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VIABILITY ASSESSMENT FOR PACIFIC SALMON AND STEELHEAD LISTED UNDER THE ENDANGERED SPECIES ACT: SOUTHWEST

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Viability assessment for Pacific salmon and steelhead listed under the Endangered Species Act: Southwest

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1 Introduction and Summary of Findings

In California, there are currently 10 distinct population segments (DPSs) or evolutionarily significant units (ESUs)[1] of Pacific salmon and steelhead listed as threatened or endangered under the Endangered Species Act (ESA) (Table 1.1). The ESA requires that the National Marine Fisheries Service (NMFS) review the status of listed species under its authority at least every five years and determine whether any species should be removed from the list or have its listing status changed. The most recent such review for ESA-listed salmonids along the West Coast occurred in 2016. NMFS is again conducting such a review (84 FR 53117).

The NMFS West Coast Region is responsible for the 5-year review process for Pacific salmon and steelhead and for decision-making regarding any proposed changes in listing status. This report by the Southwest Fisheries Science Center (SWFSC) was completed to provide the West Coast Region (Region) updated information and analyses on the biological viability of the listed ESUs or DPSs, focusing primarily on trends and status in abundance, productivity, spatial structure, and diversity to inform the 5-year reviews of 10 ESA-listed of salmon and steelhead ESUs or DPSs. Where possible, this assessment also summarizes current information with respect to recovery goals identified in recovery plans or Technical Recovery Team viability documents.

Previous viability reports that supported the current listings (Good et al. 2005; Williams et al. 2011; Spence and Williams 2011; Williams et al. 2016) may have categorized each ESU as either “in danger of extinction,” “likely to become endangered,” or “not likely to become endangered” based on the ESU’s abundance, productivity, spatial structure and diversity. In this report, we use exclusively a “high,” “moderate,” or “low” extinction risk category (viability). This change in terminology from previous assessments is to clarify that the Center’s viability assessments assess the biological extinction risk, the “status” of the listed ESUs is evualated and determined by the West Coast Region’s 5-year status review. In this assessment, for each listed ESU, we summarize whether there is new biological information to indicate that an ESU is likely to have moved from one of the three biological risk categories to another since the 2015 assessment. In addition, we also note whether each ESU appears to be stable, improving, or declining in extinction risk, whether or not such changes warrant a change in category (Table 1.1) The information in the report will be incorporated into the Region’s review, and the Region will make final determinations about whether changes in listing status are or are not warranted, taking into account not only biological information but also information on the five listing factors in section 4(a)(1) of the ESA.

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[1] For Pacific salmon, NMFS uses its 1991 ESU policy, that states that a population or group of populations will be considered a Distinct Population Segment if it is an Evolutionarily Significant Unit (ESU). The species *O. mykiss*, which has anadromous and non-anadromous forms, is under the jurisdiction of NMFS (anadromous) and the U.S. Fish and Wildlife Service (non-anadromous). NMFS uses the 1996 Joint FWS-NMFS DPS policy in making its listing determinations for anadromous *O. mykiss*. Throughout this document, ESU and DPS are used interchangeably.

Here we provide a general overview of the conclusions of the most recent viability assessments for ESA-listed Pacific salmonids in California, including any findings concerning changes to the delineations of ESA-listed ESUs/DPSs. The details of the assessment of each ESA-listed ESU/DPS are organized by Recovery Domain and we provide a summary table here of the previous assessment, the current assessment, trends in biological extinction risk (i.e., viability), and any changes in the biological extinction risk (viability) since the previous assessment (Table 1.1). Populations in many ESUs declined in abundance compared to the previous review five years ago. In most cases, these declines appear to be caused by variation in survival rates in the ocean environment. In the coming decades, climate change, including negative effects in the ocean, is expected to be a major factor impacting Pacific salmon (Crozier et al. 2019). In the near term, however, we generally viewed the recent declines as mostly being short-term and not necessarily indicative of a major underlying change in ESU/DPS status. Several populations within each ESU/DPS were evaluated to have a declining trend in overall viability (i.e., increased extinction risk) since the last review.

1.1 Overview of New Information for Consideration of ESU/DPS Delinations

As previously discussed, NMFS is required to review the status of ESA listed species every five years. As part of that process, it is necessary to evaluate the geographic or ecological delineations of listed ESUs and DPSs to determine if new information is available that suggests a change may be warranted.

Chinook Salmon

Chinook salmon are distributed in coastal basins north of the Golden Gate (entrance to San Francisco Bay) and in the Sacramento/San Joaquin River and associated Bay/Delta systems of California's Central Valley. In California, six ESUs have currently been identified. The Southern Oregon/Northern California Coastal (SONCC) ESU includes populations from Cape Blanco in the north to the lower Klamath River in the south. The California Coastal (CC) ESU includes populations from Redwood Creek in the north to the Russian River (inclusive) in the south. The Upper Klamath and Trinity Rivers ESU includes populations spawning upstream of the confluence of these two rivers. The Central Valley contains three ESUs, one of which, fall-run/late fall-run Chinook salmon, currently extends from Carquinez Strait into the Sacramento and San Joaquin rivers and their tributaries. The other two ESUs, Sacramento River winter-run and Central Valley spring-run Chinook salmon and extend into the Bay/Delta Region. The Coastal California and the Central Valley spring-run Chinook salmon ESUs are ESA listed as threatened, the Sacramento River winter-run Chinook salmon ESU is ESA listed as endangered, and the other ESUs are not listed.

The 2011 viability assessment (Williams et al. 2011) discussed the fact that populations that lie between the lower boundary delination of the Central Valley fall-run Chinook salmon ESU, which is not listed under the ESA (Carquinez Straits) and the southern boundary delination of California Coastal Chinook salmon ESU (Russian River) were not included in either ESU, despite the fact that Chinook salmon had been reported in several basins. Available genetic evidence indicated fish from the Guadalupe and Napa rivers in San Francisco and San Pablo bays had close affinity with the Central Valley fall-run Chinook salmon ESU (Garza et al., unpublished data; Garza and Pearse 2008), and it was recommended that fish from these two watersheds should be included in the Central Valley fall-run Chinook salmon ESU. At the time of 2011 assessment, evidence indicated fish in Lagunitas Creek was equivocal, with 17 samples assigned almost equally between California Coastal Chinook salmon and Central Valley fall-run Chinook salmon ESUs. Williams et al. (2011) tentatively concluded that Lagunitas Creek Chinook salmon should be considered part of the California Coastal ESU pending additional data. National Marine Fisheries Service subsequently indicated that a delineation change was under consideration (76 FR 50447); however, no action has been taken to date. There is no new genetic information to further inform whether a change is advisable (Carlos Garza, NMFS SWFSC, personal communication). We believe that the rationale for revising the placement of Lagunitas Creek Chinook salmon originally stated in Williams et al. (2011) is still accurate and appropropriate and does not warrant a revision at this time, though additional analysis may be warranted to verify the tentative findings of 2011.

At the time of listing, spring-run Chinook salmon were considered extirpated in the San Joaquin basin (64 FR 50394; 70 FR 52488). Information on the presence of fish exhibiting spring-run behavior in San Joaquin tributaries may represent passive re-establishment of Central Valley spring-run Chinook salmon into the San Joaquin River Basin. No new information suggests that the delineation of the CVSRC ESU should change at this time.

Coho Salmon

Coho salmon are distributed in coastal California basins from the Oregon border in the north to Monterey Bay in the south and historically were present in the San Francisco/San Pablo Bay system, where they are now extirpated. Populations spawning from Elk River (Oregon) in the north to Mattole River (California) in the south, inclusive, an area that extends from Cape Blanco, Oregon, to Punta Gorda, California are assigned to the SONCC coho salmon ESU, whereas populations to the south of Punta Gorda to Aptos Creek, California, are part of the Central California Coast (CCC) coho salmon ESU (77 FR 19552). The SONCC coho salmon ESU is ESA Threatened, whereas the CCC coho salmon ESU is ESA Endangered. Analysis of recent microsatellite data from 30 sites in 23 watersheds spanning the SONCC and CCC ESUs provided consistent and strong support for the current ESU delineation at Punta Gorda (Gilbert-Horvath et al. 2016). No changes to current ESU delineations are proposed in this assessment.

Steelhead

Steelhead, anadromous rainbow trout, are distributed throughout California, in coastal streams from the Oregon border in the north to the border with Mexico in the south, and throughout the Central Valley. There are a total of six steelhead DPSs in California, with one in the Central Valley and five on the coast.

The Klamath Mountains Province steelhead DPS begins at the Elk River in Oregon and extends to the Klamath/Trinity basin in California, inclusive. The Northern California steelhead DPS extends from Redwood Creek (Humboldt County) southward to but not including the Russian River. The Central California Coast steelhead DPS begins at the Russian River, contains populations in streams tributary to the San Francisco/San Pablo Bay system, and stretches south to Aptos Creek, inclusive. The South-Central California Coast steelhead DPS starts at the Pajaro River in the Monterey Bay Region and continues to Arroyo Grande in San Luis Obispo Bay. The Southern California steelhead DPS begins at the Santa Maria River, inclusive, and stretches to the border with Mexico. The California Central Valley steelhead DPS includes all populations in the Sacramento/San Joaquin River system and its delta. All of these DPSs include anadromous forms of *O. mykiss* downstream of definitive natural or manmade barriers to anadromy. The Klamath Mountains Province DPS is not ESA-listed, the Southern California DPS is ESA-listed as Endangered. All of the other California steelhead DPSs are ESA-listed as Threatened.

In the previous viability assessment (Williams et al. 2016) and 5-year review (NMFS 2016a), a change in the delineation of the California Central Valley steelhead DPS was recommended to include steelhead from Mokelumne River Hatchery (85 FR 81822). This DPS includes steelhead populations spawning in the Sacramento and San Joaquin rivers and their tributaries. Populations upstream of migration barriers are not included in this DPS. Hatchery stocks within the DPS include Coleman National Fish Hatchery (CNFH), Feather River Hatchery (FRH), and Mokelumne River Hatchery (MRH). The Nimbus Hatchery (NH) steelhead remain genetically divergent from the Central Valley DPS lineages, consistent with their founding from coastal steelhead stocks, and are not included in the DPS (Pearse and Garza 2015).

In the previous viability assessments (Williams et al. 2011; Williams et al. 2016), it was determined that new genetic population structure data not available at the time of the original ESU/DPS delineation suggest several potential changes may be warranted for coastal California DPSs. Based on these new data and information, it was recommended that a Biological Review Team (BRT) be convened to compile, review, and evaluate the best available scientific and commercial information on steelhead genetics, life history and biology, and the ecological/habitat requirements of steelhead that are relevant to evaluation current delineations and potential changes. The BRT review to determine if DPS delineations need to be modified has yet to be conducted, and therefore the existing delineations of coastal California steelhead DPSs were used in this report.

1.2 New and Developing Science

Recent Advances in Genetic Analyses and Life-history Diversity

Life-history diversity is a critical component to the resilience of salmon populations (Schindler et al. 2010). Many steelhead populations along the West Coast of the U.S. co-occur with sympatric non-anadromous *O. mykiss* (resident rainbow trout), and new research has improved our understanding of the genetic architecture of the populations exhibiting both non-anadromous (resident) and anadromous forms (Pearse et al. 2014, Pearse et al. 2019). We recognize that there may be situations where reproductive contributions from non-anadromous *O. mykiss* may mitigate short-term extinction risk for some steelhead DPSs (Good et al. 2005; 70 FR 67130).

In the Southern California Recovery Domain the viability criteria developed by the Technical Recovery Team recognized that the two listed DPSs were typically components of mixed populations of rainbow trout and steelhead, but the genetic, physiologic and ecological controls on the expression of these two life histories were poorly understood at the time. As a result of the new research discussed in Section 6 (Southern California Recovery Domain), we have improved our understanding of the genetic architecture of the populations exhibiting both non-anadromous and anadromous forms. The viability criterion for abundance for the Southern California Recovery Domain was augmented by an additional criterion for anadromous fraction, defined as the proportion of reproducing adults that exhibit the anadromous life history (Boughton 2022).

As in coastal watersheds, *O. mykiss* populations upstream of migration barriers remain excluded from the Central Valley steelhead DPS. Recent genetic information on Central Valley *O. mykiss* populations upstream and downstream of dams within the same tributaries showed that these populations were not each others' closest relative (Pearse and Garza 2015; Pearse and Cambell 2018). At this time, no changes in viability criteria have been proposed for Central Valley populations.

In addition to new genetic work exploring anadromy in *O. mykiss*, other genetic research such as genome-wide association studies have shown that run timing and associated traits are strongly associated with variation in a small portion of a single chromosome (Prince et al. 2017; Thompson et al. 2019; Thompson et al. 2020). Populations with early run timing that are otherwise clearly differentiated across the genome appear to be closely related in this small portion of the genome. It is unclear the extent to which this single loci controls other life-history traits or how this genetic information should be used to conserve specific life histories (Waples and Lindley 2018). These new genomic results warrant future consideration and may have conservation implications.

Thiamine Deficiency

Ocean conditions remain a critical component to survival and reproductive success of salmon who spend the majority of their lives in the ocean (see 2.0 Recent Trends in Marine and Terrestrial Environments and Their Likely Influence on Pacific Salmonids in California and Southern Oregon section). Thiamine deficiency can occur in adult Chinook salmon and influence reproductive success and health of their progeny (Harder et al. 2018). In fall and winter of 2019, Chinook salmon populations in the Central Valley of California (fall-, spring-, and late fall-run) were diagnosed with thiamine deficiency complex (TDC). This diagnosis was based on high rates of early life stage mortality observed in hatcheries and rapid recovery of juveniles exhibiting aberrant swimming behaviors following thiamine treatment by the USFWS California-Nevada Fish Health Center (Foott 2020). The primary hypothesis for TDC in Central Valley salmon is that a reorganization of food webs in the central California Current resulted in the dominance of northern anchovy in salmon diets. Northern anchovy possess thiaminase, an enzyme that breaks down vitamin B1, and diets high in northern anchovy can cause thiamine deficiency in their consumers, which can appear as high mortality or serious sublethal effects in subsequent progeny. It is unclear the extent to which female Sacramento River winter-run Chinook salmon have low concentrations of thiamine in their eggs that would result in acute mortality and/or latent effects in their progeny in the wild. Current research is underway to better understand this emerging stressor and potential treatment options to mitigate these nutritional deficiencies. Potential impacts to the BY2020 (Brood Year 2020) cohort due to TDC would be observed first in low juvenile passage numbers of juveniles at Red Bluff Diversion Dam in 2020, but not captured in viability criteria until adults return in 2023.

1.3 Summary of Findings

Climate plays an important role in salmon (*Oncorhynchus* spp.) habitat at every stage of their lifecycle. For instance, predictable seasonal climate variations interact with the physiography of salmon watersheds to provide predictable seasonally varying water temperature and streamflow regimes that create diverse life-history pathways for different salmon populations of the same and different species. Likewise, irregular climate and weather variations like persistent drought, episodic floods, or persistent marine heatwaves, can impact salmon populations by altering their aquatic habitats and food-webs, which in turn affect individual salmon growth and survival rates in ways that can impact salmon populations at local to regional scales. Climate variations impacting regions across 100s to 1000s of kilometers can thus impact ESU/DPS viability through impacts on abundance, productivity, spatial diversity, and distribution.

The period of 2013–2021 has been exceptional for its high frequency and magnitude of West Coast drought and terrestrial heat, widespread and severe wildfire, and record-setting marine heatwaves in the California Current Large Marine Ecosystem and broader northeast Pacific Ocean. Climate extremes from 2013–2021 have contributed to extreme bottlenecks in West Coast salmon survival rates for multiple West Coast salmon populations and subsequent declines in abundance for many DPSs and ESUs.

For nine of the ESUs/DPSs (Southern Oregon/Northern California Coast coho salmon, Central California Coast coho salmon, California Coastal Chinook salmon, Northern California steelhead, Central California Coast steelhead, South-central California steelhead, Southern California steelhead, Sacramento River winter-run Chinook salmon, and California Central Valley steelhead) the new information suggests that there has been no change in their biological extinction risk category since 2015 viability assessment (Table 1.1). For the Central Valley spring-run Chinook salmon ESU the new information suggests a change in the biological extinction risk category is warranted. The viability of Central Valley spring-run

Chinook salmon ESU appears to have declined since the 2015 assessment (Johnson et al. 2016); this ESU is a moderate to high risk of extinction.

While its extinction risk category has not increased to greater than moderate, it should be noted that the viability of the Southern Oregon/Northern California Coast has declined since the 2015 assessment (Williams 2016)**.** While its extinction risk category could not be increased to greater than "High", it should be noted that the viability of the Sacramento River winter-run Chinook salmon ESU has declined since the 2015 assessment (Johnson et al. 2016).

Table 1.1. Summary table of previous biological extinction risk, current ESA listing status, recent biological extinction risk (this assessment), and change in biological extinction risk category since 2015 assessment.

Species	ESU/DPS	2015 extinction risk category[a]	ESA-listing status	Recent extinction risk trend[b]	Change in extinction risk category[c]?
Chinook salmon					
	Central California Coast	Moderate	Threatened	Stable	No
	Sacramento River winter-run	High	Endangered	Declining	No
	Central Valley spring-run	Moderate	Threatened	Declining	Yes[d]
Coho salmon					
	Southern Oregon / Northern California Coast	Moderate	Threatened	Declining	No
	Central California Coast	High	Endangered	Stable	No
Steelhead					
	Northern California Coasts	Moderate	Threatened	Stable	No
	California Central Coast	Moderate	Threatened	Stable	
	South-central California Coast	Moderate	Threatened	Stable	No
	Southern California Coast	High	Endangered	Stable	No
	Central Valley	Moderate	Threatened	Stable	No

a – Risk category reflects the assessment of ESU/DPS viability summarized in the prior viability assessment (Williams et al. 2016).

b – Recent risk trend summarizes the overall trends in risk for each ESU/DPS since the prior status review through data from 2019, in the judgement of the chapter author(s) considering all four VSP criteria (abundance, productivity, spatial structure, and diversity). Declining = increased risk of extinction, Stable = no change, and Improving = decreased risk of extinction.

c – Extinction risk category for the assessment includes data through 2019. These risk categoires do not represent a conclusion regarding ESA-listing status.

d – FEMAT scoring captured the uncertainty of the authors suggests that the CVSRC salmon ESU is at moderate to high risk of extinction.

2 Recent Trends in Marine and Terrestrial Environments and Their Likely Influence On Pacific Salmonids in California and Southern Oregon

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2.1 Introduction

Climate plays an important role in salmon (*Oncorhynchus* spp.) habitat at every stage of their lifecycle. For instance, predictable seasonal climate variations interact with the physiography of salmon watersheds to provide predictable seasonally varying water temperature and streamflow for supporting diverse life-history pathways for different salmon populations of the same and different species. Likewise, irregular climate and weather variations like persistent drought, episodic floods, or persistent marine heatwaves, can impact salmon populations by altering their aquatic habitats and food webs, which in turn impact individual salmon growth and survival rates in ways that can impact salmon populations at local to regional scales. Climate variations impacting regions across 100s to 1000s of kilometers can thus impact ESU/DPS viability through impacts on abundance, productivity, spatial diversity, and distribution.

From 1999–2012, relatively favorable regional climate conditions supported relatively high freshwater and marine survival rates and subsequent high adult returns for many salmon populations throughout the Pacific Northwest at various times. In contrast, 2013–2021 has been exceptional for its high frequency and magnitude of West Coast drought and terrestrial heat, widespread and severe wildfire, and record-setting marine heatwaves in the California Current Large Marine Ecosystem and broader northeast Pacific Ocean. Climate extremes from 2013–2021 have contributed to extreme bottlenecks in West Coast salmon survival rates for multiple West Coast salmon populations and subsequent declines in abundance for many DPSs and ESUs.

This chapter summarizes what is known about marine and terrestrial conditions to provide environmental context when examining the viability assessments included in this report. Of primary interest are the climatic conditions that existed over the past 15–20 years, three to six generations of the Pacific salmonids that are being considered in these assessments.

2.2 Observed Environmental Conditons

Precipitation and Surface Air Temperature

A strong and persistent warming trend and large year-to-year variations in precipitation are among the most notable features of western US climate in recent decades (Figure 2.1). For both the Pacific Northwest and California, water year 2015 stands out as the warmest year on record, while water year 2018 is the second warmest year on record for California. With the exception of 2019, surface air temperatures in water years 2014–2020 in California were all much warmer than the 1981–2010 average, and all but five of California's water years between 2007 and 2020 have had below average precipitation. The combination of high temperature and low precipitation has come with a preponderance of widespread drought conditions, low snowpack and low streamflow years for California's salmon and steelhead watersheds.

Streamflow

A broad-brush overview of water year streamflow variations in northern California is provided in Figure 2.2, where stream gage data indicate substantially more low-flow than high-flow years from 2000–2019. Both the Klamath and Sacramento rivers had above average water years in 2006, 2011, and 2017, and the Sacramento River had an above average water year in 2019. Both watersheds had below average water years from 2001–2002, 2007–2010, 2012–2015 and 2018. The Klamath River also had below average water years in 2004–2005 and 2019. In 2016 water year streamflow was a bit above average in the Klamath River, but a bit below average in the Sacramento River.

California's multiyear severe drought of 2012–2016 was especially notable for the persistence and magnitude of above average surface temperatures, below average precipitation, below average snow pack, and below average streamflow throughout the state.

2015–2019 – Annual Anomalies from Recent Past

Over the past century, temperatures rose steadily, while precipitation was highly variable between years and decades. Warmer temperatures intensify the hydrological cycle within the atmosphere, causing more intense storm events and droughts (Warner et al. 2015). Within snow-dominated watersheds, warmer winters and springs reduce snow accumulation and hasten snowmelt. Reduced snowpack causes an earlier and smaller freshet in spring. Reduced snowpack also can lead to lower minimum flows and higher stream temperatures in summer (U.S. Global Change Research Program 2018). Projections of climate change in the western U.S. (U.S. Global Change Research Program 2018) indicate that both of these trends are likely to continue. Summer precipitation is projected to decline, exacerbating summertime low flows and high stream temperatures in the western U.S.

Winter conditions affect most salmon (i.e., all populations other than winter-run Chinook salmon, *O. tshawytscha*) during the egg and early rearing stages, which may be disturbed and relocated during flood events. Migrating smolts typically benefit from higher flows (Faulkner et al. 2018; Notch et al. 2020), although the impacts on migrating adults varies across populations. Summer conditions affect juveniles rearing in streams (especially coho salmon

[*O. kisutch*], steelhead [*O. mykiss*], and yearling Chinook salmon), winter-run Chinook salmon spawning and incubation, and adults migrating, holding or spawning over the summer (many Chinook salmon populations, Columbia River and Snake River sockeye salmon [O. *nerka*] and summer-run steelhead).

A recent assessment of exposure to climate change across the west coast region (Crozier et al. 2019) found that by the 2040s, average stream temperatures are likely to increase by over 2 standard deviations across most of the region, and either flooding (southern domains) or loss of snowmelt (northern domains) was also very likely to change dramatically in most ESUs and DPSs. Here we put these projected changes within the context of recent conditions (2015–2019) by expressing four metrics (summer stream temperature, low flow, high flow, and snowpack) in terms of standard deviations from the recent historical mean (1998–2014). Although they are currently anomalous years, they are likely to represent average conditions in the near future.

To facilitate interpretation of salmon dynamics within individual ESUs and DPSs, Harvey et al. (2018) averaged environmental conditions across many measurement stations within each of six ecoregions in the west coast domain from the interior Columbia River Basin to the Washington coast to southern California (Figure 2.3). We have re-analyzed these results to consider the last five years (2015–2019) specifically in relation to the mean and standard deviation of the previous 15 years (1998–2014). Deviations for each year (Y_t) were calculated from the raw value (X_t) as $Y_t=(X_t-X_{mu})/X_{sd}$ for each region, where X_{mu} and X_{sd} were the mean and standard deviation, respectively, over the 1998–2014 period.

In 2015, the combination of below-average precipitation and record-high surface air temperature brought record-low springtime snowpack to much of the west, leading to what has been called “the western snow drought.” The diminished snow pack and high surface temperatures combined with low springtime precipitation yielded especially low runoff to western watersheds in spring and early summer 2015. Temperatures returned to near normal in much of Washington and Idaho in August (which is the month shown in Figure 2.4), but then spiked again in the fall of 2015. Unusually low flows and warm stream temperatures in spring/summer 2015 caused widespread problems for salmon throughout the western United States.

In 2016, minimum flows continued to show long-term drought effects, especially in California and the unglaciated portion of the Columbia, but other indices were transitioning to more favorable high flows of 2017 in most regions.

Two ecoregions stood out in showing strongly anomalous conditions in all five years: summer temperatures were above average (>1SD) in the Salish Sea and Washington Coast region, and minimum flows were below average (>1SD) in southern California throughout the period of this viability assessment.

Particularly notable climate impacts on salmon occurred throughout the 2012–2016 drought in California. Effects of the drought on stream networks accumulated each year rather than reflecting precipitation directly. In critical coho salmon streams in the Russian River system, for example, the dewatered portion of the river network over summer increased from 28% in the first year of the drought to 58% in the third and fourth years (Deitch et al. 2018). Thus prolonged periods of dewatering occurred for two years. The previous drought from 1987–1992 likely had similar but even more prolonged effects. Using life-cycle models of coho salmon in coastal streams in Washington, Ohlberger et al. (2018) found that juvenile production has been limited historically by low-flow periods. In their projections of coho salmon production under future flow scenarios, negative population impacts followed reductions in the mean and increasing variability in annual summer low flows. Other studies (Larsen and Woelfle-Erskine 2018) found that juvenile coho salmon preferentially select pools with more groundwater intrusion, which stabilizes streams during low-flow periods. Thus, drawdown of coastal aquifers would directly affect potential habitat for these endangered salmon. During the drought, rearing juveniles adjusted their habitat use. Both coho salmon and steelhead juveniles moved between coastal lagoons and mainstem Scott Creek to regulate key physiological processes under the extreme duration (seven months longer than average) of seasonal sandbar closure (Osterback et al. 2018).

For winter-run Chinook salmon catastrophically low egg to fry survival rates (less than 5%) were observed in 2014 and 2015. Observations were based on screw trap collections of outmigrating fry (Voss and Poytress, 2017) and the corresponding spawning run size estimate, and were closely aligned with predictions from a model relating embryo survival to thermal and oxygen stress during incubation (Martin et al. 2017, Martin et al. 2020).

2.3 Ocean Conditions

Surface temperatures in the northeast Pacific Ocean were notably cooler than average from 1999–2002 and again from 2006–summer 2013, warmer than normal from 2003–2005, and at exceptionally high levels for much of the period from fall 2013–2020 (Figure 2.4). The period of exceptionally warm ocean temperatures was notable for having widespread impacts on marine life, including the food webs that West Coast salmon depend on. The 2013–2020 era of extremely high ocean temperatures was characterized by widespread declines and low abundances for many US West coast salmon and steelhead populations, and was likely a period of unusually low ocean survival rates for many of those populations.

For the California Current region, surface temperatures reached record high levels from 2014–2016, with 2015 being the single warmest year in the historical record (Jacox et al. 2018). The extreme ocean temperatures for the northeast Pacific and California Current were associated with a small number of persistent wind and weather patterns, some of which have been related to climate conditions in the tropical Pacific (Di Lorenzo and Mantua 2016; Jacox et al. 2018).

Biological Response to Marine Conditions Since 2014

A number of reports provide overviews of recent physical and biological conditions in regions of the NE Pacific Ocean that West Coast Pacific salmon may occupy during their marine residence period:

- California Cooperative Oceanic Fisheries Investigations (CalCOFI) State of the California Current (Thompson 2019),

- The Integrated Ecosystem Assessment's California Current Ecosystem Status Report (Harvey et al. 2020),

- Canadian Department of Fish and Ocean's State of the Physical, Biological and Selected Fishery Resources of Pacific Canadian Marine Ecosystems (Boldt et al. 2019), and

- Alaska Fisheries Science Center's Ecosystem Status Reports for the Gulf of Alaska (Zador et al. 2019), Eastern Bering Sea (Siddon and Zador 2019), and Aleutian Islands (Zador and Ortiz 2018).

- Southwest Fisheries Science Center Coastal Pelagic Survey reports (Stierhoff et al. 2020).

In all cases, the reports show a dramatic biological response at all trophic levels—from primary producers to marine mammals and sea birds—to the marine heat waves that have spread across the Northeast Pacific Ocean since 2013 and continued into 2020. These ecosystem changes have had large effects (both positive and negative) on Pacific salmon returns around the Pacific Rim, not just ESA-listed species on the West Coast. Here, we provide brief summaries of the biological trends described by these reports and a few other sources, with an emphasis on findings that are pertinent to salmon survival. Unless noted, the information comes from the above report series.

Overall, the NE Pacific marine heat wave in 2014–2016 had the most drastic impact on marine ecosystems in 2015, with lingering effects into 2016 and 2017. Conditions had somewhat returned to "normal" in 2018, but another marine heat wave in 2019 again set off a series of marine ecosystem changes across the North Pacific. One reason for lingering effects of ecosystem response is due to biological lags. These lags result from species impacts at larval or juvenile stages, which are typically most sensitive to extreme temperatures or changes in food supply. It is only once these species grow to adult size or recruit into fisheries that the impact of the heat wave is apparent. For example, most marine mortality for juvenile salmon and steelhead is thought to occur in the first weeks or months of ocean residence. However, whether marine survival was exceptionally high or low is not known until salmon return as adults, one to five years after ocean entry.

Primary Production

Perhaps the most dramatic change to primary producers was the largest bloom of the diatom *Pseudo-nitzschia* ever recorded in 2015 (McCabe et al. 2016; Bates et al. 2018). It stretched from southern California to the Aleutian Islands in Alaska, had some of the highest concentrations of cells ever recorded, and was particularly long lasting. *Pseudo-nitzschia* can produce domoic acid, a neurotoxin that causes amnesic shellfish poisoning, which is potentially fatal in mammals (including humans) and seabirds. In marine food webs, filter feeding molluscs (primarily bivalves) and planktivorous fishes such as Pacific sardine and Northern anchovy, consume *Pseudo-nitzschia*, and species that consume contaminated shellfish and fish become sick or die (McCabe et al. 2016; Bates et al. 2018).

The 2015 bloom caused high domoic acid levels in many commercially and recreationally important species, including razor clams, mussels and other bivalves, anchovy and sardines, and benthic scavengers Dungeness and red rock crab. Trophic transfer of domoic acid to higher trophic levels caused the stranding or death of hundreds of seabirds and marine mammals in 2015 and early 2016, and likely contributed to the large whale unusual mortality event in the Gulf of Alaska in 2015 (Bates et al. 2018).

While subsequent *Pseudo-nitzschia* blooms have not been as extensive as the 2015 bloom, they have continued to cause delays, closures, and restrictions for both razor clam and Dungeness crab fisheries in California, Oregon and Washington. Southern Oregon/northern California is particularly prone to elevated domoic acid levels in clams and crabs that exceed permissible levels for human health (20 mg/kg tissue).

Other notable primary-producer related events include harmful algal bloom of *Noctiluca* and *Heterosigma* in the Salish Sea (the Strait of Georgia and Puget Sound) in 2018 after a three-year absence. There were also more harmful algal blooms in 2018 than the previous three years in the Strait of Georgia. In the Gulf of Alaska, phytoplankton blooms were earlier and in higher concentrations in 2017–2018 relative to warm years of 2014–2016. Surface nutrient concentrations were some of the lowest on record in 2019 across the Gulf of Alaska, which, paired with elevated water temperatures, affected the offshore phytoplankton community, oceanic food webs, as well as oxygen levels and biogeochemistry.

Lower Trophic Levels: Copepods, Krill, Jellyfish, and Pyrosomes

Throughout most of the NE Pacific Ocean, the marine heat wave had profound effects on the animals at the base of the food web. Summer copepod communities are normally dominated by cold water (i.e., lipid rich) species, but during the heat wave northern species were largely or completely absent and warm water (i.e., lipid poor) species dominated. Not only were southern species abundant, but novel communities were observed in many areas. On the Newport Hydrographic Line (44.6° N), for example, 14 species of copepods that had never been observed were documented, which originated both offshore and from southern waters (Peterson et al. 2017). Other changes on the Newport line during the initial heat wave included reduced biomass of copepods and krill, and high abundances of gelatinous organisms such as larvaceans and doliolids (both types of pelagic tunicates). Similar abrupt changes in copepods, krill and gelatinous organisms were observed from southern California to the Gulf of Alaska.

To characterize this shift in biomass between major functional groups, Boldt et al. (2019) developed a “crunchy” versus “squishy” index. The index is the ratio of crunchy (zooplankton with hard chitinous exoskeleton with high protein and lipid) to squishy (zooplankton with hydrostatic skeleton, mainly gelatinous animals with high water content and low nutritional value) biomass. They show a very high squishy biomass in most areas of British Columbia in 2014–2019, which peaked in most areas in 2015. Furthermore, Boldt et al. (2020) expect that years with high squishy index equates to poor survival for juvenile fish and seabirds, which have higher survival when prey quality is high (i.e., crunchier).

The marine heat wave also negatively affected krill growth rates, abundance, and species composition from California to central Alaska. For example, krill were absent from the Seward (Alaska) line during 2014–2017 but high in early fall of 2018. Krill length, used to indicate growth, was poor in 2014–2016 but increased in 2018 on the Trinidad Head line (41.1° N) in northern California. Morgan et al. (2019) cautioned that the perceived absence of krill in some areas (Brodeur et al. 2019) was due to changes in depth distribution, rather than absence, because early larval stages were present. In general, most copepod and krill communities had returned to more "normal" conditions in 2018.

Jellyfish communities also exhibited dramatic changes from California to Alaska. In the California Current, Pacific sea nettle (*Chrysaora fuscescens*) is typically the dominant species near shore. However, starting in 2015, there was a dramatic drop in the abundance of sea nettles and concurrent increase in water jellyfish (*Aequorea* spp.) and egg-yolk jellyfish (*Phacellophora camtschatica*) (Morgan et al. 2019). These changes to the jellyfish community continued until 2017. In 2019 in the Gulf of Alaska, Zador et al. (2019) reported the highest-ever catches of Northern sea nettle (*Chrysaora melanaster*) in bottom trawls. This species was also extremely abundant in surface trawls in winter 2019 as far south as 52° N, 100s of km from shore (Pakhomov et al. 2019).

Finally, 2017 should be considered the Year of the Pyrosome in the NE Pacific, because of the enormous biomass of the pelagic colonial tunicate, *Pyrosoma atlanticum*, present throughout the region (Brodeur et al. 2018). Pyrosomes are common in warm open ocean waters throughout the tropics, but are rare north of southern California. Starting in 2014 and 2015, their abundance greatly increased in California waters, and in 2015 they were observed in offshore waters in southern Oregon. In winter 2016, their population exploded and they were everywhere including close to shore in truly staggering quantities: from southern California to northern Gulf of Alaska at densities of up to 200,000 kg/km^3.

The ecosystem effects of the pyrosome explosion are unknown but are expected to be large due to their staggering biomass and widespread distribution. Pyrosomes have low nutrient content, making them a low quality, high fiber prey. Despite this, they were observed in the diets of dozens of species from sea urchins and other demersal invertebrates to rockfishes and other commercial fishes, juvenile and adult Pacific salmon to fin whales (Brodeur et al. 2018). In spring of 2018 they were still present in large quantities off the Oregon coast but effectively absent by fall 2018, but still present off California in 2019.

Forage Fish and Squid

Like lower trophic levels, the abundance and species composition of forage fish and squid have were highly variable from 2014–2019. One species that expanded its range and abundance is the California market squid, *Doryteuthis opalescens*. Throughout the California Current, squid increased in abundance to the point that substantial commercial fisheries for California market squid have been occurring in Washington and Oregon waters since 2016, reaching the highest commercial catches ever recorded in Oregon in 2020 (>7 million tons). Squid catches have also steadily increased during juvenile salmon surveys off the Washington/Oregon coast (Morgan et al. 2019).

Other species that have increased in recent years in the California Current include Pacific pompano (*Peprilus simillimus*), juvenile rockfish (*Sebastes* spp.), adult sardine (*Sardinops sagax*) and anchovy (*Engraulis mordax*), some species of lanternfishes (Myctophidae), and both jack mackerel (*Trachurus symmetricus*) and Pacific mackerel (*Scomber japonicus*). Species with marked declines include hake (*Merluccius productus*), juvenile sardine and anchovy, Pacific herring (*Clupea pallasii*), lampfish (*Stenobrachius leucopsarus*), and juvenile salmon (especially in 2017 in the Northern California Current). Juvenile rockfish were abundant in the Gulf of Alaska in 2015 (Zador et al 2019), in northern California Current in 2016 (Morgan et al. 2019), and off the west coast of Vancouver Island in 2016–2018 (Chandler et al. 2017; Boldt et al. 2019)

The increase in Northern anchovy was particularly strong in central and southern California, where it serves as high quality prey for many species. Adult anchovy were high in 2018 and the highest ever in 2019 in central California and larval anchovies were also the highest in the CalCOFI time series in 2019. While breeding murres and Brandt's cormorants were apparently unable to take advantage of plentiful anchovy, California sea lions on the Channel Islands did, resulting in very high counts, weights and growth rates of California sea lion pups in 2018. Humpback whales were also observed congregating near shore along central California in 2013–2019 while feeding on anchovy schools.

One of the more impressive increases in abundance has been anadromous American shad (*Alosa sapidissima*), an exotic species that was introduced to the West Coast in the 1800s. Counts of shad over Bonneville Dam, the lowest mainstem dam on the Columbia River, reached 6.0 million fish in 2018, the highest ever, but were even higher in 2019 (7.4 million fish). Shad counts in 2020 at Bonneville declined slightly to 6.2 million fish.

Farther north, the biomass of Pacific herring increased in the Strait of Georgia between 2010 and 2019; in those years herring were stable off the west coast of Vancouver Island, and decreased in northern British Columbia. Northern anchovy were abundant in the Salish Sea (collectively the Strait of Georgia and Puget Sound) between 2016 and 2019, consistent with increased abundances in years following elevated coastal temperatures (Duguid et al. 2019). Eulachon (*Thaleichthys pacificus*), which have been declining throughout their range, were abundant in the Strait of Georgia in both 2015 and 2018, and in the Columbia River in 2014. Juvenile salmon of all species except chum salmon (*O. keta*) were below average off the west coast of Vancouver Island, while chum salmon were abundant. The catch of juvenile salmon in 2017 in two widely separated surveys targeting juvenile salmon were the lowest in their respective time series. Catches in Icy Strait (Alaska), which normally consists of juvenile pink salmon (*O. gorbuscha*), chum salmon, and sockeye salmon, and off the Washington/Oregon (spring-run Chinook salmon and coho salmon) were both extremely low. These surveys are used to forecast adult returns and predicted poor returns in future years, some of which have transpired (e.g., the extremely low Columbia River spring-run Chinook salmon return in 2019).

In Alaskan waters, capelin (*Mallotus villosus*) and sand lance (*Ammodytes personatus*) appear to have declined because they were low or absent in sea bird diets that normally contain them since 2014 and 2015, respectively. By contrast, Pacific herring in the eastern Gulf of Alaska and eastern Bering Sea were above long term means in recent years.

Salmon Survival/Returns

Pacific salmon populations from California to Alaska, like other guilds or trophic levels described in this section, have shown dramatic changes in abundance since 2015. While some populations (especially in northern areas) have returned at record high abundances, others have dropped to new lows. The following summary of recent North American Pacific salmon returns provides context for listed salmon populations reviewed in the previous chapters. Specifically, it demonstrates that unusually high or low returns are not restricted to any one region, species, or production type (hatchery or wild), but were continent wide. For example, recent low steelhead returns to the Columbia River basin parallel extremely low steelhead returns to the Fraser River basin. In many cases trends of listed species mirror those of hatchery- or mixed-origin (hatchery- and natural-origin) populations, indicating the critical role that recent unusual environmental conditions have had on North American Pacific salmon. Unless noted, these abundances come from the 2020 report of the Pacific Fishery Management Council (PFMC.org), the Pacific Salmon Commission website (PSC.org), Columbia River Data Access in Realtime website (cbr.washington.edu), and the Alaska Department of Fish and Game website (www.adfg.alaska.gov).

The abundances of southern Chinook salmon populations (Sacramento, Klamath, and Rogue rivers) were all at very low levels in at least some years between 2014 and 2019 to the point several stocks were declared “overfished” under management regulations. Sacramento River fall-run Chinook salmon have seen large swings in escapement (hatchery + natural), from a high of 406,000 in 2013, steadily declining to 90,000 in 2016 and reaching a low of 43,000 in 2017 (Figure 2.6). Escapement in 2018 increased to 102,000 and increased again with the 2019 return to 162,000 fish. Total run size of Klamath River Chinook salmon shows a slightly different pattern, with high in-river run size in 2012 (295,000), which declined to 24,000 in 2016, rebounded to 91,000 in 2018, but declined again to 37,000 in 2019. Indices for south migrating Oregon coast Chinook salmon (Gold Ray Dam, Rogue River and Winchester Dam, Umpqua River counts) also show a steady decline from 2015 to 2019. Peak spawner indices for north migrating Oregon coast Chinook salmon were highest in 2015 (247 adults/mile) and steadily declined to 2019 (64 adults/mile).

Chinook salmon in the Columbia River generally declined from 2015 to 2019, with details dependent on the year and run (Figure 2.5). For example, the minimum return of spring Chinook salmon to the Columbia Basin steadily declined from 2015 (420,000) to 2019 (110,000), one of the lowest levels since the 1990s. Run size for Columbia River summer

Chinook salmon also saw a steady decline from a minimum run size of 127,000 fish in 2015 to 35,000 in 2019. The minimum run size for Columbia fall Chinook salmon exceeded 1 million fish during 2013–2015 and dropped to 275,000 in 2018 (the lowest since 2007) and to 356,00 in 2019.

In Oregon, Washington, and southern British Columbia, several species show consistent patterns, suggesting a common marine cause. For example, coho salmon returns were extremely low in 2015 from the Oregon coast to the Salish Sea, with some of the lowest levels on record. The small body size of many of these adults suggested poor feeding conditions during the last summer in marine waters. Steelhead returns were extremely low in 2017 and 2018 in the same areas, to the point that Thompson and Chilcotin river (Fraser River, British Columbia) steelhead were petitioned for emergency protection under the Canadian Species At Risk Act (Neilson and Taylor 2018). One species in the region that increased in abundances were chum salmon starting in 2016, perhaps in part due to their reliance on gelatinous prey that were abundant.

Seabird Productivity

Seabirds consume forage fish that are present at predictable locations and times. Their ability to successfully feed and fledge their chicks (or themselves) is therefore a valuable indicator of the abundance and diversity of forage fish. Measures of chick success have varied widely over the last five years, and depend on the bird's mode of foraging. For example, at Semedi Island (west of Kodiak), surface-feeding black legged kittiwakes (*Rissa tridactyla*) had chick failure in 2019, while diving seabirds on the island had good success in the same season. Across reported species and locations, in general chick success was low in 2015 and 2016, rebounded in 2017 and 2018 and declined again in 2019.

There have also been several massive seabird die offs in response to the 2014–2016 northeast Pacific marine heat wave. In winter 2014/2015 there was a massive die off of Cassin's auklet (*Ptychoramphus aleuticus*) from northern California to northern Washington. It is estimated that 50,000–100,000 birds died (Coastal Observation and seabird survey team, coasst.org). These birds largely consume krill, and the late Bill Peterson (NWFSC) speculated that the warm water prevented the krill from reaching surface waters where the auklets could feed on them (B. Peterson, personal communication). A rigorous analysis suggests that reduced energy content of zooplankton paired with congregations of birds in a narrow coldwater band along the coast were to blame for the die off (Jones et al. 2018).

Another species to suffer a massive die offs were common murres (*Uria aalge*). An estimated 1 million common murres died between summer 2015 and spring 2016. The mortality event affected birds from California to Alaska. Most birds were severely emaciated and, so far, no evidence for anything other than starvation was found to explain this mass mortality (Piatt et al. 2020). Many colonies also suffered reproductive failure in 2016–2017, and another large common murre mortality event occurred along the Washington/Oregon coasts in fall 2019.

Marine Mammals

In the California Current, the most obvious impact to marine mammals was the widespread starvation of California sea lion (*Zalophus californianus*) pups in early 2015, resulting in nearly 1,500 malnourished and sick sea lion pups found along California beaches. Strandings in 2015 were the most extreme year in the 2013–2016 California sea lion unusual mortality event.

Poor feeding conditions in the California Current region in 2015 also led to a dramatic increase in the number of California sea lions farther north that summer, especially in the Columbia River, where they fed on returning adult salmon. While the number of California and Stellar (*Eumetopias jubatus*) sea lions in the Columbia River (at Bonneville Dam) in the spring has declined since the peak in 2015, the number of Stellar sea lions observed at Bonneville Dam and Willamette Falls has been increasing in the fall (Wright et al. 2014; Tidwell et al. 2019). A new Stellar sea lion rookery has been established on the north Washington Coast (Carroll Island/Sea Lion Rock complex) with over 100 pups born there in 2015 (Muto et al. 2020), which, along with a rookery off the north Oregon Coast, are likely sources of increased Stellar sea lions in the Columbia River.

Since 2015, there have been two large whale Unusual Mortality Events. The first Event occurred in the western Gulf of Alaska and British Columbia in 2015–16 (Savage 2017). A total of 52 whales were reported dead, consisting of 17 fin whales (*Balaenoptera physalus*) and 34 humpback (*Megaptera novaeangliae*) whales. A definitive cause of death could not be determined, but was generally blamed on anomalous physical and biological shifts in the marine environment. The second Event was declared for gray whales (*Eschrichtius robustus*) 2019. This event only affected a single species but over 250 whales were reported stranded from Mexico to Alaska.

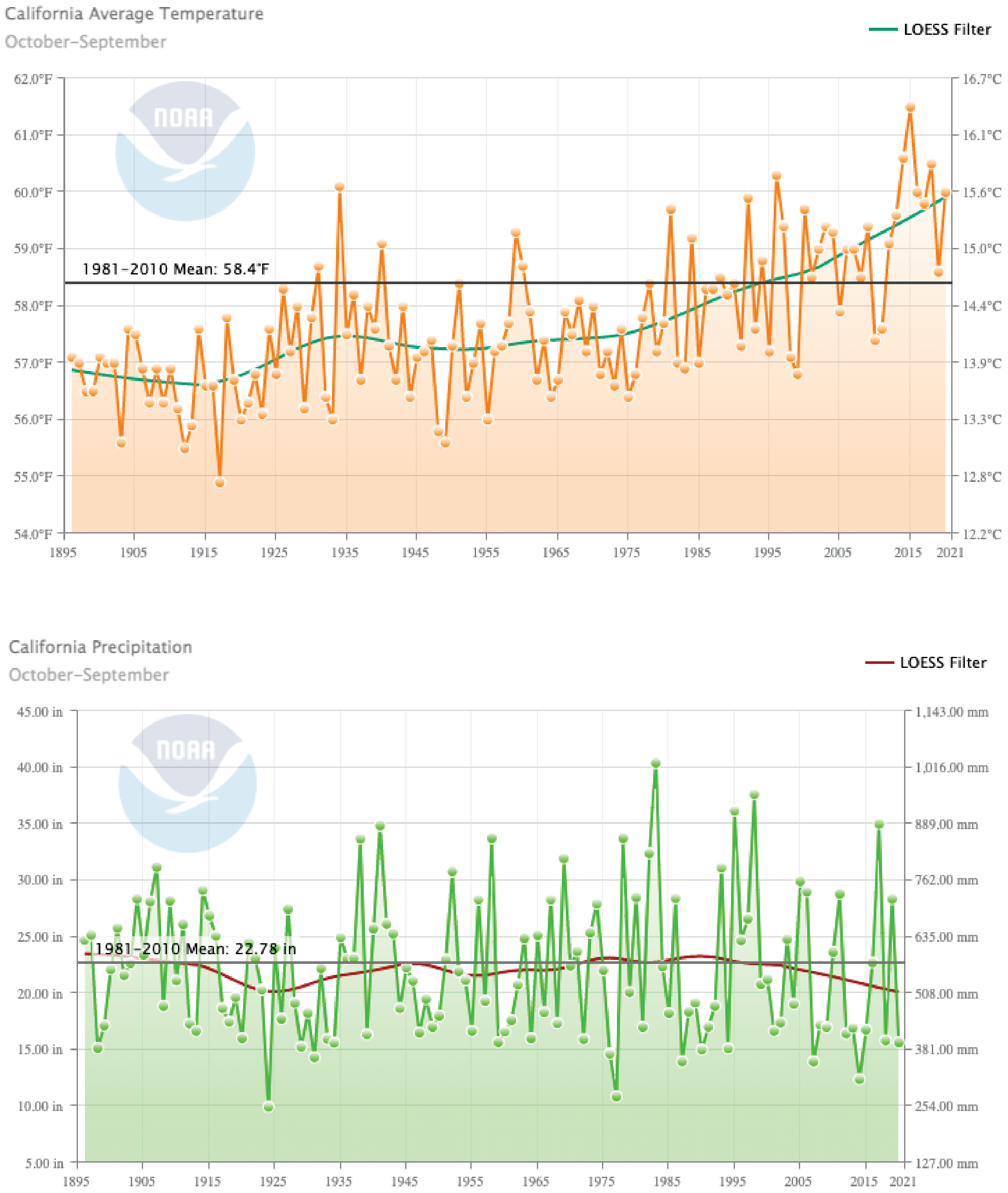

Figure 2.1. Water year (October–September) surface air temperature (top panel) and precipitation (bottom panel) for California. In each panel, the historical average for 1981–2010 is shown with the black horizontal line, and trends are shown with a LOESS smoothing curve. These figures show US Climate Division Data and were created at https://www.ncdc.noaa.gov/cag/regional/time-series.

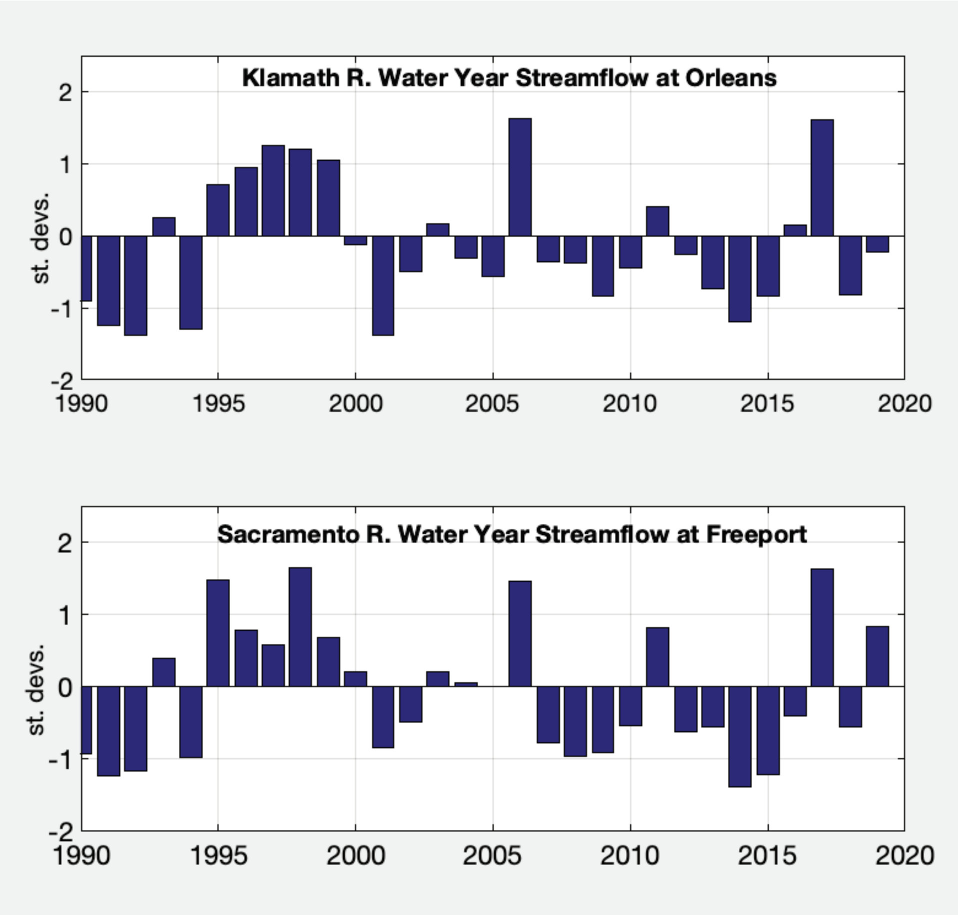

Figure 2.2. Water year streamflow anomalies (normalized with respect to the 1981–2010 mean and standard deviation) for the Klamath River and Sacramento River. Data for this figure were downloaded from the USGS (waterdata.usgs.gov).

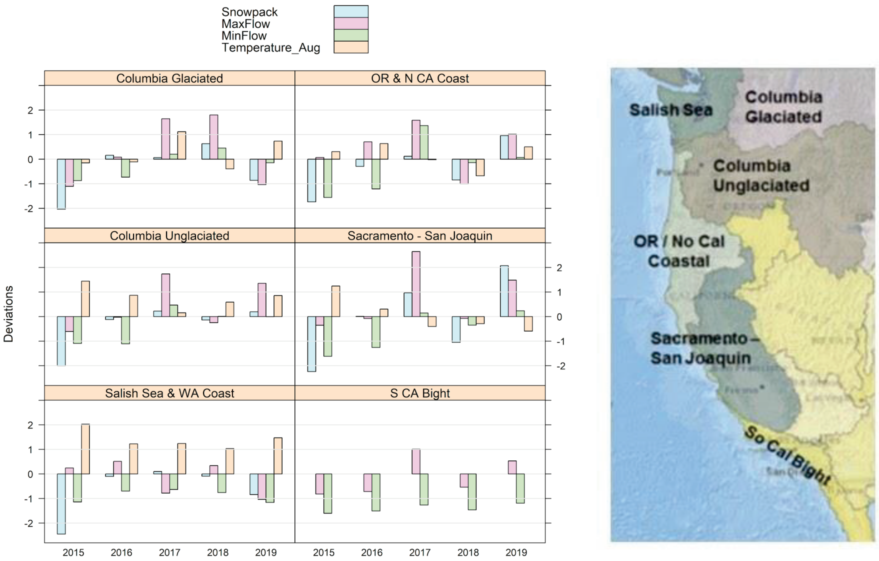

Figure 2.3. Deviations from the 1998–2014 baseline period in selected ecoregions in the maximum 1-day flow event per year (MaxFlow), the minimum 7-day flow event per year (MinFlow), snowpack on April 1, and mean August stream temperature. Map of freshwater ecoregions within which conditions were averaged. Courtesy of Harvey et al. 2019.

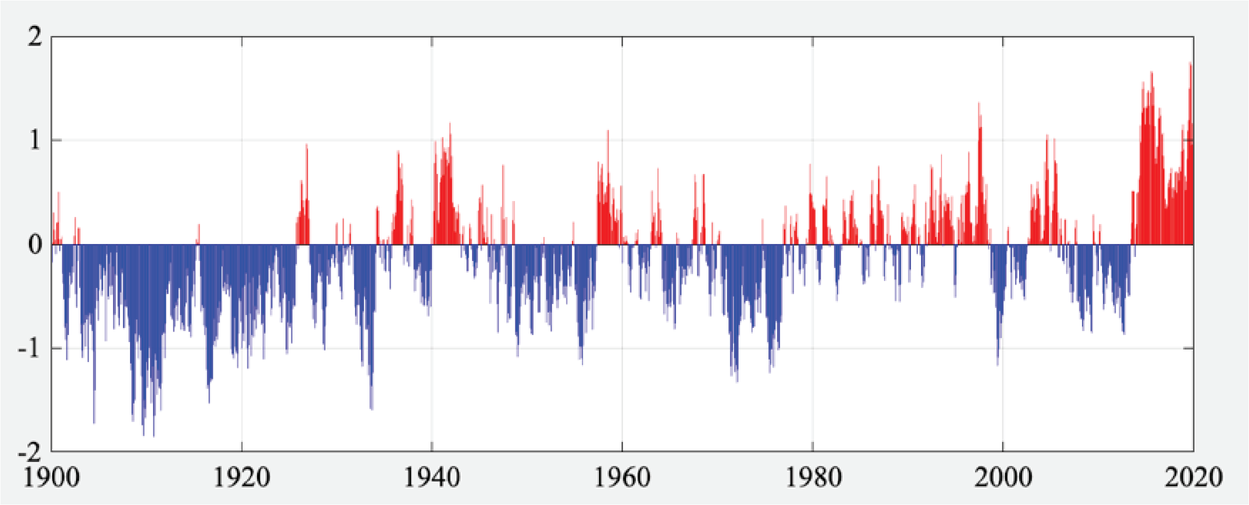

Figure 2.4. Monthly average sea surface temperature anomaly time series (in Degrees C) for the NE Pacific Arc pattern defined by Johnstone and Mantua (2014).

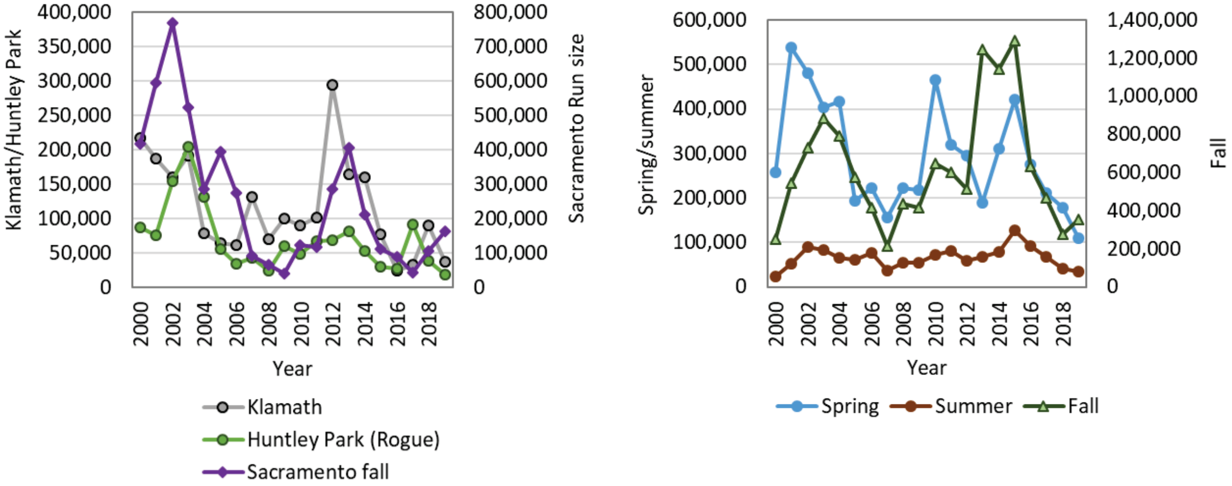

Figure 2.5. Total escapement of adult Sacramento River fall-run Chinook salmon, total in-river run size of Klamath fall-run Chinook salmon, and counts of natural fall-run Chinook salmon at Huntley Park (Rogue River) (left), and total in river run size for Columbia River spring-, summer-, and fall-run Chinook salmon (right) during 2000–2019. Data from (Pacific Fisheries Management Council 2020).

3 Southern Oregon / Northern California Coast Recovery Domain

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3.1 Southern Oregon/Northern California Coast Coho Salmon ESU

The geographic setting of the Southern Oregon/Northern California Coast (SONCC) coho salmon ESU includes coastal watersheds from Elk River (Oregon) in the north to Mattole River (California) in the south. The ESU is characterized by three large basins and numerous smaller basins across a diverse landscape. The Rogue River and Klamath River basins extend beyond the Coast Range and into the Cascade Mountains. The Eel River basin also extends well inland, including inland portions at relatively high elevation and portions that experience drier conditions and warmer temperatures during summer (Williams 2016). The numerous moderate and smaller coastal basins in the ESU experience relatively wet, cool, and temperate conditions that are in contrast to interior subbasins of the Rogue, Klamath, and Eel basins, which exhibit a range of conditions including snowmelt-influenced hydrographs, hot dry summers, and cold winters, although for the Eel Basin the influence of snowmelt on the hydrograph is fairly minor in most years. Environmental conditions in the lower portions of these large basins are more similar to the smaller coastal basins than they are to their interior subbasins (Williams 2016).

As detailed in the previous viability assessment (Williams et al. 2016), a Technical Recovery Team (TRT) for the SONCC Recovery Domain both described historical population structure (Williams et al. 2006) and developed viability criteria (Williams et al. 2008) of the ESA-listed SONCC coho salmon ESU. The approach used by the TRT conforms to the definition of independent "viable salmonid populations' developed by McElhany et al. (2000) that included consideration of diversity, spatial structure, productivity, and abundance. Williams et al. (2006 and 2008) defined historically independent and dependent populations, and proposed viability criteria for both individual populations (Table 3.1) and diversity strata (Table 3.2), which are groups of populations that likely exhibit genotypic and phenotypic similarity as a result of their exposure to similar environmental conditions and their common evolutionary history. Historically independent populations are those whose dynamics and extinction risk over a 100-year time period were not substantially altered by exchanges with individuals with other populations. Dependent populations are those whose extinction dynamics that were likely dependent on interactions with neighboring (primarily independent) populations.

The population viability criteria developed by the TRT was an extension of an approach developed by Allendorf et al. (1997) and include criteria related to population abundance (effective population size), population decline, catastrophic decline, spawner density, and hatchery influence metrics (Table 3.1). In general, the spawner density criterion, which seeks to ensure a population's ability to fulfill its historical role within the ESU, is the most conservative and preliminary viability targets for each population determined primarily by this criterion.

While the metrics used for assessing viability in this assessment are primarily based on adult abundance, the criteria were designed to address all four VSP parameters described by McElhany et al. (2000), including abundance, productivity, spatial structure, and diversity (Spence et al. 2008, Williams et al. 2008). The density-based abundance criterion was explicitly designed to, if met, ensure that spatial structure and diversity are protected at the population level (see pages 32–43 of Spence et al. 2008). Other criteria listed in Table 3.1, particularly the effective population size (or population size per generation) criteria, are highlighted when appropriate to identify instances where the population is at such low abundance that genetic risks are deemed high. Likewise, the narrative discussions of potential hatchery influence are intended to identify heightened risk to diversity associated with hatchery production.

Note that spatial structure and diversity concerns are also addressed in stratum-level criteria for representation and redundancy, and we have attempted to highlight instances where most or all populations within a diversity stratum are either extremely depressed or extirpated, indicating high concern about both spatial structure and diversity.

After the TRT developed viability criteria for the SONCC coho salmon ESU, the NMFS published the federal recovery plan for the ESU (NMFS 2014a). The recovery plan established viability criteria for populations, diversity strata, and the ESU. These recovery criteria generally follow the viability criteria developed by the TRT, but may deviate slightly for certain populations based on additional analysis.

ESU Delineation

The SONCC coho salmon ESU currently includes natural-origin coho salmon originating from the Elk River (Oregon) in the north to the Mattole River (California) in the south, inclusive, an area that extends from Cape Blanco, Oregon, to Punta Gorda, California. Data reported in the previous assessment based on samples collected in 2003 (Gilbert-Horvath et al. 2016) do not suggest the need for a re-examination of the delineations between the Central California Coast coho salmon ESU and the SONCC coho salmon ESU. The Biological Review Team for the Oregon Coast coho salmon ESU reviewed genetic data and concluded that a reconsideration of the ESU delineation between the SONCC and Oregon Coast coho salmon ESUs was not necessary (Stout et al. 2010).

Summary of Previous Assessments

Status reviews by Weitkamp et al. (1995), Good et al. (2005), NMFS (2011), and NMFS (2016) concluded that the SONCC coho salmon ESU was likely to become endangered. Risk factors identified in these early status reviews included severe declines from historical run sizes, the apparent frequency of local extirpations, long-term trends that were clearly downward, and degraded freshwater habitat and associated reduction in carrying capacity.

In the two most recent viability assessments, Williams et al. (2011 and 2016) considered that the ESU was not viable and at moderate extinction risk. The most recent assessment (Williams et al. 2016) expressed concerns that included the scarcity of long-term data on coho salmon abundances at the population unit scale and the predominantly negative abundance trends across the ESU for the populations with adequate data. The low numbers of adults counted at the Shasta River in 2014–2015 was of particular concern. Williams et al. (2016) reported that it was evident that many independent populations were well below the low-risk abundance targets, and several were likely below the high-risk depensation thresholds specified by the TRT and proposed in the Recovery Plan (NMFS 2014a). The apparent negative trends across the ESU were of great concern, as was the lack of information to determine if there had been improvement in freshwater habitat and survival. The negative trends were considered in the context of the apparent low marine survival during the period that likely contributed to the observed declines (Williams et al. 2016).

New Data and Updated Analyses

Abundance and Trends

Application of viability criteria requires population-level estimates of adult spawner abundance spanning a minimum of four generations for independent populations (Williams et al. 2008). Quantitative population-level estimates of adult spawner abundance spanning more than 9–12 years are scarce for independent or dependent populations of coho salmon in the SONCC ESU. In the Oregon portion of the ESU, population-level estimates are available for one independent population along the Oregon coast (Northern Coastal Diversity Stratum), whereas no time series at the population unit spatial scale exist in the Interior-Rogue Diversity Stratum. The 50-year time series for the Elk River independent population in the Northern Oregon Coastal Diversity Stratum was not included in previous assessments.

Since the mid-2000s, implementation of the California Coastal Monitoring Plan (CMP; Adams et al. 2011) has expanded, and shorter time series of adult spawner or redd abundance are becoming available for many populations in California. There are six independent populations currently monitored at the population scale in the California portion of the SONCC ESU, one fewer than what was available for the 2015 assessment (Smith River data are no longer collected). The CMP framework provides population abundance estimates at the appropriate spatial scale (i.e., population unit) based on redd counts from surveys of stream reaches selected using a Generalized Randomized Tessellation Survey (GRTS) design; although currently there are not appropriate life cycle monitoring stations (LCMs) to inform spawner:redd corrections in the SONCC ESU at the population unit spatial scale. Along with these estimates of redds, the two longest time series in California portions of the ESU are weir counts for the Shasta and Scott rivers independent populations. The trends informed by redd estimates will increase greatly in value as these time series become longer and we gain a better understanding of the relationship between spawner:redd ratios among populations and among years within a population. Without having appropriate LCM data (e.g., annual or regional variation in spawner:redd ratios to account for differences in observation probability among years and sites) there is uncertainty in the relationship between redd estimates and adult fish. Thus, for discussion purposes (i.e., not used for trend analysis) a ratio of 2 adult:redd will be used when comparing redd estimates with recovery targets, with acknowledgment that there are likely biases associated with interannual and spatial differences in redd observation probability.

Trends in abundance were calculated for all seven populations that had data at the independent population spatial scale since all had at least six years of data available (Table 3.1). Although only three of the time series of abundance at the independent population spatial scale meet the requisite four generations (i.e., 12 years) called for by the TRT for application of viability criteria, all still provide a substantially better basis for assessing viability compared with previous reviews and will increase greatly in value as these time series become longer.

Besides the population-unit spatial scale estimates that are required to appropriately assess population viability, there are several other data sets that provide insight into the condition of coho salmon in the ESU although at spatial scales that do not allow for direct comparison with population viability criteria. In addition, many of these partial or composite estimates are in diversity strata with no population-level estimates and thus provide valuable information to inform assessment of viability and status of the ESU. Partial and composite population counts or estimates added to this assessment include partial counts and estimates of adult coho salmon in the Upper Klamath Population and a composite estimate from the Interior-Trinity River Diversity Stratum. In addition, estimates of adult coho salmon from one (Elk River) of the four independent populations in the Northern Coastal Basins Diversity Stratum are included in this assessment. Other estimates of adult abundance for independent and dependent populations in this stratum were not considered for this assessment since issues with the surveys do not allow for the evaluation of trends or provide appropriate estimates of abundance (C. Lorion, ODFW, personal communication). The assumptions underlying all of these estimates are most likely met in the Elk River surveys, although ODFW states that even the Elk River estimate is best viewed as an index of abundance because the survey frequency varies among sites, and sampling does not occur in all areas where coho salmon spawn (Draft Rogue-South Coast Multispecies Conservation and Management Plan, ODFW).

Most California data presented in this assessment are from a dataset compiled by the California Department of Fish and Wildlife (CDFW 2020), which includes time series generated by CMP activities; however, a few datasets were acquired from alternative sources. All Oregon data were provided by ODFW.

Northern Coastal Basins Diversity Stratum. A population-level estimate of adult abundance is available for one of the four independent populations in this stratum. The 50-year average annual number of adult coho salmon in the Elk River is 166, the trend in abundance over the past 50 years is positive and significantly different from zero (slope = 0.04, p=0.007, Table 3.3, Figure 3.1). Over the past 12 years, the abundance has averaged 296 fish, far less than the recovery target of 2,400 adults and categorizing this population at a moderate extinction risk (NMFS 2014a). The trend in abundance over the past 12 years is negative and not significantly different from zero (slope = -0.029) with the 95% confidence interval including zero (Table 3.3, Figure 3.1). These estimates are based on an expansion factor applied to peak counts[2], then expanded based on distanced surveyed (km) and total spawning habitat basin (km), adjusting for observations of marked fish and a correction based on area-under-the-curve values. The assumptions underlying the expansions likely hold better in some watersheds than others; all expansions are based on small numbers of fish actually observed (C. Lorion, ODFW personal communication).

There are no stratum-wide estimates of abundance for the Northern Coastal Basins stratum; however, given the population-level information available, it is clear this stratum is well below the abundance recovery target of 7,450 adult coho salmon (NMFS 2014a).

Central Coastal Basins Diversity Stratum. A population-level estimate of adult abundance is available for one of the five independent populations in this stratum. There are now seven years of data for the Redwood Creek population (CDFW 2020). The 7-year average of redds estimated in Redwood Creek is 464. Methods for expanding these redd counts to population estimates have not yet been developed; there is no LCM in this stratum as for the purpose of developing spawner:redd ratios. Assuming an average spawner:redd ratio of 2:1, the 7-year average equates to approximately 928 adult coho salmon, 19% of the recovery target of 4,900 adults and categorizing this population as at moderate risk of extinction (NMFS 2014a). The estimated trend in abundance over the past seven years is negative but not significantly different from zero (slope = -0.067) with the 95% confidence interval including zero (Table 3.3, Figures 3.2c and 3.3c). Unfortunately, Redwood Creek data were not collected in 2016–2017; the arithmetic mean and geometric mean were based on the seven consecutive years of data that were available (2011–2016, and 2018), the harmonic mean is calculated on the 3-year running sum and therefore could only be calculated for the years 2013–2016 when data collection was not interrupted. Population-level estimates derived from redd estimates for the Smith River independent population are no longer being collected.

Mill Creek is a tributary of the Smith River in California, and represents part of the Smith River independent population. For the past seven years (2012–2018) redd counts are available that provide information on the trends of the coho salmon spawning in this portion of the Smith River Basin (CDFW 2020), but there are no estimates for the Smith River beyond this tributary. For this period of collection, the number of redds counted averaged 215 (range: 91–482) annually and the trend in those counts over the period has been negative with the 95% confidence interval not including zero (Table 3.4, Figures 3.4a and 3.5a). Methods for expanding these redd counts to population estimates have not yet been developed; there is no LCM in this stratum for the purpose of developing spawner:redd ratios. Assuming an average spawner:redd ratio of 2:1, this average number of redds equates to approximately 430 adults,

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[2] Available at: https://www.dfw.state.or.us/fish/CRP/rogue_south_coast_multi-species_conservation%20and%20Management_plan.asp

which exceeds the depensation threshold for the entire Smith River population of 325 adults (NMFS 2014a).

No stratum-wide estimate of abundance is available for the Central Coastal Basins stratum, the recovery abundance target for this stratum is 18,290 (NMFS 2014a).

Southern Coastal Basins Diversity Stratum. Population-level estimates of adult abundance are available for two of the three independent populations in this diversity stratum. Redd estimates, not estimates of adult escapement, are available for the Humboldt Bay Tributaries and Mattole River independent populations (CDFW 2020). Methods for expanding redd counts to population estimates have not yet been developed for the purpose of developing spawner:redd ratios for these populations, the only life cycle station in this stratum (Freshwater Creek) has been deemed insufficient (S. Ricker, CDFW, personal communication). The 8-year average redd estimate for the Humboldt Bay Tributary population is 908 redds with a negative trend over the past eight years (slope = -0.103) with the 95% confidence interval including zero (Table 3.3, Figures 3.2d and 3.3d). Assuming an average spawner:redd ratio of 2:1, this average equates to approximately 1,816 adult coho salmon, 32% of the recovery target of 5,700 adult coho salmon for the Humboldt Bay Tributaries independent population and placing this population at moderate risk of extinction (NMFS 2014a). There is no estimate of abundance for the Lower Eel/Van Duzen Rivers population, the other independent population in this stratum.

An estimate of spawners over the past 16 years in Freshwater Creek (CDFW 2020), a Humboldt Bay tributary, includes estimates from 2002–2003 to 2017–2018 with a trend that is negative with the 95% confidence interval including zero over the past 16 years, and positive with the 95% confidence interval including zero over the most recent 12 years (Table 3.4, Figures 3.4b and 3.5b).

There are six years of redd estimates available for the Mattole River independent population (CDFW 2020), averaging eight redds/year with a negative trend over the 6-year period with the 95% confidence interval including zero (slope = -0.634 (Table 3.3, Figures 3.2f and 3.3f). Assuming an average spawner:redd ratio of 2:1, this average equates to approximately 16 adult coho salmon annually, approximately 2% of the recovery target of 1,000 adult coho salmon for the Mattole River independent population. In addition, this average categorizes this population at high-risk based on depensation and population size per generation criteria (Table 3.1).

No stratum-wide estimates of abundance are available for the Southern Coastal Basins stratum. Given the population-level information available for Humboldt Bay and the Mattole River, the stratum is likely well below the abundance recovery target of 14,600 adult coho salmon (NMFS 2014a).

Interior-Rogue Diversity Stratum. Population-level estimates of adult abundance are not available for any of the three independent populations in this diversity stratum (Illinois River, Middle Rogue/Applegate Rivers, and Upper Rogue River).

There is a long-term composite estimate that includes portions of all four independent populations in the Interior-Rogue Diversity stratum that provides insight into trends of coho salmon in the Rogue River Basin. These estimates are derived from mark-recapture estimates based on returns to Cole River Hatchery expanded by the mark rate observed at Huntley Park and they represent a composite of four independent populations and two diversity strata (Rogue River and Northern Coastal diversity strata). The Huntly Park estimates provide the best overall assessment of coho salmon spawner abundance and trend in the basin (Good et al. 2005). The 23-year annual average of 7,233 adult coho salmon has a negative trend with the 95% confidence interval including zero (Table 3.4; Figures 3.6a and 3.7a). The average over the most recent 12 years is 5,339 adult coho salmon annually with a positive trend with the 95% confidence interval including zero over the most recent four generations (Table 3.4; Figures 3.6a and 3.7a). The Huntly Park data has been recently revised by ODFW based on a review by ODFW that identified methodological issues in the way that unmarked hatchery fish were accounted for in previous Huntly Park estimates. The new method was applied to data from 1996 to present (C. Lorion, ODFW, personal communication). Accordingly, this assessment used the revised estimates.

There are no stratum-wide estimates of abundance available for the Interior-Rogue Diversity Stratum; however, the composite Rogue estimate includes large portions of the stratum and provides a useful approximation of stratum-wide trends in abundance. The short-term (12-year) average annual estimate of 5,339 and long-term (23-year) average annual estimate of 7,233 adults suggests that the stratum is well below the abundance recovery target of 28,000 adult coho salmon (NMFS 2014a).

Interior-Klamath Diversity Stratum. Population-level estimates of abundance are available for two of the five independent population in this diversity stratum (CDFW 2020): Shasta River (18 years) and Scott River (12 years). These estimates are based on video counts at weirs.

Over the last 18 years, the number of adults passing the Shasta weir has averaged 91 adult coho salmon with a negative trend (-0.111) with the 95% confidence interval including zero (Table 3.1, Figures 3.2b and.3.3b), well below the recovery target of 4,700 fish and the high-risk threshold of 206 fish. Fewer than 50 fish have been counted in 10 of the past 12 years from 2007–08 to 2018–2019 (Giudice and Knechtle 2019) with a positive trend (0.024) with the 95% confidence interval including zero (Table 3.1). Over the past 12 years, the average spawner abundance in the three consecutive years of lowest abundance fell below the high-risk depensation threshold of 206 fish. The average abundance was low enough to categorize the population at high extinction risk for the population size per generation criterion and the population has experienced a decline in abundance of greater than 90% within one generation

($\hat{C}$ = 0.92; Table 3.3) which also categorizes this population as at high-risk of extinction (Table 3.1).

Scott River has averaged 670 adult coho salmon per year over the last 12 years, and the trend has been positive (slope = 0.037) with the 95% confidence interval including zero (Table 3.3, Figures 3.2a and 3.3a). The 12-year average of 670 fish is well below the recovery target of 6,500 (NMFS 2014a) and categorized as moderate-risk for population size per generation criterion (Williams et al. 2008).

Surveys for spawning coho salmon from Iron Gate Dam (IGD) downstream to Portuguese Creek are required in the monitoring and evaluation section of the Iron Gate Hatchery (IGH) Genetic Management Plan (NOAA 2014). The mainstem Klamath River and tributaries were surveyed by a collaboration of groups including Mid Klamath Watershed Council, Klamath National Forest Happy Camp/Oak Knoll Ranger District, Karuk Tribal Fisheries Program, California Department of Fish and Wildlife Arcata and Yreka offices, and U.S. Fish and Wildlife Service Arcata office. The Mid Klamath Watershed Council under contract by Pacific Power, conducted surveys of tributaries and reported the findings of these efforts (Dennis et al. 2017, 2018, and 2019). Not all portions of the area inhabited by this population are included in these surveys and there are a mix of field methods used providing both counts and estimates. Nevertheless, these efforts have provided the first dataset of natural-area spawning of the Upper Klamath coho salmon independent population and thus are extremely informative to understanding the condition of this population.

The average number of adult coho salmon counted by this survey during the years 2015–2016 to 2018–2019 was 273, ranging from 164 to 390 (Table 3.4, Figures 3.8c and 3.9c). Given the short-term nature of the data set, the trend was not calculated. In general, although this is a short time series and does not constitute a full population estimate, these data can be considered a minimum number of adult coho salmon spawning in natural areas of the Upper Klamath River independent population. In Figures 3.8 and 3.9, time series of abundance and trends for independent populations adjacent to the Upper Klamath River population (i.e., Scott and Shasta independent populations) are provided for context and collectively they represent three of the five independent populations in the Interior-Klamath Diversity Stratum.

There are no stratum-wide estimates of abundance are available for the Interior-Klamath Diversity stratum; however, given the population-level information available, it is clear this stratum is well below the abundance recovery target of 20,600 adult coho salmon (NMFS 2014a).

Interior-Trinity Diversity Stratum. There are no population-level estimates of adult abundance for any of the three independent populations in this stratum.

There is a long-term composite estimate that includes portions of all three independent populations (Lower Trinity River, Upper Trinity River, and South Fork Trinity River populations) and provides an estimate of the number of natural-origin adult coho salmon returning to natural spawning areas from Willow Creek weir upstream to the Trinity River Hatchery (Kier et al. 2019; CDFW 2020). The short-term (12-year) average is 1,116 adult coho salmon and the trend is negative with the 95% confidence interval not including zero (Table 3.4). The average over the 22 years (period of record) of the time series is 1,653 adult coho salmon with a negative trend with the 95% confidence interval including zero (Table 3.4, Figures 3.6b and 3.7b). In the most recent survey used in this analysis, Kier et al. (2019) reported that the most recent estimate was predominantly hatchery-origin fish; only an estimated 42 (3%) of natural-origin fish returning to the areas included in the survey upstream of Willow Creek weir in that year were of natural origin. Thus, 97% of returning adult coho salmon are considered of hatchery origin, far above the <5% required for low-risk viability threshold for the hatchery influence criterion (Williams et al. 2008).

There are no stratum-wide estimates of abundance available for the Interior-Trinity Diversity stratum. However, the composite estimate includes large portions of the stratum and provides a useful approximation of stratum-wide trends in abundance. The short-term (12-year) composite average of 1,116 and long-term (22 years) average of 1,653 indicate that the stratum is likely below the abundance recovery target of 10,370 adult coho salmon (NMFS 2014a). Moreover, returning adult coho salmon are predominantly of hatchery origin.

Interior-Eel Diversity Stratum. There is a population-level estimate of adult abundance for one of the three independent populations in this stratum (CDFW 2020). The nine-year time series of the South Fork Eel River independent population has averaged 1,223 redds per year (Table 3.3, Figures 3.2e and 3.3e). Although a negative trend, the 95% confidence interval includes zero. Methods for expanding these redd counts to population estimates have not yet been developed; there is no LCM in this stratum for the purpose of developing spawner:redd ratios. Assuming an average spawner:redd ratio of 2:1, this average equates to approximately 2,446 adult coho salmon, 26% of the recovery target of 9,300 adults and categorizing this population at moderate extinction risk.

No stratum-wide estimates of abundance are available for the Interior-Eel Diversity stratum however, given the population-level information available it is likely that the stratum is below its abundance recovery target of 18,200 adult coho salmon (NMFS 2014a).

Hatcheries

There are three hatcheries in the SONCC coho salmon ESU and all three are included in the ESA-listed ESU. The hatcheries include Cole River Hatchery (CRH) on the Rogue River, Iron Gate Hatchery (IGH) on the Klamath River, and Trinity River Hatchery (TRH) on the Trinity River. One key development since the previous assessment in 2015 is the completion of the Hatchery Genetic Management Plan (HGMP) for the TRH.

Cole Rivers Hatchery is operated as a harvest program[3] used for augmentation of fishing and harvest opportunities, and mitigation for the loss of habitat resulting from dam construction in the Rogue and Applegate rivers (ODFW 2020). An HGMP was completed in 1999. The hatchery stock is managed as an integrated stock. Approximately 75,000 smolts are released on-site annually; all fish are fin-clipped and 25,000 are coded-wire tagged (ODFW 2020). The coho salmon program at CRH provides monitoring opportunities related to ocean distribution and harvest.

The TRH coho salmon program under the HGMP will be operated as an integrated program (Bureau of Reclamation and California Department of Fish and Wildlife 2017), with the intention of allowing the natural environment to drive the adaptation and fitness of a composite population of fish that spawn both in the hatchery and in the wild (i.e., natural areas) for the purpose of increasing total adult abundance, productivity, and fitness while minimizing genetic divergence of hatchery broodstock from the natural-area spawning population. The goal of the program is to provide fish for harvest in a manner consistent with the conservation of the coho salmon population while meeting TRH mitigation requirements (Bureau of Reclamation and California Department of Fish and Wildlife 2017). The implementation of the HGMP is a positive step in progress towards meeting viability targets for Upper Trinity River independent coho salmon population, the diversity stratum, and the ESU.

With the implementation of a HGMP for IGH (California Department of Fish and Wildlife and PacifiCorp 2014), monitoring of natural area spawning in tributaries of the Upper Klamath River independent population is now providing information to better understand the presence of coho salmon in the population (see above discussion of Interior-Klamath Diversity Stratum).

Harvest Impacts[4]

Southern Oregon/Northern California Coast coho salmon are primarily distributed off the coast of California and southern Oregon. Since 1999, ocean fisheries have been managed to achieve a predicted exploitation rate of no more than 13.0% on Rogue/Klamath hatchery coho salmon (NMFS 1999a).

Coho salmon-directed ocean fisheries and coho salmon retention have been prohibited off the coast of California since 1996. Ocean fishing mortality of SONCC coho salmon results from non-retention impacts in California and Oregon Chinook salmon-directed fisheries, impacts in Oregon mark-selective coho salmon fisheries (primarily non-retention), and impacts in Oregon non-mark-selective fisheries.

Rogue/Klamath coho salmon ocean exploitation rates have been estimated for years 1986–2019 using postseason runs of the Fishery Regulation Assessment Model (FRAM).

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[3] The ODFW Hatchery Management Policy defines hatchery programs as either harvest or conservation programs. Harvest programs operate to enhance or maintain fisheries without impairing naturally reproducing populations (ODFW 2020).

[4] Michael O'Farrell (NMFS SWFSC Santa Cruz) prepared this section on harvest impacts.

Exploitation rates have been low and relatively stable since the early 1990s (average of 5.4% for years 1994–2019), which contrasts sharply with the much higher rates estimated for the 1980s and early 1990s (Figure 3.10). Since the establishment of the consultation standard in 1999, ocean fishery exploitation rate estimates have been <13%.

Direct freshwater recreational fishery impacts on SONCC coho salmon are likely relatively low given California's statewide prohibition of coho salmon retention in all non-tribal fisheries and the small-scale mark-selective coho salmon fisheries in the Oregon portion of this ESU. The impacts associated with bycatch from freshwater fisheries that target other species, such as Chinook salmon and steelhead, is unknown.

Klamath Basin tribes (Yurok, Hoopa, and Karuk) harvest coho salmon for subsistence and ceremonial purposes (CDFG 2002). The Yurok Tribe's estimated harvest rates averaged 3.4% from 1992 to 2019, and 3.5% from 2015 to 2019 (Yurok Tribe, personal communication). The harvest rates reported by the Yurok Tribe are maximum rates because escapement and harvest monitoring in the Klamath Basin is not comprehensive, which has precluded a complete estimate of run size (Williams 2015). The average harvest rate for Hoopa Valley Tribal fisheries was 2.6% for years 2001 to 2017 (Hoopa Valley Tribe, personal communication). This estimate represents the average rate of harvest on the wild coho salmon returning to the Trinity River. In 2015 the Hoopa Valley Tribe began operation of a harvest weir to selectively harvest hatchery-origin salmon in the Trinity River (Orcutt 2015). Harvest impacts on natural-origin coho salmon from the weir are included in the estimation of the harvest rate. However, zero natural-origin coho salmon were trapped or handled at the weir in 2016 and 2017. Harvest rate estimates for the Karuk Tribal fisheries are not available.

In summary, the available information indicates that the level of SONCC coho salmon fishery impacts has not changed appreciably since the 2016 status review update (NMFS 2016a).

Summary and Conclusions

In summary, the available data for populations within the SONCC coho salmon ESU indicate that the independent populations with adequate data to determine a population-specific estimate of abundance remain below recovery targets and, in two cases (Shasta River and Mattole River), are below the high-risk thresholds established by the TRT and adopted in the recovery plan (NMFS 2014a). Of the seven time series available for this assessment, positive abundance trends were observed in the Elk and Scott rivers; the Elk River abundance trend was significantly different from zero, although the annual average abundance (166) and most recent 12-year average abundance (296) are well below the population recovery target of 2,400. The 12-year average of 670 fish in the Scott River is well below the recovery target of 6,500 adults and the population is categorized as at moderate-risk of extinction for the population size per generation criterion. The remaining five populations had negative abundance trends, only the Shasta River population trend was significantly different from zero. All independent populations that are included in this assessment and were included in the previous assessment five years earlier had a lower average annual abundance in this most recent assessment, including the Scott River.

The two composite estimates for the Rogue River and the Trinity River (includes multiple independent populations) provide information at a larger spatial scale and include longer time series of abundance estimates. Neither includes the entire habitat of the associated diversity stratum, but both include large portions of the stratum area. The Rogue Basin short-term (12-year) abundance trend is positive and the long-term (23-year) abundance trend is negative, but neither trend is significantly different from zero; although the average abundance is lower in the most recent 12 years compared to the full 23-year time series. The Trinity River Basin short-term abundance trend and the long-term abundance trend are both negative, with the most recent 12-year time series having a significant negative trend with extremely high levels of hatchery-origin fish. The Trinity River Basin 12-year average abundance estimate of 1,116 is 12% of the recovery target for the diversity stratum of 9,700 fish. The Trinity River Basin adults are an estimated 97% hatchery-origin fish, failing to meet the low-risk threshold of <5% hatchery-origin fish contributing to the stratum recovery target of 9,700 fish.

These composite abundance estimates do not represent a stratum-level abundance estimate, but they do provide some relative information on the number of fish in these strata. The negative trends, including a significant decline over the past 12 years of natural-origin adult coho salmon returning to natural areas from Willow Creek weir upstream to the Trinity River Hatchery are of concern.

Trends in abundance should be considered in the context of the environmental conditions that include ocean, freshwater, and climate conditions that have occurred over the period of the times series considered. With less favorable marine and freshwater conditions one may expect to find declining trends in abundance. During their freshwater phase, coho salmon in this ESU have experienced strong and persistent warming trends and large year-to-year variations in precipitation in recent decades. For both the Pacific Northwest and California, water year 2015 stands out as the warmest year on record, while water year 2018 is the second warmest year on record for California. In terms of ocean temperature, surface temperatures in the northeast Pacific Ocean were notably cooler than average from 1999–2002 and again from 2006–summer 2013, warmer than normal from 2003–2005, and at record high temperatures for much of the period from fall 2013–2019.

As with the previous viability assessment, the lack of increasing abundance trends across the ESU for most populations with adequate data are of concern and should be considered in the context of the relatively poor ocean, freshwater, and climate conditions experienced over the past 10 to 15 years.

In summary, data availability for this ESU remains generally poor, new information available since Williams et al. (2016) suggests little improvement over the five years since the last viability assessment. For the seven independent populations with appropriate data to assess population viability, none are at low extinction risk based on population viability criteria (Williams et al. 2008). Five of the seven have negative trends in abundance including two (Shasta and Mattole rivers) that are at high risk of extinction based on viability criteria (Williams et al. 2008). Of the two populations with positive abundance trends (Elk and Scott rivers), only one has a significant positive abundance trend (Elk River). The Scott River's 12-

year average of 670 fish is well below the recovery target of 6,500 (NMFS 2014a); both the Elk River and Scott River are at moderate risk of extinction based on the spawner density criterion (Williams et al. 2008).

Based on the available data, while the extinction risk category is still moderate, the recent extinction risk trend of the SONCC coho salmon ESU is declining (i.e., less viable) since previous assessment. The ESU is considered not viable and at a moderate risk of extinction.

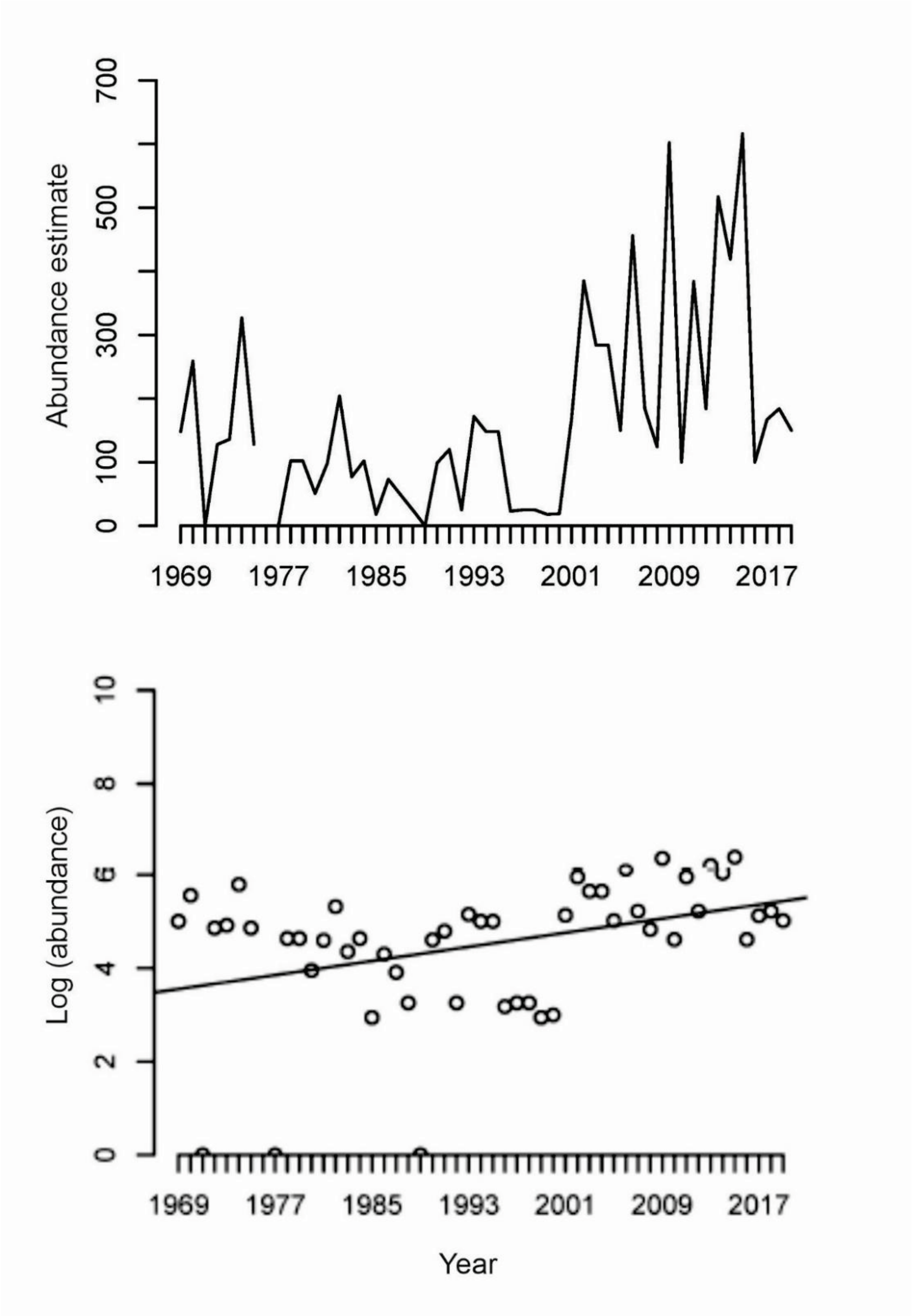

Figure 3.1. Time series of abundance estimates and population trends (log abundance) estimates for adult SONCC coho salmon in the Elk River, Oregon. Values are based on an expansion factor applied to peak counts, then expanded based on distance surveyed (km) and total amount of spawning habitat (km) in basin, adjusting for observations of marked fish. Data from ODFW (unpublished data; Available at: https://www.dfw.state.or.us/fish/CRP/rogue_south_coast_multi-species_conservation%20and%20Management_plan.asp).

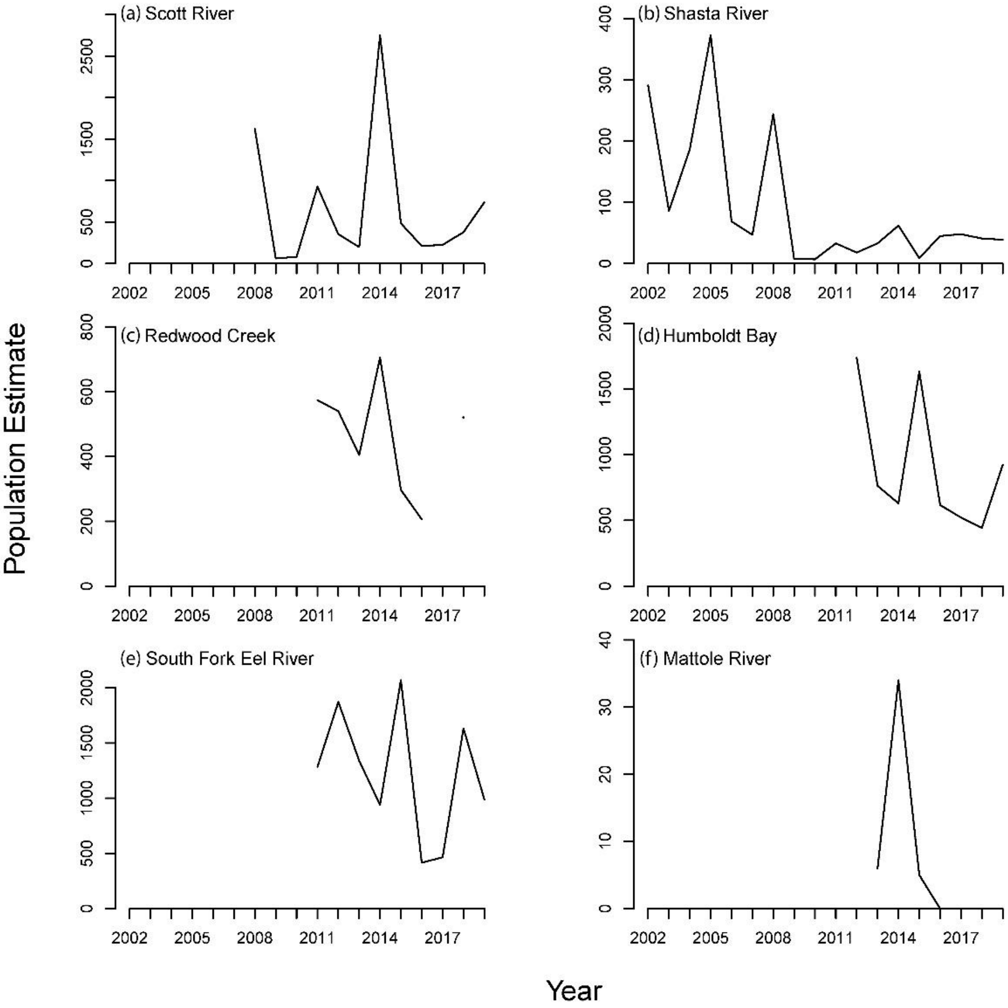

Figure 3.2. Time series of abundance estimates for independent populations of SONCC coho salmon (CDFW 2020). Values for Scott and Shasta rivers are video weir counts of adult coho salmon. All other estimates are basin-wide redd estimates.

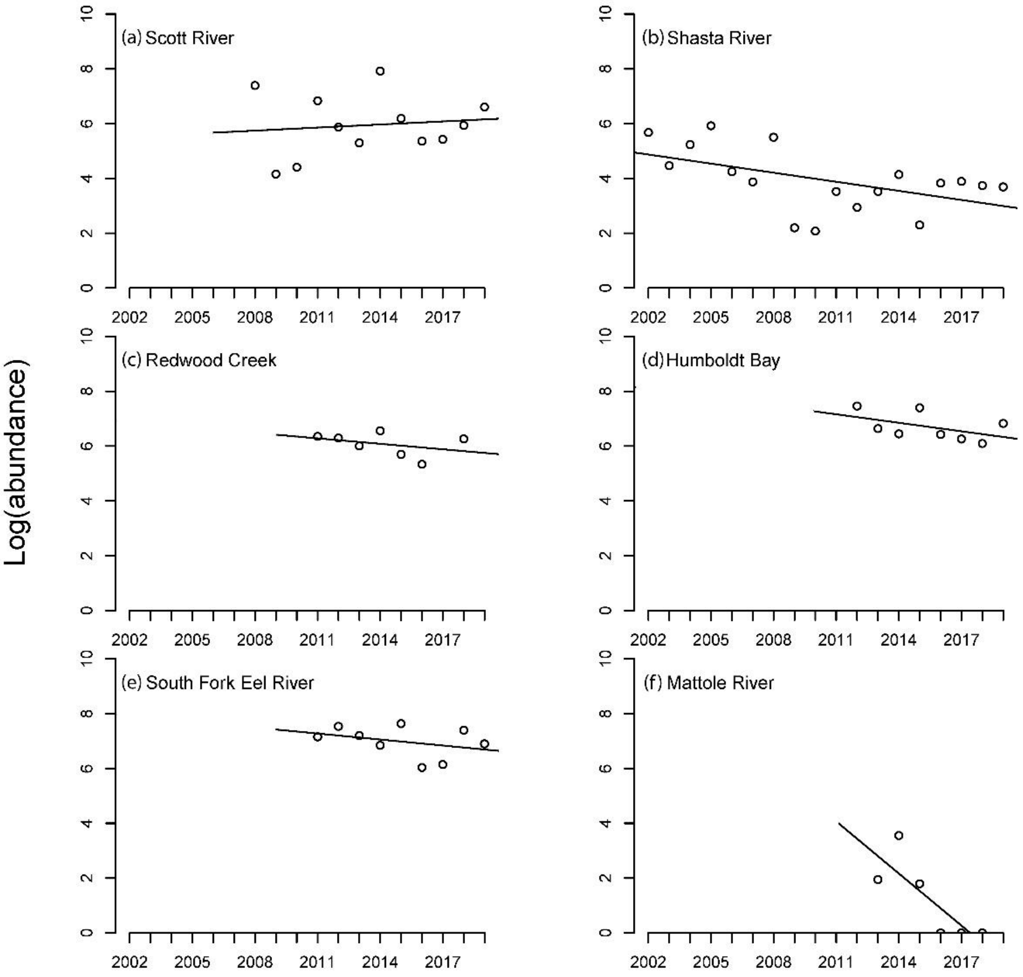

Figure 3.3. Population trends (log abundance) for independent populations of SONCC coho salmon (CDFW 2020). Values for Scott and Shasta rivers are video weir counts of adult coho salmon. All other estimates are basin-wide redd estimates.

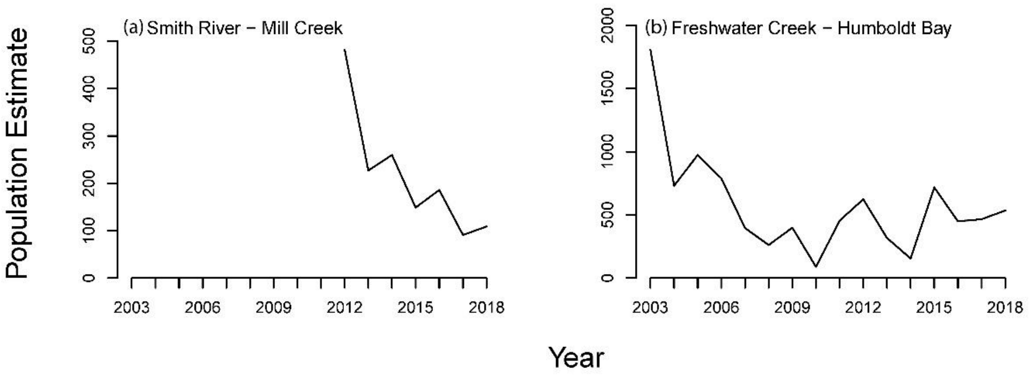

Figure 3.4. Time series of abundance estimates for partial populations of SONCC coho salmon. Values for Mill Creek, tributary to the Smith River are redd estimates (CDFW 2020). Values for Freshwater Creek are mark-recapture estimates of adult coho salmon.

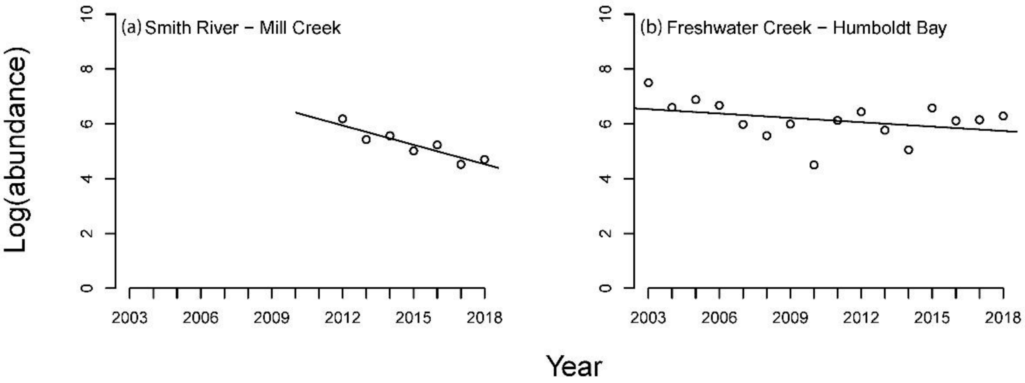

Figure 3.5. Population trends (log abundance) for partial populations of SONCC coho salmon. Values for Mill Creek, tributary to the Smith River, are redd estimates (CDFW 2020). Values for Freshwater Creek are mark-recapture estimates of adult coho salmon.

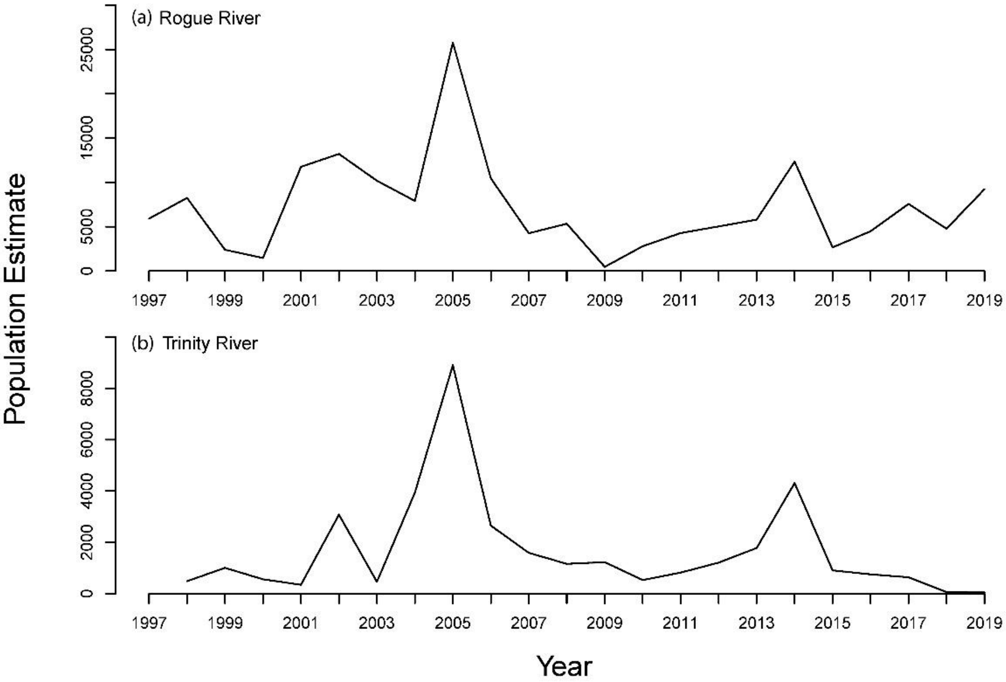

Figure 3.6. Time series of abundance estimates for composite populations (more than one independent population) of SONCC coho salmon in the Rogue and Trinity rivers. Estimates from the Rogue River are derived from mark-recapture estimates based on returns to Cole River Hatchery expanded by the mark rate observed at Huntley Park and represent a composite of four independent populations and two diversity strata (Rogue River and Northern Coastal). For the Trinity River, estimates are the number of natural-origin adult coho salmon returning to natural areas from Willow Creek weir upstream to the Trinity River Hatchery and represent a composite of three independent populations (Lower Trinity River, Upper Trinity River, and South Fork Trinity River populations) making up one diversity stratum (Trinity) (CDFW 2020).

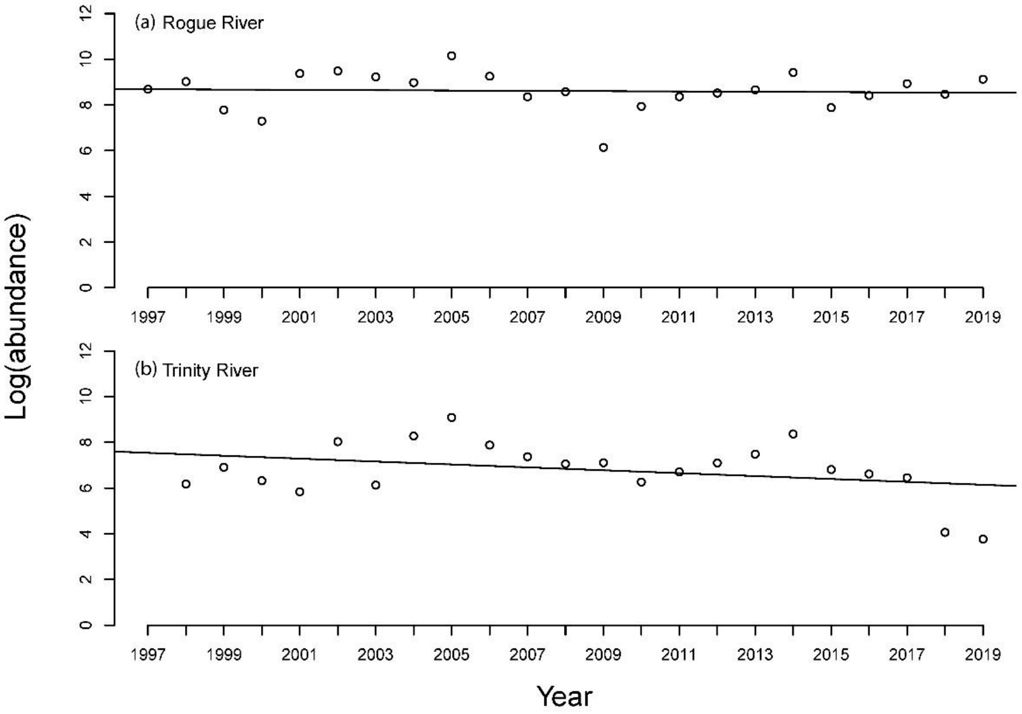

Figure 3.7. Population trends (log abundance) for composite populations (more than one independent population) of SONCC coho salmon in the Rogue and Trinity rivers. Estimates from the Rogue River are derived from mark-recapture estimates based on returns to Cole River Hatchery expanded by the mark rate observed at Huntley Park and represent a composite of four independent populations and two diversity strata (Rogue River and Northern Coastal). For the Trinity River, estimates are the number of natural-origin adult coho salmon returning to natural areas from Willow Creek weir upstream to the Trinity River Hatchery and represent a composite of three independent populations (Lower Trinity River, Upper Trinity River, and South Fork Trinity River populations) making up one diversity stratum (Interior Trinity) (CDFW 2020).

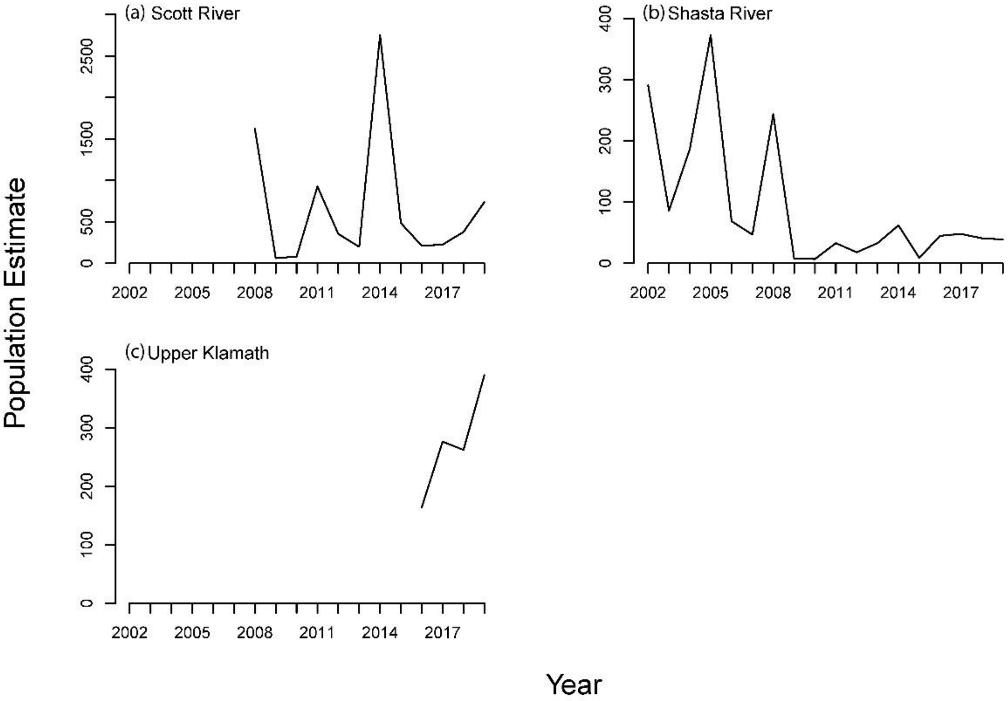

Figure 3.8. Time series for independent populations of SONCC coho salmon in the Interior Klamath Diversity Stratum. Values for Scott and Shasta rivers are video weir counts of adult coho salmon. Estimates of coho salmon in the Upper Klamath River independent population are from surveys that do not include all portions of the area inhabited by this population and based on a mix of field methods that include both counts and estimates (see text for more information; Dennis et al. 2017, 2018, and 2019).

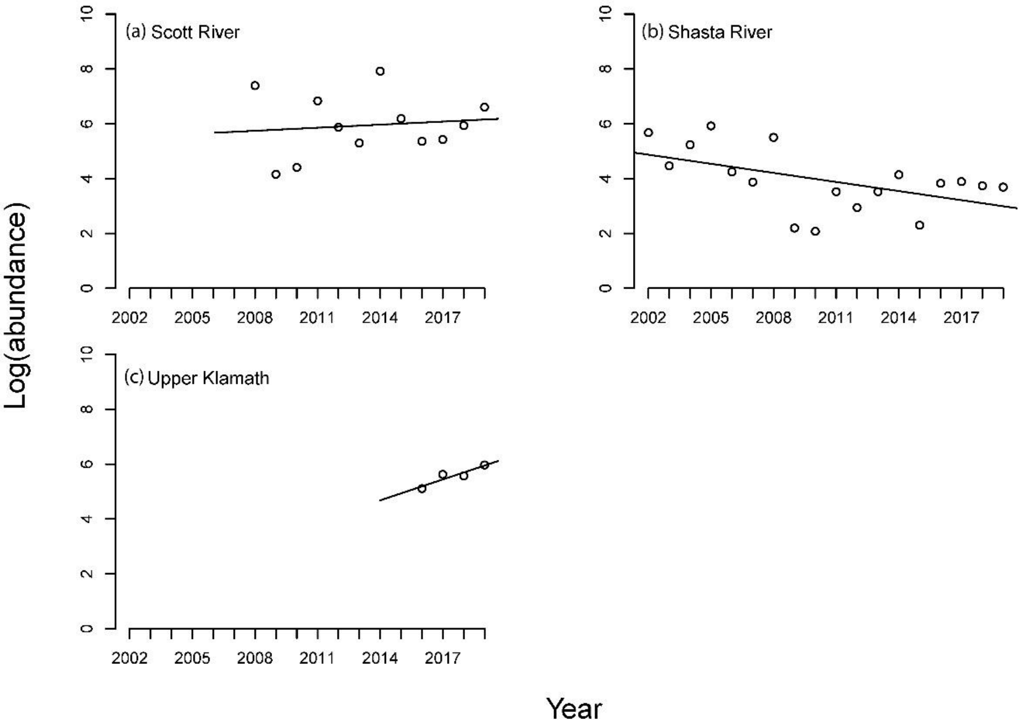

Figure 3.9. Population trends (log abundance) for independent populations of SONCC coho salmon in the Interior Klamath Diversity Stratum. Values for Scott and Shasta rivers are video weir counts of adult coho salmon (CDFW 2020). Estimates of coho salmon in the Upper Klamath River independent population are from surveys that do not include all portions of the area inhabited by this population and based on a mix of field methods that include both counts and estimates (see text for more information; Dennis et al. 2017, 2018, and 2019).

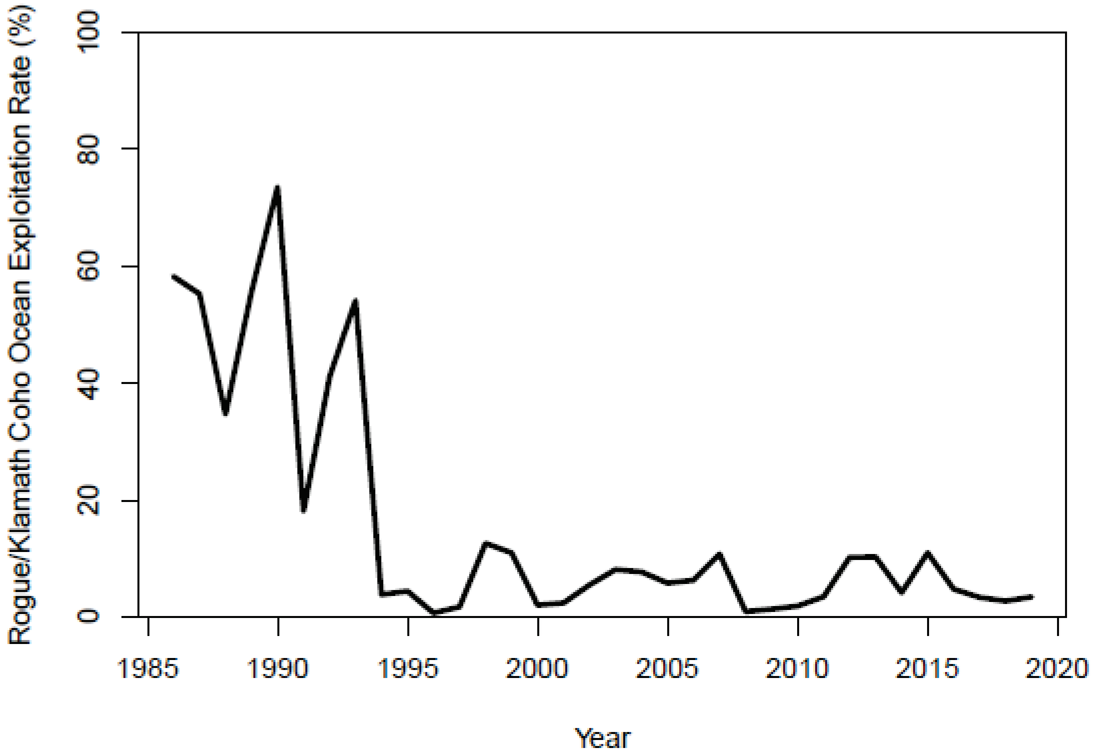

Figure 3.10. Rogue/Klamath (RK) coho salmon ocean exploitation rate estimates for years 1986–2019. Estimates provided by J. Carey, NMFS (personal communication).

Table 3.1. Viability criteria for assessing extinction risk for populations of coho salmon (*O. kisutch*) in the Southern Oregon/Northern California Coast coho salmon ESU. For a given population, the highest risk score for any category determines the population's overall extinction risk. Modified from Allendorf et al. (1997) and Lindley et al. (2007). See table footnotes for definitions of N_e, N_g, and N_a.

	Extinction risk		
Criterion	High	Moderate	Low
	- any One of -	- any One of -	- all of -
Effective population size[a]	$N_e \leq 50$	$50 < N_e < 500$	$N_e \geq 500$
- or -	*- or -*	*- or -*	*- or -*
Population size per generation	$N_g \leq 250$	$250 < N_g < 2500$	$N_g \geq 2500$
Population decline	Precipitous decline[b]	Chronic decline or depression[c]	No decline apparent or probable
Catastrophic decline	Order of magnitude decline within one generation	Smaller but significant decline[d]	Not apparent
Spawner density (adults/IP km)	$N_a / IPkm^e \leq 1$	$1 < N_a / IPkm <$ MRD[f]	$N_a / IPkm \geq$ MRD[f]
Hatchery Influence			Hatchery fraction <5%
			- in addition to above -
Extinction risk from PVA	≥ 20 % within 20 yrs	≥ 5% within 100 yrs but < 20 % within 20 yrs	< 5 % within 100 yrs[g]

a – The effective population size (N_e) is the number of breeding individuals in an idealized population that would give rise to the same variance in gene frequency under random genetic drift or the same rate of inbreeding as the population under consideration (Wright 1931); total number of spawners per generation (N_g), for SONCC coho salmon the generation time is approximately three years therefore $N_g = 3 N_a$.

b – Population has declined within the last two generations or is projected to decline within the next two generations (if current trends continue) to annual run size of $N_a \leq 500$ spawners (historically small but stable populations not included) **or** $N_a > 500$ but declining at a rate of ≥10% per year over the last two-to-four generations.

c – Annual spawner abundance (N_a) has declined to ≤ 500 spawners, but now stable **or** number of adult spawners (N_a) > 500 but continued downward trend is evident.

d – Annual spawner abundance decline in one generation < 90% but biologically significant (e.g., loss of year class).

e – *IPkm* = the estimated aggregate intrinsic habitat potential for a population inhabiting a particular watershed (i.e., total accessible km weighted by reach-level estimates of intrinsic potential; see Bjorkstedt et al. [2005] for greater elaboration).

f – Minimum required spawner density (MRD) is dependent on the amount of potential habitat available. Figure 5 of Williams et al. (2008) summarizes the relationship between spawner density and IP km.
g – For population to be considered at low-risk of extinction, all criteria must be satisfied (i.e., not just a PVA). A population viability analysis (PVA) can be also included for consideration, but must estimate an extinction risk <5% within 100 years *and* all other criteria must be met. If discrepancies exist between PVA results and other criteria, results need to be thoroughly examined and potential limitations of either approach should be carefully identified and examined.

Table 3.2. Summary of ESU viability criteria for the Southern Oregon/Northern California Coast coho salmon ESU.

ESU viability characteristic	Criteria
Representation	1. All diversity strata should be represented by viable populations.
Redundancy and connectivity	2.a. At least 50% of historically independent populations in each diversity stratum should be demonstrated to be at low risk of extinction according to the population viability criteria. For strata with three or fewer independent populations, at least two populations must be viable. AND 2.b. Total aggregate abundance of the populations selected to satisfy 2a must meet or exceed 50% of the aggregate viable population abundance predicted for the stratum based on the spawner density. 3. All dependent and independent populations not expected to meet low-risk threshold within a stratum should exhibit occupancy indicating sufficient immigration is occurring from the "core populations". 4. The distribution of extant populations, both dependent and independent, needs to maintain connectivity across the stratum as well as with adjacent strata.

Table 3.3. Viability metrics for independent populations of coho salmon (*O. kisutch*) in the SONCC coho salmon ESU. NA indicates not available or applicable; dash (-) indicates no estimate of appropriate spatial scale or sampling design for viability analysis. Trends are shown only for populations where time series is at least six years; **bold** indicates significant trend, $\hat{C}$ calculated on populations with minimum of nine years of data. California provided by CDFW (2020), Oregon data provided by ODFW (C. Lorion, ODFW, personal communication). IPkm from Recovery Plan (NMFS 2014a), $N_{a(arith)}$ target refers to minimum number of spawners to meet biological recovery criteria in recovery plan, Table 4-2 (NMFS 2014a). See Williams et al. (2008) for description of metrics reported.

Stratum/population	Yrs	$\overline{N}_{a(arith)}$	$\overline{N}_{a(geom)}$	$\overline{N}_{g(harm)}$	$\hat{C}$	$\hat{T}$ *(95% CI)*	IPkm	$\hat{D}_{dep}$	$\hat{D}_{ssd}$	$\hat{D}_{ssd}$ target	$\overline{N}_{a(arith)}$ target
Northern Coastal Basins											
Elk River											
Short-term (4-gen)	12	296	238	812	0.54	-0.029 (-0.163, 0.105)	63	**0.3**	4.7	38	2400
Long-term (period of rec)	50	166	90	262	0.82	**0.038 (0.011, 0.065)**	63				2400
Lower Rogue River							81				320
Chetco River							135				4500
Winchuck River							57				230
Central Coastal Basins											
Smith River[a]							325				6800
Lower Klamath River							205				5900
Redwood Creek[b, c]	7	464	433	1427	NA	-0.067 (-0.255, 0.122)	151	2.7	3.1	32	4900
Maple Creek/Big Lagoon[d]											
Little River							34				140
Mad River							136				550

Stratum/population	Yrs	$\overline{N}_{a(arith)}$	$\overline{N}_{a(geom)}$	$\overline{N}_{g(harm)}$	$\hat{C}$	$\hat{T}$ *(95% CI)*	IPkm	$\hat{D}_{dep}$	$\hat{D}_{ssd}$	$\hat{D}_{ssd}$ target	$\overline{N}_{a(arith)}$ target
Southern Coastal Basins											
Humboldt Bay tributaries[b]	8	908	807	2381	NA	-0.103 (-0.282, 0.075)	191	2.8	4.8	30	5700
Low. Eel/Van Duzen R.							394				7900
Bear River[d]											
Mattole River[b]	6	8	3	0	NA	-0.634 (-1.276, 0.008)	250	**0.0**	0	4	1000
Interior – Rogue											
Illinois River							590				11800
Mid. Rogue/Applegate R.							603				2400
Upper Rogue River							689				1380
Interior – Klamath											
Middle Klamath River							113				450
Upper Klamath River							425				8500
Salmon River							114				450
Scott River	12	670	382	1461	0.4	0.037 (-0.181, 0.255)	250	1.1	2.7	26	6500
Shasta River											
Short-term (4-gen)	12	49	30	111	**0.92**	0.024 (-0.164, 0.211)	206	**0.1**	0.2	23	4700
Long-term (period of rec)	18	91	49	134	0.93	**-0.111 (-0.207, -0.014)**	206				4700
Interior – Trinity											
South Fork Trinity River							242				970
Lower Trinity River							112				3600

Stratum/population	Yrs	$\overline{N}_{a(arith)}$	$\overline{N}_{a(geom)}$	$\overline{N}_{g(harm)}$	$\hat{C}$	$\hat{T}$ *(95% CI)*	IPkm	$\hat{D}_{dep}$	$\hat{D}_{ssd}$	$\hat{D}_{ssd}$ target	$\overline{N}_{a(arith)}$ target
Upper Trinity River							365				5800
Interior – Eel											
South Fork Eel River[b]	9	1223	1078	3422	0.31	-0.073 (-0.248, 0.102)	464	1.8	2.6	40	9300
Mainstem Eel River							68				2600
North Fork Eel River[d]											
Middle Fork Eel River[d]											
Mid. Mainstem Eel River							232				6300
Up. Mainstem Eel River[d]											

a – Smith River data reported in previous assessments no longer collected.

b – Redd estimate, not adult escapement estimate.

c – No sampling occurred in 2016–2017 (2017 in figures) in Redwood Creek; Personal communication, Seth Ricker, CDFW, 8 July 2020. 2017 data are missing, the mean $N_{a(arith)}$ for the 7 years where data were collected (2011–2016, and 2018), the geometric mean is calculated for the same 7 years. The harmonic mean is calculated on the 3-yr running sum, this time series is interrupted, a running sum cannot be calculated for any window that includes the missing year. Thus, the value of 1427 is the harmonic mean of the running sum of abundance values for the years 2013 through 2016.

d – Population unit designated by Williams et al. (2006 and 2008), not included in NMFS (2014).

Table 3.4. Short- and long-term trends in SONCC coho salmon ESU population abundance based on partial or composite population estimates and population indices. Trends in **bold** are significantly different from 0.0 (α = 0.05).

Spawning tributary (Population)	Years	Data type	Mean (range)	$\hat{T}$ (95% CI)	Data sources
Rogue Basin[a]	12	Composite, mark-recapture	5339 (465–12354)	0.116 (-0.024, 0.255)	ODFW
	23		7233 (465–25763)	-0.007 (-0.063, 0.049)	
Mill Creek (Smith R.)	7	Partial pop., redd count	215 (91–482)	**-0.235 (-0.361, -0.110)**	CDFW 2020
Upper Klamath[b]	4	Partial pop., counts, weir, trap[b]	273 (164, 390)		Dennis et al. 2017, 2018, and 2019.
Trinity River[c]	12	Composite, mark-recapture	1116 (60–4457)	**-0.220 (-0.426, -0.032)**	CDFW 2020
	22		1653 (60–9055)	-0.064 (-0.148, 0.021)	
Freshwater Creek[d] (Humboldt Bay)	12	Partial pop., weir-carcass mark-recapture	406 (89–718)	0.055 (-0.054, 0.164)	CDFW 2020
	16		573 (89–1807)	-0.053 (-0.133, 0.027)	CDFW 2020

a – These estimates are derived from mark-recapture estimates based on returns to Cole Rivers Hatchery expanded by the mark rate observed at Huntley Park. Data from Oregon Department of Fish and Wildlife (18 February 2020).

b – Data from Giudice and Knechte 2019; Knechtle and Giudice 2019; Dennis et al. 2017, 2018, and 2019. Counts include Iron Gate Hatchery returns (total number of fish trapped), Bogus Creek weir counts (hatchery and natural-origin adult and grilse), mainstem Klamath River between Iron Gate Dam and the Brown Bear River Access (reported by Dennis et al. 2017, 2018, and 2019) and spawning ground surveys (redd count, not expanded) in Upper Klamath River tributaries including Cottonwood, Beaver, McKinney, Doggett, Horse, Middle, Tom Martin, O'Neil, Walker, Seiad, Grider, and West creeks.

c – Trinity River run-size estimate includes all natural-origin returning adults to natural spawning areas upstream of Willow Creek weir and Trinity River Hatchery (CDFW 2020).

d – Maximum live/dead counts do not distinguish between natural and hatchery-origin spawners. Counts may include both, particularly in the early part of the time series.

4 North-Central California Coast Recovery Domain

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The North-Central California Coast Recovery Domain encompasses the geographic region from Redwood Creek (Humboldt County) south to Aptos Creek (Santa Cruz County) inclusive. Two salmon Evolutionarily Significant Units (ESUs) and two steelhead Distinct Population Segments (DPSs) lie wholly within this region: California Coastal Chinook salmon, Central California Coast coho salmon, Northern California steelhead, and Central California Coast steelhead.

As detailed in the previous viability assessment (Williams et al. 2016), a Technical Recovery Team (TRT) for the North-Central California Coast Recovery domain both described historical population structure of the four ESA-listed ESU/DPSs with the recovery domain (Bjorkstedt at al. 2005) and developed viability criteria for the listed units (Spence et al. 2008; updated for steelhead DPSs in Spence et al. 2012). These documents (1) define historically independent and dependent populations, and (2) propose viability criteria both for individual populations (Table 4.1) and diversity strata (Table 4.2), which are groups of populations that likely exhibit genotypic and phenotypic similarity due to exposure to similar environmental conditions or common evolutionary history (Bjorkstedt et al. 2005; revised in Spence et al. 2008). Historically independent populations are those whose dynamics and extinction risk over a 100-year time period were not substantially altered by exchanges with individuals with other populations. Dependent populations, in contrast, likely had extinction dynamics that were dependent on interactions with neighboring (primarily independent) populations.

The viability criteria developed by the TRT represent an extension of an approach developed by Allendorf et al. (1997). The population criteria (Table 4.1) are intended to address all four viable salmonid population attributes described McElhany et al. (2000), including abundance, productivity, spatial structure, and diversity (Spence et al. 2008). In general, the spawner density criterion represents the most conservative of the criteria, and although based on spawner abundance, it is intended to, if low-risk targets are met, ensure that spatial structure and diversity important to long-term population viability is maintained (see pages 33–43 in Spence et al. 2008). The diversity stratum criteria (Table 4.2) are intended to (1) ensure sufficient genetic and phenotypic diversity within and ESU or DPS to maintain its evolutionary potential in the face of changing environmental conditions, (2) maintain sufficient connectivity among populations within and ESU or DPS to maintain long-term demographic and evolutionary processes, and (3) to buffer the ESU or DPS against catastrophic loss of populations by ensuring redundancy (see pages 53–66 in Spence et al. 2008).

Since the TRT developed viability criteria for the NCCC Recovery Domain, NMFS recovery planning teams have completed the federal recovery plan for all four ESU/DPSs in the Recovery Domain (NMFS 2012a; NMFS 2016c). These plans include establishment of population-level and ESU-level recovery criteria for populations and diversity strata within these ESU/DPSs. These recovery criteria generally follow the viability criteria developed by the TRT, but may deviate for certain populations based on additional analysis. The recovery plans further delineate what are termed essential (or focus) populations and supporting (or supplemental) populations. Essential or focus populations are those the recovery team considered essential for recovery; these are primarily (but not exclusively) independent populations. Supporting populations include both independent and dependent populations that serve important roles in recovery, such as providing connectivity among essential populations, but that may have recovery targets that are substantially lower than those of essential populations. For the purpose of this viability assessment, we use the recovery criteria for outlined in the recovery plans as the benchmarks for assessing viability.

Application of recovery and viability criteria requires population-level estimates of adult spawner abundance spanning a minimum of four generations for independent populations (Spence et al. 2008). For most of the salmon and steelhead populations in this recovery domain, estimates meeting these criteria are lacking, though because of implementation of the Coastal Monitoring Plan (CMP; Adams et al. 2011), many time series of adult spawner abundance are now approaching the recommended duration. In other areas, indices of spawner abundance or local population estimates representing only a portion of the population constitute the best available data. If data collection has occurred in a consistent manner, these shorter time series, indices, or partial population estimates are presented herein despite the shortcomings, as they provide the best basis for evaluating current viability. However, the reader is cautioned that short-term trends in abundance or abundance indices can be highly misleading given natural variation in environmental conditions in both the freshwater and marine environments. Most data presented in this report were taken from a dataset compiled by the California Department of Fish and Wildlife (CDFW 2020[5]), which includes time series generated by CMP activities, but a few datasets were acquired from alternative sources. A complete list of data sources (by population) used in the analysis of ESU/DPSs in the North-Central California Coast Recovery Domain can be found in Appendix A.

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[5] For most time series, data considered include estimates through the 2018–2019 spawning season. Estimates from the Mendocino Coast from 2017–2019 are preliminary and subject to change due to refinement of methods; however, the estimates used herein are the best available at the time the draft report was prepared.

4.1 Central California Coast Coho Salmon ESU

ESU Delineation

The Central California Coast (CCC) coho salmon ESU was initially defined as populations from Punta Gorda southward to and including the San Lorenzo River (Weitkamp et al. 1995). In 2012, the delination was extended southward to include Soquel and Aptos creeks (77 FR 19552) based on analysis of historical and recent evidence of occurrence in Soquel Creek, as well as environmental conditions in these two watersheds (Spence et al. 2011). Coho salmon successfully reproduced in Soquel Creek during the 2015 spawning season[6] (Spence 2016). Analysis of recent microsatellite data from 30 sites in 23 watersheds spanning the Southern Oregon/Northern California Coast (SONCC) and CCC ESUs provided consistent and strong support for the current ESU delineation at Punta Gorda (Gilbert-Horvath et al. 2016). The ESU includes coho salmon from two artificial propagation programs including the Don Clausen Fish Hatchery Captive Broodstock Program and the Scott Creek/Kingfisher Flats Captive Broodstock Program (85 FR 81822).

Summary of Previous Assessments

Status reviews by Weitkamp et al. (1995) and Good et al. (2005) both concluded that the CCC coho salmon ESU was in danger of extinction, citing concerns over low abundance and long-term downward trends in abundance throughout the ESU, as well as extirpation or near extirpation of populations across most of the southern two-thirds of the ESU's historical range. Additional risk factors identified included the potential loss of genetic diversity associated with range reductions, loss of one or more brood lineages, and the historical influence of hatchery fish (Good et al. 2005). NMFS initially listed CCC coho salmon ESU as threatened in 1996 (61 FR 56138), but changed the status to endangered in 2005 (70 FR 37160). The two most recent viability assessments (Spence and Williams 2011; Spence 2016) concluded that conditions for populations in the CCC coho salmon ESU had worsened since 2005, noting negative trends for most independent and dependent populations for which longer term monitoring data were available, and the near complete collapse of populations in the Santa Cruz Mountains Diversity Stratum and associated loss of genetic diversity. NMFS concluded that the CCC coho salmon ESU remained endangered (NMFS 2016c).

New Data and Updated Analyses

Abundance and Trends

Information on population status and trends for CCC coho salmon has continued to improve with implementation of the Coastal Monitoring Plan across significant portions of the ESU. Population estimates are based on redd counts from surveys of stream reaches selected according

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[6] The spawning season for California salmonids typically straddles two calendar years; for the sake of brevity, in this section, I adopt the convention of using the calendar year at the end of the spawning season to delineate the spawning year (e.g., 2015 refers to the 2014–2015 spawning season, etc.).

to a Generalized Randomized Tessellation Survey (GRTS) design. Redd counts are then expanded to adult estimates based on spawner:redd ratios estimated at a network of life cycle monitoring (LCM) stations. Although several of the time series of abundance still do not meet the requisite four generations called for by the TRT for application of viability criteria, there are now seven independent populations and seven dependent populations with between 10 and 22 years of data. Unfortunately, monitoring of four independent populations was temporarily discontinued in 2019 due to lack of funding. Below, we review available information for each of the four diversity strata for which recovery criteria have been proposed.

Lost Coast– Navarro Point Stratum. Population-level estimates of adult abundance are available for all four independent populations for periods ranging from 10 to 17 years. Recent population estimates indicate that population sizes have averaged from 9% (Big River, Albion River) to 30% (Noyo River) of the proposed recovery targets (Table 4.3), with all populations having improved since last viability assessment. Two populations (Big River and Albion River) fell to or below the high-risk depensation threshold but both have rebounded somewhat since reaching their low point during the height of the drought in 2014, and all four populations are considered at moderate risk in relation to the effective population size criterion. Recent trends for all four populations have been positive, but significantly so only for the Big River population ($p < 0.02$) (Table 4.3; Figures 4.1a-d; Figures 4.2a-d).

Additionally, population estimates for the South Fork Noyo River (a portion of the Noyo River independent population) based on life cycling monitoring are available for the past 20 years. These estimates have been highly variable, averaging 355 fish (range 19–1,195), and as with the Noyo River population at large, there has been a positive but non-significant trend (slope = 0.034; $p = 0.46$) (Figures 4.3a, 4.4a).

For dependent populations, annual population-level estimates of abundance are available for four populations. Pudding Creek and Caspar Creek appear to be the strongest populations, with average returns of 438 and 105 adults, respectively, over the last 19–20 years (Table 4.4). These numbers are approximately 45% and 24% of recovery targets, respectively. Overall trends for these two populations, as well as for the Little River population, for the period of record are negative but not significant ($p > 0.10$) (Table 4.4; Figure 4.5b-d; Figure 4.6b-d). However, population numbers in these watersheds have rebounded since lows experienced around 2010, and particularly since the 2011–2015 drought in California. In contrast, coho salmon have not been observed in Usal Creek since 2013 (Figure 4.5a, 4.6a).

Several other dependent populations have been surveyed intermittently since 2009. Big Salmon Creek has been surveyed in four of the last 10 years, and coho salmon were observed in two of those years (mean = 27, range 0–88; Table 4.4). Juan Creek was surveyed during three seasons, and small numbers of coho salmon were observed in two of those years (mean = 15; range 0–25; Table 4.4). Cottaneva, Hare, and Wages creeks were both surveyed periodically (4–5 seasons) between 2009 and 2016 as part of the Mendocino Coast sampling program; however, no coho salmon were found in these streams and they have not been monitored since 2016. DeHaven

Creek was surveyed during a single season, and no evidence of coho salmon spawning was found (Table 4.4).

Abundance estimates for the entire Lost Coast Diversity Stratum, which includes sampling across both independent and dependent populations, indicate that stratum-wide abundance averaged 3,470 fish (range 672–7,991) between the 2009 and 2018 (Figure 4.7a). Reduced sampling during the 2019 precluded generating a stratum-wide estimate for this spawning year. The stratum average is roughly 45% of the downlisting spawner target and 22% of the delisting spawner target identified for the stratum in the CCC coho salmon recovery plan (NMFS 2012a). Overall, the trend in the stratum during this time has been positive and significant (slope = 0.22; p = 0.015) (Figure 4.8a).

Navarro Point – Gualala Point Stratum. Two of three independent populations in this stratum, the Navarro and Garcia rivers, now have time series of adult abundance spanning ten years, though basin-wide surveys were not conducted during the 2019 spawning season due to lack of funding. These data sets indicate that estimated adult population sizes in the Navarro and Garcia rivers have averaged 303 and 139 fish, respectively, with both averages increasing since the last viability assessment (Table 4.3; Figure 4.1e-f). Notably, although a basin-wide estimate for the Navarro River was not available for 2019, the LCM station in the North Fork Navarro River, which is based on a complete redd census but represents only a portion of the population, produced an estimate of 798 fish, which exceeds any basin-wide estimate for the prior 10 years. If the 2019 estimate—which constitutes a conservative "minimum" estimate for the basin—is included, the mean abundance for the Navarro since 2009 increases to 348 fish. Despite the modest improvement, both populations are less than 6% of their recovery targets, fell temporarily below the depensation high-risk threshold (but have since increased), and are considered at moderate (Navarro) or high (Garcia) risk based on the effective population size criterion. The 10-year trend for both populations is not significantly different from zero (Table 4.3; Figure 4.2e-f).

For the Garcia and Navarro rivers, there were several years where the estimated population size was fewer than ten fish, including the Garcia River in 2010, 2012, and 2014, and the Navarro River in 2014. The lack of coho salmon in the Navarro River (as well as several other Mendocino Coast streams including the South Fork Albion River, Caspar Creek, and Little River) in 2014 appears to be due to the fact the sand bar at the river mouth did not breach until February, which is typically the end of the normal spawning period for coho salmon in this region (S. Gallagher and S. Thompson, CDFW, personal communication). Remarkably, in all of these cases, population estimates three years after these years of low abundance increased substantially. In the Garcia, population estimates increased from nine in 2010 to 211 in 2013, from zero in 2012 to 163 in 2015, and from three in 2014 to 97 in 2017. Likewise, the estimated population in the Navarro River increased from zero in 2014 to 313 in 2017.

These apparent single-generation rebounds are remarkable given the predominance of the 3-year life history for coho salmon in California. These rapid recoveries have three possible explanations: (1) strays from neighboring basins helped rebuild the brood lineages that were lost or severely depressed; (2) population estimated from these years underestimated the true number of coho salmon that spawned in these watersheds in 2010, 2012, and 2014; and (3) brood lineages were "rescued" by a combination 2-year-old jacks and smolts that resided for two years in freshwater instead of the more-typical single year. Biologists with CDFW have carefully analyzed a variety of data associated with the apparent 2014 recruitment failure in four Mendocino Coast streams and have produced compelling evidence that the third hypothesis likely accounts for the majority of adults returning in 2017 (S. Gallagher and S. Thompson, CDFW, personal communication). Though we lack similar data for the Garcia River, recapture of PIT-tagged fish at outmigrant traps at three Mendocino Coast LCM stations indicates that 2-year-old smolts have occurred every year in each of these watersheds and thus may have contributed to the rebound in non-LCM rivers as well. These observations are important because exchange of individual among brood lineages reduces genetic risks that might occur if the weak cohort were rebuilt from a small population composed of highly related individuals. Specifically, if the 2014 cohorts in the Navarro and Garcia rivers were re-established primarily by smolts from the 2013 recruitment year and jacks from the 2015 recruitment year, then genetic diversity may not have been compromised, as both of these year-classes were above-average in size.

Population estimates for the North Fork Navarro River (a portion of the Navarro River independent population) based on life cycling monitoring are available for the past 7 years. These estimates have averaged 257 fish (range 0–798), and as with the Navarro Population at large, there has been a positive but non-significant trend during this short period (slope = 0.605; $p = 0.15$) (Figures 4.3b, 4.4b).

Repeated monitoring of three dependent populations in this stratum (Brush, Greenwood, and Elk creeks) was initiated in 2009 (Table 4.4). Brush Creek has been surveyed annually, though lack of funding prevented monitoring during the 2019 spawning season; no coho salmon have been recorded in the 10 years the watershed has been surveyed. Greenwood Creek has been surveyed in 4 of the last 10 years, and these surveys have produced estimates of low numbers of coho salmon (2–9) in three of those four years. Elk Creek has been surveyed in three of the last 10 years, but no coho salmon have been observed. Alder Creek and Schooner Gulch were each surveyed during a single season; an estimated 5 coho salmon were reported in Alder Creek, while Schooner Gulch surveys produced no evidence of coho salmon. Thus, recent occurrence of coho salmon has been documented in only two of the five surveyed dependent populations in this diversity stratum.

Abundance estimates for the Navarro Point Diversity Stratum indicate that stratum-wide abundance averaged 428 fish (range 2–843) between the 2009 and 2018 spawning seasons (Figure 4.7b). Reduced sampling during the 2019 precluded generating a stratum-wide estimate for this spawning year. Note that these estimates do not include the Gualala River watershed, which has not been monitored. However, coho salmon are believed to be either extirpated or at very low numbers in this watershed. The stratum average is roughly 5% of the downlisting spawner target and 3% of the delisting spawner target identified for the stratum in the CCC coho salmon recovery plan (NMFS 2012a). Overall, no trend in abundance is evident (slope = 0.04; p = 0.84) (Figure 4.8b).

Coastal Stratum. Population monitoring is ongoing for two of three independent populations in the Coastal Stratum: Russian River and Lagunitas Creek. In the early 2000s, a captive rearing program for coho salmon was initiated at Don Clausen (Warm Springs) Fish Hatchery in the Russian River basin, at which time natural-origin coho salmon were returning annually only to a single tributary (Green Valley Creek), and sporadically to a few other tributaries in the lower watershed. This conservation hatchery program has continued to the present, and now incorporates natural-origin fish from various Russian River tributaries, as well as nearby Olema Creek, into the broodstock (CDFW and USACE 2017). Hatchery spawning activities are guided by a spawning matrix developed by NOAA Fisheries Southwest Fisheries Science Center, which seeks to maximize genetic diversity and minimize genetic risk to the integrated broodstock by mating the least-related individuals. Over the last 5 years, the program has released an average of approximately 143,000 coho salmon annually to 19 tributaries and the mainstem of the Russian River. Fish have been released at various life stages including spring fry (8%), fall juveniles (57%), pre-smolts (5%), and smolts (29%) (B. White, USACE, unpublished data).

Prior to the 2015 spawning season, monitoring in the Russian River basin was spatially limited and focused on monitoring the success of outplanted hatchery-origin coho salmon from the captive rearing program into selected tributaries. Since that time, a more comprehensive program has been developed, which has produced basin-wide estimates of coho salmon redds. Over the five years of surveys, an average of 128 redds have been estimated annually. Methods for expanding redd counts to adult abundance based on LCM stations are not currently considered reliable (M. Obedzinski, California Sea Grant, personal communication). Assuming an average spawner:redd ratio of 2:1, adult numbers are less than 3% of the recovery target for this population (Table 4.3). Notably, redd estimates for the Russian River include redds produced by both hatchery- and natural-origin fish. As recovery criteria are based on returns of natural-origin fish, the population is farther from the recovery target than indicated above.

Redd surveys have been conducted in Lagunitas Creek and most of its tributaries annually since the 1998 spawning season by Marin Water, the National Park Service, and the Salmon Protection and Watershed Network. Methods for expanding redd counts to adult fish numbers have not been developed, so results are reported as the total number of unique redds observed during the season. Over the 22-year period of record, the redd count has averaged 247 (range 26–634) (Table 4.3; Figure 4.1h). Assuming an average spawner:redd ratio of 2:1, this average equates to approximately 20% of the recovery target of 2,600 for this population. The population is considered at moderate risk based on the effective population size criterion. The long-term trend is slightly downward, though not significant (p = 0.216) (Table 4.3; 4.2h). Within the past ten years (2011–2019), the population appears to have increased from a low reached during the 2009 spawning season.

Coho salmon were believed extirpated from the Walker Creek drainage; however, there has been an ongoing effort to reintroduce coho salmon into this watershed since 2004. Excess broodstock adults of Olema Creek and Russian River origin reared at the Don Clausen Fish Hatchery have been released on eight occasions since 2004 in numbers ranging between 47 and 221 (average = 98) individuals (M. Kittel, CDFW, unpublished data). Additionally, an average of approximately 6,200 juvenile hatchery-origin fish from the same program have been released annually since 2008, as well as a small (3,400 fish) release of smolts in 2007. Over the last 13 years, opportunistic spawner surveys have been conducted, and redd counts have ranged from zero to 39 over that period (E. Ettlinger, Marin Water, personal communication). The inconsistent frequency of surveys precludes formal analysis of these data, but they do document that both natural and "facilitated' reproduction is now occurring in the watershed.

Population monitoring has also been conducted by the National Park Service for two dependent populations in the stratum: Redwood Creek and Pine Gulch. For Redwood Creek, average redd count over the last 22 years has been 26 (range 0–90) (Table 4.4; Figures 4.5f, 4.6f). Assuming a spawner:redd ratio of 2:1, this total represents about 19% of the recovery target of 272. In Pine Gulch, very small numbers (range 0–3) of coho salmon were seen intermittently between 2001 and 2012, but they have not been observed in the last seven years (Table 4.4; Figures 4.5e, 4.6e). Additionally, as with Walker Creek, both juvenile (year 2008) and excess broodstock adult coho salmon (years 2008–2019) have been released into the Salmon Creek watershed. Adult broodstock releases have averaged 229 (range 62–319) over 12 years and have included natural-origin and hatchery-origin adults from both Olema Creek and the Russian River. Small numbers of juvenile salmon were collected in tributaries of Salmon Creek following the release of adults in December 2008, 2014, and 2016, and genetic analysis on juveniles collected in 2017 indicate that parents include fish of both hatchery and natural origin (M. Kittel, CDFW, personal communication).

No stratum-wide estimates of abundance are available for the Coastal stratum; however, given the population-level information available, it is clear this stratum is only at a small fraction of its recovery target of 15,300.

Santa Cruz Mountain Stratum. Monitoring of populations in the Santa Cruz Mountain stratum was initiated during the 2012 spawning season and continued through the 2019 season, with one year missing (2016) when funding was not available. Methods for assigning unidentified redds as coho salmon or steelhead are currently considered unreliable; thus, population-level estimates of redd or spawner abundance are not available for most watersheds in the stratum. Nevertheless, it is evident from the spawner survey data that coho salmon continue to be extremely rare throughout the diversity stratum. A conservation hatchery program centered at the Kingfisher Flats Hatchery in the Scott Creek watershed was established in the early 2000s. The program currently operates primarily as a captive broodstock program, with opportunistic inclusion of natural-origin fish from Scott Creek and neighboring watersheds. In recent years, fish from the Russian River program, both Olema Creek and Russian River origin, have also been used as broodstock to help improve genetic diversity. Over the last five years, an average of 21,656 smolts (range 11,346–27,812) have been released into Scott Creek. Additional releases of late-fall parr have been made into Scott Creek (14,656 over two years) and Waddell Creek

(8,954 over two years), and spring parr were released into Gazos Creek (8,203) and San Vicente Creek (4,000) in June 2018.

For the two historically independent populations, the San Lorenzo and Pescadero populations, observations of adult coho salmon have been rare since surveys began in 2012. In the San Lorenzo, small numbers (<3) of either live coho salmon or coho salmon carcasses have been observed in three of seven years surveyed during spawner surveys. In addition, in 2014, a total of 19 returning jack males were collected by seine from the lower San Lorenzo River and brought to the Kingfisher Flat Hatchery in the Scott Creek watershed for use in the captive broodstock program. All of these fish were determined through coded wire tags to be hatchery fish from the Scott Creek program. In Pescadero Creek, three coho salmon carcasses, all of hatchery origin, were recovered during the 2015 season. In some other years, a small number of redds have been classified as coho salmon redds by surveyors; however, there have been no observations of adult coho salmon or carcasses in these years to support those classifications.

The status of dependent populations in this stratum is equally precarious. An LCM station has operated on Scott Creek since 2004. This station has produced estimates of adult coho salmon for this period. However, in many years the number of returning adults marked or recaptured has been too low to produce robust estimates using mark-recapture methods; thus, the available estimates represent a combination of population estimates, adjusted weir counts, or "minimum census" estimates, where the numbers of unique adults captured at the weir, observed on spawning grounds, or recovered as carcasses are tallied (Figure 4.5g, 4.6g). As this watershed is the site of a captive rearing program, the vast majority of fish returning to the watershed are either of conservation hatchery origin or recent descendants of hatchery fish. With these caveats in mind has averaged 61 adults over the 16 years of record (Table 4.4); however, this average is strongly influenced by three years in the time series with returns estimated at between 163 and 329 fish, mostly of hatchery origin, including a high fraction of 2-year-old males. In most years, fewer than 30 adults have returned to the watershed, despite the intensive conservation hatchery effort. Small numbers of coho salmon spawners, including strays from the Scott Creek hatchery program, have been observed in other dependent populations in this stratum, including San Vicente Creek and Waddell Creek, but adult spawners have not been observed in San Gregorio, Gazos, Soquel, or Aptos creeks (Table 4.4). Juvenile coho salmon were detected in Soquel Creek on two occasions in the last 12 years (2008 and 2015), but have not been observed since. Likewise, juvenile coho salmon have been periodically detected in Laguna Creek (Chris Berry, Santa Cruz Water District, personal communication). Nevertheless, it is evident that all dependent populations in this stratum are either extirpated or at critically low levels.

No stratum-level estimates of abundance are available for the Santa Cruz Mountain stratum; however, it is clear that natural production of coho salmon throughout the region is extremely low.

Harvest Impacts[7]

No direct information exists on the harvest of CCC coho salmon. Because coho salmon-directed fisheries and coho salmon retention have been prohibited off the coast of California since 1996, the CCC coho salmon ocean exploitation rate is likely very low and attributable to non-retention impacts in California and Oregon Chinook salmon-directed fisheries, non-retention impacts in Oregon mark-selective coho salmon fisheries, and impacts in Oregon non-mark selective fisheries.

The Rogue/Klamath coho salmon ocean exploitation rate time series provides the best available proxy measure of trends in the CCC coho salmon ocean exploitation rate, given the assumption that CCC coho salmon have a similar or more southerly ocean distribution to Rogue/Klamath coho salmon. The Rogue/Klamath coho salmon exploitation rate has been low and relatively stable since the early 1990s (average of 5.4% for years 1994–2019), which contrasts sharply with the much higher rates estimated for the 1980s and early 1990s (Figure 4.9).

Freshwater fishery impacts on CCC coho salmon are likely small given California's statewide prohibition of coho salmon retention. However, in certain situations where population abundance is critically low, such as in the Santa Cruz Mountain Diversity Stratum, incidental handling and mortality from anglers targeting steelhead is a source of concern. Low-flow closure regulations have been adopted in portions of the CCC coho salmon ESU to better protect both ESA-listed and target species. In 2016, low-flow thresholds in the South Fork Gualala River were established and have been used to trigger closures for streams in Mendocino, Sonoma, and Marin counties. (In prior years, flows in the Russian River were used to trigger low-flow closures in these areas, but these were deemed inadequate to protect these populations.) A low-flow threshold for the Russian River was also adopted in 2016 to regulate closures in the Russian River. In San Mateo and Santa Cruz counties, specific low-flow closure thresholds are not defined, though closures can occur if CDFW determines that fish passage is impeded by low flows. The most recent closure occurred in 2014 (V. Gusman, CDFW, pers. comm.). These closures have likely reduced incidental capture and handling of CCC coho salmon during closure periods; however, the overall effect of these closures is difficult to quantify, as the data needed to evaluate potential temporal shifts in angler effort and encounter rates associated with the closures are not currently available.

In summary, the available information indicates that the level of CCC coho salmon fishery impacts has not changed appreciably since the 2016 salmon and steelhead status review update (NFMS 2016b).

Summary and Conclusions

In summary, the available data for populations within the CCC coho salmon ESU indicate that all independent and dependent populations remain far below recovery targets for abundance and, in some cases, are below high-risk thresholds established by the TRTs. The current viability of

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[7] Michael O'Farrell (NMFS SWFSC Santa Cruz) prepared this section on harvest impacts.

populations is progressively worse moving north to south in the ESU. Recent data from the Lost-Coast-Navarro Point and Navarro Point-Gualala Point diversity strata suggest slight improvement in viability of independent populations since the last viability assessment (Spence 2016), with most populations having rebounded somewhat since low levels reached during California's multi-year drought between 2012 and 2015 (Figures 4.1 and 4.2). This is encouraging considering both the extended drought and the unprecedented warm ocean temperatures and associated marine ecosystem impacts that began in 2014 and have persisted most years since (see Section 2). Smolt-to-adult survival estimates from four LCM stations on the Mendocino Coast indicate that marine survival of coho salmon was extremely low from brood years 2004 to 2008 (i.e., smolt outmigration years 2005–2009), but have since risen to levels more typically seen (Figure 4.10), even in years corresponding to the marine heat wave. It thus appears that near-coast conditions along the northern California coast during the springs of 2014 to 2016 may have been more favorable than occurred more generally in the Northeast Pacific Ocean. For dependent populations in these strata, while the mean abundance of some populations has increased slightly since the previous viability assessment, long-term trends have generally continued downward (Figures 4.5a-d and 4.6) and remain a concern.

Assessment of independent populations in the Coastal and Santa Cruz Mountain Diversity strata remains difficult due to the scarcity of reliable data, though the establishment of a rigorous program in the Russian River basin is a positive development. Though coho salmon numbers remain low in the Russian River population, fish are reproducing naturally in several watersheds that have received plants of fish from the ongoing captive rearing program at Don Clausen Fish Hatchery. The extremely low numbers of coho salmon in the Santa Cruz Mountain Diversity Stratum, the high dependence of population persistence on the ongoing captive rearing program, and loss of genetic diversity in the hatchery broodstock, which has necessitated infusion of out-of-stratum broodstock from Don Clausen Fish Hatchery (Olema Creek and Russian River origin) into the program, remain major concerns. Overall, the available new information since the 2016 viability assessment indicates the extinction risk has not changed appreciably, with slight improvements in the two northern-most diversity strata, but little change in the Coastal Diversity Stratum and perhaps worsening conditions in the Santa Cruz Mountain Stratum. The extinction risk for CCC coho salmon as a whole thus remains high.

4.2 California Coastal Chinook Salmon ESU

ESU Delineation

The initial status review for Chinook salmon (Myers et al. 1998) proposed a single ESU for Chinook salmon populations inhabiting coastal watersheds from Cape Blanco, Oregon, south to but not including San Francisco Bay, and including tributaries of the Klamath River downstream of its confluence with the Trinity River. Subsequent review led to division of the originally proposed ESU into the Southern Oregon and Northern California Coastal (SONCC) ESU, and the California Coastal (CC) ESU, the latter including populations spawning in coastal rivers from Redwood Creek (Humboldt County) south to the Russian River, inclusive (NMFS 1999b).

Prior viability assessments (Williams et al. 2011; Spence 2016) have noted that populations that lie between the lower delineation of the Central Valley Fall-run Chinook salmon ESU (Carquinez Straits) and the southern delineation of California Coastal Chinook salmon ESU (Russian River) are not included in either ESU, even though Chinook salmon had been reported regularly in the Guadalupe and Napa rivers in the San Francisco Bay area, as well as Lagunitas Creek in Marin County. These assessments noted that available genetic evidence indicated fish from the Guadalupe and Napa rivers in San Francisco and San Pablo bays had close affinity with Central Valley Fall-run Chinook salmon (Garza and Pearse 2008), and it was recommended that fish from these two watersheds be included in the Central Valley Fall-run Chinook salmon ESU. Evidence for fish in Lagunitas Creek was equivocal, with 17 samples assigned almost equally between California Coastal Chinook salmon and Central Valley fall-run Chinook salmon. Based on these data, the review team tentatively concluded that Lagunitas Creek Chinook salmon should be considered part of the California Coastal ESU pending additional data (Williams et al. 2011). NMFS subsequently indicated that a delineation change was under consideration (76 FR 50447); however, no action has been taken to date.

Since these reviews, Chinook salmon continue to be observed regularly in the Guadalupe and Napa rivers, and they have also appeared sporadically in other Bay Area watersheds including the Petaluma River, Coyote Creek, Sonoma Creek, Suisun Creek. In Lagunitas Creek, Chinook salmon were not observed for four consecutive years (2010–2013); however, they have been observed in each of the last six years (2013–2019), with an average of 48 adults (range 4–100) being counted during this period. To our knowledge there is no new published genetic data that helps further resolve the question of ESU membership for these populations (C. Garza, NMFS SWFSC, personal communication).

Summary of Previous Assessments

Myers et al. (1998) and Good et al. (2005) concluded that California Coastal Chinook salmon were likely to become endangered. Good et al. (2005) cited continued evidence of low population sizes relative to historical abundance, mixed trends in the few available time series of abundance indices available, low abundance and extirpation of populations in the southern part of the ESU, and the apparent loss of the spring-run life-history type throughout the entire ESU as significant concerns. Williams et al. (2011) concluded that there was no evidence to indicate a substantial change in conditions since the previous review of Good et al. (2005), but noted that the lack of population-level estimates of adults continued to hinder assessments of status. They further noted that although independent populations persisted in the North Coastal and North Mountain Interior diversity strata, there was high uncertainty about the current abundance of these populations. They also cited the apparent extirpation of populations in the North-Central Coastal stratum and the loss of all but one population (Russian River) in the Central Coastal stratum as significant concerns since this gap reduced connectivity among strata across the ESU. The most recent viability assessment (Spence 2016) concluded there was a lack of compelling evidence to suggest that the viability of these populations has improved or deteriorated appreciably since the previous assessment. The assessment reiterated concerns about the high uncertainty in northern populations such as the Eel and Mad rivers, but noted that improved monitoring in the North-Central Coastal and Central Coastal diversity strata indicated that low numbers of Chinook salmon were returning to a number of watersheds from which they were previously believed extirpated.

New Data and Updated Analyses

Abundance and Trends

The availability of data for CC Chinook salmon has improved since the previous viability assessment. Adult Chinook salmon abundance estimates include (1) sonar-based estimates on Redwood Creek and the Mad and Eel rivers, (2) weir counts at Freshwater Creek (one tributary of the Humboldt Bay population), (3) trap counts at Van Arsdale Station (representing a small portion of the upper Eel River population), (4) adult abundance estimates based on spawner surveys for six populations on the Mendocino Coast, and (5) video counts of adult Chinook salmon at Mirabel Dam on the Russian River. Prior viability assessments have included maximum live/dead counts in three index reaches in the Eel River (Sproul and Tomki creeks) and Mad River (Cannon Creek); however, these efforts have been discontinued or replaced with the more rigorous efforts to monitor populations in the Eel and Mad rivers using sonar methods. Summaries of available data are presented by diversity stratum below.

North Coastal Stratum. Population-level estimates of abundance are currently available for three of seven independent populations of Chinook salmon in the North Coastal stratum. Estimates based on sonar counts of Chinook salmon in Redwood Creek are available for eight of 10 years since the 2010 spawning season. Because sonar images do not allow easy discrimination of species, methods have been developed to apportion counts among species based on repeated snorkel surveys conducted near the site of the DIDSON camera. Population estimates have averaged 2,896 (range 1,455–4,541) over the eight years of sampling, showing a slightly positive, but not significant trend ($p = 0.31$) (Table 4.5; Figure 4.11a, 4.12a). The population mean represents 85% of the recovery target of 3,400 spawners. Conventional spawner surveys have also been conducted in Redwood Creek for all but one year between 2011 and 2018, though they occurred within a sample frame designed for coho salmon, and so have not included all potential Chinook salmon spawning habitat or season (CDFW 2020). These data are reported as total redd estimates (within the coho salmon frame) and have averaged 886 (range 740–1,063) over the seven years of monitoring. Notably, the sonar estimates are not highly correlated with redd counts in those years where the two time series overlap. Whether this reflects lack of full representation of Chinook salmon spawning areas covered in the spawner surveys, possible errors in the assignment of sonar images to species, or difficulties associated with conducting spawner surveys in larger rivers (i.e., reduced visibility and longer intervals between surveys) is uncertain (S. Ricker, CDFW, personal communication).

Estimates of Chinook salmon adult abundance based on sonar counts have also been made for the Mad River beginning in the 2014 season. Estimates have averaged just over 7,000 fish (range 2,169–12,667) over the five years of monitoring (Table 4.5; Figure 4.11b), and though the time series is too short for formal trend analysis, numbers have increased during this brief period (Figure 4.12b). The mean estimated abundance exceeds the recovery target of 3,000 for this population. This monitoring effort represents a vast improvement in information on Mad River Chinook salmon, as the Cannon Creek index counts, which have been discontinued, typically ranged from tens to low hundreds of fish over the 35-year period of record.

Spawner surveys have been conducted in the Mattole River watershed since the 2013 spawning season, with results reported as total redd estimates. During this time, redd estimates have averaged 862 (range 331–2,202) (Figures 4.11c, 4.12c). The sample frame has varied among years; thus, formal analysis of trends is not appropriate.

In addition to these population-level estimates, longer time series are available for two partial populations. Weir counts have been made at Freshwater Creek (a portion of the Humboldt Bay population) since the 2001 spawning season. These counts are considered incomplete, as the weir is not 100% effective in catching upstream migrating Chinook salmon as fish may pass over or through (smaller individuals) under certain flow conditions. Counts have averaged 29 fish (range 0–154) over the 19-year period of record (Figure 4.13a), and there has been a negative and significant downward trend ($p = 0.0001$) (Figure 4.14a). This trend was driven by high numbers of returns in the early part of the time series, which likely reflects the legacy of a small hatchery program that was discontinued in the early 2000s. Estimates of Chinook salmon redds have been made four last nine years in the South Fork Eel River. These surveys have taken place in a coho salmon sampling frame and so do not include portions of the mainstem South Fork Eel River downstream of Branscomb, which are too large to safely or effectively sample during most winters. The average estimate has been 768 (range 68–1,829) during this period, with no statistically significant trend ($p = 0.709$) (Table 4.6; Figures 4.13b, 4.14b). A sonar camera has also been operated in the South Fork Eel River since the 2018–2019 spawning season, and estimates indicate that Chinook salmon numbers were in the low thousands in the first two years of operation. However, these counts have assumed all fish observed in November and December are Chinook salmon, when it is known from spawning ground surveys that appreciable numbers coho salmon are also entering the river prior to January; thus, the reported estimates are considered provisional and not presented here. Surveys of index reaches in Sproul Creek, a

South Fork Eel River tributary, which have been reported in prior viability assessments were discontinued and are not reported here.

North Mountain Interior Stratum. The North Mountain Interior stratum contains the upper Eel River Chinook salmon population, as well as the portion of the lower Eel River population that inhabits watersheds of the interior mountains of the Eel River basin, including the Van Duzen River and Larabee Creek basins. A long-running time series (since 1947) of adult counts is available for the Van Arsdale Fish Station. The number of Chinook salmon that reach Van Arsdale Station in a given year is generally believed to be strongly influenced by flow conditions, which are affected both by natural variation in precipitation regime and water releases from Cape Horn and Scott dams. In years of low flow, fish are less likely to ascend as far as Van Arsdale Fish Station and instead spawn in areas downstream. Additionally, early counts are confounded by both variation in trap operation and the influence of hatchery plantings of Chinook salmon. For these reasons, we report here only on data collected since 1997 and include counts only of natural-origin (non-hatchery) fish. Over these 23 years, an average of 680 Chinook salmon (range 26–3,471) have been counted (Table 4.6; Figure 4.13c), and there has been no significant trend in abundance ($p = 0.709$; Figure 4.14c). Over the past 12 years, the mean abundance was higher than the 23-year average (mean = 948), but the trend was negative and marginally significant ($p = 0.084$), as high counts in 2011–2013 were followed by six years of below-average counts from 2014–2019. A sonar-based program for estimating abundance of the Upper Eel River Chinook salmon population was initiated in 2019 and produced an estimate of 3,844 fish, a year in which only 94 fish were counted at Van Arsdale. These new data highlight the fact that the Van Arsdale count represents only a small (and potentially variable) fraction of the total Upper Eel River population. Surveys of index reaches in Tomki Creek, an Upper Eel River tributary, which have been reported in prior viability assessments, are not considered reliable indicators of trend (see Spence 2016) and thus are not reported here.

North-Central Coastal Stratum. The previous viability assessment (Spence 2016) noted that, while earlier assessments had reported the apparent extirpation of Chinook salmon populations in the North-Central Coastal Stratum, implementation of the CMP in this stratum beginning in 2009 indicated that small numbers of Chinook salmon continue to return to these watersheds in most years. This continues to be true. In the Ten Mile River, adult estimates have averaged 92 fish (range 0–638 fish over the 11 years of record, (Table 4.5, Figure 4.11d), with no significant trend ($p > 0.10$). The mean represents 11–22% of the recovery target for this population, which is classified as a “supporting” population in the Federal recovery plan. The Noyo River estimate has averaged 19 (range 0–98) during this time, while Big River has averaged 16 (range 0–60) (Table 4.5, Figure 4.11e, f). These mean values are less than 1% of proposed recovery targets and fall below the depensation thresholds for high risk. Likewise, the generational averages (harmonic) fall below the high-risk threshold for effective population size.

Central Coastal Stratum. Population monitoring has continued for three of four independent populations of Chinook salmon in the Central Coastal Stratum. Monitoring of the Navarro and

Garcia river populations was initiated in 2009 and has shown sporadic occurrence of low numbers of Chinook salmon in these watersheds over the last 10 years. In the Navarro River, small numbers (n = 10) of Chinook salmon were reported in both 2010 and 2011, but they have not been observed since (Table 4.5; Figure 4.11g). In the Garcia River, estimates have averaged 34 (range 0–125) fish, with the highest numbers being reported in the last 3 years of the time series, resulting in a significant positive trend ($p = 0.04$), though the population mean is currently less than 2% of the recovery target (Table 4.5; Figures 4.11h, 4.12h). Both populations are categorized as high risk based on depensation and effective population size criteria.

Monitoring of adult Chinook salmon using video counts at Mirabel Dam on the Russian River has been conducted since 2001. An average of 2,947 (range 1,062–6,730) Chinook salmon have been counted annually over the 18-year period of record (Table 4.5; Figure 4.11i). Notably, counts for three recent spawning years (2015, 2016, 2017) are not directly comparable to other years, as the video cameras were not operational during some or all of the Chinook salmon run those years due to repairs to the dam and counting station; thus, estimates had to be derived by other means. Consequently, while there appears to be a slightly downward trend in these data (Figure 4.12i), the statistical significance of this trend cannot be evaluated. The average count represents about 32% of the viability target for the Russian River and the population is considered low risk based on the effective population size criterion.

Harvest Impacts[8]

Very limited data exists on the harvest of California Coastal Chinook salmon (CC Chinook salmon). For ocean fisheries, the Klamath River fall-run Chinook salmon (KRFC) age-4 ocean harvest rate is used as a fishery management proxy to limit harvest impacts on CC Chinook salmon. The CC Chinook salmon ocean fishery consultation standard is a maximum predicted KRFC age-4 ocean harvest rate of 16%.

The KRFC age-4 ocean harvest rate fell sharply from its average value of 44% over the 1981–1990 period (Figure 4.15). Very low KRFC age-4 ocean harvest rates were observed between 2008 and 2012, partially reflecting the widespread fishery closures in California and Oregon from 2008 to 2010. Since 2013, the KRFC age-4 ocean harvest rate has ranged from 4% to 34%, with annual rates exceeding 16% in five of seven years. The harvest rates were particularly high in 2018 (24%) and 2019 (34%), noting that the 2019 estimate is still preliminary (Pacific Fishery Management Council 2020a). The average KRFC age-4 ocean harvest rate estimated over the years since the last viability assessment update (2015–2019) is 19%. In contrast, the average KRFC age-4 ocean harvest rate estimated for years 2011–2014, as reported in the last viability assessment (Williams et al. 2016), was 13%.

Freshwater fishery impacts on CCC Chinook salmon are likely low because retention of Chinook salmon is prohibited; thus, impacts from freshwater fisheries are limited to incidental handling and mortality from anglers targeting steelhead. Low-flow closure regulations have been adopted

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[8] Michael O'Farrell (NMFS SWFSC Santa Cruz) prepared this section on harvest impacts.

in portions of the CC Chinook salmon ESU to better protect both ESA-listed and target species. In 2016, low-flow thresholds in the South Fork Gualala River were established and have been used to trigger closures for streams in Mendocino, Sonoma, and Marin counties. (In prior years, flows in the Russian River were used to trigger low-flow closures in these areas, but these were deemed inadequate to protect these populations.) A low-flow threshold for the Russian River was also adopted in 2016 to regulate closures in the Russian River. These closures have likely reduced incidental capture and handling of CC Chinook salmon during closure periods; however, the overall effect of these closures is difficult to quantify, as the data needed to evaluate potential temporal shifts in angler effort and encounter rates associated with the closures are not currently available.

In summary, the recent increases in the KRFC age-4 ocean harvest rate suggests that the level of CC Chinook salmon ocean fishery impacts has likely increased since the 2016 salmon and steelhead status review update (NMFS 2016d).

Summary and Conclusions

Data availability and reliability for the CC Chinook salmon ESU has improved since the last viability assessment, particularly in the northern part of the ESU. Relatively new sonar-based monitoring programs in the Mad and Eel rivers, which have replaced index-reach surveys in a limited number of tributaries, indicate that populations in these watersheds are doing better than believed in prior assessments, with the Mad River population currently at levels above recovery targets. Likewise, sonar-based estimates for Redwood Creek suggest that the Redwood Creek population, while somewhat variable, is approaching its recovery target in favorable years. Trends in the longer time series are mixed, with the Freshwater Creek showing a significant decline in abundance and Van Arsdale showing no significant trend in counts over the long (23-year) or short (12-year) time series, despite having below-average counts over the last 6 years. Again, interpretation of the Van Arsdale counts is potentially confounded by the relationship between stream discharge and the proportion of Chinook salmon adults reaching the counting station.

Data from populations in the more southerly diversity strata indicate that most populations (all except the Russian River) have exhibited mixed trends in abundance but remain far from recovery targets. In all Mendocino Coast populations (Ten Mile, Noyo, Big, Navarro, and Garcia rivers), surveys have failed to detect Chinook salmon in 3–10 of the 11 or 12 years of monitoring, suggesting only sporadic occurrence in these watersheds. Thus, concerns remain not only about the small population sizes, but the maintenance of connectivity across the ESU. That said, the TRT noted high uncertainty regarding the historical occurrence of independent populations on the Mendocino Coast from the Ten Mile River south to the Gualala River (Bjorkstedt et al. 2005); thus, the overall implications of low numbers in these populations on ESU viability are likewise somewhat uncertain. Only the Russian River population has consistently numbered in the low thousands of fish in most years, making it the largest population south of the Eel River. In summary, the new information available indicates that recent trends across the ESU have been mixed and that overall extinction risk for the ESU is moderate and has not changed appreciably since the previous viability assessment.

4.3 Northern California Steelhead DPS

DPS Delineation

The DPS comprises the anadromous component of *O. mykiss* inhabiting coastal watershed from Redwood Creek (Humboldt County) southward to but not including the Russian River (71 FR 833). The Mad River steelhead hatchery program, the only active steelhead hatchery program in this geographic region, has been determined to be outside the DPS (71 FR 833), as it originated with broodstock from outside the Mad River basin and was historically managed as a segregated program. However, in 2017 a new Hatchery and Genetic Management Plan (HGMP) was approved, which calls for managing the program as an integrated program that sets targets for regularly incorporating natural-origin steelhead into the hatchery broodstock and reducing the percentage of hatchery-origin fish on natural spawning grounds (CDFW 2017). Protocols for collecting natural-origin broodstock and estimating the percent hatchery origin spawners on natural spawning grounds are currently being refined. As this program matures, inclusion of this integrated program into the DPS definition should be re-evaluated.

The Northern California (NC) steelhead DPS includes both summer-run and winter-run populations in watersheds from the Mattole River northward. In November 2018, NMFS received a petition to separate summer-run steelhead in the NC steelhead DPS into a new DPS and list it as endangered under the ESA; the petition cited new information related to the adaptive genomic basis for ecotypic variation in steelhead and Chinook salmon (Prince et al. 2017). NMFS 12-month finding determined that summer-run steelhead in the NC steelhead DPS do not meet the criteria to be considered a DPS separate from winter-run steelhead (85 FR 6527).

Prior viability assessments have noted that genetic samples from contemporary populations suggest possible changes in DPS delineations for several coastal steelhead DPSs (see Section 1.1 for review). No action has been taken to modify existing DPS delineations, and there is no new information available since the prior viability assessment to suggest a change in delinations for NC steelhead is warranted (see Section 1.1).

Summary of Previous Assessments

Busby et al. (1996) and Good et al. (2005) concluded that the NC steelhead ESU/DPS was not presently in danger of extinction, but was likely to become endangered in the foreseeable future. Concerns raised by both of these biological review teams included low population abundance relative to historical estimates, recent downward trends in most stocks for which data were available, and the low abundance of summer steelhead populations. They also cited continued habitat degradation, the increasing abundance of a nonnative predator (Sacramento pikeminnow, *Ptychocheilus grandis*) in the Eel River, the influence of artificial propagation on certain wild populations, and the lack of data for this DPS as concerns and sources of risk (Busby et al. 1996; Good et al. 2005). The two most recent assessments (Williams et al. 2011; Spence 2016) concluded that there was little evidence to indicate that the viability of the NC steelhead DPS had changed appreciably in either direction since prior viability assessments, citing mixed trends in abundance among populations.

New Data and Updated Analyses

Abundance and Trends

Data availability continues to be patchy across this DPS, with population-level estimates of abundance available for more than half the independent populations in the North Coastal Stratum, the majority of independent populations in the North-Central Coastal and Central Coastal strata, but lacking entirely in the Lower Interior and North Mountain Interior strata. Data are also available for a modest number of dependent populations in the North-Central Coastal and Central Coastal strata, but under the current rotating panel design, some of these dependent populations are not sampled every year. There is no monitoring of dependent populations in the two interior strata. Partial population estimates are also available for two LCM stations on the Mendocino Coast (South Fork Noyo River and North Fork Navarro River), and the long-term times series of steelhead counts at Van Arsdale Station in the upper Eel River (which represents an aggregate of several populations) has continued. Counts of adults from snorkel surveys are available for five populations of summer-run steelhead. Summaries of available data are presented below by diversity stratum.

Northern Coastal Stratum. Implementation of the CMP for winter-run steelhead has continued for four populations in the Northern Coastal Stratum: Redwood Creek, Humboldt Bay, the South Fork Eel River, and Mattole River. These efforts have produced estimates of total redd numbers in each of these waters for the past 6–9 years (Table 4.7). Methods for expanding redd counts to population estimates have not yet been developed, as for the purpose of developing spawner:redd ratios, the lone LCM station in this stratum (Freshwater Creek) has been deemed insufficient (S. Ricker, CDFW, personal communication). Additionally, sampling targets the spawning period and habitat for coho salmon and thus may not encompass the entirety of the spawning period and space for steelhead (CDFW 2020). With these caveats in mind, the average steelhead redd estimate for Redwood Creek has been 202 (range 50–405) over seven years of surveys (Table 4.7; Figures 4.16a and 4.17a). Formal analysis of trend for Redwood Creek steelhead was not performed as the time series does not span two generations (8 years). The average redd estimate for Humboldt Bay over the nine-year period has been 109 (range 0–306), also with a positive but non-significant trend ($p = 0.56$) (Table 4.7; Figures 4.16b and 4.17b). For the South Fork Eel River, redd estimates have averaged 551 (range 5–1,125) over the last nine years, with a negative but non-significant trend ($p = 0.22$) (Table 4.7; Figures 4.16c and 4.17c). Six years of data are available for the Mattole River, with an average steelhead redd estimate of 540 (range 222–917) (Table 4.7; Figures 4.16d and 4.17d). Because surveys do not encompass the entire spawning period in some years and methods have not been developed for expanding redd estimates to adult abundance estimates, the above numbers cannot be directly compared to recovery targets. Nevertheless, unless the redd estimates grossly under-represent total population size, it appears that all four of these populations are well below viability targets (Table 4.7).

Estimates of steelhead abundance in Freshwater Creek (a portion of the Humboldt Bay population) have been generated using mark-recapture methods since 2001. Over this 19-year period, an estimated average of 175 adult steelhead (range 51–432) have returned to Freshwater Creek annually, and the trend has been negative but not significantly so ($p = 0.59$) (Table 4.8; Figures 4.18a, 4.19a).

In the Mad River, a sonar camera has recently been used to inform implementation of the Mad River HGMP. The monitoring program both enumerates upstream migrating adults and seeks to estimate the percent of hatchery-origin steelhead returning to the river. Several methods, including snorkel surveys, foot surveys, angler creel surveys, and hook and line sampling near the sonar site have been used to for species apportionment, as well as to estimate the percentage of returning steelhead that are of hatchery (adipose clipped) origin. In the three years following approval of the HGMP (2017–2019), estimates of returning adult winter-run steelhead have averaged 6,602 (range 5,655–8,224), with hatchery-origin fish making up an average of 44% (range 36%–58%) of these fish. Additionally, estimates of adults returning in late summer/early fall have also been produced and have averaged 1,399 (range 661–2,808), with hatchery fish constituting an average of 18% (range 14%–23%); it is unclear whether these fish should be considered part of the winter- or summer-run population.

A goal of the Mad River HGMP program is to ensure that the percentage of hatchery-origin fish on natural spawning grounds (pHOS) is less than 50%, a target that was exceeded in 2019 for fish identified as winter-run. However, it is recognized that these estimates are subject to uncertainty for two primary reasons. First, adipose clips are not always complete or recognizable as such, which would lead to underestimation of the hatchery fraction. Conversely, estimates of hatchery fraction near the sonar site do not account for the fact that the in-river recreational steelhead fishery upstream allows for retention of only adipose-clipped fish, which would lower the percentage of hatchery-origin fish reaching spawning grounds. Efforts are currently underway to improving both marking efficiency and pHOS estimates. The Mad River program also involves collection of natural origin adults to incorporate into the hatchery broodstock with a goal of having at least 50% (increasing over time to 67%) of spawned fish be of natural origin. To date, natural-origin fish have been collected both at the hatchery and off site. In the three years of implementation, it is estimated that the natural-origin fish used in the broodstock have constituted 0.6% to 1.5% of the total number of winter-run steelhead that have returned to the river.

No estimates of abundance are available for the Maple Creek/Big Lagoon, Little River, Price Creek, or Bear River independent winter-run steelhead populations, which are identified as “essential” populations in the Federal recovery plan (Table 4.7). Nor are there any data available for eleven dependent populations of winter-run steelhead that are identified as “supporting” populations in the recovery plan (Table 4.8).

Information on the abundance of summer-run steelhead populations is collected in three systems in the Northern Coastal Stratum: Redwood Creek, Mad River, and the Mattole River, with the upper portions of the latter two watersheds also considered part of the North Mountain Interior stratum. Dive surveys covering an index reach of approximately 41.6 km of Redwood Creek (roughly 38% of the Redwood Creek mainstem) have been conducted annually since 1981. Mean counts have averaged only nine fish during the period of record (range 0–44) (Table 4.9; Figure 4.20a). The long-term trend in counts is not significantly different from zero (p = 0.37) (Figure 4.21a); however, the short-term (16-year) trend is negative and significant (p = 0.002) (Table 4.9). Summer dive surveys were conducted annually on the Mad River between 1980 and 2005 when the effort was discontinued. The spatial extent of these surveys varied through time; thus, these early data are not appropriate for trend analysis. However, beginning in 2013, snorkel surveys were re-initiated with the goals of implementing consistent protocols and covering the river from Kadle Hole (near Hwy 101) to R.W. Matthews Dam. These surveys are believed to cover roughly 95% of the available over-summering habitat (Patrick Righter, Green Diamond Resource Company, personal communication). Over the recent seven-year period, an average of 220 adult summer steelhead have been counted annually (range 117–336) (Table 4.9; Figure 4.20b). Based on the effective population size criterion, the population within the survey reach is at approximately 26% of the recovery target (Table 4.9).

Dive counts of summer steelhead have also been made annually on the Mattole River mainstem and portions of two tributaries since 1996 by the Mattole Salmon Group. The spatial extent of sampling has varied from approximately 41 to 114 km over the years, but has been fairly consistent since 2005, with 100 km being surveyed each year. Over this 24-year period, an average of 22 adult steelhead (range 7–56) have been observed annually (Table 4.9; Figure 4.20c). Trend was estimated only for the last 15 years of data, where survey extent was consistent, and was not significantly different from zero (p = 0.48). At least 19 km of additional potential over-summering habitat in tributaries is not currently surveyed due to difficulty of access or lack of landowner permission; thus, these counts likely underestimate total population abundance to some degree. Nevertheless, it is likely that this population is currently at less than 5–10% of the recovery target.

Lower Interior Stratum. Seven independent and two dependent populations of winter-run steelhead in the Lower Interior Stratum have been identified as essential or supporting populations in the Federal recovery plan. These populations occupy tributaries that enter the Eel River primarily from the west and south between Jewett Creek and Soda Creek, inclusive. To our knowledge, there are no ongoing monitoring efforts that allow evaluation of the status of any of these populations (Tables 4.7 and 4.8).

North Mountain Interior Stratum. The North Mountain Interior Stratum includes both the upper reaches of Redwood Creek and Mad River (previously discussed), as well tributaries that enter the Eel River from the east from the Van Duzen River to the Middle Fork Eel River, and including the upper mainstem Eel River. No population-level estimates of abundance are available for winter-run populations in the Eel River portion of the stratum. The only available dataset in this region are counts of steelhead from the trap at Van Arsdale Station, which represents a composite of the Bucknell Creek and Soda Creek populations (both considered part of the Lower Interior Stratum), as well as a small portion of the historical range of the Upper

Mainstem Eel River population. Counting at this facility began in the 1930s; however, inconsistent operation and the planting of large numbers of hatchery fish confound interpretation of early data. Since 1981, counts of natural- and hatchery-origin fish have been tallied separately; thus, we analyze only data collected since 1981 for this assessment. Over 37 years of reliable records, an average of 361 (range 19–1,966) natural-origin winter steelhead have been counted at the station each year (Table 4.8; Figure 4.18b), and there has been no significant trend in abundance (p = 0.86) (Figure 4,18b). Data from the past 16 years indicate a slightly lower average (325) but no significant trend in abundance (p = 0.66) (Table 4.8). Without knowing which of the three populations these fish represent, it is difficult to evaluate these numbers against viability criteria for these populations; however, it is clear that the Upper Mainstem Eel River population far from its recovery target of 6,400 fish, which is not surprising since the majority of historical habitat lies upstream of the impassable Scott Dam.

For summer steelhead in this stratum, dive counts are available for two populations: the Middle Fork Eel River and the Van Duzen River. The Middle Fork Eel River counts date back to 1966. The long-term average abundance for the period is 753 adults (range 198–1,601) with no significant trend either over the period of record (p = 0.12) (Table 4.9; Figures 4.20d and 4.21d). The recent (16-year) average has been slightly lower at 623 with a slight negative, marginally significant trend during that time (p = 0.06) (Table 4.9). Overall, based on the effective population size criterion, the population is currently at about 80% of the recovery target for this population, and it remains the most abundant summer-run population in the DPS. Summer dive surveys have now been conducted on the Van Duzen River for the last nine seasons. These surveys cover the reach between Little Larabee Creek and Eaton Roughs (generally considered the upper extent of anadromy on the mainstem Van Duzen River), which is thought to encompass the majority of available holding pools in the river (S. Thompson, CDFW, personal communication). Over the past nine years, an average of 121 (range 54–255) steelhead has been counted each year (Table 4.9; Figure 4.20e), and the population has exhibited a slightly negative, but non-significant trend (p = 0.23). The population is currently at about 18% of the recovery target.

North-Central Coastal Stratum. Population estimates are available for all nine independent populations in the stratum, though three populations are not surveyed every year. For the three largest watersheds, adult steelhead estimates have been in the hundreds of fish. In the Ten Mile River, estimates of steelhead have averaged 416 fish (range 0–869) over ten years of record, with the trend being positive but not significant (p = 0.11) (Table 4.7; Figures 4.16f and 4.17f). Estimates of steelhead adults in the Noyo River over the last 18 years have averaged 387 fish (range 79–763), and have shown a slightly positive but non-significant trend (p = 0.14) (Table 4.7; Figures 4.16g and 4.17g). Estimates for Big River have averaged 541 (range 52–1,820) over the past 10 years, with a positive but non-significant trend (p = 0.16) (Table 4.7; Figures 4.16h and 4.17h). All three of these populations are at 11–12% of their recovery targets and are considered at moderate risk in relation to the effective population size criterion.

Partial population estimates have been produced for the South Fork Noyo River (a portion of the Noyo River population) for 20 years as part of a life cycle monitoring program. Estimation methods have varied among the years and have included fish/red expansions, AUC methods, and mark-recapture methods (CDFW 2020). Steelhead abundance in the South Fork has averaged 75 fish (range 19–153), or about 19% of the basin-wide estimate, and there has been no significant trend in abundance (p = 0.60) (Table 4.8; Figures 4.18c and 4.19c).

Annual abundance estimates are available for three independent populations in smaller watersheds. One of these, Usal Creek, is identified as "essential" in the Federal recovery plan, while the other two (Pudding Creek and Albion River) are considered "supporting" populations. For Usal Creek, estimates over 9 years of record indicate an average of 88 fish (range 5–297), with a positive marginally significant trend (p = 0.08) (Table 4.7; Figures 4.16e and 4.17e). This population is at 8% of its recovery target. Pudding Creek has averaged 89 fish (range 7–265) over 18 years with a negative and marginally significant trend (p = 0.06) (Table 4.7; Figures 4.22a and 4.23a). This supporting population is currently at 31–63% of its recovery target. The Albion River population has averaged 50 fish (range 0–182) over 10 years of record, with no trend evident (p = 0.87), and is currently at 9–17% of its recovery target (Table 4.7; Figures 4.22b and 4.23b).

Three smaller independent populations have been surveyed irregularly since 2009, including Cottaneva, Wages, and Big Salmon creeks. Average estimates for all three of these populations have been in the 60s over the 4–5 years surveyed. Cottaneva Creek, designated a supporting population in the recovery plan, has averaged 64 fish (range 0–187) and is at 25–50% of its recovery target, though it was last surveyed during the 2016 spawning season. Wages Creek, an essential population in the recovery plan, has averaged 63 fish (range 7–184) and is at 9% of its recovery target, though again, it was last sampled in 2014. Big Salmon Creek has averaged 68 fish (range 0–233), though the most recent surveys in 2018 yielded no fish. This population is not considered as either an essential or supporting population in the Federal recovery plan.

Annual data are also available for two dependent populations in this stratum: Caspar Creek and Little River. Caspar Creek, identified as essential in the recovery plan, has averaged 46 fish (range 6–145) over the past 18 years, with a negative and marginally significant trend (p = 0.07) (Table 4.8; Figures 4.24a and 4.24b). The population is at 9% of its recovery target. Over this same period, the Little River population has averaged 16 fish (range 2–34) and has likewise shown a marginally significant trend (p = 0.06) (Table 4.8; Figures 4.24b and 4.25b). This population is not identified as either essential or supporting in the recovery plan. Data for Hare Creek, which has been presented in prior viability assessments, is not presented here, as monitoring was discontinued after 2014.

Two other dependent populations in the stratum have been surveyed on an infrequent basis as part of CMP implementation. Juan Creek has been surveyed during three seasons since 2012 and estimates have averaged 27 fish (range 10–39) (Table 4.8). DeHaven Creek was surveyed in a single season (2015), but no steelhead were observed (Table 4.8). Neither of these populations are listed as essential or supporting in the Federal recovery plan.

Stratum-wide estimates have been produced for the North-Central Coastal stratum from 2009 to 2018. On average, an estimated 1,829 adult steelhead (range 274–2,883) have returned to this stratum, with a positive but non-significant trend ($p = 0.27$) (Figures 4.26a and 4.27a). The mean represents about 13% of the stratum target identified in the recovery plan.

Central Coastal Stratum. Annual population estimates are available for three independent populations in this stratum, including the Navarro and Garcia rivers and Brush Creek. The estimated return of steelhead adults to the Navarro River has averaged 399 (range 102–883) over 10 years, with a positive but non-significant trend (Table 4.7) (Figure 4.17i and 4.18i). Early in the time series, the population fell below the high-risk depensation threshold, but has since rebounded. Still the population, which is identified as essential in the recovery plan, remains at only 5% of the viability target. The Garcia River population has averaged 323 adults (range 65–492), and has also exhibited a positive but non-significant trend ($p = 0.79$) (Table 4.7; Figures 4.17j and 4.18j). This population, also an essential population, is currently at about 10% of its recovery target. Both the Navarro and Garcia River populations are considered at moderate risk with respect to the effective population size criterion. For Brush Creek, considered a supporting population, the average return of adult steelhead over 10 years has been just 9 fish (range 0–41) and has likewise exhibited a negative but non-significant trend ($p = 0.71$) (Table 4.7; Figures 4.22c and 4.23c). No current information on steelhead abundance is available for the Gualala River population; however, between 2002 and 2010, estimates were produced for the Wheatfield Fork of the Gualala using direct observation of adults in holding pools. These estimates averaged 1,735 adults annually (range 296–5,843). Although not current, these estimates suggest that this population is among the most abundant within the stratum, if not the entire DPS.

Partial population estimates for the North Fork Navarro River (a portion of the Navarro River population) are available for seven years from a LCM station in this subwatershed. These estimates have averaged 333 fish (range 183–736), or about 66% of basin-wide estimate during the same period (Table 4.8; Figure 4.18d), and there has been no apparent increase or decrease in numbers (Figure 4.19d), though trend was not formally assessed due to the short length of the time series (< 2 generations).

Elk Creek was surveyed during three seasons in the period between 2011 and 2017, producing an average estimate of 21 adult steelhead (range 0–59) (Table 4.7). The estimate represents 5–10% of the recovery target for this supporting independent population.

Three dependent populations have also been monitored in this stratum as part of CMP implementation. Greenwood Creek has been surveyed during three seasons since 2009, producing an average estimate of 15 adults (range 0–36) (Table 4.8). Alder Creek was surveyed during a single season, yielding an estimate of 11 fish. Neither of these populations are listed as essential or supporting in the recovery plan. Schooner Gulch was likely surveyed only in a single season; however, no steelhead were recorded. This population is listed as supporting in the recovery plan.

Stratum-wide estimates (excluding the Gualala River population) have been produced for the Central Coastal stratum for 2009 to 2018. On average, an estimated 810 adult steelhead (range 202–1,703) have returned to this stratum, with a positive but non-significant trend ($p = 0.18$) (Figures 4.26b and 4.27b). Without data from the Gualala River, these numbers cannot be meaningfully compared with stratum-level recovery plan targets.

Harvest Impacts[9]

Ocean harvest of steelhead is rare, and is likely an insignificant source of mortality for NC steelhead.

Potential impacts of in-river recreational fishing are more difficult to assess. Retention of natural-origin steelhead has been prohibited in all California coastal rivers south of the Smith River since 1998. Thus, any fishing impacts to CCC steelhead are limited to illegal poaching or incidental mortality of natural-origin steelhead caused by anglers targeting hatchery-origin steelhead or other co-occurring species. Since the early 1990s, anglers fishing for steelhead in anadromous portions of California waters have been required to purchase a steelhead report card. Information on the dates and locations of fishing, as well as the number of adult steelhead kept, the number of adult steelhead released, the origin of the fish caught (hatchery or wild) and the number of hours fished must be reported (Jackson 2007, CDFW 2016). Although anglers are required to report this information, average compliance rates are low, approximately 30 percent (CDFW 2016). Poor reporting of report card data and other data deficiencies preclude a rigorous assessment of harvest impacts.

Fishing regulations related to area closures appear to have changed little in California's North Coast district since the last viability assessment. Fishing closures owing to low-flow conditions occur in portions of the CCC steelhead DPS. Since 1979, low-flow closure regulations have been in place for Redwood Creek, Van Duzen River, Mattole River, Eel River, and Mad River. Since 2016, flows in the Navarro River have been used to trigger closures for this and other Mendocino County streams. These recently adopted closures in Mendocino County have likely reduced harvest impacts during the periods of closure by limiting angler effort and encounter rates at times when fish are potentially vulnerable. However, the overall effect of these closures is difficult to quantify, as the data needed to evaluate potential temporal shifts in angler effort and encounter rates associated with the closures are not currently available.

In summary, there is little direct evidence to indicate that the level of harvest impacts to CCC steelhead has changed appreciably since the previous viability assessment. Low-flow closures are likely influencing both angler effort and encounter rates in in-river recreational fisheries, but further research is needed to assess how these closures are affecting overall recreational harvest impacts.

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[9] Michael O'Farrell (NMFS SWFSC Santa Cruz) prepared this section on harvest impacts.

Summary and Conclusions

Overall, the availability of information on steelhead populations in the NC steelhead DPS has improved considerably in the past 5 years with continued implementation of the CMP across a significant portion of the DPS. However, significant gaps in information still remain, particularly in the Lower Interior and North Mountain Interior diversity strata, where there is very little information from which to assess viability. Overall, the available data for winter-run populations—predominantly in the North Coastal, North-Central Coastal, and Central Coastal strata—indicate that all essential populations remain well below recovery targets. In the North Coastal stratum, direct comparison with recovery targets is confounded by (1) the fact that survey efforts target coho salmon and thus do not encompass the entire spawning season or space for steelhead, and (2) the lack of methods for converting redd estimates to population abundance. Nevertheless, even if population sizes are several times greater than current redd estimates suggest, these populations would likely still be less than 10–15% of recovery targets. For the North-Central Coastal stratum, most essential populations are currently at 8–12% of their recovery targets, and in the Central Coastal stratum, they are at 3–10% of these targets (Table 4.7). An exception may be the Mad River population, where recent population abundances have averaged about 73% of the combined recovery targets for the lower and upper Mad River subpopulations. However, interpretation of new monitoring data is confounded by the fact that population estimates include fish of hatchery origin that are not currently considered part of the DPS. The high fraction of returning fish that are of hatchery origin (currently estimated to be 44%) is a concern; however, implementation of the new HGMP for this facility is expected to reduce potential adverse genetic impacts through time. Trends for essential independent populations have been mixed, with most either stable or showing slight (non-significant) increases in abundance since the last viability assessment (Figure 4.16). Most supporting independent populations and both essential and supporting dependent populations currently number in the tens of fish (Figures 4.22 and 4.24), and have shown downward (but non-significant) trends (Figures 4.23 and 4.25). Time series of partial or aggregate populations show essentially no trends in abundance (Figure 4.18 and 4.19). On the positive side, stratum-wide estimates for winter-run steelhead have trended slightly upwards over the last 10 years (Figures 4.26 and 4.27). Overall, the data suggest that the status of winter-run steelhead populations has not changed appreciably since the previous viability assessment (Spence 2016).

Summer-run steelhead populations remain a significant concern. The abundance of the Middle Fork Eel River population has remained remarkably stable for nearly five decades and is closer to its recovery target (~80%) than any other population in the DPS (Table 4.9). Populations in the Mad and Van Duzen rivers have averaged in the low hundreds of fish and are at 18% and 26% of their recovery targets, respectively. However, both the Redwood Creek and Mattole River populations appear very small, and little is known about other populations including the various tributaries of the Eel River. We know of no evidence of recent occurrence of summer-run steelhead in either Larabee Creek or the South Fork Eel River. There are recent reports of summer-run steelhead being observed in the North Fork Eel River, apparently the first observations in many years (Z. Ruddy, BLM, personal communication). With recent climatic trends indicating warming of air temperatures and high variability in stream discharge (see

Section 2.2), summer-run steelhead populations are likely to experience increased risk in the near future due to both decreased thermal suitability of over-summering habitats for adults and reduced access to historical spawning areas in years of low precipitation.

In summary, the new information for NC steelhead available since the previous viability assessment (Spence 2016) indicates that overall extinction risk is moderate and has not changed appreciably since the prior assessment.

4.4 Central California Coast Steelhead DPS

DPS Delineation

The Central California Coast steelhead DPS comprises the anadromous component of *O.mykiss* populations from the Russian River south to and including Aptos Creek, including all tributaries to San Francisco and San Pablo bays eastward to Chipps Island at the confluence of the Sacramento and San Joaquin rivers (85 FR 81822). Two hatchery programs, the Don Clausen Fish Hatchery Program, which includes two facilities in the Russian River basin (Don Clausen Fish Hatchery and Coyote Valley Hatchery), and the Kingfisher Flat Hatchery Program (Monterey Salmon and Trout Project) in the Santa Cruz Mountains, are considered part of the DPS. The Kingfisher Flat Hatchery program has not released steelhead since the spring of 2014 and the last return of hatchery-origin adults to Scott Creek occurred during 2018. The Monterey Bay Salmon and Trout project is currently preparing a Hatchery Genetic Management Plan for the purpose of resuming a steelhead program in the San Lorenzo River.

Prior viability assessments have noted that genetic samples from contemporary populations suggest possible changes in DPS delineations for several coastal steelhead DPSs (see Section 1.1 for review). No action has been taken to modify existing DPS delineations, and there is no new information available since the prior viability assessment to suggest a change in delineations for CCC steelhead is warranted (see Section 1.1).

Summary of Previous Assessments

The original BRT concluded that the Central California Coast (CCC) steelhead DPS was in danger of extinction (Busby et al. 1996), citing extreme risk for populations in Santa Cruz County and tributaries to San Francisco and San Pablo bays, as well as apparent substantial declines in numbers and threats to genetic integrity (caused by hatchery activities) in the Russian River. A subsequent status review (NMFS 1997) concluded that the ESU was not presently in danger of extinction but was likely to become so in the foreseeable future; the change in opinion of the BRT was prompted by new data showing that steelhead remained present in most watersheds in the Santa Cruz Mountains and were more abundant than previously thought. The DPS was listed as threatened in 2007 (62 FR 43937). Good et al. (2005) similarly concluded that the DPS was not presently in danger of extinction, but was likely to become so in the foreseeable future, and the DPS's status as threatened was reaffirmed (71 FR 833). In the most recent assessments, Williams et al. (2011) and Spence (2016) concluded that there was little information available to indicate a change in the viability of this DPS, though again acknowledged the high uncertainty surrounding most populations, particularly those entering San Francisco and San Pablo bays.

New Data and Updated Analyses

Abundance and Trends

Information on the abundance of adult steelhead in the CCC steelhead DPS remains relatively scarce. Population-level estimates of adult abundance are entirely lacking for the 25 independent populations in the North Coastal, Interior, Coastal San Francisco Bay, and Interior San Francisco Bay diversity strata identified as essential or supporting in the Federal recovery plan. A recently initiated program in the Russian River basin provides aggregate estimates of abundance for multiple independent and dependent populations within the basin, which has improved our understanding of basin-wide steelhead abundance, but the sample frame has changed through time; thus, the ability to analyze trends is limited. A few survey efforts that are targeting coho salmon do collect data on steelhead as well, but generally surveys do not encompass the entire spawning space of season for steelhead. Implementation of the CMP in the Santa Cruz Mountain stratum has been intermittent, and difficulties in assigning redds to species (steelhead versus coho salmon) confound interpretation of these data. The LCM station in Scott Creek, which has operated since 2002, provides the only data for examining longer-term trends in abundance. The lack of data continues to make it very difficult to assess the status, trends, and viability of populations in the DPS. The limited available information is summarized below by diversity stratum.

North Coastal and Interior Strata. The North Coastal stratum includes tributaries in the lower Russian River watershed downstream of the confluence of Mark West Creek, as well as coastal watersheds of Sonoma and Marin counties. The Interior Stratum includes the Russian River and its tributaries upstream of the Mark West confluence. We combine discussion of these two strata because a recently initiated adult monitoring program in the Russian River spans both strata. The program was initiated during the 2015 spawning season (targeting coho salmon), but was expanded to produce basin-wide estimates of steelhead for the past three seasons (2018–2020). The estimates produced are estimates of total redd numbers within the sample frame. In the last three years, when the sample frame was expanded, estimates have averaged 1,503 redds (range 873–2,031). Estimates for individual populations within this aggregate have not been produced. Notably, the Russian River is the site of a hatchery steelhead program that involves two hatchery facilities (Don Clausen Fish Hatchery, sometimes referred to as Warm Springs Hatchery, and Coyote Valley). These hatcheries were established as mitigation for loss of habitat upstream of the Warm Springs and Coyote Valley dams. The mitigation agreement requires release of 500,000 yearling steelhead from these two facilities (300,000 from Warm Springs and 200,000 from Coyote Valley), which are distributed throughout the upper and lower watershed. In the last 5 years, an average of 6,951 steelhead have returned to these facilities annually (E. Larson, CDFW, unpublished data), the vast majority of these (>99%) being marked fish of hatchery origin. Thus, it is evident that hatchery-origin fish outnumber natural-origin fish by several fold. In this same time period, data from spawning ground surveys indicate that 51% of all fish observed in natural spawning areas were of hatchery origin (M. Obedzinski, California Sea Grant, unpublished data). Additionally, the last two years, adult hatchery broodstock fish have been released into the system, and these have been reported during subsequent spawning ground surveys. Thus, potential introgression between hatchery and wild fish is a significant concern. A Hatchery and Genetic Management Plan for the Russian River Steelhead Program is currently in development that seeks to both incorporate natural-origin fish into the hatchery broodstock and reduce the percentage of hatchery-origin fish on natural spawning grounds.

Spawner surveys have also been conducted in the Lagunitas Creek watershed since 2002; however, these target coho salmon and do not encompass the full spawning period of steelhead, and as the temporal extent of sampling has varied, the redd counts are not considered reliable indicators of trends. With those caveats in mind, redd counts for this period, which perhaps serve as a minimum estimate for spawners, have averaged approximately 147 (range 23–321) (Table 4.10). Given the temporally limited nature of these surveys and lack of developed methods for expanding redd counts to adult estimates, it is difficult to compare these values with recovery targets. Nevertheless, these redd counts suggest that the population is well below its recovery target of 1,900 adults (Table 4.10).

Redd surveys for two dependent populations in this stratum, Redwood Creek and Pine Gulch, are regularly conducted by the National Park Service. These surveys target coho salmon and have varied in their spatial and temporal extent from year to year; thus, the resulting redd counts are not appropriate for assessing status or trends. With those caveats in mind, redd counts have averaged 9 (range 0–47) over 22 years of record for Redwood Creek and 11 (range 0–33) over 18 years for Pine Gulch (Table 4.11). Given the constraints of these data, it is unclear how close these supporting populations are to recovery targets.

Coastal San Francisco Bay Stratum. Population-level estimates of adult abundance are not available for any of the six independent or two dependent populations within this stratum identified as essential or supporting in the Federal recovery plan. However, since the previous viability assessment, several new monitoring programs have been initiated. In the Guadalupe River, juvenile surveys have been conducted since 2015, which have documented the occurrence of juvenile O. mykiss in several tributaries. Additionally, in 2018 and 2019, a VAKI camera was operated at the Alamitos fish ladder to detect migrating salmonids. Several large O mykiss (>500 mm) were observed in the 2018, indicating the presence of steelhead; however, none were observed in 2019. Spawner surveys have been conducted in San Mateo Creek downstream of Lower Crystal Springs Reservoir each year since 2015, with the exception of 2017 when stream flows were too high to conduct surveys. Redd counts have ranged from 6 to 31; however, no live fish or carcasses have been observed that would confirm the presence of anadromous O. mykiss. Outmigrant traps operated during the same periods have documented fish classified as “smolts” and “partial smolts”; thus, it appears that the potential for expression of anadromy persists. Juvenile surveys have also been conducted in Stevens Creek since 2013. These surveys have documented the continued presence of juvenile O. mykiss in the creek, though again there is no recent direct evidence of anadromous adults returning to this watershed. Collectively, while useful for confirming the continued presence of O. mykiss in these watersheds and supporting management actions in these watersheds, these new surveys do not provide the level of information needed to evaluate whether there has been any change in viability across the stratum.

Interior San Francisco Bay Stratum. Population-level estimates of adult abundance are also lacking for all nine independent populations and three dependent populations of steelhead in the Interior San Francisco Bay Stratum identified as essential or supporting in the Federal recovery plan. Spawner surveys primarily targeting Chinook salmon (but occasionally steelhead) have been conducted in recent years in selected portions of the Napa River watershed and its tributaries. These efforts have produced occasional observations of steelhead redds, live fish, or carcasses. Additionally, a rotary screw trap operated near the upper limit of tidal influence has resulted in capture of 31 to 242 smolts annually since 2009. Smolt trap efficiency has averaged about 12% during this period, suggesting that total smolt production has generally ranged from a few hundred to perhaps 2,000 fish. These efforts confirm the continued occurrence of steelhead in this watershed; however, the data are insufficient to determine if the population has increased or decreased since the previous viability assessment. Likewise, limited spawner surveys in selected tributaries of the Petaluma River produced observations of small numbers of live steelhead, carcasses, and redds in Adobe and Lichau creeks during 2015, 2016, and 2019. Again, these limited surveys confirm steelhead presence in the watershed, but do not allow conclusions to be drawn about current viability.

Since the last 5-year viability assessment, new monitoring efforts have been initiated in several additional watersheds. In Pinole Creek, new fish passage structures were installed in fall of 2016 at Interstate 80 approximately 1.5 mile upstream of the creek's mouth to retore access to the upper watershed by steelhead and other anadromous fishes. Spawning ground surveys were conducted between 2017 and 2020 to evaluate the success of the project. Redd counts ranged from 7 to 24 during those years. Although no adult steelhead or carcasses were observed during the surveys and the majority of redds were small in size and thus presumed to have been made by resident O. mykiss, from 1 to 5 redds were classified each year as likely having been produced by anadromous fish based on redd characteristics. Summer snorkel surveys conducted in Suisun Creek documented occurrence of O. mykiss in 2017; however, when revisited in 2018, most of the sites were dry or devoid of fish. In the Alameda Creek, resident O. mykiss continue to persist in the upper watershed. However, a 12-ft concrete drop structure known as the BART weir located approximately 10.5 mi upstream of the creek mouth has blocked passage by anadromous fish since its construction in the 1970s. Adult steelhead continue to be observed periodically at the base of the weir, and fish have occasionally been moved upstream of the barrier. New fish ladders at the BART weir are expected to open in early 2022, which will allow access to more than 20 miles of spawning and rearing habitat upstream. Finally, in Coyote Creek, surveys have been conducted at sites up to 5.5 miles downstream of Anderson Dam (impassable to upstream-migrating salmonids) in summer or fall each year since 2014. These surveys documented low numbers of young-of-the-year in 2014 and 2019, but none in 2015–2018. A video camera was also installed in Coyote Creek at the Coyote Percolation Dam fish ladder in 2019 and 2020 to monitor adult salmonids, but no steelhead were detected. Collectively, surveys in the Interior San Francisco Bay Stratum have been useful in both confirming the continued presence of O.mykiss and supporting management actions in the Pinole, Suisun, Alameda, and Coyote creek watersheds; however, they are insufficient to evaluate whether there has been a change in viability across the stratum.

Santa Cruz Mountains Stratum. Population-level estimates of abundance for populations in the Santa Cruz Mountain Stratum remain scarce. The Scott Creek LCM station provides the only longer-term (> 10 years) data for this stratum. Over the 16 years of record, an average of 205 steelhead (range 59–547) have returned to this watershed (Table 4.10; Figure 4.28), which is approximately 29% of the recovery target. The population trend has been negative and marginally significant ($p = 0.09$) (Figure 4.29); however, in the last year of the time series (2019), the estimated abundance was higher than any prior year in the series. Without this data point, the downward trend would likely have been significant.

Implementation of the CMP was initiated throughout the stratum during the 2012 season and has continued since then, though a lapse in funding occurred in 2016. Interpretation of data produced by this program is confounded by several factors. First, the monitoring program is designed to target coho salmon; thus, the spatial extent of surveys is limited to the coho salmon sampling frame, and surveys in most years have concluded by the end of March, before the spawning season for steelhead is typically complete. Additionally, methods for assigning species to redds (coho salmon versus steelhead) have continued to evolve through time, making it difficult to identify a consistent metric for evaluating trends through time (i.e., redd counts, expanded redd estimates, or fish:redd expanded adult estimates). Redd classification involves two steps: (1) classification of redds to species by surveyors in the field (i.e., coho salmon, steelhead, or unknown), and (2) assignment of "unknown" redds to species using the model-based approach of Gallagher and Gallagher (2005). Through time, there has been a shift in the proportion of redds classified in the field. In early years of the program, a relatively high percentage of redds were classified to species in the field based on professional judgement of surveyors (i.e., without direct observation of associated fish or carcasses), with only 9–23% of redds being classified as "unknown." In subsequent years, that percentage has risen to as high as 56–61%, as surveyors have been more conservative about assigning species in the field. Moreover, the model used to assign redds to species was developed for the Mendocino Coast region and relies to a significant degree on timing of first observation of the redd (Gallagher and Gallagher 2005). This method is currently considered to be unreliable because timing of stream entry and spawning for both coho salmon and steelhead in Santa Cruz Mountain streams is strongly influenced by hydrologic conditions, which affect when sand-bars breach and streams become accessible to anadromous fish. The net effect of these changes is that to generate basin-wide redd estimates or fish:redd expanded adult estimates would require greater dependence on redd-assignment methods that are not considered reliable. Finally, the estimated spawner:redd ratios produced at the Scott Creek LCM station varies among years, depending on flow conditions and the ability to conduct consistent surveys through the spawning season; thus, they are considered highly uncertain in some years. Although efforts to develop robust and consistent methods for analyzing these data are in progress, at the present time, no consistent metric exists for evaluating trends in Santa Cruz Mountain streams outside of Scott Creek.

With the above caveats in mind, we summarize the available data here. For Pescadero Creek, adult estimates produced for 2012 to 2015 ranged from 132 to 1407 fish. In all but 2015, coho salmon were extremely scarce in the Santa Cruz Mountain, and few redds were classified as coho redds in the Pescadero watershed. Thus, the steelhead estimates are likely minimally affected by bias in redd-assignment methods. In the last 3 years of data reviewed for this assessment, redd counts (both field classified and model assigned) have been reported, with values ranging from 17 to 51. These counts are comparable to prior years, with the exception of 2013 when 182 steelhead redds were counted. For the other larger watershed in this stratum, the San Lorenzo River, adult estimates produced for 2012 to 2015 range from 188–777. In the past three years, redd counts have ranged from 18–33; however, in 2018 when 31 steelhead redds were reported, 14 "unknown" redds were assigned to coho salmon. As no juvenile coho salmon were observed in subsequent snorkel surveys in any of the San Lorenzo River reaches, it is very likely that these redds were produced by steelhead, which would increase the total to 51. These recent redd counts are fairly comparable to the earlier years. Collectively, the evidence indicates both the Pescadero and San Lorenzo populations are well below their recovery targets (Table 4.10).

More limited data are also available for several other smaller independent populations within this stratum. For San Gregorio Creek, population estimates for 2014 and 2015 were 144 and 159, respectively. Comparable estimates are not available for the last three years, but redd counts (field classified and assigned) have been eight or less, compared with 13 and 23 in the prior two years, suggesting a possible decrease in numbers. Waddell Creek has been surveyed in five of the last eight years. Population estimates for 2012 to 2014 ranged from 34 to 89. During the 2017 and 2018 seasons, redd counts were 0 and 1, respectively, though only one reach was surveyed in each of those years. For Soquel Creek, surveys over four seasons have produced only a single redd observation. For Aptos Creek, surveys over three years have produced redd counts ranging from 5–22. A total escapement estimate of 70 was produced for the year (2013) with the highest redd count. Pilarcitos Creek was monitored in two years (2012 and 2013); seven redds were observed in 2012 but none were seen in 2013.

Data are also available for three dependent populations in this stratum. Gazos Creek has been surveyed over seven years. Between 2012 and 2015, population estimates ranged from 5 to 104 fish (based on redd counts ranging from 1–27). Population estimates are not available for the last three seasons, but redd counts have ranged from 2 to 8). San Vicente Creek has likewise been monitored over seven years. Population estimates over the first four years of surveys ranged from 0 to 120 (based on redd counts ranging from 0–76). Population estimates are not available for the last 3 years, but redd counts have varied from 0 to 14. San Pedro Creek was surveyed in the first two years of the program; no redds were observed in 2012 and 12 were counted in 2013. Surveys in this watershed have not been conducted since.

In summary, evaluating changes in status of both independent and dependent populations within the Santa Cruz Mountain diversity stratum remains extremely challenging due primarily to uncertainty associated with methods for assigning redds to species. Scott Creek remains the only population for which robust estimates are available for more than a few years, and while the population appeared to be declining, a sizable return in 2019 indicates that the population is somewhat resilient (Figure 4.28). Adult steelhead populations in the San Lorenzo River and Pescadero Creek appear to typically number in the low hundreds of fish, while other independent populations appear to number in the tens of fish. Two dependent populations (Gazos and San Vicente creeks) likewise appear to number in the tens of adult steelhead in most years, with considerable variation in numbers among years. Though uncertainty remains high for nearly all of these populations, it is clear that they are well below recovery targets.

Harvest Impacts[10]

Ocean harvest of steelhead is rare, and is likely an insignificant source of mortality for CCC steelhead.

Potential impacts of in-river recreational fishing are more difficult to assess. Retention of natural-origin steelhead has been prohibited in all California coastal rivers south of the Smith River since 1998. Thus, any fishing impacts to CCC steelhead are limited to illegal poaching or incidental mortality of natural-origin steelhead caused by anglers targeting hatchery-origin steelhead or other co-occurring species. Since the early 1990s, anglers fishing for steelhead in anadromous portions of California waters have been required to purchase a steelhead report card. Information on the dates and locations of fishing, as well as the number of adult steelhead kept, the number of adult steelhead released, the origin of the fish caught (hatchery or wild) and the number of hours fished must be reported (Jackson 2007, CDFW 2016). Although anglers are required to report this information, average compliance rates are low, approximately 30 percent (CDFW 2016). Poor reporting of report card data and other data deficiencies precludes a rigorous assessment of harvest impacts.

Fishing regulations related to area closures appear to have changed little in California’s South Central and North Central districts since the last viability assessment. Fishing closures owing to low-flow conditions occur in portions of the CCC steelhead DPS. Low flow closures based on river-specific flow thresholds have been in place for the Napa River since prior to 2015. In 2016, low-flow thresholds in the South Fork Gualala River were adopted and have been used to trigger closures for Sonoma and Marin county streams within the CCC steelhead DPS. In prior years, flows in the Russian River were used to trigger low-flow closures in Mendocino, Sonoma, and Marin counties, but these were deemed inadequate to protect these populations. At the same time, a low-flow threshold for the Russian River was adopted to regulate closures in the Russian River proper. In San Mateo and Santa Cruz counties, specific low-flow closure thresholds are not defined, though closures can occur if CDFW determines that fish passage is impeded by low flows. The most recent closure occurred in 2014 (V. Gusman, CDFW, pers. comm.). These closures have likely reduced harvest impacts during the periods of closure by limiting angler effort and encounter rates at times when fish are potentially vulnerable. However, the overall effect of these closures is difficult to quantify, as the data needed to evaluate potential temporal shifts in angler effort and encounter rates associated with the closures are not currently available.

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[10] Michael O’Farrell (NMFS SWFSC Santa Cruz) prepared this section on harvest impacts.

In summary, while no direct information is available on the level of CCC steelhead fishery impacts, it is reasonable to conclude that the level of impact has not appreciably changed in recent years.

Summary and Conclusions

The scarcity of information on steelhead abundance in the CCC steelhead DPS continues to make it difficult to assess whether conditions have changed appreciably since the previous assessment (Spence 2016).

The implementation of the Coastal Monitoring Plan in the Russian River basin has improved our understanding of the overall abundance of steelhead in the watershed, providing basin-wide estimates of abundance of steelhead (combined natural and hatchery-origin) that have ranged from about 800–2,000 over three years, but as population estimates are not produced for individual populations within the basin, direct comparison with recovery targets is not yet possible. Importantly, this monitoring program has provided quantitative evidence that hatchery-origin steelhead constitute roughly 50% of all fish on natural spawning grounds and that these hatchery fish are being observed throughout the basin. Though these hatchery-origin fish are considered part of the CCC steelhead DPS, the high fraction on natural spawning grounds is well above the recommended proportion of hatchery-origin fish on natural spawning grounds (< 30%) for integrated hatchery programs to avoid erosion of population fitness (HSRG 2015). Thus, concerns expressed in prior viability assessments (Williams et al. 2011, Spence 2016) about potential negative genetic consequences of interbreeding between hatchery and wild fish appear well founded.

Monitoring of coastal populations in the North-Coastal Stratum is designed primarily to quantify coho salmon redd abundance, and does not encompass the full temporal and spatial extent of spawning for steelhead. As such, these data have limited utility for assessing trends and provide essentially “minimum” estimates of abundance. Population-level estimates of abundance do not exist for any populations in the Interior and Coastal San Francisco Bay strata, thus, their viability remains highly uncertain. It remains likely that many Interior and Coastal San Francisco Bay populations where historical habitat is now inaccessible due to dams and other passage barriers are likely at high risk of extinction, as noted in prior viability assessments (Spence et al. 2008; Williams et al. 2011, 2016). New information from eight years of CMP implementation in the Santa Cruz Mountains Diversity Stratum has improved our understanding of steelhead in this region. However, sampling effort again focuses primarily on coho salmon and thus does not encompass the full spatial and temporal extent of steelhead spawning. Further, issues with redd assignment hinder the ability to compare data across all years. Resolving these issues should be a tractable problem and must be a priority if these data are to be useful for future status assessments. Data from Scott Creek life cycle monitoring program, which provides the only consistent, long-term time series in the entire DPS, indicates no significant trend in abundance, although without the strong 2018–2019 return, it is likely the trend would have been negative and significant.

In summary, while data availability for this DPS remains generally poor, the new information for CCC steelhead available since the previous viability assessment (Spence 2016) indicates that overall extinction risk is moderate and has not changed appreciably since the prior assessment.

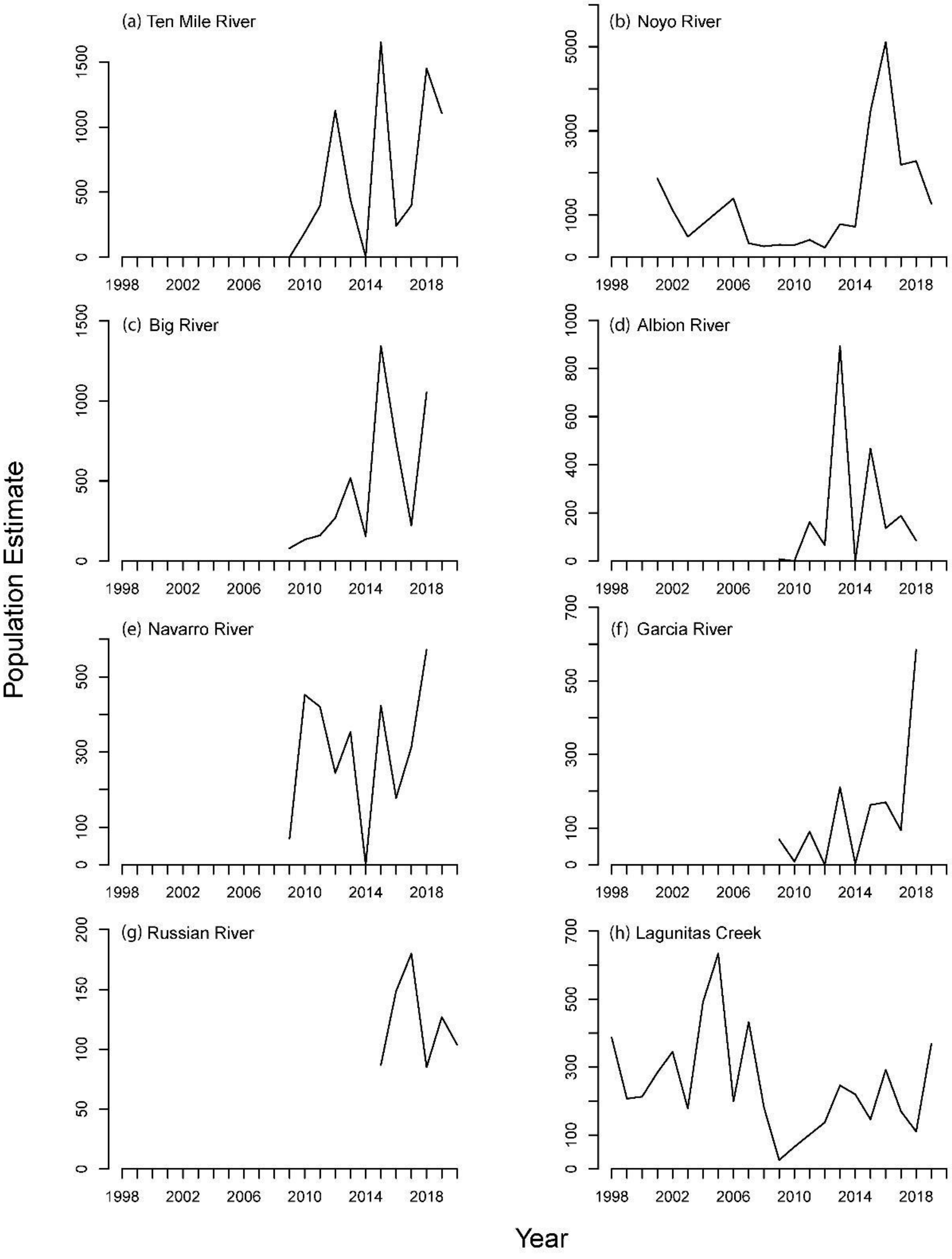

Figure 4.1. Time series of population abundance estimates for independent populations of CCC coho salmon. Values for Lagunitas Creek and Russian River are redd counts and redd estimates for the watershed, respectively. All other estimates are based on fish/redd expansions from life cycle monitoring stations.

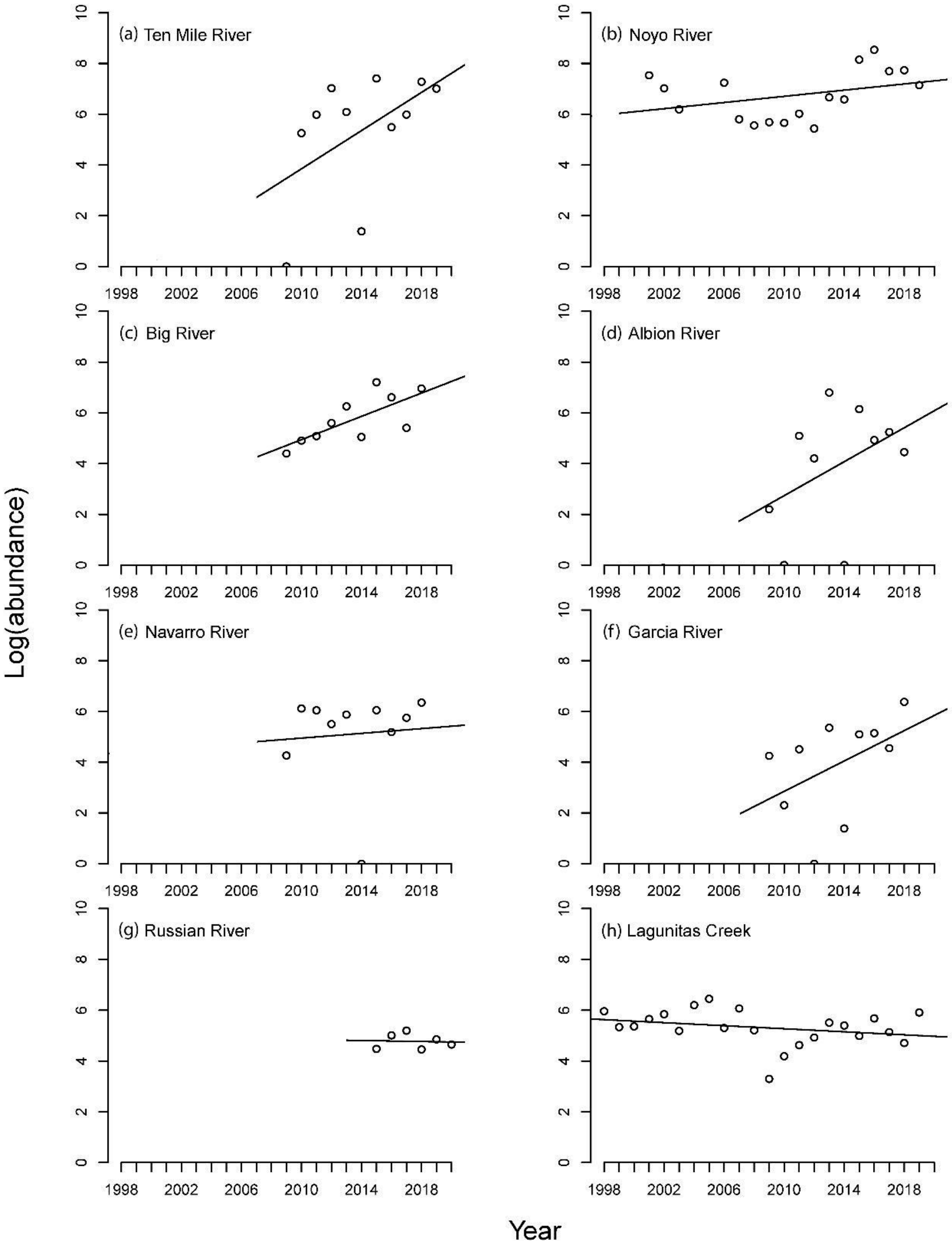

Figure 4.2. Population trends (log abundance) for independent populations of CCC coho salmon. Values for Lagunitas Creek and Russian River are based on redd counts and redd estimates for the watershed, respectively. All other estimates are based on fish/redd expansions from life cycle monitoring stations.

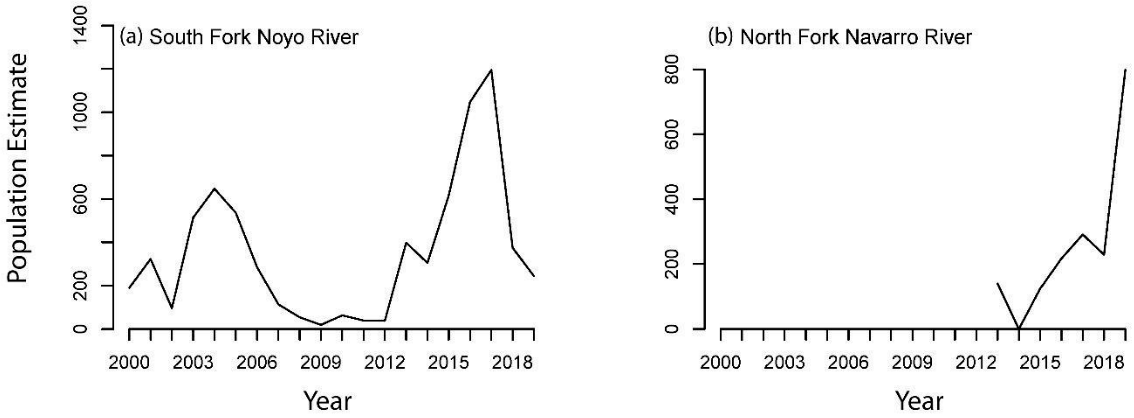

Figure 4.3. Time series of partial population abundance estimates for independent populations of CCC coho salmon. Values for South Fork Noyo River are based on mark-recapture estimates except in 2000 and 2009, which are weir counts. Values for North Fork Noyo River are AUC estimates.

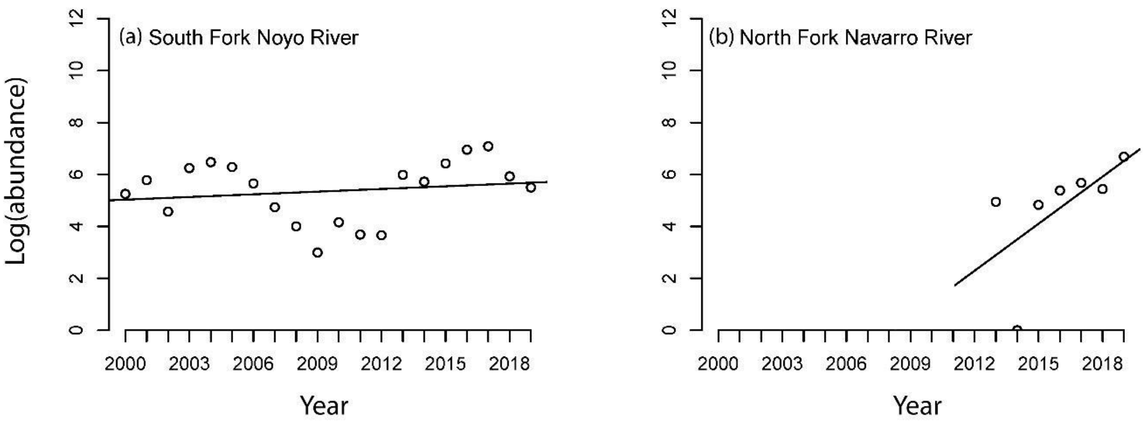

Figure 4.4. Population trends (log abundance) for partial independent populations of CCC coho salmon. Values for South Fork Noyo River are based on mark-recapture estimates except in 2000 and 2009, which are weir counts. Values for North Fork Noyo River are AUC estimates.

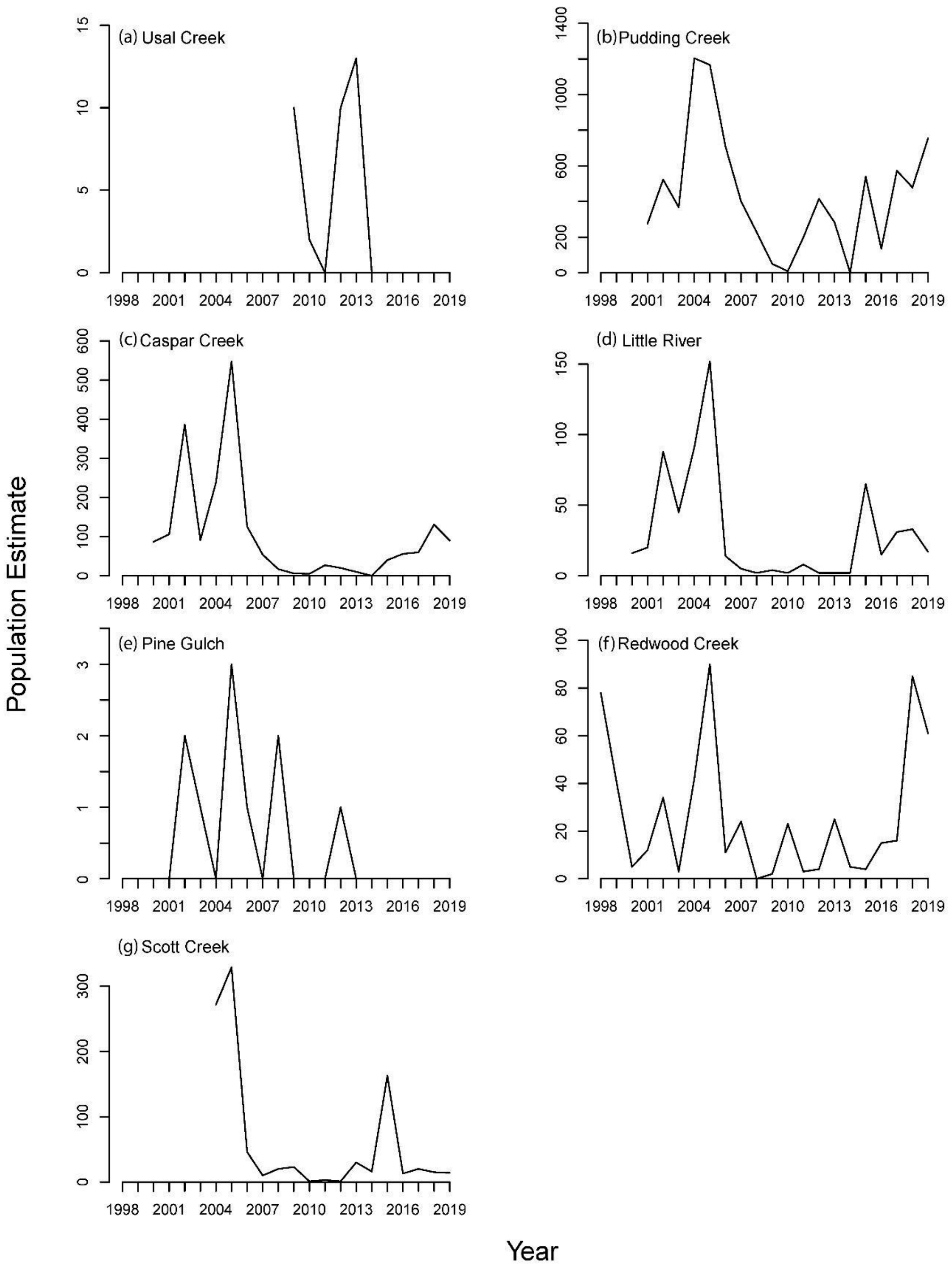

Figure 4.5. Time series of population abundance estimates for dependent populations of CCC coho salmon. Values for Redwood Creek and Pine Gulch are total redd counts for these watersheds. All other estimates are based on fish/redd expansions from life cycle monitoring stations.

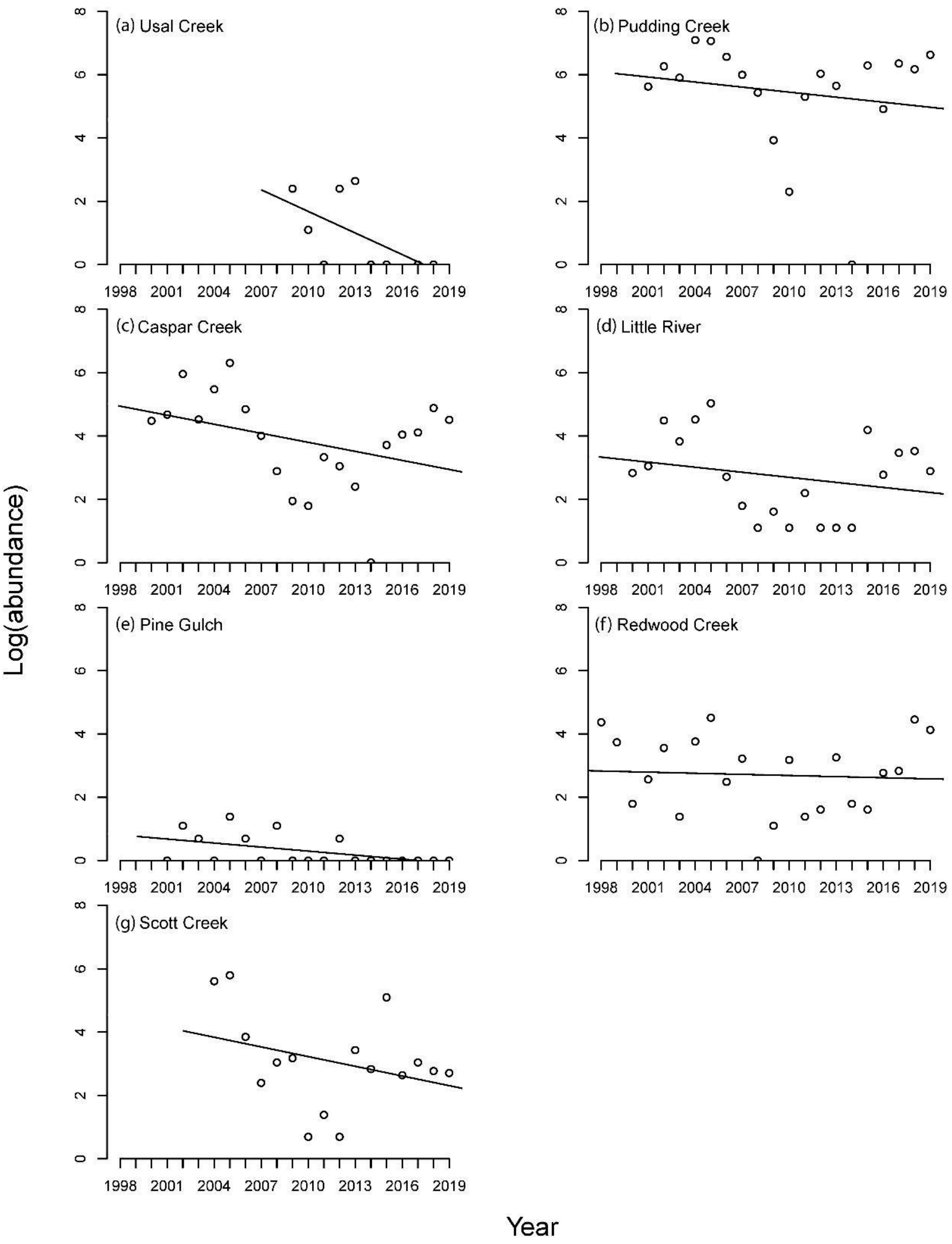

Figure 4.6. Population trends (log abundance) for dependent populations of CCC coho salmon. Values for Redwood Creek and Pine Gulch are total redd counts for these watersheds. All other estimates are based on mark-recapture estimates or fish/redd expansions from life cycle monitoring stations.

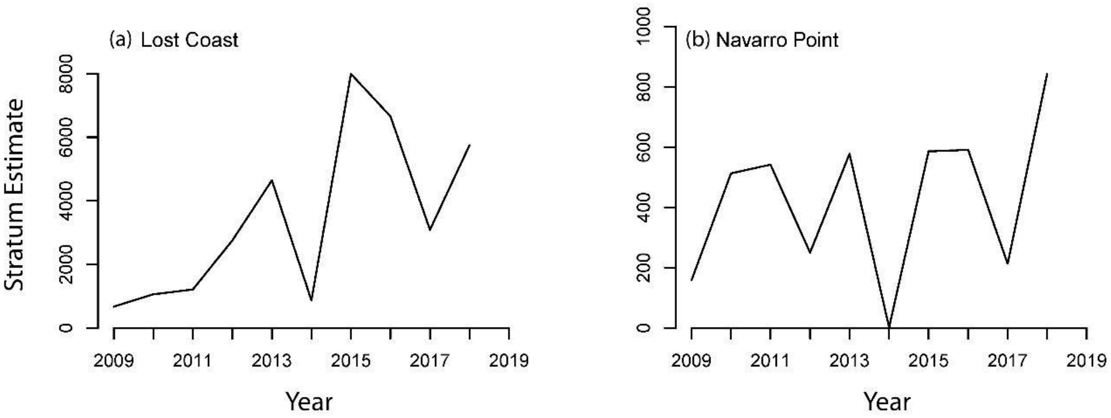

Figure 4.7. Time series of population abundance estimates for the (a) Lost Coast and (b) Navarro Point diversity strata of CCC coho salmon.

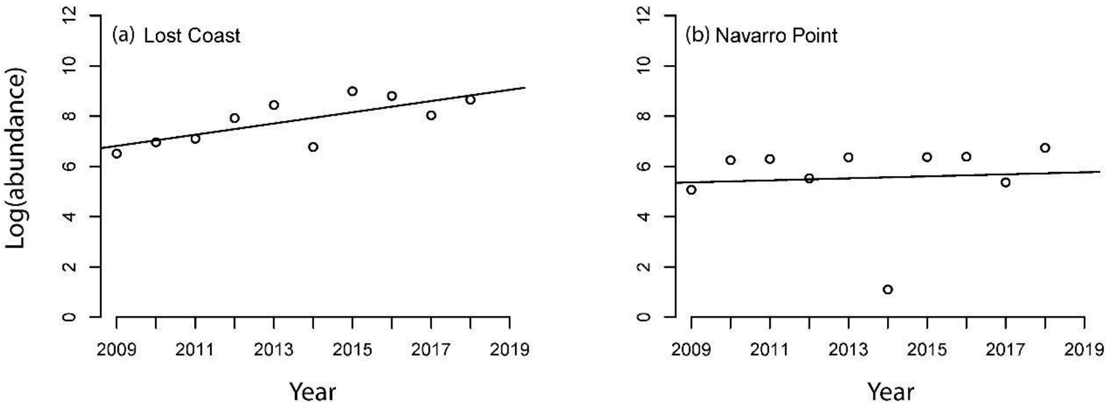

Figure 4.8. Trends (log abundance) for the (a) Lost Coast and (b) Navarro Point diversity strata of CCC coho salmon.

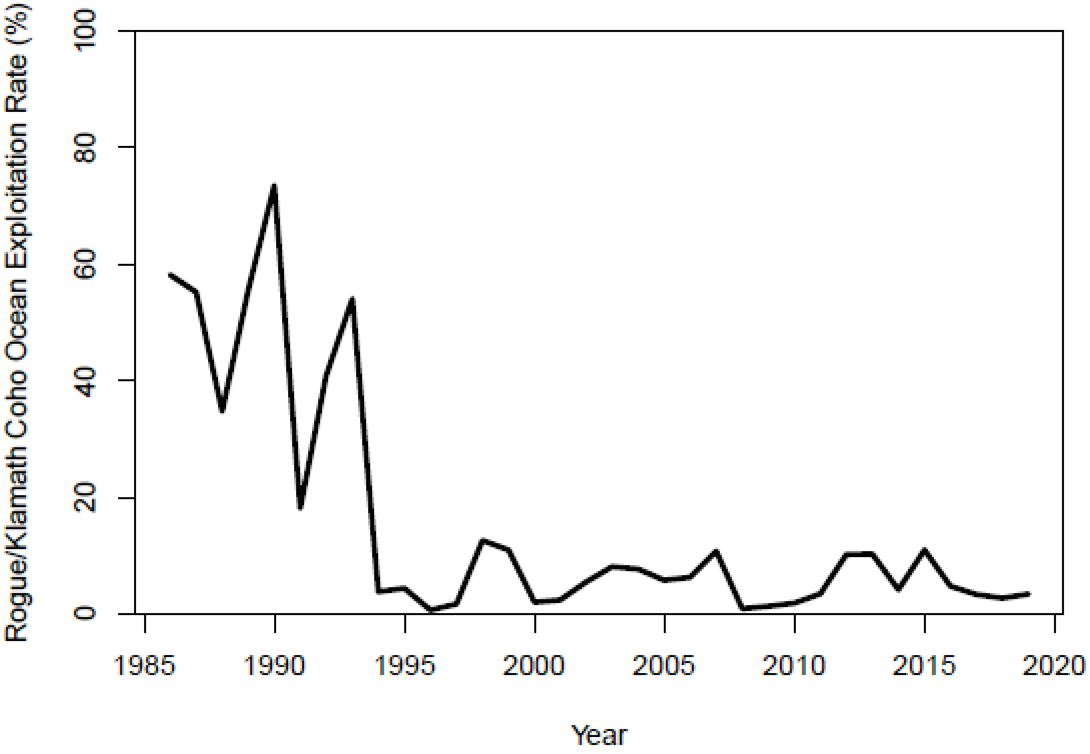

Figure 4.9. Rogue/Klamath (RK) coho salmon ocean exploitation rate estimates for years 1986–2019. Estimates provided by J. Carey, NMFS (personal communication).

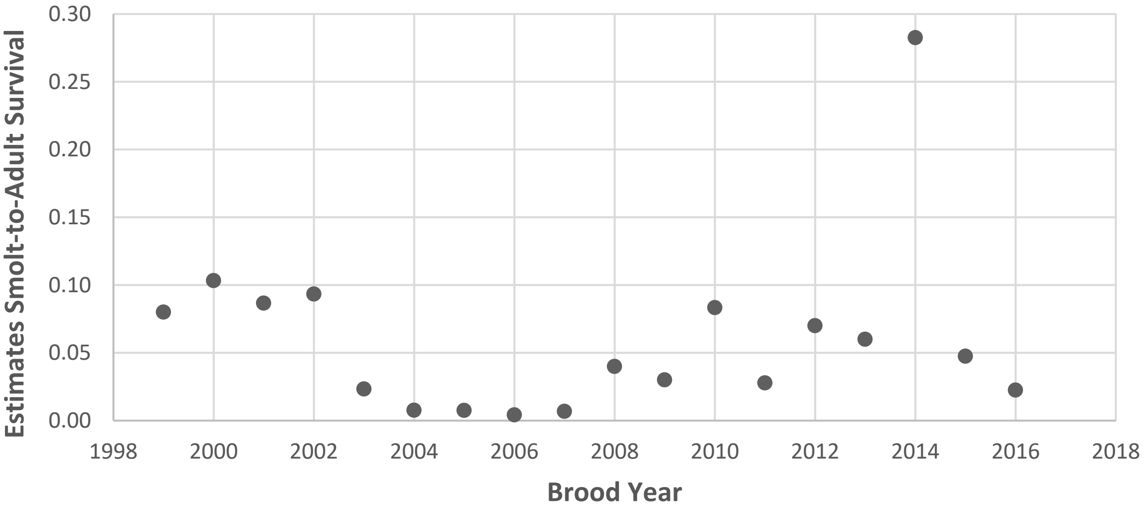

Figure 4.10. Average smolt-to-adult survival rates estimates at four life cycle monitoring stations on the Mendocino Coast (South Fork Noyo River, Pudding Creek, Caspar Creek, and Little River) for brood years 1998 to 2017. Note that the anomalously high estimate for 2014 reflects the fact that a near complete recruitment failure in 2014 populations in Caspar Creek and Pudding Creek was followed by a rapid rebound in 2017, apparently due to return of 2013 brood-year smolts that spent a second year in freshwater and migrated in spring 2015. Thus, a relatively small number of smolts in 2015 produced a surprisingly large number of adults (see text for elaboration).

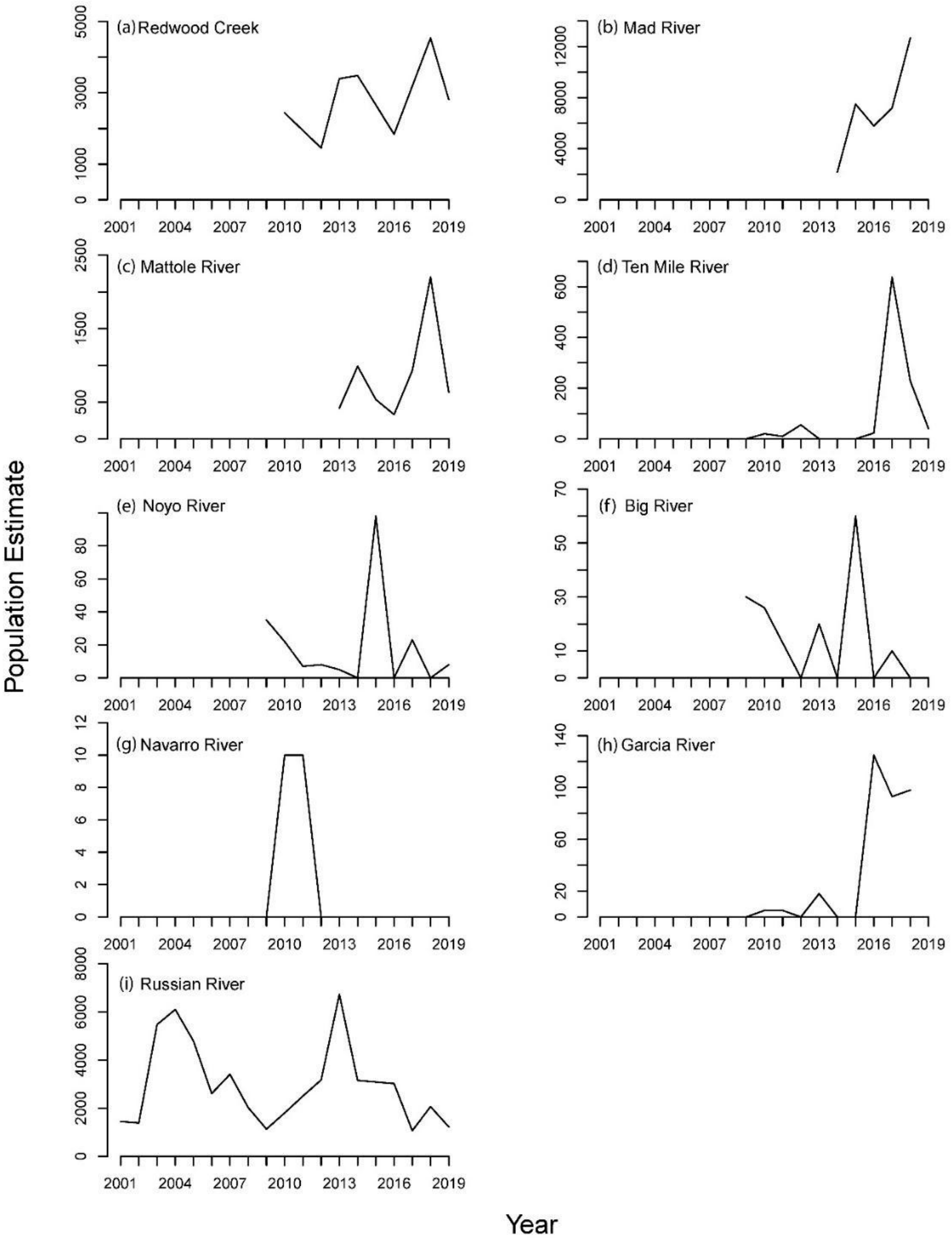

Figure 4.11. Time series of population abundance estimates for independent populations of CC Chinook salmon. Values for Redwood Creek and Mad River are based on sonar counts. Russian River values are video counts at Mirabel Dam. Mattole River values are redd estimates for the watershed. All other estimates are based on fish/redd expansions based on data from life cycle monitoring stations.

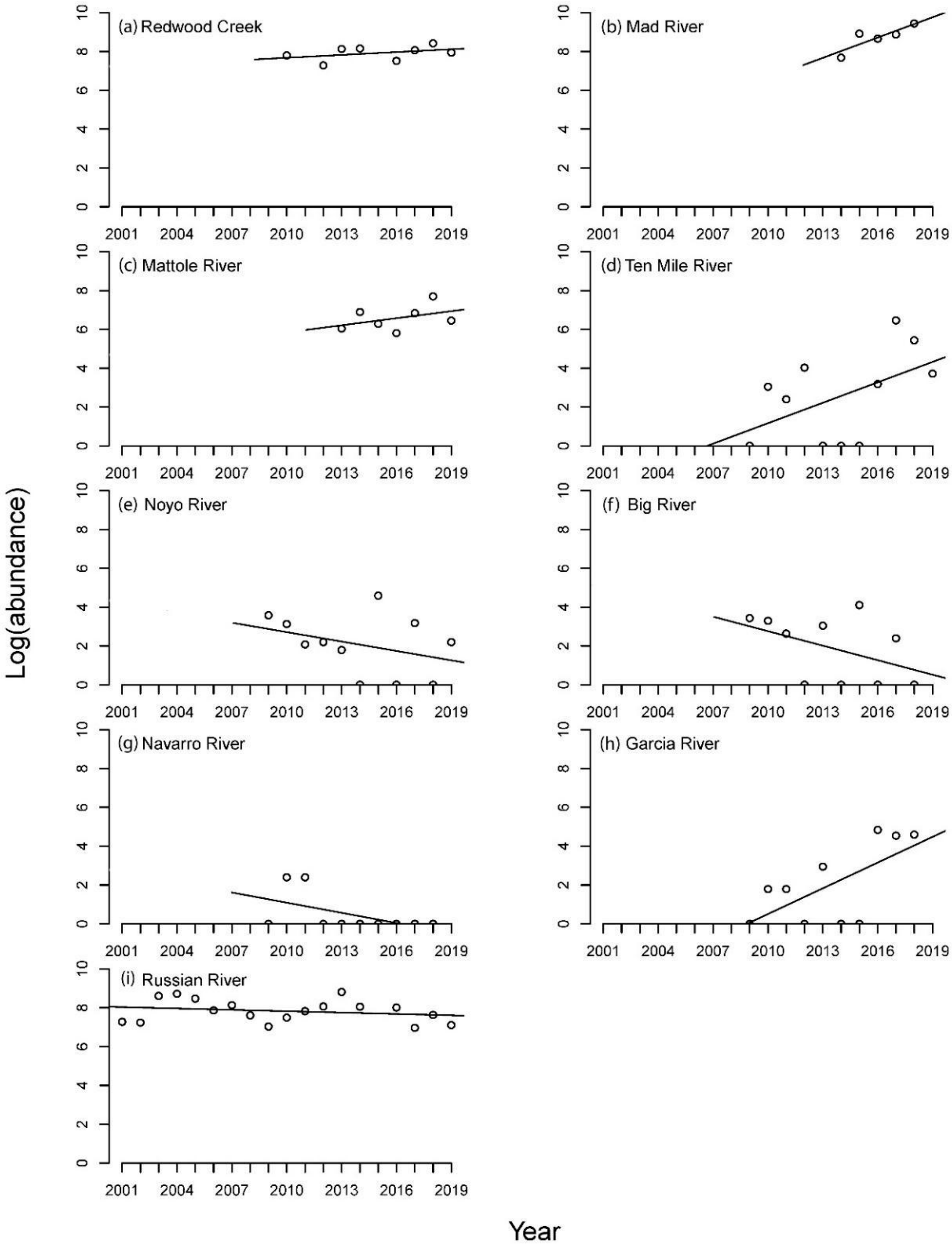

Figure 4.12. Population trends (log abundance) for independent populations of CC Chinook salmon. Values for Redwood Creek and Mad River are based on sonar counts. Russian River values are video counts at Mirabel Dam. Mattole River values are redd estimates for the watershed. All other estimates are based on fish/redd expansions based on data from life cycle monitoring stations.

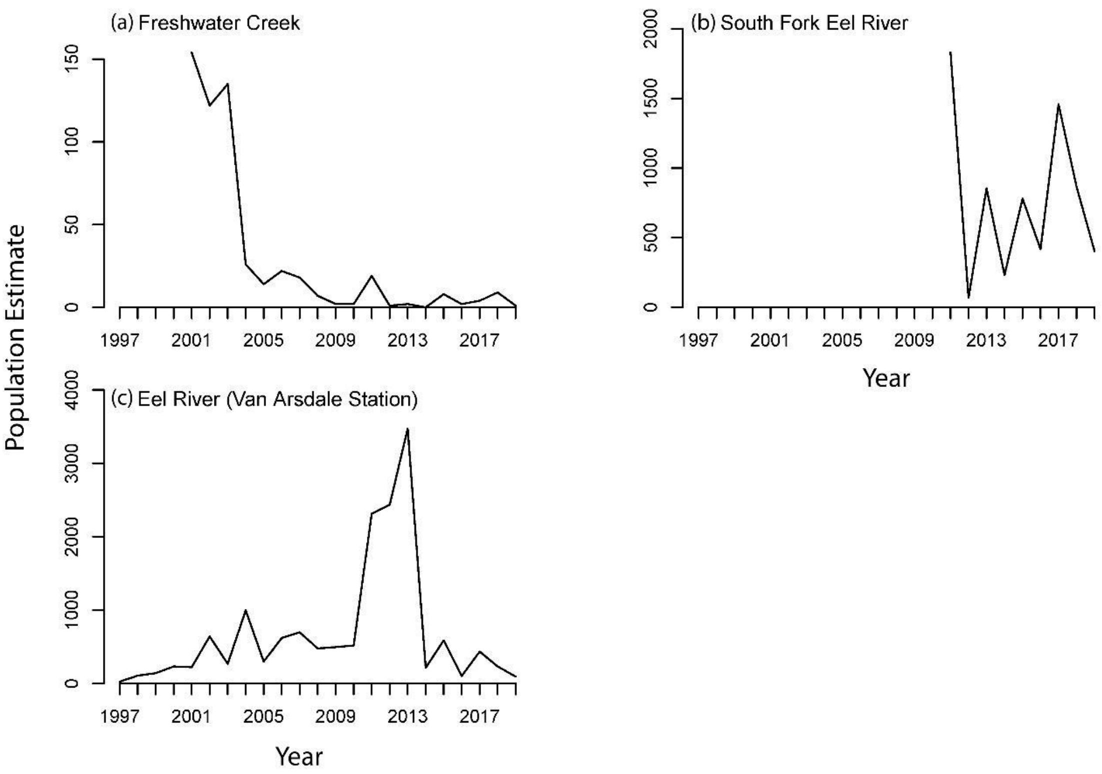

Figure 4.13. Time series of population abundance estimates for partial independent populations of CC Chinook salmon. Values for Freshwater Creek are weir counts. Values for South Fork Eel River redd estimates for the watershed. Eel River values are counts at Van Arsdale Station and include only natural-origin fish.

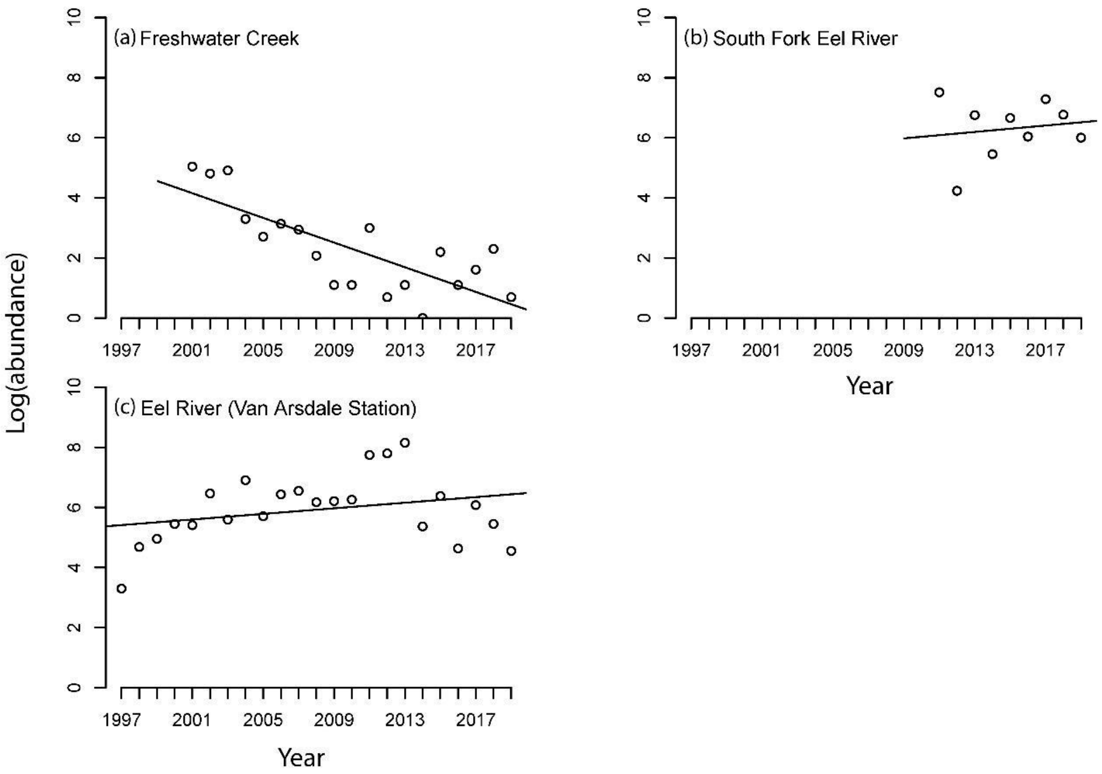

Figure 4.14. Population trends (log abundance) for partial independent populations of CC Chinook salmon. Values for Freshwater Creek are weir counts. Values for South Fork Eel River redd estimates for the watershed. Eel River values are counts at Van Arsdale Station and include only natural-origin fish.

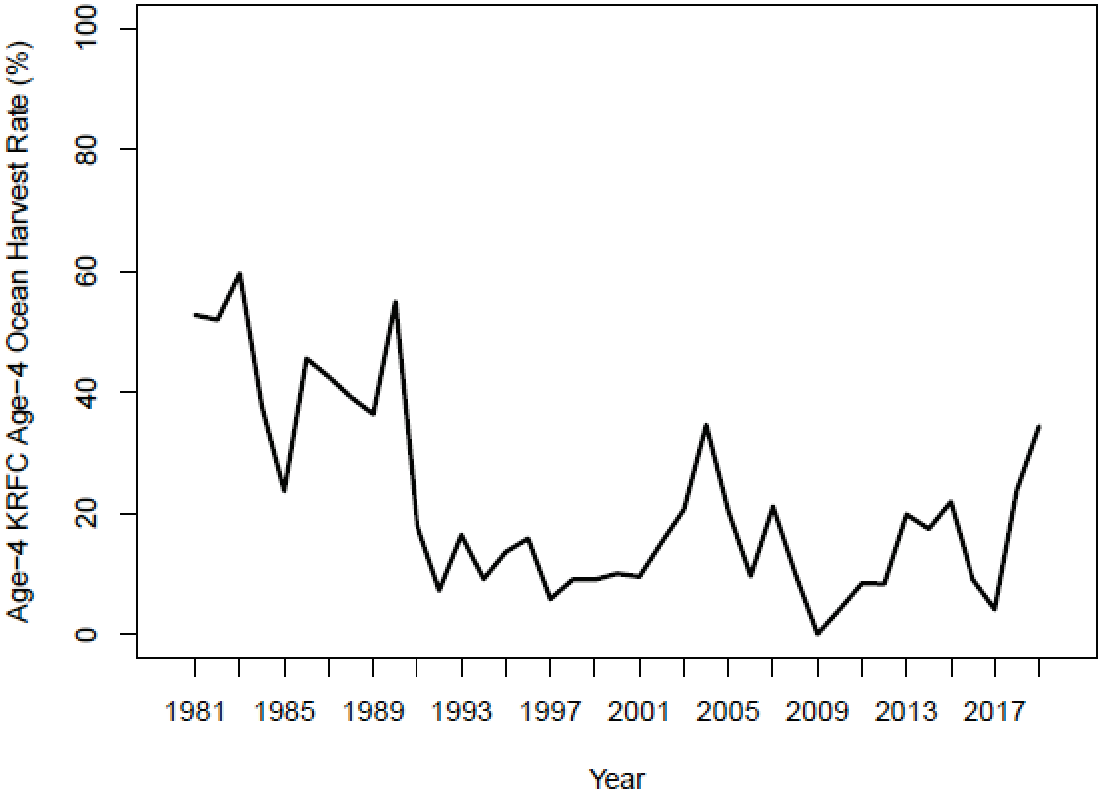

Figure 4.15. Klamath River fall-run Chinook salmon (KRFC) age-4 ocean harvest rate for years 1981–2019 (Pacific Fishery Management Council 2020).

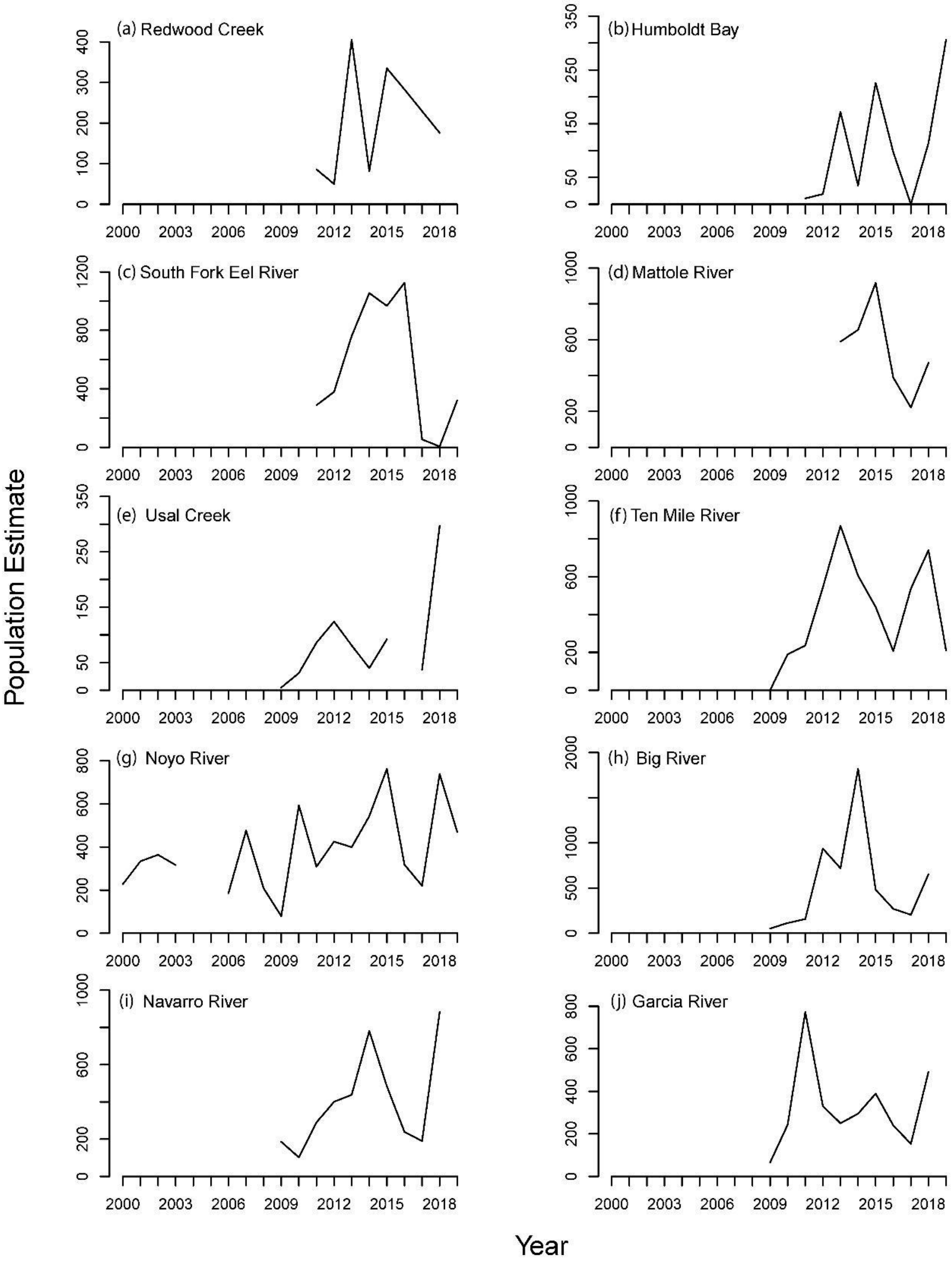

Figure 4.16. Time series of population abundance estimates for focal independent populations of winter-run NC steelhead. Estimates for Redwood Creek, Humboldt Bay, South Fork Eek River and Mattole River are redd estimates for the watershed. Estimates for all other populations are based on fish/redd expansions from life cycle monitoring stations.

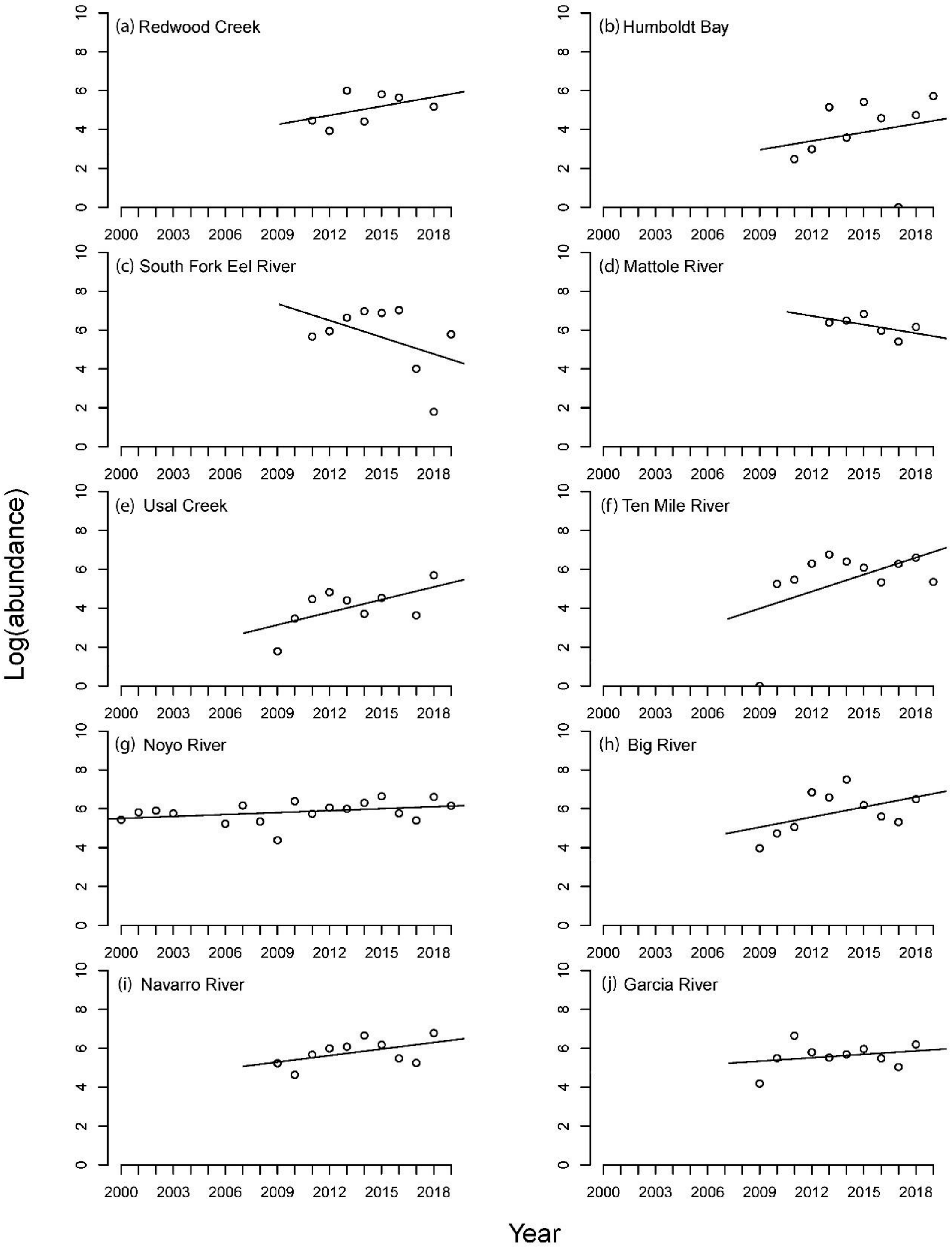

Figure 4.17. Population trends for focal independent populations of winter-run NC steelhead. Estimates for Redwood Creek, Humboldt Bay, South Fork Eek River and Mattole River are redd estimates for the watershed. Estimates for all other populations are based on fish/redd expansions from life cycle monitoring stations.

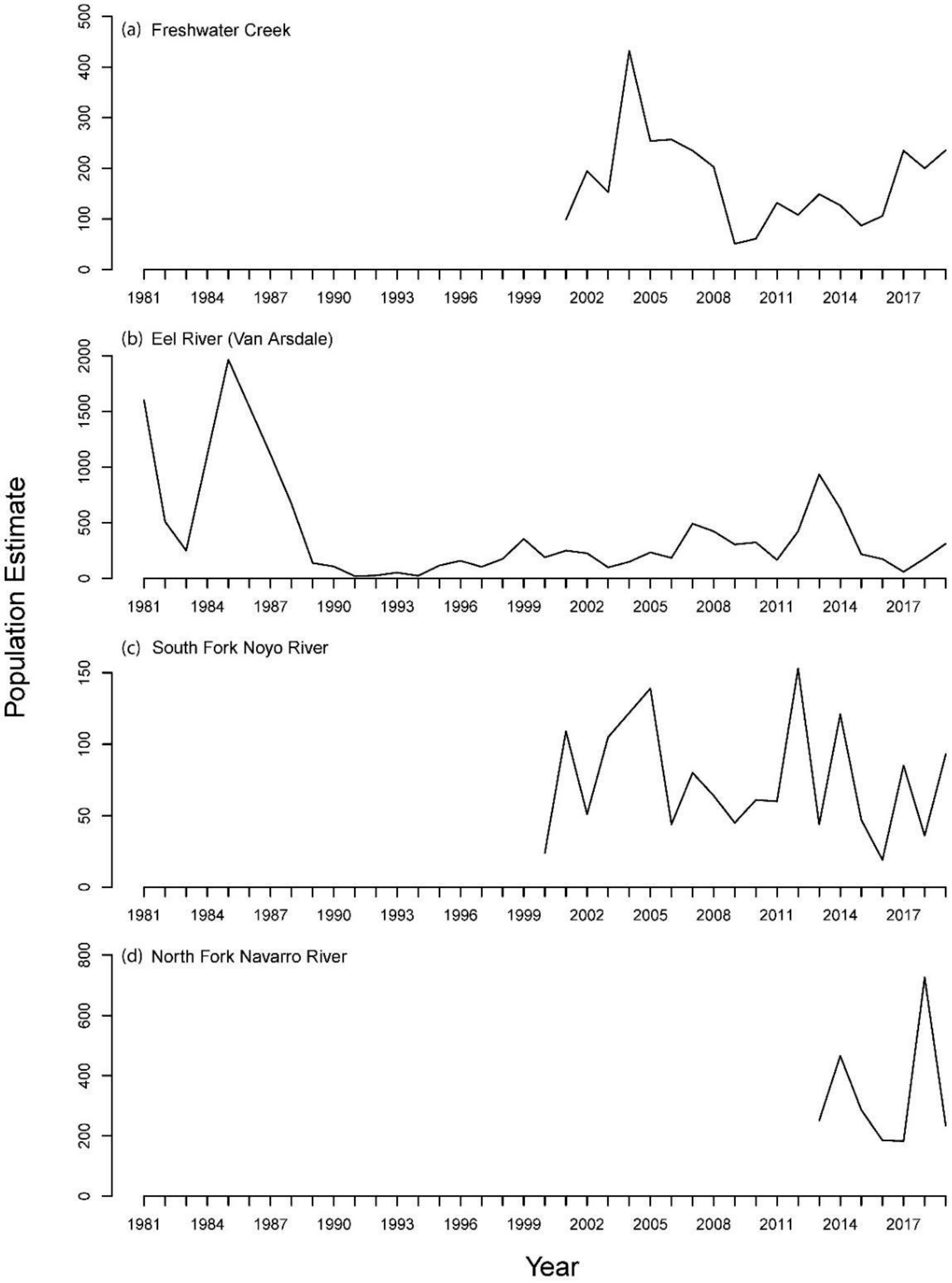

Figure 4.18. Time series of population abundance estimates for partial or aggregate populations of winter-run NC steelhead. Estimates for Freshwater Creek are mark-recapture estimates based on upstream weir captures and downstream kelt recaptures. Eel River counts are trap counts of natural-origin fish at Van Arsdale station. South Fork Noyo values based on a combination of fish/redd expansions and mark-recapture estimates (see text). North Fork Navarro values are based on AUC methods.

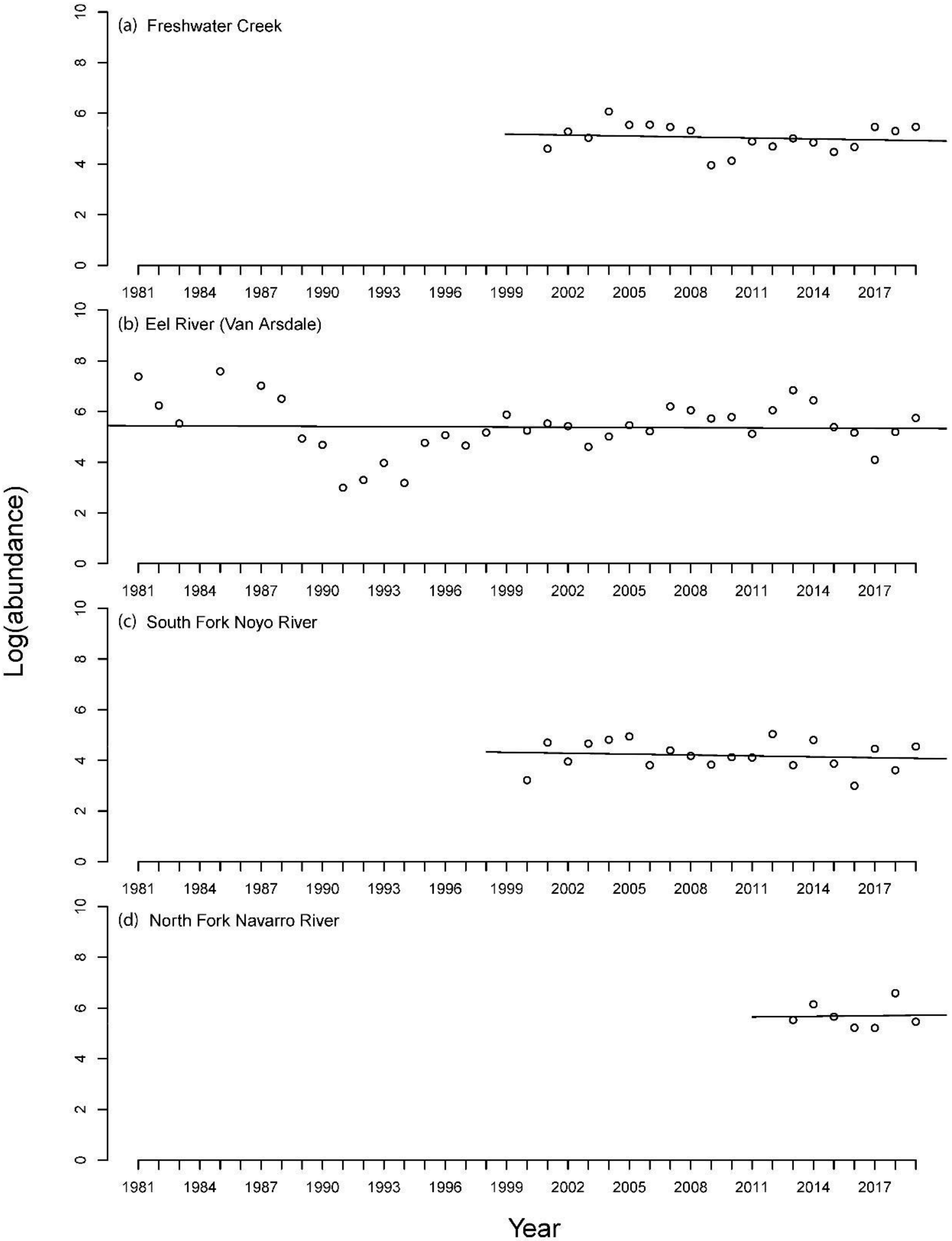

Figure 4.19. Population trends for partial or aggregate populations of winter-run NC steelhead. Estimates for Freshwater Creek are mark-recapture estimates based upstream weir captures and downstream kelt recaptures. Eel River counts are trap counts of natural-origin fish at Van Arsdale station. South Fork Noyo values based on a combination of fish/redd expansions and mark-recapture estimates (see text). North Fork Navarro values are based on AUC methods.

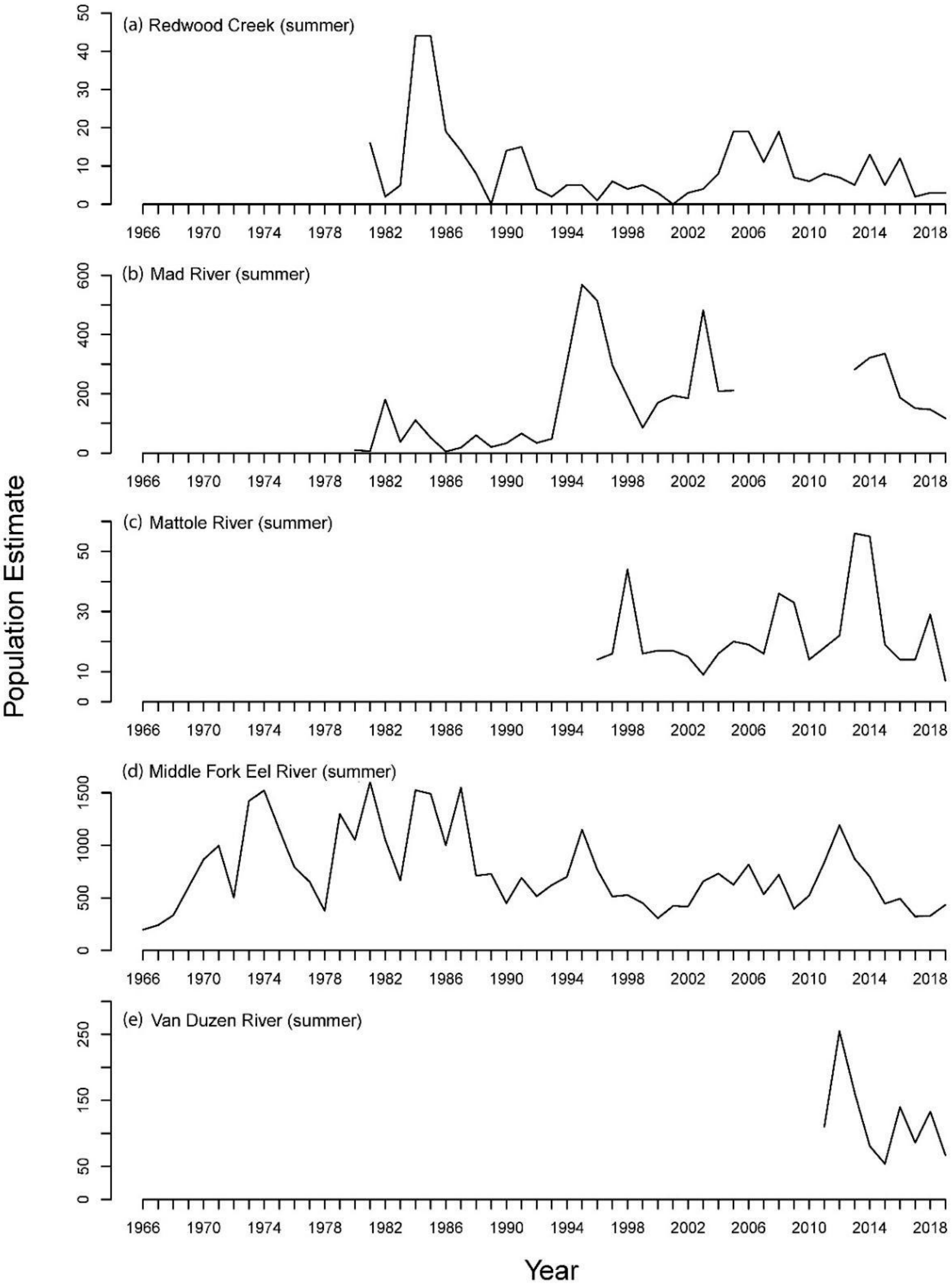

Figure 4.20. Time series of population abundance estimates for independent populations of summer-run NC steelhead. Estimates for Redwood Creek, Mad River and Mattole River are summer dive counts for index reaches. Estimates from Middle Fork Eel River and Van Duzen River are based on summer dive counts covering most available over-summering habitat.

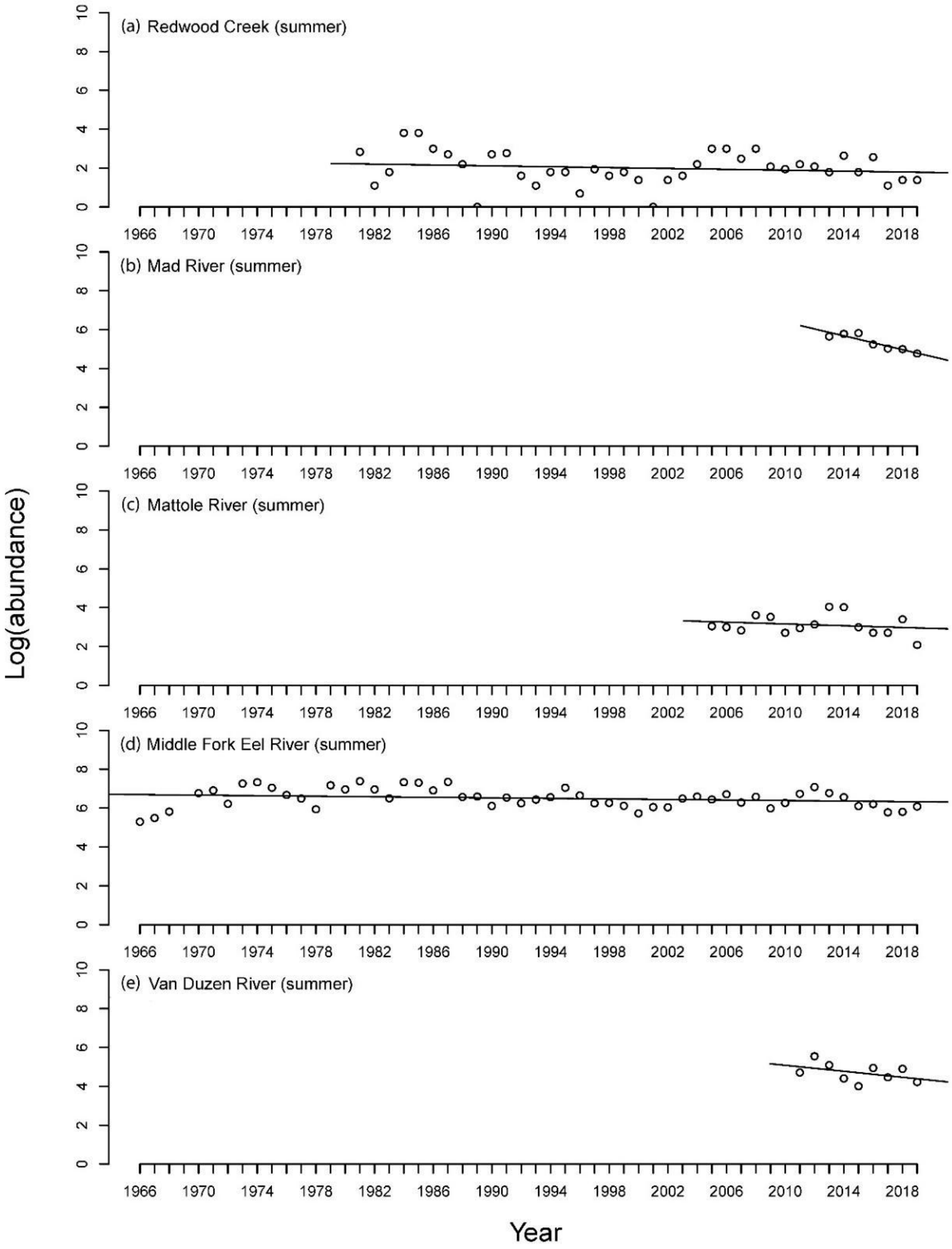

Figure 4.21. Population trends for independent populations of summer-run NC steelhead. Estimates for Redwood Creek, Mad River, and Mattole River are summer dive counts for index reaches. Trend lines for Mad and Mattole rivers are based on years where sample extent was similar. Estimates from Middle Fork Eel River are based on summer dive counts covering most available over-summering habitat.

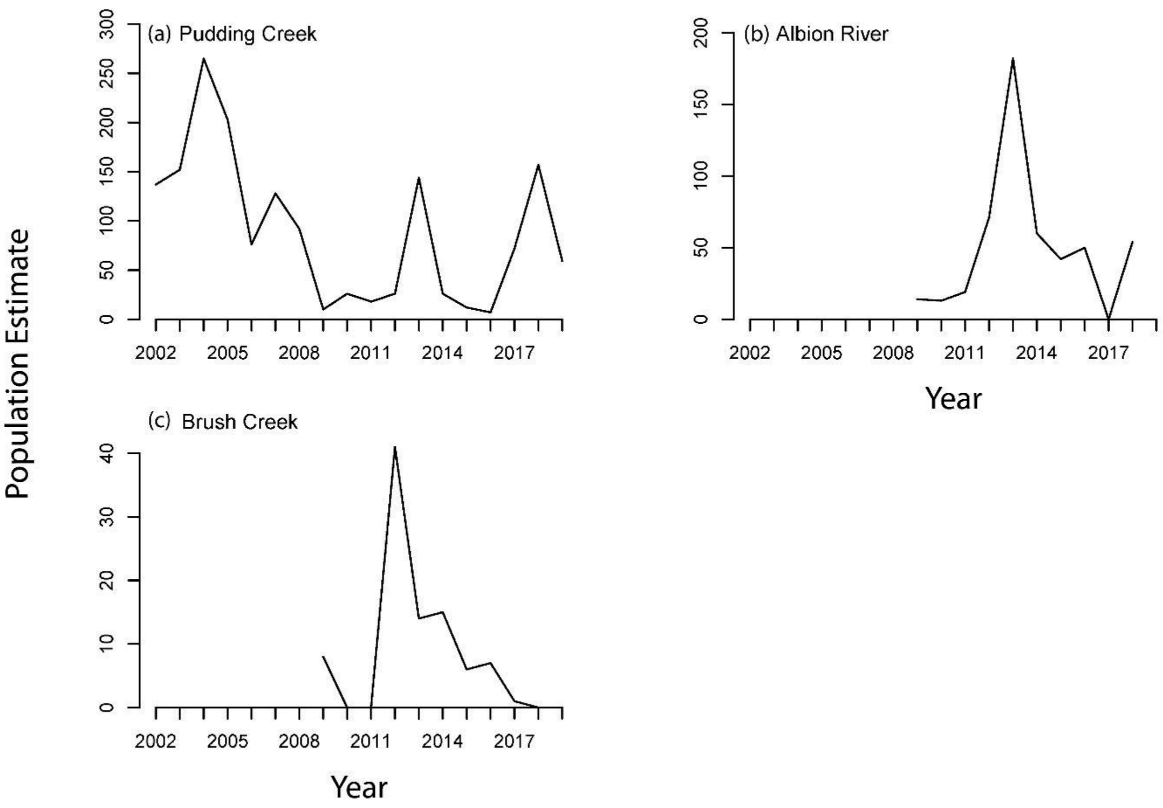

Figure 4.22. Time series of population abundance estimates for supplemental independent populations of winter-run NC steelhead. Estimation methods for Pudding Creek have varied among years (see text). Estimates for other populations are based on fish/redd expansions from life cycle monitoring stations.

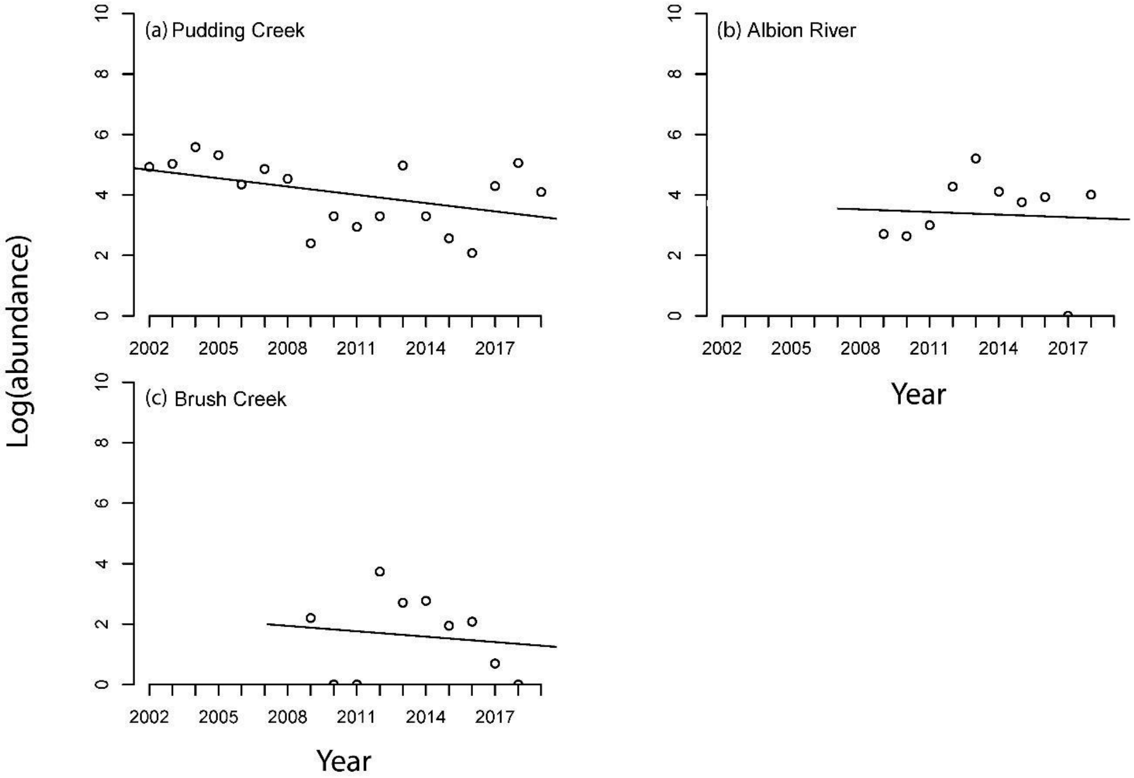

Figure 4.23. Population trends for supporting independent populations of winter-run NC steelhead. Estimation methods for Pudding Creek have varied among years (see text). Estimates for other populations are based on fish/redd expansions from life cycle monitoring stations.

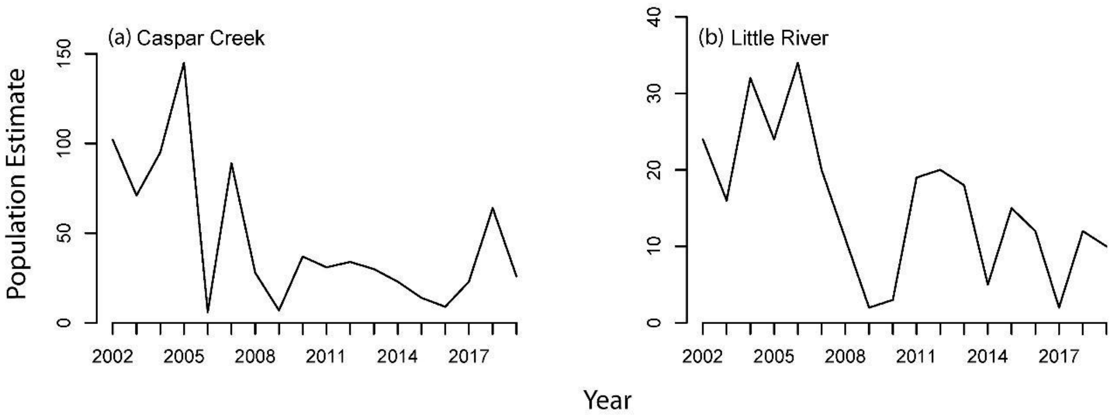

Figure 4.24. Time series of population abundance estimates for dependent populations of winter-run NC steelhead. Estimation methods for Caspar Creek have varied among years (see text). Estimates for Little River are based on fish/redd expansions from life cycle monitoring stations.

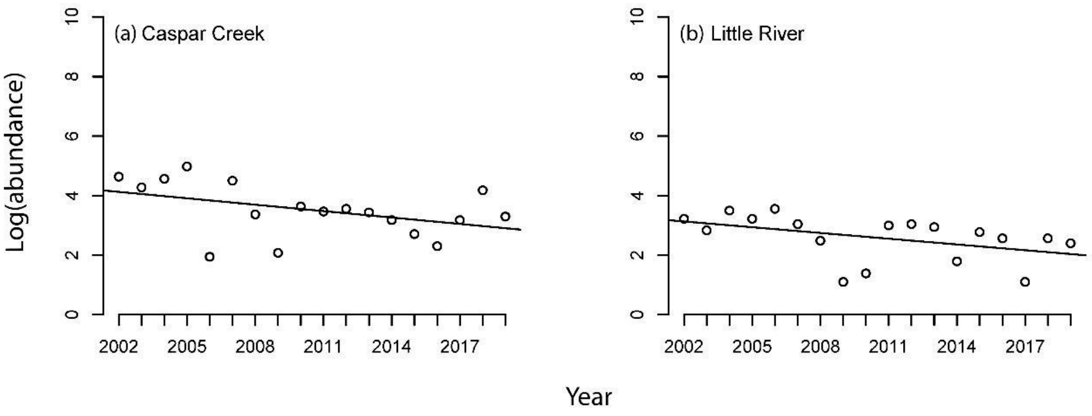

Figure 4.25. Population trends for dependent populations of winter-run NC steelhead. Estimation methods for Caspar Creek have varied among years (see text). Estimates for Little River are based on fish/redd expansions from life cycle monitoring stations.

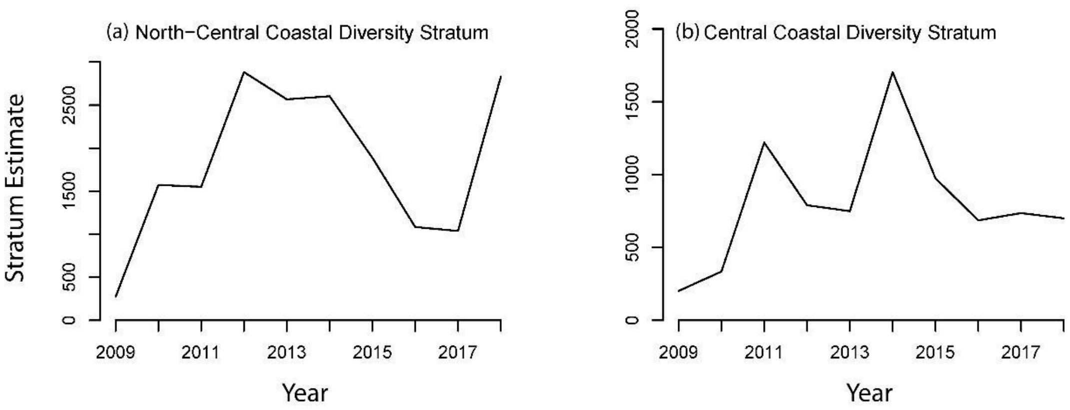

Figure 4.26. Time series of population abundance estimates for the (a) North-Central Coastal and (b) Central Coastal diversity strata of NC steelhead.

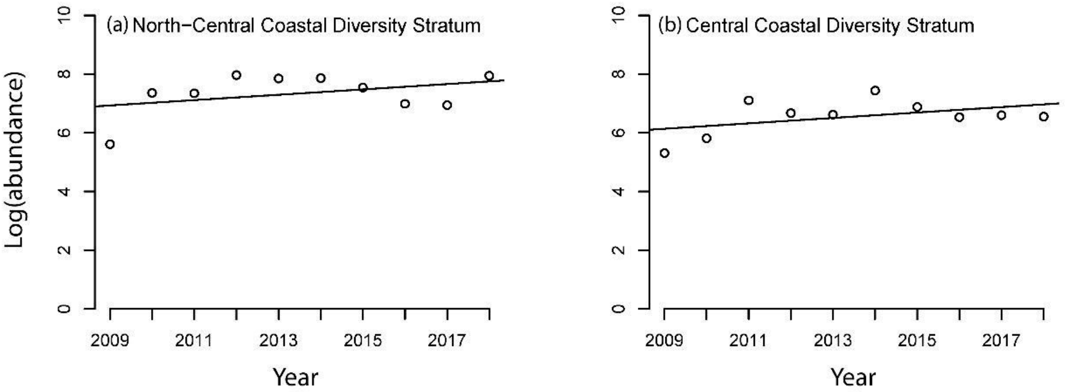

Figure 4.27. Trends (log abundance) for the (a) North-Central Coastal and (b) Central Coastal diversity strata of CCC steelhead.

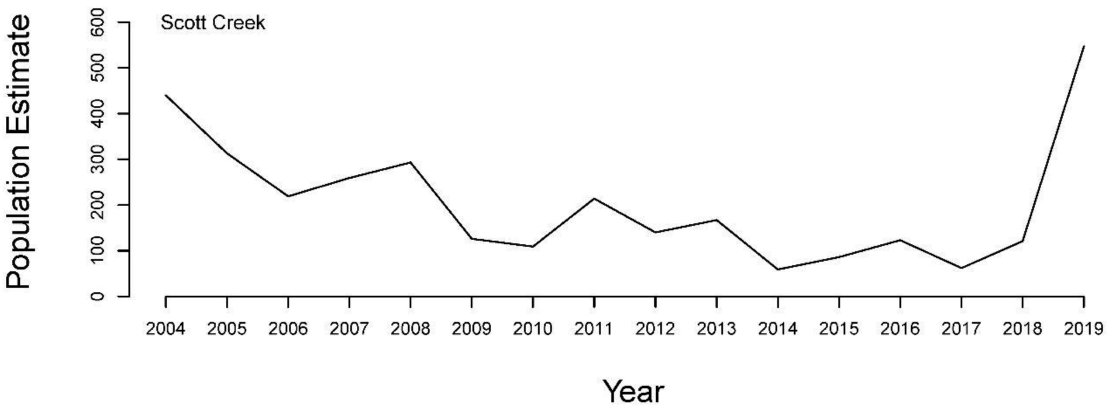

Figure 4.28. Time series of population abundance estimates for independent populations of winter-run CCC steelhead. Estimates are based on mark-recapture estimates from Scott Creek life cycle monitoring stations.

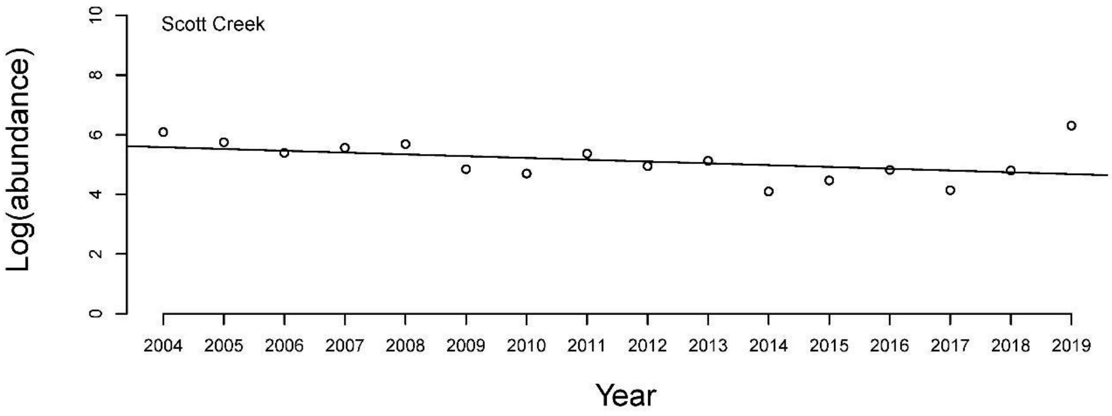

Figure 4.29. Population trends for independent populations of winter-run CCC steelhead. Estimates are based on mark-recapture estimates from Scott Creek life cycle monitoring stations.

Table 4.1. Criteria for assessing the level of risk of extinction for populations of Pacific salmonids. Overall risk is determined by the highest risk score for any category. N_g = generational sum of abundance; N_e = effective population size; and N_a = annual spawner abundance. From Spence et al. (2008).

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

a – Population has declined within the last two generations or is projected to decline within the next two generations (if current trends continue) to annual run size $N_a \leq 500$ spawners (historically small but stable populations not included) *or* $N_a > 500$ but declining at a rate of ≥ 10% per year over the last two-to-four generations.

b – Annual run size N_a has declined to ≤ 500 spawners, but is now stable *or* run size $N_a > 500$ but continued downward trend is evident.

c – Annual run size decline in one generation < 90% but biologically significant (e.g., loss of year class).

d – *IPkm* = the estimated aggregate intrinsic habitat potential for a population inhabiting a particular watershed (i.e., total accessible km weighted by reach-level estimates of intrinsic potential; see Bjorkstedt et al. [2005] for greater elaboration).

e – MRD is the population-specific minimum required density for viability.

Table 4.2. ESU-level criteria for assessing the level of risk of extinction for Pacific salmonid ESUs. From Spence et al. (2008).

Criterion	Description
Representation	All identified diversity strata that include historical functionally or potentially independent populations within an ESU/DPS should be represented by viable populations for the ESU/DPS to be considered viable -AND- Within each diversity stratum, all extant phenotypic diversity (i.e., major life-history types) should be represented by viable populations
Redundancy and Connectivity	At least 50% of historically independent populations in each diversity stratum must be demonstrated to be at low risk of extinction according to the population viability criteria outlined in Table 1 of Spence et al. (2008) -AND- Within each diversity stratum, the total aggregate abundance of independent populations selected to satisfy this criterion must meet or exceed 50% of the aggregate viable population abundance (i.e., meeting density-based criteria for low risk) for all independent populations Remaining populations, including historical dependent populations and any historical independent populations that are not expected to attain a viable status must exhibit occupancy patterns consistent with those expected under sufficient immigration subsidy arising from the "core" independent populations selected to satisfy the preceding criterion The distribution of extant populations, regardless of historical status, must maintain connectivity within the diversity stratum, as well as connectivity to neighboring diversity strata

Table 4.3. Viability metrics for independent populations of coho salmon in the CCC coho salmon ESU. NA = not available or applicable. Trends shown only for populations where time series is ≥ 6 years; **bold** indicates significant trend. IPkm includes only currently accessible habitats. $N_{a(arith)}$ target refers to recovery target (NMFS 2012a).

Stratum/population	Yrs	$\overline{N}_{a(arith)}$	$\overline{N}_{a(geom)}$	$\overline{N}_{g(harm)}$	$\hat{C}$	$\hat{T}$ *(95% CI)*	IPkm	$\hat{D}_{dep}$	$\hat{D}_{ssd}$	$\hat{D}_{ssd}$ target	$\overline{N}_{a(arith)}$ target
Lost Coast - Navarro Pt											
Ten Mile River[a]	11	637	206	1,580	-0.22	0.377 (-0.099, 0.852)	105.1	1.9	6.1	34.9	3,700
Noyo River[a]	17	1,324	849	1,887	0.72	0.061 (-0.029, 0.151)	118.0	2.4	12.2	34.0	4,000
Big River[a,b]	10	468	312	948	-2.59	**0.230** (0.053, 0.406)	191.8	0.6	2.4	28.9	5,500
Albion River[a,b]	10	201	49	437	-0.80	0.335 (-0.249, 0.918)	59.2	1.0	3.4	38.1	2,300
Navarro Pt - Gualala Pt											
Navarro River[a,b]	10	303	166	834	0.05	0.047 (-0.462, 0.555)	201.0	1.0	1.5	28.3	5,700
Garcia River[a,b]	10	139	47	241	-1.54	0.299 (-0.183, 0.781)	76.0	0.4	1.8	36.9	3,700
Gualala River	-	-	-	-	-	-	251.6	-	-	24.8	6,200
Coastal											
Russian River[c]	6	122	117	380	NA	-0.010 (-0.231, 0.212)	584.2	NA	NA	20.0	10,100
Walker Creek	-	-	-	-	-	-	76.2	-	-	36.9	2,600
Lagunitas Creek[d]	22	247	202	576	0.85	-0.030 (-0.079, 0.019)	70.4	NA	NA	37.3	2,600
Santa Cruz Mtn											
Pescadero Creek[e]	7	<1	NA	NA	NA	NA	60.6	-	-	38.0	2,300
San Lorenzo River[e]	7	<1	NA	NA	NA	NA	126.4	-	-	33.4	3,800

a – Numbers indicate the estimated number of adults based on fish/redd expansions from life cycle monitoring stations.

b – Basin-wide spawner surveys not conducted in 2018–2019 spawning season.

c – Numbers indicate basin-wide estimate of the number of redds. Methods for deriving fish/redd estimates for expansion are currently considered unreliable.

d – Numbers indicate redd census over most available spawning habitat. Methods have not been developed to derive fish/redd estimates for expansion.

e – Numbers indicate average number of adult fish observed (live adults + carcasses). Methods for redd assignment are considered unreliable; thus, population estimates are not available.

Table 4.4. Viability metrics for dependent populations of coho salmon in the CCC coho salmon ESU. (s) = supplemental population. NA = not available or applicable. Trends shown only for populations where time series is ≥ 6 years; bold indicates significant trend. $N_{a(arith)}$ target refers to recovery target (NMFS 2012a).

Stratum/population	Years	$\overline{N}_{a(arith)}$	$\overline{N}_{a(geom)}$	$\overline{N}_{g(harm)}$	$\hat{T}$ (95% CI)	$\overline{N}_{a(arith)}$ target
Lost Coast - Navarro Pt						
Usal Creek	9	4	2	15	-0.227 (-0.512, 0.057)	360
Cottaneva Creek	5	0	NA	NA	NA	469
Juan Creek (s)	3	15	NA	NA	NA	NA
DeHaven Creek (s)	1	0	NA	NA	NA	NA
Wages Creek[a]	4	0	NA	NA	NA	340
Pudding Creek	19	438	230	732	-0.053 (-0.208, 0.102)	983
Hare Creek (s)	4	0	NA	NA	NA	NA
Jug Handle Creek (s)	-	-	-	-	-	NA
Caspar Creek	20	105	45	98	-0.095 (-0.213, 0.023)	435
Russian Gulch (s)	-	-	-	-	-	NA
Little River	20	31	13	26	-0.053 (-0.159, 0.052)	NA
Big Salmon Creek	4	27	6	NA	NA	578
Navarro Pt – Gualala Pt						
Greenwood Creek	4	5	3	NA	NA	NA
Elk Creek	3	0	NA	NA	NA	NA
Alder Creek (s)	1	5	NA	NA	NA	NA
Brush Creek	10	0	NA	NA	NA	NA
Coastal						
Salmon Creek	-	-	-	-	-	1367
Pine Gulch	19	1	1	0	-0.042 (-0.080, 0.005)	394
Redwood Creek	22	26	13	47	-0.012 (-0.101, 0.077)	272
Santa Cruz Mtn						
San Gregorio Creek[b]	5	0	NA	NA	NA	1,363
Gazos Creek[b]	7	0	NA	NA	NA	279
Waddell Creek[b]	6	<1	NA	NA	NA	313
Scott Creek	16	61	19	39	-0.102 (-0.273, 0.069)	510
San Vicente Creek[b]	7	5	NA	NA	NA	105
Soquel Creek[b]	4	0	NA	NA	NA	1,122
Aptos Creek[b]	3	0	NA	NA	NA	932

a – Not surveyed since 2016 viability assessment.

b – Numbers are combined counts of live coho and carcasses observed during surveys. Redd assignment methods are considered unreliable, so population estimates are not available.

Table 4.5. Viability metrics for independent populations of Chinook salmon in the CC Chinook salmon ESU. (s) = supporting population. NA = not available or applicable. Trends shown only for populations where time series is ≥ 6 years; bold indicates significant trend. IPkm includes only currently accessible habitats. $N_{a(\mathrm{arith})}$ target refers to recovery target (NMFS 2016b).

Stratum/population	Yrs	$\overline{N}_{a(arith)}$	$\overline{N}_{a(geom)}$	$\overline{N}_{g(harm)}$	$\hat{C}$	$\hat{T}$ (95% CI)	IPkm	$\hat{D}_{dep}$	$\hat{D}_{ssd}$	$\hat{D}_{ssd}$ target	$\overline{N}_{a(arith)}$ target
North Coastal											
Redwood Creek[a]	8	2,896	2,738	8,718	NA	0.049 (-0.059, 0.156)	116.1	20.9	24.9	29.3	3,400
Little River	-	-	-	-	-	-	17.4	-	-	40.0	700
Mad River[a]	5	7,059	6,116	19,656	NA	NA	94.4	54.5	74.8	31.7	3,000
Humboldt Bay	-	-	-	-	-	-	76.6	-	-	33.7	2600
Lower Eel River (part)[b]	-	-	-	-	-	-	368.4	-	-	20.0	7,400
Bear River	-	-	-	-	-	-	39.4	-	-	37.8	1,500
Mattole River[c]	7	862	714	2,308	NA	0.121 (-0,186, 0.429)	177.5	3.4	4.9	22.5	4,000
North Mtn. Interior											
Lower Eel River (part)[b]	-	-	-	-	-	-	144.0	-	-	20.0	2,900
Upper Eel River	-	-	-	-	-	-	528.5	-	-	20.0	10,600
North-Central Coastal											
Ten Mile River[d] (s)	11	92	13	0	-9.5	0.351 (-0.105, 0.807)	67.2	NA	NA	6-12	401-804
Noyo River[d]	11	19	7	33	0.38	-0.161 (-0.492, 0.169)	62.2	0.1	0.3	35.3	2,200
Big River[d]	10	16	6	30	0.74	-0.249 (-0.657, 0.159)	104.3	0.0	0.2	30.6	3,200
Central Coastal											
Navarro River[d] (s)	10	2	2	0	1.00	-0.174 (-0.407, 0.058)	131.5	NA	NA	6-12	787-1,567
Garcia River[d]	10	34	7	22	-20.8	**0.442 (0.020, 0.864)**	56.2	0.1	0.6	36.0	2,000
Gualala River (s)	-	-	-	-	-	-	175.6	-	-	6-12	1,052-2,105
Russian River[e]	18	2,949	2,512	8,048	0.67	NA	465.2	3.1	6.8	20.0	9,300

a – Numbers are estimates based on sonar counts adjusted for species.
b – Lower Eel population spans two strata: Lower Mainstem Eel and S. Fk. Eel in the North Coastal Stratum, and Larabee Creek and Van Duzen River in the North Mountain Interior stratum.
c – Numbers indicate basin-wide estimate of the number of redds. Methods for deriving fish/redd estimates for expansion are currently considered unreliable.
d – Numbers indicate the estimated number of adults based on fish/redd expansions from life cycle monitoring stations.
e – Numbers are based on video counts at Mirabel Dam; a small but unknown percentage of adults spawn below this location, so the estimate does not include entire population.

4.6. Population information for CC Chinook salmon populations with only index data or partial population estimates. NA = not available or applicable. Trends shown only for populations where time series is ≥ 6 years, bold indicates significant trend. Short-term (12-yr) trends are shown along with long-term trends for those datasets spanning more than 18 years.

Stratum/population	Yrs	$\overline{N}_{a(arith)}$	$\overline{N}_{a(geom)}$	$\overline{N}_{g(harm)}$	$\hat{T}$ (95% CI)
North Coastal					
Freshwater Creek[a]	19	29	8	16	**-0.205 (-0.291, -0.119)**
	12	5	3	10	-0.024 (-0.188, 0.140)
SF Eel River[b]	9	768	543	1,940	0.054 (-0.272, 0.379)
North Mtn. Interior					
Van Arsdale Station[c]	23	680	372	1,065	0.047 (-0.026, 0.119)
	12	948	508	1,582	-0.169 (-0.366, 0.027)

a – Freshwater Creek represents a portion of the Humboldt Bay population. Numbers are weir counts of natural-origin fish; counts are considered incomplete, as fish may pass over or through weir under some conditions.

b – SF Eel River represents a portion of the Lower Eel River population. Numbers are expanded estimates of the number of redds within coho sampling frame. Mainstem of S. Fk Eel below Branscomb is not surveyed.

c – Van Arsdale Station counts represent a portion of the Upper Eel River population. Numbers are counts of natural-origin fish passed over the dam and represent a variable fraction of the total Upper Eel population, as the proportion of individuals reaching the dam appears highly flow dependent. Counts are available prior to 1997; however, data from earlier years are confounded by inconsistent trap operation and the occurrence of hatchery fish and so not appropriate for trend analysis.

Table 4.7. Viability metrics for independent winter-run populations of steelhead in the NC steelhead DPS. (s) = supporting population. NA = not available or applicable. Trends shown only for populations where time series is ≥ 8 years; **bold** indicates significant trend. IPkm includes only currently accessible habitats. $N_{a(arith)}$ target refers recovery target (NMFS 2016b).

Stratum/population	Yrs	$\overline{N}_{a(arith)}$	$\overline{N}_{a(geom)}$	$\overline{N}_{g(harm)}$	$\hat{C}$	$\hat{T}$ *(95% CI)*	IPkm	$\hat{D}_{dep}$	$\hat{D}_{ssd}$	$\hat{D}_{ssd}$ target	$\overline{N}_{a(arith)}$ target
Northern Coastal											
Redwood Creek[a]	7	202	157	834	NA	NA	161.1	NA	NA	20.0	3,200
Maple Cr/Big Lagoon	-	-	-	-	-	-	71.7	-	-	32.3	2,300
Little River	-	-	-	-	-	-	50.0	-	-	35.3	1,800
Mad River[b]	3	6,602	6,508	NA	NA	NA	146.3	-	-	21.9	3,200
Humboldt Bay[c]	9	109	46	391	NA	0.149 (-0.431, 0.728)	203.4	NA	NA	20.0	4,100
SF Eel River[a]	9	551	273	2,483	NA	-0.287 (-0.787, 0.214)	951.8	NA	NA	20.0	19,000
Bear River	-	-	-	-	-	-	107.8	-	-	27.2	2,900
Mattole River[a]	6	540	493	2,221	NA	NA	534.4	NA	NA	20.0	10,700
Lower Interior											
Jewett Creek (s)	-	-	-	-	-	-	16.8	-	-	6-12	99-200
Chamise Creek	-	-	-	-	-	-	36.2	-	-	37.2	1,300
Bell Springs Creek (s)	-	-	-	-	-	-	18.1	-	-	6-12	107-215
Woodman Creek	-	-	-	-	-	-	35.0	-	-	37.4	1,300
Outlet Creek	-	-	-	-	-	-	176.0	-	-	20.0	3,500
Tomki Creek	-	-	-	-	-	-	89.5	-	-	29.8	2,700
Bucknell Creek	-	-	-	-	-	-	9.0	-	-	6-12	52-106
North Mtn. Interior											
Redwood Creek (upper)	-	-	-	-	-	-	86.2	-	-	30.2	2,600
Mad River (upper)	-	-	-	-	-	-	289.6	-	-	20.0	5,800

Table 4.7. continued.

Stratum/population	Yrs	$\overline{N}_{a(arith)}$	$\overline{N}_{a(geom)}$	$\overline{N}_{g(harm)}$	$\hat{C}$	$\hat{T}$ *(95% CI)*	IPkm	$\hat{D}_{dep}$	$\hat{D}_{ssd}$	$\hat{D}_{ssd}$ target	$\overline{N}_{a(arith)}$ target
Van Duzen River	-	-	-	-	-	-	312.2	-	-	20.0	6,200
Larabee Creek	-	-	-	-	-	-	86.4	-	-	30.2	2,600
Dobbyn Creek (s)	-	-	-	-	-	-	47.0	-	-	6-12	280-562
NF Eel River	-	-	-	-	-	-	315.7	-	-	20.0	6,300
MF Eel River	-	-	-	-	-	-	472.4	-	-	20.0	9,400
Upper Mainstem Eel R.	-	-	-	-	-	-	317.5	-	-	20.0	6,400
North-Central Coastal											
Usal Creek[c]	9	88	56	304	NA	0.217 (-0.036, 0.469)	27.5	2.2	3.2	38.4	1,100
Cottaneva Creek[d] (s)	5	64	23	NA	NA	NA	21.9	NA	NA	6-12	129-261
Wages Creek[d]	4	63	33	NA	NA	NA	17.4	NA	NA	39.8	700
Ten Mile River[d]	11	416	231	1,752	NA	0.291 (-0.077, 0.660)	171.1	1.4	2.4	20.0	3,400
Pudding Creek[d] (s)	18	89	56	220	0.72	-0.091 (-0.189, 0.006)	23.9	NA	NA	6-12	141-285
Noyo River[d]	18	387	341	1,520	0.04	0.033 (-0.012, 0.079)	152.8	1.6	2.7	21.0	3,200
Big River[c]	10	541	338	2,239	NA	0.172 (-0.083, 0.427)	255.0	1.2	2.1	20.0	5,100
Albion River[d] (s)	10	50	28	204	NA	-0.029 (-0.410, 0.352)	48.6	NA	NA	6-12	290-581
Big Salmon Creek[d]	4	68	16	NA	NA	NA	18.3	NA	NA	39.7	NA
Central Coastal											
Navarro River[d]	10	399	329	1,557	NA	0.112 (-0.045, 0.269)	387.6	0.6	1.0	20.0	7,800
Elk Creek[d] (s)	3	21	6	NA	NA	NA	34.5	NA	NA	6-12	205-412
Brush Creek[d] (s)	10	9	4	36	NA	-0.060 (-0.420, 0.300)	21.4	NA	NA	6-12	126-255
Garcia River[d]	10	323	270	1,321	NA	0.059 (-0.114, 0.231)	135.4	2.0	2.4	23.4	3,200
Gualala River	-	-	-	-	-	-	396.7	-	-	20.0	7,900

a – Numbers indicate the estimated number of redds within a coho salmon sampling frame; does not encompass the entirety of steelhead spawning timing or space.

b – Numbers indicate sonar-based estimates of both natural-origin and hatchery-origin (currently not considered part of DPS) and includes fish potentially part of the upper Mad River population, which has a separate viability target.

c – Numbers indicate the estimated number of redds within a coho salmon sampling frame; covers entirety of spawning space, but not spawning timing for steelhead.

d – Numbers indicate the estimated number of adults based on fish/redd expansions from life cycle monitoring stations.

Table 4.8. Population information for dependent populations of winter-run NC steelhead or those with only partial population estimates. (s) = supporting population. NA = not available or applicable. Trends shown only for populations where time series is ≥ 8 years, bold indicates significant trend. $N_{a(arith)}$ target refers to recovery target (NMFS 2016b).

Stratum/population	Yrs	$\overline{N}_{a(arith)}$	$\overline{N}_{a(geom)}$	$\overline{N}_{g(harm)}$	$\hat{T}$ (95% CI)	$\overline{N}_{a(arith)}$ target
Northern Coastal						
Freshwater Creek[a]	19	175	153	609	-0.013 (-0.061, 0.036)	NA
Lower Mainstem Eel tribs (s)	-	-	-	-	-	996-1,995
Howe Creek (s)	-	-	-	-	-	81-165
Guthrie Creek (s)	-	-	-	-	-	53-108
Oil Creek (s)	-	-	-	-	-	62-125
McNutt Gulch (s)	-	-	-	-	-	66-134
Spanish Creek (s)	-	-	-	-	-	9-21
Big Creek (s)	-	-	-	-	-	21-44
Big Flat Creek (s)	-	-	-	-	-	33-69
Shipman Creek (s)	-	-	-	-	-	12-26
Telegraph Creek (s)	-	-	-	-	-	30-62
Jackass Creek (s)	-	-	-	-	-	39-81
North Mtn. Interior						
Van Arsdale Station[b]	37	361	216	658	-0.003 (-0.035, 0.030)	NA
	16	325	267	1,123	-0.016 (-0.095, 0.062)	NA
Lower Interior						
Garcia Creek (s)	-	-	-	-	-	83-167
Soda Creek (s)	-	-	-	-	-	92-186
North-Central Coastal						
Juan Creek[c]	3	27	23	NA	NA	NA
DeHaven Creek[c]	1	0	NA	NA	NA	NA
SF Noyo River[d]	20	75	65	291	-0.012 (-0.059, 0.035)	NA
Hare Creek[c]	9	51	14	NA	NA	NA
Caspar Creek[b]	18	46	32	129	-0.072 (-0.152, 0.008)	500
Little River[c]	18	16	12	52	-0.064 (-0.132, 0.003)	NA
Central Coastal						
NF Navarro River[e]	7	333	295	1245	NA	NA
Greenwood Creek[c]	4	15	8	NA	NA	NA
Alder Creek[c]	1	11	NA	NA	NA	NA
Schooner Gulch (s)	1	0	NA	NA	NA	44-90

a – Numbers indicate the estimated number of adults based on mark-recapture estimate from life cycle monitoring station.

b – Numbers based on counts of natural-origin fish at Van Arsdale Station fish trap; partial composite of Upper Eel River and Soda Creek populations.

c – Populations monitored but not considered essential or supporting in Recovery Plan (NMFS 2016b).

d – Partial population estimate based on fish/redd expansion from life cycle monitoring station.

e – Partial population estimate based on AUC methods at life cycle monitoring station.

Table 4.9. Population information for summer-run NC steelhead populations. NA = not available or applicable. Trends shown only for populations where time series is ≥ 8 years; bold indicates significant trend. $N_{g(harm)}$ target refers to recovery target (NMFS 2016b).

Stratum/population	Yrs	$\overline{N}_{a(arith)}$	$\overline{N}_{a(geom)}$	$\overline{N}_{g(harm)}$	$\hat{T}$ (95% CI)	$\overline{N}_{a(arith)}$ target
Northern Coastal						
Redwood Creek[a]	39	9	6	26	-0.011 (-0.036, 0.014)	2500
	16	9	7	33	**-0.087 (-0.138, -0.037)**	
Mad River[a,b]	7	220	204	641	NA	2500
SF Eel River	-	-	-	-	-	2500
Mattole River[c]	24	22	20	84	NA	2500
	15	25	21	114	-0.023 (-0.092, 0.046)	
North Mtn. Interior						
Van Duzen River[d]	9	121	109	452	-0.077 (-0.217, 0.063)	2500
Larabee Creek	-	-	-	-	-	2500
NF Eel River	-	-	-	-	-	2500
Up-Mid Mainstem Eel R.	-	-	-	-	-	2500
MF Eel River[d]	53	753	667	2772	-0.007 (-0.016, 0.002)	2500
	16	623	584	2037	-0.038 (-0.077, 0.002)	

a – The Redwood Creek and Mad River summer steelhead populations contribute to both the Northern Coastal and North Mountain Interior diversity strata. Estimates are from dive counts of standardized reaches and thus represent only a partial population estimate.

b – Prior viability assessments (Spence 2016) have reported the combined total of adult summer steelhead and half-pounders. The above numbers include only those fish classified as adults.

c – The Mattole River surveys cover only a portion of available rearing habitat and are thus a partial population estimate. Total stream miles surveyed is inconsistent from year to year; thus, calculation of trends was deemed inappropriate.

d – Summer steelhead surveys in the Van Duzen and Middle Fork Eel rivers likely cover most of the available summer holding pools for these populations.

Table 4.10. Viability metrics for independent populations of steelhead in the CCC steelhead DPS. (s) = supporting population. classified as "supporting." NA indicates not available or applicable. Trends shown only for populations where time series is ≥ 8 years; bold indicates significant trend. IPkm includes only habitats that are currently accessible. $N_{a(arith)}$ target refers to recovery target (NMFS 2016b).

Stratum/population	Yrs	$\overline{N}_{a(arith)}$	$\overline{N}_{a(geom)}$	$\overline{N}_{g(harm)}$	$\hat{C}$	$\hat{T}$ (95% CI)	IPkm	$\hat{D}_{dep}$	$\hat{D}_{ssd}$	$\hat{D}_{ssd}$ target	$\overline{N}_{a(arith)}$ target
North Coastal											
Austin Creek	-	-	-	-	-	-	95.1	-	-	29.0	2,800
Green Valley Creek	-	-	-	-	-	-	24.9	-	-	38.8	1,000
Salmon Creek	-	-	-	-	-	-	33.6	-	-	37.6	1,300
Estero Americano Cr (s)	-	-	-	-	-	-	35.4	-	-	6-12	210-423
Walker Creek	-	-	-	-	-	-	54.2	-	-	34.7	1,900
Lagunitas Creek[a]	18	147	113	NA	NA	NA	53.3	NA	NA	34.8	1,900
Interior											
Mark West Creek	-	-	-	-	-	-	164.2	-	-	20.0	3,300
Dry Creek	-	-	-	-	-	-	116.7	-	-	26.0	3,000
Maacama Creek	-	-	-	-	-	-	76.2	-	-	31.6	2,400
Upper Russian River	-	-	-	-	-	-	423.9	-	-	20.0	8,500
Coastal SF Bay											
Corte Madera Creek	-	-	-	-	-	-	19.8	-	-	39.5	800
Novato Creek	-	-	-	-	-	-	28.3	-	-	38.3	1,100
Guadalupe River	-	-	-	-	-	-	51.9	-	-	35.0	1,800
Stevens Creek	-	-	-	-	-	-	22.9	-	-	39.0	900
San Francisquito Creek	-	-	-	-	-	-	35.5	-	-	37.3	1,300
San Mateo Creek (s)	-	-	-	-	-	-	6.3	-	-	6-12	36-74

Table 4.10. continued.

Stratum/population	Yrs	$\overline{N}_{a(arith)}$	$\overline{N}_{a(geom)}$	$\overline{N}_{g(harm)}$	$\hat{C}$	$\hat{T}$ *(95% CI)*	IPkm	$\hat{D}_{dep}$	$\hat{D}_{ssd}$	$\hat{D}_{ssd}$ target	$\overline{N}_{a(arith)}$ target
Interior SF Bay											
Petaluma River	-	-	-	-	-	-	64.3	-	-	33.3	2,100
Sonoma Creek	-	-	-	-	-	-	129.0	-	-	24.3	3,100
Napa River	-	-	-	-	-	-	233.9	-	-	20.0	4,700
Green Valley/Suisun Cr	-	-	-	-	-	-	64.3	-	-	33.3	2,100
San Pablo Creek (s)	-	-	-	-	-	-	8.5	-	-	6-12	49-100
San Leandro Creek (s)	-	-	-	-	-	-	5.5	-	-	6-12	31-64
San Lorenzo Creek (s)	-	-	-	-	-	-	18.6	-	-	6-12	110-221
Alameda Creek	-	-	-	-	-	-	108.7	-	-	27.1	2,900
Coyote Creek	-	-	-	-	-	-	109.3	-	-	27.0	3,000
Santa Cruz Mtns											
Pilarcitos Creek[b]	1	NA	NA	NA	NA	NA	28.5	NA	NA	38.3	1,100
San Gregorio Creek[b]	5	NA	NA	NA	NA	NA	46.6	NA	NA	35.7	1,700
Pescadero Creek[b]	7	NA	NA	NA	NA	NA	66.1	NA	NA	33.0	2,200
Waddell Creek[b]	5	NA	NA	NA	NA	NA	10.6	NA	NA	40.0	500
Scott Creek[c]	16	205	168	448	0.54	-0.060 (-0.130, 0.010)	16.4	5.4	10.4	39.9	700
Laguna Creek (s)	-	-	-	-	-	-	4.5	-	-	6-12	25-52
San Lorenzo River[b]	7	NA	NA	NA	NA	NA	146.2	NA	NA	21.9	3,200
Soquel Creek[b]	4	NA	NA	NA	NA	NA	52.1	NA	NA	35.0	1,800
Aptos Creek[b]	3	NA	NA	NA	NA	NA	25.0	NA	NA	38.7	1,000

a – Numbers indicate the estimated minimum census of redds within selected reaches; temporal extent of surveys varies substantially among years.
b – Although surveys have been conducted, uncertainty in methods and lack of consistency in reported metrics precludes calculation of population statistics (see text for details).
c – Mark-recapture estimates from Scott Creek life cycle monitoring station.

Table 4.11. Viability metrics and abundance targets for dependent populations of steelhead in the CCC steelhead DPS. NA indicates not available or applicable. Trends shown only for populations where time series is ≥ 8 years; bold indicates significant trend. $N_{a(arith)}$ targets are recovery targets identified in NMFS (2016).

Stratum/population	Years	$\overline{N}_{a(arith)}$	$\overline{N}_{a(geom)}$	$\overline{N}_{g(harm)}$	$\hat{T}$ (95% CI)	$\overline{N}_{a(arith)}$ target
North Coastal						
Dutch Bill Creek[a] (s)	3	NA	NA	NA	NA	77-156
Freezeout Creek[a] (s)	3	NA	NA	NA	NA	6-14
Hulbert Creek[a] (s)	3	NA	NA	NA	NA	59-120
Porter Creek[a] (s)	3	NA	NA	NA	NA	60-122
Sheephouse Creek[a] (s)	3	NA	NA	NA	NA	21-44
Willow Creek[a] (s)	3	NA	NA	NA	NA	46-94
Drakes Bay tribs (s)	-	-	-	-	-	NA
Pine Gulch[b] (s)	18	11	NA	NA	NA	56-114
Redwood Creek[b] (s)	22	9	NA	NA	NA	38-78
Interior						
Crocker Creek[a] (s)	3	NA	NA	NA	NA	25-52
Gill Creek[a] (s)	3	NA	NA	NA	NA	41-84
Miller Cr Russian[a] (s)	3	NA	NA	NA	NA	17-35
Sausal Creek[a] (s)	3	NA	NA	NA	NA	65-131
Coastal SF Bay						
Arroyo Corte Madera Del Presidio (s)	-	-	-	-	-	39-81
Miller Creek (Marin) (s)	-	-	-	-	-	17-35
Interior SF Bay						
Codornices Creek (s)	-	-	-	-	-	NA
Pinole Creek (s)	-	-	-	-	-	NA
Wildcat Creek (s)	-	-	-	-	-	NA
Santa Cruz Mtn						
San Pedro Creek[c] (s)	2	NA	NA	NA	NA	NA
Tunitas Creek (s)	-	-	-	-	-	62-126
Gazos Creek[c] (s)	7	NA	NA	NA	NA	73-148
San Vicente Creek[c] (s)	7	NA	NA	NA	NA	32-66

a – Watershed part of Russian River monitoring program, but population-specific estimates are not currently reported.
b – Estimates are redd counts; surveys do not cover all potential habitat; temporal extent of surveys varies substantially among years.
c – Although surveys have been conducted, uncertainty in methods and lack of consistency in reported metrics precludes calculation of population statistics (see text for details).

Appendix A. Sources of data used in assessment of status of populations of salmon and steelhead in the NCCC Recovery Domain.

Population/Watershed	CCC coho salmon	CC Chinook salmon	NC steelhead	CCC Steelhead	Sources
Redwood Creek (Humboldt)		X	X		CDFW 2020
			X (summer)		Redwood Nat'l Park 2018; Max, 2019; K. Max, NPS, unpubl. data (2020)
Mad River		X	X		CDFW 2018, 2020, 2020b, 2021; Sparkman 2018
			X (summer)		Mad River Alliance 2020
Humboldt Bay			X		CDFW 2020
Freshwater Creek		X	X		CDFW 2020
Eel River					
S. FK. Eel River		X	X		CDFW 2020; California Trout & CDFW 2020
Van Duzen River			X (summer)		Thompson 2018, 2019
Van Arsdale Station		X	X		CDFW 2020
Middle Fork Eel River			X (summer)		S. Harris, CDFW, unpub. data (2020)
Mattole River		X	X		CDFW 2020
			X (summer)		N. Queener, Mattole Salmon Group, unpub. data (2020)
Usal Creek	X		X		CDFW 2020
Cottaneva Creek	X		X		CDFW 2020
Juan Creek	X		X		CDFW 2020
DeHaven Creek	X		X		CDFW 2020
Wages Creek	X		X		CDFW 2020
Ten Mile River	X	X	X		CDFW 2020
Pudding Creek	X		X		CDFW 2020
Noyo River	X	X	X		CDFW 2020
S. Fk. Noyo River	X		X		CDFW 2020
Hare Creek	X		X		CDFW 2020
Caspar Creek	X		X		CDFW 2020
Big River	X	X	X		CDFW 2020
Little River	X		X		CDFW 2020
Albion River	X	X	X		CDFW 2020
Big Salmon Creek	X		X		CDFW 2020
Navarro River	X	X	X		CDFW 2020
N. Fk. Navarro River	X		X		CDFW 2020
Greenwood Creek	X		X		CDFW 2020

Appendix A. continued.

Population/Watershed	CCC coho salmon	CC Chinook salmon	NC steelhead	CCC steelhead	Sources
Elk Creek	X		X		CDFW 2020
Alder Creek	X		X		CDFW 2020
Brush Creek	X		X		CDFW 2020
Garcia River	X	X	X		CDFW 2020
Schooner Gulch	X		X		CDFW 2020
Russian River	X	X		X	Obedzinski 2020; Pacific Fishery Management Council 2020b
Salmon Creek	X				M. Kittel, CDFW, unpublished data
Walker Creek	X				M. Kittel, CDFW, unpublished data
Lagunitas Creek	X				CDFW 2020
Pine Gulch	X			X	CDFW 2020
Redwood Creek (Marin)	X			X	CDFW 2020
San Francisco Bay					
Napa River				X	Napa County RCD 2018, 2019
Petaluma River				X	Robbins et al. 2015; Watland et al. 2016, 2017; Hubacker 2020
Pinole Creek				X	EBMUD 2017, 2018, 2019, 2020
Alameda Creek				X	SFPUC 2021
Coyote Creek				X	Smith 2020b; Valley Water 2020b, 2021
Guadalupe River				X	Leicester & Smith 2015, 2016; Cochran 2017; Valley Water 2018, 2019
San Mateo Creek				X	SFPUC and Stillwater Sciences 2016, 2017, 2018 2019
Stevens Creek				X	Smith 2020
San Pedro Creek				X	Jankovitz 2012, 2013[a]
Pilarcitos Creek	X			X	Jankovitz 2012, 2013[a]
San Gregorio Creek	X			X	CDFW 2020; Goin 2015a,b
Pescadero Creek	X			X	CDFW 2020; Jankovitz 2012, 2013[a]
Gazos Creek	X			X	CDFW 2020; Goin 2015a,b
Waddell Creek	X			X	CDFW 2020; Jankovitz 2012[a]; Goin 2015a
Scott Creek	X			X	CDFW 2020
San Vicente Creek	X			X	CDFW 2020; Goin 2015a
San Lorenzo River	X			X	CDFW 2020; Jankovitz 2012, 2013[a]
Soquel Creek	X			X	CDFW 2020; Goin 2015b
Aptos Creek	X			X	CDFW 2020; Jankovitz 2012[a]

[a] – Reports undated but presumed to be 2012 and 2013.

5 Central Valley Recovery Domain

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5.1 Sacramento River Winter-run Chinook Salmon ESU

ESU Delineation

The Sacramento River Winter-run Chinook salmon (SRWRC) ESU includes winter-run Chinook salmon spawning in the mainstem Sacramento River downstream of Keswick Dam and the Livingston Stone National Fish Hatchery (LSNFH) supplementation and captive broodstock programs (85 FR 81822; December 17, 2020). Within the established ESU delineations, new efforts were initiated in 2017 to establish a viable, self-sustaining, and locally adapted population of winter-run Chinook salmon in Battle Creek to add to the spatial diversity (i.e., spatial structure) and abundance of the SRWRC ESU. No new information suggests that the delineation of this ESU should change or that its status as an ESU should change.

Summary of Previous Assessments

Good et al. (2005) concluded that the status of the SRWRC ESU was endangered primarily due to a lack of spatial diversity in this ESU. The major concerns were that SRWRC ESU exist as a single population, which is restricted to spawn in the Sacramento River downstream of Keswick Dam – an area completely outside of the species' geographical range of historical spawning. The persistence of SRWRC ESU in the Sacramento River is reliant on a managed supply of cold water from Shasta Reservoir, leaving it vulnerable to impacts from catastrophic events including droughts. Using data through 2014, Johnson and Lindley (2016) found that the mainstem Sacramento River population was at moderate risk of extinction. The ESU as a whole could not be considered viable because there is only one naturally-spawning population, and it is not spawning within the range of its historical spawning habitat. An emerging concern was rising levels of LSNFH-origin fish spawning in natural areas and potential impacts from drought conditions. The average contribution of hatchery-origin spawners over the previous four generations was 13% (SD= ±8%; t=12 years) with the most recent generation at 20% hatchery influence, placing the population at a moderate risk of extinction as of 2015 (Johnson and Lindley 2016).

Brief Review of Technical Recovery Team Documents

The CV TRT delineated four historical independent populations of SRWRC (Lindley et al., 2004). The spawning areas of three of these historical populations are upstream of the impassable Keswick and Shasta dams. Battle Creek (location of the fourth population) was deemed unsuitable for winter-run Chinook salmon at the time of analysis due to high summer water temperatures. Lindley et al. (2007) developed viability criteria for Central Valley salmonids, summarized in Table 5.1.

New Data and Updated Analyses

The abundance of SRWRC ESU has declined during recent periods of unfavorable ocean conditions (2005–2006) and droughts (2007–2009, 2012–2016; see Recent Trends in Marine and Terrestrial Environments and Their Likely Influence on Pacific Salmonids in California and Southern Oregon section). Temperature conditions during egg development and fry emergence were suboptimal over the duration of SRWRC rearing in 2014 and 2015 reaching lethal levels in both years due to reduced cold-water releases from Shasta Reservoir for this life stage. The egg-to-fry survival estimate for brood year 2014 is 5%, which is a significant departure from the average of 26.4% (coefficient of variation = 37.9%) for brood years 2002–2012 measured at RBDD (Poytress et al. 2014; Johnson et al. 2017). Unusually warm temperatures in both freshwater and ocean ecosystems likely contributed to the low numbers of natural-origin adults observed in 2017 and 2018. Two consecutive years of poor returns increased the vulnerability of the overall population. Yet, water year 2017, which was one of the wettest years on record, may have contributed to the high survival of SRWRC, especially SRWRC spawning in natural areas observed in the 2019 returns (cohort replacement rate = 5.1; Killam 2020).

Since the 2015 viability assessment, routine escapement data have continued to be collected allowing viability statistics to be updated (Azat 2020; Table 5.2). The Red Bluff Diversion Dam (RBDD) gates were operated in the up/out position during some or all of the winter-run Chinook salmon immigration period since 2001 and were removed in 2012 to provide unimpaired salmon passage year-round, an action recommended in the recovery plan (b). These modifications also changed the ability to count SRWRC adults at the RBDD fish ladders (Williams et al. 2016). Population estimates from 2001 to present are derived exclusively from mark-recapture estimates from the carcass survey (Figure 5.1).

Like many other populations of Chinook salmon in the Central Valley, SRWRC have declined in abundance since 2005 with recent decadal lows of 795 of in-river spawners in 2017 (Table 5.2, Figure 5.1). Escapement in 2017 represents the second lowest run since the construction and operation of the LSNFH conservation hatchery in 1997 (Figure 5.1). Escapement improved in 2018 and 2019 such that both the current total population size (sum of last three years (2017–2019); N: LSNFH = 540, Sacramento River = 11,106) and mean population sizes ($\hat{S}$: LSNFH = 180, Sacramento River = 3,702) satisfy the low-risk criterion (N > 2500).

The point estimate for the 10-year trend in run size is 0.08, suggesting an increase of 8% per year, primarily bolstered by the large escapement in 2019 (run size = 7,853, Figure 5.1). However, the slope is not different from zero (0.00). The maximum year-to-year decline in population size has reached 75.3%, an increase from the previous two viability assessments (67% in 2015 and 38% in 2010; Williams et al. 2011; Johnson and Lindley 2016). While, the percent decline does not exceed the catastrophic decline criteria (>90% decline in one generation nor annual run size < 500 spawners, Lindley et al. 2007), the drought had a biologically significant effect on annual run sizes for natural-origin spawners in 2017 and 2018 (153 and 461 individuals) placing the population at a moderate risk of extinction.

After the first year of the four-year drought, LSNFH increased the number of adults used in the brood stock from 120 adults (when adult escapement was expected to be greater than 800 individuals) to 164, 388, and 257 in 2013–2015, respectively (Figure 5.1). The primary role of the conservation hatchery is to prevent extinction of SRWRC and they released approximately three times the usual number of juveniles from the hatchery during the drought years. This resulted in a significant increase in the proportion of hatchery fish that returned to spawn (>80%) in 2017 and 2018. The numbers of natural-origin spawners in 2017 and 2018 were low (153 and 461 individuals), indicating the significant contribution of LSNFH hatchery-origin fish during years of high in-river mortality.

The recently observed levels of hatchery influence over the last generation highlights the dependency on the hatchery. The numbers of hatchery-origin fish that returned to spawn prevented a precipitous decline designation that would have been triggered since fewer than 500 adults would have returned without the supplementation from LSNFH. Since both hatchery-origin and natural-origin returns are included in the in-river population viability metrics, the population size, population decline, and catastrophic decline all meet the low extinction risk criteria. However, the percentage of hatchery spawners over the last four generations were greater than 30%, placing the population at a high risk of extinction. This continues along a worsening trajectory, as the 'low-extinction' criteria was met in the 2010 viability assessment and the increased hatchery influence placed the stock at a 'moderate risk' in the 2015 viability assessment with the population now at a 'high risk' of extinction (Williams et al. 2011; Johnson and Lindley 2016; Figure 5.3). Prior to 2005, the proportion of LSNFH-origin spawners in the river was between 5% to 10%, consistent with guidelines from the Hatchery Scientific Review Group for conservation hatcheries (Figure 5.2; California HSRG 2012). The long-term influence of the hatchery over seven generations (e.g., 21 years) is now at 20% (Figure 5.3 and Table 5.3). The decrease in hatchery reliance in the 2019 returns and the overall return of a larger number of natural-origin adults (run = 5,000 individuals) indicates there is potential for naturally spawning SRWRC to rebuild during periods of favorable environmental conditions (Table 5.3).

Spatial structure and diversity are central tenants to the viability of populations and ESUs. The lack of population redundancy in SRWRC ESU is the primary factor contributing to its high extinction risk. The Battle Creek reintroduction efforts initiated in 2017 mark a significant milestone towards the goal of establishing a second winter-run Chinook salmon population (ICF 2016; U.S. Fish and Wildlife Service 2020a). The return of 95 adults from this first phase is promising.

Spatial structure also promotes life-history diversity which has been shown to improve the resilience of salmon populations (Schindler et al. 2010). Diverse habitats provide variation in localized temperature and food resources that influences growth and phenotypic diversity (size and timing of outmigration) in salmon populations. Recent work by Phillis et al. (2018) suggests that SRWRC ESU may rely on more diverse rearing habitats than previously considered when delineating critical habitat for SRWRC (NMFS 1993; Phillis et al. 2018). Prior to this research, modern SRWRC were thought to spawn and rear exclusively in the mainstem Sacramento River and adjacent floodplains (NMFS 1993). This work highlights the role of intermittent and perennial Sacramento River tributaries and Delta for juvenile rearing and survival (Phillis et al. 2018). Indeed, these additional habitats collectively contribute to the majority of rearing habitats used by successful SRWRC salmon that survive to adulthood (Phillis et al. 2018). These data along with previous empirical studies that document SRWRC salmon with accelerated growth in several intermittent streams suggest a reevaluation of the currently defined critical habitat for SRWRC ESU may be warranted (Maslin et al. 1997; Maslin et al. 1998; Limm and Marchetti 2009).

Harvest Impacts[11]

Sacramento River winter-run Chinook salmon have a more southerly ocean distribution relative to other California Chinook salmon populations and are primarily impacted by fisheries south of Point Arena, California. SRWRC age-3 ocean fishery impact rates for the region south of Point Arena, an approximation of the exploitation rate, are estimated annually using cohort reconstruction methods (O'Farrell et al. 2012). Age-3 impact rates have remained relatively stable, averaging 15.9% (Figure 5.4). Fisheries in 2008 and 2009 were closed south of Point Arena owing to the collapse of the Sacramento River fall-run Chinook salmon, and sufficient data do not exist to estimate the impact rate in 2010 and 2015. If years 2008-2010 and 2015 are omitted, the average age-3 impact rate is 18.0% (Pacific Fishery Management Council 2015).

There have been several layers of ocean salmon fishery regulations implemented for the protection of SRWRC beginning in the early 1990s. For example, a substantial portion of the SRWRC ocean harvest impacts once occurred in February and March recreational fisheries south of Point Arena, but fisheries at that time of the year have been closed since the early 2000s. O'Farrell and Satterthwaite (2015) hindcasted SRWRC age-3 ocean impact rates back to 1978, extending the impact rate time series beyond the range of years where direct estimation is possible. Their results suggest that there were substantial reductions in ocean impact rates prior to 2000 and that the highest impact rates occurred in a period between the mid-1980s and late-1990s.

The Reasonable and Prudent Alternative (RPA) from the 2010 biological opinion on salmon ocean harvest fishery (NMFS 2010) specified that new fishery management objectives must be established. The implementation of the RPA resulted in the development of an impact rate control rule that was first used for ocean fishery management in 2012. That impact rate control rule specified reductions in the age-3 ocean impact rate south of Point Arena when the geometric mean number of spawners from the previous three years is reduced (NMFS 2010). The limits to

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[11] Michael O'Farrell prepared this section on harvest impacts.

the impact rate imposed by the harvest control rule was an additional control on ocean fisheries which still included previously existing constraints on fishery opening and closing dates and minimum size limits south of Point Arena.

A more recent biological opinion on the salmon ocean harvest fishery (NMFS 2018) specified a new SRWRC impact rate control rule for use in managing ocean fisheries (Figure 5.5). This control rule, first implemented in 2018, specifies the maximum allowable age-3 impact rate south of Point Arena as a function of forecasted abundance, defined as the expected age-3 SRWRC escapement in the absence of fisheries. The use of an abundance forecast rather than a mean of past abundance levels to set allowable impact rates is a key feature of the current control rule, enabling fisheries management to be more responsive to recent conditions (e.g., low juvenile abundance and survival rates associated with drought). As before, the constraints on fishery opening and closing dates and minimum size limits south of Point Arena remain in place.

Between 2012 and 2020, SRWRC harvest control rules have specified maximum forecast impact rates ranging from 12.9% to 20.0% (Pacific Fishery Management Council 2020). What little SRWRC freshwater harvest that existed historically was nearly eliminated beginning in 2002, when Sacramento Basin Chinook salmon fishery season openings were adjusted so that there would be little temporal overlap with the SRWRC spawning migration and spawning period. Since that time there have been very few coded-wire tag (CWT) recoveries in Sacramento Basin river fisheries.

In summary, the available information indicates that the level of SRWRC fishery impacts have not changed appreciably since the 2016 salmon and steelhead status review update (NMFS 2016a).

Summary and Conclusions

The LSNFH improved SRWRC ESU viability demographically and genetically through increasing population abundance and through the maintenance of a larger effective population size during years of extremely poor in-river survival. Yet, reliance on production from LSNFH is increasing and remains a cause for serious concern to the long-term viability and genetic integrity of the population and the ESU (Figure 5.3). The viability of the SRWRC ESU will be improved by re-establishing winter-run Chinook salmon in their historical spawning and rearing habitat. Projects to reintroduce SRWRC into Battle Creek are on-going while reintroduction to historical habitats upstream of Shasta Reservoir are in the planning and early implementation phases. In the summer of 2020, juvenile salmon were observed in Battle Creek indicating the first successful spawning of winter-run Chinook salmon in Battle Creek in over 100 years. Further, assessments of habitat conditions in the McCloud River and achievable Chinook salmon smolt survival (70%) through the reservoir to Shasta Forebay show promise (Hansen et al. 2017; Hansen et al. 2018). If successful, the establishment of multiple self-sustaining populations of SRWRC would significantly benefit SRWRC. Genetic management plans will be critical for conserving the long-term genetic integrity of the SRWRC, the success of the reintroduction efforts, as well as achieving a low-extinction risk for the population downstream of the barrier.

Until additional SRWRC populations are established, the ESU will remain in the “High” biological extinction risk category. The overall viability of the SRWRC ESU has continued to decline since the 2015 viability assessment (Johnson and Lindley 2016), with the single spawning population on the mainstem Sacramento River no longer at a low/moderate risk of extinction (Table 5.4).

5.2 Central Valley Spring-run Chinook Salmon ESU

ESU Delineation

The Central Valley spring-run Chinook salmon (CVSRC) ESU includes spring-run Chinook salmon populations spawning in the Sacramento River and its tributaries and spring-run Chinook salmon in the Feather River Hatchery (FRH). The San Joaquin River watershed and Delta are excluded as critical habitat and San Joaquin basin populations are considered extirpated (NMFS 2013). However, information on the presence of Chinook salmon exhibiting a spring-run phenotype in San Joaquin River tributaries is provided and may represent passive reestablishment of CVSRC that are not a part of the active San Joaquin River Reintroduction effort. Thus, there is value in continuing to monitor these populations to evaluate the extent to which populations in the San Joaquin River tributaries may warrant inclusion in the ESU in future assessments. In 2014, FRH brood stock was used to actively reintroduce CVSRC into the mainstem San Joaquin River as an ESA 10(j) experimental population (NMFS 2013). Several juveniles successfully survived to adulthood and returned to spawn in 2019. No new information suggests that the delineation of the CVSRC ESU should change at this time.

Summary of Previous Assessments

Good et al. (2005) found that the CVSRC was likely to become endangered. The major concerns of the Biological Review Team (BRT) were the low diversity, poor spatial structure, and low abundance of this ESU. As part of the 2010 viability assessment, and using the criteria in Table 5.1, declines in abundance placed Mill Creek and Deer Creek populations at a high risk of extinction due to their rates of decline, and in the case of Deer Creek the level of escapement. Yet, only five years later using data through 2014, Johnson and Lindley (2016) found that the Mill Creek, Deer Creek, and Butte Creek populations were at or near low risk of extinction. Further, the performance of CVSRC on Clear and Battle creeks showed promise in reestablishing viable populations in the Northwestern California and Basalt and Porous Lava diversity groups. All populations in the Basalt and Porous Lava and the Southern Sierra Nevada groups were deemed extirpated, and only a few dependent populations persist in the Coast Range group. The ESU as a whole was not considered viable because there were no extant populations in these three diversity groups. In addition, Mill, Deer, and Butte creeks are in close proximity to each other, decreasing the independence of their extinction risks due to catastrophic disturbance (Lindley et al. 2007). However, the viability of extant populations fluctuates greatly.

Brief Review of TRT Documents

The CV TRT delineated four diversity groups, 18 or 19 historical independent populations of CVSRC (depending on the classification of Mill Creek and Deer Creek populations), along with a number of smaller dependent populations (Lindley et al. 2004). The primary criteria used to identify independent from dependent populations were data on historical accounts of the presence of spring-run Chinook salmon, isolation from other populations that exceeded a critical dispersal distance (>50 km), minimum basin size (500 km^2), and genetic information (Lindley et al. 2004).

The TRT considered multiple lines of evidence to evaluate the extent to which Mill and Deer creeks were historically independent from one another or a single panmictic population and reached no definitive conclusion. The primary evidence supporting the panmictic hypothesis included similar genetic structure, life history, and watersheds with remarkably similar habitat attributes (Lindley et al. 2004). Evidence supporting the independent designation of Mill and Deer creeks included exceeding the critical dispersal distance (89 km >50 km) and contemporary spawning abundance trends that were deemed weakly asynchronous. The TRT did conclude that CVSRC in Mill and Deer creeks are currently independent from other CVSRC populations and together with populations on Butte Creek could serve as salmon strongholds in the Northern Sierra Nevada diversity group.

New Data and Updated Analyses

Lindley et al. (2007) provide criteria to assess the level of risk of extinction of Central Valley salmon based on population size, recent population decline, occurrences of catastrophes within the last 10 years that could cause sudden shifts from a low risk state to a higher one, and the impacts of hatchery influence (Table 5.1). Figure 5.6 shows the escapement of CVSRC to various areas of the Central Valley, and Table 5.5 shows abundance and trend statistics related to viability criteria. All historically independent populations remaining (Battle, Deer, Mill, and Butte creeks) show substantially lower total population sizes (N) and mean escapement ($\hat{S}$) than the previous viability assessment in 2015. The rate of decline over the past decade coupled with low abundances place Battle, Deer, and Mill creek populations at a high risk of extinction. The Butte Creek population remains at a low risk of extinction despite having a recent decline of 76% in a single generation. All populations experienced recent declines in one generation that exceeded previous year maximums, with the exception of Deer and Antelope creeks whose largest declines in a single generation (84% and 88%), occurred at the beginning of the decadal time series (Table 5.5). Butte Creek's total population size is 17,740, which is double what was estimated in 2010 and remains by far the most abundant CVSRC population (Table 5.5). While data for the Yuba River was included in the 2015 viability assessment and showed a low extinction risk based on population size, no data were provided for escapement years 2015–2019 and therefore omitted from this assessment. The Yuba River Management Team is in the process of revising their statistical analysis previously used to refine the demarcation date that will separate spring- and fall-run Chinook salmon estimates to be retrospectively used for future assessments (P. Bratovich, HDR, personal communication; Lower Yuba River Accord River Management Team 2013).

All populations of CVSRC are still exhibiting declines in population size over time, with the exception of two dependent populations — Antelope and Clear creeks that have positive point estimates of population growth (Table 5.5). In 2015, CVSRC showed strong signs of repopulating Battle Creek, home to a historical independent population in the Basalt and Porous Lava diversity group that had been extirpated for many decades. Current viability metrics show a significant declining trend (23% decline per year) and low population size (N<250) for the

Battle Creek spring-run Chinook salmon population placing it at a high extinction risk. Similarly, the CVSRC population in Clear Creek, previously identified as increasing in abundance, has experienced recent declines in population size (N=136) down from N=822 in 2015, placing it at a high risk of extinction. Mill Creek and Deer Creek spring-run Chinook salmon populations reached low population sizes (N=590 and N=956, respectively) placing them at a moderate risk of extinction. Yet, the low run sizes in consecutive years for Mill Creek spring-run Chinook salmon following the recent droughts (~150 individuals) and precipitous decline (16% over the decade) places Mill Creek at a high risk of extinction using the criteria in Table 5.1. The highest risk score for any criterion determines the overall extinction risk for a given population. Recent declines of population size in all populations have been substantial and almost qualify as catastrophes under the criteria (>90% decline) with the main independent populations of CVSRC reaching all-time declines over one generation (Battle Creek = 77%, Butte Creek = 76%, Deer Creek = 84%, and Mill Creek = 68%).

Beginning in 2009, estimates of spawning escapement of Upper Sacramento River spring-run Chinook were no longer monitored. Historically, this estimate was derived by the total Red Bluff Diversion Dam (RBDD) counts minus the spring-run numbers in the upper Sacramento tributaries. Beginning in 2009, RBDD gates were partially operated in the up position and in 2012 they were entirely removed and thus spring-run estimates were no longer available. CVSRC on the mainstem Sacramento River are not thought to be numerous, yet in some years, the majority of fish collected in the spring and summer months in the Keswick trap as adults are genetically assigned as non-winter-run Chinook salmon. Based on when they are sampled, they are likely CVSRC. Consideration should be given to the use of genetics to improve our assessments of CVSRC in the Keswick trap sampling as well as the Sacramento River winter- and fall-run carcass survey to quantify CVSRC spawning on the mainstem Sacramento River (Prince et al. 2017; Thompson et al. 2019; Meek et al. 2020). In some years, the Sacramento River mainstem population could be more abundant than the other independent CVSRC populations.

Historical and continued introgression between Feather River spring- and fall-run Chinook salmon ESUs in the breeding program at the FRH compromises the long-term genetic integrity of the spring-run Chinook salmon population on the Feather River and poses a high extinction risk (Hedgecock et al. 2001; California HSRG 2012; Palmer-Zwahlen et al. 2019). Since 2004, spring-run Chinook salmon broodstock have been identified as phenotypic spring run trapped and tagged at the FRH between April 1 and June 30. As a result of this practice, fall run are very effectively excluded from the spring-run broodstock. Additionally, FRH has been using genetic testing of gametes of their fall-run broodstock to ensure spring-run Chinook salmon are excluded. They have effectively implemented practices to reduce introgression between spring and fall run in the hatchery. In the river, large numbers of fall- and spring-run Chinook salmon individuals from the FRH potentially spawn with natural-origin Feather River spring and fall-run Chinook salmon (Palmer-Zwahlen et al. 2019).

The majority of the FRH spring-run Chinook salmon broodstock and in-river spawning population on the Feather River were produced in the hatchery (Kormos and Palmer-Zwahlen 2012; Palmer-Zwahlen and Kormos 2013). The proportion of natural-origin fish in the broodstock is estimated to be 2% in 2015 (Palmer-Zwahlen et al. 2019). Thus, the minimum criteria of >10% of natural-origin fish in the FRH spring-run Chinook salmon broodstock is not being met annually. The lack of naturally produced fish can disrupt the balance of adaptive gene flow between hatchery and natural-area spawning populations (California HSRG 2012). The proportion of hatchery-origin spring- or fall-run Chinook salmon contributing to the natural area spawning spring-run Chinook salmon population on the Feather River remains unknown due to overlap in the spring- and fall-run spawn timing. However, the hatchery component is likely to be high. For example, 83% of spawners in the 2015 spring-/fall-run carcass survey were estimated to be from the FRH respectively (Palmer-Zwahlen et al. 2019).

Genetic studies suggest that hybridization between FRH spring-run and other Chinook salmon run types (winter-, spring-, and late fall-) in other streams has not occurred, where evaluated. For example, if FRH CVSRC have been straying extensively, the effect is not apparent in the genetic structure described by microsatellite markers for CVSRC runs in Mill, Deer and Butte creeks, or on winter- and late fall-runs of Chinook salmon that spawn in the mainstem Sacramento River (Banks et al. 2000). These findings are consistent with the generally low stray rates estimated by recovery of CWTs (Kormos and Palmer-Zwahlen 2012; Palmer-Zwahlen and Kormos 2013). Yet, there continues to be an increased stray rate associated with hatchery fish that are trucked and released off-site (Huber and Carlson 2015; Palmer-Zwahlen et al. 2019; Sturrock et al. 2019). Indeed, FRH CVSRC adults have been recovered in other Central Valley spring- and fall-run Chinook salmon populations outside of the Feather River. Over 400 FRH spring-run Chinook salmon from fish raised in net pens in the San Francisco Bay strayed as adults and were recovered in the Upper Sacramento River and other natural areas, including Clear Creek, Mill Creek, Deer Creek, and Butte Creek and potentially impacted the genetic integrity of other CVSRC populations (Palmer-Zwahlen et al. 2019). In the past, FRH strays to the Yuba River have been significant, yet in 2015 no FRH CVSRC were recovered in the Yuba River carcass survey (Palmer-Zwahlen et al. 2019). Research suggests that the practice of trucking hatchery fish downstream to the Delta and Bay for release, rather than on-site releases, increases adult straying (Huber and Carlson 2015). Prolonged influx of FRH spring-run Chinook salmon strays to other spring-run Chinook salmon populations even at levels <1% is undesirable and can cause the receiving population to shift to a moderate risk after four generations of such impact (Lindley et al. 2007; Figure 5.3). Beginning in 2014, all FRH spring-run Chinook salmon have been released in the Feather River, likely reducing straying to watersheds outside of the Feather River (California HSRG 2012; Huber and Carlson 2015; Palmer-Zwahlen et al. 2019; Sturrock et al. 2019). Additional information on the incidence of FRH spring-run Chinook salmon straying is desirable to more accurately estimate the extent to which spawning and introgression is occurring between fall- and spring-run Chinook salmon and/or between FRH CVSRC and natural-origin spring-run Chinook salmon outside of the Feather River.

Spatial structure promotes life-history diversity and phenotypic variation that is critical for the long-term persistence of species and populations, especially in highly variable environments. CVSRC express significant diversity in the duration of freshwater rearing (3–15 months) with some juveniles leaving the freshwater as sub-yearlings while others over-summer until they are much larger and migrate as yearlings. Yearlings are difficult to monitor, but have been observed in screw traps on Mill, Deer, and Butte creeks (Johnson and Merrick 2012, Ward and McReynolds 2004). The extent to which the yearling vs. sub-yearling strategies currently function to create population resilience in CVSRC populations is the source of on-going research. Unlike fall-run Chinook salmon that are not occupying the freshwater habitats in the summer, CVSRC need cold water as adults and yearlings in the summer. In order to support the yearling life history, cold over-summer temperatures are required, which are lacking on much of the valley floor. This temperature constraint in low elevation habitats likely restricts the expression and/or success of the yearling strategies to tributaries like Mill and Deer creeks that, if adequate flows remain in the streams after water diversions, retain higher elevation access and cooler summer stream temperatures. Further, juvenile smolt outmigration survival in CVSRC appears to be linked to higher springtime outmigration flows (Notch et al. 2020) which are regularly suppressed during May to store water in Shasta Reservoir for summer agricultural deliveries, Delta water quality, and Sacramento River temperature management (NMFS 2019). For example, survival of tagged smolts from Mill Creek had 8-fold higher survival during the high flows in 2017 (42.3% ± 9.1) than during the 2015 drought (4.9% ± 1.6). Further, there is often a mismatch between the ideal timing and outmigration conditions the smolts experience in Mill and Deer creeks and the poorer conditions in the Sacramento River, which is most pronounced near Tisdale Weir. Current efforts are underway to evaluate the extent to which pulse flows in the Sacramento River during May can improve CVSRC outmigration survival (NMFS 2019).

Successful reestablishment of CVSRC into multiple populations in the Southern Sierra Nevada Group would significantly increase their spatial diversity and decrease extinction risk of the ESU. CVSRC were essentially extirpated from the San Joaquin River after Friant Dam was built in the 1940s, leaving the river dry for 60 miles. For many decades, CVSRC were considered extirpated from the Southern Sierra Nevada diversity group in the San Joaquin River Basin, despite their historical numerical dominance in the Basin (Fry 1961, Fisher 1994). In 2017, the first CVSRC redds were observed in the San Joaquin River restoration area and in 2019, 168 CVSRC carcasses were detected below Friant Dam for the first time in 65 years (NMFS 2020; San Joaquin River Restoration Program 2020b). This is a result of a reintroduction program for CVSRC was initiated in 2014 as part of the San Joaquin River Restoration Program; 54,000 juvenile spring-run Chinook salmon from FRH broodstock were released into the San Joaquin River. This population of CVSRC is designated as an experimental population in accordance with the section 10(j) of the Endangered Species Act allowing the release of threatened CVSRC outside of their current range (NMFS 2013). These fish were confirmed to have originated as juveniles from the Salmon Conservation and Research Facility (SCARF) reintroduction efforts through CWT recoveries (San Joaquin River Restoration Program 2020a). In addition to the active reintroduction of CVSRC below Friant Dam, there have been recent reports of adult Chinook salmon exhibiting typical spring-run life-history characteristics including springtime migration, over-summering in deep pools, spawning in the early fall, and the occurrence of yearling sized juveniles to tributaries of the San Joaquin River including Mokelumne, Stanislaus, and Tuolumne rivers (Franks 2012; Johnson and Lindley 2016). The extent to which these phenotypic spring-run have a similar genetic lineage as other extant spring-run Chinook salmon populations and stray each generation from the Sacramento River Basin remains unknown and is the source of on-going research. It is conceivable that progeny from spring-run adults return to their natal tributaries on the San Joaquin River and thus represent early stages of reestablishing a population and a process trending towards a self-sustaining population.

No criteria exist to assess whether an ESU is at moderate or high risk of extinction. In order to summarize the extinction risk of the entire CVSRC ESU as part of this viability assessment, we first evaluated extinction risk of individual populations based on the demographic risk parameters (abundance, productivity, catastrophic declines, spatial structure and connectivity, diversity, and hatchery impacts) recently exhibited by the ESU. The highest risk score for any of the demographic parameters was used to summarize the overall extinction risk for a given population (Table 5.5). We then compared changes in the extinction risk for populations across the two previous assessments to provide context for the volatility in categorical changes in risk within and among populations. To characterize the uncertainty in determining the overall level of extinction risk facing the CVSRC ESU, we adopted the likelihood point method, often referred to as the FEMAT (Forest Ecosystem Management Assessment Team) method (FEMAT 1993). Each of the members allocated 10 points (votes) among three possible extinction risk categories (low, moderate, and high extinction risk) to reflect their own uncertainty, per established methods (Good et al. 2005; Williams et al. 2013).

In the time since the completion of the assessment which covers data through 2019, new information exists that could be incorporated into evaluating whether threats to the ESU would lessen or worsen the risk of extinction. Since data after 2019 are not formally analyzed here, for the final risk assessment, each member evaluated all the available information only reported here on current demographic status and threats to come to a single overall conclusion on the degree of extinction risk.

Harvest Impacts[12]

Attempts have been made to estimate CVSRC ocean fishery exploitation rates using CWT recoveries from natural origin Butte Creek fish (Grover et al. 2004), but due to the low number of recoveries the uncertainty of these estimates is too high for them to be reliable. Because CVSRC have a relatively broad ocean distribution, generally from central California to Cape Falcon, Oregon, that is similar to that of Sacramento River fall-run Chinook (SRFC) salmon, trends in the SRFC ocean harvest rate may provide a reasonable proxy for trends in the CVSRC ocean harvest rate. While the SFRC ocean harvest rate can provide information on trends in CVSRC fishing mortality, it has been inferred that CVSRC likely experiences lower ocean fishing mortality than SRFC. If maturation rates are similar between CVSRC and SRFC, the ocean exploitation rate on CVSRC would be lower than SRFC in the last year of life because spring-run Chinook salmon escape ocean fisheries in the spring, prior to the most extensive ocean salmon fisheries in summer. Furthermore, CVSRC tend to be smaller at age than SRFC,

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[12] Michael O'Farrell prepared this section on harvest impacts.

which would imply lower age-specific ocean fishery mortality for CVSRC (Myers et al. 1998; Satterthwaite et al. 2012).

Since the last status review (NMFS 2016a), Satterthwaite et al. (2018) reviewed available data for CVSRC and explored assessment and management options. Included in this paper was the suggestion that until CVSRC-specific stock assessments are developed, and exploitation rates can be directly estimated, trends in ocean fishing mortality rates for co-mingling stocks (SRFC, Klamath River fall-run Chinook salmon [KRFC], and Sacramento River winter-run Chinook salmon can provide information on how levels of exploitation have changed for CVSRC. Figure 5.7 displays trends in ocean fishery mortality rates for these stocks. Fishing mortality rates generally peaked in the 1980s and 1990s. Very low fishing mortality rates were estimated for 2008–2010 as fishing opportunity was either eliminated or heavily scaled back due to the collapse of the SRFC stock. Following 2010, fishing mortality rates have returned to levels generally similar to those estimated in the early to mid-2000s, but with notable increases in fishing mortality rates for SRFC and KRFC in 2019.

The level of CVSRC fishery impacts inferred from patterns in SRFC, SRWRC, and KRFC mortality rates is mixed, with recent increases in SRFC and SRWC, but little change for SRWRC. In summary, the available information suggests that ocean fishery impacts have not changed appreciably since the 2016 salmon and steelhead status review update (NMFS 2016a).

Summary and Conclusions

The viability of CVSRC has declined since the 2015 assessment with an increased risk of extinction for all independent CVSRC populations. In fact, Mill, Deer, and Battle creeks changed from low/moderate to a high risk of extinction using one or more viability criteria (Table 5.6). The total abundance of CVSRC for the Sacramento River watershed in 2019 was 26,553, approximately half of the population size in 2014 (N=56,023), and close to the decadal lows of approximately 14,000 which occurred as recently as the last two years (Azat 2020). The Central Valley-wide abundance was driven largely by the annual variation in Butte Creek returns. Butte Creek remains at low extinction risk, yet all viability metrics (except hatchery influence) are trending in a negative direction relative to 2015. The Butte Creek spring-run Chinook salmon population has become the most abundant population of CVSRC ESU in part due to extensive habitat restoration and the accessibility of floodplain habitat in the Butte Sink and Sutter Bypass for juvenile rearing in the majority of years. Most of the dependent spring-run populations in the ESU have been experiencing continued and in some cases drastic declines in abundance. For example, while adults were observed in Big Chico Creek between 2014–2018, they likely didn’t survive to spawn due to high summer temperatures resulting in zeros (0) in the escapement estimates (M. Johnson, CDFW, personal communication; Azat 2020). These results underscore the need for improved passage so that these dependent populations and habitats do not become demographic sinks for CVSRC. No adults were observed in Cottonwood Creek in 2015–2018, reflecting total loss of cohorts produced in those drought years (D. Killam, CDFW, personal communication). Counteracting these developments, CVSRC have repopulated Battle Creek, Clear Creek, and the San Joaquin River where they were once extirpated. These populations continue to fluctuate on an annual basis but previous total population estimates for

Battle and Clear creeks in 2015 suggest they have the potential to establish a self-sustaining population without significant hatchery supplementation (Johnson and Lindley 2016).

Central Valley spring-run Chinook salmon ESU populations have experienced a series of droughts over the past decade. From 2007–2009 and 2012–2016, the Central Valley experienced drought conditions and low river and stream discharges, which are strongly associated with lower survival of Chinook salmon (Michel et al 2015). The impacts of the recent drought series, and warm ocean conditions on the juvenile life stage (see Recent Trends in Marine and Terrestrial Environments and Their Likely Influence on Pacific Salmonids in California and Southern Oregon), seems to have manifested in the low run sizes in 2015–2018 for most CVSRC populations. For example, the recent drought impacted CVSRC adults on Butte Creek, which experienced lethal temperatures in holding habitats during the summer. A large number of adults (903 and 232) were estimated to have died prior to spawning in the 2013 and 2014 drought respectively (Garman 2015). Pre-spawn mortality was also observed during the 2007–2009 drought with an estimate of 1,054 adults dying before spawning in 2008 (Garman 2015). In 2015, late-arriving adults observed in sections of Butte Creek near the city of Chico experienced exceptionally warm June air temperatures, shutdown of a PG&E flume, and a corresponding fish mortality event (Garman 2015). These conditions likely influenced juvenile production and low adult returns in 2015–2018. Fortunately, the favorable hydroclimatic conditions in 2017 appear to have bolstered returns on Butte Creek to pre-drought run sizes of approximately 15,000 adults.

Current introgression between fall- and spring-run Chinook salmon in the FRH breeding program and straying of FRH spring-run Chinook salmon to other spring-run populations where genetic introgression would be possible is unfavorable and reduces population viability. . However, beginning in 2014, and expected to continue, the FRH has begun releasing spring-run production into the Feather River rather than releasing in the San Francisco Bay which is expected to reduce straying (California HSRG 2012; Huber and Carlson 2015; Palmer-Zwahlen et al. 2019; Sturrock et al. 2019).

At the ESU level, the spatial diversity within the CVSRC ESU is increasing and spring-run Chinook salmon are present (albeit at low numbers in some cases) in all diversity groups. The reestablishment of CVSRC to Battle Creek and increasing abundance of CVSRC on Clear Creek observed in some years is benefiting the viability of CVSRC. Similarly, the reappearance of early migrating Chinook salmon to the San Joaquin River tributaries may be the beginning of natural dispersal processes into rivers where they were once extirpated. On one hand, the CVSRC ESU is trending in a positive direction towards achieving at least two populations in each of the four historical diversity groups necessary for recovery with the Northern Sierra Nevada region necessitating four populations (NMFS 2014b). On the other hand, CVSRC populations have declined sharply in recent years to in most cases worryingly low levels of abundance.

Emerging threats to the CVSRC populations may include thiamine deficiency, which was responsible for early life stage mortality of FRH spring-run Chinook salmon in the hatchery (Foott 2020). Significant numbers of juvenile mortalities were observed in the Feather River rotary screw trap, early in the juvenile outmigration season, consistent with thiamine deficiency complex (TDC) observed in the hatchery. In fact, significantly fewer juveniles were observed in 2019 (N=1149) compared to 2018 (N=30,334), and 45% of juveniles in 2019 were found dead compared to 1% observed in 2018 (Kindopp 2020). It is unclear the extent to which this was a basin-wide nutritional deficiency for all CVSRC spawning in 2019. Direct mortality or latent effects that would lead to increased mortality in that cohort would not be detected in viability criteria until the dominant age class of 3-year-olds return to spawn in 2022.

The only independent population of CVSRC salmon that is not at a high risk of extinction is the population on Butte Creek. Yet, the continued existence of the Butte Creek CV spring-run Chinook population is wholly dependent on the reliable, long-term import of cold water from the West Branch of the Feather River to the anadromous habitat in Butte Creek provided by the operation of the Pacific Gas and Electric Company's (PG&E) DeSabla Centerville Project. Considerable uncertainty remains for the future of the PG&E project and the ability to transfer water from the West Branch Feather River to the anadromous habitat in Butte Creek to support the survival of CVSRC.

To conclude, the viability of CVSRC ESU has declined since the 2015 assessment and the ESU is at greater risk of extinction. The largest impacts are likely due to the freshwater drought conditions and unusually warm ocean conditions experienced by these cohorts, resulting in weakening viability metrics and greater risks of extinction to the majority of the populations since the previous viability assessment. The recent declines of many of the dependent populations, high pre-spawn mortality and poor juvenile survival during the 2012–2016 drought, unknown impacts due to warm ocean conditions and reorganization of coastal marine food webs, are all causes for increased concern for the long-term viability of the CVSRC ESU. Overall, new information on abundance, productivity, rate of population decline, spatial structure, hatchery influence, and diversity, indicate the viability of the majority of populations in the ESU has declined since the 2015 assessment. Results of the FEMAT assessment revealed that the greatest number of votes fell in the "High" risk of extinction category (47%) for the CVSRC ESU, which was only marginally higher than the "Moderate" risk of extinction (43%)- separated only by a single vote. Fewer votes fell in the "Low" risk of extinction category (10%; Table 5.7). It should be noted that the combined weights in the "Low" and "Moderate" categories is greater than the single category of "High" risk of extinction. Unlike ESA-listed Endangered Sacramento River Winter-run Chinook salmon and Central California Coast coho salmon, historically independent populations of CVSRC Chinook salmon occupy all diversity groups albeit at low numbers; it is at the diversity group spatial scale where catastrophic events are best buffered for the ESU. Extinction risks are of concern due to the low abundance of individuals, the magnitude of the abundance decline observed since the last assessment, and the ESU's pre-existing vulnerability. In the context of the occupied diversity groups yet declining populations and one population disproportionally contributing to the number of fish in the ESU, FEMAT scoring captured the uncertainty of the authors to conclude that the CVSRC salmon ESU is at moderate to high risk of extinction (Table 1.1).

5.3 Central Valley Steelhead DPS

DPS Delineation

This distinct population segment (DPS) includes steelhead populations spawning in the Sacramento and San Joaquin rivers and their tributaries. Populations upstream of migration barriers remain excluded from this DPS. Hatchery stocks within the DPS include Coleman National Fish Hatchery (CNFH), Feather River Hatchery (FRH), and Mokelumne River Hatchery (MRH). Genetic analysis showed that the steelhead stock propagated in the MRH was genetically similar to the steelhead broodstock in the FRH (Pearse and Garza 2015), consistent with documentation on the recent transfers of eggs from the FRH for broodstock at the MRH. The Nimbus Hatchery (NH) steelhead remain genetically divergent from the Central Valley DPS lineages, consistent with their founding from coastal steelhead stocks, and remain excluded from the DPS (Pearse and Garza 2015). Thus, the delineation of the Central Valley DPS was modified in the 2016 status review to include steelhead from the MRH (NMFS 2016e; 85 FR 81822).

Summary of Previous BRT Conclusions

Good et al. (2005) found that the California Central Valley (CCV) Steelhead DPS was in danger of extinction, with a minority of the Biological Review Team (BRT) viewing the DPS as likely to become endangered. The BRT's major concerns were the low abundance of natural-origin anadromous *O. mykiss*, the lack of population-level abundance data, and the lack of any information to suggest that the decline in steelhead abundance evident from 1967–1993 dams counts had stopped.

Brief Review of TRT Documents and Previous Assessments

The Central Valley (CV) Technical Recovery Team (TRT) delineated more than 80 independent populations of Central Valley steelhead, along with a number of smaller dependent populations. Spawning habitats of many of these historical populations are entirely above man-made impassable barriers and may persist as resident (non-anadromous) or adfluvial rainbow trout, although they are presently not considered part of the DPS. Impassable dams also block many anadromous populations from reaching significant portions of their historical spawning and rearing habitat.

Lindley et al. (2007) developed viability criteria for steelhead, summarized in Table 5.1. Using data through 2005, Lindley et al. (2007) found that data were insufficient to determine the viability of any of the naturally-spawning populations of CCV steelhead, except for those spawning in rivers adjacent to hatcheries, which were likely to be at high risk of extinction due to extensive spawning of hatchery-origin fish in natural areas. However, from 2000–2010, run size data from Battle Creek, which is the best population-level data available for steelhead in the Central Valley, suggested a 17% decline per year, placing the population in a high extinction risk category. The proportion of hatchery-origin fish in the Battle Creek returns averaged 29% over the 2002–2010 period, elevating the level of hatchery influence to a moderate risk of extinction. Lastly, the Chipps Island midwater trawl dataset of USFWS indicated that the decline in natural production of steelhead had continued unabated through 2010, with the proportion of adipose fin-clipped steelhead reaching 95%. In 2015, population trend data showed significant increases in abundance of CNFH and FRH populations, but data are still lacking to estimate trends in natural populations.

New Data and Updated Analyses

Population trend data remain extremely limited for the CCV Steelhead DPS. The total hatchery populations from CNFH, FRH, and MRH have significantly increased since the 2010 and 2015 viability assessments (Figure 5.8, Table 5.7). In fact, CNFH returns have steadily increased 15% per year over the last decade. Additional data are now available for Cottonwood, Antelope, Cow, Deer and Mill creeks as well as Yuba, Stanislaus, and Tuolumne rivers. Like all monitoring surveys, these data have limitations. For example, redd surveys can inflate steelhead estimates because redds can be created by non-anadromous *O. mykiss*. Some video weirs are not operated over the entire duration of adult steelhead migration. Nonetheless, these data represent the best available information. Central Valley steelhead populations in these systems were evaluated for the first time using the viability criteria (Figure 5.8, Table 5.8).

The vast majority of steelhead in the Central Valley are from CNFH, which had a total population size of 19,173 and an average run size of 6,391. Steelhead from CNFH reached a record number of returns (10,977) in 2018, which included an overall population growth rate of 17% per year over the past decade. The FRH also produces a large number of steelhead. The total population size of FRH was 5,620 with an average run size of 1,873 and population growth of 19% per year over the past decade. The in-river spawning population of steelhead on the Feather River is estimated at N=241 with an average run size of $\hat{S}$ =80 (Table 5.8). The high proportion of FRH steelhead spawning in-river and the few natural-origin steelhead used in the FRH broodstock (none used in 2017–2019) suggests the Feather River population is reliant on hatchery supplementation. While the FRH steelhead population and run sizes place this population at low risk of extinction, the reliance on the hatchery places them at a high risk of extinction.

The best population-level data come from Battle Creek, where CNFH operates a weir. Central Valley steelhead have been identified as a priority species for restoration in Battle Creek upstream of the weir as part of the Battle Creek Salmon and Steelhead Restoration Project (BCSSRP) and also are produced at CNFH. The Battle Creek watershed is thought to have high potential to support a viable independent population of CV steelhead within the Basalt and Porous Lava diversity group (NMFS 2014b). In 2002, 2,000 steelhead passed the weir into the BCSSRP area to spawn in-river. However, prior to 2003, it was not possible to differentiate all hatchery- and natural-origin steelhead, since not all juvenile hatchery fish were adipose fin-clipped and thus a large fraction of these individuals were likely from CNFH (California HSRG 2012). In recent years, so few natural-origin steelhead returned to Battle Creek that, beginning in 2009, CNFH was operated as a segregation hatchery with only hatchery-origin steelhead used in the breeding protocols, and only natural-origin steelhead passed upstream of the weir into the BCSSRP area (California HSRG 2012). Subsequently, the total numbers of adult steelhead passing the weir has declined since 2000 to a total population of approximately 900 adults with an average run size of approximately 300 adults (U.S. Fish and Wildlife Service 2020b). The low steelhead abundance, declining trend, and 20% hatchery influence places it in the moderate extinction risk category. Tradeoffs are currently being evaluated to ensure that natural-origin steelhead that could spawn upstream in the BCSSRP area have an opportunity to reproduce in the wild, while also being mindful of the value of integrating "wild" genes back into CNFH hatchery production to minimize impacts of domestication on both the hatchery and natural steelhead populations (Cramer Fish Sciences 2016).

The total steelhead population on Clear Creek has increased since it was first estimated in 2003, reaching a total population size of 663, estimated by redd counts. Over the past decade, the returns have fluctuated but have been decreasing 2.5% per year (Figure 5.8, Table 5.8).

The American River steelhead population has experienced a precipitous decline since 2003, resulting in a moderate risk of extinction. Spawner population estimates were produced from an area under the curve method (Hilborn et al. 1999) using observations of steelhead occupying redds in 2002–2005, 2007, 2011–2013, and 2015–2018 (Scriven et al. 2018). It should be noted that a significant proportion of steelhead redds on the American River are made by NH steelhead, which are not part of the DPS, and declined 8% per year over the last decade.

The NH broodstock remains a threat to the viability of steelhead populations in the Central Valley. The NH broodstock is not included in the DPS because they are genetically divergent from the CCV DPS lineages, having been founded from Eel and Mad river populations (Pearse and Garza 2015). Thus, potential straying of NH broodstock and continued introgression with natural-origin American River steelhead poses a risk to the overall DPS (California HSRG 2012). The genetic lineage of the above-barrier population was evaluated as a potential source to replace the broodstock in NH as part of the Reasonable and Prudent Alternative for the 2009 biological opinion on the long-term operations of the federal and state water projects (NMFS 2009). Results from this work concluded that some upper American River *O. mykiss* populations represent genetically appropriate sources to reestablish a native anadromous run of steelhead in the lower American River that could contribute to the recovery of the threatened CV steelhead DPS (Abadía-Cardoso et al. 2019).

Zimmerman et al. (2009) found that the progeny of anadromous females were present at all Central Valley sites sampled using otolith reconstructions, but the proportion varied among sites (0.04–0.74) and was particularly low for San Joaquin River populations. Data on the presence and numbers of adult steelhead in San Joaquin River tributaries is increasing with the installation of video weirs on the Mokelumne, Stanislaus, and Tuolumne rivers during adult steelhead migration. These counts can now be used for assessing the status and trends of steelhead in the San Joaquin River. The numbers remain low with total population sizes of 39 and five on the Stanislaus and Tuolumne rivers, respectively, and high hatchery influence, placing populations in the San Joaquin River tributaries forming the Southern Sierra Nevada diversity group at a high risk of extinction. The annual number of adult steelhead counted moving upstream through the Stanislaus River weir ranged from two to 32 during 2010–2019 (CDFW 2020). Fifty-one percent of fish in the last generation were identified as hatchery-origin, placing the Stanislaus River population at a high risk of extinction based on low numbers and high hatchery influence

(CDFW 2020). The Mokelumne River is also at a high risk of extinction with 90–100% of adult steelhead at the Woodbridge Dam video station identified as hatchery-origin, with only 0–12 natural-origin steelhead returning each year from 2010–2019 (CDFW 2020).

Populations upstream of migration barriers remain excluded from the DPS. Recent genetic information on Central Valley *O. mykiss* populations upstream and downstream of dams within the same tributaries showed that these populations were not each others' closest relative (Pearse and Garza 2015; Pearse and Cambell 2018). This is in contrast to genetic patterns observed in coastal steelhead populations. Further, surveys of genetic variation in *O. mykiss* within the upper Tuolumne and Merced Rivers revealed that most populations retained largely indigenous ancestry that expressed adfluvial migrations (Pearse and Cambell 2018). However, little genetic structure remains among downstream-of-barrier populations suggesting that any ancestral population structure that may have existed has been significantly altered (Pearse and Garza 2015). These findings highlight the need to consider genetic lineage in future reintroduction and recovery efforts. As described in Section 1.2, freshwater-resident (non-anadromous) forms of *O. mykiss* co-occur and appear to interbreed with the anadromous form in many populations, and new research has improved our understanding of the genetic architecture of the populations exhibiting both nonanadromous and andromous forms (Pearse et al. 2014, Pearse et al. 2019). Thus, while not formally considered part of the DPS, resident (nonanadromus) *O. mykiss* warrant consideration in managing for the anadromous life history.

The Chipps Island midwater trawl data provide information on the trend in abundance for the CCV steelhead DPS as a whole. Updated through 2019, the trawl data indicate that the production of natural-origin steelhead remains very low relative to hatchery production (Figure 5.9). Catch-per-unit effort has fluctuated and generally increased over the past decade, but the proportion of the catch that was adipose fin-clipped (100% of hatchery steelhead production have been adipose fin-clipped starting in 1998) has risen steadily, exceeding 90% in recent years and reaching 96% during the drought in 2015. This suggests that the vast majority of steelhead outmigrating from the Delta are of hatchery origin.

Harvest Impacts[13]

Ocean harvest of steelhead is rare, and is likely an insignificant source of mortality for Central Valley steelhead.

Since the early 1990s, anglers fishing for steelhead in anadromous portions of California waters have been required by CDFW to purchase a steelhead report card. Information on the dates and locations of fishing, as well as the number of adult steelhead kept, the number of adult steelhead released, the origin of the fish caught (hatchery or wild) and the number of hours fished must be reported (Jackson 2007; CFFW 2016). While anglers are required to report this information, average compliance rates are low, approximately 30% (CDFW 2016). Poor reporting of report card data and other data deficiencies precludes a rigorous assessment of harvest impacts.

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[13] Michael O'Farrell prepared this section on harvest impacts.

California prohibits retention of natural-origin steelhead. Fishing effort estimates are not available from report card data for recent years (post-2014). CDFW performs angler surveys on Central Valley streams, and data from these surveys are used to estimate steelhead harvest and fishing effort, however these estimates do not appear to be regularly reported. No direct information is readily available on the level of CV steelhead fishery impacts. There appears to have been little change to fishing regulations in California's Valley district in recent years. Given this relatively sparse information, it is difficult to conclude whether the level of harvest impacts on CV steelhead DPS has changed appreciably in recent years.

Summary and Conclusions

Recent data from several monitoring efforts are included for the first time in this assessment making comparisons of the trend in extinction risk across the DPS largely speculative at this time. However, the majority (11 of 16) of populations for which there are data are at a high risk of extinction based on abundance and/or hatchery influence, with no population considered to be at a low risk of extinction. Future assessments will be able to assess changes in key viability metrics over time. However, for the populations that have previously been assessed in 2010 and 2016, when it was concluded that the DPS was in danger of extinction, new data suggest steelhead viability has slightly improved. This modest improvement was driven by the increase in adult returns to hatcheries from their recent lows, but the state of natural-origin fish remains poor and largely unknown. The lack of improved natural production as estimated by exit at Chipps Island, and low abundances coupled with large hatchery influence in the Southern Sierra Nevada diversity group are causes for continued concern.

Based upon the limited information available, we find that the overall viability of the CV Steelhead DPS unchanged since the 2015 assessment. Therefore, the biological extinction risk is considered "Stable" and in the species remains in "Moderate" risk category.

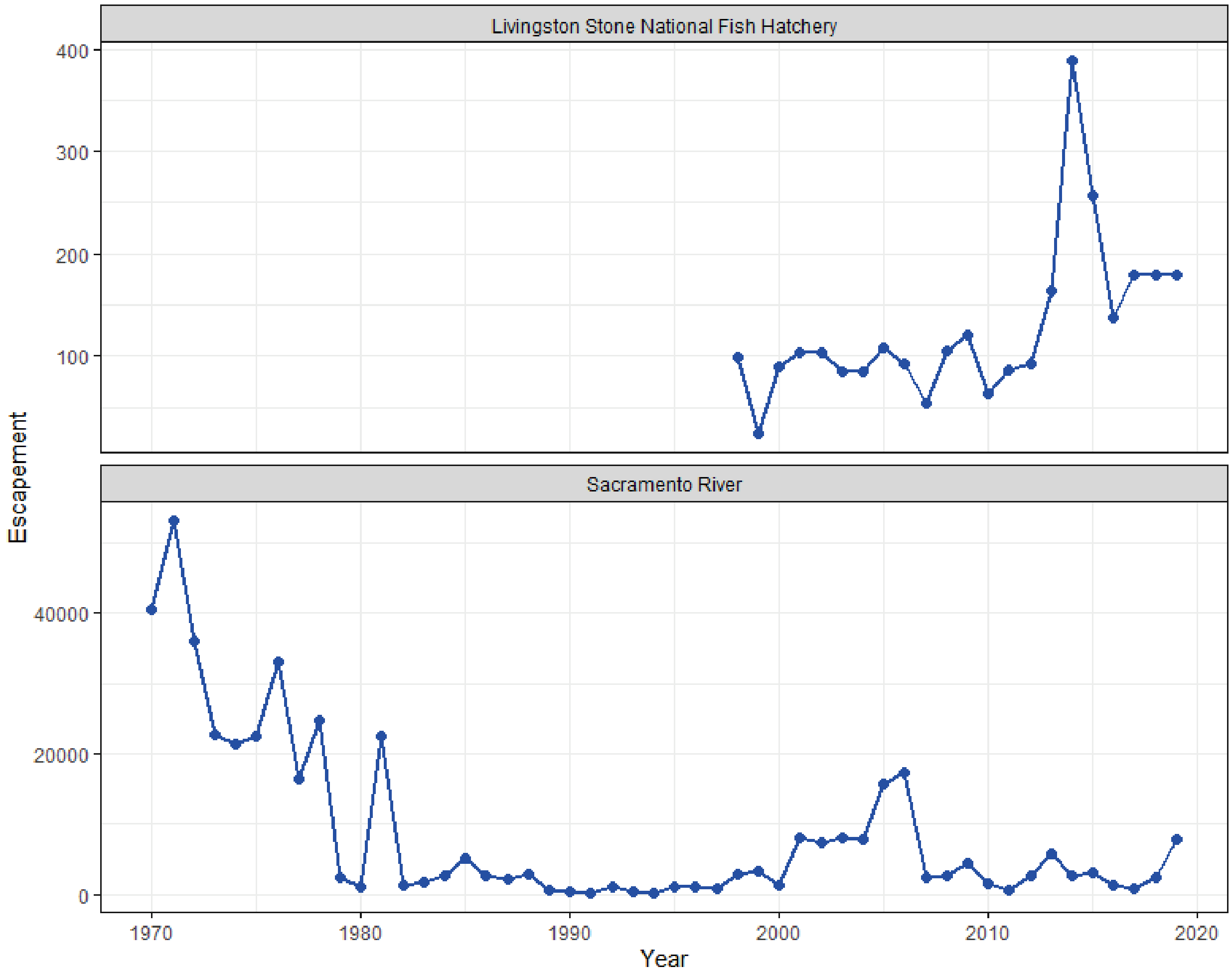

Figure 5.1. Time series of escapement for Sacramento River winter-run Chinook salmon (SRWRC) used as broodstock at (a) Livingston Stone National Fish Hatchery and (b) spawning in-river. Estimates for in-river SRWRC spawners is the number of adults counted at Red Bluff Diversion Dam and only the mark-recapture carcass survey estimates beginning in 2001; Data source: Azat (2020).

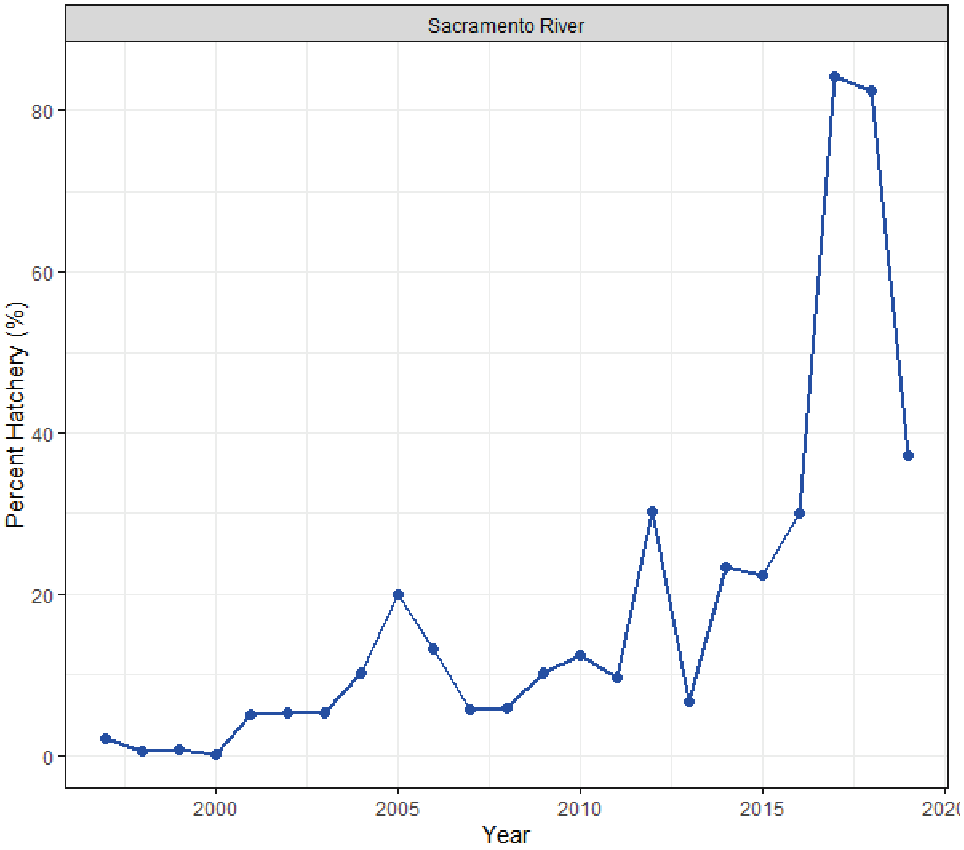

Figure 5.2. Percentage of Sacramento River winter-run Chinook salmon spawning in-river of hatchery origin; Data source: Killam 2020.

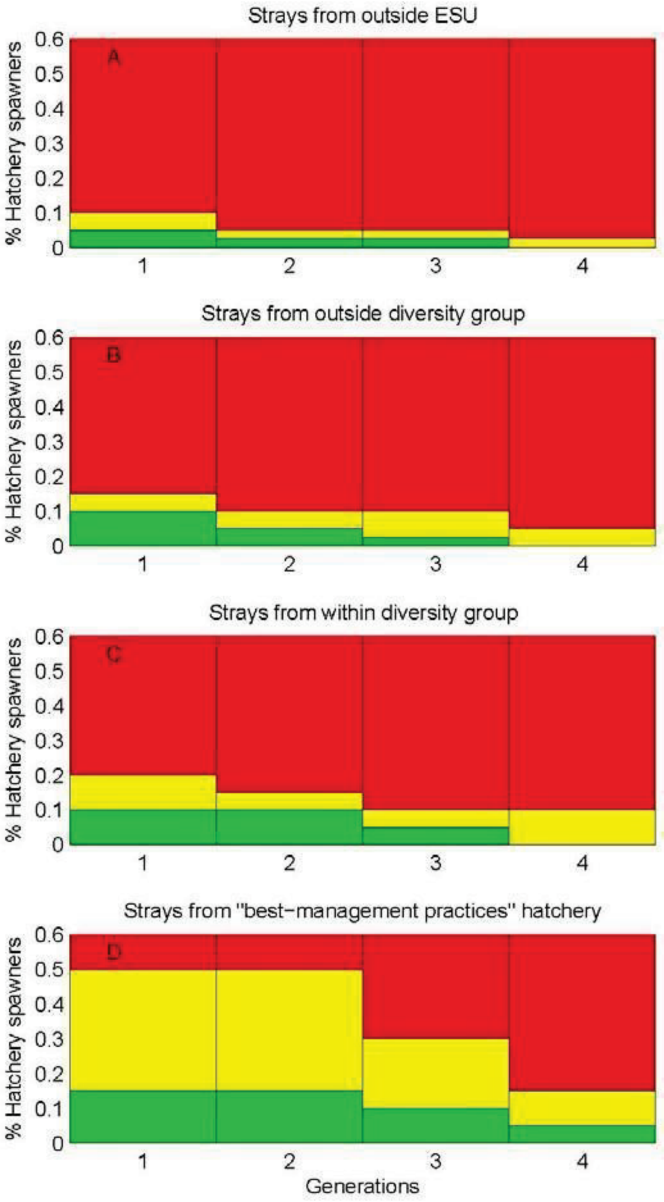

Figure 5.3. Percentage of hatchery-origin spawners and the resulting risk of extinction due to hatchery introgression from different sources of strays over multiple generations - low (green), moderate (yellow), and high (red). Model using “best-management practices” was used in the winter-run assessment based on the breeding protocols at the Livingston Stone National Fish Hatchery for Sacramento River winter-run Chinook salmon. The parameter “strays from outside ESUs” was used to assess impacts of introgression between Central Valley spring- and fall-run Chinook salmon ESUs at the Feather River Hatchery. Figure reproduced from Lindley et al. 2007.

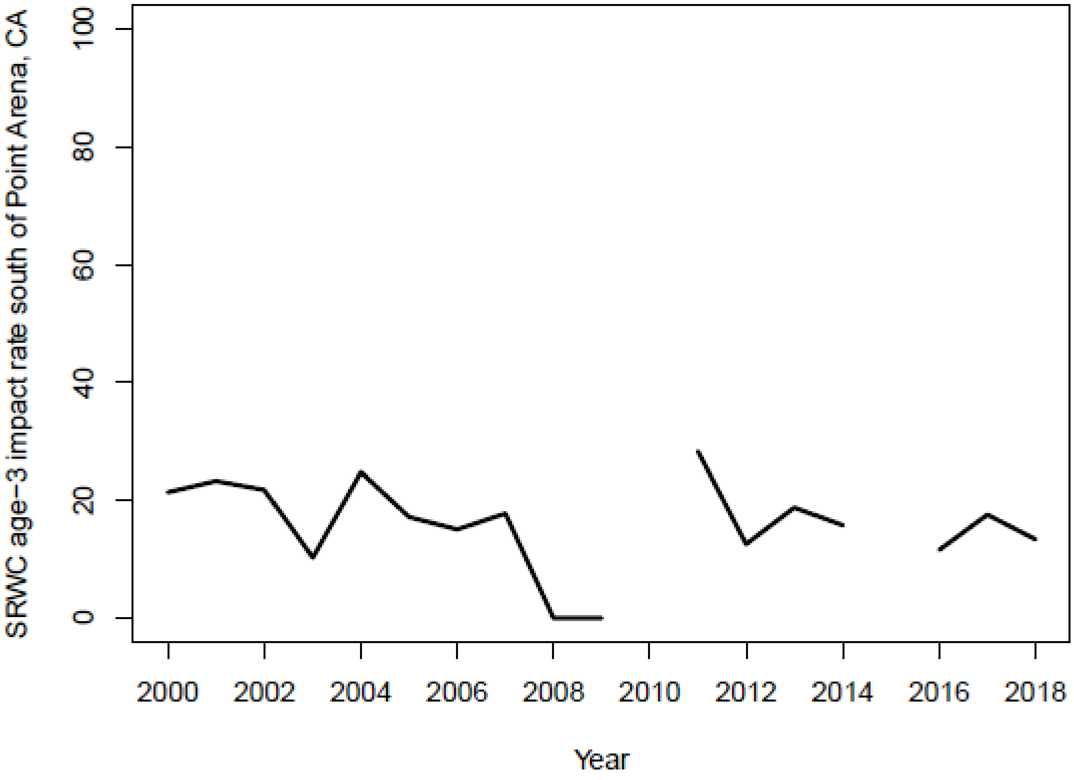

Figure 5.4. Sacramento River winter-run Chinook salmon age-3 ocean impact rate south of Point Arena for years 2000–2018. Estimates are sourced from Pacific Fishery Management Council (2020). The impact rate could not be estimated in 2010 and 2015 due to insufficient coded-wire tag recovery data.

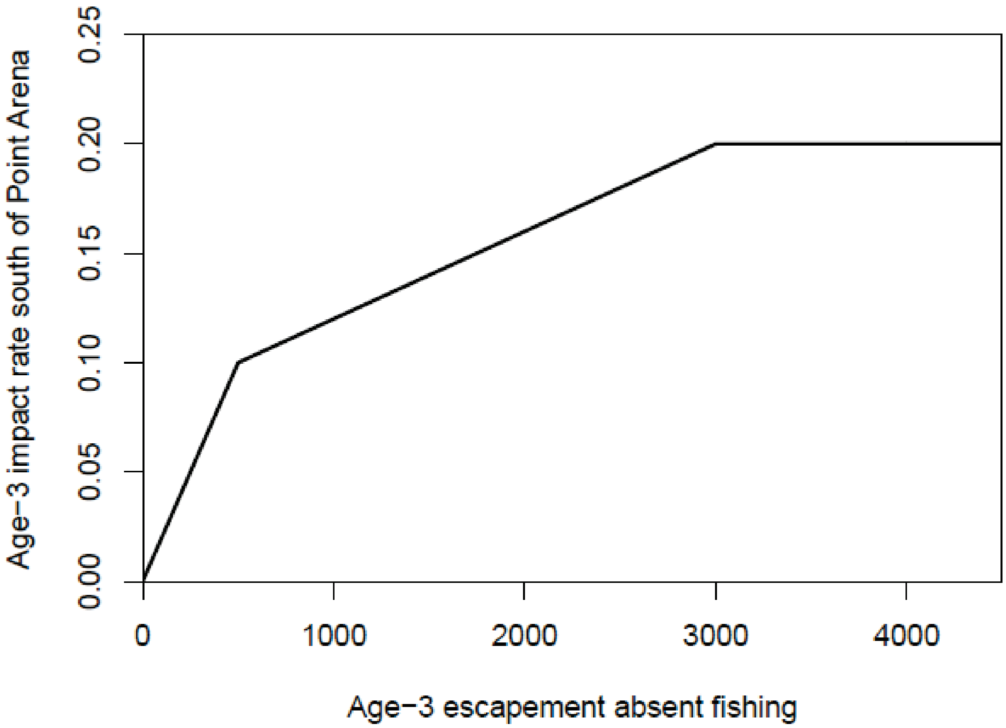

Figure 5.5. Current Sacramento River winter-run Chinook salmon harvest control rule.

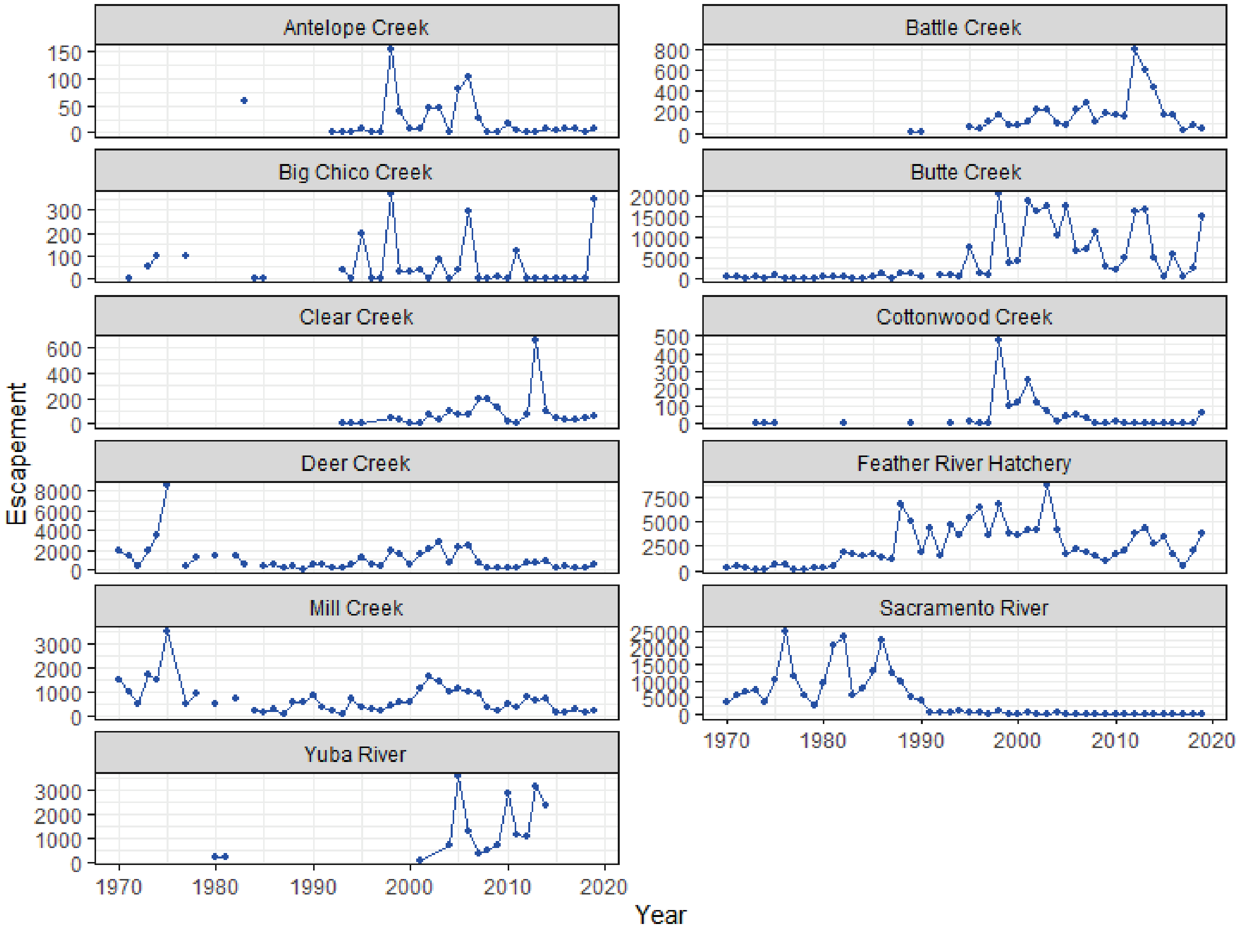

Figure 5.6. Escapement for Central Valley spring-run Chinook salmon over time. For Butte Creek populations, the mark-recapture estimates are used beginning in 2001. No data were provided for escapement years 2015-2019 for the Yuba River. The Yuba River Management Team is in the process of revising their statistical package previously used to refine the demarcation date that will separate spring- and fall-run estimates (P. Bratovich, HDR, personal communication). Note: Beginning in 2009, Red Bluff Diversion Dam estimates of spring-run Chinook salmon in the Upper Sacramento River are recorded as ‘0’ in Azat 2020.

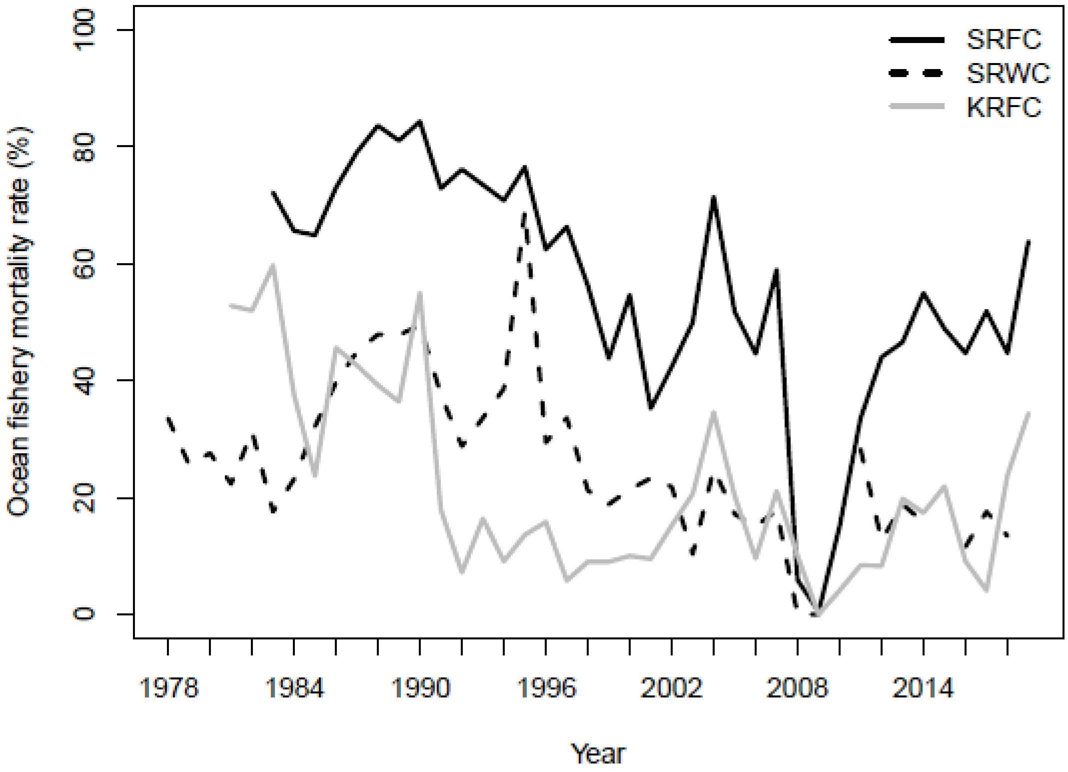

Figure 5.7. Ocean fishing mortality rates estimated for Sacramento River fall-run Chinook salmon (SRFC), Sacramento River winter-run Chinook salmon (SRWC), and Klamath River fall-run Chinook salmon (KRFC). For SRFC, the fishing mortality rate is the determined by the estimated ocean harvest divided by the Sacramento Index. For SRWC, the fishing mortality rate is represented by the age-3 ocean impact rate. For KRFC, the fishing mortality rate is determined by the age-4 ocean harvest rate. More detail regarding these estimates can be found in O'Farrell and Satterthwaite (2015), Satterthwaite et al. (2018), and Pacific Fishery Management Council (2020).

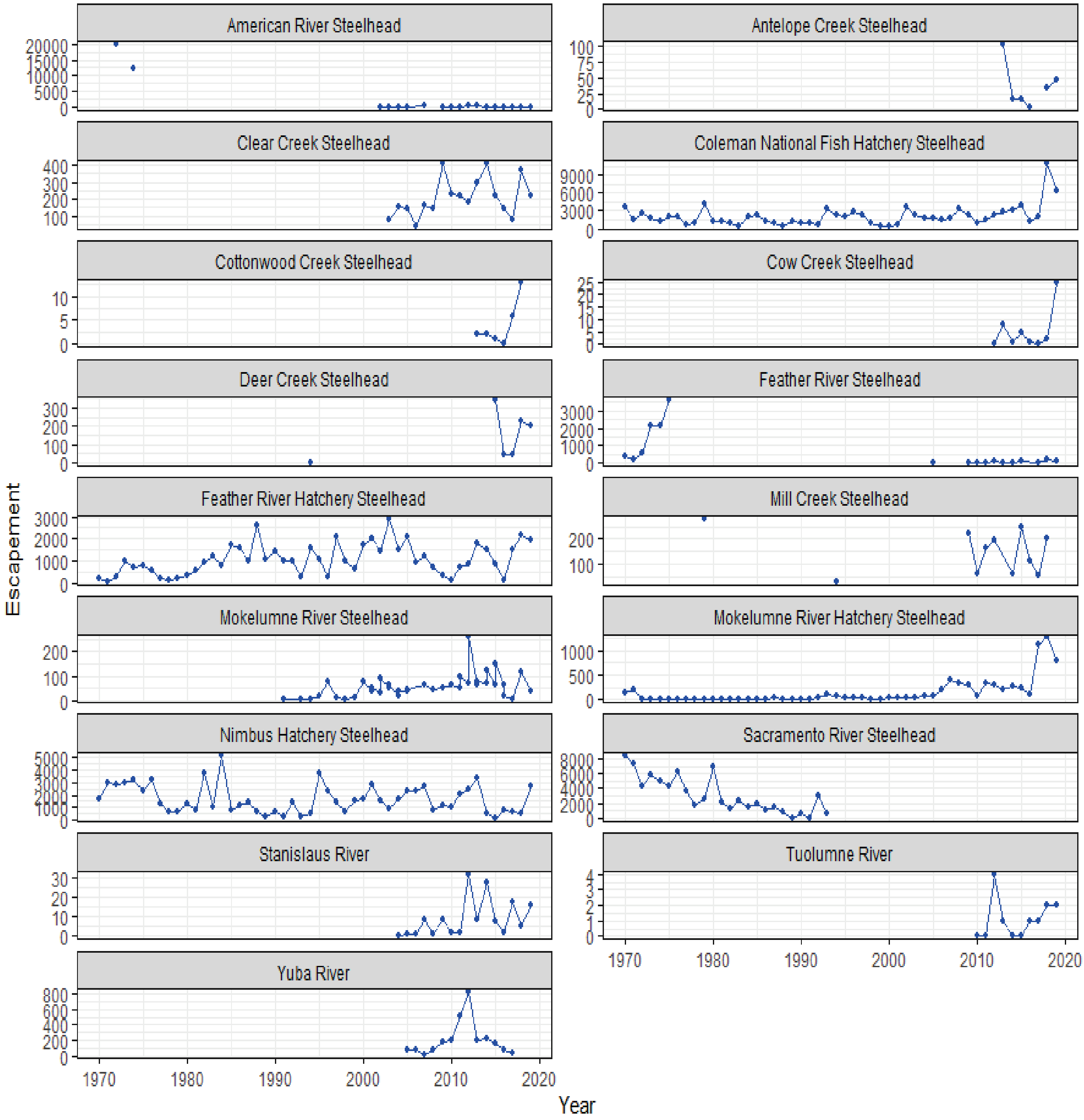

Figure 5.8. Time series of escapement for California Central Valley steelhead populations through 2019 where available.

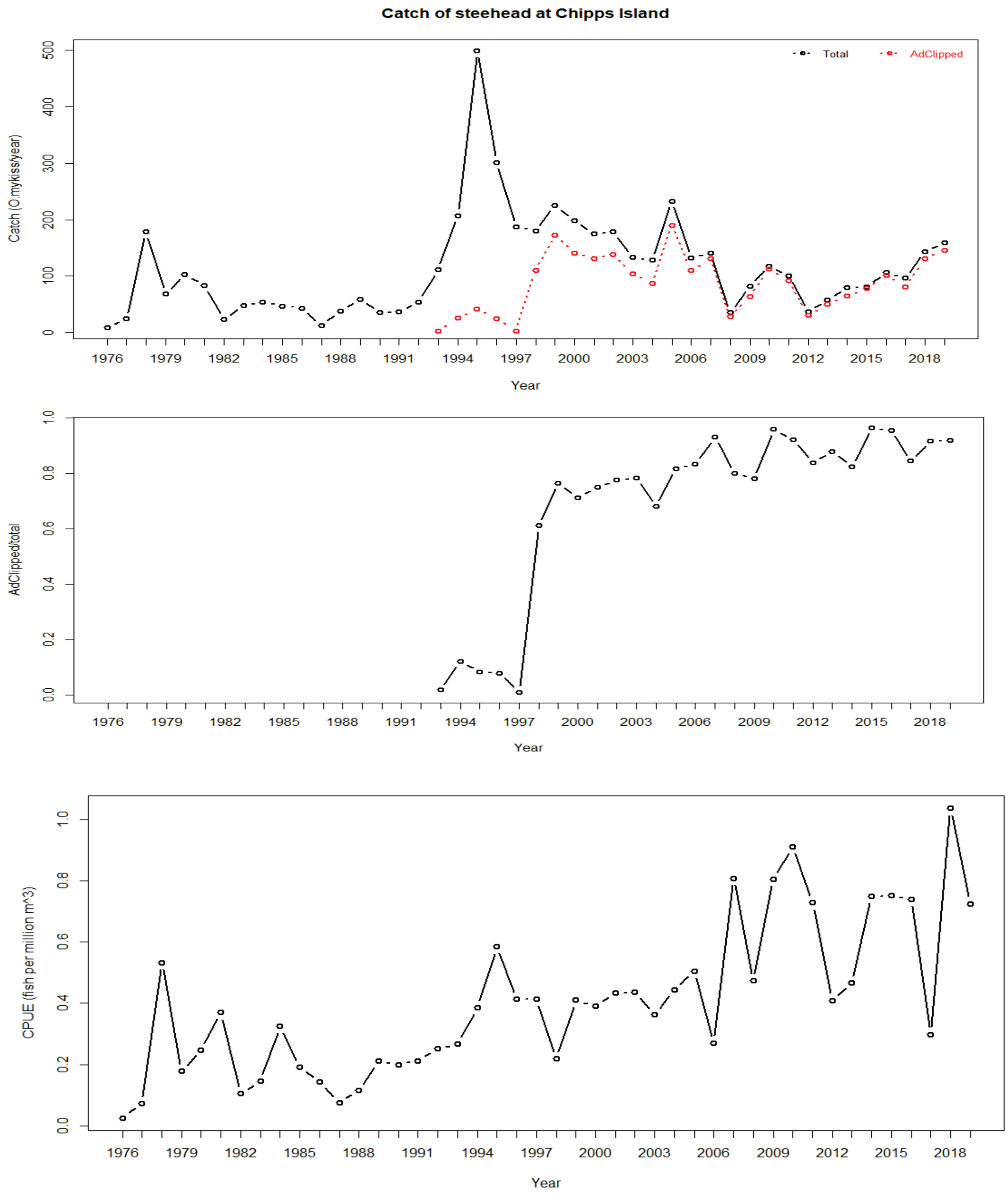

Figure 5.9. Top: Catch of steelhead at Chipps Island by the USFWS midwater trawl survey. Middle: Fraction of the catch bearing an adipose fin clip. 100% of steelhead production has been marked starting in 1998. Bottom: Catch-per-unit effort (CPUE) in fish per million m^3 swept volume. CPUE is not easily comparable across the entire period of record, as over time, sampling has occurred over more of the year and catches of juvenile steelhead are expected to be low outside of the primary migratory season.

Table 5.1. Criteria for assessing the level of risk of extinction for populations of Pacific salmonids in the Central Valley of California. Overall risk is determined by the highest risk score for any criterion (modified from Lindley et al. 2007).

	Risk of extinction		
Criterion	High	Moderate	Low
Extinction risk and PVA	> 20% within 20 yrs	> 5% within 100 yrs	< 5% within 100 yrs
	- or any ONE of -	- or any ONE of -	- or ALL of -
Population size[a]	$N_e \leq 50$	$50 < N_e \leq 500$	$N_e > 500$
	- or -	- or -	- or -
	$N \leq 250$	$250 < N \leq 2500$	$N > 2500$
Population decline	Precipitous decline[b]	Chronic decline or depression[c]	No decline apparent or probable
Catastrophe, rate, and effect[d]	Order of magnitude decline within one generation	Smaller but significant decline	Not apparent
Hatchery influence[f]	High	Moderate	Low

a – Census size N can be used if direct estimates of effective size N_e are not available, assuming $N_e/N = 0.2$.

b – Decline within last two generations to annual run size ≤ 500 spawners, or run size > 500 but declining at ≥ 10% per year over the past 10 years. Historically small but stable population not included.

c – Run size has declined to ≤ 500, but now stable.

d – Catastrophes occurring within the last 10 years.

e – Decline < 90% but biologically significant.

f – See Figure 5.3 for assessing hatchery impacts.

Table 5.2. Viability metrics for Sacramento River winter-run Chinook salmon ESU populations including those spawned at the Livingston Stone National Fish Hatchery (LSNFH) and in-river spawners through 2019. Total population size (*N*) is estimated as the sum of estimated run sizes over the most recent three years. The mean population size ($\hat{S}$) is the average of the estimated run sizes for the most recent 3 years. Population growth rate (or decline; 10 year trend) is estimated from the slope of log-transformed estimated run sizes. In order to log-transform the run data, any '0's' were replaced with '0.001'. The catastrophic metric (Recent decline) is the largest decline in a single generation over the most recent 10 such ratios (see supplemental for detailed calculations).

Population	*N*	$\hat{S}$	10-year trend (95% CI)	Recent decline (%)
LSNFH winter-run Chinook	540	180	0.103 (-0.015, 0.221)	28.4
Sacramento River winter-run Chinook	11106	3702	0.080 (-0.118, 0.277)	75.3

Table 5.3. Average percentage of SRWRC in-river spawners that are hatchery-origin over a varying (cumulative) number of years. One generation (g1) consists of the most recent three years; two generations (g2) the most recent six years; three generations (g3) the most recent nine years; four generations (g4) the most recent 12 years; five generations (g5) the most recent 15 years; six generations (g6) the most recent 18 years; seven generations (g7) the most recent 21 years. Data source: (Killam 2020).

	Generation						
	g1	g2	g3	g4	g5	g6	g7
Average hatchery influence	68%	47%	36%	30%	26%	23%	20%

Table 5.4. Summary of Sacramento River winter-run Chinook salmon extinction risk by population criteria described in Lindley et al. (2007) for the 2010, 2015, and 2020 assessment periods. Overall extinction risk is determined by the highest risk score for any criterion.

	Viability assessment		
	2010	2015	2020
Population Size	Low risk	Low risk	Low risk
Population Decline	Low risk	Moderate risk	Low risk
Catastrophe, rate, and effect	Low risk	Low risk	Low risk
Hatchery Influence	Low risk	Moderate risk	High risk

Table 5.5. Viability metrics for Central Valley spring-run Chinook salmon (CVSRC) ESU populations through escapement year 2019. Total population size (*N*) is estimated as the sum of estimated run sizes over the most recent three years for independent populations (**bold**) and dependent populations. The mean population size ($\hat{S}$) is the average of the estimated run sizes for the most recent 3 years (2017-2019). Population growth rate (or decline; 10-year trend) is estimated from the slope of log-transformed estimated run sizes. In order to log-transform the run data, any '0's' were replaced with '0.00001'. The catastrophic metric (Recent decline) is the largest decline in a single generation over the most recent 10 such ratios (see supplemental for detailed calculations).

Population	N	$\hat{S}$	10-year trend (95% CI)	Recent decline (%)
Antelope Creek spring-run	16	5.3	0.181 (-0.949, 1.312)	87.8
Battle Creek spring-run	157	52.3	-0.228 (-0.446, 0.009)	76.5
Big Chico Creek spring-run	350	116.7	-0.411 (-2.404, 1.581)	100.0
Butte Creek spring-run	17740	5913.3	-0.059 (-0.400, 0.283)	76.3
Clear Creek spring-run	136	45.3	0.044 (-0.266, 0.354)	82.9
Cottonwood Creek spring-run[a]	62	20.7	-1.073 (-2.672, 0.527)	100.0
Deer Creek spring-run	956	318.7	-0.037 (-0.191, 0.117)	83.3
Feather River Hatchery spring-run	6509	2169.7	-0.026 (-0.192, 0.140)	45.8
Mill Creek spring-run	590	196.7	-0.158 (-0.288, -0.028)	67.9
Sacramento River spring-run[b]	-	-	-	

a – Data from 2015-2018.

b – Beginning in 2009, estimates of spawning escapement of Upper Sacramento River spring-run Chinook were no longer monitored. Historically, this estimate was derived by the total Red Bluff Diversion Dam (RBDD) counts minus the spring-run numbers in the upper Sacramento tributaries. Beginning in 2009, RBDD gates were partially operated in the up position and in 2012 they were entirely removed and thus spring-run estimates were no longer available.

*Erratum: Butte Creek and Yuba River viability metrics using data from 2005-2015 reported in the 2015 viability assessment are revised below (Johnson and Lindley 2016). These changes do not influence the interpretations of the status or trends provided in the previous viability assessment:

Population	N	Ŝ	10-year trend (95% CI)	Recent decline (%)
Butte Creek spring-run	38182	12727.3	-0.018 (-0.224, 0.187)	51
Yuba River spring-run			0.067 (-0.138, 0.272)	

Table 5.6. Summary of Central Valley spring-run Chinook salmon extinction risk by population criteria described in Lindley et al. (2007) for the 2010, 2015, and 2020 viability assessment periods. Overall risk is determined by the highest risk score for any criterion.

Population	Extinction risk		
	Viability assessment year		
	2010	2015	2020
Mill Creek	High	Moderate	High
Deer Creek	High	Moderate	High
Butte Creek	Low	Low	Low
Battle Creek	High	Moderate	High
Clear Creek	High	Moderate	High
Feather River Hatchery	High	High	High

Table 5.7 Tally of the FEMAT (1993) vote distribution for extinction risk for CVSR ESU. Each of four members allocated 10 points among the three viability categories (low, moderate, high).

Risk category	Number of votes	Percent of total
Low	3	10%
Moderate	13	43%
High	14	47%

Table 5.8. Viability metrics for California Central Valley steelhead populations through escapement year 2019. Total population size (*N*) is estimated as the sum of estimated run sizes over the most recent three years for independent populations (bold) and dependent populations. The mean population size ($\hat{S}$) is the average of the estimated run sizes for the most recent three years. Population growth rate (or decline; 10-year trend) is estimated from the slope of log-transformed estimated run sizes. The catastrophic metric (Recent decline) is the largest decline in a single generation over the most recent 10 such ratios (see supplemental for detailed calculations). Note: Mill Creek and Yuba River VSP metrics include run size estimates from 2010-2018 and 2010-2017, respectively. Populations with no values in the 10-year trend have fewer than seven years of data in the time series.

Steelhead population	N	$\hat{S}$	10-year trend (95% CI)	Recent decline (%)	Extinction risk[a]
American River	362	121	-0.060 (-0.253, 0.133)	72.7	Moderate
Antelope Creek	83	28			High
Clear Creek[b]	663	221	-0.024 (-0.152, 0.105)	26.2	Moderate
Coleman National Fish Hatchery[c]	19173	6391	0.165 (0.021, 0.310)	18.8	High*
Cottonwood Creek	19	6			High
Cow Creek	27	9			High
Deer Creek	476	159			Moderate
Feather River	241	80	0.104 (-0.141, 0.349)	98.6	High
Feather River Hatchery[d]	5620	1873	0.186 (-0.071, 0.443)	63.5	High*
Mill Creek	375	125	0.079 (-0.650, 0.808)	18.2	Moderate
Mokelumne River	161	53.7	-0.138 (-0.330, 0.054)	50.0	High
Mokelumne River Hatchery	3182	1061	0.217 (0.021, 0.413)	19.1	High*
Nimbus Hatchery	3780	1260	-0.084 (-0.336, 0.169)	75.8	High*
Stanislaus River	39	13	0.100 (-0.181, 0.381)	57.1	High
Tuolumne River	5	1.7	0.277 (-0.145, 0.699)	60.0	High
Yuba River	274	91.3	-0.460 (-0.649, -0.271)	71.6	Moderate

a – Rationale for extinction risk determined primarily by abundance with (*) representing risk due to hatchery influence.

b – Clear Creek steelhead data are derived from redd counts. Some redds may be from non-anadromous *O. mykiss*.

c – Coleman National Fish Hatchery data are counts in the database and include individuals that are ultimately passed above the weir into the Battle Creek Restoration area. Those estimates can be found at U.S. Fish and Wildlife Service 2020b.

d – Feather River Hatchery numbers include repeat spawners (fish returning to the hatchery multiple times in a single year). These findings based on recent tagging studies suggest hatchery return numbers are likely slightly inflated.

6 South-Central / Southern California Coast Recovery Domain

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The recovery domain covers coastal drainages from the Pajaro River at Monterey Bay south to the Tijuana River at the U.S. border with Mexico, a relatively arid region consisting mostly of shrublands (chaparral), grasslands, and oak savannah, but with coniferous forests at high elevations and along some stream corridors, especially within the Big Sur Coast and Carmel River regions. Stream systems tend to divide into numerous small coastal creeks within the climate zone of marine influence, and fewer larger inland river systems that drain the arid interior valleys and exhibit highly variable and erratic streamflows.

The only native salmonid is *Oncorhynchus mykiss*, divided by Busby et al. (1996) into two evolutionarily significant units (ESUs). An important feature of these ESUs is that they are typically composed of mixed populations of anadromous fish (steelhead) and freshwater resident fish (rainbow trout), in which rainbow trout often vastly outnumber the steelhead depending on local conditions. The steelhead component has been given ESA protection as a distinct population segment (DPS) within the more encompassing ESU that includes both forms. Based on the viable salmon population concept of McElhany et al. (2000), Boughton et al. (2007) developed viability criteria for steelhead at the levels of population and DPS. A monitoring plan for the risk metrics was given broad conceptual outline by Adams et al. (2011) and updated by Boughton et al. (2022).

Importantly, the viability criteria recognized that the two listed DPSs—anadromous steelhead—were typically components of mixed populations of rainbow trout and steelhead, but the genetic, physiologic and ecological controls on the expression of these two life histories were poorly understood at the time. Because of this, the viability criterion for abundance was augmented by an additional criterion for anadromous fraction, defined as the proportion of reproducing adults that exhibit the anadromous life history. Because the controls on expression of anadromy were poorly understood, the criterion for anadromous fraction was set at 100% as a precautionary measure (Table 6.1, top). The underlying rationale was that viable runs of steelhead (the anadromous component) cannot be assumed to depend on rainbow trout (the resident component), without a greater understanding of the underlying mechanisms (and degree of importance) for this dependence.

Similarly, a lack of historical data on adult abundance, combined with knowledge that the region's erratic streamflows were likely to produce highly variable run sizes that increase extinction risk, led to recommendations for a precautionary approach to adult abundance criteria as well (Table 6.1, top). It was thought that better understanding of the mechanisms of environmental stochasticity in populations—especially the role of drought refugia—might eventually allow these criteria to be adjusted to a less precautionary stance.

Finally, as with salmonid ESUs in other recovery domains, it was recognized that population density was an important indicator of viability, but the specific life-stage and criterion for density were in need of further research (Table 6.1, top). To facilitate further research, recommendations were made to replace these “prescriptive criteria” with more refined performance-based criteria over time as more information became available (Table 6.1, middle).

For the DPS as a whole, viability criteria were defined in terms of collections of populations that each meet the population-level criteria, as well as additional criteria for geographic distribution and life-history expression (Table 6.1, bottom). To meet criteria for geographic distribution, viable populations would need to be distributed among the existing Biogeographic Population Groups (BPGs) in numbers meeting criteria for representation and redundancy (Table 6.2); be located in drought refugia to mitigate against recurrent drought (Table 6.1, bottom); and be separated from one another by a minimum geographic distance to mitigate risk from wildfire (Table 6.1, bottom). To meet criteria for life-history expression, viable populations would need to consistently exhibit both the resident and anadromous life history, as well as a third life history of anadromous fish that rear in estuaries for a significant time prior to smolting.

6.1 South-Central California Coast Steelhead DPS

DPS Delineation

This DPS comprises the anadromous component of Oncorhynchus mykiss inhabiting coastal streams from the Pajaro River at Monterey Bay south to, but not including, the Santa Maria River in Santa Barbara County. Freshwater-resident (non-anadromous) O. mykiss, commonly known as rainbow trout, also occur in most of these coastal streams and appear to be members of the same Evolutionarily Significant Unit (ESU) as the originally listed steelhead (Clemento et al. 2009), with which they interbreed (Pearse et al. 2019). In most populations, adult rainbow trout outnumber adult steelhead by large margins and are not considered part of the ESA-protected DPS (71 FR 833).

The DPS was divided by Boughton et al. (2007) into four BPGs:

- The Interior Coast Range BPG consisting of populations in the Pajaro and Salinas rivers (comprised of three subpopulations);
- The Carmel BPG consisting solely of the Carmel River population;
- The Big Sur Coast BPG consisting of 11 coastal populations between the Carmel River and the Monterey/San Luis Obispo County Line; and
- The San Luis Obispo Terrace BPG consisting of 15 coastal populations in San Luis Obispo County.

Summary of Previous Assessments

The first comprehensive status review of steelhead was conducted by Busby et al. (1996), who characterized Evolutionarily Significant Units (ESUs) and assessed their extinction risk. Early molecular-genetic studies of coastal steelhead populations in California found genetic diversity to be highest in south-central and southern California (Nielsen et al. 1994; Nielsen et al. 1997), leading Nielsen (1999) to propose that the diversity was a signature for a Pleistocene refugium for the species in southern California during the last ice age. She argued that this genetic diversity constituted a unique genetic legacy for the species (Nielsen et al. 2001), and presented data that it was being lost from hatchery populations (Nielsen et al. 1997). Relatively few sample sites were examined and the identification of ESU delineation by Busby et al. (1996) was based mainly on ecological factors, namely the shift from coastal redwood forest to coastal shrubland at the northern delineation of the original South-Central steelhead ESU, and the shift in zoogeographic provinces at its southern delineation (coastal mountain ranges to transverse ranges; marine zoogeographic transition at Point Conception).

At the time of the first status review (Busby et al. 1996), very few data had been collected on abundances, but those that existed suggested that anadromous adults had declined. For example, the relatively large run of wild steelhead in the Carmel River historically reached the thousands or perhaps tens of thousands, but had declined to zero during the California drought of 1986–

1992 due to a dewatered migratory corridor; the anadromous component was sustained by a captive saltwater rearing program instead (Thomas 1996). The south-central California coast steelhead ESU was subsequently listed as threatened on August 18 1997 (62 FR 43937). The listing was later modified to include only the anadromous component of the ESU on January 5 2006 (71 FR 833). The original status review of Busby et al. (1996) was updated by Good et al. (2005), Williams et al. (2011), and Williams et al. (2016), none of which led to changes in status of the DPS. A recovery plan was published in 2013 (NMFS 2013).

After the pioneering genetic work of Nielsen and coauthors, subsequent genetic studies were able to examine larger numbers of neutral alleles in greater numbers of fish from greater numbers of locations, and found that contrary to earlier findings the genetic diversity in this DPS tended to be lower than in more northerly steelhead populations (Garza et al. 2014). Thus the hypothesis of a Pleistocene refugium in southern and south-central California, with heightened genetic diversity and conservation value, was not supported by the new larger sample. However, genetic distance between populations was associated with geographic distance (either by river mile within basins or by coastal distance between basins), a classic example of the evolutionary pattern of isolation-by-distance. In addition, land-locked populations of *O. mykiss* upstream of impassible dams were found to mostly have little genetic introgression from hatchery fish that had long been stocked to reservoirs, instead being more closely related to the wild anadromous populations immediately downstream of the dams (Clemento et al. 2009).

The pattern of isolation-by-distance in neutral genetic variation supported the concept of geographically structured ESUs, but the existing ESU delineations based largely on ecological transitions did not tend to match up with the genetic breaks identified by Garza et al. (2014). In particular a large genetic break occurred at the Golden Gate rather than at the transition from redwood to shrubland along Monterey Bay. Genetic samples from the early 20th Century, preserved in museum specimens, showed that prior to the extensive fragmentation of river systems by dams, the pattern of isolation-by-distance was even stronger (Pearse et al. 2011).

Further genetic analysis identified an important component of adaptive genomic variation on chromosome 5 (Martinez et al. 2011; Pearse et al. 2014). Sometime in evolutionary history, a substantial portion of chromosome 5 (hereafter, Omy5) underwent an inversion, in which a segment of the chromosome was reversed end to end. This inversion was passed on to progeny, but for fish in which one chromosome is inverted and the other not (i.e., a parent of each type), no crossing-over can occur during meiosis, and so the set of genes on the inverted section of chromosome are tightly linked (prevented from mixing between the two chromosome types, or haplotypes). Such tightly linked sets of genes are sometimes called “supergenes.”

Pearse et al. (2014) surveyed the occurrence of these two haplotypes—the original and reversed versions of Omy5—in coastal populations of *O. mykiss* and found 1) both Omy5 haplotypes were present in most populations; 2) strong evidence for natural selection on the set of linked genes within the inversion, and 3) one haplotype dominated at sites in anadromous waters, whereas the other was more common at sites upstream of impassable dams ($p < 0.01$; Pearse et al. 2014, p. 7). Pearse et al. (2014) concluded that the patterns of allele frequency “identify a subset of [genetic] loci on Omy5 whose allele frequencies are both highly correlated [i.e. linked as a supergene] and strongly associated with the above- or below-barrier status of each population" (Pearse et al. 2014, p.4-5). This association suggested a role for the two haplotypes in expression of life-history, but did not directly demonstrate a link with migratory behavior.

New Data and Updated Analyses

Viability Criteria

Mean Annual Run Size. The original precautionary criteria for adult abundance relied on a simple model of density-independent population fluctuations (Boughton et al. 2007). The basic idea was that the highly variable rainfall and streamflows characteristic of the region drove large fluctuations in adult abundance, and these fluctuations were as large, proportionally, when abundance was low as they were when it was large. This density-independence creates a high risk of a population fluctuating to zero, and a relatively large viability criterion for mean abundance is required to compensate for that risk.

In contrast, if fluctuations in adult abundance were dampened when fish were rare, it would provide a stabilizing mechanism that would tend to protect against population extirpation, allowing a less stringent criterion. Recent data collected in the Carmel River population indicate that such density-dependent dampening appears to occur, and suggests a mechanism.

During the recent drought of 2012-2016, fish densities (juveniles + resident adults) in the mainstem Carmel River declined to very low levels. The drought was broken in 2017 by one of the wettest years on record, and data on steelhead densities were collected at random sites in the fall of that year and also in 2018 and 2019 (Boughton et al. 2020). During the drought, abundance of adult steelhead declined to zero fish in 2014 (inferred from failure of the sandbar at the mouth to open that year) and was likely very low through 2016, as judged by counts made at Los Padres Dam. Counts from Los Padres omit a substantial portion of the run—perhaps two thirds—but stayed at zero through 2016 and were below 10 in 2017, suggesting consistently small run sizes overall.

At the end of the drought in 2017, average fish density in the alluvial valley section of the mainstem was low (<0.1 fish/m^2), but within two years had climbed to about 0.3 fish/m^2 (Figure 6.1). This alluvial section of the river is vulnerable to heating and drying, which is likely the mechanism producing the low fish density observed in 2017. Similarly, fish density in the canyon section of the river also emerged from the drought relatively low (though not as low as the valley section), and then rapidly climbed from 2017 to 2019 (Figure 6.1). This section of river is regulated by flow releases from Los Padres Reservoir intended to sustain steelhead, and is not as vulnerable to drying as the alluvial channel downstream, so it is not surprising that steelhead densities were maintained at higher levels than the valley section downstream.

However, the wet uplands—the upper Carmel River above Los Padres Reservoir, and other well-watered tributaries draining the heights of the Santa Lucia Mountains—apparently maintained even higher fish densities than the canyon section of the mainstem, despite the fact that their flows were completely unmanaged (Figure 6.1). This suggests that during droughts, the distribution of O. mykiss tends to retract into the relatively reliable habitats at high elevations, where surface flow is sustained by orographic precipitation and is also less vulnerable to being lost into large alluvial groundwater basins (or extracted for out-of-stream uses). These findings suggest a mechanism by which population density becomes more stabilized at low abundance, by retracting into reliable drought refugia during periods of low-rainfall, and re-expanding into less reliable downstream habitats after the drought ends. Thus the density-independence assumption of the original precautionary viability criterion may be more stringent than necessary for populations with adequate drought refugia.

Population Density. Population density was proposed as an important risk metric by Boughton et al. (2007), but specifics of life-stage and criterion were left for further research. The original rationale was that a viable population should be characterized by good habitat conditions that sustain a population at high enough densities that density-dependent mechanisms for population stability come into play.

The only population in this DPS with a sufficiently long data series and sufficiently variable densities to assess density-dependence is likely the Carmel River population. Arriaza et al. (2017) analyzed these data and found evidence for density-dependence in the juvenile life-stage during the summer low-flow season, when the amount of freshwater habitat (wetted area) shrinks to its minimum for the year. In Figure 6.2 we present data from the Carmel River on mean fish density during the low-flow season, as a function of adult abundance the previous winter. The convex curved shape of the cloud of points illustrates the density-dependence found by Arriaza et al. (2017), and suggests that density-dependent survival of O. mykiss is most prevalent above 0.30 fish per square meter of habitat (dashed line in Figure 6.2). This is very close to the mean value of 0.29 fish per square meter reported for trout in the “Pacific Forest” region (coastal mountains from Monterey County to US border with Canada), in a meta-analysis of trout samples from the western USA during the middle of the last century (Platts and McHenry 1988). Thus we propose 0.30 fish/m^2 as a provisional viability criterion and treat it as such in this update.

Anadromous Fraction. Recent work has improved our understanding of the genetic architecture underlying mixed coastal populations of steelhead and rainbow trout. Building on the identification by Pearse et al. (2014) of two Omy5 haplotypes “A” and “R” associated with anadromy and residency, Pearse et al. (2019) delved more deeply in the genomic underpinnings of this association; Pearse et al. (2019) and Kelson et al. (2019) looked at associations with migration behavior; and Leitwein et al. (2017) and Apgar et al. (2017) examined environmental predictors for a high frequency of the “A” haplotype that is associated with anadromy. Here I summarize a number of key findings with potential implications for the viability criteria for anadromous fraction.

First, many of the genes in this inverted section of chromosome 5 are associated with circadian rhythms, sensitivity to photosensory cues, the timing of age at maturity, and other traits associated with life-history variation (Pearse et al. 2019). As described earlier, genetic recombination among these different genes of the “supergene” can occur during the generation of homozygous RR fish and AA fish, but not during the generation of heterozygous AR fish due to the inversion. This feature allows the A and R haplotypes to adaptively diverge in response to selection for two distinct life histories, while still being maintained together in the same population of O. mykiss (Pearse 2016).

Second, the two kinds of supergenes do, in fact, appear to be associated with different expression of life history. For example, Pearse et al. (2019) found that in a small steelhead population in Big Sur, juvenile females with the AA and AR genotypes were much more likely to migrate to the ocean than females with the RR genotype. Juvenile males with the AA and RR genotypes were similar to the females, but the male AR genotype was much less likely to migrate than the female AR genotype. This last observation is consistent with adaptive evolution of contrasting life-history strategies in males and females: female fitness is more associated with large body size than is male fitness, because of the energetic demands of manufacturing eggs versus sperm. Thus, females should be more likely than males to pursue anadromy because O. mykiss can generally achieve larger size at maturity in the ocean than in freshwater, and this provides more of a fitness benefit to females than to males (Pearse et al. 2019). In an independent study in the South Fork of the Eel River on the north coast, Kelson et al. (2019) made similar observations, finding that the expression of the downstream-migrant phenotype was associated both with being female and with having the A haplotype. In their smaller sample they did not detect a difference in the migration rate of AR females versus AR males, but they did find that in general the migration frequency of the AR genotype was intermediate between the RR and AA genotypes.

Third, this intermediate life-history expression of the AR genotype has potential implications for viability criteria. It provides a mechanism by which successful expression of the steelhead life history—survival of anadromous fish to maturity and reproduction—can be lost from an O. mykiss population when environmental conditions are adverse, but re-express itself when conditions favor it (although the speed of this re-expression is uncertain). When conditions are adverse, AA and AR smolts will continue to migrate toward the ocean but not survive to adulthood, and this fitness cost will favor RR and AR residents, so that frequency of the A haplotype declines over time (i.e. the population evolves). The A haplotype may become rare enough that AA individuals are very unlikely and the haplotype is maintained by resident fish carrying the AR genotype. Some of the progeny of such fish are AR rainbow trout that perpetuate the A haplotype in the resident population, whereas other progeny would be AR smolts that migrate to the ocean. These AR smolts would simply be lost to mortality when conditions for anadromy are adverse, but could reconstitute steelhead runs when conditions for anadromy become favorable.

When favorable conditions persist, sufficient AR smolts would survive to adulthood that AR x AR matings become more common rather than less common over time. One fourth of the progeny of such matings would be AA individuals, in which genetic recombination of the anadromous genome would resume and facilitate continuing adaptive evolution of the anadromous phenotype to changing conditions. A resident-only population can probably not sustain the A haplotype indefinitely because the “wasted” smolts produced by AR parents represent a fitness cost, but the loss appears to be a slow process. Apgar et al. (2017) estimated that the percentage of A haplotypes in an isolated population loses about 5 points per decade on average, although loss would likely be faster initially and then slow down, and might be somewhat decelerated for systems in which the anadromous fish can pursue their life history in reservoirs (Leitwein et al. 2017). A similar, reciprocal logic applies to the resident life history, for example providing a mechanism by which AR steelhead could colonize vacant freshwater habitat that eventually transforms to a population of rainbow trout when conditions for anadromy are adverse. Thus, even when the A haplotype is rare in a population, so that AA individuals are unlikely to occur, anadromy is still visible to natural selection due to its partial expression in AR individuals; and likewise for freshwater-residency and the R haplotype.

Fourth, the regional distribution of the two Omy5 haplotypes across coastal populations is consistent with their link to migratory phenotypes. Throughout the California coast, subpopulations above and below dams are generally each other's closest relatives when viewed from the perspective of neutral genetic variation, but are highly divergent in their frequencies of the A and R supergenes—the A haplotype is relatively common below dams, where fish have migratory access to the ocean, and R is relatively more common above dams, where anadromous migrants cannot return to reproduce (Clemento et al. 2009; Pearse et al. 2014; Pearse et al. 2019).

For example, Apgar et al. (2017) examined haplotype frequencies in 39 steelhead populations in coastal California, and found that frequency of the A haplotype at a sample site was associated with the site's degree of impact from migration barriers. Relative to similar sites without migration barriers, the frequency of the anadromous haplotype was most strongly affected by sites with complete barriers to anadromy that were longstanding (naturally occurring, such as waterfalls; -31% effect when present). The next strongest effect was of complete barriers that were more recently imposed (anthropogenic barriers; -18% effect when present), followed by recent partial barriers (-2% per barrier), with the weakest effect from longstanding (natural) partial barriers (-0.5% per barrier). Additionally, migration distance itself (river kilometers between the sample site and the ocean) had a negative effect on frequency of the anadromous haplotype. Overall, these five predictors explained 75% of the variation in haplotype frequency across sites.

Leitwein et al. (2017) had similar findings in Bay-Area populations, where haplotype frequencies showed substantial evolutionary differences between the groups of fish above and below dams, despite the groups being each other's closest relatives. They had an important finding at a set of nine reservoirs, where the A haplotype was significantly more frequent in the group below the dam (71% versus 50%, $p < 0.05$), but more variable above the dam, where it was associated with the volume of the reservoir impounded by the dam ($R^2=0.69$, $p < 0.01$). This last observation suggests that the A haplotype can be maintained not only by access to the ocean, but also by access to a large reservoir with capacity to support a migratory phenotype (sometimes called an "adfluvial" life history).

Fifth, although the A and R haplotypes are forms of adaptive genetic variation linked to anadromy and residency, respectively, they probably do not capture all the genetic variation associated with heritability of life history (Pearse 2016, Kelson et al. 2019). Moreover, the

Omy5 haplotypes may also contain adaptive variation associated with other selected traits such as growth and maturation timing (O'Malley et al. 2003; Nichols et al. 2008), and environmental factors such as realized growth also play a role in life-history trajectories (Satterthwaite et al. 2009; Ohms et al. 2014; Kendall et al. 2015). Indeed, the mean size at which fish initiate downstream migration—that is, the way life history responds to environmental factors such as food availability—is itself subject to natural selection (Phillis et al. 2016). So while there is a link between frequency of the A haplotype in a population and its expression of anadromy, numerous other genetic and environmental factors also play a role in its expression. However, since the A haplotype appears to be linked directly to the migration behavior itself, its presence would drive the selective environment experienced by the fish (freshwater vs marine), which in turn drives the selection of other genes that adapt the fish to freshwater vs marine environments, whether they are linked or not. To the degree that such adaptations are more successful than genes adapting the fish to the freshwater environment, the frequency of the A haplotype in the population will increase over time, and thus can be viewed as a lagging indicator for the fitness of the anadromous form relative to the resident form.

Finally, Pearse et al. (2019) determined that the A haplotype is ancestral to the R type, even though the R haplotype is more broadly distributed geographically, and is itself associated with anadromy in other regions (Pacific Northwest and Central Valley). After its initial appearance in a single population, it must have spread laterally to other populations via AR steelhead dispersing from their natal population to breed. This provided a mechanism for the parallel evolution of the resident phenotype across basins, in which natural selection operated on RR fish within basins, and the adaptations were moved laterally among basins by dispersing RA steelhead (Pearse 2016).

This body of work gives us much greater understanding of the way in which rainbow trout and steelhead mutually sustain each other, and indicates that the precautionary criterion of 100% anadromous fractions may be amenable to updating via an appropriate quantitative risk analysis.

Status and Trends

Status and trends are summarized for abundance of adult steelhead, and for fish density during the low-flow season.

Interior Coast Range. This Biogeographic Population Group (BPG) consists of the Pajaro River population and three populations in the Salinas River basin: the Gabilan Creek, Arroyo Seco, and southern Salinas populations. The abundance of adult steelhead has not been monitored in the Pajaro River, but the combined run of the three populations in the Salinas has been monitored intermittently since 2011; unfortunately no data have been reported since 2017 (CDFW 2020) and the time series is too short to estimate a trend. Run sizes here have been extremely small, always less than 50 fish per year and sometimes zero (Figure 6.3, bottom). The missing data from 2015 can probably be interpreted as a zero count: the fish counter was not operated because the lagoon never opened during this drought year. It is conceivable that anadromous fish entered the river via the Old Salinas River Channel, though unlikely. If anadromous fish did enter the river via the Old Salinas channel, access to suitable spawning and rearing habitat was not available due to lack of flow in the lower 30 miles of the river.

Assuming that the count in 2015 was zero, the average run size for the most recent four years ($\hat{S}$) was only 0.25 steelhead for the three Salinas populations combined (Table 3). CDFW (2020) rates the Salinas counts as full population estimates, except for 2011 and 2017 when operation of the Vaki fish counter was temporally limited. Based on observations of lagoon closure and opening, the operators of the fish counter inferred that adult steelhead may enter the estuary months before the upstream migration (e.g., the prior fall) or even mature in the estuary (Cuthbert et al. Draft).

Fish densities during the low-flow season were collected at 6 to 10 sites per year in lower Pajaro tributaries by Beck et al. (2019), using calibrated electrofishing (Figure 6.4A). The average density dipped below 0.3 fish/m^2 for five years during drought, but had recovered above this threshold as of 2019; the average density for the most recent four years was still below the threshold at 0.232 fish/m^2 (Table 6.4).

Upper Pajaro tributaries were monitored by Casagrande (2020) using uncalibrated electrofishing; since wetted widths were not reported I illustrate 1-dimensional densities (1D; fish per meter of channel) in Figure 6.4B, with 1D densities of the lower Pajaro included for comparison. Only the upper Pajaro densities showed a statistically significant downward trend ($p < 0.05$; Table 6.4).

Carmel River. Abundance of anadromous adults has been monitored for decades in the Carmel River at fish-passage facilities at two dams, San Clemente Dam and Los Padres Dam (CDFW 2020). The more complete counts at the lower dam (San Clemente) terminated in 2015 with the removal of the dam. Both series are partial counts due to steelhead spawning downstream of the dams; and both are illustrated in Figure 6.3 for completeness. The number of anadromous adults at Los Padres went to zero for three years during the drought, but has been rapidly recovering since the end of the drought in 2017 (Figure 6.3) and the removal of San Clemente Dam in 2015. For the past 20 years there has been significant downward trend averaging -21% per year ($p = 0.0007$), and despite the recent uptick the mean run size of the past four years is only 9 adults (Table 3).

Fish densities on the mainstem Carmel have been monitored for over 20 years at about 10 index sites distributed between the Los Padres Dam and the estuary. Prior to 2009 the mean density was usually above 0.3 fish/m^2 but from 2009 onward it only exceeded this threshold in two of eleven years, 2012 and 2019 (Figure 6.4C). This created a statistically significant downward trend ($p < 0.01$; Table 6.4) averaging -6% per year. The average density of the most recent four years was 0.183 fish/m^2, which is below the proposed population viability criterion. Note that this status and trends analysis omits the drought refugia in the upper watershed and southern tributaries, which were not monitored until 2017.

Big Sur Coast. Abundance of anadromous adults has been reported intermittently for the Big Sur River since 2012 (CDFW 2020), but the series is too short to estimate a trend (Figure 6.3). The average run size of the most recent four years of data was 42 fish, although these data were not considered to be full population estimates by CDFW (2020). The criterion for representation and redundancy specifies four core monitoring populations in Big Sur Coast BPG (Table 6.2), suggesting that three additional populations should be established and monitored for adult abundance.

Fish density has been reported for the steelhead population in Big Creek over the past 15 years by T. Williams and D. Rundio (personal communication). Densities here have been relatively stable, staying above 0.3 fish/m^2 except for three years at the end of the drought (Figure 6.4). Even so, this pattern created a statistically significant downward trend ($p < 0.05$), averaging -4% per year. The average density for the most recent four years captured the end-of-drought nadir at 0.258 fish/m^2 (Table 6.4).

San Luis Obispo Terrace. No data series have been reported by CDFW (2020) for this BPG, which has a viability criterion of 5 core monitoring populations with viable numbers of adult steelhead (Table 6.2).

Harvest Impacts[14]

Ocean harvest of steelhead is rare, and likely an insignificant source of mortality for South Central California Coast (SCCC) steelhead.

Since the early 1990s, anglers fishing for steelhead in anadromous portions of California waters have been required to purchase a steelhead report card. Information on the dates and locations of fishing, as well as the number of adult steelhead kept, the number of adult steelhead released, the origin of the fish caught (hatchery or wild) and the number of hours fished are must be reported (Jackson 2007; CDFW 2016). While anglers are required to report this information, average compliance rates are low, approximately 30 percent (CDFW 2016). Poor reporting of report card data and other data deficiencies precludes a rigorous assessment of harvest impacts.

California prohibits retention of natural-origin steelhead. Fishing effort estimates are not available for recent years (post-2014). There appears to have been little change to fishing regulations in the California South Central District in recent years.

Fishing closures owing to low flow conditions occur in portions of the SCCC steelhead DPS. Low flow closures based on river-specific flow thresholds are in place for the Carmel and Big Sur rivers, including adjacent streams. While these closures have the potential to reduce harvest impacts on steelhead, the data needed to evaluate the magnitude of such reductions are not currently available.

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[14] Michael O’Farrell (NMFS SWFSC Santa Cruz) prepared this section on harvest impacts.

In summary, while no direct information is available on the level of SCCC steelhead fishery impacts, it is reasonable to conclude that the level of impact has not changed appreciably in recent years.

Summary and Conclusions

The new information on genetic architecture of the steelhead life history indicates that the risk of its loss is not as great as was assumed in the precautionary stance adopted in the original viability criteria; at the same time it demonstrates the need for populations with at least periods when the frequency of AA individuals is high. The recent drought has evidently made such individuals quite rare, and the only indication that their frequency might have been high in recent decades is the relatively large number of anadromous steelhead observed in Carmel River in the late 1990s—early 2000s. Data on current adult abundances and low-flow fish densities both indicate that the recent drought had very large negative impacts on the DPS, with generally negative trends observed in all indicators, most with statistical significance (Table 3, Table 6.4). However, since the end of the drought in 2017 all indicators have ticked upward (Figure 6.3, Figure 6.4), suggesting that *O. mykiss* populations have persisted in drought refugia (e.g., lower Pajaro River tributaries, the upper Carmel River, the Big Sur Coast) and are now recovering from the drought. Yet the size of steelhead runs is extremely low, and the mean fish densities for the past four years are still below the provisional viability criterion of 0.3 fish/m^2. We do not know the frequency of AA genotypes in these populations.

A positive finding is that we now have a better understanding of the underlying genetic architecture that allows runs of steelhead to decline to zero during adverse conditions for anadromy, and then be reconstituted by populations of rainbow trout surviving in drought refugia when conditions improve. This is consistent with observations since listing of very low and/or erratic numbers of adult steelhead in the Salinas, Carmel, and Big Sur rivers, and is probably typical for the species in this region. However, additional synthesis work is needed to develop new risk-based viability criteria to replace the precautionary criteria originally developed by Boughton et al. (2007).

Unfortunately, the risk of permanently losing the anadromous phenotype over the longer term is still high and possibly increasing. Although the mechanics of the Omy5 “A” haplotype allow the anadromous fraction to decline to zero during droughts and other adverse periods and then later recover, it also effectively stops the haplotype from continuing to adapt to changing marine conditions during such periods. For long-term viability of the steelhead phenotype, populations will need periods where the anadromous fraction is high enough that AA fish occur in sufficient numbers to allow genetic recombination while buffering against genetic drift. We see no evidence for such conditions since the turn of the 21st Century, when anadromy was common in the Carmel population. Thus, even though the A haplotype provides a mechanism for the anadromous phenotype to weather droughts via rainbow trout, its continuing adaptive evolution appears to be curtailed and the viability of the anadromous phenotype thus remains at high and increasing risk.

Moreover, in subpopulations isolated above impassable dams, the A haplotype appears to be adapting to reservoir conditions, and gene flow downstream (over the dams) may further erode the anadromous phenotype in downstream populations over time if the selective pressures for reservoir life are distinctly different than the selective pressures for the marine phase of the anadromous phenotype. Since the marine phase involves a rigorous migration along thousands of miles to a broad band of habitat in the north Pacific (Atcheson et al. 2012), and reservoirs do not, it seems likely that selective pressures are indeed quite relaxed in reservoirs versus the ocean, with negative consequences for long-term viability of the anadromous phenotype.

Because of the functional association between the Omy5 “A” haplotype and outmigration, the frequency of the A haplotype in populations uninfluenced by reservoirs can be used as a lagging indicator for sustained past expression of the steelhead phenotype and thus to identify populations where it is being favored by natural selection (Pearse 2016; Funk et al 2012). Apgar et al (2017) inferred that when environmental conditions are completely adverse to anadromy over extended periods, the percent of A haplotypes in a population declines by about 5 percentage points per decade, and this gives some sense of the timescale for the loss of genetic capacity for anadromy. While the frequency of the A haplotype in a population can be viewed as a lagging indicator of successful anadromy in the past, the frequency of AA fish (i.e. homozygous fish) can be seen as an indicator for the possibility of continuing adaptive evolution in the present, which supports expression of the anadromous phenotype over the long term.

Monitoring of status and trends continues to be unsatisfactory in this DPS. A recent update to the monitoring strategy (Boughton et al. 2022) provides a resource to resolve the various ecological and methodological factors that impede effective monitoring. The main features of this monitoring strategy are:

- Estimates of mean 2D density for each BPG
- Data revealing the location and extent of drought refugia in each BPG
- Estimates of adult steelhead abundance in selected populations, sufficient to evaluate representation and redundancy
- Estimates of adult rainbow trout abundance, sufficient to evaluate total abundance of adult *O. mykiss* in the region
- Addition of routine genetic monitoring, to track the Omy5 A haplotype and AA genotype as indicators for viability
- Estimates of smolt production and marine survival in selected populations

6.2 Southern California Coast Steelhead DPS

DPS Delineation

The Southern California DPS of steelhead comprises the anadromous component of *Oncorhynchus mykiss* populations inhabiting coastal streams from the Santa Maria River system south to the US border with Mexico. Freshwater-resident (non-anadromous) *O. mykiss*, commonly known as rainbow trout, also occur in most of these coastal streams and appear to be members of the same Evolutionarily Significant Unit (ESU) as the originally listed steelhead (Clemento et al. 2009), with which they interbreed (Pearse et al. 2019). In most populations, adult rainbow trout outnumber adult steelhead by large margins and are not considered part of the ESA-protected DPS (71 FR 833). Anadromous and/or non-anadromous forms of the species also occur in some basins south of the US border, on the Baja California Peninsula (Ruiz-Campos and Pister 1995).

The DPS was divided by Boughton et al. (2007) into five biogeographic population groups (BPGs):

- The Monte Arido BPG consisting of the Santa Maria, Santa Ynez, Ventura and Santa Clara populations;
- The Conception Coast BPG consisting of coastal populations between the mouths of the Santa Ynez and Ventura rivers;
- The Santa Monica Mountains BPG consisting of coastal population between the Santa Clara River and the city of Los Angeles;
- The Mojave Rim BPG consisting of populations in the Los Angeles, San Gabriel and Santa Ana rivers; and
- The Santa Catalina Gulf Coast BPG consisting of coastal populations south of the mouth of the Santa Ana River to the U.S.-Mexico border.

Summary of Previous Assessments

Previous assessments started with Busby et al. (1996), leading to a listing of the ESU as endangered on August 18 1997. At that time the listing covered the geographic region from the Santa Maria River in the north to the Santa Monica Mountains in the south, with Topanga Creek steelhead believed to be the southern-most population. After subsequent documentation of steelhead further south (e.g., Hovey 2004), the southern range limit for the ESU was extended to the Tijuana River at the US border on 1 May 2002. The ESA listing was later modified to include only the anadromous component of the ESU on January 5 2006 (71 FR 833). The original status review of Busby et al. (1996) was updated by Good et al. (2005), Williams et al. (2011), and Williams et al. (2016), none of which led to changes in status of the DPS. A recovery plan was published in 2012 (NMFS 2012b).

The history of findings on genetic architecture summarized for the South-Central California Coast DPS largely apply to this DPS as well, and are not repeated here (see "Summary of

Previous Assessments" for that DPS). Both Omy5 haplotypes (A and R) are broadly distributed throughout populations of this DPS, but the frequency of the A haplotype has been negatively impacted by migration barriers, especially the complete barriers imposed by dams (Pearse et al. 2014; Apgar et al. 2017).

Additional genetic findings for populations in the far south were described by Jacobson et al. (2014) and Abadia-Cardoso et al. (2016), who examined the widely scattered residual populations of *O. mykiss* in the two southern-most BPGs, Mojave Rim and Santa Catalina Gulf Coast BPGs. Of the sites they examined in these two BPGs, they only found four groups with significant ancestry for coastal native steelhead: 1) sites from the San Gabriel River, 2) Coldwater Canyon Creek, a tributary of the Santa Ana River system, 3) the West Fork of the San Luis Rey River, and 4) a site in the Santa Domingo River in Baja California (currently recognized as a distinct subspecies *O. mykiss nelsoni*). Fish at other sites were found to be descended from planted hatchery strains such as the Mount Whitney strain and Kamloops strain. At one of three sites where Abadia-Cardoso et al. (2016) compared their recent samples to samples collected in the late 1990s, they found that a native strain had been largely replaced by a hatchery strain (Pauma Creek, a tributary of San Luis Rey River).

In their discussion, Abadia-Cardoso et al. (2016) stated that "the three groups in Southern California with substantial native ancestry [San Gabriel River system, Coldwater Canyon, and San Luis Rey River] … should be prioritized for conservation planning so as to ensure their persistence. However, some other populations, most notably Bear Creek in the Santa Ana River and Devil's Canyon Creek in the West Fork San Gabriel River, contained remnants of native ancestry overlaid with substantial introgressive hybridization with hatchery rainbow trout. While these populations are not pure native Southern California trout, they may be self-sustaining and adapting to the current local environment…In fact, the introduction of some novel genetic diversity from hatchery trout into these small, isolated, populations will likely increase heterozygosity, providing more variation to adapt to changing environmental conditions and reduce inbreeding."

New Data and Updated Analyses

Viability Criteria

The new material reviewed for viability criteria of the South-Central California Coast DPS largely apply to this DPS as well, and are not repeated here. For more information please see the Viability Criteria subheading under "New Data and Updated Analysis" for that DPS. Although the new analyses for the density criterion and the adult abundance criterion were derived from an analysis of the Carmel River population, I believe these findings to be applicable to this DPS as well. The overall ecology is similar though even more extreme in terms of arid conditions, such that the importance of drought refugia to population regulation is likely even more pronounced.

The findings described previously for genetic architecture and anadromous fraction were based on data collected across both the South-Central and Southern California steelhead DPSs, as well as the Central Coast steelhead DPS, and appear to be broadly applicable to coastal steelhead populations in California (Pearse et al. 2019).

Status and Trends

It should be noted that for most of California, the recent drought came to an end in 2017, but in southern California the drought conditions persisted for another two years. Figure 6.5 shows the record of annual precipitation for three important orographic drought refugia for steelhead in coastal California. In all three the drought commenced in 2012, but for the refugia in the Central Coast (Ben Lomond Mountain) and the South-Central Coast (Ventana Double Cone), annual precipitation returned to nearly average in 2016, and definitively broke the drought in 2017. However, in the southern refugium (Monte Arido), low rainfall persisted through 2018, though it was nearly average in 2017. Only in 2019 did the rainfall exceed the thirty-year average, and then by only a modest amount.

Monte Arido. Abundances of adult steelhead have been reported by CDFW (2020) for three of the four populations in this biogeographic area (Figure 6). For the Santa Ynez population, I summed counts from three separate trapping sites (Salsipuedes Creek, Hilton Creek, mainstem Santa Ynez); CDFW (2020) does not consider these to be full abundance estimates due to temporal limitation of trapping. Prior to the drought adult steelhead returned to the river in most years, peaking at 16 fish in 2008 but showing no returns in three years. Since 2012 no adult steelhead have been trapped in the Santa Ynez River at all (Figure 6), though the resulting downward trend is not statistically significant ($p = 0.12$; Table 6.5).

Similarly, the Ventura population had very modest numbers of adult returns prior to the drought, but no adult steelhead have been observed since 2010 (Figure 6). This downward trend was statistically significant ($p = 0.009$), averaging a decline of 12% per year (Table 6.5). In the Santa Clara River, adult steelhead were last reported in 2012, though no data at all were reported by CDFW (2020) after 2014 (Figure 6). Only four of the 13 years of data had nonzero abundances of adult steelhead.

Fish densities in the Santa Ynez population have been consistently monitored at four to 10 index sites per year since the 1990s (COMB Fisheries Division 2020). They used uncalibrated snorkel surveys so densities are likely biased slightly low, and they did not report wetted widths so we can only report 1D densities in Figure 7 (fish per meter of channel). Average densities were relatively stable through 2012, but then plummeted during the drought (Figure 7A); the most recent year of data (2019) gave the first hint of recovery, rising back to a density not observed since 2013. The negative trend associated with the drought was not statistically significant (Table 6.6).

No other long time series have been reported for O. mykiss density in the Monte Arido region, but 1- and 2-year series have been reported for various sites in the Santa Maria, Santa Ynez and Ventura populations (Dressler and Takata 2016; White et al. 2017; Hopkins et al. 2018; Lakish and Horgan 2018). Most of these were investigations of the downstream effects of the Zaca Fire, with some sites above impassable barriers (mostly Santa Ynez and Ventura), and some sites accessible to steelhead (mostly in the Sisquoc River and tributaries, a component of the Santa

Maria population). All were subjectively selected as index sites rather than random sites. In general these were uncalibrated snorkel surveys, so may be biased slightly low. Densities averaged 0.138 fish/m^2 at sites above barriers (not accessible to anadromous fish), but much higher at 0.564 fish/m^2 for sites accessible to anadromous fish (Table 6). One below-barrier study only counted fish in pools (omitting riffles), and found a much lower density of 0.0842 fish/m^2.

In the Santa Clara population, a single snorkel survey in the below-barrier portion of Piru Creek found a 1D density of 0.104 fish per meter of channel (Howard et al. 2015). Though this is not a 2D density estimate, it likely is below the viability threshold of 0.30 fish/m^2 because the wetted width of the creek was almost certainly wider than 1 m. In the inaccessible portion of Piru Creek above Santa Felicia Dam, Cramer Fish Sciences (2018) estimated density of O. mykiss using calibrated electrofishing at 6 random sites and 3 index sites (Table 6.6). The 1D density of the index sites was a full order of magnitude greater than at the random sites (0.144 vs. 0.0133 fish per meter of channel), but both were likely less than the 2D density criterion of 0.3 fish/m^2.

Conception Coast. The abundance of anadromous adults was monitored for three years (2014, 2016, and 2017) in the Carpenteria Creek population (CDFW 2020). Zero adult steelhead were observed each year (Figure 6), though the monitoring period was temporally limited and CDFW (2020) did not consider them to be full counts. No other data series have been reported by CDFW (2020) for this BPG, which has a viability criterion for 3 core monitoring populations (Table 6.2).

Santa Monica Mountains. Populations in this region have been very thoroughly monitored by Dagit et al. (2019). For abundance of adult steelhead, relatively long time-series of snorkel counts were collected for the Topanga, Malibu, and Arroyo Sequit populations (Figure 8). Over the past 19 to 15 years, each of these populations has fluctuated between less than 10 adults per year and zero adults per year. Only in the Malibu population did counts ever exceed two fish per year, for four of the 15 years monitored. Interestingly, no adults were observed in Arroyo Sequit from 2005 until the height of the drought, when one steelhead was observed in 2014 and two were observed in 2017. Redd counts in five other coastal streams were begun in 2014 and have yet to observe a single redd (Figure 8). CDFW (2020) did not consider either the snorkel surveys nor the redd surveys to be full counts due to spatial and temporal limitation.

For fish density, Dagit et al. (2019) conducted biweekly snorkel counts throughout the year and only reported annual averages rather than dry-season averages. One-dimensional densities averaged across Malibu, Topanga, and Arroyo Sequit are shown in Figure 7B. Assuming that these streams are at least a meter wide, mean density stayed below the threshold of 0.3 fish/m^2 during the entire period of record, and were less than 0.01 fish per meter of channel for five years during the drought. There appear to be modest upticks in density in 2017 and 2019, in part mirroring the five adult steelhead observed throughout the same three creeks in 2017 (Figure 8);

and in part mirroring the upticks in fish density observed in the Lower Santa Ynez population in 2017 and 2019 (Figure 7A).

Mojave Rim, Santa Catalina Gulf Coast. No data series have been reported for these biogeographic areas. The representation-and-redundancy criterion is 3 core monitoring populations for the Mojave Rim and 8 for the Santa Catalina Gulf Coast (Table 6.2).

Abadia-Cardoso et al. (2016) examined two linked markers for the Omy5 anadromous haplotype for fish sampled from these two BPGs. They found that the southernmost population of coastal steelhead in Baja Mexico (O. mykiss nelson), which was closely related to the remnant native population in the San Luis Rey River, had a relatively high frequency of the A haplotype, around 75% (see their Figure 6.5). In contrast the Coldwater Canyon, San Luis Rey, and San Gabriel sites with native ancestry had relatively low frequency of anadromous alleles, although the West Fork site had frequency of about 50%.

Entire range of DPS. Dagit et al. (2020) summarized the last 25 years of observations of adult steelhead in this DPS (including the recent data summarized above) and found that they are consistently very rare; indeed extremely rare—only 177 adult steelhead were observed during this period, an average of 7 per year for the entire geographic area inhabited by the DPS. Although some of this apparent rarity is due to lack of consistent monitoring, the streams reviewed above that have been consistently monitored in recent years largely corroborate the interpretation of true rarity rather than apparent rarity.

The recent findings of the genetic architecture underlying anadromy—reviewed in the section on the SCCC steelhead— provide a mechanism by which such rarity can be sustained over time, but also suggest that ongoing adaptation of the anadromous phenotype is inhibited by its very low expression. Unlike in SCCC steelhead, where favorable conditions for anadromy in the late 1990s briefly supported abundances of adult steelhead in the hundreds in the Carmel River, abundances of adult steelhead in the southern California DPS have been consistently low for a long time and indicate that ongoing evolutionary adaptation by AA fish is unlikely. This implies a high and increasing risk of permanent loss of the anadromous life history.

Harvest Impacts[15]

Ocean harvest of steelhead is rare, and likely an insignificant source of mortality for Southern California Coast (SC) steelhead. Eight coastal basins, including three not originally identified as supported steelhead populations

Since the early 1990s, anglers fishing for steelhead in anadromous portions of California waters have been required to purchase a steelhead report card. . Information on the dates and locations of fishing, as well as the number of adult steelhead kept, the number of adult steelhead released,

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[15] Michael O'Farrell (NMFS SWFSC Santa Cruz) prepared this section on harvest impacts..

the origin of the fish caught (hatchery or wild) and the number of hours fished are must be reported (Jackson 2007; CDFW 2016). While anglers are required to report this information, average compliance rates are low, approximately 30 percent (CDFW 2016).

Poor reporting of report card data and other data deficiencies precludes a rigorous assessment of harvest impacts. However, there are indicators suggesting that freshwater harvest impacts are particularly low for SC steelhead. First, California prohibits retention of natural-origin steelhead. Second, anadromous waters in the Southern District of California (south of, and including, Santa Barbara County) are closed to fishing all year. . Report card data indicates that between 2007 and 2014, only one steelhead fishing trip was reported in the SC steelhead DPS, representing less than one percent of total statewide steelhead trips.

In summary, there is no contemporary (post-2014) information on the level of SC steelhead fishery impacts, and fishing in anadromous waters of the DPS is closed. Therefore, it is reasonable to conclude that the level of harvest impacts is low for the SC steelhead DPS.

Summary and Conclusions

Unfortunately, the recent drought has been very hard on this DPS with no adult steelhead at all observed in many streams over the past five to seven years. In streams where adult steelhead runs were actually observed, the counts have been in the single digits. During the drought expression of the adult steelhead life history has nearly disappeared.

Fortunately, the recent findings on the genetic architecture of anadromy [reviewed in the section on the SCCC steelhead DPS, above] show that the anadromous phenotype can be reconstituted from populations of rainbow trout in drought refugia, if their gene pool contains the Omy5 "A" chromosome type. Prior to the era of dam construction and other anthropogenic impacts that limited that limited connectivity, this probably occurred naturally as rainbow trout populations in orographic drought refugia (perennial mountain streams) produced successful downstream migrants in years with sufficient rainfall to keep streams running to the ocean; and when enough of these "connection" years occurred with the right timing, the resulting adult steelhead then returned and were able to successfully ascend the streams and spawn.

Unfortunately, nearly all of the orographic drought refugia that might help steelhead abundance rapidly rebound are currently above impassable barriers. The extensive monitoring of the Santa Monica Mountains group of populations illustrates that it does not appear to have any substantial drought refugia that are currently accessible. The portions of Malibu Creek and tributaries upstream of Rindge Dam probably have an orographic refugium (e.g., Cold Creek), but it is currently inaccessible to steelhead. In other biogeographic population groups, important orographic refugia are almost universally isolated above impassable dams. The main exceptions are Sespe Creek and Santa Paula Creek, occupied by the Santa Clara population of steelhead, North Fork Matilija Creek occupied by the Ventura River population, and Sisquoc River, occupied by the Santa Maria population of steelhead, but these three populations suffer from accessibility problems stemming from altered flow regimes.

Thus, in contrast to the SCCC steelhead DPS, the Southern California has no drought refugia that stay connected to the ocean during droughts, and no evidence that any population has produced a run of adult steelhead larger than about 26 fish in the past 25 years; most runs in most years were apparently zero or less than 10 fish, though it is difficult to completely establish these low numbers due to lack of monitoring strategies that are temporally and spatially complete. Still, it appears that periods with anadromous runs in the hundreds or thousands of fish can be ruled out. This implies that ongoing adaptation of anadromous fish with the AA genotype has been curtailed for at least 25 years, and probably longer.

Orographic refugia with genetic resources for steelhead recovery that are isolated above impassable dams include *O. mykiss* populations in the upper Santa Ynez, upper Ventura, upper Piru Creek, and upper San Gabriel stream networks. All these populations retain the A haplotype associated with anadromy. Additional genetic resources with native ancestry are found at selected sites in the West Fork of the San Luis Rey River and a tributary of the Santa Ana River in Coldwater Canyon, although these populations had low incidence of the A haplotype. The outlying population of *O. mykiss* in the Santo Domingo River in Baja California, currently classified as a distinct subspecies, is closely related to this DPS, has a large fraction (75%) of the A haplotype, and may provide a useful genetic resource to consider for recovery.

Thus, as judged by distribution of the A haplotype, there appear to be substantial genetic resources in the isolated orographic refugia described above to meaningfully contribute to DPS recovery. However, the current situation poses two important challenges for recovery of the anadromous phenotype:

First, the anadromous phenotype in the below-barrier subpopulations is so rare that the Omy5 A haplotype is vulnerable to genetic drift and is likely prevented from recombination by a rarity of AA (homozygous) fish. This is ultimately driven by the low capacity of the below-barrier streams for *O. mykiss*. Even though below-barrier rainbow trout hold the potential to reconstitute the anadromous phenotype, they would not be expected to protect against these longer-term genetic changes.

Second, the A haplotype in the above-barrier subpopulations appears to be adapting to reservoir conditions. Since the selective regime imposed by reservoirs is probably distinctly different from the selective regime imposed by the Pacific Ocean, the genetic basis for anadromy will be progressively lost from these above-barrier populations despite the presence of the A haplotype.

Thus, the risk of permanently losing the anadromous phenotype over the longer term is very high and probably increasing due to the lack of migration corridors between drought refugia and the ocean.For long-term viability of the steelhead phenotype, populations will need periods where streams accessible to adult steelhead sustain the ecological capacity to support a high anadromous fraction in the local *O. mykiss* population. The anadromous fraction would need to be high enough that AA fish occur in sufficient numbers to allow genetic recombination while buffering against genetic drift. For example, the lower Santa Ynez River (downstream of Lake Cachuma) might have the capacity for such life-history expression in substantial quantities if it were not full of Largemouth Bass, an introduced fish that likely outcompetes juvenile steelhead in the upper end of their thermal niche (Boughton et al. 2015).

Over the shorter term, an intervention strategy is likely needed to use the existing stockpile of genetic resources in both the below-barrier subpopulations and the orographic refugia, and the establishment of rearing capacity in some form, to rebuild the anadromous phenotype.

In addition, monitoring of status and trends continues to be unsatisfactory in this DPS. A recent update to the monitoring strategy (Boughton et al. 2022) provides a resource to resolve the various ecological and methodological factors that impede effective monitoring. The main features of this monitoring strategy are:

- Estimates of mean 2D density for each BPG,
- Data revealing the location and extent of drought refugia in each BPG,
- Estimates of adult steelhead abundance in selected populations, sufficient to evaluate representation and redundancy,
- Estimates of adult rainbow trout abundance, sufficient to evaluate total abundance of adult *O. mykiss* in the region.
- Addition of routine genetic monitoring, to track the Omy5 A haplotype and AA genotype as indicators for viability
- Estimates of smolt production and marine survival in selected populations

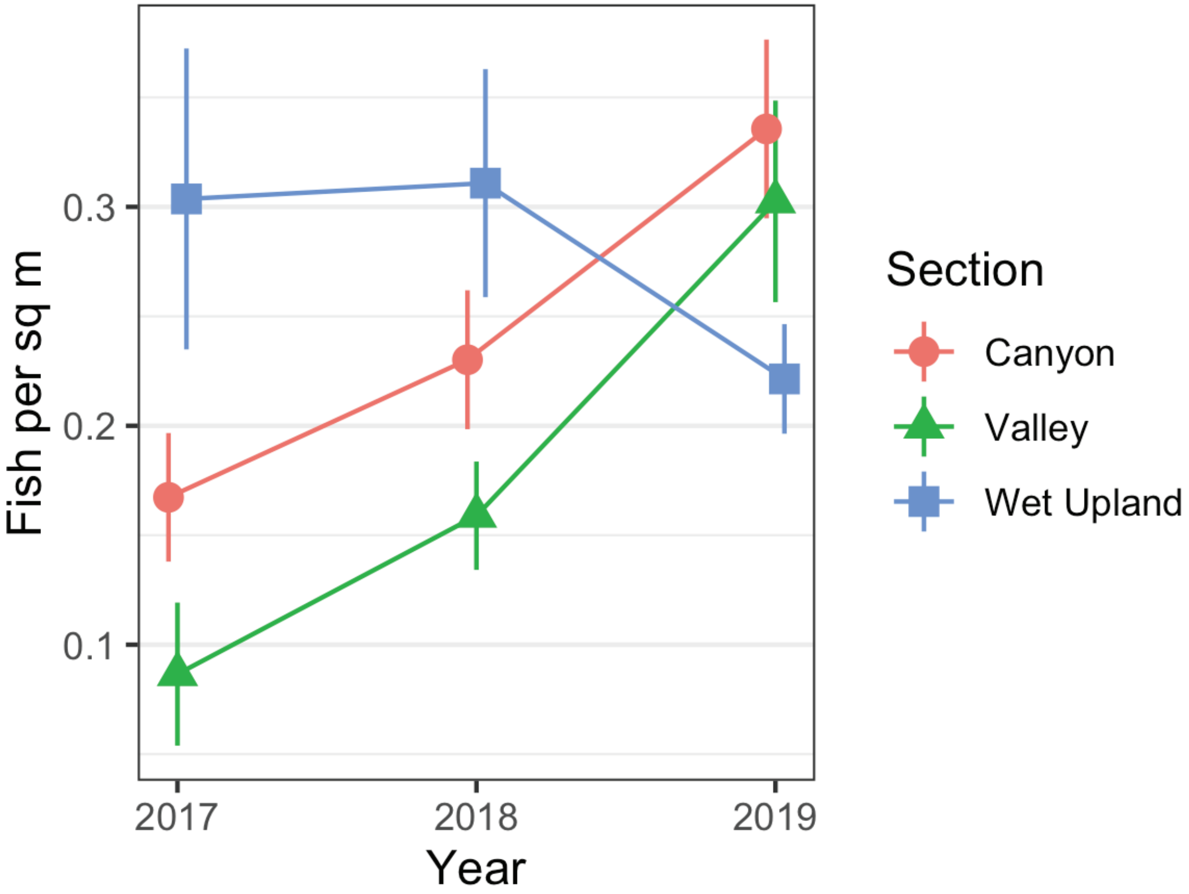


Figure 6.1. Dry-season steelhead densities in the Carmel River at the end of the recent drought, measured at sites in the Valley section of the mainstem (estuary up to Tularcitos Creek), Canyon section of the mainstem (Tularcitos Creek to Los Padres Dam), and wet upland sites (southern tributaries and the headwaters upstream of Los Padres Reservoir). Shown are mean and standard error of density for a collection of random and index sites in each section. In a statistical analysis (see Boughton et al. 2020), site type (random vs index) was not statistically significant, but section, year, and section-by-year interaction all had significant effects, demonstrating that the trend in steelhead densities differed by section.

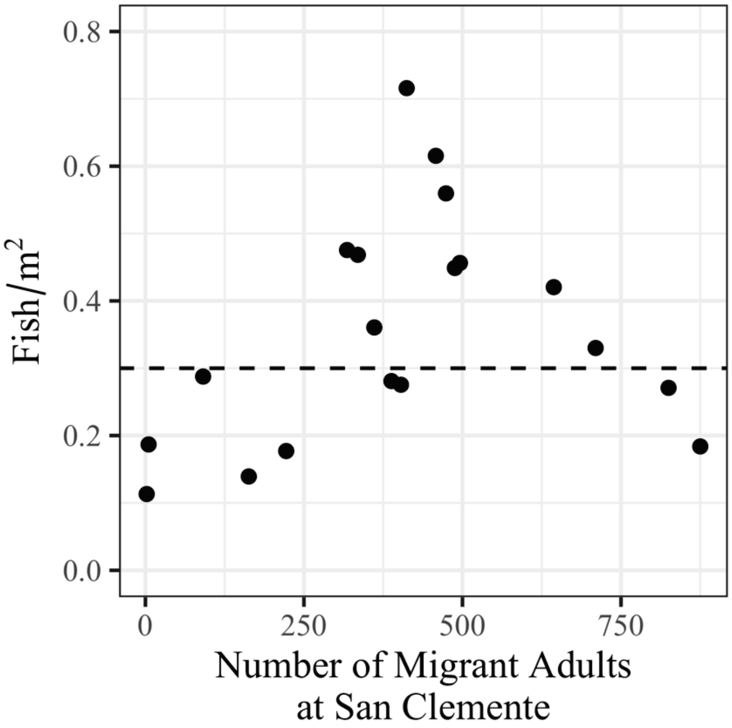


Figure 6.2. *O. mykiss* density during low-flow season in the Carmel River, as a function of the number of adult steelhead observed the previous winter. Each point represents a year between 1996 and 2015, with densities the mean of ~10 index sites distributed across the valley and canyon sections of the mainstem (the "wet upland" section of the river system from Figure 6.1 was not monitored during this period). The x-axis is the number of migrant steelhead that ascended the fish ladder at San Clemente Dam, which represents an index of run size but is only a partial count due to fish spawning downstream of the dam.

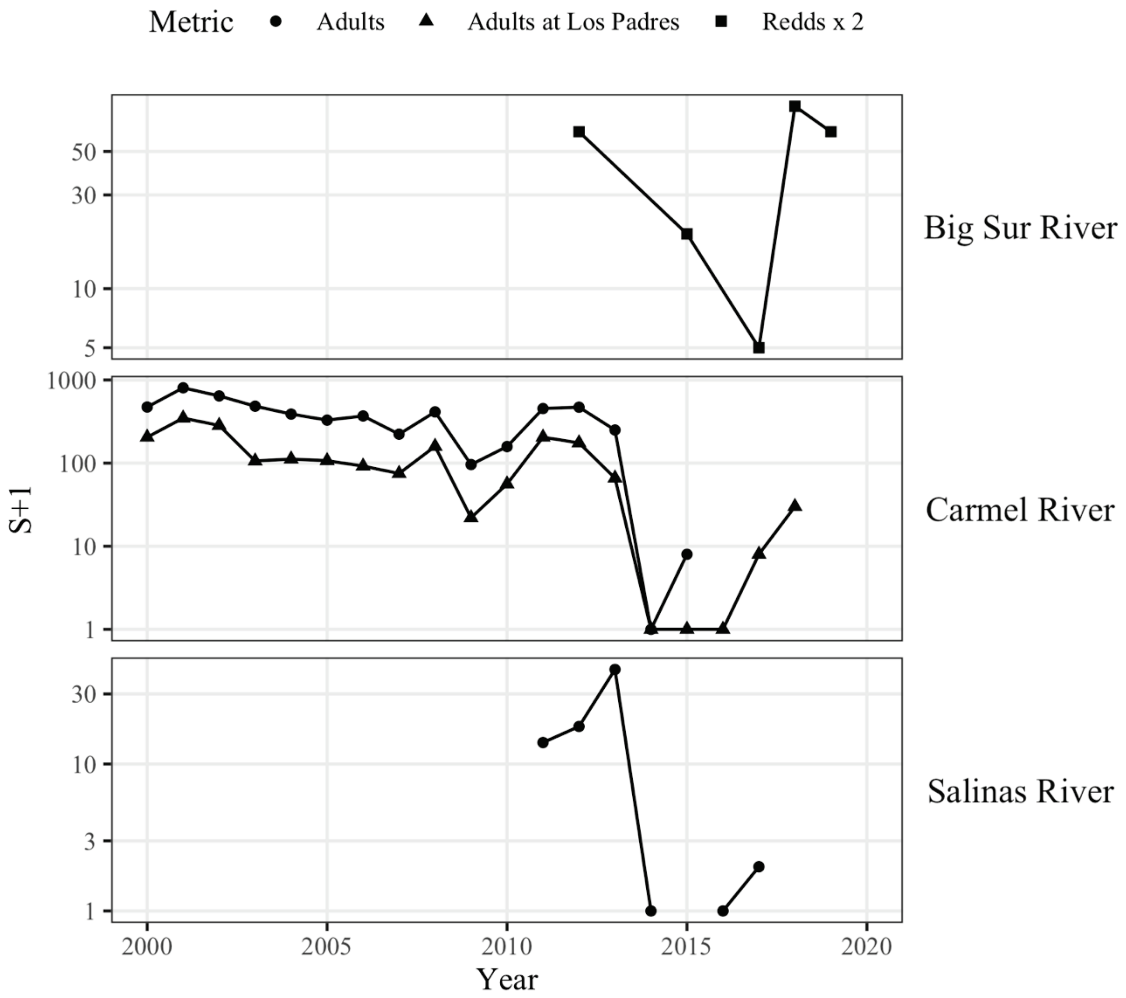


Figure 6.3. Trends in anadromous adults for populations of the South-Central California Coast DPS, compiled from various sources by CDFW (2020). In the Carmel River, round symbols show counts at San Clemente Dam, which was removed in 2015; triangles show concurrent and continuing counts at the Los Padres Dam fish trap upstream of the former San Clemente site.

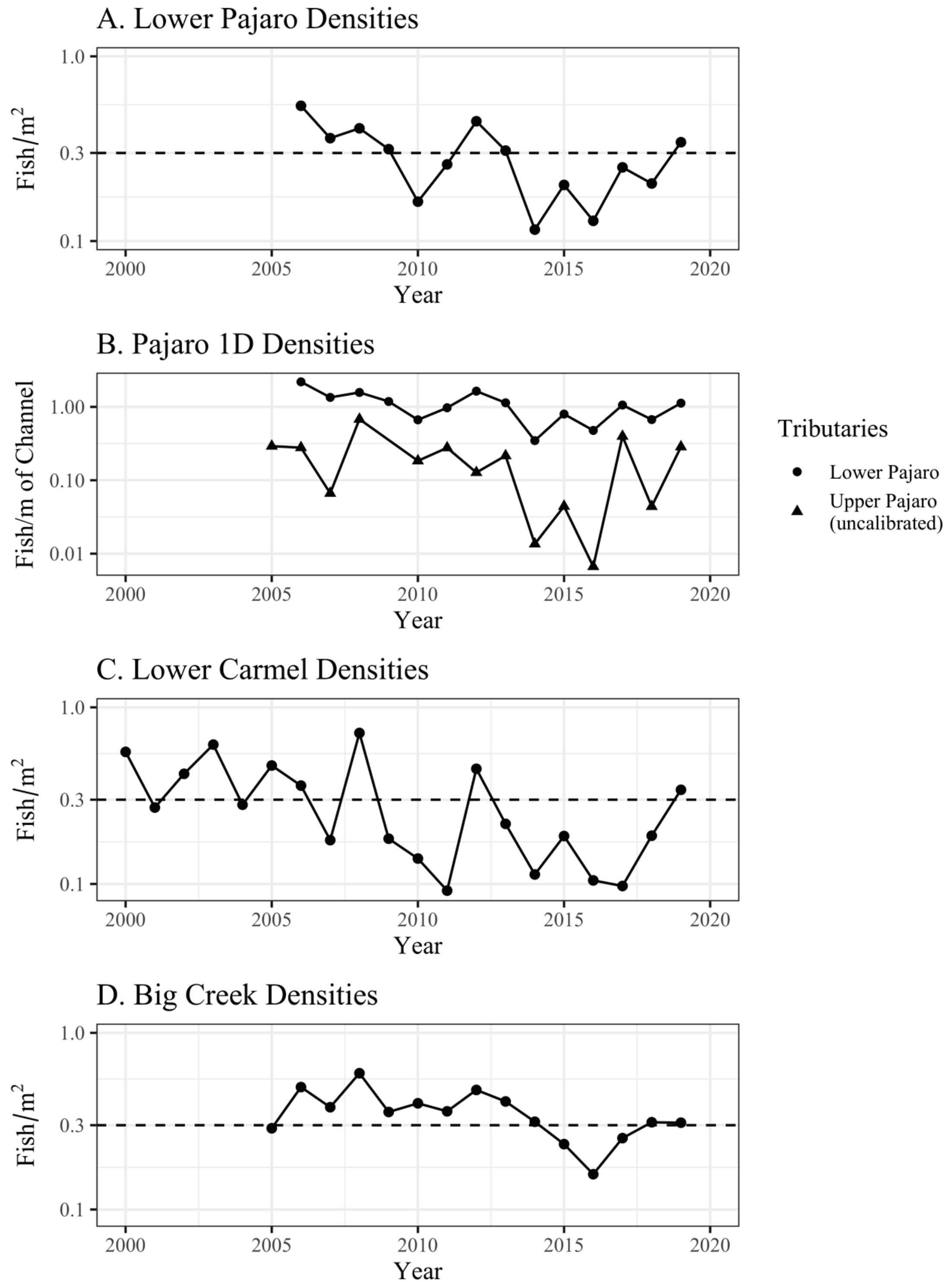


Figure 6.4. Low-flow densities of steelhead in the South-Central California Coast DPS. Densities are calibrated electrofishing estimates (depletion estimates) except for the upper Pajaro, which are from single-pass electrofishing. A – C are means for a set of index reaches.

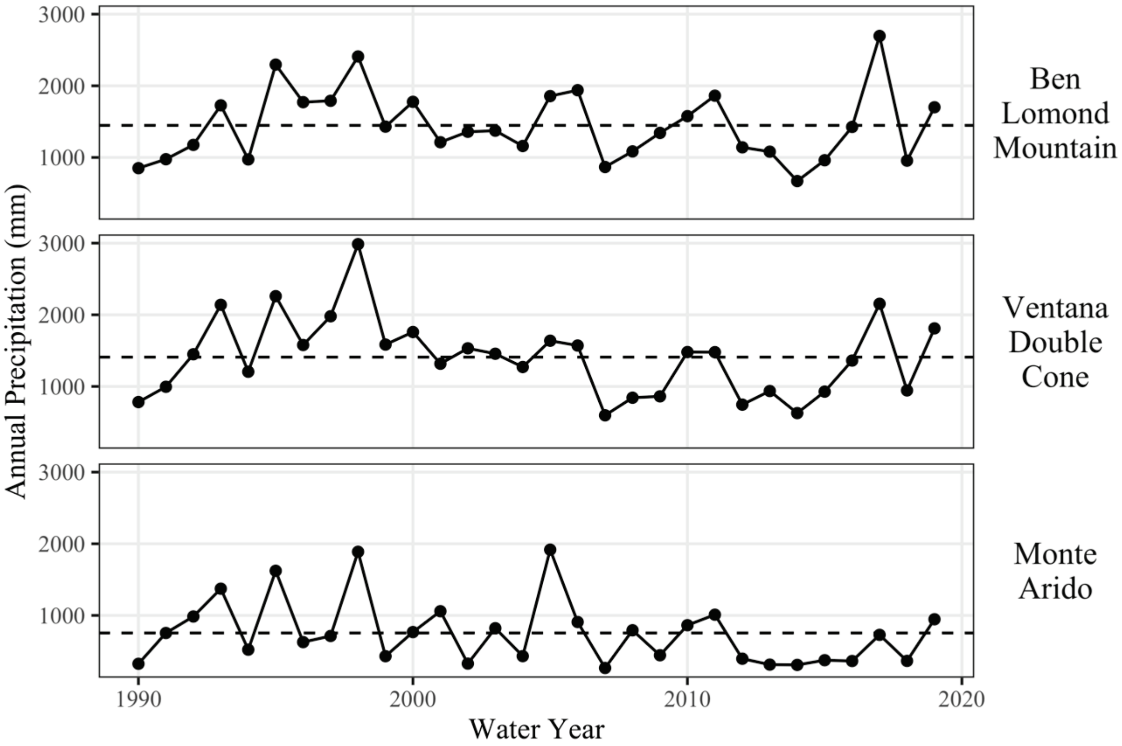


Figure 6.5. Thirty years of annual precipitation for three orographic drought refugia on the California coast south of the Golden Gate, arranged north to south. Ben Lomond Mountain supports the Scott, Waddell and San Lorenzo steelhead populations in the Central Coast DPS; Ventana Double Cone supports the Carmel, Big Sur and Little Sur populations in the South-Central Coast DPS; and Monte Arido supports the Santa Maria, Santa Ynez and Ventura populations in the Southern California DPS. Dashed lines represent the 30-year mean for each site; values are PRISM climate reconstructions (https://prism.oregonstate.edu/) for the 4-km grid-cell containing each mountain peak.

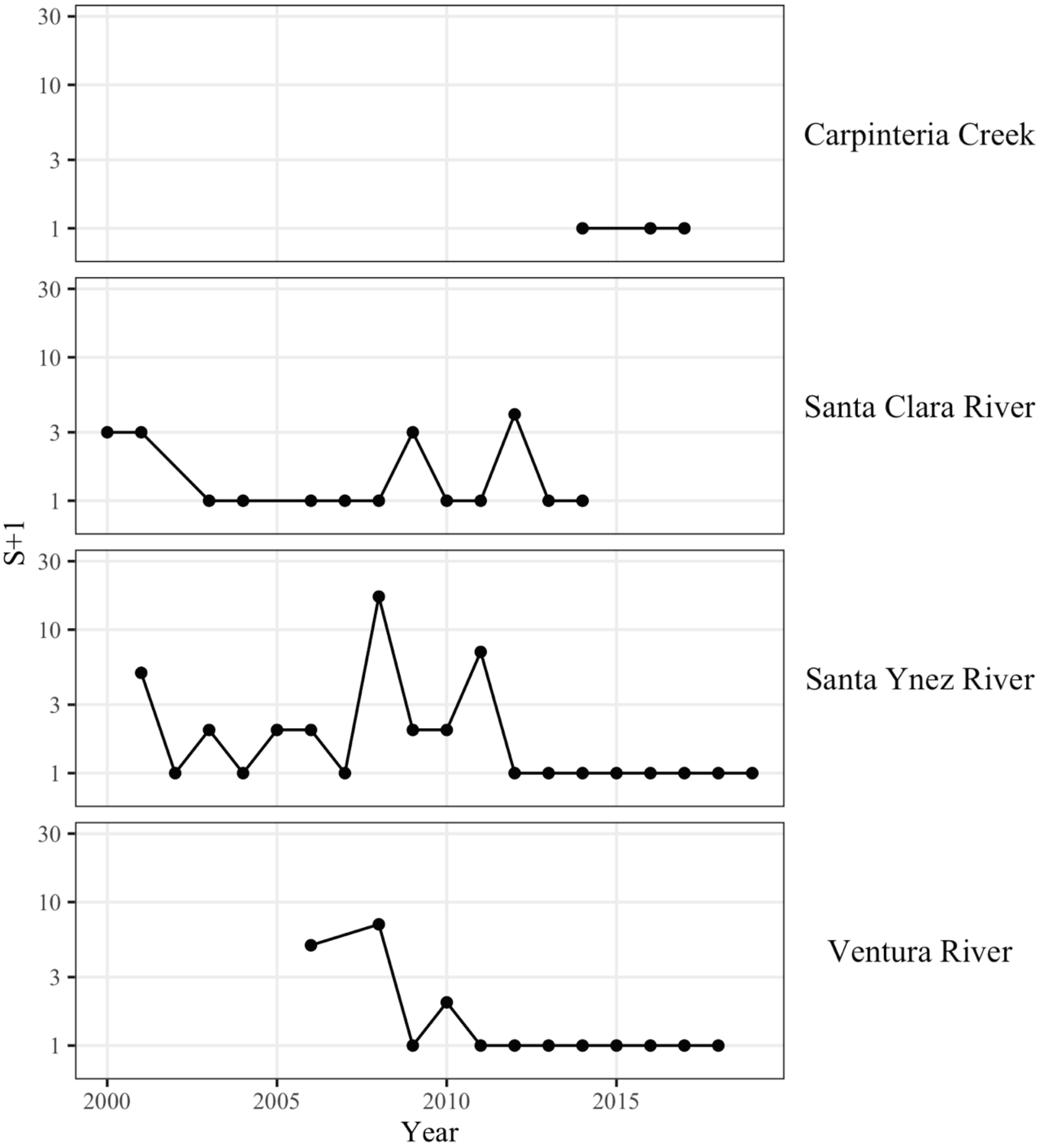
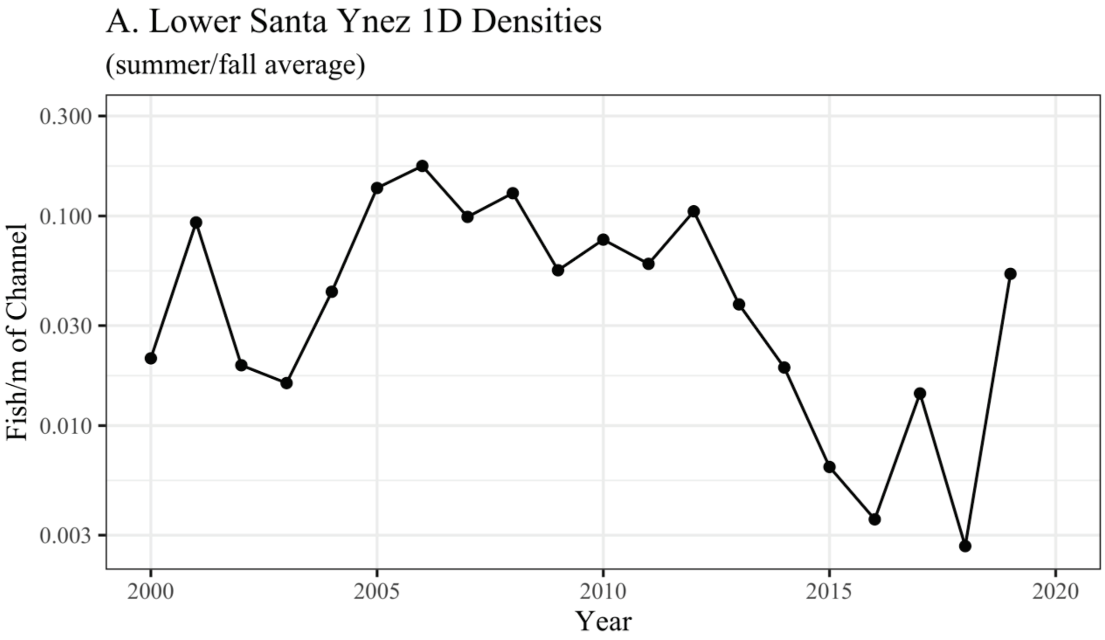


Figure 6.6. Counts of anadromous adults (possibly incomplete) for one population in the Conception Coast BPG (Carpinteria) and three populations in the Monte Arido BPG (Santa Clara, Santa Ynez and Ventura). Counts are adjusted by +1 so that zero counts show up on the log scale.



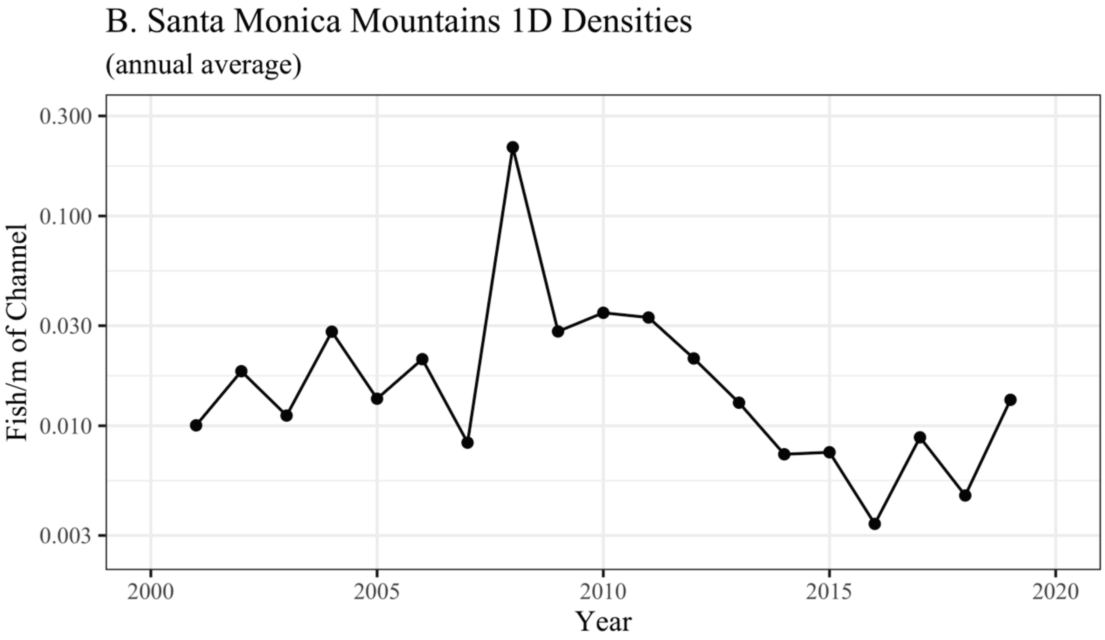


Figure 6.7. Low-flow densities of steelhead in the Southern California DPS. Densities are one-dimensional, uncalibrated snorkel counts.

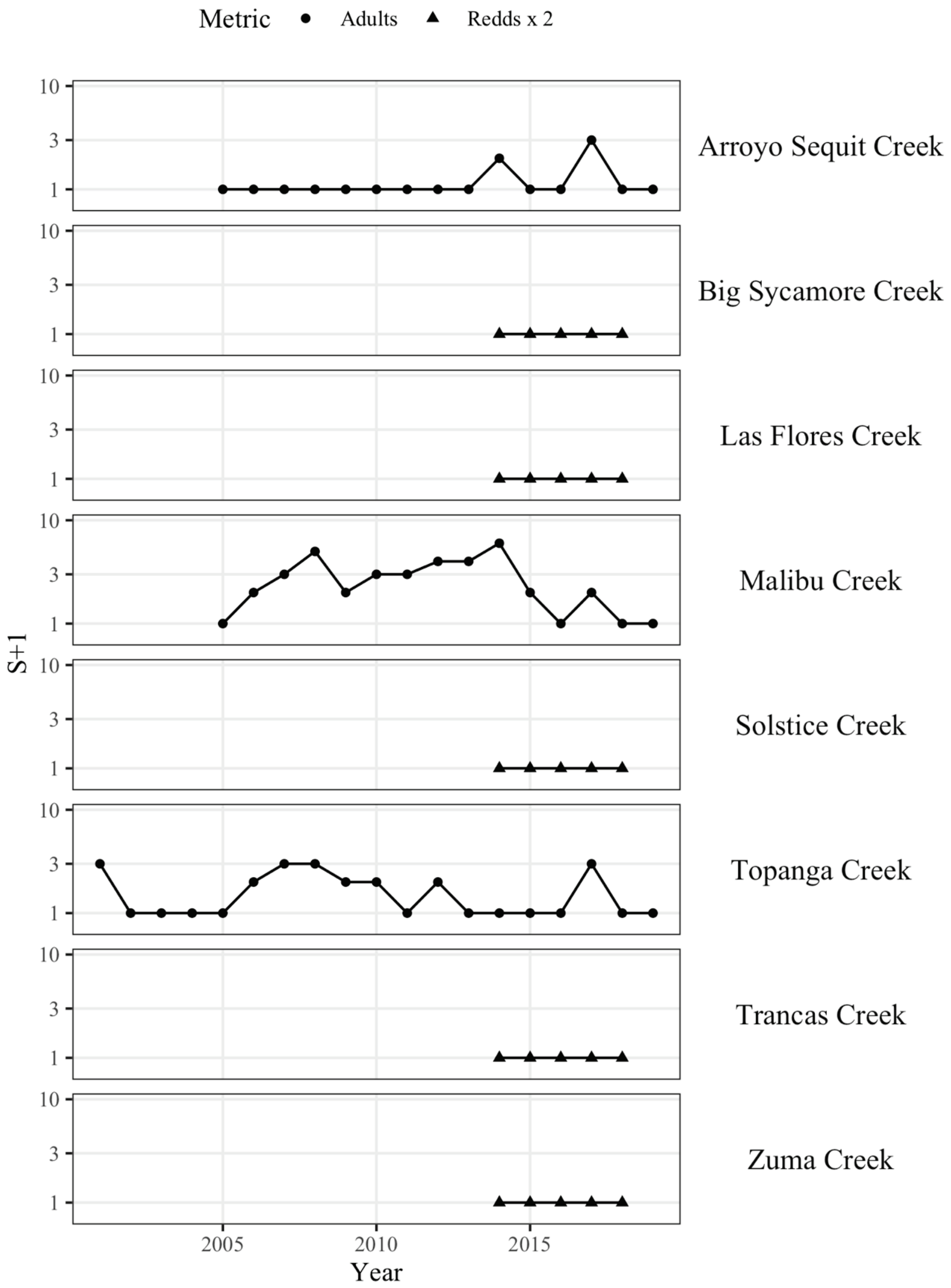


Figure 6.8. Counts of anadromous adults in eight populations of the Santa Monica Mountains BPG. Counts are adjusted by +1 so that zero counts show up on the log scale.

Table 6.1. Viability criteria for steelhead in the south-central and southern California coast recovery domain.

Criteria for Population Viability

Criterion	Viability threshold	Notes
Mean Annual Run Size	S > 4,150	Precautionary
Ocean Conditions	Size criterion met during poor ocean conditions	
Population Density	Unknown	Research Needed
Anadromous Fraction	100% of 4,150	Precautionary

Performance-Based Criteria

One or more prescriptive criteria (above) could be replaced by a quantitative risk assessment satisfying the following:

1) Extinction risk of anadromous population less than 5% in the next 100 yr.

2) Addresses each risk that is addressed by the prescriptive criteria it replaces.

3) Parameters are either a) estimated from data or b) precautionary

4) Quantitative methods are accepted practice in risk assessment/population viability analysis

5) Pass independent scientific review

Criteria for DPS Viability

Criterion	Viability threshold
Biogeographic Diversity	1) Sufficient numbers of viable populations in each biogeographic group (Table 6.2) 2) Viable populations inhabit watersheds with drought refugia 3) Viable populations in basins separated by >68km if possible
Life-history Diversity	Viable populations exhibit three life-history types (fluvial-anadromous, lagoon-anadromous, resident)

Table 6.2. Representation and redundancy criteria for viable populations.

Biogeographic Population Group	Total number of populations	Number of Core Monitoring Populations
South-Central California Coast DPS		
Interior Coast Range	4	4
Carmel Basin	1	1
Big Sur Coast	11	3
San Luis Obispo Terrace	15	5
Southern California DPS		
Monte Arido Highlands	4	4
Conception Coast	29	3
Mojave Rim	3	3
Santa Monica Mountains	5	3
Santa Catalina Gulf Coast	10	8

Table 6.3. Adult abundance and trends in the South-Central California Coast DPS. $\hat{S}$ is the average number of adult steelhead per year for the most recent four years; Trend is the slope of a regression line fit to log-transformed annual runs

Target of estimation	Years	Full population estimate?	$\hat{S}$	Trend (SE)	p
Interior Coast Range					
Salinas River[a]	6	Yes	0.25		
Carmel					
Los Padres Trap[b]	20	No	9	-0.105 (0.025)	0.0007
Big Sur Coast					
Big Sur River	5	No	42		
SLO Terrace					
No data reported					

a - Data cover period 2011-2017; no data reported for 2018–2019 or 2015. $\hat{S}$ estimated from most recent four years of data.

b - Counts at Los Padres fish trap omit on the order of two-thirds of the Carmel River steelhead run.

Table 6.4. Low-flow freshwater fish density and trends in the South-Central California Coast DPS.

Target of estimation	Yrs	Density unit	Density[a]	Trend (SE)[b]	Density of occupied[c]	Occupied[d]
			Uncalibrated electrofishing			
Interior Coast Range						
Upper Pajaro Tribs.	14	1D	0.183	-0.0256 (0.0090)*	-	-
			Calibrated electrofishing			
Interior Coast Range						
Lower Pajaro Tribs.	14	2D	0.232	-0.0234 (0.0121)	0.232	1.0
Carmel						
Lower Carmel R.	20	2D	0.183	-0.0271 (0.0091)**	0.183	1.0
Big Sur Coast						
Big Creek	15	2D	0.258	-0.0173 (0.0074)*	0.258	1.0
			No data reported			
SLO Terrace						

a – 1D densities are fish per meter or stream channel during the low-flow season; 2D densities are fish per square meter of wetted area.

b – Trend is estimated as the slope parameter from a linear regression of $\log_{10}$(density) on year. Proportional change per year is 10^x where x = trend statistic in the table. Single and double asterisks are statistical significance level at $p < 0.05$ and $p < 0.01$ respectively.

c – Density of species within occupied habitat (i.e. mean density omitting reaches where species was not observed).

d – Proportion of occupied habitat.

Table 6.5. Adult abundance and trends in the Southern California DPS.

Target of estimation	Yrs	Full population estimate?	$\hat{S}$	Trend (SE)	p
Conception Coast					
Carpenteria Cr.[a]	3	No	0		
Monte Arido					
Santa Ynez R.	19	No	0	-0.0229 (0.0140)	0.12
Ventura R.	12	No	0	-0.0577 (0.0178)	0.009**
Santa Clara R.[b]	13	No	0.75		
Santa Monica Mtns.					
Arroyo Sequit	15	No	0.5	0.0107 (0.0082)	0.22
Big Sycamore Cr.	5	No	0		
Los Flores Cr.	5	No	0		
Malibu Cr.	15	No	0.25	-0.0170 (0.0158)	0.30
Solstice Cr.	5	No	0		
Topanga Cr.	19	No	0.5	-0.0074 (0.0087)	0.40
Trancas Cr.	5	No	0		
Zuma Cr.	5	No	0		
Mojave Rim					
No data reported					
St. Catalina Gulf Coast					
No data reported					

a - Data cover period 2014-2017; $\hat{S}$ estimated from three available years of data.

b - Data cover period 2000-2014; no data reported for 2015-2019. $\hat{S}$ estimated from most recent four years of data.

Table 6.6. Low-flow freshwater fish density and trends in the Southern California DPS.

Target of estimation	Yrs	Density uUnit	Density[a]	Trend (SE)[b]	Density of occupied[c]	Occupied[d]
			Uncalibrated snorkel surveys			
Monte Arido						
Lower St. Ynez	20	1D	0.114	-0.0386 (0.0189)		
Lower Piru	1	1D	0.104		0.104	1.0
BPG below barriers	2	2D	0.564		0.564	1.0
BPG above barriers	2	2D	0.138		0.276	0.5
BPG below barriers (pools only)	1	2D	0.0842		0.112	0.75
Santa Monica Mtns.	19	1D	0.00754	-0.0252 (0.0159)		
			Calibrated electrofishing			
Monte Arido						
Upper Piru (index)	1	1D	0.144		0.217	0.67
Upper Piru (random)	1	1D	0.0133		0.020	0.67
			No data reported			
Conception Coast						
Mojave Rim						
St. Catalina Gulf Coast						

a – 1D densities are fish per meter or stream channel during the low-flow season; 2D densities are fish per square meter of wetted area.

b – Trend is estimated as the slope parameter from a linear regression of $\log_{10}$(density) on year. Proportional change per year is 10^x where x = trend statistic in the table. Single and double asterisks are statistical significance level at $p < 0.05$ and $p < 0.01$ respectively.

c – Density of species within occupied habitat (i.e. mean density omitting reaches where species was not observed).

d – Proportion of occupied habitat.

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Appendix B. Sources of data used in assessment of status of populations of salmon and steelhead in the Central Valley Recovery Domain. (Section 5).

PopID	Year	CountType	Total	TotalYears	N	logTotal	S	popseries	logPop	delta_logPop	decline
livingstonhatch.chin.win	1998	Hatchery transfer	99	22	NA	4.6	NA	NA	NA	NA	NA
livingstonhatch.chin.win	1999	Hatchery transfer	24	22	NA	3.18	NA	NA	NA	NA	NA
livingstonhatch.chin.win	2000	Hatchery transfer	89	22	212	4.49	71	NA	NA	NA	NA
livingstonhatch.chin.win	2001	Hatchery transfer	104	22	217	4.64	72	316	5.76	NA	NA
livingstonhatch.chin.win	2002	Hatchery transfer	104	22	297	4.64	99	321	5.77	NA	NA
livingstonhatch.chin.win	2003	Hatchery transfer	85	22	293	4.44	98	382	5.95	NA	NA
livingstonhatch.chin.win	2004	Hatchery transfer	85	22	274	4.44	91	378	5.93	0.18	-19.62
livingstonhatch.chin.win	2005	Hatchery transfer	109	22	279	4.69	93	383	5.95	0.18	-19.31
livingstonhatch.chin.win	2006	Hatchery transfer	93	22	287	4.53	96	372	5.92	-0.03	2.62
livingstonhatch.chin.win	2007	Hatchery transfer	54	22	256	3.99	85	341	5.83	-0.1	9.79
livingstonhatch.chin.win	2008	Hatchery transfer	105	22	252	4.65	84	361	5.89	-0.06	5.74
livingstonhatch.chin.win	2009	Hatchery transfer	121	22	280	4.8	93	373	5.92	0	-0.27
livingstonhatch.chin.win	2010	Hatchery transfer	63	22	289	4.14	96	343	5.84	0.01	-0.59
livingstonhatch.chin.win	2011	Hatchery transfer	86	22	270	4.45	90	375	5.93	0.04	-3.88
livingstonhatch.chin.win	2012	Hatchery transfer	93	22	242	4.53	81	363	5.89	-0.03	2.68
livingstonhatch.chin.win	2013	Hatchery transfer	164	22	343	5.1	114	406	6.01	0.17	-18.37
livingstonhatch.chin.win	2014	Hatchery transfer	388	22	645	5.96	215	731	6.59	0.67	-94.93
livingstonhatch.chin.win	2015	Hatchery transfer	257	22	809	5.55	270	902	6.8	0.91	-148.48
livingstonhatch.chin.win	2016	Hatchery transfer	137	22	782	4.92	261	946	6.85	0.85	-133
livingstonhatch.chin.win	2017	Hatchery transfer	180	22	574	5.19	191	962	6.87	0.27	-31.6
livingstonhatch.chin.win	2018	Hatchery transfer	180	22	497	5.19	166	754	6.63	-0.18	16.41
livingstonhatch.chin.win	2019	Hatchery transfer	180	22	540	5.19	180	677	6.52	-0.33	28.44
sacramentoriv.chin.win	1970	Dam	40409	50	NA	10.61	NA	NA	NA	NA	NA
sacramentoriv.chin.win	1971	Dam	53089	50	NA	10.88	NA	NA	NA	NA	NA
sacramentoriv.chin.win	1972	Dam	35929	50	129427	10.49	43142	NA	NA	NA	NA
sacramentoriv.chin.win	1973	Dam	22651	50	111669	10.03	37223	152078	11.93	NA	NA

PopID	Year	CountType	Total	TotalYears	N	logTotal	S	popseries	logPop	delta_logPop	decline
sacramentoriv.chin.win	1974	Dam	21389	50	79969	9.97	26656	133058	11.8	NA	NA
sacramentoriv.chin.win	1975	Dam	22579	50	66619	10.02	22206	102548	11.54	NA	NA
sacramentoriv.chin.win	1976	Dam	33029	50	76997	10.41	25666	99648	11.51	-0.42	34.48
sacramentoriv.chin.win	1977	Dam	16470	50	72078	9.71	24026	93467	11.45	-0.35	29.75
sacramentoriv.chin.win	1978	Dam	24735	50	74234	10.12	24745	96813	11.48	-0.06	5.59
sacramentoriv.chin.win	1979	Dam	2339	50	43544	7.76	14515	76573	11.25	-0.26	23.16
sacramentoriv.chin.win	1980	Dam	1142	50	28216	7.04	9405	44686	10.71	-0.74	52.19
sacramentoriv.chin.win	1981	Dam/Redd	22551	50	26032	10.02	8677	50767	10.84	-0.65	47.56
sacramentoriv.chin.win	1982	Dam/Redd	1272	50	24965	7.15	8322	27304	10.21	-1.03	64.34
sacramentoriv.chin.win	1983	Dam	1827	50	25650	7.51	8550	26792	10.2	-0.51	40.04
sacramentoriv.chin.win	1984	Dam	2662	50	5761	7.89	1920	28312	10.25	-0.58	44.23
sacramentoriv.chin.win	1985	Dam/Redd	5131	50	9620	8.54	3207	10892	9.3	-0.92	60.11
sacramentoriv.chin.win	1986	Dam	2566	50	10359	7.85	3453	12186	9.41	-0.79	54.52
sacramentoriv.chin.win	1987	Dam/Redd	2165	50	9862	7.68	3287	12524	9.44	-0.82	55.76
sacramentoriv.chin.win	1988	Dam/Redd	2857	50	7588	7.96	2529	12719	9.45	0.16	-16.77
sacramentoriv.chin.win	1989	Dam/Redd	649	50	5671	6.48	1890	8237	9.02	-0.39	32.41
sacramentoriv.chin.win	1990	Dam/Redd	412	50	3918	6.02	1306	6083	8.71	-0.72	51.43
sacramentoriv.chin.win	1991	Dam/Redd	177	50	1238	5.18	413	4095	8.32	-1.13	67.8
sacramentoriv.chin.win	1992	Dam/Redd	1203	50	1792	7.09	597	2441	7.8	-1.22	70.37
sacramentoriv.chin.win	1993	Dam/Redd	378	50	1758	5.93	586	2170	7.68	-1.03	64.33
sacramentoriv.chin.win	1994	Dam/Redd	144	50	1725	4.97	575	1902	7.55	-0.77	53.55
sacramentoriv.chin.win	1995	Dam/Redd	1166	50	1688	7.06	563	2891	7.97	0.17	-18.44
sacramentoriv.chin.win	1996	Dam/Redd	1012	50	2322	6.92	774	2700	7.9	0.22	-24.42
sacramentoriv.chin.win	1997	Dam/Redd	836	50	3014	6.73	1005	3158	8.06	0.51	-66.04
sacramentoriv.chin.win	1998	Dam/Redd	2893	50	4741	7.97	1580	5907	8.68	0.71	-104.32
sacramentoriv.chin.win	1999	Dam/Redd	3264	50	6993	8.09	2331	8005	8.99	1.09	-196.48
sacramentoriv.chin.win	2000	Dam/Redd	1261	50	7418	7.14	2473	8254	9.02	0.96	-161.37
sacramentoriv.chin.win	2001	Carcass/Redd surveys	8120	50	12645	9	4215	15538	9.65	0.97	-163.04
sacramentoriv.chin.win	2002	Carcass/Redd surveys	7337	50	16718	8.9	5573	19982	9.9	0.91	-149.62
sacramentoriv.chin.win	2003	Carcass/Redd surveys	8133	50	23590	9	7863	24851	10.12	1.1	-201.08

PopID	Year	CountType	Total	TotalYears	N	logTotal	S	popseries	logPop	delta_logPop	decline
sacramentoriv.chin.win	2004	Carcass/Redd surveys	7784	50	23254	8.96	7751	31374	10.35	0.7	-101.92
sacramentoriv.chin.win	2005	Carcass/Redd surveys	15730	50	31647	9.66	10549	38984	10.57	0.67	-95.1
sacramentoriv.chin.win	2006	Carcass/Redd surveys	17197	50	40711	9.75	13570	48844	10.8	0.68	-96.55
sacramentoriv.chin.win	2007	Carcass/Redd surveys	2487	50	35414	7.82	11805	43198	10.67	0.32	-37.69
sacramentoriv.chin.win	2008	Carcass/Redd surveys	2725	50	22409	7.91	7470	38139	10.55	-0.02	2.17
sacramentoriv.chin.win	2009	Carcass/Redd surveys	4416	50	9628	8.39	3209	26825	10.2	-0.6	45.08
sacramentoriv.chin.win	2010	Carcass/Redd surveys	1533	50	8674	7.33	2891	11161	9.32	-1.35	74.16
sacramentoriv.chin.win	2011	Carcass/Redd surveys	738	50	6687	6.6	2229	9412	9.15	-1.4	75.32
sacramentoriv.chin.win	2012	Carcass/Redd surveys	2578	50	4849	7.85	1616	9265	9.13	-1.06	65.46
sacramentoriv.chin.win	2013	Carcass/Redd surveys	5920	50	9236	8.69	3079	10769	9.28	-0.04	3.51
sacramentoriv.chin.win	2014	Carcass/Redd surveys	2627	50	11125	7.87	3708	11863	9.38	0.23	-26.04
sacramentoriv.chin.win	2015	Carcass/Redd surveys	3182	50	11729	8.07	3910	14307	9.57	0.43	-54.42
sacramentoriv.chin.win	2016	Carcass/Redd surveys	1409	50	7218	7.25	2406	13138	9.48	0.2	-22
sacramentoriv.chin.win	2017	Carcass/Redd surveys	795	50	5386	6.68	1795	8013	8.99	-0.39	32.45
sacramentoriv.chin.win	2018	Carcass/Redd surveys	2458	50	4662	7.81	1554	7844	8.97	-0.6	45.17
sacramentoriv.chin.win	2019	Carcass/Redd surveys	7853	50	11106	8.97	3702	12515	9.43	-0.05	4.74
antelopecrk.chin.spr	1983	Snorkel survey	59	29	NA	4.08	NA	NA	NA	NA	NA
antelopecrk.chin.spr	1992	Snorkel survey	0	29	NA	-11.51	NA	NA	NA	NA	NA
antelopecrk.chin.spr	1993	Snorkel survey	3	29	62	1.1	21	NA	NA	NA	NA
antelopecrk.chin.spr	1994	Snorkel survey	0	29	3	-11.51	1	62	4.13	NA	NA
antelopecrk.chin.spr	1995	Snorkel survey	7	29	10	1.95	3	10	2.3	NA	NA
antelopecrk.chin.spr	1996	Snorkel survey	1	29	8	0	3	11	2.4	NA	NA
antelopecrk.chin.spr	1997	Snorkel survey	0	29	8	-11.51	3	8	2.08	-2.05	87.1
antelopecrk.chin.spr	1998	Snorkel survey	154	29	155	5.04	52	162	5.09	2.79	-1520
antelopecrk.chin.spr	1999	Snorkel survey	40	29	194	3.69	65	195	5.27	2.88	-1672.73
antelopecrk.chin.spr	2000	Snorkel survey	9	29	203	2.2	68	203	5.31	3.23	-2437.49
antelopecrk.chin.spr	2001	Snorkel survey	8	29	57	2.08	19	211	5.35	0.26	-30.25
antelopecrk.chin.spr	2002	Snorkel survey	46	29	63	3.83	21	103	4.63	-0.64	47.18
antelopecrk.chin.spr	2003	Snorkel survey	46	29	100	3.83	33	109	4.69	-0.62	46.31
antelopecrk.chin.spr	2004	Snorkel survey	3	29	95	1.1	32	103	4.63	-0.72	51.18

PopID	Year	CountType	Total	TotalYears	N	logTotal	S	popseries	logPop	delta_logPop	decline
antelopecrk.chin.spr	2005	Snorkel survey	82	29	131	4.41	44	177	5.18	0.54	-71.84
antelopecrk.chin.spr	2006	Snorkel survey	102	29	187	4.62	62	233	5.45	0.76	-113.76
antelopecrk.chin.spr	2007	Snorkel survey	26	29	210	3.26	70	213	5.36	0.73	-106.8
antelopecrk.chin.spr	2008	Snorkel survey	3	29	131	1.1	44	213	5.36	0.19	-20.34
antelopecrk.chin.spr	2009	Snorkel survey	0	29	29	-11.51	10	131	4.88	-0.58	43.78
antelopecrk.chin.spr	2010	Snorkel survey	17	29	20	2.83	7	46	3.83	-1.53	78.4
antelopecrk.chin.spr	2011	Snorkel survey	6	29	23	1.79	8	26	3.26	-2.1	87.79
antelopecrk.chin.spr	2012	Snorkel survey	1	29	24	0	8	24	3.18	-1.7	81.68
antelopecrk.chin.spr	2013	Snorkel survey	0	29	7	-11.51	2	24	3.18	-0.65	47.83
antelopecrk.chin.spr	2014	Snorkel survey	7	29	8	1.95	3	14	2.64	-0.62	46.15
antelopecrk.chin.spr	2015	Snorkel survey	5	29	12	1.61	4	13	2.56	-0.61	45.83
antelopecrk.chin.spr	2016	Snorkel survey	7	29	19	1.95	6	19	2.94	-0.23	20.83
antelopecrk.chin.spr	2017	Snorkel survey	8	29	20	2.08	7	27	3.3	0.66	-92.86
antelopecrk.chin.spr	2018	Snorkel survey	1	29	16	0	5	21	3.04	0.48	-61.54
antelopecrk.chin.spr	2019	Snorkel survey	7	29	16	1.95	5	23	3.14	0.19	-21.05
battlecrk.chin.spr	1989	Hatchery transfer	7	27	NA	1.95	NA	NA	NA	NA	NA
battlecrk.chin.spr	1990	Hatchery transfer	2	27	NA	0.69	NA	NA	NA	NA	NA
battlecrk.chin.spr	1995	Snorkel/Redd count	66	27	75	4.19	25	NA	NA	NA	NA
battlecrk.chin.spr	1996	Snorkel/Redd count	35	27	103	3.56	34	110	4.7	NA	NA
battlecrk.chin.spr	1997	Snorkel/Redd count	107	27	208	4.67	69	210	5.35	NA	NA
battlecrk.chin.spr	1998	Snorkel/Redd count	178	27	320	5.18	107	386	5.96	NA	NA
battlecrk.chin.spr	1999	Snorkel/Redd count	73	27	358	4.29	119	393	5.97	1.27	-257.27
battlecrk.chin.spr	2000	Weir count	78	27	329	4.36	110	436	6.08	0.73	-107.62
battlecrk.chin.spr	2001	Weir count	111	27	262	4.71	87	440	6.09	0.13	-13.99
battlecrk.chin.spr	2002	Weir count	222	27	411	5.4	137	484	6.18	0.21	-23.16
battlecrk.chin.spr	2003	Weir count	221	27	554	5.4	185	632	6.45	0.37	-44.95
battlecrk.chin.spr	2004	Weir count	90	27	533	4.5	178	644	6.47	0.38	-46.36
battlecrk.chin.spr	2005	Weir count	73	27	384	4.29	128	606	6.41	0.22	-25.21
battlecrk.chin.spr	2006	Weir count	221	27	384	5.4	128	605	6.41	-0.04	4.27
battlecrk.chin.spr	2007	Weir count	291	27	585	5.67	195	675	6.51	0.05	-4.81

PopID	Year	CountType	Total	TotalYears	N	logTotal	S	popseries	logPop	delta_logPop	decline
battlecrk.chin.spr	2008	Weir count	105	27	617	4.65	206	690	6.54	0.13	-13.86
battlecrk.chin.spr	2009	Weir count	194	27	590	5.27	197	811	6.7	0.29	-34.05
battlecrk.chin.spr	2010	Weir count	172	27	471	5.15	157	762	6.64	0.12	-12.89
battlecrk.chin.spr	2011	Weir count	157	27	523	5.06	174	628	6.44	-0.09	8.99
battlecrk.chin.spr	2012	Weir count	799	27	1128	6.68	376	1322	7.19	0.49	-63.01
battlecrk.chin.spr	2013	Weir count	608	27	1564	6.41	521	1736	7.46	0.82	-127.82
battlecrk.chin.spr	2014	Weir count	429	27	1836	6.06	612	1993	7.6	1.15	-217.36
battlecrk.chin.spr	2015	Weir count	181	27	1218	5.2	406	2017	7.61	0.42	-52.57
battlecrk.chin.spr	2016	Weir count	180	27	790	5.19	263	1398	7.24	-0.22	19.47
battlecrk.chin.spr	2017	Weir count	30	27	391	3.4	130	820	6.71	-0.89	58.86
battlecrk.chin.spr	2018	Weir count	82	27	292	4.41	97	473	6.16	-1.45	76.55
battlecrk.chin.spr	2019	Weir count	45	27	157	3.81	52	337	5.82	-1.42	75.89
bigchicocrk.chin.spr	1971	Carcass survey	0	33	NA	-11.51	NA	NA	NA	NA	NA
bigchicocrk.chin.spr	1973	Carcass survey	50	33	NA	3.91	NA	NA	NA	NA	NA
bigchicocrk.chin.spr	1974	Carcass survey	100	33	150	4.61	50	NA	NA	NA	NA
bigchicocrk.chin.spr	1977	Carcass survey	100	33	250	4.61	83	250	5.52	NA	NA
bigchicocrk.chin.spr	1984	Carcass survey	0	33	200	-11.51	67	250	5.52	NA	NA
bigchicocrk.chin.spr	1985	Carcass survey	0	33	100	-11.51	33	200	5.3	NA	NA
bigchicocrk.chin.spr	1993	Snorkel survey	38	33	38	3.64	13	138	4.93	-0.59	44.8
bigchicocrk.chin.spr	1994	Snorkel survey	2	33	40	0.69	13	40	3.69	-1.83	84
bigchicocrk.chin.spr	1995	Snorkel survey	200	33	240	5.3	80	240	5.48	0.18	-20
bigchicocrk.chin.spr	1996	Snorkel survey	2	33	204	0.69	68	242	5.49	0.56	-75.36
bigchicocrk.chin.spr	1997	Snorkel survey	2	33	204	0.69	68	206	5.33	1.64	-415
bigchicocrk.chin.spr	1998	Snorkel survey	369	33	373	5.91	124	573	6.35	0.87	-138.75
bigchicocrk.chin.spr	1999	Snorkel survey	27	33	398	3.3	133	400	5.99	0.5	-65.29
bigchicocrk.chin.spr	2000	Snorkel survey	27	33	423	3.3	141	425	6.05	0.72	-106.31
bigchicocrk.chin.spr	2001	Snorkel survey	39	33	93	3.66	31	462	6.14	-0.22	19.37
bigchicocrk.chin.spr	2002	Snorkel survey	0	33	66	-11.51	22	93	4.53	-1.46	76.75
bigchicocrk.chin.spr	2003	Snorkel survey	81	33	120	4.39	40	147	4.99	-1.06	65.41
bigchicocrk.chin.spr	2004	Snorkel survey	0	33	81	-11.51	27	120	4.79	-1.35	74.03

PopID	Year	CountType	Total	TotalYears	N	logTotal	S	popseries	logPop	delta_logPop	decline
bigchicocrk.chin.spr	2005	Snorkel survey	37	33	118	3.61	39	118	4.77	0.24	-26.88
bigchicocrk.chin.spr	2006	Snorkel survey	299	33	336	5.7	112	417	6.03	1.04	-183.67
bigchicocrk.chin.spr	2007	Snorkel survey	0	33	336	-11.51	112	336	5.82	1.03	-180
bigchicocrk.chin.spr	2008	Snorkel survey	0	33	299	-11.51	100	336	5.82	1.05	-184.75
bigchicocrk.chin.spr	2009	Snorkel survey	6	33	6	1.79	2	305	5.72	-0.31	26.86
bigchicocrk.chin.spr	2010	Snorkel survey	2	33	8	0.69	3	8	2.08	-3.74	97.62
bigchicocrk.chin.spr	2011	Snorkel survey	124	33	132	4.82	44	132	4.88	-0.93	60.71
bigchicocrk.chin.spr	2012	Snorkel survey	0	33	126	-11.51	42	132	4.88	-0.84	56.72
bigchicocrk.chin.spr	2013	Snorkel survey	0	33	124	-11.51	41	126	4.84	2.76	-1475
bigchicocrk.chin.spr	2014	Snorkel survey	0	33	0	-11.51	0	124	4.82	-0.06	6.06
bigchicocrk.chin.spr	2015	Snorkel survey	0	33	0	-11.51	0	0	-10.13	-15.01	100
bigchicocrk.chin.spr	2016	Snorkel survey	0	33	0	-11.51	0	0	-10.13	-14.96	100
bigchicocrk.chin.spr	2017	Snorkel survey	0	33	0	-11.51	0	0	-10.13	-14.95	100
bigchicocrk.chin.spr	2018	Snorkel survey	0	33	0	-11.51	0	0	-10.13	0	0
bigchicocrk.chin.spr	2019	Snorkel survey	350	33	350	5.86	117	350	5.86	15.98	-874999975
buttecrk.chin.spr	1970	Snorkel survey	285	49	NA	5.65	NA	NA	NA	NA	NA
buttecrk.chin.spr	1971	Snorkel survey	470	49	NA	6.15	NA	NA	NA	NA	NA
buttecrk.chin.spr	1972	Snorkel survey	150	49	905	5.01	302	NA	NA	NA	NA
buttecrk.chin.spr	1973	Snorkel survey	300	49	920	5.7	307	1205	7.09	NA	NA
buttecrk.chin.spr	1974	Snorkel survey	150	49	600	5.01	200	1070	6.98	NA	NA
buttecrk.chin.spr	1975	Snorkel survey	650	49	1100	6.48	367	1250	7.13	NA	NA
buttecrk.chin.spr	1976	Snorkel survey	46	49	846	3.83	282	1146	7.04	-0.05	4.9
buttecrk.chin.spr	1977	Snorkel survey	100	49	796	4.61	265	946	6.85	-0.12	11.59
buttecrk.chin.spr	1978	Snorkel survey	128	49	274	4.85	91	924	6.83	-0.3	26.08
buttecrk.chin.spr	1979	Snorkel survey	10	49	238	2.3	79	284	5.65	-1.4	75.22
buttecrk.chin.spr	1980	Snorkel survey	226	49	364	5.42	121	464	6.14	-0.71	50.95
buttecrk.chin.spr	1981	Snorkel survey	250	49	486	5.52	162	614	6.42	-0.41	33.55
buttecrk.chin.spr	1982	Snorkel survey	534	49	1010	6.28	337	1020	6.93	1.28	-259.15
buttecrk.chin.spr	1983	Snorkel survey	50	49	834	3.91	278	1060	6.97	0.83	-128.45
buttecrk.chin.spr	1984	Snorkel survey	23	49	607	3.14	202	857	6.75	0.33	-39.58

PopID	Year	CountType	Total	TotalYears	N	logTotal	S	popseries	logPop	delta_logPop	decline
buttecrk.chin.spr	1985	Snorkel survey	254	49	327	5.54	109	861	6.76	-0.17	15.59
buttecrk.chin.spr	1986	Snorkel survey	1371	49	1648	7.22	549	1698	7.44	0.47	-60.19
buttecrk.chin.spr	1987	Snorkel survey	14	49	1639	2.64	546	1662	7.42	0.66	-93.93
buttecrk.chin.spr	1988	Snorkel survey	1290	49	2675	7.16	892	2929	7.98	1.22	-240.19
buttecrk.chin.spr	1989	Snorkel survey	1300	49	2604	7.17	868	3975	8.29	0.85	-134.1
buttecrk.chin.spr	1990	Snorkel survey	250	49	2840	5.52	947	2854	7.96	0.54	-71.72
buttecrk.chin.spr	1992	Snorkel survey	730	49	2280	6.59	760	3570	8.18	0.2	-21.88
buttecrk.chin.spr	1993	Snorkel survey	650	49	1630	6.48	543	2930	7.98	-0.31	26.29
buttecrk.chin.spr	1994	Snorkel survey	474	49	1854	6.16	618	2104	7.65	-0.3	26.28
buttecrk.chin.spr	1995	Snorkel survey	7500	49	8624	8.92	2875	9354	9.14	0.96	-162.02
buttecrk.chin.spr	1996	Snorkel survey	1413	49	9387	7.25	3129	10037	9.21	1.23	-242.56
buttecrk.chin.spr	1997	Snorkel survey	635	49	9548	6.45	3183	10022	9.21	1.56	-376.33
buttecrk.chin.spr	1998	Snorkel survey	20259	49	22307	9.92	7436	29807	10.3	1.16	-218.66
buttecrk.chin.spr	1999	Snorkel survey	3679	49	24573	8.21	8191	25986	10.17	0.95	-158.9
buttecrk.chin.spr	2000	Snorkel survey	4118	49	28056	8.32	9352	28691	10.26	1.05	-186.28
buttecrk.chin.spr	2001	Carcass survey	18670	49	26467	9.83	8822	46726	10.75	0.45	-56.76
buttecrk.chin.spr	2002	Carcass survey	16409	49	39197	9.71	13066	42876	10.67	0.5	-65
buttecrk.chin.spr	2003	Carcass survey	17404	49	52483	9.76	17494	56601	10.94	0.68	-97.28
buttecrk.chin.spr	2004	Carcass survey	10558	49	44371	9.26	14790	63041	11.05	0.3	-34.92
buttecrk.chin.spr	2005	Carcass survey	17592	49	45554	9.78	15185	61963	11.03	0.37	-44.52
buttecrk.chin.spr	2006	Carcass survey	6537	49	34687	8.79	11562	52091	10.86	-0.08	7.97
buttecrk.chin.spr	2007	Carcass survey	6871	49	31000	8.84	10333	41558	10.63	-0.42	34.08
buttecrk.chin.spr	2008	Carcass survey	11046	49	24454	9.31	8151	42046	10.65	-0.39	32.14
buttecrk.chin.spr	2009	Carcass survey	2687	49	20604	7.9	6868	27141	10.21	-0.65	47.9
buttecrk.chin.spr	2010	Carcass survey	1991	49	15724	7.6	5241	22595	10.03	-0.61	45.63
buttecrk.chin.spr	2011	Carcass survey	4871	49	9549	8.49	3183	20595	9.93	-0.71	51.02
buttecrk.chin.spr	2012	Carcass survey	16317	49	23179	9.7	7726	25866	10.16	-0.05	4.7
buttecrk.chin.spr	2013	Carcass survey	16782	49	37970	9.73	12657	39961	10.6	0.57	-76.86
buttecrk.chin.spr	2014	Carcass survey	5083	49	38182	8.53	12727	43053	10.67	0.74	-109.05
buttecrk.chin.spr	2015	Carcass survey	569	49	22434	6.34	7478	38751	10.56	0.4	-49.81

PopID	Year	CountType	Total	TotalYears	N	logTotal	S	popseries	logPop	delta_logPop	decline
buttecrk.chin.spr	2016	Carcass survey	5731	49	11383	8.65	3794	28165	10.25	-0.35	29.52
buttecrk.chin.spr	2017	Carcass survey	515	49	6815	6.24	2272	11898	9.38	-1.29	72.36
buttecrk.chin.spr	2018	Carcass survey	2362	49	8608	7.77	2869	9177	9.12	-1.44	76.32
buttecrk.chin.spr	2019	Carcass survey	14863	49	17740	9.61	5913	23471	10.06	-0.18	16.67
clearcrk.chin.spr	1993	Unknown	1	25	NA	0	NA	NA	NA	NA	NA
clearcrk.chin.spr	1994	Unknown	0	25	NA	-11.51	NA	NA	NA	NA	NA
clearcrk.chin.spr	1995	Unknown	2	25	3	0.69	1	NA	NA	NA	NA
clearcrk.chin.spr	1998	Snorkel	47	25	49	3.85	16	50	3.91	NA	NA
clearcrk.chin.spr	1999	Snorkel	35	25	84	3.56	28	84	4.43	NA	NA
clearcrk.chin.spr	2000	Snorkel	9	25	91	2.2	30	93	4.53	NA	NA
clearcrk.chin.spr	2001	Snorkel	0	25	44	-11.51	15	91	4.51	0.6	-82
clearcrk.chin.spr	2002	Snorkel	66	25	75	4.19	25	110	4.7	0.27	-30.95
clearcrk.chin.spr	2003	Snorkel	25	25	91	3.22	30	100	4.61	0.07	-7.53
clearcrk.chin.spr	2004	Snorkel	98	25	189	4.58	63	189	5.24	0.73	-107.69
clearcrk.chin.spr	2005	Snorkel	69	25	192	4.23	64	258	5.55	0.85	-134.55
clearcrk.chin.spr	2006	Snorkel	77	25	244	4.34	81	269	5.59	0.99	-169
clearcrk.chin.spr	2007	Snorkel	194	25	340	5.27	113	438	6.08	0.84	-131.75
clearcrk.chin.spr	2008	Snorkel	200	25	471	5.3	157	540	6.29	0.74	-109.3
clearcrk.chin.spr	2009	Snorkel	120	25	514	4.79	171	591	6.38	0.79	-119.7
clearcrk.chin.spr	2010	Snorkel	21	25	341	3.04	114	535	6.28	0.2	-22.15
clearcrk.chin.spr	2011	Snorkel	8	25	149	2.08	50	349	5.86	-0.44	35.37
clearcrk.chin.spr	2012	Snorkel	68	25	97	4.22	32	217	5.38	-1	63.28
clearcrk.chin.spr	2013	Snorkel	659	25	735	6.49	245	756	6.63	0.35	-41.31
clearcrk.chin.spr	2014	Snorkel	95	25	822	4.55	274	830	6.72	0.87	-137.82
clearcrk.chin.spr	2015	Snorkel	45	25	799	3.81	266	867	6.77	1.39	-299.54
clearcrk.chin.spr	2016	Snorkel	29	25	169	3.37	56	828	6.72	0.09	-9.52
clearcrk.chin.spr	2017	Snorkel	25	25	99	3.22	33	194	5.27	-1.45	76.63
clearcrk.chin.spr	2018	Snorkel	49	25	103	3.89	34	148	5	-1.77	82.93
clearcrk.chin.spr	2019	Snorkel	62	25	136	4.13	45	165	5.11	-1.61	80.07
cottonwoodcrk.chin.spr	1973	Unknown	0	31	NA	-11.51	NA	NA	NA	NA	NA

PopID	Year	CountType	Total	TotalYears	N	logTotal	S	popseries	logPop	delta_logPop	decline
cottonwoodcrk.chin.spr	1974	Unknown	3	31	NA	1.1	NA	NA	NA	NA	NA
cottonwoodcrk.chin.spr	1975	Unknown	3	31	6	1.1	2	NA	NA	NA	NA
cottonwoodcrk.chin.spr	1982	Unknown	0	31	6	-11.51	2	6	1.79	NA	NA
cottonwoodcrk.chin.spr	1989	Snorkel survey	0	31	3	-11.51	1	6	1.79	NA	NA
cottonwoodcrk.chin.spr	1993	Snorkel survey	1	31	1	0	0	4	1.39	NA	NA
cottonwoodcrk.chin.spr	1995	Snorkel survey	8	31	9	2.08	3	9	2.2	0.41	-50
cottonwoodcrk.chin.spr	1996	Snorkel survey	6	31	15	1.79	5	15	2.71	0.92	-150
cottonwoodcrk.chin.spr	1997	Snorkel survey	0	31	14	-11.51	5	15	2.71	1.32	-275
cottonwoodcrk.chin.spr	1998	Snorkel survey	477	31	483	6.17	161	491	6.2	4	-5355.54
cottonwoodcrk.chin.spr	1999	Snorkel survey	102	31	579	4.62	193	585	6.37	3.66	-3800
cottonwoodcrk.chin.spr	2000	Snorkel survey	122	31	701	4.8	234	701	6.55	3.84	-4573.33
cottonwoodcrk.chin.spr	2001	Snorkel survey	245	31	469	5.5	156	946	6.85	0.66	-92.67
cottonwoodcrk.chin.spr	2002	Snorkel survey	125	31	492	4.83	164	594	6.39	0.02	-1.54
cottonwoodcrk.chin.spr	2003	Snorkel survey	73	31	443	4.29	148	565	6.34	-0.22	19.4
cottonwoodcrk.chin.spr	2004	Snorkel survey	17	31	215	2.83	72	460	6.13	-0.72	51.37
cottonwoodcrk.chin.spr	2005	Snorkel survey	47	31	137	3.85	46	262	5.57	-0.82	55.89
cottonwoodcrk.chin.spr	2006	Snorkel survey	55	31	119	4.01	40	192	5.26	-1.08	66.02
cottonwoodcrk.chin.spr	2007	Snorkel survey	34	31	136	3.53	45	153	5.03	-1.1	66.74
cottonwoodcrk.chin.spr	2008	Snorkel survey	0	31	89	-11.51	30	136	4.91	-0.66	48.09
cottonwoodcrk.chin.spr	2009	Snorkel survey	0	31	34	-11.51	11	89	4.49	-0.77	53.65
cottonwoodcrk.chin.spr	2010	Snorkel survey	15	31	15	2.71	5	49	3.89	-1.14	67.97
cottonwoodcrk.chin.spr	2011	Snorkel survey	2	31	17	0.69	6	17	2.83	-2.08	87.5
cottonwoodcrk.chin.spr	2012	Snorkel survey	1	31	18	0	6	18	2.89	-1.6	79.78
cottonwoodcrk.chin.spr	2013	Snorkel survey	1	31	4	0	1	19	2.94	-0.95	61.22
cottonwoodcrk.chin.spr	2014	Snorkel survey	2	31	4	0.69	1	6	1.79	-1.04	64.71
cottonwoodcrk.chin.spr	2015	Snorkel survey	0	31	3	-11.51	1	4	1.39	-1.5	77.78
cottonwoodcrk.chin.spr	2016	Snorkel survey	0	31	2	-11.51	1	3	1.1	-1.85	84.21
cottonwoodcrk.chin.spr	2017	Snorkel survey	0	31	0	-11.51	0	2	0.69	-1.1	66.67
cottonwoodcrk.chin.spr	2018	Snorkel survey	0	31	0	-11.51	0	0	-10.13	-11.51	100
cottonwoodcrk.chin.spr	2019	Snorkel survey	62	31	62	4.13	21	62	4.13	3.03	-1966.65

PopID	Year	CountType	Total	TotalYears	N	logTotal	S	popseries	logPop	delta_logPop	decline
deercrk.chin.spr	1970	Carcass survey	2000	46	NA	7.6	NA	NA	NA	NA	NA
deercrk.chin.spr	1971	Carcass survey	1500	46	NA	7.31	NA	NA	NA	NA	NA
deercrk.chin.spr	1972	Carcass survey	400	46	3900	5.99	1300	NA	NA	NA	NA
deercrk.chin.spr	1973	Carcass survey	2000	46	3900	7.6	1300	5900	8.68	NA	NA
deercrk.chin.spr	1974	Carcass survey	3500	46	5900	8.16	1967	7400	8.91	NA	NA
deercrk.chin.spr	1975	Carcass survey	8500	46	14000	9.05	4667	14400	9.57	NA	NA
deercrk.chin.spr	1977	Carcass survey	340	46	12340	5.83	4113	14340	9.57	0.89	-143.05
deercrk.chin.spr	1978	Carcass survey	1200	46	10040	7.09	3347	13540	9.51	0.6	-82.97
deercrk.chin.spr	1980	Carcass survey	1500	46	3040	7.31	1013	11540	9.35	-0.22	19.86
deercrk.chin.spr	1982	Carcass survey	1500	46	4200	7.31	1400	4540	8.42	-1.15	68.34
deercrk.chin.spr	1983	Carcass survey	500	46	3500	6.21	1167	4700	8.46	-1.06	65.29
deercrk.chin.spr	1985	Snorkel survey	301	46	2301	5.71	767	3801	8.24	-1.11	67.06
deercrk.chin.spr	1986	Dam	543	46	1344	6.3	448	2844	7.95	-0.47	37.36
deercrk.chin.spr	1987	Snorkel survey/Dam counts	200	46	1044	5.3	348	1544	7.34	-1.11	67.15
deercrk.chin.spr	1988	Snorkel survey/Dam counts	371	46	1114	5.92	371	1415	7.25	-0.99	62.77
deercrk.chin.spr	1989	Snorkel survey/Dam counts	84	46	655	4.43	218	1198	7.09	-0.86	57.88
deercrk.chin.spr	1990	Snorkel survey/Dam counts	496	46	951	6.21	317	1151	7.05	-0.29	25.45
deercrk.chin.spr	1991	Snorkel survey/Dam counts	479	46	1059	6.17	353	1430	7.27	0.01	-1.06
deercrk.chin.spr	1992	Snorkel survey	209	46	1184	5.34	395	1268	7.15	0.06	-5.84
deercrk.chin.spr	1993	Snorkel survey	259	46	947	5.56	316	1443	7.27	0.23	-25.37
deercrk.chin.spr	1994	Snorkel survey	485	46	953	6.18	318	1432	7.27	0	-0.14
deercrk.chin.spr	1995	Snorkel survey	1295	46	2039	7.17	680	2248	7.72	0.57	-77.29
deercrk.chin.spr	1996	Snorkel survey	614	46	2394	6.42	798	2653	7.88	0.61	-83.85
deercrk.chin.spr	1997	Snorkel survey	466	46	2375	6.14	792	2860	7.96	0.69	-99.72
deercrk.chin.spr	1998	Snorkel survey	1879	46	2959	7.54	986	4254	8.36	0.64	-89.23
deercrk.chin.spr	1999	Snorkel survey	1591	46	3936	7.37	1312	4550	8.42	0.54	-71.5
deercrk.chin.spr	2000	Snorkel survey	637	46	4107	6.46	1369	4573	8.43	0.47	-59.9
deercrk.chin.spr	2001	Snorkel survey	1622	46	3850	7.39	1283	5729	8.65	0.3	-34.67
deercrk.chin.spr	2002	Snorkel survey	2195	46	4454	7.69	1485	6045	8.71	0.28	-32.86
deercrk.chin.spr	2003	Snorkel survey	2759	46	6576	7.92	2192	7213	8.88	0.46	-57.73

PopID	Year	CountType	Total	TotalYears	N	logTotal	S	popseries	logPop	delta_logPop	decline
deercrk.chin.spr	2004	Snorkel survey	804	46	5758	6.69	1919	7380	8.91	0.25	-28.82
deercrk.chin.spr	2005	Snorkel survey	2239	46	5802	7.71	1934	7997	8.99	0.28	-32.29
deercrk.chin.spr	2006	Snorkel survey	2432	46	5475	7.8	1825	8234	9.02	0.13	-14.15
deercrk.chin.spr	2007	Snorkel survey	644	46	5315	6.47	1772	6119	8.72	-0.19	17.09
deercrk.chin.spr	2008	Snorkel survey	140	46	3216	4.94	1072	5455	8.6	-0.38	31.79
deercrk.chin.spr	2009	Snorkel survey	213	46	997	5.36	332	3429	8.14	-0.88	58.36
deercrk.chin.spr	2010	Snorkel survey	262	46	615	5.57	205	1259	7.14	-1.58	79.42
deercrk.chin.spr	2011	Snorkel survey	271	46	746	5.6	249	886	6.79	-1.82	83.76
deercrk.chin.spr	2012	Snorkel survey	734	46	1267	6.6	422	1480	7.3	-0.84	56.84
deercrk.chin.spr	2013	Snorkel survey	708	46	1713	6.56	571	1975	7.59	0.45	-56.87
deercrk.chin.spr	2014	Snorkel survey	830	46	2272	6.72	757	2543	7.84	1.05	-187.02
deercrk.chin.spr	2015	Snorkel survey	268	46	1806	5.59	602	2540	7.84	0.54	-71.62
deercrk.chin.spr	2016	Snorkel survey	331	46	1429	5.8	476	2137	7.67	0.08	-8.2
deercrk.chin.spr	2017	Snorkel survey	219	46	818	5.39	273	1648	7.41	-0.43	35.19
deercrk.chin.spr	2018	Snorkel survey	159	46	709	5.07	236	977	6.88	-0.96	61.54
deercrk.chin.spr	2019	Snorkel survey	578	46	956	6.36	319	1287	7.16	-0.51	39.78
featherrivhatch.chin.spr	1970	Hatchery return	235	50	NA	5.46	NA	NA	NA	NA	NA
featherrivhatch.chin.spr	1971	Hatchery return	481	50	NA	6.18	NA	NA	NA	NA	NA
featherrivhatch.chin.spr	1972	Hatchery return	256	50	972	5.55	324	NA	NA	NA	NA
featherrivhatch.chin.spr	1973	Hatchery return	205	50	942	5.32	314	1177	7.07	NA	NA
featherrivhatch.chin.spr	1974	Hatchery return	198	50	659	5.29	220	1140	7.04	NA	NA
featherrivhatch.chin.spr	1975	Hatchery return	691	50	1094	6.54	365	1350	7.21	NA	NA
featherrivhatch.chin.spr	1976	Hatchery return	699	50	1588	6.55	529	1793	7.49	0.42	-52.34
featherrivhatch.chin.spr	1977	Hatchery return	185	50	1575	5.22	525	1773	7.48	0.44	-55.53
featherrivhatch.chin.spr	1978	Hatchery return	202	50	1086	5.31	362	1777	7.48	0.27	-31.63
featherrivhatch.chin.spr	1979	Hatchery return	250	50	637	5.52	212	1336	7.2	-0.29	25.49
featherrivhatch.chin.spr	1980	Hatchery return	269	50	721	5.59	240	906	6.81	-0.67	48.9
featherrivhatch.chin.spr	1981	Hatchery return	469	50	988	6.15	329	1190	7.08	-0.4	33.03
featherrivhatch.chin.spr	1982	Hatchery return	1910	50	2648	7.55	883	2898	7.97	0.77	-116.92
featherrivhatch.chin.spr	1983	Hatchery return	1702	50	4081	7.44	1360	4350	8.38	1.57	-380.13

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featherrivhatch.chin.spr	1984	Hatchery return	1562	50	5174	7.35	1725	5643	8.64	1.56	-374.2
featherrivhatch.chin.spr	1985	Hatchery return	1632	50	4896	7.4	1632	6806	8.83	0.85	-134.85
featherrivhatch.chin.spr	1986	Hatchery return	1433	50	4627	7.27	1542	6329	8.75	0.37	-45.49
featherrivhatch.chin.spr	1987	Hatchery return	1213	50	4278	7.1	1426	5840	8.67	0.03	-3.49
featherrivhatch.chin.spr	1988	Hatchery return	6833	50	9479	8.83	3160	11111	9.32	0.49	-63.25
featherrivhatch.chin.spr	1989	Hatchery return	5078	50	13124	8.53	4375	14557	9.59	0.83	-130
featherrivhatch.chin.spr	1990	Hatchery return	1893	50	13804	7.55	4601	15017	9.62	0.94	-157.14
featherrivhatch.chin.spr	1991	Hatchery return	4303	50	11274	8.37	3758	18107	9.8	0.49	-62.96
featherrivhatch.chin.spr	1992	Hatchery return	1497	50	7693	7.31	2564	12771	9.45	-0.13	12.27
featherrivhatch.chin.spr	1993	Hatchery return	4672	50	10472	8.45	3491	12365	9.42	-0.19	17.66
featherrivhatch.chin.spr	1994	Hatchery return	3641	50	9810	8.2	3270	14113	9.55	-0.25	22.06
featherrivhatch.chin.spr	1995	Hatchery return	5414	50	13727	8.6	4576	15224	9.63	0.18	-19.21
featherrivhatch.chin.spr	1996	Hatchery return	6381	50	15436	8.76	5145	20108	9.91	0.49	-62.62
featherrivhatch.chin.spr	1997	Hatchery return	3653	50	15448	8.2	5149	19089	9.86	0.3	-35.26
featherrivhatch.chin.spr	1998	Hatchery return	6746	50	16780	8.82	5593	22194	10.01	0.38	-45.78
featherrivhatch.chin.spr	1999	Hatchery return	3731	50	14130	8.22	4710	20511	9.93	0.02	-2
featherrivhatch.chin.spr	2000	Hatchery return	3657	50	14134	8.2	4711	17787	9.79	-0.07	6.82
featherrivhatch.chin.spr	2001	Hatchery return	4135	50	11523	8.33	3841	18269	9.81	-0.19	17.68
featherrivhatch.chin.spr	2002	Hatchery return	4189	50	11981	8.34	3994	15712	9.66	-0.27	23.4
featherrivhatch.chin.spr	2003	Hatchery return	8662	50	16986	9.07	5662	20643	9.94	0.15	-16.06
featherrivhatch.chin.spr	2004	Hatchery return	4212	50	17063	8.35	5688	21198	9.96	0.15	-16.03
featherrivhatch.chin.spr	2005	Hatchery return	1774	50	14648	7.48	4883	18837	9.84	0.18	-19.89
featherrivhatch.chin.spr	2006	Hatchery return	2181	50	8167	7.69	2722	16829	9.73	-0.2	18.48
featherrivhatch.chin.spr	2007	Hatchery return	1916	50	5871	7.56	1957	10083	9.22	-0.74	52.43
featherrivhatch.chin.spr	2008	Hatchery return	1460	50	5557	7.29	1852	7331	8.9	-0.94	61.08
featherrivhatch.chin.spr	2009	Hatchery return	989	50	4365	6.9	1455	6546	8.79	-0.94	61.1
featherrivhatch.chin.spr	2010	Hatchery return	1661	50	4110	7.42	1370	6026	8.7	-0.51	40.24
featherrivhatch.chin.spr	2011	Hatchery return	1969	50	4619	7.59	1540	6079	8.71	-0.19	17.08
featherrivhatch.chin.spr	2012	Hatchery return	3738	50	7368	8.23	2456	8357	9.03	0.24	-27.67
featherrivhatch.chin.spr	2013	Hatchery return	4294	50	10001	8.36	3334	11662	9.36	0.66	-93.53

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featherrivhatch.chin.spr	2014	Hatchery return	2776	50	10808	7.93	3603	12777	9.46	0.74	-110.18
featherrivhatch.chin.spr	2015	Hatchery return	3386	50	10456	8.13	3485	14194	9.56	0.53	-69.85
featherrivhatch.chin.spr	2016	Hatchery return	1659	50	7821	7.41	2607	12115	9.4	0.04	-3.88
featherrivhatch.chin.spr	2017	Hatchery return	532	50	5577	6.28	1859	8353	9.03	-0.43	34.62
featherrivhatch.chin.spr	2018	Hatchery return	2110	50	4301	7.65	1434	7687	8.95	-0.61	45.84
featherrivhatch.chin.spr	2019	Hatchery return	3867	50	6509	8.26	2170	8168	9.01	-0.39	32.58
millcrk.chin.spr	1970	Carcass/snorkel survey	1500	46	NA	7.31	NA	NA	NA	NA	NA
millcrk.chin.spr	1971	Carcass/snorkel survey	1000	46	NA	6.91	NA	NA	NA	NA	NA
millcrk.chin.spr	1972	Carcass/snorkel survey	500	46	3000	6.21	1000	NA	NA	NA	NA
millcrk.chin.spr	1973	Carcass/snorkel survey	1700	46	3200	7.44	1067	4700	8.46	NA	NA
millcrk.chin.spr	1974	Carcass/snorkel survey	1500	46	3700	7.31	1233	4700	8.46	NA	NA
millcrk.chin.spr	1975	Carcass/snorkel survey	3500	46	6700	8.16	2233	7200	8.88	NA	NA
millcrk.chin.spr	1977	Carcass/snorkel survey	460	46	5460	6.13	1820	7160	8.88	0.42	-52.34
millcrk.chin.spr	1978	Carcass/snorkel survey	925	46	4885	6.83	1628	6385	8.76	0.31	-35.85
millcrk.chin.spr	1980	Carcass/snorkel survey	500	46	1885	6.21	628	5385	8.59	-0.29	25.21
millcrk.chin.spr	1982	Carcass/snorkel survey	700	46	2125	6.55	708	2585	7.86	-1.02	63.9
millcrk.chin.spr	1984	Carcass/snorkel survey	191	46	1391	5.25	464	2316	7.75	-1.01	63.73
millcrk.chin.spr	1985	Carcass/snorkel survey	121	46	1012	4.8	337	1512	7.32	-1.27	71.92
millcrk.chin.spr	1986	Dam	291	46	603	5.67	201	1303	7.17	-0.69	49.59
millcrk.chin.spr	1987	Dam	90	46	502	4.5	167	693	6.54	-1.21	70.08
millcrk.chin.spr	1988	Dam	572	46	953	6.35	318	1074	6.98	-0.34	28.97
millcrk.chin.spr	1989	Dam	563	46	1225	6.33	408	1516	7.32	0.15	-16.35
millcrk.chin.spr	1990	Dam	844	46	1979	6.74	660	2069	7.63	1.09	-198.56
millcrk.chin.spr	1991	Dam	319	46	1726	5.77	575	2298	7.74	0.76	-113.97
millcrk.chin.spr	1992	Dam	237	46	1400	5.47	467	1963	7.58	0.26	-29.49
millcrk.chin.spr	1993	Dam	61	46	617	4.11	206	1461	7.29	-0.35	29.39
millcrk.chin.spr	1994	Dam	723	46	1021	6.58	340	1340	7.2	-0.54	41.69
millcrk.chin.spr	1995	Dam	320	46	1104	5.77	368	1341	7.2	-0.38	31.69
millcrk.chin.spr	1996	Dam	253	46	1296	5.53	432	1357	7.21	-0.07	7.12
millcrk.chin.spr	1997	Redd surveys	202	46	775	5.31	258	1498	7.31	0.11	-11.79

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millcrk.chin.spr	1998	Redd surveys	424	46	879	6.05	293	1199	7.09	-0.11	10.59
millcrk.chin.spr	1999	Redd surveys	560	46	1186	6.33	395	1439	7.27	0.06	-6.04
millcrk.chin.spr	2000	Redd surveys	544	46	1528	6.3	509	1730	7.46	0.14	-15.49
millcrk.chin.spr	2001	Redd surveys	1104	46	2208	7.01	736	2632	7.88	0.79	-119.52
millcrk.chin.spr	2002	Redd surveys	1594	46	3242	7.37	1081	3802	8.24	0.97	-164.21
millcrk.chin.spr	2003	Redd surveys	1426	46	4124	7.26	1375	4668	8.45	0.99	-169.83
millcrk.chin.spr	2004	Redd surveys	998	46	4018	6.91	1339	5122	8.54	0.67	-94.6
millcrk.chin.spr	2005	Redd surveys	1150	46	3574	7.05	1191	5168	8.55	0.31	-35.93
millcrk.chin.spr	2006	Redd surveys	1002	46	3150	6.91	1050	4576	8.43	-0.02	1.97
millcrk.chin.spr	2007	Redd surveys	920	46	3072	6.82	1024	4070	8.31	-0.23	20.54
millcrk.chin.spr	2008	Redd surveys	381	46	2303	5.94	768	3453	8.15	-0.4	33.18
millcrk.chin.spr	2009	Redd surveys	237	46	1538	5.47	513	2540	7.84	-0.59	44.49
millcrk.chin.spr	2010	Redd surveys	482	46	1100	6.18	367	2020	7.61	-0.7	50.37
millcrk.chin.spr	2011	Redd surveys	366	46	1085	5.9	362	1466	7.29	-0.86	57.54
millcrk.chin.spr	2012	Redd surveys	768	46	1616	6.64	539	1853	7.52	-0.32	27.05
millcrk.chin.spr	2013	Redd surveys	644	46	1778	6.47	593	2260	7.72	0.11	-11.88
millcrk.chin.spr	2014	Redd surveys	679	46	2091	6.52	697	2457	7.81	0.52	-67.6
millcrk.chin.spr	2015	Redd surveys	127	46	1450	4.84	483	2218	7.7	0.18	-19.7
millcrk.chin.spr	2016	Redd surveys	175	46	981	5.16	327	1625	7.39	-0.33	28.1
millcrk.chin.spr	2017	Redd surveys	258	46	560	5.55	187	1239	7.12	-0.68	49.57
millcrk.chin.spr	2018	Redd surveys	152	46	585	5.02	195	712	6.57	-1.14	67.9
millcrk.chin.spr	2019	Redd surveys	180	46	590	5.19	197	765	6.64	-0.75	52.92
sacramentoriv.chin.spr	1970	Dam	3652	50	NA	8.2	NA	NA	NA	NA	NA
sacramentoriv.chin.spr	1971	Dam	5830	50	NA	8.67	NA	NA	NA	NA	NA
sacramentoriv.chin.spr	1972	Dam	7038	50	16520	8.86	5507	NA	NA	NA	NA
sacramentoriv.chin.spr	1973	Dam	7175	50	20043	8.88	6681	23695	10.07	NA	NA
sacramentoriv.chin.spr	1974	Dam	3800	50	18013	8.24	6004	23843	10.08	NA	NA
sacramentoriv.chin.spr	1975	Dam	10234	50	21209	9.23	7070	28247	10.25	NA	NA
sacramentoriv.chin.spr	1976	Dam	25095	50	39129	10.13	13043	46304	10.74	0.67	-95.42
sacramentoriv.chin.spr	1977	Dam	11545	50	46874	9.35	15625	50674	10.83	0.75	-112.53

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sacramentoriv.chin.spr	1978	Dam	5669	50	42309	8.64	14103	52543	10.87	0.62	-86.01
sacramentoriv.chin.spr	1979	Dam	2856	50	20070	7.96	6690	45165	10.72	-0.02	2.46
sacramentoriv.chin.spr	1980	Dam	9369	50	17894	9.15	5965	29439	10.29	-0.54	41.91
sacramentoriv.chin.spr	1981	Dam	20655	50	32880	9.94	10960	38549	10.56	-0.31	26.63
sacramentoriv.chin.spr	1982	Dam	23156	50	53180	10.05	17727	56036	10.93	0.22	-24.07
sacramentoriv.chin.spr	1983	Dam	5647	50	49458	8.64	16486	58827	10.98	0.69	-99.83
sacramentoriv.chin.spr	1984	Dam	7823	50	36626	8.96	12209	57281	10.96	0.4	-48.59
sacramentoriv.chin.spr	1985	Dam	12913	50	26383	9.47	8794	49539	10.81	-0.12	11.59
sacramentoriv.chin.spr	1986	Dam	22058	50	42794	10	14265	48441	10.79	-0.19	17.66
sacramentoriv.chin.spr	1987	Dam	12371	50	47342	9.42	15781	55165	10.92	-0.04	3.69
sacramentoriv.chin.spr	1988	Dam	9867	50	44296	9.2	14765	57209	10.95	0.14	-15.48
sacramentoriv.chin.spr	1989	Dam	5131	50	27369	8.54	9123	49427	10.81	0.02	-2.04
sacramentoriv.chin.spr	1990	Dam	4198	50	19196	8.34	6399	31567	10.36	-0.56	42.78
sacramentoriv.chin.spr	1991	Dam	825	50	10154	6.72	3385	20021	9.9	-1.05	65
sacramentoriv.chin.spr	1992	Dam	371	50	5394	5.92	1798	10525	9.26	-1.55	78.71
sacramentoriv.chin.spr	1993	Dam	391	50	1587	5.97	529	5785	8.66	-1.7	81.67
sacramentoriv.chin.spr	1994	Dam	862	50	1624	6.76	541	2449	7.8	-2.1	87.77
sacramentoriv.chin.spr	1995	Dam	426	50	1679	6.05	560	2050	7.63	-1.64	80.52
sacramentoriv.chin.spr	1996	Dam	378	50	1666	5.93	555	2057	7.63	-1.03	64.44
sacramentoriv.chin.spr	1997	Dam	128	50	932	4.85	311	1794	7.49	-0.31	26.75
sacramentoriv.chin.spr	1998	Dam	1115	50	1621	7.02	540	2047	7.62	0	0.15
sacramentoriv.chin.spr	1999	Dam	262	50	1505	5.57	502	1883	7.54	-0.09	8.46
sacramentoriv.chin.spr	2000	Dam	43	50	1420	3.76	473	1548	7.34	-0.15	13.71
sacramentoriv.chin.spr	2001	Dam	621	50	926	6.43	309	2041	7.62	0	0.29
sacramentoriv.chin.spr	2002	Dam	195	50	859	5.27	286	1121	7.02	-0.52	40.47
sacramentoriv.chin.spr	2003	Dam	0	50	816	-11.51	272	859	6.76	-0.59	44.51
sacramentoriv.chin.spr	2004	Dam	370	50	565	5.91	188	1186	7.08	-0.54	41.89
sacramentoriv.chin.spr	2005	Dam	30	50	400	3.4	133	595	6.39	-0.63	46.92
sacramentoriv.chin.spr	2006	Dam	0	50	400	-11.51	133	400	5.99	-0.76	53.43
sacramentoriv.chin.spr	2007	Dam	248	50	278	5.51	93	648	6.47	-0.6	45.36

PopID	Year	CountType	Total	TotalYears	N	logTotal	S	popseries	logPop	delta_logPop	decline
sacramentoriv.chin.spr	2008	Other	52	50	300	3.95	100	330	5.8	-0.59	44.54
sacramentoriv.chin.spr	2009	Dam	0	50	300	-11.51	100	300	5.7	-0.29	25
sacramentoriv.chin.spr	2010	Dam	0	50	52	-11.51	17	300	5.7	-0.77	53.7
sacramentoriv.chin.spr	2011	Dam	0	50	0	-11.51	0	52	3.95	-1.85	84.24
sacramentoriv.chin.spr	2012	Dam	0	50	0	-11.51	0	0	-10.13	-15.83	100
sacramentoriv.chin.spr	2013	Dam	114	50	114	4.74	38	114	4.74	-0.97	62
sacramentoriv.chin.spr	2014	Dam	0	50	114	-11.51	38	114	4.74	0.78	-119.23
sacramentoriv.chin.spr	2015	Dam	0	50	114	-11.51	38	114	4.74	14.86	-284999975
sacramentoriv.chin.spr	2016	Dam	0	50	0	-11.51	0	114	4.74	0	0
sacramentoriv.chin.spr	2017	Dam	4	50	4	1.39	1	4	1.39	-3.35	96.49
sacramentoriv.chin.spr	2018	Dam	0	50	4	-11.51	1	4	1.39	-3.35	96.49
sacramentoriv.chin.spr	2019	Dam	33	50	37	3.5	12	37	3.61	-1.13	67.54
yubariv.chin.spr	1980	Unknown	200	14	NA	5.3	NA	NA	NA	NA	NA
yubariv.chin.spr	1981	Unknown	200	14	NA	5.3	NA	NA	NA	NA	NA
yubariv.chin.spr	2001	Unknown	108	14	508	4.68	169	NA	NA	NA	NA
yubariv.chin.spr	2004	VAKI-dam count	738	14	1046	6.6	349	1246	7.13	NA	NA
yubariv.chin.spr	2005	VAKI-dam count	3592	14	4438	8.19	1479	4638	8.44	NA	NA
yubariv.chin.spr	2006	VAKI-dam count	1326	14	5656	7.19	1885	5764	8.66	NA	NA
yubariv.chin.spr	2007	VAKI-dam count	372	14	5290	5.92	1763	6028	8.7	1.58	-383.79
yubariv.chin.spr	2008	VAKI-dam count	521	14	2219	6.26	740	5811	8.67	0.23	-25.29
yubariv.chin.spr	2009	VAKI-dam count	723	14	1616	6.58	539	2942	7.99	-0.67	48.96
yubariv.chin.spr	2010	VAKI-dam count	2886	14	4130	7.97	1377	4502	8.41	-0.29	25.32
yubariv.chin.spr	2011	VAKI-dam count	1159	14	4768	7.06	1589	5289	8.57	-0.09	8.98
yubariv.chin.spr	2012	VAKI-dam count	1046	14	5091	6.95	1697	5814	8.67	0.68	-97.62
yubariv.chin.spr	2013	VAKI-dam count	3130	14	5335	8.05	1778	8221	9.01	0.6	-82.61
yubariv.chin.spr	2014	VAKI-dam count	2336	14	6512	7.76	2171	7671	8.95	0.37	-45.04
americanriv.sthd	1972	Tag/recovery from angler census	19583	18	NA	9.88	NA	NA	NA	NA	NA
americanriv.sthd	1974	Tag/recovery from angler census	12274	18	NA	9.42	NA	NA	NA	NA	NA
americanriv.sthd	2002	AUC	300	18	32157	5.7	10719	NA	NA	NA	NA
americanriv.sthd	2003	AUC	343	18	12917	5.84	4306	32500	10.39	NA	NA

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americanriv.sthd	2004	AUC	330	18	973	5.8	324	13247	9.49	NA	NA
americanriv.sthd	2005	AUC	266	18	939	5.58	313	1239	7.12	NA	NA
americanriv.sthd	2007	AUC	504	18	1100	6.22	367	1443	7.27	-3.11	95.56
americanriv.sthd	2009	Redd count	96	18	866	4.56	289	1196	7.09	-2.4	90.97
americanriv.sthd	2010	Redd count	79	18	679	4.37	226	945	6.85	-0.27	23.73
americanriv.sthd	2011	AUC	172	18	347	5.15	116	851	6.75	-0.53	41.03
americanriv.sthd	2012	AUC	389	18	640	5.96	213	736	6.6	-0.49	38.46
americanriv.sthd	2013	AUC	437	18	998	6.08	333	1077	6.98	0.13	-13.97
americanriv.sthd	2014	AUC	91	18	917	4.51	306	1089	6.99	0.25	-27.97
americanriv.sthd	2015	AUC	65	18	593	4.17	198	982	6.89	0.29	-33.42
americanriv.sthd	2016	AUC	96	18	252	4.56	84	689	6.54	-0.45	36.03
americanriv.sthd	2017	AUC	45	18	206	3.81	69	297	5.69	-1.3	72.73
americanriv.sthd	2018	AUC	141	18	282	4.95	94	347	5.85	-1.04	64.66
americanriv.sthd	2019	AUC	176	18	362	5.17	121	458	6.13	-0.41	33.53
antelopecrk.sthd	2013	Adjusted redd count	102	6	NA	4.62	NA	NA	NA	NA	NA
antelopecrk.sthd	2014	Video count	17	6	NA	2.83	NA	NA	NA	NA	NA
antelopecrk.sthd	2015	Video count	17	6	136	2.83	45	NA	NA	NA	NA
antelopecrk.sthd	2016	Video count	3	6	37	1.1	12	139	4.93	NA	NA
antelopecrk.sthd	2018	Adjusted redd count	34	6	54	3.53	18	71	4.26	NA	NA
antelopecrk.sthd	2019	Adjusted redd count	46	6	83	3.83	28	100	4.61	NA	NA
clearcrk.sthd	2003	Redd count (adjusted)	78	17	NA	4.36	NA	NA	NA	NA	NA
clearcrk.sthd	2004	Redd count (adjusted)	151	17	NA	5.02	NA	NA	NA	NA	NA
clearcrk.sthd	2005	Redd count (adjusted)	144	17	373	4.97	124	NA	NA	NA	NA
clearcrk.sthd	2006	Redd count (adjusted)	43	17	338	3.76	113	416	6.03	NA	NA
clearcrk.sthd	2007	Redd count (adjusted)	165	17	352	5.11	117	503	6.22	NA	NA
clearcrk.sthd	2008	Redd count (adjusted)	148	17	356	5	119	500	6.21	NA	NA
clearcrk.sthd	2009	Redd count (adjusted)	409	17	722	6.01	241	765	6.64	0.61	-83.89
clearcrk.sthd	2010	Redd count (adjusted)	233	17	790	5.45	263	955	6.86	0.64	-89.86
clearcrk.sthd	2011	Redd count (adjusted)	218	17	860	5.38	287	1008	6.92	0.7	-101.6
clearcrk.sthd	2012	Redd count	178	17	629	5.18	210	1038	6.95	0.31	-35.69

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clearcrk.sthd	2013	Redd count	300	17	696	5.7	232	929	6.83	-0.03	2.72
clearcrk.sthd	2014	Redd count	406	17	884	6.01	295	1102	7	0.09	-9.33
clearcrk.sthd	2015	Redd count	225	17	931	5.42	310	1109	7.01	0.07	-6.84
clearcrk.sthd	2016	Redd count	149	17	780	5	260	1080	6.98	0.15	-16.25
clearcrk.sthd	2017	Redd count	75	17	449	4.32	150	855	6.75	-0.25	22.41
clearcrk.sthd	2018	Redd count	369	17	593	5.91	198	818	6.71	-0.3	26.24
clearcrk.sthd	2019	Redd count	219	17	663	5.39	221	812	6.7	-0.29	24.81
colemanhatch.sthd	1970	Hatchery	3742	50	NA	8.23	NA	NA	NA	NA	NA
colemanhatch.sthd	1971	Hatchery	1486	50	NA	7.3	NA	NA	NA	NA	NA
colemanhatch.sthd	1972	Hatchery	2645	50	7873	7.88	2624	NA	NA	NA	NA
colemanhatch.sthd	1973	Hatchery	1834	50	5965	7.51	1988	9707	9.18	NA	NA
colemanhatch.sthd	1974	Hatchery	1099	50	5578	7	1859	7064	8.86	NA	NA
colemanhatch.sthd	1975	Hatchery	2162	50	5095	7.68	1698	7740	8.95	NA	NA
colemanhatch.sthd	1976	Hatchery	2069	50	5330	7.63	1777	7164	8.88	-0.3	26.2
colemanhatch.sthd	1977	Hatchery	697	50	4928	6.55	1643	6027	8.7	-0.16	14.68
colemanhatch.sthd	1978	Hatchery	865	50	3631	6.76	1210	5793	8.66	-0.29	25.16
colemanhatch.sthd	1979	Hatchery	4264	50	5826	8.36	1942	7895	8.97	0.1	-10.2
colemanhatch.sthd	1980	Hatchery	1118	50	6247	7.02	2082	6944	8.85	0.14	-15.21
colemanhatch.sthd	1981	Hatchery	1275	50	6657	7.15	2219	7522	8.93	0.26	-29.85
colemanhatch.sthd	1982	Hatchery	938	50	3331	6.84	1110	7595	8.94	-0.04	3.8
colemanhatch.sthd	1983	Hatchery	529	50	2742	6.27	914	3860	8.26	-0.59	44.41
colemanhatch.sthd	1984	Hatchery	2084	50	3551	7.64	1184	4826	8.48	-0.44	35.84
colemanhatch.sthd	1985	Hatchery	2229	50	4842	7.71	1614	5780	8.66	-0.27	23.9
colemanhatch.sthd	1986	Hatchery	1176	50	5489	7.07	1830	6018	8.7	0.44	-55.91
colemanhatch.sthd	1987	Hatchery	915	50	4320	6.82	1440	6404	8.76	0.28	-32.7
colemanhatch.sthd	1988	Hatchery	492	50	2583	6.2	861	4812	8.48	-0.18	16.75
colemanhatch.sthd	1989	Hatchery	1319	50	2726	7.18	909	3902	8.27	-0.43	35.16
colemanhatch.sthd	1990	Hatchery	991	50	2802	6.9	934	3717	8.22	-0.54	41.96
colemanhatch.sthd	1991	Hatchery	870	50	3180	6.77	1060	3672	8.21	-0.27	23.69
colemanhatch.sthd	1992	Hatchery	805	50	2666	6.69	889	3985	8.29	0.02	-2.13

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colemanhatch.sthd	1993	Hatchery	3387	50	5062	8.13	1687	6053	8.71	0.49	-62.85
colemanhatch.sthd	1994	Hatchery	2185	50	6377	7.69	2126	7247	8.89	0.68	-97.36
colemanhatch.sthd	1995	Hatchery	2024	50	7596	7.61	2532	8401	9.04	0.75	-110.82
colemanhatch.sthd	1996	Hatchery	2789	50	6998	7.93	2333	10385	9.25	0.54	-71.57
colemanhatch.sthd	1997	Hatchery	2185	50	6998	7.69	2333	9183	9.13	0.24	-26.71
colemanhatch.sthd	1998	Hatchery	940	50	5914	6.85	1971	7938	8.98	-0.06	5.51
colemanhatch.sthd	1999	Hatchery	492	50	3617	6.2	1206	6406	8.76	-0.48	38.31
colemanhatch.sthd	2000	Hatchery	476	50	1908	6.17	636	4093	8.32	-0.81	55.43
colemanhatch.sthd	2001	Hatchery	812	50	1780	6.7	593	2720	7.91	-1.07	65.73
colemanhatch.sthd	2002	Hatchery	3683	50	4971	8.21	1657	5463	8.61	-0.16	14.72
colemanhatch.sthd	2003	Hatchery	2436	50	6931	7.8	2310	7407	8.91	0.59	-80.97
colemanhatch.sthd	2004	Hatchery	1743	50	7862	7.46	2621	8674	9.07	1.16	-218.9
colemanhatch.sthd	2005	Hatchery	1729	50	5908	7.46	1969	9591	9.17	0.56	-75.56
colemanhatch.sthd	2006	Hatchery	1466	50	4938	7.29	1646	7374	8.91	0	0.45
colemanhatch.sthd	2007	Hatchery	1774	50	4969	7.48	1656	6712	8.81	-0.26	22.62
colemanhatch.sthd	2008	Hatchery	3273	50	6513	8.09	2171	8242	9.02	-0.15	14.07
colemanhatch.sthd	2009	Hatchery	2338	50	7385	7.76	2462	8851	9.09	0.18	-20.03
colemanhatch.sthd	2010	Hatchery	1035	50	6646	6.94	2215	8420	9.04	0.23	-25.45
colemanhatch.sthd	2011	Hatchery	1439	50	4812	7.27	1604	8085	9	-0.02	1.9
colemanhatch.sthd	2012	Hatchery	2372	50	4846	7.77	1615	7184	8.88	-0.21	18.83
colemanhatch.sthd	2013	Hatchery	2850	50	6661	7.96	2220	7696	8.95	-0.09	8.6
colemanhatch.sthd	2014	Hatchery	3262	50	8484	8.09	2828	9923	9.2	0.2	-22.73
colemanhatch.sthd	2015	Hatchery	4059	50	10171	8.31	3390	12543	9.44	0.56	-74.6
colemanhatch.sthd	2016	Hatchery	1321	50	8642	7.19	2881	11492	9.35	0.4	-49.32
colemanhatch.sthd	2017	Hatchery	1911	50	7291	7.56	2430	10553	9.26	0.06	-6.35
colemanhatch.sthd	2018	Hatchery	10993	50	14225	9.31	4742	18284	9.81	0.38	-45.77
colemanhatch.sthd	2019	Hatchery	6269	50	19173	8.74	6391	20494	9.93	0.58	-78.33
cottonwoodcrk.sthd	2013	Video weir	2	6	NA	0.69	NA	NA	NA	NA	NA
cottonwoodcrk.sthd	2014	Video weir	2	6	NA	0.69	NA	NA	NA	NA	NA
cottonwoodcrk.sthd	2015	Video weir	1	6	5	0	2	NA	NA	NA	NA

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cottonwoodcrk.sthd	2016	Video weir	0	6	3	-11.51	1	5	1.61	NA	NA
cottonwoodcrk.sthd	2017	Video weir	6	6	7	1.79	2	9	2.2	NA	NA
cottonwoodcrk.sthd	2018	Video weir	13	6	19	2.56	6	20	3	NA	NA
cowcrk.sthd	2012	Video count	0	8	NA	-11.51	NA	NA	NA	NA	NA
cowcrk.sthd	2013	Video count	8	8	NA	2.08	NA	NA	NA	NA	NA
cowcrk.sthd	2014	Video count	1	8	9	0	3	NA	NA	NA	NA
cowcrk.sthd	2015	Video count	5	8	14	1.61	5	14	2.64	NA	NA
cowcrk.sthd	2016	Video count	1	8	7	0	2	15	2.71	NA	NA
cowcrk.sthd	2017	Video count	0	8	6	-11.51	2	7	1.95	NA	NA
cowcrk.sthd	2018	Video count	2	8	3	0.69	1	8	2.08	-0.56	42.86
cowcrk.sthd	2019	Video count	25	8	27	3.22	9	28	3.33	0.62	-86.67
deercrk.sthd	1994	Dam count	0	6	NA	-11.51	NA	NA	NA	NA	NA
deercrk.sthd	2015	Video count	340	6	NA	5.83	NA	NA	NA	NA	NA
deercrk.sthd	2016	Video count	48	6	388	3.87	129	NA	NA	NA	NA
deercrk.sthd	2017	Video count	44	6	432	3.78	144	432	6.07	NA	NA
deercrk.sthd	2018	Video count	228	6	320	5.43	107	660	6.49	NA	NA
deercrk.sthd	2019	Video count	204	6	476	5.32	159	524	6.26	NA	NA
featherriv.sthd	1970	Fish trap	422	17	NA	6.05	NA	NA	NA	NA	NA
featherriv.sthd	1971	Fish trap	156	17	NA	5.05	NA	NA	NA	NA	NA
featherriv.sthd	1972	Fish trap	576	17	1154	6.36	385	NA	NA	NA	NA
featherriv.sthd	1973	Fish trap	2184	17	2916	7.69	972	3338	8.11	NA	NA
featherriv.sthd	1974	Fish trap	2186	17	4946	7.69	1649	5102	8.54	NA	NA
featherriv.sthd	1975	Fish trap	3605	17	7975	8.19	2658	8551	9.05	NA	NA
featherriv.sthd	2005	Redd count	38	17	5829	3.64	1943	8013	8.99	0.88	-140.05
featherriv.sthd	2009	Redd count	28	17	3671	3.33	1224	5857	8.68	0.14	-14.8
featherriv.sthd	2010	Redd count	17	17	83	2.83	28	3688	8.21	-0.84	56.87
featherriv.sthd	2011	Redd count	26	17	71	3.26	24	109	4.69	-4.3	98.64
featherriv.sthd	2012	Redd count	79	17	122	4.37	41	150	5.01	-3.66	97.44
featherriv.sthd	2013	Redd count	21	17	126	3.04	42	143	4.96	-3.25	96.12
featherriv.sthd	2014	Redd count	25	17	125	3.22	42	151	5.02	0.33	-38.53

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featherriv.sthd	2015	Redd count	56	17	102	4.03	34	181	5.2	0.19	-20.67
featherriv.sthd	2017	Redd count	8	17	89	2.08	30	110	4.7	-0.26	23.08
featherriv.sthd	2018	Redd count	162	17	226	5.09	75	251	5.53	0.51	-66.23
featherriv.sthd	2019	Redd count	71	17	241	4.26	80	297	5.69	0.5	-64.09
featherrivhatch.sthd	1970	Hatchery	224	50	NA	5.41	NA	NA	NA	NA	NA
featherrivhatch.sthd	1971	Hatchery	78	50	NA	4.36	NA	NA	NA	NA	NA
featherrivhatch.sthd	1972	Hatchery	288	50	590	5.66	197	NA	NA	NA	NA
featherrivhatch.sthd	1973	Hatchery	1000	50	1366	6.91	455	1590	7.37	NA	NA
featherrivhatch.sthd	1974	Hatchery	715	50	2003	6.57	668	2081	7.64	NA	NA
featherrivhatch.sthd	1975	Hatchery	758	50	2473	6.63	824	2761	7.92	NA	NA
featherrivhatch.sthd	1976	Hatchery	573	50	2046	6.35	682	3046	8.02	0.65	-91.57
featherrivhatch.sthd	1977	Hatchery	163	50	1494	5.09	498	2209	7.7	0.06	-6.15
featherrivhatch.sthd	1978	Hatchery	131	50	867	4.88	289	1625	7.39	-0.53	41.14
featherrivhatch.sthd	1979	Hatchery	189	50	483	5.24	161	1056	6.96	-1.06	65.33
featherrivhatch.sthd	1980	Hatchery	314	50	634	5.75	211	797	6.68	-1.02	63.92
featherrivhatch.sthd	1981	Hatchery	547	50	1050	6.3	350	1181	7.07	-0.32	27.32
featherrivhatch.sthd	1982	Hatchery	891	50	1752	6.79	584	1941	7.57	0.61	-83.81
featherrivhatch.sthd	1983	Hatchery	1238	50	2676	7.12	892	2990	8	1.32	-275.16
featherrivhatch.sthd	1984	Hatchery	783	50	2912	6.66	971	3459	8.15	1.07	-192.89
featherrivhatch.sthd	1985	Hatchery	1721	50	3742	7.45	1247	4633	8.44	0.87	-138.69
featherrivhatch.sthd	1986	Hatchery	1554	50	4058	7.35	1353	5296	8.57	0.57	-77.12
featherrivhatch.sthd	1987	Hatchery	1018	50	4293	6.93	1431	5076	8.53	0.38	-46.75
featherrivhatch.sthd	1988	Hatchery	2587	50	5159	7.86	1720	6880	8.84	0.4	-48.5
featherrivhatch.sthd	1989	Hatchery	1106	50	4711	7.01	1570	6265	8.74	0.17	-18.3
featherrivhatch.sthd	1990	Hatchery	1446	50	5139	7.28	1713	6157	8.73	0.19	-21.3
featherrivhatch.sthd	1991	Hatchery	1025	50	3577	6.93	1192	6164	8.73	-0.11	10.41
featherrivhatch.sthd	1992	Hatchery	1028	50	3499	6.94	1166	4605	8.43	-0.31	26.5
featherrivhatch.sthd	1993	Hatchery	297	50	2350	5.69	783	3796	8.24	-0.48	38.35
featherrivhatch.sthd	1994	Hatchery	1594	50	2919	7.37	973	3944	8.28	-0.45	36.02
featherrivhatch.sthd	1995	Hatchery	1058	50	2949	6.96	983	3977	8.29	-0.15	13.64

PopID	Year	CountType	Total	TotalYears	N	logTotal	S	popseries	logPop	delta_logPop	decline
featherrivhatch.sthd	1996	Hatchery	269	50	2921	5.59	974	3218	8.08	-0.17	15.23
featherrivhatch.sthd	1997	Hatchery	2113	50	3440	7.66	1147	5034	8.52	0.24	-27.64
featherrivhatch.sthd	1998	Hatchery	1023	50	3405	6.93	1135	4463	8.4	0.12	-12.22
featherrivhatch.sthd	1999	Hatchery	633	50	3769	6.45	1256	4038	8.3	0.23	-25.48
featherrivhatch.sthd	2000	Hatchery	1742	50	3398	7.46	1133	5511	8.61	0.09	-9.48
featherrivhatch.sthd	2001	Hatchery	2056	50	4431	7.63	1477	5454	8.6	0.2	-22.2
featherrivhatch.sthd	2002	Hatchery	1446	50	5244	7.28	1748	5877	8.68	0.38	-45.54
featherrivhatch.sthd	2003	Hatchery	2907	50	6409	7.97	2136	8151	9.01	0.39	-47.9
featherrivhatch.sthd	2004	Hatchery	1504	50	5857	7.32	1952	7913	8.98	0.37	-45.09
featherrivhatch.sthd	2005	Hatchery	2086	50	6497	7.64	2166	7943	8.98	0.3	-35.15
featherrivhatch.sthd	2006	Hatchery	958	50	4548	6.86	1516	7455	8.92	-0.09	8.54
featherrivhatch.sthd	2007	Hatchery	1212	50	4256	7.1	1419	5760	8.66	-0.32	27.21
featherrivhatch.sthd	2008	Hatchery	679	50	2849	6.52	950	4935	8.5	-0.48	37.87
featherrivhatch.sthd	2009	Hatchery	312	50	2203	5.74	734	3161	8.06	-0.86	57.6
featherrivhatch.sthd	2010	Hatchery	86	50	1077	4.45	359	2289	7.74	-0.92	60.26
featherrivhatch.sthd	2011	Hatchery	723	50	1121	6.58	374	1800	7.5	-1.01	63.53
featherrivhatch.sthd	2012	Hatchery	830	50	1639	6.72	546	1951	7.58	-0.48	38.28
featherrivhatch.sthd	2013	Hatchery	1797	50	3350	7.49	1117	3436	8.14	0.41	-50.11
featherrivhatch.sthd	2014	Hatchery	1505	50	4132	7.32	1377	4855	8.49	0.99	-169.72
featherrivhatch.sthd	2015	Hatchery	817	50	4119	6.71	1373	4949	8.51	0.93	-153.66
featherrivhatch.sthd	2016	Hatchery	148	50	2470	5	823	4267	8.36	0.22	-24.19
featherrivhatch.sthd	2017	Hatchery	1510	50	2475	7.32	825	3980	8.29	-0.2	18.02
featherrivhatch.sthd	2018	Hatchery	2132	50	3790	7.66	1263	4607	8.44	-0.07	6.91
featherrivhatch.sthd	2019	Hatchery	1978	50	5620	7.59	1873	5768	8.66	0.3	-35.18
millcrk.sthd	1979	Dam count	280	11	NA	5.63	NA	NA	NA	NA	NA
millcrk.sthd	1994	Dam count	34	11	NA	3.53	NA	NA	NA	NA	NA
millcrk.sthd	2009	Video weir (adult count)	225	11	539	5.42	180	NA	NA	NA	NA
millcrk.sthd	2010	Video weir (adult count)	66	11	325	4.19	108	605	6.41	NA	NA
millcrk.sthd	2011	Video weir (adult count)	166	11	457	5.11	152	491	6.2	NA	NA
millcrk.sthd	2012	Video weir (adult count)	197	11	429	5.28	143	654	6.48	NA	NA

PopID	Year	CountType	Total	TotalYears	N	logTotal	S	popseries	logPop	delta_logPop	decline
millcrk.sthd	2014	Video weir (adult count)	66	11	429	4.19	143	495	6.2	-0.2	18.18
millcrk.sthd	2015	Video weir (adult count)	246	11	509	5.51	170	675	6.51	0.32	-37.47
millcrk.sthd	2016	Video weir (adult count)	116	11	428	4.75	143	625	6.44	-0.05	4.43
millcrk.sthd	2017	Video weir (adult count)	59	11	421	4.08	140	487	6.19	-0.02	1.62
millcrk.sthd	2018	Video weir (adult count)	200	11	375	5.3	125	621	6.43	-0.08	8
mokelumneriv.sthd	1991	Video count	4	38	NA	1.39	NA	NA	NA	NA	NA
mokelumneriv.sthd	1993	Video count	7	38	NA	1.95	NA	NA	NA	NA	NA
mokelumneriv.sthd	1994	Video count	8	38	19	2.08	6	NA	NA	NA	NA
mokelumneriv.sthd	1995	Video count	19	38	34	2.94	11	38	3.64	NA	NA
mokelumneriv.sthd	1996	Video count	76	38	103	4.33	34	110	4.7	NA	NA
mokelumneriv.sthd	1997	Video count	12	38	107	2.48	36	115	4.74	NA	NA
mokelumneriv.sthd	1998	Video count	6	38	94	1.79	31	113	4.73	1.09	-197.37
mokelumneriv.sthd	1999	Video count	12	38	30	2.48	10	106	4.66	-0.04	3.64
mokelumneriv.sthd	2000	Video count	80	38	98	4.38	33	110	4.7	-0.04	4.35
mokelumneriv.sthd	2001	Redd survey	40	38	132	3.69	44	138	4.93	0.2	-22.12
mokelumneriv.sthd	2001	Video count	48	38	168	3.87	56	180	5.19	0.53	-69.81
mokelumneriv.sthd	2002	Redd survey	30	38	118	3.4	39	198	5.29	0.59	-80
mokelumneriv.sthd	2002	Video count	91	38	169	4.51	56	209	5.34	0.42	-51.45
mokelumneriv.sthd	2003	Redd survey	50	38	171	3.91	57	219	5.39	0.2	-21.67
mokelumneriv.sthd	2003	Video count	62	38	203	4.13	68	233	5.45	0.16	-17.68
mokelumneriv.sthd	2004	Redd survey	18	38	130	2.89	43	221	5.4	0.06	-5.74
mokelumneriv.sthd	2004	Video count	39	38	119	3.66	40	169	5.13	-0.26	22.83
mokelumneriv.sthd	2005	Redd survey	38	38	95	3.64	32	157	5.06	-0.39	32.62
mokelumneriv.sthd	2005	Video count	44	38	121	3.78	40	139	4.93	-0.46	37.1
mokelumneriv.sthd	2007	Redd survey	65	38	147	4.17	49	186	5.23	0.1	-10.06
mokelumneriv.sthd	2008	Redd survey	43	38	152	3.76	51	190	5.25	0.19	-21.02
mokelumneriv.sthd	2009	Redd survey	51	38	159	3.93	53	203	5.31	0.38	-46.04
mokelumneriv.sthd	2010	Redd survey	66	38	160	4.19	53	225	5.42	0.19	-20.97
mokelumneriv.sthd	2011	Redd survey	53	38	170	3.97	57	213	5.36	0.11	-12.11
mokelumneriv.sthd	2011	Video count	100	38	219	4.61	73	270	5.6	0.29	-33

PopID	Year	CountType	Total	TotalYears	N	logTotal	S	popseries	logPop	delta_logPop	decline
mokelumneriv.sthd	2012	Redd survey	68	38	221	4.22	74	287	5.66	0.24	-27.56
mokelumneriv.sthd	2012	Video count	257	38	425	5.55	142	478	6.17	0.81	-124.41
mokelumneriv.sthd	2013	Redd survey	63	38	388	4.14	129	488	6.19	0.59	-80.74
mokelumneriv.sthd	2013	Video count	74	38	394	4.3	131	462	6.14	0.48	-60.98
mokelumneriv.sthd	2014	Redd survey	68	38	205	4.22	68	462	6.14	-0.03	3.35
mokelumneriv.sthd	2014	Video count	124	38	266	4.82	89	329	5.8	-0.39	32.58
mokelumneriv.sthd	2015	Redd survey	63	38	255	4.14	85	329	5.8	-0.34	28.79
mokelumneriv.sthd	2015	Video survey	152	38	339	5.02	113	407	6.01	-0.13	11.9
mokelumneriv.sthd	2016	Redd survey	61	38	276	4.11	92	400	5.99	0.2	-21.58
mokelumneriv.sthd	2016	Video count	16	38	229	2.77	76	292	5.68	-0.12	11.25
mokelumneriv.sthd	2017	Redd survey	4	38	81	1.39	27	233	5.45	-0.56	42.75
mokelumneriv.sthd	2018	Redd survey	119	38	139	4.78	46	200	5.3	-0.69	50
mokelumneriv.sthd	2019	Redd survey	38	38	161	3.64	54	177	5.18	-0.5	39.38
mokelumnerivhatch.sthd	1970	Hatchery	134	50	NA	4.9	NA	NA	NA	NA	NA
mokelumnerivhatch.sthd	1971	Hatchery	215	50	NA	5.37	NA	NA	NA	NA	NA
mokelumnerivhatch.sthd	1972	Hatchery	4	50	353	1.39	118	NA	NA	NA	NA
mokelumnerivhatch.sthd	1973	Hatchery	11	50	230	2.4	77	364	5.9	NA	NA
mokelumnerivhatch.sthd	1974	Hatchery	18	50	33	2.89	11	248	5.51	NA	NA
mokelumnerivhatch.sthd	1975	Hatchery	2	50	31	0.69	10	35	3.56	NA	NA
mokelumnerivhatch.sthd	1976	Hatchery	0	50	20	-11.51	7	31	3.43	-2.46	91.48
mokelumnerivhatch.sthd	1977	Hatchery	0	50	2	-11.51	1	20	3	-2.52	91.94
mokelumnerivhatch.sthd	1978	Hatchery	0	50	0	-11.51	0	2	0.69	-2.86	94.29
mokelumnerivhatch.sthd	1979	Hatchery	0	50	0	-11.51	0	0	-10.13	-13.56	100
mokelumnerivhatch.sthd	1980	Hatchery	0	50	0	-11.51	0	0	-10.13	-13.12	100
mokelumnerivhatch.sthd	1981	Hatchery	0	50	0	-11.51	0	0	-10.13	-10.82	100
mokelumnerivhatch.sthd	1982	Hatchery	0	50	0	-11.51	0	0	-10.13	0	0
mokelumnerivhatch.sthd	1983	Hatchery	0	50	0	-11.51	0	0	-10.13	0	0
mokelumnerivhatch.sthd	1984	Hatchery	0	50	0	-11.51	0	0	-10.13	0	0
mokelumnerivhatch.sthd	1985	Hatchery	0	50	0	-11.51	0	0	-10.13	0	0
mokelumnerivhatch.sthd	1986	Hatchery	0	50	0	-11.51	0	0	-10.13	0	0

PopID	Year	CountType	Total	TotalYears	N	logTotal	S	popseries	logPop	delta_logPop	decline
mokelumnerivhatch.sthd	1987	Hatchery	48	50	48	3.87	16	48	3.87	14	-119999975
mokelumnerivhatch.sthd	1988	Hatchery	0	50	48	-11.51	16	48	3.87	14	-119999975
mokelumnerivhatch.sthd	1989	Hatchery	7	50	55	1.95	18	55	4.01	14.13	-137499950
mokelumnerivhatch.sthd	1990	Hatchery	11	50	18	2.4	6	66	4.19	0.32	-37.5
mokelumnerivhatch.sthd	1991	Hatchery	20	50	38	3	13	38	3.64	-0.23	20.83
mokelumnerivhatch.sthd	1992	Hatchery	29	50	60	3.37	20	67	4.2	0.2	-21.82
mokelumnerivhatch.sthd	1993	Hatchery	108	50	157	4.68	52	168	5.12	0.93	-154.55
mokelumnerivhatch.sthd	1994	Hatchery	83	50	220	4.42	73	240	5.48	1.84	-531.58
mokelumnerivhatch.sthd	1995	Hatchery	25	50	216	3.22	72	245	5.5	1.3	-265.67
mokelumnerivhatch.sthd	1996	Hatchery	39	50	147	3.66	49	255	5.54	0.42	-51.79
mokelumnerivhatch.sthd	1997	Hatchery	46	50	110	3.83	37	193	5.26	-0.22	19.58
mokelumnerivhatch.sthd	1998	Hatchery	5	50	90	1.61	30	115	4.74	-0.76	53.06
mokelumnerivhatch.sthd	1999	Hatchery	0	50	51	-11.51	17	90	4.5	-1.04	64.71
mokelumnerivhatch.sthd	2000	Hatchery	32	50	37	3.47	12	83	4.42	-0.84	56.99
mokelumnerivhatch.sthd	2001	Hatchery	32	50	64	3.47	21	69	4.23	-0.51	40
mokelumnerivhatch.sthd	2002	Hatchery	43	50	107	3.76	36	107	4.67	0.17	-18.89
mokelumnerivhatch.sthd	2003	Hatchery	44	50	119	3.78	40	151	5.02	0.6	-81.93
mokelumnerivhatch.sthd	2004	Hatchery	62	50	149	4.13	50	181	5.2	0.96	-162.32
mokelumnerivhatch.sthd	2005	Hatchery	60	50	166	4.09	55	209	5.34	0.67	-95.33
mokelumnerivhatch.sthd	2006	Hatchery	189	50	311	5.24	104	355	5.87	0.85	-135.1
mokelumnerivhatch.sthd	2007	Hatchery	412	50	661	6.02	220	723	6.58	1.38	-299.45
mokelumnerivhatch.sthd	2008	Hatchery	344	50	945	5.84	315	1005	6.91	1.57	-380.86
mokelumnerivhatch.sthd	2009	Hatchery	309	50	1065	5.73	355	1254	7.13	1.26	-253.24
mokelumnerivhatch.sthd	2010	Hatchery	64	50	717	4.16	239	1129	7.03	0.45	-56.15
mokelumnerivhatch.sthd	2011	Hatchery	335	50	708	5.81	236	1052	6.96	0.05	-4.68
mokelumnerivhatch.sthd	2012	Hatchery	306	50	705	5.72	235	1014	6.92	-0.21	19.14
mokelumnerivhatch.sthd	2013	Hatchery	215	50	856	5.37	285	920	6.82	-0.2	18.51
mokelumnerivhatch.sthd	2014	Hatchery	279	50	800	5.63	267	1135	7.03	0.08	-7.89
mokelumnerivhatch.sthd	2015	Hatchery	246	50	740	5.51	247	1046	6.95	0.03	-3.16
mokelumnerivhatch.sthd	2016	Hatchery	93	50	618	4.53	206	833	6.73	-0.1	9.46

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mokelumnerivhatch.sthd	2017	Hatchery	1121	50	1460	7.02	487	1739	7.46	0.43	-53.22
mokelumnerivhatch.sthd	2018	Hatchery	1275	50	2489	7.15	830	2735	7.91	0.96	-161.47
mokelumnerivhatch.sthd	2019	Hatchery	786	50	3182	6.67	1061	3275	8.09	1.37	-293.16
nimbushatch.sthd	1970	Hatchery	1734	50	NA	7.46	NA	NA	NA	NA	NA
nimbushatch.sthd	1971	Hatchery	3033	50	NA	8.02	NA	NA	NA	NA	NA
nimbushatch.sthd	1972	Hatchery	2861	50	7628	7.96	2543	NA	NA	NA	NA
nimbushatch.sthd	1973	Hatchery	2986	50	8880	8	2960	10614	9.27	NA	NA
nimbushatch.sthd	1974	Hatchery	3237	50	9084	8.08	3028	12117	9.4	NA	NA
nimbushatch.sthd	1975	Hatchery	2295	50	8518	7.74	2839	11379	9.34	NA	NA
nimbushatch.sthd	1976	Hatchery	3181	50	8713	8.06	2904	11699	9.37	0.1	-10.22
nimbushatch.sthd	1977	Hatchery	1307	50	6783	7.18	2261	10020	9.21	-0.19	17.31
nimbushatch.sthd	1978	Hatchery	619	50	5107	6.43	1702	7402	8.91	-0.43	34.95
nimbushatch.sthd	1979	Hatchery	680	50	2606	6.52	869	5787	8.66	-0.7	50.53
nimbushatch.sthd	1980	Hatchery	1310	50	2609	7.18	870	3916	8.27	-0.94	60.92
nimbushatch.sthd	1981	Hatchery	836	50	2826	6.73	942	3445	8.14	-0.76	53.46
nimbushatch.sthd	1982	Hatchery	3808	50	5954	8.24	1985	6634	8.8	0.14	-14.64
nimbushatch.sthd	1983	Hatchery	1003	50	5647	6.91	1882	6957	8.85	0.57	-77.66
nimbushatch.sthd	1984	Hatchery	5155	50	9966	8.55	3322	10802	9.29	1.14	-213.56
nimbushatch.sthd	1985	Hatchery	763	50	6921	6.64	2307	10729	9.28	0.48	-61.73
nimbushatch.sthd	1986	Hatchery	1193	50	7111	7.08	2370	8114	9	0.15	-16.63
nimbushatch.sthd	1987	Hatchery	1431	50	3387	7.27	1129	8542	9.05	-0.23	20.92
nimbushatch.sthd	1988	Hatchery	705	50	3329	6.56	1110	4092	8.32	-0.96	61.86
nimbushatch.sthd	1989	Hatchery	296	50	2432	5.69	811	3625	8.2	-0.81	55.32
nimbushatch.sthd	1990	Hatchery	594	50	1595	6.39	532	3026	8.01	-1.04	64.58
nimbushatch.sthd	1991	Hatchery	223	50	1113	5.41	371	1818	7.51	-0.81	55.57
nimbushatch.sthd	1992	Hatchery	1359	50	2176	7.21	725	2472	7.81	-0.38	31.81
nimbushatch.sthd	1993	Hatchery	241	50	1823	5.48	608	2417	7.79	-0.22	20.13
nimbushatch.sthd	1994	Hatchery	496	50	2096	6.21	699	2319	7.75	0.24	-27.56
nimbushatch.sthd	1995	Hatchery	3803	50	4540	8.24	1513	5899	8.68	0.87	-138.63
nimbushatch.sthd	1996	Hatchery	2360	50	6659	7.77	2220	6900	8.84	1.05	-185.48

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nimbushatch.sthd	1997	Hatchery	1371	50	7534	7.22	2511	8030	8.99	1.24	-246.27
nimbushatch.sthd	1998	Hatchery	680	50	4411	6.52	1470	8214	9.01	0.33	-39.24
nimbushatch.sthd	1999	Hatchery	1577	50	3628	7.36	1209	5988	8.7	-0.14	13.22
nimbushatch.sthd	2000	Hatchery	1658	50	3915	7.41	1305	5286	8.57	-0.42	34.17
nimbushatch.sthd	2001	Hatchery	2895	50	6130	7.97	2043	6810	8.83	-0.19	17.09
nimbushatch.sthd	2002	Hatchery	1556	50	6109	7.35	2036	7686	8.95	0.25	-28.36
nimbushatch.sthd	2003	Hatchery	873	50	5324	6.77	1775	6982	8.85	0.28	-32.08
nimbushatch.sthd	2004	Hatchery	1741	50	4170	7.46	1390	7065	8.86	0.04	-3.74
nimbushatch.sthd	2005	Hatchery	2272	50	4886	7.73	1629	6442	8.77	-0.18	16.19
nimbushatch.sthd	2006	Hatchery	2339	50	6352	7.76	2117	7225	8.89	0.03	-3.48
nimbushatch.sthd	2007	Hatchery	2673	50	7284	7.89	2428	9025	9.11	0.24	-27.74
nimbushatch.sthd	2008	Hatchery	758	50	5770	6.63	1923	8042	8.99	0.22	-24.84
nimbushatch.sthd	2009	Hatchery	1135	50	4566	7.03	1522	6905	8.84	-0.05	4.43
nimbushatch.sthd	2010	Hatchery	1015	50	2908	6.92	969	5581	8.63	-0.48	38.16
nimbushatch.sthd	2011	Hatchery	2046	50	4196	7.62	1399	4954	8.51	-0.48	38.4
nimbushatch.sthd	2012	Hatchery	2489	50	5550	7.82	1850	6685	8.81	-0.03	3.19
nimbushatch.sthd	2013	Hatchery	3371	50	7906	8.12	2635	8921	9.1	0.47	-59.85
nimbushatch.sthd	2014	Hatchery	527	50	6387	6.27	2129	8433	9.04	0.53	-70.23
nimbushatch.sthd	2015	Hatchery	154	50	4052	5.04	1351	6541	8.79	-0.02	2.15
nimbushatch.sthd	2016	Hatchery	756	50	1437	6.63	479	4808	8.48	-0.62	46.1
nimbushatch.sthd	2017	Hatchery	608	50	1518	6.41	506	2045	7.62	-1.42	75.75
nimbushatch.sthd	2018	Hatchery	513	50	1877	6.24	626	2031	7.62	-1.17	68.95
nimbushatch.sthd	2019	Hatchery	2659	50	3780	7.89	1260	4536	8.42	-0.06	5.66
sacramentoriv.sthd	1970	Natural run size above Red Bluff	8423	24	NA	9.04	NA	NA	NA	NA	NA
sacramentoriv.sthd	1971	Natural run size above Red Bluff	7432	24	NA	8.91	NA	NA	NA	NA	NA
sacramentoriv.sthd	1972	Natural run size above Red Bluff	4272	24	20127	8.36	6709	NA	NA	NA	NA
sacramentoriv.sthd	1973	Natural run size above Red Bluff	5772	24	17476	8.66	5825	25899	10.16	NA	NA
sacramentoriv.sthd	1974	Natural run size above Red Bluff	4967	24	15011	8.51	5004	22443	10.02	NA	NA
sacramentoriv.sthd	1975	Natural run size above Red Bluff	4271	24	15010	8.36	5003	19282	9.87	NA	NA

PopID	Year	CountType	Total	TotalYears	N	logTotal	S	popseries	logPop	delta_logPop	decline
sacramentoriv.sthd	1976	Natural run size above Red Bluff	6328	24	15566	8.75	5189	21338	9.97	-0.19	17.61
sacramentoriv.sthd	1977	Natural run size above Red Bluff	3636	24	14235	8.2	4745	19202	9.86	-0.16	14.44
sacramentoriv.sthd	1978	Natural run size above Red Bluff	1697	24	11661	7.44	3887	15932	9.68	-0.19	17.37
sacramentoriv.sthd	1979	Natural run size above Red Bluff	2469	24	7802	7.81	2601	14130	9.56	-0.41	33.78
sacramentoriv.sthd	1980	Natural run size above Red Bluff	6811	24	10977	8.83	3659	14613	9.59	-0.27	23.9
sacramentoriv.sthd	1981	Natural run size above Red Bluff	2032	24	11312	7.62	3771	13009	9.47	-0.2	18.35
sacramentoriv.sthd	1982	Natural run size above Red Bluff	1239	24	10082	7.12	3361	12551	9.44	-0.12	11.17
sacramentoriv.sthd	1983	Natural run size above Red Bluff	2369	24	5640	7.77	1880	12451	9.43	-0.16	14.8
sacramentoriv.sthd	1984	Natural run size above Red Bluff	1406	24	5014	7.25	1671	7046	8.86	-0.61	45.84
sacramentoriv.sthd	1985	Natural run size above Red Bluff	2008	24	5783	7.6	1928	7022	8.86	-0.58	44.05
sacramentoriv.sthd	1986	Natural run size above Red Bluff	1032	24	4446	6.94	1482	6815	8.83	-0.6	45.27
sacramentoriv.sthd	1987	Natural run size above Red Bluff	1563	24	4603	7.35	1534	6009	8.7	-0.16	14.72
sacramentoriv.sthd	1988	Natural run size above Red Bluff	783	24	3378	6.66	1126	5386	8.59	-0.27	23.3
sacramentoriv.sthd	1989	Natural run size above Red Bluff	0	24	2346	-11.51	782	3378	8.13	-0.7	50.43
sacramentoriv.sthd	1990	Natural run size above Red Bluff	702	24	1485	6.55	495	3048	8.02	-0.68	49.28
sacramentoriv.sthd	1991	Natural run size above Red Bluff	0	24	702	-11.51	234	1485	7.3	-1.29	72.43
sacramentoriv.sthd	1992	Natural run size above Red Bluff	2996	24	3698	8.01	1233	3698	8.22	0.09	-9.47
sacramentoriv.sthd	1993	Natural run size above Red Bluff	553	24	3549	6.32	1183	4251	8.35	0.33	-39.47
stanislausriv.sthd	2004	Weir count	0	16	NA	-11.51	NA	NA	NA	NA	NA
stanislausriv.sthd	2005	Weir count	1	16	NA	0	NA	NA	NA	NA	NA
stanislausriv.sthd	2006	Weir count	1	16	2	0	1	NA	NA	NA	NA
stanislausriv.sthd	2007	Weir count	9	16	11	2.2	4	11	2.4	NA	NA
stanislausriv.sthd	2008	Weir count	1	16	11	0	4	12	2.48	NA	NA
stanislausriv.sthd	2009	Weir count	9	16	19	2.2	6	20	3	NA	NA
stanislausriv.sthd	2010	Weir count	2	16	12	0.69	4	21	3.04	0.65	-90.91

PopID	Year	CountType	Total	TotalYears	N	logTotal	S	popseries	logPop	delta_logPop	decline
stanislausriv.sthd	2011	Weir count	2	16	13	0.69	4	14	2.64	0.15	-16.67
stanislausriv.sthd	2012	Weir count	32	16	36	3.47	12	45	3.81	0.81	-125
stanislausriv.sthd	2013	Weir count	9	16	43	2.2	14	45	3.81	0.76	-114.29
stanislausriv.sthd	2014	Weir count	28	16	69	3.33	23	71	4.26	1.62	-407.14
stanislausriv.sthd	2015	Weir count	8	16	45	2.08	15	77	4.34	0.54	-71.11
stanislausriv.sthd	2016	Weir count	2	16	38	0.69	13	47	3.85	0.04	-4.44
stanislausriv.sthd	2017	Weir count	18	16	28	2.89	9	56	4.03	-0.24	21.13
stanislausriv.sthd	2018	Weir count	5	16	25	1.61	8	33	3.5	-0.85	57.14
stanislausriv.sthd	2019	Weir count	16	16	39	2.77	13	41	3.71	-0.14	12.77
tuolumneriv.sthd	2010	Weir count	0	10	NA	-11.51	NA	NA	NA	NA	NA
tuolumneriv.sthd	2011	Weir count	0	10	NA	-11.51	NA	NA	NA	NA	NA
tuolumneriv.sthd	2012	Weir count	4	10	4	1.39	1	NA	NA	NA	NA
tuolumneriv.sthd	2013	Weir count	1	10	5	0	2	5	1.61	NA	NA
tuolumneriv.sthd	2014	Weir count	0	10	5	-11.51	2	5	1.61	NA	NA
tuolumneriv.sthd	2015	Weir count	0	10	1	-11.51	0	5	1.61	NA	NA
tuolumneriv.sthd	2016	Weir count	1	10	1	0	0	2	0.69	-0.92	60
tuolumneriv.sthd	2017	Weir count	1	10	2	0	1	2	0.69	-0.92	60
tuolumneriv.sthd	2018	Weir count	2	10	4	0.69	1	4	1.39	-0.22	20
tuolumneriv.sthd	2019	Weir count	2	10	5	0.69	2	6	1.79	1.1	-200
yubariv.sthd	2005	Weir count (VAKI)	73	13	NA	4.29	NA	NA	NA	NA	NA
yubariv.sthd	2006	Weir count (VAKI)	77	13	NA	4.34	NA	NA	NA	NA	NA
yubariv.sthd	2007	Weir count (VAKI)	14	13	164	2.64	55	NA	NA	NA	NA
yubariv.sthd	2008	Weir count (VAKI)	71	13	162	4.26	54	235	5.46	NA	NA
yubariv.sthd	2009	Weir count (VAKI)	184	13	269	5.21	90	346	5.85	NA	NA
yubariv.sthd	2010	Weir count (VAKI)	197	13	452	5.28	151	466	6.14	NA	NA
yubariv.sthd	2011	Weir count (VAKI)	514	13	895	6.24	298	966	6.87	1.41	-311.06
yubariv.sthd	2012	Weir count (VAKI)	820	13	1531	6.71	510	1715	7.45	1.6	-395.66
yubariv.sthd	2013	Weir count (VAKI)	212	13	1546	5.36	515	1743	7.46	1.32	-274.03
yubariv.sthd	2014	Weir count (VAKI)	231	13	1263	5.44	421	1777	7.48	0.61	-83.95
yubariv.sthd	2015	Weir count (VAKI)	155	13	598	5.04	199	1418	7.26	-0.19	17.32

PopID	Year	CountType	Total	TotalYears	N	logTotal	S	popseries	logPop	delta_logPop	decline
yubariv.sthd	2016	Weir count (VAKI)	83	13	469	4.42	156	681	6.52	-0.94	60.93
yubariv.sthd	2017	Weir count (VAKI)	36	13	274	3.58	91	505	6.22	-1.26	71.58