5 Central Valley Recovery Domain

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Several important planning efforts have been completed since the last viability assessment, including a Federal recovery plan for Sacramento River Winter-run Chinook salmon (SRWRC), Central Valley Spring-run Chinook salmon (CVSRC), and Central Valley Steelhead (NMFS 2014b). The recovery plan draws on the expertise of the Central Valley Technical Recovery Team (TRT) and is guided by the scientific framework and foundation provided by Lindley et al. (2006, 2007). The recovery plan along with the science and restoration actions identified in the biological opinion for the long-term operations of the Central Valley Project and State Water Project are key decision-making documents for improving and sustaining the health of California's salmon resources (NMFS 2009a, NMFS 2014b). The further development and implementation of life-cycle models for SRWRC and CVSRC will be seminal advancements in our understanding of how water project operations and restoration actions outlined in the recovery plans influence salmon population dynamics and long-term population viability (Hendrix et al. 2014).

5.1 Sacramento River Winter-run Chinook Salmon ESU

ESU Boundary Delineation

The Sacramento River Winter-run Chinook Salmon (SRWRC) ESU includes winter-run Chinook salmon spawning in the mainstem Sacramento River below Keswick Dam and Livingston Stone National Fish Hatchery (LSNFH). No new information suggests that the boundary 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 SRWRC ESU was endangered. The major concerns of the Biological Review Team (BRT) were that there is only one extant population, and it is outside of its historical spawning distribution in an artificially maintained habitat that is vulnerable to drought. In the most recent assessment, Williams et al. (2011) found that the viability of the ESU had changed little since the 2005 review and found that it did not appear that there was a change in extinction risk.

Brief Review of TRT Documents and Previous Findings

The TRT delineated four historical independent populations of SRWRC. The spawning areas of three of these historical populations are above the impassable Keswick and Shasta dams, while Battle Creek (location of the fourth population) is presently unsuitable for winter-run Chinook salmon due to high summer water temperatures. Lindley et al. (2007) developed viability criteria for Central Valley salmonids, summarized in Table 5.1. Using data through 2004, Lindley et al. (2007) found that the mainstem Sacramento River population was at low risk of extinction. The ESU as a whole, however, 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 (mean=8%; t=10 years). However, the duration and extent of this introgression was still consistent with a low extinction risk as of 2010.

New Data and Updated Analyses

Since the 2010 viability assessment, routine escapement data have continued to be collected allowing viability statistics to be updated (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 immigration period since 2001, but were since removed in 2012 to provide unimpaired salmon passage year-round which changed the ability to count SRWRC adults at the RBDD fish ladders (NMFS 2009a). Population estimates from 2001 to present are derived exclusively from mark-recapture estimates from carcass surveys (Figure 5.1).

Table 5.2 shows the viability metrics for SRWRC abundance and trends in the LSNFH and in the Sacramento River. Like many other populations of Chinook salmon in the Central Valley, SRWRC have declined in abundance since 2005 with recent decadal lows of 827 spawners in 2011 (Figure 5.1). Escapement in 2011 represents the lowest run size since the construction and operation of the LSNFH in 1997. Both the current total population size (N; LSNFH = 645; Sacramento River = 11,125) and mean population sizes ($\hat{S}$; LSNFH = 215; Sacramento River = 3,708) satisfy the low risk criterion ($N > 2500$).

However, the point estimate for the 10-year trend in run size is negative (-0.15), suggesting a 15% per year decline in the population (Table 5.2). The slope is marginally not different than ‘0’, yet it is clear that the population has been steadily declining rather than increasing over the past decade. The maximum year-to-year decline in population size has reached 67%, an increase from 38% in the previous 2010 viability assessment (Williams et al. 2011). However, 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).

These observed levels of hatchery influence exceed the low-extinction risk criteria met in the previous viability assessment and place the genetic integrity of the population at a moderate risk of extinction (Lindley et al. 2007). Since the beginning of hatchery production at LSNFH in 1997, the proportion of hatchery-origin SRWRC spawning in the river has increased (Figure 5.1). 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). However, the hatchery proportion has increased since 2005 and reached ~20% in 2005, 2014 and >30% in 2012. The average over the last 12 years

Table 5.1. Criteria for assessing the level of extinction risk 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[e]	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. 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 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 year-to-year decline in total population size (N) over the most recent 10 such ratios.

Population	N	$\hat{S}$	10-year trend (95% CI)	Recent Decline (%)
LSNFH winter-run Chinook	645	215.0	0.102 (-0.019, 0.222)	2.7
Sacramento River winter-run Chinook	11125	3708.3	-0.155 (-0.345, 0.034)	67.4

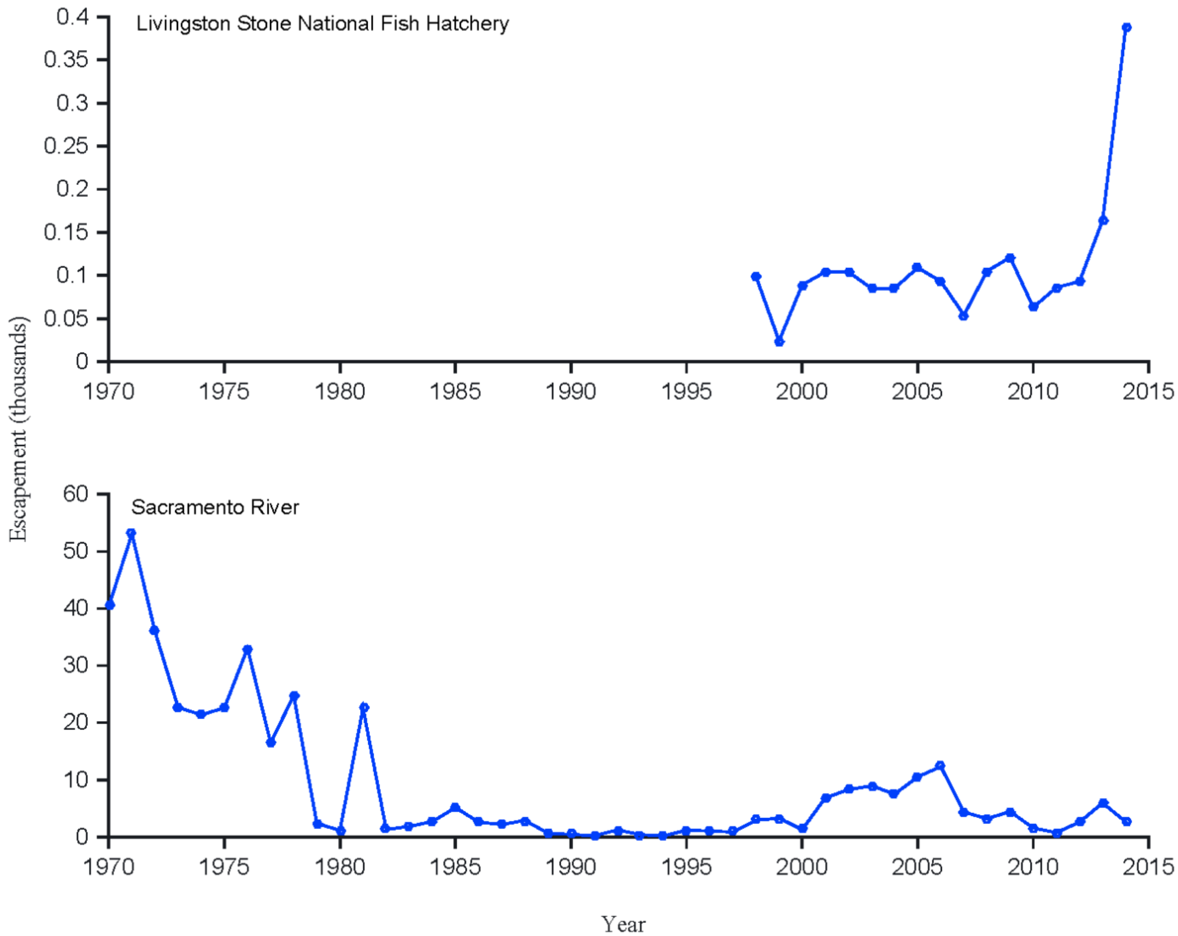

Figure 5.1. Time series of escapement for Sacramento River Winter-run Chinook Salmon used as broodstock at Livingston Stone National Fish Hatchery and Sacramento River mainstem spawners. Estimates for in-river spawners is the average number of adults counted at Red Bluff Diversion Dam and the carcass survey mark-recapture estimates (when available). Note: only mark-recapture estimates are used beginning in 2009; data from Azat (2014).

(approximately four generations) is 13% (SD= ±8%) with the most recent generation at 20% hatchery influence, placing the population at a moderate risk of extinction (Table 5.3; Figure 5.3).

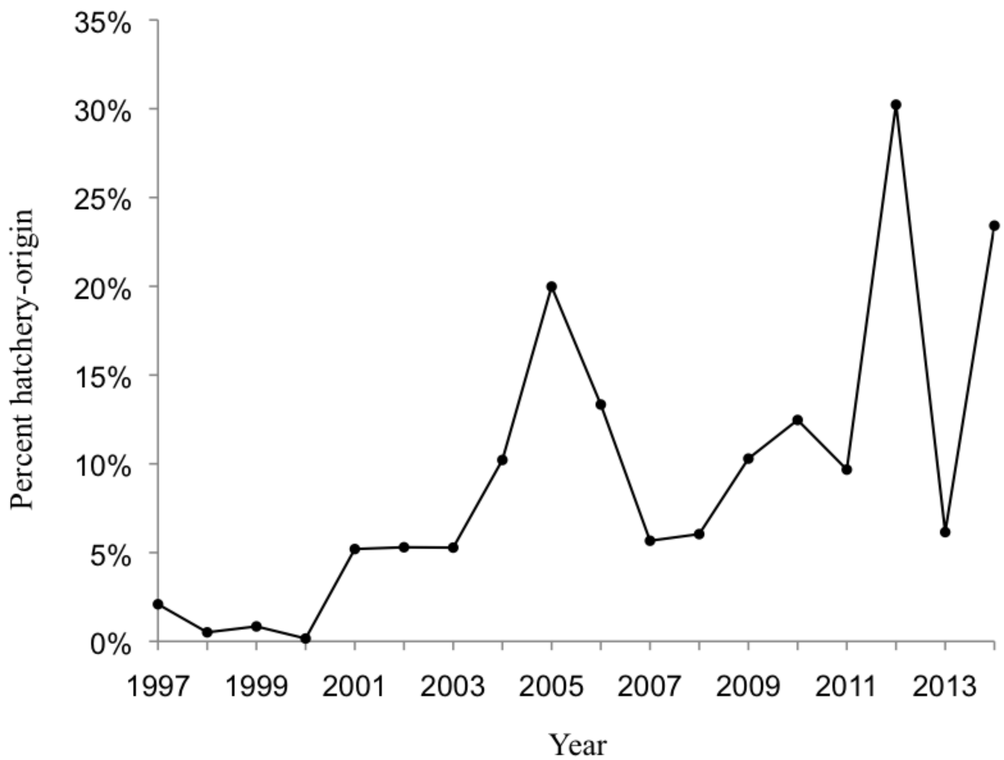

Figure 5.2. Percentage of in-river spawning Sacramento River Winter-run Chinook Salmon that are of hatchery-origin; Data source: Killam 2014.

Table 5.3. Average percentage of hatchery-origin Sacramento River Winter-run Chinook salmon spawners 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. Data source: Killam 2014).

	g1	g2	g3	g4
Average hatchery influence	20%	15%	13%	13%

Harvest Impacts[12]

Sacramento River Winter-run Chinook Salmon (SRWRC) have a more southerly ocean distribution relative to other California Chinook salmon stocks, and are primarily impacted by fisheries south of Point Arena, California. Sacramento River Winter-run Chinook Salmon age-3 ocean fishery impact rate estimates for the region south of Point Arena (an approximation of the exploitation rate) are currently available for 2000–2013, and have remained relatively stable over this time period, averaging 16% (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 stock and insufficient data exist for estimating a SRWRC impact rate in 2010. If years 2008–2010 are omitted, the average age-3 impact rate is 19% (PFMC 2015b).There have been several layers of ocean salmon fishery regulations implemented for SRWRC beginning in the early 1990s. For example, a substantial portion of the SRWRC ocean harvest impacts used to occur 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 (2000–2013). 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.

One component of the Reasonable and Prudent Alternative (RPA) from the 2010 Biological Opinion (NMFS 2010) specified that new fishery management objectives must be established. The implementation of the RPA resulted in the development of a harvest control rule which was first used for ocean fishery management in 2012. That harvest control rule specifies reductions in the age-3 ocean impact rate when the geometric mean number of spawners from the previous three years is reduced (Figure 5.5). The limits to the impact rate imposed by the harvest control rule is an additional control on ocean fisheries which still includes previously existing constraints on fishery opening and closing dates and minimum size limits south of Point Arena. Between 2012 and 2015, the SRWRC harvest control rule has specified maximum allowable forecast impact rates ranging from 12.9% to 19.0%.

What little SRWRC freshwater harvest that existed was essentially eliminated beginning in 2002 when Sacramento River Chinook salmon fishery season openings were adjusted to reduce the temporal overlap with the SRWRC spawning migration and spawning period.

In summary, the available information indicates that the level of SRWRC fishery impacts has not changed appreciably since the 2010 salmon and steelhead viability assessment (Williams et al. 2011), yet there have been additional ocean fishery regulations implemented with the purpose of reducing exploitation of SRWRC when average population size is reduced.

[12] Harvest impacts section prepared by Michael O'Farrell

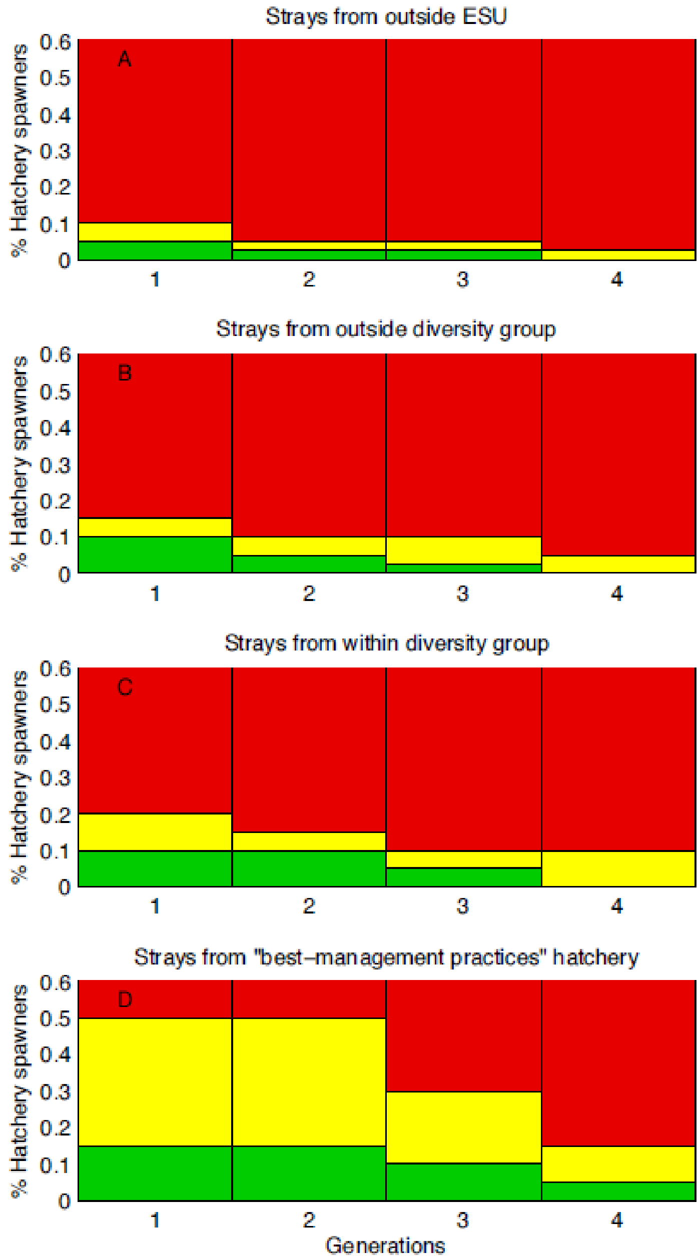

Figure 5.3. Percentage of hatchery-origin spawners and the resulting risk of extinction due to hatchery introgression from different sources of strays over multiple generations for Sacramento River Winter-run Chinook salmon. Low (green), moderate (yellow), and high (red). Model using "best-management 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 group/parameter "strays from outside of 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).

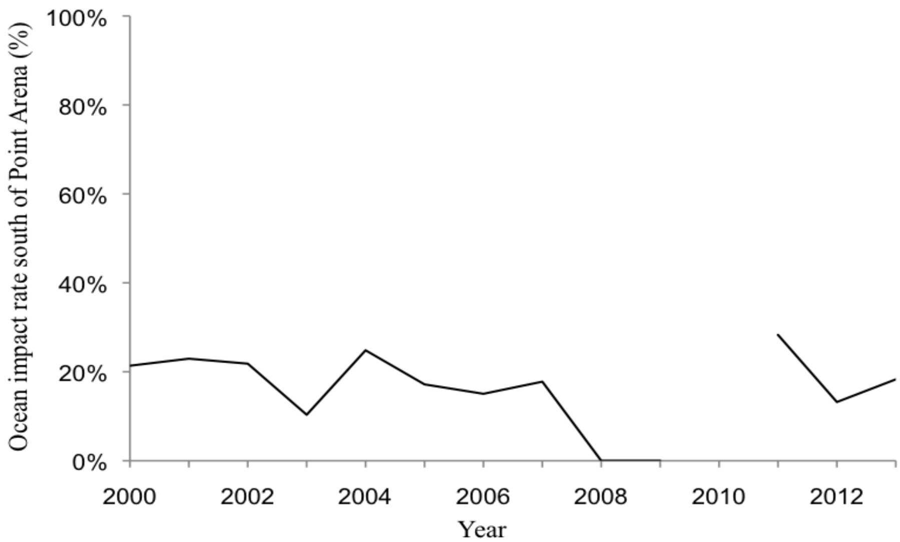

Figure 5.4. Sacramento River Winter-run Chinook Salmon age-3 ocean impact rate (percent) south of Point Arena, California for years 2000–2013. Estimates are sourced from PFMC (2015b). The impact rate could not be estimated in 2010 due to insufficient coded-wire tag recoveries.

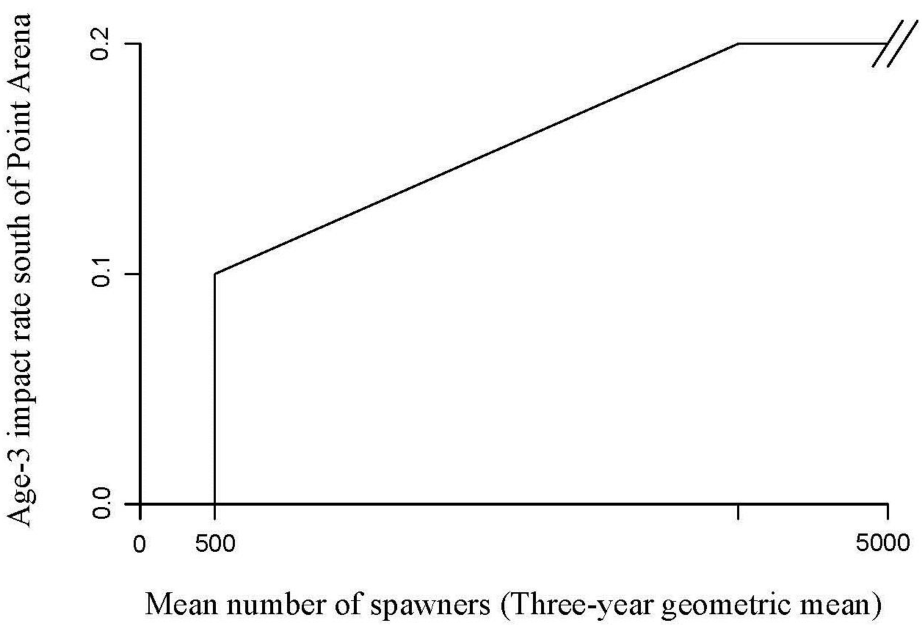

Figure 5.5. Current Sacramento River Winter-run Chinook Salmon harvest control rule. There is no explicit cap on the age-3 impact rate if the three-year geometric mean number of spawners exceeds 5000.

Summary and Conclusions

The overall viability of Sacramento River Winter-run Chinook Salmon has declined since the 2010 viability assessment, with the single spawning population on the mainstem Sacramento River. New information available since Williams et al. (2011) indicates an increased extinction risk to this ESU. The larger influence of the hatchery broodstock in addition to the rate of decline in abundance over the past decade has placed the population at an increased risk of extinction (Table 5.4).

The SRWRC population has declined during recent periods of unfavorable ocean conditions (2005–2006), and droughts (2007–2009) and are expected to continue to be low due to drought conditions in 2012–2015. The low adult returns in 2011 created a potential increase in vulnerability to a year class, yet the progeny from this cohort had relatively high survival resulting in a positive cohort replacement rate (3.5) from this numerically weak cohort (Azat 2014).

Poor early life stage survival during the most recent consecutive drought years of 2012–2015, coupled with poor ocean conditions and hatchery production practices (see Chapter 2) may further impact SRWRC survival-to-adulthood and risk of extinction. Temperature conditions during egg development and fry emergence were suboptimal over the duration of SRWRC rearing in 2014 and 2015 due to reduced cold water storage and subsequent release in/from Shasta Reservoir for this life stage. The egg-to-fry survival estimate for brood year 2014 is 5.9%, which is a significant departure from the average of 24.8% for brood years 1996–2014 measured at RBDD (Poytress et al. 2016). Potential impacts to these cohorts would be observed in viability criteria once adults return in 2015 and beyond.

Water operations can influence the routing of upper Sacramento River-origin water through agricultural fields and can create false attraction cues that cause SRWRC to deviate from the mainstem Sacramento River migration corridor and become stranded in agricultural fields behind flood bypass weirs. SRWRC have been observed to navigate up the Colusa Basin Drain for 40–70 miles before being blocked at weirs delaying and/or preventing successful migration (CALFED 2000, USFWS 2001, USBR and DWR 2012). In 2013, 600+ stranded adult SRWRC and CVSRC were observed, with a total of 312 adults relocated to the mainstem Sacramento River or the Livingston Stone National Fish Hatchery for use as broodstock (Killam et al. 2014). It is likely that survival for rescued adults that were stranded in the Colusa Basin Drain was low and that not all stranded adults were rescued. Thus, the loss of adults due to stranding prior to spawning can be demographically costly to the population.

The SRWRC ESU is likely at a lower extinction risk with a sustainable LSNFH population and naturally spawning population than it would be with just a single naturally spawning population, at least in the near-term. Yet, reliance on production from LSNFH and potential introgression between natural-origin SRWRC is increasing (Figure 5.2). In an attempt to prevent the loss of SRWRC cohorts during the 2013–2015 prolonged drought, a greater number of spawners were brought into the LSNFH as broodstock (Figure 5.1). The hatchery also produced and released three times as many juveniles. Thus, in years where mortality of natural-origin fish may be particularly high and LSNFH production is significantly increased, the contribution of LSNFH-origin fish to the

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 and 2015 review periods. Overall risk is determined by the highest risk score for any criterion.

	2010 Status Review	2015 Status Review
Population size	Low risk	Low risk
Population decline	Low risk	Moderate risk
Catastrophe, rate, and effect	Low risk	Low risk
Hatchery influence	Low risk	Moderate risk

returning adult spawners may elevate the overall risk of extinction of SRWRC due to genetic impacts from the hatchery. Potential impacts would manifest in viability criteria evaluations in escapement from the year 2016 and beyond, unless hatchery introgression is minimized through active adult management on the spawning grounds. The use of adult segregation weirs to manage gene flow between natural- and hatchery-origin fish in rivers is commonly conducted in Oregon and Washington to minimize impacts of hatchery fish on the genetic integrity of the overall population (HSRG 2014).

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 and upstream from Shasta Reservoir are in the planning phases, and if successful, would significantly benefit SRWRC. Genetic management plans will be critical for conserving the long-term genetic integrity of SRWRC, the success of the reintroduction efforts, and achieving a low-extinction risk to the portion of the population downstream Shasta Dam.

Lastly, the development and implementation of quantitative modeling tools that link water project operations, temperature management, and habitat restoration actions to SRWRC population dynamics will greatly improve our ability to make science-informed management decisions (Hendrix et al. 2014; Caldwell et al. 2015).

5.2 Central Valley Spring-run Chinook Salmon ESU

ESU Boundary 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 Delta and entire watershed is excluded as Critical Habitat and its populations considered extirpated (64 FR 50394; 70 FR 52488). Information on the presence of fish exhibiting spring-run behavior in San Joaquin River tributaries is provided and may represent passive recolonization of CVSRC into the San Joaquin River Basin. Thus, there is value in continuing to monitor these populations to evaluate the extent to which populations in the San Joaquin River tributaries warrant inclusion in the ESU boundary. No new information suggests that the boundary of this ESU should change or that its status as an ESU should change.

Summary of Previous Assessments

Williams et al. (2011) found that the viability of the ESU had probably deteriorated since the 2005 review (Good et al. 2005). Williams et al. (2011) reported improvements, evident in the viability of two populations, although these population level improvements were not enough to warrant a downgrading of the ESU extinction risk and there had been an overall increase in extinction risk to the ESU since the review by Good et al. (2005).

Brief Review of TRT Documents and Previous Findings

The TRT delineated 18 or 19 independent populations of CVSRC, along with a number of smaller dependent populations, and four diversity groups (Lindley et al. 2004). Of these 18 populations, only three are extant (Mill, Deer, and Butte creeks on the upper Sacramento River) and they represent only the Northern Sierra Nevada diversity group. All populations in the Basalt and Porous Lava group and the Southern Sierra Nevada group were extirpated, and only a few dependent populations persist in the Coast Range group. Using data through 2005 and the criteria in Table 5.1, Lindley et al. (2007) found that the Mill Creek, Deer Creek, and Butte Creek populations were at or near low risk of extinction. However in 2010, declines in abundance placed Mill and Deer Creek populations at a high risk of extinction due to their rates of decline, and in the case of Deer Creek, also the level of escapement. The ESU as a whole was not considered viable because there were no extant populations in the three other diversity groups. In addition, Mill, Deer and Butte creeks are close together geographically, decreasing the independence of their extinction risks due to catastrophic disturbance (Lindley et al. 2007).

New Data and Updated Analyses

Figure 5.6 shows the escapement of CVSRC to various Central Valley streams, and Table 5.5 shows abundance and trend statistics related to viability criteria. All independent populations (Battle, Deer, Mill, and Butte creek populations) show larger total population sizes (*N*) and mean escapement ($\hat{S}$) than the previous assessment in 2010. New data for the Yuba River suggests a low extinction risk based on population size. The Butte Creek population remains at a low risk, while Deer Creek and Mill Creek populations have shown improvements from a high risk to a moderate risk with population sizes approaching low risk abundance thresholds. In particular, Butte Creek spring-run Chinook salmon appear to be trending in a positive direction with improvements to all viability criteria since the previous 2010 assessment. Butte Creek's total population size is 20,169 which is twice the 2010 estimate and remains the largest CVSRC population (Table 5.5).

The majority of CVSRC populations are still exhibiting declines in run sizes over time, with the exception of Clear Creek, Battle Creek, and Butte Creek populations which have positive point estimates of population growth (Table 5.5). In particular, CVSRC appear to be repopulating Battle Creek, home to a historical independent population in the Basalt and Porous Lava diversity group that was extirpated for many decades. Abundance has increased 18% over the last decade (N= 1836) that qualify it for a moderate extinction risk score and trending towards a low-risk threshold of 2500 fish (Lindley et al. 2007). Similarly, the CVSRC population in Clear Creek has been increasing, although Lindley et al. (2004) classified this population as a dependent population, and thus it is not expected to exceed the low-risk population threshold of 2500 fish. While the viability of independent populations has generally improved, the majority of dependent populations have declined in abundance over the last decade (Table 5.5). Recent declines have been significant and almost qualify as catastrophes under the criteria (>90% decline) of Lindley et al. (2007) with the dependent Antelope Creek and Cottonwood Creek populations, and the independent Deer Creek population experiencing recent declines of >80% in one generation (Table 5.5).

Hatchery 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. Coded-wire tag returns confirm that fish identified as FRH spring-run Chinook salmon are intermixed at the hatchery with those identified as fall-run Chinook salmon (Hedgecock et al. 2001; California HSRG 2012). In 2011, 40% of the FRH spring-run Chinook salmon broodstock was comprised of fall-run individuals (Palmer-Zwahlen and Kormos 2013). Based on the moderate extinction risk threshold for gene-flow for one generation between ESUs (< 10%), it places the FRH and naturally spawned CVSRC in the Feather River at a high risk of extinction (Kormos et al. 2012; Palmer-Zwahlen and Kormos 2013; Figure 5.3).

The majority of the FRH spring-run Chinook salmon broodstock and in-river spawning population on the Feather River are first generation hatchery-produced fish (Kormos et al. 2012; Palmer-Zwahlen and Kormos 2013). The proportion of natural-origin fish in the broodstock is estimated to be 18% (2010) and 6% (2011) (Kormos et al. 2012; Palmer-Zwahlen and Kormos 2013). Thus, the minimum criteria of >10% of natural-origin fish

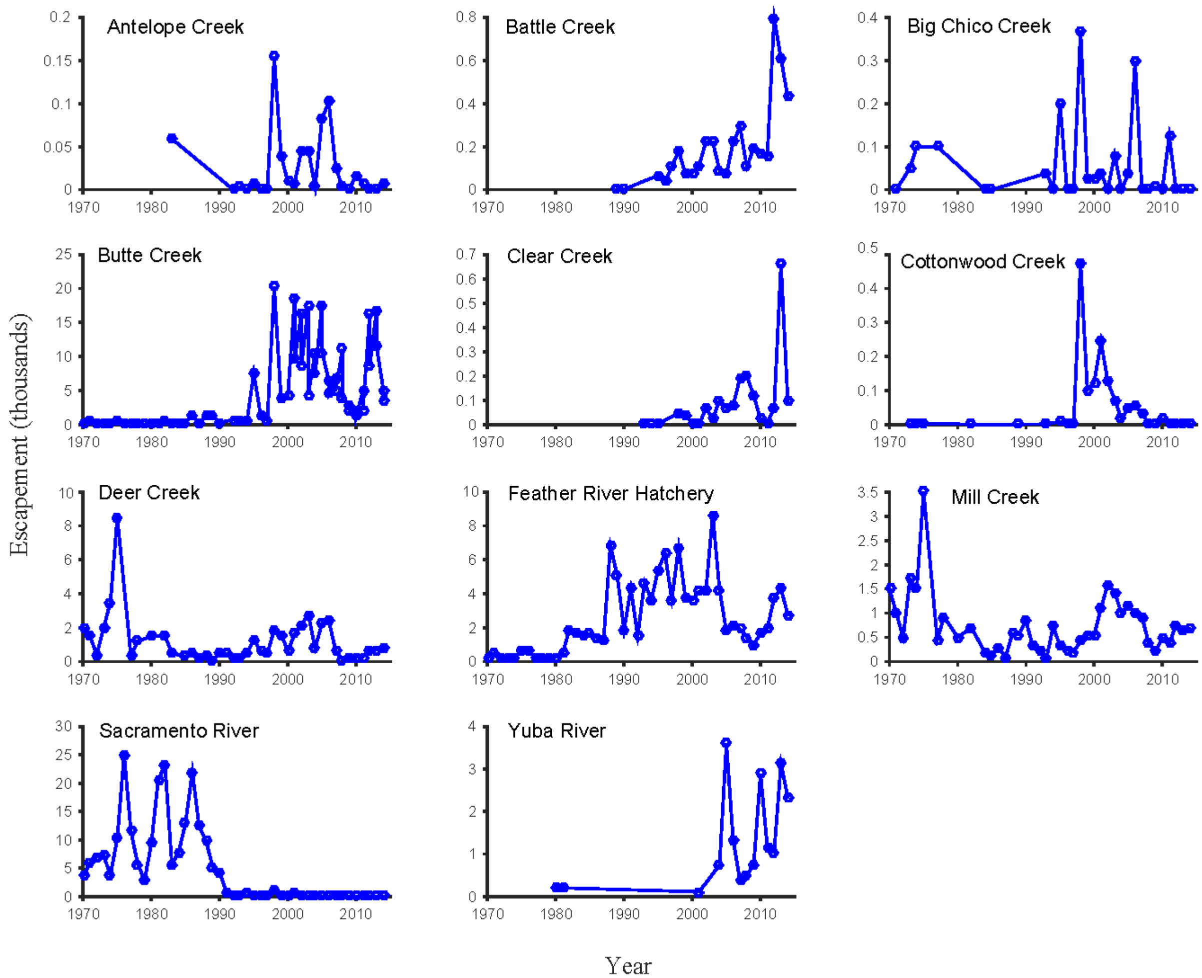

Figure 5.6. Escapement for Central Valley Spring-run Chinook Salmon populations over time in thousands of fish. Note: Beginning in 2009, Red Bluff Diversion Dam estimates of spring-run Chinook salmon in the Upper Sacramento River were no longer available.

in the broodstock is not being met annually (California HSRG 2012). The proportion of hatchery-origin spring- or fall-run contributing to the natural spawning spring-run Chinook salmon population on the Feather River remains unknown due to overlap in the spawn timing (and thus coded-wire tag recoveries in carcass surveys) of spring- and fall-run Chinook salmon. However, the hatchery component is likely to be high. For example, 78% and 90% of spawners in the 2010–2011 spring-run/fall run carcass survey were estimated to be from the FRH respectively (Kormos et al. 2012; Palmer-Zwahlen and Kormos 2013).

Table 5.5. Viability metrics for Central Valley Spring-run Chinook Salmon ESU populations. 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 year-to-year decline in total population size (*N*) over the most recent 10 such ratios.

Population	*N*	$\hat{S}$	10-yr trend (95% CI)	Recent decline (%)
Antelope Creek	8	2.7	-0.375 (-0.706, -0.045)	87.8
Battle Creek	1836	612.0	0.176 (0.033, 0.319)	9.0
Big Chico Creek	0	0.0	-0.358 (-0.880, 0.165)	60.7
Butte Creek	20169	6723.0	0.353 (-0.061, 0.768)	15.7
Clear Creek	822	274.0	0.010 (-0.311, 0.330)	63.3
Cottonwood Creek	4.0	1.3	-0.343 (-0.672, -0.013)	87.5
Deer Creek	2272	757.3	-0.089 (-0.337, 0.159)	83.8
Feather River Hatchery	10808	3602.7	0.082 (-0.015, 0.179)	17.1
Mill Creek	2091	697.0	-0.049 (-0.183, 0.086)	58.0
Sacramento River[a]	-	-	-	-
Yuba River	6515	2170.7	0.67 (-0.138, 0.272)	9.0

a – 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 Chinook salmon adult counts in the upper Sacramento River 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 are no longer available.

The spring-run Chinook salmon population in the Yuba River is at a low risk of extinction based on total population size (N=6,512) yet a high risk due to a conservative estimate of the percentage of hatchery spawners (10 year average = 19%) and likely introgression between fall- and spring-run individuals (RMT 2015). The abundance of spring-run Chinook salmon passing Daguerre Point Dam on the Yuba River was not evaluated in the previous 2010 viability report due to difficulties in differentiating spring-run from fall-run individuals in the existing monitoring. Currently, a video camera is used to count fish moving upstream in the ladders at Daguerre Point Dam. A method for delineating a temporal window for passage of spring- and fall-run passage has been developed and thus counts of spring-run Chinook salmon and fall-run Chinook salmon are reported for 2004–2011 (RMT 2015). Additionally, data on the percentage of adults with and without adipose fins based on silhouettes in the video are reported for 2004–2011(RMT 2015).

Genetic studies suggest that hybridization between FRH spring-run Chinook salmon and other Central Valley Chinook salmon runs has not occurred, where evaluated. For example, where FRH CVSRC strayed extensively, the effect is not apparent in the genetic structure described by microsatellite markers for Central Valley Spring-run Chinook salmon runs in Mill, Deer and Butte creeks, or on winter-run and late-fall-run Chinook salmon that spawn in the mainstem Sacramento River (Banks et al. 2000). These findings are consistent with the generally low straying rates estimated by recovery of coded-wire tags (Kormos et al. 2012; Palmer-Zwahlen and Kormos 2013). However, FRH CVSRC adults have been recovered in other Central Valley spring- and fall-run Chinook salmon populations outside of the Feather River. Feather River Hatchery spring-run Chinook salmon smolts released into the San Francisco Bay pose greater genetic risk to other Central Valley Chinook salmon populations than those released in-river at the hatchery based on their greater stray-rates (Kormos et al. 2012; Palmer-Zwahlen and Kormos 2013). On Clear Creek, 0%-5% of spring-run Chinook salmon carcasses above the spring-run segregation weir in 2010–2013 were from the FRH (USFWS 2014). In 2010 as many as 29% of the CVSRC were estimated to have originated from FRH on Battle Creek (USFWS 2014). A significant number of FRH spring-run Chinook salmon strays have been observed in 2015 in the Upper Sacramento River at the Keswick Dam trap (N=114) and could be interbreeding with natural-origin spring- or fall-run Chinook salmon(J. Rueth, USFWS, personal communication). 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). 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 populations outside of the Feather River.

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). More recently, there have been reports of adult Chinook salmon returning in February through June to San Joaquin River tributaries, including the Mokelumne, Stanislaus, and Tuolumne rivers (Workman 2003; Franks 2012; Guignard 2015). These spring-running adults have been observed in several years and exhibit typical spring-run life-history characteristics, such as returning to tributaries during the springtime, over-summering in deep pools, and spawning in early fall (Workman 2003; Franks 2011; Guignard 2015). For example, 114 adults were counted using a video weir on the Stanislaus River between February and June in 2013 with only 7 individuals observed without adipose fins (Guignard 2015). Since all hatchery-origin CVSRC have their adipose fins removed, these data suggest the vast majority of the adult spring-run Chinook salmon were not strays from the FRH. It is possible that they are unmarked fall-run Chinook salmon hatchery adults that strayed to the Stanislaus River from the Feather River Hatchery, Coleman National Fish Hatchery, Nimbus River Fish Hatchery, Mokelumne River Hatchery, or Merced River Hatchery. The extent to which these phenotypic spring-run Chinook salmon have a similar genetic lineage as other extant spring-run Chinook salmon populations and that they stray each generation from the Sacramento River Basin remains unknown and is the source of on-going research. It is conceivable that progeny from adult spring-run Chinook salmon return to their natal tributaries on the San Joaquin River and thus represent early stages of a recolonization process trending towards a self-sustaining population. Juveniles expressing atypical fall-run outmigration behavior, more characteristic of spring-run (e.g., yearlings) have also been observed on the Mokelumne, Tuolomne, and Stanislaus rivers (Fuller 2008; Watry et al. 2012; Bilski et al. 2013). In addition, in 2014, a reintroduction program was initiated as part of the San Joaquin River Restoration Program, and 54,000 juvenile spring-run Chinook salmon were released into the river. Successful reestablishment of CVSRC into multiple populations in the Southern Sierra Nevada Group would significantly increase their spatial diversity and decrease their risk of extinction.

Harvest Impacts[13]

Attempts have been made (Grover et al. 2004) to estimate Central Valley Spring-run Chinook (CVSRC) Salmon ESU ocean fishery exploitation rates using coded-wire tag recoveries from natural origin Butte Creek fish, but due to the low number of recoveries the uncertainty of these estimates is too high for them to be of value. However, because CVSRC have a relatively broad ocean distribution from central California to Cape Falcon, Oregon, that is similar to that of Central Valley Fall-run Chinook (CVFRC) salmon, trends in the CVFRC ocean harvest rate may provide a reasonable proxy for trends in the CVSRC ocean harvest rate. While the CVFRC ocean harvest rate can provide information on trends in CVSRC fishing mortality, it is possible that CVSRC experiences lower overall fishing mortality. If maturation rates are similar between CVSRC and CVFRC, the ocean exploitation rate on CVSRC would be lower than CVFRC 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.

The CVFRC ocean harvest rate index peaked in the late 1980s and early 1990s, but then declined (Figure 5.7). With the closure of nearly all Chinook salmon ocean fisheries south of Cape Falcon in 2008 and 2009, the index dropped to 6% and 1%, respectively. While ocean fisheries resumed in 2010, commercial fishing opportunity was severely constrained, particularly off California, resulting in a harvest rate index of 16%. Since 2011, ocean salmon fisheries in California and Oregon have had more typical levels of fishing opportunity. The average CVFRC ocean harvest rate between 2011 and 2014 is 45% which is generally similar to levels observed between the late 1990s and 2007.

The CVSRC spawning migration largely concludes before the mid- to late-summer opening of freshwater salmon fisheries in the Sacramento Basin, and salmon fishing is prohibited altogether on Butte, Deer, and Mill creeks, indicating that CVSRC river fishery impacts are relatively minor.

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[13] Harvest impacts section prepared by Michael O'Farrell

Figure 5.7. Central Valley Fall-run Chinook (CVFRC) Salmon ocean harvest index for years 1983 – 2014. The harvest rate index is computed from estimates presented in Table II-1 from PFMC (2015b).

In summary, the available information indicates that the level of CVSRC fishery impacts has not changed appreciably since the 2010 salmon and steelhead assessments (Williams et al. 2011).

Summary and Conclusions

Central Valley-wide, the viability of CVSRC has probably improved on balance since the 2010 viability assessment with improvements to Mill Creek and Deer Creek populations changing from high-risk to moderate-risk of extinction. In fact, total abundance of CVSRC for the Sacramento River watershed in 2014 (not including the FRH or Feather River but with the addition of Yuba River spring-run Chinook salmon) is 45,215, close to the decadal high of 55,827 (2004) and a factor of approximately four times higher than the decadal low of 12,207 which occurred as recently as 2012 (Azat 2014; RMT 2015). The Central Valley-wide abundance is driven largely by the annual variation in Butte Creek returns. Butte Creek remains at low risk, and all viability metrics are trending in a positive direction. The Butte Creek spring-run Chinook salmon population has increased in part due to extensive habitat restoration and the accessibility of floodplain habitat in the Sutter-Butte Bypass for juvenile rearing in the majority of years. Most dependent spring-run Chinook salmon populations have been experiencing continued and somewhat drastic declines. Counteracting these developments, CVSRC have repopulated Battle and

Clear creeks where they were once extirpated and have increased in abundance over the last decade, reaching levels of abundance that place these populations at moderate extinction risk. In the case of Clear Creek, the majority of fish spawning there are of natural origin (96%) suggesting local production may be promoting a self-sustaining population without significant hatchery supplementation (Kormos et al. 2012; Palmer-Zwahlen and Kormos 2013).

Central Valley Spring-run Chinook salmon populations have experienced a series of droughts over the past decade. From 2007–2009 and 2012–2015, the Central Valley experienced drought conditions and low river and stream discharges, which are generally associated with lower survival of Chinook salmon. The impacts of the recent drought series and warm ocean conditions on the juvenile life stage (see Chapter 2 of this assessment) will not be fully realized by the viability metrics until they manifest in potential low run sizes in 2015–2018.

The recent drought has impacted CVSRC adults on Butte Creek, which have experienced lethal temperatures in traditional and non-traditional holding habitat 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 in the vicinity of the City of Chico experienced exceptionally warm June air temperatures coupled with the Pacific Gas and Electric flume shutdown resulting in a fish die-off. Thus, while the independent CVSRC populations have generally improved since 2010 and are considered at moderate and low risk of extinction, the viability of CVSRC populations are not likely improvement over the next three years due to likely unfavorable hydroclimatic regimes.

Current introgression between fall- and spring-run Chinook salmon in the FRH breeding program and straying of FRH spring-run Chinook salmon to other non-hatchery spring-run Chinook salmon populations could compromise the genetic integrity of spring-run Chinook salmon populations. Off-site releases of FRH spring-run Chinook salmon has resulted in increased straying of hatchery fish into other spring-run Chinook salmon populations and if continued could result in a moderate risk of extinction to other spring-run Chinook salmon populations. However, beginning in 2014, and expected to continue, the FRH has released spring-run Chinook salmon juveniles into the Feather River rather than releasing them in the San Francisco Bay, which is hypothesized to reduce straying (California HSRG 2012).

At the ESU level, the spatial diversity within the CVSRC ESU is increasing and spring-run are present (albeit at low numbers in some cases) in all diversity groups. The recolonization of CVSRC to Battle Creek and increasing abundance of CVSRC on Clear Creek is benefiting the viability of CVSRC. Similarly, the reappearance of phenotypic spring-run to the San Joaquin River tributaries may be the beginning of natural recolonization processes in rivers where they were once extirpated. Active reintroduction efforts on the Yuba River and below Friant Dam on the mainstem San Joaquin River show promise and will be necessary to make the ESU viable. 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).

The viability of the CVSRC ESU has likely improved since the 2010 viability assessment. Largest improvements are due to the increase in spatial diversity with historically extirpated populations trending in the positive direction. Improvements, evident in the moderate and low risk of extinction of the three independent populations, are certainly not enough to warrant a downgrading of the ESU extinction risk. The recent catastrophic declines of many of the dependent populations, high pre-spawn mortality during the 2012–2015 drought, uncertain juvenile survival due to the drought and variable ocean conditions, as well as the level of straying of FRH spring-run Chinook salmon to other spring-run Chinook salmon populations are all causes for concern for the long-term viability of the CVSRC ESU.

5.3 California Central Valley Steelhead DPS

DPS Boundary Delineation

This Distinct Population Segment (DPS) includes steelhead populations spawning in the Sacramento and San Joaquin rivers and their tributaries. Hatchery stocks within the DPS include Coleman National Fish Hatchery (CNFH) and Feather River Hatchery (FRH); steelhead in the Nimbus Hatchery (NH) and Mokelumne River Hatchery (MRH) are currently excluded from the DPS. New genetic analysis show that the steelhead stock currently propagated in the Mokelumne River Hatchery is 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 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, we recommend a change in boundary delineation, the boundary of the Central Valley DPS should be modified to include steelhead from the Mokelumne River Hatchery.

Summary of Previous Assessments

Good et al. (2005) found that California Central Valley (CCV) Steelhead DPS was in danger of extinction, with a minority of the BRT viewing the DPS was likely to become endangered. The BRT's major concerns were the low abundance of naturally produced anadromous fish at the DPS (considered an ESU at the time of the review) level, the lack of population-level abundance data, and the lack of any information to suggest that the monotonic decline in steelhead abundance evident from 1967–1993 dams counts had stopped. Williams et al. (2011) reported that the viability of this steelhead DPS had worsened since the 2005 review when Good et al. (2005) concluded that the DPS was in danger of extinction.

Brief Review of TRT Documents and Previous Findings

The Central Valley domain Technical Recovery Team delineated more than 80 independent populations of Central Valley steelhead, along with a number of smaller dependent populations. Many of these historical populations are entirely above impassable barriers and may persist as non-anadromous or adfluvial rainbow trout, although they are presently not considered part of the DPS. Impassable dams also block many 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 Central Valley 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, 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%.

New Data and Updated Analyses

Population trend data remain extremely limited for the CCV-Steelhead ESU. The total populations on Battle Creek, CNFH, and FRH have significantly increased since the 2010 assessment with all three populations showing positive population growth estimates over the last decade (Figure 5.8; Table 5.6). Additional data are now available for the American River and Clear Creek steelhead populations and are based on redd counts. Thus, steelhead populations on the American River and Clear Creek are evaluated for the first time using the viability criteria recognizing that some redds in Clear Creek may be from non-anadromous *O. mykiss* (Figure 5.8; Table 5.6).

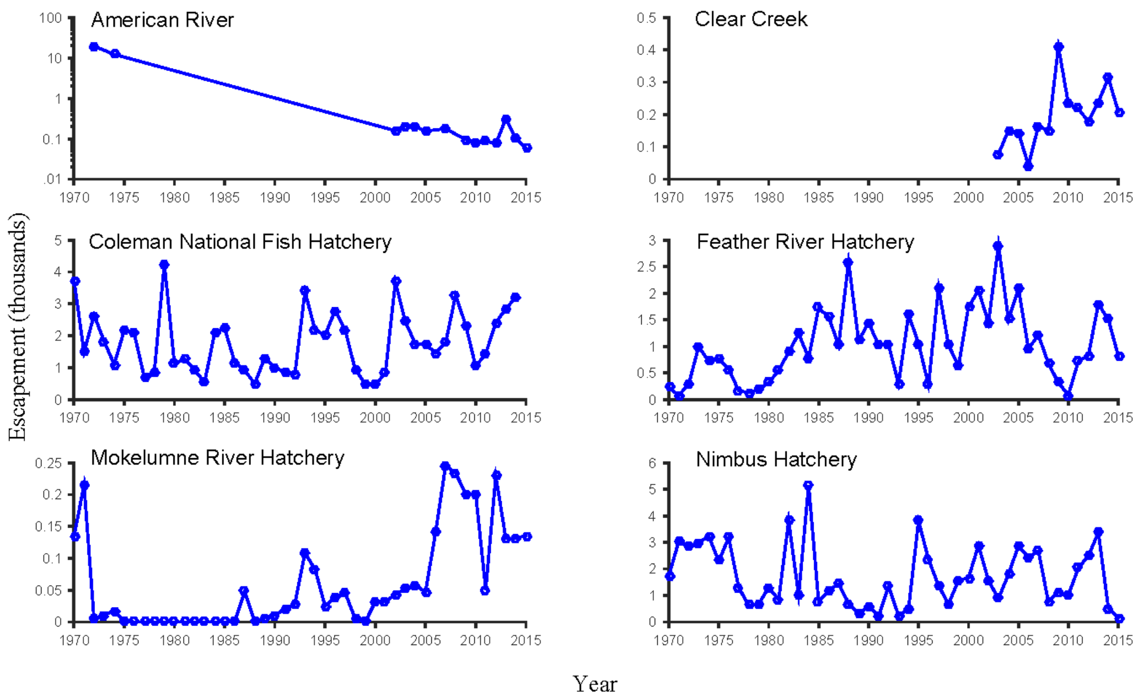

Figure 5.8. Time series of escapement for California Central Valley Steelhead populations in thousands of fish. Note that the y-axis of plot for American River steelhead is on a logarithmic scale.

The best population-level data come from Battle Creek, where CNFH operates a weir. California Central Valley steelhead have been identified as a priority species for restoration in Battle Creek above 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 CCV steelhead within the Basalt and Porous Lava diversity group (NMFS 2009a). In 2002, 2000 steelhead were passed above the weir into the BCSSRP area to spawn naturally 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 2014). 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 steelhead used in the breeding protocols, and only natural origin steelhead passed upstream of the weir into the BCSSRP area (California HSRG 2010). Between 2012 and 2014, the total population of natural-origin adults > 17 inches (size threshold identified for anadromous *O. mykiss* at CNFH; Donohoe and Null 2013) passing the weir was 510 with an average run size of 170 adults (USFWS 2015). The low abundance of natural-

Table 5.6. Viability metrics for California Central Valley steelhead populations. Total population size (*N*) is estimated as the sum of estimated run sizes over the most recent three. 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 year-to-year decline in total population size (*N*) over the most recent 10 such ratios.

Population	*N*	$\hat{S}$	10-yr trend (95% CI)	Recent decline (%)
American River[a]	472	157.3	-0.062 (-0.164, 0.039)	45.8
Clear Creek[a]	761	253.7	0.111 (-0.021, 0.244)	9.5
Coleman National Fish Hatchery	8461	2820.3	0.051 (-0.043, 0.146)	18.4
Feather River Hatchery[b]	4119	1373.0	0.061 (-0.171, 0.292)	38.3
Mokelumne River Hatchery	398	132.7	-0.051 (-0.169, 0.067)	30.5
Nimbus Hatchery	4052	1350.7	-0.155 (-0.378, 0.067)	4.5

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

origin steelhead places it in the moderate extinction risk category, albeit with lower hatchery influence than the previous 2010 assessment. Various management options and potential consequences 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 considering the value of integrating “wild” genes back into CNFH hatchery production to minimize impacts of domestication on both the hatchery- and natural-origin steelhead populations. It is difficult to assess the impact of the CNFH segregation on hatchery steelhead, as little is known about the extent to which CNFH steelhead ascend the segregation weir during high flows and spawn with natural-origin steelhead in the BCSSRP area upstream or with natural-origin steelhead downstream of the weir. In general it requires less influx of hatchery-origin fish from segregated hatcheries than from integrated hatcheries into naturally spawning populations to have significant genetic impacts (California HSRG 2014).

The total population on Clear Creek has increased since it was first estimated in 2003, reaching a total population size of 761, estimated by the number of redds counted and increasing 11% per year over the past decade (Figure 5.8; Table 5.6). American River steelhead had a precipitous decline since 2003, resulting in a moderate risk of extinction based on current total population size estimated by redd surveys. 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, but are also showing a 15% decline over the last decade.

The NH broodstock remains a high 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 Central Valley DPS lineages, having been founded from Eel and Mad River stocks (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).

Zimmerman et al. (2009) found that the progeny of anadromous females were present at all eight Central Valley populations evaluated 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 are increasing with the installation of video weirs on the Mokelumne, Stanislaus, and Tuolumne rivers during adult steelhead migration. The numbers of natural-origin adult steelhead remains low, with a high hatchery influence, placing the populations in the San Joaquin River tributaries (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 1–17 during 2005 to 2008 and 8–32 during 2011 to 2014 (Ford and Kirihara 2010; Fuller 2015.). Thirteen to fifty percent of those fish were identified as hatchery fish having clipped adipose fins, placing the Stanislaus River population at a high risk of extinction based on low numbers and high hatchery influence (Ford and Kirihara 2010; Fuller 2015). The Mokelumne River is also at a high risk of extinction with 92%–96% of adult steelhead at the Woodbridge Dam video identified as hatchery steelhead and with only 3–10 natural-origin steelhead returning to the Mokelumne River each year from 2010–2013 (EBMUD 2011, 2012, 2013).

Steelhead survival from Mossdale on the San Joaquin River to Chipps Island (Delta exit) ranged from 25%–75% in 2010 and 2011 based on acoustic telemetry studies (Buchanan 2013). These survival estimates are significantly higher that what has been observed for fall-run Chinook salmon released in the same location and under similar conditions (SJRGA 2011). The relatively high survival of steelhead is thought to be due to the larger size between the species at release. It is unclear the extent to which naturally produced steelhead experience similar survival rates as the hatchery experimental release groups. In fact, evidence from Chipps Island midwater trawl sampling by USFWS suggests either natural steelhead production and/or survival to Delta exit is very low. The Chipps Island midwater trawl data provide information on the trend in abundance for the CCV Steelhead DPS as a whole. Updated through 2013, 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 but remained level over the past decade, but the proportion of the catch that is adipose fin-clipped (100% of hatchery steelhead production have been adipose fin-clipped starting in 1998) has risen steadily, exceeding 90% in recent years, reaching 95% in 2010, and remaining very high through 2013. Because hatchery releases have been fairly constant, this implies that natural production of juvenile steelhead has been falling.

Harvest Impacts[14]

Ocean harvest of steelhead is extremely rare, and is in particular an insignificant source of mortality for the CCV-Steelhead DPS. Insufficient data are available to estimate CCV-Steelhead freshwater exploitation rates directly, though exploitation rates are likely relatively low given that retention of natural-origin steelhead is prohibited. Fishing effort estimates based on angler self-report cards, available from 2000–2014 (Figure 5.10). Jackson (2007) noted an increase in Central Valley steelhead fishing effort prior to 2000 that was accompanied by a decrease in fishing effort observed on many coastal streams, and suspected that this may be the result of regulations allowing retention of hatchery-origin steelhead in the Central Valley.

Since the 2010 assessment there have been changes in fishing regulations that could have effects on freshwater fishery impacts. In March 2010, the hatchery-origin (adipose fin-clipped) steelhead bag limit increased from 1 to 2 fish on the Sacramento and American rivers (the possession limit increased from 2 to 4 fish as well). In March 2015, the bag and possession limit on adipose fin clipped steelhead increased on the Feather River, matching the previous regulations change on the Sacramento and American rivers. Recent drought conditions have affected some steelhead fishing opportunities for this DPS. For example, the California Fish and Game Commission imposed an emergency fishery closure on the American River in February of 2014. The closure ended in April of that year.

[14] Harvest impacts section prepared by Michael O’Farrell

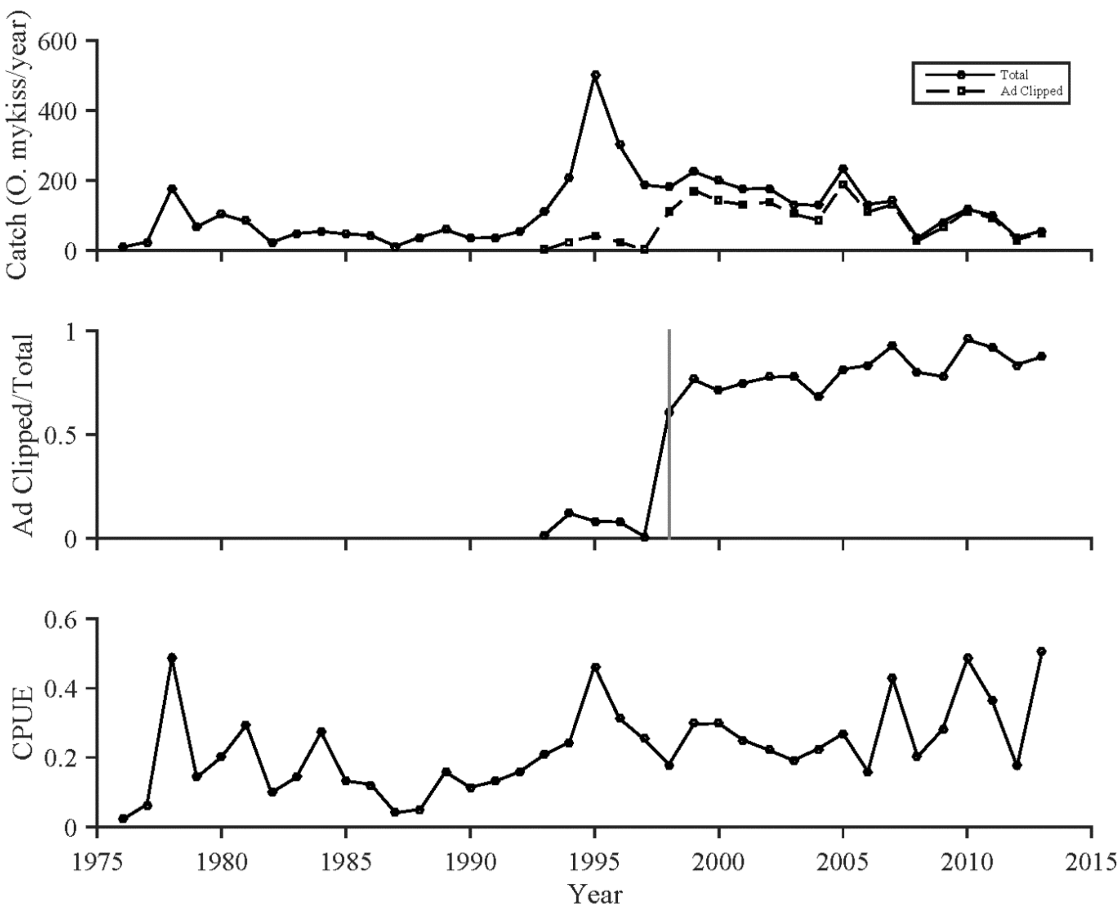

Figure 5.9. Top: Catch of steelhead at Chipps Island by the USFWS midwater trawl survey. Middle: Fraction of the catch with an adipose fin clip, 100% of hatchery steelhead have been marked (adipose fin clip) starting in 1998 (denoted with the vertical gray line). Bottom: Catch-per-unit-effort (CPUE) 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.

California Department of Fish and Wildlife 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 CCV-Steelhead fishery impacts and an assessment of whether freshwater fishery impacts have increased in response to recent regulation changes cannot yet be made. Given this sparse information, it is likely that the level of impact has either not changed since the 2010 salmon and steelhead assessment (Williams et al. 2011), or has potentially increased due to increased bag and possession limits for hatchery-origin fish.

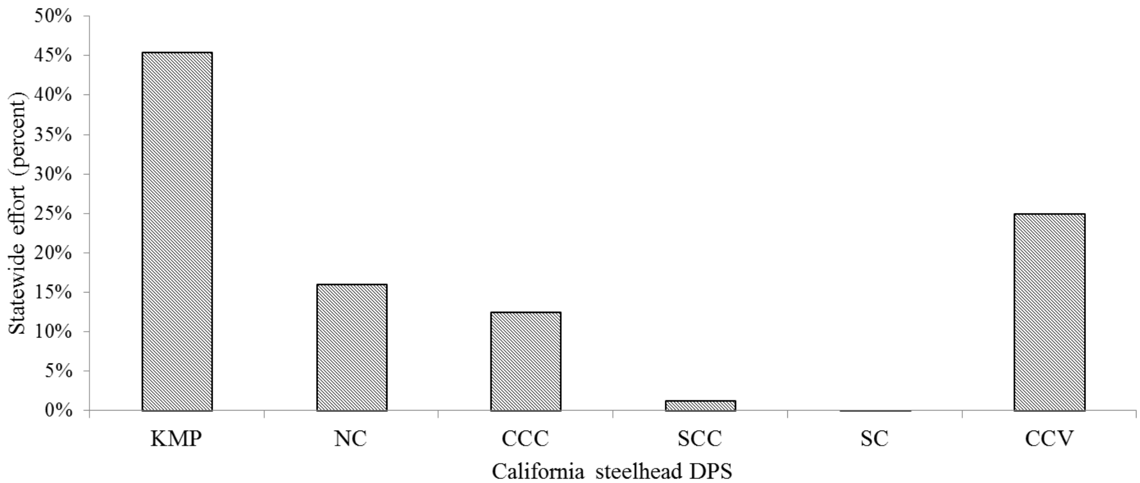

Figure 5.10. Distribution of California statewide steelhead fishing effort by DPS for years 2000−2014 (Jackson 2007; Farhat in preparation).

Summary and Conclusions

One of the greatest challenges in managing for resilient steelhead populations in our regulated rivers lies in understanding how water project operations promote, maintain, or suppress the expression and survival of the anadromous life-history form of *O.mykiss*. It is clear that some river habitats support almost exclusively abundant, non-anadromous populations, while others support the expression of anadromy (Satterthwaite et al. 2010). In the San Joaquin River tributaries specifically, there is great uncertainty in the extent to which the production of anadromous juveniles from tributaries is low and/or whether mortality of juvenile steelhead is so high during outmigration so as to preclude higher numbers of returning adult steelhead. While research suggests that the non-anadromous form can give rise to the anadromous form, it is possible that this only occurs at low levels, and it is more common for the steelhead form to give rise to the non-anadromous form (Donohoe et al. 2008). More studies are needed to understand the extent to which genes associated with the heritable components of anadromy could be lost from populations with low steelhead numbers, thus placing them at a greater risk of extinction.

The viability of the CCV-Steelhead DPS appears to have slightly improved since the 2010 assessment, when it was concluded that the DPS was in danger of extinction. This modest improvement is driven by the increase in adult returns to hatcheries from their recent lows, but the state of naturally produced fish remains poor. Improvements to the total population sizes of the three previously evaluated steelhead populations (Battle Creek, CNFH, and FRH), does not warrant a downgrading of the ESU extinction risk. In fact, the lack of improved natural production as estimated by samples taken at Chipps Island, and low abundances coupled with large hatchery influence in the Southern Sierra Nevada Diversity group is cause for concern. As in the previous assessments (Good et al. 2005; Williams et al. 2011), the CCV-Steelhead DPS continues be at a high risk of extinction.

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

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