ANALYSES OF WEIR COUNTS AND SPAWNING SURVEYS OF ADULT CHINOOK SALMON IN THE STANISLAUS RIVER

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EXECUTIVE SUMMARY

Reliable estimates of spawner abundances (escapements) of salmon provide critical information for effective management of salmon populations. To this end, the Stanislaus River Weir Project (SRWP) was developed by the Tri-Dam Project, Oakdale Irrigation District, and the South San Joaquin Irrigation District to test a portable Alaskan weir to count and characterize runs of anadromous salmonids in the Stanislaus River. To date, the weir program has enumerated and collected biological information from migrating adult fall-run Chinook salmon in each of three years: 2003, 2004, and 2005. In addition to the weir program, the California Department of Fish and Game (CDFG) conducts annual escapement surveys of Chinook salmon in the Stanislaus River.

In this report, we completed a series of statistical analyses addressing the migration and enumeration of Chinook salmon in the Stanislaus River. The specific objectives of these analyses were as follows:

- Quantify the response of migrating adults to changes in flow, in particular regulated "attraction flows", to better determine the volume and duration of flow releases required to stimulate upstream migration;
- Quantify the response of migrating adults to changes in temperature and dissolved oxygen;
- Analyze VAKI data to determine the accuracy and precision of weir counts;
- Compare weir counts and CDFG escapement estimates derived from carcass surveys to determine the accuracy of carcass surveys;
- Collect and review historic carcass survey data and escapement estimates to assess the reliability of the historic record and possible evidence of spawner condition (egg retention, pre-spawn mortality) and distribution.

In addition to these objectives, we also conducted a statistical analysis to address concerns that the CDFG historic scale record for San Joaquin Chinook salmon may not be applicable to the Stanislaus River (Simpson et al. 2006).

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Summary of key findings

<u>Accuracy and precision of weir counts</u>

We developed statistical frameworks for assessing the accuracy and precision of the Vaki system and for estimating missing counts and their variances. Based on trapping data for 2004, the Vaki system appeared to be quite accurate and reasonably precise at detecting migrating Chinook salmon. The estimated detection probability was 0.97, implying a slight undercount on average (a detection probability of one is a perfect count). The 95% confidence interval for the detection probability ranged from 0.89 to 1.05.

This detection probability was applied to the total weir counts in each year to generate estimates of total migrations of Chinook salmon. Total migration estimates and their 95% confidence intervals are as follows:

2003: 5009 (4543 – 5474)

2004: 4580 (4208 – 4952)

2005: 4258 (3912 – 4604)

The coefficient of variation (*CV,* a relative measure of precision) for each estimate of total migration was less than 5%, implying that the estimates are very precise (a *CV* of 0 implies perfect precision). Thus, weir counts of Chinook salmon appear to provide high quality estimates of migration passage.

<u>Factors affecting migration timing</u>

Short-term increases in adult salmon migration (2-3 days) can be stimulated by modest, short duration increases in flow (e.g., 200 cfs for 2-3 days). For example, in 2003, a relatively modest pulse-flow event (an increase of roughly 200 cfs), which preceded the start of attraction flows by two weeks, resulted in a two-day spike in weir passage of nearly 1,000 fish. Attraction flows in 2004 and 2005 caused similar short-term migration responses as well. In general, the timing of these events suggested the migrations were initially stimulated by modest increases in flow (e.g., 200 cfs).

In addition to immediate in-river responses to flow, there was a consistent pattern across years in which a 3-4 day increase in passage at the weir occurred in early November, roughly three weeks after flows in the San Joaquin River had markedly increased. Though speculative, these migrations may have started low in the San Joaquin River in response to flow increases. Goodwin attraction flows contributed to large, prolonged increases in San Joaquin flow in 2003 and 2005, whereas large flow increases had already occurred in 2004.

In contrast to flow, there were no obvious associations between migration timing and conditions for temperature, dissolved oxygen (DO), or turbidity. Rather, there were some unexpected patterns. Although temperatures remained above 70°F for an extended period during the 2003 migration, a larger fraction of fish migrated during September and early October in 2003 than in the later years. Similarly, DO levels were especially low in 2004, yet the migration timing in that year was no different than that observed in 2005.

Accuracy and precision of carcass surveys

We reviewed and compiled annual CDFG escapement reports, which were available dating back to 1987. Estimates of precision were often lacking, so we derived surrogate measures when possible. Across years, the escapements estimates for Chinook salmon in the Stanislaus River appeared reasonably precise. This was especially true for recent years in which sampling efforts increased dramatically. However, it is much more difficult to assess the accuracy of the historic escapement estimates. Mark-recapture estimates can be strongly biased if underlying assumptions are violated. In a few years, escapements estimates appeared highly suspect. In these years, abundances were very low and/or sampling efforts were hindered by high flows.

Despite strong CDFG sampling efforts in recent years, we found that Schaefer estimates differed markedly from the weir estimates in 2003-2005. For 2003, the Schaefer estimate was higher than the weir estimate by roughly 2,000 fish (39% higher). In contrast, Schaefer estimates were lower than weir estimates in 2004 and 2005 by roughly 1,100 fish (24% lower) and 1,200 fish (28% lower), respectively. Further, an especially large and uncertain weekly estimate prevented the overall Schaefer estimate in 2005 from dramatically underestimating annual

escapement. These results suggest that systematic biases may inadvertently occur during the carcass mark-recapture studies.

Applicability of the historic scale record

Scales collected from adult Chinook salmon in the San Joaquin Basin provide a historic record of age data that could potentially be applied to Stanislaus River Chinook salmon in an effort reconstruct cohort recruitment. However, there have only been a few years where large scale samples were analyzed for the Stanislaus River. If there is little similarity across rivers in year-to-year changes in adult age structure, then the historic scale record would not be applicable to the Stanislaus River.

To address this concern, we assessed evidence of similarities in adults age structure across San Joaquin tributaries using the following steps: (1) we used CDFG scale data to derive length criteria for distinguishing between age-2 fish and older fish; (2) we applied the criteria to length samples to estimate the proportion of age-2 spawners by river and year; and (3) we compared these age-2 proportions across rivers, and with age-2 proportions estimated from the scale data. Results of these comparisons strongly suggest that Chinook populations of the Tuolumne, Merced, and Stanislaus Rivers exhibit similar year-to-year changes in spawner age structure. It would therefore be appropriate to apply the historic scale record to Stanislaus River Chinook salmon even though few scales were represented by these fish.

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INTRODUCTION

Reliable estimates of spawning populations (escapements) of salmon are essential for effective management of these populations. Accurate escapement estimates provide critical information for monitoring population status and assessing key components of population dynamics such as cohort recruitment strength, factors affecting freshwater survival rates, and optimal spawner objectives. To this end, the Stanislaus River Weir Project (SRWP) was developed by the Tri-Dam Project, Oakdale Irrigation District, and the South San Joaquin Irrigation District to test a portable Alaskan weir to count and characterize runs of anadromous salmonids in the Stanislaus River. As discussed in the 2004 Final Weir Report (S.P. Cramer, 2004), the Stanislaus River Weir was constructed and installed at river mile 31.4 near the town of Riverbank (Figure 1).

To date, the weir program has enumerated and collected biological information from migrating adult fall-run Chinook salmon in each of three years: 2003, 2004, and 2005. In addition to the weir program, the California Department of Fish and Game (CDFG) conducts annual escapement surveys and mark-recapture studies in the Stanislaus River to determine spawning population characteristics and abundance of fall-run Chinook salmon.

In this report, we conducted a series of statistical analyses addressing issues related to the migration and enumeration of Chinook salmon in the Stanislaus River. The specific objectives of these statistical analyses were as follows:

- Quantify the response of migrating adults to changes in flow, in particular regulated "attraction flows", to better determine the volume and duration of flow releases required to stimulate upstream migration;
- Quantify the response of migrating adults to changes in temperature and dissolved oxygen;
- Analyze VAKI data to determine the accuracy and precision of weir counts;
- Compare weir counts and CDFG escapement estimates derived from carcass surveys to determine the accuracy of carcass surveys;
- Collect and review historic carcass survey data and escapement estimates to assess the reliability of the historic record and possible evidence of spawner condition (egg retention, pre-spawn mortality) and distribution.

In addition to these objectives, we also conducted a statistical analysis to address concerns that the CDFG historic scale record for San Joaquin Chinook salmon may not be applicable to the Stanislaus River (Simpson et al. 2006).

The report contains five main sections. In the first, we assess the accuracy and precision of weir counts and generate total escapements for 2003-2005 based on the weir counts. Second, we examine the affects of flow, temperature, and dissolved oxygen on the migration timing of adult Chinook spawners. Third, we review historic CDFG escapement surveys and examine data on spawner distribution. Fourth, we compare the weir and carcass-survey estimates of escapement for 2003-2005. In the final section, we assess evidence of similarities among age-structures of spawners across rivers to address concerns regarding the historic scale record.

Figure 1. Map of the Stanislaus River indicating the location of the weir near the town of Riverbank.

ASSESSMENT OF WEIR COUNTS

During each season of weir operation, fish passage was monitored using a Vaki RiverWatcher digital infrared fish counting system, by trapping, and by visual surveys upstream and downstream of the weir. Each of these methods is detailed in S.P. Cramer (2004). In brief, the Vaki system consists of an infrared digital system and computer that records the event of an object passing between two scanner plates, including the time, speed, and direction of passing object. Simultaneous photo and silhouette images allow identification of the species and other characteristics when possible (e.g., sex), as well as the depth and length of the object.

In addition, trapping in the passing chute upstream from the weir was used during limited sampling periods (i.e. no more than two consecutive days and when temperatures were below 60°F) within each season to periodically collect scale samples and other biological data from salmonids. The initial trap design was found to be ineffective (S.P. Cramer, 2004), but was modified and successfully employed during the 2004 and 2005 Chinook spawner migrations.

Comparison of the trap samples with the corresponding Vaki counts for each sampling period allows for the estimation of the detection probability of the Vaki system, which in turn allows for estimation of an adjusted weir count and confidence interval. Thus, our objective was to examine the accuracy and precision of the Vaki counting system in order to estimate the total Chinook spawner passage (with an estimate of precision) in each of the three years. As described in the following sections, we provide a statistical analysis of the 2004 trap samples and apply the results to all three years of migration data. This task included the development of algorithms for estimating missing counts that occurred in 2003.

Methods

Missing counts

In 2003, there were some days in which Vaki counts were missing because of equipment malfunctions or when weir panels were removed due to digging below the weir. As detailed below, there were some days with partial counts that were missing counts for blocks of hours. In other days, all 24 hours of data were missing, as well counts in the hours or days before and after the 24-hour period.

Estimates of missing counts by day were computed differently depending on whether there was a "partial" count for that day (i.e., not all 24 hours were missing) or no count. In the case of partial counts, all available counts in the 24-hour periods before and/or after the missing data were used to estimate missing counts on an hourly basis. Specifically, we used the mean of the available counts as the estimate for each missing hourly count, and then summed the missing hourly counts to provide an estimate of the total missing count for a given day. In statistical terms, let y denote the missing count for hour i within a continuous block of m missing hours. Now let x denote an adjacent count for hour j within continuous blocks of n available hours. The total missing count (c) was estimated as:

$$(1) \qquad \hat{c} = \sum_{i=1}^{m} \hat{y}_i = m * \bar{x} \; ,$$

where $\bar{x}$ is the average of the n available hourly counts. It can be shown that the variance of the total missing count estimate is given by:

$$(2) \qquad \text{vâr}(\hat{c}) = m^2 * \left(s_{\bar{x}} \right)^2 + m * s_x^2 = m^2 * \frac{s_x^2}{n} + m * s_x^2 \; ,$$

where s_x denotes the standard deviation of the n available hourly counts.

For days with all 24 hours of data missing, we used an explicit time-series model (e.g., Kohn and Ansley 1986) to estimate the missing daily counts and associated variances. In brief, all daily counts in 2003 were modeled in a state-space (Kalman filter) formulation consisting of two equations (Harvey 1989). The first represents the "observation" equation:

$$(3) \qquad y_t = x_t + e_t$$

where y is the logarithm of the observed daily count for day t, x denotes the time-varying mean of the daily count, and e is randomly distributed normal error. The second or "state" equation models the time trend in the mean:

$$(4) \qquad x_t = x_{t-1} + z_t$$

where x is assumed to follow a "random walk" from one day to the next with randomly distributed normal error (z). This Kalman-filter formulation was fit to the complete time series of daily counts for 2003, complete with missing days, to generate parameter estimates for the variances of the error terms (e and z), as well as estimates of daily counts for missing days ($\hat{y}_t$) and their variances.

Vaki detection probability and adjusted weir counts

Counts of adult Chinook spawners collected with the Vaki system were compared with trapping counts collected during 35 independent trapping periods (samples) in 2004 (details provided in the "Results" section below). Only trapping periods in which at least one fish was counted by either the Vaki system or in the trap were included in the analysis. The number of fish captured in the trap was assumed to represent the true number passing the weir.

To evaluate the accuracy and precision of the Vaki counts, we estimated the detection probability (p) and its associated variance using methods described in Thompson (2002). The probability of detection is estimated by the ratio

$$(5) \qquad \hat{p} = \bar{x}/\bar{y} \, ,$$

where $\bar{x}$ is the average number of fish counted by Vaki across all trapping intervals, and $\bar{y}$ is the average number of fish counted in the trap samples. The variance of $\hat{p}$ is given by

$$(6) \qquad \text{vâr}(\hat{p}) = s_d^2 / n\bar{y}^2 \, ,$$

where s_d^2 is the standard sampling variance used in ratio estimation, and is given by

$$(7) \qquad s_d^2 = \sum_{i=1}^{n} (x_i - \hat{p}y_i)^2 /(n-1) \, .$$

Here, x and y are the Vaki counts and trap counts, respectively, for each trapping interval i, and n is the total number of trapping intervals. The estimated detection probability for the Vaki system provides a measure of its accuracy, whereas the variance provides a measure of the precision the Vaki system.

The detection probability can then be used to compute an adjusted weir count (τ) based on the total Vaki count (c) across days of an annual migration. Assuming that $\hat{p}$ is an approximately unbiased estimate for p and that $\hat{p}$ is uncorrelated with the number of fish observed during each trapping interval, an estimator of the population total (i.e., the actual number of fish passing the weir) is

(8) $\qquad \hat{\tau} = \dfrac{c}{\hat{p}}$

where c is the total number of fish counted by the Vaki system. An approximate formula for the variance of the estimated total is given by

(9) $\qquad \text{vâr}(\hat{\tau}) \approx \hat{\tau}\left(\dfrac{1-\hat{p}}{\hat{p}}\right) + \dfrac{\hat{\tau}^2}{\hat{p}^2}\,\text{vâr}(\hat{p})\,.$

Approximate confidence intervals for the estimated detection probability ($\hat{p}$) and adjusted weir count ($\hat{\tau}$) were computed by assuming that the sampling distributions of the estimates were normally distributed (Thompson 2002; Zar 1999). Thus, the 95% confidence intervals were computed as the estimate plus or minus 1.96 times the standard error (SE), where the SE is simply the square-root of the variance estimate (equation 6 or 9).

Results

Missing counts

During the 2003 Chinook spawner migration, there were 10 days with partial missing counts and 7 days with complete missing counts (all 24 hours missing) (Table 1). Across days with partial counts, a total of 179 adult Chinook salmon were observed (counted by Vaki) passing the weir, whereas 204 adults (SE = 29.7) were estimated to have been missed. For complete missing counts, the estimate for missed adults was 271 (SE = 114.7) across days (Table 1). Thus, the total estimate for missing counts across all 17 days was 475 (SE = 118.5), with a 95% confidence interval ranging from 244 to 706 adults.

Table 1. Estimates of missing counts for days with partial missing counts and complete missing counts (all 24 hours) during the 2003 Chinook spawner migration. SE = standard error.

Date	Cause	Partial Missing Counts			Complete Missing Counts	
		Observed	**Estimate**	**SE**	**Estimate**	**SE**
5-Oct	Computer error	4	23	11.2		
6-Oct	Computer error	25	23	11.2		
16-Nov	Power failure	19	26	15.0		
17-Nov	Power failure	22	33	13.3		
25-Nov	Panels removed	15	28	8.6		
26-Nov	Panels removed				41	34.2
27-Nov	Panels removed				41	47.9
28-Nov	Panels removed				41	67.6
29-Nov	Panels removed				41	49.3
30-Nov	Panels removed				41	36.2
1-Dec	Panels removed	26	19	4.8		
3-Dec	Panels removed	29	20	7.4		
4-Dec	Panels removed				37	29.7
5-Dec	Panels removed	22	14	6.1		
8-Dec	Panels removed	3	10	4.3		
9-Dec	Panels removed				29	22.2
10-Dec	Panels removed	14	10	4.1		
Total		**179**	**204**	**29.7**	**271**	**114.7**

Vaki detection probability and adjusted weir counts

Of the 76 independent trapping periods conducted in 2004, 41 had zero counts for both and Vaki and trap, leaving 35 periods with counts of adult Chinook required to estimate the accuracy of the Vaki system (Table 2). Total counts varied considerably across trapping periods, ranging from 0-34 fish for Vaki counts and 0-31 fish for trap counts (Table 2). The duration of each trapping period ranged from approximately 1.75 to 15.5 hours. Turbidity levels ranged from about 0.6 to 33.2 NTU (mean = 3.3), but remained low across most trapping periods before spiking in January (Table 2).

The total number of fish counted by the Vaki system was 198, six fewer than the trap count of 204 (Table 2). Overall, there were 11 out of 35 trapping periods in which the Vaki counted fewer Chinook than found in the trap, and 7 periods for which the Vaki count was higher than the trap count. Thus, on average, the Vaki system tended to slightly undercount the true number of fish passing the weir. Absolute count errors, defined as the Vaki count minus the trap count, did not appear to be related to either trapping duration (Figure 2) or the total number of

fish captured (Figure 3). Due to low variability in turbidity across trapping periods, it was not possible to rigorously assess whether turbidity may influence the accuracy of the Vaki system.

The Vaki system was quite accurate and reasonably precise as indicated by the high estimated detection probability (0.97) and relatively narrow 95% confidence interval (i.e., 0.89 to 1.05) (Table 3). Although the Vaki system tended to slightly undercount the number of fish captured in the trap, the potential for occasional overestimates was reflected in the confidence interval for the detection probability, which included values greater than 1.

The estimated detection probability and variance developed from the 35 trapping periods in 2004 were used to estimate the total migrations of adult Chinook salmon passing the weir during 2003-2005 (Table 4). Estimates of total migration were 5,009 in 2003, 4,580 in 2004, and 4,258 in 2005. These estimates were quite precise, with standard errors of roughly 200 and coefficients of variation (*CV*, a relative measure of precision) less than 5% (Table 4).

Table 2. Summary of paired Vaki and trap count data from 35 different trapping periods in 2004.

Trapping Period	Start date	Elapsed time (hrs)	Turbidity range (NTU)	Vaki count	Trap count	Error (Vaki – trap)
1	2-Oct	13.8	1.27-0.8	2	3	-1
2	1-Nov	10.1	2.6-5.3	5	5	0
3	2-Nov	8.9	1.7-3.4	5	5	0
4	8-Nov	4.2	1.1-0.8	6	5	1
5	8-Nov	5.9	0.8	4	4	0
6	9-Nov	2.5	1.0	7	7	0
7	9-Nov	12.3	1.0-1.0	10	11	-1
8	9-Nov	9.4	1.0-1.2	19	21	-2
9	12-Nov	4.8	2.1-2.2	26	25	1
10	12-Nov	5.3	2.2-2.4	4	4	0
11	12-Nov	9.3	2.4-2.1	3	0	3
12	13-Nov	7.5	2.1-1.5	10	11	-1
13	13-Nov	7.2	1.5-1.9	5	5	0
14	13-Nov	9.8	1.9-1.2	9	9	0
15	16-Nov	12.3	1.3-1.6	5	5	0
16	16-Nov	7.8	1.6-1.5	2	3	-1
17	17-Nov	14.5	1.5-1.0	34	31	3
18	18-Nov	9.3	1.0-2.07	12	12	0
19	20-Nov	12.2	1.0-1.3	3	3	0
20	20-Nov	8.8	1.3-1.2	8	10	-2
21	21-Nov	1.8	1.20	1	0	1
22	30-Nov	11.0	0.89-0.6	3	5	-2
23	30-Nov	10.6	0.6-0.73	1	3	-2
24	1-Dec	11.8	0.73-0.89	2	4	-2
25	1-Dec	10.6	0.89-0.78	1	1	0
26	4-Dec	12.1	0.87-0.86	3	2	1
27	5-Dec	13.5	0.64-0.84	1	1	0
28	8-Dec	12.5	2.47-1.45	1	1	0
29	8-Dec	12.0	1.45-1.29	0	2	-2
30	9-Dec	11.3	1.29-1.15	2	0	2
31	9-Dec	12.8	1.15-1.48	0	2	-2
32	3-Jan	8.8	17-15.9	1	1	0
33	4-Jan	6.2	8.90	1	1	0
34	4-Jan	15.5	8.79-14.6	1	1	0
35	13-Jan	3.8	33.20	1	1	0
Total				**198**	**204**	**-6**
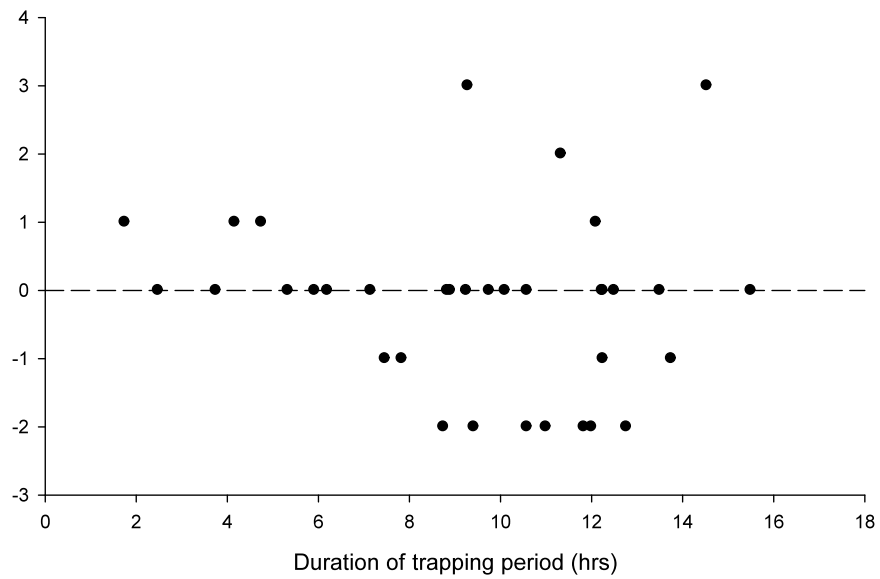
Figure 2. Relationship between absolute count error (Vaki count – trap count) and duration of the trapping period from 35 independent trapping periods in 2004.
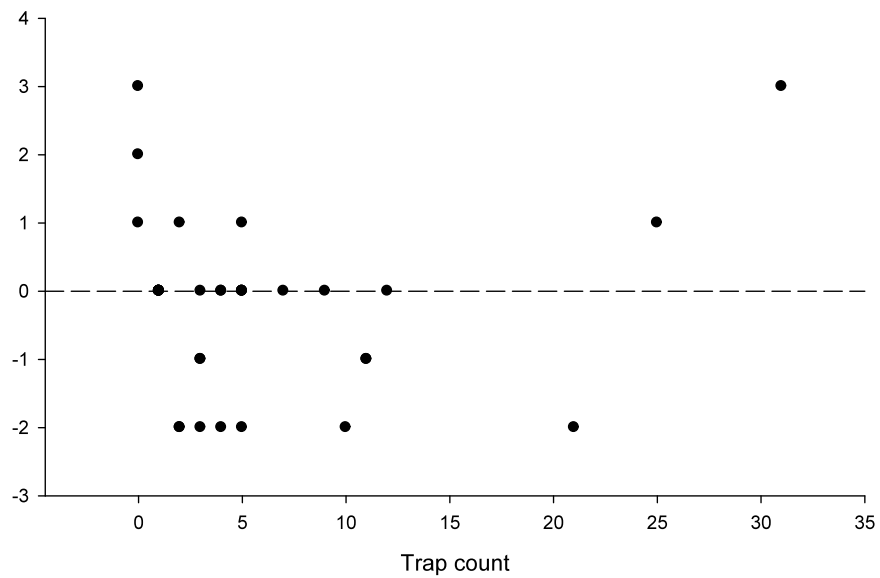
Figure 3. Relationship between absolute count error (Vaki count – trap count) and total trap count from 35 independent trapping periods in 2004.

Table 3. Estimates of the detection probability and the adjusted Vaki count across the 35 trapping periods in 2004.

Parameter	Estimate	Standard Error	95% Confidence Interval
Detection probability	0.97	0.04	[0.89, 1.05]
Adjusted Vaki count	204	8	[187, 221]

Table 4. Estimates of total Chinook salmon migrations passing the weir in 2003-2005 based on adjusted Vaki counts using the estimated detection probability from 2004 trap data. The estimate of missing counts and associated standard error are shown in parentheses for 2003. CV = coefficient of variation (100% * standard error / estimate).

Year	Unadjusted Vaki count	Population estimate (adjusted Vaki count)	Standard error	95% Confidence interval	CV (%)
2003	4862 (475)	5009	233 (119)	[4543, 5474]	4.6%
2004	4445	4580	183	[4208, 4952]	4.0%
2005	4133	4258	170	[3912, 4604]	4.0%

FACTORS AFFECTING ADULT MIGRATION

The weir data provide an excellent opportunity to assess potential factors affecting migration timing of Chinook spawners in the Stanislaus River. Because the specific time and frequency of passage events are accurately recorded by weir counts, relationships between migration timing and environmental conditions can be studied at discrete time scales (e.g., days or even hours). Such resolution and accuracy would not be possible for migration data collected by periodic visuals surveys or mark-recapture experiments.

Of key interest are the affects on migration of scheduled releases (pulse flows) from Goodwin Dam. These releases, referred to here as "attraction flows," are typically conducted in late October with the objective of increasing flows in San Joaquin River, thereby stimulating migration, reducing straying rates, and enhancing conditions during the middle of the Chinook migration period. Figure 4 depicts the scheduled releases for 2003-2005. Planned flow increases were about 800 cfs in 2003 and 600 cfs in 2004-2005, with peak flows maintained up to 9 days in 2003. In the following sections, we use the weir data to examine the affects of these attraction flows and other conditions on migration timing of Chinook salmon in the Stanislaus River.
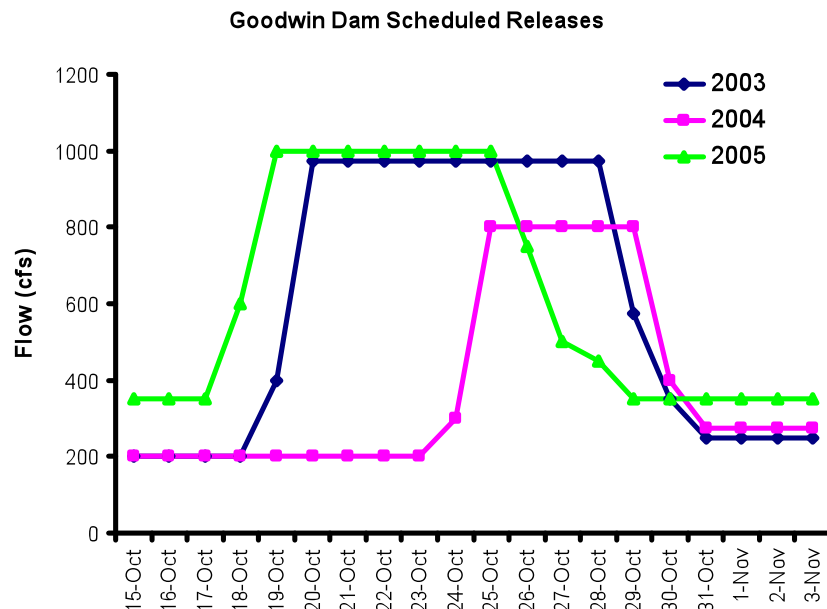
Figure 4. Scheduled releases ("attraction flows") from Goodwin Dam in 2003-2005.

Methods

We used graphical presentations and exploratory statistical models to asses affects of environmental conditions on migration timing. The environmental variables we examined included Stanislaus River flow at Ripon (RM 15.7), San Joaquin River flow at Vernalis (RM 72.3), dissolved oxygen (DO) and temperature in the San Joaquin River measured at Rough and Ready Island (RM 34.9), and turbidity measured at the Stanislaus weir.

To examine short-term migration responses to changes in flow, we used intervention analyses (Box and Tiao 1976). Intervention models are a form of time series model for testing if abrupt changes occur in time series data. Specifically, we used the following model:

(10) $$y_t = \phi y_{t-1} + bx_t + e_t,$$

where y is the weir count on day t, ϕ is the autocorrelation coefficient reflecting similarities in counts from one day to the next, b is the coefficient for the "dummy" variable x, and e is normally distributed error. In this case, the dummy variable has values of either one or zero. If we wish to test whether a particular count is significantly different from adjacent counts, then $x = 1$ for that day and zero for all other days. As discussed below, this model allowed us to examine if counts significantly increased after pulse-flow events. Alternatively, x could be a continuous variable such as temperature or turbidity.

Results

The relative timing of Chinook spawner migrations passed the weir differed somewhat across years (Figure 5). The cumulative distributions in Figure 5 show that numerous fish migrated earlier in 2003 than in 2004 and 2005. This is also reflected by the migration percentiles shown in Table 5. Five percent of the 2003 migration passed the weir by September 30, a week earlier than in later years. However, the mid-point of migration (50% percentile) was very similar across all years, occurring about November 2.
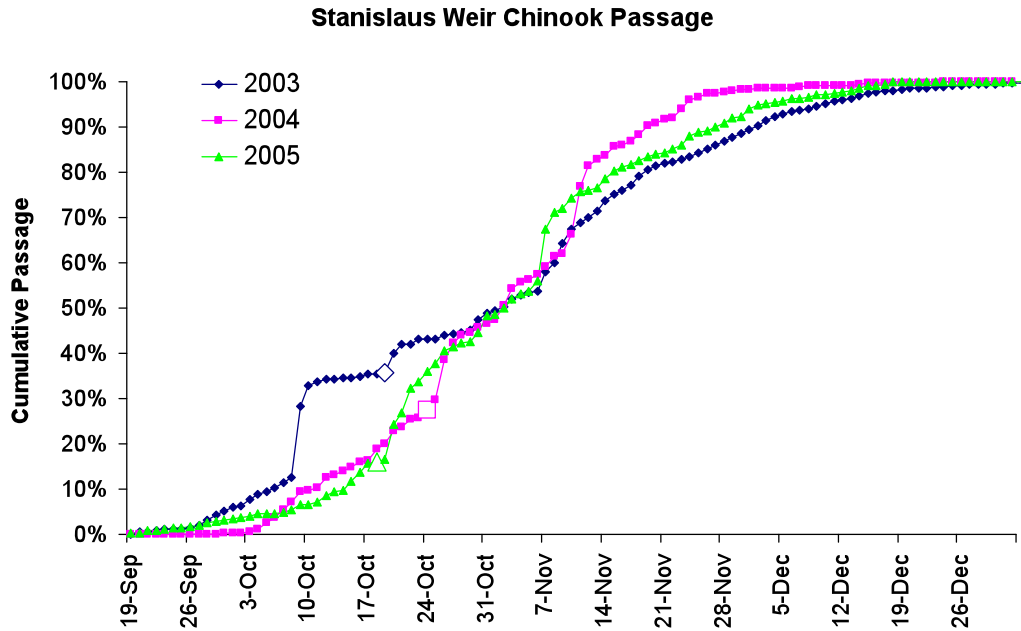
Figure 5. Cumulative passage of Chinook spawners migrating passed the Stanislaus River weir. Large open symbols denote the start date of attraction flows in a given year.

Table 5. Dates for percentiles of cumulative passage of Chinook spawners migrating passed the Stanislaus River weir.

Percentile	2003	2004	2005
5%	30-Sep	7-Oct	8-Oct
50%	2-Nov	2-Nov	3-Nov
95%	10-Dec	24-Nov	3-Dec

Effects of flow

Short-term increases in Chinook passage (2-3 days) occurred immediately at the weir following pulse-flow events in the Stanislaus River. In 2003, flows at Ripon increased roughly 200 cfs during October 6-8, which was followed by a spike in daily weir counts of 764 and 217 fish on October 9 and 10, respectively (Figure 6). A second spike in counts occurred on October 20-21 (307 fish) following a roughly 250 cfs increase in flow that began on October 17. This event appears to have just preceded the start of Goodwin attraction flows (evening of October

19) that arrived at Ripon about 24 hours later. (Note that in all figures, the large open symbols denote the start day for Goodwin attraction flows).

Similar short-term migration responses to flow were observed in 2004 (Figure 7) and 2005 (Figure 8). In 2004, there was an immediate spike in weir counts associated with Goodwin attraction flows. Attraction flows begin on October 24 and Ripon flows increased by 300 cfs a day later. Weir counts then spiked on October 25 and 26 with 397 and 162 fish passing those days. Likewise, the start of attraction flows on October 18 in 2005 was followed by three days of high weir counts from October 20-22 (569 fish in total).

We used intervention models to further explore these short-term responses to flow. In these models, we used 6-hr intervals of total weir counts and average flows at Ripon to more precisely quantify the timing and duration of the migration responses. For the two events noted above in 2003, weir counts began to increase dramatically roughly 66 and 78 hours after Ripon flows had increased by about 150 cfs. The response to the first event is shown in Figure 9, which shows statistically significant increases in weir counts lasting for roughly 36 hours. In contrast, the responses 2004 and 2005 occurred about 48 hours after an initial increase in flow of 150 cfs at Ripon. The apparent during of these events varied from roughly 24 hours to 3 days.

In addition to immediate pulse-flow responses, there was another consistent pattern in the weir counts that may be related to the Goodwin releases and subsequent increases in San Joaquin flow. In all years, there was a 3 to 4 day pulse in passage in early November (Figure 6 - Figure 8) that occurred roughly three weeks after flow increases in San Joaquin River (Figure 10). These events are summarized in Table 6. Though speculative, these large migrations may have started low in the San Joaquin River (perhaps a 100 miles downstream of the weir) in response to flow increases. It appears that Goodwin attraction flows contributed to the large, prolonged increases in San Joaquin flow in 2003 and 2005; large flow increases had already occurred in 2004 (Figure 10).

Table 6. Summary of distinct migration events occurring in early November of each year.

Year	Migration Event	Weir count	Percent of total migration	Days after Vernalis flow increase
2003	Nov. 7-10	675	14%	20
2004	Nov. 10-12	860	19%	21
2004	Nov. 6-8	720	17%	23

We examined these migration pulses using intervention models where the dummy variables represented a 3-day or 4-day migration response. As observed in Figure 11 for 2003, four-day counts were significantly greater for adjacent periods roughly three weeks after an initial 300 cfs increase in San Joaquin flow. Similar results were found in 2004 and 2005 for 3-day response models. These analyses are only exploratory in nature, but the consistency of patterns across years suggests that rising flows may stimulate pulse migrations of fish holding in the lower San Joaquin River.

In addition, flows in the San Joaquin River during September and October were much lower in 2003 and 2004 than in 2005 (Figure 10). If low flows were an important deterrent to migration, we might expect delayed migrations in 2003 and 2004. This was not evident in the weir data. In contrast, the initial migration was highest in 2003, whereas 2004 and 2005 showed similar cumulative patterns during the early migration period (Figure 5).
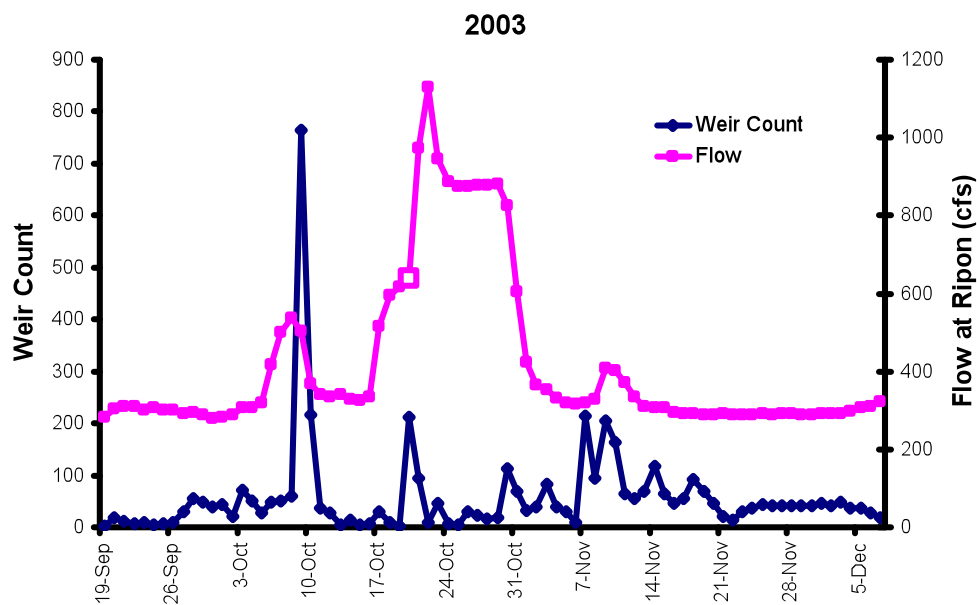
Figure 6. Weir counts of Chinook spawners and flow at Ripon for 2003. The large open symbol denotes the start date of attraction flows.

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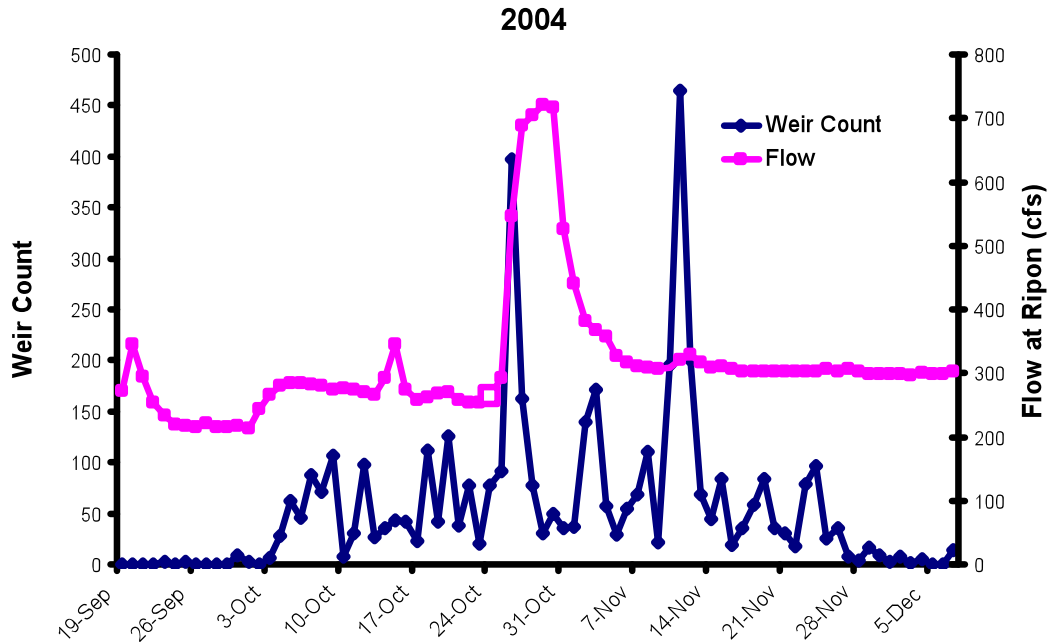
Figure 7. Weir counts of Chinook spawners and flow at Ripon for 2004. The large open symbol denotes the start date of attraction flows.
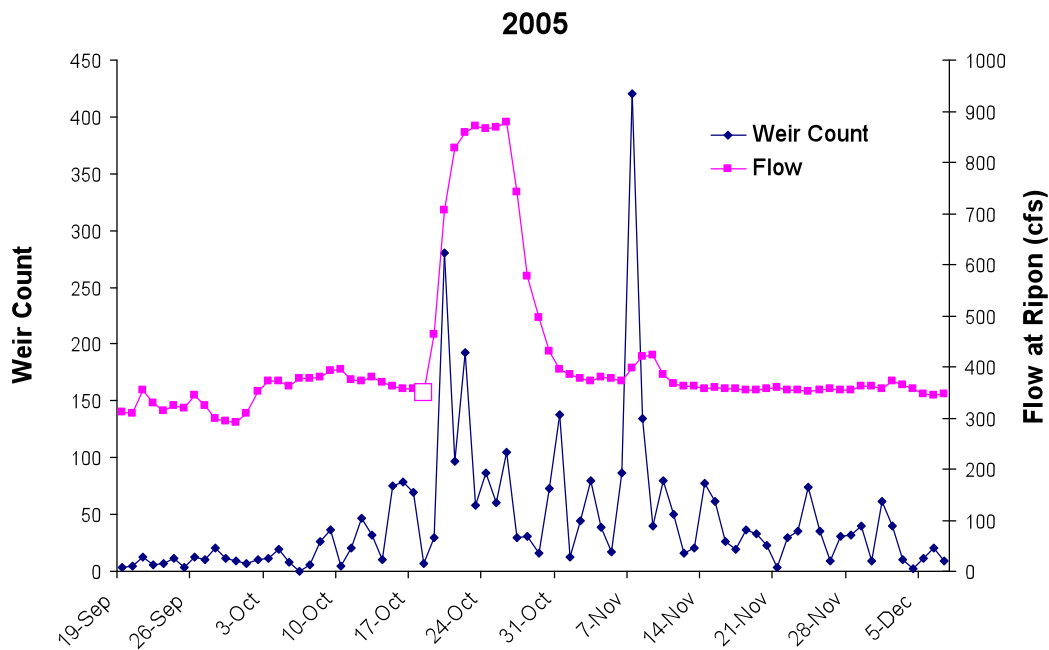
Figure 8. Weir counts of Chinook spawners and flow at Ripon for 2005. The large open symbol denotes the start date of attraction flows.

Response to Ripon Flow Increase
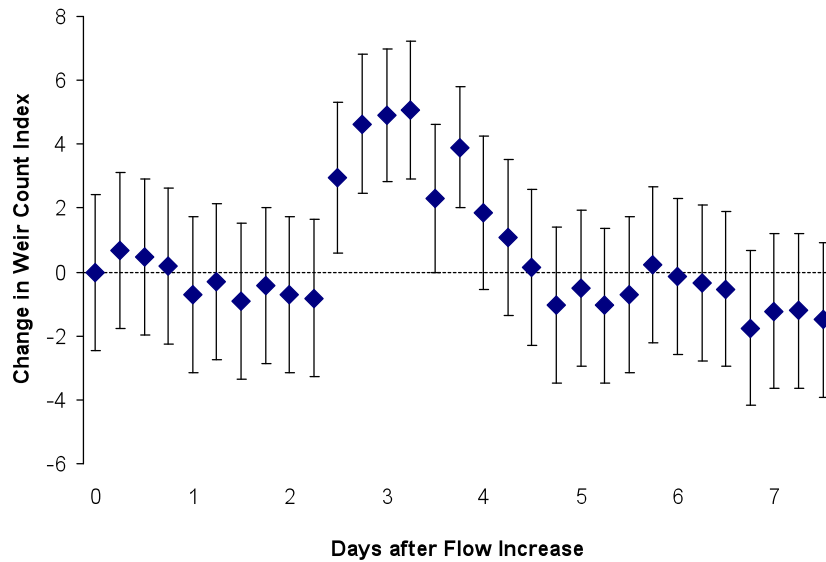
Figure 9. Intervention estimates of changes in square-root weir counts (6-hr intervals) following the pulse-flow event on October 7, 2003. Error bars denote 95% confidence intervals.
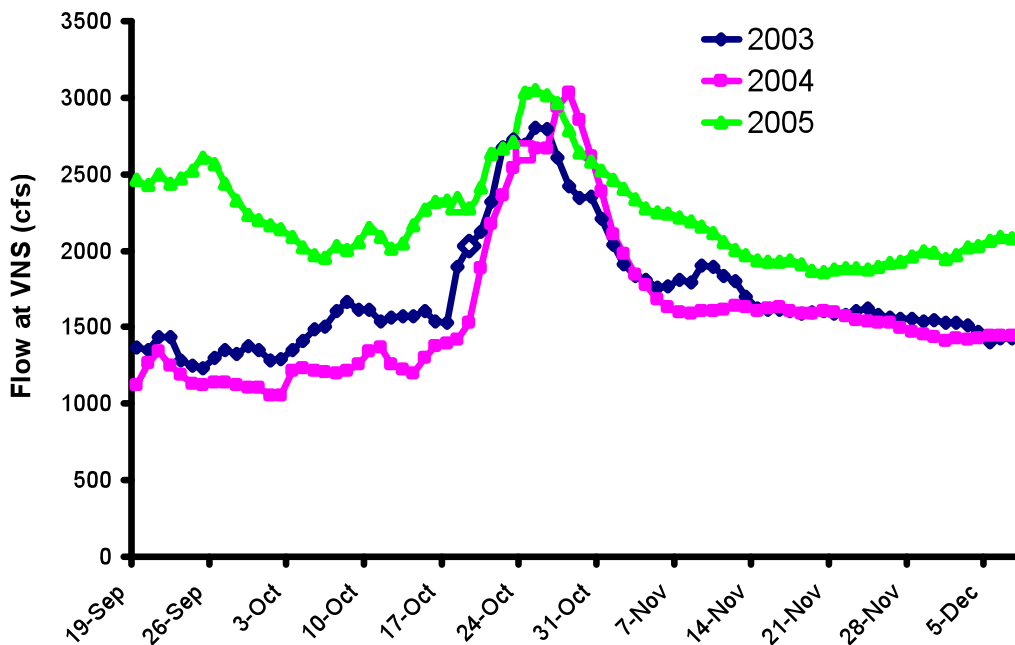
Figure 10. San Joaquin River flows at measured at Vernalis (VNS). Large open symbols denote the start date of attraction flows in a given year.

Response to Vernalis Flow Increase
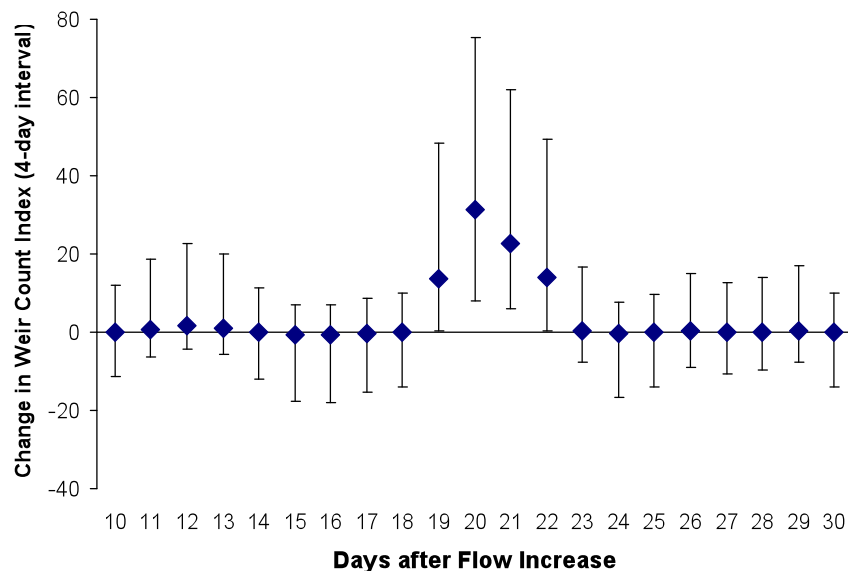
Figure 11. Intervention estimates of changes in square-root weir counts (4-day intervals) following an abrupt increase in San Joaquin flows in 2003. Error bars denote 95% confidence intervals.

Effects of other variables

There were no obvious associations between migration timing and conditions for temperature, dissolved oxygen (DO), or turbidity. Migrating salmon can be adversely affected by high water temperatures (e.g., > 70 F) or low levels of dissolved oxygen (e.g., < 5 mg/L), and such conditions could delay migration (Hallock et al. 1970). Temperatures at Rough and Ready Island (RRI) in the lower San Joaquin River typically remained above 70 F until early October, and then declined steadily thereafter (Figure 12). Interestingly, although temperatures were the highest in 2003, a larger fraction of fish migrated during September and early October in 2003 than in the later years (Figure 5).

Dissolved oxygen levels at RRI were especially low in 2004, and did not consistently exceed 5 mg/L until late October (Figure 13). However, in comparison to 2005, migration timing in 2004 did not appear to be significantly delayed despite potentially low DO levels.

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Turbidity levels at the weir remained low throughout the primary migration season in all years (Figure 14). Spikes in turbidity approaching 6 NTU were associated with the onset of Goodwin attraction flows in 2003 and 2004. In general, however, there were no obvious associations between trends in weir counts or migration timing and turbidity levels.

To confirm that temperature, DO, and turbidity conditions were not strongly related to within-year patterns in migration timing, we used each variable as the independent variable (x) in the time series model (equation 10). For all variables and years, no significant associations were found (all $P > 0.6$).
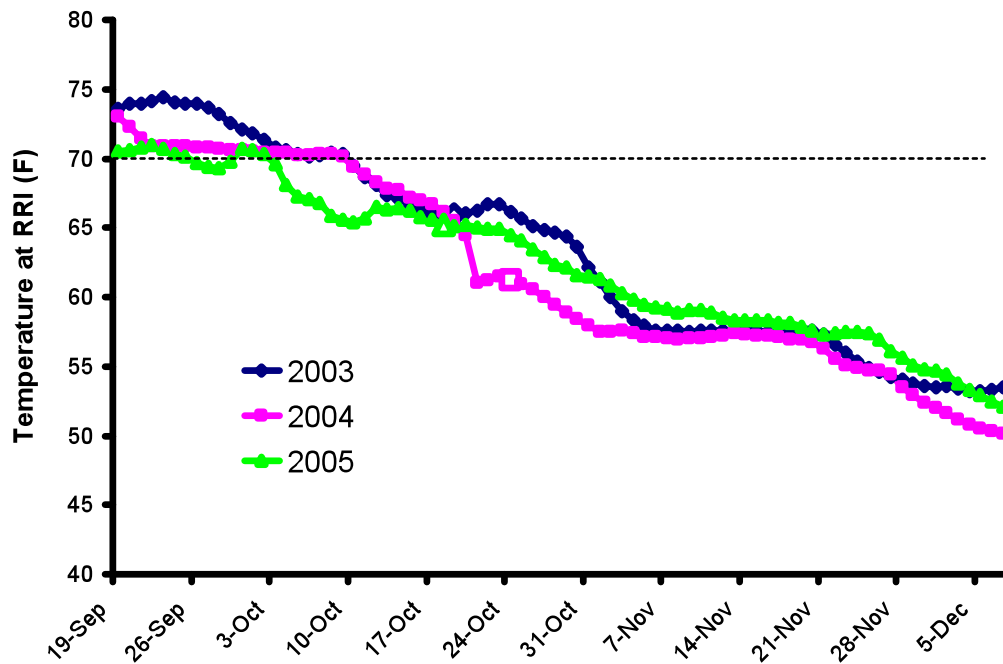
Figure 12. San Joaquin River temperatures measured at Rough and Ready Island (RRI).
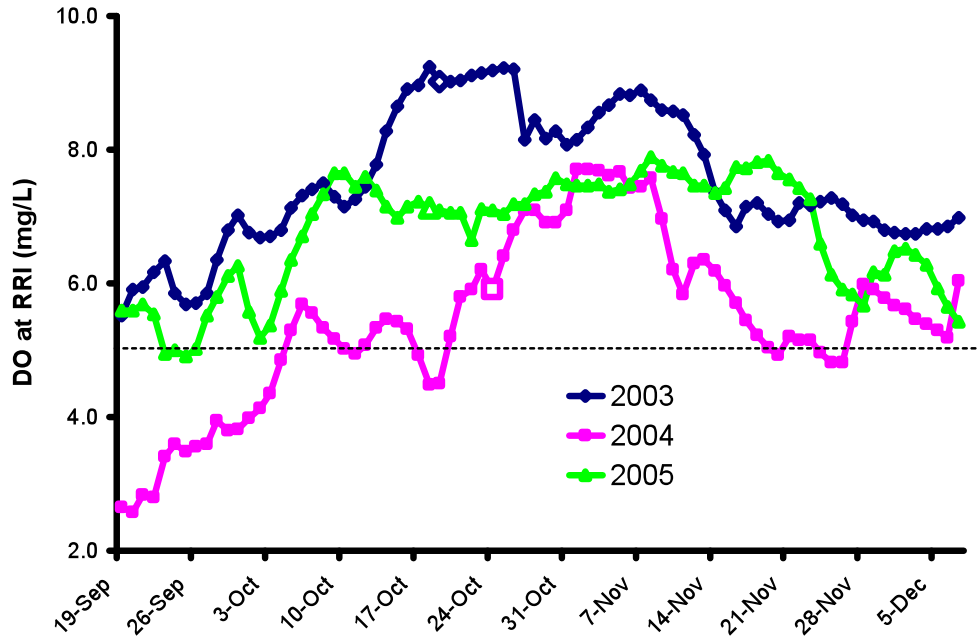
Figure 13. San Joaquin River dissolved oxygen measured at Rough and Ready Island (RRI). Large open symbols denote the start date of attraction flows in a given year.
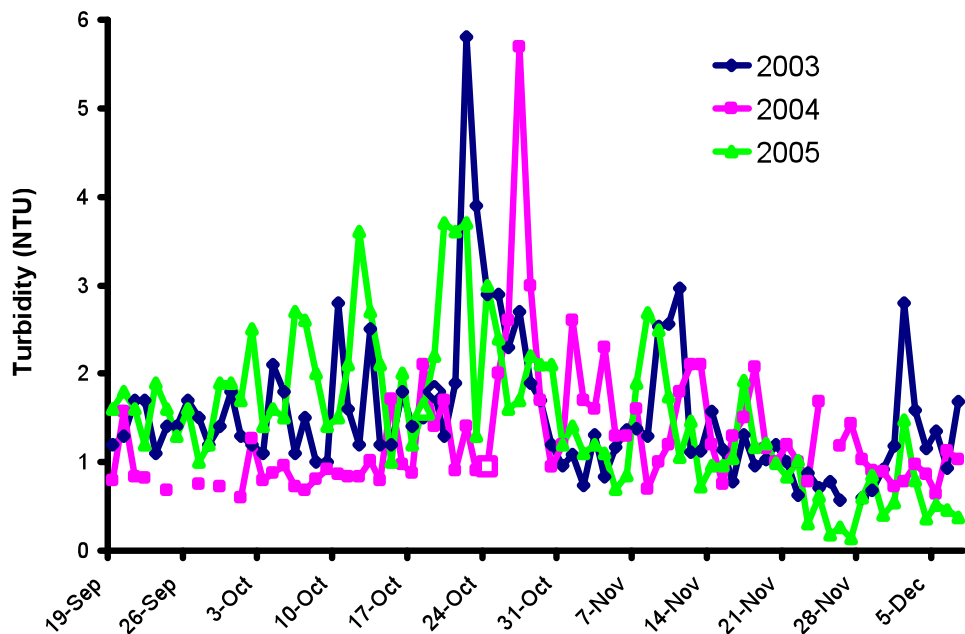
Figure 14. Stanislaus River turbidity measured at the weir. Large open symbols denote the start date of attraction flows in a given year.

Summary

Adult salmon migration can be stimulated by short duration (2 or 3 days), modest increases in flow (e.g., 200 cfs). There was no evidence of a prolonged increase in migration passing the weir during, or soon after, multi-day attraction flows. The rapidity of the responses suggests that the migrants were likely already holding within the lower Stanislaus River prior to when the pulse flows occurred.

It is unclear what effect the Goodwin attraction flows have on the migration conditions, straying rates, and other behaviors of fish holding in the San Joaquin River. It is possible that pulse flows in the San Joaquin can stimulate migration of fish holding in the lower river, based on tentative evidence indicating brief 3-4 day passage pulses at the weir some three weeks after San Joaquin flow increases. However, these pulses occurred during the latter half of the adult migration period, suggesting that dramatic flow increases were not necessary for successful migration of the initial 50% of spawners. In addition, there was no evidence that low flows (1,000 to 1,500 cfs) in the San Joaquin in 2003 and 2004 were an impediment to migration.

We did not observe any obvious association between characteristics of migration timing and measures of temperature, dissolved oxygen, or turbidity.

ASSESSMENT OF CDFG ESCAPEMENT SURVEYS

Dating back to 1953, annual escapement surveys of San Joaquin River fall-run chinook salmon have been conducted by the California Department of Fish and Game (CDFG) on the Stanislaus, Merced, and Tuolumne rivers. The escapement survey objectives have been to "determine the timing, number, size and age distribution of adults returning to the Stanislaus, Tuolumne and Merced Rivers" (CDF&G 1989). In the early 1990's the scope of the escapement surveys expanded to the following (Guignard 2005):

- Estimate the annual escapement of in-river fall-run Chinook spawning populations
- Evaluate the distribution of salmon redds throughout the study area.
- Collect fork-length and sex data.
- Collect scale and otolith samples for age determination and cohort analysis.
- Collect DNA samples for genetic analysis.
- Collect and analyze coded wire tag data from marked hatchery fish to determine escapement contribution of hatchery produced salmon, and evaluate smolt survival.

When scoping this study, we set forth to collect and review historic carcass survey data and escapement estimates with the following three objectives: (1) to assess the reliability of the historic record, (2) to explore possible patterns in spawner distribution, and (3) to examine possible evidence of spawner condition (egg retention, pre-spawn mortality). The results of the first two objectives are presented in the following sections; however, data were not available to assess spawning condition.

Survey Area

Escapement surveys in the Stanislaus River have typically covered a 25-mile reach below Goodwin Dam (RM) 58 and continuing downstream to Riverbank (RM 33) (Figure 15). The study area was divided into 4 stream sections, with Section 1 being the most upstream reach. Section 1 begins just below Goodwin Dam (RM 58.3) and continues downstream to Knight's Ferry (RM 54.5). Section 2 continues downstream from Knights Ferry (RM 54.5) to Horseshoe Road Recreation Area (RM 50.5). Section 3 extends from Horseshoe Road Recreation Area downstream to Oakdale Recreation Area (RM 39.1). Section 4 extends from Oakdale Recreation Area downstream to Jacob Myers Park (RM 33.9). Riffles within each stream section were

uniquely identified using a sequential alpha-numeric code, with letters designating river miles and numbers indicating the sequential order of riffles from upstream to downstream.
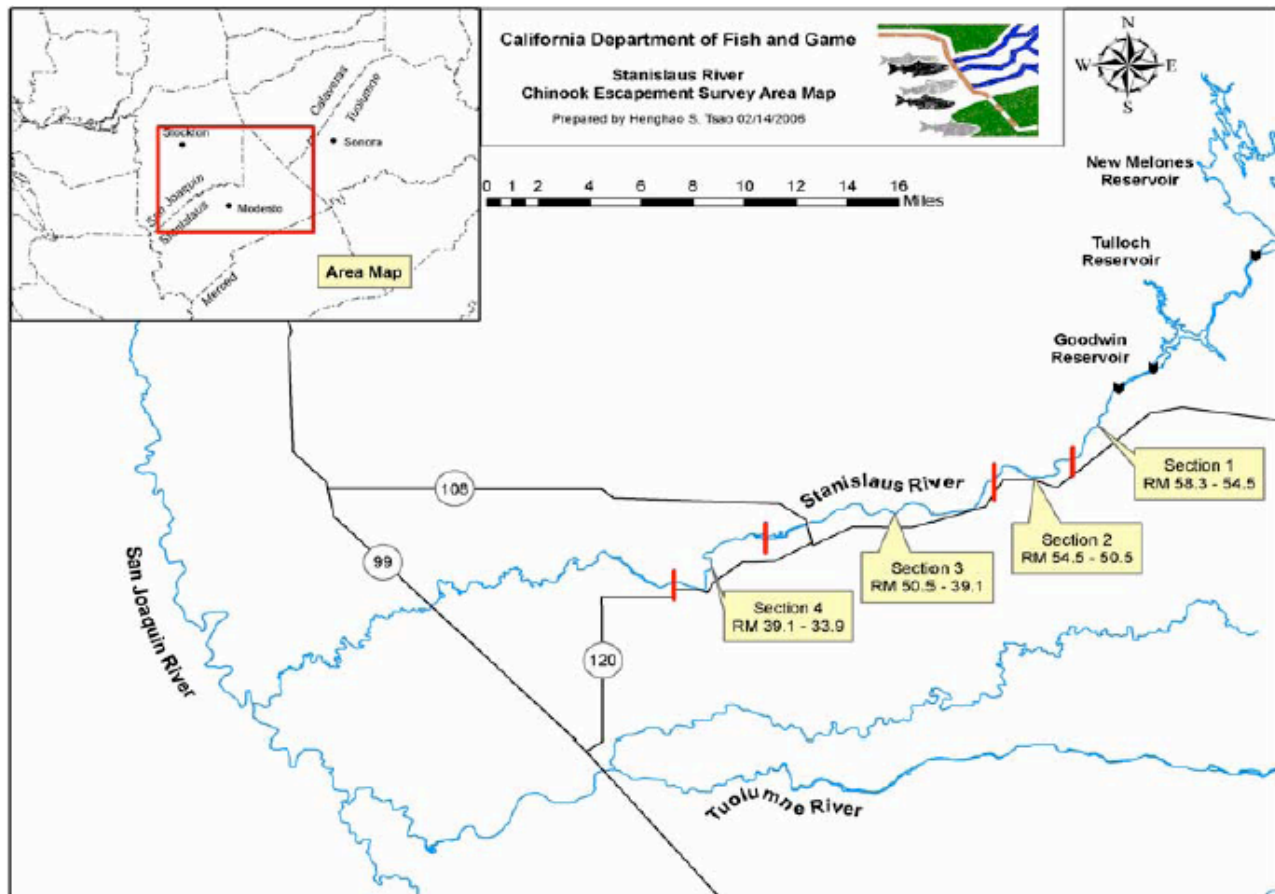
Figure 15. Site map indicating locations of the four different survey sections on the Stanislaus River (From Guignard 2006).

Reliability of Historic Surveys

<u>Methods</u>

For the purpose of this study, we reviewed and compiled annual CDFG escapement reports dating back to 1987. Schaefer (1951), Jolly-Seber (Jolly 1965, Seber 1965 & 1982) and Peterson (Chapman 1951; Ricker 1975) mark-recapture methods were employed in the annual CDFG surveys to estimate fall-run Chinook salmon escapement in the Stanislaus River. Schaefer estimates were often adjusted according to a modified Schaefer equation (Hoopaugh 1978), where marked carcasses from the second marking period onward were subtracted from the total estimate.

The specific method employed varied from year to year depending on environmental conditions (i.e., flow) and evolving analytical strategies. For example, the Schaefer mark-recapture escapement estimation model was employed since 1971; however, CDFG later included the Jolly-Seber model (Jolly 1965, Seber 1965 & 1982), which may provide more accurate population estimates under various conditions (Law 1994). The potential increase in accuracy arises from adjustments for immigration and emigration through the survey period. Further, CDFG also analyzed the Stanislaus River escapement data with POPAN-5 (Arnason et al., 1998) from 2002 through 2005. POPAN-5 is a custom software program developed for the analysis of mark-recapture data. Lastly we also obtained official CDFG Grand Tabs in-river escapement data. Unfortunately, the specific methods for arriving at escapement estimates are not provided for Grand-Tabs data. These methods appear to vary on a year-to-year basis, though in most cases the Schaefer escapement estimates appear to be a strong consideration in the final Grand-Tabs value.

In addition, we calculated Peterson escapement estimates for years in which CDFG did not report Peterson estimates but provided the requisite data for estimation (1997-2005). In contrast to the Schaefer estimator, which generates population estimates for each week across multiple weeks (i.e., multiple samples or discrete time periods), the Peterson estimator of population abundance (N) was developed for a simple two-sample experiment:

$$\hat{N} = \frac{c \times r}{m} \quad ,$$

where c is the number of carcasses tagged in the first sample; r is the number of carcasses examined in the second sample; and m is the number of tagged carcasses recovered in the second sample. For the escapement survey data examined here, the number of tagged, examined, and recovered carcasses represent pooled (summed) values across all weeks of the survey. The coefficient of variation (CV, a relative measure of precision) of the Petersen estimate can be expressed as:

$$CV = \frac{SE(\hat{N})}{\hat{N}} \approx \frac{1}{\sqrt{m}} \quad .$$

Thus, the approximate standard error (SE) for the Petersen estimate ($\hat{N}$) can be computed as:

$$SE(\hat{N}) \approx \frac{\hat{N}}{\sqrt{m}} \; .$$

Using this standard error, we estimated confidence intervals for the Petersen estimate assuming a normal sampling distribution, which provides a reasonable approximation especially when m is large (e.g., > 50).

In recent years, CDFG calculated confidence intervals around the POPAN Jolly Seber estimates to reflect the precision (i.e., the level of reliability or uncertainty) of population estimates. However, estimates of precision were not provided for Schaefer estimates in any year. To address this data gap, we generated surrogate confidence intervals for Schaefer estimates dating back to 1997 by assuming that the coefficient of variation (CV) of the Schaefer estimate was equal to the CV of the Peterson estimate. Although the Schaefer estimator is more complex in that it accounts for potential heterogeneity in capture or recovery probabilities, the CV of the Schaefer and Peterson estimators have been shown to be quite similar across a range of conditions (unpublished research by Carl Schwartz). We therefore calculated the Peterson CV as above and used it to derive confidence intervals for both the Peterson and Schaefer escapement estimates.

There was a key exception to our approach for computing surrogate confidence intervals for Schaefer escapement estimates. As discussed below in the section "Comparison of weir and survey estimates," a low recovery of tagged carcasses in the final week for 2005 resulted in an exceptionally large and uncertain Schaefer estimate for that week. In this case, using the CV of the Petersen estimate as a proxy for the CV of the Schaefer estimate would be very misleading. This is because the Petersen estimate, which is based on pooled data across all weeks, ends up being much lower and far more precise than the Schaefer estimate. A much more reasoned approach would be to treat the final week of data, which was independent of all other weeks, as one Petersen experiment, and all previous weeks as a second Petersen experiment. We therefore computed two Petersen estimates for 2005, and then summed the estimates and their variances to derive an approximate CV for the Schaefer estimate in 2005.

<u>Results</u>

In several years, there was a strong lack of consistency between the various escapement estimates (Table 7). For example, while we expect differences between methods, the Schaefer estimates in 1990 and 1991 are well outside the 95% confidence intervals of the Peterson estimates (Table 7). This was likely due to small sample sizes. Starting in 1995, the two estimators were more closely aligned and the confidence interval around each Peterson estimate contained the annual Schaefer escapement estimate. Another importance difference occurred in 2005. In this year, the Schaefer estimate (3,050) was considerably greater than the two reported Jolly Seber estimates (1,025 and 950), and well above the confidence interval reported for the POPAN Jolly Seber estimate (Table 7). Another notable discrepancy involves the official CDFG Grand Tab estimates. These estimates do not match the published Schaefer or Peterson estimates, though they bare a strong resemblance to the Schaefer estimates in most years (Table 7).

We found several inconsistencies in the reported annual escapement estimates that could not be reconciled. For example, the reported Schaefer estimates in the annual escapement reports for 1994 (1,424) and 1998 (2,699) did not match with summary escapement tables provided in recent years, which reported values of 1,079 and 2,150, respectively. It was not clear what estimates were used in the summary escapement tables for those years. In 1988, the annual report refers to adjusted Peterson estimates, but no Peterson-based results were presented. The report does however provide Schaefer and Jolly Seber estimates of escapement based on weekly recoveries from October through December. The report addressed escapement estimates as follows: "A preliminary sensitivity analysis was performed by DFG's Technical Services Branch on 1988 Stanislaus River data. This analysis indicates that Jolly-Seber may have underestimated escapement by 30%" (CDFG 1989).

Annual reporting methods and formats evolved over the years, with distinct improvements since 2001. In recent years, weekly mark-recapture data were published, whereas in the 1980s and 90s only the final estimates were provided. Schaefer estimates were calculated and reported in most years. Peterson estimates were also reported in the early to mid 1990s, often serving as the only escapement estimate during years of low returns (1992, 1993, and 1996) (Table 7). In some years, high flows hampered sampling methods and the calculation of escapement. For example, in 1996, escapement surveys were prematurely terminated due to high flows, and the escapement estimate was calculated using a Peterson method based on the mean

carcass recovery rate computed across the previous six years of escapement surveys. Starting in 1997, the reports no longer calculated the Peterson escapements estimate, but did provide the necessary data so that we could generate those estimates and their precision.

Table 7: Historical escapement estimates of Stanislaus fall-run Chinook salmon reported in CDFG annual reports and Grand Tabs, and estimates of Petersen confidence intervals and coefficient of variation (% CV = 100% * standard error / estimate).

Year	Schaefer	Jolly Seber	POPAN Jolly Seber	Petersen[a]	Peterson CV (%)	Grand Tabs
1987	6,400					6,292
1988	12,344	7,676				10,212
1989	1,543	932				1,510
1990	492	447		96 ± 22	12.0	480
1991	321	193		77 ± 27	18.0	394
1992				254 ± 98	19.6	255
1993				410 ± 232	28.9	677
1994	1,424	525		1079 ± 214	10.1	1,031
1995	622	192		611 ± 211	17.4	619
1996				160[c]		168
1997	5,583			6,466 ± 2206	17.4	5,588
1998	2,699			2,807 ± 1230	22.4	3,087
1999	4,034			3,473 ± 491	7.2	4,349
2000	8,493			11,265 ± 2678	12.1	8,498
2001	9,181			9,458 ± 1067	5.8	7,033
2002	6,960	5,533	5,710 ± 626	7,352 ± 622	4.3	7,787
2003	6,980	5,141	5,836 ± 697	7,902 ± 656	4.2	7,596[b]
2004	3,458	2,787	2,813 ± 337	4,181 ± 378	4.6	5,000[b]
2005	3,050	1,025	950 ± 200	1,714 ± 283	8.4	3,500[b]

[a] Petersen escapement estimates prior to 1996 were reported in annual surveys. Estimates after that year were calculated using published data.

[b] The Grand Tabs 2006 report these data as preliminary

[c] Due to high flow conditions in 1996, escapement surveys were prematurely terminated and the escapement estimate was calculated using an adjusted Peterson method with a mean carcass recovery rate from the Stanislaus over the last six survey years.

Precision of estimates

The <u>reliability</u> of the escapement estimates depends on the *accuracy* and *precision* of the estimates. In some respects, precision is easier to assess than accuracy. Regardless of the type of mark-recapture estimate that is used (Peterson, Schaefer, or Jolly Seber), the precision of survey estimates will depend largely on the number of carcasses that are tagged and subsequently recovered (Seber 1982). When more tagged carcasses are recovered, precision increases. As observed in Table 7, the coefficients of variation (*CV*) of Petersen estimates ranged from a low of 4.2% in 2003 to a high of 28.9% in 1993. Although these measures will tend to overstate the level of precision (the Petersen estimator contains fewer assumptions than the other models), they likely provide a reasonable basis for comparing the relative precision of survey estimates across years. In rough terms, a *CV* of 5% implies an approximate 95% confidence interval of plus or minus 10% of the estimate (i.e., plus or minus two times the *CV*). Consequently, a *CV* of 5% is generally considered excellent for escapement estimates, whereas a *CV* of 30% is considered much worse. Mark-recapture estimates of spawner abundance that yield *CVs* of 10% are considered good, while *CV*s of 20% are considered average (Cousens et al. 1982).

Thus, across years, the escapements estimates for Chinook salmon in the Stanislaus River appear reasonably precise. This was especially true from 1999-2004, for which Petersen *CV* was below 10% in most years (Table 7). In recent years, large numbers of carcasses were both tagged and subsequently recovered (Table 8), providing estimates with apparently high precision (Table 7). For example, in each of the 2002-2004 surveys, over 800 carcasses were tagged and over 400 of those were recovered (Table 8). Such high recovery rates are exemplary, and will generally yield escapement estimates that are highly precise.

As noted above, however, the *CV* of the Petersen estimate for 2005 is quite misleading. To account for the large Schaefer estimate in the final week of the 2005 survey, we computed and combined two Petersen estimates for that year as described above. The resulting new Petersen estimate was 3,363 spawners. This estimate was much closer to the reported Schaefer estimate (3,050) for 2005 than the pooled Petersen estimate (1,714) shown in Table 7. More importantly, the *CV* of the new Petersen estimate was 43%, as apposed to just 8.4% for the pooled Petersen estimate (Table 7). In other words, the Schaefer estimate for 2005 appears to be highly uncertain.

Table 8: Escapement survey results for the number of fish tagged, the number counted (handled), the number of tagged fish recovered, and the tag recovery rate for 1990-2005.

year	# tagged	# counted	# recovered	recovery rates
1990	238		70	29%
1991	94		31	33%
1992	75		26	35%
1993	88		12	14%
1994	269		98	36%
1995	139		33	24%
1996	48		*	NA
1997	227	940	33	15%
1998	116	484	20	17%
1999	609	1,095	192	32%
2000	587	1,305	68	12%
2001	744	3,839	302	41%
2002	841	4,686	536	64%
2003	1,070	4,121	558	52%
2004	888	2,208	469	53%
2005	340	711	141	41%

*High flow conditions prematurely ended escapement surveys in 1996. With only 48 tagged Chinook and no recoveries, the escapement estimate for that year was calculated using a Peterson method with a mean carcass recovery rate computed across the previous six annual surveys.

This is demonstrated by the Petersen-based 95% confidence intervals for the annual Schaefer estimates (Figure 16). The 2005 estimate has the broadest confidence interval, followed by the 2000 and 1997 Schaefer estimates. In other years, confidence intervals appear reasonably narrow. Of course, these are only surrogate confidence intervals that likely overstate precision to some extent. Nevertheless, the results of C. Schwartz (unpublished data) and the close match between the Peterson and POPAN Jolly-Seber confidence intervals for 2002-2005 (Table 7) suggest that the Petersen-based confidence intervals for Schaefer estimates provide useful insight into the relative precision of the CDFG survey estimates of escapement (Table 7; Figure 16).
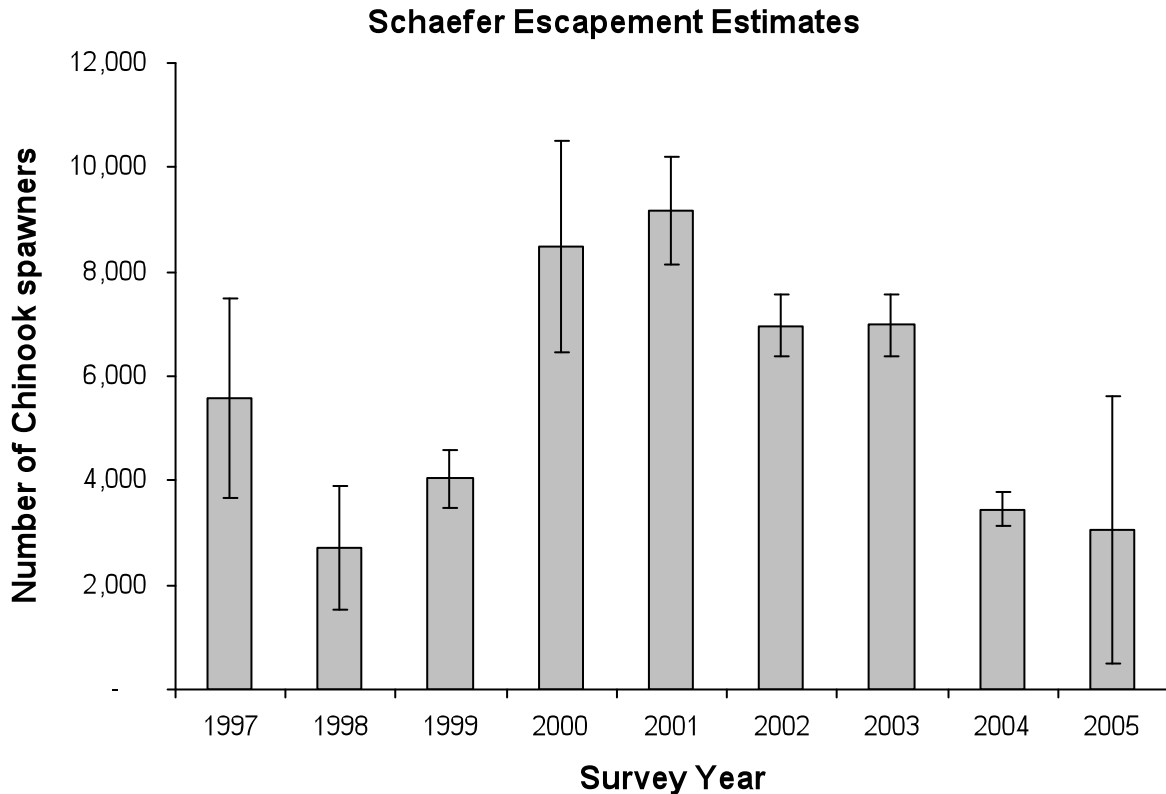
Figure 16: Schaefer estimates of Chinook escapement in the Stanislaus River based on CDFG carcass surveys. Error bars depict approximate 95% confidence intervals based on the coefficient of variation (CV) for Petersen estimates.

Accuracy of estimates

It is difficult to assess the statistical accuracy of a given survey estimate because accuracy depends on factors that may *bias* an estimate, but such factor are typically not studied or reported when escapement surveys are conducted. A given estimate may have high statistical precision, but if it is strongly biased then the estimate will be misleading and will have little value.

An estimate is likely to be biased when an assumption underlying the mark-recapture method has been violated. Many such violations are known to occur in practice (e.g., Seber 1982). In the case of escapement surveys, a critical assumption is that all carcasses have a similar probability of being detected and handled during a given sampling period. It is therefore important that tagged carcasses be randomly mixed among the full population so that detection probabilities are similar across all carcasses. In general, if tagged carcasses have a greater chance of being recovered than untagged carcasses, the population estimate will be biased low.

Similarly, if tagged carcasses have a lower chance of being handled, the population estimate will be biased high. For example, if high flow conditions limit the detection of carcasses to only a subset of spawning reaches, the tagging and recovering of carcasses would be concentrated in those reaches as weeks progress. The final population estimate would be biased low, perhaps quite strongly, because the survey sampled only a subset of the population.

One way to assess potential bias is to compare estimates across different mark-recapture models, each of which has a somewhat different set of assumptions. Across years, Jolly-Seber estimates were always lower than the Schaefer estimates, suggesting that one of the estimators is systematically biased (Table 7). Petersen estimates tended to be higher than Schaefer estimates when population estimates were large (e.g., above 5,000), but considerably lower for small population estimates (e.g., below 500) (Table 7). Unfortunately, this provides little insight into potential biases. Of course, the best way to assess bias is to have an independent measure of the population that is known to be highly accurate. As discussed below, the weir estimates provide such a measure for assessing Schaefer estimates in 2003-2005.

In conclusion, our examination of the historic survey data suggests that in most years the survey estimates appear reasonably precise, though it is difficult to assess potential biases. Surveys are hampered by higher flows, which limit the visual detection and accessibility of carcasses. In such years, estimates will tend to be less precise and may be subject to strong biases that result in severe underestimates. It is also clear that in years of low abundance, there may be insufficient detections of carcasses to provide reliable population estimates. Thus, low population estimates derived from carcass surveys should be viewed cautiously.

Spawning Distribution

Analysis of the geographic distribution of spawners and redds over a series of years can provide valuable insight into the physical processes affecting spawning habitat selection and habitat quality that would not be possible through analysis of spawner abundance alone. Anomalous patterns in spawner distributions in a given year, or changes in distributions across years, may indicate potential problems with respect to habitat quality, fish passage, or other significant alterations to the stream channel morphology.

The following analysis of spawning distribution in the Stanislaus River examined data from CDFG escapement surveys conducted from 2000-2005 (data provided by Jason Guignard, CDFG). The objective was to evaluate the annual distribution of salmon redds and live adult spawners throughout the study area. The spatial resolution was limited to the "stream section" scale as described above (Figure 15), with the four key sections ranging in length from 3.8 to 11.4 miles. As additional escapement surveys are conducted in future years, information about spawner distributions over a smaller spatial scale (i.e. riffle or reach-scale) will provide an opportunity for a more refined analysis and may provide additional information about the quality of spawning habitat within the Stanislaus River.

Methods

Weekly observations of live adult fish and redds in each stream section were conducted throughout the entire spawning season in each year. Counts were made using tally counters as the field crew drifted or walked through riffles and pools, and the riffle location and stream section were noted for each fish or redd encountered. To evaluate the distribution of live fish and redds, we calculated the proportion and density of the total number of live fish and redds encountered in each stream section for each year. This provided a general description of the geographic distribution of spawning activity within the study area and allowed for a graphical analysis of potential trends over time.

Results

Patterns in the longitudinal distribution of live adult Chinook salmon and redds within the Stanislaus River were generally consistent from 2000-2005 (Figure 17 and Figure 18). Section 2 (RM 50.5-54.5) consistently had the largest proportion of live fish and redds compared with the other stream sections (Figure 17). Section 3 (RM 39.1-50.5) had the next highest proportion of live fish and redds, followed by Section 1 (RM 54.5-58.3) and Section 4 (RM 33.9-39.1). Some consistent patterns were also observed in the density of live fish and redds in each stream section (Figure 18). The density of spawners and redds were highest in Section 2, indicating that the high proportion of fish in that section was likely related to favorable physical habitat features and not to differences in section length. Densities of both live fish and redds were lowest in Section 4. However, densities were second highest in Section 1, a section which consistently had the second lowest proportion of overall live fish and redds. This discrepancy between the density and proportion data indicates that the low proportions of live fish and redds observed in Section

1, and similarly, the high proportions observed in Section 3, are likely related to differences in section length. Section 1, having the smallest continuous stream length, would be expected to have the smallest proportion of spawners if spawning conditions (e.g., suitable habitat, water temperatures, etc.) were more or less constant across sections. However, relatively high densities in this section indicate more preferable spawning conditions relative to Sections 3 and 4.

The data suggest a subtle trend in the distribution of adult spawners in which the proportion and density of live fish encountered in Section 2 gradually declined in recent years, coinciding with an apparent increase in the proportion of live fish in Section 1 (Figure 17 and Figure 18). However, this pattern was not as apparent in the distributions of redds.

Several factors may contribute to differences in the number of live fish and redds observed within each stream section and within each year. Interannual variation in total spawner abundance as well as differences in streamflow and other environmental factors can potentially influence the spawning behavior and site selection of adult Chinook in the Stanislaus River. In addition, redd counts can be affected by time of day, stream visibility, sunlight, redd superimposition, as well as variability between observers. The problem of redd superimposition, which often results in a negative bias in redd counts, is particularly acute in stream Sections 1 and 2, while redds located in sections further downstream are more easily identified and more accurately reflect true spawner density (Guignard 2006). Given few years of data and the large number of potential confounding effects, it is difficult to identify potential factors influencing year-to-year variation in the spatial distribution of spawners. Nevertheless, the spatial data collected by CDFG during carcass surveys may provide valuable insight into future changes in spawner distributions and their potential causes and implications. Additional years of data will increase our ability to identify such trends, and finer-scale analysis examining patterns in spawner distribution across stream reaches may also help to identify critical habitat features or changes therein.
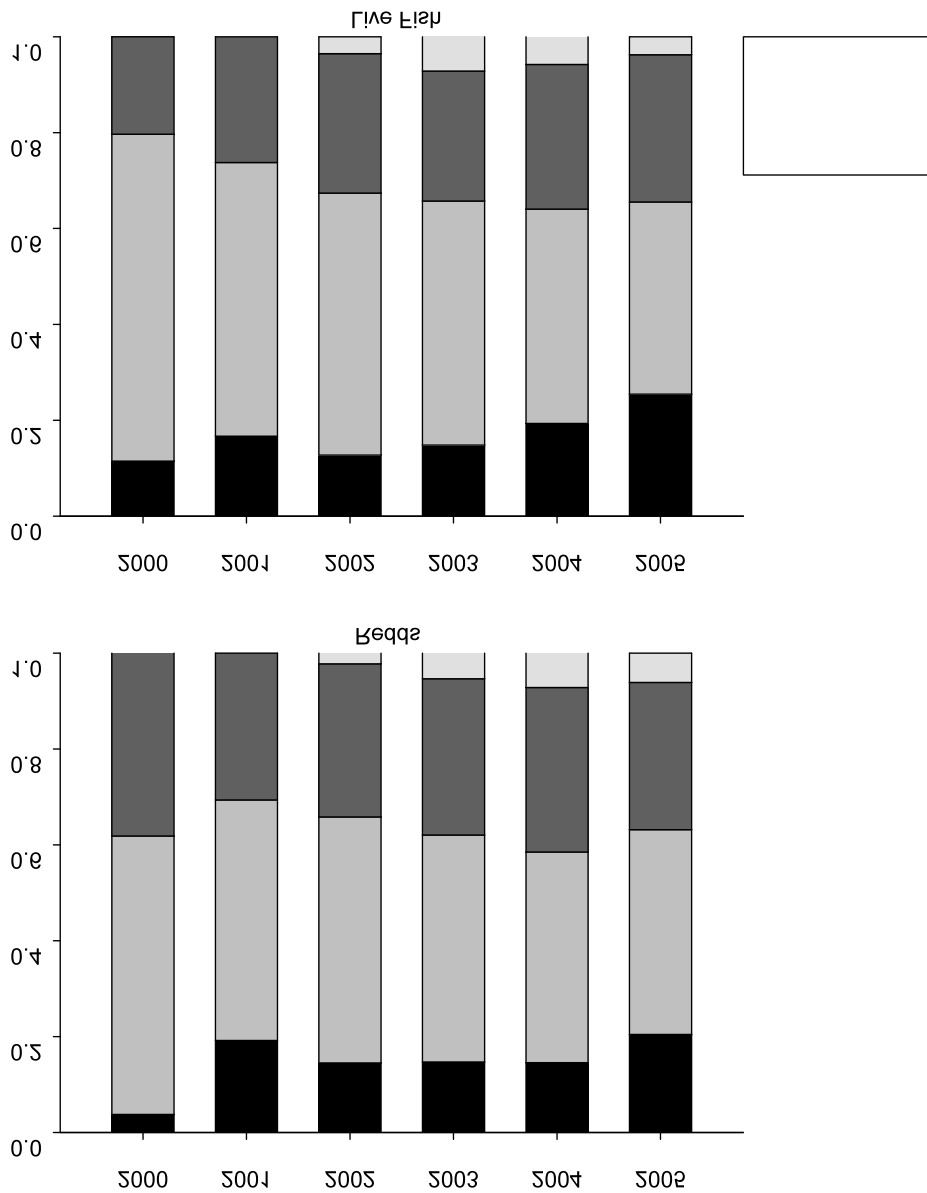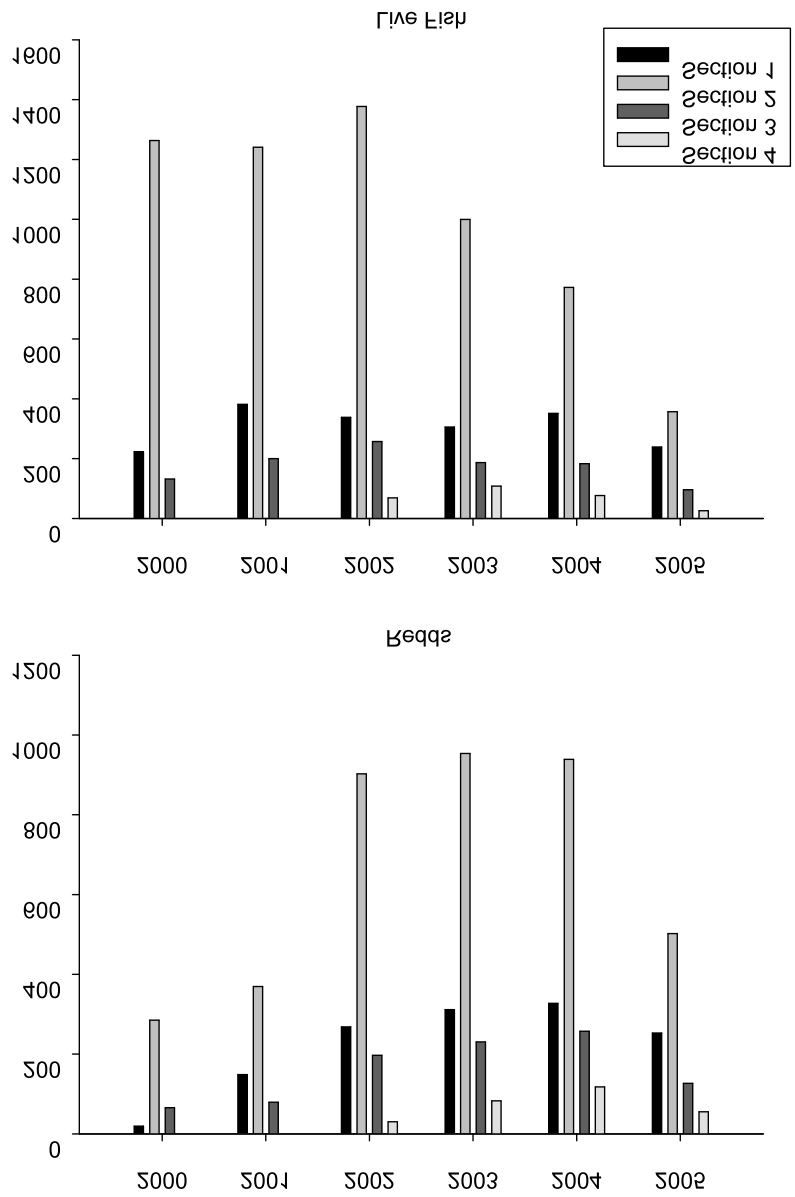
Figure 18. Density (number per mile) of live adult Chinook and redds encountered in each section of the Stanislaus River, 2000-2005. No data was available for Section 4 in 2000 and 2001.

COMPARISON OF WEIR AND SURVEY ESTIMATES

To further examine the reliability of adult Chinook escapement estimates in the Stanislaus River, we compared the weir estimates and CDFG survey estimates for 2003 through 2005. Our objective was to evaluate the consistency between weir counts and the carcass survey results using both annual estimates and weekly estimates. For the carcass survey results, we examined Schaefer estimates as well as counts of live fish, redds, and carcasses. Note that counts of live fish, redds, and carcasses were summarized by week in the CDFG reports. Because these counts were tallied across sampling days and weeks, they are likely to include numerous repeat counts of the same live fish, redd, or carcass. As such, the absolute magnitude of the total counts are not expected to reflect to the population abundance of spawners. Rather, they provide measures of the presence and persistence of each component over the course of the migration period.

Annual Estimates

There were notable differences between the annual weir and CDFG Schaefer estimates of total escapement (Table 9 and Figure 19). For 2003, the Schaefer estimate was higher than the weir estimate by roughly 2,000 fish (39% higher). In contrast, Schaefer estimates were lower than weir estimates in 2004 and 2005 by roughly 1,100 fish (24% lower) and 1,200 fish (28% lower), respectively. Further, the Schaefer estimates were outside of the 95% confidence intervals of the weir estimates in all three years (Figure 19). These are important results. To the extent that the weir estimates are highly accurate, as our analyses and previous studies of Vaki systems indicate, the discrepancies in escapement estimates must largely reflect potential bias and imprecision in the Schaefer estimates.

Total counts of live fish and redds were quite high in 2003 and 2004 (Table 9), reflecting the high intensity of CDFG sampling across weeks. Although the weir estimate was relatively high in 2005, there were markedly lower counts of live fish, redds, and carcasses that year. This likely reflects either a lower intensity of sampling or reduced visual detection due to adverse conditions such as high flow or turbidity. Indeed, flows were somewhat higher throughout the migration period in 2005. For example, the average November flow at OBB in 2005 was 373 cfs, whereas averages in 2003 and 2004 were 287 cfs and 300 cfs. It is unclear if such differences could profoundly affect visual detection.

Table 9. Annual weir estimates, Schaefer estimates, and total counts of live fish, redds, and carcasses across weeks of the CDFG survey.

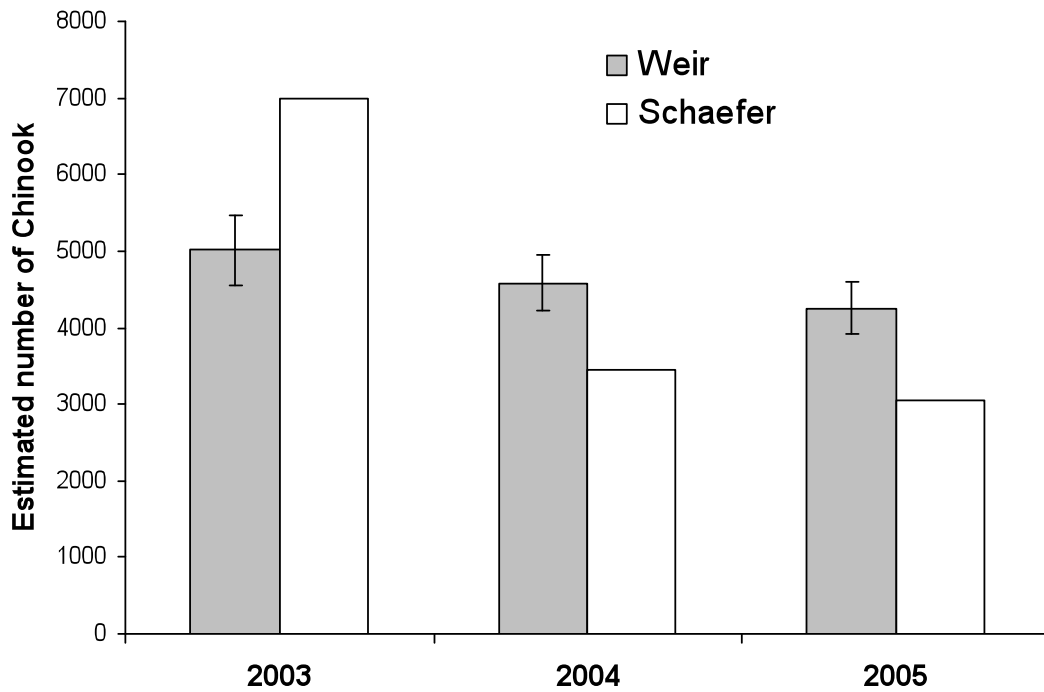| Year	Weir Estimate	Schaefer estimate	Live fish	Redds	Carcasses
2003	5,009	6,980	7,861	8,421	3,563
2004	4,580	3,458	6,900	8,280	1,959
2005	4,258	3,050	3,563	4,601	682

Figure 19. Annual weir and CFDG Schaefer estimates of total escapement. Errors bars for weir estimates denote 95% confidence intervals.

Weekly Estimates

We examined weir and survey estimates by week, where "week numbers" corresponded with those defined in the CDFG survey reports. The survey start dates differed across years, with "week 1" commencing on the following days: September 22 in 2003; October 4 in 2004; and September 26 in 2005. The duration of CDFG sampling also differed across years, with 17 weeks reported for 2003, 14 weeks for 2004, and 13 weeks for 2005.

Weekly weir counts of fish passage exhibited bimodal patterns in all years, whereas the live counts exhibited one peak in abundance (Figure 20). In each year, the live counts trailed the weir counts by roughly 4-5 weeks, and lacked evidence of the first spike in abundance recorded at the weir. The early peaks in weir passage in 2003 and 2005 represented substantial components of the migration. In 2005, 44% of the total spawners counted at the weir passed in the first 5 weeks. The percentage was even higher in 2003, when 55% of the spawners passed within the first 5 weeks. This disparity in timing between the weir and live counts may be largely due to slow upstream movements of adults into the main spawning sections.

As expected, weekly Schaefer estimates of carcasses also trailed weir counts by numerous weeks (Figure 21). Although the 2003 and 2004 weekly Schaefer estimates show sensible patterns, the large estimate in week 12 of 2005 appears highly anomalous and suspect (Figure 21). Only 2 tags were recovered in week 12 of the survey (despite 71 tagged in the preceding week), with a resulting weekly estimate of 2,060 (Table 10). Prior to week 12, Schaefer estimates were quite consistent, ranging from 213 to 370 and providing a total estimate for weeks 6-11 of only 1,328 (Table 10).

Table 10. Numbers of tagged carcasses, recovered tags, and Schaefer estimates by week for the CDFG survey in 2005.

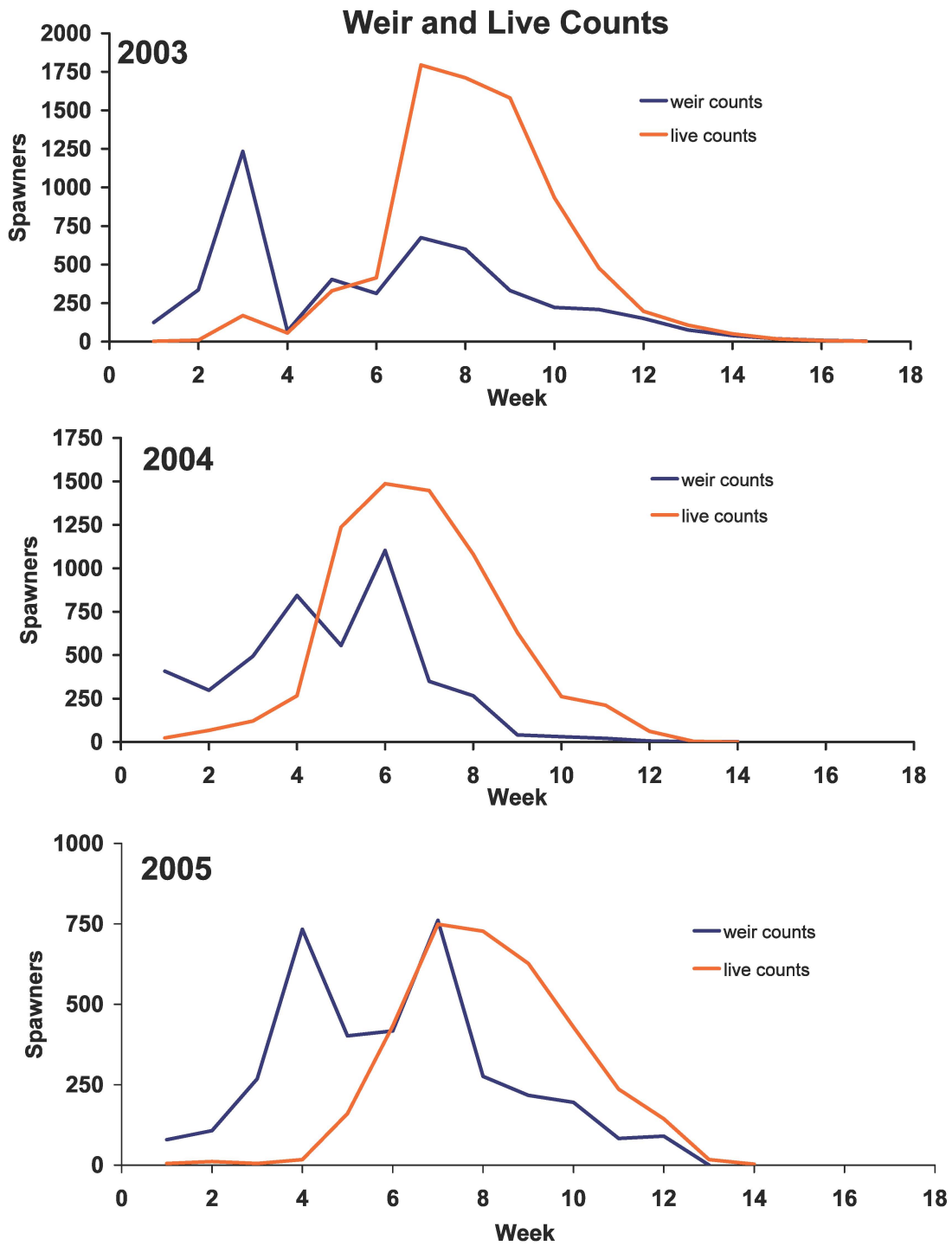| Week	Total Tagged	Number of Tags Recovered	Schaefer Estimate
6	2	1	0
7	31	6	213
8	61	24	258
9	69	24	370
10	66	35	247
11	71	47	240
12	40	2	2,060

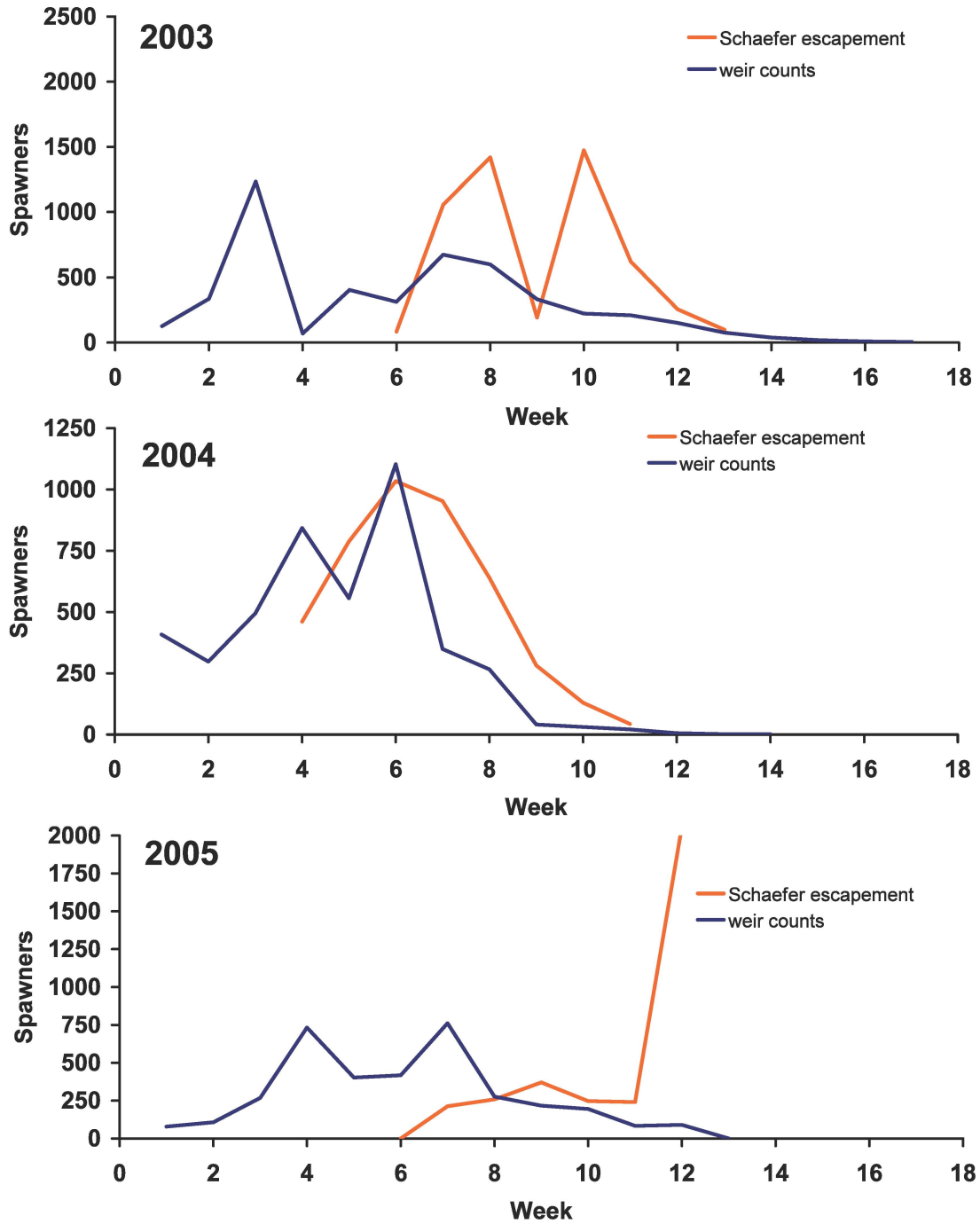**Figure 20: Weir counts and CDFG live counts by week from 2003-2005**

Figure 21: Schaefer escapement estimates and weir counts by week for 2003-2005

These results have several implications. First, because the week-12 estimate was based on only two recovered tags, the Schaefer estimate of 2,060 fish is highly uncertain (e.g., approximate $CV = 71\%$). In contrast, the estimate across weeks 6-11 appears quite precise ($CV = 9\%$ based on a surrogate Peterson estimate). Second, given estimates in previous weeks and patterns observed in 2003 and 2004, it seems highly likely that the week-12 Schaefer estimate was a gross overestimate of the carcasses remaining in the spawning reaches. Without week 12, the total Schaefer estimate for 2005 would have been roughly 3,000 fish less than the weir estimate, representing a significant underestimate of total escapement. This further suggests that strong survey biases were likely present across weeks 6-11. Finally, carcass recovery in week 12 was likely hindered by high flows, which increased rapidly that week to levels above 1,000 cfs. Though speculative, we expect that survey results may suffer greatly as flows increase.

Summary

Accurate escapement estimates provide critical and valuable information for monitoring spawner abundances and assessing key components of population dynamics such as cohort recruitment strength, factors affecting freshwater survival rates, and optimal spawner objectives. Despite strong CDFG sampling efforts in recent years, we found that Schaefer estimates differed markedly from the weir estimates in 2003-2005. Notably, an especially large and uncertain weekly estimate prevented the overall Schaefer estimate in 2005 from dramatically underestimating annual escapement. Unfortunately, intensive sampling cannot guarantee accurate mark-recapture estimates. There is considerable potential for bias in mark-recapture experiments due to non-random mixing of tagged and untagged carcasses, for example. Further, such problems are likely to increase substantially as flows increase or spawner abundances decline. In contrast, escapement estimates provided by the weir should be highly accurate regardless of spawner levels or flow conditions.

ASSESSMENT OF LENGTH AND SCALE DATA

In a recent memorandum, Simpson et al. (2006) discussed the feasibility of adult scale analysis for the purpose of aging adult Chinook salmon in the Stanislaus River. Adult Chinook salmon typically return to the Stanislaus, Tuolumne, and Merced rivers as two-, three-, and four-year-old fish, and the age composition of the run varies year to year (Baker and Morhardt 2001). More than 11,953 Chinook scale samples have been collected in tributaries to the San Joaquin River (i.e., Stanislaus, Tuolumne, and Merced rivers) and in the mainstem (i.e., Hills Ferry or Los Banos Wildlife Area) since 1976. As of August 2006, a total of 5,160 of more than 6,000 samples collected from the San Joaquin Basin between 1976 and 2000 have been analyzed by CDFG. Thus, a historic record of age data exists that could potentially be applied to Stanislaus River Chinook salmon in an effort reconstruct cohort recruitment. If successful, such estimates of cohort recruitment could greatly enhance various assessments such as stock-recruitment analyses and examinations of factors influencing survival rates of Stanislaus River Chinook salmon.

However, there have been only three years (1987-1989) where greater than 100 scale samples were analyzed for the Stanislaus River, and there are only two additional years (1998-1999) with enough archived scales that could be analyzed to provide a reasonable sample size. If variations in age structures from year to year were similar across Chinook populations in the Stanislaus, Tuolumne, and Merced rivers, then it would be reasonable and beneficial to apply the historic age data to Stanislaus Chinook salmon. Unfortunately, there is little information within the scale data to test for similarities in age structures among the populations.

To address this problem, we examined an alternative method for exploring year-to-year changes in age structure across Chinook populations of the Stanislaus, Tuolumne, and Merced rivers. In brief, we speculated that strong similarities in age structures across these populations would be evident in the length data collected by CDFG during spawning surveys. In particular, a high proportion of age-2 fish among spawners should be evident because age-2 fish are typically much smaller than age-3 and age-4 fish. Further, in contrast to the scale data, large sample sizes for length exist for all three rivers across many years. We therefore began by assessing similarities in mean length across rivers and years to determine if lengths were indeed correlated as expected. Next, we used the scale data to determine an appropriate critical length for

separating age-2 fish from older fish. We then applied that critical value to the length data sets to estimate the proportion of age-2 spawners in each system and year. Finally, we compared the age-2 proportions across rivers and with age-2 proportions derived from the scale record. These analyses, which are discussed in the following sections, provide conclusive evidence of strong similarities in the annual age structures of the different Chinook populations.

Comparison of Lengths

Length and sex data for adult Chinook salmon are regularly collected during annual spawning surveys of the Stanislaus, Tuolumne, and Merced rivers. We obtained the length and sex data for each river for years 1992-2004 (data provided by Jason Guignard, CDFG). Sample sizes were reasonably large for the Merced and Tuolumne rivers in most years except 1992 (Table 11). Sample sizes were typically lower for the Stanislaus River during 1992-2001, and no data were available for this river in 1993 and 1998. In general, however, the moderate to large samples sizes for both male and female lengths provide a strong basis for comparing trends in length across rivers and years.

Table 11. Sample sizes of adult spawners for which sex and length data were collected in the Merced, Stanislaus, and Tuolumne Rivers, 1992-2004.

1992	72	36	27	92	35	20
1993	324	0	72	215	0	97
1994	441	125	79	578	145	78
1995	135	76	197	177	68	216
1996	599	27	787	661	21	398
1997	326	274	451	450	329	605
1998	478	0	1140	555	0	1211
1999	391	320	1302	406	326	1104
2000	301	251	803	454	422	1358
2001	484	372	830	612	574	976
2002	477	766	818	500	1144	978
2003	212	785	235	337	1144	349
2004	349	440	213	435	667	310

Annual trends in mean length were similar among the three rivers (Figure 22). Both females and males showed increasing trends in mean length across years, though mean lengths of

males tended to fluctuate more between years. These similar patterns among males, females, and rivers were reflected by moderate to strong correlations among the time series of mean lengths. For example, correlations in mean length between females and males ranged from 0.75 for the Merced River to 0.92 for the Tuolumne River. Among females of different rivers, correlations in mean length ranged from 0.65 (Stanislaus versus Merced) to 0.85 (Tuolumne versus Merced). Correlations among males of different rivers were somewhat lower, ranging from 0.13 (Stanislaus versus Merced) to 0.78 (Tuolumne versus Stanislaus). However, the low correlation between Stanislaus and Merced males ($r = 0.13$) was highly influenced by the low-sample year 1992. Removing 1992 increased this correlation to 0.64, and generally increased all other correlations as well.

The strong correlations in mean lengths among rivers would be expected to arise for two key reasons: (1) adult Chinook salmon from all rivers experience similar ocean growth conditions during their final year at sea (Rogers and Ruggerone 1993, McKinnell 1995, Pyper et al. 1999), such that adults of a given age class are larger in some years and smaller in others; and (2) returning adults from all rivers share a similar age structure, and hence, a similar age-dependent length distribution (Pyper et al. 1999). It is likely that both processes are important. Nevertheless, evidence that age structure might be a key factor driving the high correlations is observed in Figure 23, which compares the length distributions of males (pooled across rivers) in 1999 and 2000. First, note that mean lengths of males for all rivers were roughly 10 cm less in 1999 than in 2000 (Figure 22). Such differences would be expected if there was a much larger proportion of age-2 males in 1999 than in 2000. Indeed, the length distribution in 1999 is distinctly bimodal, with a large fraction of males less than 70 cm (Figure 23). This is characteristic of a year with a substantial fraction of age-2 males. In contrast, the length distribution in 2000 appears much more normally distributed and lacks a significant component of fish less than 70 cm.

This one example is far from conclusive, however. We therefore attempted to estimate age-2 proportions from the length data and compare the results across, as described in the following sections.
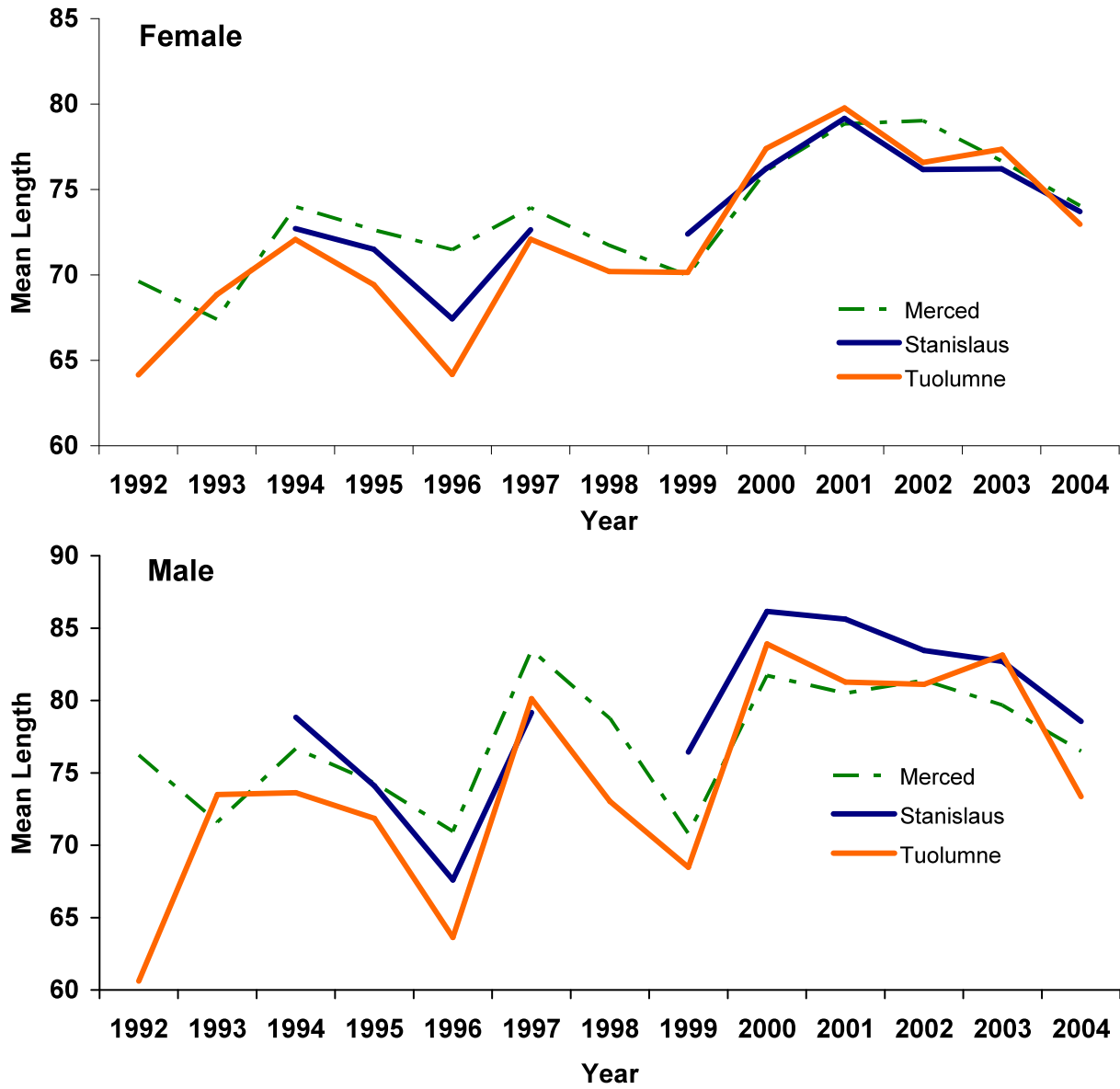
Figure 22. Mean lengths by year of male and female Chinook spawners in the Merced, Stanislaus and Tuolumne rivers.
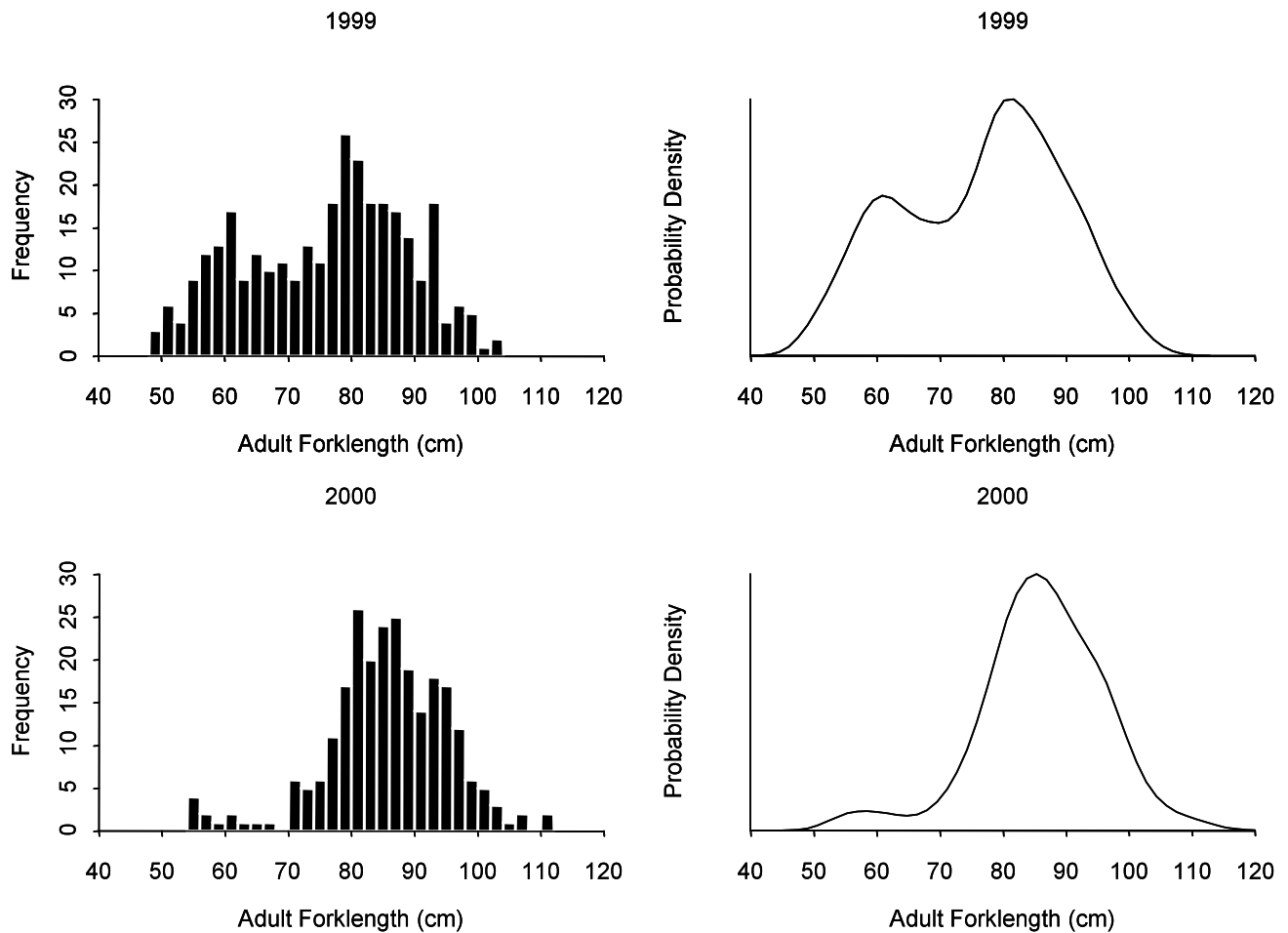
Figure 23. Histograms and probability density plots for lengths of adult males in 1999 and 2000 (data pooled across all Merced, Tuolumne and Stanislaus rivers).

Estimation of age-2 proportions from length data

To estimate the proportion of age-2 spawners in the annual length samples for each river, we used the historic scale data to establish a critical (or "cut-off") length for separating age-2 fish from older fish. As described in the memorandum by Simpson et al. (2006), the CDFG scale data represent samples from 1976-2000 that were taken at several locations within the San Joaquin River system (primarily the Tuolumne and Merced rivers, followed by the Stanislaus River). The scale data include about 4,500 useable records with data for age (consensus annulus count), sex, and fork length.

Using data from the scale record, we compared length distributions of age-2, age-3, and age-4 fish to visually determine the cut-off length distinguishing age-2 from age-3 spawners (Figure 24). This was done for males and females, as well as both sexes combined. In each case, we used all available scale records across years and sample locations.

There was considerable overlap in the length distributions of the three age classes, particularly for age-3 and age-4 fish (Figure 24). Despite this overlap, there were clear transitions between distributions of age-2 to age-3 fish occurring at lengths of roughly 71 cm for males and 66 cm for females. When both males and females were combined, the approximate cut-off length separating age-2 and age-3 fish occurred at 68 cm.

Next, we used these cut-off lengths derived from the scale data to estimate the proportion of age-2 fish in the annual length samples collected during the CFDG spawner surveys of the Merced, Stanislaus, and Tuolumne Rivers (1992-2004) (Table 11). Thus, for each year and river, we denoted all fish with lengths at or below at given cut-off length (e.g., 71 cm for males) as age-2 adults. This provided estimates of the proportion of age-2 fish by river and year. For comparison, we computed proportions separately for males and females, as well as for both sexes combined.
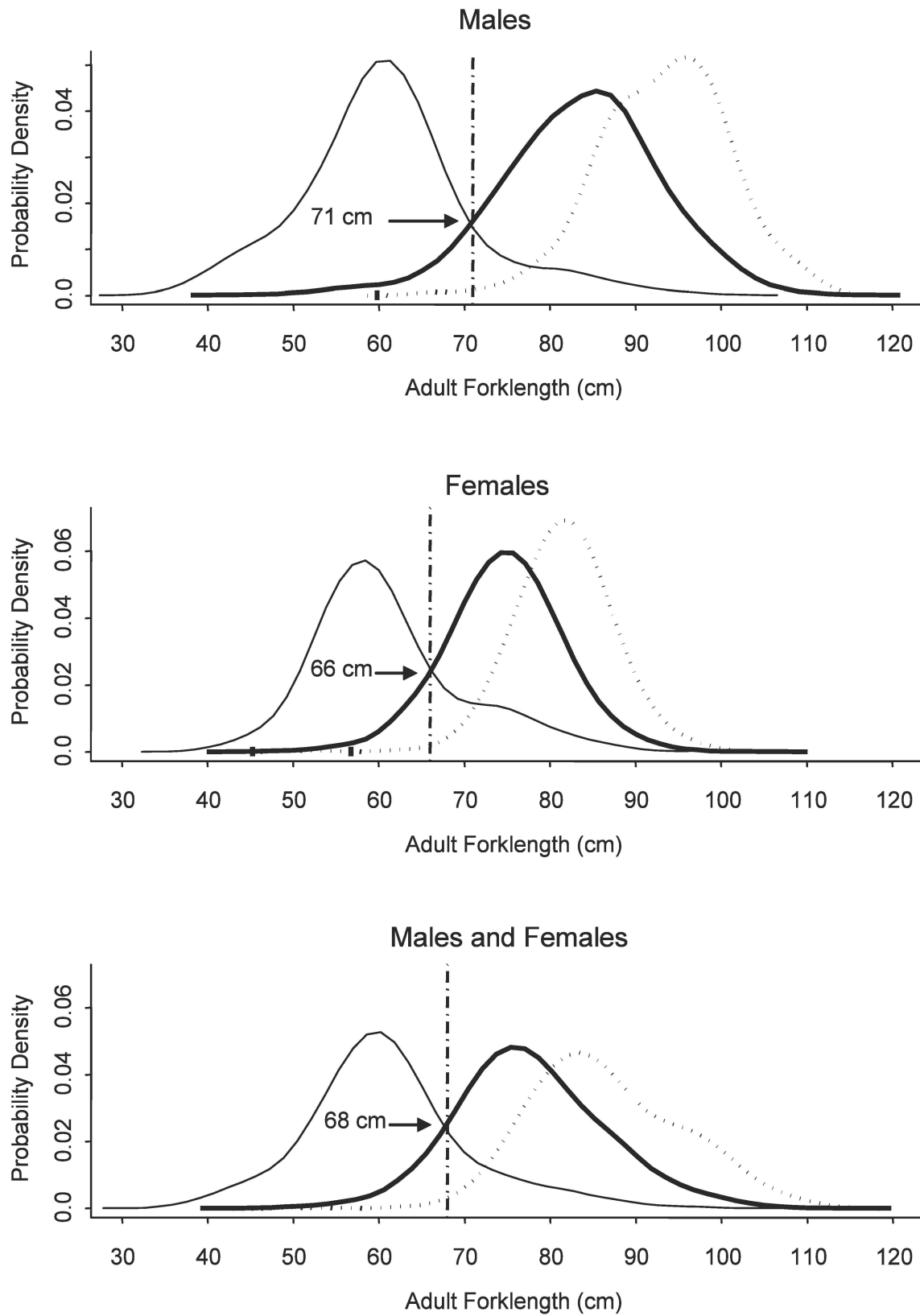
Figure 24. Length distributions (probability density plots) for age-2 adults (solid, thin line), age-3 adults (bold line), and age-4 adults (dashed line) from the CDFG historic scale data. Vertical lines denote approximate "cut-off" lengths between age-2 and age-3 distributions.

Comparison of age-2 proportions

We found generally strong correlations among the estimated age-2 proportions of spawners across rivers, suggesting that age structures of spawners in each river change in similar ways from year to year. For example, across 1992-2004, correlations for the proportion of age-2 males ranged from 0.76 (Merced versus Stanislaus) to 0.86 (Tuolumne versus Stanislaus) (Table 12). Correlations were weaker for comparisons involving female spawners of the Stanislaus River. More importantly, all correlations among the rivers were quite high (> 0.8) for the combined data (males and females) (Table 12).

Table 12. Correlations in the proportion of age-2 spawners (1992-2004) derived from length data for the Merced, Stanislaus, and Tuolumne Rivers. Length data were grouped by males, females, and both sexes combined.

	Males		Females		Combined	
	Stanislaus	**Tuolumne**	**Stanislaus**	**Tuolumne**	**Stanislaus**	**Tuolumne**
Merced	0.76	0.77	0.33	0.81	0.81	0.87
Stanislaus	--	0.86	--	0.45	--	0.87

Additional evidence of shared variation in age structures was found by comparing the length-based proportions with the actual scale estimates of age-2 proportions. The latter are based on known ages of fish, and thus represent our best estimate of the actual age-2 proportions. Note, however, that estimates for the scale data were derived from annual samples pooled across all locations and both sexes. This provided annual sample sizes of roughly 200 to 500 for years 1992-2000. For these years, the vast majority of scales came from the Merced and Tuolumne rivers. Application of the historic scale record to Stanislaus River Chinook salmon will require use of pooled scales across rivers and sexes, so the scale estimates of age-2 proportions used here are an appropriate measure.

Correlations between the scale-based estimates of age-2 proportions and length-based estimates were greater than 0.7 for all comparisons expect Stanislaus females (Table 13). Moreover, the actual estimates of age-2 proportions for the scale and length-based approaches were very similar across years regardless of river. For example, Figure 25 shows the scale estimates of age-2 proportions and the length-based estimates for data pooled across rivers and sexes. The close agreement of these estimates, and the strong correlations between the scale estimates and those of the length-based approach is strong evidence in support of the hypothesis

that annual age structures of Chinook spawners vary in similar ways across the Stanislaus, Merced, and Tuolumne rivers. Of course, these analyses were limited to proportions of age-2 fish because there is too much overlap in lengths of age-3 and age-4 fish to separate these groups. Nevertheless, it is reasonable to assume that similar variation in these proportions would be observed among rivers. Finally, correlations were general weak for Stanislaus females. While this may indicate strong differences between Stanislaus females and those of other rivers, the small sample sizes for this group in early years may have also obscured the results. For example, removing the 1992 data point for Stanislaus females always resulted in a large increase in correlations.

Table 13. Correlations between the proportion of age-2 spawners derived from the scale data (all rivers and both sexes combined) versus the age-2 proportions derived from length data for each river and sex. Correlations are across years 1992-2000.

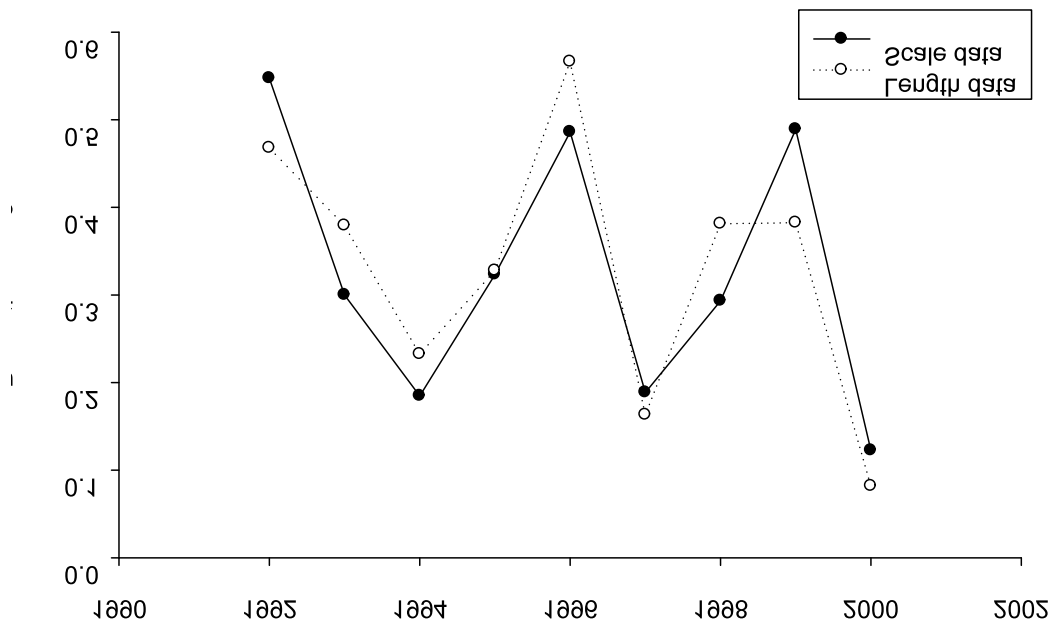|	Males	Females	Combined
Merced	0.73	0.76	0.84
Stanislaus	0.77	0.27	0.72
Tuolumne	0.93	0.88	0.94

Figure 25: Proportion of age-2 spawners based on scale and length estimates (all three rivers and sexes combined).

Summary

In summary, we assessed evidence of similarities in spawner age structure across San Joaquin tributaries using the following steps: (1) we used CDFG scale data to derive length criteria for distinguishing between age-2 fish and older fish; (2) we applied the criteria to length samples to estimate the proportion of age-2 spawners by river and year; and (3) we compared these age-2 proportions across rivers, and with age-2 proportions estimated from the scale data. Our results strongly suggest that Chinook populations of the Tuolumne, Merced, and Stanislaus Rivers exhibit similar year-to-year changes in spawner age structure. These results indicate that it would be appropriate to apply the historic scale record to Stanislaus River Chinook salmon even though few scales were represented by these fish.

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