ESTIMATING THE ABUNDANCE OF SACRAMENTO RIVER JUVENILE WINTER CHINOOK SALMON WITH COMPARISONS TO ADULT ESCAPEMENT

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Estimating the abundance of Sacramento River juvenile winter chinook salmon with comparisons to adult escapement

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Abstract.—We developed in-river quantitative methodologies for indexing juvenile winter-chinook production (JPI) in the upper Sacramento River using data collected by rotary-screw traps at Red Bluff Diversion Dam. These indices were used in conjunction with and in support of adult escapement and for evaluating year-class strengths in winter-run abundance. Estimates of juvenile winter chinook production derived from escapement estimates based on ladder counts at Red Bluff Diversion Dam, winter chinook carcass surveys and the National Marine Fisheries Service's (NMFS) juvenile production estimate (JPE) were compared to the JPI to identify possible sources of bias and determine whether these surveys were correlated in magnitude and trend. Five complete brood-year (BY) periods – July through June the following year – were monitored to index winter-run production for 1995, 1996, 1997, 1998 and 1999. Emergence and dispersal of winter-run fry (≤ 45mm FL) started in July for all brood years evaluated with peak dispersal occurring in September. Pre-smolt/smolt (> 45mm FL) emigration started in September with 100% of production passing Red Bluff Diversion Dam two to three months prior to the onset of the next brood year. Between 81% (BY98) and 44% (BY99) of winter-run production used areas below RBDD for nursery habitat, and the relative utilization above and below RBDD appeared to be influenced by river discharge during fry emergence (P=0.029, r^2=0.838, N=5). This relationship may be a useful tool for managing fry distributions in the upper river to compensate for and address dwindling habitat during dry years. The JPI was also useful for providing supportive evidence of estimated escapement. We concluded that escapement estimates from winter chinook carcass surveys and ladder counts were relative predictors for evaluating year-class strengths ($r^2 \geq 0.566$); however, no correlation was found between ladder escapement estimates and JPI's, although readers are cautioned that one data point had a large influence on this conclusion (P=0.555; r^2=0.128; N=5). Paired comparisons with JPI and JPE, a production estimate that uses female escapement as the primary variate, did not significantly differ (*carcass survey JPE*, paired t-test, P=0.903, df=3; *ladder count JPE*, paired t-test, P=0.097, df=4), yet evidence suggested that ladder count JPE underestimated in-river abundance of juvenile winter run. First, ladder count JPE fell below the 50% confidence interval of JPI in 4 of 5 brood years evaluated (probability of occurrence ≈ 0.01). Secondly, egg-to-fry survival based on ladder count data averaged 118% (± 80 SD), indicating that fry production exceeded estimated egg deposition. Egg-to-fry survival from carcass survey data, on the other hand, averaged 29% (± 9 SD), a value similar to that which has been estimated for winter-run in the upper Sacramento River. We concluded that NMFS's JPE model, based on estimates of escapement using ladder count data, underestimated juvenile winter-run production, while carcass survey escapement estimates were found to be a satisfactory replacement for RBDD ladder counts.

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Introduction

The Sacramento River system is unique in that it alone supports four seasonal runs or "races" of chinook salmon *Oncorhynchus tshawytscha*. Named for the time the majority of adults enter freshwater on their spawning migration, these four runs include the fall, late-fall, winter, and spring chinook salmon. Steelhead trout *Oncorhynchus mykiss* is another indigenous salmonid in the system. Populations of all four runs of chinook salmon, and steelhead trout, have declined in the last 25 years. The most dramatic has been the winter chinook, which have declined from a high count of almost 118,000 in 1969 to a low of 189 in 1994.

Historically, winter chinook utilized spring-fed streams that provided coldwater flows for summertime spawning, incubation, and rearing (Yoshiyama et al. 1998). Most of their historical habitat occurred in the upper Sacramento River drainage where cool-water conditions prevailed year-round from glacier and snow melt off of Mount Shasta and Mount Lassen and from cold-water springs. During the early part of the 20th Century, numerous small dams were built in the upper Sacramento River and its tributaries which began reducing the reproductive potential of winter chinook (NMFS 1996). With the construction of Shasta Dam on the Sacramento River in 1943, winter chinook were blocked from reaching their historical spawning grounds on the Little Sacramento, Pit, McCloud, and Fall Rivers (Yoshiyama et al. 1998). Fortunately, water discharged out of Shasta Lake after 1944 was sufficiently cool to allow for reproductive success in the Sacramento River in areas that had not historically supported winter chinook production. Winter chinook populations rebounded during the first two decades following completion of the dam because of the continuous cold-water releases mimicking the necessary summertime flow conditions needed for winter chinook (Yoshiyama et al. 1998). However, winter chinook populations started a steady and precipitous decline during the subsequent two decades, due in part, to the operations at Shasta Dam episodically supplying water with temperatures needed for successful egg incubation (NMFS 1997). Construction and operation of Red Bluff Diversion Dam (RBDD) in 1967 created another impediment to winter chinook migration and survival in the main stem Sacramento River. Up to 40% of winter chinook encountering the dam during gates lowered operations were blocked, and those passing upstream were delayed on average 13 days (Vogel et al. 1988). Adults blocked by the dam were forced to spawn downstream in areas where water temperatures were frequently too high for successful egg incubation (NMFS 1997). Winter chinook populations declined by almost 99% from 1966 to 1991 despite conservation measures to improve habitat and spawning conditions. Winter chinook were formally listed as a Federally threatened species in 1989 and reclassified as Federally endangered in 1994 in response to the continued decline and continued threats to the population.

Since listing, numerous protective measures have been implemented in an attempt to protect winter chinook, including managing water exports from the Central Valley Project's Tracy Pumping Plant and State Water Project's Harvey Banks Delta Pumping Plant located in the southern Sacramento-San Joaquin Delta. The United States Bureau of Reclamation (BOR) and the Department of Water Resources are authorized for incidental take of up to two percent of the estimated number of juvenile winter chinook entering the Delta by the Tracy and Harvey Banks Delta Pumping Plants (CDFG 1996). Numbers of juvenile winter chinook salmon entering the Delta are based on a production model that uses escapement, estimated from counts of salmon using fishways that provide passage over RBDD, as the primary variate (Diaz-Soltero 1995, 1997; Lecky 1998, 1999, 2000).

From 1969 through 1985, RBDD was operated throughout the entire winter chinook migration period, facilitating accurate estimation of adult passage above RBDD. Beginning in 1986, gates were raised during the non-irrigation season to allow for unimpeded passage of most winter chinook (approximately 85% of the annual migration, on average; NMFS 1996). The diversion and fishways currently operate from May 15 through September 15, which historically included only a small portion (15%) of winter chinook migration compared to previous season long counts (Snider et al. 2000). Annual escapement is now estimated by expanding the abbreviated count to upstream passage prior to May 15 when the dam is not operating. This extrapolation, based on historical run timing, can lead to large errors in estimation (43% to 230%; NMFS 1997) and has come under increased scrutiny.

Starting in 1996, the upper Sacramento River winter chinook carcass survey was undertaken by the California Department of Fish and Game and the U.S. Fish and Wildlife Service. This survey was designed to augment existing escapement estimates because of the uncertainty and imprecision associated with RBDD ladder counts, and the importance of this estimate for recovery and management of winter chinook populations (Snider et al. 2000, Snider et al. 1999, Snider et al. 1998, Snider et al. 1997). These two survey methods have, at times, produced very different estimates of escapement. For example, in 1999 66% of observed salmon from RBDD ladder counts were grilse compared to 19% in the carcass survey (Lecky 2000, Snider et al. 2000). The disparity observed between these two surveys was largely a result of differences in size composition of fish sampled (Snider et al. 2000). Additional disparities between ladder counts and carcass surveys include different adult male:female ratios. In 1999, the male:female ratio was 1:8.4 for the carcass survey. Because gender differentiation is questionable from RBDD ladder counts, an assumed 1:1 sex ratio is used for estimates. These disparities in sex ratios between surveys have large net effects on estimating effective spawner populations (i.e., the estimated number of females), which in turn, are used as the primary variate in NMFS model for generating juvenile production estimates (JPE). The estimated winter chinook adult return for 1999 was 3,208 based on RBDD ladder counts (Lecky 2000). Of this return, 34% were adults resulting in an effective female spawner population estimate of 502: once numbers of grilse (N=2,127), adults transferred to Coleman National Fish Hatchery (N=24), adults dying before spawning (N=53; 5% loss) and spawning males (N=502; 50% ♂) were removed from the estimate. Conversely, the estimated adult return based on the carcass survey for 1999 was 2,262 (Snider et al. 2000). Although escapement was 29% lower than the ladder counts, 71.9% were spawning females (N=1,577), resulting in an estimate three times greater than the estimates from RBDD ladder counts (Snider et al. 2000). Given these differences and that the JPE does not account for the success or failure of juvenile production, in-river indices of abundance were needed in conjunction with and in support of adult escapement estimates for evaluating year-class strengths in winter chinook abundance.

This study was conducted by the U.S. Fish and Wildlife Service and funded by the BOR as a component of their evaluation of the Red Bluff Research Pumping Plant (RBRPP). One of our goals was to provide the RBRPP with information regarding temporal patterns of abundance for each race of chinook salmon and steelhead trout. The RBRPP, in turn, used this information to evaluate the use of experimental water lifts (pumps) for delivery of water to the Tehama-Colusa Canal system (Borthwick and Corwin 2001). In-river estimates of abundance were needed by the RBRPP to quantify the fractional entrainment of fishes by these lifts, such that potential impacts to fish populations could be determined.

Our second goal was to develop quantitative methodologies for indexing juvenile production (JPI) in the upper Sacramento River which could then be used in conjunction with and in support of adult escapement estimates. The NMFS generated JPE's for BY95 through BY99 were compared to our JPI's to determine whether correlations existed in trend and magnitude and to help identify possible sources of bias. Finally, temporal patterns of abundance were described for juvenile winter chinook salmon emigrating from the upper Sacramento River past RBDD.

Study area

The RBDD is located at river kilometer 391 (RK391) on the Sacramento River approximately 3.2 km southeast of the city of Red Bluff (Figure 1). It was completed in 1964 and began operation in 1966 (Liston and Johnson 1992). The purpose of the dam is to divert water into the Tehama-Colusa and Corning Canal system, for agriculture and wildlife refuges. The dam consists of eleven moveable gates which can be raised or lowered to impound and divert river flows into the canal system. For 20 years the dam gates remained closed year-round, until the winter of 1986 when gates were raised during the non-irrigation season to improve upstream fish passage.

The spawning grounds for winter chinook salmon occur almost exclusively upstream from RBDD and within the main stem Sacramento River (Vogel and Marine 1991, Snider et al. 1997). The RBDD is an ideal rotary-screw trap location because multiple traps can be attached to the dam and fished simultaneously within a transect across the river (Figure 2). The structures around the dam control the channel morphology and the hydrological characteristics of the area providing for consistent fishing conditions for evaluating trends in juvenile abundance between years.

Methods

Rotary trapping.—Four 2.4 m-diameter rotary-screw traps, attached directly behind RBDD, were used to estimate abundance of juvenile winter chinook salmon emigrating from the upper river. Rotary-screw traps were fished in river margin (east and west river-margins) and mid-channel habitats. Traps were positioned within these spatial zones unless sampling equipment failed, river depths were insufficient (i.e., < 1.2 m), or river hydrology restricted our ability to fish all traps (e.g., water velocity < 0.6 m/s). Rotary-screw traps were fished continuously throughout 24-hour periods, except during high-flow events and periods of high winter chinook abundance. When this occurred, randomly selected intervals were sampled by stratifying between diurnal and nocturnal periods, and sampling one of four non-overlapping strata within each period (Figure 3). Estimates were extrapolated to un-sampled strata by dividing catch by the strata-selection probability (i.e., $P = 0.25$).

Data were collected for each trap clearing and included: (1) length of time trap was fished, (2) water velocity immediately in front of cone at depth 0.6 m, (3) number of cone rotations during the fishing period, (4) depth of cone submerged, (5) debris type and amount, (6) captured fish identification, enumeration and fork length and (7) environmental conditions including water and air temperatures, and water turbidity. Chinook salmon race was assigned from daily length tables (DWR 1992). Water velocity was measured using an Oceanic® Model 2030 flow torpedo. Water samples were taken to measure turbidity and were analyzed in the laboratory using a Model 2100A Hach® Turbidimeter. Volumes of water sampled (or sieved) by RST were estimated from the (1) area of the cone submerged, (2) average velocity of water entering the cone, and (3) duration of the sample. River discharge (Q) was obtained from the California Data Exchange Center's Bend Bridge river gauge. The percent water sampled (%Q) passing RBDD was estimated by the ratio of water volume sampled to total Q passing RBDD.

Trap efficiency.—Fish were marked with either fluorescent spray dye (Phinney 1967), bismark brown stain (Mundie and Taber 1983) or both (Gaines and Martin 2001; draft). Spray-dye marking equipment consisted of: (1) a 1.5 hp compressor and regulator valve capable of maintaining hose pressure of 150 pounds per square inch (psi); (2) a sandblast gun fitted with a one quart canister and a 2.4 mm diameter siphon orifice; (3) and fluorescent, granulated pigment. Fish were stained in bismark brown staining solution, prepared at a concentration of eight grams of bismark brown to 380 L of water. Fish were stained in the solution for 45-50 minutes and removed.

Fish marked for trap efficiency trials were held for 24 hours before being released, generally 4 km upstream from RBDD. It was assumed that negligible mark-induced mortality occurred following the 24-hour holding period (Gaines and Martin 1999, draft). Several release strategies were investigated including: (1) hatchery and wild stock releases (Roper and Scarnechia 1999); (2) diurnal (sunrise) and nocturnal (sunset) releases; (3) fish size at release for fry (≤ 45 mm FL), pre-smolt (46 - 80 mm FL) and smolts (> 80 mm FL); (4) RBDD gates lowered and gates raised releases; and, (5) location of release (4 km vs. 2 km upstream of RBDD). Relative frequencies of expected (unmarked) and observed (marked) captures in traps during efficiency trials were tested with a chi-square test to determine whether catch from multiple traps could be combined for an unbiased estimate.

Stock assessments.—Stock assessments were estimated by developing a model which predicted trap efficiency using %Q (percent river discharge sampled) as the primary variate (eq. 5). Data used to develop this model was generated by conducting 54 trap efficiency trials at RBDD. Measurements of trap efficiency from these trials was plotted against %Q to develop a least squares regression equation (eq. 5) from which daily trap efficiency was predicted.

Daily passage (P_d).—The following procedures and formulae were used to derive daily and monthly estimates of total numbers of chinook salmon passing RBDD. Define C_{di} = catch at trap *i* (*i*=1,...,t) on day *d* (*d*=1,...,*n*), and X_{di} = volume sampled at trap *i* (*i*=1,...,*t*) on day *d* (*d*=1,...,*n*). Daily salmonid catch and water volume sampled was expressed as:

1. $$C_d = \sum_{i=1}^{t} C_{di}$$

and;

2. $$X_d = \sum_{i=1}^{t} X_{di}$$

The percent river volume sampled (%Q) was estimated from the ratio of water volume sampled (X_d) to river discharge (Q_d) on day *d*.

3. $$\%Q_d = \frac{X_d}{Q_d}$$

Total salmonid passage was estimated on day *d* (*d* = 1,...,*n*) by

4. $$\hat{P}_d = \frac{C_d}{\hat{T}_d}$$

where,

5. $$\hat{T}_d = (0.0091)(\%Q_d) - 0.00252$$

$\hat{T}_d$ = Predicted trap efficiency on day d.

Monthly passage (Passage).—Population totals for numbers of chinook salmon passing RBDD by month were derived from $\hat{P}_d$ where there are *N* days within the month:

6. $$\hat{P} = \frac{N}{n} \sum_{d=1}^{n} \hat{P}_d$$

Estimated variance

7. $$Var(\hat{P}) = (1 - \frac{n}{N})\frac{N^2}{n} s^2_{Pd} + \frac{N}{n}\left[\sum_{d=1}^{n} \mathrm{var}(\hat{P}_d) + 2\sum_{i \neq j}^{n} \mathrm{cov}(\hat{P}_i, \hat{P}_j)\right]$$

The first term in Equation (7) is associated with sampling of days within the month.

8. $$s^2_{\hat{P}_d} = \frac{\sum_{d=1}^{n} (\hat{P}_d - \overline{\hat{P}})}{n - 1}$$

The second term in Equation (7) is associated with estimating $\hat{P}_d$ within the day.

9. $$Var(\hat{P}_d) \doteq \frac{\hat{P}_d(1-\hat{T}_d)}{\hat{T}_d} + \text{var}(\hat{T}_d)\frac{\hat{P}_d(1-\hat{T}_d)+\hat{P}_d\hat{T}_d}{\hat{T}_d^3}$$

where

10. $$\text{var}(\hat{T}_d) = \text{error variance of trap efficiency model}$$

The third term in equation (7) is associated with estimating both $\hat{P}_i$ and $\hat{P}_j$ with the same trap efficiency model.

11. $$\text{cov}(\hat{P}_i, \hat{P}_j) = \frac{\text{cov}(\hat{T}_i, \hat{T}_j)\,\hat{P}_i\,\hat{P}_j}{\hat{T}_i\,\hat{T}_j}$$

where

12. $$Cov(\hat{T}_i, \hat{T}_j) = \text{var}(\hat{\alpha}) + x_i\,\text{cov}(\hat{\alpha}, \hat{\beta}) + x_j\,\text{cov}(\hat{\alpha}, \hat{\beta}) + x_i x_j\,\text{var}(\hat{\beta})$$

for some $\hat{T}_i = \hat{\alpha} + \hat{\beta}\, x_i$

Seventy-five, ninety, and ninety-five percent confidence intervals (CI) were constructed around $\hat{P}$.

13. $$\hat{P} \pm t_{(\alpha/2;n-1)}\sqrt{\text{var}(\hat{P})}$$

Juvenile production indices (JPI) were estimated by summing $\hat{P}$ across months for a winter chinook brood year (July 1 through June 30 the following year).

14. $$JPI = \sum_{month=1}^{12} \hat{P}$$

Fry (≤ 45 mm FL) and pre-smolt/smolt (> 45mm FL) passage was estimated from JPI by size class. However, the ratio of fry to pre-smolt/smolts passing RBDD was variable between years, therefore, we standardized juvenile production by estimating a fry equivalent value. Fry equivalent was estimated by the summation of fry passage to a weighted pre-smolt/smolt passage (59% fry-to-presmolt/smolt survival; Hallock undated). The JPI was also compared to the National Marine Fisheries Service's (NMFS) winter chinook production estimate (JPE).

Results

Fifty-four mark/recapture trials were conducted to model %Q with trap efficiency and resulted in a significant relationship ($P<0.001$, $r^2 = 0.459$, N=53; Figure 4). Trials included periods with RBDD gates lowered (N=18) and gates raised (N=36); hatchery produced (N=23) and naturally produced (N=31) experimental fish; fry (N=10), pre-smolt (N=25) and smolt-sized (N=19) chinook salmon; and, diurnal (N=21) and nocturnal (N=21) releases (Figure 5). Ninety-two percent of recaptures occurred within 24 hours of release, 98.5% within two days, 99.5% within three days, 99.9% within four days, and 100% within five days. Four traps were fished for most trials (N=48); however, six 3-trap trials were included in the model. Efficiency for combined traps at RBDD ranged from 0.37% (excluding the zero recapture trial) to 5.27%. River discharge during trials ranged from 5,950 to 36,508 cfs. Percent Q was highest during low-flow trials (6,404 cfs; 4.09 %Q) and lowest during high-flow trials (36,508 cfs; 0.88 %Q). The square root of efficiency minimized model error when linearly regressed with %Q (Box-Cox transformations; Neter et al. 1989; Figure 4). Release group size averaged 1,035 (SD=595; range = 255 - 2,820) and number of recaptures during trials averaged 21 (SD=20; range = 0 - 100). Fork lengths of marked ($\bar{x}$ = 70.4 mm FL) and recaptured ($\bar{x}$ = 71.0 mm FL) salmon during efficiency trials did not significantly differ (P = 0.202, df = 50, paired t-test).

Highest relative frequencies of recaptured fish were observed in mid-channel habitats, although this trend was not as pronounced during gates lowered operations at RBDD (Figure 6). Expected (unmarked) and observed (marked) relative frequencies did not statistically differ between west-river margin, mid channel, and east river-margin traps ($P > 0.05$, Pearson's chi-square); however, there appeared to be a general trend for lower numbers of recaptured fish in the west river-margin and higher numbers in the east river-margin (Figure 6).

Patterns of abundance for fry at RBDD were bimodal, temporally separated and consistent with the timing of the winter and fall chinook emigration (Figure 7). Newly emerged ($\leq$ 45 mm FL) and pre-smolt/smolt (> 45 mm FL) chinook salmon emigrated past RBDD throughout the year, although numerically, three quarters of all winter run passing RBDD were $\leq$ 45 mm FL (Figure 8). Length frequency distributions for winter chinook salmon captured at RBDD were bimodal; the first mode occurring at 34-35 mm FL and the second between 54 and 64 mm FL (Figure 8). Median fork lengths for winter chinook captured during the emergent period (weeks 27 - 42; September through October) remained static, while median fork lengths for fish captured during the outmigrant period (weeks 43 - 22; November through May the following year) increased, on average, 3 mm per week (Figure 9). The transition between weeks 42 and 43 from emergent to the outmigrant period was characterized by a 15 mm increase (36 to 51 mm) in median fork length. Weekly median fork lengths were $\leq$ 36 mm or > 50 mm, illustrating that few 40-50 mm FL winter chinook salmon were captured in rotary-screw traps at RBDD (Figures 8 and 9).

Winter chinook JPI's were generated for five complete brood-years (BY95, BY96, BY97, BY98 and BY99, Table 1). The number of 24-hour samples ranged from 126 for

BY97 to 293 for BY99. Additional funding for Sacramento River flow evaluations allowed us to increase sampling effort by 43% starting in July 1998. The number of days sampled within months ranged from 30 in April 1996 (BY95) and July 1999 (BY99) to 0 in January 1997 (BY96) and February 1998 (BY97; Table 1). Passage estimates for January 1997 and February 1998 were interpolated by taking an average of the JPI for the month immediately preceding and following the non-sampled month. We accounted for the uncertainty about this estimate by using a 500% relative standard deviation in our variance estimator for these months.

Winter chinook fry (≤ 45 mm FL) were predominately captured in July, August, September and October for all brood years evaluated (Table 1). Fry passage through August was generally low (≤ 16 %), except for BY98 (42%), with most fry production passing RBDD by September (mean = 85%; range 74 - 93%) and all fry production passing by November (Table 1, Figure 10b). Pre-smolt/smolt passage was greatest in November for all brood years evaluated except BY95 (October, Figure 11c). According to a length-at-date criteria, the tail end of the emigration period was marked by a substantial decrease in the number of pre-smolt/smolts passing RBDD two to three months prior to the onset of the next brood year (Table 1, Figures11c).

The JPI was greatest for BY98 (N=4,628,597), nearly three times greater than the next highest indexed year (BY97, N=1,876,636) and thirteen times greater than the lowest indexed year (BY96; N=338,856, Table 2). Total numbers of winter chinook passing RBDD were variable between BY97, BY98 and By99; however, the number of pre-smolt/smolts passing RBDD was similar during these years (BY97=473,659, BY98=551,050 and BY99=451,228; Table 1). We observed a large variation in the proportion of fry utilizing the areas above and below RBDD as nursery habitat, and estimated, on average, 75% of winter chinook produced reared below RBDD (Table 3). A negative correlation was observed between total summer discharge and the proportion of winter chinook juveniles passing as pre-smolt/smolts (P=0.029, r^2=0.838, N=5; Figure 12a); no correlations between smolt production and autumn or winter discharges were detected (Figure 12 b and c). Mean water temperatures during the summer period averaged 56.0°F and ranged from a high of 56.6°F during BY97 to 55.7°F during BY98 and 55.4°F during BY99. No relationship between the proportion of pre-smolt/smolts passing RBDD and summer temperatures were found (P=0.785, r^2=0.046).

The JPI was significantly correlated to estimates of the number of female spawners (FS) and total escapement from the winter chinook carcass survey (P=0.026, $r^2 > 0.94$; Figure 13 a and b). Conversely, JPI was weakly correlated to ladder escapement (P=0.555, r^2 = 0.128; Figure 14a) but was correlated, although not significantly, to ladder FS (P=0.143, r^2 = 0.566; Figure 14b). Paired comparisons between JPI's and carcass derived JPE's (P=0.903; Paired t-test; df=3) and ladder JPE's (P=0.097; Paired t-test; df=4) did not significantly differ, yet evidence strongly suggested that the ladder JPE underestimated in-river estimates of juvenile winter chinook abundance (Table 4). Ladder JPE fell below the 50% confidence interval for JPI in four of the five brood years evaluated, while carcass JPE fell within for all years evaluated. Secondly, estimated egg-to-fry survival using the ladder estimates of FS averaged 118% (± 80 SD), indicating fry production exceeded estimated egg deposition from ladder FS (Table 5). Egg-to-fry survival using carcass FS averaged 29% (± 9) and was similar to the survival rate used in the JPE model (Table 5).

Discussion

JPI estimate.—Juvenile trapping was identified within the Biological Opinion (2 February 1993) for the Red Bluff Research Pumping Plant Program at RBDD to assess the effects of the plant operations on Federally threatened (now endangered) winter chinook salmon (NMFS 1993). The original goal of this project was to understand the availability of juvenile salmonids for potential entrainment into the Red Bluff Research Pumping Plant by gaining life-history information on the population of fish moving past RBDD. Specific goals included estimating the abundance of juvenile winter chinook emigrating to areas below RBDD for use as indices of juvenile production in the upper Sacramento River.

We found RBDD to be an ideal monitoring location for winter chinook because (1) the spawning grounds occur almost exclusively upstream from RBDD (Vogel and Marine 1991; Snider et al. 1997), (2) multiple traps could be attached to the dam and fished simultaneously across a transect, and (3) the dam structure controlled the channel morphology and hydrological characteristics of the area providing for consistent sampling conditions to evaluate trends in juvenile abundance. These features provided optimal conditions for developing a time invariant trap efficiency model.

To generate experimentally sound and statistically robust JPI's required that we measure our gear efficiency (rotary-screw trap efficiency). This is typically conducted through the use of mark/recapture trials. However, there are periods when trap efficiency trials are neither practical nor possible. For example, when abundance is low, capture of sufficient numbers of experimental fish for conducting trap efficiency trials is not possible. Moreover, researchers in the upper river have been restricted from use of State or Federally threatened or endangered fishes to conduct mark/recapture experiments. To reduce our program's reliance on experimental fish, as well as minimizing impacts on threatened and endangered species, we developed a model that predicted trap efficiency. Our model used the percent of river discharge sampled (%Q) as the primary variate.

Cumulative information from efficiency trials were used in developing the model, which in turn, allowed us to calibrate fish capture by rotary-screw traps and estimate juvenile emigrant abundance during periods when experimental fish were not available or sub-sampling protocols were initiated.

Additional advantages were realized from using the trap efficiency model. The juvenile monitoring program at RBDD was under strict ESA restrictions for winter chinook take and incidental mortality. During high production years, our program would have exceeded its authorized take and/or mortality limit if not for a sub-sampling program. Because sub-sampling can compromise the results of trap efficiency trials, especially if the number of recaptured fish is usually low, use of this model allowed us to produce statistically robust JPI's during these sub-sampling events. When sub-sampling was implemented, we stratified between diurnal and nocturnal periods and sampled one of four non-overlapping strata within each period. Estimates were extrapolated to un-sampled strata by dividing our catch by the strata-selection probability (P = 0.25) and expanding this estimate by the predicted trap efficiency.

Sub-sampling was also useful when sampling high-flow events (river stage rises). High flows restricted, impeded and, in some cases, eliminated our ability to gather samples. High flows in combination with heavy debris loading, which was usually the case, can jeopardize personnel safety and substantially increase the risk of equipment loss. Also, incidental fish mortality increased with river stage rise. Following implementation of our sub-sampling protocol, the monitoring program was able to routinely fish river flows in excess of 55,000 cfs in all river channel habitats (Figure 2). Data on juvenile outmigration could not have been obtained had we not sub-sampled during these periods.

Furthermore, juvenile production indices have been scrutinized in the Central Valley because of the limited ability to sample streams and rivers with rotary-screw traps during high-flow events. This bias is magnified for fall chinook juveniles which peak in abundance in January, February and March: typically the wettest months of the year and greatest variability in river discharge. Sampling bias due to incomplete or missing samples was minimized for winter chinook fry due to their temporal pattern of emigration. Winter chinook began emerging and dispersing below RBDD in mid-summer (July) and continued through early autumn (October). During this study, flows in the upper river remained stable throughout this period with small to moderate freshets starting in late autumn (November) or early winter (December; Figure 15). Winter floods generally did not occur until January or February. Two months were not sampled during this study because of high sustained flows in January 1997 and February 1998 (Table 1). To estimate our JPI during these months, we used a mean from the months immediately prior to and following these periods. Simulation tests using BY98 and BY99 data to evaluate the bias associated with interpolating missing data indicated that the January interpolated estimate (BY96, N=12,124) would be, on average, underestimated (JPI = -2.7%, pre-smolt/smolt passage = -10.6%, and fry equivalent = -3.7%) and the February interpolated estimate (BY97, N=20,220) overestimated (JPI = 1.0%, pre-smolt/smolt passage = 3.5%, and fry equivalent = 1.3%). Fry passage was not affected by this bias because fry emergence and emigration concluded by early November in all brood years evaluated (Table 1, Figure 10b). Secondly, we used a 500% relative standard deviation in our confidence interval estimation to account for the variation and uncertainty in this monthly estimate. We expect our total JPI confidence intervals to be conservative because (1) January and February account for a relatively small percentage of JPI, (2) the interpolated JPI estimate was expected to be ± 3%, and (3) monthly variation added into the variance model would account for large errors in estimation.

Experimental bias in trapping efficiency trials may cause over or underestimation of chinook population numbers (Thedinga et al. 1994). In order for efficiency trials to be unbiased, marked fish should be randomly mixed with unmarked fish. At RBDD different distributions between marked and unmarked fish will lead to a biased estimate of passage when combining catch and efficiency across traps. Relative frequencies did not statistically differ between expected and observed captures during efficiency trials; however, there appeared to be a general trend for recapturing fewer fish than expected in the west river-margin and greater numbers in the east river-margin (Figure 6). This trend was more evident for hatchery produced fish than for naturally-produced fish. Other researchers have used hatchery-produced fish to estimate trap efficiencies for wild fish (Keenen et al. 1994), but some have found that emigration behaviors may differ between naturally-produced and hatchery-produced salmonids (Roper and Scarneccia 1996). We recommend testing spatial distributions prior to pooling catch and efficiency across traps to evaluate any bias associated with estimation.

Patterns of abundance.—We used a length-at-date criteria developed by [1]Greene (1992) to assign run designation to captured chinook. The criteria was developed for differentiating between the four runs of salmon in the upper river. The accuracy of the criteria was dependent on two assumptions: (1) timing of egg deposition and (2) rates of development and growth. Errors in one or both of these assumptions may have led to incorrect run designation which may, in turn, negatively or positively bias our winter chinook JPI's. For example, some winter chinook juveniles may have been erroneously identified as spring chinook. One-half of fry passage between the weeks of 43 and 46 were assigned as spring-run production, illustrated by the fact that median fry length shadowed the lower spring chinook length-at-date criteria (i.e.,upper winter chinook criteria; Figure 7a). If we assume that the one-half of these individuals were actually miss-assigned winter chinook, the JPI would have been negatively biased, on average, by about 2%.

Fry dispersal below RBDD at the beginning of the winter chinook emergent period was characterized by a 3 mm decrease in fork length between weeks 29 and 31 (Figure 9). Prior to this decrease, fry were present at RBDD but at low levels. Misidentification of fry between weeks 21 and 29 would negligibly bias winter our chinook JPI's because of low abundance during this period.

Length frequency distributions were bimodal for winter chinook captured at RBDD (Figure 8). The first mode (30 - 40mm FL) dominated our catch numerically where, on average, three-quarters of all winter chinook production dispersed below RBDD as fry (Table 6). Below RBDD, winter chinook fry have been found above the confluence of Deer Creek (RK352) from July through September, and frequency of occurrence increases and moves slowly downstream (NMFS 1997, Johnson et al. 1992). Although entry into the lower Sacramento River and Delta has occurred as early as September, winter chinook presence in these reaches is generally noted in November (Snider and Titus 2000a, Snider and Titus 2000b). Accordingly, winter chinook fry dispersing below RBDD in summer and early autumn rear below RBDD before outmigrating to the delta in autumn and winter. We compared the number of fry and pre-smolt/smolt winter chinook passing RBDD to derive a rearing index for river reaches above and below RBDD (Table 3).

We estimated that, on average, as many as two-thirds of winter chinook fry reared below RBDD. Differential survival rates have been observed for juvenile chinook production originating in different reaches in the Nechako River, British Columbia; unfortunately, little is known about the survival of naturally-produced fish rearing in different sections of the upper Sacramento River. The river above Red Bluff is characterized by a meandering channel, with large rubble and boulders predominating the streambed (USBR 1986). The river below RBDD is classified as a gravel-bed alluvial, predominated by sand, gravel and cobbles. Compared to the river reach above RBDD, the river below Red Bluff has

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Generated by Sheila Greene, Department of Water Resources, Environmental Services Office, Sacramento (8 May 1992) from a table developed by frank Fisher, CDFG, Inland Fisheries Branch, Red Bluff (revised 2 February 1992). Fork lengths with overlapping run assignments are placed with the later spawning run.

lower gradient, occupies a larger flood plain, and has greater fluvial geomorphic activity (USBR 1986). Prior to the construction of Shasta Dam, it is unlikely that winter run utilized either of these reaches as nursery habitat because water temperatures would have exceeded the tolerance range for fry in July, August and September: months of peak emergence. Although temperature control downstream from Shasta Dam has allowed these areas to be utilized for nursery habitats, winter chinook evolved under different temperature and hydrological regimes leading us to believe that fry survival may differ between these reaches. The importance of these areas as nursery habitats, relative to each other, will not be known until fry survival below RBDD is known.

The variation in the proportion of fry utilizing areas above and below RBDD as nursery habitats prompted us to investigate whether river flows and water temperatures had an effect on winter chinook behavior in the upper Sacramento River (Figure 12). Alevins display positive rheotaxis prior to emerging from gravel (Thorpe 1981). This behavior enables emerging fry to hold position and avoid being immediately displaced downstream. Upon emerging, fry attain neutral buoyancy, swim to the surface, and ingest air to fill their gas bladder (Groot 1981). Fry will then rest on the bottom substrate either upstream or downstream from their emergence points according to the relative intensity of their rheotactic behavior and velocity of water (Thorpe 1981). Many fry will disperse downstream from the spawning grounds within 24 hours of emergence with additional downstream movements occurring until arrival at their first nursery habitat (Reimers 1973, Groot 1981, McDonald 1960).

Pronounced genetic control of fry responses to current velocity has been demonstrated for rainbow trout and sockeye populations *Oncorhynchus nerka* (Godin 1981), and it is possible that these responses may also occur for chinook salmon. In this study, we found a negative linear relationship between cumulative discharge in July, August and September and the relative proportion of winter chinook passing RBDD as pre-smolt/smolts (P=0.029, r^2 = 0.838; Figure 12a). Between 1998 and 1999, for example, average river flows during the winter chinook emergent period decreased 27% from 14,908 cfs to 10,945 cfs, while the relative proportion of winter chinook passing as smolts increased from 12% to 43%, respectively. In addition to the decrease in river discharge, water temperatures dropped during this period from 55.7°F to 55.4°F; however, the relative proportion of winter chinook passing as pre-smolt/smolts was not correlated to temperature (P=0.785, r^2=0.046, N=5). The increase in pre-smolt/smolt passage for years with low discharge, relative to fry passage, suggests that larger numbers of winter chinook emigrated past RBDD as fry during years with greater river discharge during the emergent period. We concluded, although no measurements were taken at the spawning grounds, that as river discharge increased, water velocity at redds increased because discharge is dependent on water velocity and depth (Mundie 1974). Increased water velocity, during fry emergence, may have increased numbers dispersing downstream from RBDD by increasing negative rheotaxis or by dispersing fry, on average, further downstream. Alternatively, an increase in fry-to-smolt survival during low discharge emergent periods could also explain this correlation. This explanation seems doubtful since it is unlikely that decreased flows would enhance fry survival. For example, fry survival for hatchery-produced fall chinook has been found to be positively correlated with river discharge in the upper river (K. Niemela, unpublished data, USFWS, Red Bluff). We concluded that larger proportions of winter chinook fry rear above

RBDD at lower river discharge volumes during their emergent period. This trend was believed to be a behavioral response to decreased water velocity at redds or a decrease in the distance juveniles are physically displaced, relative to high-flow years. Conditions in the upper river vary with respect to flow and temperature, and it is believed this relationship may be a useful tool for managing winter chinook stocks in conjunction with and subsequent to changing in-river conditions.

During wet years, temperatures within the tolerance range of winter chinook extend farther downstream from Red Bluff than during dry years. Fry distributions in the upper river could be managed to compensate for and address the dwindling habitat below Red Bluff during dry years. Differential survival rates of fry rearing above versus below RBDD will be needed before managers can develop management strategies that would provide in-river conditions that maximize fry-to-adult survival.

Four distinct emigration patterns were observed for juvenile winter chinook captured at RBDD. Patterns were characterized by a (1) large dispersal downstream immediately following or shortly after emerging from redds, (2) a strong tendency for holding, (3) protracted pre-smolt/smolt emigration, and (4) episodic outmigration periods following high-flow and turbidity events. Winter chinook cohorts between 41 and 49 mm fork length appeared to demonstrate, as a whole, the strongest tendency to hold and have limited downstream movement past RBDD (Figure 8). Ninety-two percent of all winter chinook captured at RBDD were $\leq$ 70 mm, yet 41 - 49 mm cohorts comprised 4.5% of total captures versus 69.3%, 8.2% and 6.8% for 31 - 39 mm, 51 - 59 mm, 61 - 69 mm cohorts, respectively. Fewer than expected 41 - 49 mm cohorts were captured in rotary-screw traps although they were frequently captured while shoreline seining near the study site (Johnson and Martin 1997). Following emergence and dispersal, winter chinook fry need to find a feeding station for stream residence because conditions in the lower river and Delta are not conducive for survival. The transition from fry to parr is characterized by the development of individualistic, territorial, positively rheotactic, stream-bed related behavior ensuring retention in the system (Thorpe 1981). It is possible that during this phase parr are less vulnerable to capture by downstream traps because of their propensity to maintain station and not move downstream. Although we would expect parr of all size classes to exhibit similar behavior, data on emigrants passing RBDD suggests that 41 - 49mm cohorts exhibited the strongest tendencies for limited downstream movements. Alternatively, if movement only occurs in the extreme river-margins, they would not be sampled by our traps.

Adult and juvenile comparisons.—Estimated adult returns for winter chinook are currently being used for two important management purposes in the Sacramento River system. The first provides a measurable indicator for evaluating the success of winter chinook restoration and providing target criteria for delisting winter chinook from the Endangered Species List. Criteria over a specified number of years must occur before winter chinook can be delisted (NMFS 1997). Criteria include population growth and numerical escapement goals to ensure that the probability of extinction is low. Criteria for delisting winter run include: (1) annual female spawning abundance over 13 consecutive years of 10,000 with a cohort replacement rate > 1.0, and (2) relative standard errors of < 25% for spawner abundance estimates (NMFS 1997). If the level of precision for the latter criteria cannot be achieved, then the sampling period over which to calculate the cohort replacement rate must be increased by one additional year for each 10% of additional error above 25% (NMFS 1997). At RBDD with the current gates raised operating scenario during the non-irrigation season, relative standard errors range from 43 to 230% for spawner abundance estimated by ladder counts (NMFS 1997). With this level of error, it could take up to 33 years to delist winter chinook salmon after achieving female spawner abundances of 10,000, annually. Secondly, NMFS manages the Central Valley Project's Tracy Pumping Plant and the State Water Project's Harvey Banks Delta Pumping Plant diversions by limiting winter chinook juvenile entrainment at these facilities to 2% of the annual JPE (NMFS 1997, Diaz-Soltero 1995 and 1997, and Lecky 1998, 1999 and 2000). These JPE's are based upon a production model that uses escapement estimates from RBDD adult ladder counts as the primary variate. The JPE does not account for inter-year variations in survival to emergence, fry to pre-smolt/smolt survival (Botsford and Brittnacher 1998; Major and Mighell 1969; Wales and Coots 1955), fecundity (Healy and Heard 1983), environmental conditions (Bigelow 1996, Reiser and White 1988, Heming 1981), losses due to pollution (Arkoosh et al. 1988), degraded water quality (Bradford 1994), density dependent and/or independent factors, infectious disease (Arkoosh et al. 1988), and behavioral patterns (e.g., adults straying and spawning in streams where temperatures become too high for successful egg incubation; Hallock and Fisher 1985). Many of these factors are expected to influence juvenile production on a year-to-year basis while others may be year specific depending on environmental and/or anthropogenic-induced conditions.

Moreover, historical run timing, developed over a period of years when gates at RBDD were lowered year-round, may not be reflective of adult timing with current gate operations. The NMFS JPE model assumes that ladder counts at RBDD represent the final 15% of adult winter chinook migrants, annually. Telemetry studies conducted during this historical period found up to 40% of winter chinook encountering the dam were blocked and delay time for fish passing averaged greater than five days (Hallock et al. 1982; Vogel et al. 1988). We would expect run timing to be skewed towards latter periods relative to run timing with current gate operations: fish that would have been delayed during year-long gates lowered operations, can now pass freely upstream of RBDD. This trend would result in fewer salmon being observed at the RBDD fish ladders and escapement estimates that underestimate juvenile production. Readers are cautioned that one data point had a large influence on this limited data set (*N*=5, Figure 14a).

In general, the JPI was correlated in trend to carcass survey total escapement and FS estimates, and FS estimates derived from RBDD ladder counts (Figure 13a and b, Figure 14b). The reader is cautioned that these conclusions were based on small sample sizes in both the carcass survey (*N*=4) and ladder count (*N*=5) comparisons to JPI. When comparing carcass escapement and JPI, two years, BY96 and BY98, provided the greatest contrast in magnitude with BY99 and BY97 falling in between (Figure 13a and b). Similar trends were observed when comparing JPI with ladder escapement, except that BY99 escapement exceeded BY98 (Figure 14a). This trend is contrary to that observed in both the JPI and carcass escapement estimates which demonstrated a decline in abundance from 77% to 59% (Table 2).

Interestingly, the BY99 ladder escapement estimate was brought back into line (Figure 14b), relative to 1998, when estimating FS from ladder escapement. Ladder escapement in

BY99 included 66% grilse, and the JPE model assumes that contribution rates by grilse to winter chinook production is negligible. Removal of grilse, from ladder escapement estimates, brought the 1999 FS estimate in line relative to 1995, 1996 and 1998 ladder estimates and JPI relationship; and, although the relationship was not statistically significant (P=0.143), it was moderately correlated (r^2=0.566; N=5; Figure 14b). Based on these trends, we concluded that carcass escapement estimates, carcass FS estimates and ladder FS estimates are relative predictors of the magnitude of juvenile winter chinook production in the upper Sacramento River: as these estimates increased there was an associated increase in JPI's (Figure 13 a and b, Figure 14b). The weak correlation (P=0.555; r^2=0.128; N=5) found between ladder escapement estimates and in-river abundance may be strengthened by removal of grilse from ladder escapement estimates.

Paired comparisons between JPI's and carcass escapement derived JPE's (P=0.903; Paired t-test; df=3) and ladder escapement derived JPE's (P=0.097; Paired t-test; df=4) did not significantly differ, yet additional evidence suggested that ladder escapement JPE's underestimated in-river estimates of winter chinook abundance. Fifty percent confidence intervals (C.I.) about JPI were used for evaluating intra-year trends between JPI and JPE estimates. These intervals were used because the wide range on a 95% C.I. provided little useful information for comparisons due to the inherent variability of estimating numbers of fish in a large river systems. Also, comparisons were based upon the theory that a random normal variate, centered about μ for a population, would fall within our 50% C.I., 50% of the time. Likewise, the probability of falling outside the C.I. was 50% (25% below and 25% above). The carcass escapement derived JPE's fell within our confidence intervals for all years evaluated while ladder escapement derived JPE's fell under the C.I. in four of five brood years evaluated (Tables 4). The probability of observing this trend in ladder JPE was estimated by:

15. $$\text{Probability of occurrence} = \left[0.25^5\right] + \left[5\left(0.25^4 \times 0.5\right)\right] \approx 0.01$$

We would not expect this outcome (1 in 100) unless the JPE derived from ladder FS underestimated numbers of juvenile winter chinook being produced.

The final comparison made between JPE's and JPI's evaluated whether egg-to-fry survival derived from juvenile and adult data were biologically plausible (Table 5). The JPE model assumptions used for estimating egg and fry production included (1) FS from both ladder counts and carcass survey, (2) pre-spawning mortality rates (5% assumed for the ladder counts and observed rates for the carcass survey), (3) number of ova per female (N=3,859), (4) 0% loss due to temperature (temperatures were adequately controlled during this study), and (5) 59% fry-to-presmolt/smolt survival. The model uses the best available information for the upper Sacramento River, although some assumptions, relative to the estimate's uncertainty, are better than others. Assumptions 2, 3, and 4 were based on empirical data and assumption 5 was based on hatchery-marking studies (Hallock undated). We used a range of values for this assumption to reflect the uncertainty associated with this estimate (Table 5). No range of values was evaluated for assumption 1 since the intent of the comparison was to determine whether egg-to-fry survival calculated from estimated egg deposition was biologically plausible.

Estimated egg-to-fry survival using the ladder FS averaged 118% (± 80 SD), indicating fry production exceeded the estimated egg deposition. Alternatively, egg-to-fry survival using carcass FS averaged 29% (± 9 SD) and was similar to that used in the JPE model (25%; Diaz-Soltero 1995 and 1997, and Lecky 1998, 1999 and 2000). Not only are the egg-to-fry survival rates estimated from the ladder FS not plausible, they are biologically impossible. Furthermore, variability was two times greater in the ladder egg-to-fry survival calculation (Table 5).

Escapement estimates are inherently variable in large river systems and the temporal bias associated with the ladder counts exacerbates this problem. The diversion and fishways currently operate from May 15 through September 15 which historically included only a small portion (15%) of winter chinook migrant adults when season long counts were possible (Snider et al. 2000). Annual escapement is now estimated by expanding the abbreviated count (15%) to upstream passage prior to May 15 when the dam is not operating. A small shift in run timing will result in errors being compounded by over six fold. This bias likely increased variability in ladder FS and increased the variation associated with our estimates of egg-to-fry survival. The carcass survey does not have a similar temporal bias (or at least not one that is magnified as much as the RBDD ladder counts) leading to more precise estimates of egg-to-fry survival.

Genetic evaluations of salmon recovered in the carcass survey, while limited to BY97, indicated that approximately 72% of sampled carcasses were winter chinook (Figure 16a). Carcass abundance (N=108) and genetic homogeneity (88% winter chinook) were greatest in July, although non-winter salmon were sampled throughout the study period (range by month 12% - 51%). Early non-winter salmon were believed to be of late-fall origin, and spring or fall origin latter in the study period. Two distinct juvenile cohorts were observed at RBDD, one of which was consistent with production expected from summertime spawning (Figure 7). Even though 28% of carcasses were identified as non-winter chinook, we believe these fish produced juveniles during the summer emergence period. This conclusion was based on the fact that carcasses were recovered after spawning and carcass FS and JPI were correlated in magnitude.

In-river conditions today are different from those in which winter chinook evolved in the Central Valley and, undoubtedly, different selective forces are driving its genotype → phenotype → genotype transition (Healy 1994). It seems likely that racial hybridization between winter and other runs has or will occur given the: (1) ability of chinook salmon to expand into new habitats, (2) loss (in whole or part) of habitat and environmental conditions that allowed for reproductive isolation (Waples 1991), and (3) managed main-stem river conditions (i.e., discharge volume and water temperature) conducive to year-round survival. Following the construction of Shasta Dam, temperature control downstream allowed for summertime spawning in the main-stem river in areas that had not historically supported production in summer. Shasta Dam has acted as a conundrum: blocking habitats that supported summer spawning but also opening new areas for salmon to continue this unique life-history strategy (Yoshiyama et al. 1998). Winter chinook were the first, but unlikely the last, to invade and exploit these areas. Healy (1994) concluded that winter chinook, or at least the fundamental phenotypic characteristics that define it (e.g., summer spawning), need not be lost in the Sacramento River if conservation efforts are directed towards providing the in-river conditions that permit these unique characteristics to persist and flourish. Recovery should not focus on genetic salvage but towards providing the opportunities that allow for and retain the diverse character of the species (Healy 1994). We believe genetic evaluations should be incorporated into the monitoring program to provide managers with information to study the trends in genetic drift for upper river production. Subsequent to these evaluations and improvements in our understanding of gene flow between stocks, protection should be extended to juveniles being produced during the summer spawning periods regardless of parentage. For example, winter and non-winter spawners should be included from the carcass FS for subsequent use in NMFS's JPE calculations.

Conclusions and Management Recommendations

- The JPI was found to be useful for evaluating year-class strengths in winter-run production and for supportive evidence of adult escapement. Given the inherent variability associated with estimating populations of adults and juveniles in large river systems, independent surveys are needed to provide supportive evidence of the success or failure of winter-run restoration actions. Without these indices, managers will make decision on less and more tenuous information.

- Between 81% and 44% of winter-run production used areas below RBDD for nursery habitat, and the relative utilization of nursery habitat above or below RBDD appeared to be influenced by river discharge during emergence. This relationship may be a useful tool for managing fry distributions in the upper river to compensate for and address dwindling habitat during dry years.

- Studies should be conducted to determine whether survival rates differ for sub-populations of juvenile winter-run salmon emigrating past RBDD as fry or pre-smolt/smolts. This information would allow resource managers to make more informed water management decisions during summer months.

- We concluded that carcass escapement, carcass SP, and ladder SP were relative predictors for evaluating year-class strengths: as these estimates increased, there was an associated increase in juvenile production. No correlation was found between ladder escapement and JPI, however, one data point had a large influence on this limited data set.

- We recommend that NMFS's JPE model be modified such that grilse are removed from RBDD ladder escapement estimates prior to JPE calculation.

- We concluded that NMFS's JPE model, based on ladder escapement, underestimated juvenile winter-run production, and that the carcass survey was a satisfactory replacement for RBDD ladder counts.

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Table 1.—Monthly juvenile production indices (JPI) for winter chinook salmon captured by rotary-screw traps at RBDD (RK391), Sacramento River, CA., for brood-years 1995 through 1999. Results include JPI's for fry, pre-smolt/smolt, fry equivalent and total production, as well as median fork length (FL) median river discharge volume (cfs) and number of completed 4-trap, 24-h samples within the month (N).

				Monthly juvenile production indices			
Month	N	Median Discharge (cfs)	Median FL (mm)	[a]Total JPI	Fry JPI	Pre-smolt /smolt JPI	[b]Fry equivalent JPI
				Brood-year 95			
Jul	21	15,609	36	751	751	0	751
Aug	23	14,623	34	81,804	81,699	105	81,877
Sep	8	12,075	35	1,147,684	1,139,431	8,253	1,153,419
Oct	5	6,351	36	299,047	207,033	92,014	362,989
Nov	6	5,847	62	66,197	2,663	63,534	110,348
Dec	9	6,592	70	13,998	0	13,998	23,725
Jan	11	8,952	97	6,523	0	6,523	11,056
Feb	2	35,098	102	35,712	0	35,712	60,529
Mar	17	22,945	124	7,015	0	7,015	11,890
Apr	30	9,546	137	236	0	236	400
May	13	10,894	—	0	0	0	0
Jun	13	14,819	—	0	0	0	0
Total	158			1,658,968	1,431,577	227,390	1,816,984
				Brood-year 96			
Jul	14	14,771	34	903	903	0	903
Aug	19	14,617	34	18,836	18,836	0	18,836
Sep	12	9,739	34	228,197	225,698	2,499	229,934
Oct	17	6,397	35	24,226	16,285	7,941	29,744
Nov	22	6,095	70	66,167	0	66,167	112,147
Dec	8	19,202	82	8,801	0	8,801	14,917
[c]Jan	0	55,092	—	12,124	0	12,124	20,549
Feb	15	12,001	114	15,429	0	15,429	26,151
Mar	16	7,649	120	7,791	0	7,791	13,205
Apr	24	7,237	126	1,378	0	1,378	2,336
May	19	10,838	137	272	0	272	461
Jun	16	15,279	—	0	0	0	0
Total	182			384,124	261,722	122,402	469,183

Table 1.—(continued).

Month	N	Median Discharge (cfs)	Median FL (mm)	Juvenile production indices [a]Total JPI	Fry JPI	Pre-smolt /smolt JPI	[b]Fry equivalent JPI
				Brood-year 97			
Jul	19	15,502	35	18,584	18,584	0	18,584
Aug	16	11,506	35	134,165	133,633	532	134,535
Sep	13	8,638	35	925,284	912,652	12,632	934,062
Oct	10	6,204	36	410,781	333,955	76,826	464,169
Nov	11	6,360	63	295,668	3,546	292,121	498,667
Dec	11	6,900	69	30,139	0	30,139	51,083
Jan	5	39,920	82	7,826	0	7,826	13,264
[c]Feb	0	68,073	—	20,220	0	20,220	34,271
Mar	11	36,441	108	32,619	0	32,619	55,286
Apr	11	14,974	138	732	0	732	1,241
May	8	19,556	—	—	—	—	—
Jun	11	19,549	—	—	—	—	—
Total	126			1,876,018	1,402,370	473,647	2,205,162
				Brood-year 98			
Jul	17	16,659	34	184,896	184,896	0	184,896
Aug	13	16,021	34	1,540,408	1,538,369	2,039	1,541,825
Sept	18	11,920	34	2,128,386	2,081,786	46,600	2,160,769
Oct	24	7,802	37	404,275	250,098	154,177	511,415
Nov	19	10,551	57	245,739	11,263	234,476	408,680
Dec	26	18,027	69	49,018	0	49,018	83,081
Jan	24	7,805	103	49,753	0	49,753	84,327
Feb	16	30,475	97	8,833	0	8,833	14,971
Mar	28	23,048	114	4,150	0	4,150	7,034
Apr	23	11,871	138	1,754	0	1,754	2,973
May	26	12,559	150	262	0	262	445
Jun	30	12,572	—	—	—	—	—
Total	264			4,617,474	4,066,412	551,062	5,000,416

Table 1.—(continued).

				Juvenile production indices			
Month	N	Median Discharge (cfs)	Median FL (mm)	[a]Total JPI	Fry JPI	Pre-smolt /smolt JPI	[b]Fry equivalent JPI
				Brood-year 99			
Jul	31	13,580	36	8,186	8,186	0	8,186
Aug	28	9,777	35	91,836	91,836	0	91,836
Sep	23	8,515	35	404,378	398,421	5,957	408,517
Oct	21	7,018	38	163,482	95,859	67,623	210,475
Nov	24	7,634	60	155,239	7,124	148,115	258,166
Dec	29	7,967	74	60,397	0	60,397	102,368
Jan	20	8,938	91	94,675	0	94,675	160,466
Feb	16	43,807	101	44,918	0	44,918	76,132
Mar	25	23,952	117	28,042	0	28,042	47,529
Apr	25	11,103	121	1,092	0	1,092	1,851
May	27	12,788	152	375	0	375	636
Jun	24	14,249	—	—	—	—	—
Total	293			1,052,620	601,426	451,194	1,366,162

[a] Total JPI represents the summation of fry production and pre-smolt/smolt production.

[b] Because the ratio of fry to pre-smolt/smolts passing RBDD was variable between years, we standardized juvenile production by estimating a fry equivalent value. Fry equivalent was estimated by the addition of fry passage to a weighted pre-smolt/smolt passage (59% fry-to-presmolt/smolt survival; Hallock undated).

[c] No rotary-screw trap sampling occurred in January of 1997 and February of 1998 due to high, sustained river flows and heavy debris loading. For these situations, JPI's were estimated by calculating a mean JPI using the JPI from the month immediately preceding and following the non-sampled months.

Table 2.—Winter chinook salmon annual production indices and confidence intervals derived from captures by rotary-screw traps at Red Bluff Diversion Dam (RK391), Sacramento River, CA., for brood-years 1995 through 1999. Results include total brood-year production (JPI), 75%, 90%, and 95% confidence intervals (C.I.) and number of days sampled within the year (N).

			Confidence intervals					
			75% C.I.		90% C.I.		95% C.I.	
Brood-year	N	JPI	Lower	Upper	Lower	Upper	Lower	Upper
1995	158	1,658,968	1,142,384	2,180,345	927,594	2,413,271	786,283	2,566,547
1996	182	384,124	245,620	581,573	189,230	669,762	152,317	727,782
1997	126	1,876,017	1,258,831	2,591,740	992,267	2,911,602	818,304	3,122,146
1998	264	4,617,473	3,794,900	5,440,046	3,427,579	5,807,366	3,195,976	6,048,987
1999	293	1,052,619	825,872	1,279,365	724,570	1,380,668	663,603	1,447,331

Table 3.—Estimated number of winter chinook fry utilizing areas above and below Red Bluff Diversion Dam (RBDD; RK391) for nursery habitat. It was assumed that fry (≤ 45mm FL) passing RBDD used areas below and that pre-smolt/smolts (> 45mm FL) passing RBDD had used areas above RK391 for nursery habitats. Pre-smolt/smolt passage was weighted by approximately 1.7 (59% fry-to-presmolt/smolt survival; Hallock undated) to estimate fry equivalents. Upper and lower estimates were calculated using a liberal (100%) and conservative (22%; USFWS, unpublished data) estimate of fry-to-presmolt/smolt survival.

	Brood year					
River reach	BY95	BY96	BY97	BY98	BY99	Mean Utilization
			Estimate			
Above RBDD	385,407	207,461	802,793	934,003	764,734	35%
Below RBDD	1,431,577	261,722	1,402,370	4,066,412	601,426	65%
ratio above/below	1:3.7	1:1.3	1:1.7	1:4.4	1:0.8	1:2.4
			Upper			
Above RBDD	227,390	122,402	473,648	551,062	451,193	25%
Below RBDD	1,431,577	261,722	1,402,370	4,066,412	601,426	75%
ratio above/below	1:6.3	1:2.1	1:3.0	1:7.4	1:1.3	1:4.0
			Lower			
Above RBDD	1,033,591	556,373	2,152,945	2,504,827	2,050,877	57%
Below RBDD	1,431,577	261,722	1,402,370	4,066,412	601,426	43%
ratio above/below	1:1.4	1:0.5	1:0.7	1:1.6	1:0.3	1:0.9

Table 4.—Comparisons between juvenile production estimates (JPE) and rotary trapping juvenile production indices (JPI). Ladder JPE and Carcass JPE were derived from the estimated adult female escapement from the adult ladder counts at Red Bluff Diversion Dam and the upper Sacramento winter chinook carcass survey, respectively. Assumptions in the adult-to-fry JPE model were as follows: (1) 5% pre-spawning mortality for Ladder JPE, (2) 3,859 ova per spawning female (♀), (3) 0% loss due to temperature, and (4) 25% survival from egg-to-fry. Fry equivalent was estimated by assuming 59% fry to pre-smolt/smolt survival and adjusting pre-smolt/smolt production by this survival rate (Hallock undated). Carcass JPE did not differ from JPI (Paired t-test, P=0.903, df=3) or fry equivalent (Paired t-test, P=0.304, df=3); and, Ladder JPE did not statistically differ from JPI (Paired t-test, P=0.097, df=4) or fry equivalent (Paired t-test, P=0.074, df=4), although these tests should be interpreted cautiously because of small sample size and low power.

Brood-year	JPE based on effective spawner population				rotary trapping JPI	
	Ladder [a]		Carcass [b]		JPI	Fry equivalent
1995	573,062	(♀ = 594)	--		1,658,968	1,816,984
1996	279,778	(♀ = 290)	527,795	(♀ = 547)	384,124	469,183
1997	434,434	(♀ = 243)	1,330,892	(♀ = 1,380)	1,876,017	2,205,163
1998	770,835	(♀ = 799)	4,446,919	(♀ = 4,609)	4,617,473	5,000,416
1999	491,058	(♀ = 509)	1,521,623	(♀ = 1,577)	1,052,619	1,366,161

[a] Fry JPE obtained from Diaz-Soltero 1995 and 1997, and Lecky 1998, 1999 and 2000.

[b] Juvenile production based on carcass survey estimates and using estimated effective spawner population from Snider et al. (1997, 1998, 1999, and 2000).

Table 5.—Estimated (est) egg-to-fry survival for winter chinook salmon in the upper Sacramento River, CA. by comparing fry production passing Red Bluff Diversion Dam (RBDD; RK391) and number of winter chinook ova deposited. Egg deposition was estimated from the product of the number of female spawners, using RBDD ladder counts and the winter chinook carcass survey, and the average number of ova per spawning female (*N*=3,859). Fry production was estimated from the number of winter chinook fry equivalents passing RBDD by weighting pre-smolt/smolt passage by approximately 1.7 (59% fry-to-presmolt/smolt survival; Hallock undated). Upper (up) and lower (low) estimates of egg-to-fry survival were estimated using a liberal (100%) and conservative (22%; USFWS, Red Bluff, unpublished data) estimate of fry-to-presmolt/smolt survival.

Brood-year	Fry production			RBDD ladder counts				Carcass survey			
	Fry equivalent JPI				% Egg-to-fry survival				% Egg-to-fry survival		
	Est	Up	Low	egg production	Est	Up	Low	egg production	Est	Up	Low
1995	1,816,984	2,465,169	1,658,967	2,292,246	79	108	72				
1996	469,183	818,096	384,124	1,119,110	42	73	34	2,111,182	22	39	18
1997	2,205,163	3,555,314	1,876,018	937,737	235	379	200	5,323,568	41	67	35
1998	5,000,416	6,571,241	4,617,475	3,083,341	162	213	150	17,787,675	28	37	26
1999	1,366,161	2,652,305	1,052,620	1,964,231	70	135	54	6,086,492	22	44	17
Mean (±SD)					118 ± 80	182 ±122	102 ± 70		29 ± 9	47 ±14	24 ± 8

Table 6.— Relative proportion (in percent) of chinook salmon (*Oncorhynchus tshawytscha*) fry (< 46 mm FL) and pre-smolt/smolts (> 45 mm FL) captured by rotary-screw traps at Red Bluff Diversion Dam (RK391), Sacramento River, CA. Sampling was conducted from July 1994 through June 2000. Data is only summarized for complete brood-years. Brood-years are defined as; (a) 1 December - November 30 for fall chinook, (b) 1 April - 31 March for late-fall chinook, (3) 1 July - June 30 for winter chinook and (4) 15 October - 14 October for spring chinook. Data is also summarized for fry, sub-yearling and yearling rainbow trout. Brood-years for rainbow trout are 1 January - 31 December. Table reproduced from Gaines and Martin (2001).

	Fall chinook		Late-fall chinook		Winter chinook		Spring chinook		Rainbow trout		
Brood-year	Fry	Pre-smolt/ smolt	Fry	Pre-smolt/ smolt	Fry	Pre-smolt/ smolt	Fry	Pre-smolt/ smolt	Fry	Sub-yearling	Yearling
1994	62.0	38.0	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
1995	90.6	9.4	73.9	26.1	86.3	13.7	4.2	95.8	5.6	65.5	28.9
1996	79.9	20.1	20.6	79.4	68.1	31.9	30.9	69.1	4.2	62.7	33.1
1997	85.1	14.9	24.2	75.8	74.8	25.2	63.9	36.1	3.1	21.8	75.1
1998	90.3	9.7	62.0	38.0	88.1	11.9	85.6	14.4	4.0	36.4	59.6
1999	90.8	9.2	37.6	62.4	57.1	42.9	11.7	88.3	7.5	66.2	26.2

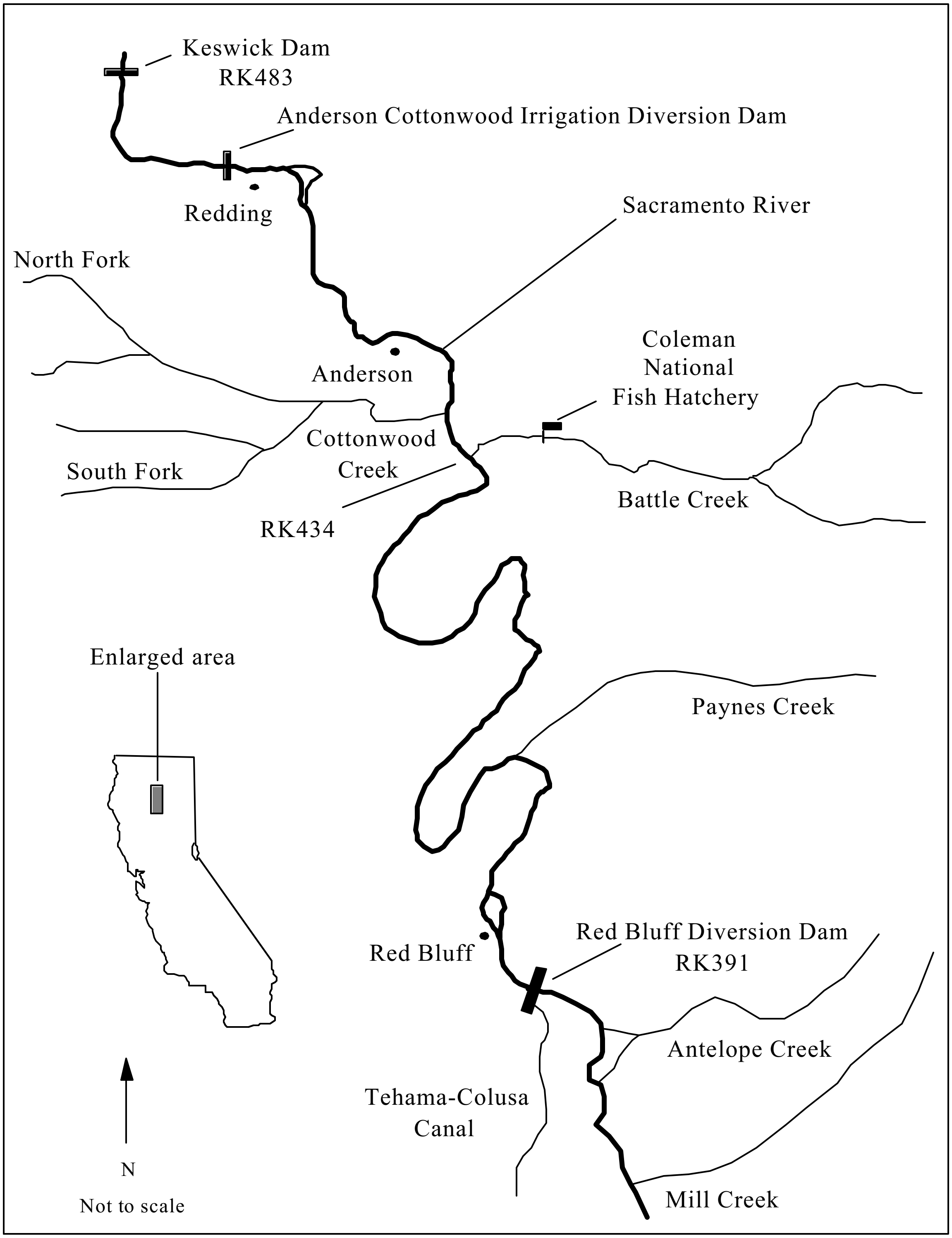

Figure 1.--Location of Red Bluff Diversion Dam on the Sacramento River at river-kilometer 391 (RK391).

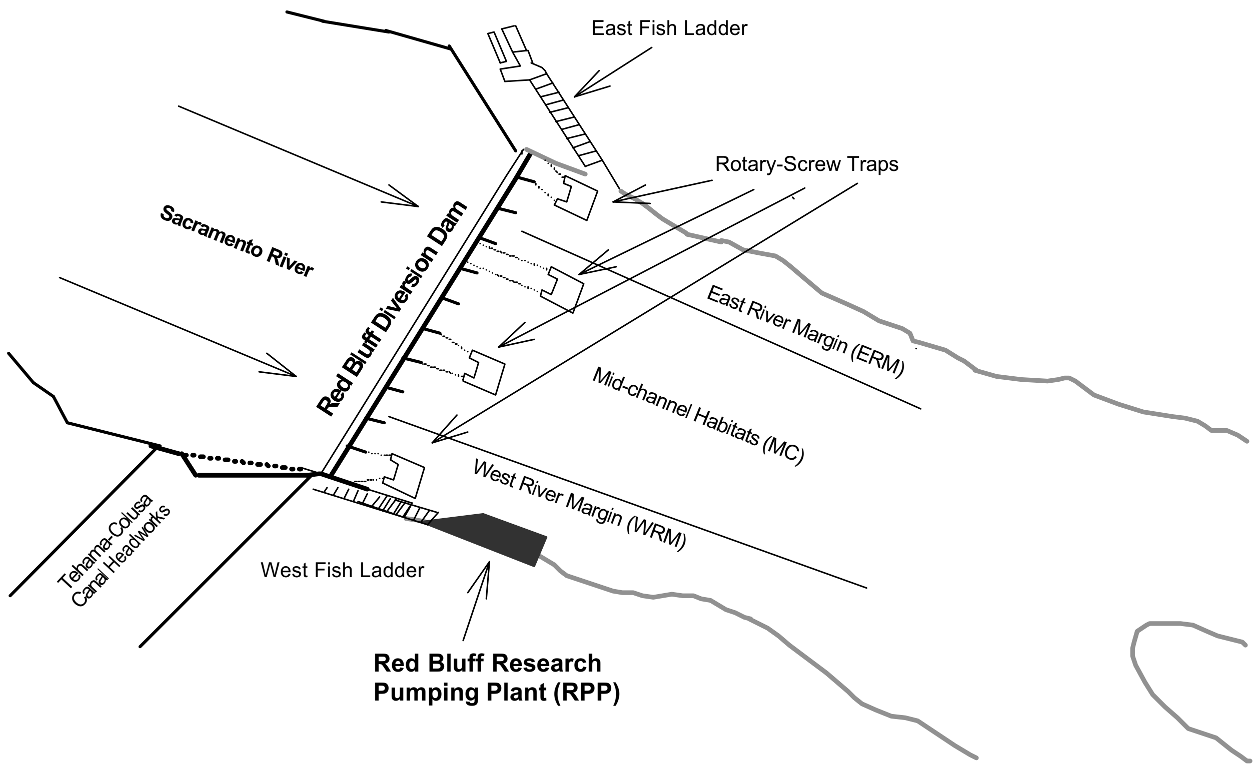

Figure 2.--Rotary-screw trap sampling transect at Red Bluff Diversion Dam (RBDD) on the Sacramento River at river kilometer 391 (RK391).

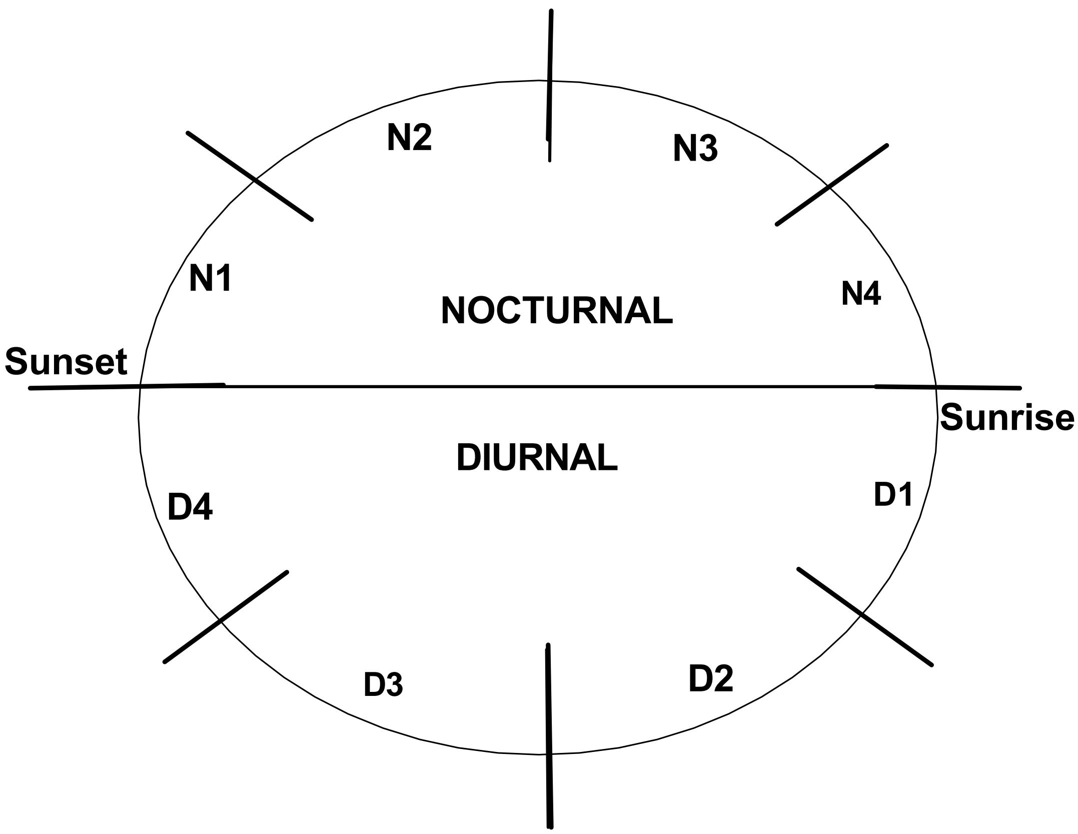

Figure 3. Sub-sampling design implemented to control "take" of juvenile winter chinook salmon during high production years and to control mortality during high-flow events. Each diurnal and nocturnal period was stratified into four non-overlapping strata. During sub-sampling, one diurnal and one nocturnal strata were selected for sampling each day using uniform probabilities (p = 0.25).

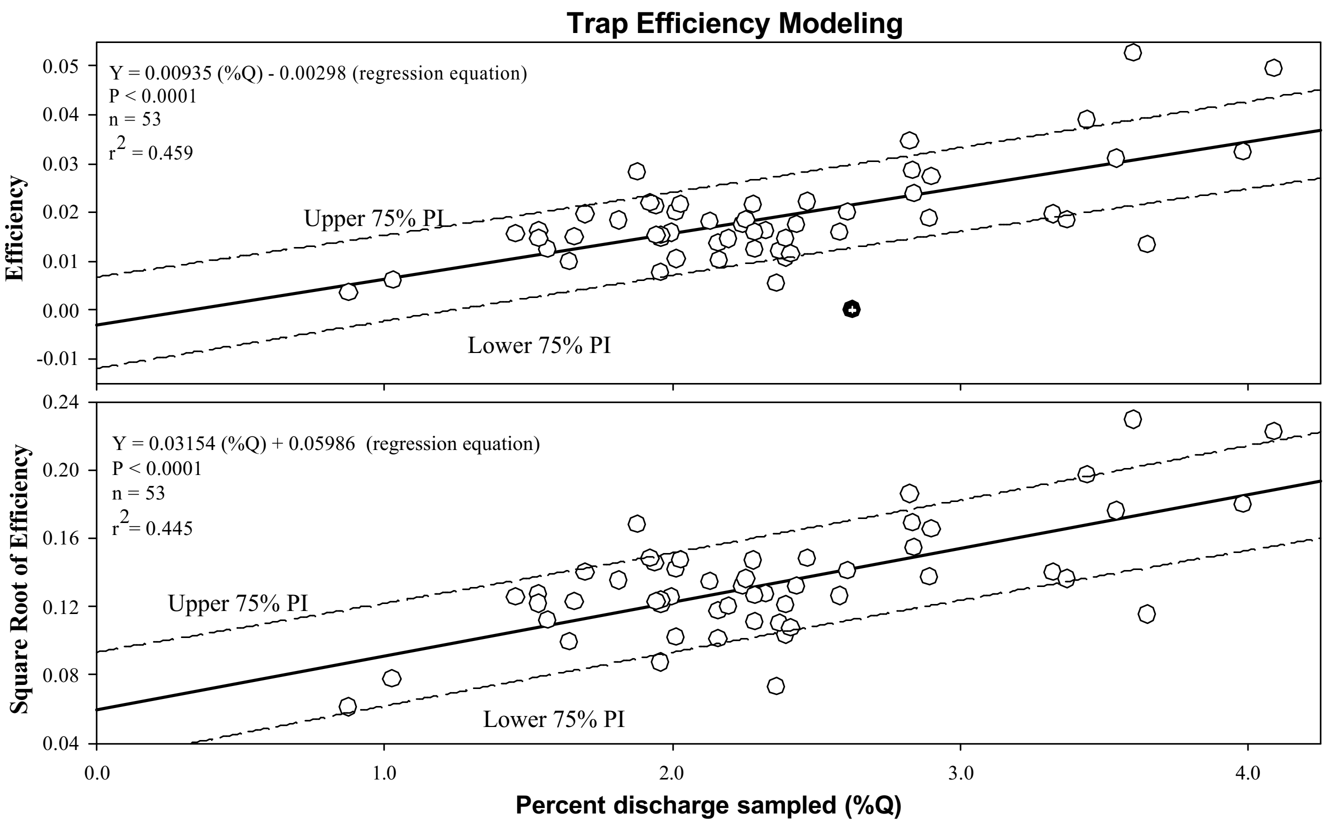

Figure 4. Monthly median fork lengths and juvenile production indices (JPI's) for **(a)** total passage (fry and pre-smolt/smolts combined), **(b)** fry and **(c)** pre-smolt/smolts. Data reported for BY95 (July 1995 through June 1996), BY96 (July 1996 through June 1997), BY97 (July 1997 through June 1998) and BY99 (July 1999 through June 2000). Vertical dashed lines denote separation between brood-years.

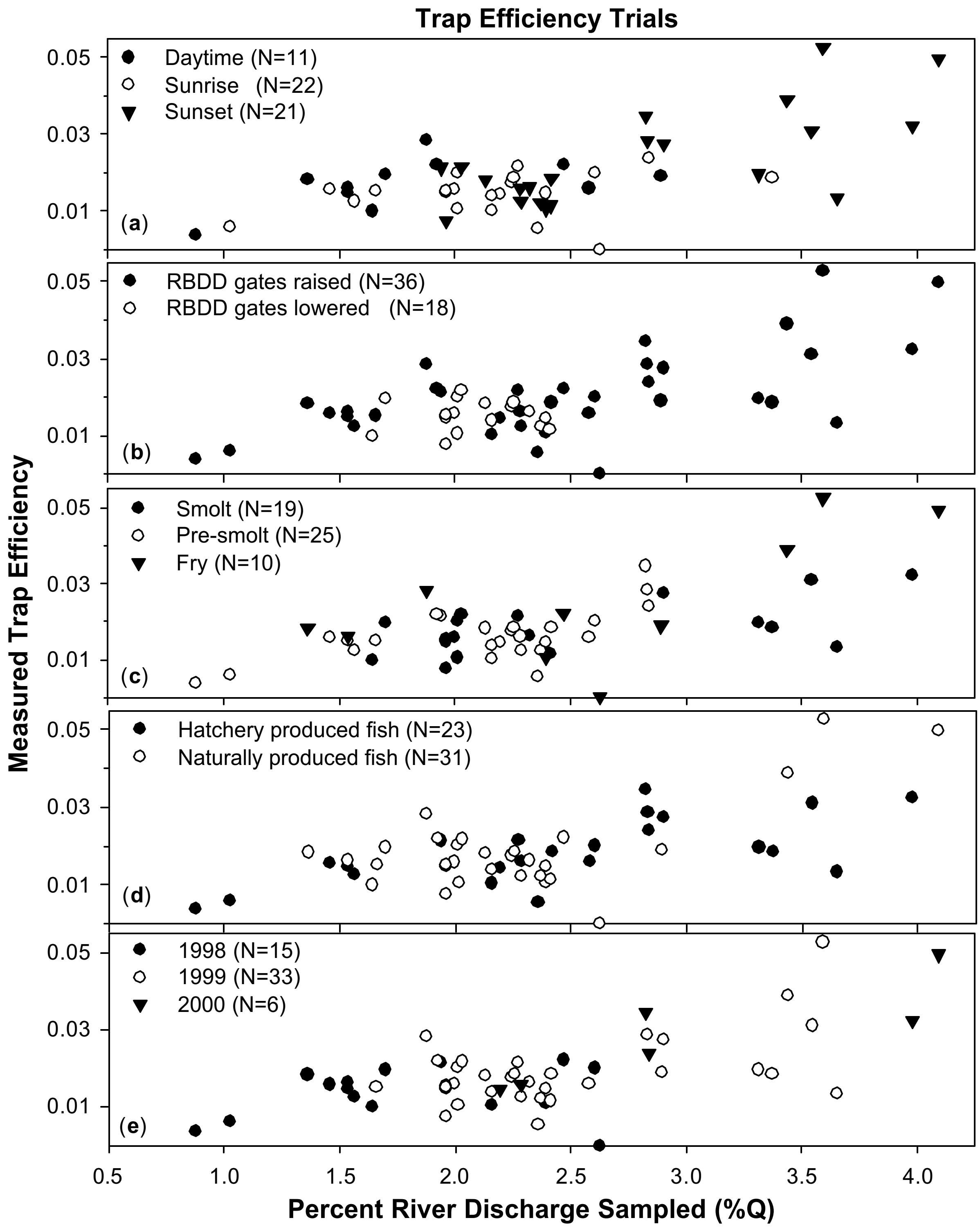

Figure 5.--Mark/recapture trials conducted for rotary-screw trap efficiency measurements at Red Bluff Diversion Dam (RK391), Sacramento River, CA. Independent trials investigated the effects of (**a**) diel behavior (sunset, sunrise or day releases), (**b**) RBDD gate operations (gates raised versus lowered), (**c**) fish fork length (FL) at release; smolt (>80 mm), pre-smolt (46-80mm), fry (< 46 mm), (**d**) fish origin (naturally produced versus hatchery produced), and (**e**) year of release.

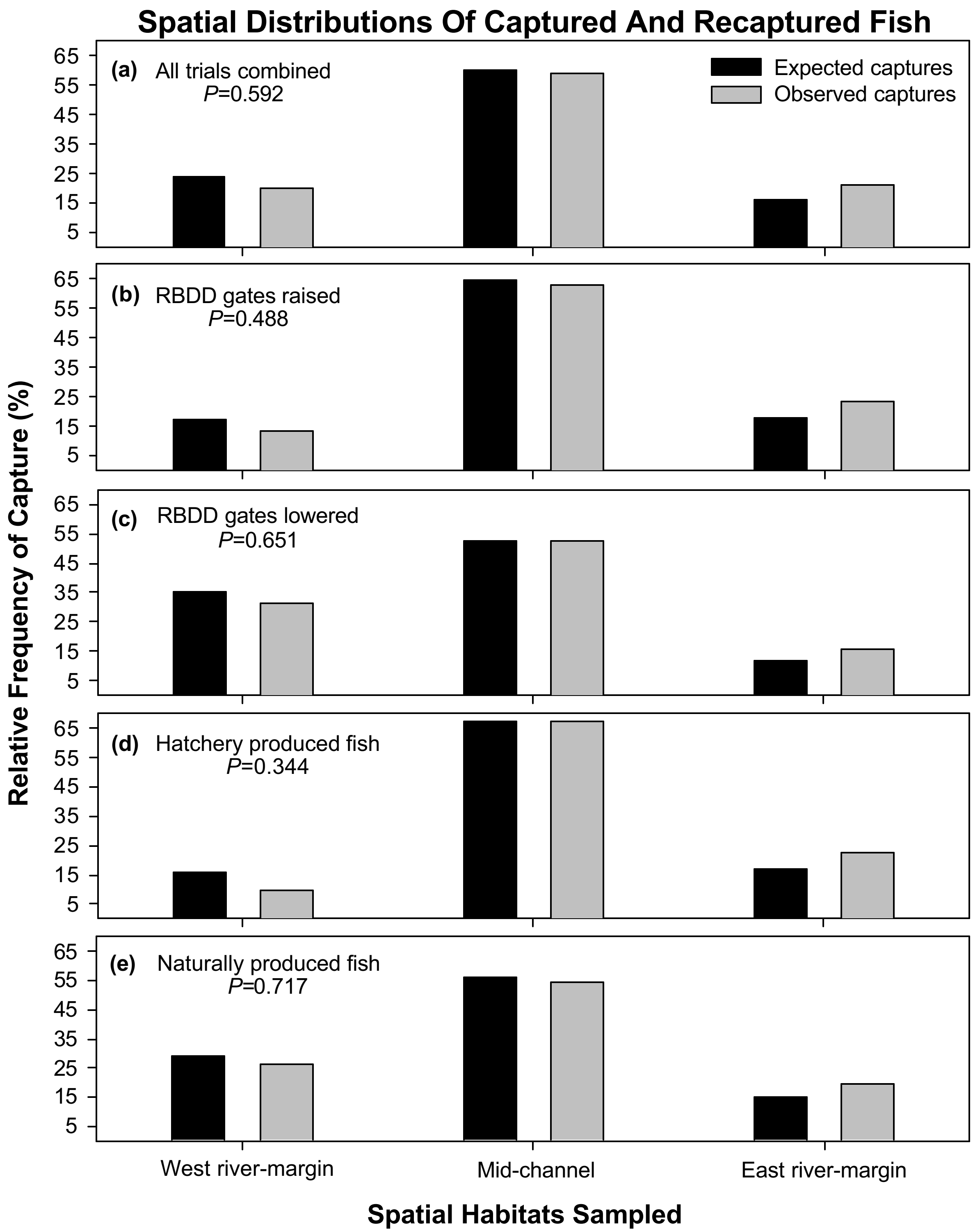

Figure 6. Spatial distributions of expected (unmarked fish) and observed (marked fish) fish captured by rotary-screw traps at Red Bluff Diversion Dam (RK391), Sacramento River, CA. We tested the assumption that marked and released fish were distributed randomly with unmarked fish. No significant differences in the spatial distributions of marked and unmarked fish were detected (P > 0.05, Pearson's Chi-square). Spatial distributions were analyzed for (a) all trials combined (N = ??), (b) hatchery produced fish, (c) naturally produced fish, (d) RBDD gates raised, and (e) RBDD gates lowered. Four trials were omitted from the analyses because spatial strata were not maintained during trials.

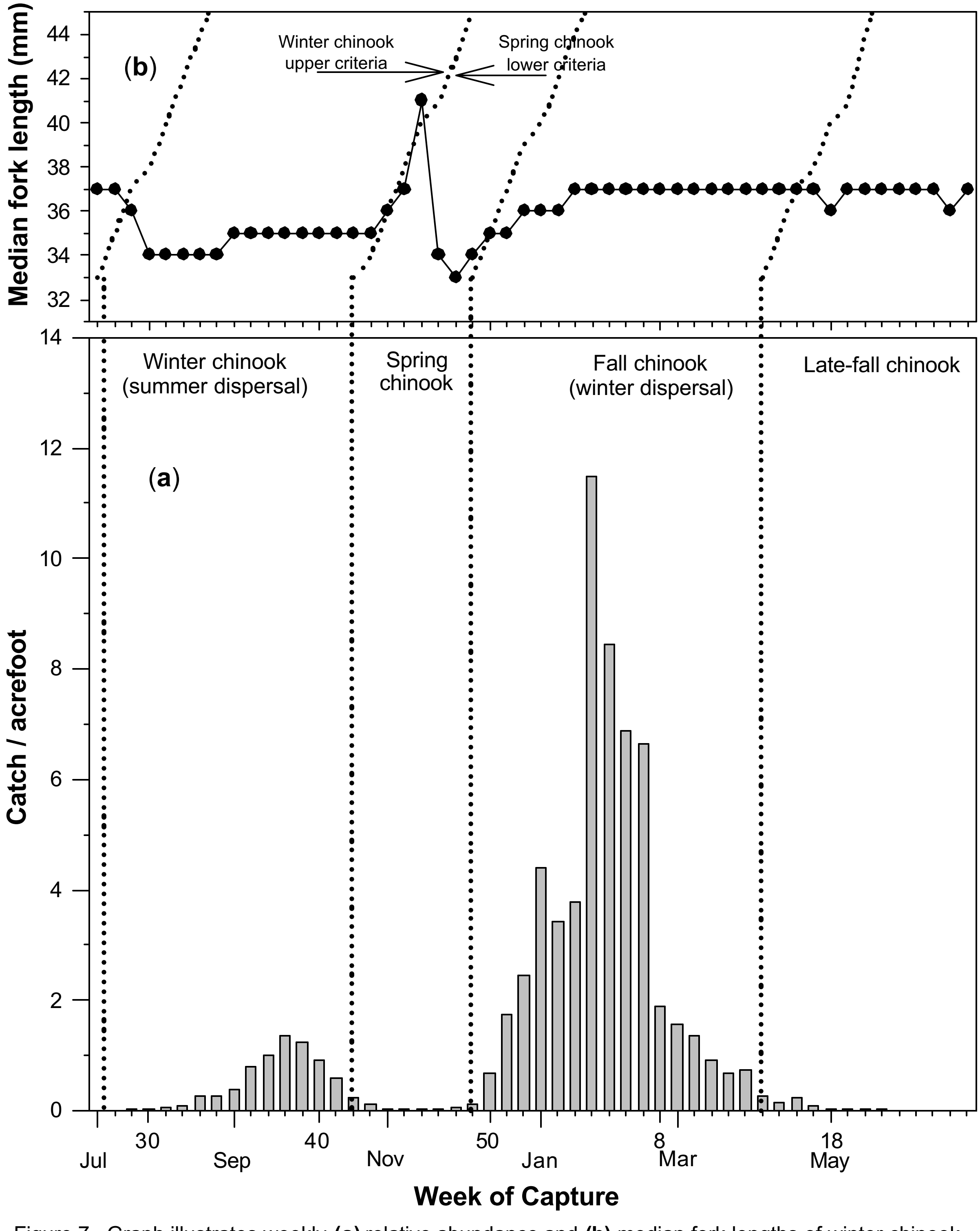

Figure 7. Graph illustrates weekly **(a)** relative abundance and **(b)** median fork lengths of winter chinook salmon captured by rotary-screw traps at Red Bluff Diversion Dam (RK391), Sacramento River, CA. Also presented is the length-at-date criteria (dotted lines, both graphs) developed by Greene (1992) for differentiating between the four “runs” of chinook salmon.

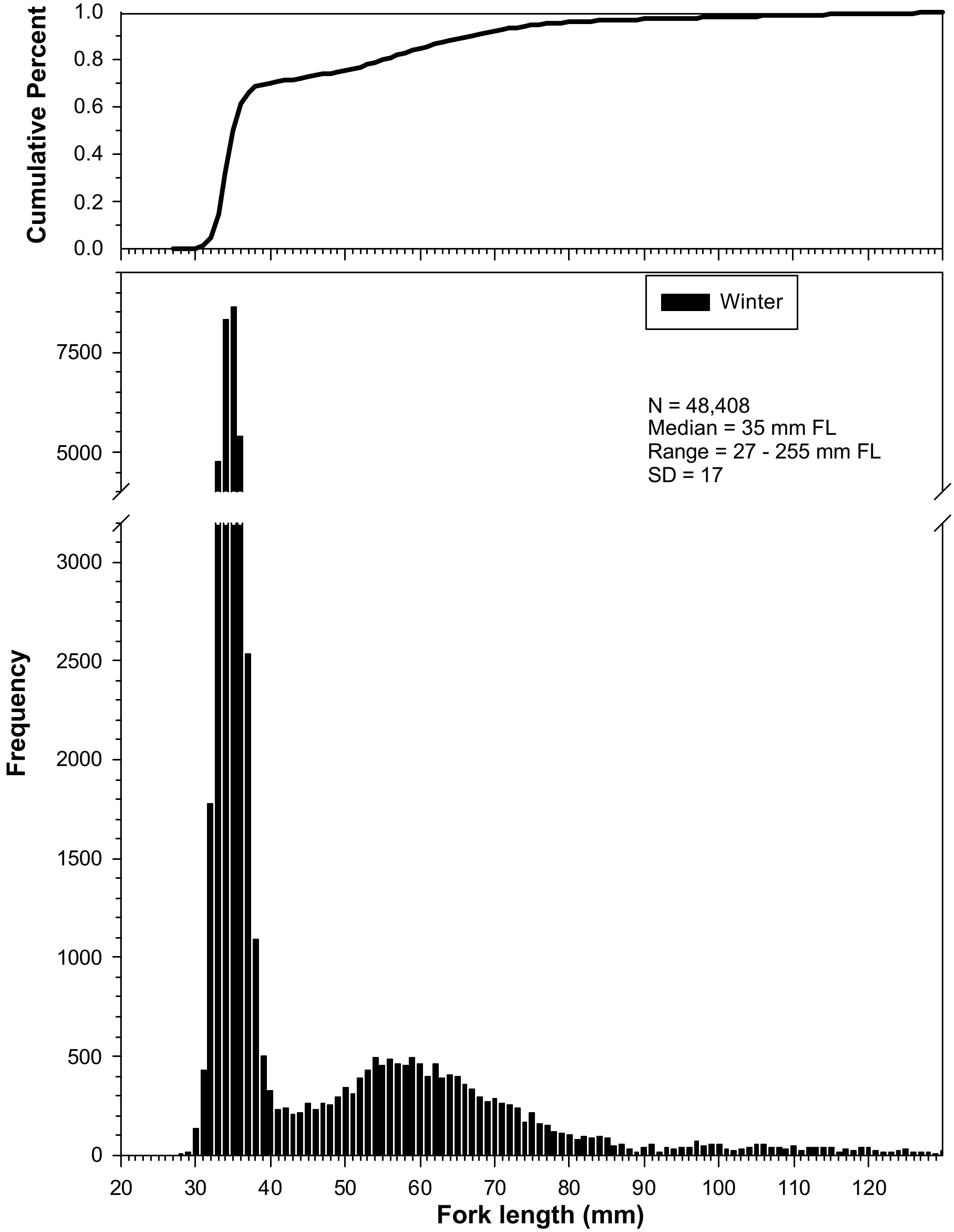

Figure 8.--Length frequency and cumulative frequency distributions for winter chinook salmon (*Oncorhynchus tshawytscha*) captured by rotary-screw traps at Red Bluff Diversion Dam (RK391), Sacramento River, CA. Data summarized from July 1994 through June 2000.

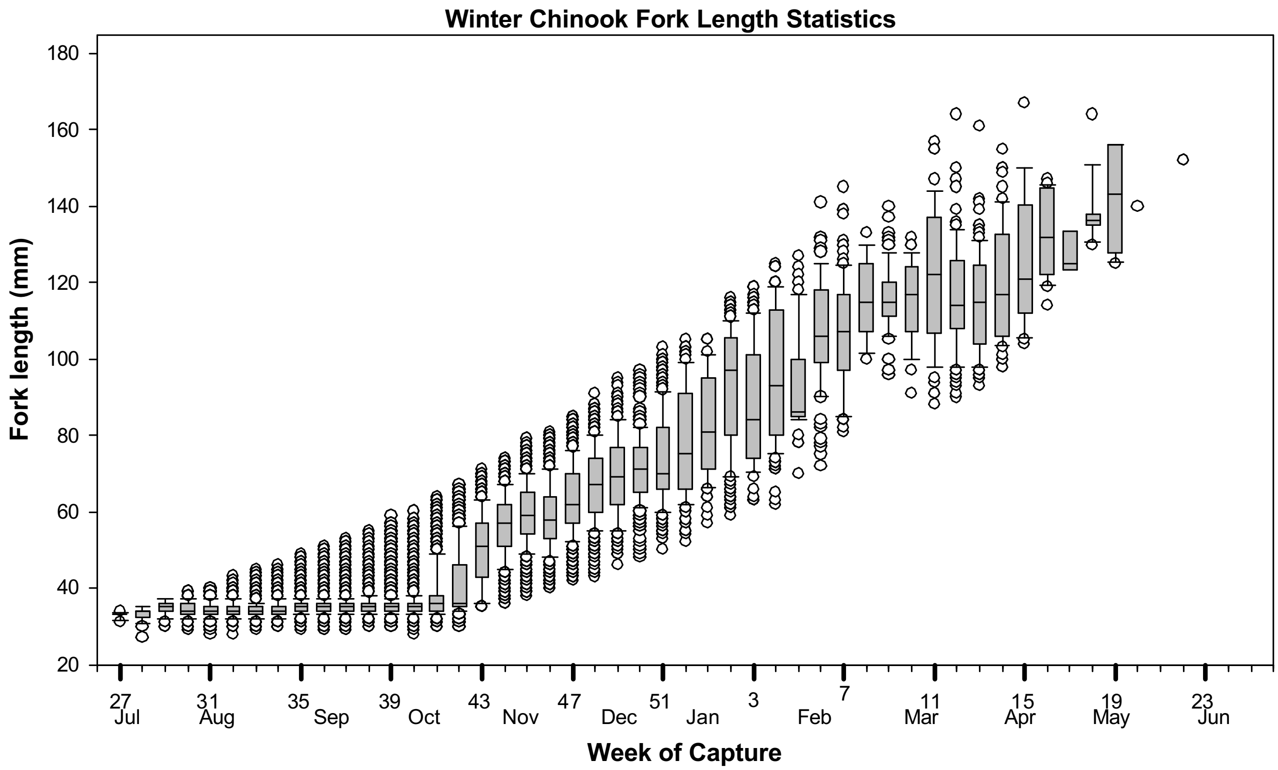

Figure 9. Box plots of weekly length distributions of winter chinook salmon captured by rotary-screw traps at Red Bluff Diversion Dam (RK391), Sacramento River, CA. Box plots denote median fork length (mm), 25th and 75th percentiles and error bars (whiskers) representing the 10th and 90th percentiles. Data points outside error bar boundaries are outliers or single captures.

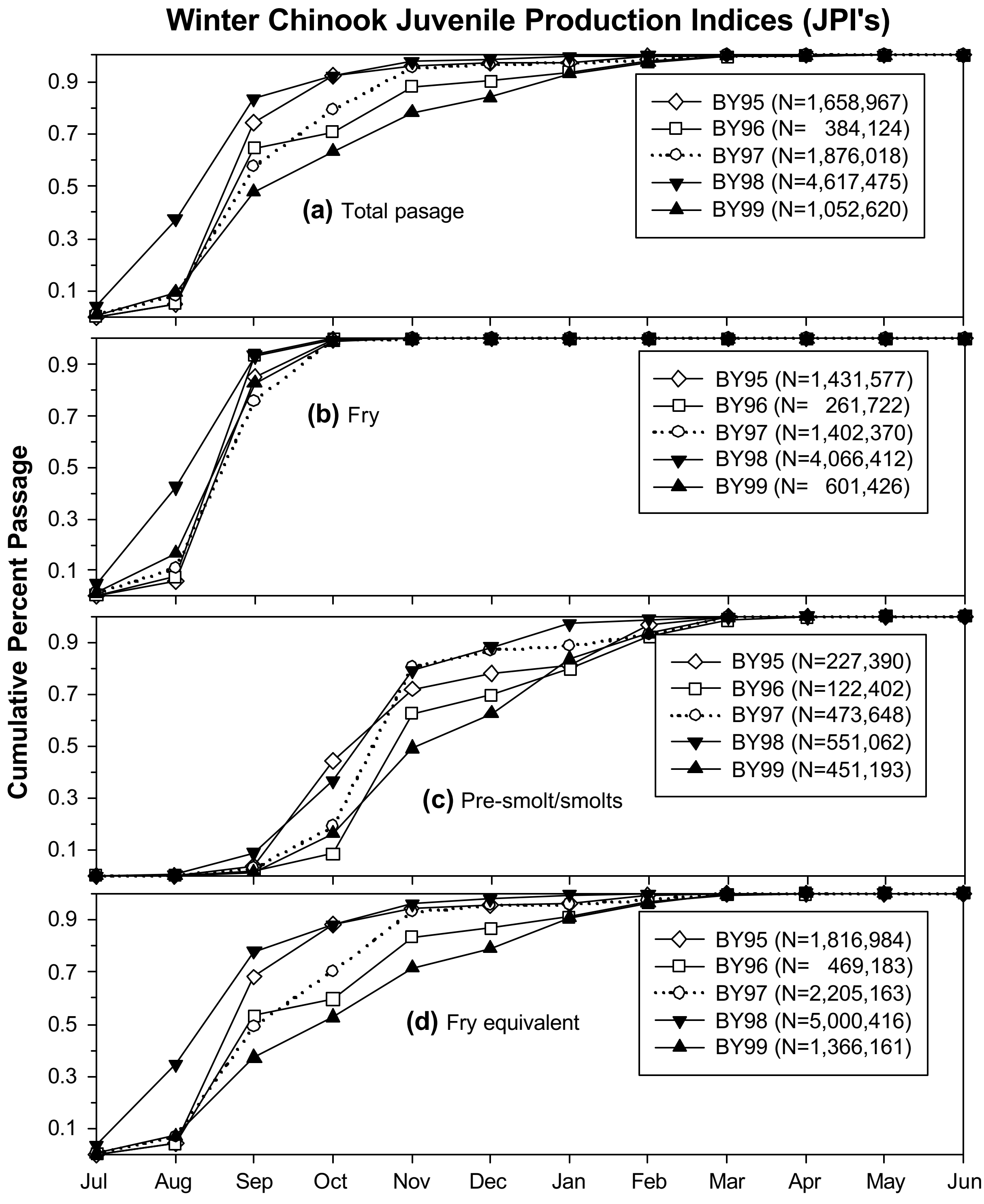

Figure 10. Monthly cumulative juvenile production indices (JPI) for winter chinook salmon captured by rotary-screw traps at Red Bluff Diversion Dam (RK391), Sacramento River, CA. Results include (a) total passage (fry and pre-smolt/smolts combined), (b) fry (≤ 45 mm FL), (c) pre-smolt/smolts (> 45 mm FL) and (d) fry equivalent (fry equivalent was estimated by the addition of fry passage to a weighted pre-smolt/smolt passage equivalent to 59% fry-to-presmolt/smolt survival; Hallock undated).

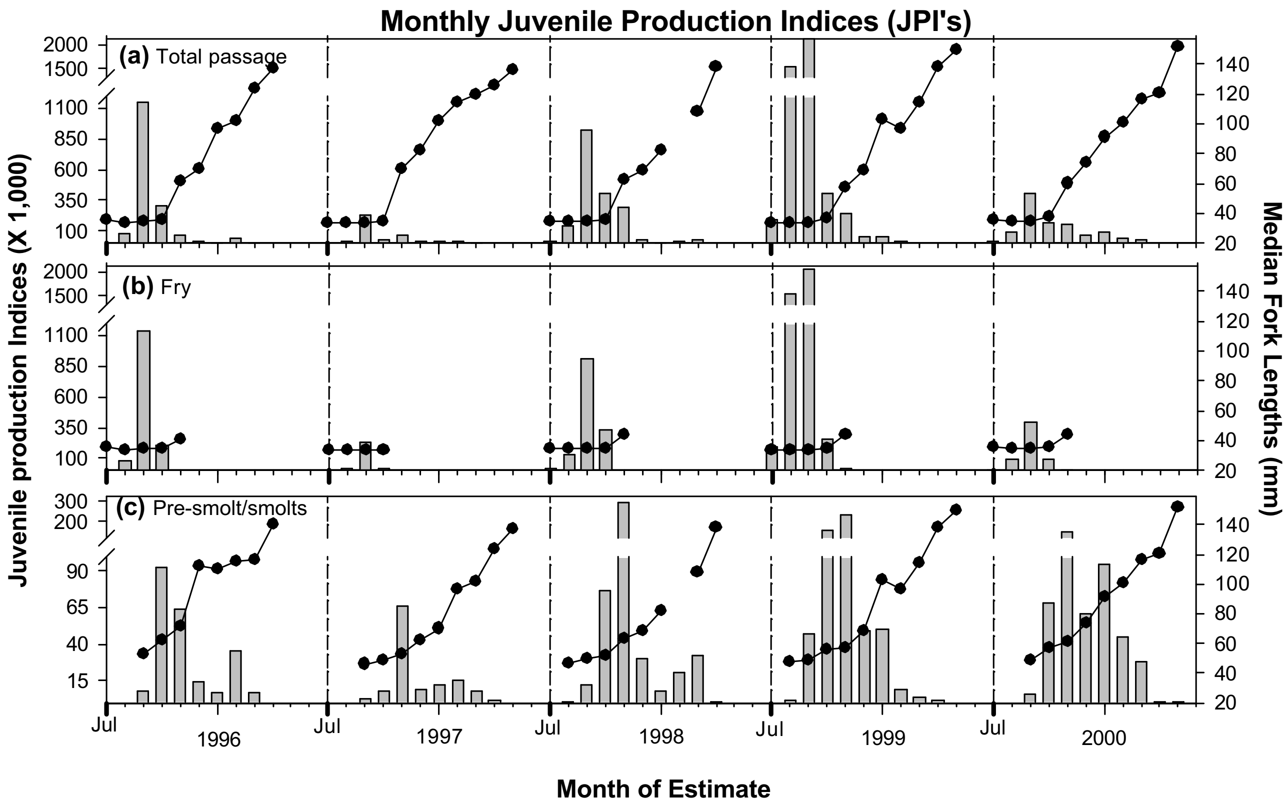

Figure 11. Monthly median fork lengths and juvenile production indices (JPI's) for **(a)** total passage (fry and pre-smolt/smolts combined), **(b)** fry and **(c)** pre-smolt/smolts. Data reported for BY95 (July 1995 through June 1996), BY96 (July 1996 through June 1997), BY97 (July 1997 through June 1998) and BY99 (July 1999 through June 2000). Vertical dashed lines denote separation between brood-years.

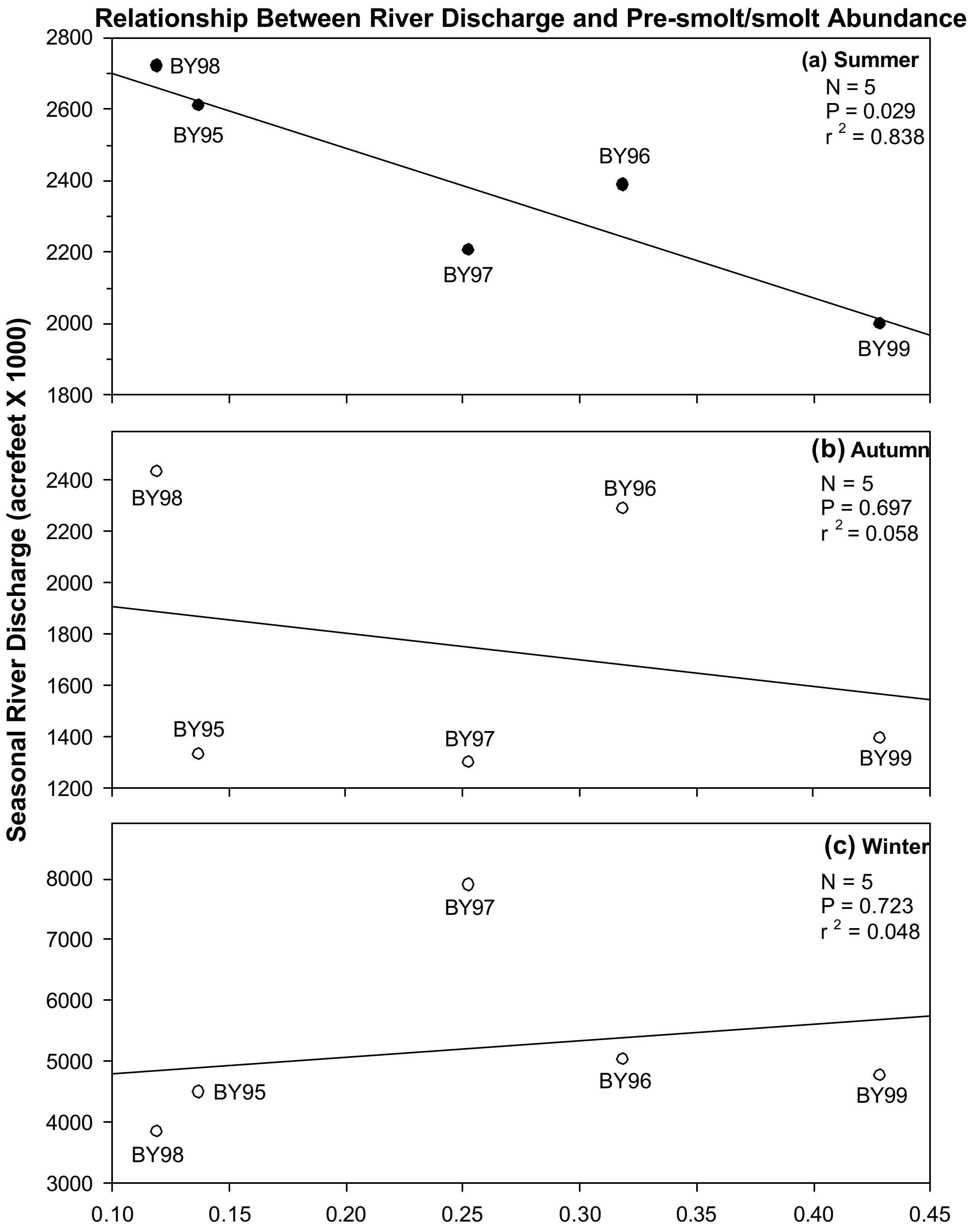

Figure 12. Linear relationships between total river discharge for (a) summer (July, Aug. and Sept.), (b) autumn (Oct., Nov. and Dec.) and (c) winter (Jan., Feb. and Mar.) and the relative abundance (in percent) of winter chinook pre-smolt/smolts passing Red Bluff Diversion Dam, Sacramento River, CA.

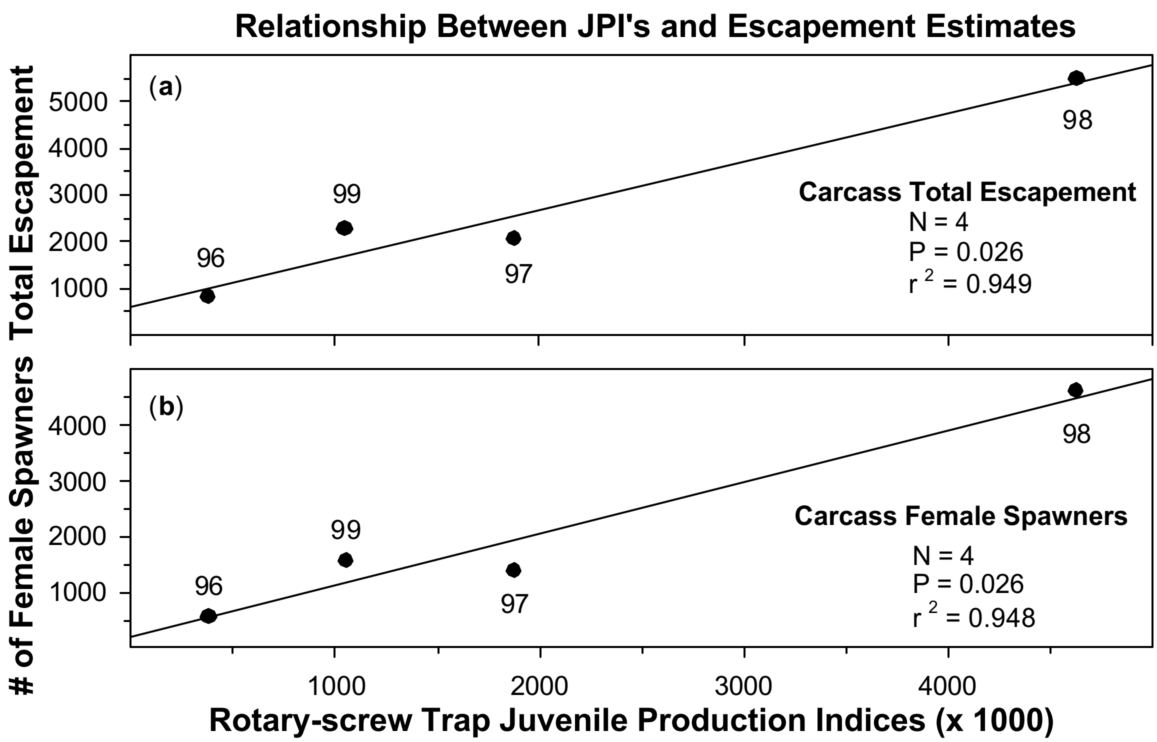

Figure 13. Linear relationship between rotary-screw trap juvenile production indices (JPI's) and **(a)** carcass survey total escapement estimates and **(b)** carcass survey estimates of the number of female spawners (FS).

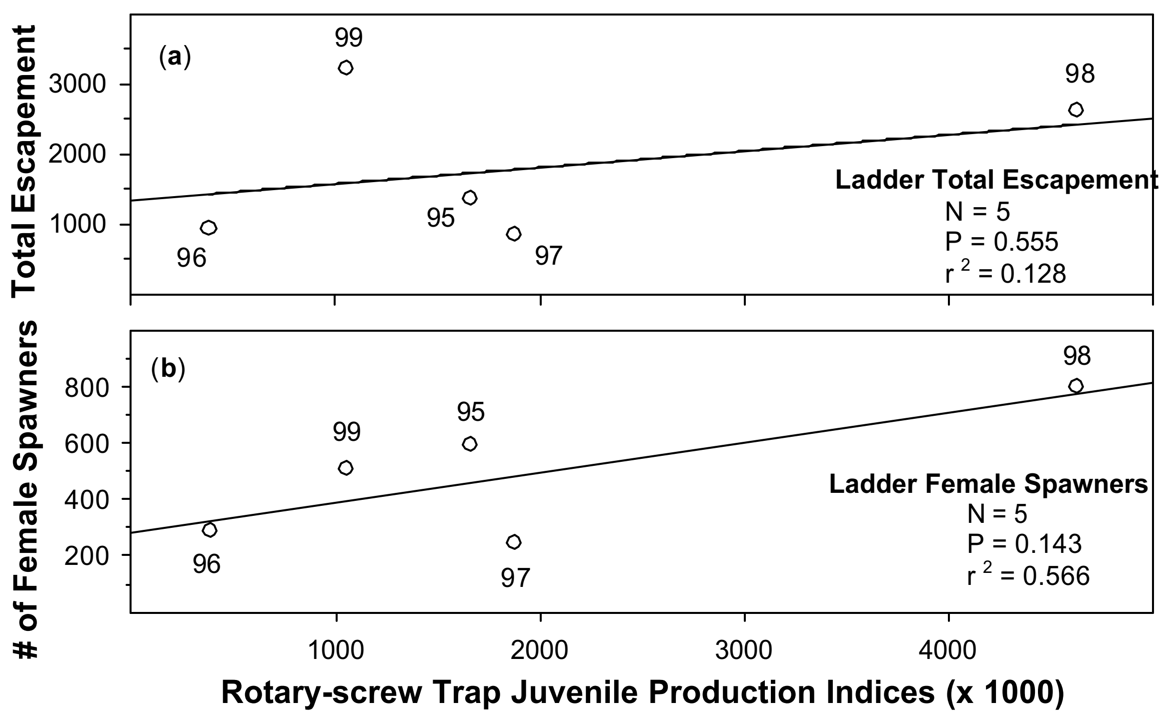

Figure 14. Linear relationship between rotary-screw trap juvenile production indices (JPI's) and RBDD ladder count derived **(a)** total escapement estimates and **(b)** estimates of the number of female spawners (FS).

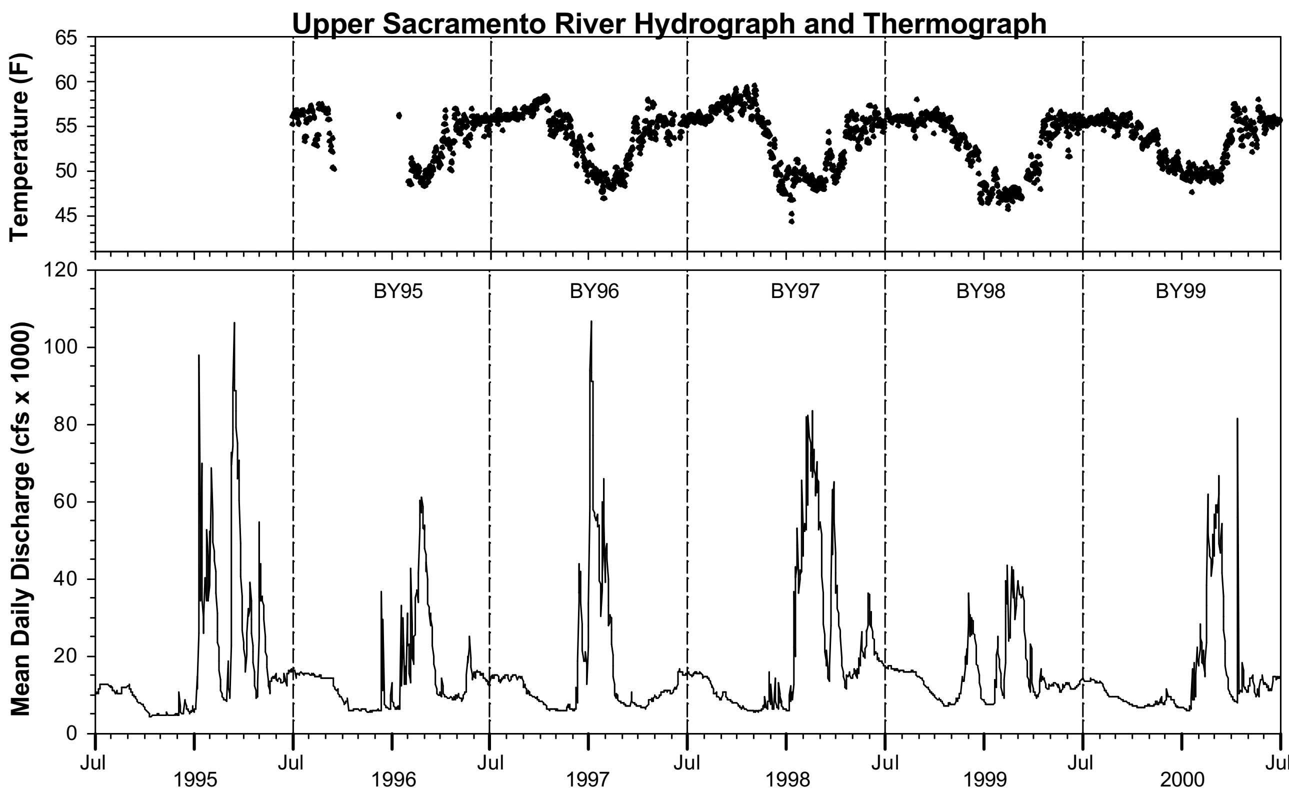

Figure 15.--Mean daily discharge in cubic feet per second (cfs) and water temperature (degrees F) at Bend Bridge (RK413) from July 1994 through June 2000. Time lines for winter-run brood year designation are denoted on top of hydrograph.

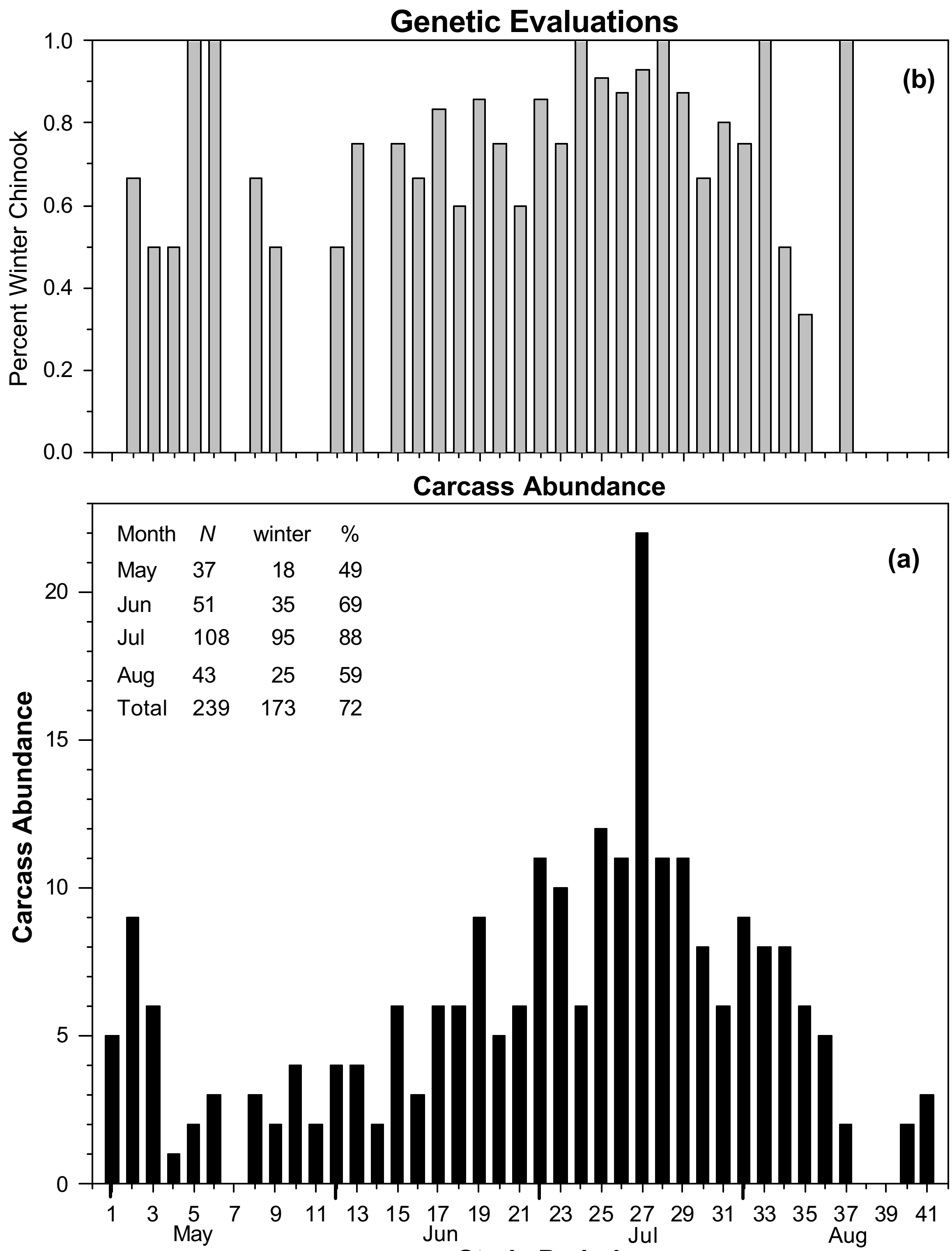

Figure 16.--Number of carcasses recovered (a) and genetic evaluations (b; winter or non-winter) by survey period during the 1997 upper Sacramento River winter-run chinook salmon escapement survey May - August 1997. Seventy-two percent of carcasses were genetically identified as winter-run.