centrarchids has been a major factor contributing to the decline of hardhead throughout its range

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

Fisheries management (harvest). From the 1950s-1980s, hardhead were one of the focal species for fish eradication programs by the CDFW, on the assumption they were competitors with trout and recreational fisheries would be improved through the elimination of so-called 'rough fish' (Moyle et al. 1982). Although these activities may have negatively affected hardhead (and other native fishes) at a localized level, in the one well-studied fish eradication program, hardhead showed considerable powers of recovery (Moyle et al. 1982). Otherwise, hardhead are rarely harvested for any purpose, although they are incidentally caught on occasion by anglers.

Alien species. It is likely that hardhead would have a much broader distribution in the absence of alien predatory fishes, especially centrarchid basses. In general, where bass are common, hardhead are absent or rare (Brown and Moyle 2005). Hardhead have largely disappeared from the upper Kings River, where smallmouth bass (*Micropterus dolomieui*) are now abundant (Brown and Moyle 1993). Gard (2004) observed that the lowermost reaches of the South Fork Yuba River (above Englebright Reservoir) had been colonized by smallmouth bass up to a waterfall. Hardhead were present both above and below the waterfall, but the hardhead below the waterfall were mostly large adults and small juveniles. The juveniles disappeared by the end of the summer, suggesting elimination by bass predation because they remained present in upstream areas. Presumably, the larger hardhead had moved down from upstream and were large enough to avoid predation. A viable hardhead population in the South Fork Yuba River is, as a consequence, confined to a relatively short stretch of river above the waterfall. More dramatically, hardhead (and other native fishes) are absent from long reaches of the Cosumnes River where they should be present, based on habitat characteristics (natural flow regime, deep pools, clear, cool water). This habitat is now occupied almost exclusively by redeye bass (*M. coosae*). Hardhead in reservoirs only persist if alien predators, especially centrarchid basses, are not abundant. Hardhead are abundant today only in those reservoirs that undergo short-term water level fluctuations (such as for power-generating flows), which impede alien species reproduction (Moyle 2002).

Effects of Climate Change: Predicted climate change impacts to hardhead habitats in California will vary greatly, given their wide distribution. In general, water temperatures are expected to increase, seasonal peak flow is expected to shift from late spring to late winter months, base flows in late summer and fall are expected to decrease (Knox and Scheuring 1991, Field et al. 1999, CDWR 2006) and the overall flow regime of streams will be altered by more frequent and extreme droughts and floods. Summer water temperatures for inland streams are predicted to increase, on average, by approximately 1-4°C by 2099, based on conversion factors developed by Eaton and Scheller (1996). Although hardhead can withstand higher temperatures than trout (Myrick and Cech 2000), exposure to higher water temperatures may increase the potential for bacterial infection. Hardhead collected from the Yuba River in 2003 were infected with bacteria (*Pseudomonad spp.*) at levels (29% of collected fish) that posed a health risk to this population (True 2004). Bacterial infections can lead to kidney disease and higher stream temperatures may reduce individual fitness by increasing physiological maintenance costs (Moyle and Cech 2004).

Elevated air temperatures associated with climate change will change the periodicity and magnitude of peak and base flows in streams due to a reduction in snow pack levels and seasonal retention. Streams may be especially impacted at lower elevations (<1000 m) in the central Sierra Nevada (Hayhoe et al. 2004), due to the already arid nature of this region, coupled with increasing urban, suburban and rural development and corresponding human water demands. Because of these combined factors, Moyle et al. (2013) rated hardhead as "critically vulnerable" to extinction from climate change.

Status Determination Score = 3.1 - Moderate Concern (see Methods section Table 2)**.** Hardhead should continue to be considered a Species of Special Concern (Table 3). NatureServe lists hardhead as Vulnerable at both the global (G3) and state level (S3).

Metric	Score	Justification
Area occupied	5	Still widely distributed in the Sacramento-San Joaquin watershed and the Russian River
Estimated adult abundance	4	Not known but large populations apparently exist
Intervention dependence	4	Monitoring is needed to establish current status but known stressors (e.g. unnatural flow regimes, alien species predation) should be mitigated
Tolerance	2	Hardhead are sensitive to habitat alterations associated with flow, turbidity and temperature
Genetic risk	4	Many populations are small and isolated
Climate change	1	Hardhead are confined to waters exceptionally vulnerable to climate change
Anthropogenic threats	2	See Table 2
Average	2.9	22/7
Certainty (1-4)	3	Information on current status (e.g. population index, abundance estimates) is largely absent

Table 3. Metrics for determining the status of hardhead in California, where 1 is a major negative factor contributing to status, 5 is factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: In general, hardhead are widely distributed but their recent downward population trend is similar to that of other California native fishes and is cause for concern (Moyle 2002). Hardhead seem especially susceptible to the combination of alien species and habitat change, especially impacts predicted by climate change models. Consequently, hardhead populations should be monitored and, where possible, cool water habitats in key portions of their range should be protected in order to prevent further declines. Management recommendations include the following:

Establish special management areas. Establishment of a number of protected areas in mid-elevation streams with natural flow regimes and high water quality will offer the best protection for hardhead in the long-term (Moyle and Yoshiyama 1992, Baltz and Moyle 1993). If managed properly, these areas would likely protect the entire native aquatic faunal assemblage and, given that they are a good indicator species for relatively undisturbed habitats, regular hardhead population monitoring should be implemented in these refuge areas (Moyle 2002).

Monitor populations. Streams known to support hardhead populations should be surveyed on a regular basis (e.g. every 5 years) in order to develop trend information. Special focus should be placed on monitoring the Napa, Russian and San Joaquin rivers, as well as Alameda Creek, from which hardhead populations appear to be disappearing rapidly and are highly isolated from other populations. Populations in the Napa River and Alameda Creek, in particular, are likely restricted to a few miles of suitable habitat; their populations should be a priority for protection because they represent remnant populations (Leidy 2007). Some watershed groups and other governmental and non-governmental organizations have begun monitoring native fishes, including hardhead. The hardhead monitoring plan in the South Fork American River seeks to establish baseline data on distribution and population structure (length, weight data) and establish 5 year interval monitoring (El Dorado Irrigation District 2007). Likewise, the Battle Creek Working Group has a plan that includes restoration within the range of hardhead, as well as the establishment of a population index (Ward and Kier 1999).

Re-establish populations where possible. The San Joaquin River is being restored and it is likely that hardhead will only reestablish themselves in this part of their native range through active reintroduction (Moyle 2008). Other areas within historic hardhead distribution where they are currently absent should, likewise, be evaluated for habitat suitability and potential reintroductions.

Manage flow regimes to favor native fishes. Major stressors to hardhead populations include changes to natural flow regimes (e.g. pulsed flows and water diversion) and predation from alien predators. These stressors can be mitigated through improving flow regimes below dams to favor native fishes. For example, in regulated rivers, flows should be managed to provide high spring flows in order to improve native fish reproductive success (May and Brown 2002, Moyle 2002).

Improve passage flows. Fish passage structures within hardhead range should be able to pass adult hardhead on spawning migrations and modified accordingly, where necessary. Velocities should not exceed 0.4 m/s, a velocity lower than is currently used as a guideline based on salmonid passage needs (Myrick and Cech, Jr. 2000). Hardhead are relatively poor swimmers and likely cannot navigate approaches managed for same-sized salmonids.

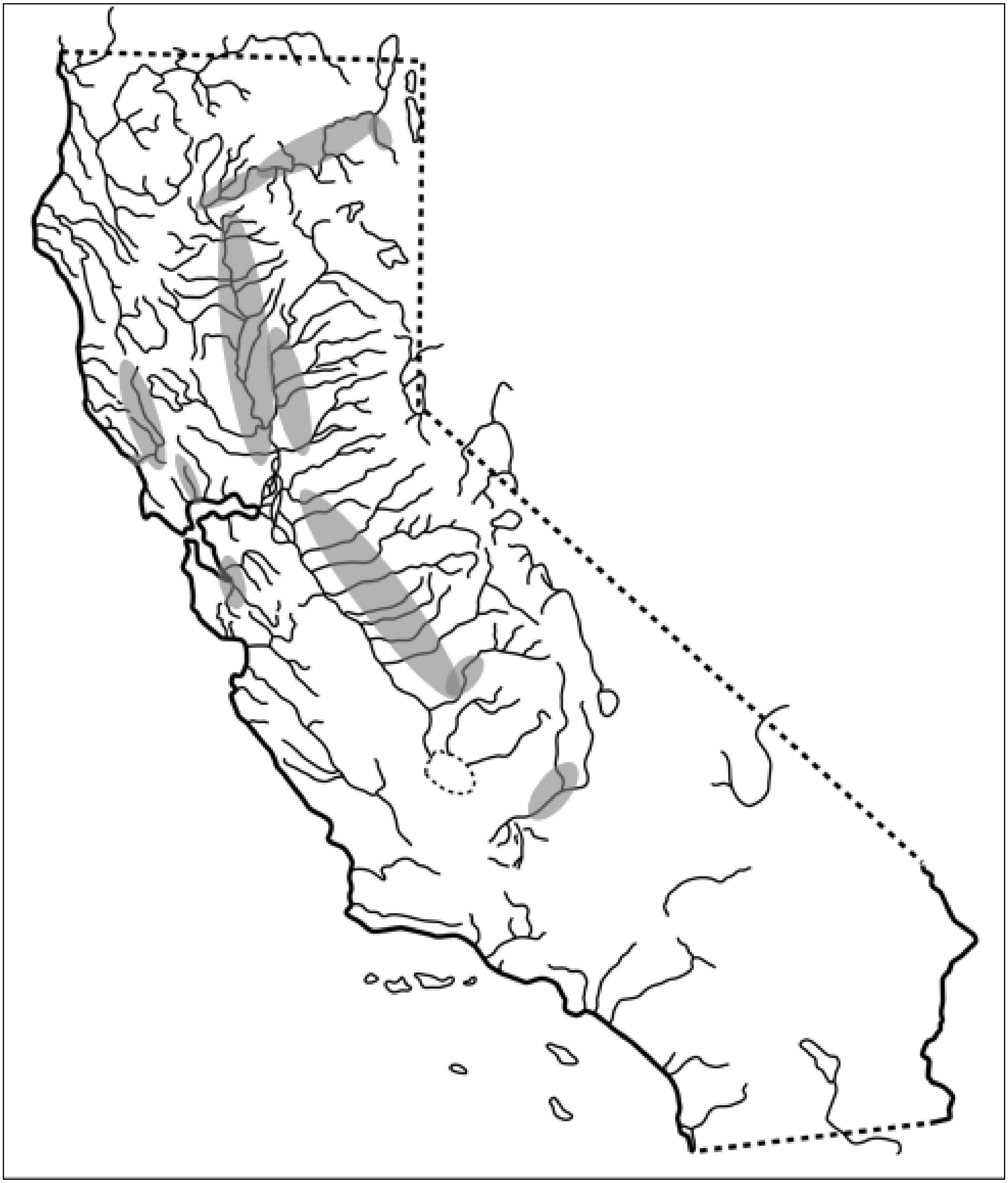


Figure 1. Generalized distribution of hardhead, *Mylopharodon conocephalus* (Baird and Girard), in California. Actual distribution is fragmented.

OWENS SPECKLED DACE
Rhinichthys osculus ssp.

Status: High Concern. The Owens speckled dace was extirpated from a majority of its historic range by the 1980s. Three populations remain, mostly isolated from one another, in Fish Slough, Round Valley, and in irrigation ditches in and near the town of Bishop.

Description: The following is a general description of speckled dace with additional information on the undescribed Owens subspecies. Speckled dace are small cyprinids, usually measuring less than 8 mm SL but occasionally reaching 11 cm SL (Moyle 2002). Although physically variable, they are characterized by a wide caudal peduncle, small scales (47-89 along lateral line), and pointed snout with a small subterminal mouth. At maturity, the dorsal fin usually has 8 rays and originates well behind the origin of the pelvic fins (Moyle 2002). The anal fin has 6-8 rays. Pharyngeal teeth (1,4-4,1 or 2,4-4,2) are significantly curved with a minor grinding surface. The maxilla usually has a small barbel at each end. The snout is connected to the upper lip (premaxilla) by a small bridge of skin (frenum). As their common name indicates, most fish larger than 3 cm have distinctive dark speckles on the dorsum and sides of the body, although some fish from highly turbid waters may lack speckles. Dark blotches present on the side can merge creating what looks like a dark lateral band. A stripe on the head, below the eye, extends to the snout, and there is a black spot on the caudal peduncle. The rest of the body is dusky yellow to olive, with the belly a paler color. Breeding adults of both sexes have fins tipped by orange or red, while males also have red snouts and lips, and tubercles on the head and pectoral fins.

Owens speckled dace are highly variable. A morphometric comparison of all extant populations in the Owens basin found that, although populations differ significantly for many characteristics, there is also high morphological overlap between populations. The frenum was well developed only in the now extirpated Little Lake population. Maxillary barbels occurred in most populations, which separates Owens Valley fish from conspecific populations in the Walker River/Lahontan basin. Speckled dace in the northern Owens Valley have maxillary barbels on at least one side, a high lateral line scale count, a moderate lateral line pore count, and moderately sized fins. Benton Valley populations were described as having low lateral line scale and pore counts, maxillary barbels on at least one side, and comparatively long pelvic fins.

The following ranges in mean counts are for four populations in the Owens River drainage: lateral line scales 59.3-70.7; lateral line pores 11.6-61.7; dorsal rays 7.8-8.0; anal rays 7.0-7.1; pectoral rays 12.0-13.9; pelvic rays 7.0-7.6; total vertebrae 36.9-38.1.

Taxonomic Relationships: The speckled dace has long been considered the most widely distributed species in the western United States and isolated populations can be found in many small streams and springs. However, its taxonomy is complex because the species is naturally so variable and mobile. Small morphological differences among speckled dace populations isolated in different watersheds (especially in the endorheic valleys of the Great Basin) led early ichthyologists to describe 12 separate species (Jordan and Evermann 1896). Later, based on the flexible nature and plastic morphology of the species, all speckled dace were collapsed into a single species, *Rhinichthys osculus* (Hubbs 1974). Recently, however, genetic analysis has supported a return to some of the original taxonomy. Today, a number of forms are recognized as separate taxa by ichthyologists due to their distinctive morphology, diverse habitats, isolation from other dace populations, and genetic differentiation. Five such forms appear to exist in the Death Valley system: the Owens speckled dace, the Long Valley speckled dace, the Amargosa Canyon speckled dace, the Ash Meadows speckled dace, and the Oasis Valley speckled dace.

Gilbert (1893) described *Rhinichthys nevadensis* from Ash Meadows, Nevada, but the subspecific name *R. o. nevadensis* has also been assigned to speckled dace in both the Amargosa River system and the Owens Valley (La Rivers 1962, Moyle 1976). However, since the 1980's, some investigators have placed speckled dace from Amargosa Canyon and the Owens Basin in separate undescribed subspecies (Williams et al. 1982, Deacon and Williams 1984). Sada et al. (1995) conducted a morphological and electrophoretic study of all extant speckled dace populations in the Death Valley region, which includes the Owens and Amargosa river systems, both of which were tributaries to pluvial Lake Manly during the Pleistocene (Miller 1946, Hubbs and Miller 1948). Their results suggest:

1. All the isolated populations in the Owens River hydrographic basin (Owens and Long valleys) show genetic and morphological differences from each other but, with one exception, not enough for them to be regarded as separate subspecies.
2. The exception is the Long Valley speckled dace population in Whitmore Hot Spring, which differs enough from other dace populations to be regarded as a separate subspecies (see account for Long Valley speckled dace).
3. Owens speckled dace are closely related to speckled dace found in the Amargosa River (*R. o. nevadensis*) of Death Valley and probably should be placed within the same subspecies, but each isolated population should be recognized as a distinct population segment for management purposes.
4. To date, studies of the Death Valley region's speckled dace complex indicate the Owens and Amargosa Canyon speckled dace should be treated as distinct population segments of a distinct taxon.

A comprehensive genetic study of *R. osculus* from throughout its entire range using mtDNA supports both the distinctive nature of the Long Valley speckled dace and the grouping of the other Owens basin dace populations with those of Amargosa and Ash Meadows (Oakey et al. 2004). The affinity between Amargosa and Owens Basin is likely the result of their occasional contact in pluvial Lake Manley (Hubbs and Miller 1948, Oakey et al. 2004).

Systematics for the five forms would thus be: Long Valley speckled dace (undescribed subspecies, *R. o. ssp.*); with Owens, Amargosa Canyon, Ash Meadows and Oasis Valley speckled dace representing distinct population segments within the subspecies *R. o. nevadensis*. Despite the fact that the Ash Meadows speckled dace was listed as a federally endangered species in 1984, the Owens and Amargosa Canyon populations remain unprotected, partially due to historic uncertainties about their taxonomic status.

Life History: Specific life-history adaptations of speckled dace from the Owens Basin are unknown. In general, speckled dace live three years and attain a maximum size of 80 mm SL in inland basins (Moyle 2002). Owens speckled dace, however, rarely exceed 50 mm SL in length. Because of the paucity of data on Owens populations, the following general description of speckled dace life history is based on data from other locations.

The subterminal mouth, pharyngeal tooth structure, and short intestine of the speckled dace are characteristic of small invertebrate feeders. Speckled dace generally forage on small benthic invertebrates, especially taxa common in riffles including hydropsychid caddisflies, baetid mayflies, and chironomid and simuliid midges, but will also feed on filamentous algae (Li and Moyle 1976, Baltz et al. 1982, Hiss 1984, Moyle et al. 1991). Not surprisingly, diet varies according with prey availability and speckled dace, in general, prey opportunistically on the most abundant small invertebrates in their habitat, which may change with season (Moyle 2002). Preference of forage items may also be influenced by the presence of other fishes that share similar habitats.

Speckled dace are usually found in loose groups in appropriate habitats, although they avoid large shoals except while breeding. Their activity is mediated by stream temperatures, apparently staying active all year if water temperatures remain above 4°C (Moyle 2002). Slight changes in growth rates are positively correlated with changes in temperature, as seen in the Colorado River (Robinson and Childs 2001). Life expectancy is approximately 3 years where maximum sizes do not exceed 80 mm FL, but dace may reach 110 mm FL and live up to six years (Moyle 2002). By the end of their first summer, dace grow to 20-30 mm SL (Moyle 2002), growing an average of 10-15 mm/yr in each subsequent year. Females tend to grow faster than males. However, growth rates can decrease in the presence of extreme environmental conditions, high population densities, or limited food supply (Sada 1990). Dace reach maturity by their second summer, with females producing 190-800 eggs depending on size and location (Moyle 2002). Females release eggs underneath rocks or near the gravel surface while males release sperm (John 1963). Eggs settle into interstices and adhere to the gravel. At temperatures of 18-19°C, eggs hatch in 6 days but larvae remain in the gravel for another 7-8 days (John 1963). Fry in streams congregate in warm shallow areas, often in channels with rocks and emergent vegetation.

When extreme conditions such as floods, droughts or winter freezing eliminate local populations, speckled dace from nearby areas can readily recolonize or repopulate available habitats if accessible (Sada 1990, Pearsons et al. 1992, Gido et al. 1997).

Habitat Requirements: Speckled dace from the Owens Basin are known to occupy a variety of habitats, ranging from small coldwater streams to hot-spring systems, although they are rarely found in water exceeding 29˚C. They also have been found in irrigation ditches in and near Bishop. Despite the large variety of habitats apparently suitable to speckled dace in the Owens Basin, their disappearance from numerous localities since the 1930s and 1940s suggests they are vulnerable to habitat modifications and predation and or competition by alien fishes. Speckled dace in the Owens Valley appear to persist in periodically disturbed human-created habitats, and areas where alien predatory fishes are excluded by poor water quality or insufficient water depth (S. Parmenter, pers. comm. 2013).

Distribution: California Department of Fish and Wildlife (CDFW) files and museum records from the University of Michigan, Museum of Zoology and the California Academy of Sciences, dating back to the 1930s, indicate that speckled dace historically occupied most small streams and springs in the Owens Valley. In the most comprehensive survey of Owens Basin aquatic habitat to date (166 survey sites), dace were found to have been extirpated from 8 of the 17 sites from which they had been historically recorded. Dace were also discovered at two new locations, from which they have since been extirpated (Sada 1989, S. Parmenter, CDFW, pers. comm. 2009).

Today, Owens speckled dace are only known to occupy three disjunct areas in the northern Owens Valley: Fish Slough, Round Valley, and areas around and in Bishop. Waterways within each of these areas are frequently or consistently interconnected. However, speckled dace dispersal among the three population areas appears to be largely severed by both the presence of alien brown trout in intervening waterways, and stream channelization.

Sada (1989) reported the extirpation of speckled dace from Benton Valley and the persistence of a single small population remaining in the East Fork Owens River drainage at Lower Marble Creek, near Benton. Subsequently, verbal accounts documented the Marble Creek population was eliminated during the Tri-Valley Flood of 1989 (S. Parmenter, CDFW, pers. comm. 2013).

Speckled dace no longer occupy irrigation ditches between Bishop and Big Pine or Little Lake (Inyo County). Nor were Dace found in Warm Springs, where CDFW biologists had planted 75 speckled dace in 1983 (Sada 1989).

Trends in Abundance: There are few data available on the historic abundance of this dace. Given its greatly diminished range, it is undoubtedly much less numerous than it once was. In the streams and irrigation ditches around Bishop, where they are widespread, speckled dace occur at low densities but quantitative abundance estimates are lacking (Sada 1989).

Nature and Degree of Threats: The causes of the decline of Owens speckled dace are complex, but the most significant threats are:

Alien species. Introduction of centrarchid predators (largemouth bass, bluegill, green sunfish and Sacramento perch) into springs and small streams can rapidly drive dace populations to extinction. Other introduced fishes that may be competitors or predators of speckled dace are western mosquitofish (*Gambusia affinis*), brown bullhead (*Ameiurus nebulosus*) and the various trouts. Spring populations can also be threatened by cattails, which can significantly reduce habitat by filling in shallow pools and marshes, alter food webs, and remove water through transpiration.

Isolation. As habitat is altered or otherwise made inaccessible to dace small, isolated, populations are created with no gene flow to other populations. These populations are particularly vulnerable to genetic drift or bottlenecking and to stochastic events which sharply increase probability of extirpation. For example, the Benton Valley populations occurred in small springs and stream segments that were "altered and occupied by introduced predators" (Sada 1989).

Diversions. Regulation of the Owens River has impacted floodplain habitat extent and quality.

Habitat alteration. Speckled dace are highly sensitive to impacts that simplify their habitat or reduce cover. In the Owens Valley, channelization and vegetation clearing may impact dace populations.

	Rating	Explanation
Major dams	Medium	Mainstem dams have likely reduced potential habitat by flow regulation and elimination of floodplain inundation
Groundwater extraction	High	Groundwater extraction has eliminated a majority of springs on the Owens Valley floor, which would have provided habitat for speckled dace
Agriculture	Low	Agricultural water demand may dry irrigation ditches which are important speckled dace habitat, particularly in the face of climate change
Grazing	Low	
Rural residential	Medium	Alteration of streams for diversion and landscaping in and around Bishop
Urbanization	Medium	Alteration of streams for diversion and flood control and conversion of ditches to covered pipelines in and around Bishop
Instream mining	N/A	
Mining	Low	Present in region but no known impacts
Transportation	Low	Roads presumably alter habitats in some areas
Logging	N/A	
Fire	Low	Grass and brush fires are rare in existing and historical dace habitat; prescribed fires are used to maintain dace habitat in Fish Slough
Recreation	Medium	Alteration to thermal spring habitats for swimming and other recreation has been, and continues to be, a substantial threat as is off-road vehicle use
Harvest	N/A	
Hatcheries	N/A	
Alien species	High	All populations are vulnerable

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

Effects of Climate Change: The thermal spring systems which comprise a major portion of Owens speckled dace habitat are fed by aquifers dependent on snow melt for recharge. It is predicted that climate change will lead to a reduction in snow pack in the eastern Sierra Nevada due to warmer temperatures and a shift in precipitation toward rainfall in late winter and early spring months. However, the Owens Valley is at the base of the southernmost portion of the Sierra Nevada, where the range attains maximum elevations. Thus, the effects of climate change may be mitigated, at least to some extent, by retention of snow pack in this portion of the range. However, Moyle et al. (2013) score this dace as "critically vulnerable" to climate change, indicating extinction is likely within the next 100 years if measures to counter climate change effects are not taken. The predicted, hotter, drier future climate, paired with an ever-increasing human demand for water resources in the Owens Basin, strongly indicates that aquatic habitats must be protected if the Owens speckled dace is to persist.

Status Determination Score = 2.6 – High Concern (see Methods section, Table 2). The Owens speckled dace has been extirpated from many of its historic locations due to habitat alteration, alien species and water withdrawal. Only a handful of populations remain, mostly isolated from one another in the norther Owens Valley. The Owens speckled dace is listed as "Critically Imperiled", G5T1S1 by (Natureserve.org) and the American Fisheries Society lists it as "Threatened" (Jelks et al 2008).

Metric	Score	Justification
Area occupied	2	Few historic populations still extant in one watershed
Estimated adult abundance	3	Moderate fragmentation of existing populations today
Intervention dependence	3	Many populations depend on irrigation water for persistence, but no active management of natural populations is indicated
Tolerance	2	Relatively tolerant but vulnerable to habitat alteration and introduction of alien fishes and salamanders
Genetic risk	4	Moderate diversity but range fragmented into subpopulations with reduced connectivity
Climate change	2	Thermal pools provide relatively stable habitat for some populations but loss of water for irrigation could imperil other populations
Anthropogenic threats	2	See Table 1
Average	2.6	18/7
Certainty (1-4)	3	

Table 2. Metrics for determining the status of Owens speckled dace, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: The most critical needs for Owens speckled dace are:

1. Maintain existing, compatible land management practices and land uses on existing habitat. Consider creating special refuge areas and establishing additional populations, particularly in upstream areas from which larval drift could seed new areas or supplement existing populations.
2. Eliminate alien fishes from springs which historically supported speckled dace and reintroduce dace from local brood stock.
3. Establish Owens speckled dace at additional sites in the Owens Valley as recommended by Sada (1989) and by the Owens Basin Wetland and Aquatic Species Recovery Plan (USFWS 1998).

4. Establish an annual monitoring program for all populations. Isolated populations of dace are susceptible to habitat loss and alteration, effects of genetic drift, stochastic events, and to the establishment of alien fishes (Williams and Sada 1985).
5. Complete formal studies on taxonomic and genetic status and publish results in a peer-reviewed journal.
6. Initiate studies into both the specific life-history and habitat requirements of Owens populations of speckled dace.

Figure 1. Presumed historic distribution of Owens speckled dace, *Rhinichthys osculus* ssp., in California. Current distribution is highly fragmented.

LONG VALLEY SPECKLED DACE
Rhinichthys osculus ssp.

Status: Critical Concern. Long Valley speckled dace face the possibility of extinction in their native range within the next 50 years because they exist only in a single, thermal-spring complex fed by the chlorinated outflow of a public swimming pool.

Description: Speckled dace are small cyprinids, usually measuring less than 8 mm but occasionally reaching 11 cm SL (Moyle 2002). Although physically variable, they are characterized by a wide caudal peduncle, small scales (47-89 along lateral line) and pointed snout with a small, subterminal mouth. At maturity, the dorsal fin usually has 8 rays and originates well behind the origin of the pelvic fins (Moyle 2002). The anal fin has 6-8 rays. Pharyngeal teeth (1,4-4,1 or 2,4-4,2) are significantly curved with a minor grinding surface. The maxilla usually has a small barbel at each end. The snout is connected to the upper lip (premaxilla) by a small bridge of skin (frenum). As their common name indicates, most fish larger than 3 cm have distinctive dark speckles on the upper and sides of the body, although some fish from highly turbid waters may lack speckles. Dark blotches present on the side can merge, creating what looks like a dark lateral band. A stripe on the head, below the eye, extends to the snout, and there is black a spot on the caudal peduncle. The rest of the body is dusky yellow to olive, with the belly a paler color. Breeding adults of both sexes have fins tipped by orange or red, while males also have red snouts and lips and tubercles on the head and pectoral fins.

Long Valley speckled dace are distinguished by high numbers of pectoral and pelvic fin rays, high lateral line scale count, low lateral line pore count, and the absence of maxillary barbels (Sada et al. 1995). The following mean counts (standard error) are from Long Valley speckled dace collected in Whitmore Hot Springs and at an unnamed spring at Little Alkali Lake (Sada 1989): lateral line scales 61.7 (1.4); lateral line pores 19.0 (5.0); dorsal rays 8.0 (0.0); anal rays 7.0 (0.0); pectoral rays 13.0 (0.4); pelvic rays 7.4 (0.2).

Taxonomic Relationships: Speckled dace from Long Valley are both morphologically distinct (Sada 1989, 1995) and monophyletic (Oakey et al. 2004), suggesting they are a distinct taxon in need of formal taxonomic description. It is not surprising that this population remains undescribed, given the confusing systematics of the naturally variable speckled dace. The speckled dace has long been considered the most widely distributed species in the western United States, with many isolated populations found in small streams and springs. Small morphological differences among speckled dace populations isolated in different watersheds (especially in the endorheic valleys of the Great Basin) led early ichthyologists to describe 12 separate species (Jordan and Evermann 1896). Later, based on the flexible nature and plastic morphology of the species, all speckled dace were collapsed into a single species, *Rhinichthys osculus* (Hubbs et al. 1974). Recently, however, genetic analysis has supported a return to some of the original taxonomy. Today, a number of forms are recognized as separate taxa based on their distinctive morphology, isolation from other dace populations, and because they are genetically distinct. Four such forms are now recognized in the Death Valley system: the Owens speckled dace, the Long Valley speckled dace, the Amargosa Canyon speckled dace, and the Ash Meadows speckled dace.

Gilbert (1893) described *Rhinichthys nevadensis* from Ash Meadows, Nevada, but the subspecific name *R. o. nevadensis* has been assigned to speckled dace in the Amargosa River canyon and Owens Basin as well (La Rivers 1962, Moyle 2002). However, in the early 1980s, research revealed that these three populations are distinct (Williams et al. 1982). As a consequence, Williams et al. (1982) and Deacon and Williams (1984) recommended that the populations from these three areas be placed in separate subspecies. Sada et al. (1995) conducted a morphological and electrophoretic study of all extant speckled dace populations in the Owens and Amargosa river systems, all of which were tributaries to pluvial Lake Manly during the Pleistocene (Miller 1946, Hubbs and Miller 1948). Their results suggest:

1. All the isolated populations in the Owens Valley show genetic and morphological differences from each other but, with one exception, not enough for them to be regarded as separate subspecies.
2. The exception is the Long Valley speckled dace population in Whitmore Hot Spring, which differs enough from other dace populations to be regarded as a separate subspecies.
3. Owens speckled dace are closely related to speckled dace found in the Amargosa River (*R. o. nevadensis*) of Death Valley and probably should be placed within the same subspecies, but each isolated population should be recognized as a distinct taxa for management purposes.
4. To date, no study of the Death Valley system speckled dace complex has been robust enough to assign subspecies names to lineages other than that in Long Valley. Until such studies are completed, the Owens and Amargosa Canyon speckled dace should be treated as distinct populations segments of *R. o. nevadensis.*

The Whitmore Hot Springs population represents the last extant population of Long Valley speckled dace. This is the only speckled dace population known with a private fixed allele (the D allele of the PEPA locus) (Sada et al. 1995). This is possibly the result of long isolation within the 700,000 year-old Long Valley Caldera (Hill et al. 1985). The speckled dace populations of the entire Death Valley system (the Owens, Amargosa and Mojave river drainages) form a monophyletic clade (Oakey et al. 2004). However, Long Valley dace are clearly differentiated within this grouping. Oakey's results, using mtDNA restriction site mapping, paired with geologic evidence, suggests that Long Valley speckled dace may retain haplotypes from an earlier period.

Life History: Little work has been conducted on the life history adaptations of speckled dace in Long Valley. The following general description is gathered from dace populations in other locations. The subterminal mouth, pharyngeal tooth structure, and short intestine of the speckled dace are characteristic of small invertebrate feeders. Speckled dace generally forage on small benthic invertebrates, especially taxa common in riffles including hydropsychid caddisflies, baetid mayflies, and chironomid and simuliid midges, but will also feed on filamentous algae (Li and Moyle 1976; Baltz et al. 1982; Hiss 1984, Moyle et al. 1991). Not surprisingly, diet varies according with prey availability and speckled dace, in general, prey opportunistically on the most abundant small invertebrates in their habitat, which may change with season (Moyle 2002). Preference of forage items may also be influenced by the presence of other fishes that share similar habitats.

Speckled dace are usually found in loose groups in appropriate habitats although they avoid large shoals, except while breeding. Their activity is also mediated by stream temperatures and they apparently stay active all year if water temperatures remain above 4°C (Moyle 2002). Slight changes in growth rates are also positively correlated with changes in temperature, as seen in the Colorado River (Robinson and Childs 2001). Life expectancy is approximately 3 years where maximum sizes do not exceed 80 mm FL, but dace may reach 110 mm FL and live up to six years (Moyle 2002). By the end of their first summer, dace grow to 20-30 mm SL (Moyle 2002), growing an average of 10-15 mm/yr in each subsequent year. Females tend to grow faster than males. However, growth rates can decrease in the presence of extreme environmental conditions, high population densities, or limited food supply (Sada 1990). Dace reach maturity by their second summer with females producing 190-800 eggs, depending on size and location (Moyle 2002). Females release eggs underneath rocks or near the gravel surface while males release sperm (John 1963). Eggs settle into interstices and adhere to the gravel. At temperatures of 18-19°C, eggs hatch in 6 days but larvae remain in the gravel for another 7-8 days (John 1963). Fry in streams congregate in warm shallow areas, often in channels with rocks and emergent vegetation.

When extreme conditions such as floods, droughts, or winter freezing eliminate local populations, speckled dace from nearby areas can readily recolonize or repopulate available habitats if accessible (Sada 1990, Pearsons et al. 1992, Gido et al. 1997).

Habitat Requirements: Speckled dace from the Owens Basin are known to occupy a variety of habitats ranging from small, cold-water streams to hot spring pools, although they are rarely found in water exceeding 29˚C. After conducting morphometric analysis of both extant and museum specimens, Sada (1989) theorized that Long Valley speckled dace were a deep-bodied form adapted to spring habitats. Despite the large variety of habitats apparently suitable for speckled dace in the Owens Basin, their disappearance from numerous localities suggests that they are quite vulnerable to habitat modifications and to invasion by alien fishes. Their present habitat is the shallow (<50 cm), clear outflow of a single spring, including two open pools in a marshy area. The narrow, nearly invisible, channel flows through a dense growth of bulrush, which provides cover for the fish.

Distribution: The entire native range of this dace lies within the 700,000 year-old Long Valley volcanic caldera, just east of Mammoth Lakes, Mono County, including Hot Creek and various isolated springs and ponds. The formation of the caldera likely led to their isolation long before other populations of the northern Owens Basin were isolated from one another. Long Valley speckled dace have been extirpated from all but one of their historic collection sites, including Hot Creek. The sole remaining population within the native range is in Whitmore Hot Springs (Sada 1989). Whitmore Hot Springs has been developed and is operated as a swimming pool by Mono County. Spring discharge of approximately 2 cfs is lightly chlorinated and feeds an alkali marsh of roughly 1 acre. In 1989, dace occupied 250 yards of stream and two large shallow ponds that did not exceed half a meter in depth. The dace population here appears to be heavily parasitized in some years (Sada 1989, S. Parmenter, CDFW, pers. comm. 2009). Surveys in 2002 and 2009 by CDFW found this population to be relatively stable (S. Parmenter, CDFW, pers. comm. 2009).

Long Valley speckled dace were translocated from Whitmore Hot Springs to an undisclosed location near Bishop (S. Parmenter, CDFW, pers. comm. 2009). On average, six additional fish from the Whitmore Springs population are translocated to the refuge population annually in an effort to minimize genetic drift.

In 1988, Sada discovered a population in an unnamed spring at Little Alkali Lake but this population was subsequently extirpated. Dace occupied an estimated 600 meters of stream between the spring source and the lake. Fish were not believed to occupy the spring source, where water temperatures exceeded 28˚C, or Little Alkali Lake itself. When last surveyed in 1999, large numbers of western mosquitofish (*Gambusia affinis*) were observed but speckled dace appeared to be absent (S. Parmenter, CDFW, pers. comm. 2009). Speckled dace were last sampled in Hot Creek in 1962 but were likely extirpated due to alterations to the system, including the creation and operation of Hot Creek Hatchery, as well as introduction of non-native trout to the stream (Sada 1989). Non-native trout are also abundant in Crowley Lake, which is connected to Hot Creek via the upper Owens River.

The Bureau of Land Management and CDFW are cooperating in an ongoing project to restore the unnamed spring tributary to Little Alkali Lake to expand the range of Long Valley speckled dace. To date, three fish barriers have been constructed and experiments are under way to eradicate mosquitofish by a combination of mechanical removal and spring diversion under freezing temperatures (S. Parmenter, CDFW, pers. comm. 2014).

Trends in Abundance: There are few data available on the historic abundance of this dace. However, the extirpation of all but one of the historically identified populations means that it is undoubtedly much less numerous than it once was. According to the US Fish and Wildlife Service (1998), it is continuing to decline.

Nature and Degree of Threats: The major causes of decline of the Long Valley speckled dace are multiple and compounded by the fact that the remaining small, isolated, populations of dace are particularly vulnerable to genetic drift and to stochastic events.

Grazing. Reduction in riparian vegetation and trampling of stream banks and springs by cattle has impacted much of their limited habitat (e.g., Whitmore Springs, Little Alkali Lake) by increasing sediment input into pools and channels, increasing solar input, and reducing habitat complexity and cover.

Recreation. The water source that supports remaining speckled dace habitat, Whitmore Hot Springs, is now a public swimming pool. The effluent maintains sufficient flows to support this population, at least in the short-term, but a spill of over-chlorinated water could extirpate them. Whitmore Hot Springs is operated by the county of Mono as a public facility, and public health laws require disinfection.

Hatcheries. Hot Creek Trout Hatchery, a CDFW facility, likely contributed to the extirpation of dace in Hot Creek and its adjacent springs through diversion of water and construction activities in the 1960s. Based on potential habitat, it is likely that the Hot Creek population was historically one of the largest populations of this dace.

Alien species. Alien fishes, especially western mosquitofish, largemouth bass (*Micropterus salmoides*), and various sunfish (*Lepomis spp.*) have been implicated in extirpating other isolated dace populations. In 1988, the only extant populations of Long Valley speckled dace were found in springs where no other fish species were present. Subsequently, the population in the unnamed spring at Little Alkali Lake became extirpated, concurrent with the discovery of mosquitofish in the spring system (S. Parmenter, CDFW, pers. comm. 2009), along with heavy damage by grazing. The single remaining population could easily be extirpated by introduction of another fish species into its limited habitat.

In addition, a population of introduced tiger salamanders exists within three miles of Whitmore Hot Springs (S. Parmenter, CDFW, pers. comm. 2009), walking distance for adult salamanders. Colonization of the springs by these predatory amphibians could eliminate the last remaining natural population of Long Valley speckled dace. Their limited habitat is also threatened by invasion of cattail (*Typha* spp.), which can significantly reduce open water habitat by rapidly colonizing shallow pools and marshes with a solid mass of plant stems.

	Rating	Explanation
Major dams	n/a	
Agriculture	Low	Little agriculture near extant populations
Grazing	High	Cattle continue to seriously degrade dace habitats
Rural residential	Low	Few residences nearby
Urbanization	Low	Past water delivery infrastructure built by LADWP (e.g., Crowley Lake, diversions, pipelines) may have altered historic dace habitats
Instream mining	n/a	
Mining	Low	No known impacts but mining present in region
Transportation	Low	No known impacts but roads border dace habitats
Logging	n/a	
Fire	Low	Grass fires can destroy protective vegetation
Estuary alteration	n/a	
Recreation	Critical	Whitmore Hot Springs, the source of water for remaining dace habitat, is a public swimming pool; chlorine release is a potentially severe threat
Harvest	n/a	
Hatcheries	High	The creation of Hot Creek Trout Hatchery probably led to extirpation of the population in Hot Creek
Alien species	Critical	No Long Valley speckled dace are found in habitats where alien fishes occur

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

Effects of Climate Change: The thermal spring systems which Long Valley speckled dace inhabit are fed by aquifers dependent on snow melt for recharge. One major predicted impact of climate change in the eastern Sierra Nevada is the reduction in snow pack due to warmer temperatures. This will have the least effect in the southern Sierra Nevada because the range reaches its highest elevations in this area, so snow pack is expected to remain consistent. Thus, snowmelt is likely to maintain flows in most Owens Valley streams. However, it is possible that snow pack will be reduced in the portion of the Sierra Nevada spanning from Bishop to June Lake; this region is most proximate to remaining Long Valley speckled dace habitats and snow pack retention will likely be critical to maintaining stream flows and aquifer recharge in Long Valley. In any case, climate change predictions indicate that snow will not persist as long into the hotter months and stream flows will likely be reduced in late summer or early fall. A hotter, drier, future climate, paired with an ever-increasing human demand for decreasing water resources in the Owens Basin, suggest that dace habitat may be threatened by drying conditions in the future. Moyle et al. (2013) regarded the Long Valley specked dace as "critically vulnerable" to extinction from climate change, along with the other substantial threats this dace faces.

Status Determination Score = 1.0 – Critical Concern (see Methods section, Table 2). The Long Valley speckled dace now exists as a single population in shallow pools fed by the chlorinated outflow of a public swimming pool. The Long Valley speckled dace is listed as "Endangered" by the American Fisheries Society (Jelks et al. 2008) and as "declining" by the US Fish and Wildlife Service (Owens Basin Wetland and Aquatic Species Recovery Plan 1998).

Metric	Score	Justification
Area occupied	1	Only one small population occupying a single thermal spring system
Estimated adult abundance	1	Fluctuates widely but was very small when the authors sampled in July, 2010
Intervention dependence	1	Refuge populations must be established; prevention of over-chlorinated releases from Whitmore Hot Springs resort critical
Tolerance	1	Extremely vulnerable to competition from alien fishes
Genetic risk	1	Bottlenecking a distinct possibility
Climate change	1	Shallow pools fed by a thermal spring are vulnerable to rises in air temperature
Anthropogenic threats	1	See Table 1
Average	1.0	7/7
Certainty (1-4)	3	Good recent data

Table 2. Metrics for determining the status of Long Valley speckled dace, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: The Long Valley speckled dace is one of the most critically imperiled non-listed fishes in California and requires intensive and ongoing management, monitoring and would benefit from additional protections and provisions afforded by either a state or federal listing (or both).

Without additional protection, this species is likely to go extinct in the wild. Actions required include:

1. Whitmore Hot Springs should be managed for Long Valley speckled dace and other spring organisms, with dace populations monitored annually.
2. A thorough survey of all potential habitats within the Long Valley Caldera should be conducted and all existing habitats given special protection (e.g., fenced from grazing).
3. The extant refuge population near Bishop should be maintained, ensuring adequate gene flow by translocation individuals from Whitmore annually.
4. Populations should be established at additional sites in Long Valley, as recommended by Sada (1989) and the Owens Basin Wetland and Aquatic Species Recovery plan (USFWS 1998). Priority reintroduction locations are:
    a. The spring system at Little Alkali Lake, but only after mosquitofish have been removed. A low head fish barrier should also be installed to inhibit recolonization of the spring system from the lake.
    b. The Hot Creek and Little Hot Creek conservation areas, as stated in the Owens Basin Wetland and Aquatic Species Recovery plan (USFWS 1998).
5. Non-native fish should be eradicated from springs which historically supported speckled dace in order to facilitate re-introduction.
6. Consider the creation of artificial refuges, such as small ponds on existing spring systems, recognizing that such ponds have limited life spans and must be actively managed.
7. Alien tiger salamander populations near Whitmore Hot Springs should be exterminated.

Figure 1. Distribution of Long Valley speckled dace, *Rhinichthys osculus* ssp., in California.

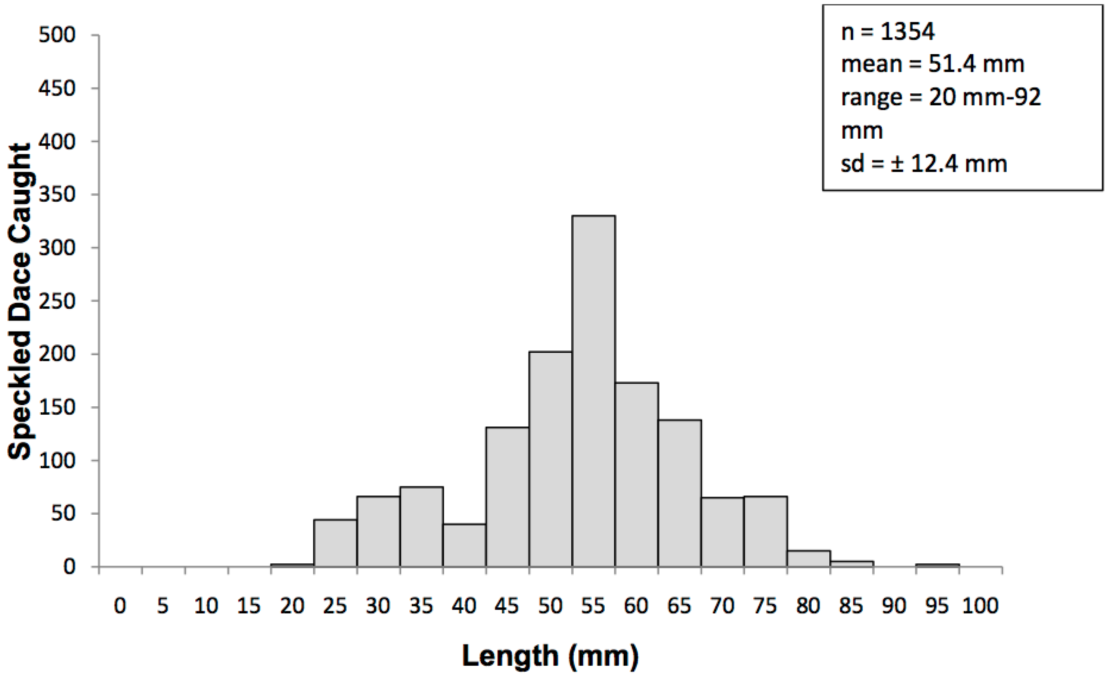
AMARGOSA CANYON SPECKLED DACE
Rhinichthys osculus ssp.

Status: Critical Concern. Amargosa Canyon speckled dace are highly vulnerable to extinction in the next 50 years because they are restricted to a single desert stream system which is under threat of dewatering and invasion by non-native plants, fishes, crustaceans and amphibians.

Description: Speckled dace are small cyprinids, usually measuring less than 8 cm SL at maturity but occasionally reaching 11 cm SL (Moyle 2002). Although physically variable, they are characterized by a wide caudal peduncle, small scales (47-89 along lateral line), and pointed snout with a small subterminal mouth. At maturity the dorsal fin usually has 8 rays and originates well behind the origin of the pelvic fins (Moyle 2002). The anal fin has 6-8 rays. Pharyngeal teeth (1,4-4,1 or 2,4-4,2) are significantly curved with a minor grinding surface. The maxilla usually has a small barbel at each end. The snout is connected to the upper lip (premaxilla) by a small bridge of skin (frenum). As their common name indicates, most fish larger than 3 cm have distinctive dark speckles on the upper and sides of the body, although some fish from highly turbid waters may lack speckles. Dark blotches present on the side can merge, creating what looks like a dark lateral band. A stripe on the head, below the eye, extends to the snout, and there is black a spot on the caudal peduncle. The rest of the body is dusky yellow to olive, with the belly being a paler color. Breeding adults of both sexes have fins tipped by orange or red, while males also have red snouts and lips, as well as tubercles on the head and pectoral fins.

Amargosa Canyon speckled dace are visually similar to other *Rhinichthys osculus* subspecies. However, dace from Amargosa Canyon are characterized by a comparatively smaller head depth, shorter snout-to-nostril length, longer anal-to-caudal length, more pectoral fin rays, and fewer vertebrae than other forms. Speckled dace captured during a summer, 2010 survey of Amargosa Canyon ranged from 20 to 92 mm in fork length with a mean of 51 mm (Scoppettone et al. 2011, Fig. 1).

Figure 1. Length frequency of speckled dace (*Rhinichthys osculus* spp.) caught during the summer, 2010 survey of the Amargosa River Canyon, California. Figure from Scoppettone et al. 2011.



Taxonomic Relationships: The speckled dace has long been considered the most widely distributed freshwater fish species in the western United States and isolated populations can be found in many small streams and springs. However, its taxonomy is poorly understood and highly confusing because the species is naturally so variable. Originally, small morphological differences among speckled dace populations isolated in different watersheds (especially in the endorheic valleys of the Great Basin) led ichthyologists to describe 12 separate species (Jordan and Evermann 1896). Later, because of the plastic morphology of the species, all speckled dace were collapsed into a single species, *Rhinichthys osculus* (Hubbs et al. 1974). Recently, however, genetic analysis has supported a return to the some of the original taxonomy. A number of forms are now recognized as separate taxa, not only because of their distinctive morphology, different habitats, and isolation from other dace populations, but also because they can be shown to be genetically distinct. Four such forms are now recognized in the Death Valley system: the Owens speckled dace, the Long Valley speckled dace, the Amargosa Canyon speckled dace, and the Ash Meadows speckled dace (Nevada).

Gilbert (1893) described *Rhinichthys nevadensis* from Ash Meadows, Nevada, but the subspecific name *R. o. nevadensis* was later also applied to speckled dace in the Amargosa River canyon and Owens Basin (La Rivers 1962, Moyle 2002). However, in the early 1980s, research revealed that these three populations are morphometrically distinct (Williams et al. 1982). Amargosa Canyon dace have a comparatively smaller head depth, shorter snout-to-nostril length, longer anal-to-caudal length, more pectoral fin rays, and fewer vertebrae than the other forms. As a consequence, Williams et al. (1982) and Deacon and Williams (1984) recommended that the populations from these three areas be treated as undescribed subspecies. In addition, the dace population in Long

Valley, in the northern Owens Basin, has been found to be morphologically distinct (Sada 1989, Sada et al. 1995) and genetically monophyletic (Oakey et al. 2004).

Each of these taxa are treated as separate subspecies in this report. It may also be valid to consider all of the above forms as comprising one taxon, *R. o. nevadensis,* while recognizing that each population represents a unique component of the overall population, on its own evolutionary trajectory. If so, it is important to realize that each DPS is in greater danger of extinction than is the entire group collectively. In such a case, the Owens, Amargosa Canyon and Ash meadows speckled daces would represent distinct population segments within the subspecies *R. o. nevadensis,* while the Long Valley speckled dace would still be recognized as an undescribed subspecies.

Although the Ash Meadows speckled dace was listed as a federally endangered species in 1984, the Owens and Amargosa Canyon populations remain unprotected, partially because they have never been formally described. Regardless of taxonomy, all populations in the Amargosa and Owens River drainages are in need of protection with individualized management plans to prevent declines in their status.

Life History: Amargosa Canyon speckled dace are active throughout the year, including the winter months. As a consequence, because growth is continuous throughout the year, they are difficult to age by scale analysis. However, length-frequency analysis of dace from other localities suggests that dace generally live for 5-6 years (Moyle 2002). In Amargosa Canyon, the most frequent size class in May was 52-54 mm TL but, in July, smaller fish averaging 31-33 mm were more common (Williams et al. 1982). However, in May there were many small fish (<30 mm TL), suggesting that peak spawning occurs in early spring (March) and that spawning activity is reduced or absent in late spring and summer. Speckled dace reproduce in their second year (Constantz 1981), so the 52-54 mm TL size class (common in May) are probably first-year fish (Williams et al. 1982).

Few data have been collected on the life history of Amargosa Canyon speckled dace. This description, therefore, is based on data from other *R. osculus* populations. Speckled dace are usually found in loose groups in appropriate habitats, although they avoid large shoals except while breeding. They can be active both day and night, although Moyle (unpubl. data) found that Lahontan speckled dace were more nocturnal in their habits when subjected to heavy bird predation in streams. Their activity is also mediated by stream temperatures; they apparently stay active all year if stream temperatures remain above 4°C, which would be typical of the Amargosa River (Moyle 2002).

Their subterminal mouth, pharyngeal tooth structure, and short intestine are characteristic of small invertebrate feeders. Not surprisingly, diet varies according with prey availability and speckled dace, in general, prey opportunistically on the most abundant small invertebrates in their habitat, which may change seasonally. Speckled dace generally forage on small benthic invertebrates, especially taxa common in riffles, including hydropsychid caddisflies, baetid mayflies, and chironomid and simuliid midges, but will also feed on filamentous algae (Li and Moyle 1976, Baltz et al. 1982, Hiss 1984, Moyle et al. 1991). Preference of forage items may also be influenced by the presence of other fishes that share similar habitats (e.g., pupfish).

Length frequency analyses have determined age and growth patterns. By the end of their first summer, dace grow to 20-30 mm SL (Moyle 2002), growing an average of

10-15 mm/yr in each subsequent year. Females tend to grow faster than males. However, growth rates can decrease under extreme environmental conditions, high population densities, and/or limited food (Sada 1990). Slight changes in growth rates are also positively correlated with changes in temperature, as seen in the Colorado River (Robinson and Childs 2001). Life expectancy is approximately 3 years, where maximum sizes do not exceed 80 mm FL; however, dace may reach 110 mm FL and live up to six years (Moyle 2002). Dace reach maturity by their second summer, with females producing 190-800 eggs, depending on size and location (Moyle 2002). Females release eggs underneath rocks or near the gravel surface while males release sperm (John 1963). Eggs settle into interstices and adhere to substrates. At temperatures of 18-19°C, eggs hatch in 6 days, but larvae remain in the gravel for another 7-8 days (John 1963). Fry in streams congregate in warm, shallow areas, often in channels with rocks and emergent vegetation.

When extreme conditions such as floods, droughts, or winter freezing eliminate local populations, speckled dace from nearby areas can readily recolonize or repopulate available habitats if accessible (Sada 1990, Pearsons et al. 1992, Gido et al. 1997). Following a devastating flood, densities of speckled dace in the Colorado River, Arizona, returned to pre-flood levels after eight months, recolonizing from upstream and stream margin areas (Valdez et al. 2001). Such recolonization may be of particular importance in the Amargosa River where large but infrequent flood events are a defining characteristic of the desert hydrograph.

Habitat Requirements: Unlike other speckled dace, which usually prefer moving water, Amargosa Canyon dace prefers pool-like habitat with deep (0.45-0.75 m), slow (<0.01 m^3 sec^{-1}) water. Williams and others (1982) found speckled dace to be rare within the Amargosa River Canyon but abundant in Willow Creek and Willow Creek Reservoir (Williams et al. 1982). In contrast, a recent survey found both speckled dace and pupfish in robust numbers in the Amargosa Canyon but found dace to be rare in Willow Creek (Scoppettone et al. 2011). Summer water temperatures ranged little (23.4-24.8°C) in Amargosa River Canyon during the 2011 survey, while dissolved oxygen ranged from 6.2 to 8.6 mg/L, conductivity ranged from 2,044 to 5,318 μS/cm, and pH ranged from 7.9 to 8.3. Water temperatures were generally warmer in the river than in Willow Creek where they ranged from 25.2 to 28.7 °C (Scoppettone et al. 2011).

Williams et al. (1982) reported the following physical characteristics for Willow Creek, a small, clear stream with low flow (1 cfs) and fine sand/silt substrates: pH of 7.7, dissolved oxygen of 5-6 mg l^{-1}, total dissolved solids of 700 ppm, and water temperatures of 21-28° C. The reservoir was turbid, with a substrate of easily roiled fines. The periphery of the reservoir has dense stands of salt-cedar and cattails (Williams et al. 1982). Scoppettone et al. (2011) made the following daytime measurements in Willow Creek: dissolved oxygen 7.1–12.1 mg/L, conductivity 1,027–1,082 μS/cm, and pH 7.6–8.4. The high dissolved oxygen (12.1 mg/L) was probably due to the lower station having shallow water (<4 cm deep), with little flow and exposure to the sun, all of which are conditions promoting higher photosynthesis.

Riparian vegetation does not appear to drive distribution, because no significant difference in abundance and density of speckled dace was observed between open water and highly vegetated reaches of Amargosa Canyon (Scoppettone et al. 2011).

Distribution: This population is confined to the Amargosa River in Amargosa Canyon and its tributaries, especially Willow Creek and Willow Creek Reservoir (Williams et al. 1982, Scoppettone et al. 2011). In the summer of 2010, it was found to be abundant throughout Amargosa Canyon, except in the lowest reaches which are subject to drying (stranded dead pupfish were observed in desiccated pools at the time of the survey). It is possible that speckled dace scarcity in this lower reach of Amargosa Canyon was due to stranding avoidance behavior (Scoppettone et al. 2011). Historically, Amargosa dace were found in a warm spring just north of Tecopa (Miller 1938) but that population is no longer present. Overall, its range may have been reduced by water diversion which may reduce surface flow in Amargosa Canyon.

Trends in Abundance: During a 1981 survey of the Amargosa Canyon that included Willow Creek, speckled dace comprised 1% and introduced western mosquitofish (*Gambusia affinis)* 40% of the fish collected (Williams et al. 1982). In the most recent survey, speckled dace were relatively abundant, representing 40% of the total catch, while mosquitofish only represented 8%, with the remaining 52% being pupfish. These latest results suggest that speckled dace populations in Amargosa Canyon fluctuate, possibly in response to flow patterns in Amargosa Canyon and interactions with introduced mosquitofish. It is likely that flood events favor native speckled dace by flushing mosquitofish from the system.

Nature and Degree of Threats: The major threat to Amargosa Canyon speckled dace is the potential dewatering of its unique habitats, the Amargosa River and tributaries, combined with interactions with invasive species (Table 1).

Agriculture, rural residential development, urbanization. These three categories are lumped because, together, they result in water withdrawals from sources which feed the Amargosa River, both far and near. The Amargosa Aquifer supplies the springs of Ash Meadows, Nevada and the Amargosa River, to which they are tributary (Riggs and Deacon 2002). It receives much of its recharge flow from areas on the northern and northeastern slopes of the nearby Spring Mountains but, along with springs on the eastern side of Death Valley, is partially dependent on regional groundwater movement through large, ancient aquifers that extend into western Utah and central Nevada (Dettinger and Cayan 1995, Deacon et al. 2007). In order to supply the city of Las Vegas, the Southern Nevada Water Authority (SNWA) proposed to mine large quantities of this water from several different valleys which lie within the Ash Meadows groundwater basin (Breen 2004, Southern Nevada Water Authority 2004, Vogel 2004). Farming operations and human settlements in the Amargosa region are withdrawing increasing amounts of water from the aquifer, producing noticeable declines in the water level of closely-monitored Devils Hole, Nevada (habitat of the endangered Devils Hole pupfish, *Cyprinodon diabolis*) (Riggs and Deacon 2004, Bedinger and Harrill 2006).

If Amargosa region water withdrawals continue to increase and if the SNWA proceeds with its planned withdrawals, it is highly likely that Amargosa River flows will be greatly reduced or even disappear entirely during dry years. Already, diversions of springs and outflows on private land in the Tecopa area have reduced flows in the river and local pupfish populations as well. Corresponding with increasing human population growth around Tecopa and the upper Amargosa Valley, potential threats to aquatic habitats in the Amargosa River from water use and flood protection also increase.

Although most land in Amargosa Canyon is owned or administered by The Nature Conservancy or the Bureau of Land Management, important habitat for the dace includes a large tract of privately owned land, China Ranch, which contains the headwaters of Willow Creek. Diversion of water from the creek or other alterations affecting water quality may be affecting dace populations.

Grazing. Although grazing is not a major land use in the region, cattle have a tendency to aggregate around water sources, particularly in arid landscapes, so their impact on aquatic habitat can be disproportionate to their actual numbers. Water is also diverted directly from the stream for cattle and pumped to grow alfalfa for feed.

Recreation. The deserts of California support high recreational use, driven by the millions of people living in the nearby major urban areas of southern California. Off-highway vehicular use (motorcycles, quads, sand rails, dune buggies, etc.) is a growing form of recreation, creating impacts to sensitive desert and aquatic habitats. These activities are difficult to regulate and, although regulations are in place that ban the use of off-road vehicles in sensitive areas, riparian and streambed habitat degradation from illegal vehicle use still occurs. The rapidly increasing popularity of off road motorized vehicle recreation represents a growing threat to the Amargosa River and its watershed.

Alien species. Although historic data are lacking, it is assumed that native fishes were likely found in greater abundance in the Amargosa River prior to the invasion of saltcedar (*Tamarisk*), crayfish, and mosquitofish, all of which have been found to negatively impact native fish populations (Scoppettone et al. 2011). Crayfish compete with and prey upon native fishes (Light 2005) and mosquitofish likely aggressively compete with speckled dace for food (Caiola and Sostoa 2005), as well as being a known predator of fish larvae and eggs (Meffe 1985, Mills et al. 2004). Similar to many other desert aquatic habitats in the American Southwest, Saltcedar is proliferating and altering habitats in Amargosa Canyon (Scoppettone et al. 2011). Historically, stochastic events such as fire and flood periodically cleared large areas of riparian vegetation, keeping stream channels open and dynamic (Benda et al. 2003, Kozlowski et al. 2010). Today, these same processes serve as agents for the spread of saltcedar (Wiesenborn 1996), threatening to form a saltcedar monoculture throughout the floodplain (Scoppettone et al. 2011). Because saltcedar has a substantially greater water demand than native vegetation, increases in saltcedar density in the riparian zone result in a corresponding increase in water lost to transpiration (Duncan and McDaniel 1998).

	Rating	Explanation
Major dams	n/a	
Agriculture	High	Water withdrawals, both locally and in and around the Pahrump Valley, threaten flows in Amargosa River
Grazing	Medium	Livestock grazing in this arid region may disproportionately impact aquatic habitats
Rural residential	Medium	Residential water use contributes to reduced stream flows
Urbanization	High	Water demands from Las Vegas threaten aquifers which feed the Amargosa River
Instream mining	n/a	
Mining	Low	No known effects but present throughout region
Transportation	Low	Roads present; possible sources of increased sediment input
Logging	n/a	
Fire	Low	Fire can intermittently affect riparian habitats
Estuary alteration	n/a	
Recreation	Medium	Recreational use in the region is fairly high, including off-road vehicle use
Harvest	n/a	
Hatcheries	n/a	
Alien species	High	Competition/predation from mosquitofish, crayfish and bullfrogs could play a major role in species decline, while saltcedar can substantially alter aquatic habitats

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

Effects of Climate Change: The predicted impacts of climate change pose a direct threat to the continued existence of Amargosa Canyon speckled dace. The Amargosa River canyon exists in an exceptionally arid region and is fed by isolated desert springs and subsurface aquifer flow; this is a precarious ecosystem, vulnerable to geologic and anthropogenic disruption. Fed by rain and snow melt at high elevation in the desert mountain ranges, desert aquifers in the Death Valley region will likely receive less recharge as the region warms (Riggs and Deacon 2004). This decline in regional water supply will be compounded by growing human demand for water both locally and in southern Nevada, which will only increase as the climate gets hotter and more arid. Moyle et al. (2013) rated the Amargosa Canyon speckled dace as critically vulnerable to climate change effects.

Status Determination Score = 1.9 – Critical Concern (see Methods section Table 2). The Amargosa Canyon speckled dace is a Bureau of Land management Sensitive Species, is listed as Critically Imperiled by Natureserve (Natureserve.com) and Endangered by the American Fisheries Society (Jelks et al. 2008). These dace are highly vulnerable to extinction in the next 50 years, because they are restricted to a single desert stream system which is under threat of dewatering and invasion by non-native plants, fishes, crustaceans and amphibians (Table 1).

Metric	Score	Justification
Area occupied	1	Endemic to Amargosa Canyon
Estimated adult abundance	3	Highly fluctuating; probably low in dry years
Intervention dependence	3	Depends on protection of stream corridor and limited water removal
Tolerance	2	Exists near edge of thermal tolerances
Genetic risk	2	Single population
Climate change	1	Water withdrawals likely to increase and flows decrease
Anthropogenic threats	1	See Table 1
Average	1.9	13/7
Certainty (1-4)	2	Recent comprehensive survey of Amargosa Canyon

Table 2. Metrics for determining the status of Amargosa Canyon speckled dace, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: The Amargosa Canyon speckled dace needs immediate attention in order to prevent its decline and possible extinction.

1. Efforts should be made to ensure natural flow of water in Willow Creek and the Amargosa River, including occasional flood flows that reduce populations of alien fishes and saltcedar. Fortunately, most of the canyon area is now owned by the Nature Conservancy or administered by the Bureau of Land Management. Amargosa Canyon is part of a BLM Area of Critical Environmental Concern and is closed to off-road vehicle use. Fences and barriers need to be properly maintained, however, because vehicle trespass has been a common problem in the past. Increased law enforcement presence in the area would likely reduce illegal off-road vehicle use impacts.
2. An evaluation should be conducted in Willow Creek to determine if complete eradication of alien species from speckled dace habitat is possible. If not feasible, invasion-proof refuges for the species (and Amargosa pupfish) should be created within the drainage.
3. Minimum base flow requirements in Willow Creek through China Ranch should be established.

4. Efforts should be made to locate the spring where Amargosa dace were documented in 1937 (Miller 1938) to determine if this spring, or another nearby spring, could again support a dace population. Frequent surveys of Amargosa Canyon are necessary to monitor habitat conditions and the presence of alien fishes, crayfish, bull frogs and saltcedar.
5. Water removal from the aquifer(s) that apparently feeds the river is a pressing threat that needs further study. The U.S. Supreme Court decision that protected the Devils Hole pupfish from water withdrawals (United States v. Cappaert 1977) may have offset some impacts, but its utility on a larger, regional, basis is uncertain. Hydrological studies should be performed to evaluate relationships between Amargosa River flow and regional aquifers and to aid in the development of models to predict how various levels of pumping in different geographic areas might affect surface flow.

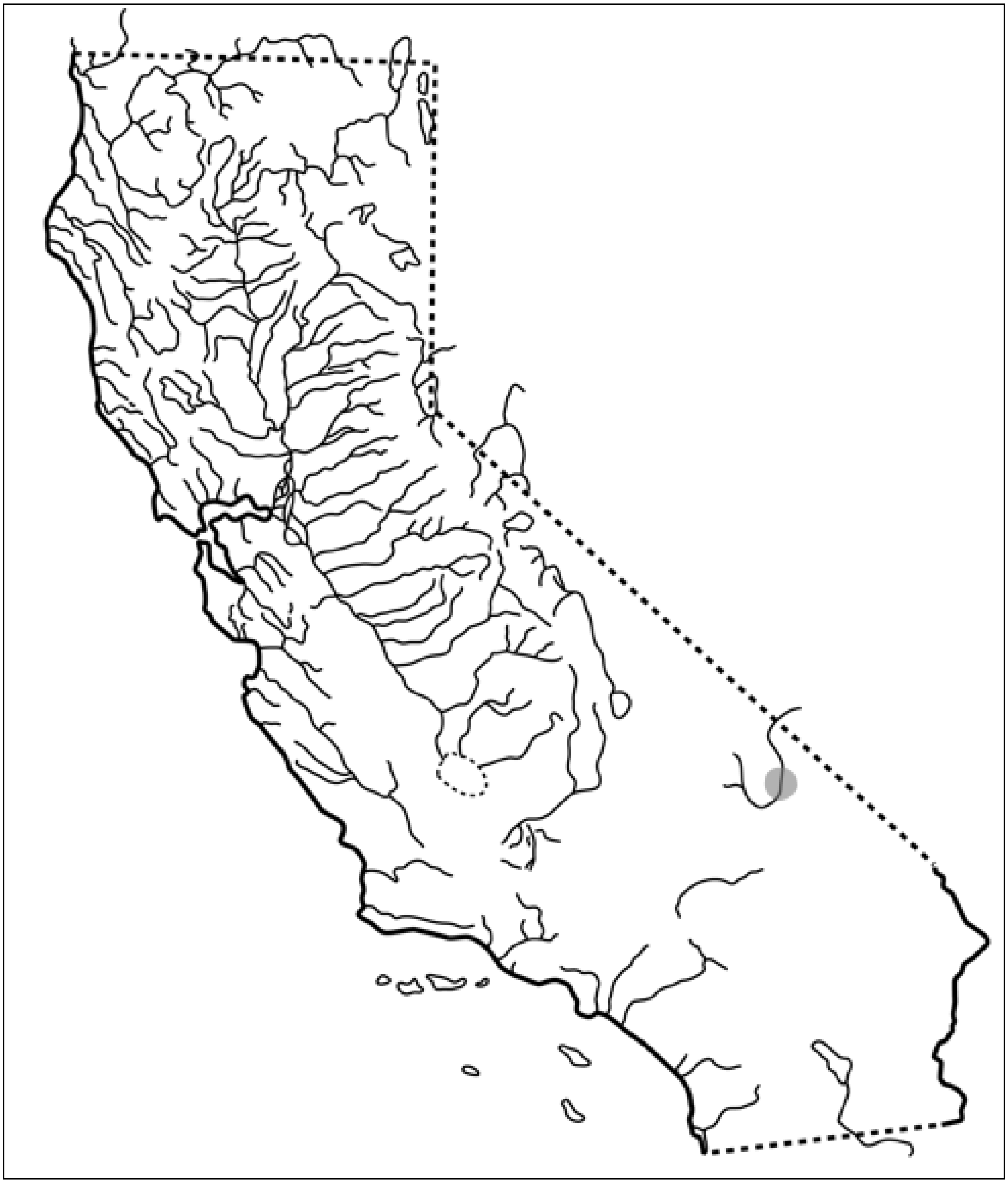


Figure 2. Distribution of Amargosa Canyon speckled dace, *Rhinichthys osculus* ssp., in the Amargosa Canyon area of the Amargosa River, California.

SANTA ANA SPECKLED DACE
Rhinichthys osculus ssp.

Status: Critical Concern. Santa Ana speckled dace are highly vulnerable to extinction within the next 50 years because their small, fragmented, populations are restricted to areas that are increasingly prone to catastrophic fire, debris flows, intensive water consumption, pollution, invasive species, and expanding urbanization and suburban development.

Description: Speckled dace are small cyprinids, usually measuring 8-11 cm SL (Moyle 2002). Although physically variable, they are characterized by a wide caudal peduncle, small scales (47-89 along lateral line) and pointed snout with a small, subterminal, mouth. Larvae have deep bodies, small eyes, overhanging snout and are characterized by 35-41 myomeres and distinctive coloration (Feeney and Swift 2008). Distinctive coloration in larvae includes large spots located on the sides of the bottom portion of the caudal peduncle and a wedge-shaped patch of spots on top of the head. Larvae have functioning eyes, mouth, and gas bladder by the time the notochord flexes at about 7-9 mm TL. A noticeable band of pigment running just below the lateral midline is visible at about 9 mm. The terminal mouth of larvae becomes subterminal at about 9.7 mm. The pectoral fins remain unpigmented until the later stages of larval development. Later stages also develop a distinctive spot on the base of the caudal fin. Scales appear when dace reach 13 mm FL (Jhingran 1948). Once fully developed, the dorsal fin usually has 8 rays and originates well behind the origin of the pelvic fins (Moyle 2002). The anal fin has 6-8 rays. Pharyngeal teeth (1,4-4,1 or 2,4-4,2) are significantly curved with a minor grinding surface. The maxilla usually has a small barbel at each end. The snout is connected to the upper lip (premaxilla) by a small bridge of skin (frenum). Most fish larger than 3 cm have distinctive dark speckles on the upper and sides of the body, a dark lateral band that extends to the snout, and a spot on the caudal peduncle. The rest of the body is dusky yellow to olive, with the belly a paler color. Breeding adults of both sexes have fins tipped by orange or red, while males also have red snouts and lips and tiny tubercles on the head and pectoral fins.

Taxonomic Relationships: The genus *Rhinichthys* is widely distributed and abundant in North America and has eight recognized species. However, most species are highly variable and may encompass complexes of unrecognized species or subspecies (Moyle 2002). Early taxonomists described different forms as separate species but later lumped them together when the variable nature of each species was discovered. For example, Jordan and Evermann (1896) described 12 separate species, which were later collapsed into a single species (Hubbs et al. 1974). However, subspecies, many of which were formerly recognized as full species, continue to be recognized on the basis of their location and isolation, provided formal scientific names exist for them. Although widely distributed, evidence continues to mount in support of the concept that isolated speckled dace populations throughout the west have long independent evolutionary histories, with distinctive adaptations to local environments. Relationships among various lineages still await resolution using modern molecular and morphometric techniques. In Oregon, speckled dace collected from five river basins exhibited high levels of divergence (0.82) among locations, and high genetic diversity (0.2, nucleotide diversity) within basins (Pfrender et al. 2004). Similarly, Oakey et al. (2004) found that speckled dace collected throughout the western United States were significantly different among sub-basins, consistent with the idea that local populations are characterized by long isolation from other populations. Based on the findings of their phylogenetic studies, Pfrender et al. (2004) proposed that populations within different basins should be considered to be Evolutionarily Significant Units for the purposes of management.

The Santa Ana speckled dace is one form that is thought to merit subspecies or, perhaps, full species designation due to its distinctive morphology (Cornelius 1969, Hubbs et al. 1979) and genetics. Although the subspecies has not yet been formally described, electrophoretic analysis supports the conclusion that this form is very different from other speckled dace (T.R. Haglund, University of California, Los Angeles, pers. comm. 1996, T. Metcalf, Calif. State Univ., San Bernardino, pers. comm. 2008). Initial mtDNA sequence analysis strongly suggests that Santa Ana speckled dace are genetically distinct from other speckled dace in California (T. Metcalf, unpubl. data, 2008). This form is apparently more closely related to speckled dace in the Colorado River basin than dace in northern California, as a result of a split in clades approximately 3.6 mya (Oakey et al. 2004, Smith and Dowling 2008). Furthermore, speckled dace in the Los Angeles basin (Santa Ana and San Gabriel rivers) were found to be only distantly related to those in the Owens or Amargosa rivers. Smith and Dowling (2008) indicate that populations of speckled dace in the Los Angeles basin have been isolated long enough (through the Pleistocene) to develop distinctive morphological characters. The long phylogenetic branch lengths associated with speckled dace from the Los Angeles basin suggests that these fish have undergone rapid molecular evolution. Oakey et al. (2004) also determined that speckled dace from the Santa Ana and San Gabriel rivers formed a monophyletic lineage.

Life History: Little has been published on the life history of Santa Ana speckled dace so this account is largely based on information from other dace populations. Their variability in body shape has allowed speckled dace to exploit a wide variety of habitats. The Santa Ana dace has a fairly streamlined body form (for a speckled dace) that indicates adaptation for living in flowing water (Moyle 2002). Although speckled dace are usually found in loose groups in appropriate habitats, such as rocky riffles, they avoid large shoals except while breeding. They can be active both day and night. Their activity is also mediated by stream temperature; they will remain active all year if stream temperatures exceed 4°C, which would be typical of streams inhabited by Santa Ana dace (Moyle 2002).

Speckled dace generally forage on small benthic invertebrates, especially taxa common in riffles, including hydropsychid caddisflies, baetid mayflies, and chironomid and simuliid midges, but will also occasionally feed on filamentous algae (Li and Moyle 1976; Baltz et al. 1982; Hiss 1984, Moyle et al. 1991). Their subterminal mouth, pharyngeal tooth structure, and short intestine are characteristic of small invertebrate feeders. Not surprisingly, diet varies according with prey availability and speckled dace, in general, prey opportunistically on the most abundant small invertebrates in their habitat, which may change with season. Speckled dace have been observed feeding by picking and grazing on cobbles in riffles and pool tail-out habitats in the East Fork San

Gabriel River (J. O'Brien, CDFW, pers. obs.). Preference of forage items may also be influenced by presence of other fishes that share similar habitats, such as sculpin or juvenile steelhead (e.g., Johnson 1985).

Length frequency analyses have determined age and growth patterns. By the end of their first summer, dace grow to 20-30 mm SL (Moyle 2002), growing an average of 10-15 mm/yr in each subsequent year. Females tend to grow faster than males. However, growth rates can decrease under extreme environmental conditions, high population densities, or limited food supply (Sada 1990). Slight changes in growth rates are also positively correlated with changes in temperature, as seen in the Colorado River (Robinson and Childs 2001). Life expectancy is approximately three years where maximum sizes do not exceed 80 mm FL, which is typical of Santa Ana speckled dace. However, in the upper San Gabriel River drainage dace over 110 mm SL are fairly common (J. O'Brien, CDFW, pers. obs.). Elsewhere, dace may reach 110 mm FL and live up to six years (Moyle 2002). Dace reach maturity by their second summer, with females producing 190-800 eggs, depending on size and location (Moyle 2002). Presumably, Santa Ana speckled dace are at the low end of this range, given their relatively small size. Spawning is generally associated with rising water temperatures and/or high flow events, suggesting that Santa Ana speckled dace most likely spawn in March-May. Spawning in lakes occurs primarily over shallow areas of gravel within the lake body itself or upstream in the edges of riffles of inlet streams. Groups of males will clear an area of algae and detritus and then surround a female when she enters the area. Females release eggs underneath rocks or near the gravel surface, while males release sperm (John 1963). Eggs settle into interstices and adhere to gravels. At temperatures of 18-19°C, eggs hatch in 6 days but larvae remain in the gravel for another 7-8 days (John 1963). Fry in streams congregate in warm shallow areas, often in channels with rocks and emergent vegetation. Santa Ana speckled dace were observed spawning within a pool during May, 2010 in Bear Creek, tributary to the West Fork San Gabriel River. A group of three to six males pursued and repeatedly attempted to spawn with several females at the head of the pool from 14:00-15:00 hours. Water temperature was 18°C, turbidity 1.5 NTU, velocity 0.8 m/s, and flow was 9 CFS. Seven days later, spawning activity was no longer observed but eggs were detected on the downstream bottom portion of a small boulder where the spawning activity occurred (J. O'Brien, CDFW, unpublished data).

Habitat Requirements: Santa Ana speckled dace are found mainly in perennial streams fed by cool springs that maintain summer water temperatures below 20°C (Moyle et al. 1995), although speckled dace in other regions of the west tolerate temperatures of 26-28°C. Surveys of trout streams in the Los Angeles basin found dace occupying shallow riffles dominated by gravel and cobble. Their habitat in the West Fork San Gabriel River was described as shallow (average depths of 15-30 cm), gravel-cobble dominated riffles with overhanging riparian vegetation (Deinstadt et al. 1990). Feeney and Swift (2008), however, characterized their preferred habitat as pools in low-gradient streams (0.5-2.5% slope) with sand to boulder substrates in slow-moving waters, noting that they were also found along stream edges by fast-moving water. O'Brien et al. (2011) observed dace in a wide variety of habitats, including riffles, runs, and pools in the San Gabriel River drainage.

Distribution: The ability of speckled dace to colonize new areas and adapt to different environments has resulted in their wide distribution. Speckled dace are the only native fish found in all major drainages in western North America. In California, their native range includes drainages in Death Valley (Amargosa River), Owens Valley, eastern Sierra Nevada (Walker River north to Eagle Lake), Surprise Valley, Klamath-Trinity basin, Pit River basin, including the Goose Lake watershed, Sacramento River basin, as far south as the Mokelumne River, San Lorenzo, Pajaro and Salinas River basins, San Luis Obispo, Pismo and Arroyo Grande Creek basins, Morro Bay, and the San Gabriel and Los Angeles basins (Swift et al. 1993).

Santa Ana speckled dace historically inhabited streams in the upland areas of the Santa Ana, San Gabriel and Los Angeles rivers systems (Moyle et al. 1995). They have since disappeared from many parts of their range, including the middle reaches of the Santa Ana River, Strawberry Creek (Santa Ana River), Mill Creek (Santa Ana River), and most of the Los Angeles River and San Jacinto River basins (Feeney and Swift 2008, G. Abbas, San Bernardino National Forest, pers. comm. 2008). Young-of-year and 2 year old fish were found in City Creek (Santa Ana River) in 2008, a location from which speckled dace were thought to have been extirpated (G. Abbas, pers. comm. 2008). Their current distribution is restricted to the headwaters of the Santa Ana and San Gabriel rivers and in Big Tujunga Creek (Los Angeles River drainage) (Moyle et al. 1995, O'Brien and Stephens 2009). A population was recently documented in Indian Creek, a headwater tributary of the San Jacinto River. Some fish were removed and held in captivity following the Esperanza Fire in 2006 to prevent total loss from flooding. They were reconfirmed as present in 2007 and 2008 (G. Abbas, pers. comm. 2008). Attempts to establish additional populations of Santa Ana speckled dace have been made through introductions into the Santa Clara and Cuyama rivers and into River Springs, Mono County. The introduction into the Santa Clara River is thought to have failed and the status of the other populations is uncertain.

Trends in Abundance: Population estimates for Santa Ana speckled dace were not found. However, their abundance is likely a small fraction of what it was in the past and populations have disappeared from two of five streams in which they were historically present (G. Abbas, pers. comm. 2008, Metcalf et al., unpubl. report). Perhaps eight populations remain in their native range, mostly small and isolated from one another. Moyle et al. (1995) declared their numbers so diminished that they were in danger of extinction.

Swift et al. (1993), Moyle et al. (1995), Abbas (pers. comm. 2008) and O'Brien (unpublished observations) indicate the status of specific populations by location, detailed below.

Big Tujunga Creek (Los Angeles River). Dace once inhabited this creek for 10-20 km below Big Tujunga Dam and were thought to be extinct due to drought conditions and establishment of red shiner (*Cyprinella lutrensis*) (Moyle et al. 1995). Red shiners directly compete for food and space with dace and prey on dace eggs. Surveys performed from 2002-2005 found a few (in the 10s) speckled dace at this and other locations in the Los Angeles River basin (Tujunga Wash, Haines Canyon, G. Abbas, pers. comm. 2008).

Surveys by CDFW indicate that dace populations have rebounded since the 2009 Station Fire and dace are common within the Tujunga Wash (O'Brien and Stephens 2009).

Fish Canyon (San Gabriel River). This population was thought to be extinct (Moyle et al. 1995). Only 6-7 fish were seen in 1988. Optimal dace habitat has been infringed upon by a rock quarry operation. However, current quarry operations are focused on restoring the streambed in order to improve dace habitat (G. Abbas, pers. comm. 2008). A few individuals were collected from this site in 2007 by ECORP. Morris dam isolates this population from other dace in the San Gabriel River, preventing genetic flow and recruitment between populations. Some recent (2002, 2006 and 2007) surveys established their presence in this location, while others did not (2005; G. Abbas, pers. comm. 2008). California Department of Fish and Wildlife surveys in 2006 and 2008 found that dace occupied a 0.8 km section of stream within the Angeles National Forest (O'Brien 2006, 2008). The U.S. Forest Service was provided specimens by rock quarry consultant and these are being analysed by Anthony Metcalf at California State University, San Bernardino, to determine genetic relationships (G. Abbas, pers. comm. 2008).

West, North, and East Fork San Gabriel River. These areas constitute the best remaining Santa Ana speckled dace habitat (Moyle et al. 1995). Populations in the West Fork in 1990 likely numbered less than 2000 (Deinstadt et al. 1990). Habitat in the West Fork is vulnerable to high water and sediment releases from Cogswell Reservoir which is managed for flood control. As of 1995, the West Fork was still recovering from major sediment releases from 1981 and 1991. These sediments buried most of the habitat used by dace until they were flushed out by rainfall and dam water releases in 1988. Multiple-pass electrofishing surveys performed in the West Fork in 1993 found 29 dace in a 68 m section of stream (Moyle et al. 1995). Surveys in 2006 found dace in only one of three locations sampled (G. Abbas, pers. comm. 2008). Dace were also abundant upstream of Cogswell Reservoir in 2005. Surveys (2005) of the North Fork found dace in one of the two days of sampling. Suveys (2005) also documented the presence of 100s of speckled dace in Cattle Creek (G. Abbas, pers. comm. 2008). Multiple-pass electrofishing surveys performed by CDFW in the middle portion of the East Fork (Heaton Flat and Shoemaker Canyon) between 1997-2010 indicated an average estimated density (fish/mile) of Santa Ana speckled dace as follows (years in parentheses): 2,143 fish/mile (1997); 4,113 fish/mile (2000); 4,640 fish /mile (2010) (Weaver and Mehalick 2010). A comprehevive survey of the upper San Gabriel River from 2007-2008 found that dace occupy 4.5 km within the North Fork, 19.5 km in the East Fork, and 20 km in the West Fork (O'Brien et al. 2011).

Santa Ana River. Speckled dace are assumed to be extirpated from most of the Santa Ana River (Moyle et al.1995, Moyle 2002). They were last seen near Rialto in 2001 (G. Abbas, pers. comm., 2008). Only a few specimens (usually <4) were documented in the mainstem in 2000 (Swift 2001) and 2005 (San Bernardino National Forest in G. Abbas, pers. comm. 2008). Recent surveys suggest that their distribution in the basin is largely limited to small areas in headwater streams, as follows:

Lytle Creek (mainstem). "The stronghold area for Santa Ana speckled dace is currently in the mainstem reach from Miller's Narrows downstream to Turk Point (approximately 1.4 river miles). The Forest Service has qualitatively monitored this reach since at least 1999. Santa Ana speckled dace have been there throughout this period with significant population fluctuations in response to drought induced low flows, major flooding (periods of declining population densities), and a couple years of sustained moderate flows (period of rapid population density increases). This reach is currently the species stronghold for this watershed, being the only place where they have persisted, and must be diligently protected from disturbance and enhanced at every opportunity. This stronghold reach is regularly threatened by encroachment into the wash by heavy equipment by a variety of forest users to protect infrastructure including public roads, public utilities and private access routes. There is perennial water above Miller's Narrows in the mainstem up to the confluence with the Middle Fork, but dace were absent here between at least 1999 and 2005.

In 2005, the Forest Service, CDFG [CDFW], and Fontana Union Water Company Consultant Jonathan Baskin conducted a reintroduction of approximately 1000 Santa Ana speckled dace from the lower mainstem of Lytle Creek to the Applewhite Picnic Area on the North Fork of Lytle Creek. In 2007 Southern California Edison reported capture of Santa Ana speckled dace in their diversion works above Miller's Narrows suggesting that some of the fish from the North Fork reintroduction had survived and migrated downstream (3.2 river miles) to this location. With this information we can now consider all of the mainstem above Turk Point occupied by Santa Ana speckled dace.

In 2007, the Forest Service and CDFG [CDFW] conducted a translocation of approximately 1300 Santa Ana speckled dace from the lower mainstem of Lytle Creek to the North Fork at Applewhite Picnic area.

In 2005 and 2006, sustained year-round flows from Turk point down to the Fontana Union Water Company (FUWC) diversion (1.8 river miles) resulted in an expansion of the speckled dace population throughout the reach with juveniles rearing in the settling pond at FUWCs intake structure (adults were also noted in the raceways of the intake structure). With this knowledge we can assume that in years when there are flows below Turk Point, that reach of the mainstem is occupied by Santa Ana speckled dace." (Excerpt from Abbas 2008).

Lytle Creek (forks). The Middle Fork Lytle Creek is a high quality water source for this watershed, consistently producing perennial waters over a 3.2 km reach. Beginning near the national forest boundary with the community of Scotland, there is a reach of the South Fork mapped as intermittent for approximately 0.6 river miles. However this reach has rarely gone dry and also supports a popular trout fishery. In 2007, the U.S. Forest Service and CDFW conducted a reintroduction of approximately 500 Santa Ana speckled dace from the lower mainstem of Lytle Creek to the Middle Fork just a few hundred meters upstream of the Scotland boundary. Surveys by the U.S. Forest Service confirmed the persistence of these fish as of 2008, but no assessment of their movement from the introduction point was conducted. There is high quality habitat available from the forest boundary with Scotland upstream at least 4.1 river km. There are no significant fish passage barriers known within this reach, so the full 4.1 km reach above Scotland is now considered Santa Ana speckled dace habitat.

Cajon Creek (tributary to Lytle Creek). Dace appear to be abundant in this drainage, predominantly congregated upstream and downstream of Interstate 15 (Moyle et al. 1995). Their presence was also documented by surveys in 2005 (G. Abbas, pers. comm. 2008). There have been several recent fires in the area. Hazardous waste spills from trucks and trains using the transportation corridor threaten aquatic habitat in this watershed. The San Bernardino National Forest, CDFW, and BNSF Railroad have been moving fish into headwater tributaries to protect them from highway or railway spills.

City Creek. Dace were seen in September, 2003, following the Bridge Fire. No dace were found following the devastating Old Fire (October, 2003) and subsequent flooding. Several surveys were conducted in 2005, 2006, and 2007 and no dace were found. However, in 2008, a small population was found in the West Fork and reconfirmed in 2009.

East Fork City Creek. Dace were observed immediately after the Bridge Fire in 2003 (G. Abbas, pers. comm. 2008). Fewer dace were seen after the Old Fire in October, 2003, and none after subsequent flooding in December, 2008. No dace were observed in 2004, 2005, 2006 and 2007.

Mill Creek. Dace were found here in the 1980s, but not in 1990 (Moyle et al. 1995). They were thus thought to be extirpated (Moyle et al. 1995), but a few were observed in a small pool created by a human-made grade control structure in 2007 (G. Abbas, pers. comm. 2008). However, dace were not seen in 2008 and are now assumed to be extirpated from Mill Creek (G. Abbas, pers. comm. 2008).

Plunge Creek. Speckled dace were observed in 2001 (9 individuals) and 2005 (G. Abbas, pers. comm. 2008). Dace were collected in 2004 to protect them from potential flooding. They were returned to the stream after the threat of flooding passed (G. Abbas pers. comm. 2008).

Silverado Canyon (Santa Ana watershed). Although dace were found in 1987, none were found in the same or nearby areas in 1990 (Moyle et al. 1995) or 2005 and 2007 (G. Abbas, pers. comm. 2008, J. O'Brien 2007).

Santiago Creek (Santa Ana River tributary). Surveys in 2005 did not find speckled dace within the mainstem or tributaries (Harding Canyon Creek, Silverado Creek; G. Abbas, pers. comm. 2008, J. O'Brien 2006-2009).

San Jacinto River. Dace were recorded in 15-30 km of stream but not since the mid-1980s (T. Haglund, in Moyle et al. 1995). Large portions of the river and the lower portion of its tributaries are now dry in the summer. Surveys in 2005 did not find speckled dace in the mainstem or in the North and South forks (G. Abbas, pers. comm. 2008).

Strawberry Creek (tributary to San Jacinto River). A small population was found in 1992 by the U.S. Forest Service (C. Swift pers. comm. in Moyle et al. 1995). Surveys did not detect dace in 2005 or 2006 (G. Abbas, pers. comm. 2008). Several surveys following the 2003 Old Fire and Christmas Flood did not find dace. They are presumed extirpated from Strawberry Creek.

Indian Creek (tributary to San Jacinto River). In 2006, Santa Ana speckled dace found in Indian Creek were relocated to the Riverside-Corona RCD for captive breeding after the Esperanza Fire (G. Abbas, pers. comm. 2008). Fish survived the fire and the population is recovering. The population has been able to sustain itself following the fire due to the lack of large flood events.

Nature and Degree of Threats: Threats to the Sana Ana speckled dace and their habitats include: 1) dams and diversions; 2) habitat loss and degradation, especially factors associated with urbanization; 3) grazing; 4) agriculture; 5) mining; 6) recreation;

7) wildfires; and 8) alien species (Swift et al. 1993, Moyle et al. 1995, Moyle 2002, Swift 2005; see Table 1).

Dams and diversions. Virtually all Santa Ana speckled dace streams contain one or more dams and diversions, so flows are generally depleted and natural flow regimes altered. Cogswell Dam on the West Fork San Gabriel River and Big Tujunga Dam on Tujunga Creek are particularly problematic for speckled dace because they block movement of fishes and capture large amounts of sediment, which often bury preferred habitats when released from the dam.

Agriculture. Agriculture is a greatly reduced threat from the past because much of the agricultural land in the Santa Ana speckled dace's range has been urbanized. However, runoff from remaining dairy and citrus operations is a source of pollution in some streams.

Urbanization. Most portions of the Los Angeles, Santa Ana and San Gabriel rivers not in public lands are highly urbanized. Extensive river channelization and impoundment has occurred in the middle and lower reaches of all rivers for flood control. These alterations result in the loss of ecological value by changing streams from riparian corridors to canals. Urbanization has also caused water quality degradation in these rivers. The Los Angeles River (lowest reach) is identified as impaired for pH, ammonia, lead, coliform, trash, scum algae, total dissolved solids and turbidity (http://ceres.ca.gov/wetlands/geo_info/so_cal/los_angeles_river.html).
The State Water Resources Control Board lists sections of the Santa Ana River as impaired by heavy metals, pathogens, bacteria, and nutrients (www.waterboards.ca.gov). The board also lists sections of the San Gabriel River as impaired by bacteria, pH, and heavy metals (lead, copper). While water quality impairment is of concern in portions of the Santa Ana speckled dace's occupied range, it is important to note some of these areas (e.g., the lowest reach of the Los Angeles River, which is an extremely altered concrete-lined channel) are no longer suitable habitat for most fishes, regardless of water quality issues.

Mining. Speckled dace in Cattle Creek (tributary to the East Fork San Gabriel River) may be adversely influenced by mining operations (Moyle et al. 1995). A rock quarry in Fish Canyon is encroaching on speckled dace habitat. However, the mining company is in the process of restoring fish habitat. Suction dredging in San Gabriel and Cajon Wash and Lytle Creek may have negatively affected habitats used by dace and other aquatic species; however, suction dredging is currently banned in California streams and the CDFW is prohibited from issuing dredging permits for an indeterminate period of time (http://www.dfg.ca.gov/suctiondredge/).

Transportation. The watersheds occupied by speckled dace have some of the highest road densities in California, due to intense urbanization in southern California. Roads exist along most speckled dace streams and impacts likely include increased siltation, pollutant input (chemical and solid (trash) wastes), as well as barriers to upstream movement. Non-paved U.S. Forest Service and other roads in the mountainous areas are also of concern, given the friable soils in this region that easily erode into streams as well as their facilitation of access for intensive human recreational use.

Fire. Fire frequency, duration and intensity have increased in southern California in the last 20+ years, increasing the risk of debris torrents and landslides. Recent wildfires and subsequent debris torrents in southern California destroyed speckled dace habitat in 2004, 2006 and 2008 (Metcalf et al. unpubl. data, G. Abbas, pers. comm. 2008). Predictions are that fire frequency, intensity, and duration will continue to increase over the next century, due to increasing temperatures and changes in precipitation patterns (Fried et al. 2004, Lenihan et al. 2008, Westerling and Bryant 2008). Catastrophic fires can accelerate the delivery of fine sediment to streams, thereby degrading the permeability of stream substrates. Fires also remove riparian vegetation along streams, increasing the amount of solar radiation input into streams and, correspondingly, water temperatures. Streams scoured during flood events after large fires generally cannot be reoccupied by natural upstream movement due to barriers (natural and artificial), stream channelization, and other factors that have altered the lower portions of nearly all rivers occupied by dace.

Recreation. Heavy recreational use in streams, including camping, dam building for waterplay, swimming, and off-road vehicle use, may displace fish from optimal habitats and further stress fish in suboptimal habitat. Swift (2003) expressed concern over the impacts of recreational activities on fish populations in the Santa Ana River as did O'Brien in the San Gabriel River (O'Brien et al. 2011). There is also concern over recreational impacts in Lytle and Mill creeks. Large portions of the San Gabriel River drainage are heavily utilized for water play, swimming, and bathing; many artificial impoundments have been built to facilitate these activities, leading to fragmentation of dace habitats (J. Weaver, CDFW, pers. obsv. 2009).

Alien species. Alien fish species are common in the reservoirs and highly altered stream reaches of the Los Angeles, Santa Ana and San Gabriel rivers. Brown trout (*Salmo trutta*), hatchery-stocked rainbow trout (*Oncorhynchus mykiss)* and red shiners can directly compete with or prey on speckled dace (Moyle et al. 1995, http://www.dfg.ca.gov/news/stocking). Bass (*Micropterus* spp.) may also prey on native cyprinids and are present in Tujunga Creek below Tujunga Dam (O'Brien pers. obs. 2012). Bullfrogs (*Lithobates catesbeiana*) and alien crayfishes may also prey on dace at various life stages. Alien aquatic vegetation can also reduce the quality of speckled dace habitat. Giant reed (*Arundo donax*) has altered aquatic habitats in some sections of the Santa Ana River so that it is no longer suitable for native fishes, including speckled dace (Bell 1997). Stream reaches where giant reed dominates the riparian vegetation are characterized by increases in pH and ammonia and decreases in dissolved oxygen. Although efforts are underway to remove *A. donax* from many streams in southern California, it is very difficult to remove (http://www.smslrwma.org/invasives/Arundo/ADRegionalMap.html) and is present in all watersheds where Santa Ana speckled dace are found.

	Rating	Explanation
Major dams	High	Dams and reservoirs are found in all major drainages
Agriculture	Medium	Agricultural runoff is a source of pollution in some streams but most areas are now urbanized
Grazing	Low	Present at low intensities in some watersheds
Rural Residential	Low	High historical threat; now greatly reduced since dace are now confined to upper portions of watershed, often on public lands
Urbanization	Medium	Urbanization and suburbanization has degraded watersheds containing dace; much higher historical threat as dace have already been largely eliminated from urban stream reaches and are now confined to upper portions of watershed, often on public lands
Instream mining	Low	Mining activities can displace dace from preferred habitat; effects mostly localized
Mining	Low	Rock quarry in Fish Canyon is encroaching on habitat
Transportation	High	Roads exist along most speckled dace streams, negatively affecting habitats through pollution and sediment inputs, along with channel constriction and barriers to instream movement
Logging	Low	Forest thinning and other practices require roads; most are unimproved, serve as sources of sediment input and provide corridors for recreational access
Fire	High	Fire frequency, duration and intensity are increasing, resulting in more frequent debris torrents and landslides
Estuary Alteration	n/a	
Recreation	Medium	Heavy recreational use may displace and stress fish as well as fragment and alter habitats
Harvest	n/a	
Hatcheries	n/a	
Alien species	High	Alien aquatic species and invasive giant reed and tamarisk threaten most populations

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

Effects of Climate Change: The most noticeable and widespread impacts of climate change on aquatic habitats in southern California will be continued increase in water temperatures and changes to the timing, frequency and duration of drought and flooding events. Water temperatures will increase by approximately 0.7°C by 2099, based on conversion factors developed by Eaton and Scheller (1996). Although this increase is seemingly small (and is probably an underestimate), it may be significant to fish already exposed to summer temperatures above 20°C. For example, elevated temperatures may stress fish so that autoimmune function is repressed, making them more susceptible to disease. White spot disease infections have already been detected in speckled dace collected from the East Fork San Gabriel River (Warburton et al. 2001).

Elevated air temperatures associated with climate change will change the periodicity and magnitude of peak and base flows in streams. Predictions are that stream flow will increase in the winter and early spring and decrease in the fall and summer (Knox and Scheuring 1991, Field et al. 1999, Stewart et al. 2004, Stewart et al. 2005, CDWR 2006, Knowles et al. 2006). Hydrographs that mimic natural flow regimes more closely may actually benefit speckled dace populations, as their populations can reestablish themselves faster than those of alien fish species (Gido et al. 1997, Valdez et al. 2001, Propst and Gido 2004). However, decreases in summer base flows may further isolate speckled dace populations. Dace in Cajon Creek, North Fork Lytle Creek, West Fork City Creek, Silverado Canyon and the San Jacinto River become isolated by the presence of dry stream reaches during most of the year, preventing repopulation and genetic mixing between stocks (Moyle et al. 1995). Fire frequency, intensity and duration will almost certainly increase in southern California over the next century due to increasing temperatures and changes in precipitation patterns (Fried et al. 2004, Lenihan et al. 2008, Westerling and Bryant 2008), further threatening the stability and quality of speckled dace habitats. Moyle et al. (2013) considered Santa Ana speckled dace to be "critically vulnerable" to the effects of climate change.

Status Determination Score = 1.6 - Critical Concern (see Methods section Table 2). The Santa Ana speckled dace is in danger of extinction in the next 50 years (Table 2). Santa Ana speckled dace are a Region 5 U.S. Forest Service Sensitive Species in the three southern California national forests to which they are native (Angeles, San Bernardino, Cleveland). However, it is now considered extirpated from the Cleveland National Forest (Santiago Creek; see above). The California Natural Diversity Database ranks speckled dace as secure at the global scale (G5) but the Santa Ana subspecies as Imperiled (T1S1; www.natureserve.org). It is considered Threatened by the American Fisheries Society (Jelks et al. 2008). Listing of the Santa Ana speckled dace under the Federal Endangered Species Act was denied after being petitioned in 1994 because it had not been described as a separate taxon from other speckled dace in the southwest (61FR4722). Recent genetic and phylogenetic studies show this form to be distinct from other speckled dace in California (Oakey et al. 2004, Smith and Dowling 2008, Metcalf, unpubl.), so it merits further taxonomic investigation and publication of findings in the peer-reviewed literature. Extinction is likely unless special protection is afforded (Swift et al. 1993, Moyle et al. 1995, Moyle 2002). The range of this form has been dramatically diminished due to urbanization in the Los Angeles region, resulting in fragmentation and reduction of populations.

Metric	Score	Justification
Area occupied	3	Distribution restricted to the headwaters of the Los Angeles, Santa Ana and San Gabriel rivers
Estimated adult abundance	2	Most populations are small (50-100 individuals)
Intervention dependence	1	Captive breeding and intensive management of streams is needed
Tolerance	2	Appears to need cooler water than most speckled dace; water quality conditions now often exceed tolerances of speckled dace in general
Genetic risk	1	Small, isolated populations are vulnerable to genetic drift and bottlenecks
Climate change	1	Because most populations are in small streams in an already dry region there is extreme range-wide vulnerability to climate change
Anthropogenic threats	1	Many factors with high degree of threat (Table 1)
Average	1.6	11/7
Certainty (1-4)	3	Information is largely from experts who work closely with Santa Ana speckled dace

Table 2. Metrics for determining the status of Santa Ana speckled dace, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: Endangered species listing of this distinctive dace at the federal level has been denied for reasons of taxonomy, so a high priority for the Santa Ana speckled dace is to have it formally recognized as a distinct taxon. Beyond that, funding should be secured to complete genetic and morphological studies and to publish the results in peer-reviewed literature.

A multi-pronged recovery process is needed to prevent the extinction of Santa Ana speckled dace, including:

- Provide special protection for all streams and watersheds occupied by speckled dace and from which Santa Ana speckled dace have been observed in the past 20 years.
- Establish a rigorous (annual) monitoring program for streams that are known to contain dace in order to generate population estimates and trend data. Establish a periodic (every 3-5 years) monitoring program for streams in which dace have been observed within the past 20 years.
- Develop a Santa Ana speckled dace recovery plan to prioritize stream restoration and other actions needed to sustain dace populations (see recommendations below).

Stream restoration. Stream channelization, impoundments and pollution have degraded aquatic habitats in the middle and lower reaches of all major southern Californa drainages. Efforts (e.g., Los Angeles River Revitalization Project) should be made to reestablish the ecological function of these streams. Associated actions may include daylighting (unearthing and removing river channels from underground canals), riparian planting, volunteer stream clean up projects, and managing flows to mimic natural flow regimes. Water treatment should be improved for any water discharged into streams. The reestablishment of wetlands along streams could help meet multiple goals by removing toxins, increasing seasonal water retention, and increasing the biodiversity of riparian corridors.

Captive breeding program. A captive breeding program should be evaluated for potential benefits to dace and include assessments of possible donor stocks from multiple populations, following a genetic management plan.

Reintroduction of extirpated populations. Streams where dace have been lost due to flooding and other causes should be considered for repopulation from captively reared fish or fish salvaged or captured from adjacent healthy populations. Many of the lower sections of streams become mostly dry in the summer, leaving dace isolated in pools. These fish can be (and have been) moved to upstream areas in order to repopulate areas where they have been flushed out or extirpated due to other causes.

Expand range into suitable habitats with barriers. Expanding the range of the species in historically and currently occupied drainages is most desirable (e.g. South Fork of Lytle Creek, North and East Fork San Gabriel River, Cajon Wash, etc.), but transplants into streams outside the range should be considered if they provide suitable habitat and impacts to other native fishes and invertebrates are deemed minimal.

Manage wildfire. Measures such as brush thinning and prescribed burning should continue to be used and monitored to determine their efficacy in minimizing impacts from wildfire in national forests (Keeley 2002, Keeley et al. 2004). Prescribed burning confined to small areas can reduce the chances of inadvertently burning entire watersheds, minimize fire intensity, and decrease the potential for flooding and debris flows in subsequent years.

Minimize impacts of alien species. Alien fish planted into reservoirs should be prevented from moving into streams, possibly with the use of screens or gates. Bass, brown trout and red shiners should, where feasible, be eradicated from streams where they are established. Efforts should continue to remove giant reed throughout the range of Santa Ana speckled dace.

Minimize impacts from agricultural and urban areas. Settling ponds, wetlands, or other wastewater treatment options should be used where point sources of pollution are identified. Living riparian buffers (riparian vegetation, wetlands) should be maintained or restored along stream channels in order to mitigate the effects of nonpoint source pollution. The use of chemicals (e.g., pesticides for mosquito abatement) that are nontoxic to vertebrates should be used wherever possible.

Limit recreational use. In heavy use areas, recreational use of speckled dace streams should be limited until dace populations recover. Area closures should be implemented if necessary to protect habitat integrity. Priority should be given to the development of educational programs and volunteer stream restoration projects in order to minimize the need for area closures.

Minimize impacts from grazing. If grazing is permitted in speckled dace watersheds, grazing allotments should be closely monitored before, during and after use. Grazing permits should include requirements that reduce impacts to streams and riparian corridors (through the establishment of herd size limits, strict timing and duration of grazing periods, management and enforcement of allotment boundaries and monitoring of landscape response metrics such as minimum vegetative cover, vegetation height, etc).

Alternative (out of stream) water sources should be provided and streams fenced where they are heavily impacted by livestock.

Evaluate mining impacts. Recreational and commercial mining should be evaluated and, where negative impacts are identified, eliminated within the range of Santa Ana speckled dace. The current ban on suction dredging in California should be assessed to determine if, and to what extent, native aquatic fauna and their habitats are benefitting from this protective measure. It is worth noting that recreational mining is increasing in popularity (especially with recent spikes in the value of gold) and occurs in many locations where suction dredging formerly occurred; therefore, impacts are ongoing and benefits from the ban on suction dredging may be difficult to detect in this densely populated region.

Manage conservatively in the face of climate change. The predicted impacts from climate change will exacerbate all existing threats to Santa Ana speckled dace. Fishes native to southern California watersheds are likely to experience severe impacts, given the already hot and arid nature of the mostly desert streams they occupy, coupled with intense urban and suburban expansion in the region. Climate change models should be employed with respect to forecasting and development of long-term best management practices to protect cool water habitats in southern California streams.

Figure 1. Generalized distribution of Santa Ana speckled dace, *Rhinichthys osculus* ssp., in California. Actual distribution is highly fragmented.

OWENS SUCKER
Catostomus fumeiventris (Miller)

Status: Low Concern. While relatively secure, with a high degree of confidence about their status, the limited distribution of Owens sucker in a highly altered water system indicates a need to continue monitoring populations.

Description: Owens suckers are stout, with large heads, large mouths, long snouts and wide caudal peduncles. Their mouths are characterized by a deeply incised lower lip. Their scales are coarse and usually number less than 80 (66-85) along the lateral line. Rows of scales are 13-16 above and 9-11 below the lateral line. Ray fin counts are 16-19 for the pectoral, 10 for the dorsal, and 9-10 for the pelvic fins. Their coloration can be very dark but is normally slate on the dorsal surface and smoky on the belly, with blue iridescence on the sides. Spawning adults develop red coloration in a line along their sides and at the tips of the paired fins.

Taxonomic Relationships: The Owens sucker was first described as a population of sandbar suckers (*C. arenarius*; Snyder 1919), a species considered to be the same as the Tahoe sucker (*C. tahoensis*). However, they were recognized as a distinctive taxon by C. L. Hubbs in 1938 (Shapovalov 1941) and formally described as a species by R. R. Miller (1973). They are closely related to the Tahoe sucker (G. R. Smith 1992) and are able to hybridize with the Santa Ana sucker (*C. santaanae*; Hubbs et al. 1943, Crabtree and Buth 1981, Buth and Crabtree 1982).

Life History: Owens and Tahoe suckers share similar life history traits (R.R. Miller 1973, Moyle 2002). They feed largely at night, ingesting aquatic insects, algae, detritus and inorganic material from stream substrates. Spawning takes place in the spring and summer, from early May to early July. In Crowley Reservoir, Owens suckers spawn, sometimes in large numbers, along the shore in springs and gravel patches or in tributary streams (C.C. Swift, pers. comm. 1999). In May, 1975, large numbers (500-1000) of spawning adults were seen in a 200 m section of Hilton Creek, while smaller numbers were seen in the reservoir at 1-2 m depth. Larvae transform into juveniles at 19-22 mm TL, then move into margins or backwaters that are dominated by sedges (Miller 1973). Growth rates are not known but they rarely grow larger than 50 cm SL.

Owens suckers have the ability to sustain populations in altered environments, by quickly repopulating new habitats and withstanding the presence of nonnative fish species. In 2007, they were the first fish species to recolonize a 35 mile stretch of the lower Owens River upon rewatering of the stream channel (M. Hill, Ecosystem Sciences, pers. comm. 2009). Although the lower Owens River had gone dry due to water diversion, populations of Owens suckers were able to survive in small lakes and ponds and other off-channel habitats. These populations were able to disperse into new habitats once flows were reestablished. Once the only fish in Convict Lake, they are now found with alien trout species (Moyle 2002). They are also found together with brown trout (*Salmo trutta*) in pools in the Owens River Gorge (S. Parmenter, CDFW, pers. comm. 2009). However, trout maintain position in stream currents, while suckers are associated with the stream bottom. Likewise, they are found in the lower Owens River with introduced bass (*Micropterus* spp.), but bass are generally found in slightly cooler and faster water than are suckers (M. Hill, pers. comm. 2009). Suckers in the lower river appear to withstand higher temperatures and lower dissolved oxygen concentrations than introduced bass (M. Hill, pers. comm. 2009).

Habitat Requirements: Owens suckers, in the Owens River and two of its tributaries, Hot Creek and Rock Creek, are most common in stream reaches with long runs and few riffles (Deinstadt et al. 1986). Habitat in these reaches is characterized by fine substrate with lesser amounts of gravel and cobble, water temperatures of 7-13°C, and pH of 7.9-8.0. In lakes and reservoirs, such as Convict Lake and Crowley Reservoir, adults are abundant near the bottom, regardless of depth. Adult suckers (> 15 cm) were also commonly found at the bottom of pools in a 10 mile reach of the Owens River Gorge (CDFW snorkel surveys 2008; S. Parmenter, CDFW, pers. comm. 2009). Recent surveys in the lower Owens River found suckers predominantly in off-channel habitats, such as backwaters (M. Hill, pers. comm. 2009).

Distribution: Owens suckers are an endemic species that are widely distributed in streams and rivers of the Owens River watershed, including the Owens River and Bishop Creek. They are most abundant in Crowley Reservoir (Mono County) and are also found in Convict Lake (Mono County) and Lake Sabrina (Inyo County). They were successfully introduced into June Lake (Mono Lake Basin), the Santa Clara River (Los Angeles County), and South Lake (Bishop Creek drainage) (Moyle 2002; S. Parmenter, CDFW, pers. comm. 2009). In the Santa Clara drainage, they are found in lower Sespe Creek, Piru Creek and Piru Reservoir ('Lake Piru'), and the outflow from Fillmore Trout Hatchery (Swift et al. 1993).

Trends in Abundance: Owens suckers are still abundant in most of their range, primarily due to their ability to adapt to life in Crowley Reservoir as well as the highly modified Owens River (Moyle 2002). Their populations in the river have increased as a result of restoration activities begun in the early 1990s (S. Parmenter, CDFW, pers. comm. 2009). In 1993, flows released from the Crowley Reservoir were increased as a result of a court decision in 1991. Improved flow management in the 'middle' Owens rewatered formerly dry reaches in the Owens River Gorge and allowed reestablishment of aquatic and riparian habitats, restoring ecosystem function to the benefit of fishes and other aquatic organisms (Hill and Platts 1998). In October, 2008, snorkel surveys of a 10 mile reach in the Owens Gorge found large numbers of Owens suckers and brown trout, often sharing the same habitats (S. Parmenter, CDFW, pers. comm. 2009). Owens suckers appear to be well established in the upper Owens River, as both juveniles and adults are commonly collected there (S. Parmenter, CDFW, pers. comm. 2009). They also appear to be well established in the lower Owens River (lower 60 miles), where they dominate species composition (M. Hill, pers. comm. 2009). Surveys in the lower river commonly find aggregations of more than 100 individuals (M. Hill, pers. comm. 2009).

Nature and Degree of Threats: Two factors may be limiting Owens sucker abundance in their range: 1) habitat degradation associated with dams and water diversions; and 2) invasive trout and bass species.

	Rating	Explanation
Major dams	Medium	Two large dams and aqueducts regulate flows in the Owens River
Agriculture	n/a	
Grazing	Low	Grazing is pervasive in the Owens River basin but few known effects on suckers
Rural residential	n/a	
Urbanization	Medium	The Owens River is a major source of water for expanding Los Angeles urban area; potential localized impacts from the town of Bishop
Instream mining	n/a	
Mining	n/a	
Transportation	n/a	
Logging	n/a	
Fire	Low	Present but with no know impacts on suckers
Estuary alteration	n/a	
Recreation	Low	Potential impacts from OHVs; effects likely minimal
Harvest	n/a	
Hatcheries	Low	Extirpated from a spring that feeds the Hot Creek Trout Hatchery
Alien species	Medium	Predation by alien species may limit abundance, but may be offset by differences in habitat utilization

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

Dams. The Owens River has been highly altered by dams and diversions, although it still supports large numbers of suckers, as does Crowley Reservoir. Owens suckers have benefited from rewatering of the upper and lower Owens River, but flows are still a fraction of what they once were. Sustained flows (40 cfs) in the lower River are approximately 5% of the river's natural capacity (S. Parmenter, CDFW, pers. comm. 2009). Groundwater extraction and surface diversion have lowered the water table and reduced water supply to riparian habitats in the Owens River Valley (Zektser et al. 2005). Given that so much of their habitat is in regulated waterways, there is an underlying threat to sucker populations through future changes in water management, especially during periods of drought.

Grazing. Livestock grazing is a pervasive land use in the Owens Valley and cattle can negatively affect riparian vegetation, stream channel morphology, stream bank stability, and water quality. However, federal agencies are establishing measures to protect stream channels from grazing in this region (S. Parmenter, CDFW, pers. comm. 2009).

Urbanization. The growing demand for water by cities in the Los Angeles basin represents an increasing threat to the Owens River and its aquatic fauna, despite recent court decisions mandating that flows be maintained in the upper and lower parts of the river.

Recreation. Recreation is a major land use in the region, including use by off-road vehicles (OHVs). However, impacts on suckers appear to be minimal and federal agencies are developing measures to protect stream channels from OHVs (S. Parmenter, CDFW, pers. comm. 2009).

Fire. While wildfires are common in the region, they rarely affect sucker populations. Thus, populations have recovered quickly after repeated wildfires in the Rock Creek drainage (S. Parmenter, CDFW, pers. comm. 2009)

Hatcheries. Pest removal practices used by the Hot Creek Trout Hatchery appear to have extirpated Owens sucker from the spring which feeds the hatchery (S. Parmenter, CDFW, pers. comm. 2009). However, this is a very small part of the historic range of the Owens sucker and hatchery impacts to Owens sucker have otherwise been minimal.

Alien species. Alien species may represent a threat to Owens sucker; however, Owens sucker populations appear to have maintained in their presence. Brown trout are common in the Owens River Gorge (Hill and Platts 1998; S. Parmenter, CDFW, pers. comm. 2009) and may prey on young suckers. Bass, common carp (*Cyprinus carpio*), and catfish (species unknown) are common in the lower Owens River (M. Hill, pers. comm. 2009; S. Parmenter, CDFW, pers. comm. 2009). Owens sucker abundance does not appear to have been limited by their interactions with these species and they presumably persist through a combination of large size, high fecundity, and distinctive life history. Suckers appear to outgrow predation pressure by the time they become adults and/or by using different habitats than alien species.

Effects of Climate Change: The most noticeable and widespread impacts of climate change on aquatic habitats in California will be continued increases in water temperatures and changes to the frequency and timing of both drought and flooding events. Air temperatures (both winter and summer) are expected to increase somewhere between 1°C and 6°C by 2100, with a similar increase in water temperatures in summer, along with decreased summer flows because of reduced snowpack (Cayan 2009, Moyle et al. 2013). Although the environmental tolerance of Owens sucker has not been studied, they appear to withstand temperatures in excess of 22°C (M. Hill, pers. comm. 2009), similar to the tolerances of Tahoe sucker, their closest relative (Moyle 2002). Climate change is not expected to increase water temperatures beyond the thermal limits of native fishes in the Owens River Valley (Parmenter 2008). However, high stream temperatures may reduce individual fitness by increasing physiological maintenance costs (Moyle and Cech 2004) and changes to hydrographs may change the spawning ecology of fishes (Parmenter 2008).

Elevated air temperatures associated with climate change will change the periodicity and magnitude of peak and base flows in streams due to a reduction in snow pack levels and seasonal retention. Streams in the Owens River basin may be not as heavily impacted as those in northern California due to the higher elevations (> 3000 m) of the southern Sierra Nevada (Hayhoe et al. 2004). Nevertheless, predictions are that stream flow will increase in the winter and early spring and decrease in the late summer and fall (Knox and Scheuring 1991, Field et al. 1999, CDWR 2006). It is worth noting that Owens suckers are found in streams that are regulated by dams and diversions, so flows could be manipulated to favor them; however, severe drought could increase water demands in southern California urban areas that might override any flow protections for fishes in the Owens Valley. As such, Moyle et al. (2013) scored the Owens sucker as "highly vulnerable" to climate change.

Status Determination Score = 4.0 – Low Concern (see Methods section Table 2). The California Natural Diversity Database and NatureServe consider the Owens sucker as G3S3, a species that has only a moderate risk of extinction. There does not appear to be any threat to the extinction of Owens sucker at the present time; however, it has been included in the last two iterations of this report with lower ratings of status, but increasing information certainty suggests higher (less vulnerable to extinction) status.

Metric	Score	Justification
Area occupied	1	Native to only one watershed, although introduced into two others
Estimated adult abundance	5	Adults are common throughout most of their range
Intervention dependence	5	None required
Tolerance	5	Owens sucker withstand high temperatures and low dissolved oxygen levels
Genetic risk	4	Possible threat from introductions of other sucker species
Climate change	4	Should persist through most foreseeable changes, unless water diversions greatly increase
Anthropogenic threats	4	See Table 1
Average	4.0	28/7
Certainty (1-4)	3	

Table 2. Metrics for determining the status of Owens sucker, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations:

Habitat restoration. Futher habitat restoration should be pursued in the Owens Valley to support Owens suckers and other native fishes, adding to ongoing efforts. Although the Owens River has been significantly altered by dams and water diversions, reestablished flows in the middle and lower portions of the river are naturally restoring ecosystem function. After five years of a managed flow regime, the Owens River Gorge was able to sustain a productive riparian system (measured by density of riparian vegetation and number of brown trout) (Hill and Platts 1998). Efforts are underway to curtail impacts from OHVs and grazing on aquatic habitats in the upper Owens River. A habitat conservation plan is currently being drafted, in part, for the recovery of native fishes in the lower river (M. Hill, pers. comm. 2008). This plan should be fully implemented in order to maintain and, where feasible, improve ecosystem function and provide for additional habitats for native fishes.

Alien species. Owens sucker appear to withstand interactions with alien species. However, studies have not been completed to test whether predation and/or competition are impacting the survival and fitness of Owens sucker populations. A general policy in the Owens Valley should be to prevent the introduction of additional species and to reduce the populations of established alien species.

Refuges. Although they have not yet been introduced into the Owens Valley Native Fish Sanctuary, located north of Bishop, due to their abundance and widespread distribution in the Owens River watershed, the option of introducing Owens suckers into this refuge (and other refuges) should be evaluated.

Monitoring. In general, Owens sucker populations appear to be stable. However, populations should be closely monitored because they have a limited geographical range and most of their populations are found in habitats that are dominated by introduced fishes, which may displace them from optimal habitats (Moyle 2002, Parmenter 2008).

Research. Studies should be initiated to determine the environmental tolerances of Owens suckers in order to better understand how their populations may respond to predicted changes in environmental conditions.

Figure 1. Distribution of Owens sucker, *Catostomus fumeiventris*, in California. The southern coastal population is introduced.

LAHONTAN MOUNTAIN SUCKER
***Catostomus lahontan* (Rutter)**

Status: Moderate Concern. The Lahontan mountain sucker does not appear to be at risk of extinction in California in the near future; however, many populations are declining and their range is fragmented.

Description: Mountain suckers are small (adults 12-20 cm TL), with subterminal mouths and full lips that are covered by many large papillae (Moyle 2002). Their lips are protrusible, have deep grooves where the upper and lower lips meet, and a cleft on the middle of the lower lip. The lower lip has two semicircular smooth areas along the inner margin next to a conspicuous cartilaginous plate that is used for scraping. The front of the upper lip is smooth. They have 75-92 scales along the lateral line and 23-37 gill rakers on the first gill arch. Fin rays typically number 10 (range 8-13) and nine for the dorsal and pelvic fins, respectively. An axillary process is easily visible at the base of the pelvic fins. Internally, their intestine is long (up to six times TL), and the lining of the abdominal cavity (peritoneum) is black. Their coloration is brown to olive green on the dorsal and lateral surfaces, white to yellow on their bellies, and dark brown in blotches in a lateral row or line. Mature males have two lateral bands, one red-orange on top of another that is black-green. Spawning males have tubercles covering their bodies and fins, with the exception of the dorsal fin. Tubercles on the enlarged anal fin become especially prominent. Spawning females also have tubercles but only on the top and sides of their heads and bodies. Larvae have relatively few dorsal-fin rays and a complete mid-ventral line of pigment from the heart to the vent (Snyder and Muth 2004).

Taxonomic Relationships: The Lahontan mountain sucker was originally described by Rutter (1903) as *Pantosteus lahontan*; the species was subsumed into *Catostomus platyrhynchus* by G.R. Smith (1966). The species was then revived by G.R. Smith et al. (2013), who described the complex taxonomic history of *Pantosteus* suckers as a distinct lineage within the genus *Catostomus*. The *Pantosteus* suckers are collectively referred to as mountain suckers, because they all tend to be small, occur mainly in mountain streams, and have a cartilaginous plate in their lower lip, used for scraping food organisms from rocks. Mountain suckers occur throughout western North America and G.R. Smith (1966) determined that there were six species within the group. Only one species, *C. platyrhynchus,* was recognized to encompass all mountain suckers in the Lahontan, Missouri, Snake, Bonneville, upper Green, and Columbia River drainages, which included California populations. However, based on combined morphometric, meristic, skeletal and mitochondrial DNA analyses, G. R. Smith et al. (2013) concluded that there were actually 11 modern species plus a number of fossil forms. *C. platyrhynchus* was re-divided into four species, including the Lahontan mountain sucker. This classification fits with the long isolation of Lahontan populations and the fact that a number of other Lahontan fishes are considered to be endemics (Moyle 2002).

Life History: Most studies on mountain suckers have been performed on other mountain sucker species outside of California; given the morphological similarity of the forms found throughout the west (Smith 1966, Smith et al. 2013), basic life history characteristics are also likely to be similar.

Mountain suckers (*C. jordani*) in Montana grow to 60-65 mm TL in their first year, 90-100 mm in their second year, and rarely exceed 17 cm TL as adults (Hauser 1969); growth rates in the first three years gradually decrease to a slow and constant rate. In Utah, *C. platyrhynchus* (as redefined by Smith et al. 2013) grow to 64 mm TL in their first year and reach 193 mm TL by the age of six years (Wydoski and Wydoski 2002). Growth is likely mediated by temperature and productivity of the stream in which they occur (Wydoski and Wydoski 2002). Lahontan mountain suckers likely have a similar growth pattern, based on length data (Moyle 2002). In populations that have been studied, males mature at 6-14 cm TL during their second or third year (Smith 1966, Marrin 1980). Females are larger, tend to mature later (second to fourth year at 9-17 cm TL), and live longer (7-9 years) than males (Smith 1966, Marrin 1980, Wydoski and Wydoski 2002). Fecundity can vary from 990 (at 13 cm TL) to 3,710 (at 18 cm TL) eggs per female (Marrin 1980) and is correlated to female total length but not age (Wydoski and Wydoski 2002). Mean egg diameter is also correlated to female total length.

Lahontan mountain suckers, unlike most stream-dwelling fishes in western North America, spawn in summer (June to early August), rather than spring (Olson and Erman 1987, Decker 1989). In California, adults have been observed moving into small streams during later July to feed on algae and to spawn (Decker and Erman 1992). Spawning probably occurs at night, in riffles located immediately below pools, at temperatures ranging from 9-19 °C (Olson and Erman 1987, Decker 1989). However, spawning adults were noted in Sagehen Creek (Nevada and Sierra counties) at temperatures ranging from 9 to 12 °C (Decker 1989). In Utah, *C. platyrhynchus* adults preferred to spawn in flowing water of 6-20 cm/s, in riffles that were 11-30 cm deep (Wydoski and Wydoski 2002). Fertilized eggs adhere to stream substrates. Larvae and juveniles move into the stream margins, favoring areas with beds of aquatic algae associated with pools (*C. jordani*, Hauser 1969). Lahontan mountain suckers hybridize with Tahoe suckers in streams where they co-occur (Decker 1989; T. Taylor, ENTRIX, pers. comm. 2009).

Lahontan mountain suckers feed primarily on algae and diatoms but will also feed on aquatic invertebrates (Smith 1966, Marrin 1980). Juveniles (< 30 mm TL) have a higher proportion of aquatic insects in their diet than adults (Marrin 1980). Adults will move into areas of filamentous algal blooms to forage (Decker 1989).

Lahontan mountain suckers have been observed shoaling with Tahoe suckers (Decker 1989), with which their abundance is positively correlated (Olson and Erman 1987). They are also often associated with alien brown and rainbow trout, which may prey on them (Moyle 2002, Olsen and Belk 2005, Giddings et al. 2006).

Habitat Requirements: Lahontan mountain suckers are characteristically found in shallow (< 2 m), clear, low-gradient streams; they are associated with diverse substrates, from sand to boulders, in areas with dense cover (macrophytes, logs, undercut banks) (Moyle 2002). They have been found in streams at elevations up to 2800 m and at temperatures of 1-25°C (Smith 1966). Cool (<20°C), clear water seemed to be the common characteristic among sites. In eastern Sierra Nevada streams, their abundance is positively correlated with pools but not riffles (Olson and Erman 1987, Decker 1989). They may also be found in larger, more turbid rivers and in some smaller lakes and reservoirs. They have not been found in large lakes such as Tahoe, Eagle, or Pyramid lakes and they seem to be largely absent from California reservoirs. In streams, they typically use habitats with water velocities of 0.1-0.5 m/sec and depths of 0.5-1.8m, especially areas with abundant cover such as root wads and emergent vegetation (Decker 1989; T. Taylor, ENTRIX, pers. comm. 2009). In the East Fork Carson River (Alpine and Douglas/Lyon (NV) counties), mountain suckers are found primarily in mainstem reaches dominated by riffles and runs with cobble-boulder substrates at elevations of 1400-1770 m; these habitats had fish assemblages of 6-8 other species, including various salmonids (Dienstadt et al. 2004).

Habitat use may shift in the presence of piscivores such as brown trout (*Salmo trutta*). Juvenile mountain suckers (*C. platyrhynchus*) in central Utah occurred in main channel pools when brown trout were absent, but occurred exclusively in backwaters and off-channel habitats when brown trout were present (Olsen and Belk 2005). Adults, in contrast, did not exhibit a shift in habitat use, probably because they escaped predation once they reached larger sizes. However, in streams in Wyoming and South Dakota, high densities of large brown trout were found to have a negative influence on occurrence of mountain suckers, regardless of age (Dauwalter and Rahel 2008).

Distribution: In California, Lahontan mountain suckers occur in the Walker, Carson, Truckee and Susan river drainages of the Lahontan basin in the eastern Sierra Nevada, but not in the Eagle Lake basin. They are also found in the North Fork Feather River (Sacramento River) drainage, mainly in Red Clover Creek, into which they were likely carried by a water diversion from the Little Truckee River (Moyle 2002). Although there is at least one specimen known from the Sacramento River, they do not appear to have spread much beyond Red Clover Creek. Lahontan suckers are also widely distributed in streams of the Lahontan Basin (e.g. Humboldt River), in the northern half of Nevada.

Trends in Abundance: Lahontan Mountain suckers appear to be in decline in their native range in California (Erman 1986, Olson and Erman 1987, Decker 1989, Moyle 2002), although Deinstadt et al. (2004) noted that numbers can be highly variable from year to year, based on electrofishing samples. The evidence of decline is mostly anecdotal, where suckers are rare or absent from streams in which they have been abundant in the past. For example, they disappeared from Sagehen Creek following construction of Stampede Reservoir, into which the creek now flows (V. Boucher and P. Moyle, unpublished data). Mountain suckers, however, apparently remain abundant in some streams, such as the East Fork Carson River and its tributary, Hot Springs Creek (Erman 1986). In the East Fork Carson River, mountain sucker densities were estimated to range from 27 to 1,922 fish per mile in the 1980s and 1990s, depending on year of sampling and reach sampled, although estimates were not regarded as very reliable (Deinstadt et al. 2004). Mountain suckers rarely persist in reservoirs in California and smaller tributary streams upstream of reservoirs generally support only small populations, making them vulnerable to extirpation (e.g., Sagehen Creek). Once thought to occur in large numbers in the upper Truckee River (Moyle 2002), Lahontan mountain suckers are now infrequently found there (T. Taylor, ENTRIX, pers. comm. 2009).

Nature and Degree of Threats: Stream impoundment, sedimentation, passage barriers (dams, culverts), interactions with alien species, and hybridization with other sucker species have been noted as threats to various species of mountain suckers (Patton et al. 1998, Wydoski and Wydoski 2002, Belica and Nibbelink 2006). In California, impoundments, predation by brown trout, and habitat degradation due to grazing have been identified as significant limiting factors (Table 1; Decker 1989, Moyle et al. 2002). Because the mountain sucker is, at best, only moderately tolerant of environmental change, the synergistic effects of multiple limiting factors that degrade habitats are presumably the causes of decline.

Major dams. Habitat degradation associated with dams (e.g., alteration of flow and thermal regimes, interruption of sediment recruitment, habitat fragmentation) negatively affects Lahontan mountain sucker abundance and distribution. In Sagehen Creek, impoundment first resulted in a decrease (88%) of the historical longitudinal distribution of mountain suckers (Decker 1989) and then their eventual elimination from this stream (Moyle, unpublished data). Impoundments reduce the amount of stream habitat available and reduce connectivity between mountain sucker habitats because mountain suckers do not colonize most reservoirs. Hybridization between mountain and Tahoe suckers may result from reduced populations of mountain suckers combined with increased populations of Tahoe suckers (which do well in reservoirs), resulting in introgressive hybridization and loss of the species.

Agriculture. The effects of agriculture upon mountain suckers have not been documented and would occur only in the lowermost reaches of streams. In these areas, there are likely impacts to aquatic habitats from channel alteration, irrigation diversions, polluted return water and similar consequences of farming along streams.

Grazing. Grazing can alter the quality of stream habitats for Lahontan mountain suckers by increasing turbidity (decreasing the quality of spawning gravel) and decreasing cover, especially undercut banks (Decker 1989, Moyle 2002). Past grazing pressure incised stream reaches in the upper Truckee River, resulting in siltation of stream substrates and loss of riparian vegetation that provided cover (T. Taylor, pers. comm. 2009).

Rural residential and urbanization. The streams in which mountain suckers occur are affected by rapidly expanding urban and suburban areas (e.g.,Truckee), or areas pressured with development of recreational homes and ski, golf and other types of resorts. The effects of increasing development on suckers has not been documented but negative effects from stream alteration, siltation from run-off, septic pollution, fertilizers and other pollutants from landscape runoff and similar stressors are likely reducing the amount of suitable mountain sucker habitat within their range.

Mining. The legacy effects of hard rock mining in the region include acid mine drainage and stream alteration but effects on mountain suckers are not well documented. Silver and gold mining during the Comstock Lode era likely contributed substantially to degradation of stream and forest habitats, with the widespread development of 'boom and bust' mining towns and their demand for natural resources, but the legacy effects on mountain suckers and other native Lahontan fishes is unknown.

Transportation. Roads are generally associated with declines in fish abundance and diversity in the Sierra Nevada (Moyle and Randall 1998). In the eastern Sierra Nevada, major highways follow the courses of large rivers (e.g. Truckee, Carson rivers) and alter habitats by confining streams, reducing riparian trees and cover, and allowing for increasing development of the region, which impacts streams through habitat alteration, pollution, and diversions. Logging, mining, and agriculture are also associated with increased densities of secondary roads, which directly impact streams through channel alteration and indirectly affect them through increased siltation, removal of riparian cover, and other environmental changes.

Logging. Logging is pervasive throughout the Lahontan mountain sucker's range and, while practices are now much more stream and fish 'friendly' than in the past, logging may still negatively impact streams in which mountain suckers occur. Of greater concern are the legacy effects of intensive logging during the 19th and 20th centuries (much of which supported Comstock Lode mines and mining towns), which dramatically altered streams, with lasting impacts that continue to impair aquatic ecosystem functions. Large rivers in the eastern Sierra Nevada (e.g., Truckee, East Fork Carson) were used as natural sluices to extract millions of board feet of timber from headwater basins during the latter part of the 19th century, causing extensive and, in some cases, lasting environmental damage. For example, large woody debris remains generally absent in many streams that would otherwise provide cover and feeding areas for mountain suckers.

Fire. Fire is a natural and ongoing occurrence in the Lahontan region but the effects of fire upon mountain suckers are unknown. Because fire has been suppressed for many decades, catastrophic fires, with the potential to greatly alter stream habitats, are now more frequent and intense. The future impacts of fire may be exacerbated by predicted climate change outcomes, which may especially affect small, isolated populations in headwater stream reaches.

Recreation. Heavy recreational use, including ski and golf resorts, has altered some streams, especially through sedimentation, pollution input, or perhaps changed behavior of fishes (e.g. through rafting, swimming, or angling). Effects on mountain suckers are not known but are likely minimal.

Alien species. The presence of alien species (e.g. brown trout) can relegate mountain suckers to suboptimal habitats and subject them to increased predation and physiological costs (Olsen and Belk 2005, Belica and Nibbelink 2006, Giddings et al. 2006). Habitat use shifts by juvenile mountain suckers can reduce growth and decrease energy available for reproduction (Olsen and Belk 2005). Nonlethal effects, due to increased physiological costs, may result in additional population declines. In the Truckee River, one of the larger mainstem rivers within their range in California, mountain suckers face threats from interactions with non-native fishes including largemouth bass, bluegill, and brown bullhead, as well as brown, brook and rainbow trouts (T. Taylor, pers. comm. 2009). A more recent threat is the rapid spread of smallmouth bass in the Truckee River watershed, apparently introduced by anglers (Moyle, personal observations).

	Rating	Explanation
Major dams	Medium	Impoundments fragment populations
Agriculture	Low	Increased turbidity and water temperatures may affect some populations
Grazing	Medium	Grazing decreases water quality, reduces riparian cover, and incises streams
Rural residential	Medium	Suburbanization is a growing problem in their range, which can reduce water and habitat quality
Urbanization	Medium	Urban areas tend to concentrate along streams that support mountain suckers
Instream mining	n/a	
Mining	Low	Present in region with toxic effluents, but effects not documented
Transportation	Medium	Highways and railroads parallel many streams, reducing edge habitat and potentially increasing sediment and pollutant input
Logging	Medium	Logging is a principal land use around mountain sucker streams and may increase sedimentation, etc.
Fire	Low	Fire is a natural and recurrent phenomenon in the region but effects on suckers are unknown; fire suppression, coupled with predicted climate change outcomes, may increase future impacts
Estuary alteration	n/a	
Recreation	Low	Heavy recreational use, including ski and golf resorts, has altered some streams, especially through sedimentation
Harvest	n/a	
Hatcheries	n/a	
Alien species	Medium	Interactions with alien species (e.g. brown trout) may interfere with mountain sucker utilization of preferred habitats and reduce populations through predation

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

Effects of Climate Change: Predicted climate change impacts on Lahontan mountain sucker populations and their habitats in California will vary by location. In general, water temperatures are expected to increase and the flow regime of streams will become more variable as the result of more frequent and extreme droughts and floods. Water temperatures are predicted to increase, on average, by at least 0.7°C by 2099, based on conversion factors developed by Eaton and Scheller (1996). Lahontan mountain suckers are generally found in water <20°C (Decker 1989, Moyle 2002). Higher stream temperatures may reduce individual fitness by increasing physiological maintenance costs (Moyle and Cech 2004) and changes to hydrographs may change the spawning ecology of fishes (Parmenter 2008).

Elevated air temperatures associated with climate change will change the periodicity and magnitude of peak and base flows in streams due to a reduction in snow pack levels and seasonal retention. Predictions are that stream flow will increase in the winter and early spring and decrease in the fall and summer (Knox and Scheuring 1991, Field et al. 1999, CDWR 2006). Because mountain suckers spawn in the summer, spawning success may be especially impacted by lower base flows. Moyle et al. (2013) consider Lahontan mountain suckers to be "highly vulnerable" to eventual extinction in California as the result of climate change, reflecting both their apparent on-going decline and the high degree of uncertainty about the their status.

Status Determination Score = 3.1 - Moderate Concern (see Methods section, Table 2). Lahontan mountain suckers are a declining species in California (Decker 1989, Moyle 2002) and probably in Nevada as well; although many populations still persist, they are fragmented and subject to localized extinction (Table 2).

Metric	Score	Justification
Area occupied	3	Found in three major watersheds
Estimated adult abundance	4	Populations in some rivers are assumed to be large
Intervention dependence	4	Persistence will require habitat improvements for most, if not all, streams
Tolerance	3	Moderately tolerant of low water quality
Genetic risk	3	Low numbers, isolation, habitat degradation and hybridization (with Tahoe suckers) threaten genetic integrity of most populations
Climate change	2	Dramatic changes to stream flows likely
Anthropogenic threats	3	See Table 1
Average	3.1	22/7
Certainty (1-4)	2	Abundance and trend data generally not available

Table 2. Metrics for determining the status of Lahontan mountain sucker in California, where 1 is a major negative factor contributing to status, 5 is factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: The apparent decline of Lahontan mountain sucker populations in California may be indicative of the reduced capacity of northeastern Sierra

Nevada streams to support large and diverse populations of native fishes (Moyle 2002), especially since associated declines have also occurred in Lahontan speckled dace and mountain whitefish populations (Olson 1988). Consequently, a number of streams should be targeted for management for native fish communities, as part of a long-term conservation strategy to maintain the biotic integrity of Lahontan basin streams (Moyle 2002). Matrix demographic models suggested that most species of mountain suckers are particularly vulnerable to mortality when they are young of year, so the habitat needs and life history requirements of early stages need special attention (Belica and Nibbelink 2006).

Lahontan mountain suckers are a poorly understood species; basic research on their life history, physiology, and ecology is needed to provide guidance for their protection and for reversing apparent declines. It would be particularly beneficial to conduct a joint research program involving the state of Nevada, given that Nevada encompasses a large portion of the Lahontan mountain sucker's range. Such a program could potentially include NDOW, CDFW, universities, and/or federal agencies performing fisheries monitoring and recovery actions in both states (e.g., USFWS, USGS, USFS). Genetic studies to compare relatedness between California populations and those in the Humboldt River and other areas in Nevada would be of value in terms of developing management strategies to protect genetic and ecological diversity within the species. Specific management recommendations include:

Dams and diversions. Management measures to mitigate impacts of impoundments and diversion should include the removal of dams wherever possible and construction of structures that provide fish passage for non-game species. Where dam removal is not feasible, flows should be managed to enhance spawning by providing colder, higher flows in the summer. Water quality in tributaries to impoundments and reservoirs can be improved by management actions that reduce erosion and sustain riparian vegetation (e.g., through establishment of wide riparian buffer strips, improvements to secondary roads, or closure and restoration of under or non-utilized roads).

Interactions with alien species. Although Lahontan mountain sucker habitat use can presumably shift in response to the presence of alien predator species, refuges from predation are often only available in channels that have not been degraded (Olsen and Belk 2005). Strategies should be developed to reduce impacts from alien species, especially brown trout, which are highly piscivorous at larger sizes. Protection for mountain suckers and other native fishes can be enhanced by increasing instream cover complexity. Restoration actions that increase riparian vegetation and channel complexity should, thus, be developed and implemented. However, restoration plans need to be carefully designed and their potential impacts closely monitored, as a reduction (65-85%) in mountain sucker abundance has been tied to restoration activities aimed at increasing trout habitat (Glover and Ford 1990 in Quinn 1994).

Loss of structural complexity. As noted, mountain suckers can benefit from stream restoration projects that increase habitat complexity and improve water quality. In California, measures to restore heavily altered streams include the creation of new channels in areas with heavy incision (T. Taylor, pers. comm. 2009). In areas where cattle grazing still occurs, benefits can accrue from cattle exclusion fencing to protect stream channels, reduced allotment sizes and quicker rotation of cattle, closure of riparian areas to grazing for experimental impact and recovery studies, and establishment of drinking water sources outside the stream channel. In other areas, roads may need to be moved away from stream banks, crossings reduced, and other measures taken to reduce their impacts.

Overall, management actions to benefit Lahontan mountain suckers will require two interrelated efforts: (1) a status survey; and (2) restoration and management of selected streams to favor native fish assemblages. A status survey should be conducted at least once every five years, as part of a general survey of the status of native Lahontan basin fishes in California. An initial survey should be set up to: (1) identify key sites for a monitoring program; (2) identify streams to manage specifically for native fishes; and (3) quantify the habitat requirements of mountain sucker. Once key native fish restoration streams are identified, efforts should be made to protect habitats in order to enhance their ability to support native fishes. For example, Martis Creek contains a nearly complete assemblage of native fishes but would benefit from restoration efforts (Kiernan and Moyle 2012). Removal of Martis Creek Dam (listed as unsafe by the Army Corps of Engineers) would provide the opportunity for natural flow regimes to be re-established in the lower creek and to eliminate Martis Creek reservoir as a source of alien fishes such as green sunfish.

Figure 1. Generalized distribution of Lahontan mountain sucker, *Catostomus lahontan* (Rutter), in the Susan, Truckee, Carson, and Walker River basins in California. Presumed introduced population in Red Clover Creek (Sacramento River drainage) not shown.

GOOSE LAKE SUCKER
Catostomus occidentalis lacusanserinus (Fowler)

Status: High Concern. The Goose Lake sucker does not face immediate extinction risk but its restricted distribution makes it vulnerable to land and water use practices, climate change, and other factors which could compromise its status.

Description: The Goose Lake sucker is a catostomid that can reach 350 mm SL. As a subspecies, it shares many characteristics with the Sacramento sucker (Ward and Fritzsche 1987), including the number of lateral line scales (64-73), scales above (12-16) and below (8-12) the lateral line, and scale rows before the dorsal fin (27-36). They also have similar numbers of fin rays (11-13 dorsal rays, 7 anal rays, 16-18 pectoral rays, 9-10 pelvic rays), lip papillae (5-6 upper-lip papillae, 5 lower-lip papillae), and gill rakers (21-27). The number of post-Weberian vertebrae in Goose Lake suckers ranges from 42 to 44. They are characterized by a caudal peduncle that is 8-10 percent of the standard length, lack of pelvic axillary processes and a black peritoneum. Body coloration is dark grey to black dorsally and light grey to dull brown ventrally. The head is steel-grey to brown dorsally, but is lighter ventrally. A darker lateral stripe is present in larger fish. The caudal, pelvic, and pectoral fins are light grey to cream. Males develop breeding tubercles on branched and unbranched anal rays and on lower caudal rays. Females have no tubercles (Martin 1967). In reproductive males, the pelvic fins become extremely enlarged, elongated and cupped, presumably to aid in dispersal of sperm during reproduction (Martin 1967).

Taxonomic Relationships: The Goose Lake sucker was first described as a subspecies of Sacramento sucker, *Catostomus occidentalis,* by Fowler (1913) from a single specimen. Since then, the original subspecific name, *lacus-anserinus*, has been modified to eliminate the hyphen and the present name is *C. o. lacusanserinus* (Shapovalov et al. 1959, Kimsey and Fisk 1960, Hubbs et al. 1979). Martin (1967) compared Goose Lake suckers with Sacramento suckers from the Pit River. He concluded that the two forms belonged to different subspecies but that the differences were minor. Ward and Fritzsche (1987), using standard meristic and morphological measurements, looked at *C. occidentalis* from a number of localities, including Goose Lake. Although their multivariate analysis could separate the suckers of Goose Lake from other populations, they concluded that the morphological differences were too small for the Goose Lake form to merit subspecies status. Both Martin (1967) and Ward and Fritzsche (1987) indicated that the Sacramento sucker is a highly variable species morphologically. Therefore, the conservative course of action is to retain the various subspecies names until a thorough genetic study is done on the Sacramento sucker throughout its range.

Life History: Little is known about the life history of the Goose Lake sucker, except that they spawn during spring in streams that are tributary to Goose Lake (Martin 1967). Adults are found in tributaries and the lake throughout the year. Young suckers 40-70 mm SL are very abundant in shallow water during summer in the lake, "packed" in among aquatic macrophytes (R. White, unpubl. data, 1989). Fish become sexually mature by the second year when they are 80-90 mm SL. Martin (1967) found several fish (141-216 mm SL), both male and female, with mature gonads at the beginning of April and concluded that Goose Lake suckers breed during April or May, depending on water temperature. J. Williams (BLM, unpubl. observ. 1984) observed 246-430 mm FL fish on a spawning migration in Willow Creek during May 14-16, 1984. Surveys in 2007 (Heck et al. 2008) found that length frequencies of Goose Lake suckers in Oregon streams represented individuals from young of year to adults, although individual age classes were not established. Goose Lake suckers positively identified by these surveys ranged in size from ~50 mm to 200 mm. Smaller (~20 mm) suckers were captured but were not separated from a group that included Modoc suckers (*Catostomus microps*). In Oregon streams, Goose Lake suckers are closely associated with speckled dace (*Rhinichthys osculus*) and northern roach (*Lavinia mitrulus*) in mid-elevation habitats (Scheerer et al. 2010). Goose Lake suckers feed primarily on algae and diatoms (Martin 1967). Like other suckers, they have a long intestine and ventral mouth adaptive to this diet.

Habitat Requirements: In streams, Goose Lake suckers are typically found in water depths of 15-150 cm and in moderate to slow water velocities (Martin 1967). The streams which they inhabit are up to 4.5 m wide, with summer water temperatures of 15-19°C. Little aquatic vegetation is present. Substrates consist primarily of rock and gravel in headwater sections and mud, silt, and gravel in lower sections. In Oregon, Goose Lake suckers are most abundant in mid-elevation streams flowing through sagebrush, with fine substrates (Scheerer et al. 2010). Goose Lake is shallow, muddy, and alkaline. Gillnetting and trawling surveys indicate that suckers are found throughout the lake (R. White, unpubl. data, 1989). Populations of Goose Lake suckers are apparently also present in small reservoirs in the Cottonwood and Thomas creek drainages, Oregon, but the characteristics of these reservoir populations are not well documented. Juvenile fish have been observed in shallow water among emergent vegetation.

Distribution: The Goose Lake sucker is endemic to the Goose Lake basin and has been reported from Goose Lake and Willow, Lassen, Davis, Branch, and Badger-Cloud Corral creeks, Modoc County, California; and from Dog, Hay, Dent, Drews, Cottonwood, Augur, Cox, Warner and Thomas creeks, Lake County, Oregon (GLFWG 1996, Heck et al. 2008). Individuals have also been documented in Drews, Dog and Cottonwood reservoirs in Oregon, but it is unknown if permanent populations are established in these reservoirs. Apparent spawning runs from these reservoirs, however, have been recorded (J. Williams, unpubl. obs., 1984), which suggests that self-sustaining populations may exist in one or more of these potential refuge sites.

Trends in Abundance: This subspecies is fairly common in streams in its limited range and is common in Goose Lake during periods when the lake is inundated. Individuals were collected in brief surveys of the lake by CDFW (King and Hansen 1966), by USFWS (J. Williams, 1984, unpubl. data), and by University of California, Davis (R. White, 1989, unpubl. data). However, their abundance presumably declined when Goose Lake dried up in 1992-1993 and again in 2010, recovering once lake levels rose again. Although only one Goose Lake sucker (320 mm SL) was caught by the authors from the lake in June, 2008 (Moyle et al. unpubl. data), juvenile and adult Goose Lake suckers are widespread in Oregon streams (Heck et al. 2008).

Nature and Degree of Threats: The principal threat to the Goose Lake sucker is loss of habitat in Goose Lake and its tributaries (GLFWG 1996, Heck et al. 2008; Table 1). Diversions, combined with loss of natural water-storage areas (e.g., wet meadows lost to bank erosion and downcutting of streams), likely cause the lake to dry up more rapidly during prolonged drought as occurred in 1986-1992. While the lake has dried up naturally multiple times (1851, 1852, 1926, 1929-1934, 1992, 2010), it may do so now more quickly or frequently, becoming too alkaline to support freshwater fishes such as suckers, even if drying is not complete.

Agriculture. Diversions, dams, culverts and other obstructions can prevent suckers from reaching spawning habitat and refuge areas in tributary streams (Tate et al. 2005). An estimated 35% of the inflow to Goose Lake is currently diverted for irrigation (Heck et al. 2008). Streams can provide refuge to fishes when low water level and poor water quality become unsuitable for fishes in the lake. Currently, high water temperatures impair ecosystem function in the lower reaches of some streams (e.g. Lassen and Willow creeks), primarily through solar input in open meadows and irrigation water return (Tate et al. 2005), although stream restoration on Lassen Creek have improved conditions in this stream. Temperature gains are partially mediated by seeps and spring along many streams, accentuating the importance of groundwater input. Nonetheless, 20% of streams draining the Oregon portion of the Goose Lake basin are listed as impaired, generally because of high temperatures (NRCS 2006).

Grazing. Livestock grazing is widespread throughout the basin and its effects are often inseparable from other agricultural practices (e.g., irrigated pasture). Most Goose Lake basin streams have experienced some habitat loss due to the effects of grazing and other factors (logging, roads etc.) that degrade watersheds. While improved management of most grazed lands has reduced the threat of grazing in the short-term, as the climate becomes warmer and more variable (see Effects of Climate Change section), there is considerable potential for grazing impacts to increase without reductions in livestock numbers or other mitigation measures (e.g., exclosure fencing along streams). Populations in Oregon reservoirs may provide sources for natural or artificial (translocation) reestablishment after periods of extended drought, provided that water levels and quality are maintained in these refuge locations.

Transportation. Virtually all streams used by Goose Lake suckers are crossed by roads, which often present passage barriers and sources of siltation. Many culverts have been improved (e.g., under Highway 395) for fish passage but most roads crossing streams are unimproved and have unknown effects on sucker populations.

Logging. The Goose Lake watershed was extensively logged in the past, although timber harvest on national forest lands is substantially reduced from historic levels. Timber harvest, however, remains a prominent use of the watershed's forests and has contributed to habitat degradation in streams through siltation, road-crossings, and other factors.

Fire. Wildfire is a natural component of the forested portions of the watershed; increased fire frequency or intensity associated with land use practices and predicted climate change impacts may increase threats, especially to smaller streams.

Alien species. Alien species that may compete with or prey on Goose Lake suckers are present in some reservoirs and streams in the basin. Alien species in the basin include: trout (*Salvelinus fontinalis*, *Salmo trutta)*; centrarchids (*Micropterus dolomieui*, *M. salmoides*, *Lepomis gibbosus*, *L. macrochirus*, *Pomoxis annularis*); yellow perch (*Perca flavescens*); fathead minnow (*Pimephales promelas*); and brown bullhead (*Ameiurus nebulosus*) (GLFWG 1996, Heck et al. 2008). Scheerer et al. (2010) found suckers were absent or scarce where alien fishes were abundant.

	Rating	Explanation
Major dams	n\a	
Agriculture	High	Water diversion and returns from irrigation lower base flow and increase water temperatures; dams may block migration
Grazing	Medium	Grazing pervasive throughout the basin
Rural residential	Low	Rural development is minimal in the basin; however, pumping for wells and septic effluents, along with other impacts from residences may negatively affect stream habitats
Urbanization	n/a	
Instream mining	n/a	
Mining	Low	Uranium mines are present in the area but their impacts are unknown
Transportation	Medium	Roads increase sediment delivery to streams and culverts block fish passage
Logging	Medium	Logging has occurred in the headwaters with decreased intensity in recent years
Fire	Low	Increased fire frequency or intensity may increase threat
Estuary alteration	n/a	
Recreation	Low	Fishing, camping, off-highway vehicles and other recreational use in the area can have negative effects on fish populations and water quality but impacts are likely low because recreation is dispersed
Harvest	n/a	
Hatcheries	n/a	
Alien species	Medium	More than 10 alien species have been introduced to the watershed; however, most are not abundant

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

Effects of Climate Change: The most noticeable and widespread impacts of climate change on aquatic habitats in the Goose Lake basin will be continued increases in water temperatures and changes to the frequency and timing of drought and flooding events. Water temperatures will likely increase by approximately 1°C or more, on average, by 2099, perhaps reducing the individual fitness of fishes already living in temperature impaired streams, such as those found in the Goose Lake basin.

Elevated air temperatures associated with climate change will change the periodicity and magnitude of peak and base flows in streams due to a reduction in snow pack levels and seasonal retention. Stream flow in the basin is primarily fed by snowmelt from the Warner and Fremont mountains, with some baseflow provided by springs (GLFWG 1996). Streams in the Goose Lake basin may be significantly impacted due to the relatively low elevations (< 3000 m) of the Fremont and Warner mountains (Hayhoe et al. 2004). Peak flow currently takes place in the spring, from April to May, but may shift earlier by as much as one month. The lake itself is also fed by a few small springs (Phillips and van Denburgh 1971, in GLFWG 1996). Predictions are that stream flow will increase in the winter and early spring and decrease in the fall and summer (Knox and Scheuring 1991, Field et al. 1999, CDWR 2006), resulting in potential changes to the spawning ecology of fishes. Fish distribution in the basin is already impacted by decreases in streamflow. During dry years (as in 2007 and 2010-12), the distribution of fishes in the basin can be affected by reduction in wetted channel availability. In 2007, 21% of the habitats sampled by Heck et al. (2008) and Scheerer et al. (2010) had gone dry. Moyle et al. (2013) found Goose Lake suckers "highly vulnerable" to extinction as the result of climate change, mainly from prolonged drought. However, Goose Lake suckers are found in some streams that are regulated by small dams that could be managed to mitigate the impacts of climate change on stream flow.

Status Determination Score = 2.3 - High Concern (see Methods section, Table 2). The limited distribution of Goose Lake sucker in California (mostly in Goose Lake itself, as well as the Lassen and Willow creek watersheds) puts this subspecies in some danger of extirpation from its limited stream habitat, especially during years when the lake is dry. The Goose Lake sucker is considered a Sensitive Species by the U.S. Forest Service and Oregon Department of Fish and Wildlife. The American Fisheries Society considers the Goose Lake sucker to be "Vulnerable" (Jelks et al. 2008), while NatureServe, ranks it as "Imperiled" (T2T3). A fundamental problem is the Goose Lake sucker's dependence on lower elevation, low gradient streams which are highly altered by diversions, farming and grazing. Populations likely expand when Goose Lake is full but declines and isolation occur when the lake dries. These same factors make it particularly susceptible to the predicted effects of climate change in this region. Extirpation of the subspecies is less likely when Oregon populations are taken into consideration.

Metric	Score	Justification
Area occupied	2	Goose Lake suckers are endemic to the Goose Lake basin, with limited distribution in California
Estimated adult abundance	2	It is unlikely that any spawning or stream population in CA contains more than 1000 adults
Intervention dependence	3	Population persistence requires active management to maintain water level and quality in streams and in Goose Lake itself
Tolerance	3	Prefers cool-water environments
Genetic risk	3	Genetics poorly understood but populations wide spread in Goose Lake basin
Climate change	1	Summer base flows are predicted to decrease throughout the Goose Lake basin
Anthropogenic threats	2	See Table 1; species may persist in refuge sites in Oregon if it disappears from California
Average	2.3	16/7
Certainty (1-4)	2	Information specific to Goose Lake suckers is limited

Table 2. Metrics for determining the status of Goose Lake sucker in California, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: In 1995, a strategy to protect the Goose Lake sucker and other native fishes was developed by the Goose Lake Fishes Working Group, which includes representatives from federal and state agencies, private landowners, and interested citizen groups (GLFWG 1996). Strategy goals include: reducing threats to fish species, stabilizing populations, and maintaining ecosystem function throughout the Goose Lake basin. Many restoration projects were identified including bank stabilization, riparian fencing, and culvert replacement. Monitoring projects such as telemetry and temperature monitoring were also identified as priorities. Other restoration activities in the basin (mainly in Oregon) have focused on addressing impacts from grazing, erosion, and nutrient influx (NRCD 2006). All of these actions should reduce aquatic impacts and provide needed data and information to inform future management and conservation actions. However, impacts from agricultural irrigation, artificial barriers, grazing, roads (especially culverts) and alien species continue to threaten Goose Lake sucker persistence, especially in California. Additionally, impacts from climate change are predicted to lower base flows, thereby reducing the amount of perennial habitat and increasing summer water temperatures in tributary streams and Goose Lake. Recognizing that persistence of the Goose Lake sucker depends on management actions in both California and Oregon, specific management recommendations include the following:

Dams. Small dams and diversions should be outfitted to allow sucker passage at different life stages. Wherever possible, dams should be removed in a manner that will not expose aquatic habitats to increased sedimentation, scouring, etc.

Agriculture. Open diversions should be replaced by pipes in order to minimize streamflow diversion and water temperature gains. Improving spawning access and increasing flows in streams in California and Oregon, especially Lassen, Willow, and Thomas creeks, would benefit suckers and other native fish species in the basin. Establishment of living buffers and wetlands may reduce the amount of nutrients delivered to Goose Lake and tributary streams, as well as moderate stream temperatures.

Grazing. Stream restoration projects should continue to be implemented, especially measures that create large pools and expand the amount and complexity of riparian vegetation. Cattle exclusion fencing should be maintained and, where appropriate, expanded. Water sources for cattle outside the riparian area should be developed. Maximum impact levels (vegetative height, minimum ground cover, etc.) should be identified, especially for meadow systems, and implemented. Areas where riparian vegetation has been removed, stream banks destabilized, and/or water quality degraded should be closed to grazing to allow ecosystem recovery.

Transportation. Seasonal roads should be storm-proofed (outsloped, inboard ditch removed) and/or decommissioned (outsloped, inboard ditch removed, access blocked, planted) in order to reduce the amount of sediment delivery to streams. Culverts should be replaced by open arches or bridges (minimum width of 1.5 bankfull width) to reduce the potential for blow outs in winter storms and improve fish passage.

Alien species. Alien species should be eradicated from streams and ponds where possible, with priority placed on the removal of predators (e.g., trout and bass species). Removal plans should be made on a site-by-site basis, using information gathered on the community assemblage and estimated abundances of species present in order to account for the potential incidental impacts to native fishes or other aquatic organisms from either chemical treatments or manual removal via electrofishing or netting.

Other actions. Little is known about the life history, habitat requirements and environmental tolerances of the Goose Lake sucker. Studies are needed in order to better understand Goose Lake sucker requirements and tolerances so that additional management measures can be identified. Establishment of refuge populations in farm ponds and other sites in the drainage should be considered. Populations throughout the basin, particularly in California, should be monitored to establish trend information and preserve genetic diversity.

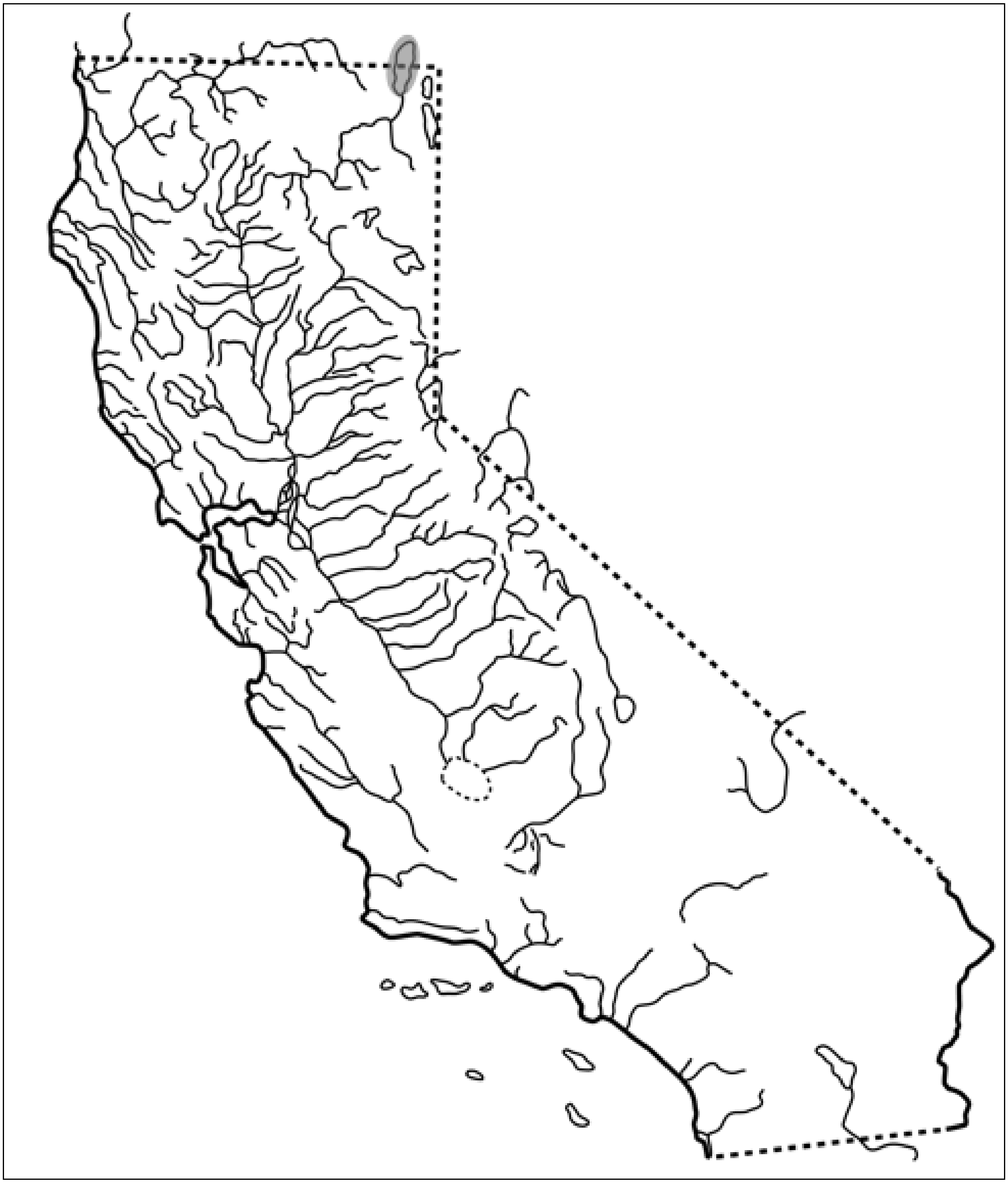


Figure 1. Distribution of Goose Lake sucker, *Catostomus occidentalis lacusanserinus,* in the Goose Lake basin, California.

KLAMATH LARGESCALE SUCKER
***Catostomus snyderi* (Gilbert)**

Status: Critical Concern. Klamath largescale sucker are found in isolated, restricted populations throughout their historic range. In California, they are reproducing in low numbers only in the Tule Lake sump, Clear Lake Reservoir, and the Lost River.

Description: Andreasen (1975) described this species as a generalized sucker, intermediate in most morphological characteristics between the Lost River sucker (*Deltistes luxatus*) and the shortnose sucker (*Chasmistes brevirostris*), with which it co-occurs. While it can reach 50 cm FL, the inferior mouth is comparatively small. The lips are papillose with a medial incision resulting in only one row of papillae extending across the lower lip (Moyle 2002). The narrow upper lip has 4 or 5 complete rows of papillae. The dorsal fin is short, with a basal length equal to or shorter than the longest dorsal ray and an insertion closer to the snout than to the caudal fin. There are 11 dorsal fin rays (may range from 11 to 12) and 7 anal fin rays. Scales are large: 67-81 along the lateral line, 11-14 scale rows above, and 8-12 rows below. Gill rakers number 30-35 but usually 32, in adults, and 25-28 in juveniles. Adults have gill rakers with well-formed processes (bony bumps). Their dorsal surface is green-hued, while their ventral surface is yellow-gold (Moyle 2002).

Taxonomic Relationships: *Catostomus snyderi* was first described from Upper Klamath Lake by Gilbert (1897). It is morphologically similar to *C. macrocheilus* of the Columbia River drainage to the north and to *C. occidentalis* of the Sacramento drainage to the south but, genetically, it is most closely related to other suckers found in the Klamath River basin; the Lost River, shortnose, and Klamath smallscale suckers (Tranah and May 2006). Hybridization among these species is so extensive that Klamath largescale and shortnose suckers from the Lost and upper Klamath rivers were found to be genetically indistinguishable (Tranah and May 2006). However, reproductive and ecological segregation between species has maintained distinct morphological identities (Moyle 2002, Ellsworth et al. 2009). Furthermore, Klamath largescale suckers in a tributary to upper Klamath Lake, the Sprague River (Oregon), appear to be genetically distinct from all other populations (Tranah 2001).

Life History: Detailed information is scant on the biology and life history of this species. Mature suckers collected during a spawning migration were aged at 5-8 years (Andreasen 1975) but these ages are probably underestimates, based on ages of similar-sized shortnose and Lost River suckers. Although growth rates have not been determined, they likely become mature at lengths of 20-30 cm FL, at ages of 4-6 years (Moyle 2002). One male was aged as 7 years old at 31 cm FL (Buettner and Scoppetone 1991). In Upper Klamath Lake, spawning migrations occur from March to May, peaking at the end of March, when ripe individuals of both sexes move up river in large numbers. Males migrate before females (Andreasen 1975). Initiation of reproduction was attributed to rising temperatures (range 5.5-19°C) and flow (Janney et al. 2007 in Ellsworth et al. 2009). In Oregon, spawning migration was initiated by water temperatures above 10°C and rising flows (Ellsworth et al. 2009). The fecundity of three females was estimated as 39,697 (353 mm SL), 64,477 (405 mm SL), and 63,905 eggs (421 mm SL). In the Sprague and Williamson rivers (Oregon), larvae moved quickly from spawning to rearing areas (9-14.5 mm TL) as surface drift at night (Ellsworth et al. 2009). The uniformity of larval drift size suggested that drift only occurs during early swim-up phases.

Historically, adults likely occupied deep lake habitats while juveniles occupied streams or lake margins. A number of larger streams currently support reproducing populations (Scoppetone and Vinyard 1991). Adults have also been found during near-shore and offshore sampling of upper Klamath Lake, suggesting that they use habitats at different depths within the lake (Burdick et al. 2008). Like other large catostomids, Klamath largescale suckers are benthic grazers, preferring invertebrates and algae (Scoppetone and Vinyard 1991, Moyle 2002). Juveniles from upper Klamath Lake fed primarily on zooplankton (Scoppetone et al. 1995).

Habitat Requirements: Although the Klamath largescale sucker is known to inhabit both lentic and lotic habitats, it seems to be primarily adapted to a riverine existence (Andreasen 1975). Little additional information on its ecology is available. They are able to withstand temperatures as high as 32°C, dissolved oxygen concentrations as low as 1 mg/L, and pH levels higher than 10 for short periods of time (Falter and Cech 1991, Scoppetone and Vinyard 1991, Castleberry and Cech 1993). However, streams occupied by Klamath largescale suckers seldom reach water temperatures higher than 25°C (Moyle 2002).

Distribution: Klamath largescale suckers are native to the Lost River-Clear Lake and Klamath River systems in Oregon and California (Moyle 2002). Andreasen (1975) reported them from Upper Klamath Lake, the Clear Lake-Lost River system, the entire Sprague River, the lower 20 km of the Sycan River, and the lower and upper (above Klamath Marsh) Williamson River. In California, they are found in Clear Lake Reservoir, Tule Lake, and the portion of the Lost River between them (USFWS and NOAA 2004, Hodge 2008, Barry et al. 2009, Courter et al. 2010). They possibly occur in the Klamath River and its reservoirs upstream of Iron Gate Dam; however, there is no evidence of self-sustaining populations in this reach.

Trends in Abundance: Abundance estimates for Klamath largescale suckers are lacking. It is likely that their populations have declined in parallel with those of Lost River and Klamath shortnose suckers, with which they co-occur. Both Lost River and shortnose suckers are California Fully Protected Fish and were listed as federally endangered in 1988 (53 FR 27130) and have not recovered (69 FR 43554). Recent surveys of the Lost River (Shively et al. 1999), Clear Lake Reservoir (Barry et al. 2009) and Tule Lake sump (Courter et al. 2010) that focused on capturing the two endangered sucker species have shown Klamath largescale suckers to be present, but in much lower numbers than either of the two listed species.

Nature and Degree of Threats: Klamath largescale suckers in northern California and Oregon have multiple threats to different life stages, including: migration barriers, flow manipulation, pollution, habitat degradation (stream alteration, loss of habitat), harvest, and predation and competition with alien species (Cooke et al. 2005, Table 1). Largescale suckers hybridize with the listed Lost River suckers and shortnose suckers. Most Oregon populations of Klamath largescale suckers appear stable, perhaps because they are largely stream dwelling (avoiding the polluted waters of upper and lower Klamath lakes) and can cross barriers if fish ladders are present. In contrast, California populations are confined to a reservoir, a highly polluted river, and sump for wastewater.

	Rating	Explanation
Major dams	Medium	Clear Lake Reservoir Dam presumably affects populations in Lost River, CA
Agriculture	High	Agriculture diverts water for irrigation and pollutes the Lost River with fertilizers, pesticides and warm return water
Grazing	Medium	Grazing has adversely impacted water quality in the Lost River watershed
Rural residential	Low	Areas within the range of Klamath largescale suckers are little developed
Urbanization	n/a	
Instream mining	Low	Instream mining has occurred and continues to occur but effects on suckers are unknown
Mining	n/a	
Transportation	Low	Impassible culverts, altered riverbanks, and siltation from roads may limit distribution
Logging	Low	Logging may continue to degrade stream habitats by causing temperature increases and siltation
Fire	Low	Wildfires are common in the Klamath River basin but specific impacts to suckers are unknown
Estuary alteration	n/a	
Recreation	n/a	
Harvest	Low	Past harvest contributed to the decline of suckers in Oregon but is largely absent today
Hatcheries	n/a	
Alien species	High	Predation by and competition with alien species (e.g. yellow perch) has most likely contributed to declines throughout their range

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

Dams. Dams in the upper Klamath basin may isolate populations from one another, inhibiting gene flow and recruitment. In California, Clear Lake Dam (which forms Clear Lake Reservoir) may negatively affect Klamath largescale sucker populations through habitat fragmentation, reduced flows, and potential increase of pollutants downstream of the dam. Klamath largescale suckers can become stranded in water diversion canals associated with major dams (Peck 2001, Gutermuth et al. 2000). A total of 432 Klamath largescale suckers were entrained by Link River (Oregon) Dam operations from 1997-1999 *(*Gutermuth et al. 2000*).*

Agriculture. The Lost River and Tule Lake sump are highly polluted with agricultural return water, reducing water quality apparently required by these riverine suckers. Water diversion for agriculture can decrease the amount and diversity of habitat available in the Lost River and often changes flow regimes. Spawning, as well as larval and juvenile drift, appears to be instigated by changes in flow and temperature. In Oregon, cyanobacteria blooms in Upper Klamath Lake are the result of anthropogenic eutrophication; these blooms adversely affect sucker adults and juveniles (due to high pH, low dissolved oxygen, high ammonia) (Bortleson and Fretwell 1993).

Grazing. Grazing has contributed to degraded water quality in the Clear Lake reservoir watershed, Lost River and Tule Lake sump, although studies directly linking grazing impacts to reductions in Klamath largescale sucker populations have not been performed.

Urbanization. Only minor urban development has occurred within their range.

Instream mining. Although impacts are unknown, mining has occurred and continues within their range.

Transportation. Klamath largescale sucker distribution may be limited by impassible culverts and habitat degradation associated with roads.

Logging. Logging continues to impact water quality in this region (increased water temperatures and sedimentation), although this is more a problem in Oregon than in California.

Fire. Wildfires frequently occur within their range but impacts to suckers are unknown.

Harvest. Harvest in Oregon historically contributed to the decline of suckers but is uncommon today.

Alien species. In California, the Lost River ecosystem has been altered by introduction of predatory alien fish species, including yellow perch (*Perca flavescens*) and Sacramento perch (*Archoplites interruptus*). Predation by such fishes on larval and juvenile suckers may have caused sucker declines in California, especially if their populations were already impacted by poor water quality, habitat fragmentation, or other factors. Likewise, competition from fathead minnows (*Pimephales promelas*), abundant in the Lost River and Tule Lake sump, may impact juvenile suckers.

Effects of Climate Change: The most noticeable and widespread impacts of climate change on aquatic habitats in California will be continued increases in water temperatures and changes in the frequency and timing of drought and flooding events. Water temperature increases may reduce the individual fitness of fishes by decreasing growth, decreasing reproductive potential and increasing susceptibility to disease (Moyle and Cech 2004). The Lost River is already a stressful system to suckers, in part because of high summer temperatures, so even small temperature increases may have dramatic impacts.

Climate change will change the periodicity and magnitude of peak and base flows in streams due to a reduction in snow pack levels and seasonal retention. This may make streams less suitable for spawning and rearing and reduce flows in the Lost River, especially during extended periods of drought. Moyle et al. (2013) determined that Klamath largescale suckers were critically vulnerable to extinction as the result of climate change interacting with other stressors.

Status Determination Score = 1.9 - Critical Concern (see Methods section, Table 2). Klamath largescale suckers are the least abundant of the three large sucker species endemic to the upper Klamath River basin, at least in California. They have been classified as Vulnerable (S3) by NatureServe and as Threatened by the American Fisheries Society (Jelks et al. 2008) due to their restricted range, few populations, and other factors that make the species vulnerable to extirpation. Klamath largescale suckers are listed as a U.S. Forest Service Sensitive Species for Upper Klamath Lake and its tributaries and as a Species of Concern by the USFWS.

Metric	Score	Justification
Area occupied	1	Distribution in California restricted to the Lost River-Clear Lake basin and Tule Lake sump
Estimated adult abundance	2	Populations are likely smaller than the two already listed (endangered) suckers in the California portion of the Klamath basin
Intervention dependence	2	Persistence or re-establishment will require intervention
Tolerance	3	Can withstand high temperatures and pH and low dissolved oxygen concentrations for short periods of time but most abundant where water quality it high
Genetic risk	1	Dams isolate populations and hybridization with shortnose suckers can influence genetic diversity
Climate change	2	Flows in the Lost and Klamath rivers will likely be negatively impacted by climate change
Anthropogenic threats	2	See Table 1
Average	1.9	13/7
Certainty (1-4)	2	Very little information is available on their abundance and ecology in California

Table 2. Metrics for determining the status of Klamath largescale sucker in California, where 1 is a major negative factor contributing to status, 5 is factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: Because so little is known about Klamath largescale suckers in California, more information is needed about their abundance, systematics, distribution, habitat requirements, and life history in the state. Management of flows and habitats in the Klamath River drainage should involve establishment of refuge locations for Klamath largescale suckers and other native fishes, preferably including the Lost River. It is quite likely that steps taken to benefit the two formally listed suckers of the upper Klamath basin will also benefit Klamath largescale sucker, but additional measures (such as protection of spawning grounds) are also needed to specifically protect the species, given that it is rarest of the three upper Klamath sucker species in California. Cooke et al. (2005) recommended establishment of freshwater protected areas for spawning and rearing as critical habitat, restoration of degraded habitats, and fish bypass facilities that are sucker-friendly to protect sucker species in streams of the Pacific Northwest and lakes of the western U.S. Furthermore, they recommend protection of natural flow regimes and water quality, eradication of alien species, habitat restoration, and dam removal. These actions should be coupled with education and outreach programs that emphasize the important ecological role (grazer, nutrient cycler) that suckers play in the habitats where they occur.

Klamath largescale suckers should also be protected because they likely contribute to the evolutionary legacy of shortnose and Lost River suckers, species already listed as endangered (Tranah and May 2006). Protection and restoration of spawning and rearing habitats may be particularly important in maintaining genetic diversity and facilitating recovery of these species.

Figure 1. Distribution of Klamath largescale sucker, *Catostomus snyderi*, in the Klamath and Lost rivers systems in California. Distribution is fragmented within shaded areas.

MOUNTAIN WHITEFISH
Prosopium williamsoni (Girard)

Status: Moderate Concern. Mountain whitefish are locally abundant, where present, but their overall abundance and distribution are reduced from historic levels. However, population estimates are generally lacking throughout their range, as are comprehensive distribution surveys, so their overall status remains uncertain.

Description: Mountain whitefish are silvery, large-scaled (74-90 on lateral line) salmonids, with a conspicuous adipose fin, a small ventral mouth, a short dorsal fin (12–13 rays), a more or less cylindrical body and a forked tail. 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 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 sometimes placed in a separate family, the Coregonidae (Moyle 2002), from other salmonids and are regarded as one species throughout their extraordinarily wide range. However, a thorough genetic analysis may reveal a number of distinct population segments within this range. The Lahontan population in California and Nevada is the one most isolated from other populations and may eventually be recognized as a distinct taxon.

Life History: Mountain whitefish are usually observed in loose shoals of 5–20 fish, close to the bottom. As their subterminal mouths and body shape suggest, they are bottom-oriented predators on aquatic insects (Moyle 2002). 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 whitefish in California come from lakes; one measuring 51 cm FL and weighing 2.9 kg came from Lake Tahoe. In Fallen Leaf Lake, the population sampled by gill nets was on average 31 cm FL, with the largest fish being 44 cm long (Al-Chokhachy et al. 2009). Rogers et al. (1996) have developed a standard length-weight relationship for mountain whitefish, based on data from 36 populations throughout their range.

"Spawning takes place in October through early December 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 lake 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 may spawn at dusk or at night, in groups of more than 20 fish. They become mature in 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 streams and rivers at elevations of 1,400–2,300 m. While they are known to occur in a few natural lakes (e.g. Tahoe), there are few records from reservoirs. In streams, they are generally associated with large pools (<1 m deep) or deep runs. In lakes, they typically live close to the bottom in fairly deep water (Al-Chokhachy et al. 2009), 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.

Environmental tolerances of mountain whitefish in California are poorly understood but they are largely found in waters with summer temperatures <21°C. More northern populations have been reported to have temperature preferences of 10-18°C, depending on season (Ihnat and Bulkley 1984). Spawning has been recorded at temperatures of 0-9°C but 2-5°C is typical, which corresponds with optimal temperatures for development of embryos (Northcote and Ennis 1994). Mebane et al. (2003) 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.

Distribution: Mountain whitefish, as the taxon is broadly recognized, are found in western North America, from California to Alaska. They are distributed throughout the Columbia River watershed (including Wyoming, Montana, Oregon, Washington, Idaho, British Columbia, and Alberta), the upper reaches of the Missouri and Colorado rivers, the Bonneville drainage, and the Mackenzie and Hudson Bay drainages in the Arctic. In California and Nevada, they are present in the lower Truckee, Carson, and Walker river drainages on the east side of the Sierra Nevada, in both states, and in the Humboldt River drainage in Nevada. Their range includes both natural lakes (e.g., Tahoe, Fallen Leaf) and streams. Curiously, they are absent from the Susan River and from Eagle Lake, Lassen Co.

Trends in 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 were apparently already reduced in numbers by the 1950s. They still appear 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 also still found in the Little Truckee River, Independence Lake and some small streams, such as Wolf and Markleeville creeks, tributaries to the East Fork Carson River. Their populations in Sierra Nevada rivers and tributaries have been fragmented by dams and reservoirs and whitefish are generally scarce in reservoirs." Severe decline in abundance of whitefish in Sagehen and Prosser creeks, and their eventual disappearance, followed construction of reservoirs that covered their lower reaches (Erman 1973, Moyle, unpublished data). However, a population in nearby Independence Lake (a natural lake) did not show an obvious decline in the period from 1997- 2005 (Rissler et al. 2006). 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. However, there is some indication from diving surveys of dramatic decline in the mountain whitefish population in the Truckee River over the past 20 years (R. Cutter, pers. comm. 2013). At present, California allows 5 whitefish per day to be taken by anglers and Nevada allows 10 whitefish per day. According to the Nevada Department of Wildlife, mountain whitefish are "much less abundant today" than they were historically (http://dcnr.nv.gov/documents/documents/nevadas-fishes-2/).

Overall, indications are that whitefish populations have declined significantly in last 10-20 years. However, existing electrofishing data within their range should be analyzed for presence/absence and trends in abundance in order to better understand their status and inform conservation and management strategies.

Nature and Degree of Threats: Mountain whitefish are little studied in California so factors affecting their abundance and distribution are poorly documented (Table 1). The keys to understanding their possible decline, however, are habitat-related: (1) they live primarily in the larger streams of the northeastern Sierra Nevada and associated lakes, (2) they do not seem to fare well in reservoirs, and (3) they require high water quality and generally low water temperatures for persistence. In general, they live in the waters most likely to be impacted by human activities, especially by expanding development (e.g., rapid expansion in areas surrounding Truckee), dams and diversions, and by highways and railroads.

Major dams. As noted, whitefish inhabit the larger stream of the eastern Sierra Nevada, many of which have been dammed or impounded for agricultural or municipal water delivery. Dams may block movements of whitefish to favored spawning and feeding grounds and create unfavorable conditions both above reservoirs and below them, especially poor water quality. For example, when Farad Dam (Nevada) on the Truckee River was blown out by high flows in 1997-98, the river below it recovered rapidly, with higher flows creating more complex habitat and cooler summer temperatures that favored whitefish and trout. Erman (1986) noted that mountain whitefish abundance dropped in Sagehen Creek following the flooding of its lower reaches by Stampede Reservoir. However, it is possible that flow releases to support trout fisheries below dams also improve conditions for mountain whitefish in certain areas.

Agriculture. Pasture and alfalfa fields line streams occupied by mountain whitefish, especially in the lower reaches of the West and East Walker rivers in California, as well as in Nevada. Attendant diversions and warm, often polluted, return water may impact whitefish populations, which generally require cold, high quality water. Diversions may also reduce stream flows and corresponding water quality required by whitefish.

Grazing and logging. The watersheds in which mountain whitefish occur in California were extensively logged and grazed in the past and continue to be actively managed for such use, although at a much lower and carefully controlled level than occurred historically. Nonetheless, continued timber harvest operations and open range and allotment grazing may contribute to increased sedimentation and water temperatures, as well as riparian and stream habitat degradation.

Urbanization. The Truckee River and tributaries to Lake Tahoe have been altered in many ways by urban and suburban sprawl, along with associated road and highway networks; however, the effects and potential impacts of such developments on whitefish are not quantified.

Harvest. Over-exploitation in the past presumably depleted whitefish numbers although this threat is now largely gone, in part because few anglers target them despite their high degree of edibility.

Alien species. Whitefish coexist in many areas with alien brown, brook, and rainbow trout and it is possible that these trouts may limit whitefish populations by preying on their fry, which have been recorded as an item in brook trout diets. In recent years, smallmouth bass have spread into some parts of the Truckee River system which may present a new predation threat.

	Rating	Explanation
Major dams	Medium	Prefer larger rivers that are most affected by dams; reservoirs provide poor habitat
Agriculture	Medium	Diversions remove water from streams; return water contributes to increased temperatures and pollutant input
Grazing	Low	Most watersheds extensively grazed; impacts to mountain whitefish unknown
Rural residential	Medium	Rural development increasing rapidly in portions of range (e.g., Truckee, Tahoe Basin)
Urbanization	Low	Increasing development of Lake Tahoe, Truckee and Reno regions may reduce habitat quality and quantity
Instream mining	Low	Effects of placer and other mining historically substantial; now greatly reduced
Mining	Low	Effluent from mines may affect local populations (e.g., Leviathan Mine in EF Carson drainage)
Transportation	Medium	Most streams affected by riparian roads, railroads, or both (e.g., Truckee River)
Logging	Low	Most watersheds extensively logged; impacts much greater in the past
Fire	Low	Fires common in watersheds; effects unknown
Estuary alteration	n/a	
Recreation	Low	Heavy use of many streams (e.g., recreational fisheries, boating, ski resorts in headwaters); impacts to whitefish unknown
Harvest	Low	Limited harvest; generally by-catch in trout fisheries
Hatcheries	n/a	
Alien species	Low	Some potential for predation by bass and alien trout to affect populations

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

Effects of Climate Change: Climate change is predicted to increase variability in stream flows, increase water temperatures by 2-4°C and increase human demand for water. The combined impact of these changes is likely to reduce suitable habitat for whitefish, especially summer rearing habitat, and may cause further population declines. Extended drought or flash flooding associated with predicted increased frequency of 'rain-on-snow' events in this portion of the Sierra Nevada may also negatively affect whitefish populations. Moyle et al. (2013) rated mountain whitefish as "highly vulnerable" to extinction in California in the next 100 years as the result of climate change severely altering their already limited habitats, assuming no major changes in water management in the large rivers (Truckee, Carson, Walker and their tributaries) that constitute the core of their habitat in California.

Status Determination Score = 3.9 – Moderate Concern (see Methods section Table 2). Mountain whitefish are locally abundant in many areas, although their overall abundance and distribution are probably reduced from the past. Because so little is known about their abundance, distribution and population trends, the conservative approach is to treat mountain whitefish as a declining species, unless evidence indicates otherwise, in spite of the fairly high score in Table 2.

Metric	Score	Justification
Area occupied	4	Present in three watersheds
Estimated adult abundance	4	Numbers appear to be fairly large in rivers where whitefish are still present
Intervention dependence	5	Populations persist; however, abundance and distribution data are needed; many habitats have been degraded and fragmented
Tolerance	4	Whitefish are more physiologically tolerant than most salmonids, live at least 5 years and are iteroparous; however, they require high water quality and low temperatures
Genetic risk	4	Genetics have not been studied but most populations are isolated from one another
Climate change	2	Whitefish are likely to be negatively affected by decreased flows, warmer temperatures and increased diversions
Anthropogenic threats	4	See Table 1
Average	3.9	27/7
Certainty (1-4)	2	Most reports are anecdotal although there is some grey literature

Table 2. Metrics for determining the status of mountain whitefish in California, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: It is clear that mountain whitefish in California would benefit from a thorough study of their biology including systematics, genetics, distribution, abundance, environmental tolerances, and habitat requirements of all life stages. Existing fisheries surveys in eastern Sierra Nevada streams where mountain whitefish occur are generally focused on trout species (both native – e.g., Lahontan cutthroat and non-native – e.g., rainbow, brown, brook) and the popular recreational fisheries they support. While mountain whitefish are often captured during these surveys (Deinstadt et al. 2004), few efforts have been made, thus far, to assess distribution or population trends. A shift in fisheries management toward native species restoration and recovery is occurring within their range but is currently focused on Lahontan cutthroat trout, which are a listed species (threatened) under the federal Endangered Species Act of 1973. Inclusion of mountain whitefish in survey data analyses, reporting, and management or restoration plans would increase their profile as likely the most abundant native salmonid in the eastern Sierra Nevada. Because of their low tolerance for high water temperatures and poor water quality, they also are a good indicator of 'health' of the Carson, Walker, and Truckee rivers, as well as of Lake Tahoe and other natural lakes. As such, perhaps the best recommendation to benefit mountain whitefish populations is to advocate that they become an integral part of ongoing management and restoration efforts currently focused on other salmonids. Specific recommendations include: (1) basic research on their biology and distribution, (2) monitoring of existing populations at least once every 5 years, (3) habitat restoration in degraded (simplified) stream reaches, and (4) maintenance of flows in regulated rivers at high enough levels so that temperatures remain below 21° C and high water quality is maintained throughout the year.

Figure 1. Distribution of mountain whitefish, *Prosopium williamsoni* (Girard), in California.

UPPER KLAMATH - TRINITY RIVERS FALL - RUN CHINOOK SALMON
Oncorhynchus tshawytscha (Walbaum)

Status: Moderate Concern. Abundance of natural spawners in most tributaries is fairly stable. However, basin-wide trends show increasing hatchery returns, with decreasing natural spawners, even within recent large runs.

Description: See the upper Klamath-Trinity rivers (UKTR) spring-run Chinook and Central Valley fall-run Chinook accounts in this report for detailed coverage of species description. Upper Klamath-Trinity rivers fall-run Chinook enter rivers as reproductively mature fish, exhibiting spawning colors.

Taxonomic Relationships: The UKTR Chinook salmon Evolutionarily Significant Unit (ESU) includes all naturally spawned populations of Chinook salmon in the Klamath River basin, upstream from the confluence of the Klamath and Trinity rivers. The UKTR Chinook salmon ESU is genetically distinguishable from other California Chinook ESUs (Waples et al. 2004). Although fall-run and spring-run Chinook salmon are both part of this ESU, the two runs are treated here as separate taxa due to the distinctive adaptive life histories characterized by each group. See the UKTR spring-run Chinook salmon account in this report for further details on taxonomy within this ESU.

Life History: Upper Klamath-Trinity rivers fall-run Chinook salmon show considerable variability in adult and juvenile life history strategies. This variability is characteristic of "ocean-type" Chinook salmon juveniles, which spend less than a year in fresh water before migrating to the ocean (see the Central Valley spring-run Chinook account for a more detailed discussion of ocean-type vs. stream-type life histories). Adult UKTR fall-run Chinook salmon enter the Klamath estuary from early July through September (Moyle 2002). They often hold in the estuary for a few weeks and initiate upstream migration as early as mid-July and as late as October. Migration and spawning both occur under decreasing temperature regimes. Fall-run UKTR Chinook seem to hold extensively in, and travel slowly through, the lower Klamath River (Strange 2005). Between 1925 and the early 1960s, the Klamathon Racks provided a counting facility and an egg collection station close to the current location of Iron Gate Dam. The earliest date that Chinook salmon passed this location between 1939 and 1958 was 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 recorded at 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 2007) 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 in these tributaries tapers off in December, although snorkel surveys at the mouth of the Scott River found Chinook holding through mid-December (Shaw et al. 1997). Fall-run Chinook salmon migration occurs in the Trinity River between September and December, with early migrating fish entering larger tributaries first; use of smaller streams for spawning occurs later in the spawning season. Spawning on the Trinity River begins earliest in suitable mainstem habitats, immediately downstream of Lewiston Dam, and extends into late November further downstream. Spawning in the South Fork has been documented to begin 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 than 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 late winter or spring. The timing of fry emergence is dictated by water temperature, so the beginning of emergence may differ by over four weeks between years in the mainstem (Shaw et al. 1997).

The timing of juvenile emigration is highly variable and dependent on river rearing conditions, which are controlled largely by water temperature and food availability. High winter flows, level of snowpack and subsequent spring runoff can reduce water temperatures (Minshall et al. 1989) and may contribute to 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 scales from returning fall-run adults to determine fry emigration patterns. Three distinct types of juvenile freshwater life history strategies for UKTR fall-run Chinook were identified: (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 quickly and forage along tributary and mainstem rivers for a short period, prior to emigrating during summer months. In the Shasta River, peak fry outmigration occurs in March or early April and from mid-April to mid-May in the Scott and Salmon rivers. Historically, in the mainstem Klamath River, Chinook juvenile emigration initiated in mid-March, before peaking in mid-June, and decreased by the end of July (Shaw et al. 1997). More recently (1997-2000), wild juveniles were not observed in the lower river earlier than the beginning of June, with a peak in mid-July (USFWS 2001).

The second juvenile rearing strategy involves extended freshwater rearing with emigration to the ocean during fall to mid-winter (Sullivan 1989). Juveniles emigrate into the mainstem 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 summer months by juveniles (USFWS 2001). On the lower Trinity River (0.4 rkm upstream of Weitchepec), naturally produced Chinook salmon emigration peaked around April 21, 2001. The first hatchery-produced Chinook salmon were not observed until six weeks later and emigration of these fish peaked in mid-October on the lower Trinity River (Naman et al. 2004). Juveniles of this life history strategy may remain in tributaries until fall 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 (probably from multiple ESUs) were more abundant in the Klamath River estuary than during high flow years, suggesting that the second strategy may involve moving into cooler estuarine water sooner than under high 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, entering the ocean the following spring as yearlings (Sullivan 1989). From 1997-2000, these yearlings 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).

A fourth life history variation has recently been described. Recent surveys have observed mature parr in the Shasta River (C. Jeffres, pers. comm. 2011). Mature parr are reproductively mature males that have never left fresh water (M. Knechtle, pers. comm. 2011).

In the ocean, Klamath River Chinook salmon (all runs) are found in the California Current system off the California and Oregon coasts. Salmon seem to follow predictable ocean migration routes and Chinook recaptured from the Klamath River generally use ocean areas that maintain 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 age composition of Chinook spawners returning to the Klamath basin, a majority are fish age 3 or 4 years. Some age 5 fish are observed but they make up a smaller proportion of the total escapement than grilse. Grilse are small, mostly male, two-year-old spawners. Between 1978 and 2006, they constituted 2-51 percent of the number of annual Klamath River Chinook salmon (CDFG 2006). Sullivan et al. (1989) observed that, in 1986, a larger proportion of four year old Chinook returned to the Salmon River (24%) than to other subbasins. 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 2004, the age structure of the Trinity River Hatchery (TRH) fall Chinook run was composed of: two (8%), three (78%), four (13%), and five (1%) year old fish (CDFG 2006a). In 2006, the Klamath River fall Chinook run was composed of: two (31%), three (21%), four (47%), and five (1%) year old individuals (KRTAT 2007).

Habitat Requirements: Upper Klamath-Trinity rivers fall-run Chinook salmon enter the Klamath estuary for only a short period prior to spawning. Unfavorable temperatures may exist in the Klamath estuary and lower river during summer and chronic exposure of migrating adults to temperatures of even 17°-20°C is detrimental (McCullough 1999). However, if water temperatures are decreasing, UKTR fall-run Chinook will migrate upstream in water temperatures as high as 23.5°C; water temperatures above 21°C generally seem to inhibit migration when temperatures are rising (Strange 2005). The thermal threshold for migration inhibition seems to be higher for UKTR fall-run Chinook than for Columbia River fall-run Chinook (>21°C; McCollough 1999). Optimal spawning temperatures for Chinook salmon are reported as less than 13°C (McCollough 1999). Water temperatures in the fall are usually within this range in the Trinity River (Quilhillalt 1999). Magneson (2006) reported water temperatures up to 14.5°C during spawner surveys in 2005. The Shasta River was historically the most reliable spawning tributary in the Klamath River system in terms of water temperatures (Snyder 1923), but diversions of cold water, combined with warm irrigation return water, have greatly diminished its capacity to support salmon. In addition, Ricker (1997) found that levels of fine sediment in 6 of 7 potential Shasta River and Park Creek spawning locations were high enough to significantly reduce fry emergence rates and embryo survival.

A majority of spawning habitat in this ESU is found in larger tributaries and in the mainstems of the Klamath and Trinity rivers. Spawning occurs primarily in habitats with large cobbles, loosely embedded in gravel, with sufficient subsurface infiltration of water to provide oxygen for developing embryos. On national forest lands in the Scott River basin, a significant portion of such Chinook spawning habitat is in poor condition (Olson et al. 1992). In a survey of Trinity River redds, Evenson (2001) found embryo burial depths averaged 22.5-30 cm, suggesting minimum depths needed for spawning gravels. Regardless of depth, the keys to successful spawning are adequate water flow and cold temperatures. Redds in the mainstem Trinity River averaged 4.4m long and 2.3m wide (Moffett and Smith 1950), where the loosened gravels permitted infiltration of oxygenated water. For maximum embryo survival, water temperatures must be between 6-12°C, with oxygen levels 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 above 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 common in the middle Klamath River during the spring/summer juvenile emigration period, so cool water inputs at tributary confluences are important refuge habitats during the day (Belchik 1997). Stratified pools and subsurface flows at the base of old landslides and gravel bars are also important thermal refuges (Klamath National Forest, unpubl. report). Elevated river temperatures (>16°C) increase mortality from *Ceratomyxa shasta* infection in Chinook salmon released from Iron Gate Hatchery, in association with lethargic behavior, reduced body mass and co-occurring bacterial infections from *Parvicapsula minibicornis*. Belchik (1997) identified 32 cool water refuge areas in the middle Klamath River mainstem. Twenty-eight of these locations were tributary confluences, including that of the Scott River. These habitats have temperatures of 10°-21.5°C and provide refuges from temperatures lethal to emigrating juveniles (Belchik 1997). Belchik (1997) determined that fish abundance in these cool water areas was significantly related to the distance from Iron Gate Dam, proximity to the nearest cool water refuge area, and minimum temperature of each refuge area.

Distribution: UKTR Chinook salmon are found in all major tributaries above the confluence of the Klamath and Trinity rivers and are raised in hatcheries below Iron Gate and Lewiston dams. Upper Klamath-Trinity rivers 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 1 Dam and further restricted by the completion of a series of dams on the Klamath, concluding with construction of Iron Gate Dam in 1964. As a result, salmon (and other anadromous fishes) were denied access to approximately 563 km of migration, spawning and rearing habitats in the upper Klamath River basin (Huntington 2006). Along the lower Klamath River, numerous tributaries provide suitable spawning habitat including: Bogus, Beaver, Grider, Thompson, Indian, Elk, Clear, Dillon, Wooley, Camp, Red Cap, and Bluff creeks. The Salmon, Shasta and Scott rivers historically supported large numbers of spawning Chinook salmon and they remain 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).

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

Trends in Abundance: It is likely that UKTR spring-run Chinook was historically the most abundant run in the Klamath and Trinity rivers (Snyder 1931, LaFaunce 1967) but, by the time records were kept, the spring run had been reduced to a minor component of Klamath salmon populations. Therefore, modern estimates of Chinook salmon numbers in the Klamath-Trinity system are generated primarily from UKTR fall-run 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 at 200,000 fish annually, using commercial fishery data collected 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 entire Klamath River system, during the period from 1915-1928. At the Klamathon Racks, a fish counting station proximate to Iron Gate Dam, an estimated annual average of 12,086 Chinook spawned in the upper basin from 1925-1949 and declined to an average of 3,000 from 1956-1969 (USFWS 1979). In 1965, the Klamath

River basin was believed to contribute 66% (168,000) of the total number of Chinook salmon spawning in California's coastal basins (CDFG 1965). This production was nearly equally distributed between the Klamath (88,000 fish) and Trinity (80,000 fish) basins, with approximately 30% of the Klamath basin's fish originating in the Shasta (20,000 fish), Scott (8,000 fish), and Salmon (10,000 fish) rivers. Snyder (1931) noted that the Shasta River was the best spawning tributary in the basin; however, the number of returning spawners has markedly declined since that time. Leidy and Leidy (1984) estimated an annual average abundance of 43,752 Chinook from 1930-1937; 18,266 from1938-1946; 10,000 from 1950-1969; and 9,328 from1970-1976. A review of recent escapement into the Shasta River found an annual escapement of 6,032 fish from 1978-1995 and an escapement of 4,889 fish from 1995-2006 (CDFG 2006b). In the Scott River, fall Chinook escapement averaged 5,349 fish from 1978-1996 and 6,380 fish from 1996-2006. Analysis of natural spawner abundances suggests that numbers are fairly stable in several tributaries (Bogus Creek, Shasta River, Salmon River, Scott River, Trinity River) (Quiñones, unpublished data). Coots (1967) estimated the annual run of Klamath River Chinook salmon at 168,000, half of which ascended the Trinity River. Hallock et al. (1970) estimated 40,000 Chinook salmon entered the Trinity River above the 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; this average declined, between 1996 and 2006, to 23,463 fish (CDFG 2007).

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). Between 1978 and 2006 the total in-river escapement of UKTR Chinook ESU ranged from 34,425 to 245,542 fish, with an average 5-year geometric mean of 112,317 fish (Figure 1). The mean number of natural spawners in the basin in recent years (2008-2012) was 79,187, which is equal to approximately 60% of the historical run of 300,000 spawners. The number of natural spawners in the basin appears to have remained steady since 1978 (Figure 1).

Hatchery operations have supplemented the abundance of UKTR Chinook salmon since completion of terminal mitigation hatcheries on the Klamath and Trinity rivers in the 1960s. The origins of hatchery stocks are principally from Klamath River fish 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 out-migrant trap were hatchery-origin fish (USFWS 2001). At the Willow Creek out-migrant 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). Hatchery-origin adults also spawn in rivers, including all major tributaries (e.g., Shasta, Scott, Salmon rivers), although straying of hatchery fish is most pronounced in areas closest to the hatcheries (e.g., Bogus Creek and Shasta River in the Klamath drainage and upper main stem Trinity River).

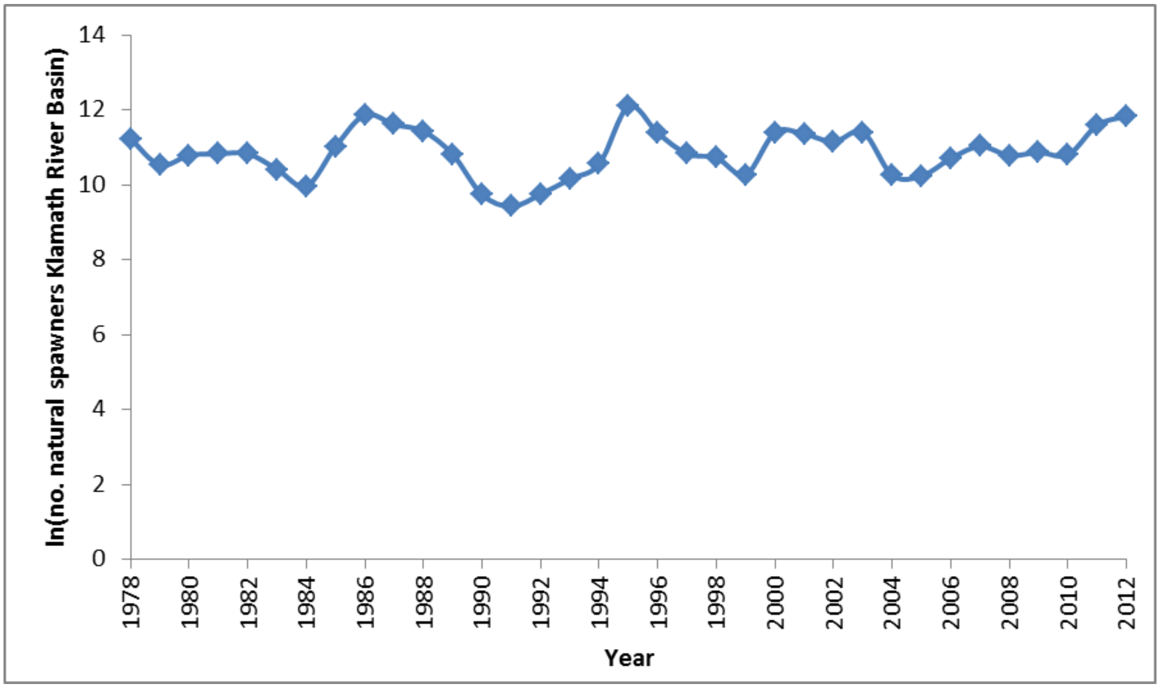


Figure 1. Number (ln) of UKTR fall-run natural spawners in the Klamath River basin, 1978-2012 (source data: CDFW 2012 Megatable). No trend in numbers was detected.

In general, historic numbers of wild UKTR fall-run Chinook probably ranged between 125,000 and 250,000 fish per year. While numbers over the past 25 years have often reached into that range, much lower numbers are typical and many fish are of hatchery origin. Of particular concern is the increasing trend of the proportion (%) of basinwide escapement made up of hatchery returns (Quiñones et al. 2013). However, factors influencing adult abundances include oceanic conditions and freshwater habitat quality; these factors differ by run and location (Quiñones 2011).

Nature and Degree of Threats: Numerous threats have influenced the status of UKTR Chinook salmon. Primary stressors include: dams, logging and other land uses, fisheries, hatcheries, and disease (Table 1).

Dams. Upper Klamath-Trinity rivers fall-run Chinook are primarily mainstem spawners, so Lewiston and Iron Gate dams negatively affected their population by changing downstream habitats (including altering seasonal flows and temperature regimes) and by blocking access to historic spawning area upstream. 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 reduce the amount of instream flow). However, dams have eliminated spawning gravel recruitment from upstream areas and reduced hydrologic variability. The lack of adequate flow releases is thought to have been a principal factor that caused a major fish kill in the lower Klamath river in September, 2002 (CDFG 2004).

Lewiston Dam and other dams on the Trinity River have substantially modified river flows and generally reduced the size and habitat complexity of the river channel. Starting in 1964, 75-90% of Trinity River flow was diverted to the Central Valley.

Declines of naturally-spawning fall-run Chinook populations were likely exacerbated by diversion of most of the river's water and corresponding reduction and degradation of 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 accomplished 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 mimicking the natural flow regime. Implementation is now underway (http://www.trrp.net/), after the commencement was delayed until 2004 by lawsuits.

Agriculture. Much of the water diverted from the Trinity River is used for agriculture in the Central Valley. Diversion of water for agriculture from the Klamath River in Oregon, as well as from the Shasta and Scott rivers, reduces stream flows and increases temperatures, making many areas of formerly suitable habitat no longer suitable for salmon spawning or rearing. Because many farms use flood irrigation, return water flows back into the streams at high temperatures, further warming streams. These impacts are particularly acute during summer and early fall months, when ambient temperatures are highest and natural flow inputs are lowest. Pumping from wells also reduces ground water tables and associated cold water inputs into rivers. The Shasta River, for example, has been converted by agricultural diversions from a cold river that supported year-round salmon production to one with degraded water quality, including temperatures too high to support salmon in summer.

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

Grazing. Livestock are grazed on many public and private lands throughout the Klamath-Trinity system. Grazing impacts occur mainly on tributary streams, where livestock can cause severe bank damage and reduce riparian vegetation, resulting in stream incision, reduction of riparian cover, and silting of spawning gravels.

Rural residential. The long history of mining and logging in the Klamath and Trinity basins has left an extensive network of roads which continue to provide access to many remote areas, facilitating rural development throughout these basins. Widespread rural development results in increased sediment delivery to streams, particularly in the steep, mountainous terrain of this region, effluent from septic tanks and other pollutants, water diversion, deforestation and habitat fragmentation.

	Rating	Explanation
Major dams	High	Much former habitat is above dams; dams have decreased habitat quality downstream
Agriculture	High	Habitats have been degraded through diversions, warm return water, and associated pollutant inputs
Grazing	Medium	Livestock are pervasive on public and private lands; impacts concentrated in smaller tributary streams
Rural residential	Medium	Cumulative effects of numerous roads and rural development can negatively affect salmon habitats
Urbanization	Low	Urban areas are few, small, and restricted to main rivers
Instream mining	Medium	Legacy effects are still severe in some areas, while dredge mining can alter habitat and disturb fish (currently banned)
Mining	Low	Legacy effects of hard-rock mining are potentially severe in localized areas
Transportation	Medium	Roads present along many streams; sources of sediment and pollutant input along with habitat fragmentation
Logging	Medium	Both legacy effects and ongoing impacts degrade aquatic habitats; much greater historical impact
Fire	Medium	Fires predicted to become more frequent and severe, potentially degrading important headwater tributary habitats
Estuary alteration	Low	The Klamath River estuary is less altered than most north coast estuaries
Recreation	Low	Human use of rivers may impact behavior of spawning fish and juveniles
Harvest	Medium	Legal and illegal harvest, combined, may be negatively affecting abundance
Hatcheries	Medium	Principal run raised in Iron Gate and Trinity hatcheries
Alien species	Low	Few alien species in range, although brown trout present in Trinity River

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

Mining. Mining has dramatically altered river and stream habitats in the Klamath-Trinity Province, with lasting legacy impacts in many areas. Intensive hydraulic and dredge mining occurred in the 19th century and, depending on location, these activities caused severe stream degradation and alteration to channel morphology. Mining was a principal cause of decline of UKTR Chinook in the Scott River and large areas in the Trinity River, followed by some level of recovery after large-scale mining ceased. The Scott River was heavily altered in the Scott River Valley and remains so today, where a degraded river winds through immense piles of dredge tailings. Historic mining impacts still affect the Salmon River Chinook population, as the estimated 16 million cubic yards of sediment disturbed between 1870 and 1950 are slowly transported through the basin (J. West, U.S.F.S., pers. comm. 1995). Mining and its legacy effects have disconnected and constricted juvenile salmon habitats, filled in adult holding habitats, degraded spawning grounds and altered the annual hydrograph of many streams. Pool in-filling is a particular problem because high stream temperatures have been demonstrated to reduce survival of both holding adults and rearing juveniles (West 1991, Elder et al. 2002).

Suction dredging for gold can also negatively affect fall-run UKTR populations, although there is currently a moratorium in place. See the UKTR spring-run Chinook account in this report for more details.

Transportation. Roads are present along many streams, resulting in sediment or pollutant inputs; many roads in this region were constructed to provide access for timber harvest and mining and built at a time when little attention was paid to environmental impacts. Many roads have been improved and/or closed to public access, but impacts to stream habitats and water quality remain. Culverts and other passage structures often create migration barriers, although restoration projects have mitigated many of these impediments.

Fire. Wild fires are predicted to become more frequent and severe under climate change scenarios, so may pose increasing threats to spawning and holding habitats, as well as contribute to increasing water temperatures and sediment input.

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

Harvest. The Pacific Fisheries Management Council (PFMC) has paid particular attention to upper Klamath and Trinity River Chinook salmon in recent years because annual escapement goals have not met the Council's minimum escapement objective for natural adult spawners in 17 out of 35 years. In November, 2006, the PFMC accepted new fisheries guidelines that are intended to result in annual 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 (2005-2006); (2) the risk that populations were dropping below critical genetic thresholds; (3) prevailing ocean conditions; and (4) Federal Endangered Species Act recovery actions for other species (PFMC 2007). Poor ocean conditions can severely impact escapement, especially when combined with high rates of harvest. In 2008, the minimum escapement goal was raised to 40,700 fish, partly to account for recent high returns. Harvest goals are often difficult to set because consistently poor conditions in freshwater, coupled with reliance on hatchery fish to support the fishery, means that ocean conditions become increasingly important in determining levels of adult returns. This results in extreme population fluctuations, as evidenced in recent years.

Because the status of both Central Valley and Klamath River salmon stocks is highly variable, the ocean fishery (and probably the inland sport fishery as well) is likely to be periodically restricted to prevent overharvest of wild fish, unless a mark-selective fishery is instituted (e.g., all hatchery fish are marked and all non-marked (wild) fish are released).

Hatcheries. Although most tributary spawning stocks are apparently comprised mainly of wild fish, the spawning stocks in the mainstem Trinity and Klamath rivers are increasingly supported by hatcheries. Hatchery operations have likely influenced the age of maturation and spawning distribution of UKTR Chinook salmon and reduced life history diversity in the Klamath-Trinity basin. Hatcheries first began operating on the Klamath River for 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 from 1978-82 to 0.26 from 1991-95 and 0.29 from 2001-2006 (CDFG 2007, Myers et al. 1998). In 1999, 73% of redds were located between Iron Gate Hatchery and the Shasta River and this proportion has increased over time (Bartholomew and Hendrikson 2006). Similar observations have been made on the Trinity River. Historically, most fall Chinook in the Trinity River spawned between the North Fork and Ramshorn Creek (Moyle et al. 2008). More than 50% of out-migrating smolts observed between 1999 and 2000 at the Willow Creek monitoring traps were fish clipped at hatcheries. This proportion increased to more than two-thirds during the fall monitoring period (USFWS 2001), although this may attributed to the fact that most naturally produced Chinook in the basin are ocean type and emigrate in the spring and summer and hatchery releases of yearling fish occur in October. Large numbers of hatchery fish in the Klamath-Trinity system may impact naturally produced Chinook juveniles through competition, predation, and/or disease transmission. Competition and predation may be enhanced when releases of large (compared to wild fish) hatchery juveniles occupy shallow water refuge habitats used by naturally spawned juveniles (NRC 2004), which may also increase the incidence of disease transmission. Wild populations are also threatened with reduced fitness through interbreeding with hatchery fish (Quiñones et al. 2013).

Hatchery returns are likely replacing natural escapement of at least some wild populations of UKTR fall-run Chinook. The proportion (percent of basin-wide escapement) of fall-run Chinook natural escapement has significantly decreased ($p = 0.001$), concurrent with significant increases in hatchery returns to IGH and TRH. Since the 1980s, returns of Chinook salmon to IGH significantly increased ($p = <0.0002$), as did the number of hatchery strays throughout the basin ($p = 0.013$). Basin-wide fall-run Chinook adult abundance was significantly correlated to returns to both hatcheries ($r(27) =0.53$, $p = <0.05$). Fall-run Chinook natural escapement to Bogus Creek was significantly correlated to returns to both IGH ($r(27) = 0.60$, $p = <0.05$) and TRH ($r(27) = 0.58$, $p = <0.05$). Fall-run Chinook natural escapement to the Salmon River was significantly correlated to returns to IGH ($r(27) = 0.36$, $p = <0.05$). Fall-run Chinook natural escapement to the Trinity River was significantly correlated to returns to both IGH ($r(27) = 0.41$, $p = <0.05$) and TRH ($r(27) = 0.72$, $p = <0.05$) (Quiñones et al. 2013). These patterns suggest increasing dependence on hatchery propagation but may, alternately, signal similar responses of natural and hatchery spawners to environmental conditions. In either case, more research is needed to understand the full extent of hatchery influence on natural production and genetics of fall-run Chinook in the Klamath-Trinity system. See the Central Valley fall-run Chinook account in this report for further discussion on hatchery effects.

Synergistic impacts. Recent large scale die-offs of UKTR salmon and other fish in the Klamath River provide examples of how multiple factors can affect salmon runs. 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 predominantly UKTR fall-run Chinook adult salmon perished in the lower Klamath River. The immediate cause of death 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 this massive die-off are still not fully understood, but were likely a combination of: (1) high water temperatures, (2) crowded conditions, and (3) low flows. In response to high water temperatures and low flows, fish apparently ceased migration and concentrated in large numbers in pools. These conditions allowed for a disease epidemic to sweep through the population of highly stressed fish. The contribution of low flows to this unfortunate and highly publicized event is underscored by the finding that increased base flows likely reduce pathogen transmission risk during Chinook salmon migration (Strange 2007).

In juvenile UKTR Chinook salmon, high water temperatures and low flows can also 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 not always abundant nor do the conditions necessary for infecting large numbers of Chinook salmon occur regularly. *C. shasta* is known to occur 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 fall-run Chinook were historically infected by these diseases at low levels, but rarely did widespread epidemics occur because contributing factors such as high temperatures, low flows, poor water quality and lack of access to upper watersheds (greater spatial distribution and lower concentration of spawners) did not exist to the extent they do now. Although *C. shasta* does not appear to occur in the Shasta, Scott, and Trinity rivers (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). When high densities of infected fish and warm temperatures exist in combination, *C. shasta* infection appears to be accelerated (Foott et al. 2003). Large releases of hatchery fish may, therefore, be particularly susceptible and spread disease to wild fish. *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). It is also likely that most juvenile Chinook from the Scott and Shasta rivers do not survive their exposure during emigration through the lower Klamath and these diseases may, therefore, ultimately select for juvenile UKTR Chinook that emigrate at times when temperatures in the main river are cooler, increasing the potential for more frequent disease outbreaks.

Effects of Climate Change: The 'ocean' life history strategy of UKTR fall-run Chinook makes them least vulnerable of all runs to climate change, although warm temperatures in the Klamath River are already a substantial threat. Elevated water temperatures have been identified as a factor limiting anadromous salmonid abundance in the Klamath River basin, as the result of multiple land and water use impacts, combined with climate change. Water temperatures have increased approximately 0.5°C/decade and has resulted in the loss of about 8.2 km of cool summer water in the mainstem each decade (Bartholow and Hendrikson 2006). Bartholow and Hendrikson (2006) documented that the timing of high temperatures potentially stressful to Chinook has moved forward seasonally 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 also influence the survival of UKTR fall-run Chinook. See the UKTR spring-run Chinook account in this report for further information on potential climate change impact to salmon populations in this region. Moyle et al. (2013) rated the UKTR fall-run Chinook as "highly vulnerable" to extinction in the next 100 years as the result of the added impacts from climate change.

Status Determination Score = 3.0 - Moderate Concern (see Methods section, Table 2). UKTR fall-run Chinook are not in immediate danger of extinction, although their numbers have declined in recent decades. There is increasing reliance on hatcheries to maintain fisheries and hatchery production may be masking a decline of wild production in the Klamath-Trinity basins, which does not bode well for the longer-term persistence of wild salmon stocks (Quiñones 2011). The UKTR Chinook salmon ESU was determined to not warrant listing under the Federal Endangered Species Act on March 9, 1998. Upper Klamath-Trinity rivers fall-run Chinook are a U.S. Forest Service Sensitive Species. They are managed by CDFW for sport and ocean fisheries, and by PFMC for tribal, ocean sport and commercial fisheries.

Metric	Score	Justification
Area occupied	5	Widely distributed in Klamath and Trinity basins
Estimated adult abundance	4	Abundant, with several large populations, but minimum escapement goal is not always met
Intervention dependence	3	Major intervention is required to maintain fisheries, primarily through hatchery propagation and flow regulations
Tolerance	3	Moderate physiological tolerance
Genetic risk	2	One genetically diverse population but heavily influenced by hatcheries
Climate change	2	Vulnerable to increasing temperatures in mainstem rivers, changes in flow regimes in tributaries, and variable ocean conditions
Anthropogenic threats	2	Two threats rated "high" and eight "medium" (Table 1)
Average	3.0	21/7
Certainty (1-4)	4	Most studied of Klamath River Chinook runs

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

Management Recommendations: There are many ongoing, as well as potential, management options for the Klamath and Trinity rivers to benefit UKTR Chinook salmon. The Trinity River Restoration Program (TRRP) is focused on maintaining and recovering populations of UKTR Chinook salmon by taking a holistic approach to restoration. The TRRP approach involves flow manipulations and focused restoration activities to meet the habitat requirements of Chinook and other keystone aquatic species. A similar program should be implemented as part of the Klamath River Restoration Program. Models evaluating limiting factors and habitat availability for UKTR Chinook salmon suggest that crucial short-term actions are required to increase UKTR fall-run Chinook spawners and prevent further declines (Bartholow and Henrikson 2005). Restoration objectives for the TRRP provide reasonable targets for ameliorating limiting factors and increasing suitable habitat quantity and quality in 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 large-scale restoration efforts and improved flows to protect salmon populations.

Water temperatures may be more important to UKTR Chinook salmon than a restored natural flow regime per se, although the two are often interrelated. Protecting and restoring cool water habitats throughout the Klamath and Trinity watersheds will be essential to conserving these fish. Bartholow (2005) modeled a changing thermal regime in the Klamath River that could eventually eliminate UKTR Chinook spawning in the mainstem and disconnect critical spawning tributaries from the lower mainstem, an important migratory corridor. Both adult immigrants and juvenile emigrants are often exposed to water temperatures that are bioenergetically suboptimal or even lethal, especially in relation to increased incidence of disease outbreaks. The behavioral plasticity displayed by Chinook salmon indicates strong potential for management strategies that increase juvenile survival through maintenance of multiple life history patterns, rather than reliance upon hatchery production which may lead to loss of life history diversity. In main-stem habitats, Belchik (1997) demonstrated that UKTR Chinook use cool water areas as refuges; use of such habitats increases adult spawner and juvenile outmigrant survival. These key habitats should be conserved, monitored and, where possible, expanded. Many of the recommendations for conservation of UKTR spring-run Chinook also apply to fall-run Chinook (see UKTR spring-run Chinook account in this report).

Figure 2. Distribution of upper Klamath-Trinity rivers fall-run Chinook salmon, *Oncorhynchus tshawytscha* (Walbaum), in California.

UPPER KLAMATH-TRINITY RIVERS SPRING-RUN CHINOOK SALMON
Oncorhynchus tshawytscha (Walbaum)

Status: Critical Concern. Small, self-sustaining, wild populations occur in the Salmon River and South Fork Trinity, where they are highly vulnerable to climate change, poaching, and other stressors. Recent basin-wide abundances are thought to be approximately 3% of historical run size.

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 black coloration of the lower jaw make Chinook distinguishable from other sympatric salmonid species. Klamath River Chinook possess significant differences from Sacramento River Chinook in the number of gill rakers and pyloric caeca, with 12-13 rough, widely spaced gill rakers on the lower half of the first gill arch and 93-193 pyloric caeca (Snyder 1931, McGregor 1923). Dorsal fin ray, anal fin ray and branchiostegal counts are significantly different from Columbia River Chinook (Snyder 1931, Schreck et al. 1956). They have 10-14 major dorsal fin rays, 13-16 anal fin rays, 14-19 pectoral fins rays and 10-11 pelvic fin rays. Branchiostegal rays number 13-18 and there are 131-147 scales along the lateral line.

Spawning adult Chinook 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 Chinook (Snyder 1931). In 2004, Trinity River fall-run Chinook averaged 69 cm FL, with a maximum grilse size of 56 cm FL (CDFG 2006). 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 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; however, a more distinguishing adipose fin character is that of a pigmented upper edge and clear center and base.

Upper Klamath-Trinity rivers (UKTR) spring-run Chinook salmon enter natal streams during spring and early summer months as silvery, sexually immature adults and lack the breeding colors or elongated kype seen in fall-run Chinook salmon (Snyder 1931).

Taxonomic Relationships: The UKTR Chinook salmon ESU includes all naturally spawned populations of Chinook salmon in the Klamath River basin, upstream from its confluence with the Trinity River. This ESU is genetically distinguishable from other California Chinook ESUs (Waples et al. 2004).

Within the UKTR Chinook ESU, genetic analyses have demonstrated that stock structure mirrors geographic distribution (Banks et al. 2000). Fall- and spring-run Chinook salmon from the same subbasin appear more closely related to one another than each is to fall or spring-run Chinook from adjacent basins. This pattern is distinct from Chinook of different run timings in the Sacramento and Columbia rivers, where spring-run Chinook from different basins are more similar to one another than they are to fall-run Chinook within the same basin. Furthermore, fall-run Chinook salmon populations from both the Klamath and Trinity subbasins appear more similar to the respective spring-run Chinook populations within a given subbasin than they are to fall-run Chinook in Lower Klamath River tributaries.

Despite the lack of strong genetic differentiation from UKTR fall-run Chinook, the UKTR spring-run is treated here as a distinct taxon because it represents a life history strategy that is an essential adaptive component of the ESU and requires separate management strategies.

Life History: Adult UKTR spring-run Chinook salmon enter fresh water before their gonads are fully developed and hold in cold water streams for 2-4 months before spawning. They enter the Klamath estuary during spring and summer months, beginning in March and tapering off in July, with a peak between May and early June (Moffett and Smith 1950, Myers et al. 1998). A majority of late-entry fish are apparently of hatchery origin (Barnhardt 1994, NRC 2004). Leidy and Leidy (1984) noted that adult Trinity River spring-run Chinook migration continued until October. However, given this late-entry timing, it is unclear if these fish are sexually mature and capable of spawning with spring-run Chinook adults already in the system. Because this late spring-run type is limited to the Trinity River, it is possible these fish represent hybrid spring and fall-run Chinook from hatchery stocks. Biologists at the Trinity River Hatchery (TRH) classified Chinook salmon entering between September 3 and October 15, 2004, as spring-run Chinook (CDFG 2006). However, entry timing into the hatchery was artificially delayed until early September due to the fish ladder being closed. Spring–run Chinook have not been successfully held over for long periods of time in the hatchery due to space constraints and mortality (W. Sinnen, CDFW, pers. comm. 2013). Moffett and Smith (1950) noted that spring-run Chinook migrated quickly through the watershed; more recent work (Strange 2005) has confirmed this rapid migration pattern 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. Spring-run Chinook in the South Fork Trinity River begin spawning in late September, with a peak in mid-October (LaFaunce 1967). Trinity River spawning typically is 4-6 weeks earlier than that of fall-run UKTR Chinook in the same basin (Moffett and Smith 1950). Overlap between fall and spring-run Chinook spawning areas was historically minimal. In the South Fork Trinity River, the majority of spring-run Chinook spawning occurred upstream of Hitchcock Creek, above Hyampom Valley, while fall-run Chinook spawned below this point (LaFaunce 1967, Dean 1996). However, Moffett and Smith (1950) noted that spawning of the fall and spring-runs overlapped in October on suitable spawning riffles between the East Fork and North Fork Trinity River and that redd superimposition and hybridization may have occurred. In the Salmon River, overlap exists between spawning times of fall- and spring-run Chinook, although redds constructed upstream of the confluence of Matthews Creek are predominantly those of spring-run Chinook (Olson et al. 1992). Overall, spatial separation between the two runs in the Klamath-Trinity system occurs at approximately 518 m elevation.

Upper Klamath-Trinity rivers spring-run Chinook fry emerge from gravels 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-run Chinook. More recent reports (Leidy and Leidy 1984) suggest emergence begins as early as November in the Trinity River and December in the Klamath River, lasting until February.

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

Other life history attributes are similar to UKTR fall-run Chinook and other Chinook salmon taxa (Moyle 2002).

Habitat Requirements: UKTR spring-run Chinook enter the Klamath estuary when river water temperatures are at or above optimal holding temperatures (10-16°C; McCullough 1999). Temperatures in the Lower Klamath River typically rise above 20°C in June and can reach 25°C during August. Spring-run Chinook use thermal refuges in the estuarine salt wedge and associated nearshore ocean habitats 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. 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-run Chinook for more than 24 hours in 2005 (Strange 2005). Optimal adult holding habitat is characterized by pools or runs >1 m deep with cool summer temperatures (<20°C), all-day riparian shade, little human disturbance, and underwater cover such as bedrock ledges, boulders or large woody debris (West 1991). Because the Salmon River and its forks regularly warm to summer daytime peaks of 21-22°C, the best holding habitats are deep pools that have cold water sources, such as those at the mouths of tributaries or those deep enough to thermally stratify.

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

Distribution: Upper Klamath-Trinity rivers spring-run Chinook were once found throughout the Klamath and Trinity basins in suitable reaches of larger tributaries (e.g., Salmon River) or, flows permitting, utilizing smaller tributaries for holding and spawning. Historically, they were apparently 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 (Moffett and Smith 1950, Campbell and Moyle 1991). Their distribution is now restricted by dams, which block access to the upper Klamath and Trinity rivers. Passage of spring-run Chinook, through Upper Klamath Lake, to access holding and spawning grounds in the Sprague, Williamson and Wood rivers, was blocked in 1918 by completion of Copco 1 Dam (Hamilton et al. 2005). Currently, the Salmon River and its two forks and the South Fork Trinity River maintain self-sustaining populations in the Klamath River basin, with little hatchery influence. Approximately 177 km of habitat is accessible to spring-run Chinook in the Salmon River (West 1991) but most of it is underutilized or unsuitable. The South Fork Salmon River supports the majority of the spawning population, although spring Chinook redds have been found in some smaller tributaries of the Salmon River basin including Nordheimer, Knownothing, and Methodist creeks. In addition, there are dwindling populations of spring-run Chinook in Elk, Indian, Clear and Wooley creeks.

In the Trinity River basin, spring-run Chinook salmon historically spawned in the East Fork, Stuart Fork, Coffee Creek and the mainstem upper Trinity River (Campbell and Moyle 1991). The completion of Trinity Dam in 1962 and Lewiston Dam in 1963 blocked access to 56 km of what was considered to be prime spawning and nursery habitat on the mainstem as earlier recorded by Moffett and Smith (1950). Currently, Trinity River spring-run Chinook are present in small numbers in Hayfork and Canyon creeks, as well as in the North Fork Trinity, South Fork Trinity and New rivers, but only the South Fork population appears to maintain itself through naturally-spawned fish (W. Sinnen, CDFW, pers. comm. 2013). LaFaunce (1967) found spring-run Chinook spawning in the South Fork Trinity River, from about 3 km upstream of Hyampom and in Hayfork Creek up to 11 km above its mouth. 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).

Trends in Abundance: UKTR spring-run Chinook populations once likely totaled greater than 100,000 fish (Snyder 1931, Moyle 2002). The spring-run was thought to be the main run of Chinook salmon in the Klamath River, but the stocks had been depleted by the early 20th century as the result of irrigation, overfishing, mining, and other causes (Snyder 1931). Historic run sizes were estimated by CDFW to be at least 5,000 in each of the following Klamath tributaries: Sprague River (Oregon), Williamson River (Oregon), Shasta River and Scott River (CDFG 1990). The runs in the Sprague, Wood, and Williamson rivers were extirpated after the construction of Copco 1 Dam in 1918. Approximately 500 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-run Chinook returned to the hatchery 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 anthropogenic causes that depleted flows and altered habitats (Moyle 2002). In the middle reaches of the Klamath, spring-run Chinook have been extirpated from their historic habitats except the Salmon River and one of its tributaries, 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-run Chinook summer counts are highly variable over time (Figure 1). Both the lowest (90 in 2005) and highest (1593 in 2011) numbers on record have been documented in recent years. Overall, the number of spring-run Chinook salmon adults appears to be increasing ($p = 0.0015$; Quiñones 2011), but numbers continue to be a fraction of historical runs (Hamilton et al. 2011). Quiñones (2013) found significant cross correlation ($r(27) = 0.50$, $p = <0.05$) between spring-run Chinook returning to the Salmon River and TRH returns but trends may reflect similar responses of both wild and hatchery-reared fish to changing environmental conditions, rather than hatchery supplementation. Spring-run Chinook adult return numbers to the Trinity River were also significantly correlated to spring-run Chinook returns to TRH ($r(23) = 0.83$, $p = <0.05$). Spring-run Chinook returns to TRH fluctuated during the years of 1985 (increase), 1986 (increase), 1989 (decrease) and 1990 (increase), while Salmon River spring-run Chinook steadily increased over the same time period. The 1989 (decrease) and 1990 (increase) in returns of Spring-run Chinook to TRH may be explained by ocean conditions during those years. However, other increases (1985, 1986) likely reflected modification of hatchery infrastructure (construction of cement raceways) in the early 1980s that improved hatchery production (N. Hemphill, Trinity River Restoration Program, pers. comm. 2010) and probably led to an increase in short-term adult returns.

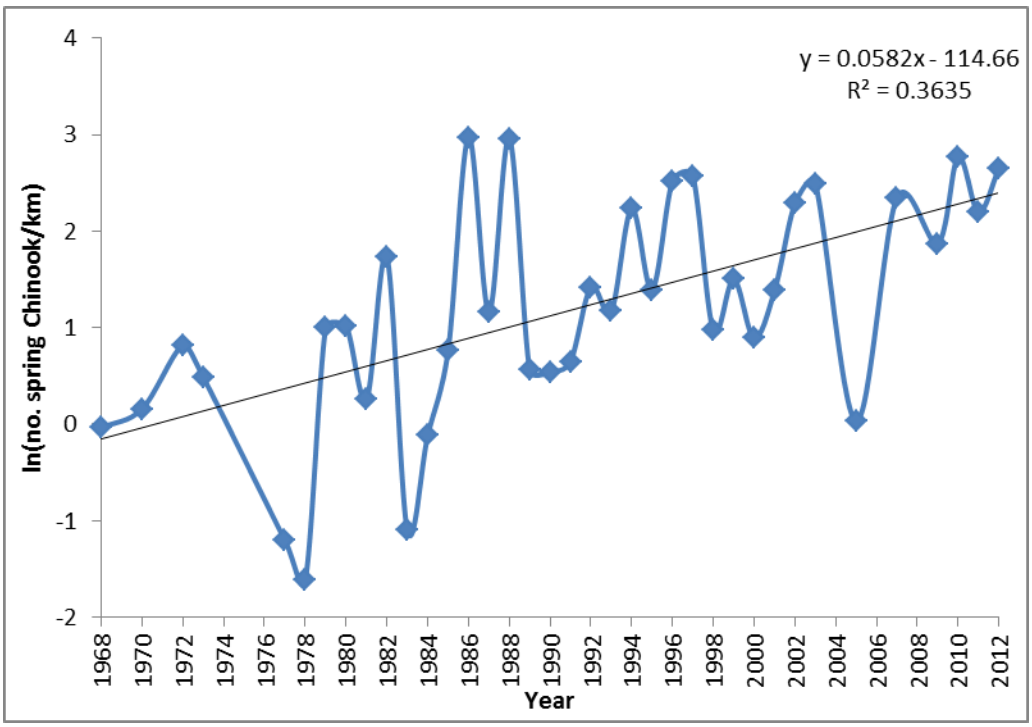


Figure 1. Number (ln) of spring-run Chinook salmon per kilometer observed in the Salmon River basin, excluding Wooley Creek, 1968-2012 (see Quiñones et al. 2013 for methods).

In the Trinity River, spring Chinook runs above Lewiston Dam have been extirpated but historically included more than 5,000 adults in the upper Trinity River and 1,000-5,000 fish in each of the Stuart Fork Trinity River, East Fork Trinity River and Coffee Creek (CDFG 1990). An average of 263 fish have been counted annually, over roughly the last thirty years, in the South Fork Trinity River, with runs 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, more recently, 232 fish between 2000-2005. Historically, 7,000-11,000 spring-run Chinook entered this stream (LaFaunce 1967) and outnumbered fall-run Chinook in the watershed. From 1980-2004, an average of 18,903 Chinook with spring-run life history returned above Junction City on the mainstem Trinity River.

In 2004, 16,147 spring-run Chinook salmon were estimated to migrate into this area, with 6,019 fish (37%) classified as spring-run Chinook entering Trinity River Hatchery.

While spring-run Chinook salmon are still scattered throughout the lower Klamath and Trinity basins, the only viable wild population appears to be limited to the Salmon River and South Fork Trinity River. Mainstem Trinity River numbers are presumably largely influenced by fish from the TRH, as are most tributary populations. Even if Trinity River tributary spawners are considered to be wild fish, the total number of spring-run Chinook in the UKTR system rarely exceeds 1000 fish and may drop to <300 in many years. Even recent large runs (~2500) of spring Chinook returning to the South Fork Trinity and Salmon River, combined, represent less than 3% of the total number of spring Chinook historically spawning in the basin.

In recent years, efforts have been made to compile spring-run Chinook numbers for all survey areas in a "mega-table" maintained by CDFW. However, these numbers represent varying degrees of effort (number of stream miles surveyed) between years, even within the same location, making trend analysis without standardization of the data unreliable (e.g., number of fish per kilometer; R. Quiñones, pers. observations, 2001-2011).

Nature and Degree of Threats: UKTR spring-run Chinook have been largely extirpated from their historic range because their life history makes them extremely vulnerable to the combined impacts from dams, mining, habitat degradation and fisheries, as well as many other anthropogenic (Table 1) and natural factors (e.g., ocean conditions).

Dams. A significant portion of the historic UKTR spring-run Chinook range 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 habitats of varying quality upstream of it and three other dams (Hamilton et al. 2005). These barriers to adult holding and spawning habitats, as well as juvenile nursery areas, have reduced the resilience of spring-run Chinook populations due to smaller population sizes, loss of available habitat, and reduction in spatial segregation between spring and fall-run Chinook. This has likely led to significant interbreeding between fall- and spring-run Chinook in the Trinity River (Myers et al. 1998). Dams and diversions have also led to the extirpation of spring-run Chinook in the Klamath and Shasta rivers due to alteration of water quality and temperature, channel simplification, and disconnection of mainstem river channels from floodplains below dams.

Alternately, there is potential for UKTR spring-run Chinook salmon to be restored to large portions of their former range in the Klamath-Trinity basins through dam removal, especially on the mainstem Klamath, and habitat restoration.

Agriculture. Most spring-run Chinook holding and rearing habitats are upstream of areas heavily influenced by agriculture (e.g., Scott and Shasta valleys); nonetheless, pasture and crops along the Shasta and Scott rivers are irrigated with cold water from rivers that would otherwise be available for instream flow. Agricultural return waters are generally warm, with low water quality and often deliver pesticides, fertilizers and other pollutant to streams.

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

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

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

Logging. Logging and associated road building have dramatically altered aquatic habitats in the Klamath and Trinity River basins (NRC 2004). Intensive and widespread logging began in the mid-19th century and legacy effects continue to affect rivers and streams in this region. Historic logging and the development of most early access roads occurred with little regard for environmental impacts. The steep and unstable slopes of this region, combined with local geology, make them particularly prone to erosion following road development and timber harvest (NRC 2004). Primary and ongoing impacts from this long history of timber operations in the Klamath-Trinity province include: increased erosion rates (delivering large amounts of sediments into streams which often imbed spawning areas and fill pools needed by holding adults in the summer), increased surface run-off of precipitation (and corresponding decreased aquifer recharge capacity, leading to increased frequency of flash flooding in streams), and increased summer stream temperatures due to lack of aquifer recharge and reduced canopy and riparian vegetation (instream shading). Thus, the low numbers of spring-run Chinook salmon currently using the heavily-logged South Fork Trinity River may be due to severe habitat degradation from the catastrophic 1964 flood, which altered channel morphology and hydrology and triggered landslides that filled in holding pools and covered spawning beds. Without the influence of long-standing timber harvest in this drainage, the impacts of the 1964 flood would likely have been considerably reduced. Other logging impacts include elimination of large, senescent trees that, under natural forest succession conditions, historically provided large wood as cover in streams for salmon and corresponding habitat complexity for all life history stages. As discussed in the UKTR fall-run Chinook salmon account, increasing stream temperatures are a growing threat to salmonids in these basins.

Fire. Altered forests in the region have also become more prone to large-scale catastrophic fires and increased erosion as a result. For example, over 50% of the Salmon River watershed, one of the few remaining strongholds in the Klamath Basin for UKTR spring-run Chinook, has been severely burned in the past 100 years (NRC 2004).

Mining. Mining has dramatically altered river and stream habitats in the Klamath-Trinity Province, with lasting legacy impacts in many areas. Intensive hydraulic and dredge mining for gold occurred in the 19th century and, depending on location, these activities caused severe stream degradation and alteration to channel morphology. Mining was a principal cause of decline of spring-run Chinook in the Scott River and large areas in the Trinity River, followed by some level of recovery after large-scale mining ceased. The Scott River was heavily altered in the Scott River Valley and remains so today, where a degraded river winds through immense piles of dredge tailings. Historic mining impacts still affect the Salmon River spring-run Chinook population as the estimated 16 million cubic yards of sediment disturbed between 1870 and 1950 are slowly transported through the basin (J. West, U.S.F.S., pers. comm. 1995). Mining and its legacy effects have disconnected and constricted juvenile salmon habitats, filled in adult holding habitats, degraded spawning grounds and altered the annual hydrograph of many streams. Pool in-filling is a particular problem because high stream temperatures have been demonstrated to reduce survival of both holding adults and rearing juveniles (West 1991, Elder 2002).

Although greatly reduced in scale from the past, mining continues in the region and may pose an increasing threat as the price of gold has increased sharply in recent years. Instream suction dredge mining has been particularly damaging to spring-run Chinook habitats, although suction dredging is currently under a moratorium in California. Suction dredging may cause chronic unnatural disturbance (noise, turbidity, sediment movement) in stream habitats that are already stressed by other factors and can, therefore, negatively impact fishes, benthic macroinvertebrates and other aquatic organisms (Harvey and Lisle 1998). Direct effects may include entrainment and possible mortality of invertebrates (food for juveniles) and small fish in dredges, habitat alteration including changes to channel structure and complexity, and increased turbidity, which may interfere with foraging of juvenile salmon and other fishes. Suction dredging (and the accompanying presence of people in stream channels, often for long periods of time) can also present a continuous disturbance to holding adults and juveniles during summer, increasing stress and probability of premature mortality. Of particular concern, in the Klamath, Salmon and Scott rivers and their tributaries, is the creation of piles of suction dredge tailings that may be utilized by spawning salmonids. Although these tailing piles are often comprised of suitable substrates for salmon redd creation and successful spawning, they are so unstable that they are likely to be mobilized during high flows, greatly reducing survival of embryos within the gravels. For more details on the effects of suction dredging see Harvey and Lisle (1998).

Harvest. Both illegal harvest of holding adults, as well as legal harvest of fish in the ocean and river fisheries, can limit the abundance of spawning populations. Holding adults are extremely vulnerable to poaching, although the extent to which poaching affects spring-run Chinook populations is largely undocumented. Because UKTR spring- and fall-run Chinook belong to the same Evolutionarily Significant Unit, they are taken legally in sport and commercial fisheries in the ocean. In 2013, CDFW regulations were as follows:
The Klamath River is "open to Chinook salmon fishing from Jan. 1 through Aug. 14 with a daily bag and possession limit of two salmon. The take of salmon is prohibited on the Klamath River from Iron Gate Dam downstream to Weitchpec from Jan. 1 through Aug. 14."
The Trinity River is "open to Chinook salmon fishing from Jan. 1 through Aug. 31. The daily bag and possession limit is two Chinook salmon. The take of salmon is prohibited from the confluence of the South Fork Trinity River downstream to the confluence of the Klamath River from Jan. 1 through Aug. 31." September 1 to December 31, a fall-run Chinook quota is in place, with a four fish limit, only three of which can be over 22 inches. All tributary waters along the main rivers are closed to fishing. These regulations provide some, but not full, protection from harvest pressures on spring-run Chinook, so recreational angling has the potential to limit the abundance of already small spring-run Chinook populations.

Hatcheries. The Trinity River Hatchery, below Lewiston Dam, is the only remaining hatchery in the Klamath basin that still cultures spring-run Chinook salmon. The impacts of hatchery propagation on wild spring-run Chinook salmon in the Trinity basin may be substantial; however, mixed runs of wild and hatchery-reared fish tend to segregate themselves above Cave Junction, with most hatchery fishes returning to TRH. Consequently, most naturally spawning fish are considered to be of wild origin (W. Sinnen, CDFW, pers. comm. 2013). Artificial selection in a hatchery environment has been demonstrated to reduce fitness in fish reproducing in the wild (Araki et al. 2007, 2009). Hatchery-reared spring-run Chinook are also more likely to hybridize with fall-run Chinook because of shifts in run timing and increased rates of straying of both spring- and fall-run fish.

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

Effects of Climate Change: UKTR spring-run Chinook have declined from being the most abundant run in the Klamath-Trinity system to one that is in danger of extinction. Climate change is predicted to lead to decreased snow pack (reduced instream flows), increased water temperatures and more variable flow fluctuations in many portions of their range. For example, the Salmon River already reaches summer temperatures of 21-23°C, approaching lethal temperature thresholds for salmonids. A 1-2°C increase in stream temperatures could greatly reduce the amount of suitable habitat available for spring-run Chinook and interfere with spawning and recruitment success. Reduced reservoir recharge may limit thermal stratification and the amount of cold water pool available for environmental flows via dam releases, which may be particularly acute in the Klamath River given the extensive network of dams. Climate change is also predicted to increase the frequency and intensity of both drought and flashy floods, both of which will likely limit spring-run Chinook abundance.

Climate change may also increase the incidence of disease outbreaks, due to warmer water temperatures, and lead to increased stress of adult salmon. For example, warmer temperatures favor epizootic outbreaks of *Ichthyophthirius multifiliis* and transmission of the bacteria *Columnaris*. Columnaris disease is associated with pre-spawn mortality of spring-run Chinook that are exposed to above-optimal water temperatures. Increased base flows likely reduce pathogen transmission risk during Chinook salmon migrations (Strange 2007), thus the predicted impacts from climate change (e.g., lower flows, warmer water temperatures, reduced availability of suitable habitats and corresponding increased densities of spawning adults or out-migrating juveniles) may enhance conditions favorable to disease outbreak. Moyle et al. (2013) rated the UKTR spring-run Chinook salmon as "critical vulnerability" to extinction because of the added effects of climate change on already diminished populations.

Status Determination Score = 1.7 - Critical Concern (see Methods section Table 2). The principal self-sustaining wild populations of UKTR spring-run Chinook exist in the Salmon and South Fork Trinity rivers; most other populations are small in number, influenced or supported by hatchery fish, and may not be self-sustaining. Upper Klamath-Trinity rivers spring-run Chinook are considered a Sensitive Species by the USDA Forest Service.

Metric	Score	Justification
Area occupied	2	Only Salmon River and South Fork Trinity River support wild, self-sustaining populations
Estimated adult abundance	2	Only a few hundred natural spawners support the population, with attendant impacts of small populations and hatchery influence
Intervention dependence	3	Hatchery stocks appear to be maintaining the run in the mainstem Trinity; dam removals in Klamath system needed to restore access to historic range; runs dependent upon dam flow releases
Tolerance	2	Narrow temperature tolerance (<20ºC) during holding and spawning migrations; temperatures and other factors in summer holding areas limit suitable habitat
Genetic risk	1	Hybridization with fall-run and/or hatchery spring-run is occurring in some watersheds; fitness reduction may result from hybridization with hatchery stocks
Climate change	1	Increased temperatures, reduction in suitable habitats, increased density of adults and juveniles, and potential increase of disease outbreaks will further limit populations
Anthropogenic threats	1	See Table 1
Average	1.7	12/7
Certainty (1-4)	3	Fairly well studied

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

Management Recommendations: Monitoring of spring-run Chinook occurs annually throughout the Klamath-Trinity system. However, data collected need to be standardized so that trend analyses can be performed. Data from existing surveys demonstrate that suitable habitat exists for adult holding and spawning, yet spring-run Chinook abundance, while showing an upward trend, continues to fluctuate at low numbers (Quinones et al. 2013). Over-summering behavior and associated habitat requirements are the most distinctive life history attributes of spring-run Chinook. The rarity of cool water refuges throughout the UKTR Chinook ESU range is already a significant threat to spring-run Chinook persistence. Spring-run Chinook may be particularly susceptible to warming trends, especially in the face of predicted climate change impacts. As such, reconnecting historic habitats in the upper watersheds of the Klamath and

Trinity rivers and their tributaries to lower main stem river habitats below major dams is necessary for long-term persistence of this run. Such restoration efforts would increase habitat availability for spring-run Chinook and remove barriers, which negatively impact water quality and quantity. UKTR spring-run Chinook are a good indicator species, due to their narrow tolerances to water quality and temperature, as well as their presence during some of the most challenging portions of the year for riverine habitation. The near extirpation of this sentinel species in the Klamath River subbasin indicates that other anadromous stocks that rely on freshwater habitats during their juvenile and adult life histories may also be at risk. Specific management recommendations for spring-run Chinook salmon in the Klamath and Trinity basins include:

- Remove dams on the mainstem Klamath to allow access to historic upstream spawning and rearing areas. Of all salmonids in this drainage, spring-run Chinook would likely benefit the most from increased access to cold-water habitats.
- Restore the Shasta River as a cold-water refuge for all salmonids in the Klamath basin by recapturing spring flows in the river, reducing ground water extraction and, possibly, removing Dwinnell Dam.
- Manage the Salmon River as a spring-run Chinook and summer steelhead refuge by restricting extractive resource use of the river in summer (e.g., continue moratorium on suction dredging).
- Develop a program to investigate impact(s) of the TRH on spring-run Chinook populations (e.g., number of hatchery-reared fishes spawning in the wild, genetic shifts in population) and manage hatchery production accordingly.
- Develop restoration actions and priorities for reducing the impacts of sediment inputs from roads, logging and other activities into rivers of the Klamath-Trinity system, especially on public lands.
- Determine the harvest rate of sport, commercial and traditional fisheries on UKTR spring-run Chinook to improve fisheries management.
- Limit harvest to a mark-selected, in-river, fishery for TRH-produced spring-run Chinook.

Figure 2. Distribution of upper Klamath-Trinity rivers spring-run Chinook salmon, *Oncorhynchus tshawytscha*, in California.

SOUTHERN OREGON – NORTHERN CALIFORNIA COASTAL CHINOOK SALMON
Oncorhynchus tshawytscha

Status: Moderate Concern. Distribution of Southern Oregon - Northern California Coastal (SONCC) Chinook salmon in California is limited to a few watersheds. Their status is threatened by interactions with hatchery stocks, habitat degradation (especially estuary alteration) and fisheries harvest.

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

Adult lengths can be greater than 140 cm SL but usually fall between 75 and 80 cm SL. Adult Klamath River Chinook salmon are considered to be among the smallest of the Chinook salmon found in the Pacific Northwest and, when compared to Sacramento River Chinook of the same length, are more rounded and heavier (Snyder 1931). Adult Chinook salmon in California can reach weights of 38.6 kg, but average between 9-10 kg. Sexually mature adults are uniformly colored in dark burgundy or olive brown. Males develop humped backs and hooked jaws and are usually darker than females. Chinook juveniles have 6-12 parr marks equal in width or wider than the spaces between them and an adipose fin with dark coloration along the upper edge only although some parr develop spots on the dorsal fin as they grow, most have clear dorsal fins.

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

Life History: Most SONCC Chinook spawning adults migrate into rivers in the late fall, when increases in stream flow facilitate access into Klamath Mountain Province streams. Adults enter tributaries of the lower Klamath River from September through December and spawning occurs in the latter part of this period and into January (Leidy and Leidy

1984). In the Smith River, spawning typically occurs between October and February. Chinook salmon enter Blue Creek in September and spawning peaks after fall rains, usually in November (Gale et al. 1998). However, pulses of spawning fish continued to enter Blue Creek through December, perhaps reflecting differences in reproductive maturity between earlier and later arrivals (Gale et al. 1998). Differences in reproductive behavior were observed for females in Smith River tributaries. The amount of time that a female spent on a redd decreased from 10-21 days to 5-10 days as the spawning season progressed (Waldvogel 2006). Spawners in Blue Creek are primarily 3 years old with a few age 4 and age 5 fish; in addition there are a few grilse, reproductively mature age 2 fish (Gale et al. 1998). In Mill Creek, from 1993-2002, most spawners were 3 year old fish (62%) but, from 1981-1992, 4 year old females comprised the majority of spawners (66%) (Waldvogel 2006).

Chinook salmon fry emerge in lower Klamath tributaries from February through mid-April and most migrate into the ocean in the same year (Leidy and Leidy 1984). In 1995-96, fry outmigration from Blue Creek began before mid-March, peaked in late April and late May, and continued into August (Gale et al. 1998). Fry grew to 103 mm FL throughout the period of outmigration (Gale et al. 1998). Early outmigrants traveled quickly into the estuary. However, larger juveniles can spend months rearing in fresh water before beginning outmigration (Sullivan 1989). In the Smith River, juvenile Chinook were most commonly observed in low salinity zones (<5%) in the upper estuary and were associated with abundant cover from overhanging riparian vegetation (Quiñones and Mulligan 2005). Ocean survival is likely enhanced by longer periods of rearing in fresh water. Of the juvenile outmigrants from Blue Creek in 1996, 28% reared extensively in freshwater. Approximately 5% of the juveniles rearing in Hurdygurdy Creek (Smith River) in 1987 and 1988 remained in the stream to rear after spring flows receded (McCain 1994). However, high flows in the spring of 1988 likely shortened the length of freshwater residency in that year. Juvenile Chinook salmon in tributaries of the Sixes River, Oregon, (northern range of SONCC Chinook) also displayed varying degrees of freshwater residency; some moved into the ocean within weeks of emergence, while others reared in freshwater from two months to more than one year (Reimers 1971). Scale aging revealed that most adults returning to spawn had reared in freshwater for two to six months as juveniles (Reimers 1971). Once in the ocean, Chinook seem to follow defined migration routes but can alter migration patterns to use regions with temperatures of 8°-12°C (Hinke et al. 2005).

Habitat Requirements: Spawning habitats are characterized by large cobbles and sufficient flows to facilitate oxygen delivery to developing embryos. Most SONCC Chinook salmon spawn in the middle reaches of coastal tributaries, but in the Smith River small tributaries are also commonly used for spawning. In Blue Creek, holding spawners favored deep pools and areas with runs and pocket water with fast flows (Gale et al. 1998). Adults have been observed spawning at depths ranging from a few centimeters to several meters, with water velocities of 15-190 cm/sec; however, preferred spawning habitat was at depths between 25 to 100 cm, with water velocities from 30 to 80 cm/sec. Embryo survival is enhanced when water temperatures are between 5°-13°C and oxygen levels close to saturation (Healey 1991). Water temperature requirements of Chinook salmon are discussed in Moyle et al. (2008).

Embryos incubating in optimal conditions generally hatch within 40-60 days, but remain in the gravel as alevins for an additional 4-6 weeks, usually until the yolk sac is absorbed. Juveniles will continue to rear in streams throughout the summer if water temperatures remain <20°C (Gale et al. 1998). Rearing habitats are characterized by shallow water in areas with overhanging riparian vegetation that provides cover, food and habitat complexity.

Distribution: Southern Oregon - Northern California Coastal Chinook salmon are found in streams from Cape Blanco, OR (south of the Elk River) south to the Klamath River, including Klamath River tributaries from the mouth to the Trinity River confluence. In California, SONCC Chinook salmon were found historically in the many small tributaries of the lower Klamath River that are within the ocean-influenced fog belt (USFWS 1979). In 1977 and 1978, SONCC Coastal Chinook were found in Hunter, Terwer, McGarvey, Tarup, Omagar, Blue, Surpur, Tectah, Johnson, Mettah, and Pine creeks (USFWS 1979). In 2000, they were also found in Hoppaw, Saugep, Waukell, Bear, Pecwan, and Roaches creeks, but not in Omagar and Surpur creeks (Gale and Randolph 2000). Chinook salmon from the Rogue and Smith rivers have different ocean migration patterns than Chinook salmon in ESUs to the south (Gale et al. 1998), with a greater tendency for adults in the ocean to stay north of Cape Blanco (Brodeur et al. 2004). Klamath River Chinook salmon stocks tend to associate with the California Current further south.

Trends in Abundance: Southern Oregon - Northern California Coastal Chinook in California are currently found in only a few, small lower Klamath tributaries, including Blue Creek, as well as the Smith River. The abundance of the fall-run appears stable, although populations in the Klamath basin have been adversely affected by land use practices, particularly logging. Spring-run Chinook salmon appear to have largely disappeared from this ESU (Moyle 2002). The majority of SONCC Chinook salmon originate from the Rogue River in Oregon. Individuals from the lower Klamath River tributaries and Smith River contribute to the population to a lesser extent. Historically, some 2,000 – 3,000 adult Chinook salmon spawned in the lower Klamath River each year (Moyle 2002). In 1960, an estimated 4,000 Chinook salmon spawned in lower Klamath tributaries (USFWS 1979) while, in 1978-79, the number of spawners dropped to 500 (USFWS 1979). However, there is considerable natural variability in the number of spawners observed from year to year. In 1995 and 1996, respectively, 236 and 807 fall Chinook salmon were observed in Blue Creek (Gale et al. 1998). A study of Chinook salmon spawning in Blue Creek indicated that surveys observed about half the actual number of spawners, with spawner estimates in survey years (1995- 2009) ranging from 100-2400 fish (Antonetti 2009). The numbers of late fall-run Chinook spawning in Blue Creek from 1988 to 2009 showed an increasing trend (Quiñones at al. 2014 a,b). However, the time series also showed a significant correlation to hatchery returns, suggesting that numbers are supplemented by hatchery strays or that hatchery fish encounter similar ocean conditions as naturally produced fish of the same cohort. Annual numbers of adult Chinook salmon in the Smith River are estimated to range from 15,000 - 30,000 fish (Moyle 2002), but robust population estimates have not been established. There is no evidence of a decline in fall-run spawner abundance in the Smith River. The numbers of spring-run Chinook adult in the Smith River were probably always low, with 0-21 fish counted in recent years (34 to 53 miles surveyed) (Reedy 2005).

Nature and Degree of Threats: Although poorly documented, SONCC Chinook salmon abundance in California appears to be mostly limited by habitat alteration, hatcheries and fisheries harvest.

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

Agriculture. In the Smith River estuary, construction of dikes and reclamation of lands for agriculture and grazing have reduced the amount of juvenile rearing habitat by more than 40% (R. Quinones, pers. observations, 2007). Diversions of water for flower bulb cultivation, alfalfa production and other purposes in the Smith River drainage may affect salmon outmigration, depending on seasonal timing and volume of water diversions.

Grazing. Grazing of riparian areas by feral cattle has been identified as significant cause of habitat degradation in Blue Creek drainage, causing stream bank sloughing and reduced riparian vegetation (Beesley and Fiori 2008). Cattle grazing along the Smith River estuary has also degraded stream banks and reduced or eliminated riparian vegetation (R. Quinones, pers. observation, 1997-2002).

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

Logging. The coastal watersheds of northern California have been heavily logged, beginning in the mid-19th century (USFWS 1979). Logging has altered most coastal streams by increasing solar input and water temperatures through reduced tree canopy cover, introduction of heavy loads of fine sediments that bury spawning gravels and fill pools, and increased surface runoff of precipitation, leading to increased frequency of flash flooding in streams. In many streams, extensive networks of logging roads (mostly unimproved) in north coastal drainages have blocked salmon spawning migrations. Improperly built stream crossings (culverts, bridges and other structures) have created fish passage barriers, impeding fish passage although, in recent decades, many passage impediments have been rectified. Road construction in lower Blue Creek has altered stream morphology and reduced recruitment of large woody debris into the stream channel (Beesley and Fiori 2008). Roads have increased fine sediment delivery to streams in the Smith River basin (Six Rivers National Forest 2011).

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

	Rating	Explanation
Major dams	Low	No dams on the Smith River but dams on Klamath River may affect migration patterns and reduce habitat suitability
Agriculture	Low	Agriculture is a primary land use in the Smith River estuary; estuary alteration from wetland reclamation, diking, diversions, pollutant and pesticide inputs; however, potential effects have not been studied
Grazing	Medium	Cattle grazing in the Smith River estuary has contributed to habitat degradation; cattle grazing in the Blue Creek drainage has substantially impacted riparian and aquatic habitats
Rural residential	Medium	Rural development is increasing in north coastal California watersheds, contributing to habitat degradation, water diversion, and pollutant inputs into streams
Urbanization	n/a	
Instream mining	n/a	
Mining	n/a	
Transportation	Medium	Roads are primary sources of sediment inputs in SONCC watersheds
Logging	Medium	Most watersheds have been heavily logged in the past; legacy effects remain in many watersheds
Fire	Low	Predicted increases in severe wildfires may lead to increased habitat degradation, especially outside the fog belt
Estuary alteration	Medium	Land reclamation for agriculture in the Smith River estuary has reduced juvenile rearing habitats
Recreation	Low	Most habitats are in smaller tributaries not heavily used by swimmers and boaters
Harvest	Medium	Harvest has presumably reduced Chinook numbers to a fraction of historic numbers
Hatcheries	Medium	Hatchery fish probably have negative effects on Klamath River populations but impacts to the main population in the Smith River are likely minimal
Alien species	Low	Few alien species are reported for the Klamath and Smith rivers

Table 1. Major anthropogenic factors limiting, or potentially limiting, viability of populations of SONCC Chinook salmon in California. Factors were rated on a five-level ordinal scale where a factor rated “critical” could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated “high” could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated “medium” is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated “low” may reduce populations but extinction unlikely as a result; and a factor rated “no” has no known negative impact to the taxon under consideration. Certainty of these judgments is moderate. See methods section for descriptions of the factors and explanation of the rating protocol.

Estuary alteration. As discussed under agriculture, the capacity of the Smith River estuary to support juvenile salmon rearing has been greatly reduced due to prevailing land uses and associated habitat degradation.

Harvest. Commercial, sport, and tribal fisheries have likely already reduced SONCC Chinook salmon abundance in the past. However, recent regulations to protect Upper Klamath-Trinity fall-run Chinook from overharvest (e.g., closure of fishery in 2006 by Pacific Fisheries Management Council) may have reduced harvest rates of SONCC Chinook salmon from the lower Klamath and Smith rivers in recent years.

Hatcheries. Although hatcheries are not operated in tributaries to the lower Klamath River, SONCC Chinook in the basin are likely interacting with salmon produced by hatcheries located upstream on the main-stem Klamath (Iron Gate Hatchery) and Trinity (Trinity River Hatchery) rivers. Hatchery-produced juvenile Chinook salmon migrate through the middle Klamath River in late summer (USFWS 2001), around the same time that wild SONCC Chinook are also outmigrating. Hatchery-produced adults may stray into lower Klamath tributaries, perhaps interbreeding with and altering the genetic makeup of, wild SONCC Chinook salmon. The abundance of adult Chinook salmon returning to Blue Creek was found to be significantly correlated with returns of adult Chinook salmon to Trinity River Hatchery, suggesting that hatchery strays are contributing to the population (Quiñones 2013). In the Smith River basin, about 50 female Chinook salmon are spawned each year by Rowdy Creek Hatchery juveniles are released in the spring and have been observed displacing other salmonids (e.g., steelhead trout) from estuarine habitats (Quiñones, personal observations, 1997-2001).

Effects of Climate Change: Predicted climate change impacts to north coastal streams are expected to be less than those to inland waters in California, since the maritime climate and associated fog belt will likely offset air temperature increases. However, coastal areas have already experienced a 33% reduction in fog frequency since the early 20th century and further reduction is predicted to increase summer drought frequency and duration along the west coast (Johnstone and Dawson 2010). Predicted increases in air temperatures (up to 10°C by 2100; Dettinger 2005), in combination with reduced fog frequency and associated increases in evapotranspiration, may negatively impact juvenile rearing habitats decrease (e.g., warmer water temperatures, lower flows). Poor ocean conditions (e.g., reduced upwelling, higher temperatures), may also reduce ocean survival and limit gene flow between more northern populations. In addition, sea level rise will likely reduce rearing habitats in estuaries, unless similar habitats become available in upstream areas as estuaries 'back upstream' as a result of sea level rise. Moyle et al. (2013) rated the SONCC Chinook salmon ESU "critically vulnerable" to extinction in 100 years due to the added impacts of climate change, although uncertainty in this regard is high.

Status Determination Score = 3.3 - Moderate Concern (see Table 2 and Methods section). The SONCC Chinook salmon ESU in California is limited to a few watersheds that are impaired, to varying degrees, by habitat degradation associated with land and water use practices (Table 1). This ESU was determined by NMFS on September 16, 1999 to not warrant listing under the Federal Endangered Species Act, although SONCC

Chinook salmon 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 the principal populations, along with smaller populations in tributaries
Estimated adult abundance	3	No systematic surveys have been performed but between 5,000 and 50,000 spawners in the Smith is probable in most years; <1000 spawners in Klamath River tributaries annually
Intervention dependence	4	California populations are largely self-sustaining but some supplementation by hatcheries is likely
Tolerance	3	Multiple juvenile life histories and spawner age diversity demonstrate physiological tolerances
Genetic risk	3	Limited hatchery operations in California portion of range but some concern for hybridization with hatchery 'strays' from other ESUs
Climate change	3	Fall-run is least vulnerable to climate change in north coastal streams of California, since they spawn later in the year and scouring of redds is less likely to influence juveniles; possible sea level rise may negatively affect important rearing habitats in estuaries; Smith River likely to retain runs under worst-case scenarios through end of the century
Anthropogenic threats	3	Multiple threats rated as "medium" (Table 1)
Average	3.3	23/7
Certainty (1-4)	2	Least studied of Klamath River Chinook runs

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

Management Recommendations: The persistence of the two largest populations of SONCC Chinook salmon (in Smith River and Blue Creek in the lower Klamath River) suggests that conservation of this ESU within California is largely reliant upon protection of spawning and rearing habitats in these two watersheds. Increased protection of these populations would also facilitate recolonization of other degraded streams in the ESU, as habitats recover and are restored, potentially expanding the distribution and increasing the abundance of SONCC Chinook salmon.

It has been shown that interactions of wild Pacific salmon with hatchery-produced conspecifics can reduce both the overall fitness of a population (Araki et al. 2008) and its local adaptability (Reisenbichler and Rubin 1999). To determine the status of Chinook salmon within this ESU, both population monitoring and genetic studies are needed to determine levels of introgression between wild and hatchery stocks and to determine the status of spring-run Chinook salmon within this ESU. Such studies may be of particular value in the Smith River drainage (the largest free-flowing river system in the state) which has been designated a National Recreation Area, and is included in the National Wild and Scenic River program. These designations imply that priority should be given to maintaining self-sustaining, wild populations of native salmonids and other organisms. The introduction of hatchery salmon from Rowdy Creek Hatchery on the Smith River may therefore be in conflict with these designations and also with the status of the Smith River as a 'stronghold' for wild salmon.

Figure 1. Distribution of Southern Oregon - Northern California Coastal Chinook salmon, *Oncorhynchus tshawytscha,* in California.

CENTRAL VALLEY FALL-RUN CHINOOK SALMON
Oncorhynchus tshawytscha ESU

Status: High Concern. The abundance of Central Valley (CV) fall-run Chinook salmon has varied significantly in recent years, but the run is widespread and the number of spawners typically exceeds 100,000 fish. The run continues to be of concern because it is supported, to a large extent, by hatchery production which has ecological and genetic impacts on the sustainability of the run. Reliance upon hatchery stocks to augment low numbers of natural spawning (wild) CV fall-run Chinook is unlikely to be sustainable and likely will lead to, if not already, a largely homogenized population with reduced life history variability. Central Valley fall-run Chinook salmon are also one of the main populations contributing to California and Oregon ocean and inland fisheries. It is unknown what impacts the fisheries may be having on the wild stocks in the run.

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

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

Taxonomic Relationships: Central Valley fall-run Chinook salmon are part of the CV Chinook complex consisting of four life-strategy runs differentiated by genetic and life history characteristics, including time of spawning migrations, maturity of fish entering fresh water, spawning location, incubation times, and out-migration timing of juveniles (Moyle 2002). The seasonal runs of CV Chinook salmon (winter, spring, fall and late fall) are more closely related to each other than they are to populations outside the CV (Williams 2006). Winter- and spring-runs are recognized as distinct ESUs, while the National Marine Fisheries Service groups the fall-run and late fall-run in a single ESU. This report differs from that taxonomy in that we regard the late-fall run to be a distinct life-history strategy, with specific management concerns. CDFW continues to work with NMFS in its scientific evaluation of the genetic relationship of late-fall-run (see Williams 2006).

Life History: Chinook salmon life history strategies are differentiated by the timing of immigration, a fact implicit in the naming of the different "runs" according to the season of their spawning migration. However, movement between habitat types, synchronized to specific life stages, defines the entire life history of salmon. Adult spawning migration timing is only one differentiating characteristic of the multiple life history attributes of CV Chinook salmon (Table 1). This account focuses on life history and migratory characteristics specific to the CV fall-run which have classic "ocean type" life history that minimizes time spent in fresh water. Because both fry and smolts out-migrate in spring before water temperatures become too warm in summer, the fall-run can exploit the extensive lower elevation reaches of Central Valley rivers and streams, where temperatures exceed thermal tolerances during summer and early fall. In contrast, spring and winter-run Chinook salmon exhibit a stronger "stream-type" life history, which is dependent upon year-round cool freshwater habitat; as such, spawning locations for these runs are restricted to higher elevation stream reaches where year-round cool water is found.

Adult CV fall-run Chinook salmon enter rivers as mature individuals and move relatively quickly to spawning grounds. Spawning usually occurs within several weeks to two months of freshwater entry. Peak spawning time is typically in October-November, but can continue through December and into January. Juveniles typically emerge from the gravel in December through March and rear in fresh water for 1-7 months, usually moving downstream into large rivers within a few weeks. Salmon smolts initiate migration during storm events and flow is positively correlated with migration rate (McCormick et al. 1998, Michel et al. 2013). In the clear upper reaches of the Sacramento River, out-migrating smolts employ a nocturnal migration strategy, a behavior likely influenced by predation. Turbidity also has a strong positive relationship with increased survival during out-migration, likely by decreasing predation efficiency. However, this relationship is also influenced by the strong positive association between turbidity and large flow events (Michel et al. 2013). The slowest movement rates were observed in the estuary, with intermediate rates observed in the lower Sacramento River (Michel et al. 2013).

In the past, before entering the San Francisco Estuary, CV fall-run juveniles likely foraged extensively on floodplains. Today, less than 10% of historical CV wetland habitats remain accessible to CV salmon (Frayer et al. 1989). Juvenile fish foraging in these highly productive habitats grow much more quickly than those in major river channels (Sommer et al. 2001, Jeffres et al. 2008). Historically, this rapid growth before ocean entry was likely very important to the survival of fall-run juveniles, which enter the ocean at relatively small size and young age compared to other CV runs.

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

There are many exceptions to this general life cycle, including juveniles that spend as long as one year in freshwater. However, the general attributes of fall-run Chinook salmon that have made them so well adapted to low-elevation regulated rivers have also made them the preferred run for use in hatcheries; they can be spawned as they arrive and juveniles can be reared for a short time before being released.

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

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

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

Chinook salmon use the largest substrate of any California salmonid for spawning, a mixture of large gravel and small cobble. Such coarse material allows sufficient water flow through the substrate to provide oxygen for developing embryos, while simultaneously removing their metabolic waste. As a result, the selection of redd sites is often a function of gravel permeability and subsurface water flow. 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. Because females dig the redds, redd size is a function of female size as well as the degree of substrate mobility. Redds are typically over 2-15 m^2 in size, where the loosened gravels permit steady interstitial flow of well oxygenated water (Healey 1991). For maximum embryo survival, water temperatures must be between 5º and 13º C and oxygen levels 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.

Once alevins emerge and become fry, they tend to aggregate along stream edges, seeking cover in vegetation, swirling water, and dark backgrounds. As they grow larger and become increasingly vulnerable to avian predators, especially herons and kingfishers, they move into deeper (>50 cm) water. Larger juveniles may utilize the tails of pools or other moderately fast-flowing habitats, where food is abundant and some protection from predators is afforded. As juveniles 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 preferred by migrating juveniles as daytime refuges.

Juveniles use off-channel habitats, including floodplains, for rearing where they grow faster because of warmer temperatures and abundant food (Sommer et al. 2001, Limm and Marchetti 2006, Jeffres et al. 2008). Historically, these habitat types were widespread along the valley reaches of rivers and likely contributed to the large numbers of salmon produced in the past.

Off-channel habitat was also important in the San Francisco Estuary (e.g., tidal marshes), but these habitats are now largely unavailable, cut off from main river channels behind levees.

The route by which Sacramento River smolts pass through the Delta has a significant effect on survival. Those that migrate through the interior Delta have higher mortality rates than fish remaining in the mainstem Sacramento River (Perry et al. 2010).

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

Distribution: Central Valley fall-run Chinook salmon historically spawned in all major rivers of the CV, migrating as far as the Kings River to the south and the Upper Sacramento, McCloud, and Pit rivers to the north. Today, in the Sacramento and San Joaquin River watersheds, they spawn upstream as far as the first impassible dams. Passage into the mainstem San Joaquin River, above the confluence with the Merced River, is intentionally blocked at the CDFW-operated weir at Hills Ferry. Overall, it is estimated that over 70% of spawning habitat has been blocked by dams (Yoshiyama et al. 2001), although coldwater releases from dams now allow spawning where it did not formerly exist (Yoshiyama et al. 1998). Habitat for fall-run Chinook salmon spawning has been impacted less by dam construction than spawning habitat for winter and spring-run Chinook salmon, because the fall-run historically spawned only in low elevation reaches, up to 500 – 1,000 feet above sea level (Yoshiyama et al. 2001). Levees also block access for juveniles to the historic floodplain and tidal marsh rearing habitats.

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

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

Trends in Abundance: The historic abundance of fall-run Chinook salmon is difficult to estimate, because populations declined before extensive monitoring occurred and good records were kept. Hydraulic mining operations during the Gold Rush Era buried spawning and rearing areas under mining debris before the first estimates of salmon numbers were made. Likewise, Chinook salmon were extensively harvested in-river during the 19th century and accurate, detailed records of run and river source were not documented. The best estimates of historic numbers suggest that fall-run Chinook salmon were one of the largest runs in the CV, with about a million spawners returning per year (Yoshiyama et al. 1998).

Yoshiyama et al. (1998) reported that exploitation by fisheries and alteration of California rivers during the Gold Rush had already reduced fall-run Chinook salmon abundance to about 10% of historical numbers by the 1940s. Construction of large dams throughout the CV in the 1940s-60s further reduced wild Chinook numbers. However, the extent of these impacts on CV Chinook populations is uncertain because artificial propagation began in this era and no effort was made to differentiate wild Chinook from those produced in hatcheries. Until recent years, escapement estimates for CV fall-run salmon included both hatchery and natural-origin fish with the relative proportions unknown. .

From 1967 to 1991, an average of 250,000 adult fish returned to spawn with an additional 375,000 harvested each year in the commercial and sport fisheries (USFWS 2011). From 1992 to 2006, average escapement was nearly 400,000 with an annual average of 484,000 harvested in the fisheries. In 2007, escapement plummeted to fewer than 100,000 fish with about 121,000 harvested in fisheries, prompting the first-ever closure of the California ocean salmon fishery. Returns dropped to 71,000 in 2008 and, in 2009, escapement reached a record low of 53,000 spawners, even though the ocean fisheries remained closed (CDFW GrandTab 2011). Escapement in 2010 increased to 163,000 with a limited ocean fishing season, harvesting 20,400 fish. Central Valley escapement continued to rebound to approximately 228,000 fish in 2011 and 342,000 fish in 2012.

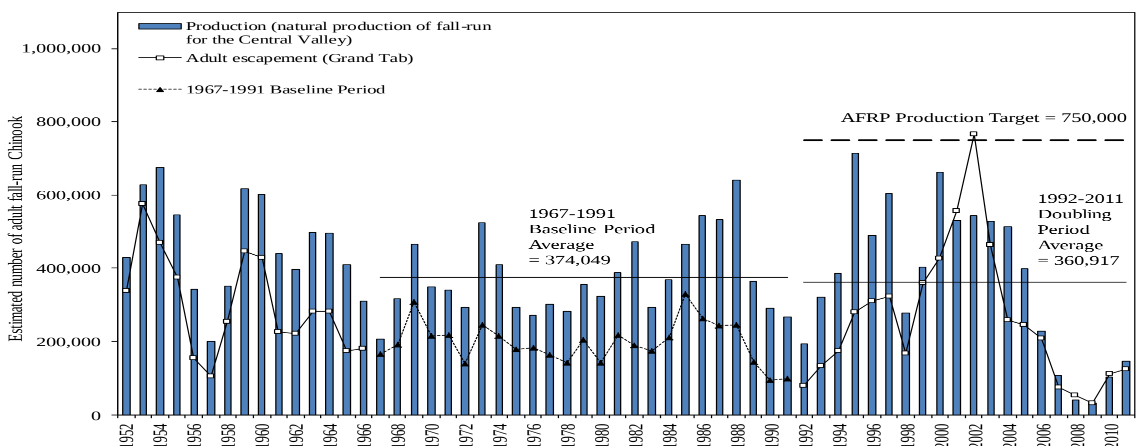


Figure 2. Estimated yearly natural production and in-river escapement of adult fall-run Chinook salmon in the Central Valley rivers and streams. 1952 - 1966 and 1992 - 2011 numbers are from CDFG Grand Tab (Apr 24, 2012). 1967-1991 Baseline Period numbers are from Mills and Fisher (CDFG, 1994).

Figure 1. Estimated natural production and in-river escapement of adult fall-run Chinook salmon in Central Valley rivers and streams. 1952-1966 and 1992-2011 data are from CDFW GrandTab (updated April 24, 2012). 1967-1991 Baseline Period data are from Mills and Fisher (CDFG 1994).

The effects of hatchery production on abundance and population dynamics of CV fall-run Chinook has been poorly documented, but recent studies are allowing a better analysis of stock composition in the CV. Data from the CV Constant Fractional Marking Program indicates that a high proportion of fall-run Chinook salmon spawning in-river are of hatchery origin, particularly in streams with large hatchery facilities. Recent studies of otolith michrochemistry suggest the same (Barnett-Johnson et al. 2007, Johnson et al. 2012, Kormos et al. 2012). In addition, stray rates between river basins are variable and in some cases relatively high (Kormos et al. 2010). Genetic evidence suggests that CV fall-run Chinook populations are now genetically homogenous (Williamson and May 2003, Lindley et al. 2009).

Nature and Degree of Threats: Widespread and intensive development of the CV over the last 150 years has simplified river, floodplain, and estuarine habitats, altered ecological processes (i.e., hydrology, sediment transport, nutrient cycling) and fundamentally altered the CV Chinook salmon complex, from a diverse collection of numerous wild populations employing diverse life histories to one dominated by fall-run Chinook salmon produced in four large hatcheries (Lindley et al. 2009). Important factors continuing to limit population viability of CV fall-run Chinook salmon include: water management, habitat loss and alteration, climate change, and hatchery practices.

Dams. Large dams on the Sacramento River and its tributaries have blocked fall-run Chinook salmon access to historic spawning grounds. Habitat downstream of the dams has been altered; some changes have negatively impacted remaining spawning and rearing habitats. Regulated flows and resulting water temperatures are sometimes unsuitable for salmon spawning and rearing. Spawning gravel can be limited by lack of recruitment from upstream areas and deposition of fine sediments. Most large dams now have flow requirements for salmon spawning, rearing, egg incubation and juvenile emigration, but flows may not provide optimum habitat or water conditions. Large quantities of gravel are now trucked to spawning areas below dams to improve spawning habitat; however, effectiveness of these restoration actions at the population level is not well documented and require regular, human intervention (Mesick 2001, Wheaton et al. 2004).

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

The largest diversions in the Central Valley are those of the State Water Project (SWP) and the federal Central Valley Project (CVP) in the south Delta, which export water for both agricultural and urban use. They entrain large numbers of fall-run Chinook salmon (as well as salmon of other runs), especially from San Joaquin River tributaries (Kimmerer 2008). These diversions have louver screens that divert salmon to be salvaged from the projects by capture, trucking, and then release downstream in the Delta. However, both direct and indirect mortality associated with these operations is likely high (Kimmerer 2008). Direct mortality is also caused by high predation rates in Clifton Court Forebay, from which the SWP pumps water prior to running it through the salvage facility.

	Rating	Explanation
Major dams	Medium	Dams prohibit access to large geographic areas that supported historic spawning, alter flows, and simplify stream geomorphology; however, flow releases generally provide adequate water quality and temperatures below major dams; lack of gravel recruitment below dams necessitates augmentation in many lower river reaches
Agriculture	Medium	Diverted water reduces stream flow and entrains juvenile salmon; levees protecting agricultural lands limit salmon access to floodplains, tidal marshes, and other important habitats
Grazing	Low	Relatively little grazing takes place on the CV valley floor
Rural residential	Low	Generally minimal impact on large river systems (e.g., Sacramento), but increasingly connected to urbanized areas
Urbanization	Medium	Urban areas widespread and growing in many portions of historic range; urban landscapes generally simplify habitats, impair aquatic ecosystem function and pollute streams
Instream mining	Medium	Gravel pits in rivers are problematic in some locations, particularly in the San Joaquin River basin
Mining	Low	Legacy effects of hydraulic and hard rock gold mining remain; impacts may still be severe at a localized scale
Transportation	Low	Most Chinook streams have roads and railroads along them, often leading to habitat simplification
Logging	Low	Little logging in the CV although logging may affect upper portions of CV watersheds
Fire	Low	Little threat of fire in the CV although fire may affect upper portions of CV watersheds and effects can be propagated downstream
Estuary alteration	High	San Francisco Estuary is a highly altered system; fall-run Chinook salmon, however, have short residence periods in the estuary; the Sacramento-San Joaquin Rivers delta is greatly altered and current physical and water habitat conditions impact effective migration of adults and juveniles in both river basins
Recreation	Low	Recreation can disturb redds and spawners
Harvest	Medium	Ocean and inland fisheries may harvest natural-origin (wild spawned) fish at unsustainable rates
Hatcheries	Medium	A large proportion of fall-run Chinook are produced in hatcheries
Alien species	Low	Introduced species may increase predation, competition, or decrease food supply

Table 3. Major anthropogenic factors limiting, or potentially limiting, viability of populations of Central Valley fall-run Chinook salmon. Factors were rated on a five-level ordinal scale where a factor rated “critical” could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated “high” could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated “medium” is unlikely to drive a species to extinction by itself but

contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction is unlikely as a result. A factor rated "n/a" has no known negative impact. Certainty of these judgments is high. See methods section for descriptions of the factors and explanation of the rating protocol.

Indirect mortality resulting from changes in Delta hydrology due to project operations is likely considerably higher than direct mortality. Salmon are often diverted into unfavorable parts of the Delta where habitat conditions are poor and predation is high. In general, when flows are higher and diversion rates are lower, survival of outmigrants tends to be higher, although there is no simple relationship between diversion rates and salmon survival (Brandes and McLain 2001). San Joaquin fall-run Chinook salmon are affected to a greater extent by Delta pumping, because juveniles emigrate in the vicinity of the export facilities and are, therefore, vulnerable to entrainment.

Agriculture in the CV also contributes to loss of juvenile habitat by limiting access, via an extensive network of flood protection levees, to the shallow riverine habitats needed for feeding and protection from predators during migration, management of floodplain for agriculture and not fish habitat, and limiting expansion of native riparian habitat. Construction of levees to channelize 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. Bank hardening has been enhanced by the reduction of peak flows. Reduction of floodplain habitat has likely contributed to population declines of CV fall-run Chinook salmon. Recent studies have demonstrated the importance of floodplains for increased juvenile salmon growth and survival (Sommer et al. 2001, Jeffres et al. 2008).

Agricultural development can also degrade water quality conditions for Chinook salmon rearing in CV streams and the Delta. A new threat is the use of pyrethroid pesticides, which are particularly toxic to fish. Although mortality events are periodically recorded, the interacting effects of multiple pollutants on juvenile salmon survival are largely unknown. Even if pollutants are sublethal in concentration, they can stress both adult and juvenile fish, making them more vulnerable to disease, predation and other stressors.

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

Mining. Historic (and, to a lesser degree, ongoing) gold and gravel mining have dramatically altered many CV streams. Hydraulic and dredge mining in the 19^{th} and early 20^{th} centuries caused major morphological and hydrological changes in many rivers, degrading salmon spawning and rearing habitats. Many of these waterways are still recovering. Deep gravel pits in a number of CV rivers (e.g. Tuolumne, Merced, San Joaquin) reduce water velocities and allow for the aggregation of predatory fishes, potentially increasing mortality of juvenile salmon moving downstream. In the past, Iron Mountain Mine, northwest of Redding, drained highly acidic water laden with heavy metals into the Sacramento River, resulting in acute mortality to Chinook salmon. Although discharge is now highly controlled, failure of the Spring Creek retention reservoir could result in impacts to aquatic life.

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

Despite long-term monitoring, causes of apparent high mortality rates as fish pass through the estuary are poorly understood. General observations suggest that rearing conditions in the estuary are often poor; highest survival occurs during wet years, when passage through the estuary is likely most rapid (Brandes and McLain 2001, Baker and Morhardt 2001). Flooding in wet years also increases rearing habitat in the Delta and Yolo Bypass, which may also have a positive effect. To improve survival, most hatchery juveniles are transported and released downstream of the Delta. Transporting smolts improves survival, but it also increases rates of straying upon return as adults. High straying rates contribute to homogenization of population structure and reductions in fitness by facilitating gene flow between populations in different streams, thus reducing biocomplexity within the CV Chinook salmon complex.

The Delta ecosystem is as, if not more, altered than the estuary. Land and water management practices have altered the delta's landscape and ecological processes such that fall-run Chinook salmon and other native fishes encounter poor to extremely poor habitat conditions when migrating through the Delta's waters.

Harvest. In most years, salmon populations support major sport and commercial fisheries along the California and Oregon coasts and major inland sport fisheries in freshwater. Hatchery fish can sustain higher harvest rates than wild fish, but the two cannot be discriminated within the fishery. It is, therefore, possible that existing recreational fisheries, in spite of being highly regulated and managed, may harvest natural-origin fish at unsustainable rates (Williams 2006). Wild-spawned fish, while a fraction of the overall fall-run, may be of particular importance in maintaining genetic attributes that increase life history diversity and adaptability to localized selection processes, particularly in the face of changing environmental conditions, such as those predicted under climate change models (e.g., Hayhoe et al. 2004, Mote et al. 2005).

Fisheries also affect Chinook salmon populations through continual removal of larger and older individuals. This selection results in spawning runs made up primarily of two and three-year-old fish, which are smaller and, therefore, produce fewer eggs per female. The removal of older fish also removes much of the buffering that salmon populations have against natural disasters, such as severe drought, that may eliminate an entire cohort. Under natural conditions, the four- and five-year-old fish still in the ocean help to buffer against population declines due to short-term environmental changes. In order to protect the low stock of Sacramento River fall-run Chinook salmon, ocean salmon fisheries were greatly restricted in 2006-2010 by the National Marine Fisheries Service and the Pacific Fisheries Management Council (Congressional Record, 50 CFR Part 660). The Chinook salmon sport fishery in the Sacramento River system was also severely restricted in 2008 and 2009. Since that time, ocean and inland fisheries have not been limited by low abundance of CV fall-run Chinook.

Hatcheries. Returns of CV fall-run Chinook were very low in 2007 and 2008. The proximate cause of the poor returns is thought to have been poor ocean conditions that resulted in low juvenile survival when outmigrating smolts first entered the Gulf of the Farallones (Lindley et al. 2009). However, the homogenizing influence of hatcheries on population diversity has made the fall-run more susceptible to adverse conditions, such as drought and corresponding low flows in freshwater habitats, or periods of reduced upwelling in coastal waters (Moyle et al. 2008, Carlson et al. 2011). The negative effects of hatchery production on wild stocks can be divided into ecological and genetic impacts, although the two interact considerably.

Ecological effects include competition, predation, and disease transfer from hatchery stocks to wild populations (Allendorf and Ryman 1987). Competition between hatchery and naturally-produced Chinook can reduce abundance (Pearsons and Temple 2010), growth rate (Williams 2006) and survival of wild juveniles in river, estuarine and marine habitats (Nickelson et al. 1986, Levin et al. 2001, Levin and Williams 2002, Nickelson 2003). Hatchery releases can even exceed the carrying capacity of ocean habitats, particularly in times of low ocean productivity (Beamish et al. 1997, Levin et al. 2001), resulting in high ocean mortality (Beamish et al. 1997, Heard 1998, Kaeriyama 2004). Historically, a high degree of genetic variation and the availability of complex and diverse habitats resulted in diverse salmon behavior and many distinct life history strategies which ensured persistence in California's extremely variable climate. Hatchery propagation has not only narrowed this behavioral variation in hatchery stocks (most fish are released over a short time period), leaving them vulnerable to climatic anomalies (ocean conditions, drought, etc.) and management decisions (water releases, storage), but it has also resulted in domestication of the stock, favoring a salmon genome that is well adapted to comparatively stable hatchery conditions but may be unfit under variable natural conditions.

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

Effects of Climate Change: Climate change may be one of the biggest threats to the persistence of CV salmon (Williams 2006, Katz et al. 2012). At the southern edge of the Chinook salmon range along the Pacific Coast, the CV fall-run, at times, already experiences environmental conditions near the limit of its tolerance (Moyle et al. 2008). For instance, summer temperatures in some streams already exceed 22°C (California Data Exchange Center 2009). Thus, small thermal increases in summer water temperatures could result in suboptimal or lethal conditions and consequent reductions in distribution and abundance (Ebersole et al. 2001, Roessig et al. 2004). Changes in precipitation patterns in California may also significantly alter CV fall-run habitats. Climate change models predict that a larger proportion of annual precipitation will fall as rain, rather than snow, running off quickly and earlier in the season. With less water stored in snowpack, reservoirs will potentially have less water available for fishery releases, particularly during summer and fall months. The available water is also likely to be warmer. During summer and fall, high water temperatures will be exacerbated due to the lower base flows resulting from reduced snowpack (Hamlet et al. 2005, Stewart et al. 2005).

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

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

Status Determination Score = 2.7 – High Concern (see Methods section Table 2). The Central Valley fall-run Chinook is listed as a species of special concern by NMFS. The NMFS status review concluded that "…high hatchery production combined with infrequent monitoring of natural production make assessing the sustainability of natural production problematic, resulting in substantial uncertainty regarding this ESU (Myers et al. 1998)".

Metric	Score	Justification
Area occupied	2	Some indication of natural self-sustaining populations in the upper Sacramento River watershed
Estimated adult abundance	4	Annual spawning returns generally exceed 100,000 fish
Intervention dependence	2	The majority of remaining spawning and rearing habitat is dependent on instream flow releases from major dams, gravel augmentation and other ongoing efforts; population appears largely dependent on hatchery augmentation
Tolerance	3	Moderate physiological tolerance, multiple age classes
Genetic risk	2	High hatchery production has resulted in genetic homogenization of the run, reducing overall fitness

Climate change	3	The ‘ocean’ life history strategy makes them the least vulnerable of all CV Chinook runs to extirpation; however, models suggest dramatic changes to lower elevation CV rivers and streams
Anthropogenic threats	2	See Table 3
Average	2.7	19/7
Certainty	4	Well studied although high uncertainty about ocean stage

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

Management Recommendations: Before CV winter and spring-run Chinook salmon were listed, virtually all salmon conservation actions were focused on fall-run Chinook, because it was the most abundant run that supported fisheries. Prior to the passage of the Central Valley Project Improvement Act by Congress in 1992, which established the Anadromous Fish Restoration Program (AFRP), actions to protect fall-run salmon were either focused on 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 salvage salmon entrained at the large pumping plants in the south Delta, barriers to passage were removed in some streams, and monitoring continued. The AFRP and its associated agencies began to take additional actions to enhance natural salmon populations, including evaluating 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 is charged to plan "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-run Chinook.

The listing of winter-run Chinook salmon as endangered (1989 by CDFW; 1994 by NMFS) and spring-run Chinook as threatened in 1998 (both State and federal listings) increased the urgency of salmon restoration efforts and actions to benefit these two runs have benefited fall-run Chinook salmon 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), largely came through CALFED, established in 1994. Fall-run Chinook salmon should also benefit considerably from additional measures required by NMFS (e.g., http://swr.nmfs.noaa.gov/ocap.htm) to enhance winter and spring-run Chinook salmon populations in the river.

In the San Joaquin tributaries, considerable effort has been made to improve conditions for fall-run Chinook salmon, including modified flow regimes, better habitat management, reducing impacts of instream gravel pits and other actions. However, these actions have not prevented continued declines in fall-run Chinook numbers, presumably as the result of factors outside the San Joaquin basin, especially in the south Delta.

There are four general directions management actions could take: (1) improving population monitoring, (2) improving habitats, (3) adjusting water management, and (4) improving hatchery management practices.

Improving population monitoring. Expanded monitoring of fall-run Chinook salmon in Central Valley streams and the ocean is essential for improved management. At present, our understanding of the relationships between ocean conditions and salmon survival is largely unstudied, with studies and restoration actions implemented long (sometimes years) after a significant event affecting populations occurs. An investment in research on the effects of ocean conditions on survival of juvenile Chinook salmon would have large benefits for improved salmon management and population recovery.

In 2012, the CDFW completed the *Central Valley Adult Chinook Salmon In-stream Escapement Monitoring Plan*. The plan reviewed existing monitoring programs and made recommendations for program improvements. Implementation of the recommendations, already in progress, is expected to yield more accurate estimates of Chinook salmon escapement to the Central Valley for use in harvest management and restoration planning.

Additional emphases need to be given to where fish are naturally spawning and rearing, the relative importance of specific rivers to the run, the genetic diversity across the Central Valley and with the various hatchery stocks, and the genetic differences between fall-run and late-fall-run Chinook salmon.

Habitat improvement. In the Central Valley, recovery actions have focused on habitat restoration. Because habitat diversity is essential to maintaining life-history diversity, conservation strategies that restore and improve physical habitat quality, extent, and connectivity are essential tools in improving the resilience of salmon populations. For example, efforts should be made to reconnect river channels to floodplains. Infrastructure and operational changes needed to increase habitat value for salmon and other native fishes in the Yolo Bypass and other floodplains should be prioritized. Modification of flows below many dams could also improve habitat conditions for salmon. Improving habitat for rearing in the Delta and San Francisco Bay, reducing inputs of toxins to the estuary, continuing with improvements of upstream habitats, managing floodplain areas such as the Yolo Bypass for salmon, and restoring the mainstem San Joaquin River are all important management actions that need further attention and resources.

Adjusting water management. Water management from each major reservoir, in-river (both the Sacramento and San Joaquin rivers), and in the Delta greatly affects spawning, rearing, and migration of both juvenile and adult life stages. A comprehensive plan for the run as well as specific evaluation and planning for rivers where salmon spawn and rear would both directly benefit the run, as well as have important ramifications for habitat restoration and sustainability, response to climate change, and ensuring sustainable river and ocean fisheries.

Improving hatchery practices. In 2012, the California Hatchery Scientific Review Group completed a comprehensive review of California's anadromous fish hatcheries, including those hatcheries rearing Central Valley fall-run Chinook salmon (California HSRG 2012). Implementation of the review's recommendations over the next ten years will significantly reduce the genetic and ecological impacts of hatchery production on Central Valley fall-run Chinook salmon.

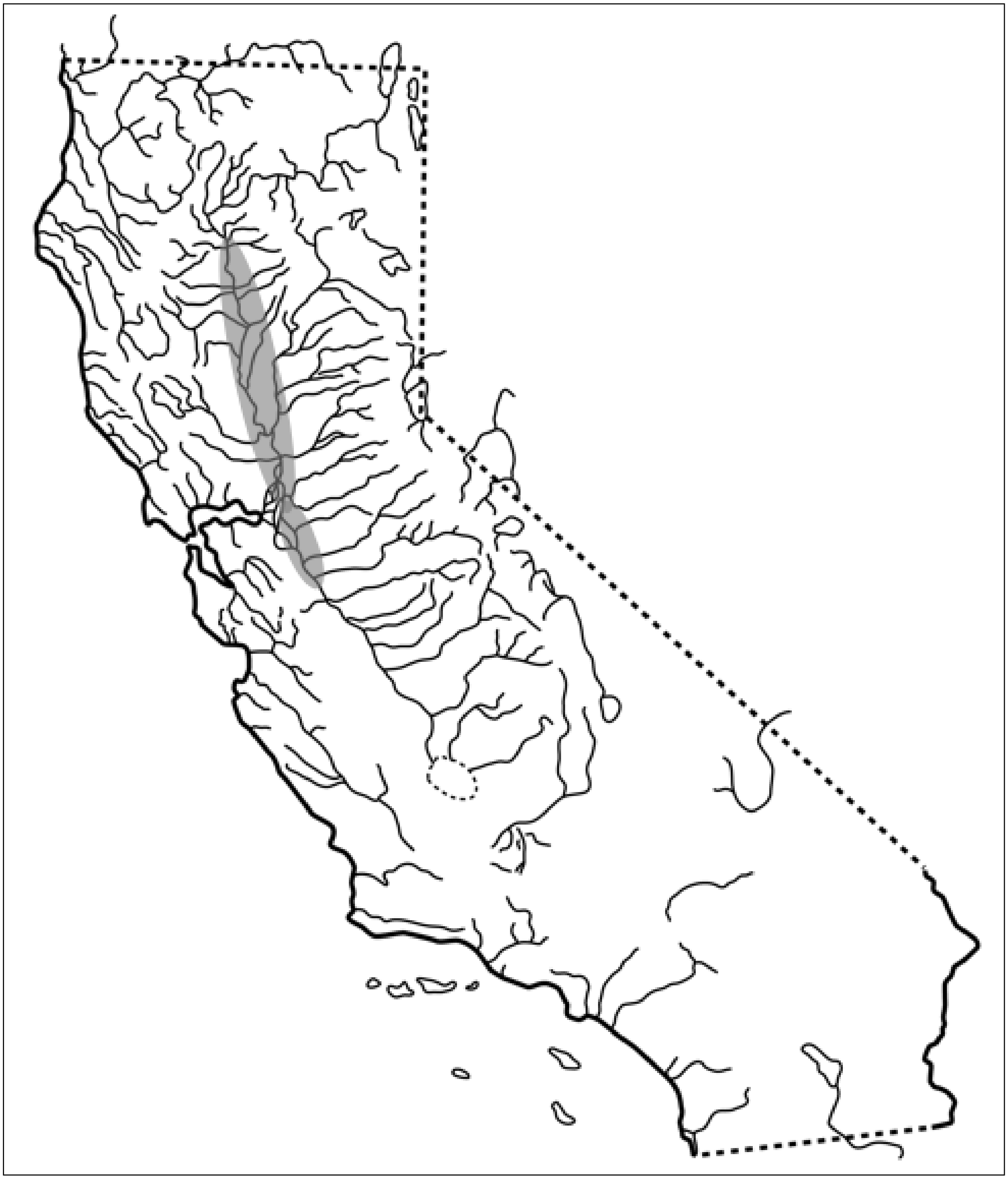


Figure 2. Distribution of Central Valley fall-run Chinook salmon, *Oncorhynchus tshawytscha* ESU, in the Sacramento and San Joaquin rivers of California.

CENTRAL VALLEY LATE FALL-RUN CHINOOK SALMON
***Oncorhynchus tshawytscha* ESU**

Status: High Concern. The Central Valley (CV) late fall-run Chinook salmon have been extirpated from the majority of their native spawning habitat, which now lies upstream of Shasta Dam. Although late fall-run Chinook salmon occur in tributary streams to the Sacramento River, most spawn in the main river. The primary population depends on dam operations for maintenance of suitable habitat. While affected to a lesser degree than fall-run Chinook salmon, this run remains of ongoing concern due to the strong influence of salmon hatchery stocks in the CV and associated potential ecological and genetic impacts to the sustainability of the run.

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

Spawning adults are olive brown to dark maroon, without streaking or blotches on the sides. 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 (juveniles) 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 the Evolutionary Significant Units (ESUs) of Chinook salmon in the CV, so separation is based on genetic data and life history characteristics.

Taxonomic Relationships: The four runs of Chinook salmon in the CV differ in 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, Table 1). For management purposes, juvenile salmon are assigned to winter, spring, fall, and late fall-runs by size criteria, reflecting different spawning times and rearing conditions. While these criteria are useful, they are not very precise, given natural variability in lengths for any population and the presence of so many hatchery fish in the system (e.g., juvenile hatchery fish tend to be larger than wild members of the same run). The utility of the size criteria decreases rapidly downstream from Red Bluff Diversion Dam (RBDD).

All populations within the CV are more closely related to each other than they are to populations outside the valley. Because of their similar arrival time to fall-run Chinook, late fall-run Chinook were only recognized as a distinct run in 1966, after the construction of RBDD allowed careful observation of run timing for the first time. As salmon passed through the dam, two distinct peaks were observed. NMFS currently groups late fall-run with the fall-run ESU, though there are life history differences between the two runs. Yoshiyama et al. (1998), Moyle (2002), and Williams (2006) and others in recognizing the Central Valley late fall-run to be a distinct taxonomic entity with a unique evolutionary trajectory (as evidenced by a distinct life history strategy) and with specific management concerns of its own. It is still unclear if this is a unique ESU. Williams (2006) described genetic techniques being applied to late-fall-run Chinook salmon to investigate if late-fall run can be distinguished from the other runs. Currently, CDFW recognizes late-fall run as a unique life history strategy and, partnering with federal scientists, is further investigating the genetic relationship of this run with other runs in the Central Valley. Currently, late fall-run and fall-run are considered races under a single ESU.

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

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

Life History: Chinook salmon life history strategies are differentiated by immigration timing, a fact implicit in the naming of the different "runs" according to the season of their spawning. However, movement between habitat types synchronized with changes in developmental life stage defines the entire life history, not simply adult spawning migration (Table 1). For instance, the fall-run has classic "ocean type" life history that minimizes time spent in freshwater. Because both fry and smolts out-migrate before water temperatures become too warm in summer, the fall-run can exploit extensive valley floor reaches of the CV where temperatures exceed thermal tolerances during summer and early fall. In contrast, spring and winter-run exhibit a "stream-type" life history that is dependent upon year-round, cool, freshwater habitat for both adults (which arrive in spring and mature while over-summering in foothill streams) and juveniles, which regularly spend more than a year in rivers before out-migration. Spring-run spawning and rearing habitat is, therefore, restricted to the higher elevation portions of the CV, where cool summer temperatures can be found in snow melt-fed rivers. The basic life history of late fall-run Chinook salmon is intermediate to the "ocean type" fall-run and the "stream type" spring-run, because adults arrive in fresh water already mature and spawn quickly after arriving (similar to fall-run) but juveniles regularly over-summer, out-migrating in their second year of life (similar to spring-run). The details of late fall-run life history, however, are much less well known than those of other CV runs because of the comparatively recent recognition of this run, coupled with its tendency to ascend and spawn at times when the Sacramento River is likely to be high, cold and turbid. This combination of factors makes this run particularly difficult to study.

Late fall-run Chinook salmon migrate upstream in December and January as mature fish, although their migration has been documented from November through April (Williams 2006). Historically, the spawning adults would have been comprised of a mixture of age classes, ranging from two to five years old. Currently, most of the run is composed of three-year olds. Spawning occurs primarily in late December and January, shortly after the fish arrive on spawning grounds, although it may extend into April in some years (Williams 2006). Emergence from the gravel begins in April and all fry have usually emerged by early June. 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 may out-migrate at younger ages and smaller sizes during most months of the year (Williams 2006).

Habitat Requirements: The specific habitat requirements of late fall-run Chinook salmon have not been determined but they are presumably similar to other CV Chinook salmon runs. It is believed that optimal conditions fall within the range of physical and chemical characteristics of the unimpaired Sacramento River above Shasta Dam. For a more detailed review of CV Chinook salmon requirements, see Williams (2006), Stillwater Sciences (2006), and Moyle (2002).

Distribution: The historic distribution of late fall-run Chinook salmon is not well documented, but they most likely spawned in the upper Sacramento and McCloud rivers, in reaches now blocked by Shasta Dam and flooded by Shasta Reservoir, as well as in portions of major tributaries that provided 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).

Currently, late fall-run Chinook are found primarily in the Sacramento River, where most spawning and rearing of juveniles takes place in the reach between RBDD and Redding (Keswick Dam). Varying percentages of the total run spawn downstream of RBDD in some years. In 2003, for example, 3% of the late fall-run spawned below the dam, while, in 2004, no spawning occurred below the dam (Kano 2006a, b). R. Painter (CDFW, pers. comm. 1995) indicated that late fall-run Chinook have been observed spawning in Battle Creek, Cottonwood Creek, Clear Creek, Mill Creek, Yuba River and Feather River, but these are presumably a small fraction of the total population. The Battle Creek spawners are likely derived from fish that strayed from Coleman National Fish Hatchery.

Trends in Abundance: The historic abundance of Central Valley late fall-run Chinook is not known because it was recognized as distinct from fall-run Chinook only after RBDD was constructed in 1966. In order to pass the dam, salmon migrating up the Sacramento River ascended 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 counts, although salmon passed over the dam during every month of the year. In the first 10 years of counting (1967-1976), the late fall-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, when operation of the RBDD was changed, estimates of abundance have been less accurate but, from 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. Reduced accuracy in fish counts resulted from the opening of RBDD gates to provide free passage of the listed winter-run Chinook salmon from September 15 to May 15 each year, starting in 1992. This made estimation of late fall-run Chinook spawner numbers more difficult because many late fall-run fish swam freely through the open gates and could not be counted as they had been previously while ascending the fish ladders. From 1992-1996, estimates were made by extrapolating from counts made on only part of the run. These numbers are extremely low and unreliable. In 1998, CDFW initiated surveys based on carcass and redd counts from airplanes and estimated that over 35,000 late fall-run Chinook had spawned above RBDD. Subsequent surveys have resulted in lower estimates (e.g. 5,000 in 2003), with variability from year to year. Spawner surveys and estimates seem to indicate that measures taken to benefit winter-run Chinook salmon have also benefited the late fall-run. Fish from Coleman National Fish Hatchery on Battle Creek are contributing at a low rate to the spawning population in the mainstem Sacramento River.

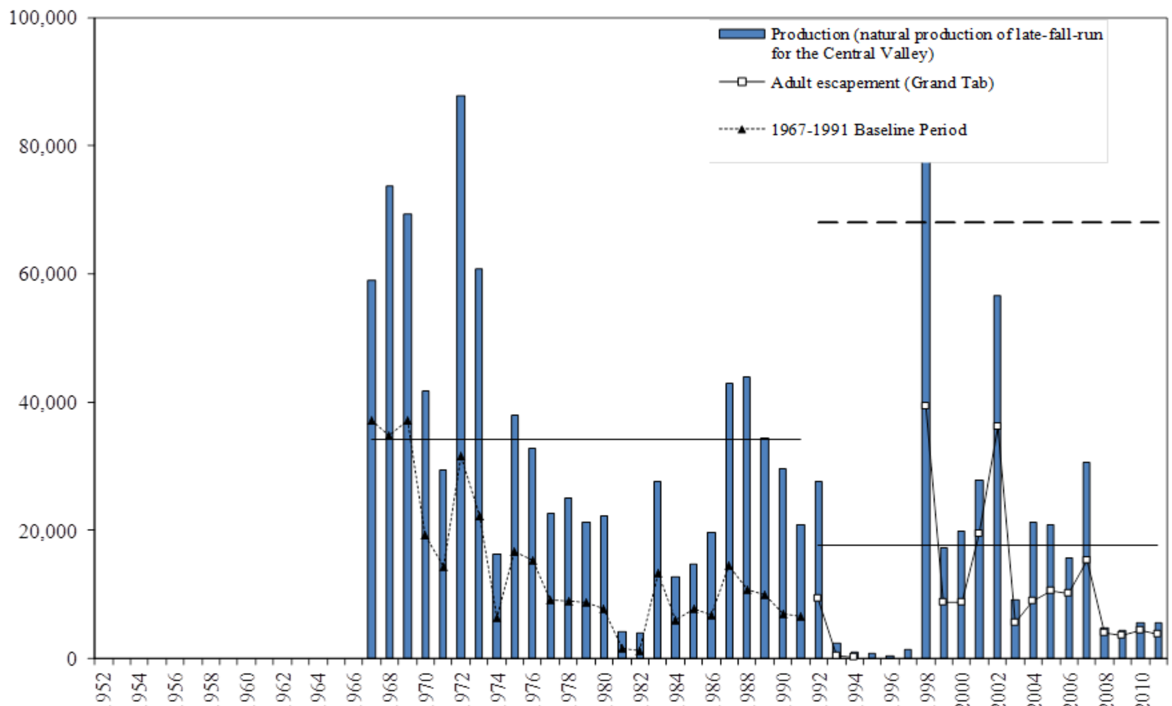


Figure 1. Estimated annual adult natural production and in-river adult escapement estimates for late fall-run Chinook salmon in the Central Valley. 1992 - 2011 numbers are from CDFW Grand Tab (Apr 24, 2012). 1967-1991 baseline period numbers are from Mills and Fisher (CDFG 1994).

	Rating	Explanation
Major dams	High	Dams block access to the majority of historic spawning grounds; however, current operation of Shasta Dam creates some replacement habitat
Agriculture	Medium	Levees reduce access to floodplains and other important habitats; diversions and agricultural return water decrease water quantity and quality
Grazing	Low	Little grazing on valley floor
Rural residential	Low	Source of minor changes to river banks and pollution
Urbanization	Low	Urban areas along Sacramento River and tributaries may restrict habitat and decrease water quantity and quality
Instream mining	Low	Gravel mining and legacy effects of placer mining may continue to impair habitats
Mining	Low	Discharge from Iron Mountain Mine has been attenuated, now posing only a slight risk to water quality in the upper Sacramento River
Transportation	Low	Roads line banks and cross rivers, contributing to habitat simplification and sediment or pollutant input
Logging	Low	Generally low impact; occurs at higher elevations
Fire	Low	Few impacts on mainstem river likely
Estuary alteration	High	San Francisco Estuary is a highly altered system; fall-run Chinook salmon, however, have short residence periods in the estuary; the Sacramento-San Joaquin rivers Delta is greatly altered and current physical and water habitat conditions impact effective migration of adults and juveniles in both river basins
Recreation	Low	Recreation (boating, wading, angling) can disturb spawners and migrants
Harvest	Medium	Ocean and inland fisheries may harvest natural-origin (wild spawned) fish at unsustainable rates
Hatcheries	Medium	Based on recent coded-wire tag recoveries, a small proportion of the spawning population is of hatchery-origin but still of concern
Alien species	Low	Predation and competition from introduced fishes is a growing concern

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

Nature and Degree of Threats: The causes of population decline from pre-dam numbers for late fall-run Chinook salmon are poorly documented, compared to the other three runs. Some of principal factors specifically affecting late fall-run Chinook salmon status and abundance, past and present, are: (1) dams, (2) loss of habitat, (3) fisheries, (4) outmigrant mortality, (5) water management, and (6) hatcheries (Table 2).

Dams. When Shasta and Keswick dams were built in the 1940s, they blocked late fall-run 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 year-round. At present, late fall-run Chinook salmon are largely dependent on cold-water releases from Shasta Reservoir. Large dams on the Sacramento River and its tributaries have not only blocked salmon access to historic spawning grounds, but they have reduced or eliminated recruitment of spawning gravels into the river beds below dams and altered temperature regimes. Loss of spawning gravels in the Sacramento River below Keswick Dam is regarded as a serious problem; large quantities of gravel are now trucked and placed in the river. Warm water temperatures are potentially a problem in this reach during drought years, when the cold-water pool in Shasta Reservoir is reduced. However, modification of Shasta Dam to provide cooler water in summer for winter-run Chinook has presumably also benefited late fall-run Chinook.

The effects of RBDD were more subtle. 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 apparently alleviated much of this problem. The gates are now open year-round, allowing uninhibited passage of adult and juvenile late fall-run Chinook salmon.

Agriculture. Outmigrant mortality of both fry and smolts is a factor affecting late fall-run Chinook abundance, as it is for all runs of salmon in the Sacramento-San Joaquin drainage. Small numbers of outmigrants are presumably entrained at larger irrigation diversions along the Sacramento River that are operating during the migration period. At the same time, extensive bank alteration to benefit agricultural operations has reduced the amount of cover available to protect outmigrants from striped bass, terns, herons and other predators. Given the extensive agricultural land use in the CV, it is likely that return waters negatively affect water quality, even in systems as large as the Sacramento River. Levees to protect agricultural fields from flooding have substantially degraded riparian habitats and eliminated connectivity of main stem river channels to historically widespread (and ecologically important) floodplain habitats.

Urbanization. Urbanization simplifies and pollutes Chinook salmon habitats. By diverting water and denying access to floodplain areas, the simplification process is similar to that discussed above for agriculture.

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

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

Estuary alteration. There is growing appreciation of the importance of "biocomplexity" for the persistence of salmon in a variable environment (Hilborn et al. 2003), including those in the CV (Carlson and Satterthwaite 2011). Biocomplexity is defined as multiple variations in life history that improve the ability of populations to persist in changing environmental conditions. Loss of diverse habitats in the San Francisco Estuary has essentially eliminated aspects of life history diversity and the best strategy for juvenile salmon, today, appears to be to move through the estuary as quickly as possible. Large pumping stations in the southern Sacramento-San Joaquin Rivers Delta (Delta) divert approximately 40% of the historic Delta flows, resulting in substantial modifications in flow direction (Nichols et al. 1986). Pumping also increases the likelihood of out-migrating smolts entering the interior delta, where longer migration routes, impaired water quality, increased predation, and entrainment result in higher mortality rates (Perry et al. 2010).

Despite long-term monitoring, causes of apparent high mortality rates as fish pass through the estuary are poorly understood. General observations suggest that rearing conditions in the estuary are often poor; highest survival occurs during wet years, when passage through the estuary is likely most rapid and water quality is higher (Brandes and McLain 2001, Baker and Mohrhardt 2001). Flooding in wet years also increases rearing habitat in the Delta and Yolo Bypass, which may also have a positive effect. Additionally, recent studies documented that the further downstream a group of late fall-run smolts is released, the longer the group takes to reach the ocean. These finding suggest that environmental cues that trigger migration in the upper watershed may be subdued or absent in the lower river (Michel et al. 2013).

The Delta ecosystem is as, if not more, altered than the estuary. Land and water management practices have altered the delta's landscape and ecological processes such that fall-run Chinook salmon and other native fishes encounter poor to extremely poor habitat conditions when migrating through the Delta's waters.

Harvest. The effects of harvest on CV salmon, in general, are discussed by Williams (2006). The actual harvest rates of late fall-run Chinook salmon are not known, but it is highly likely that they are harvested at similar rates as fall-run Chinook salmon. Although hatcheries are operated to sustain fisheries and hatchery fish can sustain higher harvest rates than wild fish, fisheries do not discriminate between them. Fisheries may, therefore, be taking a disproportionate number of natural-origin late fall-run Chinook salmon. Other effects are discussed in the 2015 fall-run Chinook salmon account.

Hatcheries. Late fall-run Chinook salmon have been reared at Coleman National Fish Hatchery on Battle Creek since the 1950s, even though the run was not formally recognized until 1973 (Williams 2006). The current production goal is one million smolts per year, which are released into Battle Creek from November through January (Williams 2006). Hatchery broodstock selection for late fall-run fish includes both fish returning to Coleman National Fish Hatchery and those trapped below Keswick Dam. Large numbers are needed because survival rates are low (0.78% at Coleman). Hatchery production may have impacts to the naturally-spawning population, although a low proportion of hatchery-origin fish have been found in the in-river spawning surveys (Kormos et al. 2010).

Alien species. Over the past 150 years, numerous fish species have been introduced to the Bay-Delta system. Several species of introduced fishes prey upon Chinook salmon, including striped bass, largemouth bass, smallmouth bass, and spotted bass. Striped bass can consume large numbers of juvenile salmon, particularly at diversion structures, or where hatcheries release large numbers of juvenile fish.

Effects of Climate Change. The effects of climate change on late fall-run Chinook salmon are similar to those of other runs of Chinook salmon. However, particularly critical for late fall-run Chinook salmon, is maintaining a cold water pool in Shasta Reservoir to keep water in the Sacramento River cold enough to support late fall-run habitat requirements year-round. Maintaining the cold water pool will be increasingly difficult during periods of extended drought and in the face of predicted increasing air and water temperatures. Thus, spring-fed Battle Creek may be crucial as a refuge during periods of drought. Moyle et al. (2013) found late fall-run Chinook salmon to be "critically vulnerable" to extinction from the effects of climate change because of the run's dependence on cold water released from dams.

Status Determination Score = 2.6 – High Concern (see Methods section Table 2). Late fall-run Chinook have been extirpated from a considerable portion of their historic spawning grounds. In the past 10 years, numbers of CV late fall-run Chinook salmon have fluctuated but appear to be comparable to numbers in the 1970s and 1980s. According to NMFS, they "continue to have low, but perhaps stable, numbers." (pdf http://www.fisheries.noaa.gov/pr/species/fish/chinook-salmon.html). Nevertheless, CV late fall-run Chinook may be vulnerable because of their relatively small population size and limited spawning distribution (Figure 1). Lack of access to (and degradation of) spawning and rearing habitats may make this population exceptionally vulnerable to changes in water quality and flow in the Sacramento River, as in the case of an extended drought, changes in water management, or a major spill of toxic materials. Their persistence depends on operation of water projects (Shasta Dam) and hatchery operations. The late fall-run Chinook salmon is considered a Species of Concern by the National Marine Fisheries Service (combined, single ESU with two races, late fall-run and fall-run Chinook salmon).

Metric	Score	Justification
Area occupied	2	Only one primary population concentrated in the upper Sacramento River; some tributary spawning and rearing
Estimated adult population	3	Total escapement has averaged approximately 10,000 spawners in recent years; hatchery contribution is low
Intervention dependence	3	Primary population is dependent on dam operation for flows and gravel injection for spawning habitat improvement
Tolerance	4	Moderate physiological tolerance, multiple age classes
Genetic risk	2	Hybridization with other runs may occur

Anthropogenic threats	2	See Table 2
Climate change	2	Snow pack or cold spring-fed flow dependent
Average	2.6	18/7
Certainty (1-4)	2	Least studied of CV Chinook runs

Table 3. Metrics for determining the status of Central Valley late fall-run Chinook salmon, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: Currently, less management is directed to benefit late fall-run Chinook salmon than for any other run in the Sacramento River, because little is known about the run and it is considered a race within the fall-run Chinook ESU. A key to conserving late fall-run Chinook is to develop and implement specific measures tailored to its unique life history.

This run should benefit considerably from measures being taken to enhance winter and spring-run Chinook salmon populations in the upper Sacramento River. However, specific studies should be undertaken to better understand the environmental requirements specific to the late fall-run, because this population needs protection at all stages of its life cycle. The Anadromous Fish Restoration Program has set a goal in their final restoration plan of an average production (escapement plus catch in fishery) of 44,000 fish per year, although the official doubling goal (required in the Central Valley Project Improvement Act) is 68,000 natural-origin fish (http://www.delta.dfg.ca.gov/afrp/). Whether or not existing habitat is adequate to sustain a population at either level is uncertain. Spawning and rearing ground monitoring specific to the run as well as additional genetic studies should be conducted for late fall-run Chinook salmon.

Restoration will require: (1) continuing to provide improved passage of adults to holding and spawning areas, (2) protecting adults in spawning areas, (3) establishing additional spawning populations (e.g., Battle Creek), (4) providing passage flows for out-migrating juveniles to move through the Delta as rapidly as possible, (5) maintaining and expanding rearing habitats for juvenile fish in the mainstem river and floodplains, and (6) ensuring ocean and inland fisheries regulations minimize impacts. Most of these require continuous, adaptive management as well as improved monitoring and population evaluation programs for both hatchery and naturally-produced fish (Williams 2006). Recent oversight by the California Hatchery Scientific Review Group (2010), improving hatchery practices, and the release of a comprehensive monitoring plan for Central Valley salmon are promising signs that efforts are being made to focus on better understanding and protecting salmon stocks, minimizing impacts of hatchery stocks to wild-spawned stocks across all runs of Chinook salmon in the CV and elsewhere, and strengthening regulatory protection of at-risk stocks.

Figure 2. Distribution of Central Valley late fall-run Chinook, *Oncorhynchus tshawytscha*, in the Sacramento River and tributaries of California.

KLAMATH MOUNTAINS PROVINCE STEELHEAD
Oncorhynchus mykiss irideus

Status: High Concern. Klamath Mountains Province (KMP) steelhead appear to be in long-term decline. Stream-maturing forms (mostly summer steelhead) are more limited in distribution and face a higher likelihood of near-term extinction than ocean-maturing forms (winter steelhead).

Description: Steelhead are anadromous coastal rainbow trout which return from the ocean as large, silvery fish with numerous black spots on their tail, adipose and dorsal fins. The spots on the tail are typically in radiating lines. Their dorsal coloration is iridescent blue to nearly brown or olive. Their sides and belly appear silver, white, or yellow, with an iridescent pink or 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 rays. 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 the lateral line (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 the interspaces wider than the parr marks themselves. 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 (non-anadromous) adult coastal rainbow 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, "*Salmo" mykiss* which had been recognized as a species before the North American form. As a result, rainbow trout, including steelhead, are officially recognized by the American Fisheries Society as *Oncorhynchus mykiss*. Two major genetic groups of *O. mykiss* have been identified as inland and coastal groups, separated by the crest of the Cascade Range (Busby et al. 1994). Coastal rainbow trout of North America, including coastal steelhead, have been identified in the subspecies *O. m. irideus* (Behnke 1992).

Historically, Evolutionarily Significant Unit (ESU) criteria were created by the National Marine Fisheries Service (NMFS) for management of endangered salmonids (56 FR 58612). In 2005, a joint policy with the U.S. Fish and Wildlife Service (USFWS) and NMFS designated Distinct Population Segment (DPS) criteria for steelhead (61 FR 4722). 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." While the boundaries for designation of a steelhead population as a DPS did not change much from an ESU designation, the DPS designation allowed for the listing of anadromous forms under state or federal endangered species acts, while not listing resident forms (although the two forms can interbreed). Six west coast steelhead DPSs occur within California. The NMFS and CDFW recognize distinct life history variations of steelhead in the KMP DPS, based upon their timing of freshwater entry, reproductive biology and spawning strategy (Busby et al. 1996). These KMP steelhead DPS variations have been defined as: winter, fall and summer, with a distinctive variant known as 'half-pounder,' that may be derived from any of the three DPSs. Genetic data do not support the hypothesis that winter, fall and summer steelhead populations are separate monophyletic units (Reisenbichler et al. 1992, Busby et al. 1994); thus, all life history variations within the KMP DPS are considered a single population source, although there is some degree of genetic differentiation among steelhead groups or clusters within the basin (Pearse et al. 2006, Pearse et al. 2011).

Genetic analyses from samples collected between the Klamath River estuary and the confluence of the Trinity River supports at least two discrete migrating populations, based on timing of freshwater entry (Papa et al. 2007). This correlates with the observed run-timing for the ocean-maturing (winter) and stream-maturing (summer, fall) ecotypes (Table 1). Pearse et al. (2007) analyzed genetic samples collected from 30 sites throughout the Klamath River watershed and three Trinity River sites. Results indicated that geographically proximate populations were most similar genetically. Steelhead sampled from the Klamath River below the Trinity River confluence (Turwar, Blue, Pecwan, Cappell, and Tully creeks) expressed limited gene flow with steelhead sampled upstream of the confluence. Steelhead sampled nearest the mouth of the Klamath River (Blue and Hunter creeks) had genetic similarity to populations in the Smith River and Wilson Creek, showing that migration of nearby coastal stream populations provides a source of additional variation. Populations sampled in the middle regions of the Klamath River basin clustered closely together. However, steelhead from the Shasta and Scott rivers were genetically distinct from steelhead sampled in other mid-Klamath basins and clustered closely to steelhead from Iron Gate Hatchery, suggesting that influence of hatchery gene flow (possibly from straying) to these nearby tributaries has occurred (Pearse et al. 2007). Samples collected from Trinity River Hatchery steelhead clustered most closely with the relatively homogeneous mid-Klamath steelhead, perhaps due to decades of egg transfers from the mid-Klamath basin to the hatchery (Busby et al. 1994). Steelhead from the only in-river collection site on the Trinity River (Horse Linto Creek) grouped with steelhead from the lower Klamath River, below the confluence.

Genetic studies of KMP summer steelhead indicate that they are more closely related to KMP winter steelhead than to summer steelhead outside the KMP (Reisenbichler et al. 1992). Recent genetic studies of summer and winter steelhead show a low level of differentiation between the two runs over multiple years, but also identified potentially greater levels of differentiation between spatially isolated reproductive populations (Papa et al. 2007, Pearse et al. 2007). Genetic studies on steelhead from the Eel River (Northern California Steelhead DPS) also found that winter and summer populations were more closely related to each other than they were to winter and summer populations from other rivers (Clemento 2006). Nevertheless, non-genetic factors (physical, physiological, ecological, and behavioral factors) indicate that the stream-maturing life history is distinct (presumably with a genetic basis) and that these fish are largely segregated from winter steelhead.

Life History: Two basic reproductive strategies have been identified for steelhead: ocean-maturing and stream-maturing. Ocean-maturing steelhead enter fresh water with well-developed gonads and spawn relatively soon thereafter, while stream-maturing steelhead enter freshwater with immature gonads and require several months to mature and then spawn (Burgner et al. 1992, Busby et al. 1996). Ocean-maturing steelhead typically begin spawning migration between November and April and are generally referred to as winter steelhead. Stream-maturing steelhead enter fresh water between May and October and are generally referred to as summer steelhead (Burgner et al. 1992). In the KMP, the term "fall steelhead" is used to distinguish a distinct run that enters fresh water between August and November, whereas summer steelhead enter between April and June. Both summer and fall steelhead are considered stream-maturing and fall steelhead are often lumped with, or described as, summer steelhead (Busby et al. 1996), yet their run timings are clearly discrete. Because of overlaps in spawning migration and timing, differentiating between winter, fall, and summer steelhead can be difficult (Table 1). The KMP steelhead life history variant referred to as 'half-pounder' is comprised of subadults that spend 2-4 months in the Klamath estuary or nearshore marine habitats, overwinter in the lower and middle Klamath River, and return to the ocean the following spring. Half-pounders are most common downstream of Seiad Valley (Kesner and Barnhardt 1972). Winter, fall and summer steelhead are all known to exhibit a half-pounder strategy. A total of 33 different steelhead life history categories at maturity were identified by Hodge (2010) in the Klamath Basin, including non-anadromous and anadromous forms.

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. Klamath Mountains Province steelhead run timing.

The following is a description of the three principal steelhead runs recognized in the KMP, as well as a description of half-pounder and early life history stages of all runs.

KMP winter steelhead: Klamath Mountain Province winter steelhead become reproductively mature in the ocean. Winter steelhead typically enter fresh water from potentially as early as September (though more typically November) to March, as mature adults, spawning shortly after migrating to suitable spawning areas (Busby et al. 1996). Spawning peaks before March. Population data are sparse for winter steelhead due to their run-timing, which is concurrent with higher winter flows and turbidity levels. As such, monitoring this run using traditional weirs or spawner surveys is not feasible.

KMP fall steelhead: Klamath Mountain Province fall steelhead enter the Klamath Basin between July and November (USFWS 1979, Hopelain 1998). Fall steelhead migrate into the Klamath and Trinity rivers between August and November and spawn in the mainstem and tributaries during the months of January through May. The fall steelhead run is more abundant than the summer steelhead run and, based upon mark-recapture data (Sinnen et al. 2009), is the main run of fish utilized for hatchery production at Trinity River Hatchery. However, based on trapping data from Willow Creek (Trinity) and the Klamath, a substantial number of non-marked (wild) fish enter the system at the same time as hatchery fall-run steelhead, supporting the separation of fall from summer steelhead in the KMP (W. Sinnen, CDFW, pers. comm. 2014). Nonetheless, fall steelhead are similar to summer steelhead in their level of sexual maturation and spawn timing and probably represents a continuum of the stream-maturing ecotype that encompasses both summer and fall steelhead, perhaps as a result of hatchery practices.

KMP summer steelhead: Summer steelhead in California typically enter rivers during spring months (April-June), while still sexually immature. They then mature in-river over the course of several months (Shapovalov and Taft 1954, Busby et al. 1996, Moyle 2002). Summer steelhead spawn in upstream reaches that are typically not utilized by fall or winter steelhead (Roelofs 1983), including smaller tributary/headwater streams. In the Rogue River, Oregon, spawning begins in late December and peaks in January (Roelofs 1983) and this early spawn timing is apparently also found throughout the KMP. However, in the Trinity River, while summer steelhead are found in tributaries by June, they only appear in the mainstem Trinity above Lewiston by August. In the Klamath River, summer steelhead presumably ascend into summer holding areas by June. Holding areas are typically deep, bedrock pools in remote stream reaches, with subsurface flow or great enough depth to allow for thermal stratification, keeping temperatures cool during low flow periods. Steelhead also utilize thermal refuge plumes emanating from tributary mouths as holding areas in the mainstem Klamath River. While many KMP summer steelhead die after spawning, about 40-64 percent are repeat spawners (Hopelain 1998). Based on their occupancy of headwater streams with relatively low (< 50 CFS) winter flows (Roelofs 1983), fry are assumed to move out of smaller natal streams into larger tributaries soon after emerging.

KMP half-pounder: Half-pounders are small, generally sexually immature, fish (25-35 cm FL), that return to the river in late summer and early fall (between late August and early October); the majority are subadults who have spent only 2-4 months in the Klamath estuary or near-shore environments before returning to the river to overwinter and forage in the lower and mid-Klamath river reaches (Kesner and Barnhart 1972). Recent information suggests that a small proportion (8%) of half-pounders may attain sexual maturity (Hodge 2010). Half-pounder run timing in the Rogue River is generally about a month earlier than in the Klamath (ODFW seine numbers peak in early to mid-August). They return to the ocean the following spring. The presence of half-pounders is uncommon above Seiad Valley in the Klamath River (Hopelain 1998), as are summer steelhead in tributaries above this location. While half-pounders do not typically mature or reproduce in fresh water, they are often encountered during snorkel surveys for adult summer steelhead (and spring Chinook) in the Salmon (Klamath), New (Trinity), and South Fork Trinity rivers (J. Israel, R.Quiñones, and J. Weaver, pers. obs.). However, the presence of over-summering half-pounders with adult summer steelhead is not typically discussed in the literature (Kesner and Barnhardt 1972, Hopelain 1998). Because it is difficult to distinguish between half-pounders and large stream-resident *O. mykiss*, half-pounders are typically not included during snorkel surveys for adult summer steelhead

(E. Wiseman, USFS, pers. comm. 2012). The presence of higher numbers of half-pounders appears to decrease the size of adults at first-spawn. Lower Klamath winter steelhead had the lowest occurrence of half-pounders and the greatest first-year growth rate (Hopelain 1998).

Early life stages: Fry emerge in the Trinity River beginning in April and migrate downstream from May through July (Moffett and Smith 1950). Fry initially move into shallow habitats along stream margins (Moyle 2002) but later establish territories, through aggressive behaviors, in or below riffles (Shapovalov and Taft 1954). In the Trinity River, fry were most common in tributary streams but moved downstream in the early summer, prior to their first winter (Moffett and Smith 1950). Further downstream movement occurred in late fall and winter during periods of higher flows and lower water temperatures. Parr moved the most near the end of their first year and spent their second year in the mainstem. In the Klamath River, relatively equal portions of young-of-year (34%), age-1+ (37%) and age-2+ (27%) steelhead were captured emigrating downstream by rotary screw traps near Orleans from 1997-2000 (USFWS 2001). Most (86%) steelhead returning to the Klamath River apparently spend two years in fresh water before migrating to the ocean (Hopelain 1998). However, steelhead rearing in fresh water for longer periods had shorter downstream migrations (Kesner and Barnhart 1972). Klamath mountain province steelhead live one to three years in the ocean before beginning upstream spawning migrations. Migration patterns in the ocean are unknown.

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 for steelhead are often found farther inland and in smaller streams than those utilized by Chinook and coho salmon (Moyle 2002). Adult steelhead require high flows, with depths of at least 18 cm 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 adults (see Table 2) (Moyle 2002). Steelhead require loose gravels at pool tail-outs for optimal conditions for redd construction and spawning success. 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 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 reduced dissolved oxygen concentrations available for incubating eggs (McEwan and Jackson 1996). When fine sediments (< 2.0 mm) compose > 26% of the total volume of substrate, poor embryo survival is observed (Barnhart 1986). Once out of the gravel, emerging fry can survive at a greater range of temperatures than embryos, but have difficulty obtaining oxygen from the water at temperatures above 21°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 bioenergetic costs for steelhead and can lead to reduced growth and increased mortality (Table 2). As temperatures become stressful, juvenile steelhead will move into faster riffles to feed, due to increased prey abundance, and seek out cool-water refuges associated with tributary confluences and gravel seeps. Optimal temperatures for growth are estimated to be around 10-17°C (Table 2). However, juvenile steelhead can live in streams that regularly exceed 24°C for a few hours each day if food is plentiful (Moyle 2002).

	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	<4ºC	5-11°C	12-17°C	>17°C	This is the most temperature-sensitive phase of life cycle.
Juvenile Rearing	<10ºC	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 2. Temperature requirements of steelhead, from Richter and Kolmes (2005), McEwan and Jackson (1996), and Moyle (2002). Values may vary according to acclimation history of individuals 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. Hawkins and Quinn (1996) found that the critical swimming velocity for juvenile steelhead was 7.7 body lengths/sec, compared to 5.6-6.7 body lengths/sec. for juvenile cutthroat trout. Adult steelhead swimming ability is hindered at water velocities above 3.0-3.9 m/sec (Reiser and Bjornn 1979, in Spence et al. 1996). Preferred holding velocities are much slower and range from 0.19 m/sec for juveniles to 0.28 m/sec for adults (Moyle and Baltz 1985). Physical structures such as boulders, large woody debris and undercut banks are important habitat components that create hydraulic heterogeneity, increase cover from predators, provide visual separation of juvenile territories, and afford refuges during high flows.

Because upstream migration often coincides with high flows, winter steelhead are able to move into smaller tributaries inaccessible to other salmonids during low flows. Inhabited streams may include those in medium-sized watersheds with confluences that are not passable in the summer/fall seasons due to high rates of sedimentation and associated subsurface flows. They can also migrate into the headwaters of low-order streams, where flows would otherwise be too low to be accessible by large fish.

Over-summering habitat for adult summer steelhead includes pools of moderate size (200-1,000 m^2), with minimum depths of 1.0 to 1.4 m. Although localized areas of cool water (0.2 to 3.8°C lower than the mean hourly pool temperature of 18.0°C) were observed in some pools, Nakamoto (1994) did not find a significant positive relationship between adult fish density and mean hourly pool temperature in the New River. Habitat use was more often associated with physical habitat characteristics such as pool size, substrate embeddedness (<35%), shade from riparian vegetation, and instream cover (Nakamoto 1994, Baigun 2003). Most (99%) of summer steelhead observed in the New River used cover during the day; bedrock ledges and boulders were used more frequently than depth (>1m) or shade (Nakamoto 1994).

Fall steelhead, which migrate during periods of decreasing stream temperatures, are less reliant on deep pools for holding and tend to hold in the mainstem for extended periods of time. It is assumed that these fish enter tributaries after the first series of rainfall-driven freshets in the fall. Fall steelhead trapped and tagged in the lower Trinity River from September through November often do not appear at Trinity River Hatchery until January through March.

Spatial segregation of spawning habitats between winter and summer steelhead reproductively isolates the two runs, facilitating low levels of genetic differentiation (Barnhart 1986, Papa et al. 2007). Summer steelhead often spawn in the upper portions of watersheds in isolated and/or intermittent streams; juveniles move into perennial streams soon after emergence (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, in small stream habitats, embryos are less susceptible to scouring by high flows and juveniles are less vulnerable to predation by adults, due to lower adult densities in smaller streams. Water velocities and depths measured at redds were 23-155 cm sec^{-1} and 10-150 cm, respectively, and diameters of the gravels were typically 0.64-13 cm. The concept of spawning spatial segregation is based largely on summer steelhead distribution and habitat utilization, since little is known about the spawning distribution of fall and winter steelhead throughout the KMP.

Distribution: Klamath Mountain Province steelhead are found in the Klamath River basin and streams north to the Elk River, Oregon, including the Smith (California) and Rogue (Oregon) rivers. In the Klamath River, the upstream limit of steelhead migration is Iron Gate Dam. Historic range likely included tributaries to Upper Klamath Lake, prior to dam construction (Hamilton et al. 2005). In the Trinity River, upstream migration is blocked by Lewiston Dam (Moffett and Smith 1950).

It is likely that the steelhead runs that migrated into the upper Klamath Basin before construction of Copco Dam were KMP summer steelhead. Recent genetic analysis concluded that anadromous steelhead (coastal group) were genetically distinct from redband trout (*O. mykiss newberri*; inland group), which currently persist in the upper basin (Pearse et al. 2011).

Trends in Abundance: Few data are available to evaluate trends in abundance. Adult spawners in the Smith River during the 1960s were estimated at 30,000; recent estimates for the 2010 and 2011 seasons were 16,000 and 15,000 respectively (Larson 2013). Rough estimates for annual size of all steelhead runs combined in the Klamath basin in the 1960s were between 283,000 (CDFG 1965) and 222,000 (Busby et al. 1994). Estimates declined to 87,000-181,000 from 1977 to 1983 (Hopelain 2001), with the winter steelhead run declining to 10,000-30,000 in the main stem Klamath River. Fall steelhead adult runs in the Trinity River were estimated between 7,833 and 37,276 during the 1980s. Returns to Iron Gate Hatchery are highly variable but appear to be in decline (Figure 1; Quiñones et al. 2013). Returns to Trinity River Hatchery, in contrast, appear to be increasing (Figure 1). See below for information related to trends in winter, fall and summer KMP steelhead.

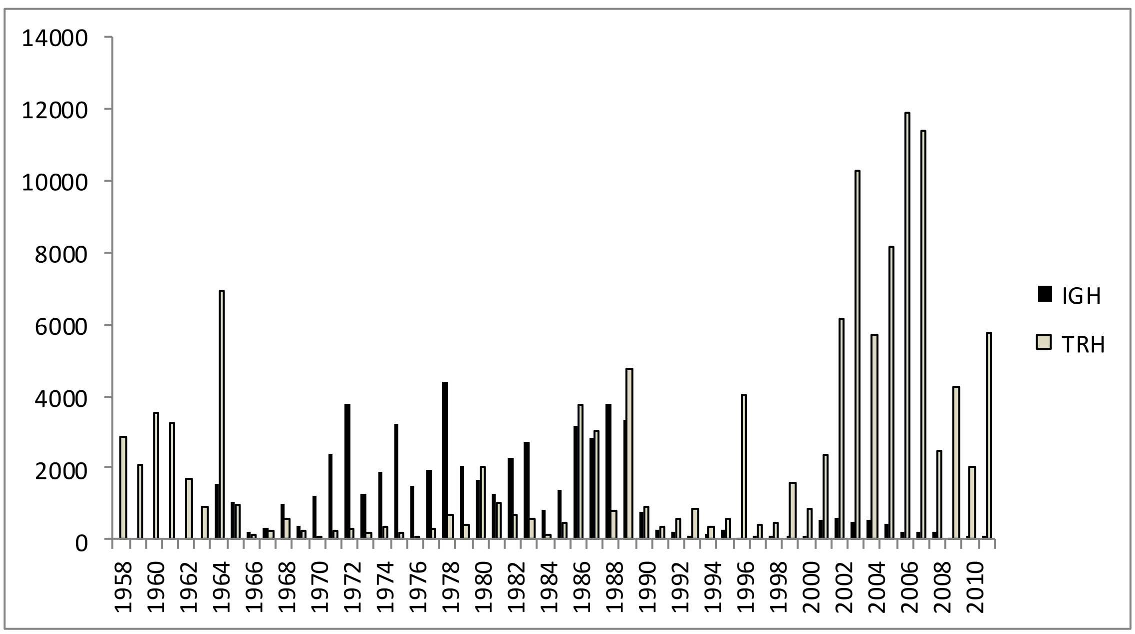


Figure 1. Fall steelhead returns to Iron Gate and Trinity River hatcheries, Klamath River basin, 1958-2011 (Iron Gate and Trinity River hatcheries, unpublished data).

KMP winter steelhead: Data are particularly sparse for KMP winter steelhead, due to the difficulties (high flows, turbidity, surveyor safety concerns) associated with monitoring during the winter months. Recently, DIDSON sonar counts were used to estimate the abundance of winter steelhead in the Smith River (Larson 2013). Estimated winter steelhead abundance for the 2010-11 and 2011-12 seasons was 16,000 and 15,000, respectively. Winter steelhead are the predominate run in the Smith River. There are no long-term (or even recent) estimates for the Klamath Basin for winter steelhead abundance.

KMP fall steelhead: Fall steelhead in the KMP are largely a stream-maturing run and have been classified as summer steelhead by NMFS (Busby et al. 1994, Busby et al. 1996), adding to the confusion related to identifying and managing discrete KMP steelhead runs; however, data from the Trinity River (Sinnen et al. 2009) suggest the fall steelhead run peaks in the lower mainstem Trinity River between September and November. This spawning peak occurs considerably later than summer steelhead counts in tributaries (which occur in August), suggesting that there is a discrete fall steelhead run. Adult steelhead numbers for the fall migration period have been generated for a number of years on the Trinity River and indicate stable, albeit heavily hatchery-supported, runs (Table 3). The run has averaged 15,182 fish/year for the years for which data are available. The wild component has ranged from 1,349 fish to 16,645 fish, averaging 5,579 fish. The hatchery component of this run has ranged between 1,315 and 46,379 fish, averaging 12,350 fish. Populations of fall steelhead, based on run-timing, also exist in upper Klamath tributaries. CDFW counts at two video weirs on the Shasta (CDFW 2013) and Scott rivers (CDFG 2012a) for the 2011-12 season indicate bimodal peaks of migration that occur in mid-October and late December/early January. Steelhead were observed at both weirs throughout September through early January, after which the weirs were removed due to high flows. A total of 251 adult steelhead were observed in the Scott River (CDFG 2012a) and 180 in the Shasta (CDFW 2013) during the operational time frame.

KMP summer steelhead: Little is known about the historical abundance of summer steelhead in the KMP; quantitative records of summer steelhead numbers exist only for recent decades (Roelofs 1983). Given the limited amount of habitat now available since large portions of the upper Klamath and Trinity basins were blocked by dams, it is likely that summer steelhead in the Klamath Basin currently represent only a small fraction of their original numbers. Some summer steelhead populations (e.g., Salmon River) have declined precipitously in the past 30-40 years (Quiñones et al. 2013), while others have shown increases in recent years (e.g., New and North Fork Trinity rivers; E. Wiseman, USFS, pers. comm. 2013). Snorkeling counts for summer steelhead are prone to difficulties such as counting half-pounders as adult steelhead, incomplete spatial surveys, observational biases by surveyors, and low water clarity from rainfall events, and sediment inputs, especially from suction dredging (at least in the past, given the current moratorium on dredging in California). Therefore, survey numbers likely represent the minimum fish present. The majority of estimates for California populations have been less than 100 fish at each location for the past decade, with a few exceptions. In 1989-1991, the three-year average exceeded 500 fish in the North Fork Trinity River and New River, which also had more than 500 fish in 1999-2001 and 2002-2004. These two tributaries averaged more than 800 fish in 2009-2012. Three year averages also exceeded 500 fish for some years in Dillon Creek (2000-2004) and Clear Creek (2001-2003) (T. Jackson, CDFW, pers. comm. 2011). Out of 20 summer steelhead populations surveyed in the Klamath-Trinity basins, eleven averaged <100 fish annually and nine averaged < 20 fish each for the years they were surveyed (Table 4). Average counts for the combined 20 populations for the years 1981 through 1985 were 1,919 fish. The more recent period, 1996 through 2012, averaged 2,923 annually. It appears the larger tributary populations within the KMP, excluding the Salmon River, have stabilized or increased and smaller tributaries continue to support considerably smaller summer steelhead populations. Because effective (breeding) population sizes are likely less than actual counts, many populations may be close to or below the minimum size needed for long-term persistence (Lindley et al. 2007). These abundance estimates are generated from adult fish observed in midsummer (July and August), so mortality prior to the winter spawning period is not accounted for. Most populations were severely affected by the extraordinary floods of 1964, which dramatically altered most KMP stream and river habitats. Although habitats are gradually recovering over time, the abundance of summer steelhead has fluctuated widely in recent years. The status of each major population or subpopulation of KMP summer steelhead is as follows:

Mainstem Trinity River. Moffett and Smith (1950) indicated that summer steelhead were common in the upper mainstem Trinity River in the 1940s. Utilization of this portion of the river persisted through the early 1960s (CDFG 1992), with individuals still present at Junction City (W. Sinnen, CDFW, pers. comm. 2011). Suitable water temperatures downstream of Lewiston Dam provide habitat for summer steelhead; however their current abundance in this section is unknown. It is likely that a large proportion of fish observed in the upper mainstem Trinity River originate from the Trinity River Hatchery.

North Fork Trinity River. There is little historical information on summer steelhead utilization and abundance in this stream, but relatively recent data (1979-2005) indicate that the population fluctuates between 200 and more than 1,200 fish per year (T. Jackson, CDFW, pers. comm. 2011). Summer steelhead distribution has changed relatively little during recent decades of monitoring and the majority of holding habitat occurs in the middle reaches. Their distribution in upper portions of the watershed appears to depend on sufficient flows, while high temperatures may limit their use of 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 more abundant in the past (Roelofs 1983). Canyon Creek, a tributary close to the North Fork Trinity River, continues to support very small numbers of summer steelhead and the average count for 24 of 30 years was 19 fish (Table 4).

South Fork Trinity River. There is no historical information on summer steelhead in this stream. Recent counts were as low as 11 fish in 1996; however, in 2006 and 2007, more than 200 fish were observed and, in 2011 and 2012, more than 300 fish were observed. Surveys performed in 2002 indicated that summer steelhead adults were less common than half-pounders, although with a similar distribution (Garrison 2002). All South Fork Trinity River counts of adult summer steelhead are combined with half-pounder steelhead and, in some years, the number of half pounders is substantial.

New River. This tributary to the Trinity River supports the second largest population of summer steelhead in California (T. Jackson, CDFW, pers. comm. 2011). The estimated average abundance for 1979-2006 was 647 summer steelhead. The estimated abundance reached a high of 2,108 fish in 2003, averaged 977 between 2004-2006, and, most recently, averaged 903 between 2007 and 2012.

Klamath River tributaries. Since 1985, summer steelhead counts were generally less than 100 fish in six tributaries: Bluff, Red Cap, Camp, Indian, Thompson, and Grider creeks (J. Grunbaum, pers. comm. 2010). The summer steelhead populations in Elk Creek averaged about 110 fish during this same period. Dillon and Clear creeks have the largest summer steelhead populations on the Klamath River, averaging more than 300 fish annually during the years they were surveyed. While there is no clear trend among the smaller populations, summer steelhead populations in Dillon and Clear creeks were estimated to be over 1,000 fish in 2002 (Table 4). These estimates have decreased over the past few years and the 2005-2009 average was 207 and 139, respectively.

Salmon River. Adult summer steelhead counts in the Salmon River (North Fork, South Fork, mainstem) were usually less than 150 fish per year each (Klamath National Forest, unpublished data 1990-2011). These watersheds were heavily mined during the late 19th century and smaller scale mining continues in the river during summer. Adult escapement decreased significantly from 1968 to 2009 ($p = 0.00074$; Quiñones et al. 2013). Within the general decreasing trend, adult escapement increased in 1973 and decreased in both 1980 and 1990. Favorable ocean conditions may explain increases in steelhead abundances during the early 1970s, years during cold PDO phases (Mantua and Hare 2002). Likewise, decreases in numbers may reflect unfavorable ocean conditions (warm PDO phase) in the mid- to late-1980s and early 1990s. However, correlation trend data between IGH hatchery steelhead and Salmon River summer steelhead suggest that hatchery stocks are influencing adult escapement trends (Quiñones et al. 2013). Further investigation is needed to explore adult escapement and population trends between hatchery and wild steelhead.

Wooley Creek. As with the Salmon River, to which Wooley Creek is tributary, this stream has maintained a summer steelhead population that is estimated to be between 100-300 fish per year. However, this population declined to an average of 50 individuals annually between 1990 and 2000. Counts increased to 288 fish in both 2003 and 2004, although counts in recent years are similar to those in the 1990s.

Smith River. Only 10-20 fish are estimated to occur annually in each of five tributaries since surveys began in 1978 (T. Jackson, CDFW, pers. comm. 2011); however, the Smith River watershed may never have supported summer steelhead in large numbers (Roelofs 1983), so these small numbers may not reflect actual declines.

Overall, KMP summer steelhead numbers in recent decades appear to have ranged between 1,400 and 4,000 fish in the entire KMP system per year. These estimates almost certainly represent only a small fraction of historic numbers, based on the fact that large areas of formerly accessible habitats are now blocked above dams, that summer steelhead generally utilize these same types of habitats (e.g., smaller tributary headwater streams), and human land and water uses have altered many remaining accessible habitats. Increases in numbers have been documented in some tributaries in recent years, presumably due to a combination of good ocean conditions, recovering stream habitats and restrictive sport fishing regulations.

Table 3. Fall-run adult steelhead (>41cm FL) run-size, spawner escapement, and angler harvest estimates for the Trinity River upstream of Willow Creek weir, 1977 - 2012.

	Run-size estimate					Spawner escapement						Angler harvest		
						Natural Area Spawners[a]			Trinity River Hatchery					
	Hatchery[b]		Wild[c]			Hatchery	Wild	Total	Hatchery	Wild	Total	Hatchery	Wild	Total
Year	Number	Percent	Number	Percent	Total									
1977			No estimates				No estimates		269	16	285		No estimates	
1978			"				"		628	55	683		"	
1979			"				"		329	53	382		"	
1980	8,449	33.7	16,645	66.3	25,094	5,101	14,462	19,563	1,903	102	2,005	1,445	2,081	3,526
1981			No estimates				No estimates		892	112	1,004		No estimates	
1982	2,106	20.0	8,426	80.0	10,532	971	6,889	7,860	634	79	713	501	1,458	1,959
1983	No estimates for hatchery/wild component				8,605			6,661			599			1,345
1984		"			7,833			6,430			142			1,261
1985		No estimates				No estimates					461	No estimates		
1986		"					"				3,780		"	
1987		"					"				3,007		"	
1988	No estimates for hatchery/wild component				12,743			11,926 [d]			817		"	
1989		"			37,276			28,933			4,765			3,578
1990		"			5,348			3,188			930			1,230
1991		"			11,417			8,631			446			2,340
1992	1,315	43.2	1,731	56.8	3,046	759	1,540	2,299	430	25	455	126	166	292
1993	1,894	58.4	1,349	41.6	3,243	801	1,176	1,977	875	10	885	218	163	381
1994	1,477	34.8	2,767	65.2	4,244	878	2,410	3,288	403	8	411	196	349	545
1995	1,595	37.2	2,693	62.8	4,288	1,424	1,867	3,291	24	681	705	147	145	292
1996	8,598	82.4	1,837	17.6	10,435	4,127	1,703	5,830	3,964	48	4,012	507	86	593
1997	No estimates for hatchery/wild component				5,212	No estimates		4,267	No estimates		429	No estimates		516
1998		"			2,972	"		2,463	"		441	"		68 [e]
1999		"			5,470	"		3,817	"		1,571	"		82 [e]
2000		"			8,042	"		7,097	"		768	"		177 [e]
2001		"			12,638	"		9,938	"		2,333	"		367 [e]
2002	14,408	75.6	4,650	24.4	19,058	7,730	4,566	12,296	5,966	42	6,008	697	57	754 [e]
2003	19,245	83.0	3,947	17.0	23,192	8,717	3,837	12,554	10,182	42	10,224	346	68	414 [e]
2004	15,038	75.7	4,817	24.3	19,855	8,937	4,732	13,669	5,688	37	5,725	413	48	461 [e]
2005	14,049	72.4	5,363	27.6	19,412	5,782	5,280	11,062	8,080	63	8,143	187	20	207 [e]
2006	32,609	78.8	8,781	21.2	41,390	20,272	8,660	28,932	11,509	38	11,547	828	83	911 [e]
2007	46,379	86	7,506	14	53,885	31,923	7,405	39,328	11,366	31	11,397	3,090	70	3,160 [e]
2008	9,538	64	5,477	36	15,015	6,680	5,415	12,095	2,471	24	2,495	386	38	424 [e]
2009	13,314	73	5,047	27	18,361	7,704	4,877	12,581	4,234	17	4,251	1,376	154	1,530 [e]
2010	4,640	55	3,811	45	8,451	2,468	3,749	6,217	2,000	37	2,037	172	25	197 [e]
2011	15,243	68	7,059	32	22,302	8,690	6,977	15,667	5,700	50	5,750	853	33	886 [e]
2012[f]	12,405	59	8,507	41	20,912	6,281	8,385	14,666	5,685	52	5,737	439	70	509 [e]

a/ Natural area spawners includes both wild and hatchery fish that spawn in areas outside Trinity River Hatchery.
b/ Trinity River Hatchery-produced steelhead.
c/ Naturally produced steelhead.
d/ The natural spawner escapement reflects an overestimate due to the unknown number of fish harvested by anglers upstream of Willow Creek Weir.
e/ Harvest was limited to hatchery-produced fish only. Hatchery fish are those with an adipose fin-clip.
f/ Preliminary data only.

Table 4. Observed number of summer steelhead in Klamath Mountain Province stream and rivers. Numbers should be regarded as indicators of relative abundance, rather than population estimates, since survey efforts may have differed by year and location. Some survey results include half pounders.

Watershed	Source	1966	1967	1968	1969	1970	1971	1972	1973	1974	1975	1976	1977	1978	1979	1980
Bluff	A,B	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	41	37
Red Cap	A,B	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns
Camp	A,B	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns
Dillon	A,B	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	236
Clear	A,B	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	1810	79	241
Elk	A,B	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	408	ns	90
Indian	A,B	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	421	ns	ns
Thompson	A,B	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns
Salmon mainstem	A,B,C	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	65
Wooley	A,B,C	ns	ns	33	ns	20	ns	45	ns	ns	124	ns	510	105	160	165
NF Salmon	A,C	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	69
SF Salmon	A,C	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	166
Grider	B	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns
Canyon	A,B	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	6
NF Trinity	A,B	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	200	320	456
SF Trinity	A,B,D	ns	ns	ns	ns	2	ns	ns	ns	1	ns	ns	ns	ns	91	ns
New	A,B	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	341	320
NF Smith	A,D	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns
MF Smith	A,D	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns
SF Smith	A,D	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns

Watershed	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995
Bluff	16	87	23	48	23	73	73	91	58	91	212	149	31	15	20
Red Cap	ns	45	12	11	18	ns	29	25	25	7	2	31	8	4	3
Camp	ns	ns	ns	ns	ns	ns	ns	ns	18	ns	1	7	ns	2	2
Dillon	187	295	300	200	162	ns	77	294	38	74	88	ns	161	ns	122
Clear	270	18	257	156	162	428	524	693	934	117	39	100	178	134	175
Elk	47	249	ns	18	ns	ns	ns	69	150	57	44	72	61	110	61
Indian	ns	15	ns	ns	ns	ns	ns	46	154	21	8	271	67	117	39
Thompson	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	4
Salmon mainstem	ns	100	ns	ns	ns	ns	ns	ns	13	15	24	24	16	11	25
Wooley	245	353	78	92	290	ns	280	357	234	73	25	17	49	22	34
NF Salmon	5	41	ns	ns	8	8	4	8	17	12	17	15	20	10	11
SF Salmon	16	225	ns	ns	9	9	14	154	ns	21	26	59	26	22	21
Grider	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns
Canyon	3	20	3	20	10	ns	0	32	ns	15	3	6	24	45	23
NF Trinity	219	193	160	180	57	ns	300	624	347	554	837	367	605	990	830
SF Trinity	ns	27	ns	8	3	73	ns	26	37	66	8	21	23	22	42
New	236	114	ns	335	ns	ns	ns	500	699	381	748	358	368	427	817
NF Smith	ns	2	ns	ns	ns	ns	ns	12	4	8	0	13	0	0	4
MF Smith	ns	2	ns	ns	ns	ns	ns	21	1	18	11	13	5	2	11
SF Smith	ns	2	ns	ns	ns	ns	ns	12	4	8	13	8	4	5	4

Watershed	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012
Bluff	15	2	15	5	9	9	35	31	20	10	7	18	11	23	10	11	3
Red Cap	6	1	6	3	0	2	9	23	20	10	6	4	0	2	2	2	4
Camp	1	0	4	0	0	2	4	5	3	13	0	15	0	7	2	1	1
Dillon	91	180	151	209	679	929	1108	576	437	216	448	58	ns	107	119	166	119
Clear	102	85	68	65	186	538	1034	238	268	108	158	129	222	78	97	141	132
Elk	96	33	490	23	77	212	200	55	112	34	37	33	68	56	38	87	37
Indian	ns	42	ns	ns	ns	ns	ns	4	ns	ns	30	87	71	442	51	70	29
Thompson	14	13	ns	ns	ns	ns	ns	46	17	9	13	21	9	36	27	0	3
Salmon mainstem	27	13	23	35	17	81	35	46	56	7	1	19	37	47	60	31	31
Wooley	14	18	14	13	32	74	143	240	75	39	ns	53	ns	26	37	24	58
NF Salmon	9	9	22	13	14	24	19	7	18	6	6	10	25	19	19	121	121
SF Salmon	35	8	17	20	14	21	39	11	34	24	35	29	68	45	37	24	24
Grider	ns	ns	0	ns	ns	ns	29	0	44	3	8	16	2	1	7	0	ns
Canyon	5	26	42	16	27	33	13	40	24	7							
NF Trinity	361	328	149	187	380	977	985	1042	453	443	420	399	ns	827	820	1082	1219
SF Trinity	11	95	57	38	221	131	77	144	114	95	214	409	ns	94	322	322	324
New	307	651	495	538	515	995	1500	2108	1156	843	932	898	222	1088	894	1084	1230
NF Smith	4	ns	ns	ns	ns	ns	ns	ns	ns	ns	0	ns	ns	ns	ns		
MF Smith	11	6	6	0	6	0	ns	1	6	2	14	0	5	10	0		
SF Smith	9	0	ns	0	13	1	ns	2	8	ns	11	9	1	ns	3		

Sources: A=McEwan and Jackson 1995; B=USFS, LeRoy Cyr and Jon Grunbaum, Six Rivers and Klamath National Forest; Eric Wiseman, Shasta/Trinity National Forest; C=USFS, Rebecca Quiñones, Klamath National Forest; D=Friends of the Smith River

Effects of Climate Change: Streams in the Klamath Basin downstream of Iron Gate Dam are projected to be warmer and drier during the summer and fall months, due to reduction in total snowpack and seasonal retention of snow (Hamlet et al. 2005, Stewart et al. 2005). Snow pack water content in the last 50 years has already significantly declined at several monitoring stations in the Klamath Basin (Van Kirk and Naman 2008). Lower flows further exacerbate increasing water temperatures, as river depth is often inversely related to water temperature (Allan and Castillo 2007). Climate change may also alter stream flow patterns by increasing winter runoff as rain rather than snow, likely decreasing spring and summer stream flows, and increasing the occurrence of winter floods and summer droughts (Knox and Scheuring 1991, Field et al. 1999). With increased temperatures causing earlier snowmelt, the timing of peak flows has already changed by 10 to 30 days (Stewart et al. 2005), with peak flows occurring earlier in more recent decades (Cayan et al. 2001). Flows in snowmelt-fed rivers (e.g., Salmon River) in the Klamath Basin usually peak in winter with a second, smaller, peak in spring and then gradually decrease to lowest levels in summer. If changes in flow regimes continue at the current rate, then stream flows in the Klamath River Basin are expected to decrease by 10%-50% in the spring and summer, while the frequency of extreme high and low flows are predicted to increase by 15%-20% (Leung et al. 2004, Kim 2005).

Increases in water temperatures will strongly affect the physiology and behavior of salmonids throughout their life histories. Changes in movement patterns are likely to be the most obvious response of individual salmonids to climate change, particularly as fish are exposed to increases in water temperature and changes in stream flow patterns. Most behavioral responses in salmonids are triggered by temperature thresholds and changes in flow (Groot and Margolis 1991). Because temperature increases will hasten developmental rates, and stream flows are predicted to peak earlier in the year, the migration patterns of Klamath salmonids may, correspondingly, shift to earlier in the year. However, photoperiod (day length) at a given site can also influence the initiation of salmonid migrations; thus, migration initiation and timing may become unsynchronized with temperature (Feder et al. 2010).

Another behavioral response of salmonids to increased temperatures is movement into colder waters as a method of thermoregulation. Salmonids use cold water pockets (thermal refuges) in rivers during juvenile rearing and adult migration when water temperatures exceed 22°C (Nielsen et al. 1994, Ebersole et al. 2003, Strange 2010). In summer, use of thermal refuges may make juveniles less susceptible to disease (Foott et al. 1999). Climate change influences could diminish or eliminate cold water pockets as temperatures increase. The reduction of suitable freshwater habitat is expected to result in a northward and/or higher elevational shift in the range of cold water fishes (Mohseni et al. 2003, Battin et al. 2007). As a result, steelhead in the KMP may experience local extinctions and range contractions, particularly since most higher elevation, headwater streams are inaccessible behind large dams.

Altered flow regimes, due to changes in precipitation patterns, may impair salmonid embryo development and juvenile survival. Extreme high flows can scour redds, flush juveniles into suboptimal habitats before they reach critical size, and desynchronize juvenile outmigration timing with the spring oceanic phytoplankton bloom (Mote et al. 2003). Fine (< 4 mm) sediment introduced by intense storm events and associated runoff can smother redds, preventing oxygen from reaching developing embryos or acting as a physical barrier to fry emergence (Furniss et al. 1991). Decreases in summer and fall flows may increase juvenile mortality through stranding and changes in the timing of peak spring and fall base flows may reduce survival of juveniles migrating from rivers into the ocean (Lawson et al. 2004). Increases in winter flows may decrease adult survival or reproductive success, due to the higher metabolic cost of upstream migration at higher flow stages.

The predicted impacts of climate change may particularly affect KMP summer steelhead adults because of decreased summer and fall base flows, increased summer water temperatures, and increased variability of seasonal flow patterns in the upper watersheds summer steelhead occupy. The cumulative impact of these changes is a likely reduction in suitable habitat available for spawning and over-summering (Moyle et al. 2013).

Nature and Degree of Threats: KMP steelhead stocks are the most abundant in California; however, as with all west coast steelhead DPSs, their abundance appears reduced from historic levels, to varying degrees, depending on the run and/or geographical area. Major factors likely contributing to the decline of KMP steelhead include: 1) dams, 2) diversions, 3) logging, and 4) agriculture.

Dams. Like many rivers in California, the Klamath and Trinity rivers have been dammed. Three dams that directly affect KMP steelhead in the Klamath basin are Iron Gate, Dwinnell, and Lewiston dams. All are part of larger projects and these three dams, alone, have blocked access to large portions of formerly utilized KMP steelhead habitats, especially important spawning and rearing grounds in the middle and upper portions of both systems. However, removal of Iron Gate and other upstream dams under the Klamath Basin Hydroelectric Agreement, and concordant Klamath Basin Restoration Agreement, may open up hundreds of kilometers of potential steelhead habitat in the future.

Iron Gate Dam is the downstream-most dam on the Klamath River and is one of six dams that make up USBR's Klamath Project, which has altered the main stem by regulating flows, increasing water diversions (Lewis et al. 2004) and degrading water quality (Hamilton et al. 2011). Iron Gate Dam has no fishway and, therefore, completely blocks access to historic upstream spawning and rearing habitats. Dam operations have decreased the variability, magnitude, duration, and timing of flows in the Klamath River. As a result, base flows have decreased and peak flows have been dampened. Peak flow timing has also shifted to at least a month earlier than prior to dam construction (Hamilton et al. 2011). Lower flows are of particular concern in the summer because daytime water temperatures can reach 24-26°C across large portions of the Klamath system, reducing available rearing habitat. Juvenile steelhead likely persist in the main stem because of abundant food resources and the presence of thermal refuges. Nevertheless, warm temperatures can be stressful if they alter movement, feeding, or growth patterns.

Dwinnell Dam has blocked access to > 30 km of habitat in the upper Shasta River, a tributary to the Klamath River, since its construction in 1928. The dam, in combination with multiple diversions, has reduced flows in the lower Shasta River. Dwinnell Dam has also altered the natural hydrograph, eliminating peak flows that could improve habitat conditions for steelhead and other salmonids (Lewis et al. 2004).

Minimum daytime water temperatures in summer below the dam are usually higher than 20°C, peaking above 22-24°C, which can create conditions stressful to steelhead. As a result, the quality and quantity of steelhead habitat in the lower Shasta River has been greatly reduced.

Lewiston Dam has blocked access to >170 km of habitat on the Trinity River since 1963. Along with Trinity Dam, located just upstream, the dam has greatly reduced flows and altered the natural hydrograph of the main stem Trinity River. The quality and quantity of steelhead habitat has been substantially reduced as a result. In an effort to restore main stem habitat, the Trinity River Restoration Program (initiated in 2000 as part of the Trinity River Record of Decision) was implemented with the goal of restoring up to 48% of flows into the Trinity River. Since its implementation, restoration has included augmentation of summer flows, habitat improvements, reconnection between the stream channel and floodplain, and spawning gravel supplementation.

Agriculture. Agriculture, especially for alfalfa irrigation, has affected many KMP streams by altering flows and degrading water quality. Flows in many streams within the KMP steelhead range have been decreased by agricultural diversions and pumping from wells adjacent to streams. In some streams, this may be the biggest factor affecting steelhead abundance. Diversions in the Scott and Shasta rivers, in particular, have major impacts on fishes by decreasing flows and returning "excess" water to rivers (Lewis et al. 2004), thereby reducing the amount of suitable habitat. Return water is typically much warmer than that in the river, after passing through ditches and fields, and is also often polluted with pesticides, herbicides, fertilizers, or animal wastes. Although many diversions in the Scott and Shasta valleys are screened to prevent juvenile salmonid entrainment, the effectiveness of such screening has not been adequately evaluated. Better agricultural practices and appropriate mitigation measures could dramatically improve salmonid production in the Shasta and Scott valleys (Lewis et al. 2004). Large-scale marijuana cultivation on public lands in the KMP (one of the more heavily used areas of the state for illegal cultivation) may be negatively impacting riparian and aquatic habitats through water diversion, increased sediment inputs, fertilizer and herbicide or pesticide inputs and solid waste inputs (trash dumps or abandoned growing supplies), although this issue requires further investigation and is confounded by safety risks and law enforcement involvement, limiting the opportunities to document impacts from this widespread activity.

Grazing. Livestock grazing is common throughout KMP watersheds and, in certain areas, contributes to degradation of aquatic and riparian habitats. Stream bank trampling and removal of riparian vegetation by livestock can cause bank sloughing, stream channel lie-back and head-cutting in meadows, leading to increased sediment loads and higher water temperatures in streams (Spence et al. 1996). Impacts may also include reduction in canopy cover (shading) over stream channels, siltation of pools necessary for juvenile rearing (Moyle 2002), or sedimentation of spawning gravels. In areas grazed by large herds or where grazing occurs for extended periods without allotment rotation or exclusion fencing, fecal matter from livestock can also impair water quality and increase nutrient loading, leading to eutrophication.

Instream mining. Gold dredging has occurred in KMP streams since the mid-19th century. Suction dredging can be an important limiting factor because dredgers often concentrate in preferred steelhead habitats in remote areas, where disturbance of habitats and disruption of fish habitat utilization may be particularly acute. A moratorium on suction dredging was implemented by CDFW in 2011; however, in 2012, a state law was enacted requiring CDFW to develop alternatives to a complete moratorium (new regulations and a proposed fee structure for dredging permits) by 2013. Depending on the outcome of this process, some level of suction dredging may continue to occur in KMP streams, although this activity will likely be more heavily regulated.

Mining. Legacy effects of 19th century hydraulic mining still negatively affect KMP steelhead habitats in many areas. Historic mining was widespread and intensive in this region and, in combination with logging (often to support mining), devastated many watersheds. Legacy effects from historic mining may be difficult to distinguish from contemporary impacts from logging, rural development, and other land uses that require road building, vegetation removal, or other landscape alterations that contribute to destabilization of the steep slopes of the Klamath Province and increased sediment loads in rivers and streams. Evidence of direct impacts from mining, historic and current, is apparent in many watersheds in the region (e.g., extensive tailing piles, active mining claims and associated equipment or refuse piles, cable crossings, etc.), indicating that mining may still affect KMP steelhead habitats, although historic impacts were almost certainly greater than they are today.

Transportation. Most KMP steelhead streams are paralleled or crossed by roads, often in many locations. Unsurfaced and unimproved roads (mining, logging, rural residential access) are abundant in the Klamath and Trinity basins and culverts associated with road crossings block access to habitat in many streams, while runoff of fine sediments and pollutants associated with roads can degrade water and habitat quality.

Logging. Contemporary logging, along with associated roads and widespread legacy effects from extensive historic timber harvest, has increased erosion rates of steep hillsides that are prone to landslides and mass wasting in this region, greatly increasing sediment loads in KMP streams (Lewis et al. 2004). Both private and public forest lands in the Klamath Basin have been heavily logged in the past century. In the Smith River basin and other protected coastal streams in the KMP, current logging practices are well managed but legacy effects from past, unregulated, timber harvest may continue to reduce steelhead production in some areas. Adverse impacts are especially acute in tributaries used by steelhead for spawning and rearing (Borok and Jong 1997, Jong 1997, Ricker 1997). Increased sedimentation in spawning areas results in lower egg survival and fry emergence rates in the Shasta and South Fork Trinity rivers. High sediment loads fill deep pools with gravel, embed spawning gravels in fine materials, and create shallower runs and riffles, negatively affecting all life stages of steelhead. Juvenile production can decrease significantly due to pool infilling, loss of cover, and increased water temperatures (Burns 1972). The potential for further mass wasting in the Trinity and Klamath basins is high, due to ongoing timber harvest operations. Deteriorating legacy road crossings are prone to failure in large storm events and recent forest fires may be further contributing to soil instability.

Fire. Most lower KMP tributaries, as well as the lower main stems of larger rivers, are within the marine fog belt, with cooler temperatures and higher fuel moisture levels that inhibit wildfires; however, inland portions of KMP watersheds are subject to frequent fires (e.g., Forks, Salmon, and Corral complex fires, 2013) that, under predicted climate change scenarios, are likely to increase in frequency and intensity. Fires can increase water temperatures of important holding and rearing headwater streams, cause landslides, increase sediment loading, and remove shading canopy cover, all to the detriment of steelhead.

Recreation. Recreational activities in KMP steelhead streams include: angling, boating, gold panning (and other forms of mining), swimming, hiking, and other outdoor activities. The impacts from recreation upon steelhead, especially at the population level, are likely minimal. Intensive motorized boating (e.g., lower Klamath River) may disrupt movement patterns and, potentially, habitat utilization, but this has not been substantiated.

Harvest. Current fishing regulations prohibit the take of wild steelhead and only hatchery (adipose fin-clipped) steelhead are legal to harvest. The influence of recreational angling on steelhead abundance is not known, but is assumed to be minimal. Tribal net fisheries generally do not target steelhead; however, nets are an indiscriminate method of fishing and may capture both wild and hatchery steelhead, especially larger fish, due to the large net mesh size typically deployed for Chinook salmon. Klamath Mountain Province summer steelhead are particularly susceptible to poaching during summer months. Summer steelhead are unusually vulnerable because they are large and conspicuous, aggregate in pools, and are prevented from exiting holding areas by low stream flows. Roelofs (1983) indicated that the most stable populations of summer steelhead are in the most inaccessible streams on public lands, whereas those that are showing signs of severe decline are in areas that are most easily accessible. Roelofs (1983) also indicated that poaching was a factor affecting populations of summer steelhead in, at least, the North Fork of the Trinity, New River, and some tributaries to the Klamath River, although current levels of poaching, 30 years later, are largely unknown. The impact of marine (commercial and recreational) fisheries on steelhead, in general, is poorly known and adult steelhead are rarely documented as by-catch; however, these activities may account for some level of ocean mortality.

Hatcheries. Iron Gate and Trinity River hatcheries are operated to mitigate for the loss of habitat upstream of Iron Gate and Lewiston dams. Current mitigation production goals are 200,000 and 800,000 steelhead smolts, respectively. Rowdy Creek Hatchery, on the Smith River, is a privately operated enhancement hatchery and produces approximately 100,000 steelhead smolts annually (Rowdy Creek Hatchery Five Year Management Plan: 2011/12 through 2015/16) (CDFG 2012b). These three hatcheries, combined, produce about 1,100,000 smolts annually (Lewis et al. 2004). While use of native (within watershed) broodstock is the current practice, fish from outside the Klamath Basin have also been used for broodstock in the past. Fish were transferred from the Sacramento, Willamette, Mad and Eel rivers prior to 1973 (Busby et al. 1996), with unknown consequences related to the genetics of native stocks. Recent studies, however, suggest that hatchery propagation can deleteriously affect the genetics of wild stocks (Goodman 2005, Araki et al. 2008, Chilcote et al. 2011). Interactions between wild and hatchery steelhead are recognized as needing further evaluation (CDFG 2001). In April, 2012, the California Hatchery Scientific Review Group (HSRG) released the California Hatchery Review Report (California HSRG 2012). The report focused on California anadromous fish hatcheries, including Iron Gate and Trinity River hatcheries. The goal of the HSRG review was to ensure that hatchery programs are managed and

	Rating	Explanation
Major dams	High	Major dams block access to large areas of spawning and rearing habitat, alter temperature regimes, and otherwise modify downstream habitats
Agriculture	Medium	Agriculture and water diversions in the KMP, including those for illegal marijuana cultivation, reduce flows and degrade water quality
Grazing	Medium	Cattle/livestock grazing may have substantial but localized impacts
Rural residential	Low	Rural development is widely dispersed but increasing in the region
Urbanization	Low	Minimal urban development within the KMP
Instream mining	Low	Suction dredging has been common throughout KMP watersheds and legacy effects of past gold mining still exist in many areas
Mining	Low	Impacts from hardrock mines and their effluents, while widespread in the KMP, appear to be low
Transportation	Medium	Most primary streams have roads along almost their entire length; roads along rivers degrade water quality and simplify stream habitats
Logging	Medium	Logging is pervasive in KMP watersheds and continues to degrade habitats; legacy effects in watersheds without recent logging continue to limit steelhead production
Fire	Medium	Wildfires are common in KMP watersheds and can result in high levels of sedimentation; fire frequency and intensity predicted to increase with climate change
Estuary alteration	Medium	The Klamath River estuary is relatively unaltered; however, the Smith River estuary has lost ~50% of its historic rearing habitat (Quiñones and Mulligan 2005)
Recreation	Low	Habitats used by summer steelhead for holding are particularly sensitive to recreational use
Harvest	Low	The sport fishery in the KMP is well regulated - illegal to take wild steelhead; poaching may be a limiting factor in some areas
Hatcheries	Medium	KMP hatcheries produce ~ one million steelhead a year; interactions between wild and hatchery steelhead may be detrimental and require further study
Alien species	Low	Alien species are uncommon within KMP watersheds with no known impacts to steelhead

Table 5. Major anthropogenic factors limiting, or potentially limiting, viability of populations of KMP steelhead in California. Factors were rated on a five-level ordinal scale where a factor rated “critical” could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated “high” could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated “medium” is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated “low” may reduce populations but extinction unlikely as a result; and a factor rated “no” has no known negative impact to the taxon under consideration. Certainty of these judgments is moderate. See methods section for descriptions of the factors and explanation of the rating protocol.

operated to meet one or both of the primary purposes for hatcheries: 1) aid in the recovery and conservation of naturally spawning salmon and steelhead populations; and, 2) supporting sustainable fisheries while minimizing impacts to natural populations. The report includes recommendations for improving steelhead management and production at both Iron Gate and Trinity River hatcheries.

Status Determination Score = 2.9 - High Concern (see Methods section, Table 2). The original KMP steelhead ESU (now DPS) was first determined to be "not warranted" for listing under the federal ESA by NMFS in March, 1998. A court decision overturned the ruling in 2000, finding that NMFS relied too heavily on expected effects of future conservation efforts. A final decision was reached on April 4, 2001, and the listing of KMP steelhead ESU under the ESA was again determined to be unwarranted. KMP steelhead are listed by the U.S. Forest Service, Pacific Southwest Region, as a Sensitive Species and are managed by CDFW for sport fishing.

Due to the distinctive life history variations (winter, summer, fall, half-pounder, resident forms), diverse watershed characteristics and impairments, and the difficulties in monitoring during periods of high flow/turbidity, abundance estimates for the entire KMP steelhead DPS are not available. Instead, abundance is determined on a smaller scale, focusing on seasonal timing for individual watersheds. Based on seasonal conditions and survey feasibility, summer and fall adult steelhead have the largest data sets. Relatively few data are available for winter steelhead; however, new monitoring technologies (DIDSON) are providing estimates on the Smith River.

Decreases in hatchery abundances are most noticeable in the Klamath Basin, where recent estimates are well below estimates from just two decades ago. Statistically significant decreases in adult returns were detected for steelhead returning to Iron Gate Hatchery ($p = 0.0004$; Quiñones 2011). Most of the steelhead returning to Iron Gate Hatchery are assumed to be fall steelhead, while the abundance of wild winter steelhead in the Klamath Basin is unknown. Hatchery steelhead returns to Iron Gate Hatchery experienced significant changes in 1969 (increase), 1970 (increase), 1989 (decrease), 1990 (increase), 1995 (decrease), and 2000 (increase). Favorable ocean conditions may explain increases of steelhead abundances during the mid-1960s to early 1970s and in 2000, years during cold Pacific Decadal Oscillations (PDO) phases (Mantua and Hare 2002). Likewise, decreases in numbers may reflect unfavorable ocean conditions (warm PDO phase) in the mid- to late-1980s and early 1990s. Continued and proposed restoration efforts (dam removal, improved agricultural practices) could improve the health of Klamath Basin populations. Returns for fall steelhead to the Trinity River Hatchery have fluctuated over the past decade and, although returns are lower than historical records, the population appears stable. The wild population of fall steelhead on the Trinity also appears to be stable. The Smith River watershed is still largely undisturbed and wild steelhead abundance, although reduced from historic estimates, appears to be stable.

KMP summer steelhead have a spotty distribution throughout the KMP and specialized habitat requirements, making each subpopulation more susceptible to environmental changes and anthropogenic threats than adult winter and fall steelhead. Adult abundance for summer steelhead was historically small and continues to be so, but most populations appear to be stable (Table 4). However, specific threats (i.e., fire, sedimentation, climate change, poaching) are more likely to impact subpopulations, leading to local extirpations. Currently, there are no coordinated, basin-wide, management plans in place for the protection of summer steelhead.

Metric	Score	Justification
Area occupied	4	KMP steelhead are found throughout KMP watersheds; adult summer steelhead have the most restricted distribution due to their life history requirements and lack of access to upper watershed portions of historic range
Estimated adult abundance	3	KMP winter steelhead abundance is largely unknown; summer steelhead subpopulations are small and isolated; fall steelhead are the most abundant, although numbers are heavily supplemented by hatcheries
Intervention dependence	2	Continuous management actions needed for habitat restoration/protection, improved water quality/quantity, law enforcement for the sport fishery and poaching
Tolerance	3	Require clear, cool water; adult summer steelhead require cold water refuges
Genetic risk	3	Presumably generically diverse; however, potential hybridization risk with hatchery steelhead
Climate change	3	All KMP watersheds are projected to be negatively affected by climate change; seasonal water temperatures and flows are already marginal in many areas
Anthropogenic threats	2	See Table 5
Average	2.9	20/7
Certainty (1-4)	3	Data are particularly sparse for KMP winter steelhead; summer steelhead are relatively well studied and monitored annually; fall steelhead are well documented, for both wild and hatchery runs

Table 6. Metrics for determining the status of KMP steelhead in California, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: Restoration and management recommendations for KMP steelhead have been outlined in several plans (Jones et al. 1980, Roelofs 1983, McEwan and Jackson 1996, Voight and Waldvogel 2002). The California Department of Fish and Wildlife, along with partnering agencies/organizations, are dedicated to implementing key elements of these plans to effectively protect and manage KMP steelhead. Objectives include:

1. Increasing naturally-produced stocks of steelhead through the protection of 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. This has already taken place to a certain extent; for example, the Trinity ROD stipulates that ~50% of annual inflow goes to the river whereas, historically, up to 90% was diverted at Lewiston.
3. Restoring favorable instream conditions to benefit multiple species and desired ecosystem functions, rather than focusing on single species management. This concept recognizes that steelhead in the Klamath Basin are a component of a larger community of native fishes, including other salmonids, and restoration efforts should strive to benefit the entire aquatic community.
4. Complete management plans for each subpopulation of summer steelhead throughout the KMP. This task was referenced as "being prepared by DFG" (McEwan and Jackson 1996, p 139), but has not yet been completed.
5. Reduce impacts of hatchery steelhead on wild steelhead populations. Hatchery genetic management plans are being drafted and current hatchery operations are being evaluated based on recent independent scientific review (California HSRG 2012).

Watersheds identified by McEwan and Jackson (1996) as high priority areas for stream restoration to benefit KMP steelhead included: the South Fork of the Trinity River, Scott River, and Shasta River. Many subbasins of the Klamath River are predominantly surrounded by USFS administered public lands and were designated key watersheds as part of the Northwest Forest Plan. Additional measures, such as conservation easements, are required on private lands to restore functioning aquatic habitats and steelhead populations. Fish and watershed monitoring and restoration projects, along with popular sport fisheries, are playing an increasing role in the local economies of the Klamath River Basin, whereas extractive resource industries (timber, mining, etc.) dominated in the past. However, without improved flow management and suitable water quality (i.e. cool and sediment-free), the effectiveness of restoration in many areas will be marginalized (Wu et al. 2000).

In recent years, significant resources have been directed toward mitigating many of the detrimental effects road building and timber harvest have had on KMP steelhead and their habitats (e.g., see http://www.dfg.ca.gov/fish/Administration/Grants/FRGP/). Additionally, private landowners that graze livestock in riparian areas and divert water for agriculture have increased protection efforts such as fencing of riparian areas in the Scott and Shasta valleys. 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, in order to benefit 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 inputs into rivers and streams are ongoing needs.

More research is needed on the life history diversity of KMP steelhead, especially in the Klamath Basin. Managers would benefit from a better understanding of the physical and biological cues that lead to their wide variety of migration and habitat utilization patterns. Determination of survival and escapement rates for wild steelhead is essential to understanding the viability and persistence of individual subbasin populations. For an accurate assessment of the status, distribution and abundance of all populations, monitoring must expand and be well coordinated within the KMP. Additional information regarding the genetics, ecology, and behavior of KMP steelhead is also needed and will help inform management and conservation strategies.

The highest degree of protection for KMP steelhead (and other fishes) is found in the Smith River (Del Norte County), which is the largest river in California without a major dam. In 1990, the Smith River National Recreation Area Act was signed by President George H. W. Bush as Public Law 101-612, which provides some degree of protection. As in the Klamath Basin, where intergovernmental cooperation among tribes, state, and federal agencies, and non-governmental organizations has played an important role in protecting steelhead habitat, a local conservation group (Smith River Alliance) is actively involved in working with federal and state agencies, local stakeholders, tribal representatives, and others to protect the Smith River and its fish fauna. The conservation strategy of acquiring large tracts of private lands to protect important watersheds, such as Goose, Mill, and Hurdygurdy creeks, is a valuable mechanism for conserving steelhead sanctuaries.

Special management consideration should be afforded to KMP summer steelhead populations. Conservation measures should focus on reducing human impacts and improving habitats, especially in ways that improve minimum base flows and maintain cool water temperatures. Summer steelhead populations would benefit, in particular, from restoration actions that reduce impacts from logging and mining (and the many roads created to facilitate these activities). Summer holding and rearing habitat has been repeatedly identified as a critical limiting factor to summer steelhead populations. Land management strategies that seek to reduce sedimentation, increase cover, and minimize other stressors that negatively affect over-summering habitat for adults are critical to recovering populations.

Summer steelhead management should address: (1) improving enforcement of fishing and land use regulations in over-summering areas, (2) identifying watershed management approaches that minimize sediment delivery to streams and maintain high water quality, (3) improving management and, where necessary, implementing restoration of downstream reaches to favor out-migrating smolts, (4) rebuilding present populations through identifying and affording protection to key refuge streams, and (5) restoring populations that have become extirpated. Strategies should incorporate approaches from the Steelhead Restoration and Management Plan for California (McEwan and Jackson 1996).

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 suitable summer holding areas, (3) spatial distribution of spawning areas and whether they require special protection, (4) habitat requirements of out-migrating smolts, and (5) effects of poaching and disturbance from recreation or other human activities on adults. For most populations, there is a need to continue monitoring surveys, as well as identify and mitigate the factors that limit their abundance.

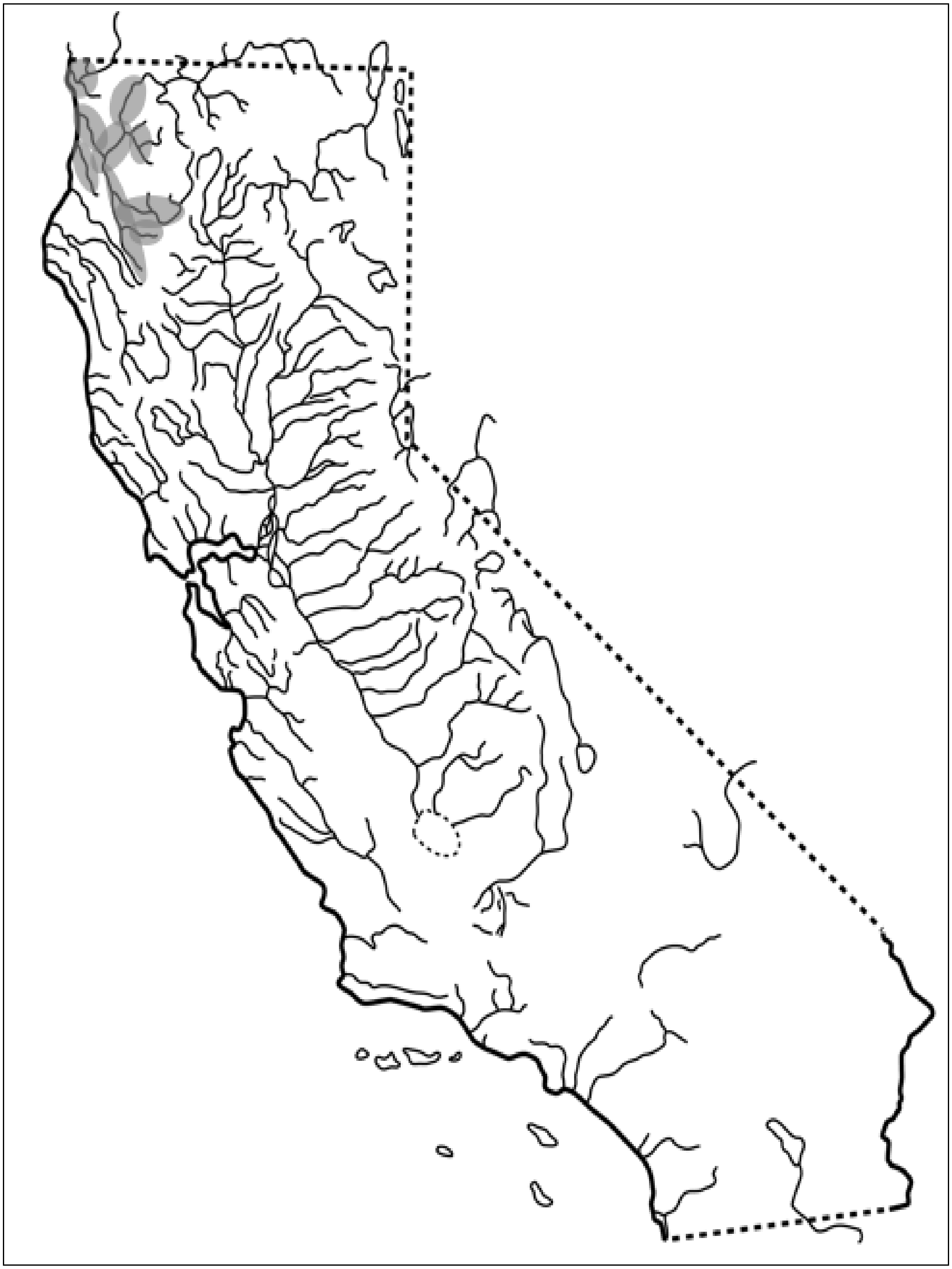


Figure 2. Distribution of Klamath Mountain Province steelhead in California. Individual runs occupy varying portions of the range.

McCLOUD RIVER REDBAND TROUT
Oncorhynchus mykiss stonei (Jordan)

Status: Critical Concern. Because of ongoing and recently increased interest and management, McCloud River redband trout are in no immediate risk of extinction but their populations are small, fragmented, and exist in limited habitats so status could change rapidly, particularly related to predicted climate change impacts.

Description: The following description is based on the Sheepheaven Creek population (Hoopaugh 1974, Gold 1977), which appears to have a somewhat narrower range of meristic characters than the other known populations found in Swamp, Edson, and upper Moosehead creeks. Behnke (1992), however, considered 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 as exemplified by rainbow trout. It has a yellowish to orange body color with a brick-red lateral stripe. The dorsal, anal, and pelvic fins are white tipped. Adults retain parr marks. Gill rakers number from 14-18 (average 16), which is the lowest number known from any rainbow trout population (Behnke 1992). Pyloric caeca number is 29-42, which is also low. However, the numbers of scales along the lateral line (153-174) and above the lateral line (33-40) are greater than in most rainbow trout. Pelvic fin rays are 9-10 and branchiostegal rays range from 8-11. Many, but not all, McCloud River redband trout have basibranchial teeth, 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 U.S. Fish Commissioner, Livingston Stone, who was responsible for a fish hatchery located on the river. However, the lower portion of the McCloud River (below Middle Falls) was historically inhabited by coastal rainbow trout, including steelhead, the anadromous form and other fishes. It is uncertain whether redbands were distributed in these lower reaches and, if so, whether Stone identified them as distinct. The redband trout we recognize today are varieties of inland resident rainbow trout that 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 U.S. to consist of a number of distinct lineages, each independently derived from early invasions of ancestral forms of trout into headwater systems, with populations then becoming isolated through geologic events. Behnke (2002) indicated that McCloud River redband trout are part of a Northern Sacramento River basin trout complex in which all populations are, or were, tied to the headwaters of the Sacramento, McCloud, Pit, and Feather rivers. In theory, the subspecies name *O .m. stonei* could be applied to any population in these headwaters but only the upper McCloud River watershed populations apparently retain unhybridized redbands; these fish are now the exclusive possessors of the subspecies epithet (Behnke 2002).

The population in Sheepheaven Creek, described above, is so distinctive, even from other McCloud River redband trout, that Behnke suggested it should be classified as a separate subspecies. Genetic studies by Berg (1987), using electrophoretic techniques, by Nielsen et al. (1999) using microsatellites, and more recently by Stephens (2007) using nuclear DNA methods, support the conclusion that the Sheepheaven Creek form is distinct but the most recent study (Simmons et al. 2009), using both nuclear and mitochondrial single nucleotide polymorphisms indicates that Sheepheaven Creek and fish from three other streams should be considered together as the McCloud River redband trout group. Of the tributaries to the Upper McCloud River, upper Moosehead, Sheepheaven, Edson and Swamp creeks were found to contain relatively “pure” populations, with few introgressed alleles from coastal rainbow trout. Trout Creek (northern tributary) and most of the southern tributaries to the McCloud River contain redband populations with higher levels of introgression with rainbow trout. Trout in the Upper McCloud River itself apparently retain some genetic and physical characteristics of redband trout but are hybridized with coastal rainbows (Simmons et al. 2009).

Life History: Available information suggests that the life history of McCloud River redband trout is similar to that of other *O. mykiss* populations, including golden trout, in small streams. Redband trout caught from Sheepheaven Creek were in reproductive condition in June, indicating that they spawn in late spring (May-June), as do other rainbow trout at high elevations. The largest fish recorded during a 1973 survey (Hoopaugh 1974) was 208 mm FL, and the population was then estimated at 250 fish over 80 mm FL. Four size classes were found in the stream. Observations in August, 2008, suggest the same age classes were still present (J. Katz, R. Quinones, and P. Moyle, unpublished observations). However, recent (2011) CDFW surveys of Sheepheaven Creek indicated a lack of younger age classes, extremely low abundance, and limited distribution within suitable habitat (J. Weaver, CDFW, pers. comm. 2012).

Habitat Requirements: Habitat requirements for the McCloud River redband are derived from conditions in Sheepheaven Creek (Hoopaugh 1974, Moyle 2002) and the McCloud River, based on descriptions in the 1998 Redband Trout Conservation Agreement (RTCA), 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. During periods of drought, flows are greatly reduced and streams in the upper McCloud basin become intermittent; as a consequence, summer water temperatures can exceed 22°C. 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 and dominated by riffles and runs with under-cut banks. 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). All watersheds are wholly or partially located on the Shasta-Trinity National Forest. Historically, they were apparently present in the mainstem McCloud River above Middle Falls and perhaps in the lower river and its tributaries as well, especially in reaches not accessible to anadromous steelhead. Redband trout from Sheepheaven Springs (McKay Creek) were transplanted into Swamp Creek in 1972 and 1974 and into Trout Creek in 1977 (RTCA 1998). They are now established in both streams. According to a 2011 CDFW survey, putative redband trout exist in streams with a total length of about 8.9 km, with a total estimated population of 3,560 fish (Weaver and Mehalick 2011). Potential habitat, including the upper McCloud River, is about 98 km, or about 50 km in dry years (RTCA 1998). Most of these tributary streams remain isolated from the upper McCloud River due to subsurface flows and may only experience limited connectivity with the McCloud River during high flow events. One exception is Moosehead Creek, which can have subsurface flows during drier periods, but also has an artificial barrier 2.2 km from the confluence with the McCloud River to prevent upstream migration of non-native or hybridized trout.

Stream	Summer Flow class	Redband status	Isolation	Comments
Sheepheaven (McKay)	1	1	3	Key "pure" population
Trout	2	3	3	Introduced from Sheepheaven
Swamp	1	1	3	Introduced from Sheepheaven
Edson	1	1	3	
Tate	2	3	1	
Moosehead (upper)	1	1	2	
Raccoon	1	3	2	
Blue Heron	1	3	2	Possibly extirpated
Bull	1	3	2	
Dry	1	3	2	
Upper McCloud	3	0	1	Dominated by hybridized and non-native trout

Table 1. Redband trout streams in the upper McCloud River. Summer flow class (1 = <1 cfs, 2 =1-5 cfs, and 3 = >5 cfs in late summer in most years). Redband status (1 = 'pure' population, 2 = relatively 'pure', little introgression 3 = good redband population but slightly higher levels of hybridization, 0 = all trout 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).

Trends in Abundance: McCloud River redband presumably once had large, interconnected populations in the Upper McCloud River and tributaries, so the present isolated populations represent greatly reduced remnants of historic populations. Recent genetic analyses indicate that all populations sampled from across the upper McCloud watershed shared alleles in common with the distinctive Sheepheaven Creek population, indicating that redband trout with common ancestry were once widely distributed throughout the basin (Simmons et al. 2009). Existing redband trout creeks were surveyed a number of times from 1975-1992 and in 2011 (Table 2 in RTCA 1998; Weaver and Mehalick 2011). Numbers of fish estimated were highly variable and depended on the stream and habitat sampled; numbers ranged from 53 to 1100 per km. Repeated drought cycles (e.g., 1976-1977, 1987-1992), combined with the predominance of loamy volcanic soils in the watershed, have intermittently reduced surface flows in most McCloud basin streams and limited populations of McCloud redband trout. The same is expected under future drought conditions and may be exacerbated by the effects of climate change. If population estimates are confined to the unintrogressed populations in Sheepheaven, Edson, upper Moosehead and Swamp creeks, then abundance is estimated at 3,560 putative McCloud redband trout (Weaver and Mehalick 2011).

It is likely that habitat conditions and consequent abundance of McCloud River redband trout have improved in the past 10 years, except in extremely dry years. An increase would be the expected response to many ongoing habitat restoration and protection efforts that have taken place. Presumably, habitat protection and restoration, including protection of springs, has moderated population fluctuations and reduced vulnerability to drought.

Nevertheless, it will take considerable effort to maintain McCloud redband trout populations, especially through extended droughts. A particular threat is climate change and potential reduction in stream flows in 25-50 years (once the full effects of global warming hit the Mt. Shasta region). Until then, it is likely that redband populations will continue to maintain themselves, as long as active management continues.

Nature and Degree of Threats: Long-term survival of populations of McCloud River redband trout confined to small, isolated, streams such as Sheepheaven Creek is tenuous because stream habitats are largely diminshed during drought years, a process which can be accelerated by poor watershed management practices impacting upland and riparian areas (Table 2). Fortunately, interest in conservation of McCloud River redbands has resulted in a recent reversal of downward trends in abundance and habitat quality. Factors which threaten McCloud River redband trout populations are: (1) grazing, (2) roads, (3) logging, (4) fire, (5) harvest, and (6) alien species, especially coastal rainbow trout. Upper McCloud streams can be regarded as exceptionally vulnerable to these factors due to their geologic and hydrologic nature.

Grazing. Grazing by cattle and sheep has taken place in the McCloud River watershed for over 125 years and was especially intense in the first half of the 20th century. Heavy grazing, especially by cattle, reduced trout habitat by eliminating streamside vegetation, collapsing banks, making streams wider and shallower, increasing temperatures, reducing bank undercutting, polluting the water with feces and urine, silting up spawning beds, and generally making the habitat less complex and suitable for trout. The reduction of grazing pressure in the late 20th century and the increasing willingness of land managers to implement improved grazing practices has led to better condition of small streams in the McCloud River watershed and improved habitat for redband trout. Today, much of Sheepheaven and lower Trout creeks have been fenced to exclude cattle. The grazing allotment associated with Sheepheaven Creek has not been active for several years, but this could change in the future.

Roads (transportation). Roads, mainly from logging, are numerous and widespread throughout the upper McCloud River basin, providing a source of sediment and pollutant input into streams (potentially covering spawning gravels) and providing easy access to most redband streams in the watershed.

	Rating	Explanation
Major dams	Low	Major dams are downstream of remaining McCloud redband habitat but their construction may have contributed to fragmentation of habitat in the past
Agriculture	n/a	
Grazing	Medium	Historically pervasive in the area but currently limited on private and U.S. Forest Service lands through attrition and better grazing management
Rural residential	n/a	
Urbanization	n/a	
Instream mining	n/a	
Mining	n/a	
Transportation	Medium	Roads are widespread in the upper McCloud basin and are sources of sediment and pollutant input into streams
Logging	Medium	The major land use in the region; associated water drafting may reduce stream flows and cause direct or indirect mortality
Fire	Medium	Headwater areas could be altered by more severe fires than occurred historically
Estuary alteration	n/a	
Recreation	Low	Off-road vehicles a potential threat
Harvest	Low	Light angling pressure in most streams; special fishing regulations to protect key redband populations
Hatcheries	n/a	
Alien species	High	Major potential threat & cause of limited distribution

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

Logging. The region in which McCloud River redband trout live contains a checkerboard of private and public ownership, with most public lands as part of the Shasta-Trinity National Forest. According to the RTCA (1998):

"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, including culverts potentially blocking or limiting instream movement, removal of water for dust control on dirt roads, erosion of sediment from roads, and similar factors. Fortunately, greatly improved logging practices have reduced the effects of logging and logging roads on streams, in good part because both private and public land managers recognize the uniqueness of the McCloud River redband trout and their habitats (RTCA 1998).

Fire. The 1998 RTCA considered fire a potential threat to this subspecies because fire suppression has greatly increased the amount of fuels in surrounding forests and increased the potential for high intensity fires. Such fires can cause direct mortality to fishes (high water temperatures), as well as indirect impacts from increased sedimentation and reduction in riparian vegetation and associated instream shading.

Harvest. It is likely that harvest was never a major problem in the small streams of the McCloud basin but redband trout populations are small enough that even occasional harvest by anglers or scientific collectors could reduce populations (RTCA 1998). Special angling regulations are in place for the following streams: Sheepheaven, Edson and Moosehead creeks (closed to all fishing all year); Swamp Creek (last Saturday in April through November 15 – zero limit, artificial lures with barbless hooks only).

Alien species. Coastal rainbow trout (*O. mykiss*), brown trout (*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, although stocking of coastal rainbow trout in the upper McCloud River was discontinued in 1994 (RTCA 1998). Generally, where alien trout are present, redband trout are absent or have become hybridized. The exact causes of redband trout disappearance from the McCloud River itself have not been documented, but presumably it was a combination of predation on young (brown trout), competition for space (all species), disease introductions (all species), and hybridization (rainbow trout, next section). Fortuitously, a number of 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 by unknown means.

Hybridization between coastal rainbow trout and redband trout is a natural event: both are native to California and hybridization would have occurred where their populations overlapped (e.g. lower McCloud River and tributaries). However, due to planting of rainbows above natural barriers, hybridization has become the primary threat to headwater redband populations which were formerly isolated from coastal rainbow trout. Once hybridization occurs, the rainbow trout phenotype tends to dominate, resulting in a loss of the distinctive, brightly-colored redband trout phenotypes. This is likely coupled with a loss of adaptivity to the unique streams redband trout have evolved in. Rainbow trout and rainbow-redband hybrids have presumably replaced McCloud River redbands in the majority of their historic range, perhaps presenting the greatest threat to redband trout persistence in this basin.

Effects of Climate Change: The fact that existing redband trout streams are so small and flow through highly permeable volcanic soils means that they are exceptionally vulnerable to stressors such as floods, drought and fire, which, in turn, are likely to be more extreme under climate change scenarios. However, the persistence of distinctive trout in Sheepheaven Creek is due to the springs that maintain some level of surface flow (albeit for a short distance), even during severe drought. Presumably, most of the other streams occupied by McCloud River redbands have similar ‘safe’ water sources. If, however, this is not the case, drying of key stream reaches due to climate change may be a critical limiting factor to their persistence. It is also worth noting that spring flows can be eliminated by even minor volcanic or seismic activity and these streams are located in a relatively active region. Additionally, most streams currently inhabited by redbands are already subject to seasonal reductions in flow (during non-drought periods), so increases in air temperature or reductions in snow pack may dramatically reduce available habitat. Moyle et al. (2013) consider McCloud redband trout to be “critically vulnerable” to climate change because of the small size of their streams, warmer temperatures, and the potential effects of lengthy drought.

Status Determination Score = 1.7 – Critical Concern (see Methods section Table 2). Long-term drought, fire, or other factors that affect stream flows or habitat suitability, coupled with genetic risks associated with isolation of small populations, threaten McCloud redband with possible extinction. McCloud redband populations are especially vulnerable to rapid changes in status due to their small, isolated populations. While high levels of interest and management scrutiny seem to preclude *immediate* risk of extinction, recent events such as rescue efforts and movement of vulnerable populations into artificial refuge sites is of concern. In longer time frames, extinction probability will increase as the climate becomes warmer and droughts more frequent. Genetic risks increase with habitat reductions, potentially leading to bottlenecks in small, isolated populations.

The McCloud River redband trout is considered to be Vulnerable by American Fisheries Society (Jelks et al. 2008) because of its limited distribution and exposure to multiple threats. It was considered to be a Candidate Species for listing by the USFWS in 1994 but, following the signing of the RTCA by the USFS and other cooperators in 1998, it was removed from consideration. However, the conservation agreement does not actually preclude listing if needed (M. Dege, CDFW, pers. comm. 2013). The USDA Forest Service lists it as a Sensitive Species, while NatureServe considers it to be an imperiled subspecies.

Metric	Score	Justification
Area occupied	1	Isolation of at least four populations provides some security, although “pure” populations are clustered fairly close to each other and all are found in Upper McCloud watershed
Estimated adult abundance	2	Minimum total population today is probably more than 3,000 adults, although individual populations presumably have effective sizes of 100-500 fish in drought years
Intervention dependence	2	Recent drought (2012-2014) has necessitated rescue of several populations and relocation to refuge holding facilities until natural conditions improve; continual monitoring, habitat protection and possible installation of barriers required; ongoing implementation and recent revision and expansion of Conservation Strategy is critical
Tolerance	3	It is likely they are fairly tolerant of high temperatures, as are other redband trout, but water quality in their small streams can become too extreme
Genetic risk	1	Hybridization risk with rainbow trout is high; small isolated populations during drought can create genetic bottlenecks and lead to inbreeding depression
Climate change	1	Vulnerable in all streams because of small size
Anthropogenic threats	2	Alien trout, fires, and reduced flows are constant threats; See Table 2
Average	1.7	12/7
Certainty (1-4)	3	Most published information is on Sheepheaven Creek population

Table 3. Metrics for determining the status of the McCloud River redband trout, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: Conservation of McCloud River redband trout is active and ongoing, thanks to the leadership of the McCloud Redband Core Group (RCG), a multi-partner organization (California Department of Fish and Wildlife, Shasta-Trinity National Forest, U.S. Fish and Wildlife Service, private landowners, and others), which is dedicated to the conservation of the McCloud River redband trout. The forging of an expanded and updated draft RTCA (2013), based on the original agreement of 1998, 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. Recent attention has focused on the broader populations within the upper basin and four ‘core conservation populations’ (Sheepheaven, Edson, Swamp, and Moosehead) have been identified and will be managed collectively (J. Weaver, CDFW, pers. comm. 2012). Private and public landowners actively cooperate on conservation, particularly those who comprise the RCG. On private lands, considerable effort has been made to improve roads in ways that minimize impacts to streams, to fence streams from livestock, and to assist in restoration and management activities. The conservation agreement is an effort to provide a systematic framework for all restoration and management activities in the watershed. It is crucial that this agreement be finalized as the working plan to improve conditions for McCloud River redband trout. The following recommended actions to increase protection for redband trout and their habitats are largely drawn from this agreement. Recommendations are not in order of importance.

1. Establish a McCloud River Redband Refuge. A portion of the upper McCloud River basin should be managed for the protection and enhancement of McCloud redband populations and their habitats. The refuge should include the main stem McCloud River and its tributaries above the confluence with Bundoora Spring Creek and, within this broader refuge, a ‘core conservation area’ should be established to provide further protections for populations with low (or no) levels of introgression with coastal rainbow trout (Sheepheaven, Swamp, Edson, and Moosehead creeks). While the refuge area contains all the streams known to contain presumed redband trout at the present time, suitable reaches of other perennial streams should, nevertheless, be evaluated for their potential as future translocation/restoration sites. Streams that have potential for expanding the range of redband trout (particularly within-basin, but also outside of the McCloud basin as warranted) would be of great value in terms of offsetting climate change impacts or stochastic events that may lead to the extirpation of one or more existing populations. Management plans that include eradication of non-native trout should be developed and construction of barriers to prevent alien trout invasions considered. In particular, the upper McCloud River itself should be evaluated as a refuge during periods of reduced stream flow caused by prolonged drought or climate change.

2. Maintain and enhance existing habitats. McCloud River redband trout survive in remarkably small and fragile habitats, so continued work is needed to improve the ability of these habitats to support redband trout and to reduce the impacts of human activities. Of particular concern are grazing and logging practices, but other factors such as fire protection, angling, and off-road vehicles have also been taken into consideration. While management plans and agreements are in place to protect streams, continued vigilance is required to avoid long-term loss of habitat. 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 genetic integrity of existing populations. The present populations of McCloud River redband trout are highly vulnerable to loss of genetic integrity (and phenotypic distinctiveness) due to hybridization with introduced rainbow trout and potential for genetic bottlenecking due to complete isolation of existing redband populations from one another. Efforts are needed, therefore, to protect populations from further inappropriate introductions (e.g., by making vehicle access difficult) or from ‘natural’ invasions from downstream areas (e.g., through construction of barriers). This program should include genetic and phenotypic monitoring as part of the assessment of population health. Consideration should also be given to active movement of putative redbands in order to promote and restore gene flow and increase genetic heterozygosity, in order to offset potential impacts from past and ongoing isolation of existing populations (e.g., donor stock from Swamp Creek moved back in to Sheepheaven Creek).

4. Continue to develop and enforce angling regulations appropriate for protection of redband trout. Sheepheaven, Edson, and Moosehead creeks are closed to all fishing all year. Catch-and-release angling is allowed in Swamp Creek from the last Saturday in April to November 15th, using artificial lures with barbless hooks. These regulations need to be strictly enforced with frequent monitoring of streams.

5. Complete genetic evaluations of all populations. Expansion upon recent genetic research (Simmons et al. 2009), to include additional samples from throughout the upper McCloud basin, is planned (M. Dege, CDFW, pers. comm. 2012) and should allow for the development of a genetic management plan, including the potential for enhancing local genetic diversity by translocating fish between populations. Such translocations must be carefully planned and implemented with both a short and long-term strategy in mind, in order to minimize impacts to donor populations and ensure the genetic integrity of all core populations.

6. 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).

7. *Develop emergency (contingency) plans for rescue of trout from extreme drought conditions, fire, reduction in genetic fitness, or other stressors.* An extended severe drought or catastrophic fire has the potential to reduce or even eliminate stream flows in redband trout streams. Given the existing limited distribution (and isolation) of relatively genetically ‘pure’ McCloud River redbands, a plan for salvaging fish from drying streams or critically low populations and rearing redbands in captivity or elsewhere is imperative, so action(s) can be taken quickly as needed in a planned and methodical manner.

Figure 1. Distribution of McCloud River redband trout, *Oncorhynchus mykiss stonei* (Jordan), in California.

GOOSE LAKE REDBAND TROUT
Oncorhynchus mykiss ssp.

Status: Moderate Concern. While Goose Lake redband trout do not face immediate extinction risk, California populations are not entirely secure because they are largely isolated from one other, most are small, and, during drought periods, the lake population disappears and stream populations contract.

Description: Goose Lake redband trout are similar in appearance to other rainbow/redband trout. Their bodies are a yellowish to orange color with a brick-red lateral stripe. The dorsal, anal, and pelvic fins are white-tipped. Stream-dwelling adults retain parr marks, while lake-dwelling adults become silvery-grey in color. The Goose Lake redband trout has two ecological types: a lake-dwelling form that attains lengths of 45-50 cm TL and a stream-dwelling form that rarely grows larger than 25 cm TL. Behnke (1992) examined six specimens collected by J. O. Snyder in 1904 from Cottonwood Creek, in the Oregon portion of the basin. These fish had 21-24 (mean, 23) gill rakers, 61-64 (mean, 63) vertebrae, and averaged 30 scale rows above the lateral line and 139 scales in the lateral series. See Behnke (2002) for color plates of both lake and stream forms.

Taxonomic Relationships: Redband trout are inland forms of rainbow trout (Behnke 1992, 2002) and the Goose Lake redband belongs in the group that Behnke (2002) calls "redband trout of the northern Great Basin." The Goose Lake redband trout is most similar to redband trout of two adjacent basins: the Warner Basin, California, Oregon and Nevada, and the Chewaucan Basin, Oregon (Behnke 2002). This conclusion is based on the lower vertebral counts and higher gill-raker counts of redband trout in these basins and distinct genetic markers (Behnke 2002). The Goose Lake redband trout has not been assigned a subspecific name but Behnke (2002) suggests that Goose Lake redband 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 (amplified fragment length polymorphism AFLP technique) indicates a close relationship with Warner Valley redband trout (M. Stephens 2007). Simmons (2011), using both mitochondrial and nuclear DNA, found that redband trout from the upper Pit watershed, including Goose Lake, formed a distinctive lineage. No genetic differences between the lake and stream forms in the Goose Lake drainage have been documented. The USFWS lumped Goose Lake redband trout with five other Great Basin redband trout as one Distinct Population Segment when considering a petition for listing them as threatened under the Federal Endangered Species Act (Federal Register 65(54), March 20, 2000, 14932-14936). Although the Goose Lake watershed may have had connections to other Great Basin watersheds during wetter climatic periods, it is clearly isolated from other basins today and, presumably, has been for thousands of years. Regardless of its ultimate taxonomic designation, the Goose Lake redband trout is clearly a distinct evolutionary unit, confined to the Goose Lake basin and nearby headwater streams in the upper Pit River.

Life History: Goose Lake redband trout have two life history strategies: a lake-strategy and a headwater-strategy. Lake-strategy fish live in Goose Lake, where they grow to large size and spawn in tributary streams. Headwater-strategy fish remain small and may spend their entire life cycle in streams. It is almost certain that the two forms represent one population because the aperiodic desiccation of Goose Lake presumably has eliminated the lake form repeatedly in the past. This was demonstrated in 1992 when the lake dried up entirely during a prolonged drought. In the next two years, the lake refilled and, about three years later, small runs of large trout again appeared in the streams. It is assumed that the lake dwelling form was reestablished from tributary stream-resident populations. In the small, cold streams of the Warner Mountains to the east of Goose Lake, scattered populations of resident trout persist, completing their entire life cycle in these streams. They look quite different from lake fish because of small size and more vibrant color patterns, reflecting responses to a stream environment. Many of these populations are above potential barriers to upstream movement of fish from the lake. Presumably, small numbers of headwater redbands always move downstream, a natural mechanism for dispersing to new habitats or for recolonizing streams wiped out by drought or other natural disasters. Some of these fish reach the lake and, a few years later, they mature and spawn, renewing the cycle. It is also possible that progeny of lake trout can persist in some lower-elevation tributaries (e.g., Cold Creek).

In California, the lake-dwelling form spawns in Cottonwood, 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 occur following snow melt and rain in the spring, usually during late March or in April. Spawning fish are rather pale looking, perhaps as a result of time spent in Goose Lake's highly turbid waters. Adults return to the lake following spawning. Young trout apparently spend one or more years in streams before dispersing downstream (if they leave at all) into Goose Lake. In the lake, the trout likely feed on Goose Lake tui chub, tadpole shrimp, and other super-abundant food. Growth appears rapid; scales from 6 spawning fish (27-48 cm TL) taken in 1967 indicated that they were all 3 years old (CDFG unpublished data).

The life history of the stream-dwelling form has not been studied but it is thought to be similar to other redband and rainbow trout that live in small, high-elevation streams. Surveys by CDFW (CDFG unpublished data; Hendricks 1995) indicate that headwater streams have 4-5 length classes of trout, with a maximum size around 24 cm TL. It appears that fish in their third summer are 9-12 cm TL. Lake fish were observed spawning May 14-15, 2007 (CDFG unpublished data), though spawning time is highly dependent on variable water years and amount of runoff.

Habitat Requirements: Goose Lake is a large, alkaline lake that straddles the California border; it is shallow (mostly < 3 m when full), extremely turbid, and highly variable in area (about 500 km^2). Because of its high elevation (1430 m), the lake generally remains cool (<22°C) although summer temperatures in the lake may reach 24°C or higher during the day. During calm days, water temperatures stratify with warm water within the first

25-50 cm of the surface; on most days the wind causes temperatures to be uniformly cool (R. White and P. Moyle , unpublished data, 1989). Goose Lake redbands nevertheless survive warm temperatures, high alkalinities, and high turbidity that exist in Goose Lake during summer months. Presumably, a major factor contributing to their survival is the extraordinarily high abundance of fish, tadpole shrimp (*Lepidurus lemmoni*) and other food in the lake (P. Moyle and R. White, unpublished observations).

Spawning takes places in March-May, whenever flows in Willow and Lassen creeks are high enough to attract trout for an upstream migration (M. Yamagiwa, USFS, and S. Reid, pers. comm. 2007). Most spawning areas are located in reaches and tributaries with permanent flows, such as Cold Creek, a tributary to Lassen Creek about 15 km upstream from the lake. Spawning sites are reaches with clean gravels and riparian cover that maintain cool water temperatures. Goose Lake redbands have been observed to spawn in the lower reaches of Willow and Lassen creeks when access to upstream areas is blocked (P. Chappell, pers. comm. 1995), but most spawning areas are upstream of the Highway 395 crossing. However, spawning migrations and behavior of Goose Lake redband trout has been poorly recorded in California.

Tate et al. (2005) evaluated temperatures in the two largest California tributaries to Goose Lake, Lassen and Willow creeks. Lassen Creek, the larger of the two (1-2 cfs flows in late summer), became progressively warmer from headwaters to mouth, so that headwater reaches were typically <16°C in summer, while lower reaches typically averaged 18-21°C, all reasonable temperatures for trout. However, in the summer of 2007, temperatures in some reaches supporting trout regularly reached 24-26°C (S. Purdy, unpublished data). Likewise, Tate et al. (2005) found temperatures in Willow Creek (< 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. Streams in the Warner Mountains are generally dominated by riffles with undercut banks. Pools in meadow areas provide habitat for larger fish. Dense overhanging vegetation, especially willows, provide essential cover.

The environmental tolerances of Goose Lake redband trout have not been measured but it can be inferred that they can survive temperatures of 24°C for short periods on a regular basis, highly turbid, alkaline water (pH 8-9), and dissolved oxygen levels at <50% of saturation, although growth may be inhibited under more extreme conditions.

Distribution: Goose Lake redband trout are endemic to Goose Lake and its major tributaries and a few tributaries to the upper Pit River. In California, Lassen and Willow creeks are their principal streams although they are also present in smaller streams (Pine, Cottonwood, Davis, 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 North Fork 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). Simmons (2011) identified genetically similar fish in North Fork Fitzhugh Creek, tributary to South Fork Pit River and in Parker Creek, Tributary to North Fork Pit River, south of Goose Lake. In addition, two populations in the eastern Warner Mountains above Surprise Valley seem to be Goose Lake redbands, perhaps as the result of historic introductions (Stephens 2007).

Trends in Abundance: According to local history, in the 19th century these trout were once abundant enough in the lake that they were harvested commercially and sold to logging camps. Conversations with local residents (P.B. Moyle 1989) indicated that both sport and commercial fisheries existed for Goose Lake redband trout and that large runs occurred in local creeks, especially Thomas Creek in Oregon. The Goose Lake redband trout population historically has undergone major fluctuations, being depleted during series of dry years and recovering in wet periods. The lacustrine population was severely depleted during the 1976-1977 drought, recovered during the wet early 1980s, and dropped precipitously during the 1986-1992 drought. Most recently, the lake was dry in 2010 and remained very low through 2012.

In California, Lassen Creek and its tributary, Cold Creek, have been the principal spawning streams. Numbers of spawning fish have fluctuated from ten or so individuals to several hundred, but the creek appears to have the potential to support perhaps 1,000 spawning fish under optimal flow conditions (E. Gerstung, CDFW, pers. comm. 1995). The only large run documented in recent years in Lassen Creek (1988) was comprised of several hundred spawners (J. Williams, unpubl. data), which suggests that there were fewer than 1,000 adults from California streams in Goose Lake, assuming many of the lake fish were immature one and two year old fish. In 1989, in the middle of a drought, only about a dozen fish appeared in the creek and there was no evidence of successful spawning.

Goose Lake dried up in 1992 but, by March, 1997, a run was reported in Lassen Creek and spawning was reported in April in Cold Creek (M. Yamagiwa, USFS, pers. comm. 2007). In May, 1999, S. 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 in a relatively short period of time.

The stream form of Goose Lake redband trout apparently exists in about 20 small headwater streams. ODFW (2005) estimated that about 102,000 trout (+/-32%) age 1+ and older ($0.14/m^2$) live in 13 Oregon streams under typical conditions; this number is presumably low compared to numbers that existed before streams were degraded by grazing and other activities. Surveys of California streams made in 1993 and 1999, showed 600-1600 trout per km in Lassen Creek, which suggests that densities/numbers in California and Oregon streams are roughly comparable (unpublished surveys, CDFW). More recent CDFW multiple-pass electrofishing surveys (Weaver and Mehalick 2010) showed 114-747 trout per km in Lassen Creek and 313-451 trout per km in Cold Creek, considerably lower than previous estimates but with the caveat that section lengths were estimated in 1999 (J. Weaver, CDFW, pers. comm. 2013), so abundance estimates may or may not be accurate for that year.

ODFW (2005) indicated that most Oregon redband trout streams are impaired to some degree by accumulated 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, largely due to extensive habitat restoration efforts. Overall, the carrying capacity of Goose Lake streams is presumably a fraction of their historic carrying capacity. Since 1995, conditions for Goose Lake redband trout in California have steadily improved because large sections of Lassen Creek and other streams have been protected from grazing and otherwise restored. These conservation measures have likely improved habitat conditions and allowed runs of lake fish to re-establish themselves. Presumably, headwater populations have increased as well, thanks to better management.

Nature and Degree of Threats: Goose Lake redband trout populations have been affected by many stressors, but habitat degradation and diversions have been the greatest threats (Table 1). ODFW (2005) indicated that these two factors, combined, put Goose Lake redband trout "at risk" in 80% of Oregon streams. Overexploitation and introduced species are, at present, minor problems. However, all threats are exacerbated during periods of severe drought. Goose Lake dried up in the 1420s, in the 1630s, 1926 (with low lake levels from 1925 to 1939), 1992, and 2010-2012. Thus, the key to survival of the Goose Lake redband trout (and other Goose Lake fishes) is maintenance of populations in tributaries that may have severely reduced habitat during these drier periods.

Agriculture. Populations of the lake-dwelling form were reduced because access to spawning areas was blocked by dams, diversions, culverts, and channelization in the lower reaches of many streams but, since 1995, most of these impacts have been mitigated or eliminated. Much of the critical stream habitat for Goose Lake redband trout is on private land and, at times, large volumes of water are diverted to irrigate fields. On some streams, small diversion dams are barriers to fish movement (ODFW 2005). Diversions may have disproportionate impacts in dry years because they have the potential to dry longer stream reaches that are refuges for trout and other fishes when the lake is dry.

Grazing. Headwater streams containing redband trout have been heavily grazed, resulting in reduced riparian cover and, in places, down-cutting to bedrock. The impact of grazing has been reduced in recent years through a combination of fencing, rotational grazing, installation of erosion control structures, and planting of willows.

Transportation. All streams in the watershed have been degraded by roads to some degree. Highway 395 crosses all tributaries to the east side of the lake and culverts under the highway were once a partial barrier to migration, an issue which has largely been fixed. Roads also impact headwater streams, especially where culverts may be barriers to fish movement or where the road-cuts are a source of silt. Some streams face multiple threats from poor water quality as the result of road building, channelization, and waste materials from uranium mines.

Logging. Timber harvest is a prominent use of the watershed's forests and has contributed to habitat degradation in streams through siltation, road-crossings, and other factors. Logging impacts were more severe historically; many regulations exist today to protect stream habitats from the effects of timber harvest operations.

Harvest. When lake-dwelling fish are moving upstream to spawn, they are 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. At present, only catch-and-release angling for redband trout is permitted in Goose Lake's California tributaries.

	Rating	Explanation
Major dams	n/a	
Agriculture	High	Water diversion and return flows from irrigation lower base flow and increase water temperatures; dams may block migration
Grazing	High	Pervasive in the area, especially in meadows with redband streams; reduced impacts in recent decades with improved management
Rural residential	n/a	
Urbanization	n/a	
Instream mining	n/a	
Mining	Low	Old uranium mines in watershed; unknown impacts
Transportation	Medium	Roads are a source of sediment input into streams and culverts have blocked access in the past
Logging	Medium	Logging and associated roads have likely contributed to stream degradation; greater impacts in the past
Fire	Low	Fire suppression, coupled with increasing aridity, predicted with climate change, may contribute to increased fire frequency and intensity
Estuary alteration	n/a	
Recreation	Low	Off road vehicles a potential threat but not demonstrated
Harvest	Medium	Poaching is potentially a problem; legal fishing pressure is light and limited to catch-and-release
Hatcheries	n/a	
Alien species	Medium	Major potential threat in streams if introduced; less so in lake

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

Alien species. Brook, brown, and rainbow trout have been introduced into streams of the Goose Lake drainage and brown trout are known to persist in California in Davis and

Pine creeks (Hendricks 1995, S. Purdy, UC Davis, unpublished data, 2006, P. Divine, CDFW, pers. comm. 2012). Brook trout are still present in at least one Oregon stream (ODFW 2005, Scheerer et al. 2010). California has not stocked any rainbow trout in the drainage since 1980, when electrophoretic studies indicated that the native redband 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 rainbow trout (ODFW 2005). Behnke (1992) thought that some Goose Lake redband trout populations in California showed evidence of past hybridization with rainbow trout, based on meristic measurements, but there is no biochemical evidence of this.

The potential for future unauthorized, illegal introductions to impact native trout and other sensitive Goose Lake fishes remains although is unlikely. Possible effects to native fishes could occur through disease, hybridization, predation, or competition; however, some past introductions of warm-water fishes were largely unsuccessful because of the lake's extreme environment.

Although it is uncertain whether beavers were historically distributed in the Goose Lake basin, beaver dams in Lassen Creek's middle reaches have created intermittent barriers to upstream migration and may have blocked recent lake fish runs from reaching preferred spawning habitat (J. Weaver, CDFW, unpublished observations, 2012). The California Department of Fish and Wildlife has, in the past, periodically used explosives to remove beaver dam complexes in Lassen and Willow creeks in order to improve upstream passage for Goose Lake redband trout, although this practice is no longer utilized (P. Divine, CDFW, pers. comm. 2012). Beaver dams may need to be evaluated in the future to determine if fish passage is being impeded.

Effects of Climate Change: Goose Lake is located in an arid, high desert region so any reduction in precipitation or increased frequency of droughts will further stress streams and the lake. Both are predicted by climate change models (Moyle et al. 2012). During low flow periods, streams in the Goose Lake basin already reach temperatures (24-26°C) that are lethal or nearly so to redband trout. Thus, an increase in air temperature, especially when combined with reductions in stream flow through diversions, could reduce or even eliminate most California populations. An increase in fire frequency or intensity could reduce riparian shading, add sediment, and otherwise impair streams in which redband trout are found. In addition, increased frequency of Goose Lake's known aperiodic dessication or increased temperatures in the lake could have negative effects on the lake dwelling and migratory part of the population. Moyle et al. (2013) rated Goose Lake redband trout as critically vulnerable to climate change, with extinction likely in California in the next 100 years if present climate change trends continue.

Status Determination Score = 3.3 - Moderate Concern (see Methods section Table 2). Goose Lake redband trout face no immediate extinction risk (Table 2) but their populations are not entirely secure because: (a) the 19 extant populations, 6 in California and 13 in Oregon, are largely isolated from each other, (b) most stream populations are small, and (c) drought periods are predicted to increase over the coming century, during which the lake population disappears and stream populations shrink. Warmer temperatures will also reduce the quantity and quality of stream refuges.

The Goose Lake redband trout has been given various designations by state and federal agencies: (a) USFWS, Category 2 Candidate Species (now, Species of Concern); (b) USFS, Region 5, Management Indicator Species; (c) USFS, Region 6, Sensitive Species, and (d) ODFW, Vulnerable or At Risk species. The American Fisheries Society lists it as "Vulnerable," while NatureServe lists it as "Imperiled" (T2) (Jelks et al. 2008).

In 1997, the USFWS was petitioned to list Great Basin redband trout, which includes Goose Lake redband trout, as threatened or endangered. In 2000, the petition was denied (Congressional Record, March 20, 2000:65 (54):14932-14936) 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."

Metric	Score	Justification
Area occupied	4	Present in six streams in California and 13 in Oregon
Estimated adult abundance	4	Lake spawners are <1000 but headwater populations presumably contain more fish
Intervention dependence	4	Long-term decline reversed by restoration actions which must continue to protect remaining habitats
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 is possible; potential impacts from isolation of headwater populations need investigation
Climate change	2	Distribution in isolated, small streams increases probability of extirpation in California due to prolonged drought
Anthropogenic threats	2	See Table 1
Average	3.3	23/7
Certainty (1-4)	2	Mostly 'grey' reports and expert opinion

Table 2. Metrics for determining the status of Goose Lake redband trout in California, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

The USFWS analysis also cites the many successful restoration projects in the Goose Lake Basin as further reason for finding that listing was not justified. However, because fish in California depend largely on just two streams, Lassen and Willow creeks, for survival, they could face extirpation from California even if there are viable populations in Oregon. It is likely that better and more current information on California populations and better resolution of levels of movement (or lack thereof) of lake dwelling fish between tributaries, both in Oregon and California, would change their status.

Management Recommendations: 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, the Goose Lake Fishes Working Group was formed in 1991, made up of representatives from both California and Oregon, and comprised of private landowners, state and federal agencies, 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, restoration actions included making road under-crossings passable to trout. A fish ladder was installed over a major diversion dam on Thomas Creek in 1992 by the Oregon Department of Fish and Wildlife. In Willow and Lassen creeks, the California Department of Fish and Wildlife has removed natural and artificial migration barriers. Headcut control, bank stabilization, stream fencing, planting of riparian vegetation, modified grazing practices and other protective measures have also been undertaken on a number of streams in recent years. These measures have greatly improved habitat and water quality in Goose Lake tributaries, including the lower reaches that flow through agricultural land. Monitoring of water quality, insects, and fish demonstrate the improvements (Tate et al. 2005); however, continued effort is needed to maintain (and ideally increase) the populations of trout and other fishes, especially during periods of severe drought.

Management recommendations (not in order of priority) include:
1. Identification and modification of barriers to fish movement, especially diversion dams.

2. Identification, protection, and improvement of stream reaches that are critical for spawning, rearing, and refuge during drought. Cold Creek (tributary to Lassen Creek) and Buck Creek (tributary to Willow Creek) have already been identified as important habitats. At present, a diversion structure often diverts the flows of lower Buck Creek. Lower Willow Creek habitat conditions are poor (bank sloughing, minimal riparian or instream cover, heavy sedimentation), along with multiple diversion dams. Although these dams were, at some point, improved with fish ladders, some of these structures appear badly deteriorated and fish passage needs to be reevaluated (J. Weaver, CDFW, unpublished observations, 2012).
3. Regular quantitative monitoring (every 3-5 yrs) of fish populations in both upstream and downstream reaches of Lassen and Willow creeks, and at least qualitative monitoring of fishes in other streams. In 2012, CDFW received a Sport Fish Restoration Act grant from the US Fish and Wildlife Service for the purposes of implementing quantitative fish population and habitat monitoring in California tributaries to Goose Lake, so data gaps should be filled and trend monitoring can occur. Collaborative planning between CDFW and ODFW is occurring and basin-wide monitoring strategies should be developed in the next several years (P. Divine, CDFW, pers. comm. 2012).
4. Improved management of headwater areas to protect streams from livestock grazing and other stressors, including predicted impacts of climate change.
5. Prevent the illegal importation/stocking of non-native fish in the Goose Lake basin, including eradicating existing populations where possible. The abundant tui chubs and aquatic invertebrates in Goose Lake 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 otherwise negatively impact native fishes should continue to be banned and enforced.
6. Adult lake-form trout attain large sizes and spawn in small streams; as such, they are susceptible to poaching. Regular patrol by wardens and others should be conducted to prevent poaching as adults amass in pools and shallow spawning areas.
7. The Goose Lake Fishes Conservation Strategy should be fully implemented and revisited periodically to ensure it is up to date. The continued involvement of private landowners and public agencies is crucial to this effort, as is the continued involvement of University of California Cooperative Extension, which has provided coordination and scientific studies to support conservation efforts.

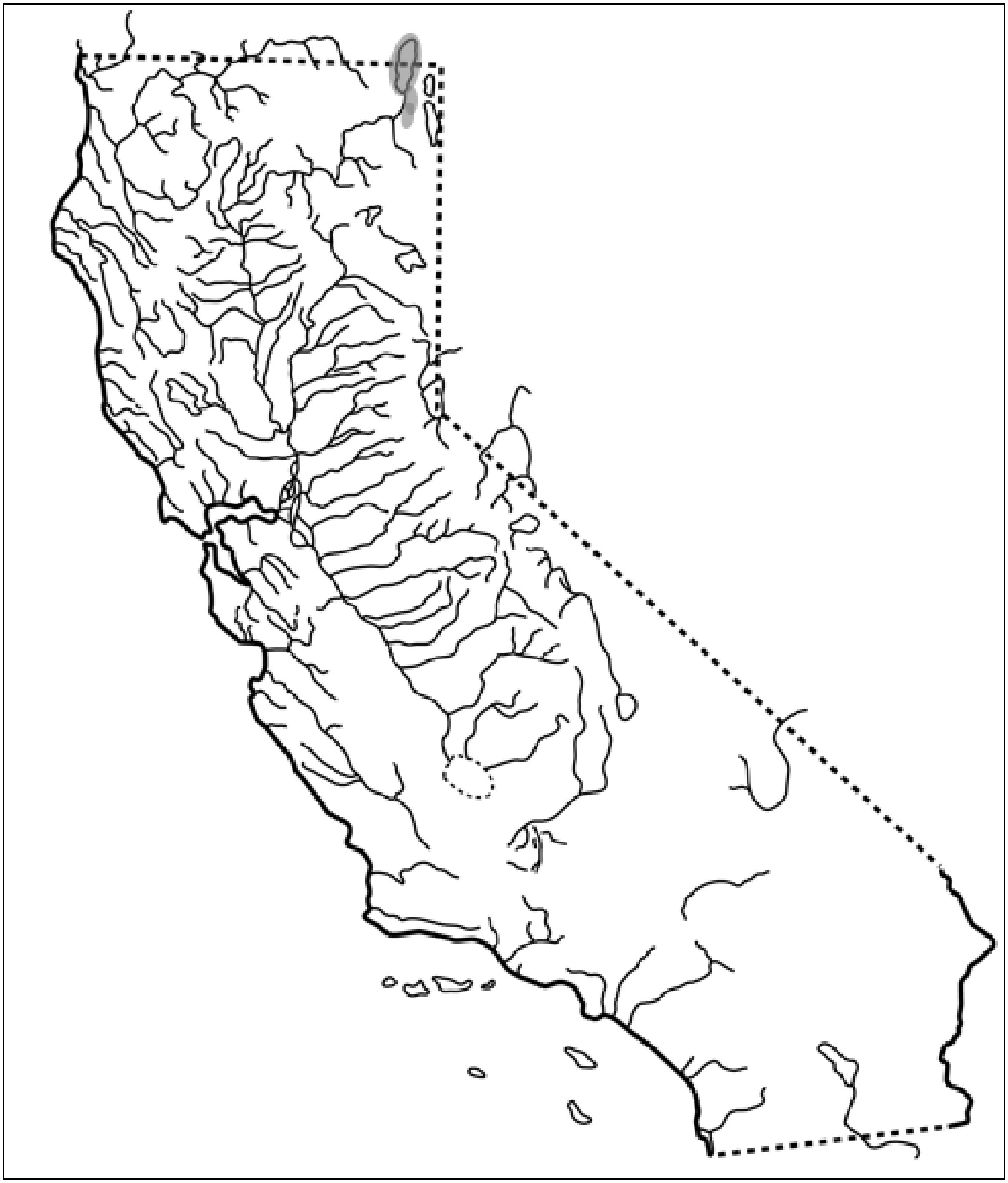


Figure 1. Distribution of Goose Lake redband trout, *Oncorhynchus mykiss* ssp., in Goose Lake, upper Pit River, and above Surprise Valley, in California.

EAGLE LAKE RAINBOW TROUT
Oncorhynchus mykiss aquilarum (Snyder)

Status: High Concern. The Eagle Lake rainbow trout (ELRT) does not exist as a self-sustaining wild population because of dependence on hatchery propagation. Habitat degradation and the presence of alien brook trout in Pine Creek, the ELRT's principal spawning grounds, along with continued reliance on hatchery production to maintain the ELRT population will make it increasingly difficult to re-establish a wild population.

Description: This subspecies is similar to other rainbow trout in gross morphology (see Moyle 2002), but differs slightly in meristic counts, especially in having finer scales than coastal rainbow trout. It is also distinctive in possessing 58 chromosomes, rather than the 60 typical of other rainbow trout (Busack et al. 1980).

Taxonomic Relationships: Snyder (1917) described this trout as a subspecies of rainbow trout, *Salmo gairdneri aquilarum.* However, Hubbs and Miller (1948) examined Snyder's specimens and concluded that ELRT were derived from hybridization between native Lahontan cutthroat trout (presumed to have occupied Eagle Lake prehistorically) and introduced rainbow trout. Miller (1950) later retracted the hybridization theory. Needham and Gard (1959) then suggested that ELRT were descended from introduced or immigrant rainbow trout from the Feather or Pit River drainages. Behnke (1965, 1972) proposed a redband-rainbow hybrid origin, although redband trout are now considered to be rainbow trout subspecies. Busack et al. (1980), in an extensive electrophoretic, karyotypic and meristic analysis, suggested that ELRT were derived either from immigration or an unrecorded introduction of a rainbow trout with 58 chromosomes. The distinctive morphology, ecology, and physiology of this form all point to ELRT being derived from natural colonization from the Sacramento River drainage. Behnke and Tomelleri (2002) speculated that Lahontan cutthroat trout were the original inhabitants of Eagle Lake but that they disappeared during the Pleistocene during an extended period of drought. During a wetter period, rainbow trout managed to invade through an unspecified headwater connection (Behnke and Tomelleri 2002). Recent genetic studies (ALFP DNA techniques) suggest that the closest relatives of ELRT are rainbow trout from the Feather River (M. Stephens 2007, Simmons 2011). Given the relatively recent volcanism and resulting uplift and mountain building in the vicinity of Lassen National Park (near the headwaters of the Feather River), it is plausible that historic wetted connectivity existed between the Feather River and Pine Creek, Eagle Lake's main tributary (R. Bloom, CDFW, pers. comm. 2012).

Life History: Eagle Lake rainbow trout are late maturing (usually in their third year for females) and were historically long-lived, up to 11 years (McAfee 1966). Trout older than five years are rare in the lake today, although individuals as old as 8-9 years have been caught (CDFW, unpublished data). Historically, the trout spawned primarily in Pine Creek, which flows into the lake on the western shore and, presumably, on occasion, in the much smaller Papoose and Merrill creeks, which feed the southern end of Eagle Lake. Upstream migrations took place in response to snowmelt-fed high flows in March, April, or May. In the Pine Creek drainage, principal spawning areas were presumably gravel-bottomed, spring-fed creeks, such as Bogard Spring Creek, and headwaters in meadows, especially Stephens Meadows, about 45 km from the lake. In the past, it is likely that the trout spent at least their first 1-2 years of life in these stream habitats before migrating to the lake, much like coastal steelhead. However, it is possible some became stream-resident, while retaining the capability of producing migratory progeny, similar to steelhead and other lake-dwelling trout populations, such as Goose Lake redband trout (Moyle 2002). In recent years, progeny of adults transported to the upper basin have been found to be as old as four years. It is also possible that ELRT spawned successfully in the lower reaches of Pine Creek, with fry washing into the lake. In 2010 and 2011, 26 (21 male and 5 female) and 150 adult spawners (60 male and 40 female PIT tagged fish, along with 50 others), respectively, were released above the weir in lower Pine Creek in April. In June, fry (30-40 mm TL) were collected from the trap downstream (P. Divine CDFW, pers. comm. 2012). It is not known if these fish can survive in the lake.

Yearling ELRT from hatchery plantings grew to about 40 cm by the end of their first year in the lake, 45-55 cm in the third, and up to 60 cm in the fifth year (McAfee 1966). These fish could (at least in the past) apparently reach 3-4 kg and 65-70 cm FL (McAfee 1966). Data from the last 10 years shows that mature females produce an average of 3,300 eggs (Crystal Lake Hatchery, CDFW, unpublished data, 2009). Rapid growth is the result of abundant forage in Eagle Lake, combined with a delay in maturity until 2-3 years of age. This latter trait has made them highly desirable as a hatchery fish (Dean and Chappell 2005).

The life history of these fish has been significantly altered because access to spawning grounds in Pine Creek has been obstructed since the late 1950s. As fish move up Pine Creek in the spring, they are trapped at a permanent weir installed by CDFW and artificially spawned. The fertilized eggs are then taken to Crystal Lake and Darrah Springs hatcheries where they are hatched and the young reared for 14-18 months. The first generation fish that originate from parents captured in the trap are planted in Eagle Lake at 30-40 cm FL (CDFW, unpubl. data). 160,000-180,000 fish are planted in the lake each year; about half in the fall near the mouth of Pine Creek, in the vicinity of Spaulding, and the other half are planted in the spring in the south basin. In addition, between 5,000 and 10,000 1+kg ‘bonus’ fish have been planted each year for the sport fishery. Progeny of the fish captured in the Pine Creek trap are also reared in other hatcheries in California and planted widely in reservoirs (Carmona-Catot et al. 2011).

All trapped fish are marked in order to prevent sibling crosses (reduce inbreeding), avoid using fish that have been more than one generation in the hatchery, and to select for longer-lived fish to compensate for longevity reductions that may have been caused by past hatchery practices (R. L. Elliott, CDFW, pers. comm. 1998). Currently (beginning in 2001), no ELRT are planted that have been more than one generation in the hatchery (P. Divine, CDFW, pers. comm. 2012). Formerly, a hatchery program for rearing ELRT was maintained at Mt. Shasta Hatchery by using wild-caught fish as brood stock for one generation. The progeny of these fish were originally 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 U.S. Eagle Lake rainbow trout are prized because of their delayed maturity, rapid growth and longevity. As noted, all fish reared in hatcheries for planting in Eagle Lake are first generation ELRT from the Pine Creek trap, although fish from hatchery broodstock were planted in combination with first generation fish from the Pine Creek trap into Eagle Lake in the past (P. Divine, CDFW, pers. comm. 2009).

Despite this long (60+ year) history of hatchery selection, there is evidence that ELRT can still spawn successfully in Pine Creek. Fish that were trucked to the upper reaches of Pine Creek in the 2000s produced young which survived and grew for two years. A thorough survey of Bogard Spring Creek revealed the presence of at least 170 ELRT in 2007, with most fish lengths between 105 and 150 mm FL; in 2008, only 25 ELRT were captured with lengths between 130-165 mm FL, while 34 ELRT were captured in 2009 (Figure 1; Carmona-Catot et al. 2010, 2011). These fish survived and grew despite the presence of about 5,300 brook trout in the same reach of stream in which they were found (see management section below for details). There is some evidence that two year old fish will try to migrate downstream to the lake during periods of high spring flow (P. Moyle, unpublished observations, 2006). In spring, 2009, an ELRT was captured in Pine Creek at 800 meters downstream from the confluence with Bogard Spring Creek. This fish was fin clipped in September, 2008 in Bogard Spring Creek (Moyle and Carmona, unpublished data). In 2011, a single male ELRT managed to migrate the entire distance from the weir to the upstream spawning areas (T. Pustejovsky, pers. comm. 2011).

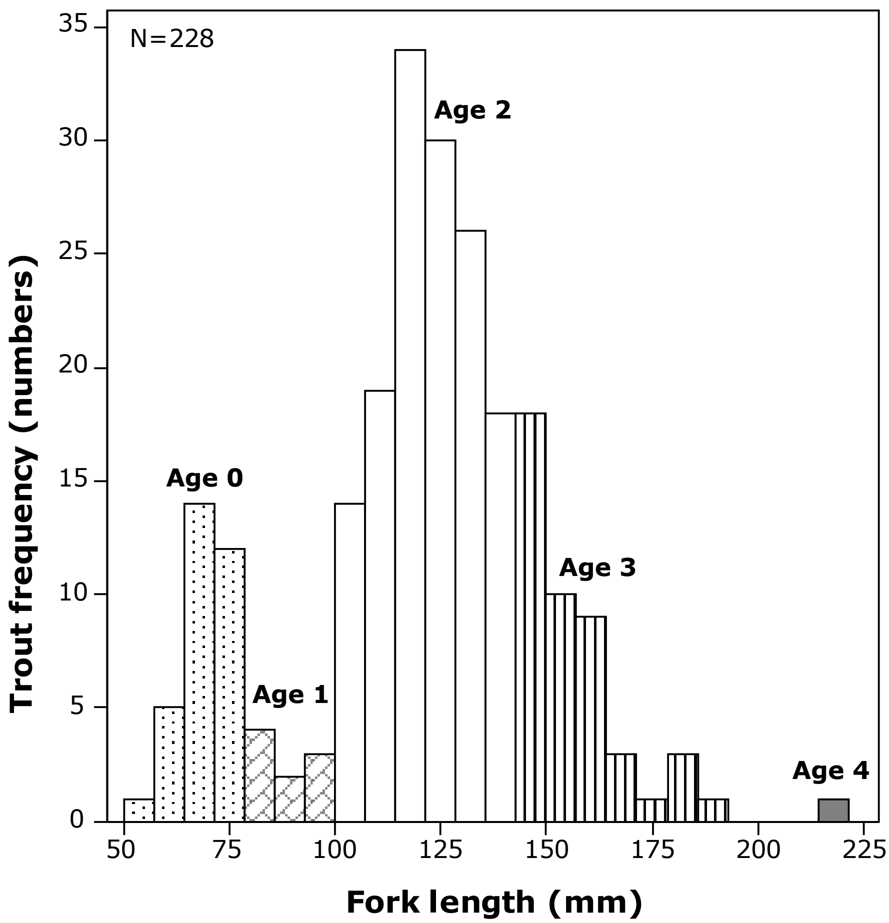


Figure 1. Fork lengths and ages of Eagle Lake rainbow trout in Bogard Spring Creek sampled in 2007, 2008, and 2009. Age distributions are inferred from scales of 71 fish. From Carmona-Catot et al. (2011).

The diet of ELRT varies with age and season. Newly planted trout in their first year in the lake feed mainly on zooplankton, including *Daphnia* spp. and *Leptodora kindti*, as well as on benthic invertebrates, especially leeches and amphipods. By August, most of the trout switch to feeding on young-of-year tui chubs (King 1963, Moyle 2002, Eagles-Smith 2006).

Habitat Requirements: Eagle Lake rainbow trout spend most of their life in Eagle Lake, a large (24 km long by 3-4 km wide), highly alkaline lake. The lake consists of three basins: two of them average 5-6 m deep in most years, but drop to 2-3 m during severe drought and the third averages 10-20 m, with a maximum depth of about 30 m. The shallow basins are uniform in their limnology and water temperatures may exceed 20°C in the summer. The deep basin stratifies so, in late summer, most of the trout are in the deeper, cooler water of this basin. Otherwise, they are found throughout the lake. Currently (2012-13) the lake is at near record low levels, so the upper basins are only about 2-3 m deep. How this has affected the ELRT population in the lake is not known.

During the summer, upper Pine Creek is a cold, spring-fed stream, flowing at .03-0.14 m^3/s through meadows and open forest, with modest gradients. Bogard Spring Creek is also a spring-fed creek, with flows of 0.01-0.02 m^3/s. The meadow streams have deep pools and glides with deeply undercut banks, providing abundant cover for trout. The Pine Creek watershed is described in detail by Pustejovsky (2007). Unfortunately, the trout present today in the Pine Creek watershed are almost entirely alien brook trout in high densities (Carmona-Catot et al. 2010).

Environmental tolerances of ELRT are high for a trout. In Eagle Lake, they live in highly alkaline water (pH 8.4-9.6), in which dissolved oxygen is usually at or close to saturation (except in the hypolimnion of the south basin during months of thermal stratification). They have been observed foraging in shallow water at temperatures of 22-23°C but generally retreat to deeper, cooler areas (<20°C) as lake temperatures increase. The requirements of spawners and juveniles in streams have not been well studied but are presumably similar to those of other rainbow trout (see Moyle 2002).

Distribution: Eagle Lake rainbow trout are endemic to Eagle Lake, Lassen 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 weir and fish trap at the mouth of Pine Creek. In the past, hatchery trout have 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, although supporting data are largely absent.

Trends in Abundance: Naturally-spawned ELRT were once abundant in the lake. According to Purdy (1988), "In the spring months of the 1870s and 1880s, when trout were spawning, huge quantities were being caught. It was not unusual to hear that wagon loads of trout, some weighing as much as 600 pounds, were being brought into Susanville where they were sold at local markets for twenty-five cents a pound (p. 14)." This exploitation occurred at the same time as extensive logging in the drainage, heavy grazing of the basin's meadows, and the first construction of railroad grades and roads across meadows and streams, all of which altered stream hydrology and morphology. When the ELRT was described by Snyder (1917), he noted its numbers were low. Although commercial fishing for trout was banned in California in 1917, ELRT populations remained low, presumably because of the poor condition of Pine Creek and the establishment of predatory largemouth bass and brown bullheads in the lake. By 1931, trout were scarce in the lake and Pine Creek (Snyder 1940).

During the 1930s, trout populations were further stressed as lake levels dropped dramatically when diversion of water through Bly Tunnel combined with prolonged drought to reduce spawning access to Pine Creek. In 1939, biologists with the Lassen National Forest expressed concern over impoundments further reducing flows of drought-stricken Pine Creek (Pustjevoksy 2007). Meanwhile, logging, railroad construction, and other human alterations to the basin further degraded the Pine Creek watershed. Fortunately, high alkalinities brought on by dropping lake levels also eliminated bass from the lake, although bullheads persisted into the 1970s. Even with the return of wetter conditions, the trout population showed little sign of recovery. In 1949 and 1950, CDFW collected 35 and 75 adult ELRT, respectively, from the mouth of Pine Creek, spawning them for hatchery rearing (Dean and Chappell 2005). The 258 progeny from the 1949 fish were planted in Pine Creek, where brook trout had recently become established, but probably did not survive. The spawning of fish in 1950 was more successful and the hatchery-reared progeny were planted in the embayment at the mouth of Pine Creek. From 1951-1958, some artificial propagation also took place, although the records are not clear as to how many fish were produced (Dean and Chappell 2005). Prior to hatchery propagation, trout presumably persisted only because occasional wet years permitted successful spawning despite degraded stream channels and the presence of brook trout in the spawning reaches of Pine Creek (McAffee 1966). It is possible that these actions by CDFW biologists prevented extinction of ELRT although, based on recent genetic evidence, a small component of the population may have been able to migrate upstream during larger flow events until all access to upstream areas was blocked in 1995 (Carmona-Catot et al. 2011).

In 1959, an egg taking station was built at the mouth of Pine Creek, including a wooden weir/dam to block upstream passage of most fish (Dean and Chappell 2005). Regular trapping operations began in 1959, when 16 trout were captured and spawned; in the next five years the numbers captured varied from 45 to 391 (McAfee 1966). From 1959 through 1994, a few trout were able to make it over the barrier during wet years, allowing some potential for natural spawning (Pustejovsky 2007, Moyle, unpublished data). It is unknown, however, if spawning was successful, if progeny survived in degraded stream habitats and in the presence of abundant brook trout, or if any outmigrants during this period were able to return to the lake.

In 1995, the weir was rebuilt to more effectively prevent erosion and prevent upstream movement of all ELRT (Pustjevoksy 2007), based on the assumption that adults migrating up Pine Creek would become stranded as the lower portions of Pine Creek dried and would be lost to the lake population and recreational fishery. The spawning of ELRT then became entirely under human control. At present, eggs and milt are stripped from the fish at the egg taking station. The embryos are then transported to Crystal Lake Hatchery, from where they are distributed to other hatcheries across California (Carmona-Catot et al. 2011). To provide fish for planting, hundreds of trout are trapped each year and between 1 and 6 million fertilized eggs per year are taken for hatchery rearing. Thus, in 2009, 1,737 females were spawned, producing 5,985,880 eggs for the hatchery while, in 2008, the take was 2,757,420 eggs, and, in 2007, 1,113,980 eggs (P. Divine, CDFW, pers. comm. 2009). It should be noted that the passage of California Assembly Bill 7 (AB-7) in 2005 required the CDFW to increase production of native trout forms in hatcheries, thus the incremental increase in egg take from 2007-2009. The egg quotas are developed every year by CDFW hatchery personnel in order to achieve the broodstock hatchery and statewide goals (Carmona-Catot et al. 2011). There is no recent evidence (although no studies have been performed) 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 fins with distorted fin rays or missing and/or eroded fins. The trap was modified in 2012 in order to allow passage of adults, a significant stride toward restoring some level of natural reproduction in the population (P. Divine, CDFW, pers. comm. 2012). The CDFW stocked ca. 1,000 "half pound" fish in Pine Creek intermittently prior to 2006, ostensibly for the purpose of experimentally reducing brook trout abundance through predation (Dean and Chappell 2005). However, no studies were conducted to confirm that this practice had the desired effect. Subsequent sampling suggests that few of these fish persisted for long in the creek (Carmona-Catot et al. 2011).

Actual population size of trout in Eagle Lake has not been studied but it is presumably dependent on the stocking allotments every year. Creel censuses indicate that catch per hour from 1983 through 2007 ranged from 0.2 to 0.6, with a mean of 0.3, while average length of fish caught increased over the years (Carmona-Catot et al. 2011). The number of mature females captured at the trap while migrating and spawned by the CDFW ranged from ca. 600 to 1,700, although no estimates were made of size of the entire spawning run.

Genetic studies provide some insights into minimum population sizes in the lake. Carmona-Catot et al. (2011) found individuals in the lake population had an $F_{IS,}$ or inbreeding value, of 0.064, significantly higher than zero, although no genetic evidence of a bottleneck was detected. The effective population size (size of breeding population) was estimated at 1,125 fish, with a confidence interval from 151-∞, indicating in all years there was a fairly large population contributing to reproduction. Given the presumed small number of fish used to establish the original hatchery-based population, it is interesting that no genetic bottleneck was detected. The original bottleneck could have been masked by the number of generations that have passed since the bottleneck and/or efforts of the hatchery breeding program to maximize genetic diversity (by breeding as many individuals as possible), as seen in the population's now high effective population size. It is also possible that the population left in the lake in the 1950s was larger than trapping efforts on Pine Creek indicated and multiple years of naturally-spawned fish contributed to the initial hatchery stock. The slight, if significant, F_{IS} value is still something of concern and worth monitoring, although it is comparable to levels found in other lake-stream systems in the region such as Goose Lake (Simmons 2011).

Overall, the population appears to be stable because it is maintained by hatchery production, which may be selecting against fish capable of reproducing naturally. For example, Chilcote et al. (2011) show that wild populations of three species of anadromous salmonids from the Pacific Northwest have greatly reduced ability to remain self-sustaining when fish of hatchery origin are also present. There is ample evidence that hatchery rearing has an impact on the genetics and behavior of fish released into the wild, affecting their ability to persist (e.g., Waples 1999, Araki et al. 2007, 2008, Kostow 2008). Recent evidence suggests that fitness reductions may not just be limited to fish raised in the hatchery but, instead, continues into subsequent generations (Araki et al. 2009).

Nature and Degree of Threats: The greatest historical cause of the near-extinction of ELRT has been the degradation of the Pine Creek watershed and the establishment of brook trout in historic spawning and rearing areas. The watershed was severely altered as the combined result of logging, grazing, diversions, and railroad and road building among other threats (Carmona-Catot et al. 2011). These factors do not operate independently but, instead, must be viewed in aggregate, along with other less pressing threats (Table 1), as cumulative and synergistic watershed impacts.

	Rating	Explanation
Major dams	n/a	
Agriculture	Low	Bly Tunnel was built to divert water for agriculture but was fully closed in 2012
Grazing	Medium	This was a major historic cause of degradation to the watershed but recent actions have substantially reduced impacts from grazing
Rural residential	Low	Septic tank effluents and ground water removal may be an ongoing threat; many septic issues resolved with recent construction of waste water treatment plants; however, diversion of water to evaporation ponds may negatively affect lake levels
Urbanization	n/a	
Instream mining	n/a	
Mining	n/a	
Transportation	Medium	Culverts (now fixed) have been past barriers to migration but roads continue to affect Pine Creek and lake (sedimentation, etc.)
Logging	Medium	Major activity in watershed
Fire	Low	Has potential to negatively impact entire Eagle Lake basin, especially with risk of more frequent and severe fires
Estuary alteration	n/a	
Recreation	Low	Recreation is a major human use of the basin; impacts (other than the recreational fishery) to ELRT are unknown
Harvest	Medium	Major impact in past; trophy fishery drives management; current fishing regulations in place to manage harvest rates
Hatcheries	Medium	Almost all fish have been produced in hatcheries for 60+ years; however, ELRT hatchery operations currently focus on minimizing artificial selection processes; hatchery diseases a possible threat
Alien species	High	Brook trout dominance in Pine Creek watershed is a major barrier to restoration and establishment of self-sustaining wild ELRT population; alien diseases are a possible threat

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

Agriculture. In the past, Eagle Lake was viewed as a potential source of water for the otherwise arid agricultural region around Susanville and the Honey Lake Basin. This resulted in the construction of Bly Tunnel, which was completed in 1923, to send Eagle Lake water into

Willow Creek for use in crop irrigation. This project largely failed to deliver the water promised. During the 1930s, lake levels dropped as the result of diversion of water through the tunnel in combination with a severe, prolonged drought. Although it was blocked off with a concrete plug in 1986, the tunnel continued to passively leak, through an eight-inch bypass pipe in the plug, 0.034 cubic m/s (1.2 cubic ft/s) of Eagle Lake water into Willow Creek for downstream water right holders. Due to lack of surface flow diversion, some questions remained as to whether the water was coming directly from Eagle Lake or was, instead, percolating from groundwater into the tunnel. Water chemistry analysis revealed that most of the leakage was Eagle Lake water because of its unique chemical similarity to water sampled directly from Eagle Lake (Moyle et al. 1991). Based upon a position paper issued by the California Department of Fish and Wildlife to the State Water Resources Control Board in late 2011, the Bureau of Land Management, who administers the lands surrounding Bly Tunnel, closed the pipe in Feburary, 2012, thus eliminating direct discharge of Eagle Lake water via Bly Tunnel.

Grazing. Livestock grazing in the Eagle Lake basin started in the mid-1800s and was unregulated until 1905. Past grazing impacts to the Pine Creek watershed were substantial but are now greatly reduced because of improved grazing management (Pustejovsky 2007; Carmona-Catot et al. 2011). However, the legacy effects of past grazing continue, especially in the lower 40 km of Pine Creek, where the streambed has down cut and become enlarged in places, much of the riparian vegetation has been removed, and riparian meadows have presumably become drier, making them more likely to be invaded by sagebrush and similar xeric vegetation. Although stream flow records are lacking, it is likely that Pine Creek flows have also become more intermittent during summer, with spring flows decreasing more rapidly after snowmelt. At present, the lower creek (below Highway 44) usually stops flowing in late May or early June. The legacy effects of past grazing practices may have contributed to this altered hydrological regime; however, habitat conditions in recent years have been steadily improving (Pustejovsky 2007).

Rural residential. Eagle Lake has a number of residential tracts on its shores that depend on groundwater pumping (connected to lake levels) for water supplies. Although the potential connection between aquifer pumping and lake levels is poorly understood, the impacts may be substantial (especially during drought periods). Leakage of septic tank effluents into the lake is also a potential problem. This was resolved in 2007 at Spaulding Tract, with the development of a waste water treatment facility. Waste water is now diverted to evaporation ponds in Spalding Tract, Stones Landing, and South Shore campgrounds, which may result in significant loss of ground water in the basin, potentially exacerbating low lake levels during drought periods.

Transportation. Past road and railroad building to support historic and ongoing logging activities (see below) negatively affected habitat conditions and fish passage in Pine Creek. Culverts created barriers to upstream fish migration and road or railroad crossings created constriction points which may have altered stream hydrology. Wet road crossings contributed to stream bank erosion and sediment input. The more recent construction of State Highway 44, parallel to the railroad, forced Pine Creek through several culverts. The combination of culverts and channelized stream created a nearly-impassible velocity barrier for spawning ELRT. All potential barriers created by roads or other infrastructure have been removed or modified in lower Pine Creek. In spring, 2011, ELRT migration to the perennial sections of Pine Creek was verified through the use of PIT tags.

An additional concern is that part of the spring flow of Bogard Spring Creek is being diverted to provide water for a rest stop facility on Highway 44, reducing already minimal flow in this small, but important, tributary to Pine Creek.

Logging. Timber harvesting officially began in the Lassen National Forest in 1909, although the highest production took place in the 1970s and 1980s. The direct effects of timber harvest on stream habitats and flows may have been minimal because of the rapid infiltration capacity of the volcanic soils of the region, which reduces erosion rates (Platts and Jensen 1991). However, the roads constructed to facilitate logging were (and generally still are) very erosion-prone. Railroad lines were constructed across the Pine Creek drainage in the 1930s and 1940s to support logging activities, which restricted instream flows and led to channelized streambeds. Timber harvest is still very active in the area and the road networks utilized to support logging may serve as source inputs of sediments into streams.

Fire. Fires are common in the dry, heavily altered forests of the Eagle Lake watershed. The effects of fire on Pine Creek and its fishes have not been documented but the potential exists for severe damage to the upper watershed, with subsequent erosion, and perhaps direct mortality of fish in small streams. Historical photos (and surveys documenting stand densities and sizes) of the area show open stands of large conifers, with little understory or ladder fuels prior to fire suppression and logging in the basin (P. Divine, CDFW, pers. comm. 2012). Current forest conditions are quite different, with increased stand densities and widespread growth of firs which are not well adapted to fire and serve as ladder fuels (J. Weaver, CDFW, unpublished observations, 2012). This change in forest structure may increase the risk of high intensity, catastrophic fires, especially when coupled with predicted climate change outcomes, which may have dramatic impacts on riparian habitats and stream hydrodynamics in the Eagle Lake basin.

Recreation. The major use of Eagle Lake and its watershed is increasingly for recreation, much of which is focused on the widely popular recreational fishery ELRT support. The impacts from recreational angling, other than from harvest, which is closely regulated, are minimal. Other recreational impacts may include off-road vehicle use.

Harvest. As noted, in the 19th century, ELRT were once heavily exploited by a commercial fishery, which probably contributed to their initial decline. Since the 1950s, however, demand to support the lake sport fishery has been the principal reason its population has been maintained. However, a high percentage of the trout produced are planted in places other than Eagle Lake and the actual carrying capacity of the lake for rainbow trout is not known. It is possible that planting fewer fish would result in higher survival rates and more rapid growth rates. If a run becomes re-established in Pine Creek, the trout fishery in the creek will have to be managed in ways that do not negatively affect recruitment to the lake. In 2012 and 2013, the number of ELRT stocked into Eagle Lake was reduced by 20,000 to improve quality/condition of ELRT in the lake (P. Divine, CDFW, pers. comm. 2013).

Hatcheries. Eagle Lake rainbow trout are, at present, most likely completely dependent on hatchery production for survival (Moyle 2002). Prior to the1950s, they presumably persisted only because occasional wet years permitted access to upstream spawning areas through degraded stream channels and because ELRT were exceptionally long-lived for rainbow trout. A potentially negative outcome of hatchery reliance is that fish are being selected for survival in the early life history stages in a hatchery environment, rather than in the wild, perhaps for early spawning (as has happened in steelhead, Araki et al. 2007). In addition, fish may have been directly selected for large sizes for planting the lake (Carmona-Catot et al. 2011). However, sizes of angler-caught fish appear to be fairly static or slightly increasing over time (Figure 2).

Eggs taken from spawned fish at the Pine Creek Trap are sent to several hatcheries for rearing and then stocking into recreational waters. Crystal Lake Hatchery and Darrah Springs Hatchery rear fish to stock back into Eagle Lake. Darrah Springs also has a broodstock select program and rear these selected fish for 1.5 to 2 years. They are then transferred to Mt. Shasta Hatchery where they are used for production broodstock for statewide hatchery programs.

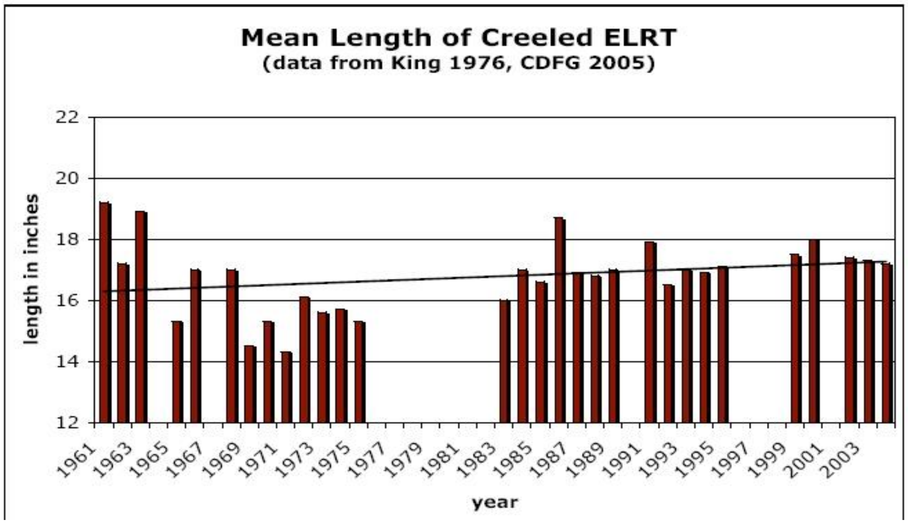


Figure 2. Mean lengths of Eagle Lake rainbow trout caught by anglers, 1961-2005 (from Pustejovsky 2007).

Genetic changes to ELRT have likely occurred as the result of continued hatchery selection, which may reduce the ability of trout planted in the lake to spawn naturally and produce young that can survive in streams or retain the predisposition to outmigrate back to the lake. Complete dependence on hatcheries for maintaining the species is undesirable because survival of the species then becomes dependent on vagaries of hatchery funding and management. Survival is further threatened by disease in hatcheries, loss of adaptation for life in the wild, loss of life history diversity, and potential inbreeding. Hatchery impacts may be particularly detrimental to a species with notable longevity (e.g., possibly eliminating the adaptation of ELRT toward a 10+ year life span, which has likely served as a buffer against extended periods of drought and periodic lack of access to spawning grounds). National Marine Fisheries Service guidelines indicate that a salmonid population dependent on hatchery production cannot be regarded as viable in the long-term (McElhany et al. 2000), a policy supported by recent studies (e.g., Chilocote et al. 2011).

The Pine Creek Coordinated Resource Management and Planning (CRMP) group (Pustejovsky 2007) has functioned over the past 25 years and is focused on restoration actions to provide for natural spawning of ELRT in Pine Creek. These efforts, if carried out completely, will result in a stream again capable of supporting a self-sustaining, wild population of ELRT. While hatchery production to sustain the trophy fishery has historically been regarded as a higher priority than re-establishment of a wild population (Dean and Chappell 2005), management shifts in recent years are increasingly focused on restoring a wild population, which is likely to happen only if brook trout are eliminated from Pine Creek so high production of ELRT juveniles can be assured (P. Divine, CDFW, pers. comm. 2012).

Another threat to the survival of ELRT is exotic disease, which could be introduced in hatcheries or into the lake by hatchery-reared fish, potentially severely affecting the lake's ELRT population (and possibly other fishes). However, hatchery protocols require routine examination of fish and water quality to reduce the threat of disease and ELRT are reared at two separate facilities to provide a redundant system, in the event that disease outbreak affects one or the other hatchery (P. Divine, CDFG, pers. comm. 2013).

Alien species. Many different species have been introduced into Eagle Lake in the past but none have persisted because of the lake's alkalinity. Nonetheless, because of Eagle Lake's large size and accessibility, it is possible that other species will be introduced illegally and, eventually, one may succeed, perhaps 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. The only alien species that persists in the drainage is brook trout, which is abundant in upper Pine Creek. Predation and competition by brook trout in Pine Creek may prevent reestablishment of ELRT, so a program to eliminate this species from the watershed is needed and is currently in the planning stages (J. Weaver, CDFW, pers. comm. 2013). The high densities and biomass of brook trout in upper Pine Creek indicates good capacity for rearing ELRT in large numbers in the absence of brook trout (Carmona-Catot et al 2010, 2011), with the potential for contributing wild fish back into the lake population.

Effects of Climate Change: Climate change is likely to have two major impacts on the Eagle Lake watershed: decreased stream flows and changing lake conditions. Reduced snowpack in the mountains surrounding the Pine Creek watershed will presumably reduce the output of springs that feed Pine Creek. The magnitude of this effect, however, will depend on the timing and amount of rain and snowfall and how well meadows are managed to increase their ability to retain water and release it during summer months. Reduced inflow into the lake could potentially increase alkalinities to lethal levels for trout although, if average precipitation remains roughly the same, the lake should maintain itself. Unfortunately, the lake is now (2013) at near-record low levels and has been so for several years, so changing water chemistry is an increasing concern. Surface temperatures of the lake could potentially increase 2-3°C but, presumably, a cold water refuge for trout will continue to exist in the deepest basin of the lake. If climate change produces extended droughts that dry Pine Creek early or for longer periods of time, resulting in increased lake alkalinity and temperatures, ELRT could be driven to extinction in its native range, relegating it to a hatchery fish. Fires, coupled with predicted climate change outcomes, may become more frequent and catastrophic, especially in the dry headwaters of the basin and may interfere with ongoing and planned restoration efforts in the Pine Creek watershed. For these reasons, Moyle et al. (2013) scored the species as "critically vulnerable" to climate change and threatened with extinction by 2100 without human intervention.

Status Determination Score = 2.1 - High Concern (see Methods section Table 2). While this score reflects improved understanding of ELRT genetics, the subspecies is likely to experience further genetic change and become a semi-domestic hatchery fish if actions to restore a naturally spawning population are not implemented. Genetic degradation may occur because continued hatchery selection is likely to select against the ability of ELRT to maintain a natural life history. Stochastic events such as elimination of hatchery or lake stocks through a disease epidemic, severe drought, illegal introductions of invasive species, parasites, or other factors put ELRT at high risk in its native habitat given that they are endemic to only one watershed. Remarkable progress has been made in restoring stream habitats and natural spawning in the past 5-10 years but continued restoration is needed, particularly regarding the elimination of brook trout from the Pine Creek watershed.

A petition for federal listing 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 California State Fish and Game Commission in 2004. In both cases, the reason given for not listing was insufficient information. However, the USFWS issued a 90-day finding in 2012 (Federal Register 77 (172) 54548-54553, September 5, 2012), indicating listing may be warranted and is currently performing a 12-month review to gather additional information and make a status determination. The ELRT is regarded as a Species of Special Concern by the California Department of Fish and Wildlife and as an R5 Sensitive Species by the U.S. Forest Service. The American Fisheries Society lists it as Threatened, while NatureServe lists it as "Critically Imperiled" (Jelks et al. 2008). Eagle Lake is a designated Heritage Trout Water (one which supports a fishery for native trout forms in their historic range), managed under CDFW's Heritage and Wild Trout Program.

Listing under either federal or state ESA, while potentially justifiable, is not desirable because so much progress is being made toward their conservation and management. Listing could inhibit the ability of agencies or local conservation groups to efficiently implement restoration tasks by increasing permitting delays or disallowing certain activities intended to benefit the species. Nevertheless, it is important to underscore the need to connect habitat restoration with re-establishment of a wild population, provide additional incentives to eradicate brook trout, and continue to address other stressors.

Metric	Score	Justification
Area occupied	1	Endemic to a single watershed
Estimated adult abundance	4	Includes hatchery fish
Intervention dependence	1	Persistence depends on trapping fish for hatchery spawning and rearing and restocking lake annually
Tolerance	4	One of most tolerant, long-lived forms of rainbow trout
Genetic risk	3	Although operated to maximize diversity and minimize artificial selection processes, hatchery rearing has presumably altered genetics; possible selection against longevity and fitness in the wild is of concern; accidental hybridization in hatcheries possible
Climate change	1	Reduced stream flows or increased alkalinity of lake could further impact population; lake already at very low levels
Anthropogenic threats	2	See Table 1
Average	2.3	16/7
Certainty (1-4)	4	Well documented

Table 2. Metrics for determining the status of Eagle Lake rainbow trout, where 1 is a major negative factor contributing to status, 5 is factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: The management of ELRT is an ideal opportunity to institute principles of adaptive management, where management actions are treated as experiments to inform future management (Carmona-Catot et al. 2011). The first step in the adaptive management process is to continue efforts to restore a wild, naturally-spawning population, rather than relying on maximizing egg 'take' for hatchery reproduction and maintenance of the recreational fishery. Substantial take of eggs to meet hatchery goals and targets can likely take place even if 10-20% of the adult fish are diverted for natural spawning and for experimental migration studies. A plan is currently being developed to guide management of the Pine Creek trap to allow for increased numbers of ELRT to migrate through the trap via the fish-way constructed in 2012 (P. Divine, CDFW, pers. comm. 2012). Additionally, CDFW, in collaboration with the CRMP, is currently (as of 2013) engaged in drafting a conservation strategy for ELRT, much of which will focus on restoration actions in the Pine Creek watershed, including a subcomponent addressing strategies to eradicate brook trout, along with options for enhancing spawning success and improving natural recruitment of ELRT in Pine Creek (J. Weaver, CDFW, pers. comm. 2013). These recent developments indicate that natural spawning and recruitment of wild stocks into the population have been identified as priorities for the recovery and management of ELRT.

As studies are developed and actions identified, three basic questions should be considered:

1: Can ELRT successfully migrate upstream from the lake in most years and successfully spawn?

2: Does re-establishment of a self-sustaining population of ELRT require complete eradication of brook trout from Pine Creek?

3: Can progeny from natural spawning return to the lake and contribute to the fishery?

Given that ELRT have undergone more than 60 years of artificial selection for reproduction and survival under hatchery conditions for a significant part of their life cycle, it is imperative to reverse that process as soon as possible. This underlying issue has long been recognized and was one of the justifications for the formation of the CRMP group in 1987, followed by many projects on Pine Creek to improve flow and remove passage barriers (Pustejovsky 2007). In order to implement adaptive management and begin the process of restoring natural spawning of ELRT, it is likely that a program of experimental release of adults above the Pine Creek weir and possible trapping and trucking of juveniles downstream past low-flow portions of the creek will be necessary. Recent research demonstrated that trapping and trucking may be a viable option for helping to recreate a naturally reproducing ELRT population; the study suggested that if spawners are allowed to migrate upstream naturally early in the season, they could successfully spawn and perhaps emigrate back to the lake (with trap and truck assistance as needed) following spawning (Carmona-Catot et al. 2010, 2011). The costs of this type of alternative management would presumably be comparable to costs of rearing hatchery fish but with fewer genetic consequences (e.g., Waples 1999, Araki et al. 2007, Kostow 2008).

Evidence exists that ELRT, at least during wet years, can migrate to the upper reaches of Pine Creek and spawn successfully. In the 1980s, a few juvenile rainbow trout were found below Stephens Meadow, suggesting adults made it over the weir, migrated upstream and successfully spawned (Moyle, unpublished data). In 1999-2005, biologists from CDFW, USFS and UC Davis placed radio transmitters in a small number of adult fish, which were then released above the weir (L. Thompson, UC Davis, pers. comm.). In 1999, one of these fish apparently made it to the Pine Creek headwaters, as its transmitter was recovered in Bogard Springs Creek, a tributary to Pine Creek above Highway 44 (T. Pustejovsky, pers. comm.). From 2002-2006, CDFW biologists released about 500 unspawned trout from the fish trap into Pine creek above Highway 44. In September, 2006, a crew from UC Davis, CDFW, and the USFS sampled Pine Creek to document the presences of ELRT (Carmona-Catot et al. 2011). They found evidence that ELRT had spawned successfully in the creek in the past two years because small numbers of juvenile rainbow trout were found at several locations in Pine Creek. About 100 m of Bogard Spring Creek were electrofished and 10 juvenile rainbow trout (76-90 mm FL) were captured, along with about 170 brook trout of varying sizes. Presumably, the rainbow trout were YOY or yearlings. The rainbow trout tended to be in faster water than brook trout, in reaches with deep overhanging cover. The UC Davis crew also found 3-4 small rainbows in Pine Creek, below the Bogard Spring Creek confluence, as well as a couple of rainbow trout in the 145 mm range in a creek filled with brook trout of all sizes, speckled dace, Lahontan redside, and Tahoe sucker. Curiously, several large trout from the lake that had been planted in the spring were still surviving in the pool below the culvert under Highway 44. Likewise, three spawners were found alive in a culvert about 5 km below the highway, in a largely dry section (no surface flow), along with a rainbow trout that was 142 mm SL. In 2007, at least 10 large ELRT (40-50 cm FL) were found downstream from the gauging station weir on Pine Creek (G. Carmona-Catot, pers. comm.). Successful spawning and migration was observed in 2010 and 2011, with juveniles reaching the trap and one tagged adult migrating from the weir to upper Pine Creek (T. Pustejovsky, pers. comm.).

From 2007-2012, Bogard Spring Creek was electrofished to remove brook trout to determine if spawning success of transplanted adult rainbow trout could be improved and to assess whether a three-pass electrofishing removal can successfully depress brook trout populations. In 2007, 4,887 brook trout were removed from the 2.5 km long creek (ca. 2,000 fish /km), which is remarkable considering the creek is less than 1 meter wide for all of its length and mostly less than 40 cm deep. During 2007, 170 juvenile ELRT were captured and returned to the creek; most fish were under 150mm FL, which indicates that they were not hatchery fish planted in the stream by CDFW at larger sizes (Carmona-Catot et al. 2010). Similar results were obtained in following years, along with evidence of a greatly diminished brook trout population. This evidence strongly indicates that a wild spawning population of ELRT can be reestablished, especially if brook trout populations are largely eliminated (Carmona-Catot et al. 2011).

As noted, major efforts have been undertaken in recent decades to fix passage problems and address habitat restoration needs in Pine Creek through the CRMP process (Pustejovsky 2007). As a result, sections of the creek have been fenced to exclude livestock, off-stream watering stations have been provided, an impassible culvert under Highway 44 has been replaced with a passable one, and a structure to divert water from Pine Creek near the Bogard Campground has been removed (and the meadow fenced). However, the meadows along lower Pine Creek and Bogard Spring Creek are still grazed by cattle, potentially affecting instream habitats and reducing the capacity for meadows to store and slowly release water into streams.

Elements of an adaptive management strategy for ELRT should include:

- Develop a management plan that is flexible enough to be adapted to changing conditions. A basic assumption of such a management plan should be that both hatchery-based and wild spawning populations will be maintained, as mutual insurance policies. As noted, CDFW, in collaboration with the CRMP, is currently (2013) drafting a conservation strategy for ELRT, which should provide the framework for future management. The

CDFW is also in the process of developing a genetics management plan for ELRT (P. Divine, CDFW, pers. comm. 2013), which should be incorporated into a broader conservation strategy.

- Continue efforts to ensure that restoration of a wild, naturally-spawning ELRT population remains a high priority.
- Develop an eradication strategy for brook trout in Pine Creek using either piscicides or other means (e.g., installation of artificial barriers and manual removal via electrofishing). If piscicides are proposed, a thorough investigation of the aquatic insect and herpetofauna of the watershed should be conducted in order to determine potential impacts of piscicides on their populations. Adaptive management and experimentation will be at the core of eradication efforts, particularly if piscicides are not employed, and successful removal of all brook trout from the Pine Creek drainage will likely be a costly, challenging and lengthy process. Nonetheless, CDFW recognizes the importance of this key step in the long-term conservation of ELRT and funding and resources are being allocated within the Department to enable focused, long-term, on-the-ground field work to benefit ELRT and other native trout forms across the state; installation of one or more barriers and experimental manual removal of brook trout in Bogard Springs Creek is slated to begin in October, 2013 (P. Divine and J. Weaver, CDFW, pers. comm. 2013).
- Finalize and implement plans to allow adult ELRT passage above the now modified Pine Creek trap as soon as spring snow-melt flows allow, in order to maximize potential for natural migration and spawning. Continue and expand upon existing instream movement monitoring studies (e.g., PIT tagging, radio telemetry) and incorporate assessments of passage improvement using these technologies, where applicable.
- Depending on water year type, develop plans to establish trapping and trucking operations for both adults (if natural migration of adults released above the weir does not occur) and out-migrating juveniles until there are signs the population is self-sustaining.
- Continue habitat improvements in the Pine Creek watershed with the goal of improving the quantity and duration of flow, following the recommendations in Pustejovsky (2007). Continue improvements in grazing practices and other activities that may affect stream habitat conditions.
- Increase flows in Bogard Spring Creek by eliminating the diversion that provides water to the rest station on Highway 44.
- Develop a comprehensive monitoring plan to assess habitat conditions, brook trout abundance, adult ELRT instream movement, spawning success, and juvenile ELRT abundance and outmigration success.
- Determine the feasibility of using Papoose Creek for establishment of a small spawning population.
- Conduct a thorough study of the survival and growth of trout planted in Eagle Lake to determine its actual carrying capacity for ELRT. Planting of trout in the lake (150,000+ per year) is based on maintaining catches of at least 0.4 fish per hour (Dean and Chappell 2005), rather than on biological constraints. It is possible that planting fewer trout may improve trophy angling.

Figure 3: Distribution of Eagle Lake rainbow trout, *Oncorhynchus mykiss aquilarum*, in California (native range only).

KERN RIVER RAINBOW TROUT
***Oncorhynchus mykiss gilberti* (Jordan)**

Status: Critical Concern. The Kern River rainbow trout has a high probability of disappearing as a distinct entity in the next 50-100 years, if not sooner. The greatest threat continues to be hybridization with coastal rainbow trout, but competition and predation from invasive brown trout and brook trout may also be contributing to its decline.

Description: This subspecies is similar to coastal rainbow trout but its coloration is brighter, with a slight tinge of gold; it has heavy, fine spotting over most of its body (Moyle 2002). The spots are more irregular in shape than those of the round spots of the other two Kern basin golden trouts. On many larger fish, there is a broad rosy-red band along the sides. There are also minor differences in meristics from the other two golden trouts (Schreck and Behnke 1971).

Taxonomic Relationships: The taxonomic status of this subspecies is controversial because of its complex evolutionary history and exposure to introduced varieties of rainbow trout. In 1894, D. S. Jordan designated this fish as a distinctive subspecies of rainbow trout; this analysis was accepted until Schreck and Behnke (1971) described it as a population of golden trout. Their decision was based mostly on comparisons of lateral scale counts and on aerial surveys that led them to believe that there were no effective barriers on the Kern River which might have served to isolate trout in the Kern River from those in the Little Kern River [in particular, barriers to downstream movement of golden trout into the Kern River, which also applies to Golden Trout Creek]. However, in a subsequent analysis, Gold and Gall (1975) determined that golden trout populations were effectively isolated genetically and physically. Meristic (Gold and Gall 1975) and genetic (Berg 1987) characteristics of *O. m. 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). The AFLP analysis indicated that Kern River rainbow trout represent a distinct lineage that is intermediate between coastal rainbow trout and Little Kern golden trout, although there was also some evidence of recent hybridization with coastal rainbows, presumably of hatchery origin. Erickson (2103) performed a detailed genetic analysis of upper Kern Basin trout in the historic range of Kern River rainbow trout, using single nucleotide polymorphism (“SNP”) and microsatellite markers to evaluate extent of introgression. He found that introgression with coastal rainbow trout, California golden trout, and Little Kern golden trout is widespread throughout the basin, although a distinct genetic signature of the Kern River rainbow trout could be detected in most populations, particularly in isolated tributaries. A number of tributary populations showed no or little introgression with other rainbow trout. A number of these populations, however, have limited genetic diversity and show signs of genetic bottlenecks (Erickson 2013).

Life History: No life history studies have been performed on this subspecies, but its life history is assumed to be similar to other rainbow trout populations in large rivers (e.g., Moyle 2002).

Historically, fish found in the mainstem Kern River grew to large sizes, as much as 71 cm TL and 3.6 kg (Behnke 2002), although fish over 25cm TL are rare today (S. Stephens et al. 1995).

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

Distribution: This subspecies is endemic to the Kern River and 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) downstream. Today, remnant populations live in the Kern River above Durrwood Creek, in Rattlesnake and Osa creeks, and, possibly, upper Peppermint Creek (S. Stephens et al. 1995). Bagley and Gall (1998), using a variety of genetic techniques, determined that several populations, mostly located in the middle section of the Kern River drainage, were relatively unhybridized Kern River rainbow trout: Rattlesnake Creek (in Sequoia National Park), Kern River at Kern Flat, Kern River above Rattlesnake Creek, Boreal Creek, Chagoopa Creek, Kern River at Upper Funston Meadow, Kern River above Redspur Creek, and Kern River at Junction Meadow. These populations are in the middle of the historic range and lack hybridization with either California golden trout (seen in the upper sections of the Kern) or with coastal rainbow trout (seen in the lower sections). While Behnke (2002) doubted that pure Kern River rainbow trout still exist in their native range, recent genetic analyses suggest that at least some unhybridized populations exist as indicated above. Much of their remaining habitat is in Sequoia National Forest (29+ km) and Sequoia National Park (40+ km). In addition, there are distinctive introduced populations in the Kern-Kaweah River and Chagoopa Creek which have maintained their genetic identity (M. Stephens 2007).

Trends in Abundance: Kern River rainbow trout were once abundant and widespread in the upper Kern Basin and grew to large sizes. As a result, they were subject to intensive removal by angling. Since the 19th century, overexploitation, combined with habitat degradation and, most importantly, hybridization with other trout, has reduced populations to a small fraction of historic numbers. In 1992, a study of Kern River rainbow trout abundance in the Kern River in Sequoia National Park indicated there were about 360-840 trout per km (600-1400 trout per mile) of all sizes (Stephens et al.1995). There are no data on current abundance but, if it is 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. These estimates are highly questionable given natural variation in numbers, smallness of sample sizes upon 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. Most of the least hybridized populations are isolated from other populations, as shown in recent genetic assays (Erickson 2013). Thus, the status of Kern River rainbow trout could deteriorate rapidly as populations disappear or become heavily hybridized.

Nature and Degree of Threats. Erickson (2013) found 7 populations that showed low or no hybridization (i.e. 75% or more of the fish sampled genetically were assigned to Kern River rainbow trout), scattered among creeks and lakes in the upper Kern Basin or nearby basins (from introductions). Another 14 populations showed a genetic signature of Kern River rainbow trout. The entirety of their habitat is on public land, including Sequoia National Forest and Sequoia National Park. The primary threats to remaining populations are identical to those facing other endemic trout of the southern Sierra, which center on interactions with non-native trout: (1) hybridization with hatchery rainbow trout, which are still planted in the upper Kern Basin, though not in Sequoia National Park, (2) hybridization with golden trout historically planted, that may continue 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, especially through angler-assisted introductions. In addition, habitat loss from the region's long history of grazing, logging and roads, as well as stochastic events such as floods, drought and fire can degrade habitats, negatively affecting already isolated populations and their persistence (Moyle 2002). For a full discussion of these regional stressors, see the California golden trout account in this report.

	Rating	Explanation
Major dams	Medium	Isabella Reservoir has fragmented its range and allowed for introduction of alien species
Agriculture	n/a	
Grazing	Medium	Pervasive in the area, although less severe than in the past
Rural residential	Low	Few residences; most of the subspecies range is within national forest or national park lands
Urbanization	n/a	
Instream mining	n/a	
Mining	n/a	
Transportation	Low	Trails and off-road vehicle routes can be a source of sediment influx into streams; however, most of range is in areas with minimal transportation impacts
Logging	Low	This is an important land use in the region but probably has little direct effect on local streams
Fire	Low	Despite fire suppression, fish-killing fires are unlikely given the sparse plant communities in the Kern Basin; fires generally allowed to burn in national parks with unknown impacts to fish populations
Estuary alteration	n/a	
Recreation	Medium	Off road vehicles a potential threat, but more so in past
Harvest	Medium	Heavily harvested in past; present harvest, legal and illegal, may affect some populations
Hatcheries	High	Constant threats of introgression, competition and predation from hatchery fish
Alien species	Critical	Non-native trout are the major cause of limited distribution via hybridization, competition, predation and possible disease transfer

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

Effects of Climate Change: The major predicted impacts from climate change in the range of the Kern River rainbow trout are a reduction in snow pack due to warmer temperatures, as well as a seasonal shift in peak runoff. However, the southern Sierra Nevada is the highest part of the mountain range and this may offset substantial reductions in snowpack, as is predicted in the northern Sierra Nevada and other regions of the state. Thus, snowmelt is likely to maintain flows in Kern River rainbow trout streams. Nevertheless, more precipitation may come as rain, potentially earlier in the season, which may lead to increased 'rain on snow events' and corresponding flash flooding. This may be particularly acute in the Kern River, which drains a large geographic area and may suffer substantial habitat alteration or degradation associated with flood events. Since snowpack is predicted to melt earlier in the season, meadows and forests surrounding Kern River rainbow habitats are likely to become drier by the end of summer, with reduced flows in streams. Elimination of grazing and other activities that compact meadows (reducing their ability to store water) and reduce riparian cover and shade may mitigate, in part, for the predicted effects of climate change. Temperatures in streams are likely to increase and it is possible that spawning times may occur earlier, with unknown consequences. For these reasons, Moyle et al. (2013) list wild populations of Kern River rainbow trout as "critically vulnerable" to extinction via climate change, assuming the small, isolated, first and second order streams that support most populations would be subject to increased frequency and extent of drying and warmer temperatures. Kern River rainbow trout occupying the main stem Kern may be less subject to threats of habitat loss due to drying but may be negatively affected by flood-based habitat degradation, warmer water temperatures, lower flows, and other factors.

Status Determination Score = 1.7 – Critical Concern (see Methods section Table 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 (Table 2). It is listed as a Special Concern (formerly Category 2) species by the USFWS, indicating that it is a candidate for listing as threatened but that there is inadequate information to make the determination. The American Fisheries Society considers it to be Threatened (Jelks et al. 2008), while NatureServe considers it as Critically Imperiled.

Kern River rainbow trout are confined to a handful of streams that are subject, independently and collectively, to natural and human-caused disturbance, such as landslides and fire, even through most are in protected areas, including Sequoia National Park. The greatest single threat continues to be invasions of alien rainbow trout, brown trout, and brook trout into their remaining streams, either through natural invasions, stocking programs, or through angler-assisted introductions. Protection of remaining populations, therefore, requires constant vigilance and the ability to react quickly to counter new threats.

Metric	Score	Justification
Area occupied	1	Found only in 4-6 small tributaries and short reaches of the Kern River
Estimated adult abundance	2	Much uncertainty about size of populations
Intervention dependence	2	Barriers must be maintained, planting of hatchery fish managed (preferably eliminated), grazing managed, and other ongoing protective activities
Tolerance	3	Presumably fairly tolerant, as are most rainbow trout, but not tested
Genetic risk	1	Hybridization with introduced rainbow trout a constant high risk to its distinctiveness
Climate change	2	Potential for large flood events and associated habitat alteration, as well as drying of small streams
Anthropogenic threats	1	Continued stocking of hatchery rainbow trout in Kern River is an ongoing threat, along with other stressors (see Table 1)
Average	1.7	12/7
Certainty (1-4)	3	This is least studied of the three native trout taxa found in the Kern River watershed

Table 2. Metrics for determining the status of Kern River rainbow trout in California, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: A multi-agency management plan for the upper Kern River basin, written in 1995, has as its goal to "restore, protect, and enhance the native Kern River rainbow trout populations so that threatened or endangered listing does not become necessary" (S. Stephens et al. 1995, p 9). While this plan has been implemented, almost 20 years later the trout may still merit listing. Problems addressed in the plan still exist, including stocking of non-native trout (including hatchery rainbow trout), grazing in riparian areas, and heavy recreational use of the basin, including angling. Future management actions should be based upon recommendations in this plan and updates to address developments in the past two decades should be performed (especially data and other gap analyses). Abundance and distribution data are much needed in order to better assess the current status of the Kern River rainbow trout and establish a baseline from which to monitor trends over time.

The Edison Trust Fund is supposed to provide at least $200,000 each year to implement the management plan and improve fisheries in the upper Kern Basin, including developing a conservation hatchery for Kern River rainbow trout, increasing patrols of wardens in areas where recreational angling occurs, and for funding studies on genetics. However, the recent financial crisis in the United States has reduced the availability of funds from the Trust.

Figure 1. Distribution of Kern River rainbow trout, *Oncorhynchus mykiss gilberti* (Jordan), in California.

CALIFORNIA GOLDEN TROUT
***Oncorhynchus mykiss aguabonita* (Jordan)**

Status: High Concern. While the Golden Trout Creek (GTC) population is relatively secure, the South Fork Kern River (SFKR) population is threatened by introgression with rainbow trout and predation and competition from introduced brown trout (*Salmo trutta*).

Description: The California golden trout is named for its bright colors. Behnke (2002) describes their coloration as follows: "The color of the back is brassy or copper, becoming bright golden yellow just above the lateral line. A deep red stripe runs along the lateral line and the golden yellow body color intensifies below. A deep crimson color suffuses the ventral region from the anal fin to beneath the lower jaw… (p. 105)." Fish from GTC are particularly brightly colored. Young and most adults have about 10 parr marks centered along the lateral line. The parr marks on adults are considered to be a distinctive characteristic (Needham and Gard 1959), but they are not always present, especially in larger fish from introduced lake populations. Large spots are present, mostly on the dorsal and caudal fins and on the caudal peduncle. The pectoral, pelvic, and anal fins are orange to yellow. The anal, dorsal, and pelvic fins have white to yellow tips, preceded by a black band. Basibranchial teeth are absent and there are 17-21 gill rakers. Other characteristics include 175-210 scales along the lateral line, 34-45 scales above the lateral line, 8-10 pelvic rays, 25-40 pyloric caeca, and 58-61 vertebrae (Schreck and Behnke 1971).

Taxonomic Relationships: The complex history of golden trout taxonomy and nomenclature is reported in Behnke (2002) and is presented here in a simplified version. Originally, three species of golden trout were described from the upper Kern River basin: *Salmo aguabonita* from the SFKR, *S. whitei* from the Little Kern River, and *S. roosevelti* from GTC. However, the first two forms were eventually recognized as subspecies of *S. aguabonita*: *S. a. aguabonita* and *S. a. whitei*. *S. roosevelti* was shown to be a color variant of *S. a. aguabonita* (Moyle 2002). Berg (1987) concluded that the two recognized subspecies of golden trout are more closely related to the Kern River rainbow trout (*O. m. gilberti*) than either are to each other. However, Bagley and Gall (1998) and M. Stephens (2007), using improved genetic techniques, found that California golden trout and Little Kern golden trout represent two independent lineages derived from coastal rainbow trout. *O. m. aguabonita* is referred to in some lists as South Fork Kern golden trout or as Volcano Creek golden trout but California golden trout seems more appropriate, given its status as the official state freshwater fish of California.

Life History: California golden trout live in cold, clear alpine streams. They have comparatively slow growth rates due to the truncated growing season and low productivity of high elevation streams in their native range (Knapp and Dudley 1990, Knapp and Matthews 1996). In streams, they are usually 3-4 cm SL at the end of their first summer of life, 7-8 cm SL at the end of their second summer, 10-11 cm SL at the end of their third summer, and grow 1-2 cm per year thereafter; they reach a maximum size of 19-20 cm SL and a maximum age of 9 years (Knapp and Dudley 1990). In alpine lakes, individuals from introduced populations grow to 4-5 cm FL, 10-15 cm FL, 13-23 cm FL, and 21-28 cm FL at the end of their first through fourth years, respectively (Curtis 1934); they can reach 35-43 cm FL by the seventh year. The largest on record from California weighed 4.5 kg, from Virginia Lake, Madera County, in 1952. However, most records of golden trout growth in lakes are suspect because populations were established from introductions that may have been hybridized with rainbow trout.

Golden trout spawn when they are three or four years old, when water temperatures exceed 10°C, with daily maximums of 16-18°C in late June and July (Stefferud 1993; Knapp and Vredenburg 1996). Average daily temperatures for spawning are around 7-10°C and spawning occurs in gravel riffles in streams. Spawning behavior is typical of other members of the rainbow trout group, although they spawn successfully in finer substrates (decomposed granite) more than most other trout (Knapp and Vredenburg 1996). Females produce 300-2,300 eggs, depending on body size (Curtis 1934). Embryos hatch within 20 days at an incubation temperature of 14°C. Fry emerge from the gravel two to three weeks after hatching, at which time they are about 25 mm TL. In introduced lake populations, fry move into lakes from spawning streams when they are about 45 mm TL.

In streams, golden trout are active at all times of day and night but tend to stay in the same areas for long periods of time (Matthews 1996a). They feed on both terrestrial and aquatic invertebrates, mostly adult and larval insects, taking whatever is most abundant. In lakes, they feed mainly on benthic invertebrates, especially midge pupae (Chironomidae) (T. Armstrong, UC Davis, unpublished data). Although bright coloration makes them highly visible, there are very few natural predators in their range (Moyle 2002). Their tendency to be more active during the day than most trout also suggests low predation. Thus, their bright coloration may have evolved for reproductive advantage. However, bright coloration has also been implicated as providing camouflage against the bright colors of the volcanic substrates in the clear, shallow streams within their range (Needham and Gard 1959). When these trout are removed from mountainous streams and brought down to low elevation streams, they may lose their brightness and take on dull gray and red colors (Needham and Gard 1959). In lakes, they become paler in color, often appearing silvery.

Habitat Requirements: Golden trout evolved in streams of the southern Sierra Nevada, at elevations above 2,300 m. The valleys of the Kern Plateau are broad, flat, and filled with glacial alluvium, which results in wide meadows through which streams meander. These streams are small, shallow, and have only limited riparian vegetation along the edges. The exposed nature of the streams California golden trout inhabit is largely the result of heavy grazing of livestock on a fragile landscape, which began in the 1860s. Grazing causes compaction of soils, collapse of stream banks, and elimination of riparian plant cover (Odion et al. 1988, Knapp and Matthews 1996, Matthews 1996b). Stream bottoms are mostly volcanic sand and gravel, with some cobble. The water is clear and mostly cold, although summer temperatures can fluctuate from 3 to 20°C (Knapp and Dudley 1990). California golden trout generally prefer pool habitat and congregate near emergent sedges and undercut banks (Matthews 1996a).

Environmental tolerances are presumably similar to those of coastal rainbow trout.

Distribution: California golden trout are endemic to the SFKR, which flows into Isabella Reservoir, and to GTC (including its tributary, Volcano Creek), which flows into the Kern River (Berg 1987). Initially (1909 and earlier), California golden trout were collected from GTC and transported north by pack train, extending their range by some160 km by 1914 (Fisk 1969). They were also translocated into many other waters within and outside California, including Cottonwood Lakes, not far from the headwaters of GTC, and headwaters of the SFKR, such as Mulkey Creek (Stephens et al. 2004). Cottonwood Lakes served as a source of golden trout eggs for stocking other waters beginning in 1917 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 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. However, some unhybridized populations apparently still exist from early transplants in the Sierra Nevada and elsewhere but they appear to have limited genetic diversity due to small numbers used to establish these populations (Stephens and May 2011).

Trends in Abundance: California golden trout populations suffered major declines during the 19th and first half of the 20th Century from overfishing and heavy grazing. Invading brown trout displaced California golden trout, including hybrids, from all reaches below artificial barriers, so golden trout are now confined to a few kilometers of stream in the GTC watershed and in the South Fork Kern watershed. Within their native range, California golden trout occur at both low densities (0.02 - 0.17 fish per m^2 in streams) (Knapp and Dudley 1990) and at high densities (1.3-2.7 fish per m^2). Low densities are most likely to be in found in grazed reaches of stream with little cover and food, with some exceptions (see next paragraph). Presumably, densities were much higher, on average, before livestock began grazing the drainage. Although California golden trout were widely introduced outside their native range during the 19th and 20th century, the introduced 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) estimated that golden trout streams typically support 8-52 fish/ 100 m of stream, although a recent estimate for Mulkey Creek, a tributary to the SFKR which supports an introduced population, was 472 fish/100m (Carmona-Catot and Weaver 2006). If the Knapp and Dudley figures are accepted as correct then, in 1965, when the first major CDFW habitat management plan was issued (CDFG 1965), there would have been 2400-15,600 individuals in GTC (30 km) and 4000-26,000 in the South Fork Kern (50 km). Curiously, the high numbers in the SFKR are found in reaches that have been degraded by grazing, presumably because the reaches contain decomposed granite substrates that are used for spawning (S. Stephens, pers. comm. 2008). The lack of cover in these reaches selects for smaller fish, which are more numerous (but which may have lower fecundity due to small body size and reduced egg production).

At present, if unhybridized fish exist only in 5 km of Volcano Creek, then there are only 400-2600 'pure' golden trout left in their native range, a decrease of at least 95%

from historic numbers. The percentage of these fish that are reproductive every year is not known but likely to be small. A caveat on this very rough calculation is that it is based on genetic studies (Stephens et al. 2004) that show many fish that are counted as hybrids have a very low incidence of 'foreign' genes; thus it may not be necessary to eliminate all rainbow trout genes from introgressed populations through eradication, if there is no impact on phenotypes. If golden trout populations with phenotypes that show low introgression of rainbow trout genes are considered to have conservation value, then the numbers of golden trout would be considerably higher and might include fish both within and outside their native range as well. For example, the introduced population in Mulkey Creek may be as large as 40,000 fish (>75 mm FL) in roughly 10 km of habitat, with very low levels of introgression (2%; Stephens 2007). Nevertheless, because golden trout had already been eliminated through hybridization and predation from most of the lower SFKR by 1965, where populations would have been most dense, the 95 percent decline figure for the native range may still be valid, even if populations with low introgression are counted.

As noted, California golden trout in the upper SFKR and GTC are introgressed with non-native rainbow trout. However, the levels of introgression are markedly different in these two streams. In the SFKR, there is a cline of introgression from the lower Kennedy Meadows area (94%) upstream to the headwaters (2%). Nearly all SFKR trout are introgressed with rainbow trout to some degree. Kennedy Meadows also contains dense populations of brown trout. In many reaches of GTC, 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). Nevertheless, genetically 'pure' populations exist in only a few kilometers of streams and this is likely to continue for the short term (<5 yrs).

Overall, unhybridized California golden trout are much less abundant than they have been in the past in their native range. In areas where they still persist, numbers may be higher than they were in the days of heavy harvest and grazing, but these numbers are still presumably less than historic highs (pre-1800s) because of the continued presence of hybridized fish, grazing, and other human impacts.

Nature and Degree of Threats: The principal threats to California golden trout are grazing and, most importantly, interactions with alien trout species.

Grazing. Livestock grazing is permitted in designated Wilderness Areas, such as the Golden Trout Wilderness Area; grazing occurs around GTC and the SFKR where California golden trout reside. According to the USFWS (October 11, 2011, 76 FR 63094), about 95 percent of areas around golden trout streams have been grazed by livestock for 130 years. Not surprisingly, some sections of stream and entire meadows have been severely damaged by grazing. 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). Grazing impacts to instream and riparian habitats include: reducing the amount of streamside vegetation, collapsing banks, making streams wider and shallower, reducing bank undercutting, polluting waters with feces and urine, increasing temperatures, silting up spawning beds (smothering embryos), and generally making habitats less complex and suitable for trout. These impacts may result in declines in trout populations.

Levels of cattle grazing have been reduced in recent years and the USFS has adopted guidelines to allow heavily grazed areas to recover (USFWS October 11, 2011, 76 FR 63094). Two of the four grazing allotments on the Kern Plateau have been rested since 2001 (S. Stephens et al. 2004). Future management of grazing for the four allotments is being considered by the USFS with a decision concerning grazing yet to be determined. Herbst et al. (2012) show that eliminating grazing in meadows results in improved streambank structure and macroinvertebrate abundance, more so than does fencing of short sections of stream. Such improvements are likely to be reflected in larger, more robust golden trout populations. Thus, this decision and the enforcement of improved grazing practices will have major impacts, positively or negatively, on the health of golden trout populations in their native range.

Recreation. Although California golden trout waters are entirely within Sequoia and Inyo National Forests and largely within the Golden Trout Wilderness, they are still impacted by human activities, including off-road vehicles (in the lower portions of the SFKR) and recreational damage by hikers, horse riders and pack stock. A particular threat is off-road vehicle use in the vicinity of Monache Meadows and the severe degradation of the lower SFKR due to multiple causes throughout that area.

Harvest and hatcheries. Recreational fishing within the Golden Trout Wilderness is allowed from the last Saturday in April through November 15, is restricted to artificial lures with barbless hooks, and a five fish daily bag and possession limit is allowed. Harvest rates are unknown, but are presumably low due to the remote nature of most golden trout-bearing streams, along with shifts in angler preference toward catch-and-release fishing, particularly for native or unique forms of trout with limited distribution. Golden trout, usually partially hybridized, are still raised in hatcheries for the purpose of supporting recreational fisheries, but these fish are not planted within the native range.

Alien species. The major threats from alien species are hybridization with rainbow trout and competition and predation from brown and rainbow trout. There is a long history of planting rainbow trout in the upper Kern River basin to improve recreational angling. The peak of stocking was probably 1931-1941, when 85,000-100,000 rainbows were planted every year (Gold and Gold 1976). Stocking of hatchery rainbows in the SFKR at Kennedy Meadow occurred in the past but was ceased in 2008 (B. Beal, CDFW, pers. comm. 2012). This portion of the SFKR also supports a fishery for wild brown trout. In addition, golden trout were introduced in Cottonwood Lakes in 1891, with a subsequent egg-taking station established by 1918; this population, the source of most golden trout transplants to other watersheds, was apparently contaminated with rainbow trout fairly early in its history.

In the SFKR, brown trout were eliminated from headwaters in the early 1980s and Ramshaw, Templeton and Schaeffer barriers were constructed to prevent their reinvasion. Even so, brown trout still dominate about 780 km of stream in the basin (Stephens et al. 2004). Unfortunately, rainbow trout were able to move upstream over the deteriorated Schaeffer Fish Barrier 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 because the original barriers have been replaced with better ones. This combination of events has resulted in rainbow trout or rainbow trout-golden trout hybrids invading most streams in the native range of California golden trout in the SFKR and hybridizing with them (Cordes et al. 2006). In

GTC, hybridization affects only a small percentage (about 5%) of the trout. The populations in Volcano Creek and some smaller tributaries have escaped this problem but may have relatively low genetic diversity. In the SFKR basin only a few headwater populations may have escaped hybridization (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 or the golden trout originated from hybridized stocks (Cottonwood Lakes). Hybridization with rainbow trout results in fish that are likely to be less brightly-colored than the native golden trout. The rainbow trout phenotype eventually becomes dominant, so the fish look more like rainbow trout. This has been well demonstrated in the lower SFKR, where hatchery rainbow trout had been planted annually from the 1930s until the late 2000s and the few wild golden trout left are heavily hybridized, having a rainbow trout appearance. After 2004, only sterile triploid rainbow trout were stocked in the lower SFKR with stocking entirely discontinued in 2008. Hybridization can ultimately result not only in the loss of the uniquely colored variety of trout but 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 retain characteristic golden trout coloration, a high degree of genetic fitness, and adaptivity to their habitats.

In addition to threats from rainbow trout, predation and competition from introduced brown trout are a continuous threat. In 1993, CDFW biologists found a reproducing population of brown trout above the lowermost barrier (Schaeffer) and a population was also found in Strawberry Creek in 2003 (S. Stephens et al. 2004). How they arrived there is not known, but it would have been relatively easy for anglers to move fish over the barrier. By the early 1990s, both Templeton and Schaeffer fish barriers had deteriorated and the Schaeffer Barrier allowed upstream fish passage. Both barriers were replaced with substantial concrete structures in 1996 and 2003, respectively. In these reaches, golden-type trout (goldens of varying degrees of hybridization) coexist with both brown trout and native Sacramento sucker (Carmona-Catot and Weaver 2006), although the long-term viability of this assemblage is not known. While barriers that prevent fish from migrating upstream can eliminate or reduce gene flow among golden trout, they may be the only solution to preventing additional upstream movement of alien trout. An additional barrier is possible near Dutch John Flat, upstream of Kennedy Meadows, to create an additional isolated area (B. Beal, CDFW, pers. comm. 2012).

	Rating	Explanation
Major dams	n/a	All major dams are outside the native range of California golden trout
Agriculture	n/a	
Grazing	Medium	Ongoing threat but greatly reduced from the past
Rural residential	n/a	
Urbanization	n/a	
Instream mining	n/a	
Mining	n/a	Historic mines are present but have no known impacts
Transportation	Low	Trails and off-road vehicle routes can be a source of sediment and pollution input into streams; direct habitat impacts from wet route crossings
Logging	Low	This is an important land use in the broader region but probably has no direct effect on golden trout streams
Fire	Low	Because of fire suppression, headwater areas could be impacted by hot fires, although this is unlikely given sparse plant communities in region
Estuary alteration	n/a	
Recreation	Low	Pure populations within the GTC watershed are entirely within designated wilderness; South Fork populations with conservation value are also within designated wilderness
Harvest	Low	Potential impact but light pressure and most fishing is catch and release
Hatcheries	Low	Residual effects of hybridization with hatchery fish
Alien species	High	Major cause of limited distribution in South Fork Kern; however, very limited introgression with rainbow trout and no brown trout in waters within GTC watershed

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

Effects of Climate Change: The major predicted impacts of climate change in the Sierra Nevada are reduction in snow pack, increased likelihood of rain-on-snow events, and shifts in peak runoff from late spring/early summer months to late winter/early spring months due to warmer temperatures. This will have the least effect in the southern Sierra

Nevada because the mountain elevations are highest there and may continue to retain a great deal of snow. Thus, snow melt is likely to maintain flows in golden trout streams. Nevertheless, snow pack may not persist as long in the extensive meadows of the Kern Plateau and meadows are likely to become drier by the end of summer, with reduced base flows in streams. Elimination of grazing and other activities that compact meadows (reducing their ability to store water) and reduce riparian cover and shade can mitigate, in part, for the effects of climate change. Temperatures are likely to increase earlier in the season in golden trout streams and it is possible that spawning times may become earlier, with unknown consequences. Moyle et al. (2013) rated California golden trout as "critically vulnerable" to climate change, indicating that extirpation from its native range is likely by 2100 if present trends continue.

Status Determination Score = 2.1 – High Concern (see Methods section, Table 2).

The California golden trout is listed as a Species of Concern by the USFWS and as a Sensitive Species by the USDA Forest Service. The American Fisheries Society lists it as Threatened, while NatureServe lists it as "Critically Imperiled" (Jelks et al. 2008).

A petition to the USFWS to list California golden trout as federally endangered was submitted by Trout Unlimited in 2000 (Behnke 2002). The USFWS determined in a 90-day finding that the proposal deserved additional consideration. After a 10 year review, the USFWS concluded (October 11, 2011, 76 FR 63094) that listing was not warranted because of all the collaborative efforts taking place to protect the trout, particularly the ongoing and active implementation of the Conservation Assessment and Strategy for the California Golden Trout (1994). This cooperative conservation agreement, signed by state and federal agencies and concerned NGOs, indicated that listing the fish would provide few, if any, additional benefits to it. According the Federal Record (76 FR 63094): "The purposes of the Conservation Strategy are to: (1) Protect and restore California golden trout genetic integrity and distribution within its native range; (2) Improve riparian and instream habitat for the restoration of California golden trout populations; and (3) Expand educational efforts regarding California golden trout restoration and protection." Until recently, the California golden trout was perceived as secure because it had been widely introduced throughout the Sierra Nevada and the Rocky Mountains. However, these introduced populations are likely on a different evolutionary trajectory from the native populations (most are in lakes) and they have also largely hybridized with rainbow trout. Nonetheless, Stephens and May (2011) show a number of populations do exist outside the native range that are unhybridized or only slightly introgressed. As Stephens and May (2011) point out:

> "…it is possible that these populations could be preserved *in situ* as an insurance policy against the loss of CAGT [California golden trout] within their native range or possibly utilized in other conservation or restoration efforts. Any introduction of these fish into the native CAGT range should be considered with caution: 1) future genetic analysis may reveal introgression previously undetected, 2) they do not appear to contribute any unique allelic diversity not already represented in the extant native range populations, and 3) they may have experienced substantially different selection regimes in their watersheds, possibly rendering them less (or more) fit than extant CAGT (p. 12)."

Meanwhile, even slightly hybridized populations in the native range can only be maintained through constant intervention such as building and repairing of barriers and eradication of non-native trout and golden-rainbow hybrids (Behnke 2002).

Metric	Score	Justification
Area occupied	1	“Pure” California golden trout are confined to a few small tributaries in one watershed
Estimated adult abundance	3	Volcano Creek populations may be <1,000 but, if other populations with conservation value within native range are counted, the numbers would be much higher, perhaps 50,000
Intervention dependence	3	Annual monitoring of barrier performance required; continued implementation of Conservation Strategy is critical
Tolerance	3	Generally tolerant of a wide range of conditions and habitats within their native range
Genetic risk	1	Hybridization with rainbow trout is a constant high risk
Climate change	2	Smaller streams may be negatively impacted by changing climate; improved watershed management may offset some impacts
Anthropogenic threats	2	See Table 1
Average	2.1	15/7
Certainty (1-4)	4	Well documented

Table 2. Metrics for determining the status of California golden trout, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: The overarching goal of California golden trout management should focus on the maintenance of self-sustaining populations in refuges that can persist through long periods of less intensive management and/or extended drought. Populations in their native range have persisted because of continuous, cooperative actions by the California Department of Fish and Wildlife, US Fish and Wildlife Service, and US Forest Service, along with volunteers from multiple groups. Ever since it was realized in 1968 that California golden trout in the SFKR were threatened by alien trout, mainly brown trout, major efforts have been undertaken to create refuges for golden trout in the upper reaches of the SFKR by constructing three barriers (Ramshaw, Templeton, Schaeffer) and then applying rotenone and antimycin to eradicate all unwanted fish above or between barriers. From 1969 through 2000, 10 treatments were carried out, with varying degrees of success (Stephens et al. 2004). In addition, gill netting of selected headwater lakes (e.g. Chicken Spring Lake, Rocky Basin lakes) to remove hybridized fish has been successful and these lakes are now fishless. The future focus of conservation should be protection of the original gene pools of golden trout in GTC and SFKR as: (1) a source for future fish transplants into restored streams, (2) stocks that can be genetically compared with introduced populations, and (3) an aesthetic measure. However, special protection should also be provided to demonstrably unhybridized populations outside the native range, as an insurance policy against the potential for complete loss of unhybridized fish from within the native range.

A major impediment to the protection and restoration of California golden trout is funding and staff shortages within management agencies. Implementation of the Conservation Strategy for California golden trout should reduce the threat of extinction through management of hybrids, maintenance of multiple barriers (redundancy in case one fails), improved management of watersheds, and elimination of non-native trout populations (S. Stephens et al. 2004). This strategy continues to be implemented and several key goals of this document have been met. These include the replacement of two failing fish barriers and increased genetic research to better understand the overall status of California golden trout. An additional barrier in the lower portions of the South Fork Kern drainage is being explored. Two of the four grazing allotments have been rested since 2001. Additional management actions needed include: (1) repair or replacement of barriers, (2) eradication of all rainbow trout and brown trout populations that threaten California golden trout, (3) utilization of recent genetics techniques to refine management, (4) improved management of livestock grazing, (5) modified recreation management strategies, and (6) expanded efforts to further implement the Conservation Strategy.

Barrier improvement. Barriers to prevent alien trout from invading golden trout waters are important, if ultimately short-term, management measures. Templeton and Schaeffer barriers were replaced with major concrete structures in 1996 and 2003 respectively, and have reduced the probability of unwanted invasions. However, because accessible barriers that have golden trout on one side and brown trout on the other are inherently flawed (by the ease of moving fish over the barrier), other solutions 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 at Dutch John Flat is in the early planning stages about 10 km upstream of Kennedy Meadows. Whether such a structure will ever be built in designated wilderness remains uncertain (S. Stephens, pers. comm. 2008).

Eradication of aliens. Eradication of non-native trout continues to be a necessary and important measure. Unfortunately, such eradication generally requires the use of the controversial piscicide, rotenone. Alternate toxins (e.g., antimycin) have yet to be approved in California so are unavailable for use. Given the controversial nature of the use of toxins, albeit natural ones, a thorough risk analysis should be conducted for streams in which their use is proposed. The analysis should include 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 occurring and necessary in order to allow for genetics-based management. A genetics management plan (GMP) for California golden trout was completed in 2013 (M. Stephens, UCD, pers. comm. 2013). The best management approach in the GTC watershed (now that introgressed trout have been removed from headwater lakes) is to monitor populations at intervals of five years or more to assess estimates of introgression from SNPs and microsatellite analyses. Establishment of refuge populations elsewhere for fish with high genetic integrity from the GTC drainage should be considered.

There is a cline of hybridization in the SFKR with levels of introgression with non-native rainbow trout increasing downstream (Stephens 2007). It appears the golden trout in GTC and the SFKR are slightly different genetically (Stephens 2007) and they will continue to be regarded as separate management units as recommended in the GMP. Plans to install a new fish barrier at Dutch John Flat should be pursued. Using the guidance of the GMP and the Conservation Strategy managers should develop appropriate plans and take steps needed to eradicate brown trout and hybrid golden trout considering the system of SFKR barriers. These activities may take years to accomplish but offer large rewards for golden trout in terms of greatly expanded range and protection from hybridization, competition and predation.

Grazing. Improvements have been made in livestock grazing management in the Golden Trout Wilderness Area in recent decades but further refinement and restrictions may be necessary to protect golden trout populations and their habitats. Continued resting of grazing allotments (or elimination of allotments altogether) should result in recovery of riparian vegetation and associated shading, improved stream channel morphology, and increased abundance of invertebrate food supplies for fish (Herbst et al. 2012). According to the USFWS (2011, Federal Register 76 FR 63094), changes in grazing management practices for the past 10 years or so, including resting allotments, have removed grazing as a primary threat to golden trout but the practice may still cause degradation of streams. If complete elimination of grazing is infeasible, then intense management of grazing to reduce impacts on streams should be continued and expanded, including the use of allotment rotation, seasonal closures during periods when meadows are wet, herd size reduction, expanded fencing, and active herd management to keep cattle away from streams. Monitoring of grazing practices needs to continue in order to document compliance with appropriate USDA Forest Service guidelines.

Recreation management. Improvement of recreation management is needed, which should include better enforcement of existing laws and increased public education programs. Forest Road (Route) closures should be implemented where needed (e.g., eliminate off-road vehicles from areas where they are currently directly impacting streams).

Integrated management. The CDFW performs regular monitoring of populations in the native range (Carmona-Catot and Weaver 2006, Weaver and Mehalick 2008, Weaver and Mehalick 2009), and these surveys should continue in order to determine population status and to document the presence and distribution of non-native trout. The CDFW plans greatly expanded genetics, population structure and abundance, and habitat monitoring in the near future which will include random stratified sampling of sites throughout the SFKR and GTC drainages (J. Weaver, CDFW, pers. comm. 2013). This level of sampling will provide scientifically rigorous and objective data to inform future management on a much broader spatial scale than ever performed. Beyond expanded monitoring, two kinds of refuges in the native range should also be established for managing California golden trout: (1) streams containing unhybridized populations and (2) streams containing populations with low levels of hybridization (S. Stephens et al. 2004). Defensible streams 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 periodic genetic assessments of refuge populations. In addition, populations of unhybridized California golden trout found outside the native range should also receive special protection and management, as described for populations in the native range. These would serve as additional refuge populations and could be used for experiments in management (e.g., modified grazing practices, introductions from other populations to increase genetic diversity) without compromising genetically ‘pure’ populations within the native range. For information on additional management measures, see Stephens et al. (2004) and Sims and McGuire (2006).

Figure 1. Distribution of California golden trout, *Oncorhynchus mykiss aguabonita* (Jordan), in the upper Kern River basin, California.

COASTAL CUTTHROAT TROUT
***Oncorhynchus clarkii clarkii* (Richardson)**

Status: Moderate Concern. Coastal cutthroat trout populations in California are small and face multiple threats, including predicted outcomes of climate change in their range.

Description: Coastal cutthroat trout are similar to coastal rainbow trout (*O. mykiss*) but have heavier spotting, particularly below the lateral line, and heavy spots on paired and anal fins. The spots become nearly invisible when fish become silvery during migrations to and from sea. Mature fish in fresh water have a dark coppery or brassy appearance (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 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 for individuals from landlocked populations to exceed 30 cm FL.

Taxonomic Relationships: The coastal cutthroat has long been recognized as distinct and it was the first cutthroat trout described by John Richardson in 1836. He used the name "*Salmo clarkii*" so *clarkii* with a 'double-' ending is the correct name, even if not widely used (Trotter 2007). Behnke (1992, 1997) proposed that, approximately one million years ago, cutthroat trout diverged into two major lineages, the coastal cutthroat (*O. c. clarkii*) and all the interior subspecies (with complex evolutionary histories). The coastal cutthroat are characterized by 68 chromosomes and interior cutthroat subspecies are characterized by either 66 or 64 chromosomes. The 64 chromosome fish include Lahontan cutthroat (*O. c. henshawi*) and Paiute cutthroat (*O. c. seleneris*) in California (Trotter 2007). The coastal cutthroat has numerous populations that spend their entire life cycle in fresh water but are genetically connected to sea-run populations (Trotter 2007). Coastal cutthroat have colonized coastal rivers from northern California to Prince William Sound in Alaska; their populations can be divided into a number of Distinct Population Segments (Johnson et al. 1999). California's populations are at the southern end of the coast range lineage and include both sea-run and freshwater populations. The populations in California are considered part of the Southern Oregon-California Coast DPS (Johnson et al. 1999; Trotter 2007).

Life History: Coastal cutthroat trout possess variable life history strategies (DeWitt 1954; Pauley et al. 1989, Moyle 2002). This plasticity is among the most extreme in Pacific salmonids and variations in migratory behavior are found both between and within populations. Trotter (2007) categorizes this diversity into four main groups: (1) amphidromous (sea-run) life history, (2) lacustrine life history, (3) riverine (potadromous) life history, and (4) stream-resident. The amphidromous forms are not considered strictly anadromous because they can move back and forth between fresh and salt water multiple times to feed (often on other salmonids), although they also migrate into fresh water to spawn. Lacustrine coastal cutthroat use large lakes like the ocean (but do not occur in California). Potadromous forms are found in rivers and make seasonal migrations up and down these rivers. Resident populations are typically found above natural barriers, in headwaters. Offspring of resident fish can become amphidromous and vice-versa (Trotter 2007). The Smith and Klamath rivers in California have both amphidromous populations and resident populations isolated in small streams upstream of barriers (e.g., Little Jones and Tectah creeks). Sea-run cutthroat trout generally make their first migrations 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 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 migratory behavior and residency time (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 salmonids; this flexibility may release cutthroat trout from competition and predation pressures at certain times of year, while allowing them to track the movements of juvenile salmonids as prey (Trotter 2007).

Coastal cutthroat trout have ecological requirements analogous to those of resident rainbow trout and steelhead. When the two species co-occur, cutthroat trout occupy smaller tributary streams, while the competitively dominant steelhead occupy larger tributaries and rivers. As a consequence, cutthroat trout tend to spawn and rear higher in watersheds than steelhead. While cutthroat and rainbow trout can naturally hybridize, this spatial segregation is likely a key reproductive barrier that functions in many streams where their distribution overlaps. 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 sea-run fish rarely spawn before age 4 (Johnson et al. 1999). Sexually mature trout can demonstrate precise homing capabilities in their migrations to natal streams. In northern California, coastal cutthroat trout migrate upstream to spawn after the first significant rain, beginning in fall. Peak spawning occurs in December in larger streams and January to February in smaller streams (Johnson et al. 1999). Ripe or nearly ripe females have been caught from September to April in California streams, indicating a prolonged spawning period.

Females dig redds in clean gravels with their tails, predominantly in the tails of pools in low gradient reaches, often with low flows (less than 0.3 m^3/second summer flows) (Johnston 1982, Johnson et al.1999, Trotter 2007). The completed redds average around 35 cm in diameter by 10-12 cm deep. After spawning is completed, the female covers her redd with about 15-20 cm of gravel. Each female may mate with numerous males. Fecundity ranges from 1,100 to 1,700 eggs for females between 20 and 40 cm TL. Coastal cutthroat trout are iteroparous with a higher incidence of repeat spawning than steelhead. They can spawn every year but post-spawning mortality can be quite high. Maximum age recorded for coastal cutthroat is 14 years, from Sand Creek, Oregon (Trotter 2007).

Eggs hatch after 6-7 weeks of incubation, depending on temperature. Alevins emerge as fry between March and June, with peak emergence during mid-April, then spend the summer in backwaters and stream margins (Johnson et al. 1999). Juveniles remain in the upper watershed until approximately 1 year in age, at which point they may move extensively throughout 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 et al. 1999). Smolts or adults entering the saltwater environment remain close to the shore and do not normally venture more than about 7 km from the edge of the coast (Johnson et al. 1999). Typically, they stay in or close to the plume of the river in which they were reared (Trotter 2007). Individuals can spend prolonged periods (months) in estuaries, often moving in and out of fresh water, likely taking advantage of different feeding and rearing habitats. Cutthroat trout up to ~350 mm were captured in the Smith River estuary from May- October 1997-2001 (R. Quiñones, unpublished observations). A similar pattern is observed in the Klamath River estuary (M. Wallace, CDFW, pers. comm. 2013).

Adults feed on benthic macroinvertebrates, terrestrial insects in drift and small fish, while juveniles feed primarily on zooplankton, macroinvertebrates, and microcrustaceans (Wilzbach 1985, Romero et al. 2005). White and Harvey (2007) found that cutthroat trout of all sizes in small creeks fed mainly on aquatic insects in low numbers, but that earthworms washed in by winter storms may be bioenergetically most important for overwintering survival. Cutthroat captured in Prairie Creek appeared to feed opportunistically on migrating Chinook salmon fry during peak migration periods (M. Sparkman, CDFW, pers. comm. 2011) and cutthroat captured in the Klamath estuary regurgitated salmon eggs during late summer, when large numbers of adult salmon were being caught and cleaned (M. Wallace, CDFW, pers. comm. 2013). In the marine environment, cutthroat trout feed on various crustaceans and fishes, including Pacific sand lance (*Ammodytes hexapterus*), salmonids, herring and sculpins. 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 ample cover and deep pools for holding in summer. They prefer small, low gradient coastal streams and estuarine habitats, including lagoons. 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, although they have been observed spawning in velocities as low as 0.01-0.03 in small streams in Oregon (Pauley et al. 1989).

Optimal stream temperatures are less than 18°C, with preferred temperatures being around 9-12°C. This may explain why they occur mainly in more northern streams in California, within the coastal fog belt. In Washington streams, most rapid growth occurred at 8-10°C, in early summer, with rates declining as temperatures rose to 12-14°C (Quinn 2005). Spawning has been recorded at temperatures of 6-17° C, with preferred temperatures of 9-12° C (Pauley et al.

1989, Moyle 2002). Coastal cutthroat require high dissolved oxygen levels and will avoid areas with less than 5 mg/L DO in summer months (Pauley et al. 1989). Feeding and movement of adults are impaired at turbidities of greater than 35 ppm. Embryo survival is greatly reduced at turbidities >103 ppm and dissolved oxygen levels <6.9 mg/l.

Distribution: Coastal cutthroat trout are distributed from the Seward River, in Southern Alaska, to tributaries to the Salt River, a tributary to the Eel River estuary in Humboldt County, California. There are anecdotal reports of cutthroat in lower Eel River tributaries, near Fortuna, and, in 1992, coastal cutthroat trout were identified in Barber Creek, a lower Van Duzen River tributary (S. Downie, CDFW, pers. comm. 2012). North of the Eel River, their range coincides closely with that of temperate coastal rain forest (Trotter 2007). The interior range of the subspecies in Washington, Oregon, and California is bounded by rain forests on the western slope of the Cascade Range; their range rarely extends inland more than 160 km and is usually less than 100 km (Johnson et al. 1999). In California, this band is 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), where they still persist (J. Weaver, CDFW, pers. comm. 2011). Cutthroat from other parts of their range have also been successfully transplanted to Indian Creek, also in the Klamath River watershed (M. McCain, USFS, pers. comm. 2011).

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 may 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 basins where coastal cutthroat trout occur are the Smith, Mad and lower Klamath rivers. Cutthroat trout also rear in approximately 1875 ha of habitat in several coastal lagoons and ponds: Big, Stone, and Espa lagoons, and the Lake Earl-Talawa complex (Gerstung 1997). 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) indicated that the lower Mad River is another area of high cutthroat occupancy, but more recent assessments indicate that it contains only a small population (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.

Trends in Abundance: There are a limited number of long-term data sets available to evaluate population trends in coastal cutthroat trout, primarily of adults in Oregon and Washington. Data are spotty, scattered, and typically unpublished. Records suggest that coastal cutthroat trout were more abundant historically and, in some locations, supported substantial fisheries (Gerstung 1997). Current coastal cutthroat trout abundance is thought to generally be low in most waters, particularly where juvenile steelhead are present (Johnson et al. 1999, Griswold 2006). Effective population size in California streams is difficult to determine, but Gerstung (1997) estimated that there are likely less than 5,000 spawners each year in all of California.

The largest population apparently exists in the Smith River, where a local watershed group, the Smith River Alliance (SRA) and U.S. Forest Service conduct annual snorkel surveys for salmon and trout. Figure 1 summarizes results for surveys of all cutthroat observed from 2003 to 2011, with the exception of 2004 for which data are unavailable. A dedicated CDFW biologist is now conducting fisheries monitoring in the Smith River watershed (M. Gilroy, CDFW, pers. comm. 2012). Additionally, CDFW's Heritage and Wild Trout Program conducted watershed-wide population and habitat surveys in 2010 and has provided assistance to the annual snorkel survey population counts coordinated by the Smith River Alliance (J. Weaver, CDFW, pers. comm. 2012). Previous population and trend data collections from the Smith River have been intermittent and represent only a small portion of the range with inconsistent locations and methods over the years. The Yurok Tribe has conducted anadromous salmonid surveys on the lower Klamath River and many of its tributaries and found cutthroat widely distributed in medium to high densities in nearly all lower Klamath tributaries downstream of Mettah Creek (Gale and Randolph 2000). Figure 2 summarizes results for the number of adults observed during surveys in Blue Creek, one of the most productive coastal cutthroat streams in the Klamath basin, for most years 1999-2009. Although both populations appear to be increasing in recent years, analysis by Quiñones (2011) was unable to detect significant trends for the Blue Creek data. Longer time series are also difficult to interpret (Figure 3; Johnson et al. 1999).

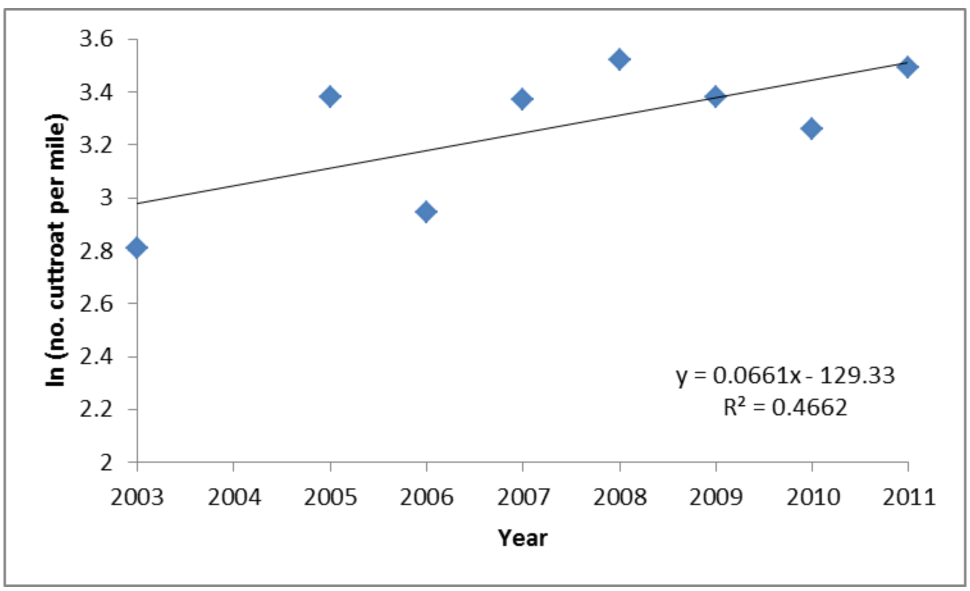


Figure 1. Coastal cutthroat abundance (ln(fish number/mile)) in the Smith River watershed, 2003-2011.

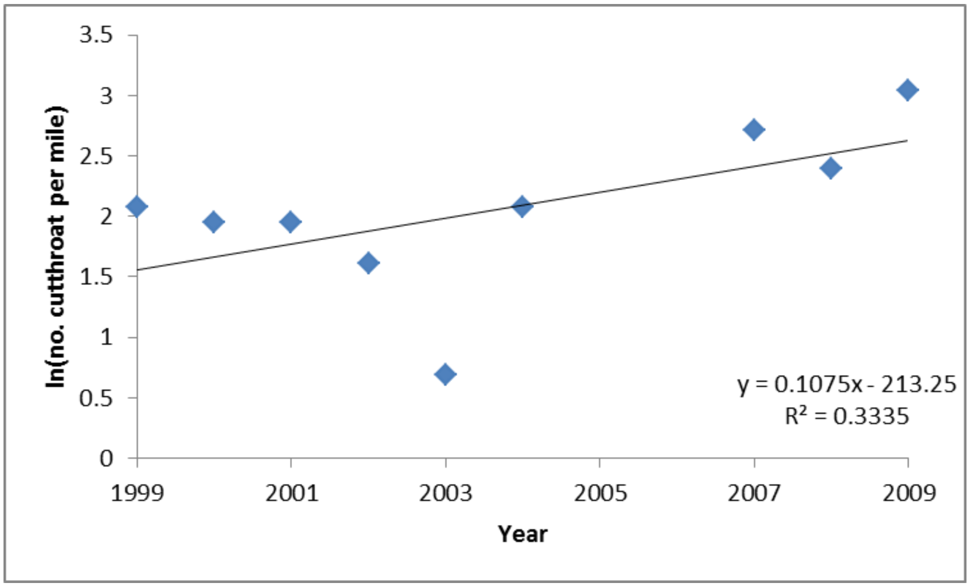


Figure 2. Adult coastal cutthroat abundance (ln(fish number/mile)) in Blue Creek, Klamath basin, 1999-2009.

Because quantitative measures of historical abundance are lacking, it is difficult to determine whether populations are in decline, increasing, or stable (Johnson et al. 1999, Griswold 2006). Declines in coastal cutthroat numbers are likely due to extensive changes made to estuaries, watersheds, and streams throughout their range in California. Fortunately, there is increasing protection in some areas (e.g., Smith River, streams in Del Norte Coast Redwoods State Park), in part to protect listed coho salmon.

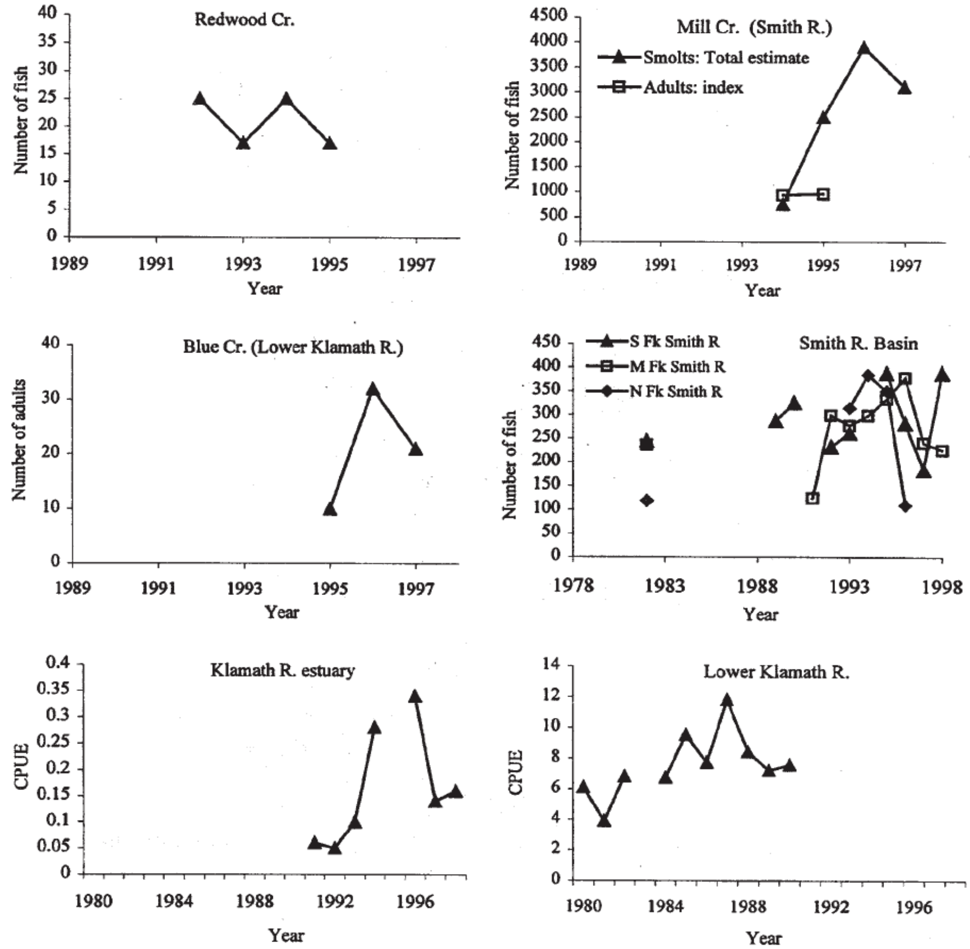


Figure 3. Coastal cutthroat trout abundances from Johnson et al. (1999). Data include both snorkel surveys and electrofishing efforts.

Nature and Degree of Threats: Major factors affecting the status of coastal cutthroat trout are discussed below. Populations are affected differentially by one or more stressors, depending on location (Table 1). According to Gregory and Bisson (1997), degraded habitat is associated with more than 90% of documented extinctions or declines of Pacific salmonid stocks. Coastal cutthroat trout stocks are no exception to the rule. Major anthropogenic land-use activities, including agriculture, forestry, grazing, water diversions, urban and industrial development, road construction and mining, have resulted in the alteration and loss of cutthroat trout habitat and a subsequent 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, presence of tidal gates, the loss of large wood in channels, and road impacts on small headwater streams are all associated with habitat degradation in the coastal cutthroat trout's range. While treated separately in this account, the various contributing causes of decline are multiple and often interact synergistically. A unique problem relates to the effects of habitat alteration on interactions between steelhead and cutthroat trout. The two species naturally co-occur and 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.

Dams and diversions. Dams and diversions have altered flows in 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.

	Rating	Explanation
Major dams	Medium	Dams present on some streams
Agriculture	Medium	Conversion of estuarine wetlands to agricultural lands, diversions, influx of fertilizers and other pollutants into estuaries
Grazing	Medium	Some impacts in lowland areas, especially where estuary marshes have been converted to pasture
Rural Residential	Medium	Effects localized, but increasingly an issue in Humboldt Bay tributaries and the Crescent City area
Urbanization	Low	Increasingly an issue in Humboldt Bay tributaries
Instream mining	Low	No known impact but occurs in some streams
Mining	n/a	
Transportation	Medium	Roads are an ongoing source of sediment input, habitat fragmentation, and channel alteration
Logging	Medium	Major activity in many watersheds; dramatic historic impacts in many areas
Fire	Low	Increased stream temperatures and sediment input may be a factor in some inland watersheds
Estuary alteration	Medium	Estuaries are important habitat and have been significantly altered
Recreation	Low	Probably minor but may affect populations in heavily used streams
Harvest	Low	Harvest is generally light but not widely monitored; data mostly limited to CDFW Heritage and Wild Trout Program angler survey boxes at lagoons
Hatcheries	Medium	Possible hybridization or competition with hatchery steelhead
Alien species	Low	Alien species are common throughout range; impacts to coastal cutthroat are unknown but assumed to be minimal at present

Table 1. Major anthropogenic factors limiting, or potentially limiting, viability of coastal cutthroat trout populations in California. Factors were rated on a five-level ordinal scale where a factor rated “critical” could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated “high” could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated “medium” is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated “low” may reduce populations but extinction unlikely as a result; and a factor rated “no” has no known negative impact to the taxon under consideration. Certainty of these judgments is moderate. See methods section for descriptions of the factors and explanation of the rating protocol.

Agriculture. Agricultural practices that most impact cutthroat trout are likely reclamation of estuarine marshes, water diversions and 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). Increasingly, marijuana growing (particularly along the north coast region of California) is a threat to aquatic habitats because growers divert water from headwater streams, convert large areas of former timber lands to monocultural crop production, pollute streams with pesticides and other material, and degrade stream habitats. Unfortunately, there are no known studies that document such impacts specifically for coastal cutthroat trout.

Grazing. Grazing occurs in most of the former wetlands surrounding the Smith River estuary, Humboldt Bay, and the Eel River estuary which are present and historic ‘hot spots' for coastal cutthroat. Grazing and other agriculture occurs in the lower Mad River, Little River, and Redwood Creek. In all instances, these rivers have been isolated from their surrounding riparian habitat to the detriment of cutthroat trout. In addition, complete blockage of access to estuarine marsh channels by tide gates and dikes to keep pastures from flooding greatly reduce rearing habitat.

Rural residential. Residential areas are scattered throughout the range of coastal cutthroat trout and likely impact fish through habitat alteration, diversion of water, and pollution from septic tanks or surface runoff. These effects are mostly localized but, cumulatively, could pose significant threats during drought periods.

Urbanization. Urbanization plays an important role in reducing cutthroat trout habitat in urban streams in the Humboldt Bay region and around Crescent City (T. Weseloh, pers. comm. 2008). These streams generally have reduced cover, shallower pools, and poorer water quality than less disturbed streams.

Transportation. Roads or railroads line most streams, most dating from past eras of heavy exploitation of natural resources. They continue to be a major source of habitat loss for cutthroat trout through continued bleeding of sediment into streams and poorly constructed or placed culverts that prevent access to headwater areas. In addition, roads, railroads, and other infrastructure associated with transportation and urbanization limit habitat restoration projects because ‘hardened' banks are very difficult and expensive to restructure into viable habitat for fish.

Logging. Logging and associated road networks have caused tremendous impacts to coastal cutthroat trout habitats with massive landslides and erosion stemming from excessive tree removal and road construction on steep, unstable soils found in 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 such as timber harvest. Johnson et al. (1999) cite numerous studies showing the importance of riparian vegetation to fish production and note 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 to intolerable levels, as well as bury spawning gravel, alter rearing habitats, and fill pools. Additionally, clear cutting in headwater basins has decreased shading and reduced the absorption capacity of soils. In certain areas, this is likely to have increased stream temperatures and incidence of flash flooding, as well as reduced late summer and early fall base flows. While the (especially legacy) impacts from logging to coastal cutthroat trout may merit a ‘high' threat score (Table 1), historic impacts were much greater and, thanks to strict timber harvest regulations and many restoration efforts, current impacts are substantially reduced in many watersheds.

Harvest. Gerstung (1997) indicated that historical runs of coastal cutthroat trout were quite large and that, in some areas, substantial commercial and sport fisheries existed for them. Today, fisheries for coastal cutthroat occur mainly in coastal lagoons, where populations tend to be largest. Fisheries elsewhere are small and largely catch-and-release, although impacts from harvest on coastal cutthroat trout populations are unknown. In general, coastal cutthroat trout receive considerably less attention from anglers than the more popular salmon and steelhead fisheries of the north coast.

Hatcheries. Coastal cutthroat trout are generally competitively subordinate to all other species of salmonids (Johnson et al. 1999) and hatchery steelhead, in particular, are likely to affect their numbers through predation and competition, as well as disease (Johnson et al. 1999).

Estuarine alteration. Estuaries are important for cutthroat trout rearing and passage, yet most in California have been severely altered, usually for agriculture. In general, there is much less habitat available in the larger estuaries (e.g. Eel and Smith rivers, Humboldt Bay) than in the past.

Alien species. Alien species occur throughout the range of coastal cutthroat trout but impacts appear to be small. Aliens with potential impacts include: (1) New Zealand mud snail in lower Klamath, Big and Stone lagoons, Lake Earl, lower Smith River, and Redwood Creek; (2) largemouth bass (*Micropterus salmoides*) in the Big Lagoon watershed; (3) striped bass (*Morone saxatilis*) in several estuaries; and (4) Sacramento pikeminnow (*Ptychocheilus grandis*) in the Eel River (M. Gilroy, CDFW, pers. comm. 2011). The threat from pikeminnow may increase if they spread beyond the Eel River; CDFW has captured small numbers in Martin Slough, a tributary to Elk River, which flows into Humboldt Bay.

Effects of Climate Change. Climate change will further stress coastal cutthroat trout populations in California that have already been depleted over the last 50 years; existing numbers suggest that the overall population in California is low (Johnson et al. 1999). Coastal cutthroat occur primarily in north coast streams close to the ocean, which may seem to be relatively protected from predicted temperature increases, due to the influence of fog, although the effects of climate change on ocean currents and coastal fog is poorly understood (Quinones and Moyle, in press). However, their requirements for exceptionally cool water (<18ºC) may allow even small temperature increases to have a major effect on growth and survival. The suitability of estuarine habitats may also decrease as sea levels rise and more extreme tides and storm surges alter salinity profiles that define food webs. Sea level rise will move estuarine conditions farther upstream, potentially causing more competition and/or hybridization between cutthroat trout and steelhead (M. Wallace, CDFW, pers. comm. 2013). For these reasons, Moyle et al. (2013) regarded coastal cutthroat trout as "critically vulnerable" to extinction in California as the result of the added effects of climate change.

Status Determination Score = 3.0 – Moderate Concern (see Methods section Table 2)**.** Coastal cutthroat trout are apparently in no immediate risk of extinction throughout their range in California, but there is a high degree of uncertainty about their status in the state and most populations can decline rapidly in response to environmental change (Table 2). Coastal cutthroat trout persist in many streams on the northern California coast, although most populations are rarely monitored. They are listed as a Sensitive Species in California by the U.S. Forest Service. Their populations are now entirely dependent on natural reproduction. This makes them unique among the more abundant north coast salmonids, so they are, therefore, a good indicator of condition of streams in their range. Nevertheless, coastal cutthroat trout are a non-commercial, non-listed, widely-distributed, and somewhat cryptic salmonid that support a minor sport fishery. Until there is evidence to the contrary, coastal cutthroat trout should be assumed to be in decline in California; significant habitat alteration throughout their range, coupled with their fairly narrow environmental tolerances, means cutthroat populations can become extirpated one at a time. Monitoring coastal cutthroat populations to document the effects of climate change on north coast rivers is of particular value because of their lack of hatchery influence, dependence upon intact estuarine conditions, low exploitation rates, wide distribution, intolerance of warm temperatures, and preference for smaller streams.

Metric	Score	Justification
Area occupied	5	Found in many watersheds from Eel River north
Estimated adult abundance	3	This would score '5' if assumed all populations are genetically interconnected; most appear to be small and fragmented within California at southern end of range
Intervention dependence	3	Persistence requires improved management of heavily logged watersheds and extensively altered estuaries
Tolerance	2	Prefer water temperatures below 12ºC
Genetic risk	3	Little information on genetics available; hybridization with steelhead may affect populations in some streams
Climate change	2	Most populations are in small streams or depend upon existing estuary conditions; considerable range-wide vulnerability to climate change
Anthropogenic threats	3	See Table 1
Average	3.0	21/7
Certainty (1-4)	3	Information was compiled for recent status review

Table 2. Metrics for determining the status of coastal cutthroat trout in California, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: The greatest conservation need for coastal cutthroat trout is updated information on their status and distribution so appropriate management measures can be taken. The NMFS team, writing the 1999 status review of coastal cutthroat trout in Washington, Oregon and California, concluded 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). A 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 in November, 2006, as the Coastal Cutthroat Trout Executive Committee (currently referred to as the Coastal Cutthroat Trout Interagency Committee) (Griswold 2006). Nearly a decade after the 1999 status report, the 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;

their findings for California were that virtually all aspects of coastal cutthroat biology, status, and distribution needed updating. However, as a result of their ongoing efforts, many sources of data have been identified and continue to be compiled in a georeferenced database across the species' range (K. Griswold, pers. comm. 2013). In California, research and monitoring of coastal cutthroat trout is being performed by the California Department of Fish and Wildlife, Humboldt State University, the Yurok Tribe, Green Diamond Resource Company, USFS (Six Rivers National Forest, Redwood Sciences Laboratory), and other agencies and groups. California's Fisheries Monitoring Plan aims to provide consistent methods for monitoring (M. Gilroy, CDFW, pers. comm. 2011). The interagency committee has planned a coastal cutthroat trout rangewide status assessment for 2014-15. The California portion of the range is scheduled first and will be the first comprehensive assessment in California since Gerstung (1997). Developing long-term management strategies for coastal cutthroat trout is heavily dependent on improved monitoring and assessment.

Griswold (2006) noted "it should be recognized that a voluntary effort that tackles difficult scientific and monitoring issues for a non-listed non-commercial subspecies requires considerable leadership and good will from Federal and State agencies." The development of a multi-agency cutthroat trout management team, along with significant resources by state and federal agencies, will hopefully fill much needed data gaps and provide the framework for future coastal cutthroat trout conservation. Such measures are particularly needed in California, where coastal cutthroat trout populations are fragmented at the southern end of their range and, therefore, may be exceptionally vulnerable to climate change or other stressors.

The many measures, both local and regional, taken (or proposed) to protect steelhead and salmon populations should benefit coastal cutthroat trout, although direct benefits remain largely unstudied. Continued management of the Smith River as a free-flowing, wild river that is a refuge for all salmonids, including the seemingly abundant cutthroat trout, is of particular importance. Recent conservation measures have included acquisition and protection of much of the Goose Creek, Mill Creek, Hurdygurdy Creek, Little Jones Creek, and Siskiyou Fork watersheds. Mill Creek has benefited from numerous habitat restoration projects (M. McCain, USFS, pers. comm. 2012). Other targeted restoration efforts include: Lake Earl, Jordan Creek, Stone Lagoon, tributaries to Lake Earl, Big Lagoon, and many creeks in Humboldt and Del Norte counties, including Blue Creek in the Klamath basin (T. Weseloh, pers. comm. 2008, M. Gilroy, CDFW, pers. comm. 2011, R. Quiñones, pers. comm. 2011). Blue Creek has become a Salmon Sanctuary of the Yurok Tribe, to protect its diverse salmonids; the lower reaches are being acquired on behalf of the tribe by the Western Rivers Conservancy (pending successful fund raising, 2013).

Figure 4. Generalized distribution of coastal cutthroat trout, *Oncorhynchus clarkii clarkii,* in California.

SARATOGA SPRINGS PUPFISH
Cyprinodon nevadensis nevadensis (Miller)

Status: High Concern. Saratoga Springs pupfish numbers appear to be stable; however, they should be monitored closely because limited distribution in extreme habitats increases their vulnerability to anthropogenic and natural stressors.

Description: All Amargosa pupfish (*Cyprinodon nevadensis)* subspecies are small, rarely exceeding 50 mm TL. The body is deep, especially in reproductive males. The head is blunt and slopes steeply to a small, terminal, oblique mouth. There is one row of tricuspid teeth on each jaw, with the central cusps being truncated or pointed. *Cyprinodon nevadensis* is a variable species, but can be distinguished by the following morphometric characteristics: (1) the scales are large, the circuli lack spine-like projections, and the interspaces are reticulated; (2) there are 23-28 scales (usually 25-26) along the lateral line and 15-24 scales (usually 16-18) anterior to the dorsal fin; (3) the pelvic fins are reduced and may even be absent; (4) there are 8-11 anal fin rays (usually 10), 11-18 pectoral fin rays (usually 15-17), 0-9 pelvic fin rays (usually 6), and 14-22 caudal fin rays (usually 16-19); gill rakers range from 14-22 (usually 15-17) and preopercular pores from 7-17 (usually 12-14). Reproductive males in breeding colors are bright blue with a black band at the posterior edge of the caudal fin. Reproductive females are drab olive-brown and develop 6-10 vertical bars along the sides which may be distinct or faint. An ocellus (eyespot) is typically present on the posterior base of the dorsal fin of females.

Cyprinodon n. nevadensis can be distinguished from other subspecies by its deeper, broader body, anteriorly placed pelvic fins, and a greater average number of scales (Table 1). Scales are narrow and larger, with very dense and extensive reticulations and a high number of scale radii. Males of this subspecies have an intense blue coloration (Soltz and Naiman 1978).

Taxonomic Relationships: The fossil record and past geologic events suggest that the *Cyprinodon* species differentiated relatively recently, with most differentiation occurring during the pluvial-interpluvial fluctuations of the early to mid-Pleistocene (Miller 1981). Some differentiation may have occurred in the last 10,000 years, following the final recession of pluvial waters. As water table height receded in the Great Basin during the Pleistocene, numerous scattered lakes and streams shrank, isolating remnant populations of pupfishes, which led to allopatric speciation of *C. nevadensis*.

Cyprinodon nevadensis, the complex of subspecies commonly referred to as Amargosa pupfish, was first described from Saratoga Springs by Eigenmann and Eigenmann (1889). Following this initial description, the species was lumped with desert pupfish (*Cyprinodon macularius)* until Miller (1943) separated it out again. In subsequent studies, Miller (1948) recognized and described six subspecies of *C. nevadensis*, four of which occurred in California: the Saratoga Springs pupfish (*C. n. nevadensis),* the Amargosa River pupfish (*C. n. amargosae),* the Shoshone Spring pupfish (*C. n. shoshone),* and the Tecopa pupfish (*C. n. calidae*). Two more subspecies occur in Nevada: the Ash Meadows pupfish (*C. n. mionectes)* and the Warm Springs pupfish (*C. n. pectoralis*). *Cyprinodon n. calidae* is now extinct (Moyle 2002).

Measure/ Count	*C. n. amargosae*			*C. n. nevadensis*			*C. n. shoshone*		
	male	ALL	female	male	ALL	female	male	ALL	female
Standard length (mm)		36			40			34	
*Body width	256		265	274		269	231		229
*Head length		305			312			307	
*Head depth	330		304	367		343	331		311
*Head width	240		259	257		256	233		231
*Snout length		101			97			89	
*Mouth width		117			115			114	
*Mandible length		198			95			93	
*Anal origin to caudle base	338		346	394		362	371		355
*Caudle peduncle length	264		237	277		253	263		251
*Anal fin base length	116		105	111		105	108		101
*Anal fin length	330		304	227		195	217		190
*Pelvic fin length	98		89	95		87	90		77
Anal fin ray count		10			10			10	
Dorsal fin ray count		10			10			10	
Pelvic fin ray count		6			6			4	
Pectoral fin ray count		16			16			16	
Caudal fin ray count		18			17			18	
Lateral line scales		26			26			26	
Predorsal scale count		19			18			18	
Dorsal fin to pelvic fin scale count		11			10			9	
Caudal peduncle circumference scale count		16			16			15	
Body circumference scale count		27			25			23	

*Expressed as percent of standard length x 1000.

Table 1. Comparative average morphometrics and meristics of *Cyprinodon nevadensis* subspecies. Adapted from Miller (1948).

Life History: Pupfish inhabit a wide variety of habitats and exhibit many adaptations to thermal and osmotic extremes (Miller 1981). Optimal temperature for growth is 22˚C, with growth ceasing below 17˚C and above 32˚C. At optimal temperatures, growth is extremely rapid and fish reach sexual maturity within four to six weeks (Miller 1948). Such a short generation time enables small populations to remain viable. Generation time varies among the subspecies, with populations living in widely fluctuating environmental conditions exhibiting shorter generation times (Moyle 2002). Young adults (15-30 mm SL) of *C. nevadensis* usually constitute a majority of the biomass throughout the year (Naiman 1976). Reproductive activity in Saratoga Springs peaks during the spring, tapers off during the summer, and is virtually nonexistent during fall and winter. This produces an annual population cycle with a low of about 800 pupfish in March and a high of about 2700 in September (LaBounty 2003).

Saratoga Springs pupfish, like other spring-dwelling subspecies, exhibit different reproductive behaviors than riverine forms (Kodric-Brown 1981). Males of spring-dwelling subspecies establish territories over substrate with topographic complexity suited for oviposition. Both sexes are promiscuous and a single female may lay eggs in a number of different territories. The demersal eggs are sticky and thus adhere to substrates. Females may lay a few eggs each day (not necessarily on consecutive days) throughout the year. Territorial defense by males confers some protection of the eggs from predators, but otherwise parental investment is limited to gamete production (Kodric-Brown 1981).

Little additional work has been done on the biology of Saratoga Springs pupfish; for more general information on the biology of the Amargosa pupfish species complex, see the Shoshone Spring pupfish account in this report and Moyle (2002).

Habitat Requirements: Saratoga Springs is roughly circular, approximately 10 m in diameter, 1-2 m deep (Miller 1948) and has a soft sand and silt bottom through which the spring inflow enters (P.B. Moyle and J. Katz, personal observations 2010). The spring water is clear and temperature is a constant 28-29°C. The spring overflows into a pond and then into a marsh 4-6 ha in area, ringed by sand dunes. The marsh has a grassy bottom with substrate consisting of mud and sand. Water temperatures fluctuate in the marsh area according to daily ambient temperature and may vary from 4 to 49°C, depending on season. Pupfish are largely inactive from late November to late January in water temperatures less than 7-10ºC. During summer, peak activity is concentrated at temperatures of 31-35ºC (LaBounty 2003). Pupfish tend to avoid temperatures exceeding 35°C, selecting areas along shore in 40-50 cm of water and between 20 and 30°C, when possible. As temperatures rise above 35-38ºC, fish will burrow into the marsh mud and, as temperatures cool in the fall, roughly a third of the population has been observed burrowing into marsh substrates for thermal refuge. Pupfish will move from shoreline areas into marshy meadow habitats when disturbed.

Reproduction occurs at temperatures between 28-35ºC, and reproductive behavior and reproductive colors fade at 35-38ºC (Labounty 2003). Juvenile fish are found in the marsh but are absent from the main spring, suggesting that spawning occurs only in the marsh. In 1995, fish abundance in the marsh exceeded that in the spring-pool by as much as two orders of magnitude, and length-frequency distributions differed between the two habitats. The spring-pool population was always dominated by adults, whereas juvenile fish dominated the marsh population. Length-weight regressions also showed that body condition of spring-pool fish exceeded that of marsh fish (Sada 2003).

Distribution: *Cyprinodon n. nevadensis* occur naturally only in Saratoga Springs and its outflow marsh in Death Valley National Park, San Bernardino County, California. This spring is located at an elevation of 70 m and is tributary to the Amargosa River (Miller 1948). Saratoga Springs pupfish were also introduced into "Lake" Tuendae (an artificial, spring-fed pond) at Zyzzyx, San Bernardino County, where they became established (Turner and Liu 1976). Inadvertent introduction of mosquitofish (*Gambusia affinis*) and accidental draining of a portion of Lake Tuendae during restoration appear to have led to a significant decrease in the pupfish population (Hughson and Woo 2004). More recent surveys, however, indicate a persistent population (S. Henkanaththegedara, pers. comm. 2008).

Trends in Abundance: Comparison of survey data collected in 1966 and 1995 at Saratoga Springs indicates that this population is stable and occupies all available habitat. Pupfish abundance estimates were similar between the two studies, with 1966 abundance estimates ranging from 761 to 3833, and 1995 estimates from 686 to 2993 (Sada 2003). While no systematic population survey has been performed on the Lake Tuendae population, incidental capture of pupfish during surveys for tui chub in 2006 and 2008 suggest an increasing population. In 2008, 1500 pupfish were captured. These capture data indicate that Lake Tuendae remains a viable refuge population (S. Henkanaththegedara, pers. comm. 2008).

Nature and Degree of Threats: The major threat to Saratoga Springs pupfish is the possibility that its unique habitat may become dewatered due to predicted climate change impacts, coupled with increasing human demand upon groundwater aquifers in this region. Saratoga Springs and the other springs on the eastern side of Death Valley are partially dependent on regional groundwater movement through large, ancient aquifers that extend into central Nevada and western Utah (Dettinger et al. 1995, Riggs and Deacon 2002, Deacon et al. 2007).

Agriculture. Ground water pumping for irrigation, even at great distances, could affect flow in Saratoga Springs. Agricultural impacts in this region have decreased as water supplies are increasingly captured by urban areas.

Urbanization. In order to meet increasing water demand by the city of Las Vegas, the Southern Nevada Water Authority (SNWA) proposed mining large quantities of water from several different valleys within the groundwater basin (Southern Nevada Water Authority 2004). The potential impacts of distant aquifer pumping were exemplified at Devils Hole in Nevada, which had reduced water levels as a result of water being pumped for irrigation from the Ash Meadows flow-system aquifer. The U.S. Supreme Court (United States v. Cappaert 1977) protected Devils Hole and the Devils Hole pupfish (*Cyprinodon diabolis*) by ordering that pumping stop (Deacon and Williams 1991). After rising and then stabilizing for a number of years, the water level in Devils Hole is now dropping again, likely the result of groundwater pumping a considerable distance away (Bedinger and Harrill 2006). If water withdrawals in the Amargosa region continue to increase and Las Vegas proceeds with its planned withdrawals, it is very likely that flows in the Amargosa River and tributary springs will be greatly reduced or eliminated during dry years. Demand for water and flood control is also on the rise with increasing human development of Tecopa and the upper Amargosa Valley.

Recreation. While Saratoga Springs is located in an area seldom visited by Death Valley tourists, public access is allowed and its protection relies largely on voluntary compliance with park rules. Isolation in limited, fragile, habitat increases risk to Saratoga Springs pupfish through potential contamination and introduction of exotic species and pathogens.

Alien species. Although Saratoga Springs is in Death Valley National Park, it is accessible to the public and, therefore, is vulnerable to intentional introduction of alien fishes or invertebrates. Alien species may compete with or prey on pupfish or introduce disease. The decline of the Lake Tuendae population of *C. n. nevadensis* after introduction of western mosquitofish to the lake (Hughson and Woo 2004) may serve as an example of the consequence of alien introduction. Concurrent with the introduction of mosquitofish, Mojave tui chub (*Siphateles bicolor mohavensis)* in Lake Tuendae were found to be infected with Asian tapeworm (*Bothriocephalus achelognathii*) and a perennial algal bloom began. It is uncertain whether mosquitofish caused the algae bloom but they were almost certainly the tapeworm vector. It is not known if the tapeworm is deleteriously affecting the pupfish population.

	Rating	Explanation
Major dams	n/a	
Agriculture	Medium	Ground water pumping for irrigation even at great distances could affect flow in Saratoga Springs
Grazing	n/a	
Rural Residential	n/a	
Urbanization	High	Groundwater pumping by the city of Las Vegas has potential to intercept aquifer water flowing to Saratoga Springs
Instream mining	n/a	
Mining	Low	Mining was a prominent land use in the area but no known impact of abandoned mines on Saratoga Springs
Transportation	n/a	
Logging	n/a	
Fire	n/a	
Estuary alteration	n/a	
Recreation	Medium	Public access to isolated habitat increases risk
Harvest	n/a	
Hatcheries	n/a	
Alien species	High	Introduction of alien species into Saratoga Springs may threaten pupfish population

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

Effects of Climate Change: Although all pupfishes of the American Southwest are remarkably well adapted to the wide range of salinity and temperature characterized by their desert habitats, they are also remarkably vulnerable to change. Isolated desert springs and rivers fed by subsurface flow systems are precarious ecosystems, vulnerable to geologic and anthropogenic disruption. Fed by rain and snow melt at high elevation in the desert mountain ranges (Riggs and Deacon 2002), desert aquifers in the Death Valley region will likely receive less recharge as the region warms. Reduced recharge will be compounded by growing human demand for water in southern Nevada which already exceeds supply in this arid region. Predicted increases in air temperature may also have direct impacts on Saratoga Springs pupfish if water temperatures in the spawning habitat of Saratoga Springs Marsh exceed thermal tolerances for egg production. Moyle et al.

(2013) rated Saratoga Springs pupfish as "highly" vulnerable to extinction in the next 100 years due to climate change impacts.

Status Determination Score = 2.3 - High Concern (see Methods section Table 2). Isolated in a single spring system, the Saratoga Springs pupfish remains vulnerable to both anthropogenic and natural perturbations. NatureServe assigns the Saratoga Springs pupfish a conservation rank of "Critically Imperiled" (G2T1, http://www.natureserve.org/explorer/) due to small range and threats of hybridization and predation, while the American Fisheries Society considers it to be "Threatened" (Jelks et al. 2008). As a consequence of proposed aquifer pumping, the Superintendent of Death Valley National Monument (E. L. Rothfuss, letter to B. Bolster of CDFW, 27 May 1992) recommended the Saratoga Springs pupfish be listed as threatened. This recommendation was endorsed by the Desert Fishes Council (E. P. Pister, pers. comm.).

Metric	Score	Justification
Area occupied	2	Native range confined to Saratoga Springs with one small refuge population in Lake Tuendae
Estimated adult abundance	3	Population stable with short generation time
Intervention dependence	3	Continuous protection of spring required
Tolerance	2	Although remarkably adapted to temperatures and salinities that would kill most other fishes, Saratoga Springs pupfish exist at the very edge of their reproductive tolerances
Genetic risk	3	Single population but no apparent signs of genetic bottleneck or inbreeding depression
Anthropogenic threats	2	See Table 2
Climate change	1	Threatened by reduced recharge capacity in base aquifer and increases in temperature
Average	2.3	16/7
Certainty (1-4)	4	Small population easily monitored

Table 3. Metrics for determining the status of Saratoga Springs pupfish, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: The Saratoga Springs complex is protected by the National Park Service and is in near-pristine condition. However, flow reduction associated with groundwater pumping and increasing aridity remains a threat. Hydrological studies should be initiated to determine whether or not Saratoga Springs (and other springs in the region) is/are connected to the aquifer being pumped. A contingency plan should be developed that includes the identification of natural and/or artificial habitats to temporarily hold pupfish from Saratoga Springs in the event population loss appears imminent. The population at Zyzzxx should be monitored and maintained as a refuge population.

Figure 1. Distribution of Saratoga Springs pupfish, *Cyprinodon nevadensis nevadensis*, in California. Dashed grey circle represents refuge population at Zyzzxx (Lake Tuendae).

AMARGOSA RIVER PUPFISH
Cyprinodon nevadensis amargosae (Miller)

Status: High Concern. The Amargosa River pupfish needs to be monitored closely because its status could change quickly if river flows are further reduced or new non-native species invade.

Description: All Amargosa pupfish (*Cyprinodon nevadensis)* subspecies are small fish that rarely exceed 50 mm TL. The body is deep, especially in reproductive males. The head is blunt and slopes steeply to a small, terminal, oblique mouth. There is one row of tricuspid teeth on each jaw, with the central cusps being truncated or pointed. *C. nevadensis* is a variable species, but can be distinguished from other pupfishes by the following characteristics: (1) the scales are large, the circuli lack spine-like projections, and the interspaces are reticulated; (2) there are 23-28 scales (usually 25-26) along the lateral line and 15-24 scales (usually 16-18) anterior to the dorsal fin; (3) the pelvic fins are reduced and may even be absent; (4) there are 8-11 anal fin rays (usually 10), 11-18 pectoral fin rays (usually 15-17), 0-9 pelvic fin rays (usually 6), and 14-22 caudal fin rays (usually 16-19); (5) gill rakers range from 14-22 (usually 15-17) and preopercular pores from 7-17 (usually 12-14). Reproductive males in breeding colors are bright blue with a black band at the posterior edge of the caudal fin. Reproductive females are drab olive-brown and develop 6-10 vertical bars along their sides which may be distinct or faint. An ocellus (eyespot) is typically present on the posterior base of the dorsal fin of females.

The Amargosa River subspecies is similar to *Cyprinodon n. nevadensis* but has more scales around the body and fewer scale radii than other subspecies (see Table 1).

Taxonomic Relationships: *C. nevadensis amargosae* is one of three extant subspecies of *C. nevadensis* found in California. The fossil record and past geologic events suggest that the *Cyprinodon* species differentiated relatively recently, with most differentiation occurring during the pluvial-interpluvial fluctuations of the early to mid-Pleistocene (Miller 1981). Some differentiation may have even occurred in the last 10,000 years, following the final recession of pluvial waters. As water table height receded in the Great Basin during the Pleistocene, numerous scattered lakes and streams shrank, isolating remnant populations of pupfishes which led to allopatric speciation of *C. nevadensis*.

C. nevadensis is a complex of subspecies commonly (and confusingly) referred to as Amargosa pupfish. The species was first described from Saratoga Springs by Eigenmann and Eigenmann (1889) but, following the initial description, it was lumped with the desert pupfish (*Cyprinodon macularius)* until Miller (1943a) separated it again. In subsequent studies, Miller (1948) recognized and described six subspecies of *C. nevadensis*, four of which occurred in California: Saratoga Springs pupfish (*C. n. nevadensis)*, Amargosa River pupfish (*C. n. amargosae)*, Shoshone Springs pupfish (*C. n. shoshone)*, and Tecopa pupfish (*C. n. calidae*). Two more subspecies occur in Nevada: Ash Meadows pupfish (*C. n. mionectes)* and Warm Springs pupfish (*C. n. pectoralis*). *Cyprinodon n. calidae* is now extinct (Moyle 2002).

Measure/ Count	*C. n. amargosae* male	*C. n. amargosae* ALL	*C. n. amargosae* female	*C. n. nevadensis* male	*C. n. nevadensis* ALL	*C. n. nevadensis* female	*C. n. shoshone* male	*C. n. shoshone* ALL	*C. n. shoshone* female
Standard length (mm)		36			40			34	
*Body width	256		265	274		269	231		229
*Head length		305			312			307	
*Head depth	330		304	367		343	331		311
*Head width	240		259	257		256	233		231
*Snout length		101			97			89	
*Mouth width		117			115			114	
*Mandible length		198			95			93	
*Anal origin to caudle base	338		346	394		362	371		355
*Caudle peduncle length	264		237	277		253	263		251
*Anal fin base length	116		105	111		105	108		101
*Anal fin length	330		304	227		195	217		190
*Pelvic fin length	98		89	95		87	90		77
Anal fin ray count		10			10			10	
Dorsal fin ray count		10			10			10	
Pelvic fin ray count		6			6			4	
Pectoral fin ray count		16			16			16	
Caudal fin ray count		18			17			18	
Lateral line scales		26			26			26	
Predorsal scale count		19			18			18	
Dorsal fin to pelvic fin scale count		11			10			9	
Caudal peduncle circumference scale count		16			16			15	
Body circumference scale count		27			25			23	

*Expressed as percent of standard length x 1000.

Table 1. Comparative average morphometrics and meristics of three *Cyprinodon nevadensis* subspecies. Adapted from Miller (1948).

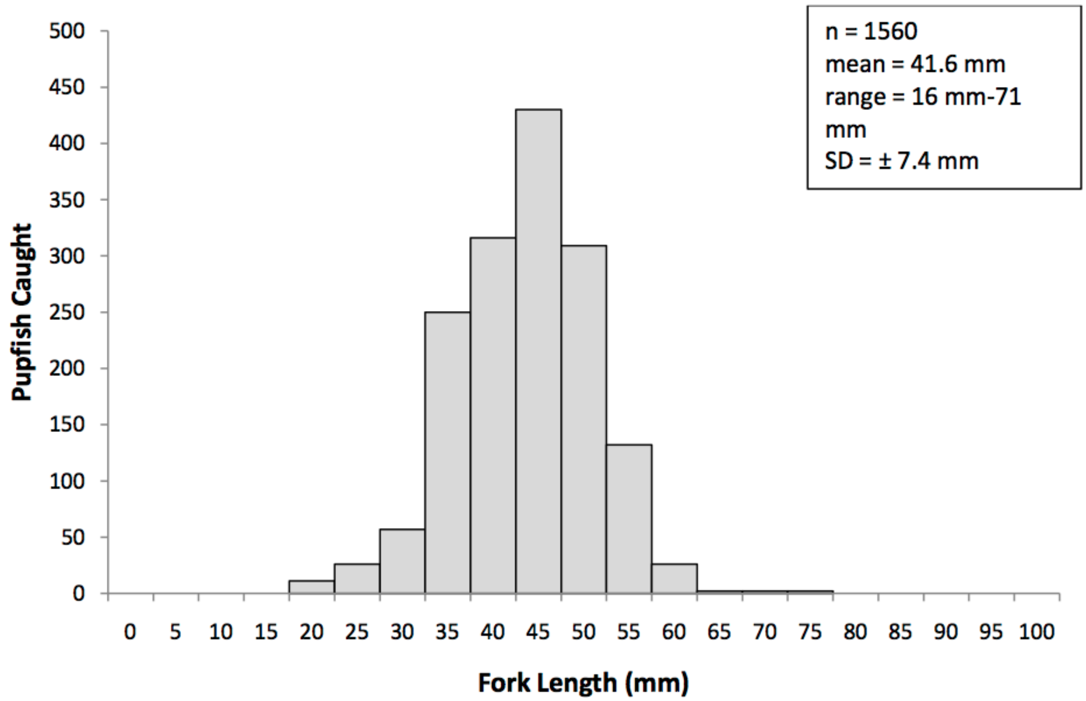


Figure 1. Length frequency of Amargosa River pupfish (*Cyprinodon nevadensis amargosae*) captured during a summer, 2010 survey of the Amargosa River Canyon, California. Figure from Scoppettone et al. 2011.

Life History: The life history of Amargosa River pupfish is similar to that of Saratoga Springs pupfish. Being riverine, however, its reproductive strategies differ from spring-dwelling pupfishes. Males do not establish and defend territories in leks, as do males of spring-dwelling subspecies. Instead, they are group spawners (Kodric-Brown 1981). Spawning may take place in the center of the group but, most often, a reproductive male will direct a receptive female to the periphery of the group to spawn. Highest densities and peak breeding season occur during summer, when water temperatures are higher and food is abundant (Kodric-Brown 1977). However, breeding may occur year-round in thermally stable habitat.

Pupfish feed primarily on blue-green cyanobacteria and algae, but they also feed seasonally on lesser quantities of small invertebrates, mostly chironomid larvae, ostracods, and copepods (Naiman 1975, 1976). They can also be effective predators of mosquito larvae in heavy vegetation (Danielsen 1968). They forage continuously throughout the day but are less active at night.

Few Amargosa River pupfish live longer than a year in the wild and they rarely exceed 65 mm FL; most are less than 50 mm FL (Figure 1). However, these pupfish can reach sexual maturity in a few months at 30 mm FL, so are capable of multiple generations in a year (Moyle 2002).

Habitat Requirements: The Amargosa River is an intermittent desert stream. For most of its course, the river flows underground except after infrequent rain events. Pupfish habitat in the lower Amargosa River is divided into two distinct reaches of perennial flow separated by 16 km of dry riverbed. The upstream reach, near Tecopa, is itself divided into two sections, the first being characterized by broad marshes fed by hot springs and the second by a narrow, steep-sided canyon where the river is only 2 m wide but reaches depths of 2.5 m. In the canyon, flows are swift between pools and substrates consist of gravel and sand, with some boulder and rubble (Miller 1948, Williams et al. 1982). Pools are numerous, both within the river and on the flood plain, the largest being about 8 x 5 m. Substrates in pools are primarily mud and clay and shoreline vegetation is abundant. Gravel riffles are not preferred habitat (Williams et al. 1982). Pupfish seem to prefer depths between 10-35 cm (Williams et al. 1982). A 2010 BLM survey of Amargosa Canyon found that pupfish were most abundant in habitat reaches associated with native vegetation and scarce where salt cedar dominated the riparian corridor (Scoppottone et al. 2011). The water during this survey was clear and saline, with pH ranging from 8.2-8.7 and dissolved oxygen ranging from 7.3-11.6 mg l^{-1}. Total dissolved solids were fairly high and variable at 1,390-3,890 ppm.

In the Tecopa area, this subspecies also inhabits the torrid outflows of hot springs, habitats formerly occupied by *C. n. calidae* (extinct). One of the most unusual habitats is Tecopa Bore, an outflow of an artesian well. Water temperature at the head of the bore is 47.5˚C; in winter it can cool to nearly freezing only 1 km downstream. In the bore, pupfish tend to congregate at water temperatures near their thermal maxima of 42˚C because cyanobacteria abundance is greater in the warmer water (J.H. Brown 1971). Pupfish follow cooler water blown upstream by winds into ungrazed areas otherwise outside their thermal tolerances. When the wind dies, pupfish caught in hot water (> 42˚C) will die unless they are washed downstream into cooler water.

The downstream reach is in Death Valley National Park at an elevation of 33 m (Miller 1948). Here, the river bottom consists of fine silt, clay, mud, and sand; there are no macrophyte beds. The current is moderate to swift between pools, which are 0.75-1.25 m deep. Water temperature varies seasonally from 10 to 38 °C, except during severe winters when temperatures may approach freezing. Younger fish tolerate higher water temperatures than adults (Shrode 1975) and are commonly found in the warmer, shallower (ca. 5 cm) water close to shore. Habitats along the shore may serve as refuge from predation or competition for food (Miller 1948). This reach is characterized by large diel variation in water temperature and by vertical temperature stratification.

Distribution: *C. n. amargosae* is the most widely distributed subspecies of *C. nevadensis*, inhabiting two perennial sections of the lower Amargosa River and Tecopa Bore, Inyo County. The upper section begins upstream of Tecopa and flows through Amargosa Canyon for about 11 km until it approaches Sperry, where it dries, except after rare periods of heavy rainfall upstream. The second, lower, section flows through Death Valley northwest of Saratoga Springs, approximately 32 km downstream of Sperry, and continues for about 3 km. Differences in meristic characteristics between the two populations suggest that they are effectively isolated from each other (Miller 1948), except, perhaps, in times of floods. In 1940, R. R. Miller planted 350 Amargosa pupfish in River Springs, Adobe Valley, Mono County. This population was extant and flourishing; however, because *C. s. salinus* was planted at the same time, studies are needed to determine whether, and to what extent, hybridization between the two taxa may have occurred (E. Pister, CDFW, pers. comm. 1999).

Trends in Abundance: The Amargosa River pupfish is the most widespread of any *C. nevadensis* subspecies and is fairly common in the lower Amargosa River, particularly around Tecopa (unpublished data, the authors) and in Amargosa Canyon (Scoppottone et al. 2011). Pupfish populations in Amargosa Canyon appear relatively stable between surveys in 1982 (Williams et al. 1982) and 2010 (Scoppettone et al. 2011). The Amargosa River pupfish also occurs in an isolated downstream reach of river in Death Valley National Park. Historic abundance records are lacking but Amargosa River pupfish may be less abundant now than formerly because water diversions have long reduced Amargosa River flows.

Nature and Degree of Threats: The major threat to Amargosa River pupfish is potential dewatering of its unique habitats, the Amargosa River and tributaries, by a combination of local surface water diversions and groundwater withdrawals. The Amargosa Aquifer, which supplies springs in Ash Meadows, Nevada and the Amargosa River, receives much of its recharge flow from areas on the northern and northeastern slopes of nearby Spring Mountains (Riggs and Deacon 2004), but is also dependent on regional groundwater movement through large, ancient aquifers that extend into western Utah and central Nevada (Dettinger and Cayan 1995, Deacon et al. 2007).

Agriculture. In the late 1970s and early 1980s, substantial groundwater development reduced spring discharge in Ash Meadows, causing a dramatic decline in water levels at Devils Hole, habitat of the endangered Devils Hole pupfish, *C. diabolis*. After a Supreme Court decision halted pumping, spring discharge in Ash Meadows recovered and the groundwater table rose steadily until 1987. However, a slow decline began in 1988 and continues to the present (Riggs and Deacon 2004). Analysis by Bedinger and Harrill (2006) indicates that the decline is not correlated to climate but, instead, is due to agricultural water withdrawal from as far as 30 km away.

Grazing. Livestock grazing is prohibited within Death Valley National Park but may be an issue on private lands around Tecopa. Even small herds can significantly alter habitat required by pupfish by trampling springs or eliminating native riparian vegetation.

Urbanization. In 2004, the Southern Nevada Water Authority (SNWA) proposed to mine large quantities of water from several different valleys that lie within the Ash Meadows groundwater basin (Breen 2004, Southern Nevada Water Authority 2004, Vogel 2004). If the Amargosa region withdrawals continue to increase and if Las Vegas proceeds with its planned withdrawals, it is highly likely that the springs which feed the Amargosa River could be greatly reduced or even dry completely, especially during drought periods (Deacon 2011). As the increasing human population of Tecopa and the upper Amargosa Valley seek protection from episodic flood events, potential flood control modifications to the basin can pose a threat to Amargosa River aquatic habitats.

Recreation. Unrestricted public access to the entire limited range of Amargosa River pupfish increases risks of introduction of alien species, contamination, and novel pathogens. Off-road vehicles can damage stream habitats and negatively affect water quality.

Alien species. Western mosquitofish (*Gambusia affinis*), associated with declines of other pupfish species, are abundant in Amargosa Canyon, yet Amargosa River pupfish appear able to coexist with them (Williams et al. 1982). Flash floods periodically reduce mosquitofish populations, to the advantage of pupfish. If flood control measures were implemented upstream, this natural purging of exotics would be reduced. Maintaining natural disturbance regimes has been shown to be of prime importance for the persistence of desert aquatic ecosystems (Kodric-Brown and Brown 2007) and every effort should be made to ensure that periodic flood flows continue in the Amargosa River. The possibility of additional introductions of alien fishes into the Amargosa River also exists. The Amargosa River is highly accessible to the public and, as such, there is an increased threat of the introduction of competitors, predators, and pathogens. Of particular concern are largemouth bass (*Micropterus salmoides),* which are easily moved, handle warm temperatures well, and are voracious predators. Even a few individuals can quickly consume a pupfish population. Pupfish have no effective defense against such predation (Moyle 2002).

Other alien species also represent a threat. Although historic data are lacking, it is assumed that native fishes were likely found in greater abundance in the Amargosa River prior to the invasion of saltcedar (*Tamarisk*), crayfish and mosquitofish, all of which have been found to negatively impact native fish populations (Scoppettone et al. 2011). Similar to many other desert aquatic habitats in the American Southwest, saltcedar is proliferating and altering habitats in Amargosa Canyon. Historically, stochastic events such as fire and flood periodically removed substantial amounts of riparian vegetation, keeping the stream channel open and dynamic (Benda et al. 2003; Kozlowski et al. 2010). Today, these same processes serve as agents for the spread of saltcedar (Wiesenborn 1996), threatening to form a saltcedar monoculture throughout the floodplain (Scoppettone et al. 2011). Because saltcedar has a substantially greater water demand than native vegetation, increases in saltcedar density in the riparian zone lead to corresponding increases in water lost to transpiration (Duncan and McDaniel 1998).

	Rating	Explanation
Major dams	n/a	
Agriculture	High	Ground water pumping and surface diversions threaten Amargosa River base flows
Grazing	Medium	Limited but even small herds can damage sensitive aquatic desert habitats
Rural residential	Low	Local surface flow diversion and groundwater pumping has the potential to reduce Amargosa River flows
Urbanization	High	Groundwater pumping by the city of Las Vegas has the potential to intercept aquifer water flowing to Amargosa River
	n/a	
Instream mining	n/a	
Mining	Low	Present in region but no known impact
Transportation	Low	Roads run along or across riparian or instream habitats in some areas, potentially increasing sediment and pollutant input
Logging	n/a	
Fire	n/a	
Estuary alteration	n/a	
Recreation	Medium	Entire limited range publicly accessible; off-road vehicle use popular in area
Harvest	n/a	
Hatcheries	n/a	
Alien species	High	Alien species are diverse and widespread

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

Effects of Climate Change: Isolated desert springs and rivers fed by subsurface flow systems are precarious ecosystems that are particularly vulnerable to geologic and anthropogenic disruption. Climate change, therefore, poses a direct threat to the continued existence of Amargosa River pupfish. Fed by rain and snow melt at high elevation in the desert mountain ranges, desert aquifers in the Death Valley region will likely receive less recharge as the region warms (Riggs and Deacon 2002). This decline in regional water supply will be compounded by growing human demand for water in southern Nevada, which will only increase as the climate gets hotter and more arid. Although well-adapted to extreme salinity and temperature fluctuations characterized by its desert habitats, the Amargosa River pupfish exists at the limit of it thermal tolerances and is, therefore, remarkably vulnerable to small increases in temperature. Moyle et al. (2013) rated this pupfish as "highly vulnerable" to extinction as the result of the added impacts of climate change.

Status Determination Score = 2.3 – High Concern (see Methods section Table 2). The Amargosa River pupfish may be at risk of extinction because of its limited distribution in an extreme environment. Its status could change quickly if river flows are further reduced or new non-native species invade (Tables 2, 3). The Amargosa River pupfish is considered a Sensitive Species by the Bureau of Land Management, while the American Fisheries Society lists it as Vulnerable because of its limited distribution and threats to habitat (Jelks et al. 2008). NatureServe ranks this subspecies as Critically Imperiled (G2T1, http://www.natureserve.org/publications/NEscor2006.pdf). The Superintendent of Death Valley National Monument once recommended that Amargosa River pupfish be listed as a threatened species (E. L. Rothfuss, Superintendent of Death Valley National Monument, letter to B. Bolster of CDFG, May 27, 1992). This recommendation was endorsed by the Desert Fishes Council (E. P. Pister, pers. comm. 2008).

Metric	Score	Justification
Area occupied	1	Occupies only three areas in one watershed in its native range
Estimated adult abundance	3	Fluctuates highly with season and flow
Intervention dependence	3	Requires protection of habitats and water sources
Tolerance	3	Although remarkably adapted to high temperatures and salinities, they exist at the edge of their tolerances
Genetic risk	3	Flood waters likely connect Amargosa River populations periodically
Anthropogenic threats	1	Groundwater pumping and alien species could change status rapidly (see Table 1)
Climate change	2	Long-term natural and anthropogenic reductions in aquifer discharges plus increases in temperature will affect viability
Average	2.3	16/7
Certainty (1-4)	3	Well-documented biology

Table 3. Metrics for determining the status of Amargosa River pupfish, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: Efforts should be made to maintain natural flows in the Amargosa River, including periodic flood flows that reduce populations of introduced fishes and the abundance and distribution of invasive riparian vegetation. Management strategies should focus on protecing populations in both the upstream segment (Tecopa area and Amargosa Canyon) and the downstream segment (Death Valley) to maintain genetic diversity. Fortunately, the Bureau of Land Management has designated 21,552 acres surrounding Amargosa River from just south of Shoshone to Sperry as an Area of Critical Environmental Concern. In addition, The Nature Conservancy has acquired much of the canyon not administered by the BLM. Fences and barriers need to be properly maintained since vehicle trespass is a common problem. The downstream section in Death Valley is managed by the National Park Service but is dependent on water availability from upstream.

The greatest management concern is current and likely increasing levels of water removal from the aquifer that feeds the river. The U.S. Supreme Court decision (United States v. Cappaert 1977), which protected Devils Hole pupfish from water withdrawals, set a precedent for the extension of federal water rights from surface waters to include groundwater. Known as the Winters Doctrine (established in Winters vs. United States 1908), the ruling states that when the federal government reserves land, such as Death Valley National Park, by implication, it also reserves sufficient water rights to accomplish the purposes of that reservation. In the case of the Devils Hole ruling, the "purposes of the reservation" was the continued existence of the Devils Hole Pupfish. However, the application of the Winters Doctrine to ground water resources on a regional basis remains uncertain. The proposed, massive, groundwater pumping by Las Vegas should be rigorously evaluated to determine whether or not the Amargosa River depends on the aquifer and how surface flows would be reduced by pumping. Protection of the pupfish could, thus, help to protect an entire unique desert ecosystem.

Given the uncertainties of persistent flow in the Amargosa River, a contingency plan should be developed that would include the identification of habitats or facilities to temporarily hold pupfish from both upstream and downstream populations in the event population loss appears imminent (captive rearing and/or establishment of one or more refuge populations).

Figure 2. Distribution of Amargosa River pupfish, *Cyprinodon nevadensis amargosae*, in the lower Amargosa River, California.

SHOSHONE PUPFISH
Cyprinodon nevadensis shoshone (Miller)

Status: Critical Concern. The Shoshone pupfish faces extinction in the wild, given that just one confirmed population exists in a small, tenuous, artificial habitat.

Description: The Shoshone pupfish is a subspecies of the Amargosa pupfish (*Cyprinodon nevadensis).* All *C. nevadensis* subspecies are deep-bodied (especially in reproductive males), with total lengths that rarely exceed 50 mm. The head is blunt and slopes steeply to a small, terminal, oblique mouth. There is one row of tricuspid teeth on each jaw, with the central cusps being truncated or pointed. The morphology of the Shoshone subspecies is similar to the Saratoga Springs pupfish, *C. n. nevadensis*. However, it is characterized by larger scales and a somewhat narrower, more slender body. It also has fewer pelvic fin rays and scales than other subspecies of *C. nevadensis* (See Table 1). Reproductive males in breeding colors are bright blue with a black band at the posterior edge of the caudal fin. Reproductive females are drab olive-brown and develop 6-10 vertical bars along the sides which are often indistinct. An ocellus (eyespot) is typically present on the posterior base of the dorsal fin of females.

Taxonomic Relationships: The fossil record and past geologic events suggest that the western *Cyprinodon* species differentiated relatively recently, with most of this differentiation occurring during the early to mid-Pleistocene (Miller 1981). As the numerous large pluvial lakes in the Great Basin shrank after the Pleistocene, remnant populations of pupfish survived in isolation, leading to subspecific differentiation within *C. nevadensis*. Speciation of the Devils Hole pupfish (*C. diabolis*), which, together with *C. radiosus*, *C. nevadensis* and *C. salinus*, constitutes the *C. nevadensis* species complex, possibly occurred less than 20,000 years ago (Echelle and Echelle 1993). Further differentiation may have occurred in the last 10,000 years, following the final recession of pluvial waters.

C. nevadensis was first described from Saratoga Springs (now in Death Valley National Park) by Eigenmann and Eigenmann (1889). Following the initial description, the species was lumped with the desert pupfish (*C. macularius)* until Miller (1943) separated it again. Miller (1948) described six subspecies of *C. nevadensis*, four of which occur in California (The Saratoga Springs pupfish (*C. n. nevadensis)*, the Amargosa River pupfish (*C. n. amargosae)*, the Shoshone pupfish (*C. n. shoshone)*, the Tecopa pupfish (*C. n. calidae*) and two in Nevada (The Ash Meadows pupfish (*C. n. mionectes)* and the Warm Springs pupfish (*C. n. pectoralis*). *C. n. calidae* is now extinct (Moyle 2002).

At the time of their formal description, Shoshone pupfish were understood to exhibit a high degree of morphological variation between the headspring and downstream areas. Miller (1948) speculated on the origins of this variation and recommended but did not pursue studies to separate potential genetic and environmental causes. No pupfish have been observed below Old Highway 127 since before 2010, so the extant pupfish at Shoshone probably all descended from the captive breeding programs and reintroduction (S. Parmenter, CDFW, pers. comm. 2013) described in the Trends in Abundance section of this account.

Measure/ Count	*C. n. amargosae* male	*C. n. amargosae* ALL	*C. n. amargosae* female	*C. n. nevadensis* male	*C. n. nevadensis* ALL	*C. n. nevadensis* female	*C. n. shoshone* male	*C. n. shoshone* ALL	*C. n. shoshone* female
Standard length (mm)		36			40			34	
*Body width	256		265	274		269	231		229
*Head length		305			312			307	
*Head depth	330		304	367		343	331		311
*Head width	240		259	257		256	233		231
*Snout length		101			97			89	
*Mouth width		117			115			114	
*Mandible length		198			95			93	
*Anal origin to caudle base	338		346	394		362	371		355
*Caudle peduncle length	264		237	277		253	263		251
*Anal fin base length	116		105	111		105	108		101
*Anal fin length	330		304	227		195	217		190
*Pelvic fin length	98		89	95		87	90		77
Anal fin ray count		10			10			10	
Dorsal fin ray count		10			10			10	
Pelvic fin ray count		6			6			4	
Pectoral fin ray count		16			16			16	
Caudal fin ray count		18			17			18	
Lateral line scales		26			26			26	
Predorsal scale count		19			18			18	
Dorsal fin to pelvic fin scale count		11			10			9	
Caudal peduncle circumference scale count		16			16			15	
Body circumference scale count		27			25			23	

*Expressed as percent of standard length x 1000.

Table 1. Comparative average morphometrics and meristics of three *Cyprinodon nevadensis* subspecies. Adapted from Miller (1948).

Life History: Shoshone pupfish exhibit many characteristics that adapt them to live in habitats with thermal and osmotic extremes (Miller 1981). The life-history characteristics of this subspecies, however, have not been studied in detail but are likely similar to the Saratoga Springs pupfish (*C. n. nevadensis*), for which the following characteristics are known. Optimal temperature for growth is 22˚C, with growth ceasing below 17˚C and above 32˚C. At optimal temperatures, growth is extremely rapid and fish reach sexual maturity within four to six weeks (Miller 1948). Such short generation time enables small populations to remain viable. Generation time varies among subspecies, with populations living in widely fluctuating environmental conditions exhibiting shorter generation times (Moyle 2002). Young adults (15-30 mm SL) of *C. nevadensis* usually constitute a majority of the biomass throughout the year (Naiman 1976). Highest densities and peak breeding season occur during summer, when water temperatures are higher and food is abundant (Kodric-Brown 1977). However, breeding may occur year-round in thermally stable habitat. Like the Saratoga Springs pupfish, the Shoshone pupfish presumably once bred year-round.

The Shoshone pupfish, like other spring-dwelling subspecies, exhibits reproductive behavior different from riverine forms (Kodric-Brown 1981). The males of spring-dwelling subspecies establish display territories. Both sexes are promiscuous and a single female may lay eggs with different males over time. The demersal eggs are sticky and, thus, adhere to substrates. Females may lay a few eggs each day (not necessarily on consecutive days) throughout the year. Territorial defense by males may confer some protection of eggs from predators, but otherwise parental investment is limited to gamete production (Kodric-Brown 1981).

Despite the Shoshone pupfish's ability to survive in a wide range of extreme conditions, their reproductive tolerance limits are likely narrow, 24-30° C, optimal being 28-29° C. The most sensitive phase of life history to thermal stress is oogenesis (Gerking 1981). Extreme temperatures affect egg production and egg viability (Shrode and Gerking 1977, Gerking 1981) and reproduction is greatly diminished at pH levels below 7 (Lee and Gerking 1980). Furthermore, reproductive performance does not improve despite generation-long acclimation to suboptimal temperatures (Gerking et al. 1979). Thus, any alterations to their habitat that would result in temperatures outside the range of their reproductive temperature optima would be potentially deleterious. Fertilized eggs, however, become resistant to environmental stresses within hours of being laid.

Shoshone pupfish, like other pupfishes, likely feed primarily on blue-green cyanobacteria and algae but will feed seasonally on small invertebrates, mostly chironomid larvae, ostracods, and copepods (Naiman 1975, 1976). Pupfishes forage continuously from sunrise to sunset and become inactive at night and have long convoluted guts characteristic of aquatic herbivores and teeth adapted for nipping (Moyle 2002).

Habitat Requirements: Historically, two holes in the upper portion of the Shoshone Springs province above the Old State Highway 127 provided velocity refuge for Shoshone pupfish from the channel's swift flows (Miller 1948). The larger, upper hole (known as Squaw Hole) was about 1 m in diameter and 0.75 m deep. The water was clear, with overhanging banks, and the pool bottom was muddy. Shoshone Spring water had lower salinity and boron content than other springs in the region but more calcium.

Miller (1948) noted the largest numbers of pupfish occurred on either side of Old Highway 127, where temperatures were cooler and more variable and channel slope was reduced. Since Miller's description in 1948, Shoshone Springs underwent severe alteration (Taylor et al. 1988, Castleberry et al. 1990). Notably, the present habitat for Shoshone pupfish also hosts one of 5 known populations of Sanchez's Springsnail (*Pyrgulopsis sanchezi*), *Tryonia variegata* (a regionally endemic springsnail), abundant non-native Red-rimmed Melania snails (*Melanoides tuberculata*); however, mosquitofish are absent (S. Parmenter, CDFW, pers. comm. 2013).

Distribution: The Shoshone pupfish was formerly found in Shoshone Spring and throughout its outlet creek in Inyo County (Miller 1948, Taylor et al. 1988). The spring source is at an elevation of 518 m, about 170 m above State Highway 127 on the east slope of a rocky lava hill. By approximately 50 years ago, most of the spring vents were enclosed and their water diverted to supply the town of Shoshone. In the 1980s, a small refuge pond was dug and supplied with water flowing from the last uncaptured spring vent. The refuge pond is approximately 5 m X 13 m, with banks vegetated by three-square bulrush (Scirpus americanus). In recent years, cattail infestation along the outflow ditch has been controlled and pupfish now utilize 135 meters of flowing channel, including a total of 5 pools between 5 and 50 m2. An additional isolated 20 m2 unconnected refuge pond is under construction, to be supplied by piped water from the enclosed spring vents. Plumbing has been installed to provide a backup water supply should flow fail from the single uncapped spring, which solely supplies the extant pupfish habitat. Shoshone pupfish are expected to colonize or be introduced into additional habitat, consisting of a former catfish aquaculture pond downstream of Old Highway 127, in the near future. Below Highway 127, the ouflow ditch provides perennial flow to a short segment of the Amargosa River. Mosquitofish occur in the floodplain, but not in the spring channel managed for pupfish (S. Parmenter, CDFW, pers. comm. 2013).

Trends in Abundance: The Shoshone pupfish was once considered to be extinct (Selby 1977, CDFG 1980) but was rediscovered in 1986 (Taylor et al. 1988). Although the pupfish was found in "large numbers" through the outflow creek in the summer of 1986 (Taylor et al. 1988), its numbers had dwindled to perhaps less than 20 individuals by 1988 (J. Williams, unpubl. data). The decline may have been precipitated by the invasion of western mosquitofish (*Gambusia affinis*) into the outflow creek. Taylor et al. (1988) hypothesized that Shoshone pupfish survived in very low numbers until conditions became more favorable, when the population expanded. The pupfish may have passed through a genetic bottleneck during the period of severely reduced population size.

Because of the lack of suitable habitat and the abundance of mosquitofish, most Shoshone pupfish were removed from the wild and small stocks of approximately 12 fish each were kept at the University of Nevada, Las Vegas (UNLV), and the University of California, Davis (UCD). In the 1990s, captive-raised individuals from both UCD and UNLV were introduced into the refuge pond (Swift et al. 1993). By May, 2002 Shoshone pupfish still persisted in the pond, which was found to be 100% overgrown with cattail and only a handful of fish were detected. Since then, the property owner and CDFW have cooperated to improve and increase the available habitat. Sampling with Gee traps has produced captures of over 200 individuals, but no formal population estimates have been made (S. Parmenter, CDFW, pers. comm. 2013). Fish were common in July, 2010 when three of the authors visited the site.

Nature and Degree of Threats: Shoshone pupfish survive today due to human intervention in an intensively managed refuge habitat. They are particularly threatened by their extremely limited distribution and potential for additional genetic impacts, should their population size again shrink due to reduced flows, habitat alteration, invasion of alien species or other factors.

Agriculture and urbanization. As discussed in the Saratoga Springs pupfish account, pumping of ancient aquifers to supply water for farming, industrial-scale solar development, and, increasingly, the City of Las Vegas, has the potential to eventually reduce or eliminate flows from Shoshone Spring, if it is connected to the pumped aquifers. For potential threats to the groundwater source of Shoshone Spring, see the Amargosa River pupfish account in this report.

Rural residential. The existence of Shoshone pupfish is entirely dependent on the maintenance of artificial habitats. The spring is privately owned and its water is used as a water supply for the town of Shoshone, as well as for the pupfish refuge. Despite the fact that the current owners of the site are dedicated to the preservation of this unique fish, the fact that Shoshone pupfish are now largely confined to a small, artificial, habitat means that they are extremely vulnerable to random acts of vandalism, to the introduction of other fishes and pathogens into their habitat, to the degradation of the spring habitat by colonization of cattail, which can completely crowd out open water habitat, and to the potential for changes in land ownership and decreased commitment to their preservation.

Alien species. Introduced mosquitofish undoubtedly preyed on pupfish eggs and young and were the immediate factor threatening the pupfish with extinction before they were rescued. While no mosquitofish were observed in the refuge spring, they may prevent the re-establishment of pupfish in the outflow ditch, even if water quality issues were corrected (e.g. chlorinated water from the swimming pool).

	Rating	Explanation
Major dams	n/a	
Agriculture	Medium	Regional ground water pumping for agriculture, even at great distances, has potential to affect flow in Shoshone Spring
Grazing	n/a	
Rural residential	Critical	Diversion of Shoshone Springs for use as a water supply for the town of Shoshone nearly caused extinction
Urbanization	Low	Regional ground water pumping to support Las Vegas and to service industrial-scale solar farms has potential to affect flow in Shoshone Spring
Instream mining	n/a	
Mining	Low	Occurs in area but no known impact
Transportation	Low	Spring outlet affected by road
Logging	n/a	
Fire	n/a	
Estuary alteration	n/a	
Recreation	Medium	Public access to the only known habitat increases risk of pollution, as well as introduction of alien species and pathogens
Harvest	n/a	
Hatcheries	n/a	
Alien species	High	In the past, interactions with mosquitofish was major factor in declines, but the greatest threat is now degradation of open water habitat by cattails

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

Effects of Climate Change: As one might expect of oasis species, climate change poses a direct threat to the continued existence of desert pupfish species, including Shoshone pupfish. Although all pupfishes of the American southwest are remarkably well adapted to the wide range of salinity and temperature found in their arid range, they are also remarkably vulnerable to change. Isolated desert springs and rivers fed by subsurface flow systems are precarious ecosystems, vulnerable to geologic and anthropogenic disruption. Fed by rain and snow melt at high elevation in the desert mountain ranges (Riggs and Deacon 2004), desert aquifers in the Death Valley region will likely receive less recharge as the regions warms. This will be compounded by the growing human demand for water in southern Nevada, which will invariably increase as the climate becomes hotter and drier. Moyle et al. (2013) considered Shoshone pupfish as "highly vulnerable" to extinction as the result of climate change, but other factors are more likely to drive it to extinction first.

Status Determination Score = 1.1 – Critical Concern (see Methods section, Table 2). The only purported population of Shoshone pupfish exists in a small, artificial, habitat and is one of the most endangered fishes in California (Table 2). It is possible that this population is comprised of hybrids, in which case the Shoshone pupfish may already be extinct. Jelks et al. (2008) list it as Endangered, while NatureServe considers it to be "Imperiled" with a high risk of extinction due to very restricted range.

It remains unclear if the fish currently in Shoshone Spring are distinct from Amargosa River pupfish (*C. n. amargosae*). If genetic analysis indicates that the two populations are the same, then the Shoshone pupfish is either extinct or was never genetically distinct to begin with. If the results, instead, confirm a distinct genetic lineage, then the Shoshone pupfish will also bear the dubious distinction of being one of the most threatened fish in California.

Metric	Score	Justification
Area occupied	1	Confined to one heavily modified springbrook
Estimated adult abundance	1	Population extremely small
Intervention dependence	1	Additional refuge populations must be established
Tolerance	2	Although remarkably adapted to high temperatures and salinities, outside their refuge they encounter conditions at the very edge of their reproductive tolerances
Genetic risk	1	Single, small, population is vulnerable to genetic bottlenecking and/or drift
Climate change	1	Threatened by increases in temperature and reductions of flow resultant from limited recharge of the aquifer, compounded by increasing human water demand
Anthropogenic threats	1	See Table 2
Average	1.1	8/7
Certainty (1-4)	4	Well studied

Table 3. Metrics for determining the status of Shoshone pupfish, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: Two artificial pools were created in the headsprings area of Shoshone Spring during 1988 and stocked with Shoshone pupfish from refuge populations at UCD and UNLV. The pools were subsequently enlarged into a single pool that serves as the principal refuge for the species. The headsprings area should be managed as a preserve and the refuge pool monitored frequently (at least monthly) to establish baseline water conditions and check for presence of alien fishes. In the past, cattails (*Typha* spp.) have threatened to completely take over the refuge pool, which would eliminate open water habitat (S. Parmenter, CDFW, pers. comm. 2009). Cattail control entails laborious hand removal. In an effort to prevent cattail reestablishment, S. Parmenter (CDFW) has cultivated naturally occurring three-square bulrush (*Schoenoplectus americanus*). Once established, the bulrush competes with and prevents cattail reinvasion and has the added benefit of not colonizing deeper water, thereby stabilizing open water habitat. The initial phase of this “gardening” is labor intensive, but is critical to the maintenance of preferred Shoshone pupfish habitat and, therefore, must continue. Funds should be permanently allocated for this endeavor.

The concrete ditch between the Old State Highway and State Highway 127 has experienced at least two fish kills due to chlorinated outflow from the swimming pool (D. Castleberry and B. Bolster, pers. comms.). Chlorinated discharges, therefore, must be avoided. Conditions in the outflow creek between State Highway 127 and the Amargosa River should be monitored to determine if it is suitable for pupfish reintroduction.

A groundwater study is also needed to identify the source aquifer to guard against overexploitation.

Figure 1. Distribution of Shoshone pupfish, *Cyprinodon nevadensis shoshone* (Miller), in California.

SALT CREEK PUPFISH
***Cyprinodon salinus salinus* (Miller)**

Status: High Concern. While Salt Creek pupfish seem fairly secure, given their restricted distribution in the protected lands of Death Valley National Park, the threat of extinction is elevated due to their isolation and dependence upon a single water source.

Description: Salt Creek pupfish are slender bodied compared to most other pupfishes. They can reach 63 mm but rarely get longer than 50mm TL. The scales are small, oval to circular in shape, with reticulated interspaces between the circuli; they are intermediate to *Cyprinodon nevadensis* and *C. macularius* in the number of radii (15-22, usually 18). There are 28-29 scales in the lateral series. The preorbital region of the head lacks scales. Lateral line pores, especially the preopercular pores, are well developed. The mouth is slightly supraterminal and has tricuspid teeth with prominent median ridges. The dorsal fin is set behind the midpoint of the body. The pelvic fins are reduced and may even be absent. There are 8-11 dorsal fin rays (usually 9-10); 9-11 anal fin rays (usually 10); 14-17 pectoral fin rays (usually 15-16); 15-19 caudal fin rays (usually 16-17); 0-6 pelvic fin rays (the pelvic fin may be absent); gill rakers number 18-22 (usually 19-21) and are shorter and more compressed than in other pupfishes.

The back of reproductive males is purple and the sides are deep blue with 5-8 broad black bands that may be continuous or interrupted. The caudal fin has a prominent black terminal band and the anterior profile of males is noticeably arched (Miller 1943b). Females have 4-8 vertical lateral bars that are, except during spawning, less intense than the barring pattern of males. Females are more slender bodied than males and less conspicuously colored, being brownish with a silvery sheen.

Taxonomic Relationships: The Salt Creek pupfish was first described by Miller (1943b) from Salt Creek in Death Valley. Similarity of morphological characteristics to the Amargosa pupfish, such as reduced or absent pelvic fins, posterior position of the dorsal fin, short head, small eyes, and low fin-ray counts, indicate the two species are closely related (Miller 1943b). Mitochondrial DNA analysis confirms this close relationship but suggests that they began diverging before the desiccation of Lake Manly in the late Pleistocene (Echelle and Dowling 1992). *Cyprinodon salinus* is divided into two subspecies, *C. s. salinus* from Salt Creek and *C. s. milleri* from Cottonball Marsh, into which Salt Creek overflows.

Life History: While Salt Creek pupfish usually live one year or less, they become sexually mature at 30-40mm TL and have a generation time of 2-3 months, enabling them to reproduce several times a year (Sigler and Sigler 1987). Such a short generation time allows large populations to build rapidly during favorable high water conditions, resulting in colonization of areas beyond the limits of permanent water. During these periods, pupfish numbers have been estimated in the millions (Miller 1943b). While this estimate is likely high, densities of 527 fish per square meter have been measured (Sada and Deacon 1995). When flood waters recede, many fish are trapped in side pools or on drying flood plains and perish. Flash flooding also results in population losses, as fish become isolated in downstream pools that eventually dry (Williams and Bolster 1989).

Like other desert pupfishes, Salt Creek pupfish largely subsist on cyanobacteria and algae but will also feed on aquatic insects, crustaceans, and snails that share their habitats (Moyle 2002). Reproductive behavior and other aspects of their life history are similar to the riverine Amargosa pupfish (*Cyprinodon nevadensis ssp.)*.

Habitat Requirements: Salt Creek is located 49 m below sea level in North America's driest desert, where summer air temperatures can be greater than 50˚C, so it is among the most severe habitats inhabited by any fish. Beginning as a series of seepages on the floor of Death Valley, upper Salt Creek contains surface water only during winter and spring. This upper section is fishless and traverses Mesquite Flat for 2 km before abruptly entering a narrow, shallow canyon. Augmented by inflow from Mclean Springs, flow within the canyon provides 1.5 km of year-round habitat for pupfish. Within the canyon, the stream channel incised 3-7 m into the alkaline mud substrate and created a series of large (10 x 25 m by 2 m deep), interconnected pools which form the core of Salt Creek pupfish habitat. Canyon pools contain heavy growths of aquatic plants and are protected by overhanging salt grass, pickleweed, and saltbush, making them ideal refuges for pupfish. Below this canyon section the stream becomes shallow and exposed, quickly disappearing into the floor of Death Valley during normal water years. During periods of high flow, when surface water in Salt Creek expands downstream from the canyon, Salt Creek pupfish may inhabit as much as 5 km of stream habitat. However, most fish in this reach perish as high waters on the floodplain recede and downstream pools dry.

Water temperatures in Salt Creek fluctuate from near freezing in the winter to greater than 40°C in the summer. However, the temperature in deeper pools seldom exceeds 28°C and may provide temperature refuges, especially for reproduction. Salinity is also high, in summer approaching that of sea water (LaBounty and Deacon 1972). The levels of boron (39 ppm) and total dissolved solids (23,600 ppm) are remarkably high for any inland fish habitat (Miller 1943b).

Given the extreme conditions found in Salt Creek, it is not surprising that these fish are physiologically adapted to tolerate wide temperature and salinity fluctuations. Under experimental conditions, Salt Creek pupfish tolerate temperatures of 38˚C and can survive short-term exposure up to 43°C. They also survived salinities of up to 67 ppt, but died at 79 ppt (LaBounty and Deacon 1972).

Distribution: The Salt Creek pupfish is naturally restricted to Salt Creek, Death Valley National Park, Inyo County. However, they were introduced into Soda Lake, San Bernardino County, and into River Springs, Mono County (Miller 1968). The Soda Lake population no longer exists and the pupfish in River Springs have apparently hybridized with *Cyprinodon nevadensis amargosae,* which were introduced into the same spring. Thus, genetically pure *C. s. salinus* are restricted to Salt Creek and its associated marshes, about 1.5-6 km below McLean Springs. Their actual range varies by water year (Swift et al. 1993).

Trends in Abundance: The numbers of Salt Creek pupfish fluctuate widely based on season and water year, but there is no indication that they are less abundant now than they were in the past.

Nature and Degree of Threats: In spite of protections afforded by Salt Creek's relatively pristine state and location within Death Valley National Park, the Salt Creek pupfish population still faces potential threats, especially given their extremely restricted distribution. The springs which feed Salt Creek may be connected to the aquifer that provides water to Furnace Creek, the center of Death Valley's tourism, so potential for excessive pumping and reduced stream flow exists. Public access to their only known habitat increases risk of contamination and introduction of exotic species and novel pathogens, although these risks are small given the severity of the environment.

	Rating	Explanation
Major dams	n/a	
Agriculture	n/a	
Grazing	n/a	
Rural Residential	Low	Groundwater pumping by Furnace Creek, the town center of Death Valley, could reduce base flows in Salt Creek
Urbanization	n/a	
Instream mining	n/a	
Mining	Low	Present in basin but no known impact
Transportation	n/a	
Logging	n/a	
Fire	n/a	
Estuary alteration	n/a	
Recreation	Low	Limited potential for source of invasive species or pathogens
Harvest	n/a	
Hatcheries	n/a	
Alien species	Low	Harsh conditions favor Salt Creek pupfish and limit opportunities for colonization by alien species

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

Effects of Climate Change: The predicted effects of climate change pose a particular threat to the continued existence of Salt Creek pupfish and their unique desert habitat. As an oasis species, Salt Creek pupfish are remarkably well adapted to widely varying salinity and temperature characterized by their habitat; however, they also exist at the edge of their thermal tolerances, so slight increases in water temperature during summer could impact reproduction and survival.

Isolated desert springs and rivers fed by subsurface flow systems are precarious ecosystems, vulnerable to geologic and anthropogenic disruption. Fed by rain and snow melt at high elevations in desert mountain ranges (Riggs and Deacon 2002), desert aquifers in the Death Valley region will likely receive less recharge as the regions warms. For the reasons described above, Moyle et al. (2013) considered Salt Creek pupfish to be "critically vulnerable" to extinction as the result of climate change effects.

Status Determination Score = 2.7 – High Concern (see Methods section Table 2). While this pupfish seems fairly secure in its isolated setting within a national park, its single, isolated, population is particularly vulnerable to stochastic events and anthropogenic threats, especially aquifer depletion, along with predicted impacts associated with climate change (Table 2).

Metric	Score	Justification
Area occupied	1	Confined to Salt Creek
Estimated adult abundance	3	Population fluctuates widely
Intervention dependence	4	Protection afforded within Death Valley National Park
Tolerance	2	Adapted to extreme temperatures and salinities that would kill most other fishes but exist at the very edge of their tolerances; could be threatened by small changes, esp. increased water temperatures or decreased surface flow
Genetic risk	2	Frequent population fluctuations increase the risk of genetic bottlenecks and may reduce heterozygosity
Climate change	1	Threatened by potential water temperature increases and reduction in aquifer recharge
Anthropogenic threats	5	See Table 1
Average	2.6	18/7
Certainty (1-4)	3	Population has been studied in the past; no recent data available

Table 2. Metrics for determining the status of the Salt Creek pupfish, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for explanation of scoring procedures.

In 1992, E. L. Rothfuss, then superintendent of Death Valley National Monument, recommended that the Salt Creek pupfish be listed as a threatened species for the following reasons: "During spring the population expands and disperses throughout the braided stream system, but by mid-summer habitat has contracted due to seasonal evaporation of water, and the fish are confined to several source pools south of MacLean Spring. We believe listing is warranted in light of the restricted extent of the habitat during this portion of the year. While restricted to these pools, the fish are vulnerable to intentional contamination, introduction of exotic competitors, and stochastic events," (letter to B. Bolster, CDFW, June 1, 1992). These threats have not changed in the intervening 20+ years. The Salt Creek pupfish is considered to be Vulnerable by the American Fisheries Society and “Critically Imperiled” by NatureServe (Jelks et al. 2008). The Cottonball Marsh subspecies is listed as Threatened by the State of California.

Management Recommendations: Present management by Death Valley National Park is adequate to maintain Salt Creek pupfish populations, as well as the entire unique Salt Creek ecosystem. However, potential refuge sites should be located and contingency plans developed in case reductions in Salt Creek pupfish abundance drop below minimum viable levels. Given their restricted range and limited perennial habitat, Salt Creek pupfish should remain a Species of Special Concern with a status review every 5-10 years to document abundance trends and habitat quality and quantity.

Figure 1. Distribution of Salt Creek pupfish, *Cyprinodon salinus salinus* (Miller), in Salt Creek, Inyo County, California.

BIGEYE MARBLED SCULPIN
***Cottus klamathensis macrops* (Rutter)**

Status: Moderate Concern. There is no immediate extinction risk for bigeye marbled sculpin. However, populations may have experienced long-term declines and are subject to the negative effects of fragmentation and intensive land use (agriculture, grazing, logging) within their limited range.

Description: All subspecies of marbled sculpin (*Cottus klamathensis)* have large, dorsally flattened heads with two chin pores; large, fan-like pectoral fins with four elements; and small pelvic fins that are positioned ventrally between the pectorals (Moyle 2002). Marbled sculpin are distinguished from other *Cottus* species by 7-8 dorsal fin spines, joined dorsal fins, an incomplete lateral line with 15-28 pores, and relatively smooth skin (Daniels and Moyle 1984). A few prickles can sometimes be found below the lateral line. They also lack palatine teeth and have only one preopercular spine. Fin ray counts are: 18-22 in the second dorsal fin, 13-15 in the anal fin, 14-16 in the pectoral fin, and 11-12 (principal rays) in the caudal fin (Moyle 2002). All other sculpin species in California possess a split dorsal fin and more than 7 dorsal spines. Marbled sculpin are generally green-hued with a dark circular spot at the posterior end of the dorsal fin and alternating dark and light spots on the pectoral fin rays. Fish from the Klamath River are generally lighter and more marbled than those from the Pit River (Moyle 2002). Other marbled sculpin characteristics include: a wide interorbital region, a wide head and blunt snout, a maxillary rarely extending beyond the anterior half of the eye, and unjoined preoperculer mandibular canals, but these characteristics are shared with one or more other species (Daniels and Moyle 1984). The subspecies *C. klamathensis macrops* is distinguished from other marbled sculpins by having few (if any) axillary prickles, a short preopercular spine (<1 percent of SL), a large orbit diameter, and a long predorsal length (Daniels and Moyle 1984). They tend to be rather plain in patterning with relatively inconspicuous barring on the body and fins.

Taxonomic Relationships: *Cottus klamathensis* was first described by Gilbert (1897) from the Klamath River system. Rutter (1908) then described *Cottus macrops* from the Fall River, a large tributary to the Pit River, and noted that it closely resembled *C. klamathensis*. Robins and Miller (1957), upon review of specimens and recent collections, concluded that the two species were not sufficiently different to warrant separate species designations and considered *C. macrops* synonymous with *C. klamathensis*. Daniels and Moyle (1984), however, on the basis of meristic and mensural differences in fish from the Pit and Klamath river systems, concluded that *C. klamathensis* could be divided into three subspecies: (1) *C. k. klamathensis* (upper Klamath marbled sculpin), the nominate subspecies found in the Upper Klamath River drainage; (2) *C. k. polyporus* (lower Klamath marbled sculpin), found in the lower Klamath River, in some of its larger tributaries, and possibly in the Trinity River system; and (3) *C. k. macrops* (bigeye marbled sculpin), found in the Pit River system downstream from the confluence of the Fall River to the Pit 7 Reservoir, and in three tributaries: Hat Creek (downstream of the Rising River system), Burney Creek (downstream of Burney Falls), and the Fall River system (with the exception of Bear Creek). However, bigeye marbled sculpin may constitute a separate species due to its distinctive morphology, ecology and behavior (Moyle 2002).

Life History: Bigeye marbled sculpin grow quickly, attaining 35% of their maximum length in their first year and live about five years (Daniels 1987). Growth occurs from spring to early fall. Average sizes are 39 mm at the age of 1 year, 55 mm at 2 years, 62 mm at 3 years, 70 mm at 4 years, and 79 mm at 5 years. Although fish over 80 mm are rare, one specimen was recorded at 111 mm. Marbled sculpin attain sexual maturity after 2 years, during the winter (Moyle 2002). Spawning occurs from late February to March. Fecundity is low, with females producing 139-650 large eggs each. Adhesive eggs are deposited in clusters in nests under flat rocks. Eggs from different females may be present in the same nest. Nests are usually guarded by males (Daniels 1987). Embryos number from 826-2,200 per nest. Larvae measure 6-8 mm upon hatching, are benthic, and likely rear close to their nests (Moyle 2002). Because bigeye marbled sculpin have low fecundity, mature late and live relatively long, they are well-adapted to relatively stable environments such as spring-fed rivers (Daniels 1987).

Habitat Requirements: Bigeye marbled sculpin are well-adapted to large, clear, cool (< 20°C summer temperatures) spring-fed streams but also adjust to the conditions found in some reservoirs. Brown (1988) found that the acute preferred temperature was about 13°C (range 11-15°C) for fish acclimated at 10°, 15°, and 20°C. Temperatures above 15°C caused stress, particularly when associated with wide temperature fluctuations, and prolonged exposure to temperatures above 25°C was lethal. They are usually found in water with moderate flows (mean bottom velocity = 9.7 ±3.0 (1 S.E.) cm sec^{-1}; mean water column velocity = 23.1 ±4.5 cm sec^{-1}) and depths (mean 64.3 ±7.3 cm). Habitat use does not differ between adults and juveniles with respect to water velocity, but juveniles are found in shallower water. Typically, bigeye marbled sculpin are found in low-gradient runs and pools with abundant aquatic vegetation and coarse substrates, especially cobble, boulder, and gravel (Daniels 1987). In artificial streams, when given a choice of cobble and sand, they always selected cobble (Brown 1988). However, habitat use may shift in the presence of competitors such as Pit sculpin in riffles of the Pit River (Moyle 2002).

Distribution: The bigeye marbled sculpin is distributed throughout the middle reach of the Pit River system (Daniels and Moyle 1982). In this region, it is found in the main river below Britton Reservoir, lower Hat Creek, Sucker Springs Creek, and Clark Creek. It is the dominant sculpin in the sections of Lower Hat Creek and Burney Creek just above Britton Reservoir. The bigeye marbled sculpin also is found in the lower reaches of streams flowing into reservoirs of the lower Pit River, the lower Pit River itself, and Fall River.

Trends in Abundance: Bigeye marbled sculpin are the least abundant of the three sculpins endemic to the Pit River drainage (Moyle 2002). There are no trend data available, but it seems likely that modification of the lower Fall River and the creation of reservoirs (especially Britton Reservoir) has reduced their already limited range. Unlike rough sculpin, they are rarely found in reservoirs (Daniels and Moyle 1982) and populations in various stream reaches are now isolated from one another. Rutter (1908) found them to be the most abundant sculpin in the Fall River, whereas the rough sculpin is most abundant today. Overall, both the range and abundance of bigeye marbled sculpin appear to have declined over the past century.

Nature and Degree of Threats: Bigeye marbled sculpin are adapted to cold spring systems, such as Hat Creek and the Fall River. Land uses or other activities that change or disrupt these habitats are likely to affect marbled sculpin populations (Table 1). The habitat of this sculpin is similar, in large part, to that occupied by rough sculpin and the endemic Shasta crayfish (*Pascifasctus fortis*), both protected species. However, the disappearance of the crayfish from most its habitats in this region may indicate changing conditions, including the invasion of the aggressive signal crayfish (*P. lenuisculus*), that may cause reductions in bigeye marbled sculpin populations (Light et al. 1995). Thus, the apparent decline of Fall River populations may indicate the occurrence of long-term, subtle habitat degradation (Moyle 2002).

Dams. The Fall River, lower Pit River, Hat Creek and numerous tributaries have been almost completely harnessed for hydropower, so native fishes often have to exist in highly regulated and, in some cases, dramatically fluctuating hydrological conditions. The Fall River, for example, ends abruptly at Fall River Mills and is diverted into a penstock. The rocky, high gradient stretch at the mouth of the Fall River is consequently dry much of the year, fragmenting the system and inhibiting fish movement. This reach was quite likely good habitat for bigeye marbled sculpin in the past, with the combination of coarse substrate and cold water. Further fragmentation occurs with the series of four dams and their reservoirs on the Pit River, although some habitat for marbled sculpins is present downstream of the dams where cold-water releases are provided for hydropower production (Moyle and Daniels 1982).

Agriculture. Water is diverted from the Fall River directly, or indirectly, through wells for filling of paddies for wild rice or for flood irrigation of pasture. Excess water is returned to the river and is likely warmer and potentially polluted with agricultural chemicals and manure. The effects of these practices on sculpins and other fishes are not known, but unlikely to be favorable.

Grazing. Grazing is pervasive in the Fall River Valley and, in riparian areas, may degrade aquatic habitats by making them warmer and polluted. Cattle graze river banks in a number of places along the Fall River and Hat Creek. However, water quality in the system remains high, according to a study by the State Water Resources Control Board (http://fallriverconservancy.org/issue/water-quality/).

Logging. The Pit River watershed has a long history of repeated logging on private and public land (Shasta-Trinity National Forest), resulting in heavy sediment loads in tributary streams. This is presumably a major reason the river below tributaries has a distinct chocolate cast to it during periods of high run-off. The heavy loads of sediment coming down much logged and roaded Bear Creek, a tributary to the Fall River, were reduced only after a privately funded meadow restoration project created an area in which sediment could be deposited.

Recreation. The Fall River and Hat Creek are largely protected because of their importance to trout anglers, but heavy use by anglers can result in disturbance of sculpin (and other fish) habitat by wading in shallow water and by disturbing riparian vegetation on the banks; however, impacts to sculpin are unknown.

Alien species. The streams in which marbled sculpin occur are largely managed for wild rainbow trout (*Oncorhynchus mykiss*) and brown trout (*Salmo trutta*) fisheries. Generally, native rainbow trout have dominated the streams and introduced brown trout have been relatively uncommon. Changes to habitats or management activities that favor brown trout might have negative effects on marbled sculpin by increasing predation, given the more predatory nature and often larger size of brown trout (Moyle 2002). The invasion of aggressive signal crayfish into the spring systems of this region may have resulted in the displacement of marbled sculpin from under-rock shelters, making them more vulnerable to predation, much as has happened with the native, non-aggressive Shasta crayfish. A newer threat is the presence of piscivorous largemouth bass (*Micropterus salmoides*), smallmouth bass (*M. dolomieu*) and spotted bass *(M. punctulatus*) in the Pit River and its reservoirs. Their effects on the native fishes of this system need to be evaluated.

	Rating	Explanation
Major dams	Medium	The complex Pit River hydropower system fragments populations but fishes may benefit from habitat created by some dams
Agriculture	Medium	In the Fall River, water quality may be negatively affected by agricultural effluent and warmer temperatures from return flows
Grazing	Medium	Grazing is common in both the Fall River and upper Pit River drainages
Rural residential	Low	Runoff and effluent from Fall River Mills, Burney, and other communities may affect marbled sculpin habitats, as may diversions
Urbanization	Low	Few urban areas in region
Instream mining	n/a	
Mining	Low	Only known mining is for diatomaceous earth near Hat Creek and Britton Reservoir
Transportation	Low	Most habitats are crossed or paralleled by roads
Logging	Low	Sedimentation of Fall River and other watersheds in species range is an ongoing stressor; may have disproportionate impact on benthic species like sculpins; impacts much greater historically
Fire	Low	Wildfires are common in the region but impacts on bigeye marbled sculpin are unknown
Estuary alteration	n/a	
Recreation	Low	Most areas containing bigeye sculpin are heavily fished by trout anglers
Harvest	n/a	
Hatcheries	n/a	
Alien species	Medium	Predation and competition can reduce populations

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

Effects of Climate Change: Stream flow in the key spring streams occupied by bigeye marbled sculpin (Fall River, Hat and Burney creeks) depends on water percolating into volcanic landscapes, especially the Modoc Plateau (resulting in spring outflows of 1500-2000 cfs into the Fall River). Thus, flows will depend on how climate change affects precipitation patterns and associated water supplies long distances from these rivers, which remains largely unknown. A likely assumption is reduced or more variable flows, but stream temperatures remaining cold (because most flow is subsurface through aquifers). For more seasonal streams, predictions are that stream flow will increase in the winter and early spring and decrease in the fall and summer (Knox and Scheuring 1991, Field et al. 1999, CDWR 2006); however, this may not have much effect on core bigeye marbled sculpin populations, given that they mostly occupy larger, perennial, spring-fed streams. However, three factors suggest some vulnerability of bigeye sculpin to climate change: (1) they are a cold water-dependent species; (2) temperatures are likely to increase in below-dam habitats; and (3) the effects of changes in precipitation (likely less or more variable) and possible impacts to the lava-dominated watersheds that feed the region's spring systems are unknown. Potential climate change-induced alterations to operation of hydroprojects may also affect habitats in unknown ways. Moyle et al. (2013) rated bigeye marbled sculpin as being on the cusp between high and moderate vulnerability to extinction due to the added impacts of climate change, with low certainty.

Status Determination Score = 3.0 - Moderate Concern (see Methods section Table 2)**.** The bigeye marbled sculpin does not seem to be at risk of extinction at present, despite fairly large-scale changes to streams in its native range. This sculpin is largely protected by its occupation of spring-fed rivers with expansive subsurface catchments. NatureServe ranks bigeye marbled sculpin in California as Vulnerable to extirpation due to a restricted range, few populations, recent declines and/or other factors. They estimate the global abundance of the subspecies at 2500-10,000 with recent declines of 10-30%, but there seems to be no firm basis for this conclusion. The rationale for this status determination is detailed in Table 2.

Metric	Score	Justification
Area occupied	1	Endemic to the Pit River drainage
Estimated adult abundance	4	There appear to be multiple, fairly large populations
Intervention dependence	3	Population persistence may eventually require habitat improvements (management of flows, removal of alien species)
Tolerance	2	Bigeye marbled sculpin prefer constant (flow), cold (< 20°C summer temperatures), low gradient habitats
Genetic risk	4	Populations may become isolated due to dams and reservoirs
Climate change	3	Spring-fed streams probably a refuge, but high uncertainty
Anthropogenic threats	4	See Table 1
Average	21/7	3.0
Certainty (1-4)	2	Little information specific to bigeye marbled sculpin is available

Table 2. Metrics for determining the status of bigeye marbled sculpin, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: One of the biggest challenges to management of bigeye marbled sculpin is lack of data on abundance, genetic structure, and distribution in relation to hydroprojects. Periodic status surveys (about every 5 years) should be made of the endemic fishes and invertebrates of Fall River and Hat Creek to ensure the unique fauna remains self-sustaining. Future studies should also include genetic analyses of marbled sculpin subspecies to test whether any should be elevated to separate species status. Other recommendations are to protect and/or enhance aquatic habitats through active management of water and land use practices, including the lava catchments that feed the area's spring systems. For instance, changes in management of hydroelectric projects or trout fisheries should take into account habitat requirements and other needs of native fauna, including bigeye marbled sculpin. Water released from dams should mimic natural flow regimes in scale and periodicity. Recent changes to Pit River dam releases, as part of a FERC relicensing agreement, were implemented to more closely match natural flow regimes, including increased summer/fall base flows, increased flows during winter and spring months and intermittent freshet pulse flows (spikes) to flush substrates and vegetation. Agricultural and grazing practices should buffered from riparian areas sufficiently to protect against nonpoint source pollution and streambank destabilization.

Figure 1. Distribution of bigeye marbled sculpin, *Cottus klamathensis macrops* (Rutter), in California.

LOWER KLAMATH MARBLED SCULPIN
Cottus klamathensis polyporus Daniels and Moyle

Status: Moderate Concern. No immediate extinction risk exists for lower Klamath marbled sculpin. However, very little is known about this subspecies and it should be treated with moderate concern until information is available to demonstrate otherwise.

Description: All subspecies of marbled sculpin (*Cottus klamathensis)* have large, dorsally flattened heads with two chin pores; large, fan-like pectoral fins with four elements; and small pelvic fins that are positioned ventrally between the pectorals (Moyle 2002). Marbled sculpin are distinguished from other *Cottus* species by having 7-8 dorsal fin spines, joined dorsal fins, an incomplete lateral line with 15-28 pores, and relatively smooth skin (Daniels and Moyle 1984), although a few prickles can sometimes be felt below the lateral line. They also lack palatine teeth and have only one preopercular spine (Moyle 1976). Fin ray counts are: 18-22 in the second dorsal fin, 13-15 in the anal fin, 14-16 in the pectoral fin, and 11-12 (principal rays) in the caudal fin (Moyle 2002). All other sculpin species in California possess a split dorsal fin and more than 7 dorsal spines. Marbled sculpin are generally green-hued with a dark circular spot at the posterior end of the dorsal fin and alternating dark and light spots on the pectoral fin rays. Fish from the Klamath River are generally lighter and more marbled than those from the Pit River (Moyle 2002). Other marbled sculpin characteristics include: a wide interorbital region, a wide head and blunt snout, a maxillary rarely extending beyond the anterior half of the eye, and unjoined preoperculomandibular canals; however, these characteristics are shared with one or more other species (Daniels and Moyle 1984). Lower Klamath marbled sculpin are identified by 22-28 lateral line pores (Moyle 2002). Other marbled sculpin subspecies have 15-22 pores along the lateral line.

Taxonomic Relationships: *Cottus klamathensis* was first described by Gilbert (1898) from the Klamath River system. Rutter (1908) then described *Cottus macrops* from the Fall River, a large tributary to the Pit River, and noted that it closely resembled *C. klamathensis*. Robins and Miller (1957), upon review of specimens and recent collections, concluded that the two species were not sufficiently different to warrant separate species designations and considered *C. macrops* synonymous with *C. klamathensis*. Daniels and Moyle (1984), however, on the basis of meristic and mensural differences in fish from the Pit River and Klamath River systems, concluded that *C. klamathensis* could be divided into three subspecies: (1) *C. k. klamathensis* (upper Klamath marbled sculpin), the nominate subspecies found in rivers upstream of Klamath Falls and in the headwaters of the Lost River; (2) *C. k. polyporus* (lower Klamath marbled sculpin), found in the lower Klamath River downstream of Klamath Falls, in some of its larger tributaries, and possibly in the Trinity River system; and (3) *C. k. macrops* (bigeye marbled sculpin), found in the Pit River system downstream from the confluence of the Fall River to the Pit 7 Reservoir, and in three tributaries: Hat Creek (downstream of the Rising River system), Burney Creek (downstream of Burney Falls), and the Fall River system (with the exception of Bear Creek). Baumsteiger et al. (2012), using molecular techniques, confirmed that the three subspecies do represent three separate lineages.

Life History: Although specific data were not available, lower Klamath marbled sculpin life history likely mimics that of bigeye marbled sculpin in the Pit River. Bigeye marbled sculpin grow quickly, attaining 35% of their maximum length in their first year and live about five years (Daniels 1987). Growth occurs from spring to early autumn. Average sizes are 39 mm at the age of 1 year, 55 mm at 2 years, 62 mm at 3 years, 70 mm at 4 years, and 79 mm at 5 years. Although fish over 80 mm are rare, one specimen was recorded at 111 mm. Marbled sculpin attain sexual maturity after 2 years during the winter (Moyle 2002). Spawning occurs from late February to March. Adhesive eggs are deposited in clusters in nests under flat rocks. Eggs from different females may be present in the same nest. Nests are usually guarded by males (Daniels 1987). Embryos number from 826-2,200 per nest. Larvae measure 6-8 mm upon hatching, are benthic, and likely rear close to their nests (Moyle 2002).

Habitat Requirements: The habitat requirements of lower Klamath marbled sculpin are not well documented but they seem to occupy a wide variety of habitats, much like the upper Klamath marbled sculpin. Bond et al. (1988) found upper Klamath marbled sculpin were most likely to be collected in water with summer temperatures of 15-20°C, in coarse substrates (cobble and gravel) where water velocities ranged from slow to swift, in streams with widths greater than 20 m. Bond et al. (1988) characterized the marbled sculpin as a slow water species. Markle et al. (1996) noted that, while found in waters with temperatures ranging from 8-24°C, they appear to prefer temperatures of 10-15°C.

Distribution: Lower Klamath marbled sculpin are common in the Klamath River drainage from Iron Gate Dam downstream to the mouth of the Trinity River (Moyle 2002). They are apparently rare or absent in the Klamath River drainage downstream of the Trinity River and in the Trinity River itself, although Voight (2006) recorded them in McGarvey Creek, a tributary to the lower river.

Trends in Abundance: Although survey data do not exist, it is assumed that lower Klamath marbled sculpin are common throughout their native range (Moyle 2002).

Nature and Degree of Threats: Major anthropogenic factors that limit the viability of lower Klamath marbled sculpin populations are not described but factors known to affect stream-dwelling sculpins in the Klamath Basin appear in Table 1. Generally, any factors that alter water quality or cause sedimentation or compaction of substrates likely negatively affect this species. Thus, alteration of stream flow by dams and the effects of poor watershed management (logging, grazing, roads, water diversions) may have impacted lower Klamath marbled sculpin, although supporting data are largely absent.

	Rating	Explanation
Major dams	Medium	Six major dams have presumably resulted in reduced habitat quality and quantity
Agriculture	Low	Diversions a problem mainly in Shasta and Scott valleys
Grazing	Medium	Present throughout Klamath Basin
Rural residential	Low	Drainages where lower Klamath marbled sculpin occur are little developed
Urbanization	n/a	
Instream mining	Low	Changes to channel morphology and aquatic habitats are localized but unstudied
Mining	n/a	
Transportation	Low	Roads line most streams, delivering sediment, pollutants, etc.
Logging	Medium	Major land use in basin that may degrade habitat quality in streams
Fire	n/a	
Estuary alteration	n/a	The Klamath River estuary is assumed to be outside of their range
Recreation	n/a	
Harvest	n/a	
Hatcheries	n/a	
Alien species	Low	Introduced species (e.g., brown trout, bluegill, bullfrog) occur throughout their range and may prey on sculpin

Table 1. Major anthropogenic factors limiting, or potentially limiting, viability of populations of lower Klamath marbled sculpin in California. Factors were rated on a five-level ordinal scale where a factor rated "critical" could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated "high" could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction unlikely as a result; and a factor rated "no" has no known negative impact to the taxon under consideration. Certainty of these judgments is moderate. See methods section for descriptions of the factors and explanation of the rating protocol.

Major dams. Habitats and flows of the Klamath River have been altered by five mainstem dams and one on the Shasta River. Lower Klamath marbled sculpin appear to remain abundant below the lowest-most dam (Iron Gate) but, presumably, suitable habitat has been lost through creation of reservoirs and may be impaired in the interdam reaches (Moyle 2002). Like other native fishes, this sculpin may be negatively impacted by dam releases that do not closely mimic natural flow regimes.

Agriculture. Agriculture within lower Klamath marbled sculpin range is limited to valleys along the Shasta and Scott rivers. In these areas, flows and water quality (especially temperature) are impacted by agricultural diversions, with unknown but probably negative effects on sculpins.

Grazing. Grazing is present throughout the Klamath Basin. In riparian areas, grazing can degrade aquatic habitats by eliminating vegetation and associated shading, eroding stream banks, increasing sediment input and stream temperatures and adding fecal contamination, making these areas less suited for sculpins (Moyle 2002). Open range and allotment grazing are common throughout the range of the lower Klamath marbled sculpin.

Rural development and urbanization. Drainages where lower Klamath marbled sculpin occur are little developed. However, development in places like Yreka may affect local water quality, as may recreational developments (e.g., summer homes around upper Shasta River).

Instream mining. The Scott River (and other streams) now largely flows through an exposed channel surrounded by mining tailings, as a result of intensive historic mining. It is likely that average summer water temperatures are now higher and flows lower than they were prior to mining-related stream alteration. Suction dredging for gold, while currently banned in California, likely disrupted preferred riffle habitats throughout the basin. In both cases, potential effects on marbled sculpin are unstudied.

Logging. Logging on public and private lands is common throughout the range of lower Klamath marbled sculpin. Logging practices can degrade aquatic habitats by increasing sediment delivery to streams and removing riparian vegetation (Moyle 2002). Culverts along logging roads can prevent longitudinal movement, potentially isolating populations.

Alien species. Introduced species (e.g., brown trout, bullfrog) occur throughout their range and may prey on sculpin but, for the most part, alien species are not abundant in sculpin habitats and potential direct or indirect impacts remain unknown.

Effects of Climate Change: The predicted impacts of climate change on aquatic habitats in California include increases in water temperatures and changes to the frequency and timing of drought and flooding events. Water temperature increases may reduce the individual fitness of fishes by decreasing growth, decreasing reproductive potential, and increasing susceptibility to disease. However, specific impacts to lower Klamath marbled sculpin are unknown.

Elevated air temperatures associated with climate change will change the periodicity and magnitude of peak and base flows in streams due to a reduction in snow pack levels and seasonal retention. Stream flow in the Klamath River basin is fed by snowmelt from the Cascade Mountains and springs primarily associated with the Shasta River. Flows in the Scott River, Salmon River, and other snowmelt-fed tributaries may be significantly reduced due to the low elevations (< 3000 m) of the Cascade Mountains in northern California (Hayhoe et al. 2004). Stream flows are predicted to increase in the winter and early spring and decrease in the fall and summer (Knox and Scheuring 1991, Field et al. 1999, CDWR 2006), perhaps changing the spawning ecology of fishes. If increased winter and spring flows make floodplain habitats accessible, spawning lower Klamath marbled sculpin may benefit from the additional productivity associated with floodplain habitats. However, if lower Klamath marbled sculpin continue to spawn in main channels, increased winter and spring flows may mobilize stream sediments to the detriment of nests and eggs. Effects to some lower Klamath marbled sculpin populations may be mitigated by dam releases (in the mainstem Klamath River) and spring inputs (Shasta River). Moyle et al. (2013) found the lower Klamath marbled sculpin to be "highly vulnerable" to climate change, but with a low degree of certainty.

Status Determination Score = 3.9 - Moderate Concern (see Methods section, Table 2)**.** NatureServe ranks marbled sculpin in California as apparently Secure (S4), although no specific status is noted for the lower Klamath subspecies. The rationale for this status determination is detailed in Table 2 and is driven by the fact that so little is known about this distinctive sculpin.

Metric	Score	Justification
Area occupied	5	Distributed in the Klamath River and tributaries from Iron Gate Dam to the mouth of the Klamath River
Estimated adult abundance	5	Apparently abundant although robust population and distribution estimates are not available
Intervention dependence	4	Restoration activities that improve salmonid stream habitats should improve conditions for this subspecies
Tolerance	3	Lower Klamath marbled sculpin appear to withstand some environmental fluctuation
Genetic risk	3	No information on genetic structure but some populations may be fragmented
Climate change	3	Reaches that are solely fed by snowmelt may have reduced habitat quantity and/or quality under predicted scenarios
Anthropogenic threats	4	See Table 1
Average	3.9	27/7
Certainty (1-4)	1	Little information specific to lower Klamath marbled sculpin is available

Table 2. Metrics for determining the status of lower Klamath marbled sculpin, where 1 is a major negative factor contributing to status, 5 is factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: Management of lower Klamath marbled sculpin is challenged by the lack of data on abundance, environmental tolerance and population structure. Baseline surveys are needed to establish relative abundance of this subspecies within its range. Subsequent surveys (recommend every 5 years) will help determine general abundance trends. Studies are needed to establish the environmental tolerances of this subspecies, especially to factors likely affected by land use and climate change, including: temperature, turbidity, sedimentation, and water velocity. These studies would be complemented by a detailed investigation of the life history and genetic structure of meta- and sub-populations. Other recommendations are to protect and/or enhance aquatic habitats through active management of water and land use practices. For instance, changes in management of hydroelectric projects or actions to favor salmonids should take into account the needs of the native fauna, including lower Klamath marbled sculpin. Water releases from dams should mimic natural flow regimes in scale and periodicity. Also, buffers from grazing and logging activities should be established to protect stream habitats against nonpoint source pollution and stream bank destabilization.

Figure 1. Distribution of lower Klamath marbled sculpin, *Cottus klamathensis polyporus,* in California.

UPPER KLAMATH MARBLED SCULPIN
***Cottus klamathensis klamathensis* Daniels and Moyle**

Status: Critical Concern. No immediate extinction risk exists for upper Klamath marbled sculpin in Oregon but it is at risk of localized extirpation in California because of its limited distribution in a single, highly modified, watershed.

Description: All subspecies of marbled sculpin (*Cottus klamathensis)* have large, dorsally flattened heads with two chin pores; large, fan-like pectoral fins with four elements; and small pelvic fins that are positioned ventrally between the pectorals (Moyle 2002). Marbled sculpin are distinguished from other *Cottus* species by having 7-8 dorsal fin spines, joined dorsal fins, an incomplete lateral line with 15-28 pores, and relatively smooth skin (Daniels and Moyle 1984), although a few prickles can sometimes be felt below the lateral line. They also lack palatine teeth and have only one preopercular spine (Moyle 1976). Fin ray counts are: 18-22 in the second dorsal fin, 13-15 in the anal fin, 14-16 in the pectoral fin, and 11-12 (principal rays) in the caudal fin (Moyle 2002). All other sculpin species in California possess a split dorsal fin and more than 7 dorsal spines. Marbled sculpin are generally green-hued with a dark circular spot at the posterior end of the dorsal fin and alternating dark and light spots on the pectoral fin rays. Fish from the Klamath River are generally lighter and more marbled than those from the Pit River (Moyle 2002). Other marbled sculpin characteristics include: a wide interorbital region, a wide head and blunt snout, a maxillary rarely extending beyond the anterior half of the eye, and unjoined preoperculo-mandibular canals, but these characteristics are shared with one or more other species (Daniels and Moyle 1984). Upper Klamath marbled sculpin are identified by 15-22 lateral line pores, indicating a shorter lateral line than lower Klamath marbled sculpin (Gilbert 1897, Daniels and Moyle 1984). Other marbled sculpin subspecies have 22 or more pores along the lateral line.

Taxonomic Relationships: *Cottus klamathensis* was first described by Gilbert (1898) from the Klamath River system, including Upper Klamath Lake. Rutter (1908) then described *Cottus macrops* from the Fall River, a large tributary to the Pit River, and noted that it closely resembled *C. klamathensis*. Robins and Miller (1957), upon review of specimens and then recent collections, concluded that the two species were not sufficiently different to warrant separate species designations and considered *C. macrops* synomymous with *C. klamathensis*. Daniels and Moyle (1984), however, on the basis of meristic and mensural differences in fish from the Pit and Klamath river systems, concluded that *C. klamathensis* could be divided into three subspecies: (1) *C. k. klamathensis* (upper Klamath marbled sculpin), the nominate subspecies found in rivers upstream of Klamath Falls and in the headwaters of the Lost River; (2) *C. k. polyporus* (lower Klamath marbled sculpin), found in the lower Klamath River downstream of Klamath Falls and in some of its larger tributaries, and possibly in the Trinity River system; and (3) *C. k. macrops* (bigeye marbled sculpin), found in the Pit River system downstream from the confluence of the Fall River to Pit 7 Reservoir and in three tributaries: Hat Creek (downstream of the Rising River system), Burney Creek (downstream of Burney Falls), and the Fall River system (with the exception of Bear Creek). Baumsteiger et al. (2012), using molecular techniques, confirmed that the three subspecies do represent three separate lineages.

Life History: Upper Klamath marbled sculpin life history remains largely unknown but is likely similar to that of bigeye marbled sculpin in the Pit River, based on similarity of habitats. Bigeye marbled sculpin grow quickly, attaining 35% of their maximum length in their first year and live about five years (Daniels 1987). Growth occurs from spring to early autumn. Average sizes are 39 mm at the age of 1 year, 55 mm at 2 years, 62 mm at 3 years, 70 mm at 4 years, and 79 mm at 5 years. Although fish over 80 mm are rare, one specimen was recorded at 111 mm. Marbled sculpin attain sexual maturity after 2 years during the winter (Moyle 2002). Spawning occurs from late February to March. Fecundity of upper Klamath marbled sculpin is fairly high for sculpin, with 8-9 cm DL females producing about 1,200 eggs each (Markle et al. 1996). Adhesive eggs are deposited in clusters in nests under flat rocks. Eggs from different females may be present in the same nest. Nests are usually guarded by males (Daniels 1987). Embryos number from 826-2,200 per nest. Larvae measure 6-8 mm upon hatching, are benthic, and likely rear close to their nests (Moyle 2002).

Habitat Requirements: Upper Klamath marbled sculpin occur in a wide variety of habitats, from Upper Klamath Lake to headwater streams. Bond et al. (1988) found that they were most likely to be collected in water with summer temperatures of 15-20°C, in coarse substrates (cobble and gravel), where water velocities ranged from slow to swift; most streams had widths greater than 20 m. Bond et al. (1988) characterized marbled sculpin as a slow water species. Markle et al. (1996) noted that, while found at temperatures of 8-24°C, they appear to prefer temperatures of 10-15°C. In Upper Klamath Lake, they occur on soft bottom substrates and come off the bottom to feed at night (Markle et al. 1996).

Distribution: The upper Klamath marbled sculpin is apparently common in the upper Klamath Basin in Oregon but, in California, has been found recently only in Willow and Boles creeks, Modoc County, California (Markle et al. 1996). Based on what is known about its habitats elsewhere, it may have once occurred in much of the Lost River, before water quality and habitats became degraded.

Trends in Abundance: Most fish surveys of the Lost River basin have been focused on endangered suckers (Catostomidae), so sculpins may be underrepresented in existing data sets. Nevertheless, some records exist. V. King (CDFW, unpublished memo to E. Bailey, October 18, 1972) found a few marbled sculpin in pools below Clear Lake Reservoir dam. Sonnevil (1972) found marbled sculpin to be "common" in Willow Creek and "present" in Boles Creek. Koch et al. (1975) collected only seven sculpins, all from Willow Creek, by electrofishing. Shively et al. (1999) sampled the entire watershed in Oregon, using a variety of techniques, and collected only 11 marbled sculpins out of over 5,000 fish collected. A majority of the fish were alien species that had invaded since the 1970s. This spotty evidence suggests that Upper Klamath marbled sculpins have become rare in the Lost River watershed. In California, they may persist only in upper Boles and Willow creeks.

	Rating	Explanation
Major dams	High	Major dams have altered flows and changed habitats throughout the species range
Agriculture	High	The mainstem Lost River contains mainly agricultural return water of poor quality
Grazing	High	Cattle grazing is the major land use in the Willow and Boles creek watersheds
Rural residential	Low	Few residences in range
Urbanization	n/a	
Instream mining	n/a	
Mining	n/a	
Transportation	Low	Roads line most streams and are potential sources of sediment input and possible habitat fragmentation (e.g., culverts)
Logging	Medium	Limited historic and present logging probably contributes to degraded aquatic habitats in CA portion of range
Fire	n/a	
Estuary alteration	n/a	
Recreation	n/a	
Harvest	n/a	
Hatcheries	n/a	
Alien species	Medium	Alien fishes occur throughout range and presumably prey on sculpin, as they do other native fishes

Table 1. Major anthropogenic factors limiting, or potentially limiting, viability of populations of upper Klamath marbled sculpin in California. Factors were rated on a five-level ordinal scale where a factor rated "critical" could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated "high" could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction unlikely as a result; and a factor rated "no" has no known negative impact to the taxon under consideration. Certainty of these judgments is low. See methods section for descriptions of the factors and explanation of the rating protocol.

Nature and Degree of Threats: Upper Klamath marbled sculpin in California are apparently now restricted to highly modified and degraded habitats in limited portions of the Lost River drainage in California (Table 1).

Major dams. Lost River flows and habitats have been altered by multiple dams in California and Oregon. The principal dam in California creates Clear Lake Reservoir, capturing water from the Willow-Boles Creek drainage. Dam releases have greatly reduced natural flows in the Lost River, creating generally poor water quality and increased temperatures (Snively et al. 1999).

Grazing. Grazing is the dominant land use around Willow and Boles creeks, the principal habitats of upper Klamath marbled sculpin in California. In riparian areas, grazing can degrade aquatic habitats by removing riparian vegetation and associated stream shading, reducing depth, increasing sediment and nutrient input, and increasing stream temperatures so habitats become less suited for sculpins (Moyle 2002).

Logging. Logging occurs in Modoc National Forest but impacts on streams are undocumented. In general, logging practices degrade aquatic habitats by increasing sediment delivery to streams and removing riparian vegetation (Moyle 2002). Culverts along logging roads can prevent longitudinal movement, potentially isolating populations.

Alien species. The Lost River is dominated by alien species, including various centrarchids, brown bullhead (*Ameiurus nebulosus*), and fathead minnow (*Pimephales promelas*). While predation and competition are of concern, the presence of these species is more indicative of poor water quality (Shively et al 1999). Sacramento perch (*Archoplites interruptus*) are now one of the most common fish in Willow Creek and they may be predators on, or competitors with, marbled sculpin (Moyle 2002).

Effects of Climate Change: The most noticeable and widespread predicted impacts of climate change on aquatic habitats in California will be increased water temperatures and changes to the frequency and timing of drought and flooding events. Water temperature increases may reduce the individual fitness of fishes by decreasing growth, decreasing reproductive potential, and increasing susceptibility to disease (Moyle and Cech 2004). While specific impacts to upper Klamath marbled sculpin remain unknown, climate change increases the likelihood that Willow and Boles creeks will become less suitable as sculpin habitat, including large sections of stream drying completely during extended drought periods. Upper Klamath marbled sculpin occur in an already arid portion of the state, with instream flows highly dependent on both snowmelt (headwater tributaries) and dams and diversions and associated releases (mainstem rivers). Reduction in snowpack, or precipitation in general, coupled with modified dam operations associated with reduced reservoir recharge, will likely negatively impact marbled sculpin and other native fishes. These changes will also likely favor alien species and potentially allow for expansion of their distribution. Moyle et al. (2013) scored upper Klamath marbled sculpin as being “highly vulnerable” to extinction from the combination of climate change effects and other stressors.

Status Determination Score = 1.7 - Critical Concern (see Methods section, Table 2)**.** NatureServe ranks marbled sculpin as apparently Secure (S4), although no specific status is noted for the upper Klamath subspecies. The rationale for this status determination (see Table 2) relates to the fact that little is known about upper Klamath marbled sculpin distribution and abundance in California. The limited empirical data suggest that their populations may be critically low.

Metric	Score	Justification
Area occupied	1	Restricted to the Lost River drainage; mainly occurs in Boles and Willow creeks
Estimated adult abundance	2	Unknown; habitat limited and records few
Intervention dependence	2	Limited headwater habitats need to be managed to benefit sculpins if they are to persist in CA
Tolerance	3	Upper Klamath marbled sculpin appear to withstand some environmental fluctuation
Genetic risk	2	No information on genetic structure but populations are small and isolated from one another
Climate change	1	Increased likelihood of warmer water temperatures, reduced flows, and potential drying of large portions of existing habitat
Anthropogenic threats	1	See Table 1
Average	1.7	12/7
Certainty (1-4)	1	Little information specific to upper Klamath marbled sculpin is available

Table 2. Metrics for determining the status of upper Klamath marbled sculpin, where 1 is a major negative factor contributing to status, 5 is factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: Willow and Boles creeks should be managed as refuges for upper Klamath marbled sculpin (and other native fishes), with protected water sources and stream banks protected from grazing. A survey of all native fishes of the entire Lost River watershed in California and Oregon should be undertaken, expanding upon existing efforts to assess sucker and trout populations. A comprehensive survey, repeated at some level of frequency in order to establish trend information, would clarify the status of upper Klamath marbled sculpin and allow for improved management of their populations and habitats.

Figure 1. Distribution of upper Klamath marbled sculpin, *Cottus klamathensis klamathensis***,** in Willow and Boles creeks, tributaries to the Lost River, Modoc County, California.

CLEAR LAKE PRICKLY SCULPIN
Cottus asper ssp. (Richardson)

Status: Moderate Concern. The Clear Lake prickly sculpin cannot be regarded as secure because of continual changes in water quality and high abundance of alien species in its lake habitats.

Description: Prickly sculpins can be distinguished from other sculpins by their long dorsal and anal fins (Moyle 2002). For Clear Lake prickly sculpin, fin spine and ray counts are 8-9 soft spines in the first dorsal fin, 20-21 rays in the second dorsal fin, and 17-19 rays in the anal fin (Hopkirk 1973). The dorsal fins join at the base. Pelvic fins have one spine fused with the first ray and 3 additional rays. The pectoral fins have 17-18 rays. Prickling, which gives the body a rough feel, is well developed on the body (Hopkirk 1973). Prickly sculpin, in general, have 5-6 gill rakers, 6 branchiostegal rays on each side, and 2-3 pre-opercular spines, of which only one is usually observable. The lateral line has 28-43 pores and is complete. Palatine teeth are easily observable. They normally have one but, occasionally, two pore(s) on the chin. The caudal peduncle is rounded and narrow in relation to body depth. Coloration varies but is usually mottled reddish brown to dark brown with 4-5 dark saddles on the dorsal surface and light yellow to white on the belly. During breeding, males turn very dark and both sexes develop orange edges on the first dorsal fin. Non-breeding males can be distinguished from females by their long, V-shaped genital papilla. Clear Lake prickly sculpin tend to have smaller adult body size (usually <60 mm TL) than most other California prickly sculpins.

Taxonomic Relationships: Prickly sculpin are highly variable morphologically, widely distributed, and have poorly studied systematics (Kresja 1965, 1970). Although pelagic larvae allow for wide dispersal, it is likely that genetic differences exist between subgroups. Three distinct forms of prickly sculpin exist in California: the coastal form, Central Valley form, and Clear Lake form (Hopkirk 1973). Hopkirk (1973,1988) indicated that the Clear Lake form merited subspecies status based on the number of anal (ca. 18) and pectoral (17-18) fin rays, the partial prickling on adults and trophic adaptations for feeding on small benthic invertebrates. Recent genetic studies support the distinctiveness of the Clear Lake sculpin, perhaps at the species level, with the most closely related populations occurring in Cache Creek downstream of the lake (Baumsteiger et al. 2012).

Life History: Clear Lake prickly sculpin are most commonly found near shore, associated with beds of tules and gravel substrates (Week 1982). Assuming they have similar life history to other forms, they spend most of their time lying on the bottom during the day and become more active at night. Prickly sculpin are not especially gregarious, although aggregations of prickly sculpin have been observed near lake shores in British Columbia (Northcote and Hartman 1959). They are not territorial outside of the breeding season.

Clear Lake prickly sculpin complete their entire life cycle within the lake (Broadway and Moyle 1978). Spawning takes place in March and April, as indicated by presence of larvae in the water column. Presumably, like other prickly sculpins, they spawn under rocks and logs. Males build nests by excavating a small area beneath rocks and then clean the underside of the rock to which eggs are attached. Males lure females into the nest and courtship occurs mostly at night (Kresja 1965, 1970). Once spawning is complete, males chase females from the nest and guard embryos until they hatch. Males move in the nest to facilitate water circulation over the eggs and ensure hatching (Moyle 2002). Fecundity ranges from 280 to 11,000 eggs per female, depending on size and age (Patten 1971); presumably fecundities of Clear Lake sculpins are on the low side of this range because of their relatively small size. In lakes, such as Clear Lake, larvae swim up into the water column upon hatching (5-7 mm TL), live as plankton for 3-5 weeks, and eventually settle as juveniles (15-20 mm) on the bottom. Juveniles in lakes move into shallow water upon settling (McLarney 1968). Clear Lake sculpins move offshore during the day and move inshore to feed at night, although some are found in shallow water at all times of day (T. Ford, UC Davis, unpubl. report 1977, Broadway and Moyle 1978).

Prickly sculpin feed primarily on benthic invertebrates and small fish. In Clear Lake, 74% of their summer diet was historically chironomid midge larvae and pupae (Cook 1964). Amphipods became an abundant prey item after invasion of Mississippi silverside, *Menidia audens* , which greatly reduced midge abundance (Broadway and Moyle 1978, L. Decker and M. LeClaire, UC Davis, unpubl. report 1978). However, amphipods were more commonly eaten by sculpins captured inshore, while chironomid larvae were more common in fish captured offshore. Clear Lake prickly sculpin feed at all times of the day and night but more intensely at sunrise and sunset. The rate for complete digestion of one chironomid larva is about 7 hours (T. Ford, UC Davis, unpubl. report 1977). Diets of fish collected from sandy substrates are less varied than in those collected from rocky substrates. Diets vary little with size with the exception of pelagic larvae, which feed on planktonic copepods and cladocerans (Broadway and Moyle 1978, Eagles-Smith et al. 2008b). However, ontogenetic shifts in diet, from small invertebrates to larger invertebrates and fish, have been noted for prickly sculpin in Lake Washington (Tabor et al. 2007). Prickly sculpin are prey to other species but are not common in diets even where they are abundant, as in Clear Lake.

Clear Lake prickly sculpin growth is highly variable. One study found young-of-year, on average, measured 26 mm SL, while 1+, 2+, 3+, and 4+ age fish measured 34 mm, 44 mm, 48 mm, and 55 mm, respectively (L. Decker and M. LeClaire, UC Davis, unpubl. report 1978). Another study determined Clear Lake prickly sculpin to measure 28 mm SL at 1+ and 35-45 mm at 2 to 5+ (T. Ford, UC Davis, unpubl. report 1977). One individual was aged at 5+ at 95 mm SL, yet lengths for 3+ fish ranged from 30-90 mm. The length-weight relationship for Clear Lake prickly sculpin was determined by Ford (1977) to be $W = 1.02 \times 10^{-5} SL^{3.19}$, where SL is standard length.

Habitat Requirements: The Clear Lake prickly sculpin is adapted to life in a warm (summer temperatures 25-28°C), shallow (average depth 6.5 m), lake with mostly sandy or soft bottom substrates. The lake has been highly productive for thousands of years as a result of shallow, warm, well-mixed waters. Clear Lake and lower Blue Lake are thus eutrophic, alkaline (pH of ca. 8), and fairly turbid (Secchi depth, <2m) (Suchanek et al. 2008). Upper Blue Lake, in contrast, is clear and cool. Sculpins show no apparent preference for substrates within the lake and are abundant on soft and sandy bottoms; they have been found at depths up to 10 m (Broadway and Moyle 1978). While spawning has not been directly observed, it is likely they require logs, rocks and similar substrates for their nests. Clear Lake prickly sculpin do not inhabit streams of the Clear Lake basin, although prickly sculpin are found in Cache Creek and its tributary, Bear Creek, a tributary downstream from the lake (Hopkirk 1973; J. Baumsteiger, UC Merced, unpublished data, 2013).

Distribution: Clear Lake prickly sculpin are found in Clear Lake, Lake County, a large, natural lake, and in Upper and Lower (presumed) Blue lakes, in the Clear Lake basin (Hopkirk 1973). They may also occur in small numbers in Cache Creek, the outlet of the lake, although this population appears to be genetically and ecologically distinct from the Clear Lake population (J. Baumsteiger, UC Merced, unpublished data, 2013). Clear Lake is the largest freshwater lake in California (not counting Lake Tahoe, which is partly in Nevada). It is located in the Coast Ranges at 402 m elevation and has a surface area of about 17,670 ha (Moyle 2002).

Trends in Abundance: Prickly sculpin are apparently abundant in Clear Lake (Broadway and Moyle 1978). However, survey data indicate that, while the population experiences wide fluctuations, the general trend is declining (Figure 1). Eagles-Smith et al. (2008) found sculpin to be one of the most common fish in the lake, with no significant changes in density (based on area seined) from 1986 through 2004.

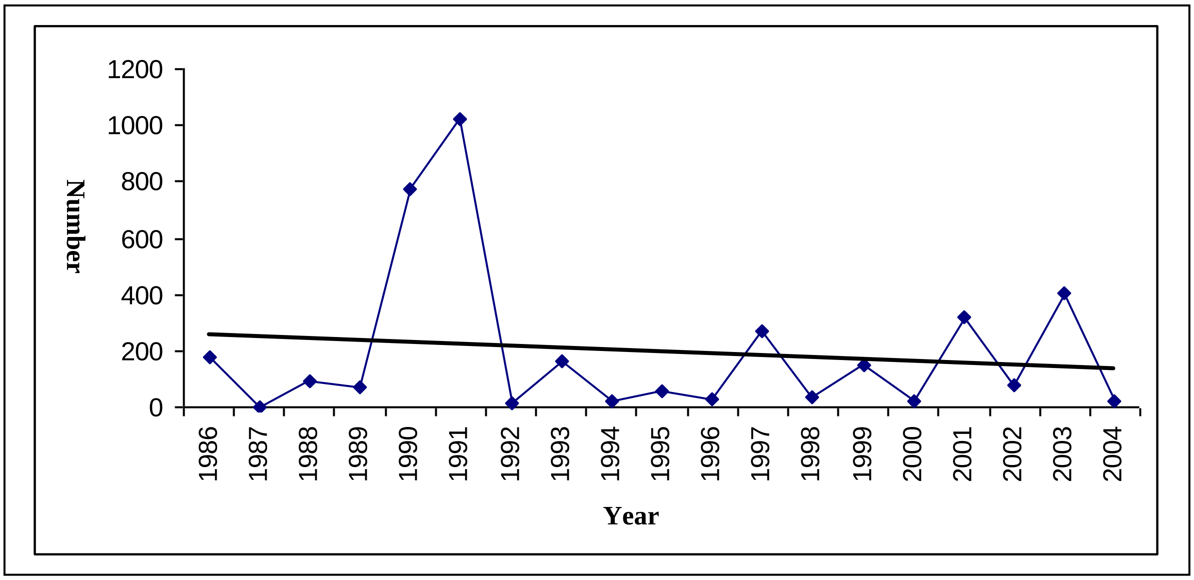


Figure 1. Number of Clear Lake prickly sculpin caught through standardized seining, Clear Lake, 1986-2004. Data from the Clear Lake Vector Control Agency.

Nature and Degree of Threats: Clear Lake is a highly altered natural lake. It is polluted with nutrients, sediment, heavy metals, and pesticides from numerous sources and has experienced invasions of many species of alien fish, invertebrates, and plants (Table 1). These changes resulted in the disappearance of five native species of fish from the lake, including the endemic Clear Lake splittail (*Pogonichthys ciscoides*). Five native species have persisted despite these stressors, including Clear Lake prickly sculpin. Whether or not prickly sculpin can continue to persist long-term in the face of rapid change and cumulative impacts, especially predation and competition from large populations of alien fishes is, however, in doubt, despite their present abundance.

Major dams. Cache Creek Dam was built in 1914 to provide water for Yolo County agriculture by storing water in Clear Lake. The effects on sculpin populations are unknown.

Agriculture. The Clear Lake basin is utilized for fairly intensive agriculture, including expanding viticulture, which sends effluents carrying fertilizers, sediments, and pesticides into the lake, although these impacts were greater historically than they are today. Fertilizers and sediments contributed to accelerated eutrophication of Clear Lake in the 20th century that resulted in major blooms of blue-green algae. While Clear Lake prickly sculpin persisted through periods of impaired water quality, die-offs of prickly sculpin elsewhere have been attributed to low dissolved oxygen concentrations associated with the senescence of algal blooms (Martin et al. 2007), including that of cyanobacteria found in Clear Lake.

Grazing. Heavy grazing of Clear Lake watersheds has occurred since the 1870s and has likely contributed to sedimentation and nutrient loading of the lake (Suchanek et al. 2002). Effects on sculpin are unknown.

Rural development. As Clear Lake became popular as a resort area in the 19th century, the lakeshore became increasingly developed with vacation and permanent homes. This development filled wetlands on the lake margin, important for trapping sediment and nutrients, added septic tank effluent to the lake, and caused large-scale application of pesticides to the lake to control pestiferous gnats. Sculpin persisted despite these changes to lake characteristics.

Urbanization. Many small towns around the lake also contribute to eutrophication through sewage spills, increase in sedimentation, and removal of wetlands. Local residents were leading proponents of applying pesticides to the lake. In particular, dichloro-diphenyl-dichloroethane (DDD) was applied (1949, 1954, 1957) to control gnat populations. DDD accumulates in the fatty tissues of fishes, perhaps affecting survival and reproduction (Hunt and Bischoff 1960).

Mining. The Sulphur Bank Mercury Mine dumped mining waste containing mercury directly into the Oaks Arm of the lake and shore from 1922-1947 and 1955-1957; these wastes contaminated the lake ecosystem with mercury and arsenic (summarized in Suchanek et al. 2002). Elevated levels of mercury have been found in fish and waterfowl within the basin. A current health advisory (first issued in 1986) recommends that not more than one fish from Clear Lake be consumed per week. The water column does not seem to contain high concentrations of methyl mercury, in contrast to some lake sediments. Mercury concentrations in Clear Lake fishes appear to be directly correlated with extent of benthic foraging, making prickly sculpin particularly susceptible to mercury bioaccumulation (Eagles-Smith et al. 2008). Indirect effects from mercury exposure include behavior disruption (prey capture, inhibition of reproduction), reduced growth rate, and disruption of physiological functions (olfaction, thyroid function, blood chemistry; Suchanek et al. 2008). However, the physical and biological attributes of the lake, including its size, alkalinity, and lack of a developed hypolimnion, appear to diminish the effects of mercury on aquatic organisms (Suchanek et al. 2008a, b).

	Rating	Explanation
Major dams	Low	Cache Creek Dam regulates lake levels, potentially exposing important near-shore habitats during draw-downs
Agriculture	Medium	Agricultural runoff contributes to eutrophication and pesticide loads
Grazing	Low	Overgrazing has occurred since the 1800s and has contributed to sedimentation and nutrient loading in the lake; greater impact in the past
Rural residential	Medium	Development has drastically altered shorelines and increased eutrophication
Urbanization	Medium	Urban runoff is a source of nutrients and pesticides; development along the lake shore has degraded habitats
Instream mining	n/a	
Mining	Medium	Mercury levels in lake fishes are highest in benthic foragers such as prickly sculpin
Transportation	Low	Roads along the lake shore can contribute pollutants and sediments
Logging	Low	Erosion from timberlands have likely increased the amount of fine sediment delivery to the lake
Fire	Low	Wild and human-induced fires are common in Clear Lake watersheds and can increase sediment delivery to the lake
Estuary alteration	n/a	
Recreation	Low	Motorized boats can contribute to pollution from oil and gas and disrupt fish habitat use
Harvest	n/a	
Hatcheries	n/a	
Alien species	High	At least three alien species likely compete with prickly sculpin; introduced piscivores may prey on sculpin

Table 1. Major anthropogenic factors limiting, or potentially limiting, viability of populations of Clear Lake prickly sculpin in California. Factors were rated on a five-level ordinal scale where a factor rated “critical” could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated “high” could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated “medium” is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated “low” may reduce populations but extinction unlikely as a result; and a factor rated “no” has no known negative impact to the taxon under consideration. Certainty of these judgments is moderate. See methods section for descriptions of the factors and explanation of the rating protocol.

Transportation. Roads follow the lake shores for long distances (e.g. highway 20), facilitating pollution from road run off and siltation by road drainage.

Logging. Clearing of forestlands around Clear Lake began in the 1840s but accelerated post-World War II, contributing to eutrophication and siltation of the lake (Suchanek et al. 2002).

Recreation. Extensive use of gas-powered watercraft in Clear Lake may negatively affect the health of prickly sculpin. Polycyclic aromatic hydrocarbons (PAH), a contaminant that enters water bodies from the combustion or oil wastes of personal watercrafts, has been implicated in causing physiological changes in prickly sculpin in Auke Lake, Alaska (Moles and Marty 2005). Prickly sculpin exposed to high concentration of PAH experienced lower condition factors and fewer lymphocytes than sculpin collected from lakes where motorized watercraft were banned. Sculpin collected from Auke Lake also had more liver lesions indicative of chronic toxicity than sculpin collected from other locations. Boats and other watercraft can also disrupt fish habitat use in shallower waters.

Fire. Natural and human-induced fires are common in the watersheds that drain into Clear Lake (Suchanek et al. 2002). Catastrophic fires can increase erosion rates and sediment delivery to the lake, contributing to eutrophication. Fire frequency and intensity are expected to increase in the future, as is the duration of 'fire season' under climate change models, potentially putting Clear Lake at higher risk for continued habitat degradation associated with sedimentation and eutrophication.

Alien species. Historically, 10 native fish species were found in Clear Lake (Moyle 2002). Presently, only five (hitch, blackfish, tule perch, prickly sculpin, Sacramento sucker) exist in numbers, along with at least 16 alien fish species. Sculpin persist in large numbers despite the introduction of many potential predators and competitors. They can be major prey of largemouth bass (Murphy 1949) although, so far, they have sustained populations despite potential predation impacts. It is also possible that predation on larvae by introduced planktivorous fishes, such as Mississippi silversides and threadfin shad, could reduce sculpin numbers, as could competition for benthic prey. Planktivores switch to benthic invertebrates in the lake if zooplankton is depleted by grazing, although prickly sculpin did not undergo a dietary shift when threadfin shad became extremely abundant in the lake for a short period (Eagles-Smith et al. 2008). The study of Eagles-Smith et al. (2008) suggests that Clear Lake has a highly variable community of alien fishes. An unexpected shift in this community or the invasion of a new species could impact sculpin populations.

Effects of Climate Change: Predicted increases in temperatures may increase the extent and intensity of algal blooms in Clear Lake. Coupled with reduction of tributary stream inputs in the summer, these conditions can lead to areas in the lake with very low dissolved oxygen concentrations, limiting suitable habitat for native fish species. Climate change predictions also state that the frequency and intensity of storm events will increase, potentially increasing sedimentation, nutrient loading and pollution (from mine wastes and urban or suburban runoff and effluents) into Clear Lake (Suchanek et al. 2002). In a separate analysis of 10 metrics, Moyle et al. (2013) rated the Clear Lake sculpin as 'highly vulnerable' to climate change, indicating that if present conditions in Clear Lake and the Blue Lakes significantly worsen as the result of climate change (e.g., water temperatures and eutrophication increase), extinction risks increase dramatically.

Status Determination Score = 3.3 - Moderate Concern (see Methods section Table 2)**.** The Clear Lake prickly sculpin have a limited distribution and face many threats that, in combination, could contribute to further population declines and potentially cause its extinction.

Metric	Score	Justification
Area occupied	2	Clear Lake prickly sculpin are only found in Clear Lake and in Upper and Lower Blue lakes
Estimated adult abundance	5	Current abundance is not known but population assumed to be large
Intervention dependence	4	Many stressors threaten the viability and health of Clear Lake prickly sculpin, although they have proven remarkably resilient
Tolerance	4	Prickly sculpin, in general, are tolerant of a wide range of natural environmental factors; however, they are likely at the limits of their tolerance in Clear Lake
Genetic risk	4	Genetic risks unknown
Climate change	2	Increased temperatures have the potential to change the base of food webs and decrease productivity
Anthropogenic threats	2	See Table 1
Average	3.3	23/7
Certainty (1-4)	3	Seine sampling provides reasonable assessment of status

Table 2. Metrics for determining the status of Clear Lake prickly sculpin, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: The following recommendations will enhance our understanding of this form and bolster conservation efforts:

1. The Clear Lake prickly sculpin should be formally described as a subspecies (as recommended in Hopkirk 1973) and as supported by new genetic data from J. Baumsteiger, UC Merced (2013).
2. The distribution and ecology of Clear Lake prickly sculpin should be more thoroughly documented as part of a systematic sampling program for Clear Lake native fishes. In particular, its status in Upper and Lower Blue Lakes should be determined.
3. Population abundance indices should be established and determined frequently, allowing for trend monitoring.
4. Environmental tolerances specific to Clear Lake prickly sculpin should be established. Parameters studied should include: temperature, dissolved oxygen, siltation (to determine spawning success), as well as exposure to methyl mercury, pesticides, and other gas/oil derivatives.
5. A conservation plan for all fishes native to the Clear Lake basin should be developed and

implemented (see the Clear Lake tule perch account in this report).

6. Use existing laws and regulations to protect remaining shoreline habitats in order to improve spawning and rearing conditions for prickly sculpin and other native fishes, which depend on these important habitats.

Figure 2. Distribution of Clear Lake prickly sculpin, *Cottus asper* ssp. (Richardson), in Clear Lake, California.

RIFFLE SCULPIN
***Cottus gulosus* (Girard)**

Status: Moderate Concern. The riffle sculpin has a fragmented distribution and faces numerous threats that, in combination with climate change, could conceivably cause extinction of genetically distinct populations, leading to reduced diversity and further isolation. The taxon here appears to represent several species or subspecies.

Description: Riffle sculpins are 'generic' sculpins with no single definitive identifying external characteristics, although quite distinct genetically. According to Moyle (2002) they "are defined by the following combination of characteristics: four pelvic elements (1 spine and 3–4 rays); 7–8 soft spines on the first dorsal fin; 16–19 rays in the second dorsal fin; 15–16 rays in each pectoral fin, some of which may be branched; 12–16 rays (usually 13–15) in the anal fin; palatine teeth that are usually present; prickles that are present only behind the pectoral fin (axillary patch); 2–3 preopercular spines; a lateral line that is complete or incomplete with 22–36 pores; and dorsal fins that are usually joined. The mouth is large, so the maxillary may reach as far as the rear edge of the eye. The pelvic fins usually do not reach the vent when depressed. There is usually one median chin pore. They have the typical sculpin mottled body color, with a large black blotch on the rear of the first dorsal fin. Spawning males are dark, often with an orangish edge to the first dorsal fin (p. 350)."

Taxonomic Relationships: As Moyle (2002) states "Riffle sculpin were originally described by Charles Girard in 1854, from San Mateo Creek, San Mateo County, as *Cottopsis gulosus.* The identity of local populations has been in a state of confusion ever since (p. 351)." Fortunately, Baumsteiger et al. (2012) and Baumsteiger (2013) have used molecular phylogenetics to resolve many aspects of riffle sculpin systematics, using both mitochondrial and nuclear DNA. These studies show the following:

1. The anomalous populations in Oregon and Washington, long considered part of *C. gulosus* (Moyle 2002), belong to a quite different, distantly related species. This makes riffle sculpin a species endemic to California.

2. Riffle sculpins in streams tributary to the San Joaquin River are distinct from other riffle sculpin populations. They also show considerable genetic differences (structure) among populations, indicating that each stream contains an isolated population with little historic gene flow to other populations.

3. Riffle sculpins in the Sacramento River and tributaries are distinct from San Joaquin riffle sculpins, reflecting an undefined relationship (e.g., ancient hybridization, shared ancestry) with Pit sculpin (*C. pitensis*).

4. Coastal populations of riffle sculpin are separate lineages from sculpins in Central Valley tributaries and seem to be more closely related to prickly sculpin (*C. asper*) than to other riffle sculpins. The populations from the Russian River also appear to be distinct from other coastal populations. Because the original description of riffle sculpin was

based on a coastal population, future taxonomy may designate these populations as *C. gulosus,* and other populations as separate species.

The evidence presented by Baumsteiger et al. (2012) and Baumsteiger (2013) indicates that California populations of riffle sculpin potentially represent four species or subspecies (associated with San Joaquin, Sacramento, Pajaro-Salinas, and Russian river watersheds). The presence of such cryptic taxa has been found within other "species" of *Cottus* as well (Kinzinger et al. 2005, Lemoine et al. 2014). However, further work is needed to define taxon boundaries and to look for morphological and meristic differences as well. Until such work is completed, all populations in California should continue to be treated as part of one species, while excluding the Oregon and Washington populations, which are widely separated geographically from the other populations.

Life History: The sculpins grouped together here as riffle sculpins are found exclusively in permanent cold-water streams. Despite genetic differences, we assume the habitat similarities among disparate populations indicate similar life history adaptations, following the general pattern described in Moyle (2002).

The disjunct distribution pattern of riffle sculpins reflects their narrow habitat requirements and the poor dispersal abilities of both adults and young. Following a severe drought, it took over 18 months for sculpins in the Pajaro River to recolonize a riffle that went dry only 500 m downstream from a large permanent population (Smith 1982). The fact that their larvae are benthic (rather than planktonic) and do not move far after hatching greatly reduces their ability to quickly recolonize areas from which they have been extirpated, especially if there are barriers that restrict recolonization.

Riffle sculpins eat mainly benthic invertebrates, primarily active insect larvae such as those of caddisflies, stoneflies, and mayflies (Moyle 2002). However, they will consume other prey that is readily available, such as amphipods and small fish, including other sculpins. They appear to feed mainly at night, although their stomachs can contain food at any time of the day.

Age and growth of riffle sculpin has not been well studied and is based mainly on length-frequency distributions (Moyle 2002). Most adults are 60–80 mm long (standard length) and are assumed to be 2-3 years old. Older fish, probably 3-4 year old males, measure 75–100 mm. Larger fish are rare but, when food is abundant, they can reach 100–160 mm TL and 4+ years old. The maximum age for the species is not known.

Riffle sculpins are thought to mature at the end of their second year, spawning in February, March, and April (Moyle 2002). Spawning takes place under rocks in swift riffles or inside cavities in submerged logs. Males choose nesting sites and will spawn with multiple females. Embryo counts range from 462 to more than 1,000 per nest; embryos may be in different stages of development, the result of multiple spawnings. Males stay in the nest to guard embryos and fry, often becoming emaciated in the process. Embryos hatch in 11 (at 15°C) to 24 (at 10°C) days. After absorbing the yolk sac, at about 6 mm TL, fry assume their benthic existence and remain close to the nest.

Habitat Requirements: Riffle sculpins live in permanent, cool, headwater streams where riffles and rocky substrates predominate (Moyle 2002, Leidy 2007). Such streams are clear and shaded, with moderate gradients. In Deer Creek (Tehama County), they occupy areas in fairly shallow (mean depth of 38–39 cm), fast-flowing water (mean water column velocity of 42–44 cm/sec), typical of rocky riffles. However, they live in areas sheltered from strong currents, under rocks or logs (mean water velocity of 8–9 cm/sec). Consequently, they also live in small pools that contain undercut banks, rubble, or other complex cover. They are most abundant in water that does not exceed 25–26°C for extended periods of time; temperatures over 30°C are usually lethal. Dissolved oxygen levels must be at or near saturation, a requirement that also restricts them to areas with flowing water. In most streams, they occur with 3-6 species of other native fishes, most typically with rainbow trout (*Oncorhynchus mykiss*).

Distribution: Riffle sculpin are found in many increasingly isolated watersheds in the Central Valley drainage and the central coast. In tributaries to the San Joaquin River, they are present from the Mokelumne River south to the Kaweah River. They are mostly present in mid-elevation reaches, although they are present below dams with coldwater releases (e.g. Kings and Tuolumne rivers, Moyle 2002). They are absent from the Cosumnes River (Moyle et al. 2010). In the Sacramento River drainage, they are present in Putah Creek on the west side and most tributaries on the east side, from the American River north to the upper Sacramento and McCloud rivers. However, the exact boundaries between riffle and Pit sculpin (*Cottus pitensis*) distributions still need to be determined. In the San Francisco Bay region, they are still found in about a quarter of the watersheds, including Coyote Creek, the Guadalupe River, the Napa River, Sonoma Creek, Corte Madera Creek, and Green Valley Creek (Leidy 2007, Leidy et al. 2011). They are absent today from San Mateo Creek, from which they were originally described (Leidy 2007). They are found in coastal streams that have had historical connections to the Central Valley drainage, including the Pajaro and Salinas rivers and Salmon and Redwood creeks (Marin County). They are also present in Russian River tributaries. Although they have been identified in the Navarro River, recent surveys have failed to locate riffle sculpin (Moyle 2002), indicating past records represent misidentification of other sculpin species. The absence of riffle sculpins from many tributary streams in which they might be expected within their known range demonstrates the difficulties this species has in re-colonizing a stream, once a population has been lost.

Trends in Abundance: Most fish surveys in California do not identify sculpins to species so trend data is largely absent. However, the studies of Leidy (2007) and others (Moyle 2002) indicate they were more widely distributed in the past. They are absent from the South Fork Yuba watershed, in which they were presumably once present. Populations are present below dams on a number of rivers and creeks (e.g., Kings, Mokelumne, Tuolumne and Yuba rivers, Putah Creek), which suggests they can persist if there are adequate cold water flows. The large population in the upper Sacramento River below Dunsmuir was wiped out by the 1991 Cantara toxic fungicide spill, but showed apparent complete recovery by 1998. Presumably, the reach was recolonized by fish from upstream or from tributaries. Likewise, the population in the North Fork Feather River was able to survive repeated piscicide treatments that were supposed to eradicate "nongame" fish species.

Nature and Degree of Threats: Riffle sculpins are abundant and widely distributed in many streams, although each genetic group has more limited distribution and, consequently, a higher vulnerability to the threats noted here. Most populations are increasingly isolated from other populations and are thus vulnerable to local extinction, with limited potential for recovery. Physiologically, they are exceptionally vulnerable to habitat changes that reduce flows or increase temperatures.

	Rating	Explanation
Major dams	Medium	Dams fragment populations; however, some populations likely benefit from cold water releases below dams
Agriculture	Medium	Agricultural runoff and diversions pollute water and contribute to fragmentation
Grazing	Medium	Grazing can reduce riparian vegetation and negatively affect habitat quality in some streams
Rural residential	Low	Localized effects; impacts largely unknown
Urbanization	Medium	Urban runoff is a source of nutrients and pesticides
Instream mining	Medium	Dredging, currently banned, particularly affects benthic fishes such as sculpin and their habitats
Mining	Medium	Legacy effects of gold mining still impair habitats in many streams within historic distribution
Transportation	Low	Roads can channelize streams and contribute pollutants and sediment
Logging	Medium	Erosion from timber harvest have likely increased the amount of fine sediments in streams, reducing habitat suitability for sculpins
Fire	Low	Wild and human-induced fires can increase sediment delivery to streams and reduce canopy cover and associated shading, often leading to increased stream temperatures
Estuary alteration	n/a	
Recreation	Low	Off-road vehicles and other activities can negatively affect streams but impacts are generally localized
Harvest	n/a	
Hatcheries	n/a	
Alien species	Medium	Absent from waters where alien species are abundant

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

Major dams. Dams occur in virtually every watershed inhabited by riffle sculpins. Because these sculpins cannot use fish ladders designed for salmonids, nor survive in reservoirs, dams effectively isolate populations, preventing recolonization if local populations are extirpated. While cold-water releases below dams create refuges for riffle sculpins, potential cessation of such flows during severe drought may lead to loss of these populations, indicating that their dependence upon such artificially maintained habitats is tenuous. Baumsteiger and Aquilar (2014) found that where riffle sculpins are found below dams, their presence in the river predates dam construction, so each below-dam population represents a further isolation event.

Agriculture. Agricultural diversions and polluted, warm return water make large sections of rivers (e.g., San Joaquin) uninhabitable for riffle sculpins. A growing threat is diversion of water for production of marijuana in many areas throughout their historic range, although direct impacts to fishes and other aquatic organisms need further study.

Grazing. Most headwater streams inhabited by sculpins flow through livestock grazing lands. Cattle reduce riparian shade, trample banks, increase local sedimentation, and generally reduce habitat quality for riffle sculpins.

Rural residential. Many streams are affected by suburban or rural development, resulting in degradation of riparian habitat, effluent from septic tanks, diversions, and other localized, yet cumulative, impacts.

Urbanization. Streams in urban areas are often highly altered for flood control, and many are channelized and polluted from storm water and surface runoff, although protected reaches (especially with coldwater sources) can act as refuges (Leidy et al. 2011). However, most populations in urban areas are isolated in limited areas of suitable habitat.

Mining. Instream mining is largely detrimental to sculpins, given their benthic habitat occupation across all life history stages, as Harvey (1986) demonstrated for gold dredging, a practice currently banned in California. Other effects from mining are mainly legacy effects of hydraulic mining (e.g., elimination of riffle sculpin from the South Yuba River) and polluted drain water from abandoned hardrock mines.

Transportation. Roads and railroads often run along one or both sides of riffle sculpin streams and bridges and/or unimproved roads with culverts cross them. Impacts may include channelization, habitat fragmentation, narrowing of stream channels, increased sedimentation, and increased likelihood of contaminant delivery; the latter was dramatically demonstrated by the 1991 fungicide spill in the Sacramento River, when a train derailed at the Cantara Loop and fell into the river, killing most aquatic organisms in the river for many miles downstream of the spill.

Logging. Timber harvest and associated road development and erosion are common in the riffle sculpin's range, especially in the Sierra Nevada. Such land use increases the likelihood of local extinctions of already fragmented populations.

Alien species. Riffle sculpins are generally absent from stream reaches in which alien fishes, such as smallmouth bass, redeye bass, and brown trout, are common, or even present. This is largely a reflection of habitat quality, because cool water streams tend to favor native species. But it also indicates vulnerability to predation by alien predators.

Effects of Climate Change: Riffle sculpin require cool water habitats that will become increasingly restricted to higher elevations and northern latitudes as stream temperatures increase and summer base flows decrease. During periods of extended severe drought, cold water releases below most dams may disappear, with severe consequences to sculpin populations. As a result, Moyle et al. (2013) rated the riffle sculpin as "critically vulnerable" to climate change.

Status Determination Score = 3.0 - Moderate Concern (see Methods section Table 2). The riffle sculpin has a fragmented distribution and faces many threats that, in combination, could eventually cause extinction of one or more of the genetically distinct population segments (Baumsteiger 2013).

Metric	Score	Justification
Area occupied	5	Riffle sculpin are present in multiple watersheds in four distinct geographical regions
Estimated adult abundance	4	Current abundance is not known but assumed to be locally abundant in a number of streams
Intervention dependence	3	Many stressors threaten the viability and health of riffle sculpin; different for each population
Tolerance	2	Requires high quality cold water environments
Genetic risk	3	Values range from 1 to 4 depending on populations
Climate change	1	All populations exceptionally vulnerable
Anthropogenic threats	3	See Table 1
Average	3.0	21/7
Certainty (1-4)	3	Reasonable knowledge of many populations

Table 2. Metrics for determining the status of riffle sculpin, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See Methods section for further explanation.

Management Recommendations: A major step toward protecting riffle sculpin would include a more extensive study of the genetics, morphometrics, and meristics of sculpins from diverse populations, to determine the identity of cryptic species or subspecies indicated by the work of Baumsteiger et al. (2012) and Baumsteiger (2013). Genetically distinct population segments occupying four geographical areas (San Joaquin drainage, Sacramento drainage, central coast watersheds, and Russian River; Figure 1) have varying levels of vulnerability to extinction although all are threatened, especially by climate change.

A comprehensive assessment and monitoring program should be developed across all four regions to assess abundance and distribution of riffle sculpin and to identify threats to all local populations. Potential refuge watersheds or stream reaches should be evaluated, along with identification of coldwater sources that can sustain populations during severe drought and in the face of climate change. Environmental flows should be provided, including during drought periods, which would protect a viable portion of the population below major dams; such flows would also benefit other native fishes and aquatic organisms.

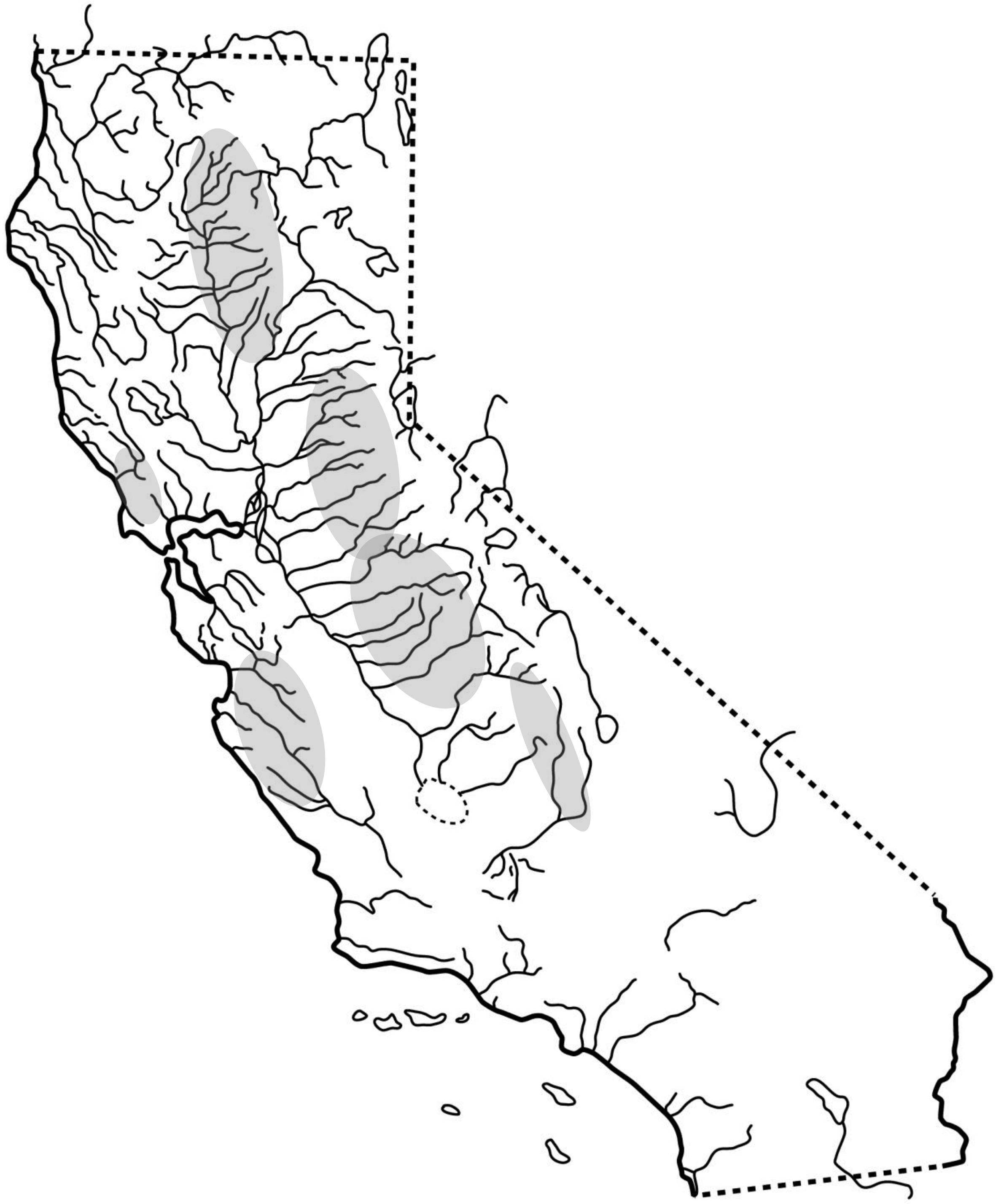


Figure 1: Genetically distinct populations of riffle sculpin (*Cottus gulosus*, Girard) in California (based on Baumsteiger 2013). There are four distinct genotypes: (1) San Joaquin basin and lower Sacramento River (2) upper Sacramento River basin, (3) Pajaro-Salinas basin, and (4) Russian River basin.

SACRAMENTO PERCH
Archoplites interruptus (Girard)

Status: Critical Concern. The Sacramento perch is already extinct in its native range and most, if not all, populations outside its native range could become extinct within the next 100 years.

Description: Moyle (2002, p. 376) describes Sacramento perch as follows: "Sacramento perch are deep-bodied (depth is up to 2.5 times the standard length) and laterally compressed, with long dorsal (12-14 spines, 10-11 rays) and anal (6-8 spines, 10-11 rays) fins. The mouth is large and oblique, with the maxilla extending just below the middle of the eye. Numerous small teeth are present on the jaws, tongue, and roof of the mouth. The 25-30 gill rakers are long. The scales are fairly large, numbering 38-48 along the lateral line. The spiny portion of the dorsal fin is continuous with the soft-rayed portion. Pectoral fin rays number 13-15 while vertebrae number 31-32, intermediate between the counts for bass and sunfish (Maybee 1993). Depending on the watershed in which they occur, live fish are brownish to silvery on the sides and top with a metallic green to purplish sheen and 6-7 brown/black irregular vertical bars on the sides, with the most anterior bar extending down onto the top of the opercula. Their bellies are silvery to white. Breeding males become darker, especially on the opercula, which may turn purple. Males may also develop a distinct silvery spotting that shows through the darker sides, but in females the color is more uniform." Although this color pattern is distinctive, it is also highly variable.

Taxonomic Relationships: As the only member of the family Centrarchidae native to waters west of the Rocky Mountains, the Sacramento perch is unique. All existing populations (unless there is still a population in Clear Lake) are derived from introductions. As a result, all populations are inbred to varying degrees and each population is genetically distinct from one another (Schwartz and May 2008, Crain and Moyle 2011).

Life History: The life history of Sacramento perch is reviewed in Crain and Moyle (2011) and the information here is condensed from that account.

Sacramento perch spawn for the first time in their second or third year of life, depending on size. They are highly fecund, with females producing up to 125,000 eggs, although most females (12-20 cm FL) produce 8,000-20,000 eggs per year. Spawning takes place at water temperatures of 18-28°C from the end of March through October, although most spawning takes place in March and April. Spawning is typically initiated when males move into shallow water (15-60 cm deep, although spawning has been observed down to 3 m depth) and set up territories over beds of aquatic macrophytes, rocks covered with filamentous algae, or flooded terrestrial plants. Each male typically clears out a depression or other area which is surrounded by plants. These territories are set up in loose aggregations and are defended from other males, as well as potential egg predators.

Females swim in groups close to the spawning area and actively seek out territorial males for spawning. Following a brief courtship period, each female is accepted by a male. Spawning occurs when both the male and female turn on their sides, with vents in close proximity, releasing eggs and sperm. The fertilized eggs attach to the vegetation or other debris in or around the nest. Males guard the nest for 2-4 days after spawning.

Embryos hatch in less than 72 hours, depending on temperature, and in another 2-4 days the larvae (<2mm TL) are able to swim freely. The larvae have a small filament attaching their heads to the egg capsules for 1-4 days. After the attachment is lost, larvae remain in the substrate for another 2-4 days before swim-up. Once they begin actively swimming, larvae either become planktonic or (mostly) live among aquatic plants. Small juveniles (15 -50 mm TL) shoal together in shallow water, gradually moving into deeper water as they grow larger. Individuals eventually become solitary or form only loose aggregations, usually in association with submerged tree branches or other types of structure.

Growth rates depend on temperature, food availability and other environmental conditions. At the end of years 1, 2, 3, 4, 5, and 6, fish are typically 6-13 cm FL, 12-19 cm, 17-25 cm, 20-28 cm, 21-32 cm, and 28-36 cm, respectively (Moyle 2002). Perch can live as long as nine years and reach 61 cm TL and 3.6 kg. The oldest fish known (9 years) were from Pyramid Lake, Nevada, with lengths ranging from 38-41 cm FL. However, the largest fish caught by angling was 43 cm TL, weighing 2.2 kg. Females grow faster and larger than males. Females also have higher survival rates after their first year, so fish that are four years and older tend to be females.

Sacramento perch are ambush predators that feed upon invertebrates and fish, with prey size increasing with mouth gape. Their mouth is quite large, so they can feed on relatively large prey in relation to their body size. Larvae and small juveniles feed on planktonic crustaceans and early instars of insects, especially midges and mosquitoes. Although juveniles are fairly opportunistic, they typically feed on chironomid midge larvae and pupae, as well as amphipods. Larger insects and small fish become increasingly important in the diet of larger perch and those >9 cm FL feed almost exclusively on fish, especially minnows and other soft-rayed native fishes. They feed most actively at dawn and dusk.

Habitat Requirements: This section is based on the studies of Woodley (2007) and the review of Crain and Moyle (2011).

Sacramento perch are adapted for life in sloughs, slow moving rivers, and large lakes, including floodplain lakes, of the Central Valley. These habitats often become very warm and alkaline during periods of drought or in late summer. Their distribution in such habitats led 19th century biologists to conclude that Sacramento perch actually preferred harsh conditions including high alkalinity and salinity. As a result, Sacramento perch were planted as game fish in alkaline waters throughout the western United States. Recent studies suggest that, while Sacramento perch have considerable capacity to survive under such conditions, their preferred habitats are in rivers, large lakes, and estuaries that are fairly cool and fresh much of the year.

Sacramento perch can live in alkaline waters (pH 8-10), but tend to have physiological problems when alkalinities reach 1500 mg/L, with reproduction ceasing at 2000 mg/L. However, they can withstand salinity levels of 24-28 ppt and can grow at salinities in the 10 ppt range, suggesting that they once lived, in part, in estuarine habitats.

In the laboratory, juveniles tolerate temperatures of 7-37°C, including withstanding abrupt temperature shifts of 11-16°C; optimal temperatures for growth are 18-23°. Adult perch require somewhat cooler water, with upper tolerance limits of approximately 29°C. Optimal temperatures for growth appear to be about 15-22° C. Both adults and juveniles can live in lakes that ice over in winter, so they can persist through periods of low temperature as well. While Sacramento perch appear to require cooler water than most other centrarchids, their oxygen requirements at a given temperature are lower, so they can survive relatively low dissolved oxygen conditions for extended periods of time. Likewise, Sacramento perch have a greater capacity to swim in flowing water than similar deep-bodied centrarchids. These attributes suggest that the historic habitats of Sacramento perch were varied and included alkaline valley floor lakes, rivers, floodplains, and estuaries.

Distribution: The historic range of Sacramento perch has been determined from limited collection records and remains in middens left by native peoples. Their range included the Tulare and Buena Vista basins to the south, the San Joaquin River basin, the San Francisco Estuary and its tributaries, and the Sacramento Valley (Moyle 2002, Crain and Moyle 2011). Other populations existed in the Pajaro-Salinas drainage and in Clear Lake, Lake County. The Central Valley populations were distributed in valley floor waters and, presumably, perch did not ascend streams more than a few hundred meters in elevation. It is possible a population also once existed in the Russian River but evidence is equivocal.

Sacramento perch have been widely introduced outside their native range, mainly to alkaline waters where other game fishes generally do not survive. In California, populations were established in stock ponds (no recent records of establishment outside of Yolo County), in the Owens Valley (mainly in Crowley Reservoir), in the Walker River watershed (mainly in Bridgeport Reservoir), in the Cedar Creek drainage (West Valley and Moon reservoirs), in Clear Lake Reservoir within the Lost River drainage (spreading into the Lost River, Copco Reservoir, and Sheepy and Indian Tom lakes), in Abbott's Lagoon in Point Reyes National Seashore, and in a few other small reservoirs (Table 1). Declines have occurred in many of these reservoirs. Moyle (2002) recorded their presence in 28 waters in California (22 if the four Upper Klamath and two Cedar Creek populations are lumped together as one population), but Crain and Moyle (2011) determined that they have been extirpated from at least eight of these waters (Table 1). If the six populations of unknown status are counted as extirpated, which is likely, then the total number of populations in California is 22 (16 independent). Outside of California, as of 2008, nine populations existed in Nevada, one in Utah, and one in Colorado. In all, there are 25 independent populations, mostly in reservoirs, still known to exist with a high degree of certainty as of 2008.

Location	County	Status (2008)
Calaveras Reservoir	Alameda/Contra Costa	Extirpated
Alameda Cr. gravel ponds	Alameda	Extirpated
Lake Anza	Contra Costa	Extirpated
Jewel Lake	Contra Costa	Present
Lagoon Valley Reservoir	Solano	Present
Hume Lake	Fresno	Present
Sequoia Lake	Fresno	Present
San Luis Reservoir	Merced	Present
Middle Lake	San Francisco	Extirpated
Almanor Reservoir	Plumas	Present
Butt Valley Reservoir	Plumas	Unknown
Abbott's Lagoon	Marin	Present
Sonoma Reservoir	Sonoma	Unknown
West Valley Reservoir[1]	Modoc	Present
Moon Reservoir	Lassen	Present
Honey Lake	Lassen	Unknown
Clear Lake Reservoir	Modoc	Present
Lost River and Tule Lake	Modoc	Present
Copco Reservoir	Siskiyou	Present
Sheepy and Indian Tom Lake	Siskiyou	Unknown
Bridgeport Reservoir	Mono	Present
East Walker River	Mono	Present
West Walker River	Mono	Unknown
Topaz Lake	Mono	Unknown
Gull, June, Silver, and Grant Lakes	Mono	Present
Crowley Reservoir	Mono	Present
Lower Owens River, Pleasants Valley Reservoir	Mono	Present

Table 1. Major water bodies listed as containing Sacramento Perch in California in the 1990s by Moyle (2002) with a determination of status in 2008. Populations labeled unknown are likely extirpated.

Trends in Abundance: Sacramento perch have been in a steady decline since the 19th century, when they were once abundant enough to be fished commercially to supply the San Francisco markets. Prior to that, perch were a major food source for Native Americans who lived in the Central Valley. The decline of Sacramento perch was noted by the early 20th century and it was regarded as scarce in its native range by the 1930s. However, little attention was paid because they had been widely introduced outside their native range, replaced within the native range by desirable non-native centrarchids, and restoration to historical abundance was deemed unlikely because alien species were

[1] West Valley Reservoir and Moon (Tule) Reservoir are both in the Cedar Creek watershed so are interconnected. The population was apparently extirpated in the 1980s when water levels were low and the reservoirs became ice-covered in winter. Sacramento perch were subsequently reintroduced (P. Chappell, CDFW, pers. comm. 1995).

perceived to be the principal cause of decline. In the 1950s and 1960s, agency biologists planted Sacramento perch in isolated lakes and reservoirs around the state but they were extirpated from most of the native range by the 1970s, except for a population in Clear Lake (Crain and Moyle 2011). At present, there are 25 confirmed independent populations, nine of them in three other western states. In California, it appears that six populations have been lost in the recent past. In addition, a few populations have been established in farms ponds in Yolo County, including the UC Davis campus, but such populations are ephemeral (P. Crain, UCD, unpublished data, 2010).

The sizes of existing populations are unknown but some are apparently very small (Jewel Lake) while others (Lost River basin and Crowley Reservoir) may be quite large. All have limited genetic diversity, however, because of the small numbers of fish used to start each initial population (Schwartz and May 2008, Crain and Moyle 2011). Each isolated population is different from others genetically and, as populations are lost, the genetic diversity of Sacramento perch is further reduced.

Nature and Degree of Threats: Sacramento perch have declined because of the combined effects of habitat loss and interactions with alien species (Table 2). Given their physiological tolerances, it is likely they would persist today in parts of their native range in the absence of alien fishes. However, their native valley floor habitats were already so heavily altered by the late 19th century, their distribution and abundance were likely severely restricted even prior to introduction of alien fishes.

Major dams. The decline of Sacramento perch in its native range coincided with the construction of dams, including the large 'rim' dams, around the Central Valley. The capture and export of water by these dams was accompanied by conversion of habitats in valley floor rivers and lakes to farms and cities, especially through the draining of wetlands and construction of dikes and levees, which isolated rivers from their floodplains.

Agriculture. Recent physiological and behavioral studies suggest that Sacramento perch were especially well adapted for living in large valley floor rivers and spawning on floodplains (Crain and Moyle 2011). Farming permanently and severely altered these habitats, including drying of the San Joaquin River, draining Lake Tulare and Lake Buena Vista, channelizing the Sacramento River, and building vast networks of levees to protect 'islands' (now sinks often many meters below water level) across the Delta. In addition, agricultural return waters are often warm and laden with pesticides and fertilizers, creating poor water quality for most fishes, including Sacramento perch.

Urbanization. Many California cities are built on floodplains along major rivers in areas that were once habitat for Sacramento perch and other native fishes. The impacts of massive urban and suburban expansion post-WWII in California upon already diminished populations of Sacramento perch likely contributed substantially to their fragmentation and further decline.

Mining. Hydraulic mining had an enormous impact on foothill rivers and valley floor habitats in the 19th century and severely altered many riverine habitats. Although relatively short-lived, the legacy of hydraulic mining still affects habitat quantity and quality for Sacramento perch and other fishes, potentially limiting their distribution and capability of expansion into formerly suitable habitats.

Estuary alteration. The Sacramento-San Joaquin Delta and brackish areas of the rest of the San Francisco Estuary were once major habitats of Sacramento perch, as indicated by large numbers taken in 19th century fisheries. The decline of perch was coincident with the loss of complex estuarine habitats, especially in the Delta, as well the introduction and establishment of non-native centrarchids.

	Rating	Explanation
Major dams	Medium	Dams contribute to reduced and highly manipulated flows and allowed for development of farms and cities in the past
Agriculture	High	Agriculture is a dominant land use across range; diversions, low quality return water, channel alteration, draining of lakes, levee building and other impacts are pervasive
Grazing	Low	Some impacts in lowland areas on aquatic habitats, although most grazing occurs at higher elevations than historic perch range
Rural residential	Medium	Rural development has expanded dramatically across the range of Sacramento perch, contribution to habitat degradation and simplification
Urbanization	Medium	Cities contribute to extensive alteration or loss of habitats, input of pollutants and municipal water demand
Instream mining	Low	Placer and gravel mining presumably altered habitats
Mining	Medium	Hydraulic mining in 19th century may have affected populations and legacy effects remain in many areas
Transportation	Low	Most habitats lined with roads, etc.; potential sources of pollutant inputs
Logging	Low	Impacts mostly indirect from sedimentation; logging largely occurs at higher elevations than historic perch range
Fire	n/a	
Estuary alteration	Medium	Once abundant in complex habitats of Delta; habitats now greatly altered and floodplains mostly disconnected from rivers
Recreation	n/a	
Harvest	Medium	Heavy harvest in 19th century may have contributed to initial declines
Hatcheries	n/a	
Alien species	Critical	Greatest cause of decline; especially acute in the Delta

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

Harvest. Sacramento perch were an important food fish in San Francisco fish markets in the 19th century, with 40,000-432,000 pounds of fish harvested per year (Skinner 1962). It is likely that these fish came from the lower Sacramento River and Delta. It is possible that heavy harvest contributed to their decline, making it easier for other species of fish to invade.

Alien species. The negative effects of alien species on Sacramento perch populations have long been documented. Jordan and Evermann (1896) thought their decline was due to carp and catfish "infesting their spawning grounds." Alien centrarchids, especially bluegill and black crappie, spread throughout California in the early 20th century. Their similarity in ecology and spawning habits to Sacramento perch was noted and, combined with their more aggressive behavior, they were consequently thought to eliminate Sacramento perch wherever they came in contact. Given the rather weak and short protection time of the nest given by male Sacramento perch, it is highly likely that embryo and larval predation by alien fishes has played a major role in their decline (Crain and Moyle 2011). In general, the only habitats where Sacramento perch persist today are those that lack alien sunfish and crappie.

Effects of Climate Change: The Sacramento perch exists mainly in populations in reservoirs or ponds, many of them quite small. Reservoir populations are subject to widely varying habitat quality and potential desiccation of the reservoirs during extended drought periods or when reservoirs are drawn down for dam repairs or other purposes. For example, Crowley Reservoir (Mono County), could drop to low levels and become too alkaline for perch reproduction, as could Bridgeport Reservoir (Mono County). In addition, climate change will likely cause additional stress on remaining Sacramento perch populations through increasing water temperatures and increasing alkalinities as lake levels drop. Moyle et al. (2013) rated the Sacramento perch as 'highly vulnerable' to climate change because of the likely impacts of drought and other factors on their limited, mostly artificial, habitats.

Status Determination Score = 1.9 - Critical Concern (see Methods section Table 2). There are only 25 isolated populations of Sacramento perch remaining and these populations have been declining or disappearing at a steady rate in recent decades. In addition, these isolated populations have relatively low genetic diversity. The American Fisheries Society considers Sacramento perch to be Threatened (Jelks et al. 2008), while NatureServe lists them as Vulnerable (G3). Sacramento perch were included as a declining species in the Delta Native Fishes Recovery Plan (USFWS 1996). Overall, the Sacramento perch appears to be extinct in its native range with most outside populations declining, genetically bottlenecked, and restricted to artificial and potentially insecure bodies of water.

Metric	Score	Justification
Area occupied	1	One native population may remain in historic range (Clear Lake); status unknown
Estimated adult abundance	3	Existing populations are limited and isolated but some appear to be fairly large (sizes unknown)
Intervention dependence	1	Re-establishment in native range requires active rearing program and isolation of suitable habitats from alien species
Tolerance	4	Very tolerant, except of extremely warm water
Genetic risk	2	All populations bottlenecked
Climate change	1	Drought and increasing temperatures will have negative effects on their limited, mostly artificial, habitats
Anthropogenic threats	1	See Table 1
Average	1.9	13/7
Certainty (1-4)	4	

Table 3. Metrics for determining the status of Sacramento perch, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: The following ten recommendations are from a conservation proposal developed by Moyle and Crain (2011, p. 30):

1. Establish backup populations for all existing populations, including those outside of California. Ideally, these would be located in habitats within the native range of Sacramento perch but ponds or lakes under controlled conditions are probably necessary.

2. Re-establish a genetically diverse source population for future planting programs through a program that brings genotypes together from isolated populations. This program would have to be implemented under carefully controlled conditions with genetic monitoring of fish produced as new source stock.

3. In order to accomplish recommendations 1 and 2, establish a Sacramento perch rearing facility in the Central Valley, with facilities for selective breeding and ponds for large-scale rearing of fish for planting where suitable habitats exist. It may be necessary to maintain this facility indefinitely as a source of Sacramento perch for recreational ponds and reservoirs and as an insurance policy for wild populations.

4. Reintroduce fish into habitats that are determined to be suitable (e.g., other species present/absent, appropriate environmental conditions). Physiological and ecological studies suggest that habitats may exist, from which Sacramento perch were extirpated decades ago, that have changed enough so that they may once again be suitable. Some of these habitats are listed in Crain and Moyle (2011).

5. Develop a strategy to use floodplain ponds in order to allow Sacramento perch to colonize natural environments during periods of flooding, linked with a more general

strategy to develop flow regimes and habitats below dams that are generally more favorable to native fishes. A successful reintroduction will require a fairly large number of fish, distributed across a broad geographic area. This strategy could take advantage of previous studies of restoration of flooded habitat on the McCormick-Williamson Tract (CALFED project #99-B193) and the Cosumnes River Floodplain (CALFED Project #99-N06) (Moyle et al. 2007).

6. Develop a source-sink strategy by locating rearing ponds next to streams or sloughs so the ponds can 'leak' Sacramento perch on regular basis into natural habitats. Populations of Sacramento perch have been established in ponds on the UC Davis campus and small numbers of perch now occur in nearby Putah Creek which, presumably, was colonized via interconnected drainage canals.

7. Rear Sacramento perch in large numbers in ponds and other artificial facilities for large-scale introduction into the wild. This is the least desirable options but may be necessary if a large propagule size is required for re-establishment in the wild. This strategy may be especially important for re-establishment or bolstering of Sacramento perch populations in Clear Lake, Lake County, historically one of the last strongholds of wild Sacramento perch in their native range.

8. Conduct comprehensive trawl and seine surveys of Clear Lake to determine if Sacramento perch remain, estimate their abundance, assess population structure, and potentially acquire tissue samples for genetic analyses. If surveys indicate that Sacramento perch exist in Clear Lake in low abundance, captured perch should be taken into captivity so they can be protected and propagated.

9. Develop and maintain an annual monitoring program for all known Sacramento perch populations in California. Monitoring will provide crucial information as to which populations are either maintaining themselves or declining. Genetic monitoring of wild populations should be performed in concert with population monitoring.

10. Promote the use of Sacramento perch in recreational fisheries, especially farm ponds and city fishing programs. Such a program could both acquaint the public with an edible native sport fish and increase the likelihood of Sacramento perch being maintained in greater numbers of private ponds. This, in turn, would increase the probability that some may escape to the wild.

Figure 1. Distribution of Sacramento perch, *Archoplites interruptus* (Girard), in California. All populations shown are introduced outside the historic range.

RUSSIAN RIVER TULE PERCH
Hysterocarpus traskii pomo (Hopkirk)

Status: Moderate Concern. Populations of Russian River tule perch are large but remain of concern because the subspecies is endemic to one highly altered river system.

Description: Tule perch are small (up to 150 mm SL), deep-bodied fish that are green, bluish or purple dorsally, and white to yellow ventrally. Three color variants are described, based on their lateral barring patterns: wide-barred, narrow-barred, and bars absent. The narrow-barred color variant predominates (99%) in the Russian River population, with few broad-barred (1%) fish (Hopkirk 1973). The unbarred variant is absent. Bars on Russian River fish may be bright yellow (Chase et al. 2005). Adults have a pronounced hump (nuchal concavity) immediately anterior to the dorsal fin. The dorsal fin has 15-19 spines and 9-15 rays; the anal fin, 3 spines and 20-26 rays; the pectoral fins, 17-19 rays. There are 34-43 scales along the lateral line (Moyle 2002). Body proportions and gill-raker morphology of Russian River tule perch differ from the other two subspecies in California (Hopkirk 1973, Moyle and Baltz 1981).

Taxonomic Relationships: The tule perch is the only freshwater species in the marine family Embiotocidae. Russian River tule perch, *Hysterocarpus traskii pomo,* were described by Hopkirk (1973) as one of three subspecies. Morphometric analyses by Baltz and Moyle (1981) showed that *H. t. pomo* is different from *H. t. lagunae* (from the Clear Lake drainage basin) and from *H. t. traskii* (from the main Sacramento-San Joaquin drainage). The three subspecies also show genetic divergence (Baltz and Loudenslager 1984), as well as striking differences in life-history patterns (Baltz and Moyle 1982).

Life History: Tule perch are the only viviparous (live-bearing) native freshwater fish in the state. Like other members of the predominantly marine family Embiotocidae, females produce young that are surprisingly large considering the size of the mother. As a result, females have reduced swimming abilities while pregnant.

Russian River tule perch are adapted to a flow regime that varies widely by both season and year (Baltz and Moyle 1982). Because flows in the Russian River are driven by the heavy winter rains and dry summers of California's Mediterranean climate, flows are high in winter but, for six months or more (June- October), there is little rainfall and the river drops to minimum flows. Currently, the Sonoma County Water Agency (SCWA) maintains minimum summer flows at 125 cfs by releasing water from Sonoma Reservoir into Dry Creek (a tributary to the Russian River) and from Lake Mendocino on the East Fork of the Russian River, which is augmented by Eel River water via the Potter Valley Project (PVP). Before the PVP was implemented in 1923, portions of the lower Russian River likely became intermittent in the late-summer/early-fall of dry years and flow would become subsurface between large pools. Because rainfall in this region shows extreme variation from year to year, peak flows are unpredictable both in extent and timing. Following heavy storms, stream flow may peak rapidly and the river often floods.

This highly variable flow pattern resulted in the evolution of a life history quite different from that of other tule perch populations, one which reflects low survival rates of fish in most years (Baltz and Moyle 1982). High winter flows presumably flushed fish, particularly pregnant females, into poor habitats. During periods of drought, small, shallow pools and other habitats would become stagnant or too warm to support tule perch. Although deep, cool water refugia would have existed in larger pools, limited suitable summer habitat likely restricted population size, especially of adults.

The reproductive strategy of Russian River tule perch is an adaptation to this unpredictable environment (Baltz and Moyle 1982). They are relatively short-lived (typically <2 years, maximum 3-4 years), compared with the two other subspecies. The viviparous females produce more young per brood and reproduce at smaller sizes than those of other subspecies. Mating occurs from July through September and sperm is stored within the female until January, when fertilization takes place. During the mating season, males may hold and defend territories, usually under overhanging branches and among plants close to shore. Courtship and mating can, however, occur away from territories (Moyle 2002). Young are born during May-June, when food is abundant in most years (Moyle 2002). The young are released into areas with complex cover and remain associated with such cover for their first summer, often in daytime aggregations of dozens of individuals.

Except when breeding, tule perch are gregarious and adults forage and swim in small groups while smaller fish congregate in larger groups. The terminal mouth of Russian River tule perch, with its protrusible upper jaw and coarse gill rakers, is adapted for feeding on a wide variety of benthic and plant-dwelling aquatic invertebrates (Baltz and Moyle 1981). The number and length of gill rakers of this subspecies are intermediate to the two other subspecies. The lake-dwelling *H. t. lagunae* has a greater number of longer gill rakers and feeds on zooplankton, while *H. t. traskii* feeds largely on benthic invertebrates (Baltz and Moyle 1981).

Habitat Requirements: This subspecies requires clear, flowing water (Cech et al. 1990) and abundant cover, such as beds of aquatic macrophytes, submerged tree branches, overhanging plants, and large boulders. Large cover is essential for near-term females and young, serving as refuge from predators and velocity associated with high flow events. Although Russian River tule perch sometimes feed in riffles or in flowing water at the heads of pools, they congregate in deep (>1 m) pools during summer and will use rip-rap and fallen trees in deep water for cover. They are usually absent from reaches with poor water quality.

With the exception of Clear Lake, tule perch are rarely found in water where temperatures exceed 25˚C for extended periods of time; they generally prefer temperatures below 22˚C (Knight 1985). Indicative of the surfperch family's physiology, tule perch have high salinity tolerance. Sacramento tule perch thrive in salinities that fluctuate annually from 0 to 19 ppt and have been found at salinities as high as 30 ppt. Presumably, Russian River tule perch have similar tolerances because they are consistently found in small numbers in the Russian River estuary where salinity levels fluctuate from 0 to as high as 32 ppt (Cook 2006). However, tule perch in the estuary seem to inhabit plumes of relatively fresh water at the mouths of tributaries, often remaining near the surface.

Distribution: This subspecies is confined to the Russian River and its tributaries in Sonoma and Mendocino counties, California (Hopkirk 1973). Recent sampling from 1991-2009 (Figure 1) has documented tule perch in the main stem Russian from Ukiah (Mendocino County) downstream to the river mouth near Jenner (Sonoma County), as well as in the lower reaches of tributaries (Fawcet 2003, Cook 2003, Chase et al. 2005, Cook 2006, Cook et al. 2010). Historical records (Figure 2) exist from the North Fork above the present day location of Lake Mendocino; however, recent surveys have failed to document tule perch in the North Fork above or below the lake (Cook et al. 2010).

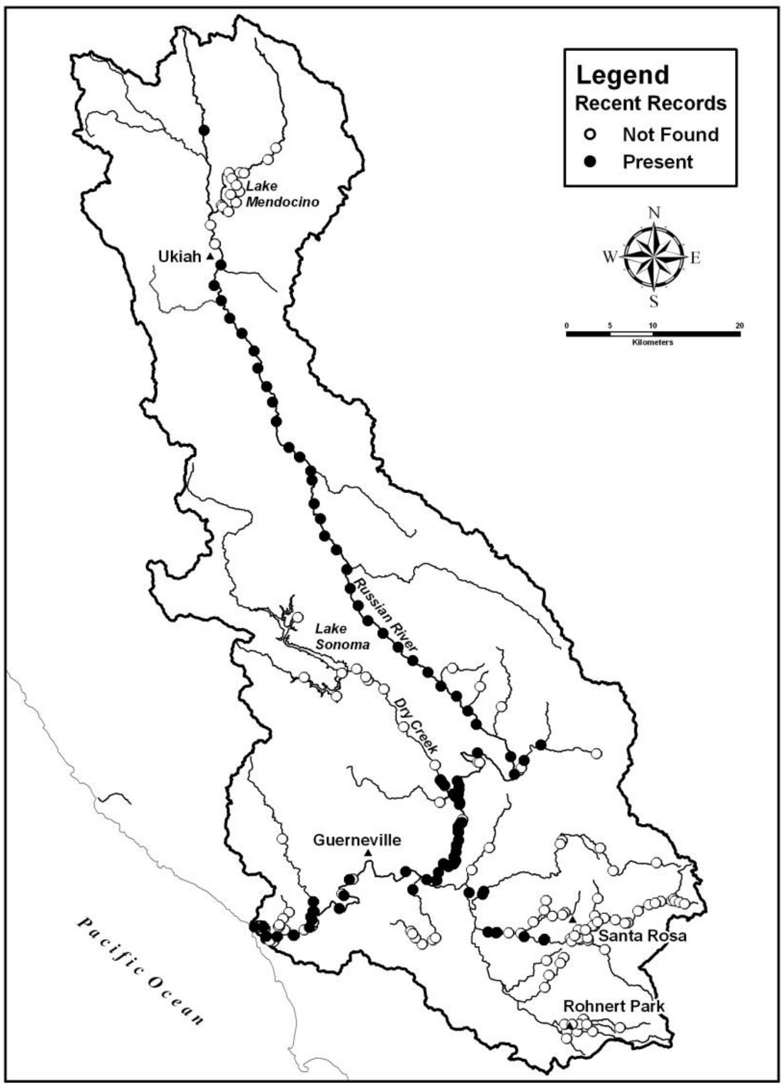


Figure 1. Recent distribution of Russian River tule perch based on records from 1991-2009. Figure from Cook et al. (2010).

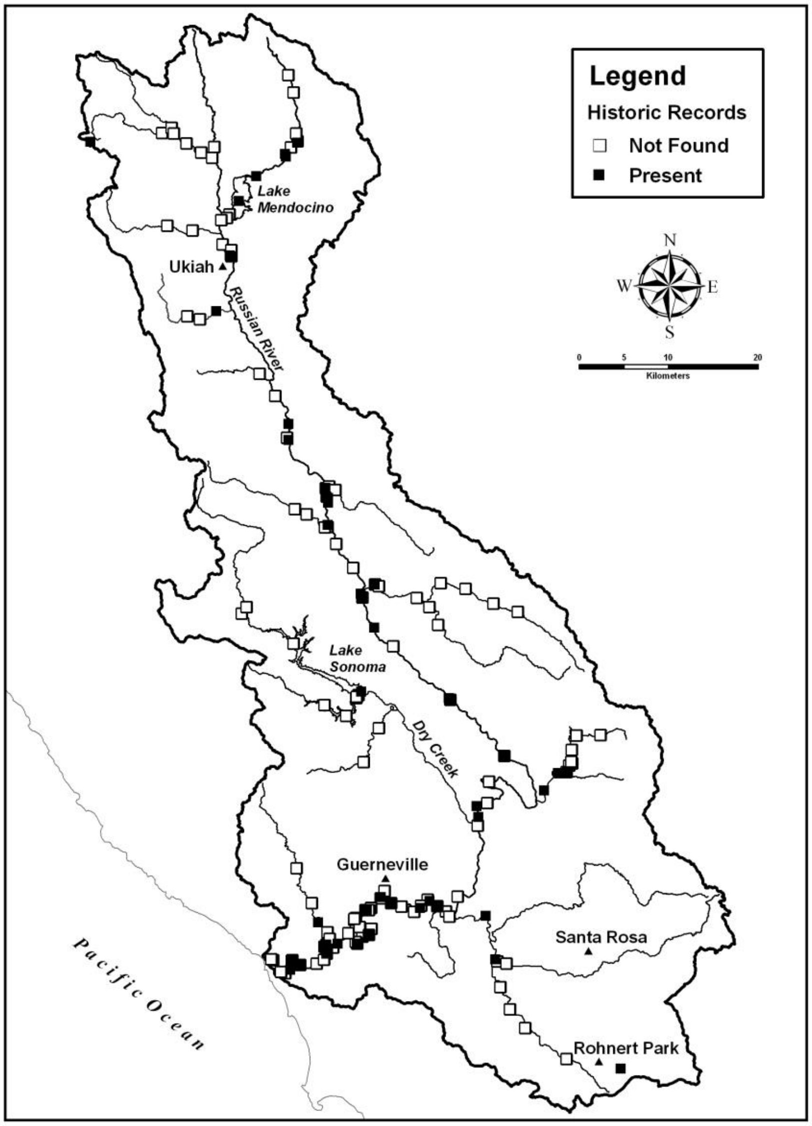


Figure 2. Historic distribution of Russian River tule perch based on records from 1897-1990. Records within the footprint of Lake Sonoma and Lake Mendocino are prior to reservoir construction. References include: Hopkirk (1973), Pintler and Johnson (1958), and unpublished data from the Sonoma County Water Agency. Figure from Cook et al. (2010).

Trends in Abundance: Extensive sampling of the Russian River by SCWA from 2000-2004 revealed that tule perch were widely distributed in the river and fairly abundant. In a 2003 snorkel survey of the upper Russian River from Coyote Dam (Mendocino Reservoir) to the confluence of Dry Creek below Healdsburg, 5,657 tule perch were counted. Tule perch accounted for between 3% and 9% of fish observed in each surveyed reach (Cook 2003). A total of 37 segments were sampled, which equaled approximately 18% of the upper Russian River. Tule perch appear to be even more common in the middle river, between Healdsburg and Forestville, where they made up 17% (329 tule perch of 1902 fish) of the catch in electro-fishing sampling conducted by SCWA in 2004 (Chase et al. 2005). From 2003-2005, tule perch were caught in beach seine-net surveys of the Russian River estuary. Fish densities appeared to be highest near the mouth of perennial Austin Creek, where salinities remained near 0 ppt. Downstream, tule perch abundance decreased as salinity increased (Cook 2005, 2006). Because of vertical stratification of fresh and saline waters in the estuary, exact salinities at the locations of capture could not be determined.

In the mid-1950s, as part of a project aimed at "steelhead trout habitat improvement," the California Department of Fish and Wildlife (CDFW) performed chemical (rotenone) treatments of the Russian River and larger tributaries, in an effort to reduce presumed competition between native nongame fishes and steelhead and salmon. During this effort, tule perch represented 3% of the fish eradicated in the stretch between Ukiah and Healdsburg and 3.5% from Healdsburg downstream to Duncans Mills (Johnson 1958). In 1979, Hopkirk found that tule perch accounted for only 1% of his catch in a beach seine survey (Hopkirk and Northen 1980). A seine survey of 15 sites between Hopland and Jenner conducted in 1984 found tule perch accounted for 1.5% of the total catch (Cox 1984). Another seine survey in 1988 also found that tule perch were uncommon compared to other fishes in the river (A. Phelps, unpublished M. S. thesis).

It is likely that sampling bias accounts for much of the discrepancy between historic and modern relative abundance estimates. Tule perch favor habitat around heavy structure and vegetation, precisely the habitats most difficult to sample with a seine. Because most historic surveys were conducted with seines, it seems likely that tule perch were under-represented in catch reports until surveys by SCWA, beginning in 2000, utilized more efficient snorkeling and electrofishing methods.

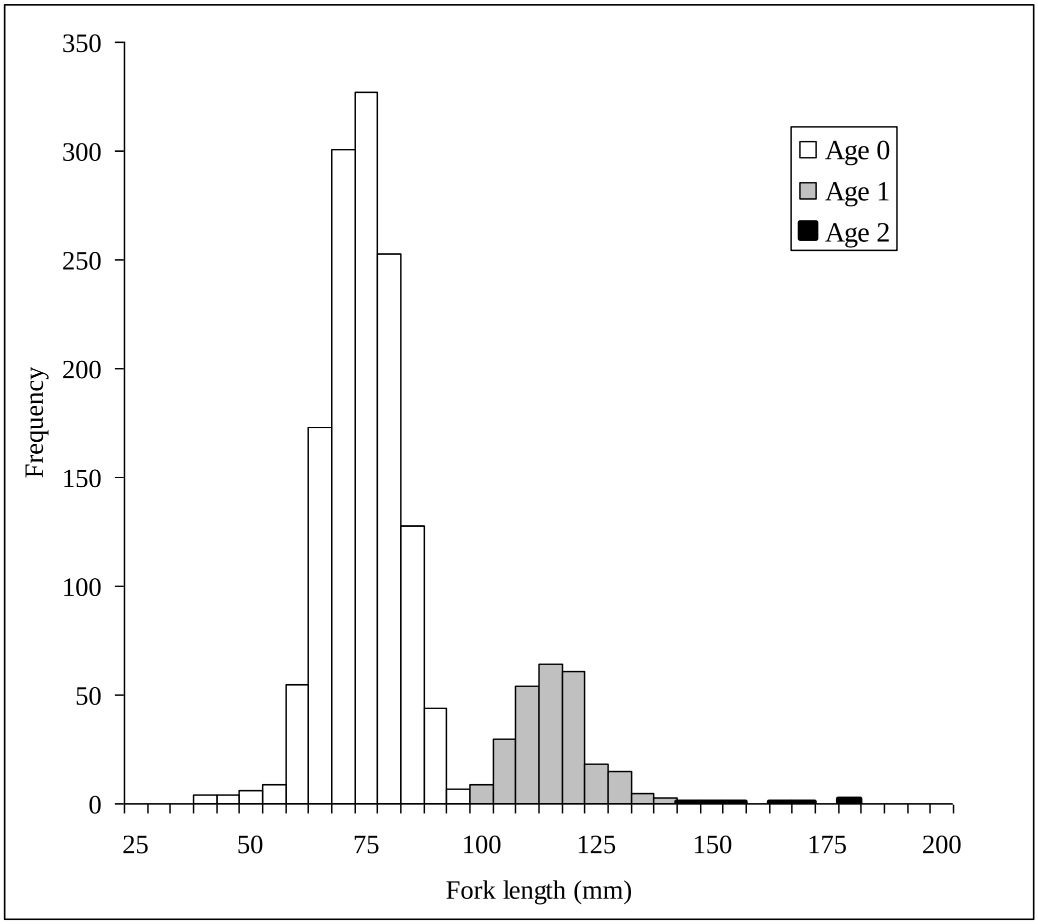


Figure 3. Length frequency histogram for tule perch in Wohler Reservoir, Russian River basin. Fish were collected annually in August by boat electrofishing from 1999-2004 and 2006 (Chase et al. 2005; Chase unpublished data). Fish captures included 1,435 (age 0; young-of-year), 286 (age 1), and 7 (age 2). Figure from Cook et al. (2010).

Nature and Degree of Threats: The limited distribution and short life span of Russian River tule perch makes their populations vulnerable to a number of factors that could reduce their numbers (Table 1). The most important threats to their persistence are: (1) regulation and alteration of stream flows, (2) pollution, (3) changes in water quality, (4) alterations to habitats, (5) gravel mining, and (6) alien species. These threats are not listed in order of severity and should be viewed as cumulative and synergistic impacts that, in combination, can threaten tule perch populations and other native fishes of the Russian River system.

Flows. Since the construction of Coyote Dam on the East Fork of the Russian River (1959) and Warm Springs Dam on Dry Creek (1983), flows in much of the watershed have become more predictable, with a decrease in frequency and duration of high flow events and an increase in summer base flows. The increased summer flows are partly the result of water being diverted into the Russian River from the Eel River, through the PVP, which began in 1923. While data are lacking, it is likely that tule perch populations have benefited from reduced flow variability due to increased survival of pregnant females and juveniles. However, the long-term effects of this highly controlled flow regime are uncertain given that: (1) the PVP may be shut down or operations modified at some time in the future, (2) summer flows may be reduced under new management guidelines aimed at improving juvenile salmonid rearing habitat, and (3) stabilized flows may increase populations of alien fishes.

Pollution. Tule perch are a remarkably resilient species, given that they survived large-scale chemical treatments of the Russian River in 1952 –1954, 1958 and 1963 by CDFW, which were aimed at reducing abundance of all non-game fishes in the river. While chemical treatments of this nature, scope and magnitude are unlikely to occur again, other events such as pesticide and oil spills from accidents on Highway 101 (which parallels long sections of the river) or more chronic and pervasive inputs from agricultural return waters (especially from viticulture) may pose ongoing threats. Pollution from waste water may be a specific threat to tule perch because females can pass heavy metals (e.g., mercury) and pesticides they accumulate directly to their young.

Water quality. Dam regulation and associated summer flow increases have improved water quality for tule perch by decreasing temperatures in some areas and diluting pollutants. However, this benefit is likely to diminish as diversions increase to meet growing agricultural and municipal water demands, including pumping of ground water. Human development of the Russian River watershed and landscape conversion to agriculture (especially viticulture) is rapidly increasing and water quality in the river may decline as a consequence, without strict controls on both water removal and effluent in water returns.

Habitat modification. The Russian River and its tributaries are increasingly confined by levees and bank stabilization projects designed to reduce the natural tendency of streams to meander and cut into agricultural fields, roads and towns. Much of the river is lined with a highway or road on at least one bank, increasing the tendency to stabilize banks wherever possible. Rip-rap, summer dams and other structures may actually create favorable habitat for tule perch in the short-run; however, longer-term simplification of habitats (e.g., decreasing pool size and depth, removal of trees that fall into the river for flood control and safety, instream gravel mining) will ultimately reduce the amount of suitable tule perch habitat.

Instream mining. Deep gravel mining pits that are separated from the river channel by narrow levees can be captured by the river during flood events. Such “pit capture” has the potential to significantly alter the hydrologic function of the entire middle reach of the Russian River and poses a threat to tule perch habitat. Flooded mining pits often harbor populations of alien species that, under flood conditions, can escape from mining pit habitats into adjacent rivers or streams. Removal of surface gravel from bars (skimming) may also reduce habitat complexity and change flow patterns.

Mining. Legacy effects of mercury and other hardrock mining still exist but appear to be currently minor. The presence and ongoing input of residual mercury in Russian River aquatic food webs may disproportionally affect tule perch, since females can pass bioaccumulated mercury to their young. Increased demand for crushed rock for use as aggregate has enlarged rock quarries, amplifying sedimentation risks from these sources.

Alien species. Although tule perch, in general, seem to coexist with alien species better than most other native fishes, introduced predators already present such as smallmouth bass (*Micropterus dolomieui*) may limit tule perch distribution, especially if flow and habitat changes increasingly favor smallmouth and other alien species, including striped bass (*Morone saxatilis*) or other black bass species (*Micropterus spp.)*.

	Rating	Explanation
Major dams	Low	Decreased flow variability and increased summer flows likely benefit tule perch; long-term negative impacts possible
Agriculture	Medium	Water withdrawals and polluted return water impair water quality; bank protection reduces cover
Grazing	Low	Still occurs in many areas but more extensive and greater threat in the past
Rural residential	Medium	Increasing water withdrawal by residential users from tributaries and groundwater aquifers decrease surface water quality and quantity
Urbanization	Medium	Urban water use affects quality and quantity of stream flows; urban development generally simplifies aquatic habitats, reduces habitat quality and quantity, and contributes to pollutant input
Instream mining	Low	Gravel mining can simplify habitats and increase turbidity, as can instream bar-skimming operations; fairly localized impacts
Mining	Low	Mainly legacy effects from past mining; possible source(s) of mercury input
Transportation	Low	Much of the river and tributaries are bordered by roads, leading to habitat simplification and increased sediment and/or pollutant input
Logging	Low	Legacy effects may still exist but logging in the Russian River basin is much reduced from the past
Fire	Low	Fire may increase sedimentation of river and reduce riparian vegetation
Estuary alteration	Low	Limited use of estuary by tule perch
Recreation	Low	Recreational use of the river is heavy; associated reduction of habitat complexity through removal of tree hazards, etc.
Harvest	n/a	
Hatcheries	n/a	
Alien species	Medium	Alien predators appear to have minimal impact at present; potentially a greater threat in future with changes in flows and water quality that may favor alien species

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

Effects of Climate Change: The unique life history, environmental tolerances, and population resilience of Russian River tule perch would appear to make them relatively resistant to the effects of climate change, which are predicted to increase flow variability and water temperatures. The most severe impacts would likely occur during extended drought periods, when there would be long periods of low river flows, coupled with impaired water quality and high water temperatures. Under these conditions, aquatic habitats in the Russian River drainage will become increasingly unsuitable for tule perch, especially if human water demand continues to increase and ground water storage capacity is reduced through landscape conversion to agriculture (i.e., viticulture) or development. Moyle et al. (2013) rated the Russian River tule perch as "highly vulnerable" to extinction as the result of climate change, as flows will likely decrease due to increased human water demand and water temperatures are predicted to increase.

Status Determination Score = 3.7 – Moderate Concern (see Methods section Table 2). Russian River tule perch do not face immediate threat of extinction (Table 2) but this subspecies is confined to a single, highly altered, watershed. The Russian River watershed is undergoing rapid change through development of vineyards and urban areas, while flows in the river are artificially controlled by water projects. Although tule perch are very resilient, they are also short-lived so extended periods of artificially enhanced drought could cause severe declines. The abundance and distribution of this subspecies is a good indicator of habitat and water quality in the mainstem Russian River and their populations should be closely monitored as a metric of overall watershed health.

Metric	Score	Justification
Area occupied	1	Limited to Russian River and major tributaries
Estimated adult abundance	5	Populations large
Intervention dependence	5	Little tule perch-specific management needed
Tolerance	4	Fairly tolerant of conditions in the Russian River although susceptible to warm temperatures or turbid conditions
Genetic risk	5	No genetic risks known
Climate change	2	Reduced stream flows may restrict available habitats; likely worsened by rapidly increasing water demand in region
Anthropogenic threats	4	See Table 1
Average	3.7	26/7
Certainty (1-4)	3	Good recent surveys

Table 2. Metrics for determining the status Russian River tule perch, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: The Russian River should be managed to maintain its assemblage of native fishes by maintaining high water quality, diverse habitats, and suitable flow releases from dams. A flow regime should be implemented that assures the river will not go dry or become intermittent in reaches important to tule perch and other native fishes. Because tule perch are a good indicator species for river health, a regular fisheries monitoring program of the Russian River, which includes monitoring of all native fishes, should be continued to determine their population status, distribution, and trends. The fish monitoring program of SCWA is a good model and should be continued.

Figure 4. Distribution of Russian River tule perch, *Hysterocarpus traskii pomo* (Hopkirk), in California.

CLEAR LAKE TULE PERCH
Hysterocarpus traskii lagunae (Hopkirk)

Status: High Concern. The Clear Lake tule perch is endemic to three highly altered lakes which have already lost the majority of their native fishes. Tule perch populations seem to have dropped to very low levels in Clear Lake; they are probably absent from Lower Blue Lake, but still common in Upper Blue Lake.

Description: Tule perch are small (up to 150 mm SL), deep-bodied fish, bluish to purple dorsally, and white to yellow ventrally. Three color variants have been described, based on their lateral barring patterns: wide-barred, narrow-barred, and bars absent but, in Clear Lake, the unbarred form is absent, and most (73%) are narrow-barred. Adults have a pronounced hump (nuchal concavity) immediately anterior to the dorsal fin, which is deeper on Clear Lake fish than other subspecies. The dorsal fin has 15-18 spines and 9-15 rays; the anal fin, 3 spines and 20-26 rays; the pectoral fins, 17-19 rays. There are 38-43 scales along the lateral line (Baltz and Moyle 1981). Clear Lake tule perch are deeper bodied than other subspecies (Hopkirk 1973, Baltz and Moyle1981).

Taxonomic Relationships: The tule perch is the only freshwater species in the marine family Embiotocidae. *Hysterocarpus traskii lagunae* was described by Hopkirk (1968) as one of three subspecies. Morphometric analyses by Baltz and Moyle (1981) confirmed that the Clear Lake tule perch is different from Russian River tule perch (*H. t. pomo*) and Sacramento tule perch (*H. t. traskii*). The three subspecies also show genetic divergence (Baltz and Loudenslager 1984), as well as striking differences in life-history patterns (Baltz and Moyle 1982).

Life History: Tule perch are deep-bodied livebearers. Females produce young that are large, considering the size of the mother. As a result, females have reduced swimming abilities while pregnant. These attributes drive the life history adaptations of this unusual fish. See the Russian River tule perch account in this report for an overview of tule perch life history in a riverine environment. Tule perch have inhabited Clear Lake, one of the oldest lakes in North America, for a long period of time. Their scales have been found in sediment cores from the lake bottom that cover 25,000 years, but they have presumably been in the lake much longer (Hopkirk 1988).

From an evolutionary perspective, Clear Lake represents a remarkably stable environment which is reflected in the life history strategy of Clear Lake tule perch, compared to those of the other two subspecies (Baltz and Moyle 1982). Clear Lake tule perch are relatively long-lived (6-7 years). Females delay reproduction until their second or third year at lengths of 110 to 120 mm; they give birth to 25-35 free swimming young of relatively large size, presumably because both adults and young have high survival rates at larger sizes. Curiously, the populations in Upper Blue Lake show signs of being stunted (all fish < 100 mm SL) but they maintain the same basic life history strategy, although brood sizes are small, with 10-12 young (Baltz and Moyle 1982).

Clear Lake tule perch are gregarious and are usually found in aggregations, especially during the day, that may include several hundred fish (Moyle unpublished observations). The Clear Lake tule perch, with its terminal mouth, protrusible jaw, and long gill rakers is adapted for selective feeding on larger zooplankton species. Cook (1964) found that tule perch fed mostly on zooplankton but switched to feeding on midge (Chironomidae) larvae when midges were abundant.

Habitat Requirements: The main population of Clear Lake tule perch spends it entire life cycle in Clear Lake, which is warm (summer temperatures 25-28°C) and shallow (average depth 6.5 m), with primarily sandy or soft bottom substrates. Clear Lake is eutrophic, alkaline (pH of ca. 8) and fairly turbid (Secchi depth, <2m) (Suchanek et al. 2008). Historically, smaller populations occured in Lower Blue Lake, which is similar to Clear Lake in its environmental attributes, and in Upper Blue Lake, which is clearer and cooler. Clear Lake and the Blue Lakes are quite different from habitats occupied by other tule perch subspecies which live in rivers, usually with clear, cool water, or in the turbid brackish water of the San Francisco Estuary. Their presence in this wide range of habitats suggests that tule perch are very tolerant of environmental variables. However, their absence from the San Joaquin Valley floor (Brown 2000) suggests that poor water quality limits their distribution in this part of their historic range (Moyle 2002). Laboratory studies indicate that Sacramento tule perch can withstand temperatures up to 30°C but they are rarely found in the wild at temperatures greater than 25-27°C (Cech et al. 1990). Clear Lake tule perch presumably have slightly higher temperature tolerances, although this has not been tested.

A key habitat requirement of Clear Lake tule perch is cover, especially for pregnant females and small juveniles. They are usually found in small shoals in deep (3+m) tule beds, among rocks (especially along steep rocky shores), or among the branches of fallen trees. Piers may also provide some cover but, in Clear Lake, such cover is usually occupied by alien sunfishes.

Distribution: This subspecies is confined to Clear Lake and to Upper and Lower Blue lakes, in Lake County (Hopkirk 1973, Moyle 2002). Presence in Lower Blue Lake has not been confirmed in recent years.

Trends in Abundance: Early accounts indicated that tule perch were one of the more common fishes in Clear Lake (e.g., Stone 1873). In sampling performed with three kinds of gear from 1961-1963, Cook et al. (1964) found tule perch to be the 5th most abundant fish in their catches. In his review of the status of native fishes in the lake, Cook (1966) found tule perch to be "reasonably abundant" throughout the lake. In July, 1977, Broadway and Moyle (1978) likewise found tule perch to be the fifth most abundant fish captured in 78 seine hauls. Abundance of tule perch in Clear Lake in recent years is not known but they were found to be uncommon or absent in more recent sampling. Tule perch favor habitat around heavy structure and vegetation, such as tule beds, and these habitat types are difficult to sample using conventional methods such as seine nets. The Clear Lake Vector Control Agency samples a number of areas by beach seine around the lake each year. In 2005, eight perch were caught, in 2007, seven perch, in 2010, six perch, and in 2012, seven perch (J. Scott, pers. comm. 2013). Because tule perch are not common in near-shore habitats along beaches, it is possible that they may have been underrepresented in seine catch reports (e.g., Broadway and Moyle 1978). However, boat electrofishing surveys should provide good indications of at least presence/absence.

Electrofishing surveys by CDFW collected 37 perch in 1999, 25 in 2000, four in 2001, three in 2002, one in 2008, six in 2010, and one in 2012 (J. Rowan, CDFW, pers. comm. 2013). In short, tule perch appear to have become very scarce in Clear Lake in the past 10-20 years. Both sampling methods used in recent years, however, have biases that select again capturing tule perch, although past sampling suggests they were common regardless of technique used.

Clear Lake has rarely been 'clear' in the past so visual surveys are not employed, although tule perch are readily visible in Upper Blue Lake, where they still appear to be common (J. Rowan, CDFW, pers. comm. 2011). Their status in Lower Blue Lake is not known but conditions there are similar to Clear Lake (shallow, turbid, dominated by non-native species).

Nature and Degree of Threats: The threats to this subspecies reflect large-scale anthropogenic changes to the Clear Lake basin. Osleger et al. (2008) examined sediment cores from the lake and found an abrupt change in sediment characteristics starting around 1927, when the Sulphur Bank Mercury Mine opened up on the edges of the lake. Core analyses revealed that the cultural eutrophication of Clear Lake "began with the advent of large-scale open-pit Hg mining in 1927 and subsequent human-induced landscape modification involving heavy earthmoving equipment. These activities resulted in increased erosion/sedimentation rates and associated nutrient input into the lake, culminating in algae blooms and reduced surface water quality through the rest of the 20th century (Osleger et al. 2008, p. A255)."

Major dams. Cache Creek Dam was built in 1914 to control lake outflows and levels to provide water for Yolo County agriculture. This causes lake levels to be higher, at times, than they naturally would be and fluctuate more than they did historically. The effects of lake drawdown on tule perch populations are not known but it is possible that young-of-year (YOY) perch could be forced from cover as water levels drop, making them more vulnerable to predation.

Agriculture. The Clear Lake basin supports widespread agriculture, especially orchards and vineyards, which sends effluent, including fertilizers, sediments and pesticides into the lake. These impacts were probably more severe historically than they are today. Agricultural effluents and other pollutants contributed to accelerated eutrophication in the 20th century that resulted in major blooms of bluegreen cyanobacteria (Osleger et al. 2008). Although tule perch persisted in spite of significant reductions in water quality, their abundance may have been greatly reduced by these conditions.

Grazing. Heavy grazing of Clear Lake watersheds has occurred since the 1870s and has likely contributed to sedimentation and nutrient loading of the lake (Suchanek et al. 2002). Impacts were greater historically but the legacy effects of erosion, soil compaction, stream degradation, and loss of meadow and wetland habitats may still be influencing water quality and habitat suitability in Clear Lake. Effects on tule perch are unknown.

	Rating	Explanation
Major dams	Low	The level of Clear Lake is partly regulated by a dam on its outlet, Cache Creek
Agriculture	Medium	Contributes to eutrophication, sedimentation and pollution of the lake
Grazing	Low	No current lakeshore grazing but heavy historic grazing may continue to contribute to loss of habitat quantity and quality
Rural residential	High	Clear Lake and the two Blue lakes are surrounded by housing which reduces shoreline habitat and contributes pollutants
Urbanization	Medium	Towns around the lake contribute to pollution and degraded aquatic habitats, especially nearshore and shoreline habitats
Instream mining	Low	Gravel mining simplifies habitats and increases turbidity
Mining	Medium	Contamination of foodwebs from mercury may especially affect tule perch because of live-bearing life history
Transportation	Medium	Roads contribute sediment and other pollutants, as well as modify habitats along lakeshore
Logging	Low	Legacy effects of sedimentation, etc.
Fire	Low	Fire may increase sedimentation rates; fire frequency and intensity predicted to increase
Estuary alteration	n/a	
Recreation	Low	Removal of tule beds, fallen trees, etc. to improve boat access or reduce hazards reduces habitat quantity and quality
Harvest	n/a	
Hatcheries	n/a	
Alien species	High	Competition and predation from alien species are substantial

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

Rural residential. As Clear Lake became popular as a resort area in the 19th century, the lakeshore became increasingly developed with vacation and permanent homes. This development removed wetlands, which trapped sediment and nutrients, added septic tank effluent to the lake, and led to large-scale application of pesticides to the lake to control pestiferous gnats. While tule perch persisted despite changes to the shoreline and lake habitats, it is likely they declined in abundance as cover, such as tule beds and dead trees, became less abundant. Such cover is especially important to pregnant females and to their small young, immediately after birth.

Urbanization. Many small towns around the lake also contribute to eutrophication through sewage spills, increase in sedimentation, and removal of tule beds and wetlands. Local residents were leading proponents of applying pesticides to the lake. In particular, dichloro diphenyl dichloroethane (DDD) was applied (1949, 1954, 1957) to control gnat populations. DDD accumulates in the fatty tissues of fish, perhaps affecting survival and reproduction (Hunt and Bischoff 1960).

Mining. The Sulphur Bank Mercury Mine dumped mining waste (~193,600 cubic yards) containing mercury directly into the Oaks Arm of the lake and shore from 1922-1947 and 1955-1957. These wastes contaminated the lake ecosystem with mercury and arsenic (summarized in Suchanek et al. 2002). Elevated levels of mercury have been found in fish and waterfowl in the Clear Lake basin. A current health advisory (first issued in 1986) recommends that not more than one fish from Clear Lake be consumed per week. The water column does not seem to contain high concentrations of methyl mercury, in contrast to some lake sediments. Indirect effects from mercury exposure include behavioral disruption (prey capture, inhibition of reproduction), reduced growth rate, and disruption of physiological functions (olfaction, thyroid function, blood chemistry; Suchanek et al. 2008), potentially making tule perch more vulnerable to predation. Female tule perch pass mercury and other contaminants on to their young, so this may affect survival of juveniles.

Transportation. Roads along the edge of the lake have reduced available cover (e.g. downed trees). Drainage from roads can also increase fine sediment delivery to the lake, adding to the lake's eutrophication problem, as well as various pollutants with unknown effects on tule perch.

Logging. Clearing of forest lands around Clear Lake began in the 1840s. By 1905, approximately 1.5×10^6 board feet of lumber were being processed locally (Suchanek et al. 2002). Erosion from timber harvest lands likely contributed to siltation of the lake and eutrophication and legacy effects may still be affecting aquatic habitats in the basin.

Recreation. Pollution from extensive use of gas-powered watercraft in Clear Lake may stress tule perch. Removal of tule beds, fallen trees and other obstacles to improve boat access or reduce boating hazards reduces habitat for perch, especially juveniles and pregnant females.

Fire. Natural and human-induced fires are common in the watersheds that drain into Clear Lake (Suchanek et al. 2002). Catastrophic fires can increase erosion rates and sediment delivery to the lake, contributing to eutrophication. Fire frequency and intensity are predicted to increase in the future under climate change models, potentially leading to further degradation of water quality and habitat suitability in Clear Lake.

Alien species. Historically, 10 native fish species were found in Clear Lake (Moyle 2002). Presently, only five (hitch, blackfish, tule perch, Sacramento sucker, prickly sculpin) of these species still exist in the lake, along with at least 16 alien fish species. Some native species extirpated from the lake maintain populations in tributaries streams (see the Clear Lake hitch account in this report). Until recently, tule perch seem to have persisted in at least small numbers despite the introduction of many competitors for zooplankton and benthic invertebrates. The high abundance of threadfin shad and Mississippi silverside in recent years may have seriously depleted both the zooplankton and benthic insects that tule perch depend on; when zooplankton is depleted, most planktivorous fish switch to feeding on benthic invertebrates (Eagles-Smith et al. 2008). Increased predation from alien species and decreased availability of forage base, in combination with decreased habitat quality (sedimentation, impaired water quality, removal of cover), may be working together to negatively affect tule perch populations. In recent years, largemouth bass, especially larger fish, have been abundant in Clear Lake, increasing the likelihood of predation impacts on tule perch, especially during years when alternative prey populations are low (Eagles-Smith et al. 2008). It is also possible that during periods of high threadfin shad and Mississippi silverside abundance in the lake (most years), zooplankton food resources are reduced and predator densities, especially fish-eating birds, may increase. Increased capture of tule perch as incidental prey by predators may also negatively affect their populations and pregnant females may be particularly susceptible due to their impaired swimming ability, as are newly-born young.

Effects of Climate Change: The life history, broad environmental tolerances and population resilience of Clear Lake tule perch should make them relatively resistant to the effects of climate change, especially in Upper Blue Lake. However, increasing water temperatures and more frequent lower lake levels could cause additional stress to tule perch and other fishes through decreased water quality, reduced cover availability, improved conditions for alien predators and other factors. Pregnant females and juveniles would be particularly vulnerable to these changes. Climate change predictions also state that the frequency and intensity of storm events will increase, potentially increasing sedimentation, nutrient loading and pollution (from mine wastes) into Clear Lake (Suchanek et al. 2002). In addition, indirect effects of climate change, such as increasing algal blooms or abundance of competing alien species also have potential to negatively affect tule perch. In an independent analysis using 10 metrics, Moyle et al. (2013) found that Clear Lake tule perch are “highly vulnerable” to extinction under predictions of standard climate change models.

Status Determination Score = 2.3 - High Concern (see Methods section Table 2). The Clear Lake tule perch is confined to Clear Lake and the two Blue Lakes but appears to be increasingly uncommon in Clear Lake and absent from Lower Blue Lake. However, few focused abundance or distribution data exist, so its actual status remains uncertain.

Metric	Score	Justification
Area occupied	1	Restricted to the Clear Lake basin; present in Upper Blue Lake (UBL)
Estimated adult abundance	3	Abundance not known but likely small in Clear Lake; the largest population may be in UBL
Intervention dependence	3	Population in Clear Lake may need intensive management or reintroduction
Tolerance	3	Tolerant of conditions in Clear Lake although susceptible to warm temperatures and pollutants
Genetic risk	2	Genetic risks unknown; could be severe if UBL contains the principal remaining population
Climate change	2	UBL is possible refuge
Anthropogenic threats	2	See Table 1
Average	2.3	16/7
Certainty (1-4)	1	Few studies or published reports exist

Table 2. Metrics for determining the status of Clear Lake tule perch, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Management Recommendations: Little is known about the habitat requirements, overall abundance, or population trends of Clear Lake tule perch. Even less is known about their interspecific dynamics with introduced fishes in Clear Lake and in Upper Blue Lake. These attributes need further study in order to develop appropriate management strategies and to bolster conservation efforts. In particular, it is important to understand the requirements of YOY perch and the effects of predation and competition from alien fishes on their survival. If tule perch populations are declining in Clear Lake and have been extirpated from Lower Blue Lake in recent years, as the few available surveys suggest, then Upper Blue Lake may, ultimately, be their only refuge.

Specific recommendations include:

1. Implement comprehensive fish surveys of Clear Lake, Upper Blue Lake and Lower Blue Lake to establish baseline status of the species. Surveys should be performed using a variety of sampling gear including electrofishers, gill nets, trawls and seines and repeated at intervals (3-5 years) to develop trend information. Visual surveys should be conducted in upper Blue Lake and, when possible, in Clear Lake.
2. Using information collected in comprehensive fish surveys, establish standardized monitoring programs for the native fishes of all three lakes.
3. Establish a conservation facility for Clear Lake fishes, including tule perch, that maintains captive populations of species in decline and works closely with the

monitoring program to determine conservation strategies and priorities.

4. Determine likely causes of decline in Clear Lake and what actions, if any, can be taken to restore populations. Develop genetic and physiological studies to determine if perch from Upper Blue Lake can be used as substitutes for fish from Clear Lake in conservation strategies (e.g. reintroduction).
5. Conduct a thorough investigation of the limnology, fishes, and other aspects of Upper Blue Lake to determine what factors, if any, might threaten its value as a refuge for Clear Lake tule perch.
6. Conduct physiological studies to establish the environmental tolerances of Clear Lake tule perch in order to determine likely impacts of climate change. Parameters studied should include: temperature, dissolved oxygen, as well as exposure to methyl mercury, pesticides, and other pollutants.
7. Investigate and implement ways to improve tule perch habitat, especially for pregnant females. Use existing laws and regulations to protect remaining shoreline habitats. Habitat improvements could include: increasing cover in areas along the lakeshore, including expanding tule beds, allowing fallen trees to stay in the water, and creating artificial cover patches (‘reefs’) in places.
8. Develop and implement a conservation strategy for Clear Lake and Upper and Lower Blue lakes to improve water and habitat quality to benefit all native fishes.

Figure 1. Distribution of Clear Lake tule perch, *Hysterocarpus traskii lagunae* (Hopkirk). Historically, they were found only in Clear Lake, Lower Blue Lake, and Upper Blue Lake.

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