COMPENDIUM REPORT OF RED BLUFF DIVERSION DAM ROTARY TRAP JUVENILE ANADROMOUS FISH PRODUCTION INDICES FOR YEARS 2002-2012

Prepared for:
California Department of Fish and Wildlife Ecosystem Restoration Program
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Compendium Report of Red Bluff Diversion Dam Rotary Trap Juvenile Anadromous Fish Production Indices for Years 2002-2012

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Abstract.— Fall, late-fall, spring, and winter-run Chinook salmon (*Oncorhynchus tshawytscha*) and Steelhead/Rainbow trout (*Oncorhynchus mykiss*) spawn in the Sacramento River and tributaries in California's Central Valley upstream of Red Bluff Diversion Dam (RBDD) throughout the year. Sampling of juvenile anadromous fish at RBDD allows for year-round quantitative production and passage estimates of all runs of Chinook and *O. mykiss*. Incidental capture of Green Sturgeon (*Acipenser medirostris*) and various Lamprey species (*Lampetra spp. and Entosphenus tridentatus)* has occurred throughout juvenile Chinook monitoring activities since 1995. This compendium report addresses, in detail, juvenile anadromous fish monitoring activities at RBDD for the period April 4, 2002 through September 30, 2013.

Sampling was conducted along a transect using four 8-foot diameter rotary-screw traps attached via aircraft cables directly to RBDD. Trap efficiency (i.e., the proportion of the juvenile salmonid population passing RBDD captured by traps) was modeled with percent of river discharge sampled (%*Q*) to develop a simple least-squares regression equation. Chinook and *O. mykiss* passage were estimated by employing the trap efficiency model. The ratio of fry to pre-smolt/smolts passing RBDD was variable among years. Therefore, juvenile passage was standardized to determine juvenile production by estimating a fry-equivalent Juvenile Production Index (JPI) for among-year comparisons. Catch per unit volume (CPUV) was used as an index of relative abundance for Green Sturgeon and Lamprey species. Abiotic data collected or calculated throughout sample efforts included: water temperature, flow, turbidity, and moon illuminosity (fraction of moon illuminated). The abiotic variables were analyzed to determine if relationships existed throughout the migration periods of the anadromous species.

A trap efficiency model developed in 2000 to estimate fish passage demonstrated improved correlation between 2002 and 2013 with the addition of 85 mark-recapture trials. The model's *r*-squared value improved greatly with the addition of numerous mark-recapture trials that used wild fry size-class salmon over a variety of river discharge levels. Total passage estimates including annual effort values with 90% confidence intervals (CI) are presented, by brood year, for each run of Chinook. Fry and pre-smolt/smolt Chinook passage estimates with 90% CI's are summarized annually by run in Appendix 1. Comparisons of relative variation within and between runs of Chinook were performed by calculating Coefficients of Variation (CV). Fall Chinook annual total passage estimates ranged between 6,627,261 and 27,736,868 juveniles for brood years 2002-2012 ($\bar{y}$ = 14,774,923, CV = 46.2%). On average, fall Chinook passage was composed of 74% fry and 26% pre-smolt/smolt size-class fish (SD = 10.3). Late-fall

Chinook annual total passage estimates ranged between 91,995 and 2,559,519 juveniles for brood years 2002-2012 ($\bar{y}$ = 447,711, CV = 159.9%). On average, late-fall Chinook passage was composed of 38% fry and 62% pre-smolt/smolt size-class fish (SD = 22.5). Winter Chinook annual total passage estimates ranged between 848,976 and 8,363,106 juveniles for brood years 2002-2012 ($\bar{y}$ = 3,763,362, CV = 73.2%). On average, winter Chinook passage was composed of 80% fry and 20% pre-smolt/smolt size-class fish (SD = 11.2). Spring Chinook annual total passage estimates for spring Chinook ranged between 158,966 and 626,925 juveniles for brood years 2002-2012 ($\bar{y}$ = 364,508, CV = 45.0%). On average, spring Chinook passage was composed of 54% fry and 46% pre-smolt/smolt size-class fish (SD = 20.0). Annual total passage estimates for *O. mykiss* ranged between 56,798 and 151,694 juveniles for calendar years 2002-2012 ($\bar{y}$ = 116,272, CV = 25.7).

A significant relationship between the estimated number of adult females and fry-equivalent fall Chinook production estimates was detected (r^2 = 0.53, df = 10, P = 0.01). Recruits per female were calculated and ranged from 89 to 1,515 ($\bar{y}$ = 749). Egg-to-fry survival estimates averaged 13.9% for fall Chinook. A significant relationship between estimated number of females and fry-equivalent late-fall Chinook production estimates was detected (r^2 = 0.67, df = 10, P = 0.002). Recruits per female were calculated and ranged from 47 to 243 ($\bar{y}$ = 131). Egg-to-fry survival estimates averaged 2.8% for late-fall Chinook. A significant relationship between estimated number of females and fry-equivalent winter Chinook production estimates was detected (r^2 = 0.90, df = 10, P < 0.001). Recruits per female were calculated and ranged from 846 to 2,351 ($\bar{y}$ = 1,349). Egg-to-fry survival estimates averaged 26.4% for winter Chinook. No significant relationship between estimated number of females and fry-equivalent spring Chinook production estimates was detected (r^2 = 0.00, df = 10, P = 0.971). Recruits per female were calculated and ranged from 1,112 to 8,592 ($\bar{y}$ = 3,122). Egg-to-fry survival estimates averaged 61.5% for spring Chinook. Spring Chinook juvenile to adult correlation values appear unreasonable and well outside those found for other runs and from other studies.

Catch of Green Sturgeon was highly variable, not normally distributed and ranged between 0 and 3,701 per year (median = 193). Catch was primarily composed of recently emerged, post-exogenous feeding larvae. The 10-year median capture total length averaged 27.3 mm (SD = 0.8). Green Sturgeon annual CPUV was typically very low and ranged from 0.0 to 20.1 fish/ac-ft ($\bar{y}$ = 2.5 fish/ac-ft, SD = 5.9). Data were positively skewed and median annual CPUV was 0.8 fish/ac-ft.

Lamprey species sampled included adult and juvenile Pacific Lamprey (*Entosphenus tridentatus*) and to a much lesser extent River Lamprey (*Lampetra ayresi*) and Pacific Brook Lamprey (*Lampetra pacifica*). Unidentified lamprey ammocoetes and Pacific Lamprey composed 99.8% of all captures, 24% and 75%, respectively. River Lamprey and Pacific Brook Lamprey composed the remaining 0.2%, combined. Lamprey captures occurred throughout the year between October and September. Lamprey ammocoete annual relative abundance ranged from 3.6 to 11.7 fish/ac-ft ($\bar{y}$ = 6.8 fish/ac-ft, SD = 2.6). Overall, these data were normally distributed as median annual CPUV was 6.5 fish/ac-ft, similar to the mean value. Pacific Lamprey macropthalmia annual relative abundance was generally higher than ammocoete relative abundance and ranged from 2.1 to 112.8 fish/ac-ft ($\bar{y}$ = 41.0 fish/ac-ft, SD = 34.7). Overall, Pacific Lamprey data was slightly positively skewed and median CPUV was 34.1 fish/ac-ft.

Tabular summaries of the abiotic conditions encountered during each annual capture period were summarized for each run of salmon, *O. mykiss*, Green Sturgeon and Lamprey species. The range of temperatures experienced by Chinook fry and pre-smolt/smolts in the last 11 years of passage at RBDD have been within the optimal range of temperature tolerances for juvenile Chinook survival. Green Sturgeon have likely benefitted from temperature management efforts aimed at winter Chinook spawning and production, albeit less comprehensively. Lamprey species have also likely benefitted from temperature management as temperatures for early life stages of Lamprey in the mainstem Sacramento River appear to have been, on average, optimal in the last 11 years.

The relationship between river discharge, turbidity, and fish passage are complex in the Upper Sacramento River where ocean and stream-type Chinook of various size-classes (i.e., runs, life stages and ages) migrate daily throughout the year. Fish passage increases often coincided with an increase in turbidity which were sampled more effectively than increases in river discharge. A positive bias of fish passage estimates may result if the peak turbidity event was sampled following an un-sampled peak flow event. The importance of the first storm event of the fall or winter period cannot be overstated. Smolt passage and juvenile Lamprey passage increase exponentially and fry passage can be significant during fall storm events.

Rotary trap passage data indicated fry size-class winter Chinook exhibit decreased nocturnal passage levels during and around the full moon phase in the fall. Pre-smolt/smolt winter Chinook appeared less influenced by nighttime light levels and much more influenced by changes in discharge levels. Spring, fall and late-fall Chinook fry exhibited varying degrees of decreased passage during full moon periods, albeit storms and related hydrologic influx dominated peak migration periods.

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Introduction

The United States Fish and Wildlife Service (USFWS) has conducted direct monitoring of juvenile Chinook salmon (*Oncorhynchus tshawytscha)* passage at Red Bluff Diversion Dam (RBDD; RM 243) on the Sacramento River, CA since 1994 (Johnson and Martin 1997). Martin et al. (2001) developed quantitative methodologies for indexing juvenile Chinook passage using rotary-screw traps to assess the impacts of the RBDD Research Pumping Plant. Absolute abundance (production and passage) estimates were needed to determine the level of impact from the entrainment of salmonids and other fish community populations through experimental 'fish friendly' Archimedes and internal helical pumps (Borthwick and Corwin 2001). The original project objectives were met by 2000 and funding of the project was discontinued.

In 2001, funding was secured through a CALFED Bay-Delta Program grant for three years of annual monitoring operations to determine the effects of restoration activities in the Upper Sacramento River aimed primarily at winter Chinook[1] salmon. Through various amendments, extensions, and grant approvals by the CALFED Ecosystem Restoration Program, the State of California based funding source lasted until 2008. At this point, the State of California defaulted on their funding agreement and internal USFWS funding sources through the Central Valley Project Improvement Act (CVPIA) bridged the gap for a period of time until State funding was restored. The US Bureau of Reclamation, the primary proponent of the Central Valley Project (CVP) of which this project provides monitoring and abundance trend information, has funded this project since 2010 due to regulatory requirements contained within the Biological Opinion for the Operations and Criteria Plan for the CVP (NMFS 2009).

Protection, restoration, and enhancement of anadromous fish populations in the Sacramento River and its tributaries is an important element of the CVPIA Section 3402. The CVPIA has a specific goal to double populations of anadromous fishes in the Central Valley of California. Juvenile salmonid production monitoring is an important component authorized under Section 3406 (b)(16) of CVPIA and has funded many anadromous fish restoration actions which were outlined in the CVPIA Anadromous Fisheries Restoration Program (AFRP) Working Paper (USFWS 1995), and Draft Restoration Plan (USFWS 1997; finalized in 2001).

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[1] The National Marine Fisheries Service first listed Winter-run Chinook salmon as threatened under the emergency listing procedures for the ESA (16 U.S.C.R. 1531-1543) on August 4, 1989 (54 FR 32085). A proposed rule to add winter Chinook salmon to the list of threatened species beyond expiration of the emergency rule was published by the NMFS on March 20, 1990 (55 FR 10260). Winter Chinook salmon were formally added to the list of federally threatened species by final rule on November 5, 1990 (55 FR 46515), and they were listed as a federally endangered species on January 4, 1994 (59 FR 440). Critical habitat for winter Chinook salmon has been designated from Keswick Dam (RM 302) to the Golden Gate Bridge (58 FR 33212; June 16, 1993). Winter Chinook salmon have been listed as endangered under the CESA since September 22, 1989 (California Code of Regulations, Title XIV, Section 670.5). Their federal endangered status was reaffirmed in June 2005 (70 FR 37160).

Since 2002, the USFWS rotary trap winter Chinook juvenile production indices (JPI's) have primarily been used in support of production estimates generated from carcass survey derived adult escapement data using the National Oceanic and Atmospheric Administration's (NOAA) Juvenile Production Estimate Model. Martin et al. (2001) stated that RBDD was an ideal location to monitor juvenile winter Chinook production because (1) the spawning grounds occur almost exclusively above RBDD (Vogel and Marine 1991; Snider et al. 1997, USFWS 2011), (2) multiple traps could be attached to the dam and sample simultaneously across a transect, and (3) operation of the dam could control channel morphology and hydrological characteristics of the sampling area providing for consistent sampling conditions for purposes of measuring juvenile fish passage.

Fall, late-fall, spring, and winter-run Chinook salmon and Steelhead/Rainbow Trout (*Oncorhynchus mykiss*) spawn in the Sacramento River and tributaries upstream of RBDD throughout the year resulting in year-round juvenile salmonid passage (Moyle 2002). Sampling of juvenile anadromous fish at RBDD allows for year-round quantitative production and passage estimates of all runs of Chinook and Steelhead/Rainbow trout. Timing and abundance data have been provided in real-time for fishery and water operations management purposes of the CVP since 2004[2]. Since 2009, confidence intervals, indicating uncertainty in weekly passage estimates, have been included in real-time bi-weekly reports to allow better management of available water resources and to reduce impact of CVP operations on both federal Endangered Species Act (ESA) listed and non-listed salmonid stocks. Currently, Sacramento River winter Chinook are ESA listed as endangered. Central Valley spring Chinook and Central Valley Steelhead (hereafter *O. mykiss)* are listed as threatened within the Central Valley Endangered Species Unit.

Incidental capture of Green Sturgeon (*Acipenser medirostris*) and various Lamprey species (*Lampetra spp. and Entosphenus sp.)* has occurred throughout juvenile Chinook monitoring activities at RBDD since 1995 (Gaines and Martin 2002). Although rotary traps were designed to capture outmigrating salmonid smolts, data from the incidental capture of sturgeon and lamprey species has become increasingly relied upon for basic life-history information and as a measure of relative abundance and species trend data. The Southern distinct population segment of the North American Green Sturgeon was proposed for listing as threatened under the Federal ESA on April 7, 2006 (FR 17757) which then took effect June 6, 2006. Pacific Lamprey (*Entosphenus tridentatus*) are thought to be extirpated from at least 55% of their historical habitat and have been recognized by the USFWS as a species needing a comprehensive plan to conserve and restore these fish (Goodman and Reid 2012).

The objectives of this compendium report are to: (1) summarize the estimated abundance of all four runs of Chinook salmon and *O. mykiss* passing RBDD for brood

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[2] Real-time biweekly reports located for download at: http://www.fws.gov/redbluff/rbdd_biweekly_final.html

years (BY) 2002 through 2012, (2) estimate annual relative abundance of Green Sturgeon and Lamprey species production for eleven consecutive years, (3) define temporal patterns of abundance for all anadromous species passing RBDD, (4) correlate juvenile salmon production with adult salmon escapement estimates, (5) perform exploratory data analyses of potential environmental covariates driving juvenile fish migration trends, and (6) describe various life-history attributes of anadromous juvenile fish produced in the Upper Sacramento River as determined through long-term monitoring efforts at RBDD.

This compendium report addresses, in detail, our juvenile anadromous fish monitoring activities at RBDD for the period April 4, 2002 through September 30, 2013. This report includes JPI's and relative abundance estimates for the 2002-2012 brood year emigration periods and will be submitted to the California Department of Fish and Wildlife to comply with contractual reporting requirements for Ecosystem Restoration Program Grant Agreement Number P0685507 and to the US Bureau of Reclamation who funded in part or in full the surveys from years 2008 through 2013 (Interagency Agreement No. R10PG20172).

Study Area

The Sacramento River originates in Northern California near Mt. Shasta from the springs of Mt. Eddy (Hallock et al. 1961). It flows south through 370 miles of the state draining numerous slopes of the coast, Klamath, Cascade, and Sierra Nevada ranges and eventually reaches the Pacific Ocean via San Francisco Bay (Figure 1). Shasta Dam and its associated downstream flow regulating structure, Keswick Dam, have formed a complete barrier to upstream anadromous fish passage since 1943 (Moffett 1949). The 59-river mile (RM) reach between Keswick Dam (RM 302) and RBDD (RM 243) supports areas of intact riparian vegetation and largely remains unobstructed. Within this reach, several major tributaries to the Sacramento upstream of RBDD support various Chinook salmon spawning populations. These include Clear Creek and Cottonwood Creek (including Beegum Creek) on the west side of the Sacramento River and Cow, Bear, Battle and Payne's Creek on the east side (Figure 1). Below RBDD, the river encounters greater anthropogenic impacts as it flows south to the Sacramento-San Joaquin Delta. Impacts include, but are not limited to, channelization, water diversion, agricultural and municipal run-off, and loss of associated riparian vegetation.

RBDD is located approximately 1.8 miles southeast of the city of Red Bluff, California (Figure 1). The dam is 740-feet (ft) wide and composed of eleven, 60-ft wide fixed-wheel gates. Between gates are concrete piers 8-ft in width. The USBR's dam operators were able to raise the RBDD gates allowing for run-of-the-river conditions or lower them to impound and divert river flows into the Tehama-Colusa and Corning canals. USBR operators generally raised the RBDD gates from September 16 through May 14 and lowered them May 15 through September 15 during the years 2002-2008. As of the spring of 2009, the RBDD gates were no longer lowered prior to June 15 and were raised by the end of August or earlier (NMFS 2009) in an effort to reduce the impact to spring Chinook salmon and Green Sturgeon. Since the fall of 2011, the RBDD gates have been left in the raised position allowing unobstructed upstream and downstream passage of adult and juvenile anadromous fish. The RBDD has been replaced by a permanent pumping plant upstream of the RBDD and the facilities have been relinquished to the Tehama Colusa Canal Authority as of spring 2012. Mothballing of the RBDD infrastructure was scheduled to occur in 2014.

Methods

Sampling Gear.—Sampling was conducted along a transect using four 8-ft diameter rotary-screw traps (E.G. Solutions® Corvallis, Oregon) attached via aircraft cables directly to RBDD. The horizontal placement of rotary traps across the transect varied throughout the study but generally sampled in the river-margin (east and west river-margins) and mid-channel habitats simultaneously (Figure 2). Rotary traps were positioned within these *spatial zones* unless sampling equipment failed, river depths were insufficient (< 4-ft), or river hydrology restricted our ability to sample with all traps (water velocity < 2.0 ft/s).

Sampling Regimes.—In general, rotary traps sampled continuously throughout 24-hour periods and samples were processed once daily. During periods of high fish abundance, elevated river flows, or heavy debris loads, traps were sampled multiple times per day, continuously, or at randomly pre-selected periods to reduce incidental mortality. When abundance of Chinook was very high, sub-sampling protocols were implemented to reduce listed species take and incidental mortality in accordance with National Marine Fisheries Service (NMFS) Section 10(a)(1)(A) research permit terms and conditions. The specific sub-sampling protocol implemented was contingent upon the number of Chinook captured or the probability of successfully sampling various river conditions. Initially, rotary trap cones were structurally modified to only sample one-half of the normal volume of water entering the cones (Gaines and Poytress 2004). If further reductions in capture were needed, the number of traps sampled was reduced from four to three. During storm events and associated elevated river discharge levels, each 24-hour sampling period was divided into four or six non-overlapping strata and one or two strata was randomly selected for sampling (Martin et al 2001). Estimates were extrapolated to un-sampled strata by dividing catch by the strata-selection probability (i.e., P = 0.25 or 0.17). If further reductions in effort were needed or river conditions were intolerable, sampling was discontinued or not conducted. When days or weeks were unable to be sampled, mean daily passage estimates were imputed for missed days based on weekly or monthly mean daily estimates (i.e., interpolated).

Data Collection.—All fish captured were anesthetized, identified to species, and enumerated with fork lengths (FL) measured to the nearest millimeter (mm). When capture of Chinook juveniles exceeded approximately 200 fish/trap, a random sub-sample of the catch to include approximately 100 individuals was measured, with all additional fish being enumerated and recorded. Chinook salmon race was assigned using length-at-date criteria developed by Greene[3] (1992). Juvenile salmon were assigned to a fry or pre-smolt/smolt life stage based on their fork length. Individuals ≤ 45 mm were classified as fry, and individuals ≥ 46 mm were classified as pre-smolt/smolts.

O. mykiss between 80 and 200-mm fork length were weighed to the nearest gram using a digital scale with a stated accuracy of +/- 0.5 grams. This size range was selected to reduce the influence of measurement error for fish lengths <80 mm (Pope and Kruse 2007). Additionally, state and federal permit regulations restricted the use of anesthetizing agents for fish that may be consumed by the public (i.e., fish >200mm). *O. mykiss* were visually assessed and assigned a life-stage rating based on morphological features following protocols developed by the Comprehensive Assessment and Monitoring Program (CAMP; USFWS 1997). Furthermore, *O. mykiss* annual weight-length regression coefficients were generated by transforming (Log_{10}) the weight and fork length data to create a linear regression equation:

$$Log_{10}(\text{Total Weight}) = b(Log_{10}\text{Fork Length}) + a$$

Confidence interval overlap between the annual slope coefficients was used to test if the annual *O. mykiss* growth rates between years were significantly different (Pope and Kruse 2007). If the 95% confidence intervals around any two slope coefficients did not overlap they were considered significantly different.

Green Sturgeon and Lamprey species were measured for total length (TL) to the nearest mm. Identification of Green Sturgeon larvae was possible based on meristics for individuals > 46 mm TL and assumed for all individuals <46 mm[4]. Lamprey species were identified to the genus level during the ammocoete stage and described as ammocoetes. Adult and macropthalmia (eyed juveniles) were identified to the genus and species level using dentition patterns, specifically by the number of inner lateral horny plates on the sucking disk (Moyle 2002).

Trap Effort. Data quantifying effort by each rotary trap were collected at each trap sampling and included the length of time each trap sampled (expressed as sample weight with 1440 minutes equal to 1.0 for 24-hour samples), water velocity immediately in front of the cone at a depth of 2-ft, and depth of cone "opening" submerged. Water velocity was measured using a General Oceanic® Model 2030 flowmeter. These data collectively were used to calculate the estimated volume of water sampled by traps (X_i)

[3] Generated by Sheila Greene, California Department of Water Resources, Environmental Services Office, Sacramento (May 8, 1992) from a table developed by Frank Fisher, California Department of Fish and Game, Inland Fisheries Branch, Red Bluff (revised February 2, 1992). Fork lengths with overlapping run assignments were placed with the latter spawning run.

[4] To confirm the identification of larval sturgeon, samples were transferred to UC Davis to be grown-out between 1996 and 1997 (Gaines and Martin 2002) and annual subsamples of larvae were sent to UC Davis for genetic analyses between 2003 and 2012 (Israel et al 2004, Israel and May 2010). To date, all samples have been confirmed to be Green Sturgeon.

in acre-feet (ac-ft). Trap effort data were then standardized to a sample weight of 1.0 for within- and between-day comparisons. Individual (X_i) data were summed for the number of traps operating within a 24-hour sample period to estimate daily water volume sampled (X_d). The percent river volume sampled by traps (%Q_d) was estimated as the ratio of river volume sampled (X_d) to total river volume passing RBDD in acre-feet. River volume (Q_d) was obtained from the United States Geological Survey gauging station at Bend Bridge at RM 258 (USGS site no. 11377100, http://waterdata.usgs.gov/usa/nwis/uv?site_no=11377100). Daily river volume at RBDD was adjusted from Bend Bridge river flows by subtracting daily RBDD diversions, when applicable.

Sampling Effort.— Annual rotary trap sampling effort was quantified by assigning a value of 1.00 to a sample consisting of four, 8-ft diameter rotary-screw traps sampling 24 hours daily, three hundred and sixty-five days a year. Annual values <1.00 represent occasions where less than four traps were sampling, traps were structurally modified to sample only one-half the normal volume of water, or when less than the entire year were sampled. Annual passage estimate effort was calculated by summing the total number of days passage was estimated, based on 3 or 4 traps sampling (minimum required to generate passage estimate; Martin et al. 2001), and divided by the sum of the annual total number of days sampled plus the number of days unsampled.

Mark-Recapture Trials.— Chinook collected as part of daily samples were marked with bismark brown staining solution (Mundie and Traber 1983) prepared at a concentration of 21.0 mg/L of water. Fish were stained for a period of 45-50 minutes, removed, and allowed to recover in fresh water. Marked fish were held for 6-24 hours before being released 2.5-miles upstream from RBDD after official sunset. Recapture of marked fish was recorded for up to five days after release. Trap efficiency was calculated based on the proportion of recaptures to total fish released (i.e., mark-recapture trials). Trials were conducted as fish numbers and staffing levels allowed under a variety of river discharge levels and trap effort combinations.

Trap Efficiency Modeling.— To develop a trap efficiency model, mark-recapture trials were conducted as noted above. Estimated trap efficiency (i.e., the proportion of the juvenile population passing RBDD captured by traps; $\hat{T}_d$) was modeled with %*Q* to develop a simple least-squares regression equation (eq. 5). The equation (slope and intercept) was then used to calculate daily trap efficiencies based on daily estimated river volume sampled. Each successive year of mark-recapture trials were added annually to the original trap efficiency model developed by Martin et al. (2001) on July 1 of each year.

Daily Passage Estimates ($\hat{P}_d$).—The following procedures and formulae were used to derive daily and weekly estimates of total numbers of unmarked Chinook and *O. mykiss* passing RBDD. We defined C_{di} as catch at trap *i* (*i* = 1,…,*t*) on day *d* (*d* = 1,…,*n*), and X_{di} as volume sampled at trap i (i = 1,…t) on day d (d = 1,…n). Daily salmonid catch and water volume sampled were expressed as:

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

and,

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

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

3. $$\%\hat{Q}_d = \frac{X_d}{Q_d}$$

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

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

where,

5. $$\hat{T}_d = (a)(\%\hat{Q}_d) + b$$

and, $\hat{T}_d =$ estimated trap efficiency on day d.

Weekly Passage ($\hat{P}$).—Population totals for numbers of Chinook and *O. mykiss* passing RBDD each week were derived from $\hat{P}_d$ where there are N days within the week:

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

Estimated Variance.—

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

The first term in eq. 7 is associated with sampling of days within the week.

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

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

9. $$Var(\hat{P}_d) = \frac{\hat{P}_d(1-\hat{T}_d)}{\hat{T}_d} + Var(\hat{T}_d)\frac{\hat{P}_d(1-\hat{T}_d) + \hat{P}_d{}^2\hat{T}_d}{\hat{T}_d^3}$$

where,

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

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

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

where,

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

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

Confidence intervals (CI) were constructed around $\hat{P}$ using eq. 13.

13. $$P \pm t_{\alpha/2,n-1}\sqrt{Var(\hat{P})}$$

Annual JPI's were estimated by summing $\hat{P}$ across weeks.

14. $$JPI = \sum_{week=1}^{52}\hat{P}$$

Fry-Equivalent Chinook Production Estimates.—The ratio of Chinook fry (<46 mm FL) to pre-smolt/smolts (>45 mm FL) passing RBDD was variable among years. Therefore, we standardized juvenile production by estimating a fry-equivalent JPI for among-year comparisons. Fry-equivalent JPI's were estimated by the summation of fry JPI and a weighted (1.7:1) pre-smolt/smolt JPI (inverse value of 59% fry-to-presmolt/smolt survival; Hallock undated). Rotary trap JPI's could then be directly compared to determine variability in production between years.

Relative Abundance.—Catch per unit volume (CPUV; Gaines and Martin 2002) was used as an index of relative abundance (RA) for Green Sturgeon and Lamprey species at RBDD.

15. $$RA_{dt} = \frac{C_{dt}}{V_{dt}}$$

RA_{dt}	=	relative abundance on day *d* by trap *t* (catch/acre-foot),
C_{dt}	=	number of fish captured on day *d* by trap *t*, and
V_{dt}	=	volume of water sampled on day *d* by trap *t*.

The volume of water sampled (V_{dt}) was estimated for each trap as the product of one-half the cross sectional area (wetted portion) of the cone, water velocity (ft/s) directly in front of the cone at a depth of 2-feet, cone modified (multiplied by 0.5) or not (multiplied by 1.0), and duration of sampling.

Exploratory Data Analyses.—The sampling of four runs of Chinook, *O. mykiss*, Green Sturgeon, and Lamprey occurred over 11 years and a variety of environmental conditions. Abiotic data collected or calculated throughout sample efforts included water temperature, flow, turbidity, and moon illuminosity (fraction of moon illuminated). The abiotic factors were analyzed to determine if patterns or trends existed throughout the migration periods of the various species. Additional statistical analyses were performed, when applicable, and additional methods are noted within the results section for species-specific data trends analyzed.

Results

Sampling Effort.—Annual sampling effort varied throughout the 11-year period of reporting. The reasons for less than 100% effort varied by time of year and run sampled due to numerous factors. These factors can be categorized as either intentional or unintentional decreases in effort. Intentional decreases in effort were primarily due to ESA Section 10(a)1(A) take and incidental mortality limits, the desire to decrease potential impacts to ESA listed fish or hatchery released production groups, or when staffing levels were not appropriate for the conditions encountered. Unintentional decreases in effort were due primarily to storm activity and related debris flows or conditions considered too dangerous to sample. Additionally, during the years RBDD was in operation (2002-2011), many days were not sampled due to operational requirements imposed by USBR operators (e.g., lowering or raising of the dam gates).

Annual sample effort was assigned a value of 1.0 based on sampling four traps 365 days a year. Annual sample effort values by salmonid species and run are described in Table 1. Overall, annual sample effort for all salmonids combined ranged from 0.53 to

0.91 ($\bar{y}$ = 0.80, SD = 0.10) following annual juvenile salmonid brood year cycles. The lowest values corresponded to the year 2002 when sampling did not begin until mid-April of the year. The highest value corresponded to the year 2007 when flow events were mild, staffing levels were optimal, and permit restrictions did not dictate major sampling effort reductions (Table 1).

Mark-Recapture Trials.—Trap efficiency estimates were calculated by conducting mark-recapture trials (Volkhardt et al. 2007) using unmarked salmon collected from daily trap samples. Trials were conducted when trap catch values allowed the release of 1,000 fish per trial, generally, as well as when staffing and river conditions would allow. Mark-recapture trials were also employed to validate daily trap efficiency estimates by comparing actual with predicted (modeled) estimates. This was especially important during peak salmon outmigration periods.

The number of trials conducted each calendar year ranged from 0 in 2010 to 21 in 2004 ($\bar{y}$ = 7.7) and totaled 85 trials between 2002 and 2013 (Table 2). Trials were conducted with four rotary traps (*N* = 74) or three traps (*N* = 11). Some trials were conducted with cones modified to sample half the volume of water (*N* = 25) or mixed (*N* = 1), but primarily unmodified and sampling full effort (*N* = 59). Trap efficiencies were tested with the RBDD gates raised (*N* = 72) and lowered (*N* = 13) during the years when RBDD was in operation (Table 2).

Trials were conducted through a variety of flow and trap effort conditions representing actual sampling conditions detected throughout various fish migration periods (Table 2). Estimates of the percentage of river water volume sampled by traps (*%Q*) ranged from 0.72 to 6.87% ($\bar{y}$ = 3.10, SD = 1.32). Efficiency estimates for the 85 trials ranged from 0.34 to 5.48% ($\bar{y}$ = 2.37%, SD = 0.01).

Released fish groups ranged from 340 to 5,143 individuals ($\bar{y}$ = 1,598) and recaptured fish numbers ranged from 7 to 119 ($\bar{y}$ = 36) per trial. Trials were conducted predominantly with fry size-class (<46 mm fork length), naturally produced fall Chinook (67%) and to a lesser extent winter Chinook (22%). Trials were conducted in some years using unmarked pre-smolt/smolts (11%) following annual Coleman National Fish Hatchery Fall Chinook production releases[5] during spring, as conditions and staffing levels allowed (Table 2).

Average fork lengths of release groups in the fry size-class had fork lengths ranging from 35.5 to 57.1 mm ($\bar{y}$ = 37.2 mm). Recaptured fork lengths ranged from 34.6 to 62.4 mm ($\bar{y}$ = 37.3 mm). Average fork lengths of fish released in the pre-smolt/smolt size-class ranged from 68.7 to 81.2 mm ($\bar{y}$ = 75.3 mm). Recaptured fork lengths ranged from 61.3 to 80.2 mm ($\bar{y}$ = 75.3 mm; Table 2). A paired t-test was performed on the average

[5] Coleman National Fish Hatchery is located upstream of RBDD on Battle Creek a tributary to the Sacramento. Fall Chinook production fish (~12 million per year) were adipose clipped (i.e., marked) in varying proportions over the years of study between 0 and 25%. Unmarked fish were included in some efficiency trials as they could not be distinguished from naturally produced fish.

release and recaptured fish lengths for all trials and indicated no significant difference between the released and recaptured fish sizes ($P = 0.759$, $df = 83$, $t = -0.308$).

Trap Efficiency Modeling.—Between 1998 and 2000, Martin et al. (2001) developed a trap efficiency model for the RBDD rotary trapping operation by conducting 58 mark-recapture trials (one trial excluded due to zero efficiency value). These data were used as the basis of the trap efficiency model to calculate daily passage estimates. The model was further developed between 2002 and 2013 with the addition of 85 mark-recapture trials. Trap efficiency was positively correlated to (*%Q)*, with higher efficiencies occurring as the relative percentage of discharge volume sampled by rotary traps increased. Trap efficiency was inversely related to river discharge (Q), as river discharge increased, trap efficiency decreased.

As mark-recapture trials were conducted, the trap efficiency model was typically updated one time each year. The newest model was applied on July 1 of each year, the beginning of the annual winter Chinook juvenile brood year period. Between 2002 and 2013 nine different models were utilized. The specific dates and model parameters with P-values used throughout the reporting period are listed chronologically below the groups of mark-recapture trials incorporated into the models in Table 2. The net result over the 11-year period was stabilization and improvement of the trap efficiency model with the addition of 85 mark-recapture trials. Overall, the P-values indicated a high level of significance for the parameter *%Q* in all years ($P< 0.001$). The model's r-squared value dropped in the first few years and then improved greatly with the addition of numerous naturally produced fry size-class mark-recapture trials over a variety of river discharge levels (Table 2; Figure 3).

Over the 11 years' data was collected a wide range of *%Q* values were sampled (0.44 to 6.86%, $\bar{y}$ = 2.90, SD = 0.01). On 10 occasions, extremely low *%Q* values (<0.72%) were sampled outside of the range of values tested through efficiency trials (Figure 3). The net result was that trap efficiency values were extrapolated outside the range of the model on a mere 10 of 3,315 days sampled (0.3%).

Chinook Capture Fork Length Analyses.—Chinook run assignment based on length-at-date (LAD) criteria was originally developed from growth data in the Upper Sacramento River at the Tehama Colusa Fish Facility using fall Chinook production records from 1972 through 1981 (Fisher 1992). An estimate of apparent growth rate was originally developed from fall Chinook < 90 mm FL as fish migrated or were depleted from the spawning channels by this size (Fisher 1992). Johnson et al. (1992) further developed (extrapolated) the data to predict run for fish ≥ 90 mm and ≤ 250 mm FL. The data was further refined by Frank Fisher of the California Department of Fish and Game, whereby estimated growth curves were produced for all runs based on adult timing, water temperatures, and juvenile emergence timing and growth (Brown and Greene 1992). The growth curves were fitted to a table of daily growth increments (i.e., fork length at age in days) by the California Department of Water Resources in the early

1990's (Brown and Greene 1992; Greene 1992). The following fork length data encompassed fish sampled by rotary traps using the LAD tables up to 180 mm FL, as fish were rarely captured above this length (i.e., extreme outliers).

Fall Chinook sampled from brood years 2002-2012 were heavily weighted to the fry size-class category (<46mm). On average, 75.7% of all fish sampled as fall could be described as fry (SD = 6.9) with 71.0% of the fry measuring less than 40 mm FL (Figure 4a). The remaining 24.3% (SD = 6.9) were attributed to the pre-smolt/smolt category (>45 mm) with fish between 70 and 89 mm composing 71.0% of that value. Overall, fall Chinook were sampled between 30 and 134 mm annually, with trivial numbers below or above this range (Figure 4b). Fall Chinook showed little growth, on average, between December and March, followed by a significant increase in length in April, followed by more moderate and variable growth through November (Figure 4c). The growth pattern exhibited by fall Chinook appears strongly influenced by the duration of the fall Chinook spawning period and the LAD criteria. Beginning on April 1, newly emerged fry were classified as late-fall Chinook instead of fall Chinook thereby significantly increasing the median fork length of fall Chinook during the first two weeks of April.

Late-fall Chinook sampled from brood years 2002-2012 were not heavily weighted to the fry size-class category (<46mm). On average, 24.9% of all fish sampled as late-fall could be described as fry (SD = 12.8) with 96.3% of the fry measuring less than 40 mm FL (Figure 5a). The remaining 75.1% (SD = 12.8) were attributed to the pre-smolt/smolt category (>45 mm) with fish between 70 and 89 mm composing 48.3% of that value. Overall, late-fall Chinook were sampled between 26 and 180 mm annually (Figure 5b). Late-fall Chinook showed little growth, on average, between April and May, followed by a significant increase in length in June and July, followed by more moderate and variable growth between late-September and February (Figure 5c). The growth pattern exhibited by late-fall Chinook appears modestly influenced by the LAD criteria. Beginning on July 1, newly emerged fry were classified as winter Chinook instead of late-fall Chinook slightly increasing the median fork length of late-fall Chinook during the first few weeks of July. In mid-September and to a lesser extent in late-December, the overall fork length distribution for late-fall Chinook increases from one week to the next and was likely a result of decreased sampling effort due to RBDD gate operations and initial winter storms.

Winter Chinook sampled from brood years 2002-2012 were heavily weighted to the fry size-class category (<46mm). On average, 77.9% of all fish sampled as winter could be described as fry (SD = 8.8) with 92.8% of the fry measuring less than 40 mm FL (Figure 6a). The remaining 22.1% (SD = 8.8) were attributed to the pre-smolt/smolt category (>45 mm) with fish between 46 and 69 mm composing 85.3% of that value. Overall, winter Chinook were sampled between 27 and 162 mm annually (Figure 6b). Winter Chinook showed little growth, on average, between July and October, followed by a significant increase in length in mid-October, followed by more moderate growth through December. The growth pattern was then highly variable between January and

April (Figure 6c). The growth pattern exhibited by winter Chinook appears moderately influenced by the LAD criteria. Beginning on October 16, newly emerged fry were classified as spring Chinook instead of winter Chinook thereby significantly increasing the median fork length of winter Chinook during the last two weeks of October.

Spring Chinook sampled from brood years 2002-2012 were slightly weighted to the fry size-class category (<46mm). On average, 58.6% of all fish sampled as spring could be described as fry (SD = 19.6) with 90.0% of the fry measuring less than 40 mm FL (Figure 7a). The remaining 41.4% (SD = 19.6) were attributed to the pre-smolt/smolt category (>45 mm) with fish between 70 and 89 mm composing 69.2% of that value. Overall, spring Chinook were sampled between 28 and 143 mm annually (Figure 7b). Spring Chinook showed moderate growth, on average, between October and mid-December, followed by more consistent increasing growth through May (Figure 7c). Spring Chinook disappear from the catch typically by June with sporadic capture of large smolts in July of some years. The growth pattern exhibited by spring Chinook appears moderately influenced by the LAD criteria. Beginning on December 1, newly emerged fry were classified as fall Chinook instead of spring Chinook likely resulting in positive size-class bias for spring Chinook.

O. mykiss Capture Size Analyses.—Following the conventions used by Gaines and Martin (2002) size categorization for *O. mykiss* followed a slightly different pattern than Chinook and was organized by fork length as fry (<41 mm), sub-yearling (41–138 mm), and yearling (>138 mm). Moyle (2002) described Sacramento River *O. mykiss* populations as highly variable, but typically reaching 140-150 mm FL in their first year. The focus of our data reporting is age-0 and the focus of our size-class analyses was primarily < 139mm and secondarily < 200 mm for length-weight analyses.

O. mykiss sampled from calendar years 2002-2012 were heavily weighted towards the 41-80 mm size-class (79.2%; Figure 8a) which fell into the sub-yearling category (Figure 8b). On average, a modest 8.2% could be categorized as fry (Table 3). Overall, *O. mykiss* yearling and estimated age-2 fish were annually sampled at rates of 2.4% and 0.6%, respectively (Table 3). There was little variation detected within any size-class between categories, yet variance in weekly captures was high throughout the year (Figure 8c). The variable life-history strategies of *O. mykiss* resident and anadromous forms was evident from our size-class capture data. In general, newly emerged fry occurred in early-April and increased in size to early July. Thereafter, a second cohort of either resident trout or summer steelhead[6] was sampled which demonstrated a secondary growth pattern through December (Figure 8c).

O. mykiss CAMP Program Life-Stage Comparisons.— *O. mykiss* capture patterns appeared to be different than that of Chinook salmon as relatively few *O. mykiss* were captured as fry ($\bar{y}$ = 8.3%) and the majority were sampled as sub-yearlings ($\bar{y}$ = 88.7%;

[6] Summer steelhead are believed to be extirpated since the construction of dams blocked access to headwater habitat (Moyle 2002).

Table 3; Figure 8b). Fry capture was highest in 2002 and 2006 (11.2% and 17.5%) although these years sampled the first and third fewest *O. mykiss* of the 11 years, respectively. Yearling and age-2 capture was generally low averaging only 3.0%.

Life stage classification of fry was uniform throughout all years ($\bar{y}$ = 6.8%, SD = 2.6%) and did not vary greatly in 2002 and 2006 in contrast to age classification. Parr and silvery-parr accounted for 91.5% of the *O. mykiss* handled at RBDD although there was a large difference between the two categories, 74.0% and 17.5% respectively. Annual variability in parr and silvery-parr classifications (SD = 15.5 and 16.8) seemed to change after 2005 and was likely due to a protocol change or interpretation of morphological characteristics by field staff. Juveniles showing signs of anadromy (i.e., smolts) made up only 1.6% of individuals sampled.

O. mykiss Weight-Length Analysis.—Log_{10} transformed *O. mykiss* weight-length data showed a strong overall relationship between the two variables (r^2 = 0.942, Table 4). The annual slope coefficients for the 11-year period varied slightly, ranging from 2.858 to 3.052. The variability in growth was not considered significant as the 95% CI annual slope coefficients encompassed the slope coefficient of the overall mean (Table 4). Typical of most weight-length models (Pope and Kruse 2007), the variability about the regression increased with the overall length of the fish (Figure 9).

Salmonid Passage.—Passage estimates for the four runs of Chinook were calculated weekly as fry and pre-smolt/smolt passage. The sum of the weekly fry and pre-smolt/smolt passage values equal the weekly *total* passage values. Confidence intervals (CI) were calculated at the 90% level for all runs for weekly passage estimates. Weekly CI values were summed to obtain the annual CI's around the annual passage estimate (i.e., summed weekly passage estimates). Negative CI values were set to zero and result in some years CI's being asymmetrical around the annual passage estimate. Annual passage estimates (i.e., total passage estimates), by brood year, with CI's and annual effort values are presented for Chinook within Tables 5a-5d and graphically in Figures 10, 12, 14, and 16. Fry and pre-smolt/smolt Chinook passage estimates with 90% CI's summarized annually by run can be found in Appendix 1 (Tables A1-A8). Comparisons of relative variation within and between runs of Chinook were performed by calculating Coefficients of Variation (Sokal and Rohlf 1995) of passage estimates.

Fall Chinook annual passage estimates ranged between 6,627,261 and 27,736,868 juveniles for brood years 2002-2012 ($\bar{y}$ = 14,774,923, CV = 46.2%; Table 5a). On average, fall Chinook passage was composed of 74% fry and 26% pre-smolt/smolt size-class fish (SD = 10.3). Proportions as low as 56% and as high as 87% fry were detected (Table 5a). Annual effort values resulted in interpolations of between 9 and 60% of annual passage estimates ($\bar{y}$ = 28%). In general, the effect of annual effort on CI width indicated greater spread of CI's with decreasing effort (Figure 10).

On average, weekly fall passage equated to 5% of total annual fall Chinook passage between mid-January and early March (Figure 11a). Weekly passage varied considerably during this period with some weeks' passage totals accounting for >25% of annual passage values. Between BY 2002 and 2012, 75% of average annual passage occurred by the end of March, signifying January through March as the greatest period of migration. A second, albeit much diminished, mode of passage occurred between late April and May of each year due to the release of unmarked fall Chinook production fish from Coleman National Fish Hatchery. These fish could not be distinguished from wild fish due to fractional marking processes that varied over the 11-year period from 0 to 25%. Overall, fall passage was complete by the end of July each year with sporadic small pulses of smolts through November (Figure 11b).

Late-fall Chinook annual passage estimates ranged between 91,995 and 2,559,519 juveniles for brood years 2002-2012 ($\bar{y}$ = 447,711, CV = 159.9%; Table 5b). On average, late-fall Chinook passage was composed of 38% fry and 62% pre-smolt/smolt size-class fish (SD = 22.5). Proportions as low as 11% and as high as 72% fry were detected (Table 5b). Annual effort values resulted in interpolations of between 9 and 56% of annual passage estimates ($\bar{y}$ = 31%). The effect of annual effort on CI width indicated greater spread of CI's with decreasing effort due to hatchery fish releases, in general (Figure 12).

On average, weekly late-fall passage started abruptly and held at ≤ 5% of total annual passage between April and May (Figure 13a). Weekly passage varied considerably during this period with some weeks' passage totals accounting for >35% of annual passage values. A second, similar magnitude mode of passage occurred between July and August in most years. A third, albeit diminished, mode occurred during October and November with passage accounting for up to 35% of the annual run in some years. Between BY 2002 and 2012, 75% of average annual passage occurred by mid-September, signifying April through September as the greatest period of migration. Overall, late-fall passage was complete by the end of December each year with sporadic small pulses of smolts through February (Figure 13b).

Winter Chinook annual passage estimates ranged between 848,976 and 8,363,106 juveniles for brood years 2002-2012 ($\bar{y}$ = 3,763,362, CV = 73.2%; Table 5c). On average, winter Chinook passage was composed of 80% fry and 20% pre-smolt/smolt size-class fish (SD = 11.2). Proportions as low as 53% and as high as 90% fry were detected (Table 5c). Annual effort values resulted in interpolations of between 8 and 42% of annual passage estimates ($\bar{y}$ = 18%). The effect of annual effort on CI width indicated greater spread of CI's with decreasing effort due to subsampling measures during peak migration periods (i.e., take or impact reduction), in general (Figure 14).

On average, weekly winter passage increased consistently through September to a peak into early October. Weekly passage varied considerably during August through December with some weeks' passage totals accounting for >20% of annual passage values. Between BY 2002 and 2012, 75% of average annual passage occurred by mid-

October. Weekly passage between October and December indicated wide variability over the 11-year period, yet the trend showed steady decreases followed by a second increase or mode of winter passage in November and December (Figure 15a). Overall, winter passage was 99% complete by the end of December each year with sporadic pulses of smolts through March that contributed minimally to the annual total winter passage estimate (Figure 15b).

Spring Chinook annual passage estimates ranged between 158,966 and 626,925 juveniles for brood years 2002-2012 ($\bar{y}$ = 364,508, CV = 45.0%; Table 5d). On average, spring Chinook passage was composed of 54% fry and 46% pre-smolt/smolt size-class fish (SD = 20.0). Proportions as low as 24% and as high as 91% fry were detected (Table 5d). Annual effort values resulted in interpolations of between 1 and 49% of annual passage estimates ($\bar{y}$ = 29%). The effect of annual effort on CI width indicated a slightly greater spread of CI's with decreasing effort due to subsampling during winter storm events, in general (Figure 16).

On average, weekly spring passage started abruptly and held at roughly 5% of total annual passage between mid-October and mid-November (Figure 17a). Weekly passage varied somewhat during this period with some weeks' passage totals accounting for up to 20% of annual passage values. A second, increased magnitude mode of passage occurred during December in most years with a single week accounting for nearly 50% of the annual passage estimate. Between BY 2002 and 2012, 75% of average annual passage occurred by mid-April, signifying October through April as the greatest period of migration. A third mode of similar magnitude to the second mode occurred during April and May with passage accounting for up to 45% of the annual run in some years. This could be characterized as an erroneous increase in spring passage. Unmarked fall production fish exceeded the size-class for fall run and therefore fell within the spring run category using LAD criteria. Between 2007 and 2012, on average, 4.3% of the marked fall production fish fell within the spring-run size-class using LAD criteria. Assumedly, a similar proportion of the unmarked fish were added into the spring-run passage estimates as they could not be distinguished from naturally produced fish. Overall, spring Chinook passage was complete by the end of May each year (Figure 17b).

O. mykiss passage estimates were generated using trap efficiency estimates calculated using the Chinook-based trap efficiency model. Caution should be exercised when interpreting the following results as Chinook and *O. mykiss* trap efficiency values likely differ, perhaps greatly. Irrespective of the accuracy of the magnitude of passage estimates based on Chinook efficiency trials, the trends in abundance remain plausible due to the standardization of effort and catch. Unlike Chinook, *O. mykiss* were not attributed to a fry or pre-smolt/smolt category and passage estimates with 90% CI's were calculated that included all size-classes and life-stages combined.

Annual passage estimates for *O. mykiss* ranged between 56,798 and 151,694 juveniles for calendar years 2002-2012 ($\bar{y}$ = 116,272, CV = 25.7%; Table 5e). Annual effort values resulted in interpolations of between 4 and 56% of annual passage estimates ($\bar{y}$ = 22%). The effect of annual effort on CI width indicated a slightly greater spread of CI's with decreasing effort, in general (Figure 18).

On average, weekly *O. mykiss* passage was low (<5% on average) from April through July of each year with some variability. In 11 years of sampling only once did passage exceed 10% of annual passage during these months. Weekly passage between July and August increased to peak values ranging from 5% to nearly 25% (Figure 19a). Between 2002 and 2012, 75% of average annual passage occurred by mid-August. Weekly passage generally declined between September and October. Overall, *O. mykiss* passage was negligible between December and the following February each year (Figure 19b).

Fry-Equivalent Chinook Production Estimates.—Juvenile Chinook passage values were standardized to *fry-equivalent* production estimates for within- and between-year comparisons. As noted above, the various runs were sampled with oftentimes considerable variability in fry to pre-smolt/smolt ratios over the 11–year sample period (Table 5a-5d). By multiplying 1.7 to all fish sampled in the pre-smolt/smolt category (>45mm) within each run, annual Chinook production above the RBDD transect could be estimated. These standardized production estimates could then be compared to adult escapement estimates calculated from the California Central Valley Chinook Population Report (Azat 2013) or carcass survey data in the case of winter Chinook (USFWS 2006-2011 and 2013). Moreover, by comparing production to the number of adult Chinook females each year (by run) and estimating fecundity data from CNFH and Livingston Stone National Fish Hatchery (LSNFH) hatchery production records, estimated recruits per female and egg-to-fry survival estimates were generated.

Fall Chinook fry-equivalent production estimates between 2002 and 2012 ranged from 7,554,574 to 30,624,209 ($\bar{y}$ = 17,262,473, CV = 43.2%). Lower and upper 90% CI's were generated for each week, summed annually, and averaged between 6,670,475 and 30,707,529 (Table 6a).

Adult fall Chinook escapement estimates above RBDD (mainstem Sacramento River plus tributaries reported) estimated escapement between 12,908 and 458,772 ($\bar{y}$ = 93,661) for the same years. Fall Chinook carcass survey data collected by California Department of Fish and Wildlife (CDFW) provided annual female:male sex ratio estimates averaging 0.46:0.54 (D. Killam, unpublished data). A significant relationship between estimated number of females and fry-equivalent fall Chinook production estimates was detected (r^2 = 0.53, df = 10, P = 0.01; Figure 20a). Recruits per female were calculated ranging from 89 to 1,515 ($\bar{y}$ = 749). Assuming an average female fecundity value of 5,407, based on fall Chinook spawning records from CNFH between 2008 and 2012 (K. Brown, unpublished data), resulted in an egg-to-fry survival estimate averaging 13.9% for fall Chinook (Table 6a).

Late-fall Chinook fry-equivalent production estimates between 2002 and 2012 ranged from 116,188 to 4,041,505 ($\bar{y}$ = 669,939, CV = 169.8%). Lower and upper 90% CI's were generated for each week, summed annually, and averaged between 222,044 and 1,236,432 (Table 6b).

Adult late-fall Chinook escapement estimates above RBDD estimated escapement between 2,931 and 36,220 ($\bar{y}$ = 9,108) for the same years. Late-fall Chinook annual female:male sex ratio estimates relied on an assumption of the average ratio found for fall Chinook (i.e., 0.46:0.54). A significant relationship between estimated number of females and fry-equivalent late-fall Chinook production estimates was detected (r^2 = 0.67, df = 10, P = 0.002; Figure 20b). Recruits per female were calculated ranging from 47 to 243 ($\bar{y}$ = 131). Assuming an average female fecundity value of 4,662 based on late-fall Chinook spawning records from CNFH between 2008 and 2012 (K. Brown, unpublished data) resulted in an egg-to-fry survival estimate averaging 2.8% for late-fall Chinook (Table 6b).

Winter Chinook fry-equivalent production estimates between 2002 and 2012 ranged from 996,621 to 8,943,194 ($\bar{y}$ = 4,152,547, CV = 70.1%). Lower and upper 90% CI's were generated for each week, summed annually, and averaged between 2,265,220 and 6,124,494 (Table 6c).

Adult winter Chinook escapement estimates above RBDD (USFWS/CDFW carcass survey data; available at http://www.fws.gov/redbluff/he_reports.aspx) estimated escapement between 824 and 17,205 ($\bar{y}$ = 6,532) for the same years. Winter Chinook annual female:male sex ratio estimates were estimated during the annual carcass surveys (Table 6c). A highly significant relationship between estimated number of females and fry- equivalent winter Chinook production estimates was detected (r^2 = 0.90, df = 10, P < 0.001; Figure 20c). Recruits per female were calculated ranging from 846 to 2,351 ($\bar{y}$ = 1,349). Annual female fecundity values were estimated based on winter Chinook spawning records from LSNFH between 2008 and 2012 (USFWS Annual Propagation Reports; available at http://www.fws.gov/redbluff/he_reports.aspx) and resulted in an egg-to-fry survival estimate averaging 26.4% for winter Chinook (Table 6c).

Spring Chinook fry-equivalent production estimates between 2002 and 2012 ranged from 207,793 to 747,026 ($\bar{y}$ = 471,527, CV = 40.9%). Lower and upper 90% CI's were generated for each week, summed annually, and averaged between 199,365 and 792,668 (Table 6d).

Adult spring Chinook escapement estimates above RBDD (mainstem Sacramento River plus tributaries reported) estimated escapement between 77 and 399 ($\bar{y}$ = 195) for the same years. Spring Chinook annual female:male sex ratio estimates relied on an assumption of the average ratio found for fall Chinook (i.e., 0.46:0.54). No significant relationship between estimated number of females and fry-equivalent spring Chinook production estimates was detected (r^2 = 0.00, df = 10, P = 0.971; Figure 20d). Recruits per female were calculated ranging from 1,112 to 8,592 ($\bar{y}$ = 3,122). Assuming an average female fecundity value of 5,078, based on averaging of 5 years of fall and late-fall Chinook spawning records from CNFH and 10 years of winter Chinook spawning records from LSNFH, resulted in an egg-to-fry survival estimate averaging 61.5% for spring Chinook (Table 6d).

Green Sturgeon Data.—Capture of young of the year sturgeon occurred annually between calendar years 2002 and 2012, except in 2008. Catch was highly variable, not normally distributed, and ranged between 0 and 3,701 per year (median = 193; Table 7). Sturgeon sampled by rotary traps could be positively identified as Green Sturgeon in the field *above* total length of 46 mm. At this size, lateral scutes were fully developed and could be counted to distinguish between White (*Acipenser transmontanus*) and Green Sturgeon (Moyle 2002). Of 2,912 sturgeon measured in the field, 99.14% were less than 46 mm. In all years, except 2007 and 2008, sub-samples of larval and/or juvenile sturgeon rotary trap catch (up to 50% in some years) were supplied to UC Davis for genetic research and all were determined to be Green Sturgeon (See Israel et al. 2004; Israel and May 2010). We therefore assumed all sturgeon captured in rotary traps were Green Sturgeon based on the results of genetic analyses. Moreover, Green Sturgeon were the only confirmed spawning Acipenserids sampled at or above the RBDD transect between 2008 and 2012 during sturgeon spawning surveys (Poytress et al. 2009-2013).

Green Sturgeon catch was primarily composed of recently emerged, post-exogenous feeding larvae with a 10-year median capture total length averaging 27.3 mm (SD = 0.8; Table 7). Sturgeon were sampled between 18 and 188 mm, but those sampled above 40 mm were considered outliers (N = 51; Table 7; Figure 21a).

The temporal pattern of Green Sturgeon captures occurred, on average, between May 1 and August 28 of each year. Green Sturgeon capture trends indicated annual variability, but on average 50% were sampled by the end of June each year and nearly 100% by the end of July (Figure 21b), with outliers (i.e., juveniles) captured in August, September and as late as November (e.g., 188 mm TL) in some years.

Relative abundance of Green Sturgeon was measured as catch per estimated water volume sampled (CPUV in ac-ft) through rotary trap cones and summed daily. Daily values were summed annually to produce each year's annual index of abundance. Absolute abundance estimates, via trap efficiency trials, could not be calculated due to low numbers of sturgeon sampled on a daily basis and the fragile nature of newly emerged exogenous feeding larvae.

Green Sturgeon annual CPUV was typically low and ranged from 0.0 to 20.1 fish/ac-ft ($\bar{y}$ = 2.5 fish/ac-ft, SD = 5.9). Data were positively skewed and median annual CPUV was 0.8 fish/ac-ft. Relative abundance distribution data were highly influenced by samples collected in 2011 that equated to two orders of magnitude higher than any other year's index (Figure 21c). Overall, variability in CPUV between years was relatively high as the CV was 236% for the eleven-year period (Table 7).

Lamprey Species Data.—Capture of multiple lamprey species occurred between water year (WY; October - September) 2003 and 2013. WY 2002 was excluded from analyses as less than 50% of the entire year was sampled. Lamprey species sampled included adult and juvenile Pacific Lamprey and to a much lesser extent River Lamprey (*Lampetra ayresi*), and Pacific Brook Lamprey (*Lampetra pacifica)*. Unidentified lamprey ammocoetes and Pacific Lamprey (PL) composed 99.8% of all captures, 24% and 75%, respectively. River Lamprey and Pacific Brook Lamprey combined, composed the remaining 0.2% of all captures. Annual catch, length, and relative abundance information for River and Pacific Brook Lamprey can be found in Appendix 1 (Tables A9 and A10) and are not discussed further due to very low capture rates.

Annual catch of ammocoetes was relatively stable and ranged between 385 and 1,415 individuals per year ($\bar{y}$ = 757, median = 657; Table 8a). The catch coefficient of variation for ammocoetes was 38.5%. Minimum TL of lamprey ammocoetes was 14 mm and maximum TL was 191. Over the eleven complete years sampled, the average minimum and maximum TL's were 32 and 164 mm, respectively ($\bar{y}$ =105, SD = 4.7; Figure 22a).

Annual catch of PL macropthalmia and a small fraction of adults was variable and ranged between 204 and 5,252 individuals per year ($\bar{y}$ = 2,335, median = 2,747; Table 8b). The catch coefficient of variation for PL was 75.3%. Minimum TL of PL was 72 mm and maximum TL was 834. Over the eleven years sampled, the average minimum and maximum TL's were 88 and 665 mm, respectively ($\bar{y}$ = 150, SD = 37.3; Figure 23a).

Lamprey captures occurred throughout the year between October and September. Ammocoete capture trends indicated annual variability, but on average 25% were sampled by the end of January, 50% were sampled by the end of March, 75% were sampled by the end of May and 100% by the end of September (Figure 22b). Transformed PL (macropthalmia and adult) capture trends indicated a different pattern of capture and annual variability compared to ammocoetes. On average, 5% were sampled through October, 50% were sampled through December, 75% were sampled through February, 90% by the beginning of April with a 100% by the end of September (Figure 23b).

Relative abundance of ammocoetes and PL were measured as CPUV through individual rotary trap cones and summed daily. Daily values were summed annually to produce each year's annual index of abundance. Absolute abundance estimates employing mark-recapture methods could not be calculated due to the sporadic capture of adequate numbers of juveniles (e.g., > 1,000 individuals) that would be needed for mark-recapture trials. Moreover, emphasis was placed on conducting Chinook mark-recapture trials at times of pronounced lamprey abundance.

Ammocoete annual relative abundance ranged from 3.6 to 11.7 fish/ac-ft ($\bar{y}$ = 6.8 fish/ac-ft, SD = 2.6; Figure 22c). Overall, ammocoete data were normally distributed as median CPUV was 6.5 fish/ac-ft, similar to the mean value. Variability in CPUV between years was modest and the coefficient of variation was 39% for the eleven-year period (Table 8a).

PL annual relative abundance was generally higher than ammocoete relative abundance and ranged from 2.1 to 112.8 fish/ac-ft ($\bar{y}$ = 41.0 fish/ac-ft, SD = 34.7; Figure 23c). Overall, PL data was slightly positively skewed and median CPUV was 34.1 fish/ac-ft. Variability in CPUV between years was moderate and the coefficient of variation was 85% for the eleven-year period (Table 8b).

Abiotic Conditions.—Tabular summaries of the abiotic conditions that were encountered during each annual capture period were summarized for each run of salmon, *O. mykiss*, Green Sturgeon and Lamprey species. Tabular summaries associated with each species annual captures are located in Tables 9a-9f and include: dates of capture, peak daily water temperature, peak daily river discharge levels and mean daily turbidity values. A series of exploratory plots comparing the above daily environmental data variables plus an index of moon illuminosity were generated for fry and pre-smolt Chinook daily passage estimates for visual analyses. Winter Chinook fry and pre-smolt/smolt plots are included in Appendix 2 (Figures A1-A23) for reference.

Annual environmental covariate data for fall Chinook salmon can be found in Table 9a. Results presented below describe data averaged over 11 brood years. Fall Chinook were sampled over a period of 250 to 273 days per year ($\bar{y}$ = 264 days, SD = 7). Water temperatures ranged from 45 to 62 °F ($\bar{y}$ = 55°F, SD = 0.8). Sacramento River discharge ranged from 5,605 to 72,027 CFS ($\bar{y}$ = 14,844 CFS, SD = 5,442). Turbidity values ranged from 1.5 to 298.7 NTU ($\bar{y}$ = 14.4 NTU, SD = 6.3).

Annual environmental covariate data for late-fall Chinook salmon can be found in Table 9b. Results presented below describe data averaged over 11 brood years. Late-fall Chinook were sampled over a period of 270 to 338 days per year ($\bar{y}$ = 300 days, SD = 24). Water temperatures ranged from 46 to 62 °F ($\bar{y}$ = 56°F, SD = 0.7). Sacramento River discharge ranged from 5,536 to 67,520 CFS ($\bar{y}$ = 12,580 CFS, SD = 2,829). Turbidity values ranged from 1.4 to 272.0 NTU ($\bar{y}$ = 11.3 NTU, SD = 6.2).

Annual environmental covariate data for winter Chinook salmon can be found in Table 9c. Results presented below describe data averaged over 11 brood years. Winter Chinook were sampled over a period of 207 to 278 days per year ($\bar{y}$ = 250 days, SD = 20). Water temperatures ranged from 46 to 61 °F ($\bar{y}$ = 55°F, SD = 0.8). Sacramento River discharge ranged from 5,349 to 66,800 CFS ($\bar{y}$ = 11,952 CFS, SD = 3,767). Turbidity values ranged from 1.3 to 290.2 NTU ($\bar{y}$ = 12.5 NTU, SD = 5.1).

Annual environmental covariate data for spring Chinook salmon can be found in Table 9d. Results presented below describe data averaged over 11 brood years. Spring Chinook were sampled over a period of 221 to 250 days per year ($\bar{y}$ = 232 days, SD = 9). Water temperatures ranged from 46 to 62 °F ($\bar{y}$ = 53°F, SD = 0.6). Sacramento River discharge ranged from 5,349 to 68,720 CFS ($\bar{y}$ = 13,370 CFS, SD = 6,116). Turbidity values ranged from 1.4 to 305.9 NTU ($\bar{y}$ = 16.0 NTU, SD = 7.0).

Annual environmental covariate data for *O. mykiss* can be found in Table 9e. Results presented below describe data averaged over 10 calendar years. *O. mykiss* were sampled over a period of 331 to 363 days per year ($\bar{y}$ = 349 days, SD = 12). Water temperatures ranged from 46 to 63 °F ($\bar{y}$ = 56°F, SD = 0.8). Sacramento River discharge ranged from 5,333 to 67,610 CFS ($\bar{y}$ = 12,519 CFS, SD = 3,551). Turbidity values ranged from 1.4 to 263.7 NTU ($\bar{y}$ = 11.4 NTU, SD = 4.1).

Annual environmental covariate data for Green Sturgeon can be found in Table 9f. Results presented below describe data averaged over 11 calendar years. Green Sturgeon were sampled over a period of 56 to 151 days per year ($\bar{y}$ = 88 days, SD = 27). Water temperatures ranged from 55 to 61 °F ($\bar{y}$ = 58°F, SD = 0.9). Sacramento River discharge ranged from 9,639 to 23,538 CFS ($\bar{y}$ = 13,483 CFS, SD = 2,181). Turbidity values ranged from 2.4 to 93.9 NTU ($\bar{y}$ = 8.5 NTU, SD = 6.9).

Due to the large amount of variability and lack of a normal distribution, all environmental covariate CPUV data analyses for Green Sturgeon were performed using natural log transformed data (Sokal and Rohlf 1995). Environmental covariates were regressed against the natural log of daily CPUV estimates for Green Sturgeon in a linear regression setting (Figure 24). Maximum daily water temperature was the only variable found to be significantly related to Green Sturgeon relative abundance, albeit the relationship explained ~5% of the variability around daily relative abundance (r^2= 0.045, df = 315, $P < 0.001$).

Annual environmental covariate data for Lamprey *spp.* can be found in Table 9g. Results presented below describe data averaged over 11 water years. Lamprey were sampled over a period of 358 to 364 days per year ($\bar{y}$ = 362 days, SD = 2). Water temperatures ranged from 46 to 63 °F ($\bar{y}$ = 56°F, SD = 0.7). Sacramento River discharge ranged from 5,347 to 68,873 CFS ($\bar{y}$ = 12,595 CFS, SD = 4,177). Turbidity values ranged from 1.2 to 306.8 NTU ($\bar{y}$ = 11.9 NTU, SD = 4.4).

Due to the variability and lack of a normal distribution, all environmental covariate CPUV data analyses for Lamprey *spp.* were performed using natural log transformed data. Environmental covariates were regressed against the natural log of daily CPUV data for Lamprey *spp.* in a linear and multiple regression setting. All four independent variables appear to contribute to predicting Lamprey *spp*. relative abundance and were significantly related to abundance levels (r^2= 0.223, df = 1999, $P < 0.001$). Individual variable linear regression analyses indicated turbidity, water temperature, discharge, and full moon illuminosity were correlated in descending order of magnitude (Figure 25). None of the covariates tested explained more than ~16% of the variability associated with daily CPUV data.

Discussion

Trap Efficiency Modeling.—Over the past 11 years, annual mark-recapture trials added 85 data points to the RBDD rotary trap efficiency linear regression model (Figure 3). Explanation of the variability associated with trap efficiency and *%Q*, in terms of the associated r-squared value, was reduced for the first few years and then steadily increased in more recent years. The reduction was due, in part, to more precise *%Q* calculations over the initial model when diversions from RBDD were not subtracted from daily river discharge values. Diversions were able to be removed from the total discharge (*Q*) passing the transect as these data became available in real-time starting in 2002.

The addition of a multitude of fry size-class trials over a variety of discharge levels greatly increased the accuracy of trap efficiency estimates. Fry size-class fish are the predominant size-class sampled at RBDD (i.e., fall and winter Chinook) thereby making them the best representatives for use in mark-recapture trials. The original trap efficiency model developed by Martin et al. (2001) employed primarily hatchery-raised smolts, as these fish were all that were available in large quantities and permitted for use in experiments to develop the initial model. However, hatchery fish weakly represented the primary fish size-class sampled by RBDD rotary traps. Roper and Scarnecchia (1996) and Whitton et al. (2008) found significant differences in trap efficiency when conducting paired mark-recapture trials using hatchery and wild caught fish. The most recent years of RBDD data support this concept.

While a simple linear regression model has worked well over the years for our real-time data output needs, analysis of the data within the model, other possible covariates, and other more advanced modeling techniques has been warranted. Analysis incorporating additional potential explanatory variables was conducted using a generalized additive model technique (GAM; Hastie and Tibshirani 1990). From this analysis, variables including turbidity, fish size and run, water temperature, weather condition, lunar phase, and river depth were explored in addition to *%Q*. The result was that only *%Q* and weather were found to be significant model explanatory variables (r^2 = 0.68; df = 141, $P < 0.01$). The weather variable needs focused testing by conducting more mark-recapture trials under a variety of weather conditions to determine the applicability or mechanism of this variable. The GAM modeling technique may be employed in the future as an improved statistical format to interpolate missed sample days.

At minimum, an update to the 142 trial linear trap efficiency model (Figure 3) needs to be implemented for future passage estimate calculations. The update will include the removal of hatchery fish trials (*N*=23) used as surrogates for natural stocks. Removal of all RBDD "gates in" mark-recapture trials (*N*=31) due to the cessation of RBDD dam operations since 2011 (NMFS 2009) is also warranted.

The loss of annual maintenance and RBDD gate lowering operations at the rotary trap sample site (Figure 1) will allow the river channel's geometry to change more frequently due to natural flow driven substrate transport mechanisms. RBDD operations of the past virtually "reset" the sample site to facilitate pumping during the gates-out period and improve fish passage at the fish ladders during the gates-in period. As the sample site's channel configuration is allowed to fluctuate in the absence of dam operations, the overall effect could be differing trap efficiency values in relation to flow compared to previous years' data. Annual mark-recapture trials will be needed to evaluate this phenomenon, which has been observed in other uncontrolled channel sampling locations (e.g., Clear Creek; Greenwald et. al. 2003). The use of a GAM model may also be of benefit in this situation as it could be constructed and employed annually to account for wide variation in annual trap efficiency values; albeit at the expense of being able to produce real-time data summaries.

A linear model that also removed the remaining pre-2002 trials (*N*=16) which estimated *%Q* in a less precise manner, would result in the most representative trap efficiency model. A post-RBDD wild Chinook model of this type would incorporate 72 mark-recapture trials with a high degree of significance (*N*=72, $r^2 = 0.669$, F = 141.5, $P < 0.001$) and be most representative of current sampling conditions in terms of fish size-class and environmental conditions.

Chinook Capture Size Analyses.—Overall capture of Chinook salmon by RBDD rotary traps was heavily weighted towards fry size-class less than 40mm in fork length. All four runs' greatest proportion of fish were found in this size-class, albeit in a range of proportions from 24% for late-fall (Figure 5b) to over 72% for winter run (Figure 6b). The capture size-class results fit well with the migratory strategies of 'stream' and 'ocean type' as noted in Moyle (2002) for late-fall/spring and fall/winter Chinook, respectively. The question of size selectivity or capture bias of rotary traps, a passive sampling gear (Hubert 1996), comes into question when dealing with two very different migration strategies.

A two sample t-test was performed to evaluate the potential for size-class bias by comparing fry (fall and winter Chinook) size-class trap efficiency values (*N*=43) to pre-smolt/smolt (fall) trap efficiency values (*N*=10) between similar river discharge conditions. The t-test results did not indicate any significant difference between the mean efficiency values (t = -0.398, df = 51, $P = 0.624$). Interestingly, the mean efficiency and standard deviation of the values were identical ($\bar{y} = 2.1\%$, SD = 0.01) between groups. We recommend further study of the relationship between pre-smolt/smolt size-class and trap efficiency to determine if differences or bias may exist between or among Chinook runs. Additional sampling effort would be needed to capture substantially more pre-smolts in the numbers required for efficiency trials in the Sacramento River to further test this potential bias. Smolting salmonids also appear to succumb to stress induced mortality at a much greater rate than fry, particularly in warmer water conditions due to relatively high respiration levels, adding to the difficulty in testing this potential bias.

O. mykiss Life-Stage and Growth.— Catch of *O. mykiss* was scattered throughout the year with multiple modes in abundance of predominately sub-yearling parr and silvery-parr occurring in early May and August. *O. mykiss* fry (<41 mm) made up 17.5% of the total *O. mykiss* catch in 2006 and was 2.4 standard deviations from the 11-year mean. In contrast, yolk-sac fry, made up only 9.4% of the *O. mykiss* catch in 2006 and varied less than 1 standard deviation from the 11-year mean (Table 3). Elevated spring discharge resulted in poor sampling conditions which reduced sampling effort, possibly scoured redds, and ultimately resulted in low overall *O. mykiss* catch in 2006. Regardless of the cause of low catch rates, it is unlikely the migration patterns of *O. mykiss* changed in 2006 and the variability in age-class distribution was likely due to our sampling effort in that year.

The small percentage of *O. mykiss* smolts that showed signs of anadromy were generally migrating during March through June which was consistent with outmigrating smolts found in Battle, Mill, and Deer Creeks (Johnson and Merrick 2012; Colby and Brown 2013). Interpretation of *O. mykiss* data collected at the RBDD was complicated as a robust resident (non-anadromous) population exists throughout the Upper Sacramento River and its' tributaries. Populations of anadromous and resident *O. mykiss* life history forms are often sympatric and may inter-breed (Zimmerman and Reeves 2000; Docker and Heath 2003), thereby reducing our abilities to separate the anadromous and non-anadromous components of this species. Donahue and Null (2013) conducted research using otolith Strontium/Calcium ratios to determine whether *O. mykiss* returning to a hatchery were progeny of anadromous or resident females. A similar analysis could be conducted using juvenile *O. mykiss* collected at the RBDD. Data from juveniles might provide incite as to whether temporal separation in spawn timing exists between anadromous and resident forms of O. mykiss coexisting within the Upper Sacramento River basin.

Linear regression equations developed using weight-length data obtained from *O. mykiss* showed a strong correlation between the two variables (r^2= 0.942). The annual slope coefficient varied slightly between 2.858 and 3.052. Carlander (1969) suggested that slopes less than 3.0 might indicate a crowded or stunted population. However, permit restrictions may have introduced bias into our results as we were unable to anesthetize and weigh fish >200 mm thereby reducing the slope of the regression compared to that of a complete analysis of the population.

Sample Effort Influence on Passage Estimates.—Sampling effort had profound effects on the precision of passage estimates and confidence intervals (Figures 10, 12, 14, 16, and 18). In general, as sampling effort decreased, variance within weekly passage estimates increased and the width of confidence intervals subsequently increased. This effect was most prominent when effort was reduced during peak periods of outmigration or for long periods of time (> 1 week) when sharp increases or decreases in fish abundance occurred. Unfortunately, sampling of outmigrant Chinook on a large river system such as the Sacramento River is invariably subject to discharge events that are insurmountable for variable periods of time.

Logistical factors including staffing and permitting restrictions can also have significant effects on the precision of estimates. For example, a comparison of BY 2002 and BY 2005 winter Chinook passage with equivalent effort values (0.64) shows less precision of BY 2002 passage estimates over BY 2005 (Table 5c). The basis of the relatively low effort in 2002 was capture restrictions prompted by ESA Section 10(a)(1)(A) NMFS permits for endangered winter Chinook. Moreover, staff levels were initially low as the program was reinstated after a nearly two-year hiatus and substantial sub-sampling measures (i.e., standardized sub-sampling of repeated weeks) had to be taken during record abundance levels. The net effect was that sampling of fry, the predominant size-class of ocean type Chinook (Moyle 2002; Figure 6a/b), was reduced in terms of the number of days each week and hours of each night sampled during the peak emigration period. The overall net effect was 20% wider CI's about the 2002 estimate (i.e., less precision) compared to BY 2005. This was due to interpolation of 45% of the fry data which comprised 90% of the 2002 annual estimate. In contrast, BY 2005 sampled 90% of the fry data which comprised 90% of the annual estimate. Effort was reduced 36% in 2005 as a result of winter storms whereby sampling ceased for 3 straight weeks due to high river discharge levels. The effect of that lost sampling time in January did little to reduce the precision of the BY 2005 estimate as it was during a period when a mere fraction of a percent of total passage for winter Chinook typically occurs (Figure 15). The impact to the BY 2005 *fall Chinook* passage estimate, on the other hand, was very wide CI's about the estimate due to the lowest effort of all 11 years during a critical time period for that run's outmigration (Table 5a, Figure 11).

In summary, the precision of passage estimates can vary widely for numerous reasons within runs and among years. Inter-annual variability in environmental conditions will always be a factor when attempting to sample a riverine environment. Making good sampling decisions with knowledge of the species of interest and riverine conditions coupled with tenacity to sample critical periods of outmigration (Volkhardt et al. 2007) are key to generating passage estimates with an acceptable level of precision. Applying effort throughout each period of interest needs to be balanced between the value of data collected, an acceptable level of precision required of the data, the cost to attain the required precision, the impact sampling may have to a particular species, and the feasibility to appropriately sample the species of interest.

Chinook Passage Variability.—Juvenile Chinook passage by one to four runs occurs every single day of the year in varying proportions at RBDD. The sources and degree of variability of juvenile Chinook passage are as diverse as the life-history and migration strategies of the runs they encompass. The magnitude of run-specific adult spawners appears to have the greatest influence on the overall magnitude of juvenile Chinook passage and associated variability.

In recent decades, fall Chinook adults consistently dominated the Upper Sacramento River spawning salmon populations (Williams 2006, Azat 2013). Throughout the past decade, we witnessed a 'collapse' of the Sacramento River fall Chinook adult population and accordingly tracked declines in juvenile passage (Figure 10). Lindley et al. (2009) analyzed the freshwater and marine components of fall Chinook outmigrants from BY 2004 and 2005 through their return as adults in 2007 and 2008. They indicated BY 2004 and 2005 juveniles encountered poor marine conditions upon ocean entry in the spring of 2005 and 2006 which resulted in the marked decline in fall Chinook adult abundance starting in 2007.

Juvenile fall Chinook had the greatest mean annual passage value (14,774,923) of the four runs sampled at RBDD (Table 5a). Fall Chinook passage also exhibited the second smallest degree of variability with a CV of 46.2%. Notably, fall Chinook annual production by the CNFH averages 12 million juveniles, a similar value to the mean passage value of unmarked fall Chinook[7]. Fall Chinook production fish from CNFH contributed heavily to the relative stability of the annual returning fall Chinook adult population (Williams 2006) and, consequently, juvenile passage estimates over the past eleven years (i.e., basis of fall Chinook population).

Temporal abundance patterns of fall Chinook indicate the primary passage of juveniles occurs between late December and March (Figure 11a/b). Over half the run passed RBDD by mid-February, yet this varied over the 11-year period by +/- one month. Fall run passage on the American River (Williams 2006), Clear Creek (Earley et al. 2013a) and Stanislaus River (Pyper and Justice 2006) in California generally subsides to low values by the end of March. This would be consistent with the ocean type migration strategy as noted by Moyle (2002). The remaining fall run smolts and subsequent 'jump' in abundance in April to May was a result of the unmarked proportion of the CNFH production releases. Reduced variability in weekly passage was observed in the final 20% of annual fall Chinook passage (Figure 11b).

Spring Chinook had the lowest average passage value of 364,000 juveniles and the lowest CV of 45% (Table 5d). The low value of spring Chinook passage at RBDD can be attributed to a relatively small number of adults spawning primarily in Battle and Clear Creeks (Figure 1). Some extant populations appear to inhabit Beegum Creek, a tributary to Cottonwood Creek (CDFG 2001), and in the mainstem Sacramento River (Killam 2009, Azat 2013). Of particular interest with respect to the accuracy of spring Chinook

[7] Fall Chinook passages estimates do not include the marked proportion (0-25%) of CNFH production fish. Unmarked fish of hatchery origin are included in annual passage estimates and their occurrence is evidenced by increased passage values primarily in May through June of each calendar year (Figure 11b).

juvenile passage at RBDD is the annual spawn timing of adult spring Chinook and expected juvenile emergence timing. USFWS rotary trapping operations on Battle and Clear Creeks between 2003 and 2012 have not predicted emergence (i.e., through temperature unit analyses; Beacham and Murray 1990) nor sampled juvenile spring Chinook prior to November of each year. On average, the first spring Chinook juvenile migrants from Battle and Clear Creeks were sampled during the week of November 26th each year (USFWS, unpublished data). As a result, LAD criteria used to identify juvenile spring Chinook at RBDD are noticeably inaccurate as fish sampled prior to late November were not sampled upstream in primary production areas at that time of year.

Simulating a removal of all LAD spring run between October 16 and November 25 of each year sampled would result in *decreased* spring run passage estimates by 19%, on average (range 2.6 to 44.2%). The effects of removing incorrectly assigned fry annually did not indicate a statistically significant difference between annual estimates (paired t-test, N = 11, P < 0.001). When incorrectly assigned fry are removed, the slightly more accurate simulated spring Chinook annual passage values remain within the 90% CI of standard estimates.

Furthering the simulation by adding the weekly October through November spring Chinook estimated passage to the winter Chinook passage estimates (i.e., late spawning or emerging winter run most likely candidate; see USFWS 2013), had minimal effect on the magnitude of winter Chinook passage. The average *increase* to winter Chinook passage was a mere 2.6% (range 0.6 to 8.8%) and simulated passage remained within the 90% CI of the annual winter Chinook estimates in all years.

Winter Chinook average annual juvenile passage was the second highest of the four runs estimated at 3,763,362 (Table 5c). The CV of the annual estimates was 73.2%; higher than fall or spring, but moderately dispersed. Overall, passage in years 2002, 2003, 2005, and 2006 surpassed the highest previous value of winter Chinook passage since juvenile monitoring began in 1995 (Gaines and Martin 2002). Similar to fall Chinook, winter Chinook adult escapement and subsequent juvenile passage began a marked decline in 2007 (Figure 16). Juvenile winter Chinook have been determined to enter the ocean during March and April of each spring (Pyper et al. 2013). Overall, it is believed that juvenile winter Chinook suffered the same fate as juvenile fall Chinook with poor marine conditions upon ocean entry in the spring of 2005 and 2006. Winter Chinook juvenile cohort replacement rates dropped below 1.0 starting with BY 2007, similar to adult fall run as noted in Lindley et al. (2009). The lowest passage estimate between 2002 and 2012 for winter Chinook occurred in 2011 at 848,976. Not until 2014 will we know if adult or juvenile cohort replacement rates will improve to a value of 1.0 or greater. Winter Chinook passage estimates between BY 1999 to BY 2002 (Gaines and Poytress 2003) indicate that replacement rates can vary substantially and replacement rates of 3.0 or greater have been estimated between juvenile cohorts.

Late-fall Chinook passage averaged 447,711 juveniles for the 11-year period and exhibited the greatest amount of variability with a CV of 159.9%. Late-fall Chinook juvenile passage estimates are likely affected by LAD criteria similar to spring Chinook in terms of potential for overestimation. The variability associated with weekly late-fall passage shows a decrease in median abundance by the beginning of June each year which may be more representative of actual late-fall emergence. Additionally, as demonstrated by Figures 13 a/b, the late-fall migration starts abruptly unlike for fall and winter Chinook which follow a more bell-shaped pattern in abundance (See Figures 11a/b and 15 a/b). It was highly likely that early emergent late-fall fry were, in fact, late emerging fall Chinook. Run specific genetic monitoring (Banks et al. 2000, Banks and Jacobsen 2004) could assist in determining the magnitude of the error in run assignment.

Sampling effort during mid-April to mid-May, the early late-fall run emergent period, was also typically low in an effort to reduce impacts to CNFH fall Chinook production fish caught in rotary traps. Within trap predation of fry by CNFH production smolts could also negatively bias late-fall juvenile production estimates. Sub-sampling of portions of the day and night (≤25% of each period) were only feasible with full staffing in some years which can reduce potential bias. During all other years, multiple sample days were typically sacrificed to allow peaks in CNFH production fish to recede ultimately reducing the accuracy of late-fall passage estimates.

Fry-Equivalent Chinook Production Estimates.—Estimation and analyses of the productivity of salmon runs in the Upper Sacramento River basin can provide valuable information to a variety of interests. Management of California's complex water resources for agriculture, municipal, commercial, and ecological uses is an increasingly controversial and complex endeavor. Knowledge of the effects of manipulating water storage and river processes on the productivity of the Sacramento River fish populations can only benefit fishery and water operations managers in an attempt to balance the competing demands on the system. Reducing uncertainty associated with threatened and/or endangered fish population dynamics by employing knowledge of the abundance, migration timing, and variability of those populations over time can then inform the decision making processes guiding management of water and fishery resources into the future.

Fall Chinook fry-equivalent juvenile production indices (FEJPI; Table 6a) indicate a significant and moderate correlation with fall Chinook escapement estimates (Figure 20a). Approximately 53% of the variation associated with fall FEJPI's was attributed to the estimated number of females in the system above RBDD each year (Figure 20a). The CV of estimated fall run females was greater than 132% indicating wide dispersion of contributors to the juvenile population over the eleven-year period. Conversely, the CV of FEJPI's was relatively low valued at 43%. Furthermore, recruits per female and similarly egg-to-fry survival demonstrated moderately low average values of 749 and

13.9%, respectively, when compared to the estimated values for winter Chinook (Table 6a).

As noted in Kocik and Taylor (1987), factors limiting production are typically a combination of biotic and abiotic factors. The sources of variability relating to fall FEJPIs are directly and indirectly related to adult abundance, but abundance alone does not explain the low CV in fall run juvenile production. A simple, albeit incorrect, conclusion might be that adult escapement of fall Chinook in some years exceeds the useable spawning area of the system (Bovee 1982, Connor et al. 2001) or optimal spawning efficiency (Wales and Coots 1955). Upon closer examination of the likely origin(s) of juvenile production, the data indicate substantial variability in the distribution of fall run adults between the mainstem Sacramento River and tributaries, including Clear Creek and Battle Creek, between years. Proportions of returning adults within the mainstem and Battle Creek have demonstrated high degrees of variability (Figure 26). The overwhelming return of fall run to Battle Creek in 2002 resulted in the lowest value of fall Chinook recruits per female (N = 89) which was outside two standard deviations of the average (Table 6a). The number of adults returning to the CNFH clearly overwhelmed the capacity of Battle Creek to produce juveniles. Sub-optimal wetted useable spawning area (Bovee 1982), red superimposition (McNeil 1968, Heard 1978), and female stress resulting in egg retention (Neave 1953, Foerster 1968) were likely just some of the factors that reduced the overall productivity of the 2002 fall Chinook adults returning to the Upper Sacramento River.

In years when estimates of fall Chinook production were at their highest in terms of recruits/females (Table 6a), the proportions spawning in the mainstem and combined tributaries were closest to 50:50. Further examination indicates that when contributions from the Battle and Clear Creeks accounted for equal proportions (i.e., 25% each), peak values of ~1,500 recruits/females were estimated to have been produced resulting in the highest net spawning efficiency (Wales and Coots 1955). Optimal natural juvenile fall Chinook production values in the Upper Sacramento River system could result under some conditions if integration of restoration projects on Battle and Clear Creeks integrate with mitigation projects (e.g., CNFH production) for the mainstem Sacramento River. The effect of consistent hatchery fall Chinook production on Battle Creek irrespective of natural fish production in the Sacramento and Chinook-bearing tributaries should be considered for further evaluation as was noted in Williams (2006). The effects of restoration of Clear Creek appear to be providing production benefits on stream and basin wide scales. Management prerogatives and actions related to the CVP affect both factors, to varying degrees, and decisions should be prioritized to attain optimal results for both fisheries and water operations.

Late-fall Chinook FEJPIs indicated high variability (CV = 170%; Table 6b), but a strong correlation with escapement estimates (r^2 =0.67; Figure 20b). The magnitude of late-fall FEJPIs were consistently an order of magnitude less than FEJPIs of fall Chinook. One exception was 2002, which increased the CV for the eleven-year period by 100%

(Table 6b). The fall and late-fall adult Chinook escapement values of 2001 and 2002 were high compared to the other 10 years of data (Azat 2013). A large run of late spawning fall run may also have contributed to the large number of juvenile fish falling within the late-fall size-class according to LAD criteria, but the adult estimate could have suffered similar inaccuracies in run assignment. Variability in CV values of anadromous fish was described by Rothchild and Dinardo (1987) as being inversely related to the number of years included within the time series analyses. While 2002 appears to be an outlier in this data set, it is likely with more years of data collection and analyses the CV associated with late-fall production would be more commensurate with other runs of Chinook.

The stream-type migration strategy noted by Moyle (2002) and our size classification method categorized the majority of late-fall outmigrants as smolts ($\bar{y}$ = 62%) which inflated the late-fall FEJPIs greatly at times (Table 5b, Table 6b). Recruits per female and similarly egg-to-fry survival had low CVs and the lowest average values of 131 and 2.8%, respectively, in comparison to other runs (Table 6b). This was unexpected as this metric does not appear to apply well to a run that was sampled primarily as smolts ($\bar{y}$ = 62%) over eleven years. Moreover, fry-equivalent calculations based on a static fry-to-smolt survival estimate of 59% (Hallock undated) was unlikely to be an accurate constant for late-fall Chinook as it was calculated from hatchery-based fall Chinook survival data. The fact that correlations with adult escapement were determined to be significant and moderately strong was unexpected given the vagaries of sampling late-fall Chinook smolts and the use of the static 59% survival estimate inversely applied to the majority of the run sampled. Additionally, difficulties with performing carcass surveys for late-fall Chinook due to low visibility, winter flow events or logistical issues (Killam 2009 and 2012) typically result in sub-optimal sampling conditions and, assumedly, would reduce the accuracy of the adult estimate.

Overall, production of late-fall Chinook appears low and the run has been characterized by some as vulnerable to extinction (Moyle et al. 2008, Katz et al. 2012). Greater attention to the relatively low abundance levels and juvenile rearing habitat needs of this genetically distinct run (Banks et al. 2000, Garza et al. 2007, Smith et al. 2009) with its unique over-summering, relatively long freshwater residency (Randall et al. 1987) and large size-at-outmigration strategy (Zabel and Achord 2004) should be afforded. The life-history strategies of late-fall Chinook have likely allowed them to persist in the Upper Sacramento River system as they occupy a distinct ecological niche. Juvenile monitoring of this run could benefit greatly if confidence in the accuracy of run assignment of juveniles was examined using non-lethal genetic techniques (Harvey and Stroble 2013).

Comparisons between winter Chinook adults and juvenile production began early using data generated by this monitoring project. Martin et al. (2001) demonstrated a strong relationship with only 5 years of data. The annual analyses of the winter FEJPI and adult estimates continually indicated a strong relationship with the addition of each year's data (See Gaines and Poytress 2003, Poytress and Carrillo 2008, Poytress and Carrillo 2012). The analysis of the most recent 11 years of data continues to indicate a strong relationship between the two variables even as adult escapement values have varied an order of magnitude.

Winter Chinook FEJPIs indicated mild variability (CV = 67%; Table 6c) and a very strong level of significance and correlation with female adult escapement estimates (r^2 =0.90; Figure 20c). Intensive adult and juvenile monitoring for this ESA listed endangered species coupled with superlative sampling conditions, in most years, appears to have resulted in very high quality information regarding the status and trends in adult and juvenile population abundance.

Egg-to-fry survival estimates generated from annual winter Chinook data indicate a range of values between 15 and 49% (Table 6c). At first glance, this appeared counterintuitive based on the highly regulated Sacramento River system (e.g., flow and water temperatures) that typically exists during the winter Chinook spawning period. The average egg-to-fry survival estimate of 26% is considerably higher than that determined from other studies on Pacific salmonids ($\bar{y}$ = 15%; e.g., Wales and Coots 1955) but was consistent with highly regulated aquatic systems (Groot and Margolis 1991). A very low CV of 38% also appeared consistent with a regulated system. Recruits per female, similarly, indicated a low CV of 36% and the second highest average value of 1,349 (Table 6c).

Natural log transformed adult female estimates influenced juvenile production and a significant relationship was determined accounting for roughly half of the variability associated with egg-to-fry survival rates (r^2 = 0.51, *df* = 10, *P* = 0.012). Densities of winter Chinook spawners are much lower currently than in the years estimated following the completion of Shasta Dam (USFWS 2001). Completion of the re-engineered Anderson-Cottonwood Irrigation District fish ladders in 2001 resulted in greater access and subsequently a greater concentration of spawners in the uppermost reaches accessible to anadromous fish (USFWS 2006-2011). Competition for optimal spawning habitat can result in lower juvenile production if sub-optimal wetted useable spawning area (Bovee 1982), red superimposition (McNeil 1968, Heard 1978), and female stress resulting in egg retention (Neave 1953, Foerster 1968) occur to varying degrees. Low resolution carcass recovery data (e.g., reach specific) indicate an abundance of spawners utilizing the uppermost 6 river miles of the Sacramento River (USFWS 2006-2011) even as seemingly suitable habitat has been made available for approximately 20+ river miles downstream of the terminus at Keswick Dam (RM 302). Geist et al. (2002) studied physiochemical characteristics affecting redd site selection preferences by Chinook and different growth and development rates have been attributed to different segments within the same river (Wells and McNeil 1970). High resolution redd surveys or spawning area mapping employing a GIS spatial analytical framework (Earley et al. 2013b) may shed light on the variability associated with winter Chinook spawning habitat over a variety of adult abundance levels. Analyses of these types of data could result in less uncertainty over the annual specific density dependent mechanisms affecting juvenile production and provide direction for future restoration activities for winter Chinook.

Spring run Chinook FEJPIs were the lowest of all four runs monitored and indicated the lowest variability (CV = 41%; Table 6d). No relationship with female adult escapement estimates was detected (r^2 =0.00; Figure 20d) and may be attributed substantially to measurement error (Sokal and Rohlf 1995). Estimates of recruits per female averaged 3,122 and the egg-to-fry survival value averaged 61.5%. These values appear unreasonable outside of a hatchery environment and well above those found for other runs (this report) and other studies (e.g., Wales and Coots 1955, Groot and Margolis 1991). Individual annual estimates varied moderately (CV= 70.8%) and nearly half appeared highly unlikely, with some values exceeding the number of eggs deposited by spawners (Table 6d).

Spring Chinook juvenile fish production estimates at RBDD were the least accurate and currently constitute 2.1%, on average, of total annual Chinook production above RBDD. Mainstem Sacramento River spawner estimates ranged from a low of 0 to a high of 370 between 2002 and 2012. Annual indexes of spring Chinook adult abundance above RBDD during the same years constitute 2.7% of the total escapement estimated in the Sacramento River system (Azat 2013). Given the relatively sporadic and low adult abundance levels, vagaries of using LAD criteria and annual CNFH fall Chinook production releases with fractional mark rates, no relationship could be found between adult escapement and spring Chinook FEJPIs when attempting to use methods to correct for these inaccuracies. The effects of inaccurate spring run assignment did not appear to affect the FEJPIs of other runs (e.g., winter or fall run) and therefore were not considered biologically significant. Genetic monitoring of fry in the fall after emergence from tributaries where emergence and migration data is collected (e.g., Earley et al. 2013a) may allow for more accurate estimation of the contributions of this run to the Upper Sacramento River outmigrant population.

Green Sturgeon Capture Dynamics.—Rotary traps were originally constructed to sample outmigrating salmonid smolts, but have been effective in sampling a variety of downstream migrating fish (Volkhardt et al. 2007). Rotary traps sampling at RBDD have been effective at monitoring temporal and spatial trends in relative abundance of Green Sturgeon since 1995 (Gaines and Martin 2002).

Annual adult Green Sturgeon aggregations were observed behind the RBDD when gates were lowered each spring (Brown 2007). Green sturgeon larvae were captured in 2012 (Table 7), the first year the RBDD gates were not lowered as it was replaced by a permanent pumping plant (NMFS 2009). Spawning was determined to have occurred in multiple locations as far as 20 river miles upstream of RBDD (Poytress et al. 2009-2013). The location of the RBDD rotary traps has been confirmed to be within the Green

Sturgeon spawning grounds as eggs were sampled directly below the RBDD and upstream of the RBDD traps in multiple years (Poytress et al. 2009, 2010, 2012).

Total length distribution data from Green Sturgeon collections at RBDD indicate a narrow and consistent size-class of larvae (Figure 21a). These data are consistent with laboratory-based studies conducted by Kynard et al. (2005) on the behavior of early life intervals of Klamath River Green Sturgeon. Their study determined that larvae migrated during two distinct periods (i.e., two-step migration). The first migration of newly exogenous feeding larvae was determined to be an initial dispersion from production areas. The second migration (of juveniles) to overwintering areas occurred in the fall some 180 days after hatching, on average. Our rotary trap data suggest we are sampling exclusively the initial redistribution of larvae from egg incubation and hatching areas.

Benthic D-net sampling conducted by Poytress et al. (2010-2011) targeted the lowest portion of the water column (inverse of rotary traps) and consistently captured Green Sturgeon larvae of the same size-class and temporal distribution pattern as rotary traps. D-net samples were collected between May and early-August (See Figure 21b for corresponding RST data only) downstream of spawning areas in years 2008-2011; even as no larvae were collected by rotary traps in 2008. Larvae were sampled by both methods primarily in the thalweg and in river velocities >/= 1.3 ft/sec[8]. Conversely, zero *juveniles* were collected with benthic D-nets in a pilot study (Poytress et al. 2013) targeting this life-stage and habitat type in the benthos during the fall period. Rotary traps have collected a few sporadic juveniles (e.g., outliers; Figure 21a) over the entire sample record of the project. These data indicate that Green Sturgeon juveniles are no longer utilizing our sampling region or more likely using a different habitat type (Hayes et al. 1996). Accordingly, rotary traps appear to be a relatively ineffective gear type for sampling the secondary juvenile sturgeon migration.

Protections afforded to ESA listed southern distinct population segment of Green Sturgeon (since 2006), limited quantities of larvae, and the small size at capture have not allowed their drift distances (Auer and Baker 2002), rates (Braaten et al. 2008), or rotary trap efficiencies to be calculated for the initial dispersion migration of Sacramento River Green Sturgeon at RBDD. Relative abundance indices for Green Sturgeon were highly variable, typically low valued at <1.0 fish/ac-ft sampled (Table 7), and contained one extraordinarily strong year-class (Figure 21c). As noted by Allen and Hightower (2010), variations in recruitment by orders of magnitude between years is common among fish stocks. Moreover, strong and weak year classes greatly influence adult fish populations. Green sturgeon relative abundance indices should not be interpreted as recruitment to the adult population, but should be viewed as a production metric influencing recruitment (e.g., age-0 year class strength). Alternately,

[8] Rotary traps generally require a minimum water velocity of 1.2 ft/sec to operate properly. D-nets sampled velocities ranging from 1.3 – 6.6 ft/sec. RST' sampled velocities ranging from 1.3 – 6.3 ft/sec.

Green Sturgeon larvae relative abundance indices could be viewed as an indirect metric for adult spawning population densities *upstream* of RBDD if genetic monitoring were conducted consistently (Israel and May 2010).

Lamprey Capture Dynamics.— Similar to Green Sturgeon, rotary trap sampling for Chinook salmon has provided the additional benefit of capturing out-migrating lamprey ammocoetes and juveniles. Greater attention to this ancestor of the earliest vertebrates (Moyle 2002) has recently been paid by the USFWS since it was petitioned for listing under the ESA in 2003 (Nawa et al. 2003). Although not listed due to inadequate data on the species' range and threats, the USFWS has engaged in a strategy to collaboratively conserve and restore Pacific Lamprey throughout their native range. Through the formation and development of the Pacific Lamprey Conservation Initiative, an assessment of Lamprey populations in California has recently been completed (Goodman and Reid 2012). The assessment noted that Lamprey species had been extirpated from at least 55% of their historical habitat north of Point Conception, CA by 1985. Long-term monitoring data sets including the RBDD rotary trap data, utilizing temporal and spatial distribution patterns as well as size-class and relative abundance levels of lamprey, can aid in the assessment and conservation of this ecologically vital species (Close et al. 2002).

Variability in annual size-class total length distributions was typically minor for both lamprey life stages sampled (Figure 22a and Figure 23a). Ammocoetes were slightly smaller than macropthalmia and slightly more variable in their annual average length distributions valued at 110 mm TL (CV= 4.6%; Table 8a). Pacific Lamprey macropthalmia were the dominant life stage sampled and the median size at capture was consistently near 125 mm TL (CV= 1.6%; Table 8b). Adults, typically noted as outliers, were encountered in much lower frequencies and were considered upstream migrants inadvertently captured when the RBDD gates were lowered as they sought upstream passage around the partial migration barrier.

Temporal distribution patterns indicated that ammocoetes and macropthalmia migrate past RBDD year-round. Ammocoetes, on average, were sampled regularly throughout the year (Figure 22b), whereas macropthalmia moved, en masse, episodically between November and March (Figure 23b). These data are consistent with studies of macropthalmia in the Columbia River system as noted by Close et al. (1995) and Kostow (2002).

Relative abundance indices of ammocoetes (Figure 22c) varied little between years and little overall when compared with macropthalmia (Figure 23c). Macropthalmia abundance indices varied considerably between years (Table 8b). On average, macropthalmia relative abundance was six times that of ammocoetes indicating metamorphosis and redistribution to different habitats from those used for rearing by ammocoetes (Goodman and Reid 2012). Differences in the relative abundance CV's of the two life stages likely indicates differences in catchability (Hubert and Fabrizio 2007) or habitat use (Hayes et al. 1996), variable migration trigger effects, or variability in sampling effort that often occurred during periods of macropthalmia migration.

Water Temperature and Juvenile Fish Dynamics.—Slight variation within and among salmonid runs (including *O. mykiss*) and years was noted for water temperatures found at RBDD (Tables 9a-e). Nonetheless, Upper Sacramento River salmonids were subjected to a relatively wide 20 degree range of water temperatures. Temperatures were recorded between 44 and 64 degrees with the average being 55 degrees each year. As summarized in Vogel and Marine (1991), the range of temperatures experienced by Chinook fry and pre-smolt/smolts in the last 11 years of passage at RBDD have been within the optimal range of thermal tolerances for survival.

Sacramento River water temperatures below Shasta/Keswick dams can be managed at certain times of the year under some conditions through discharge management to provide selective withdrawal at submerged intakes (USBR 1991 & 1994, Vermeyen 1997). Ambient air temperatures typically regulate river water temperatures during winter and early spring periods while storage and flood control operations are preeminent. The water temperatures recorded during the last 11 years appear to have been favorable for extant spring run spawners, and more so for fall and late-fall run Chinook and *O. mykiss* spawner and outmigrant populations.

The most vulnerable Chinook run to temperature management operations conducted by the USBR is winter Chinook (NMFS 2009). Temperature management of the Sacramento River via Shasta/Keswick releases by the USBR for winter Chinook appeared to be effective during the last 11 years as evidenced by the relatively favorable and stable egg-to-fry survival estimates (Table 6c). Moreover, temperature management of the upper 50 river miles of the Sacramento River aimed at winter Chinook resulted in benefits to over-summering late-fall Chinook pre-smolts and a relatively small proportion of fall Chinook smolts.

Temperature management during the summertime aimed at winter Chinook may have indirectly favored the resident form of *O. mykiss*. As noted by Lieberman et al. (2001), altering the thermal regime and food web structure by way of temperature management likely affects the proportion of anadromous to resident forms in large rivers. Lamprey species have likely benefitted from temperature management as temperatures for early life stages of lamprey in the mainstem Sacramento River appear to have been, on average, optimal (Meeuwig et al. 2005) in the last 11 years (Table 9g).

Green Sturgeon have likely benefitted from temperature management efforts aimed at winter Chinook spawning and production, albeit less comprehensively. Van Ennennaam et al. (2005) determined Green Sturgeon egg development temperatures to be optimal between 57.0 and 63.5° F. Mayfield and Cech (2004) determined optimal temperatures for larval development to be between 59.0 and 66.2°F. Temperatures recorded at RBDD during larval capture periods averaged 58.3°F and were generally within sub-optimal (lower end) to optimal ranges (Table 9f). A weak negative relationship between Green Sturgeon CPUV and water temperatures was detected in our analysis indicating greater capture rates at lower water temperatures (Figure 24d). The slightly sub-optimal temperatures might result in larvae migrating from incubation areas prematurely. Conversely, the optimal thermal environment of the lab-based migration data from Kynard et al. (2005) resulted in very similar migration timing between the lab and larval captures in rotary traps in terms of days post hatch (Poytress et al. 2013). Sacramento River Green Sturgeon larvae appear to be following their natural life-history migration patterns as opposed to being coerced from their incubation areas due to sub-optimal water temperatures at RBDD. This may not be true for larvae migrating some 20 miles upstream where the effects of temperature management may have a more pronounced negative effect on Green Sturgeon larvae (Poytress et al. 2013). Temperature management for Chinook may also have the indirect negative effect of redirecting the spawning habitat of Green Sturgeon adults by 20 river miles. A habitat comparison study on the relative value of the upper 20 river miles of the Sacramento River versus 20 lower river miles of habitat currently benefitting Green Sturgeon adult spawners and eggs from temperature management efforts should be conducted.

River Discharge, Turbidity, and Juvenile Fish Dynamics.—Volkhardt et al. (2007) stated that "flow" (i.e., discharge) was a dominant factor in juvenile trapping operations. Trapping efficiency and migration rates are affected by flow and the RBDD rotary trap passage data reflect these statements well. Exploratory plots demonstrating fry (Appendix 2, Figures A1-A11) and pre-smolt/smolt winter Chinook passage (Appendix 2, Figures A12-A23) were produced to illustrate the effects of environmental variables on fish migration. Turbidity was plotted, but not included in the final plots presented as the effects could not be deciphered from discharge at the daily scale of analyses.

The effects of river discharge on turbidity and resultant fish passage are complex in the Upper Sacramento River where ocean and stream-type Chinook of various size-classes (i.e., runs, life stages and ages) migrate daily throughout the year. Decreases in discharge in the Shasta/Keswick dam regulated Sacramento River, typical of late summer to early winter periods, appear to coincide with relatively clear water conditions and low turbidity (e.g., ~ 1.5 NTU) at RBDD. Fall or early winter freshets and winter rain-driven storm events result in highly variable increases in discharge levels and turbidity measures in terms of the magnitude and duration depending upon the source(s) of run-off.

A course scale analyses of fish passage and river discharge and turbidity measurements during storm events typically indicates a pattern that fish passage increases with simultaneous increases in both variables. Inspection of Chinook passage on a daily time step typically demonstrate a reduction in fish passage a day prior to a storm or rain-event during periods of stable river discharge. As storms produced increases in run-off or discharge from tributary inputs outside of the Shasta/Keswick dam complex, mean daily turbidity typically increased and fish passage began to increase. When storm related increases in discharge diminished, turbidity diminished, but Chinook passage often increased greatly for 24-72 hours after the peak flow event.

One problem confounding the results of storm and fish passage observations and analyses was that sampling during large storm run-off/discharge events often ceased due to safety concerns, concerns for fish impacts or simply due to the inability to sample the river when woody debris stop rotary traps from operating properly. In some years, storm events resulted in discharge levels too great to sample effectively or damaged traps which resulted in numerous days or weeks un-sampled afterwards. The results are typically negative bias in passage estimates if days following the peak discharge or concurrent turbidity events are un-sampled. Alternately, the direction of bias can be positive depending on time of year, interpolation methods, sample effort during extended storm periods, or fish developmental stage.

A fine scale, hourly analysis of fish passage, river discharge and turbidity during storm events indicated a more intricate relationship between the variables. As a comparison, two separate storm events (December 2005 and November 2012) were analyzed (Figure 27a/b). In 2005, 24-hour samples were conducted prior to and after the peak flow period which was missed due to an inability to sample the river as it more than quintupled in discharge (i.e., 7,000 CFS to ~35,000 CFS). During this storm event, sampling was conducted following the peak of river discharge as river stage decreased, but while turbidity continued to peak (Figure 27a). The planned 24-hour sample had to be cut short due to the huge influx of fry and smolt passage that occurred during the turbidity increase (i.e., from 10's to 1,000's per hour) and the need to reduce the potential impact to listed winter Chinook.

During a November 2012 storm event, a different strategy was employed to collect data more effectively throughout the storm period. For this event, we randomly sampled portions of the day and night in an attempt to manage the huge influx of fish anticipated to occur during the year's first storm event. Between 11/17/12 and 11/23/12, the project was able to collect 7-randomly selected samples that occurred throughout the first major river stage increase (Figure 27b). Samples were collected during increases and decreases in river stage. Samples were also collected prior to, during, and following a substantial increase in turbidity that lagged behind the initial stage increase by nearly 12 hours (Figure 27b). Fry and pre-smolt/smolt Chinook and juvenile lamprey fish passage increased exponentially. The peak period of fish capture occurred following the peak in river stage and during the increase and peak periods of turbidity measurements taken at RBDD. Capture rates subsided in the following days, but then increased greatly during the night-time period at the beginning of the next stage increase (Figure 27b).

Overall, it appears that flow and turbidity are important drivers for fish passage. The RBDD rotary trap data indicate that increased turbidity often results in greater fish passage than increases in river discharge or stage alone which often occur as part of water management operations at Shasta Dam. The two variables generally increase sequentially with discharge increases followed by turbidity increases (Figure 27a/b). Fish passage increases often coincide with the increase in turbidity which can often be sampled more effectively than increases in river discharge and may result in positive bias of juvenile fish passage estimates if the peak turbidity event is sampled compared to the peak flow event.

The importance of the first storm event of the fall or winter period cannot be overstated. Chinook smolt and juvenile lamprey passage increased exponentially and fry passage can be significant if first storms occur as fall Chinook begin to emerge. Fishery and water operations managers should be aware of the importance of the first Sacramento River stage increases following the summer and fall Sacramento River flow regulation period. The redistribution of winter and over-summering fall and late-fall Chinook smolts, or more generally, all anadromous juvenile fish[9] migrating from the Upper Sacramento River to the lower river and Sacramento San-Joaquin Delta with the first storm events of each water year should be incorporated into management plans for Delta operations.

Moon Illuminosity and Juvenile Fish Dynamics.—As noted in Hubert and Fabrizio (2007), species and life stages within species exhibit differing behaviors and therefore catchability in response to light levels. Gaines and Martin (2002) determined that Chinook passage occurred primarily during nocturnal periods except when turbidity levels and discharge increased with storm events. Further analyses of the effects of moon phase and ambient light levels in a statistical framework may be warranted for Chinook salmon as trends were detected based on observations. Rotary trap passage data indicated winter Chinook fry exhibit decreased nocturnal passage levels during and around the full moon phase in the fall (Appendix 3, Figures A1-A11). Pre-smolt/smolt winter Chinook appeared less influenced by night-time light levels and much more influenced by changes in discharge levels (Appendix 3, Figures A12-A23). A similar phenomenon was noted by Reimers (1971) for juvenile fall Chinook in Edson Creek, Oregon. Alternately, more data concerning night time cloud cover may further clarify the behavior associated with moon illuminosity as pre-smolt/smolts were more likely to encounter unclear night time weather between late October and December each year.

Spring, fall and late-fall Chinook fry exhibited varying degrees of decreased passage during full moon periods, albeit storms and related hydrologic influx dominated peak migration periods. *O. mykiss* relative abundance was not analyzed with respect to moon illuminosity. Lamprey CPUV regression analyses indicated a significant, but nearly imperceptible relationship (Figure 25a) likely due to the fact that lamprey are captured throughout the year under nearly all conditions. Green Sturgeon regression analysis

[9] Juvenile Green Sturgeon have been captured sporadically during the first flow events along with large numbers of Pacific Lamprey juveniles and ammocoetes.

indicated no significant linear relationship between moon illuminosity and relative abundance (Figure 24a). Migration of age-0 Green Sturgeon larvae has been determined to occur during nocturnal hours (Kynard et al. 2005) primarily between 21:00 and 02:00 using D-nets (Poytress et al. 2011) and was presumed to be similar for rotary traps as periodic diel sampling events have not collected sturgeon during daytime sample periods.

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Tables

Table 1. Summary of annual RBDD rotary trap sample effort by run and species for the period April 2002 through September 2013, by brood year (BY).

BY	Fall	Late-Fall	Winter	Spring	*O. mykiss*
2002	0.76	0.57	0.64	0.75	0.53
2003	0.81	0.76	0.81	0.81	0.76
2004	0.85	0.88	0.84	0.85	0.83
2005	0.56	0.73	0.64	0.57	0.83
2006	0.90	0.70	0.83	0.89	0.59
2007	0.88	0.90	0.89	0.89	0.91
2008	0.79	0.89	0.87	0.85	0.89
2009	0.84	0.72	0.75	0.79	0.76
2010	0.75	0.86	0.81	0.77	0.85
2011	0.87	0.77	0.82	0.86	0.76
2012	0.85	0.89	0.89	0.86	0.86
Min	0.56	0.57	0.64	0.57	0.53
Max	0.90	0.90	0.89	0.89	0.91
Mean	0.81	0.79	0.80	0.81	0.78
SD	0.094	0.104	0.088	0.091	0.122
CV	11.7%	13.2%	10.9%	11.3%	15.6%

Table 2. Summary of mark-recapture experiments conducted by RBDD rotary trap project between 2002 and 2013. Summaries include trap effort data, fish release and recapture group sizes (*N*) and mean fork lengths (FL), percentage of river discharge sampled (%Q) and estimated trap efficiency for each trial (%TE). Model data below each trial period indicate dates model was employed, total trials incorporated into model and linear regression values of slope, intercept, p-value and coeeficient of determination.

Date	Run	# Traps Sampling	Traps Modified	RBDD Gates	Release Group *N*	Release Group FL (mm)	Recapture Group *N*	Recapture Group FL (mm)	%Q	%TE
6/26/2002	Fall[1]	4	Yes	Lowered	805	68.7	8	61.3	1.58	0.99
8/6/2002	Fall[1]	4	Yes	Lowered	743	69.7	16	80.2	1.66	2.15
8/20/2002	Fall[1]	3	Yes	Lowered	340	76.5	7	77.7	1.41	2.06

Model	Employed	#Trials	Slope	Intercept	*P*	R^2
7/1/2002 -	6/30/2003	61	0.00792	0.00003205	<0.0001	0.394

Date	Run	# Traps Sampling	Traps Modified	RBDD Gates	Release Group *N*	Release Group FL (mm)	Recapture Group *N*	Recapture Group FL (mm)	%Q	%TE
1/28/2003	Fall	4	Yes	Raised	5,143	36.8	33	37.0	0.75	0.64
2/5/2003	Fall	4	Yes	Raised	2,942	36.7	10	37.9	1.36	0.34
2/10/2003	Fall	4	Yes	Raised	3,106	37.8	29	37.9	1.59	0.93
2/21/2003	Fall	3	Yes	Raised	3,256	37.4	15	37.3	0.72	0.46
2/26/2003	Fall	4	Yes	Raised	2,019	37.0	22	37.2	1.14	1.09
3/1/2003	Fall	4	No	Raised	1,456	37.0	31	37.0	3.31	2.13
3/4/2003	Fall	4	No	Raised	1,168	37.1	28	37.4	3.76	2.40
3/7/2003	Fall	4	No	Raised	1,053	37.4	22	36.6	3.58	2.09
3/20/2003	Fall	3	No	Raised	1,067	38.2	17	38.3	2.83	1.59
9/2/2003	Winter	4	No	Lowered	1,119	37.1	14	36.1	2.03	1.25
9/5/2003	Winter	3	No	Lowered	1,283	36.7	26	37.2	2.52	2.03
9/8/2003	Winter	3	No	Lowered	1,197	37.3	30	37.1	2.57	2.51
9/23/2003	Winter	3	No	Raised	1,012	35.5	18	35.6	2.20	1.78

9/27/2003	Winter	4	No	Raised	1,017	36.9	28	36.6	2.93	2.75
10/1/2003	Winter	4	No	Raised	1,064	37.6	20	36.7	3.09	1.88
10/6/2003	Winter	4	No	Raised	999	37.2	22	36.8	2.82	2.20
10/10/2003	Winter	4	No	Raised	1,017	38.1	16	38.3	3.06	1.57
10/15/2003	Winter	4	No	Raised	1,209	38.0	26	37.6	2.98	2.15

Model	Employed	#Trials	Slope	Intercept	*P*	R^2
7/1/2003 -	6/30/2004	79	0.00752	0.00046251	<0.0001	0.426

		# Traps	Traps		Release Group		Recapture Group			
Date	Run	Sampling	Modified	RBDD Gates	*N*	FL (mm)	*N*	FL (mm)	%Q	%TE
1/18/2004	Fall	4	Yes	Raised	2,074	37.1	26	37.1	1.52	1.25
1/24/2004	Fall	4	Yes	Raised	2,018	38.4	36	37.4	1.79	1.78
1/31/2004	Fall	4	Yes	Raised	2,024	37.7	33	37.6	1.61	1.63
2/6/2004	Fall	4	Yes	Raised	1,999	37.9	31	38.0	1.61	1.55
2/9/2004	Fall	4	Yes	Raised	2,017	37.8	27	37.0	1.69	1.34
2/13/2004	Fall	4	Yes	Raised	2,009	37.2	31	38.3	1.87	1.54
3/14/2004	Fall	3	No	Raised	1,401	38.3	18	39.6	1.98	1.28
3/23/2004	Fall	3	No	Raised	815	38.8	15	39.1	2.50	1.84
4/28/2004	Fall[1]	4	Yes	Raised	1,304	72.9	33	71.7	1.94	2.53
5/4/2004	Fall[1]	4	No	Raised	814	75.5	18	75.1	3.35	2.21
5/18/2004	Fall[1]	4	No	Lowered	867	80.2	10	75.1	3.20	1.15
5/26/2004	Fall[1]	4	No	Lowered	1,096	81.2	27	80.2	2.83	2.46
6/2/2004	Fall[1]	4	No	Lowered	888	76.2	28	77.2	2.77	3.15
6/15/2004	Fall[1]	4	No	Lowered	691	76.4	12	79.1	2.17	1.74
8/31/2004	Winter	4	No	Lowered	1,096	36.5	41	36.0	3.00	3.74
9/3/2004	Winter	4	No	Lowered	1,153	36.6	50	35.6	3.23	4.34
9/17/2004	Winter	4	No	Raised	1,023	36.0	14	35.4	2.52	1.37

9/20/2004	Winter	4	No	Raised	1,017	35.8	21	35.4	2.48	2.06
9/23/2004	Winter	4	No	Raised	2,006	36.0	31	35.1	2.62	1.55
9/27/2004	Winter	4	No	Raised	1,918	36.1	36	36.1	2.77	1.88
10/1/2004	Winter	4	No	Raised	1,682	36.4	24	36.0	3.11	1.43

Model	Employed	#Trials	Slope	Intercept	*P*	R^2
7/1/2004 -	6/30/2006	99	0.007464	0.00087452	<0.0001	0.385

		# Traps	Traps		Release Group		Recapture Group			
Date	Run	Sampling	Modified	RBDD Gates	*N*	FL (mm)	*N*	FL (mm)	%Q	%TE
1/23/2005	Fall	4	No	Raised	1,283	36.6	41	37.2	4.21	3.20
2/1/2005	Fall	3	Yes	Raised	1,971	36.6	31	36.0	1.35	1.57
2/10/2005	Fall	4	No	Raised	1,763	36.6	46	36.7	4.06	2.61
3/10/2005	Fall	4	No	Raised	1,216	36.6	27	36.5	3.93	2.22
3/13/2005	Fall	4	No	Raised	1,328	36.3	43	35.6	4.06	3.24
4/1/2005	Fall	4	No	Raised	1,949	57.1	50	62.3	3.49	2.57
9/11/2005	Winter	4	No	Lowered	1,437	35.6	14	38.9	2.22	0.97
10/4/2005	Winter	4	No	Raised	1,587	35.9	14	36.1	1.83	0.88
10/13/2005	Winter	4	No	Raised	1,577	35.7	21	36.6	2.33	1.33
2/15/2006	Fall	4	No	Raised	1,610	37.4	33	36.6	3.19	2.05
2/23/2006	Fall	4	No	Raised	1,503	37.2	38	36.6	2.68	2.53
1/21/2007	Fall	4	No	Raised	1,520	0.0	33	37.8	4.02	2.17
1/28/2007	Fall	4	Yes	Raised	1,987	37.6	18	37.8	3.65	0.91
2/5/2007	Fall	3	Yes	Raised	2,909	37.5	29	37.3	1.62	1.00
2/16/2007	Fall	4	No	Raised	1,782	37.9	34	38.5	3.51	1.91
3/2/2007	Fall	4	No	Raised	1,591	38.5	54	38.6	3.68	3.39
3/15/2007	Fall	4	No	Raised	953	37.6	26	37.6	4.29	2.73
3/20/2007	Fall	4	No	Raised	835	37.6	23	38.8	4.18	2.75

Date	Run	# Traps Sampling	Traps Modified	RBDD Gates	Release Group N	Release Group FL (mm)	Recapture Group N	Recapture Group FL (mm)	%Q	%TE
3/24/2007	Fall	4	No	Raised	944	37.7	23	38.0	4.24	2.44

Model	Employed	#Trials	Slope	Intercept	*P*	R^2
7/1/2006 -	6/30/2007	118	0.006653	0.00240145	<0.0001	0.420

		# Traps	Traps		Release Group		Recapture Group			
Date	Run	Sampling	Modified	RBDD Gates	*N*	FL (mm)	*N*	FL (mm)	%Q	%TE
1/23/2008	Fall	4	No	Raised	2,234	38.4	50	38.2	3.99	2.24
2/7/2008	Fall	4	Yes	Raised	2,324	38.1	60	37.9	2.19	2.58
2/14/2008	Fall	4	Mixed	Raised	1,993	38.4	83	38.8	3.40	4.16
2/20/2008	Fall	4	No	Raised	1,703	37.2	48	36.8	5.29	2.82
2/28/2008	Fall	3	No	Raised	2,080	37.6	63	38.3	3.45	3.03

Model	Employed	#Trials	Slope	Intercept	*P*	R^2
7/1/2007 -	6/30/2008	123	0.00645	0.00303101	<0.0001	0.414

		# Traps	Traps		Release Group		Recapture Group			
Date	Run	Sampling	Modified	RBDD Gates	*N*	FL (mm)	*N*	FL (mm)	%Q	%TE
1/23/2009	Fall	4	No	Raised	1,923	36.1	54	37.1	4.53	2.81
2/5/2009	Fall	4	No	Raised	1,868	36.8	58	37.4	4.65	3.10

Model	Employed	#Trials	Slope	Intercept	*P*	R^2
7/1/2008 -	6/30/2010	125	0.006332	0.00328530	<0.0001	0.425

		# Traps	Traps		Release Group		Recapture Group			
Date	Run	Sampling	Modified	RBDD Gates	*N*	FL (mm)	*N*	FL (mm)	%Q	%TE
1/20/2011	Fall	4	No	Raised	1,834	36.9	79	35.9	3.92	4.31
1/26/2011	Fall	4	No	Raised	1,989	37.6	109	36.0	4.56	5.48
2/1/2011	Fall	4	No	Raised	1,593	36.4	61	36.0	5.04	3.83

2/11/2011	Fall	4	No	Raised	1,582	35.7	81	37.4	5.34	5.12

Model	Employed	#Trials	Slope	Intercept	*P*	R^2
7/1/2010 -	6/30/2012	129	0.007297	0.00123101	<0.0001	0.493

		# Traps	Traps		Release Group		Recapture Group			
Date	Run	Sampling	Modified	RBDD Gates	*N*	FL (mm)	*N*	FL (mm)	%Q	%TE
1/30/2012	Fall	4	No	Raised	1,319	36.3	46	36.1	4.08	3.49
2/4/2012	Fall	4	No	Raised	1,146	35.8	51	35.4	5.52	4.45
2/16/2012	Fall	4	No	Raised	1,465	35.7	73	35.0	5.36	4.98
2/28/2012	Fall	4	No	Raised	1,228	35.5	57	34.6	5.40	4.64

Model	Employed	#Trials	Slope	Intercept	*P*	R^2
7/1/2012 -	6/30/2012	133	0.007676	0.00037735	<0.0001	0.561

		# Traps	Traps		Release Group		Recapture Group			
Date	Run	Sampling	Modified	RBDD Gates	*N*	FL (mm)	*N*	FL (mm)	%Q	%TE
1/16/2013	Fall	4	Yes	Raised	1,991	35.6	72	35.8	2.56	3.62
1/23/2013	Fall	4	Yes	Raised	1,965	35.9	39	35.3	2.61	1.98
1/30/2013	Fall	4	Yes	Raised	1,981	36.3	44	35.6	2.57	2.22
2/3/2013	Fall	4	Yes	Raised	1,998	36.5	42	36.1	2.69	2.10
2/13/2013	Fall	4	Yes	Raised	2,079	36.3	48	36.2	2.62	2.31
2/18/2013	Fall	4	Yes	Raised	2,156	36.1	35	36.8	2.89	1.62
2/22/2013	Fall	4	No	Raised	2,439	36.7	119	36.6	6.52	4.88
2/26/2013	Fall	4	No	Raised	1,400	36.1	65	37.3	6.87	4.64
3/3/2013	Fall	4	No	Raised	899	36.5	37	36.9	6.71	4.12

Model	Employed	#Trials	Slope	Intercept	*P*	R^2
7/1/2013 -	9/30/2013	142	0.007255	0.00150868	<0.0001	0.587

[1] Denotes Coleman National Fish Hatchery Fall Chinook production fish used during trial.

Table 3. Annual capture fork length summary of *O. mykiss* by age and life-stage classification from the RBDD rotary trap project between April 2002 through December 2012 by calendar year (CY).

Age Classification (%)					Life Stage Classification (%)					
CY	Fry <41 mm	Sub-Yearling 41-138 mm	Yearling 139-280 mm	2+ >280 mm	CY	Yolk-sac Fry	Fry	Parr	Silvery-parr	Smolt
2002	11.2	86.7	1.6	0.5	2002	0.0	6.3	54.4	37.2	2.1
2003	8.1	89.5	2.3	0.0	2003	0.0	5.6	57.7	34.9	1.8
2004	9.8	89.7	0.5	0.0	2004	0.0	4.6	60.2	34.7	0.5
2005	3.5	93.2	3.1	0.2	2005	0.0	2.8	48.7	45.6	2.9
2006	17.5	75.3	5.6	1.5	2006	0.2	9.2	78.9	9.2	2.4
2007	6.5	91.2	1.7	0.6	2007	0.1	8.7	85.3	5.3	0.6
2008	6.3	92.3	0.9	0.5	2008	0.1	8.2	79.4	12.0	0.4
2009	9.0	87.7	2.1	1.2	2009	0.0	10.7	82.8	5.1	1.4
2010	7.7	89.8	1.7	0.8	2010	0.3	9.7	87.4	1.7	1.0
2011	4.6	89.7	5.0	0.6	2011	0.1	3.5	90.9	2.8	2.7
2012	6.6	90.0	2.3	1.1	2012	0.2	5.9	88.2	4.2	1.5
Mean	8.3	88.7	2.4	0.6	Mean	0.1	6.8	74.0	17.5	1.6
SD	3.8	4.8	1.6	0.5	SD	0.1	2.6	15.5	16.8	0.9

Table 4. Annual linear regression equations with 95% confidence intervals (CI) for Log_{10} transformed juvenile (80-200 mm) *O. mykiss* weight-length data sampled at the RBDD rotary traps from April 2002 through December 2012 by calendar year (CY).

CY	Weight-Length Equation	R^2	Slope Lower 95% CI	Slope Upper 95% CI
2002	Log_{10}(weight)=2.843(Log_{10}FL)-4.616	0.903	2.648	3.039
2003	Log_{10}(weight)=2.968(Log_{10}FL)-4.886	0.968	2.885	3.052
2004	Log_{10}(weight)=3.005(Log_{10}FL)-4.941	0.952	2.879	3.132
2005	Log_{10}(weight)=3.03(Log_{10}FL)-5.009	0.952	2.929	3.132
2006	Log_{10}(weight)=3.052(Log_{10}FL)-5.085	0.917	2.811	3.293
2007	Log_{10}(weight)=2.961(Log_{10}FL)-4.864	0.947	2.853	3.069
2008	Log_{10}(weight)=2.939(Log_{10}FL)-4.819	0.942	2.833	3.044
2009	Log_{10}(weight)=3.017(Log_{10}FL)-4.981	0.974	2.922	3.112
2010	Log_{10}(weight)=2.977(Log_{10}FL)-4.911	0.934	2.836	3.118
2011	Log_{10}(weight)=2.911(Log_{10}FL)-4.778	0.939	2.743	3.078
2012	Log_{10}(weight)=2.858(Log_{10}FL)-4.662	0.903	2.746	2.970
Mean	Log_{10}(weight)=2.946(Log_{10}FL)-4.840	0.942	2.913	2.979

Table 5a. RBDD rotary trap fall Chinook total annual effort and passage estimates (sum of weekly values), lower and upper 90% confidence intervals (CI), ratio of fry to pre-smolt/smolt passage and ratio of estimated passage (Est) and interpolated passage (Interp) for brood year (BY) 2002-2012.

BY	Effort	Total	Low 90%CI	Up 90% CI	Fry	Smolt	Est	Interp
2002	0.76	17,038,417	857,106	47,315,257	0.86	0.14	0.54	0.46
2003	0.81	27,736,868	8,839,840	50,653,446	0.85	0.15	0.74	0.26
2004	0.85	14,108,238	5,079,300	24,967,671	0.56	0.44	0.70	0.30
2005	0.56	18,210,294	3,500,275	39,096,017	0.64	0.36	0.40	0.60
2006	0.90	16,107,651	6,522,666	26,414,402	0.63	0.37	0.85	0.15
2007	0.88	12,131,603	6,130,892	18,170,520	0.79	0.21	0.84	0.16
2008	0.79	9,115,547	4,381,560	13,849,709	0.73	0.27	0.81	0.19
2009	0.84	8,532,377	3,064,273	14,052,588	0.81	0.19	0.56	0.44
2010	0.75	8,842,481	4,727,816	13,252,907	0.71	0.29	0.79	0.21
2011	0.87	6,271,261	3,431,940	9,125,109	0.71	0.29	0.82	0.18
2012	0.85	24,429,420	16,028,521	33,112,943	0.87	0.13	0.91	0.09
Mean	0.81	14,774,923			0.74	0.26	0.72	0.28
SD	0.09	6,825,382			0.10	0.10	0.16	0.16
CV	11.7%	46.2%			13.9%	40.3%	22.0%	57.4%

Table 5b. RBDD rotary trap late-fall Chinook total annual effort and passage estimates (sum of weekly values), lower and upper 90% confidence intervals (CI), ratio of fry to pre-smolt/smolt passage and ratio of estimated passage (Est) and interpolated passage (Interp) for brood year (BY) 2002-2012.

BY	Effort	Total	Low 90%CI	Up 90% CI	Fry	Smolt	Est	Interp
2002	0.57	2,559,519	659,986	4,953,910	0.17	0.83	0.52	0.48
2003	0.76	346,058	78,407	911,270	0.57	0.43	0.56	0.44
2004	0.88	147,160	74,930	220,231	0.17	0.83	0.91	0.09
2005	0.73	143,362	41,800	333,415	0.35	0.65	0.71	0.29
2006	0.70	460,268	125,197	902,089	0.62	0.38	0.44	0.56
2007	0.90	535,619	271,079	800,447	0.27	0.73	0.86	0.14
2008	0.89	91,995	46,660	138,310	0.11	0.89	0.89	0.11
2009	0.72	219,824	97,294	342,652	0.13	0.87	0.73	0.27
2010	0.86	183,439	61,775	305,937	0.62	0.38	0.61	0.39
2011	0.77	97,040	28,738	165,997	0.72	0.28	0.53	0.47
2012	0.89	140,534	42,673	249,500	0.48	0.52	0.80	0.20
Mean	0.79	447,711			0.38	0.62	0.69	0.31
SD	0.10	715,999			0.23	0.23	0.16	0.16
CV	13.2%	159.9%			58.8%	36.5%	23.8%	52.5%

Table 5c. RBDD rotary trap winter Chinook total annual effort and passage estimates (sum of weekly values), lower and upper 90% confidence intervals (CI), ratio of fry to pre-smolt/smolt passage and ratio of estimated passage (Est) and interpolated passage (Interp) for brood year (BY) 2002-2012.

BY	Effort	Total	Low 90%CI	Up 90% CI	Fry	Smolt	Est	Interp
2002	0.64	7,119,041	2,541,407	12,353,367	0.90	0.10	0.58	0.42
2003	0.81	5,221,016	3,202,609	7,260,798	0.85	0.15	0.86	0.14
2004	0.84	3,434,683	1,998,468	4,874,794	0.90	0.10	0.82	0.18
2005	0.64	8,363,106	4,558,069	12,277,233	0.90	0.10	0.89	0.11
2006	0.83	6,687,079	3,801,539	9,575,937	0.87	0.13	0.76	0.24
2007	0.89	1,440,563	931,113	1,953,688	0.80	0.20	0.92	0.08
2008	0.87	1,244,990	776,634	1,714,013	0.85	0.15	0.77	0.23
2009	0.75	4,402,322	2,495,734	6,311,739	0.81	0.19	0.74	0.26
2010	0.81	1,285,389	817,207	1,756,987	0.68	0.32	0.92	0.08
2011	0.82	848,976	576,177	1,122,022	0.75	0.25	0.88	0.12
2012	0.89	1,349,819	904,552	1,795,106	0.53	0.47	0.92	0.08
Mean	0.80	3,763,362			0.80	0.20	0.82	0.18
SD	0.09	2,753,256			0.11	0.11	0.11	0.11
CV	10.9%	73.2%			13.9%	57.5%	12.8%	59.6%

Table 5d. RBDD rotary trap spring Chinook total annual effort and passage estimates (sum of weekly values), lower and upper 90% confidence intervals (CI), ratio of fry to pre-smolt/smolt passage and ratio of estimated passage (Est) and interpolated passage (Interp) for brood year (BY) 2002-2012.

BY	Effort	Total	Low 90%CI	Up 90% CI	Fry	Smolt	Est	Interp
2002	0.75	277,477	110,951	494,590	0.57	0.43	0.59	0.41
2003	0.81	626,915	249,225	1,053,421	0.80	0.20	0.67	0.33
2004	0.85	430,951	174,174	710,419	0.36	0.64	0.78	0.22
2005	0.57	616,040	131,328	1,382,036	0.69	0.30	0.58	0.42
2006	0.89	421,436	239,470	603,952	0.41	0.59	0.80	0.20
2007	0.89	369,536	229,766	510,868	0.91	0.09	0.99	0.01
2008	0.85	164,673	66,515	262,959	0.24	0.76	0.62	0.38
2009	0.79	438,405	176,952	700,959	0.50	0.50	0.51	0.49
2010	0.77	158,966	62,563	261,105	0.56	0.44	0.67	0.33
2011	0.86	184,290	101,443	272,769	0.48	0.52	0.85	0.15
2012	0.86	320,897	173,312	469,137	0.42	0.58	0.74	0.26
Mean	0.81	364,508			0.54	0.46	0.71	0.29
SD	0.09	164,135			0.20	0.20	0.14	0.14
CV	11.3%	45.0%			36.4%	43.0%	19.7%	47.6%

Table 5e. RBDD rotary trap *O. mykiss* total annual effort and passage estimates (sum of weekly values), lower and upper 90% confidence intervals (CI), and ratio of estimated passage (Est) and interpolated passage (Interp) for calendar year (CY) 2002-2012.

CY	Effort	Total	Low 90%CI	Up 90% CI	Est	Interp
2002[1]	0.53	124,436	27,224	244,701	0.53	0.47
2003	0.76	139,008	54,885	243,927	0.78	0.22
2004	0.83	151,694	86,857	218,132	0.95	0.05
2005	0.83	85,614	32,251	152,568	0.76	0.24
2006	0.59	83,801	20,603	169,712	0.44	0.56
2007	0.91	139,424	73,827	205,647	0.89	0.11
2008	0.89	131,013	69,331	193,584	0.88	0.12
2009	0.76	129,581	62,350	197,795	0.83	0.17
2010	0.85	100,997	47,050	155,692	0.74	0.26
2011	0.76	56,798	23,494	89,369	0.76	0.24
2012	0.86	136,621	78,804	194,892	0.96	0.04
Mean	0.78	116,272			0.78	0.22
SD	0.12	29,912			0.16	0.16
CV	15.6%	25.7%			20.9%	72.2%

[1] Incomplete year; sampling began in April 2002.

Table 6a. Fall Chinook fry-equivalent production estimates, lower and upper 90% confidence intervals (CI), estimates of adults upstream of RBDD (Adult Estimate), estimated female to male sex ratios, estimated females, estimates of female fecundity, calculated juveniles per estimated female (recruits per female) and egg-to-fry survival estimates (ETF) by brood year (BY) for Chinook sampled at RBDD rotary traps between December 2002 and September 2013.

BY	FRY EQ Passage	Lower 90% CI	Upper 90% CI	Adult Estimate	Sex Ratio (F: M)[1]		Estimated Females	Fecundity[2]	Recruits per Female	ETF
2002	18,683,720	1,216,244	51,024,926	458,772	*0.46*	*0.54*	211,035	5,407	89	1.6%
2003	30,624,209	10,162,712	55,109,506	140,724	0.57	0.44	79,509	5,407	385	7.1%
2004	18,421,457	6,224,790	33,728,746	64,276	0.48	0.52	31,045	5,407	593	11.0%
2005	22,739,315	4,235,720	49,182,045	80,294	0.47	0.53	37,738	5,407	603	11.1%
2006	20,276,322	8,670,090	32,604,760	78,692	0.54	0.46	42,730	5,407	475	8.8%
2007	13,907,856	7,041,759	20,838,463	31,592	0.54	0.46	16,996	5,407	818	15.1%
2008	10,817,397	5,117,059	16,517,847	36,104	0.46	0.54	16,644	5,407	650	12.0%
2009	9,674,829	3,678,373	15,723,368	12,908	0.51	0.49	6,531	5,407	1,481	27.4%
2010	10,620,144	5,637,617	15,895,197	29,321	0.24	0.76	7,008	5,407	1,515	28.0%
2011	7,554,574	4,171,332	10,960,125	31,931	0.29	0.71	9,260	5,407	816	15.1%
2012	26,567,379	17,219,525	36,197,837	65,664	0.50	0.50	32,635	5,407	814	15.1%
Mean	17,262,473	6,670,475	30,707,529	93,662	0.46	0.54	44,648		749	13.9%
CV	43.2%	64.0%	51.7%	134.7%			132.4%		57.2%	57.2%

[1] Sex ratios based on RBDD fish ladder data between 2003 and 2007 and CNFH data between 2008 and 2012. Average, in italics, input for 2002 due to lack of available data.
[2] Female fecundity estimates based on average values from CNFH fall Chinook spawning data collected between 2008 and 2012.

Table 6b. Late-fall Chinook fry-equivalent production estimates, lower and upper 90% confidence intervals (CI), estimates of adults upstream of RBDD (Adult Estimate), estimated female to male sex ratios, estimated females, estimates of female fecundity, calculated juveniles per estimated female, and egg-to-fry survival estimates (ETF) by brood year (BY) for Chinook sampled at RBDD rotary traps between April 2002 and March 2013.

BY	FRY EQ Passage	Lower 90% CI	Upper 90% CI	Adult Estimate	Sex Ratio (F: M)[1]		Estimated Females	Fecundity[2]	Recruits per Female	ETF
2002	4,041,505	1,063,720	7,808,619	36,220	0.46	0.54	16,661	4,662	243	5.2%
2003	451,230	133,225	1,067,819	5,513	0.46	0.54	2,536	4,662	178	3.8%
2004	233,106	124,245	342,837	8,924	0.46	0.54	4,105	4,662	57	1.2%
2005	209,066	70,548	441,133	9,610	0.46	0.54	4,421	4,662	47	1.0%
2006	582,956	186,984	1,086,699	7,770	0.46	0.54	3,574	4,662	163	3.5%
2007	809,272	426,272	1,192,625	13,939	0.46	0.54	6,412	4,662	126	2.7%
2008	149,049	80,500	218,597	3,747	0.46	0.54	1,724	4,662	86	1.9%
2009	353,003	159,726	546,546	3,792	0.46	0.54	1,744	4,662	202	4.3%
2010	232,279	89,343	376,286	3,961	0.46	0.54	1,822	4,662	127	2.7%
2011	116,188	38,688	194,400	3,777	0.46	0.54	1,737	4,662	67	1.4%
2012	191,672	69,229	325,189	2,931	0.46	0.54	1,348	4,662	142	3.0%
Mean	669,939	222,044	1,236,432	9,108			4,190		131	2.8%
CV	169.8%	134.4%	178.7%	105.5%			105.5%		48.1%	48.1%

[1] Sex ratio value of (0.46:0.54) is equivalent to the average ratio for fall Chinook between 2003 and 2012 used in Table 6a.

[2] Female fecundity estimates based on average values from CNFH late-fall Chinook spawning data collected between 2008 and 2012.

Table 6c. Winter Chinook fry-equivalent production estimates, lower and upper 90% confidence intervals (CI), estimates of adults upstream of RBDD (Adult Estimate), estimated female to male sex ratios, estimated females, estimates of female fecundity, calculated juveniles per estimated female (recruits per female) and egg-to-fry survival estimates (ETF) by brood year (BY) for Chinook sampled at RBDD rotary traps between July 2002 and June 2013.

BY	FRY EQ Passage	Lower 90% CI	Upper 90% CI	Adult Estimate	Sex Ratio (F: M)[1]		Estimated Females	Fecundity[2]	Recruits per Female	ETF
2002	7,635,469	2,811,132	13,144,325	7337	0.77	0.23	5,670	4,923	1,347	27.4%
2003	5,781,519	3,525,098	8,073,129	8133	0.64	0.36	5,179	4,854	1,116	23.0%
2004	3,677,989	2,129,297	5,232,037	8635	0.37	0.63	3,185	5,515	1,155	20.9%
2005	8,943,194	4,791,726	13,277,637	15730	0.56	0.44	8,807	5,500	1,015	18.5%
2006	7,298,838	4,150,323	10,453,765	17205	0.50	0.50	8,626	5,484	846	15.4%
2007	1,637,804	1,062,780	2,218,745	2488	0.61	0.39	1,517	5,112	1,080	21.1%
2008	1,371,739	858,933	1,885,141	2850	0.51	0.49	1,443	5,424	951	17.5%
2009	4,972,954	2,790,092	7,160,098	4537	0.60	0.40	2,702	5,519	1,840	33.3%
2010	1,572,628	969,016	2,181,572	1533	0.53	0.47	813	5,161	1,934	37.5%
2011	996,621	671,779	1,321,708	824	0.51	0.49	424	4,832	2,351	48.6%
2012	1,789,259	1,157,240	2,421,277	2581	0.58	0.42	1,491	4,518	1,200	26.6%
Mean	4,152,547	2,265,220	6,124,494	6,532	0.56	0.44	3,623	5,167	1,349	26.4%
CV	70.1%	64.0%	74.9%	85.7%	17.9%	22.9%	83.4%	6.7%	35.5%	37.9%

[1] Annual sex ratio values based on annual carcass survey estimates of female recoveries.

[2] Female fecundity estimates based on annual values from LSNFH winter Chinook spawning data collected between 2002 and 2012.

Table 6d. Spring Chinook fry-equivalent production estimates, lower and upper 90% confidence intervals (CI), estimates of adults upstream of RBDD (Adult Estimate), estimated female to male sex ratios, estimated females, estimates of female fecundity, calculated juveniles per estimated female (recruits per female) and egg-to-fry survival estimates (ETF) by brood year (BY) for Chinook sampled at RBDD rotary traps between October 16, 2002 and September 30, 2013.

BY	FRY EQ Passage	Lower 90% CI	Upper 90% CI	Adult Estimate	Sex Ratio (F: M)[1]		Estimated Females	Fecundity[2]	Recruits per Female	ETF
2002	360,352	142,134	657,043	608	0.46	0.54	280	5,078	1,288	25.4%
2003	714,086	293,095	1,187,827	319	0.46	0.54	147	5,078	4,866	95.8%
2004	624,079	255,886	1,029,162	575	0.46	0.54	265	5,078	2,359	46.5%
2005	747,026	146,488	1,695,236	189	0.46	0.54	87	5,078	8,592	169.2%
2006	594,511	328,845	860,757	353	0.46	0.54	162	5,078	3,661	72.1%
2007	392,451	242,563	544,184	767	0.46	0.54	353	5,078	1,112	21.9%
2008	251,795	96,737	406,863	305	0.46	0.54	140	5,078	1,795	35.3%
2009	591,549	238,710	945,904	314	0.46	0.54	144	5,078	4,095	80.7%
2010	207,793	80,320	344,475	208	0.46	0.54	96	5,078	2,172	42.8%
2011	251,444	130,051	382,077	167	0.46	0.54	77	5,078	3,273	64.5%
2012	451,705	238,187	665,825	868	0.46	0.54	399	5,078	1,131	22.3%
Mean	471,527	199,365	792,668	425			195		3,122	61.5%
CV	40.9%	41.7%	51.5%	56.8%			56.8%		70.8%	70.8%

[1] Sex ratio value of (0.46:0.54) is equivalent to the average ratio for fall Chinook between 2003 and 2012 used in Table 6a.
[2] Female fecundity estimates based on average of winter, fall, and late-fall hatchery data provided by CNFH and LSNFH; Table 6a-6c above.

Table 7. Green Sturgeon annual capture, catch per unit volume (CPUV) and total length summaries for sturgeon captured by RBDD rotary traps between calendar year (CY) 2002 and 2012.

CY	Captures	CPUV fish/ac-ft	Min TL (mm)	Max TL (mm)	Mean (mm)	Median (mm)
2002	35	0.3	23	52	28.8	27.5
2003	360	1.9	22	188	27.8	27
2004	266	1.0	21	58	30.5	29
2005	271	1.1	24	65	28.9	27
2006	193	0.8	21	79	30.5	28
2007	19	0.1	25	49	29.6	27
2008	0	0.0	-	-	-	-
2009	32	0.2	24	47	28.0	26
2010	70	0.5	20	36	27.1	27
2011	3701	20.1	18	86	27.4	27
2012	288	1.4	21	41	27.2	27
Ave	475.9	2.5	21.9	70.1	28.6	27.3
SD	1077.4	5.9	2.1	44.4	1.3	0.8
CV	226.4%	236.3%	9.7%	63.3%	4.5%	2.9%

Table 8a. Unidentified Lamprey ammocoetes annual capture, catch per unit volume (CPUV) and total length summaries for ammocoetes captured by RBDD rotary traps between water year (WY) 2003 and 2013.

WY	Captures	CPUV Fish/ac-ft	Min TL (mm)	Max TL (mm)	Mean (mm)	Median (mm)
2003	908	7.30	14	144	98	100
2004	925	6.80	27	191	105	108
2005	1415	11.65	22	159	104	108
2006	657	4.45	52	186	112	115
2007	556	5.16	29	155	105	111
2008	385	3.64	41	146	101	108
2009	593	5.53	41	150	106	112
2010	935	11.45	45	166	111	114
2011	859	7.07	30	186	111	117
2012	455	5.11	27	155	100	104
2013	632	6.45	25	160	103	107
Mean	756.4	6.8	32.1	163.5	105.1	109.5
SD	291.3	2.6	11.3	16.8	4.7	5.0
CV	38.5%	38.5%	35.1%	10.3%	4.5%	4.6%

Table 8b. Pacific Lamprey macrothalmia and adult annual capture, catch per unit volume (CPUV) and total length summaries for macrothalmia captured by RBDD rotary traps between water year (WY) 2003 and 2013.

WY	Captures	CPUV Fish/ac-ft	Min TL (mm)	Max TL (mm)	Mean (mm)	Median (mm)
2003	204	2.16	100	693	261	131
2004	478	3.91	96	630	149	125
2005	4645	45.00	72	665	137	126
2006	417	5.62	98	700	136	125
2007	3107	34.08	96	660	150	128
2008	5252	40.29	78	580	139	128
2009	2938	81.24	91	834	132	124
2010	699	32.30	80	819	136	125
2011	2747	68.18	92	620	140	129
2012	3464	112.76	86	500	136	127
2013	1734	25.63	88	617	131	127
Mean	2335.0	41.0	88.8	665.3	149.7	126.8
SD	1759.4	34.7	9.0	97.1	37.3	2.1
CV	75.3%	84.5%	10.2%	14.6%	24.9%	1.6%

Table 9a. Summary of fall Chinook abiotic sample conditions at RBDD rotary traps during dates of capture by brood year (BY).

	Dates of Capture			H_20 Temperature (°F)			Discharge (CFS)			Turbidity (NTU)		
BY	Initial	Final	Days	Min	Max	Ave	Min	Max	Ave	Min	Max	Ave
2002	4-Dec	30-Aug	269	47	61	55	6,390	86,500	17,471	0.5	240.2	19.6
2003	9-Dec	15-Aug	250	46	62	55	7,380	92,800	18,707	2.0	413.5	21.8
2004	8-Dec	29-Aug	264	46	63	56	5,390	76,200	13,315	1.9	626.5	24.6
2005	3-Dec	29-Aug	269	47	61	53	6,450	118,000	27,279	1.6	731.7	22.5
2006	10-Dec	26-Aug	259	46	62	55	6,030	45,400	10,628	1.6	90.0	8.0
2007	7-Dec	2-Sep	270	44	62	55	5,210	44,600	10,127	1.5	233.3	11.1
2008	5-Dec	4-Sep	273	45	64	56	4,160	33,000	9,297	2.1	129.8	12.0
2009	10-Dec	21-Aug	254	45	61	54	5,260	95,100	17,531	1.3	162.6	10.3
2010	7-Dec	29-Aug	265	45	61	54	5,260	95,100	17,331	1.3	162.6	10.2
2011	10-Dec	2-Sep	267	45	65	55	4,800	35,200	10,281	1.4	180.6	8.8
2012	2-Dec	23-Aug	264	44	64	56	5,330	70,400	11,323	1.5	315.5	9.9
Mean	7-Dec	27-Aug	264	45	62	55	5,605	72,027	14,844	1.5	298.7	14.4
SD			7	1.1	1.4	0.8	890	28,600	5,442	0.4	209.6	6.3
CV			3%	2%	2%	1%	16%	40%	37%	28%	70%	44%

Table 9b. Summary of late-fall Chinook abiotic sample conditions at RBDD rotary traps during dates of capture by brood year (BY).

	Dates of Capture			H_20 Temperature (^oF)			Discharge (CFS)			Turbidity (NTU)		
BY	Initial	Final	Days	Min	Max	Ave	Min	Max	Ave	Min	Max	Ave
2002	19-Apr	14-Jan	270	47	62	57	6,176	86,500	12,981	0.4	59.7	11.3
2003	3-Apr	6-Mar	338	46	61	55	6,310	92,800	16,650	0.9	413.5	20.9
2004	2-Apr	21-Jan	294	46	62	57	5,170	57,000	10,983	1.4	470.0	8.0
2005	2-Apr	22-Jan	295	48	63	57	6,050	118,000	17,431	1.6	731.7	24.4
2006	1-Apr	13-Jan	287	46	61	55	6,610	80,900	15,374	2.0	178.0	8.8
2007	4-Apr	9-Jan	280	46	62	57	5,490	38,600	10,035	1.3	198.0	5.7
2008	2-Apr	2-Mar	334	45	64	56	4,160	33,000	8,775	1.5	129.8	6.9
2009	3-Apr	1-Mar	332	46	64	57	3,920	60,400	9,855	1.9	250.6	14.2
2010	1-Apr	12-Jan	286	47	62	56	5,900	50,600	11,831	1.1	220.3	7.3
2011	1-Apr	27-Jan	301	45	61	55	5,570	57,400	11,888	2.0	68.5	5.5
2012	2-Apr	11-Jan	284	46	62	56	5,536	67,520	12,580	1.4	272.0	11.3
Mean	4-Apr	29-Jan	300	46	62	56	5,536	67,520	12,580	1.4	272.0	11.3
SD			24	0.9	1.0	0.7	849	25,109	2,829	0.5	198.7	6.2
CV			8%	2%	2%	1%	15%	37%	22%	34%	73%	55%

Table 9c. Summary of winter Chinook abiotic sample conditions at RBDD rotary traps during dates of capture by brood year (BY).

	Dates of Capture			H_20 Temperature (°F)			Discharge (CFS)			Turbidity (NTU)		
BY	Initial	Final	Days	Min	Max	Ave	Min	Max	Ave	Min	Max	Ave
2002	4-Jul	8-Apr	278	47	61	55	6,176	86,500	14,081	0.4	240.2	13.5
2003	16-Jul	17-Mar	245	46	61	54	6,310	92,800	16,809	0.9	413.5	22.8
2004	22-Jul	25-Mar	246	46	62	55	5,170	57,000	9,817	1.4	470.0	12.1
2005	25-Jul	17-Feb	207	48	61	55	6,450	118,000	19,174	1.6	731.7	19.7
2006	16-Jul	10-Mar	237	46	59	54	6,030	45,400	9,788	1.6	90.0	7.2
2007	18-Jul	4-Apr	261	44	62	54	5,210	44,600	9,318	1.3	233.3	11.3
2008	30-Jul	24-Apr	268	45	64	55	4,160	33,000	7,647	1.5	129.8	8.2
2009	26-Jul	30-Mar	247	46	64	55	3,920	60,400	9,303	1.9	250.6	15.0
2010	18-Jul	7-Apr	263	45	61	54	5,260	95,100	14,941	1.1	162.6	8.6
2011	12-Aug	31-Mar	232	45	60	53	4,800	35,200	8,646	1.7	180.6	7.0
2012	23-Jul	19-Apr	270	46	61	55	5,349	66,800	11,952	1.3	290.2	12.5
Mean	22-Jul	28-Mar	250	46	61	55	5,349	66,800	11,952	1.3	290.2	12.5
SD			20	1.1	1.5	0.8	843	27,776	3,767	0.4	185.4	5.1
CV			8%	2%	2%	1%	16%	42%	32%	31%	64%	41%

Table 9d. Summary of spring Chinook abiotic sample conditions at RBDD rotary traps during dates of capture by brood year (BY).

	Dates of Capture			H_20 Temperature (^oF)			Discharge (CFS)			Turbidity (NTU)		
BY	Initial	Final	Days	Min	Max	Ave	Min	Max	Ave	Min	Max	Ave
2002	16-Oct	29-May	225	47	61	54	6,176	86,500	16,877	0.4	240.2	19.1
2003	16-Oct	11-Jun	239	46	62	54	6,310	92,800	17,267	0.9	413.5	23.0
2004	16-Oct	3-Jun	230	46	63	54	5,170	76,200	11,612	1.4	626.5	27.6
2005	16-Oct	3-Jun	230	47	61	52	6,450	118,000	28,158	1.6	731.7	25.3
2006	16-Oct	26-May	222	46	62	53	6,030	45,400	8,630	1.6	90.0	8.3
2007	16-Oct	12-Jun	240	44	61	53	5,210	44,600	8,823	1.3	233.3	11.4
2008	16-Oct	7-Jun	234	45	64	54	4,160	33,000	7,841	1.7	129.8	10.1
2009	16-Oct	25-May	221	46	62	54	3,920	60,400	9,495	1.9	250.6	17.1
2010	16-Oct	12-Jun	239	45	61	53	5,260	95,100	16,656	1.3	162.6	9.9
2011	16-Oct	27-May	224	45	65	53	4,800	35,200	8,344	1.7	180.6	8.8
2012	16-Oct	23-Jun	250	46	62	53	5,349	68,720	13,370	1.4	305.9	16.0
Mean	16-Oct	4-Jun	232	46	62	53	5,349	68,720	13,370	1.4	305.9	16.0
SD			9	1.0	1.4	0.6	843	27,696	6,116	0.4	205.5	7.0
CV			4%	2%	2%	1%	16%	40%	46%	30%	67%	43%

Table 9e. Summary of *O. mykiss* abiotic sample conditions at RBDD rotary traps during dates of capture by calendar year (CY).

	Dates of Capture			H_20 Temperature (^oF)			Discharge (CFS)			Turbidity (NTU)		
CY	Initial	Final	Days	Min	Max	Ave	Min	Max	Ave	Min	Max	Ave
2002[1]	-	-	-	-	-	-	-	-	-	-	-	-
2003	19-Jan	30-Dec	345	46	61	56	6,310	56,800	13,677	0.9	240.2	16.4
2004	6-Jan	17-Dec	346	46	62	56	5,170	92,800	14,613	1.4	413.5	9.3
2005	1-Jan	29-Dec	362	46	63	56	5,890	94,700	12,661	1.6	626.5	20.1
2006	3-Jan	30-Dec	361	47	61	54	6,610	82,900	20,803	2.0	190.5	11.4
2007	16-Jan	27-Dec	345	46	62	56	5,510	45,400	9,596	1.3	74.5	6.4
2008	6-Jan	28-Dec	357	44	64	56	4,610	44,600	9,478	1.5	233.3	9.0
2009	12-Jan	25-Dec	347	45	64	57	4,020	33,000	8,775	1.9	129.8	10.3
2010	15-Jan	12-Dec	331	47	62	56	5,150	60,400	11,194	1.1	250.6	12.4
2011	1-Jan	30-Dec	363	45	61	55	5,260	95,100	13,833	1.3	162.6	7.2
2012	17-Jan	14-Dec	332	45	65	56	4,800	70,400	10,557	1.2	315.5	11.0
Mean	10-Jan	23-Dec	349	46	63	56	5,333	67,610	12,519	1.4	263.7	11.4
SD			12	0.9	1.3	0.8	783	22,986	3,551	0.3	159.1	4.1
CV			3%	2%	2%	1%	15%	34%	28%	24%	60%	37%

[1] Sampling did not begin until mid-April of 2002 and this year not included in analyses.

Table 9f. Summary of Green Sturgeon abiotic sample conditions at RBDD rotary traps during dates of capture by calendar year (CY).

	Dates of Capture			H_20 Temperature (^oF)			Discharge (CFS)			Turbidity (NTU)		
CY	Initial	Final	Days	Min	Max	Ave	Min	Max	Ave	Min	Max	Ave
2002	7-May	16-Jul	70	55	60	58	9,317	15,680	13,038	0.9	16.3	3.5
2003	13-Jun	11-Nov	151	52	61	58	6,950	16,000	10,802	0.9	48.6	6.5
2004	4-May	29-Jul	86	55	60	58	9,560	16,700	14,210	3.0	18.3	4.9
2005	7-May	13-Aug	98	54	61	58	10,200	76,200	18,614	2.3	626.5	26.4
2006	10-Jun	25-Aug	76	56	59	57	12,800	15,600	14,579	3.4	13.9	5.7
2007	11-May	24-Jul	74	55	61	58	9,790	17,000	12,905	1.7	50.4	4.5
2008	-	-	0	-	-	-	-	-	-	-	-	-
2009	11-May	16-Jul	66	58	64	61	9,460	13,700	11,226	4.1	34.4	13.5
2010	26-May	29-Aug	95	55	61	58	9,150	18,300	13,143	1.6	22.0	5.4
2011	16-May	27-Aug	103	52	61	58	10,400	24,800	14,059	3.6	23.5	6.8
2012	1-May	26-Jun	56	55	61	58	8,763	21,398	12,258	2.2	85.4	7.7
Mean	17-May	12-Aug	88	55	61	58	9,639	23,538	13,483	2.4	93.9	8.5
SD			27	1.7	1.2	0.9	1,464	18,782	2,181	1.1	188.4	6.9
CV			31%	3%	2%	2%	15%	80%	16%	47%	201%	81%

Table 9g. Summary of Lamprey *spp.* abiotic sample conditions at RBDD rotary traps during dates of capture by water year (WY).

	Dates of Capture			H_20 Temperature (^oF)			Discharge (CFS)			Turbidity (NTU)		
WY	Initial	Final	Days	Min	Max	Ave	Min	Max	Ave	Min	Max	Ave
2003	1-Oct	27-Sep	361	47	61	56	6,176	86,500	15,033	0.4	240.2	15.1
2004	1-Oct	29-Sep	364	46	62	55	6,310	92,800	15,528	0.9	413.5	16.3
2005	2-Oct	29-Sep	362	46	63	56	5,170	76,200	11,800	1.4	626.5	18.6
2006	1-Oct	29-Sep	363	47	61	54	6,450	118,000	22,724	1.6	731.7	17.9
2007	1-Oct	29-Sep	363	46	62	55	6,030	45,400	9,832	1.6	90.0	7.3
2008	1-Oct	29-Sep	364	44	63	56	5,210	44,600	9,342	1.3	233.3	8.8
2009	1-Oct	29-Sep	363	45	64	57	4,160	33,000	8,791	1.6	129.8	10.5
2010	1-Oct	30-Sep	364	46	62	56	3,920	60,400	10,241	1.1	250.6	12.1
2011	3-Oct	30-Sep	362	45	61	55	5,260	95,100	15,022	1.3	162.6	8.4
2012	3-Oct	27-Sep	360	45	65	55	4,800	35,200	9,753	1.2	180.6	7.1
2013	5-Oct	28-Sep	358	44	64	56	5,330	70,400	10,479	1.1	315.5	8.5
Mean	2-Oct	29-Sep	362	46	63	56	5,347	68,873	12,595	1.2	306.8	11.9
SD			2	1.1	1.3	0.7	843	27,701	4,177	0.3	205.5	4.4
CV			1%	2%	2%	1%	16%	40%	33%	29%	67%	37%

Figures

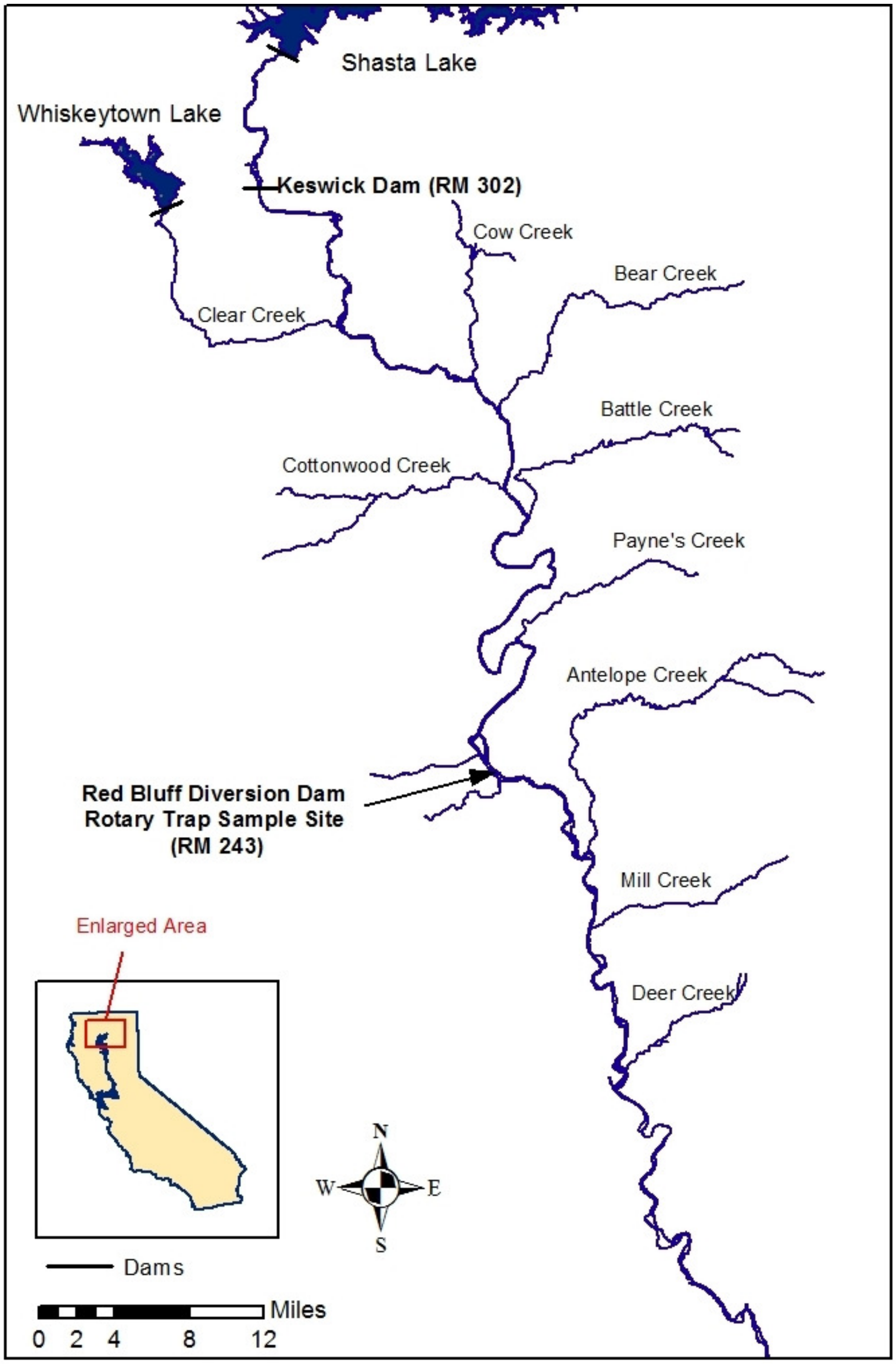


Figure 1. Location of Red Bluff Diversion Dam rotary trap sample site on the Sacramento River, California (RM 243).

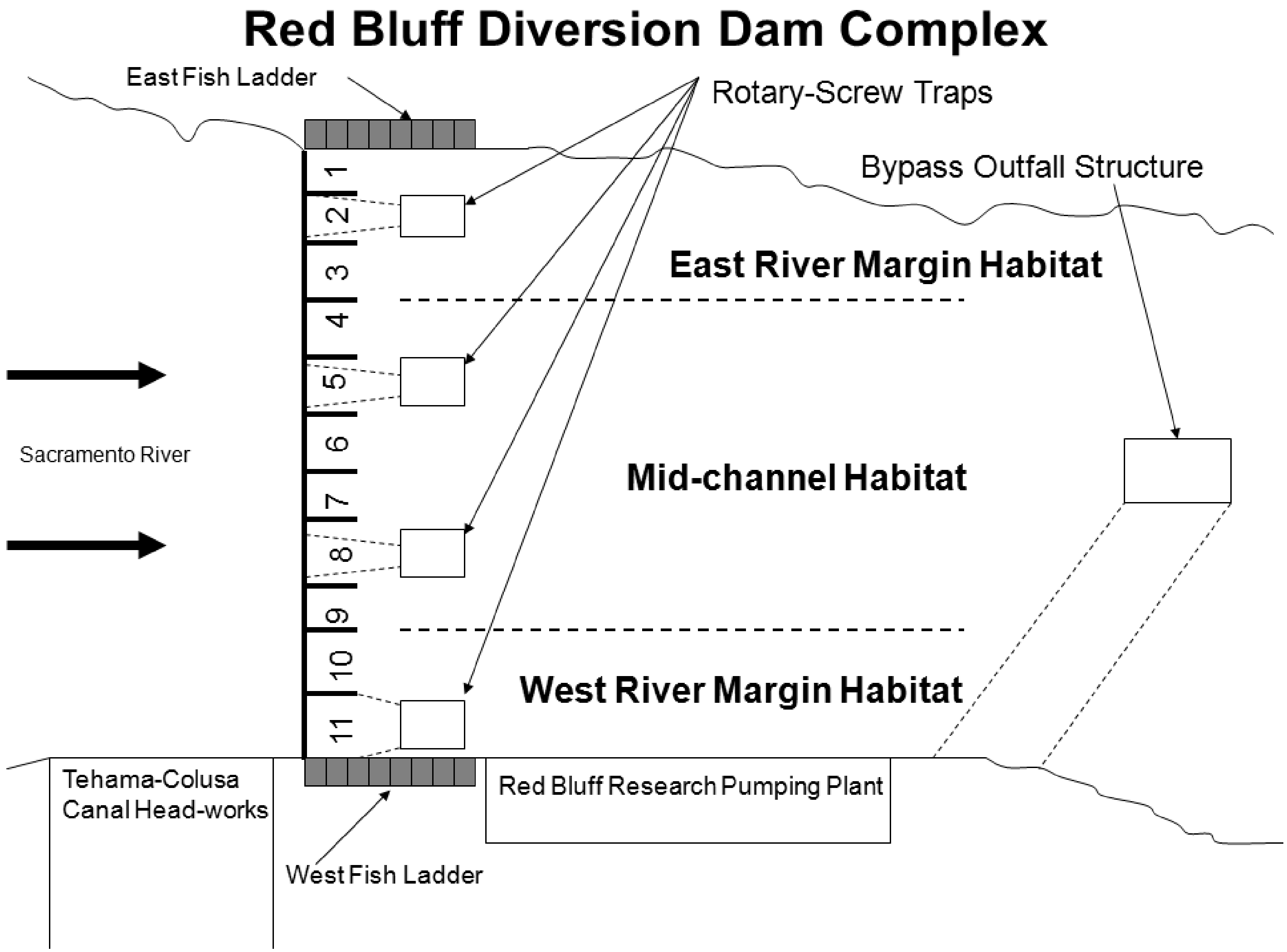


Figure 2. Rotary-screw trap sampling transect at Red Bluff Diversion Dam Site (RM 243) on the Sacramento River, California.

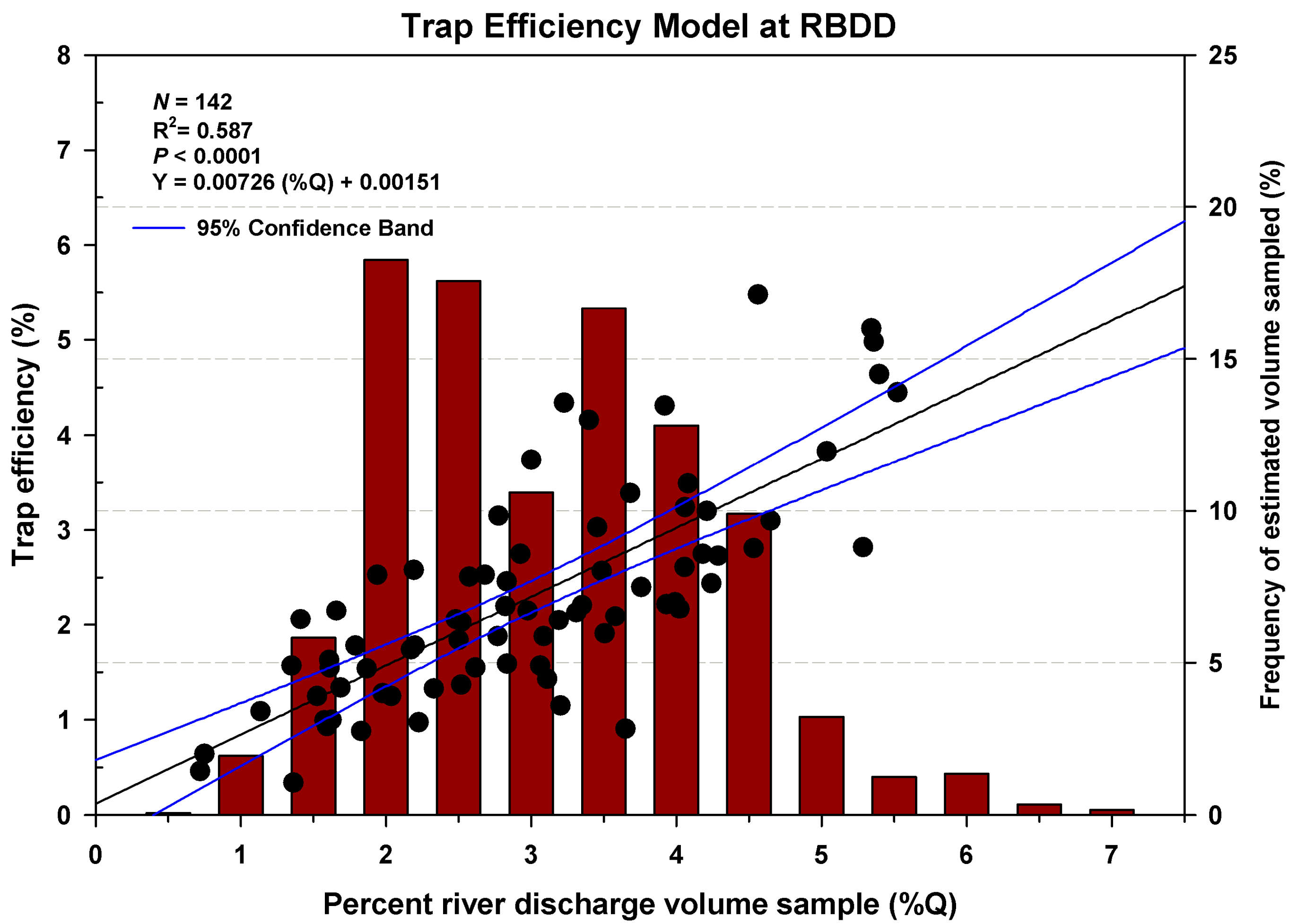


Figure 3. Trap efficiency model for combined 8-ft diameter rotary traps at Red Bluff Diversion Dam (RM 243), Sacramento River, CA. Mark-recapture trials (*N* = 142) were used to estimate trap efficiencies. Histogram indicates percentage of time traps sampled various levels (half percent bins) of river discharge between April 2002 and September 2013.

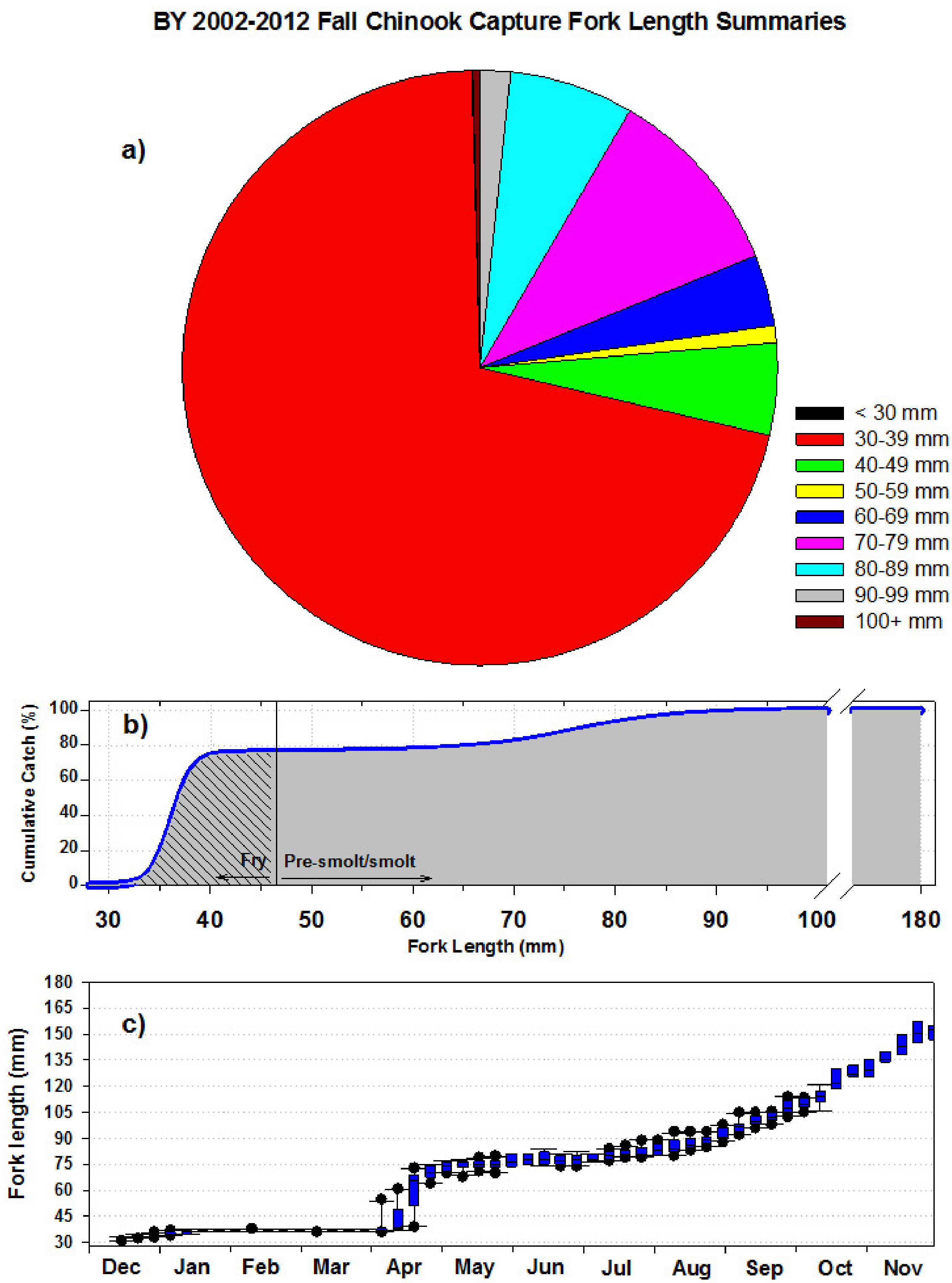


Figure 4. Fall Chinook fork length (a) capture proportions, (b) cumulative capture size curve, and (c) average weekly median boxplots for fall Chinook sampled by rotary traps at RBDD between December 2002 and September 2013.

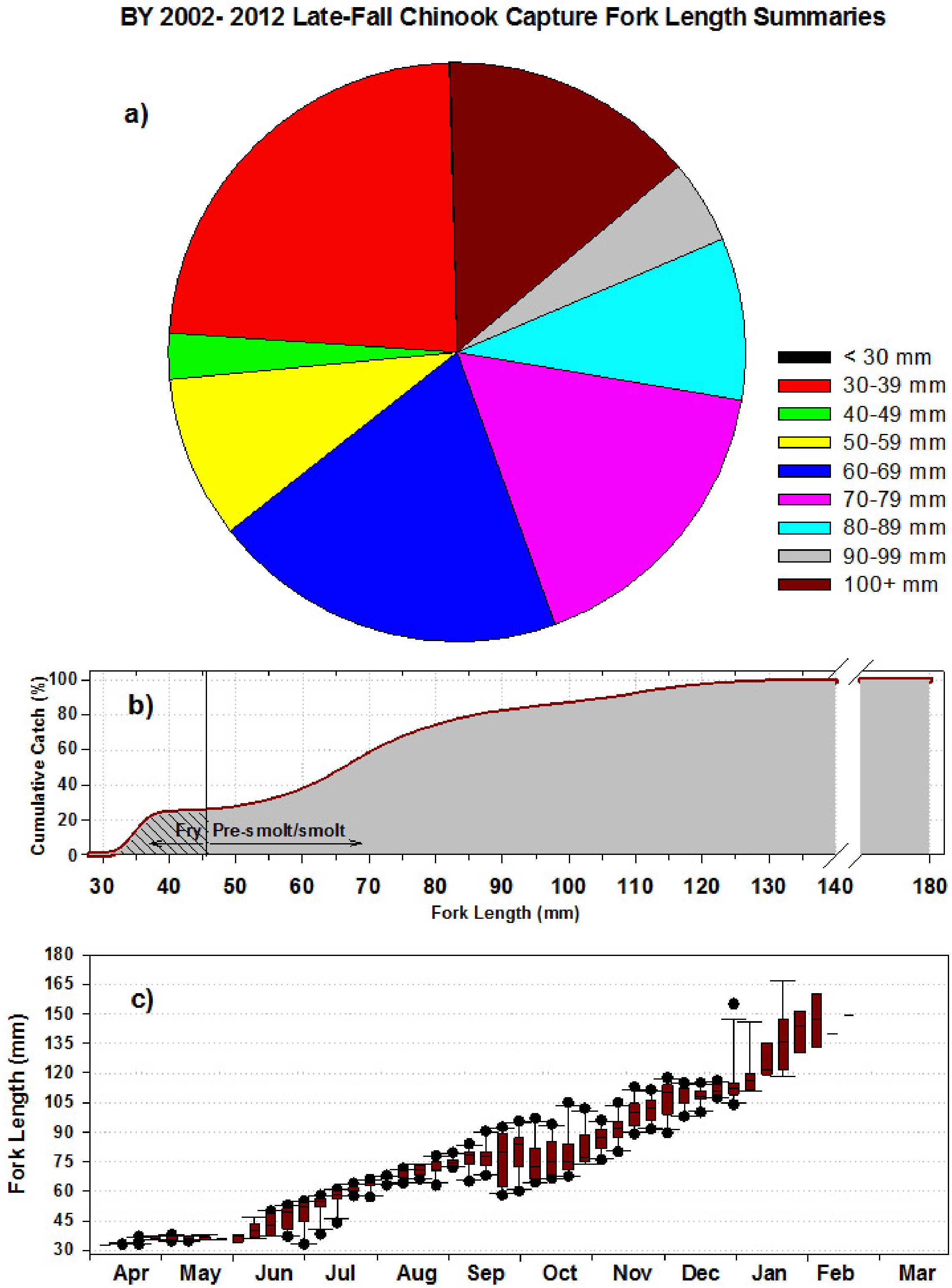


Figure 5. Late-fall Chinook fork length (a) capture proportions, (b) cumulative capture size curve, and (c) average weekly median boxplots for late-fall Chinook sampled by rotary traps at RBDD between April 2002 and March 2013.

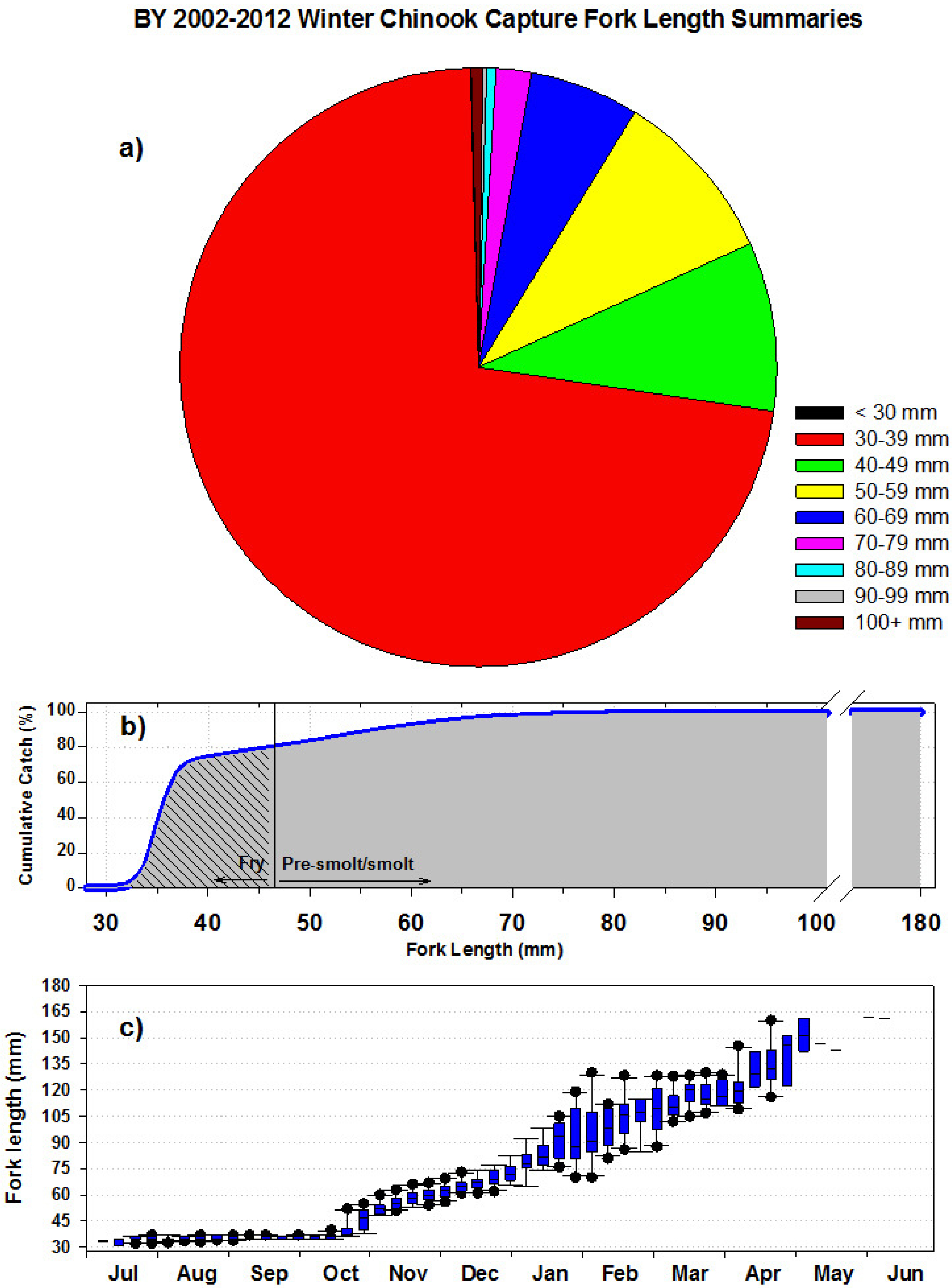


Figure 6. Winter Chinook fork length (a) capture proportions, (b) cumulative capture size curve, and (c) average weekly median boxplots for winter Chinook sampled by rotary traps at RBDD between July 2002 and June 2013.

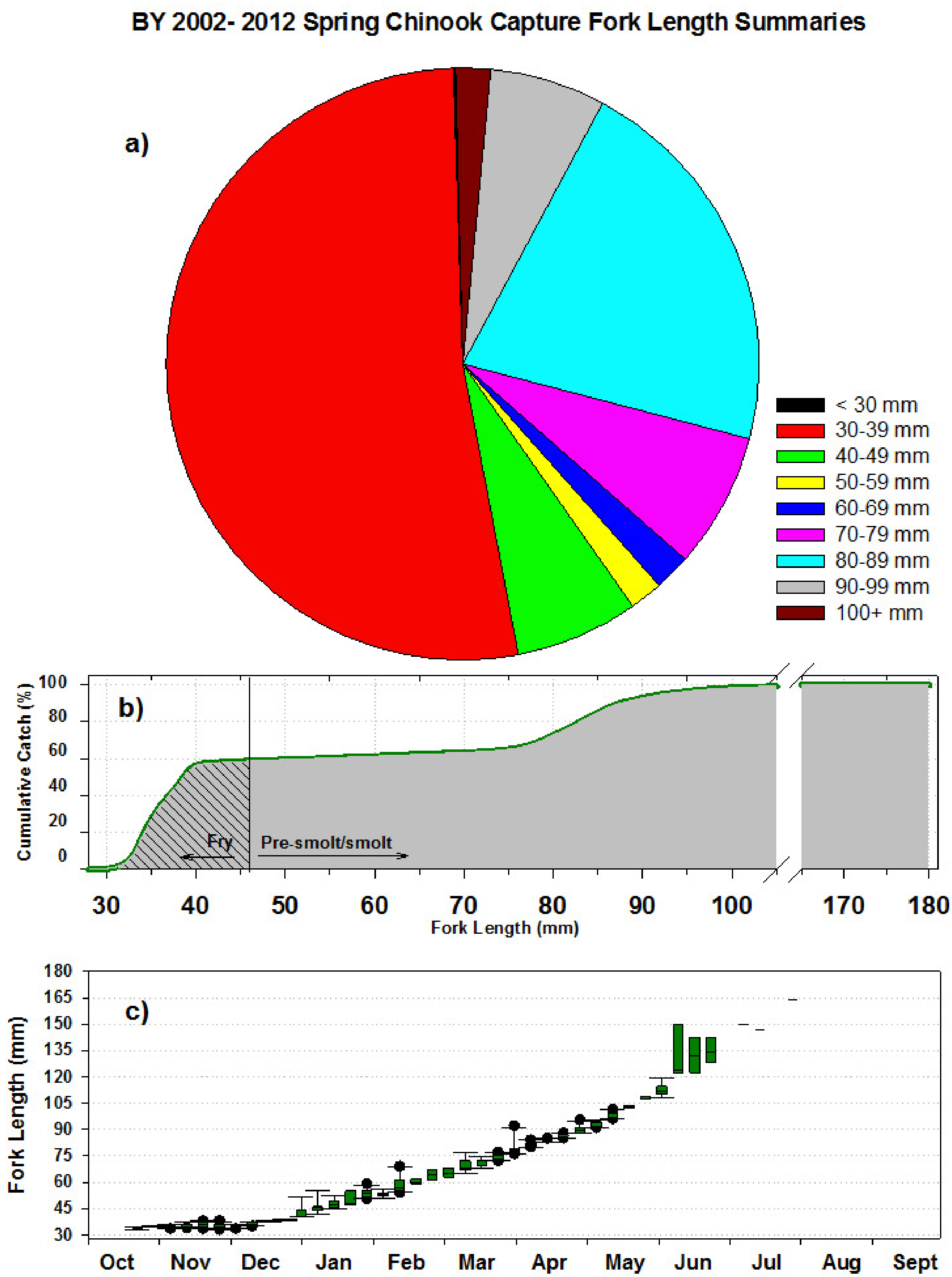


Figure 7. Spring Chinook fork length (a) capture proportions, (b) cumulative capture size curve, and (c) average weekly median boxplots for spring Chinook sampled by rotary traps at RBDD between October 2002 and September 2013.

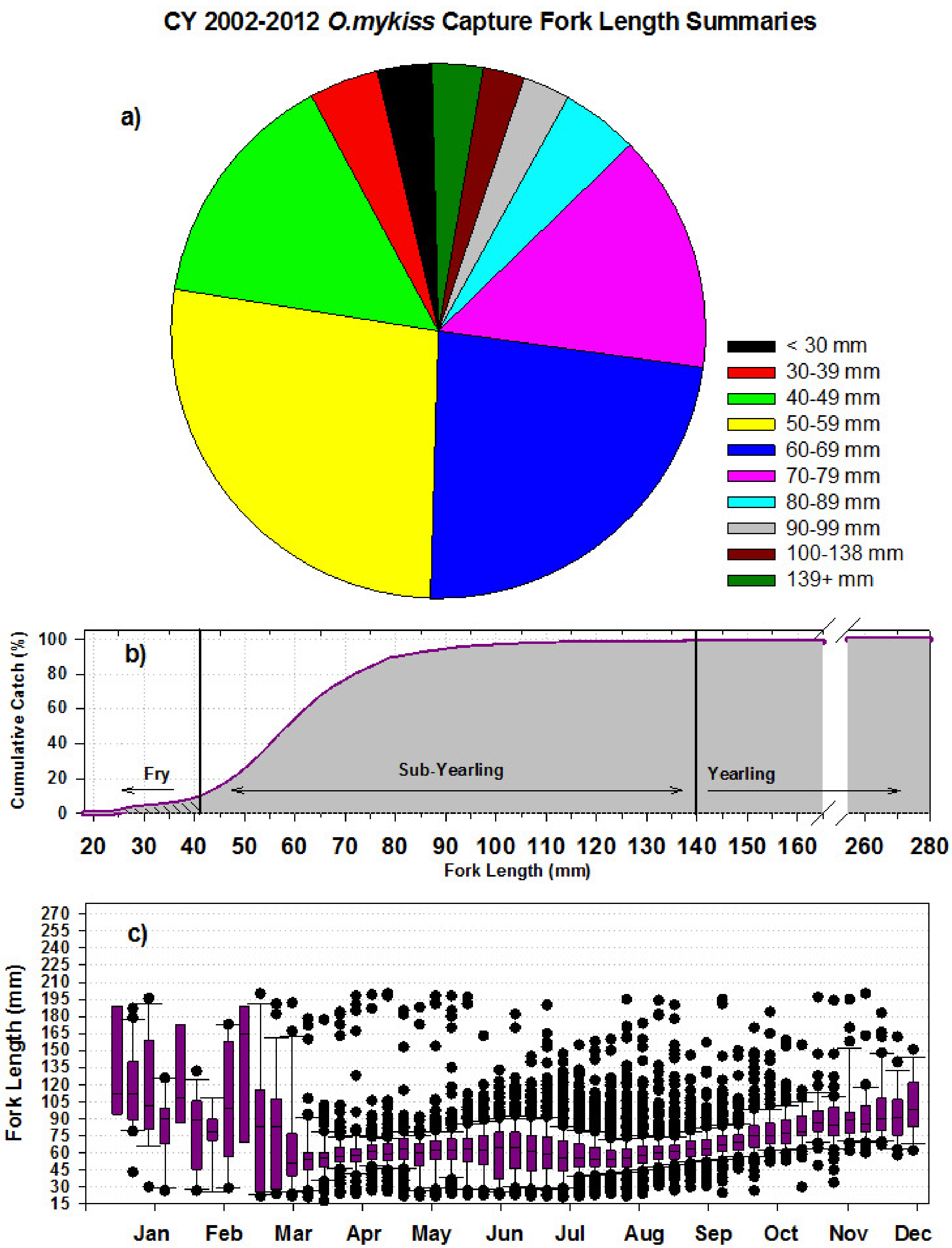


Figure 8. *O. mykiss* fork length (a) capture proportions, (b) cumulative capture size curve, and (c) average weekly median boxplots for *O. mykiss* sampled by rotary traps at RBDD between April 2002 and December 2012.

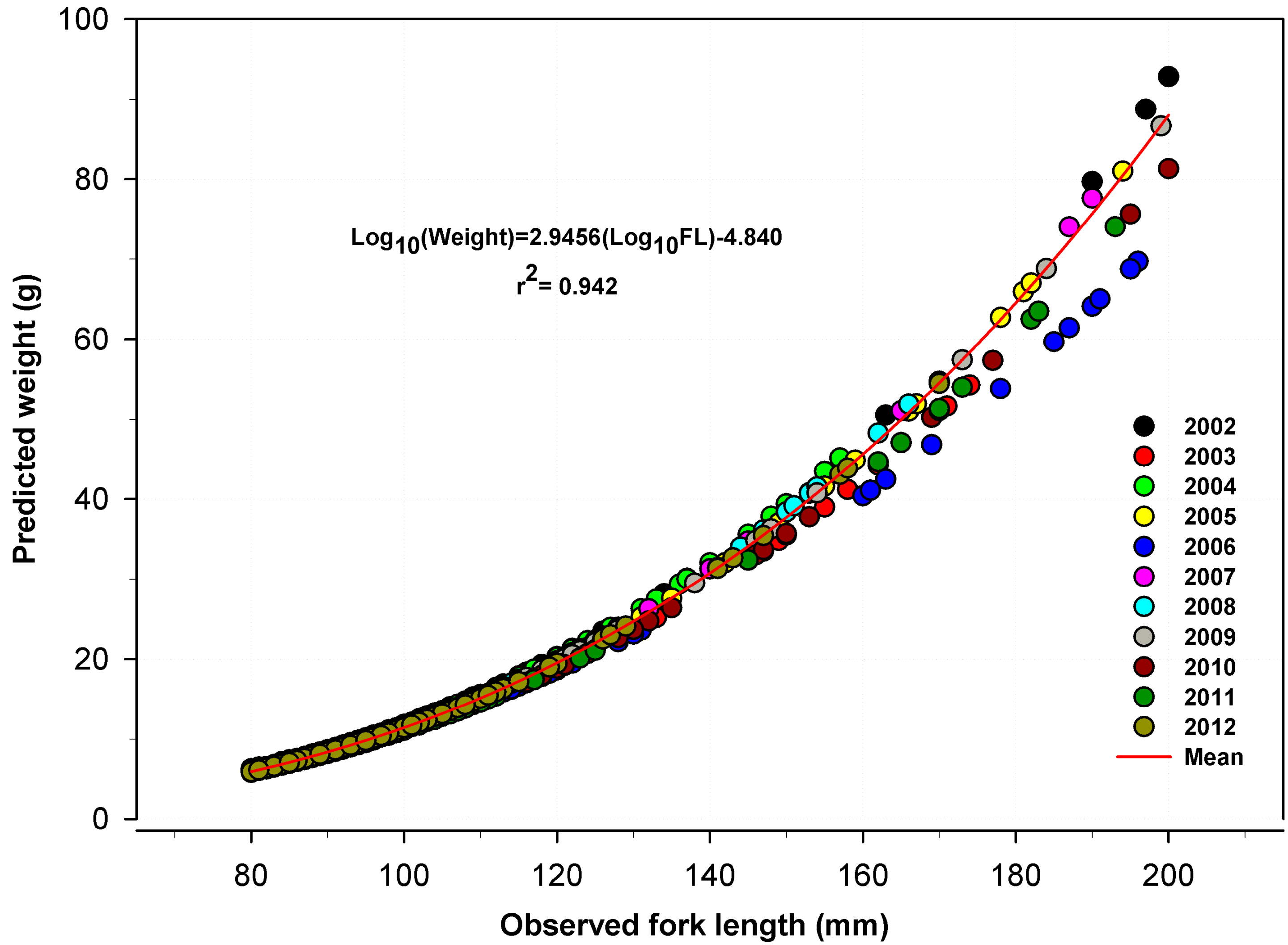


Figure 9. Predicted weight (g) for *O. mykiss* with measured fork lengths (FL) between 80 and 200 mm using annual weight-length regression equation.

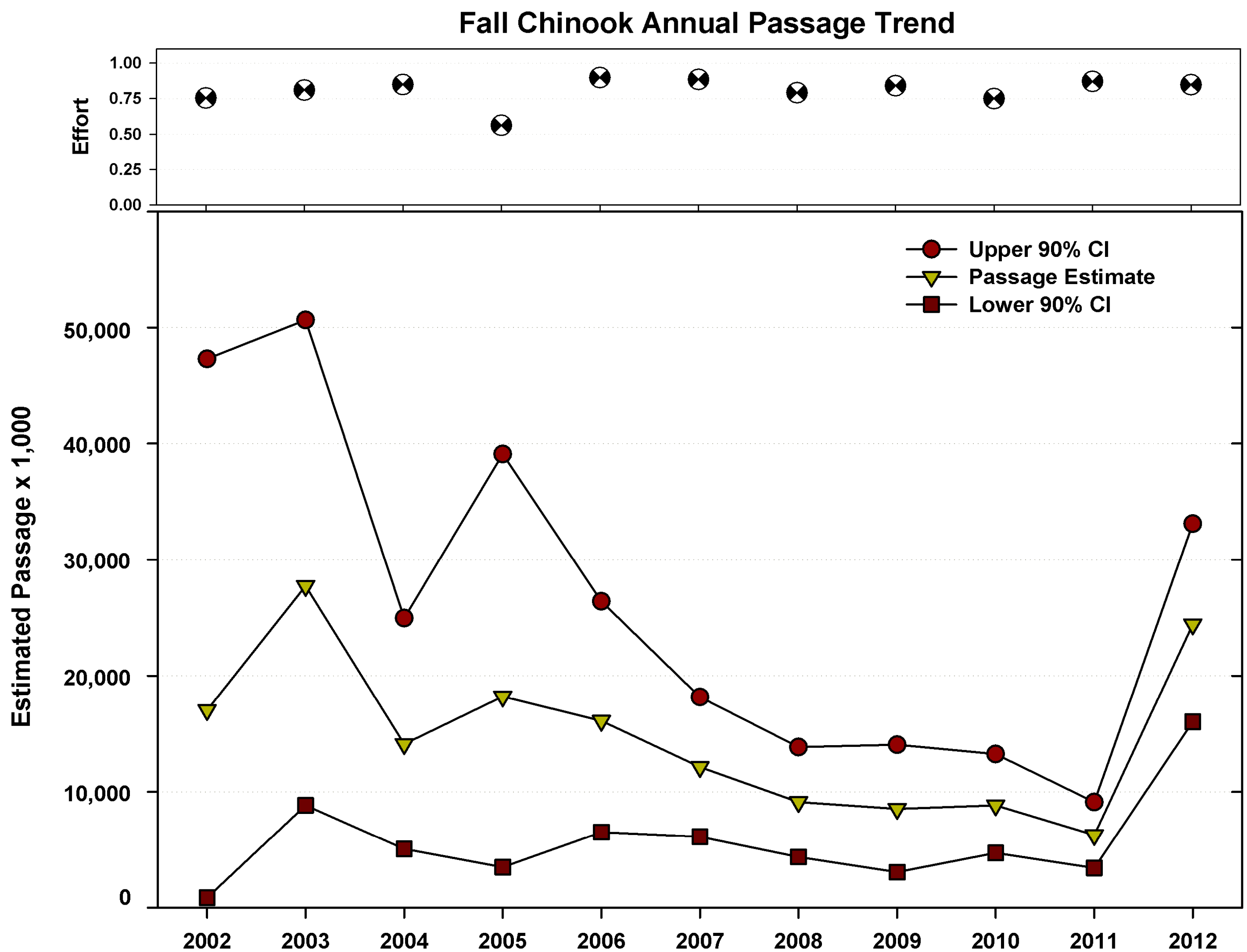


Figure 10. RBDD rotary trap fall Chinook annual sample effort and passage estimates with 90% confidence intervals (CI) for the period December 2002 through September 2013

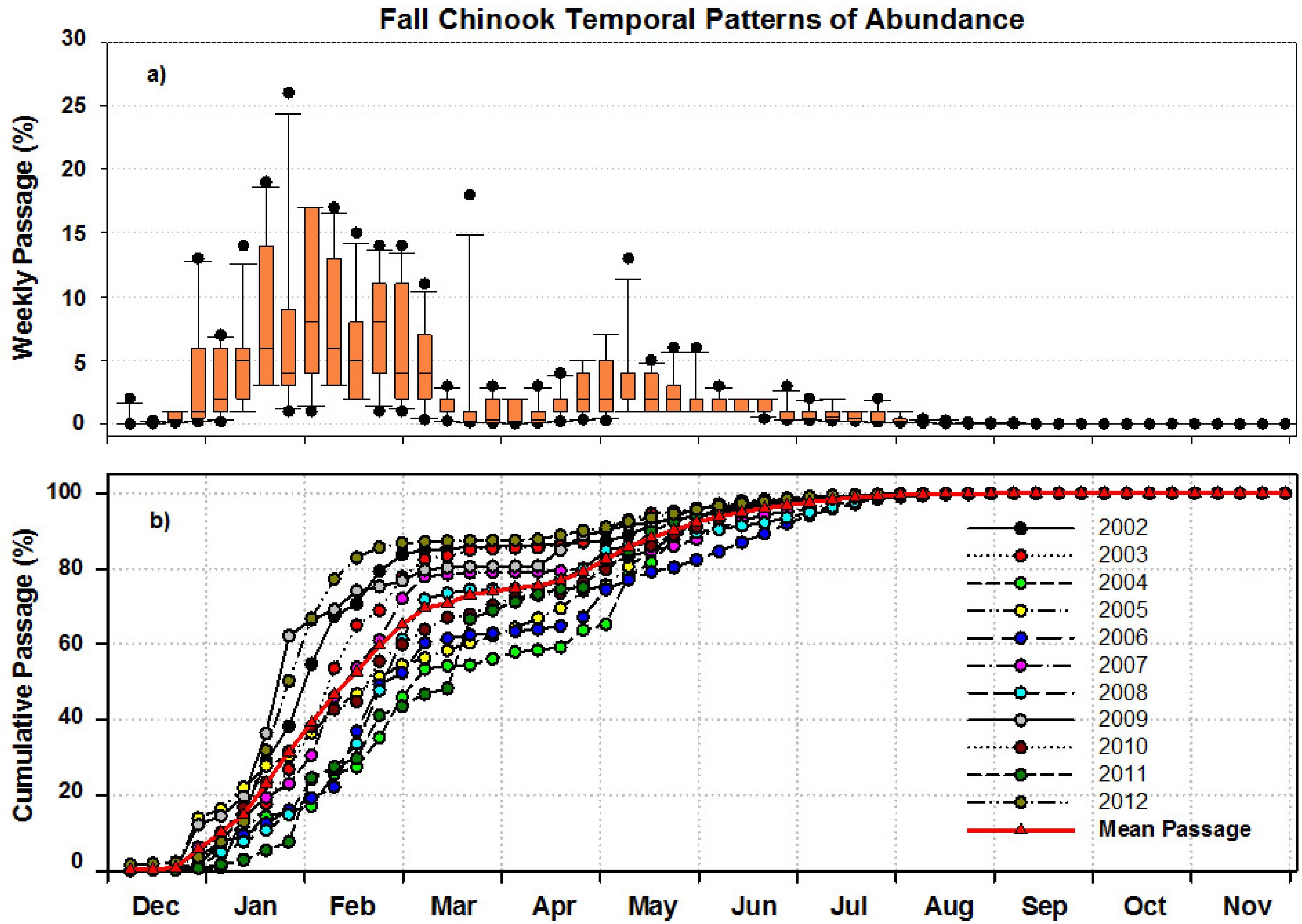


Figure 11. RBDD rotary trap fall Chinook (a) boxplots of weekly passage estimates relative to annual total passage estimates and (b) cumulative weekly passage with 11-year mean passage trend line for the period December 2002 through September 2013.

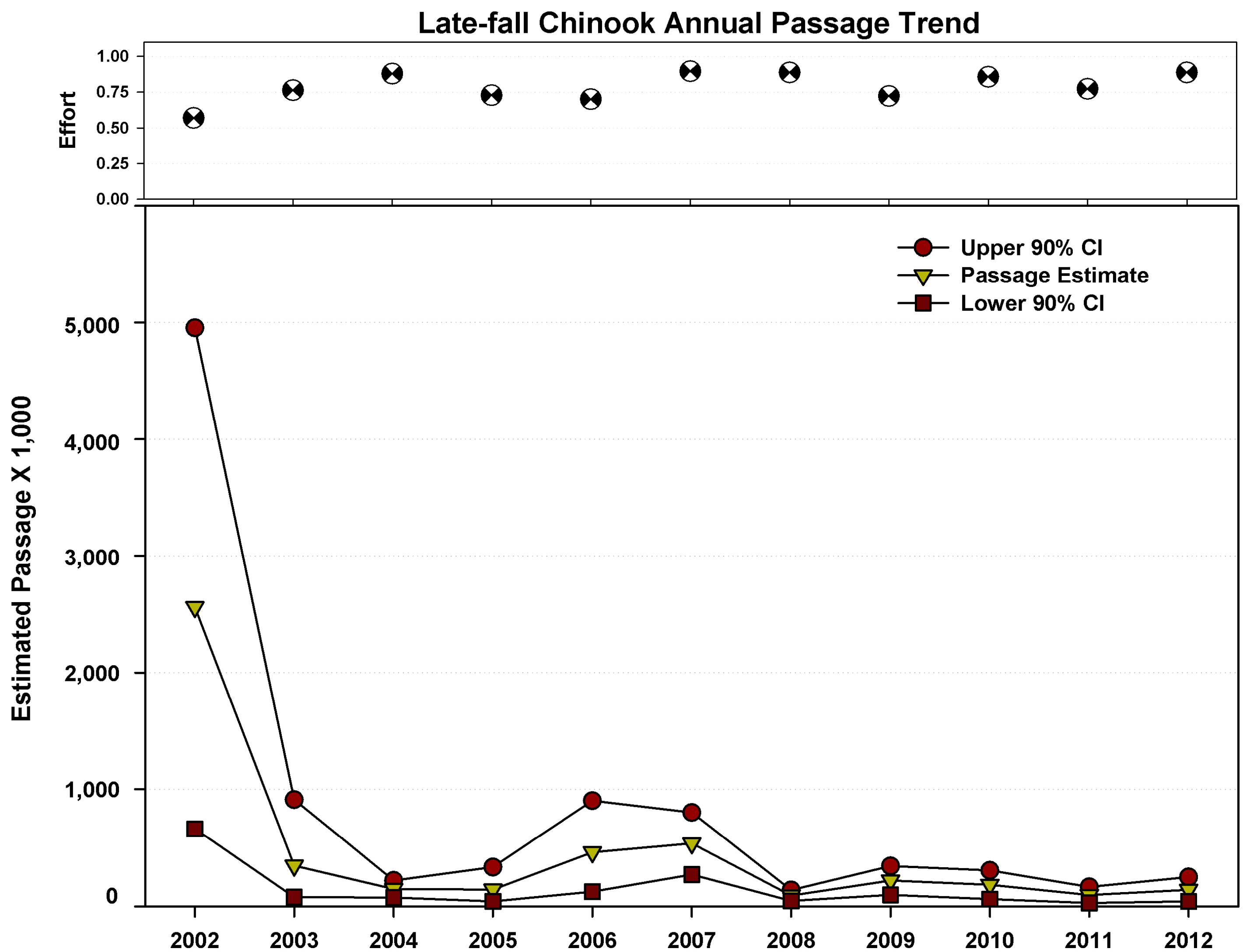


Figure 12. RBDD rotary trap late-fall Chinook annual sample effort and passage estimates with 90% confidence intervals (CI) for the period April 2002 through March 2013.

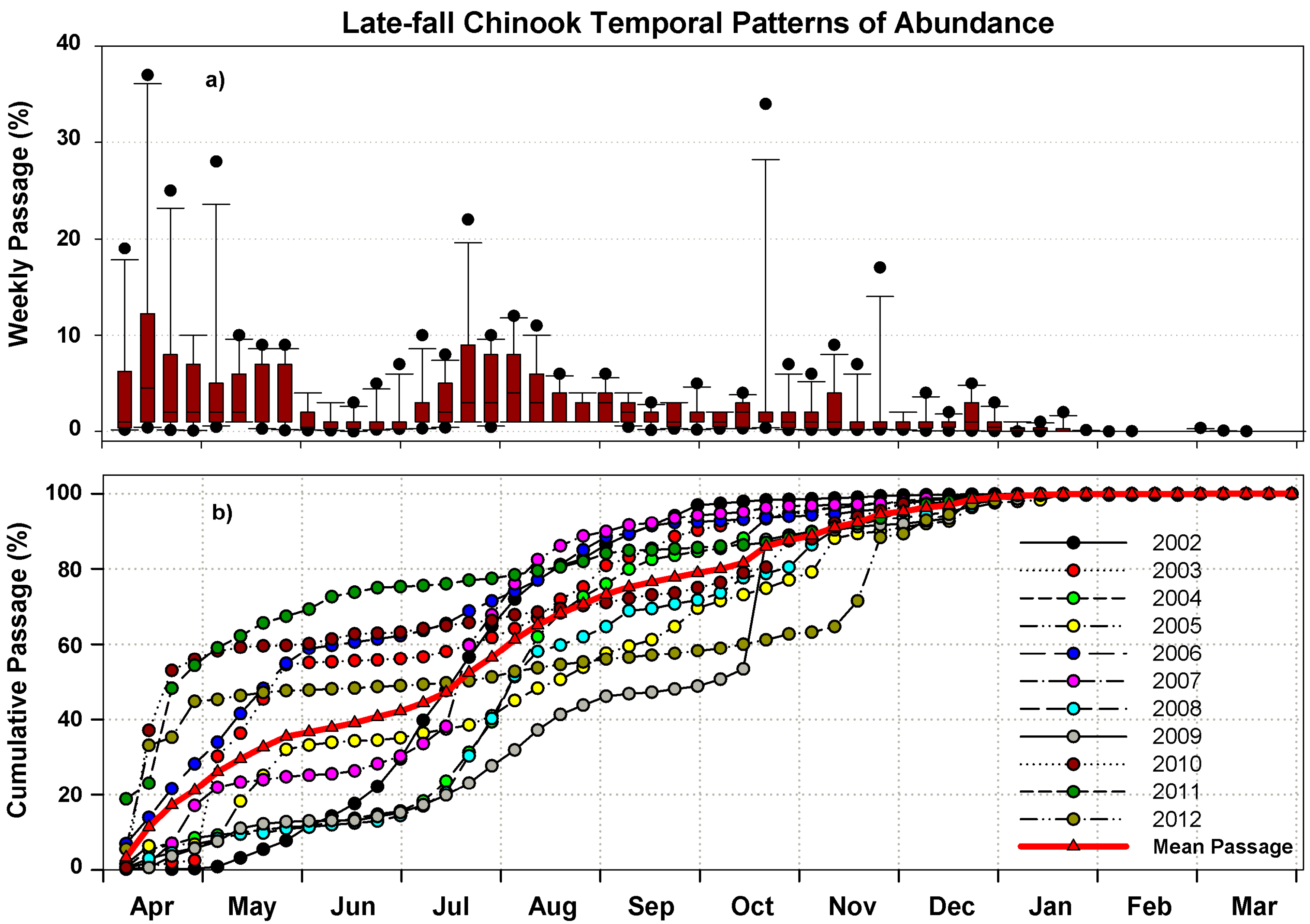


Figure 13. RBDD rotary trap late-fall Chinook (a) boxplots of weekly passage estimates relative to annual total passage estimates and (b) cumulative weekly passage with 11-year mean passage trend line for the period April 2002 through March 2013.

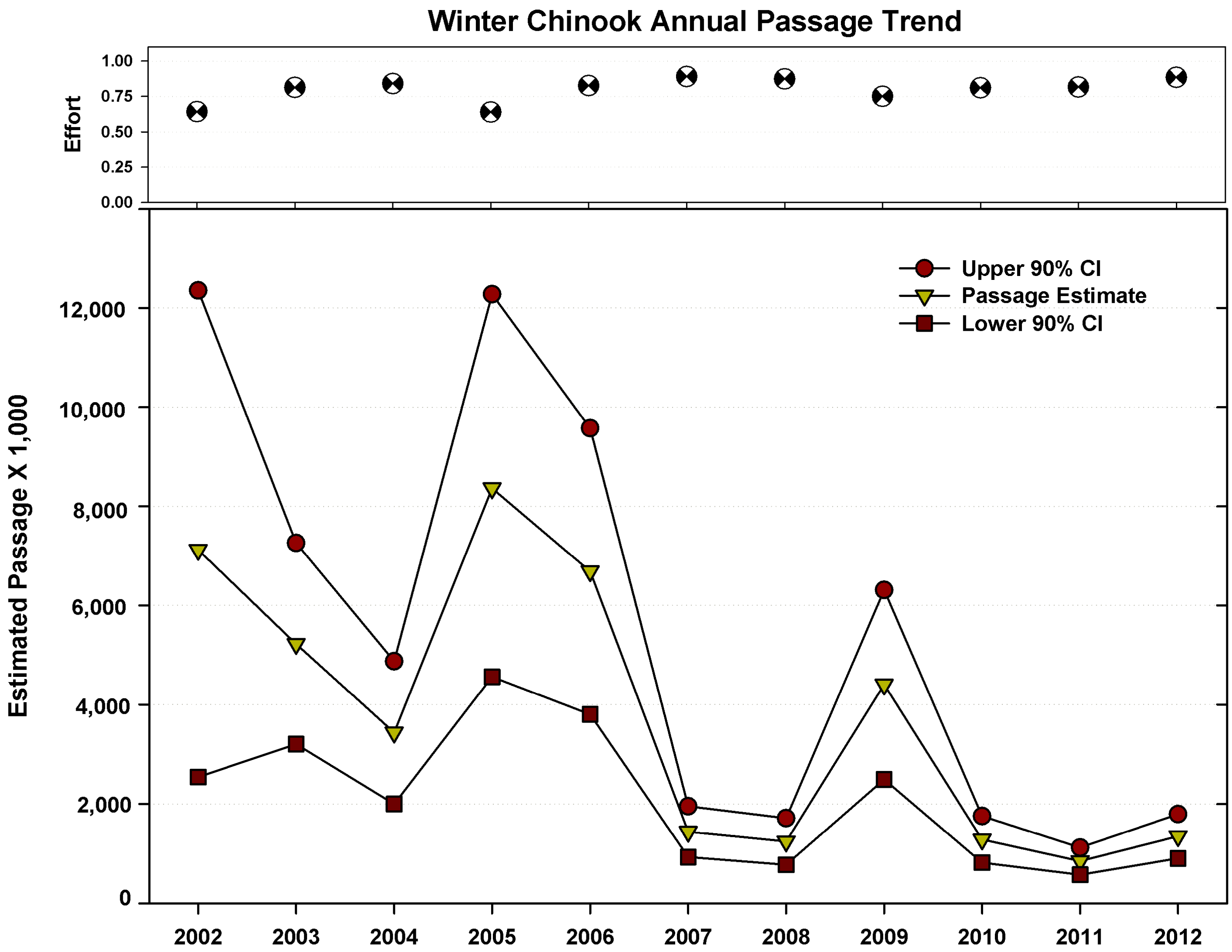


Figure 14. RBDD rotary trap winter Chinook annual sample effort and passage estimates with 90% confidence intervals (CI) for the period July 2002 through June 2013.

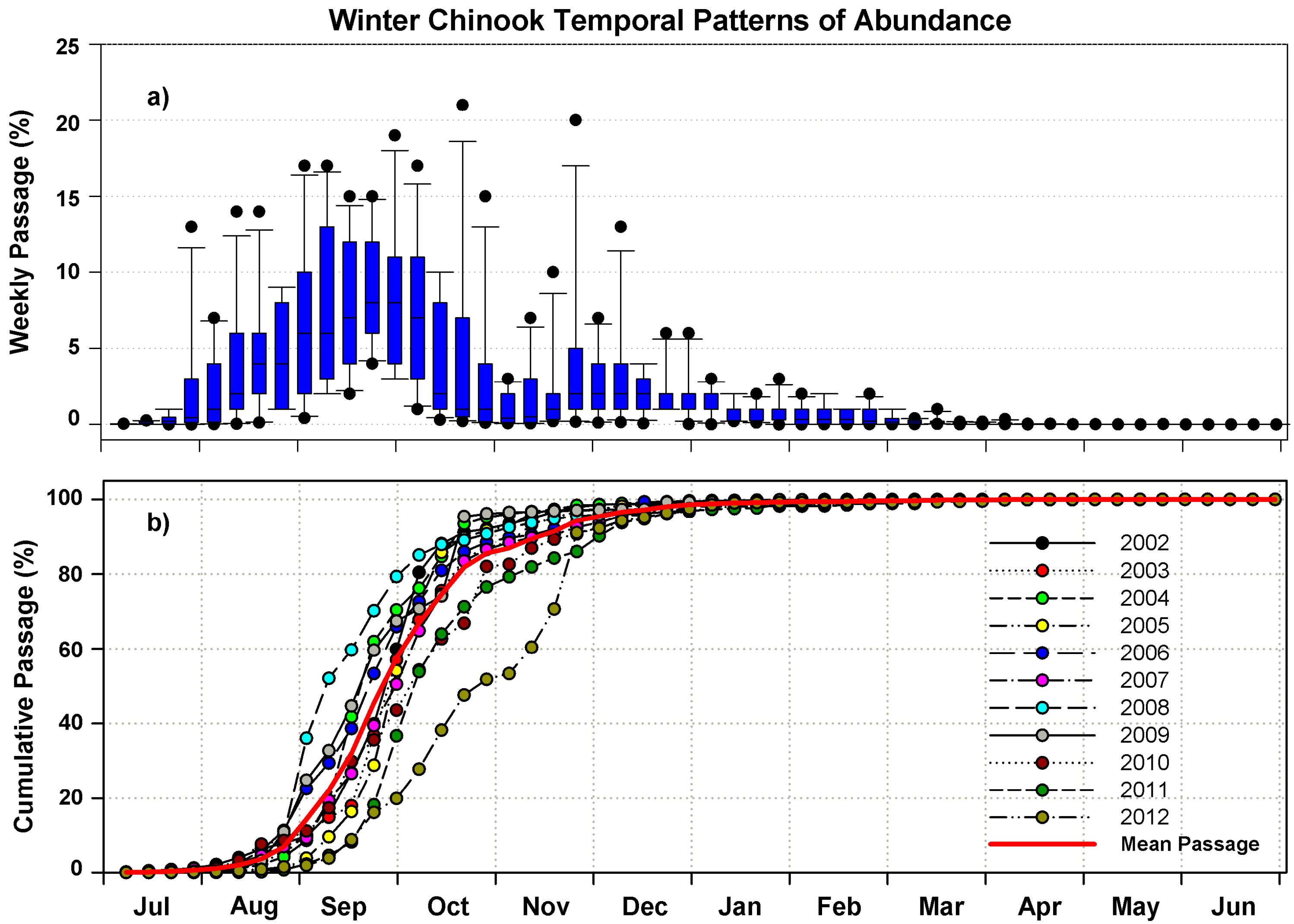


Figure 15. RBDD rotary trap winter Chinook (a) boxplots of weekly passage estimates relative to annual total passage estimates and (b) cumulative weekly passage with 11-year mean passage trend line for the period July 2002 through June 2013.

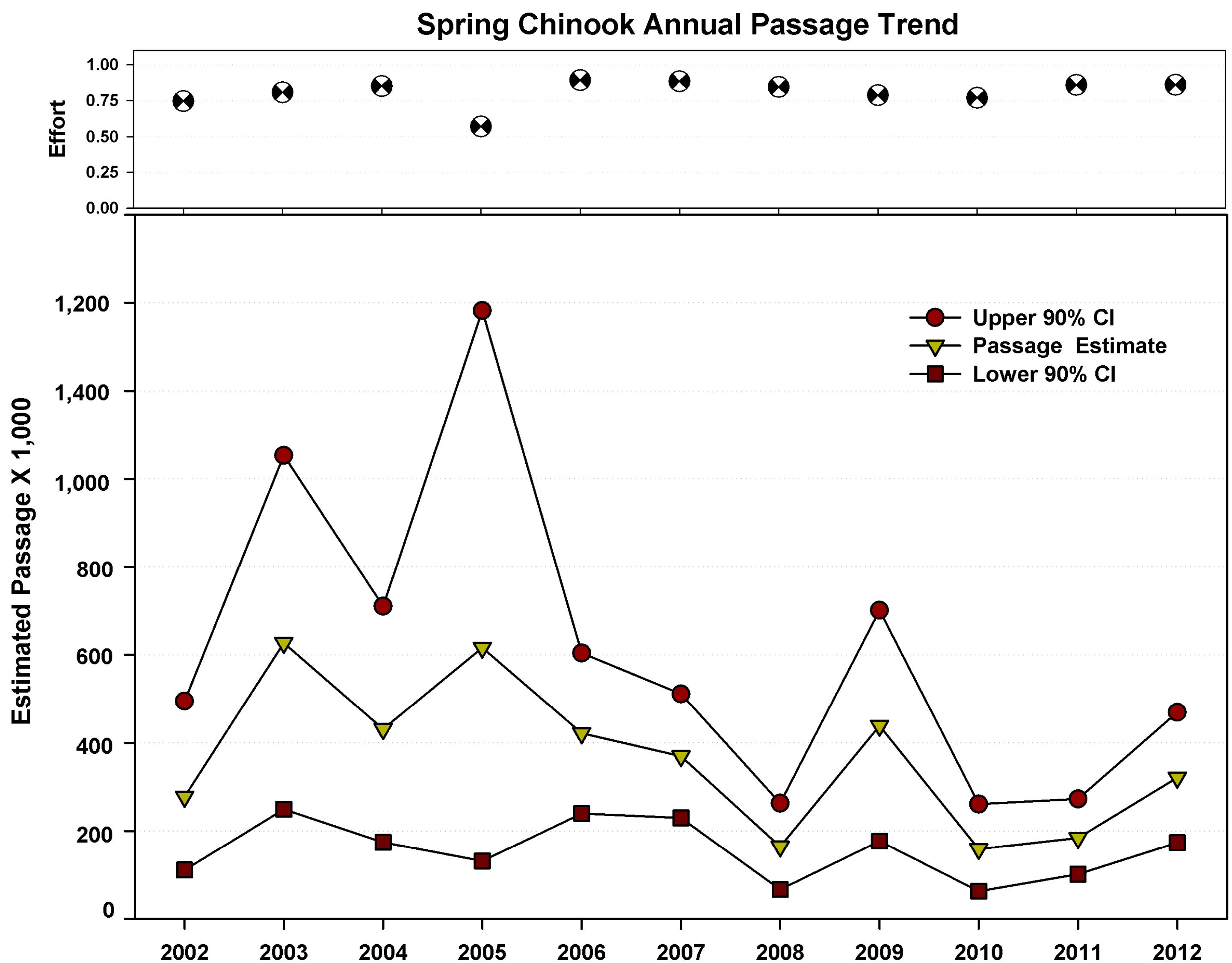


Figure 16. RBDD rotary trap spring Chinook annual sample effort and passage estimates with 90% confidence intervals (CI) for the period October 2002 through September 2013.

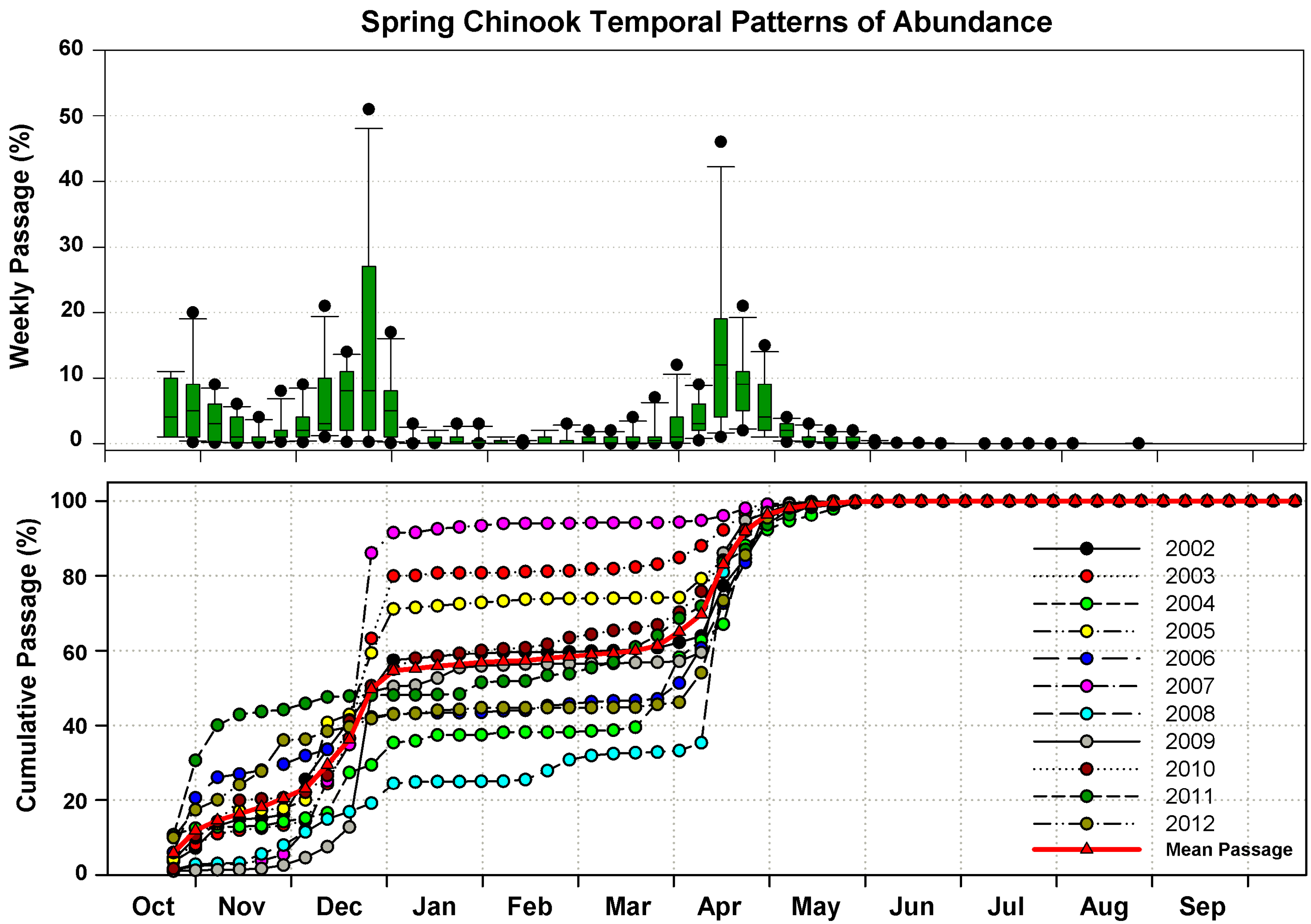


Figure 17. RBDD rotary trap spring Chinook (a) boxplots of weekly passage estimates relative to annual total passage estimates and (b) cumulative weekly passage with 11-year mean passage trend line for the period October 2002 through September 2013.

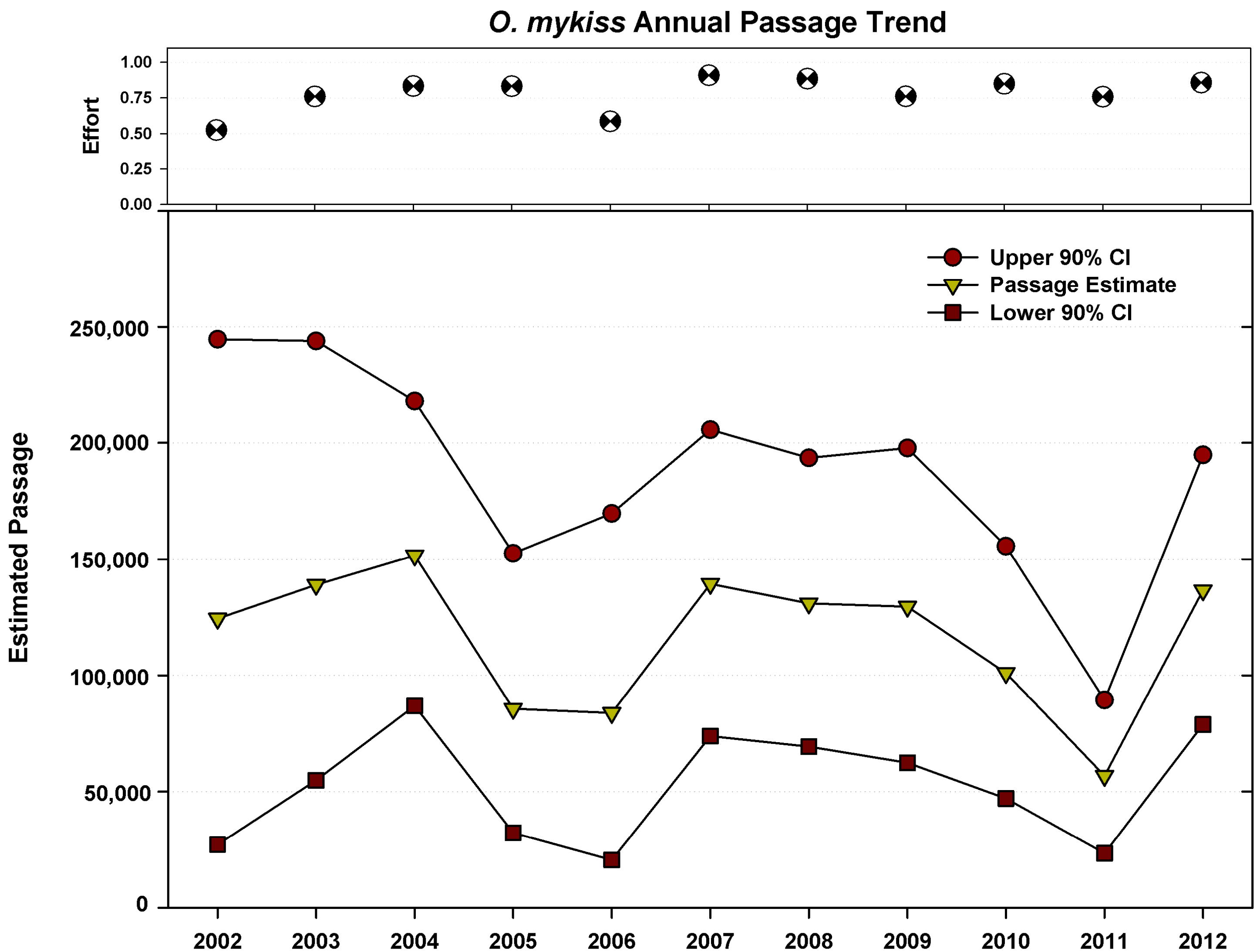


Figure 18. RBDD rotary trap *O. mykiss* annual sample effort and passage estimates with 90% confidence intervals (CI) for the period April 2002 through December 2012.

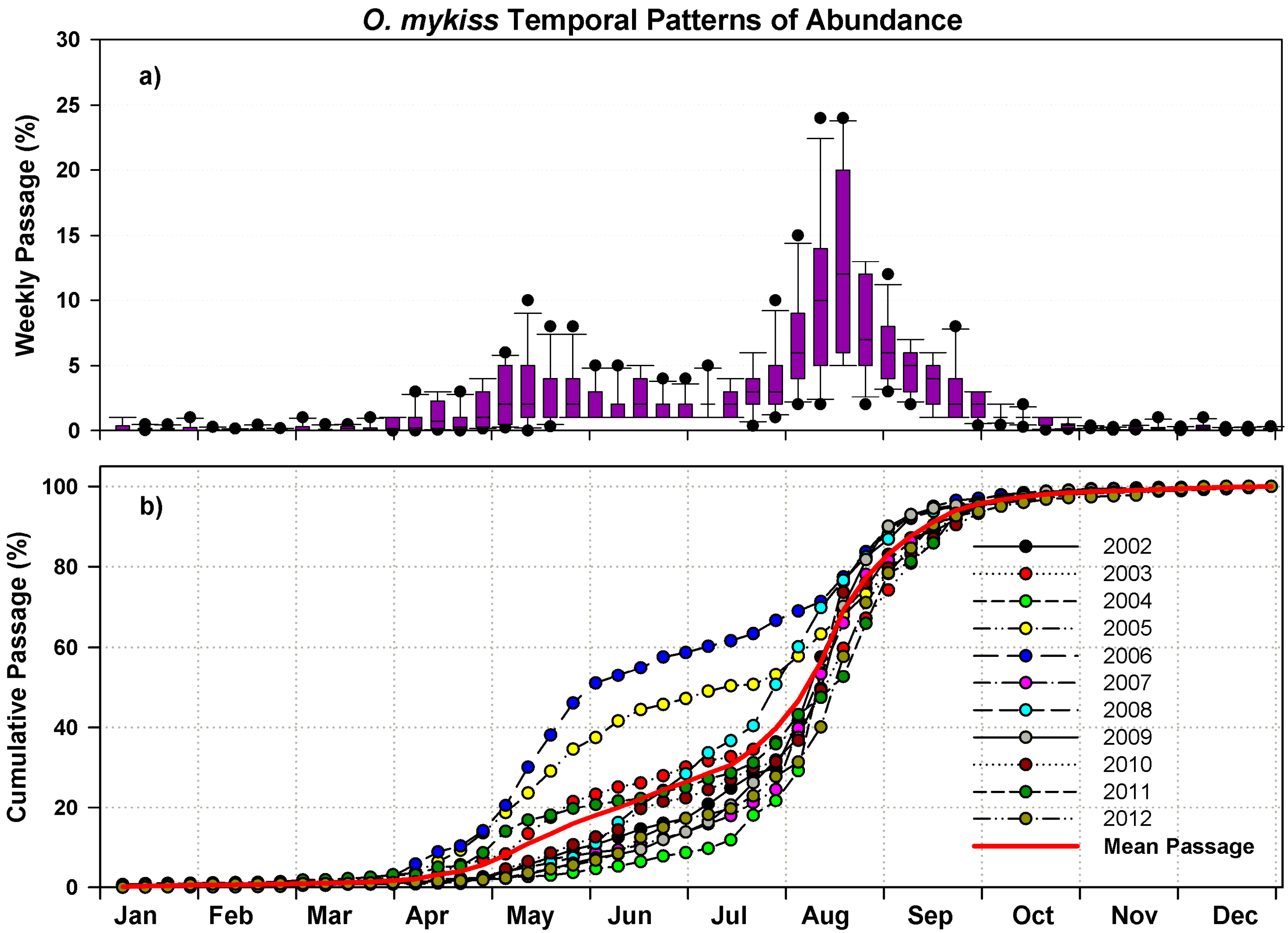


Figure 19. RBDD rotary trap *O. mykiss* (a) boxplots of weekly passage estimates relative to annual total passage estimates and (b) cumulative weekly passage with 11-year mean passage trend line for the period April 2002 through December 2012.

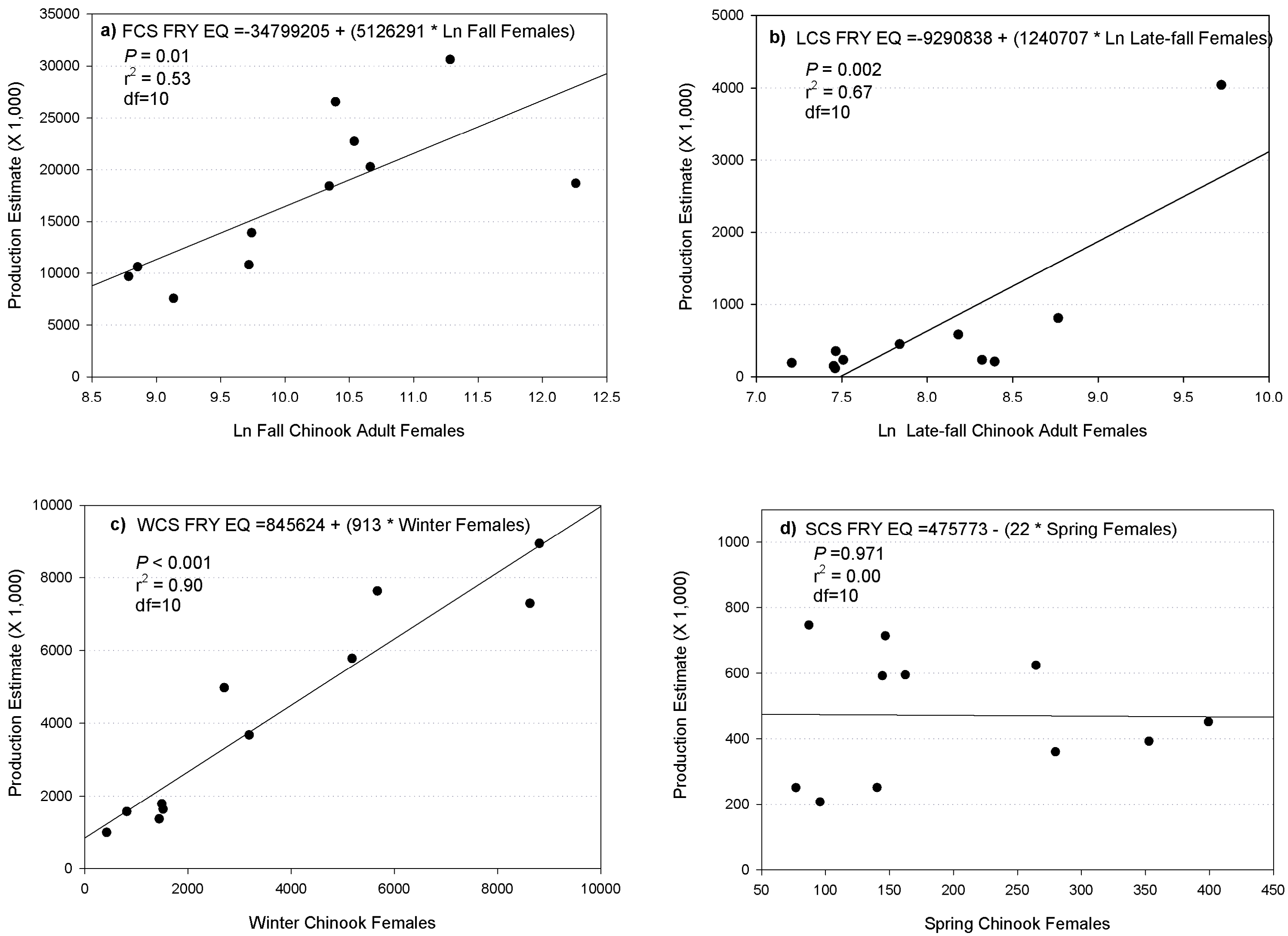


Figure 20. Relationships between a) fall, b) late-fall, c) winter, and d) spring Chinook fry-equivalent production estimates and estimated number of female adult Chinook salmon upstream of RBDD between 2002 and 2012. Note: fall and late-fall adult females were natural log transformed due to extraordinary escapement values estimated for the year 2002.

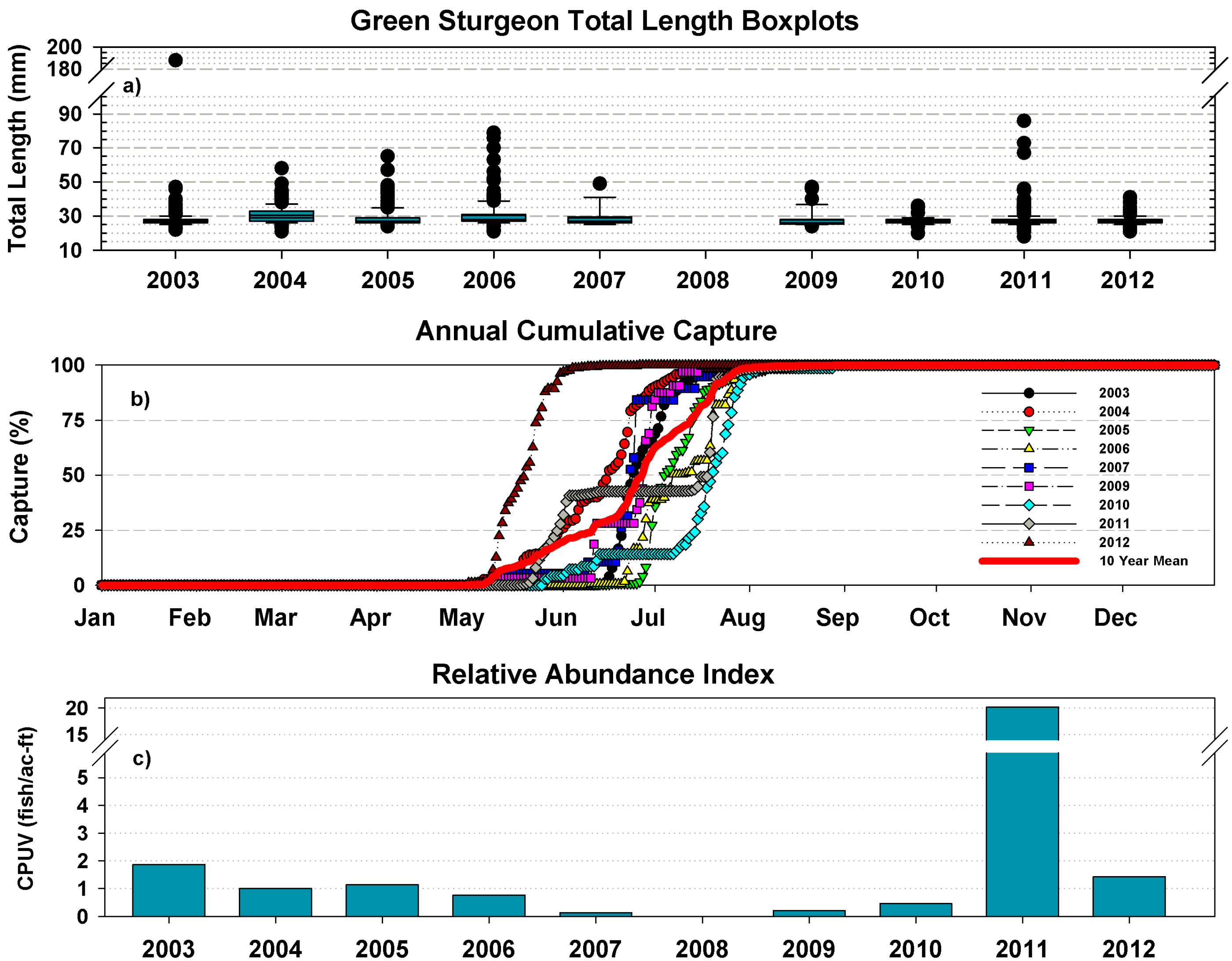


Figure 21. Green sturgeon a) annual total length capture boxplots, b) annual cumulative capture trends with 10-year mean trend line, and c) relative abundance indices. All fish captured by rotary trap at RBDD (RM 243) on the Upper Sacramento River, CA between 2003 and 2012. Data from 2002 excluded from analysis due to limited effort and USBR Crown Flow study resulting in incomparable sampling regimes and results.

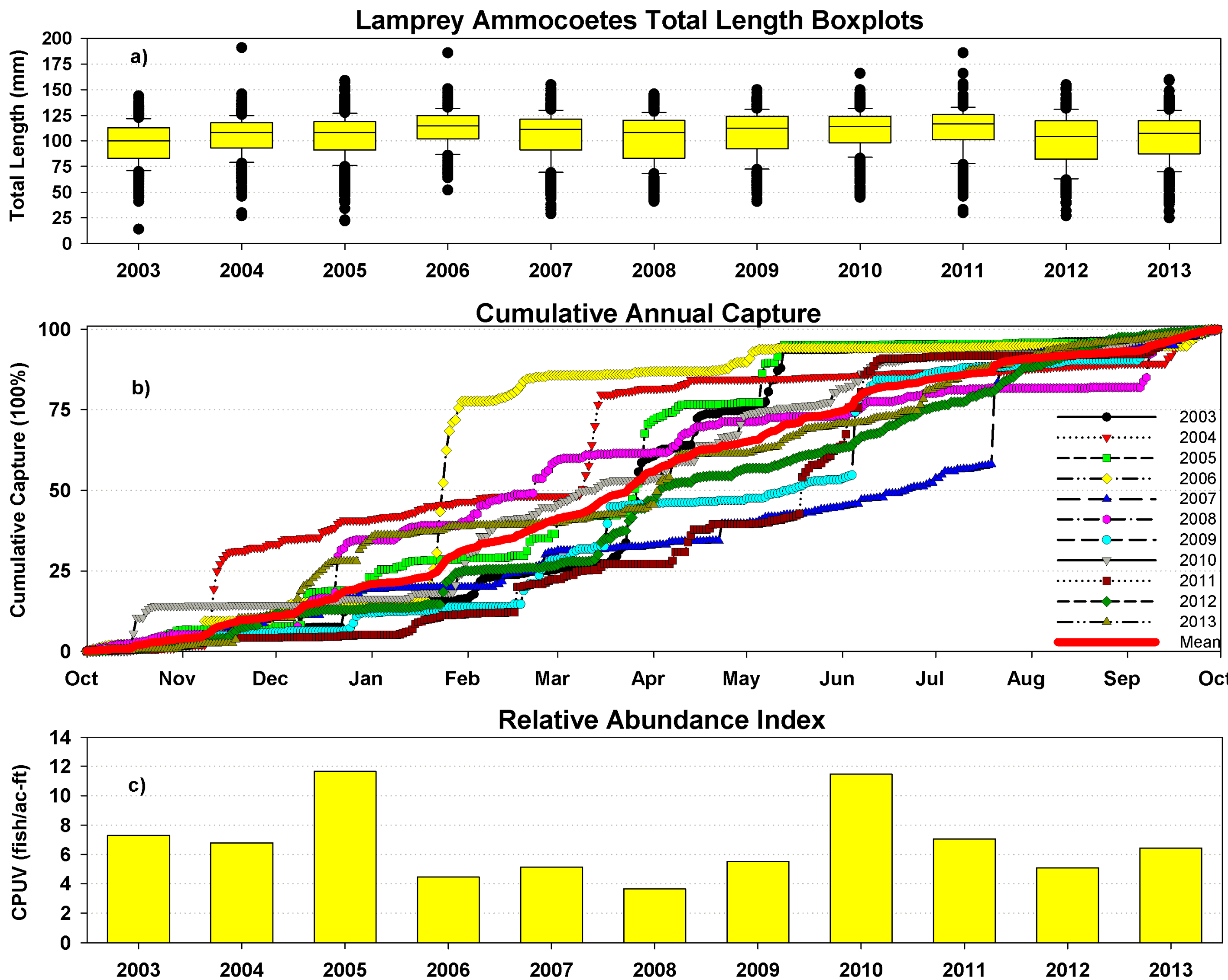


Figure 22. Unidentified lamprey ammocoetes a) total length distribution box plots, b) cumulative annual capture trends, and c) relative abundance indices from rotary trap samples collected between October 1, 2002 and September 30, 2013 by water year from the Sacramento River, CA at the RBDD (RM 243).

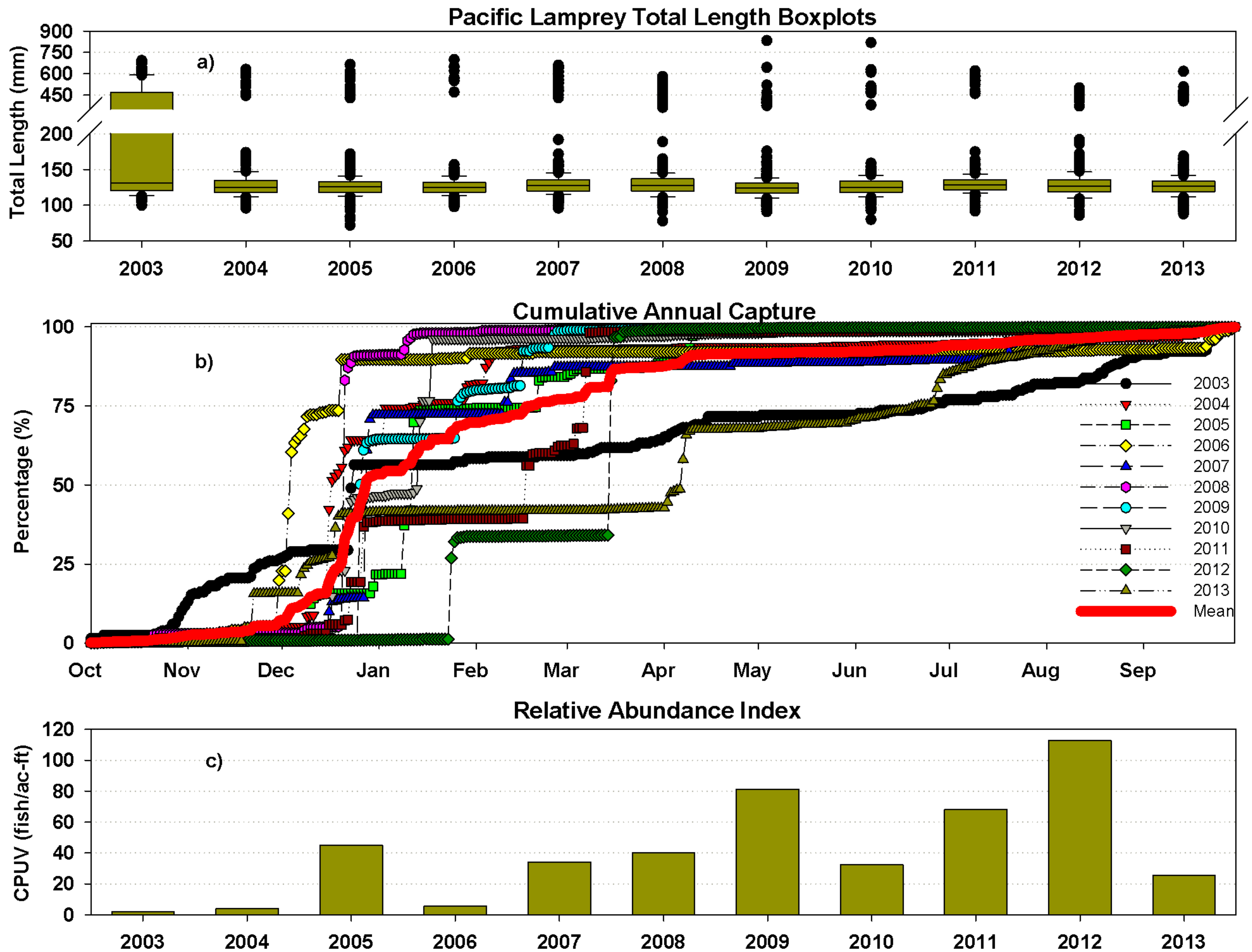


Figure 23. Pacific Lamprey (macropthalmia and adults) a) total length distribution box plots, b) cumulative annual capture trends, and c) relative abundance indices from rotary trap samples collected between October 1, 2002 and September 30, 2013 by water year from the Sacramento River, CA at the RBDD (RM 243).

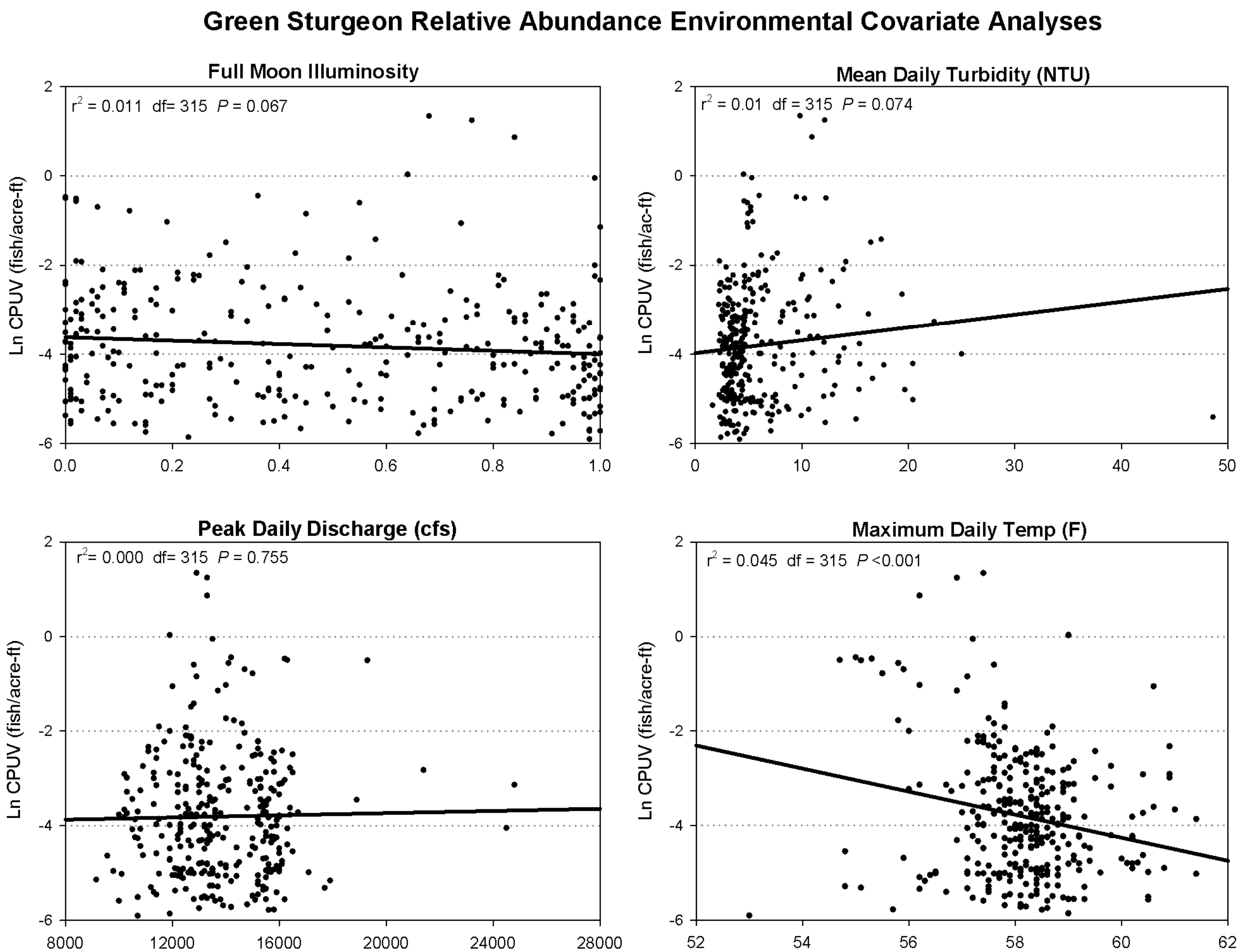


Figure 24. Regression analysis results of natural log (Ln) Green Sturgeon catch per unit volume (CPUV) and a) full moon illuminosity, b) mean daily turbidity, c) peak daily discharge and d) maximum daily temperatures at RBDD. All fish captured by rotary trap at RBDD (RM 243) on the Upper Sacramento River, CA between 2003 and 2012. Data from 2002 excluded from analysis due to limited effort and USBR Crown Flow study resulting in incomparable sampling regimes and results.

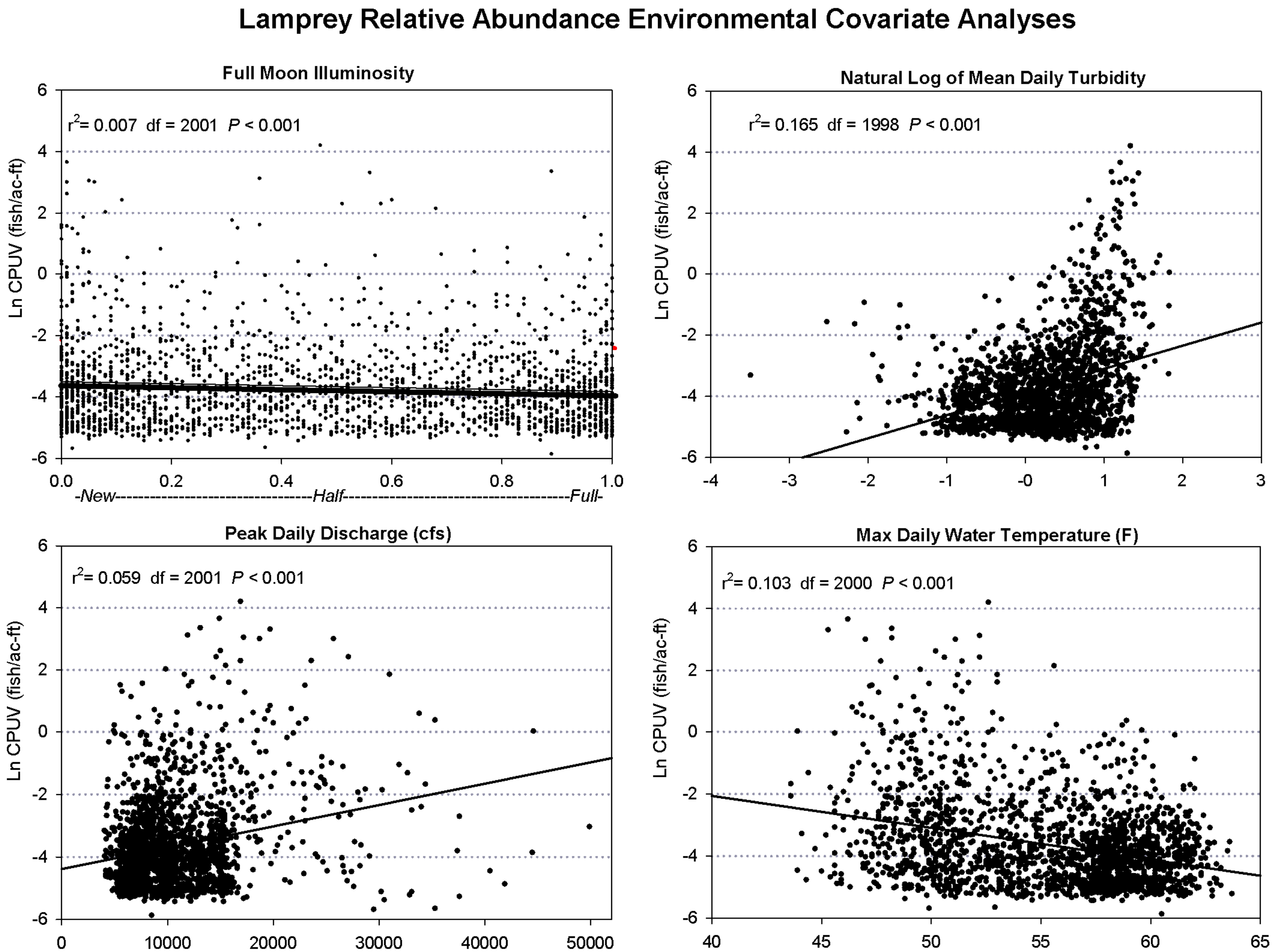


Figure 25. Regression analysis results of natural log (Ln) Lamprey *spp.* catch per unit volume (CPUV) and a) full moon illuminosity, b) Ln mean daily turbidity, c) peak daily discharge and d) maximum daily temperatures at RBDD. All fish captured by rotary trap at RBDD (RM 243) on the Upper Sacramento River, CA between water year 2003 and 2013.

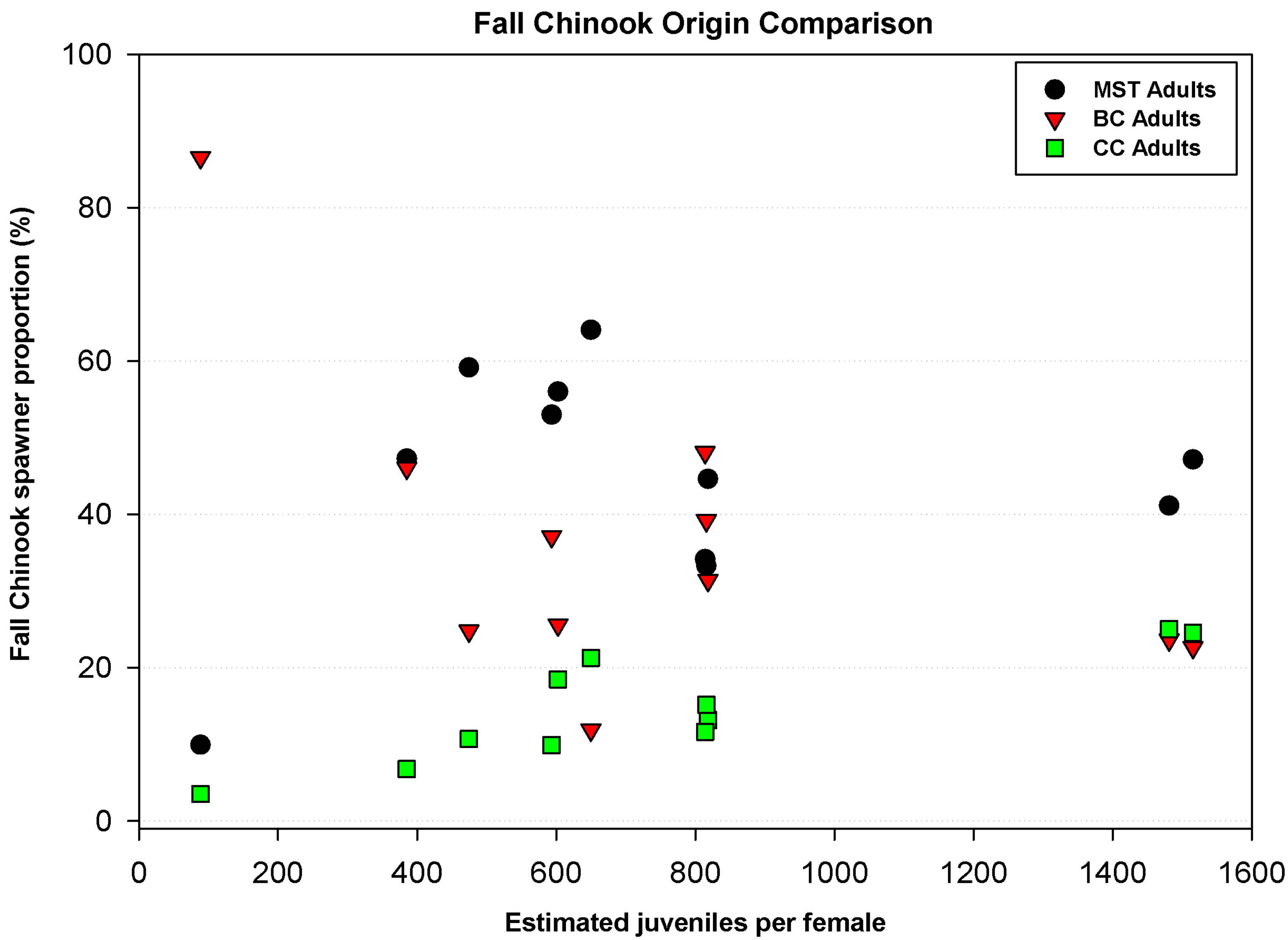
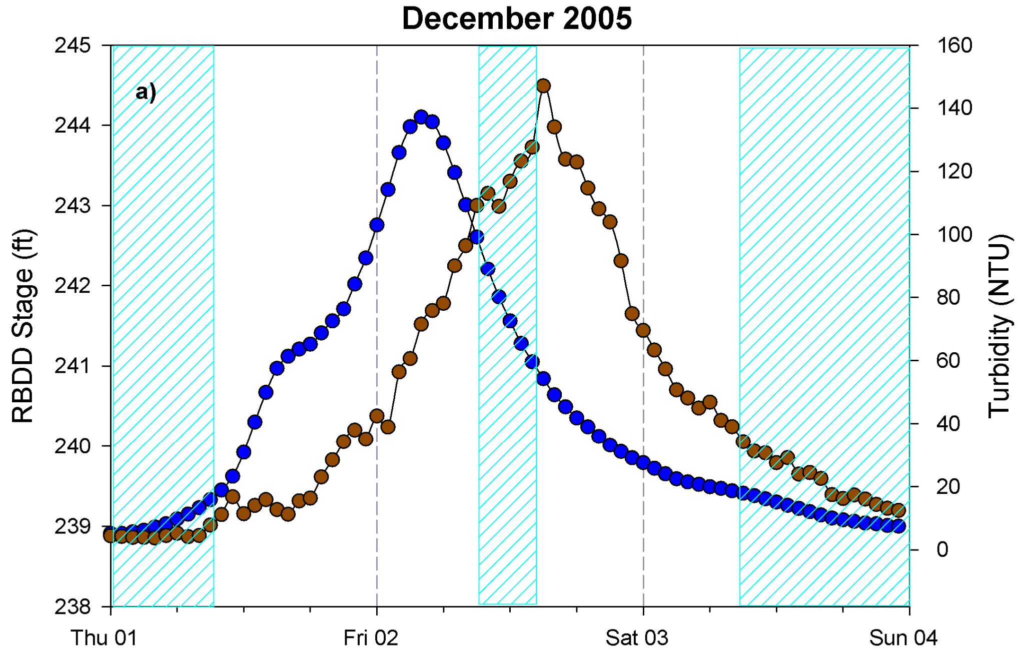


Figure 26. Comparison of estimated juveniles produced per estimated number of females in relation to distribution of fall Chinook spawners in the mainstem Sacramento River (MST), Battle Creek (BC), and Clear Creek (CC) between years 2002 and 2012.



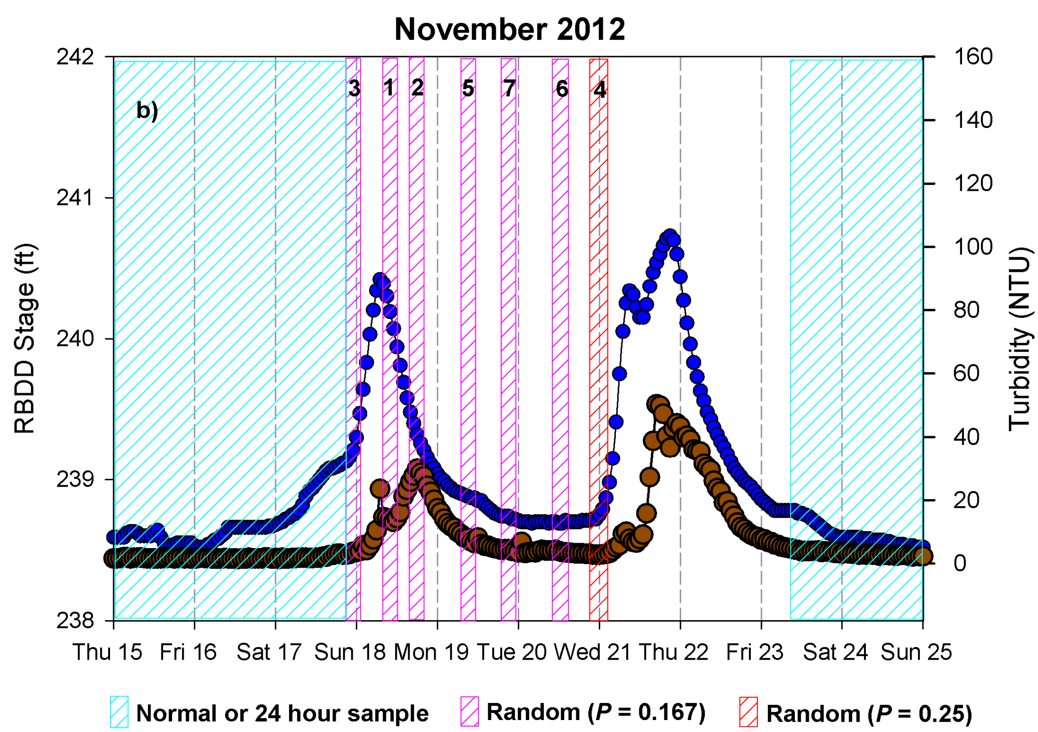


Figure 27. Timing comparison of RBDD stage (i.e., discharge level) and turbidity measurements along with sample collection times for storm events on a) December 1-4, 2005 and b) November 15-25, 2012. Numerals within sample period boxes in figure b indicate rank of standardized Chinook passage totals from greatest (1) to least (7).

APPENDIX 1

Appendix 1: List of Tables

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Fall Chinook

Table A1. Summary of RBDD rotary trap annual effort, fall Chinook fry (<46 mm FL) passage estimates and lower and upper 90% confidence intervals (CI), by brood year for the period December 2002 through September 2013.

Brood Year	Effort	Estimated Fry Passage	Low 90% CI	Up 90% CI
2002	0.76	14,687,984	348,386	42,027,818
2003	0.81	23,612,094	6,953,966	44,283,689
2004	0.85	7,946,496	3,449,094	12,447,378
2005	0.56	11,740,225	2,452,034	24,687,255
2006	0.90	10,152,406	3,458,524	17,567,355
2007	0.88	9,594,099	4,834,813	14,353,810
2008	0.79	6,684,332	3,335,617	10,033,164
2009	0.84	6,900,302	2,190,210	11,662,489
2010	0.75	6,302,961	3,432,017	9,502,694
2011	0.87	4,437,956	2,380,436	6,498,878
2012	0.85	21,375,192	14,332,396	28,700,826

Table A2. Summary of RBDD rotary trap annual effort, fall Chinook pre-smolt/smolt (>45 mm FL) passage estimates and lower and upper 90% confidence intervals (CI), by brood year for the period December 2002 through September 2013.

Brood Year	Effort	Estimated Smolt Passage	Low 90% CI	Up 90% CI
2002	0.76	2,350,433	505,837	5,318,021
2003	0.81	4,124,773	1,879,521	6,393,281
2004	0.85	6,161,742	1,626,946	12,527,167
2005	0.56	6,470,030	1,041,939	14,426,210
2006	0.90	5,955,245	3,056,683	8,855,302
2007	0.88	2,537,504	1,291,848	3,821,912
2008	0.79	2,431,215	1,034,851	3,827,754
2009	0.84	1,632,074	868,002	2,396,298
2010	0.75	2,539,519	1,288,830	3,850,851
2011	0.87	1,833,305	1,029,403	2,637,509
2012	0.85	3,054,227	1,692,494	4,416,322

Late-Fall Chinook

Table A3. Summary of RBDD rotary trap annual effort, late-fall Chinook fry (<46 mm FL) passage estimates and lower and upper 90% confidence intervals (CI), by brood year for the period April 2002 through March 2013.

Brood Year	Effort	Estimated Fry Passage	Low 90% CI	Up 90% CI
2002	0.57	442,393	84,832	901,368
2003	0.76	196,271	4,562	683,458
2004	0.88	24,382	8,802	40,591
2005	0.73	50,274	5,723	175,598
2006	0.70	284,999	41,006	634,496
2007	0.90	144,688	54,397	235,201
2008	0.89	10,489	4,347	17,813
2009	0.72	29,568	13,126	46,360
2010	0.86	113,667	26,705	200,935
2011	0.77	69,686	18,487	120,996
2012	0.89	67,479	9,925	136,431

Table A4. Summary of RBDD rotary trap annual effort, late-fall Chinook pre-smolt/smolt (>45 mm FL) passage estimates and lower and upper 90% confidence intervals (CI), by brood year for the period April 2002 through March 2013.

Brood Year	Effort	Estimated Smolt Passage	Low 90% CI	Up 90% CI
2002	0.57	2,117,122	569,453	4,093,545
2003	0.76	149,976	72,089	230,841
2004	0.88	122,779	64,498	181,783
2005	0.73	93,407	35,067	160,738
2006	0.70	175,269	82,005	273,572
2007	0.90	390,932	213,642	568,595
2008	0.89	81,506	41,983	121,166
2009	0.72	190,256	83,201	297,652
2010	0.86	69,771	33,929	106,575
2011	0.77	27,354	9,535	45,914
2012	0.89	73,055	32,567	113,633

Winter Chinook

Table A5. Summary of RBDD rotary trap annual effort, winter Chinook fry (<46 mm FL) passage estimates and lower and upper 90% confidence intervals (CI), by brood year for the period July 2002 through June 2013.

Brood Year	Effort	Estimated Fry Passage	Low 90% CI	Up 90% CI
2002	0.64	6,381,286	2,156,758	11,217,962
2003	0.81	4,420,296	2,743,637	6,096,955
2004	0.84	3,087,102	1,812,619	4,361,584
2005	0.64	7,533,380	4,225,130	10,841,630
2006	0.83	5,813,140	3,307,323	8,318,957
2007	0.89	1,158,791	744,804	1,572,817
2008	0.87	1,063,919	662,381	1,465,748
2009	0.75	3,587,134	2,076,422	5,098,125
2010	0.81	875,049	603,549	1,146,644
2011	0.82	638,056	441,983	834,289
2012	0.89	722,048	545,751	898,345

Table A6. Summary of RBDD rotary trap annual effort, winter Chinook pre-smolt/smolt (>45 mm FL) passage estimates and lower and upper 90% confidence intervals (CI), by brood year for the period July 2002 through June 2013.

Brood Year	Effort	Estimated Smolt Passage	Low 90% CI	Up 90% CI
2002	0.64	737,755	373,538	1,149,079
2003	0.81	800,719	453,256	1,169,559
2004	0.84	347,581	179,502	519,265
2005	0.64	829,302	324,860	1,442,763
2006	0.83	873,940	487,244	1,264,701
2007	0.89	281,773	180,254	387,123
2008	0.87	181,071	110,592	252,089
2009	0.75	815,188	410,512	1,222,586
2010	0.81	410,341	210,252	613,810
2011	0.82	210,920	130,861	291,312
2012	0.89	627,771	354,764	900,897

Spring Chinook

Table A7. Summary of RBDD rotary trap annual effort, spring Chinook fry (<46 mm FL) passage estimates and lower and upper 90% confidence intervals (CI), by brood year for the period October 2002 through September 2013.

Brood Year	Effort	Estimated Fry Passage	Low 90% CI	Up 90% CI
2002	0.75	159,084	67,900	255,023
2003	0.81	502,386	189,371	857,899
2004	0.85	155,053	59,655	250,451
2005	0.57	427,719	111,396	925,898
2006	0.89	174,186	114,642	233,907
2007	0.89	336,714	212,765	460,712
2008	0.85	40,213	26,016	54,448
2009	0.79	219,627	91,683	347,845
2010	0.77	89,213	39,829	138,597
2011	0.86	88,355	63,469	113,274
2012	0.86	134,028	82,843	185,271

Table A8. Summary of RBDD rotary trap annual effort, spring Chinook pre-smolt/smolt (>45 mm FL) passage estimates and lower and upper 90% confidence intervals (CI), by brood year for the period October 2002 through September 2013.

Brood Year	Effort	Estimated Smolt Passage	Low 90% CI	Up 90% CI
2002	0.75	118,393	43,022	239,870
2003	0.81	124,529	59,434	197,777
2004	0.85	275,898	113,564	460,990
2005	0.57	187,828	19,676	460,441
2006	0.89	247,250	123,621	371,968
2007	0.89	32,787	15,894	51,271
2008	0.85	124,460	40,130	208,954
2009	0.79	218,778	83,930	354,607
2010	0.77	69,753	21,938	123,577
2011	0.86	95,935	37,782	159,702
2012	0.86	186,869	89,566	284,936

Table A9. River Lamprey, *Lampetra ayresi*, annual capture, catch per unit volume (CPUV) and total length summaries for River Lamprey captured by RBDD rotary traps between water year (WY) 2003 and 2013.

WY	Catch	CPUV Fish/ac-ft	Min TL (mm)	Max TL (mm)	Mean (mm)	Median (mm)
2003	0	0.00	-	-	-	-
2004	1	0.01	102	102	102	-
2005	0	0.00	-	-	-	-
2006	0	0.00	-	-	-	-
2007	0	0.00	-	-	-	-
2008	0	0.00	-	-	-	-
2009	0	0.00	-	-	-	-
2010	1	0.01	110	110	110	-
2011	26	0.23	99	151	121	121
2012	4	0.02	128	168	144	140
2013	0	0.00	-	-	-	-
Mean	2.9	0.02	109.8	132.8	119.3	130.5
SD	7.8	0.07	13.0	31.8	18.2	13.4
CV	266.5%	279.2%	11.9%	24.0%	15.3%	10.3%

Table A10. Pacific Brook Lamprey, *Lampetra pacifica,* annual capture, catch per unit volume (CPUV) and total length summaries for Pacific Brook Lamprey captured by RBDD rotary traps between water year (WY) 2003 and 2013.

WY	Catch	CPUV Fish/ac-ft	Min TL (mm)	Max TL (mm)	Mean (mm)	Median (mm)
2003	6	0.06	98	132	116	114.5
2004	1	0.01	159	159	159	-
2005	0	0.00	-	-	-	-
2006	0	0.00	-	-	-	-
2007	0	0.00	-	-	-	-
2008	0	0.00	-	-	-	-
2009	0	0.00	-	-	-	-
2010	1	0.02	120	120	120	120
2011	1	0.01	147	147	147	147
2012	6	0.04	112	156	138	142
2013	21	0.12	110	148	124	122
Mean	3.3	0.02	124.3	143.7	134.0	129.1
SD	6.3	0.04	23.6	14.9	16.9	14.4
CV	192.8%	159.7%	19.0%	10.4%	12.6%	11.2%

APPENDIX 2

Appendix 2: List of Figures

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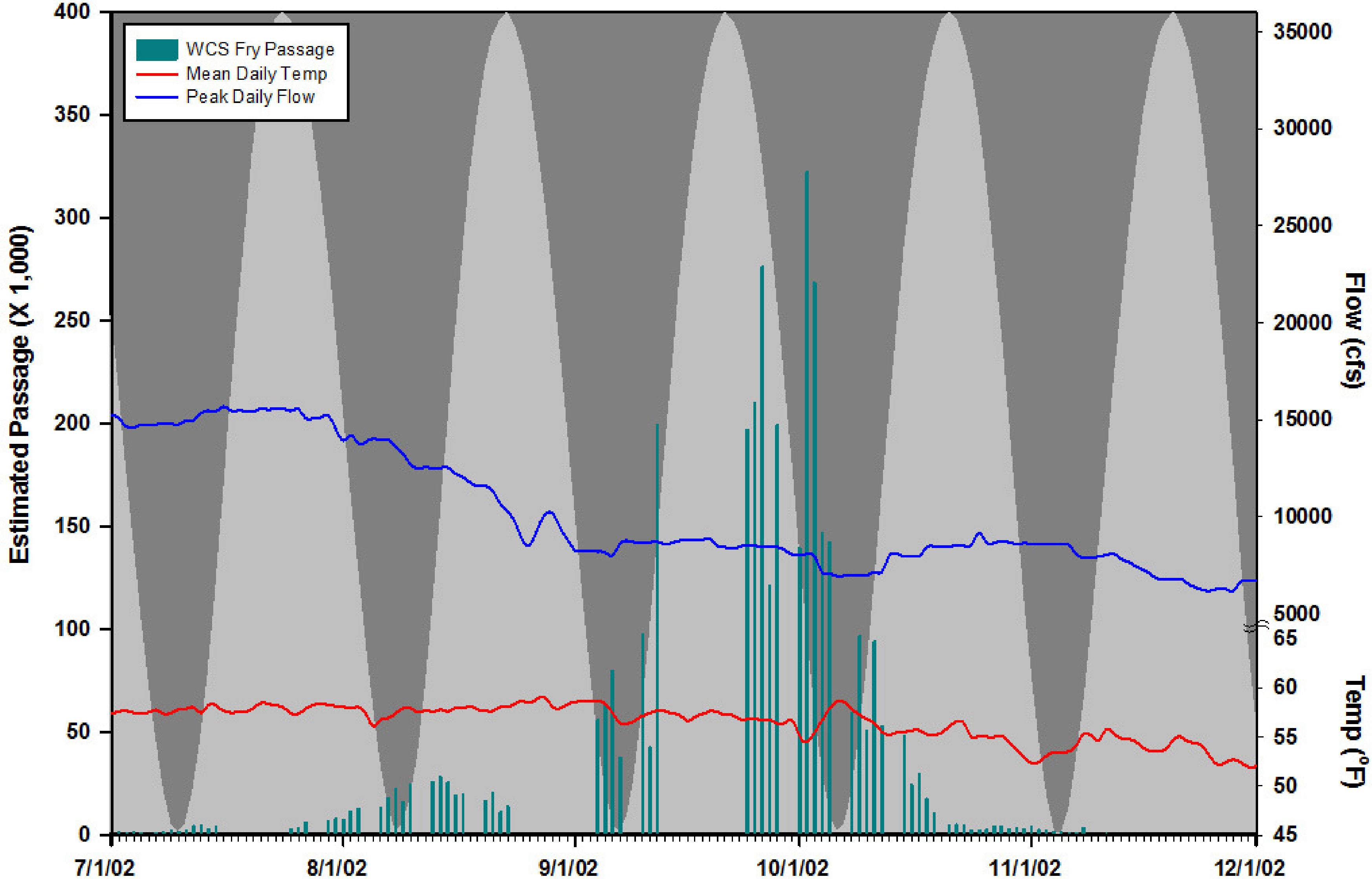

Figure A1. Brood Year 2002 winter Chinook fry passage with moon illuminosity indicated by back ground shading (peak of light gray equals full moon), mean daily water temperatures (red), and peak daily flows (blue) at Red Bluff Diversion Dam.

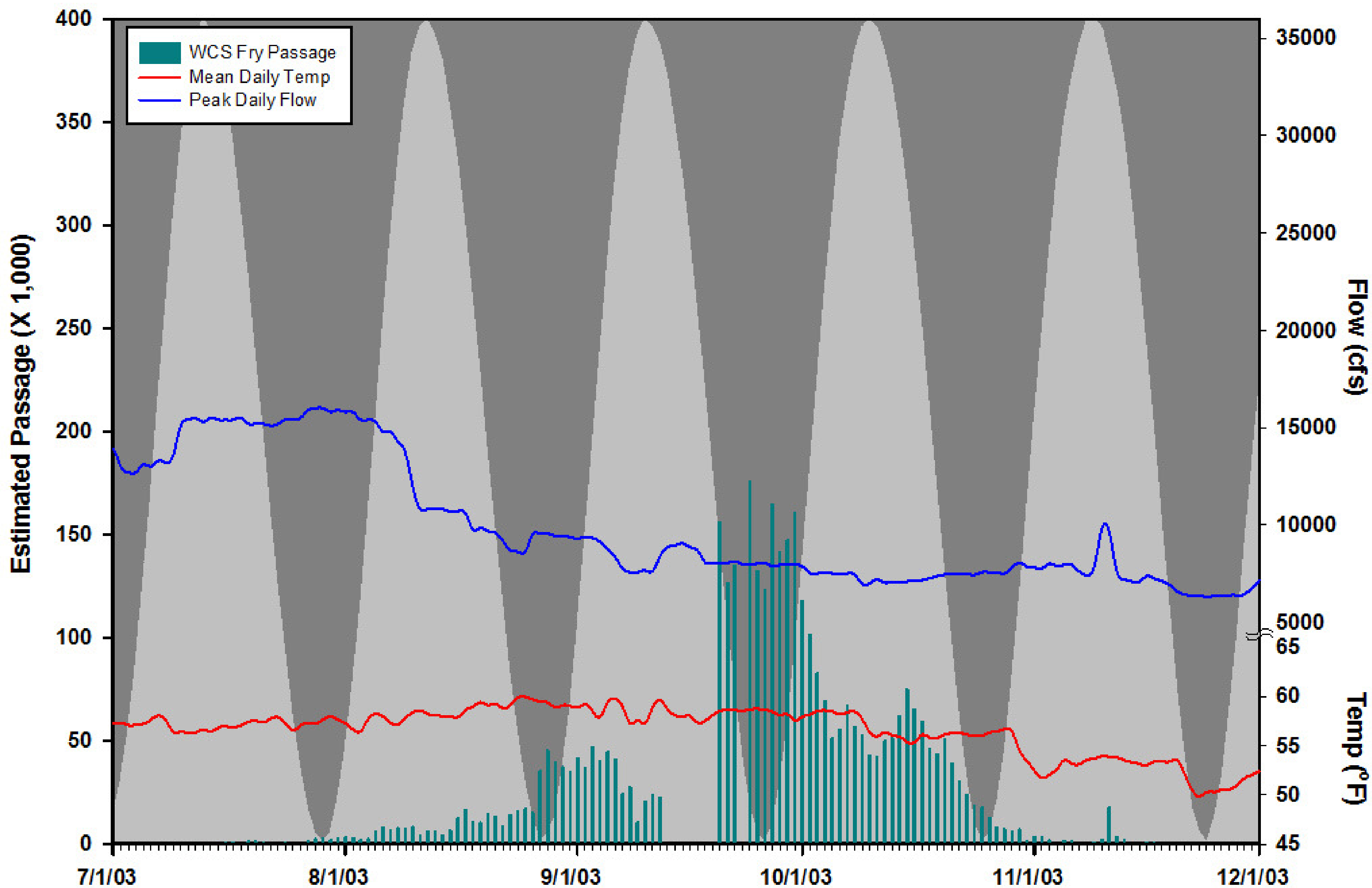

Figure A2. Brood Year 2003 winter Chinook fry passage with moon illuminosity indicated by back ground shading (peak of light gray equals full moon), mean daily water temperatures (red), and peak daily flows (blue) at Red Bluff Diversion Dam.

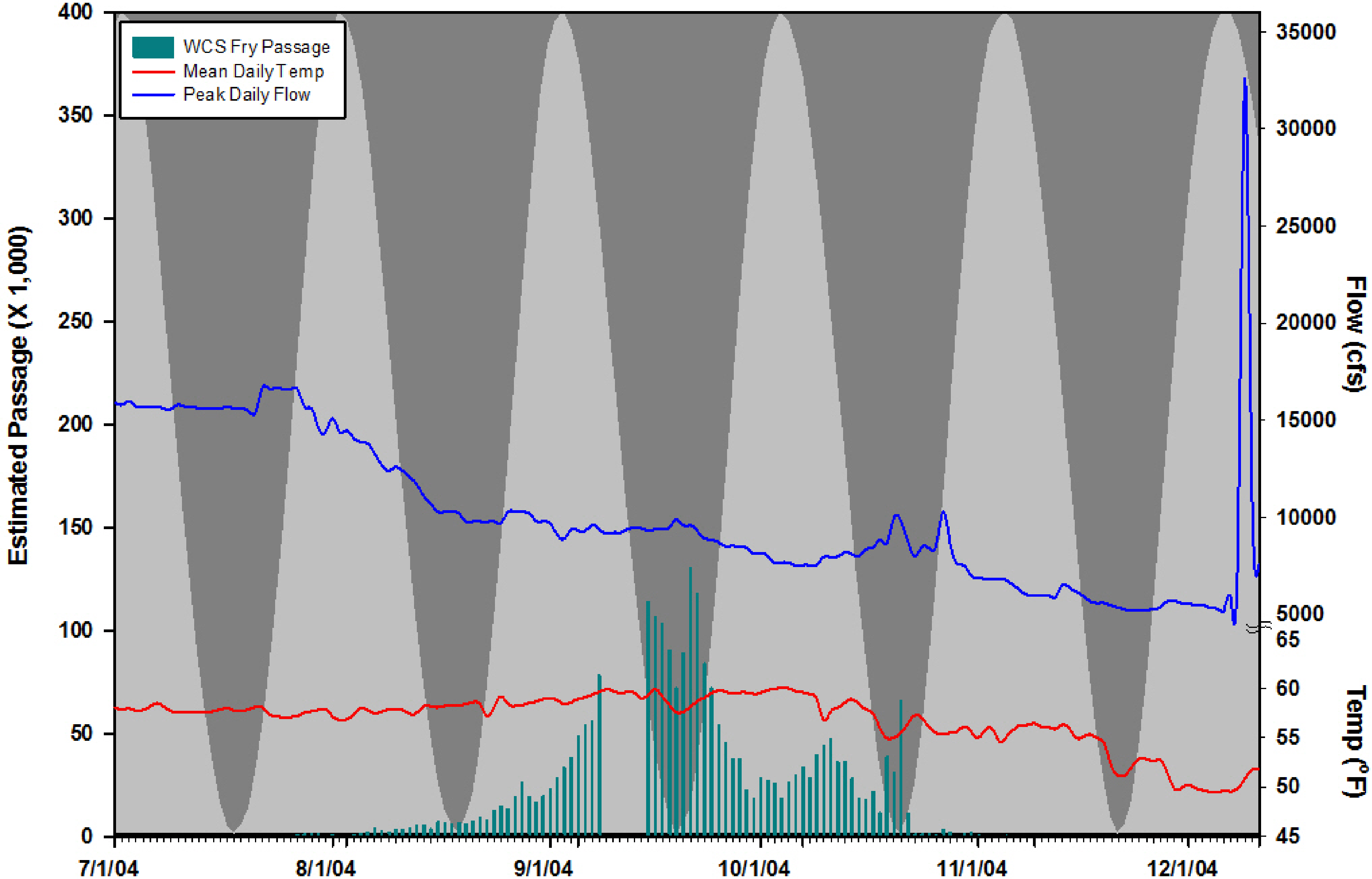

Figure A3. Brood Year 2004 winter Chinook fry passage with moon illuminosity indicated by back ground shading (peak of light gray equals full moon), mean daily water temperatures (red), and peak daily flows (blue) at Red Bluff Diversion Dam.

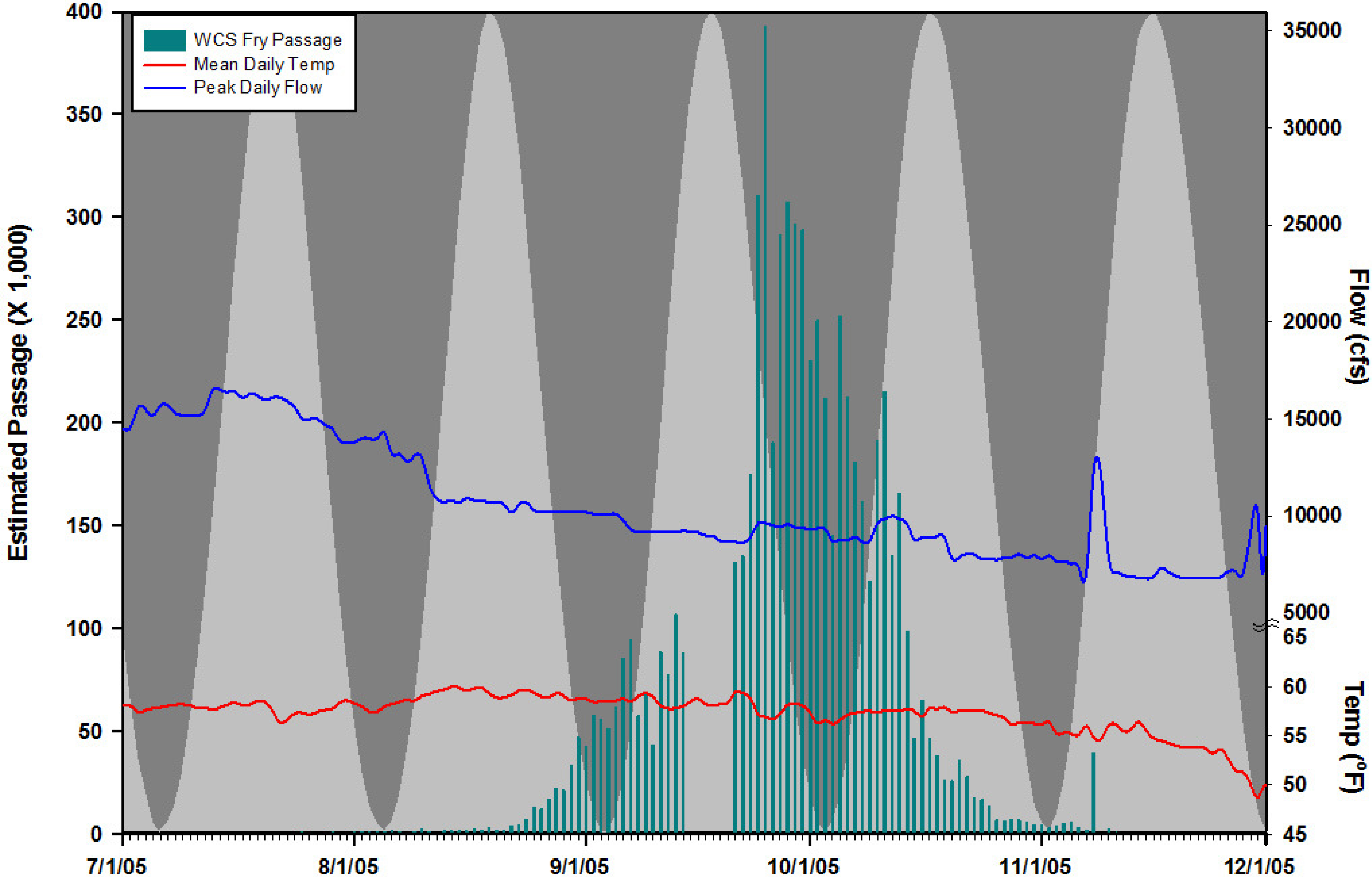

Figure A4. Brood Year 2005 winter Chinook fry passage with moon illuminosity indicated by back ground shading (peak of light gray equals full moon), mean daily water temperatures (red), and peak daily flows (blue) at Red Bluff Diversion Dam.

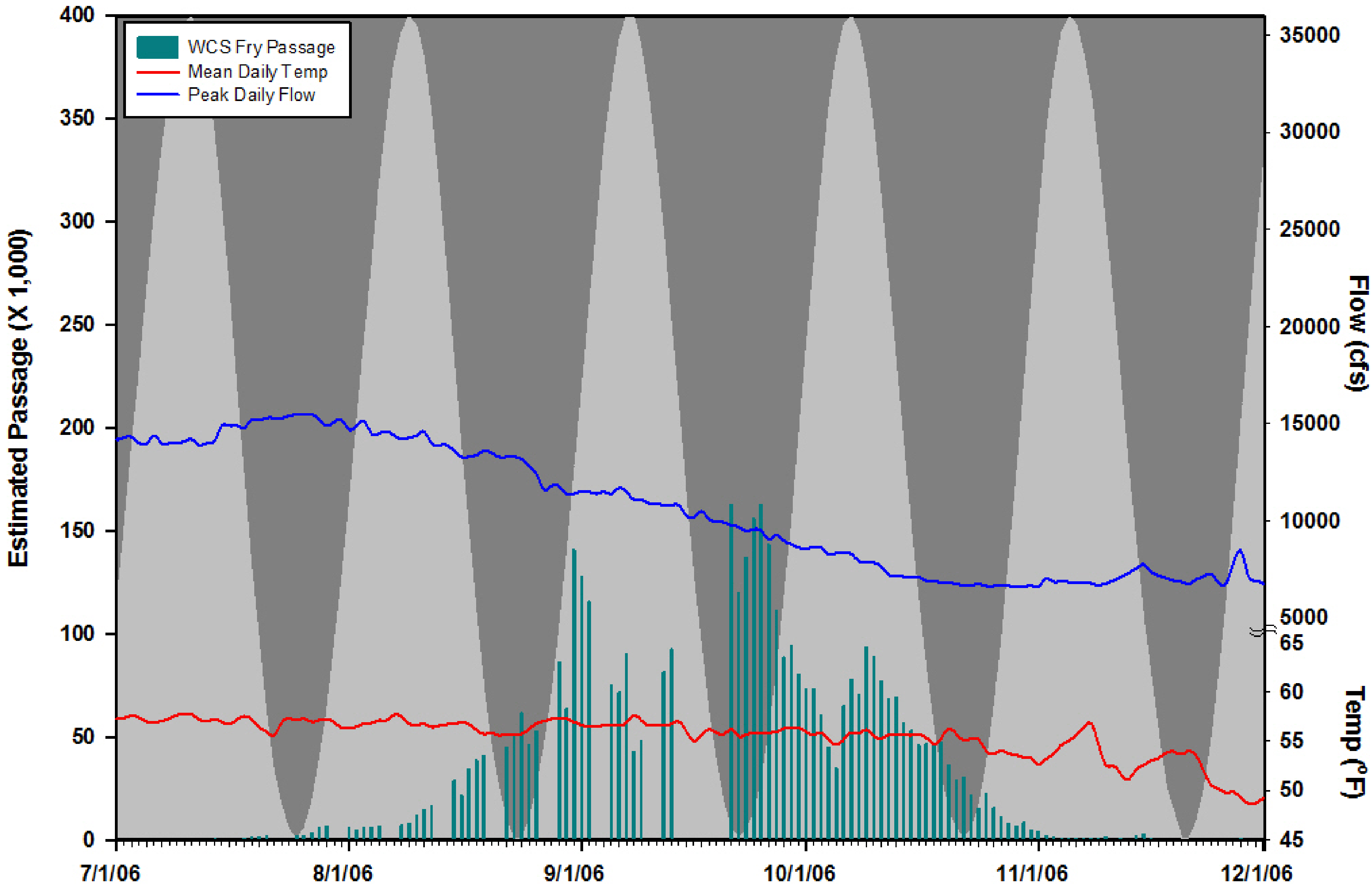

Figure A5. Brood Year 2006 winter Chinook fry passage with moon illuminosity indicated by back ground shading (peak of light gray equals full moon), mean daily water temperatures (red), and peak daily flows (blue) at Red Bluff Diversion Dam.

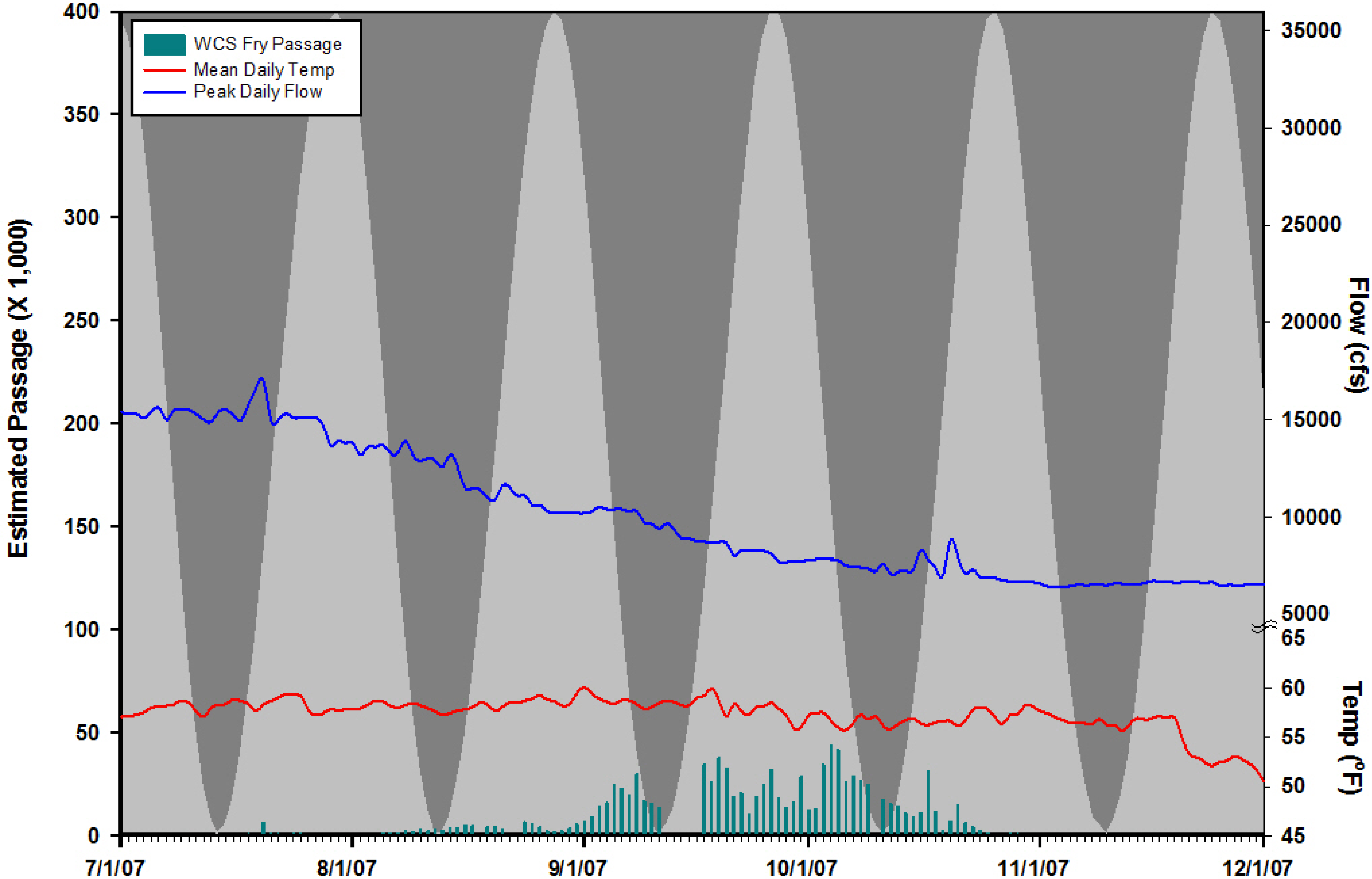

Figure A6. Brood Year 2007 winter Chinook fry passage with moon illuminosity indicated by back ground shading (peak of light gray equals full moon), mean daily water temperatures (red), and peak daily flows (blue) at Red Bluff Diversion Dam.

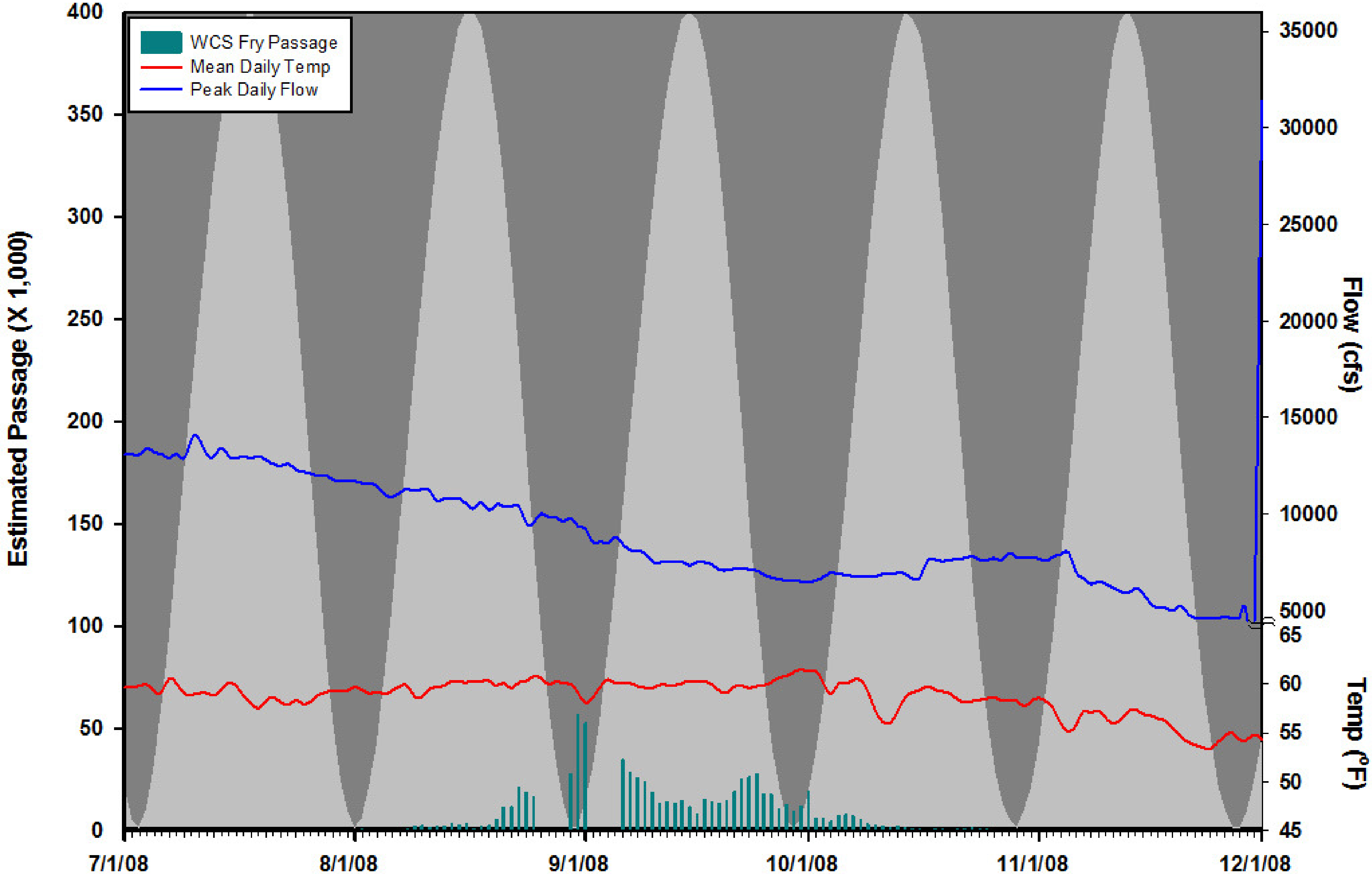

Figure A7. Brood Year 2008 winter Chinook fry passage with moon illuminosity indicated by back ground shading (peak of light gray equals full moon), mean daily water temperatures (red), and peak daily flows (blue) at Red Bluff Diversion Dam.

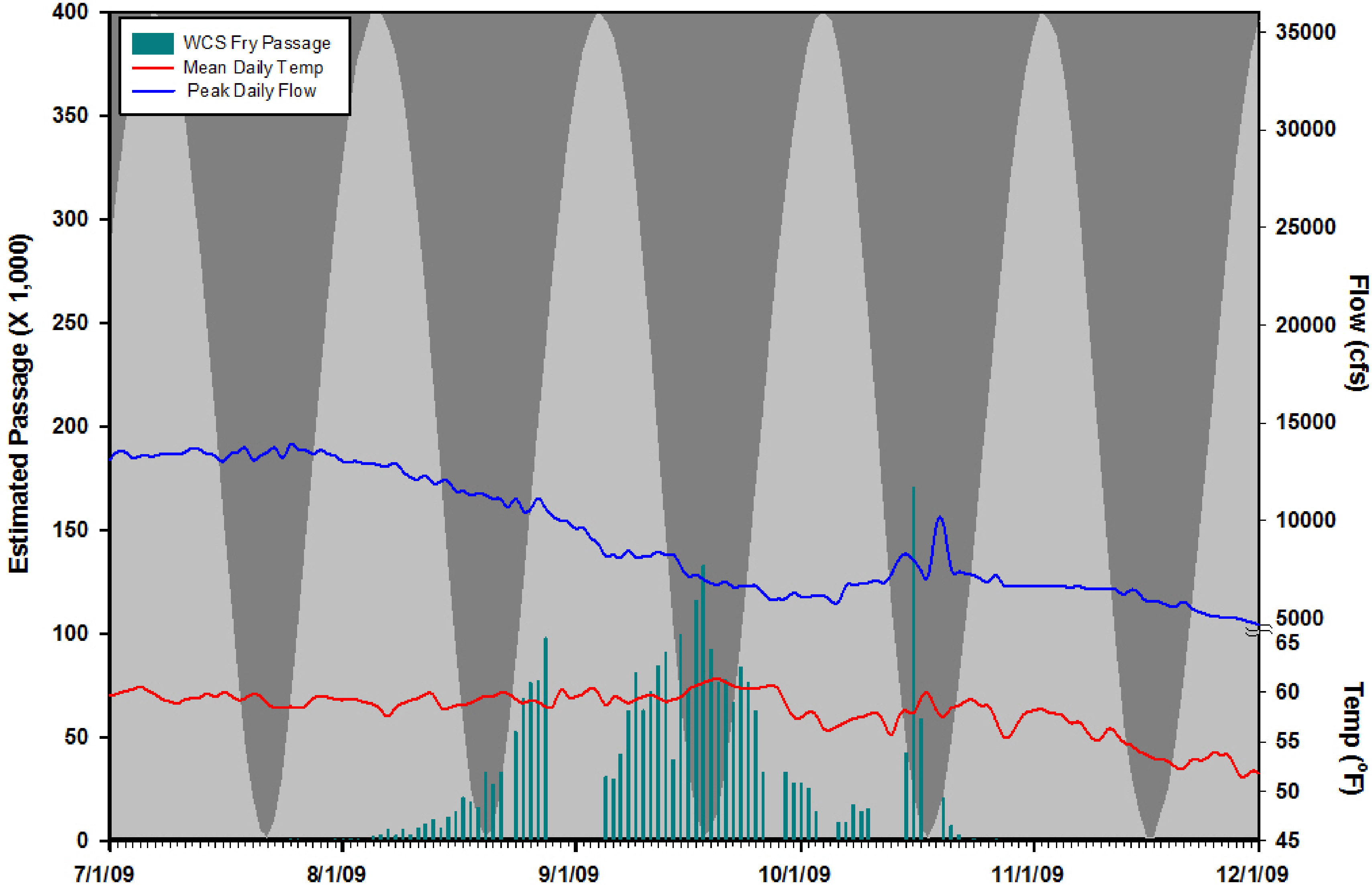

Figure A8. Brood Year 2009 winter Chinook fry passage with moon illuminosity indicated by back ground shading (peak of light gray equals full moon), mean daily water temperatures (red), and peak daily flows (blue) at Red Bluff Diversion Dam.

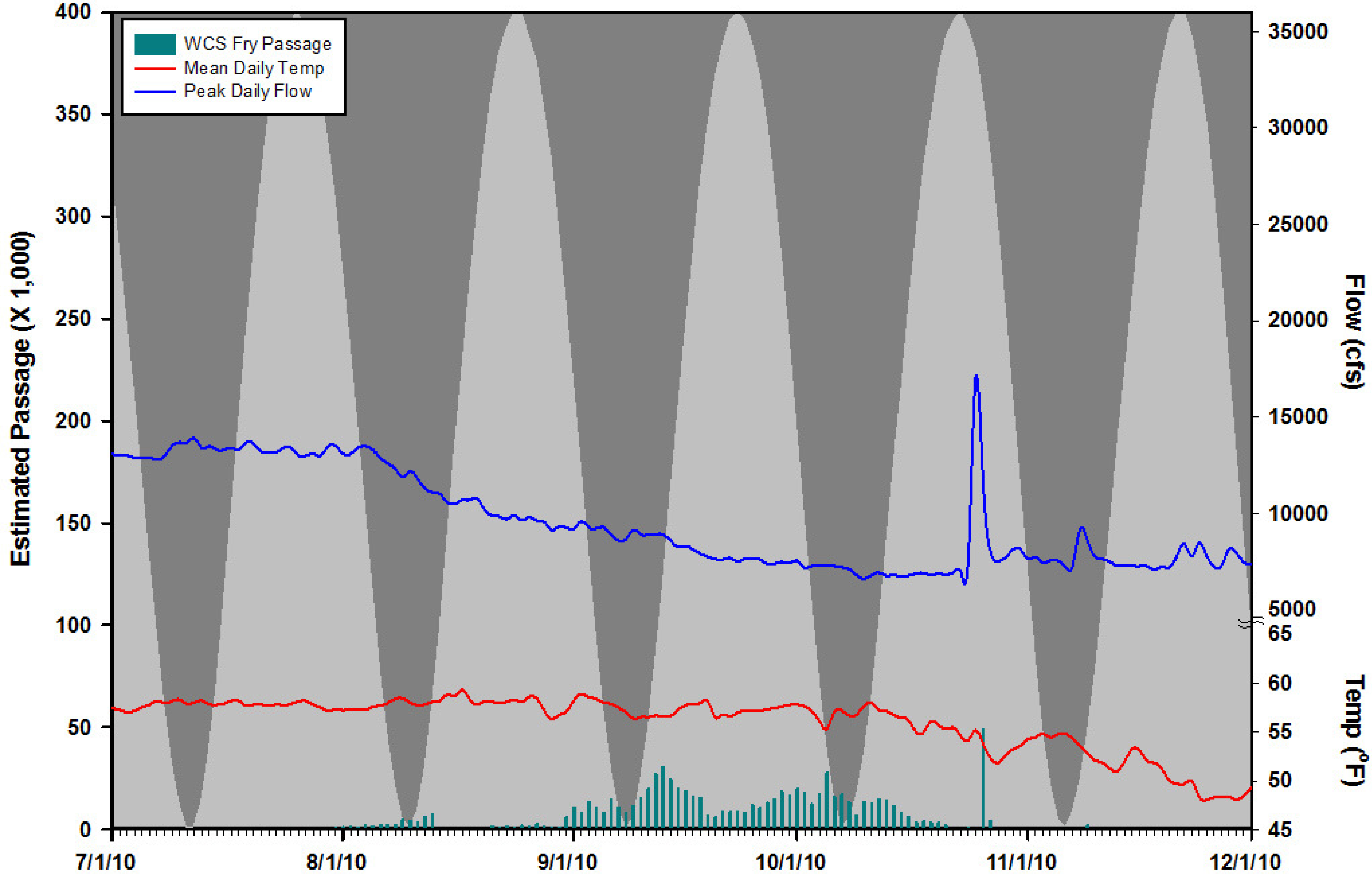

Figure A9. Brood Year 2010 winter Chinook fry passage with moon illuminosity indicated by back ground shading (peak of light gray equals full moon), mean daily water temperatures (red), and peak daily flows (blue) at Red Bluff Diversion Dam.

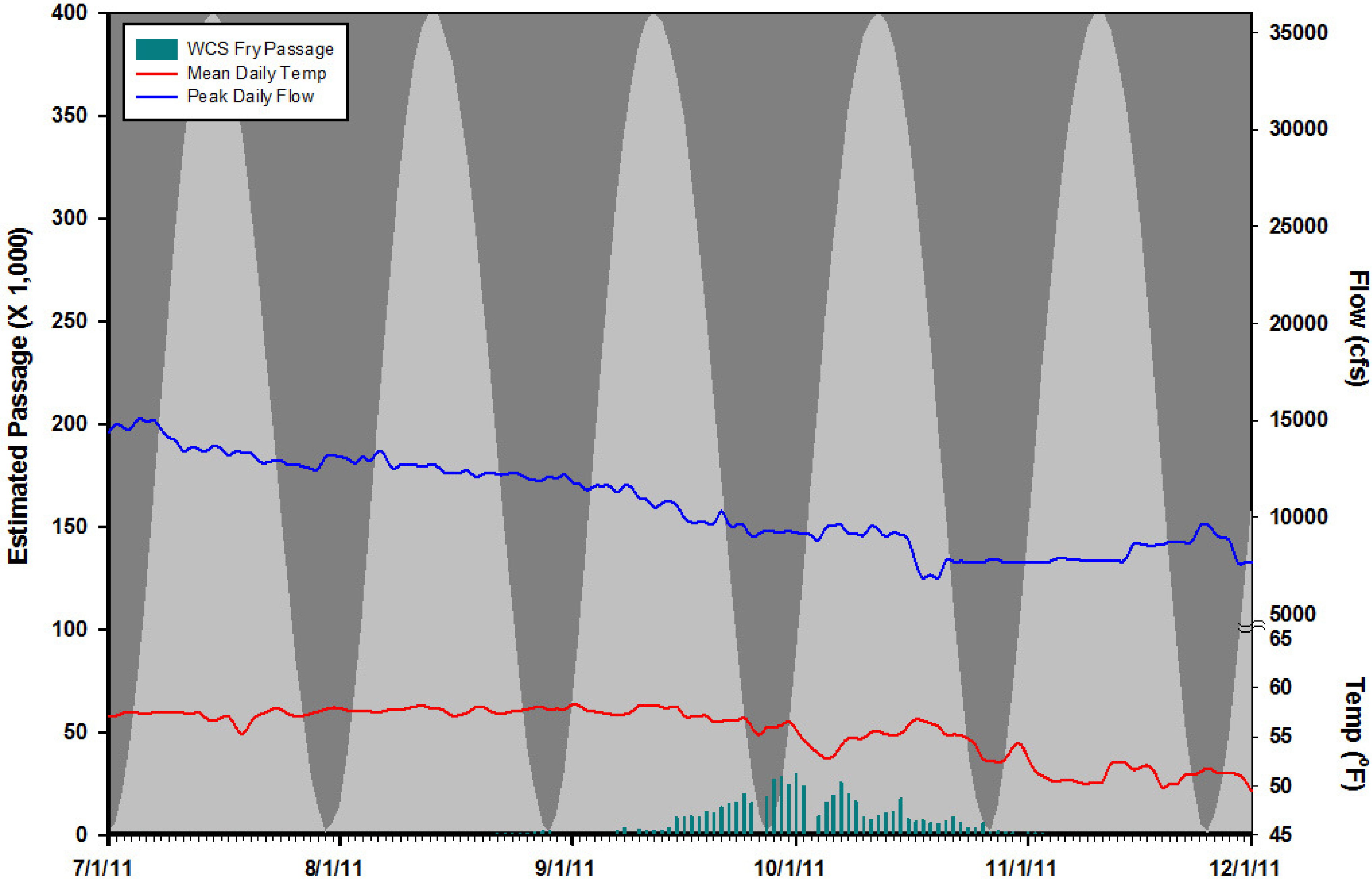

Figure A10. Brood Year 2011 winter Chinook fry passage with moon illuminosity indicated by back ground shading (peak of light gray equals full moon), mean daily water temperatures (red), and peak daily flows (blue) at Red Bluff Diversion Dam.

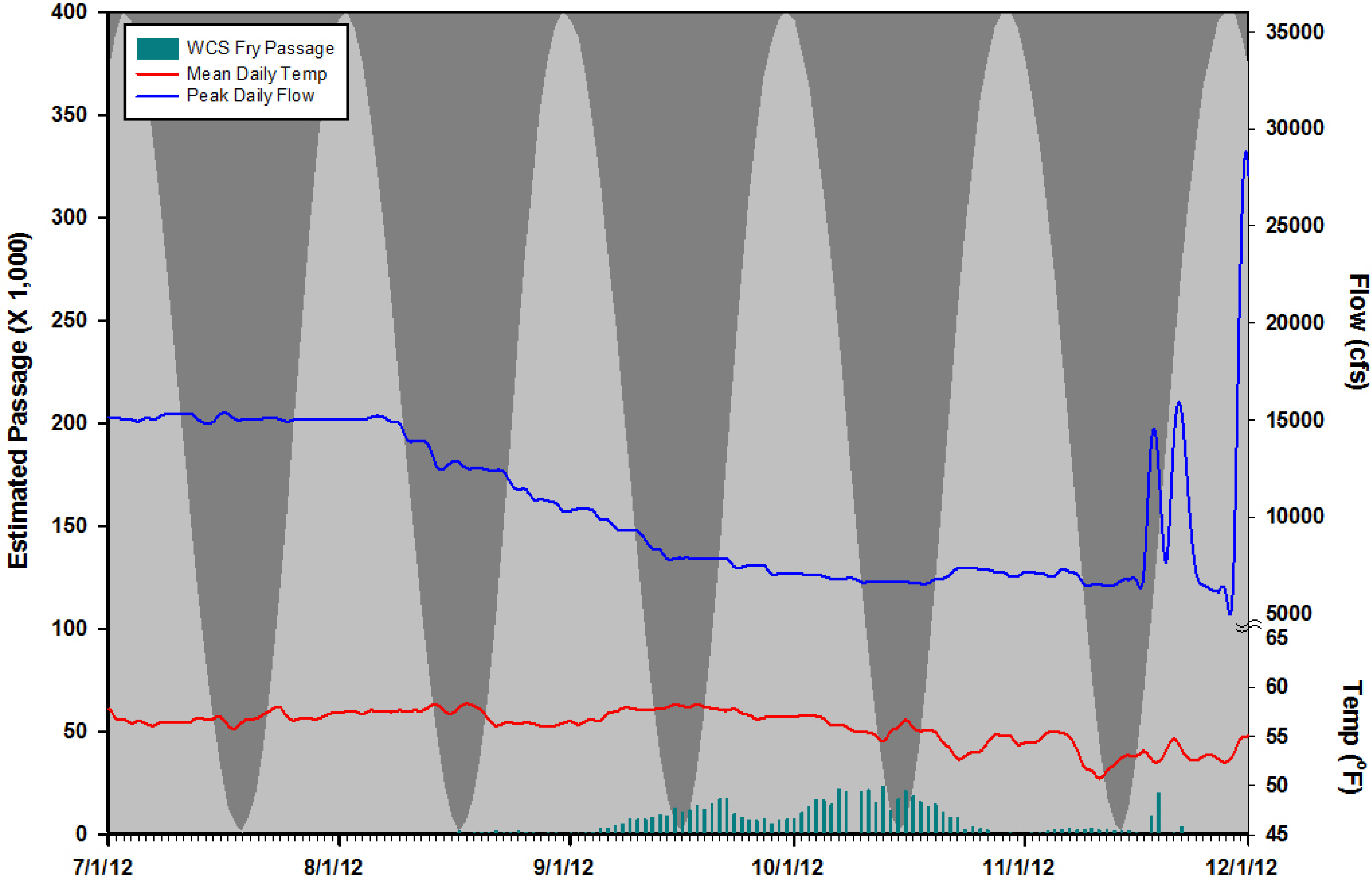

Figure A11. Brood Year 2012 winter Chinook fry passage with moon illuminosity indicated by back ground shading (peak of light gray equals full moon), mean daily water temperatures (red), and peak daily flows (blue) at Red Bluff Diversion Dam.

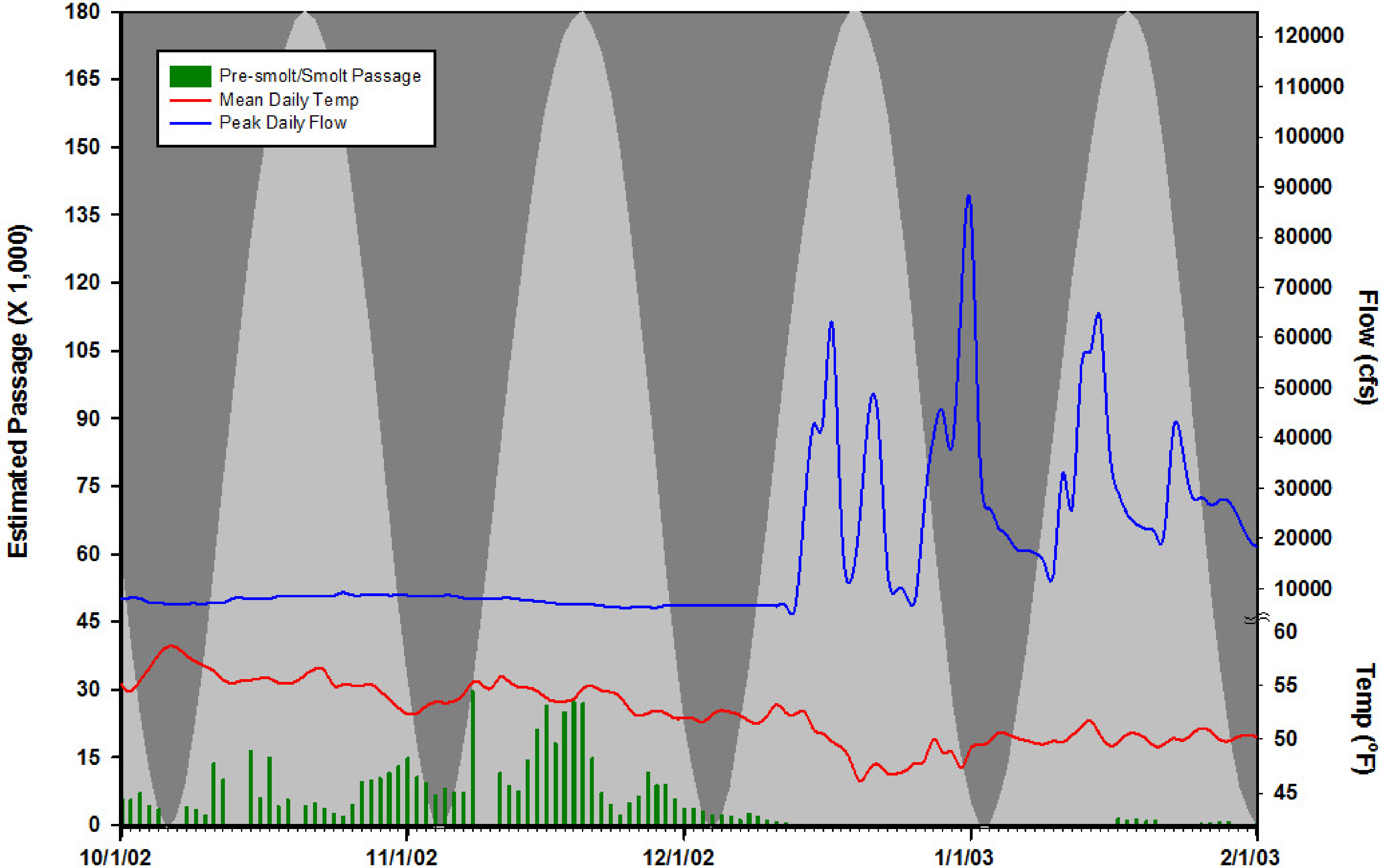

Figure A12. Brood Year 2002 winter Chinook pre-smolt/smolt passage with moon illuminosity indicated by back ground shading (peak of light gray equals full moon), mean daily water temperatures (red), and peak daily flows (blue) at Red Bluff Diversion Dam.

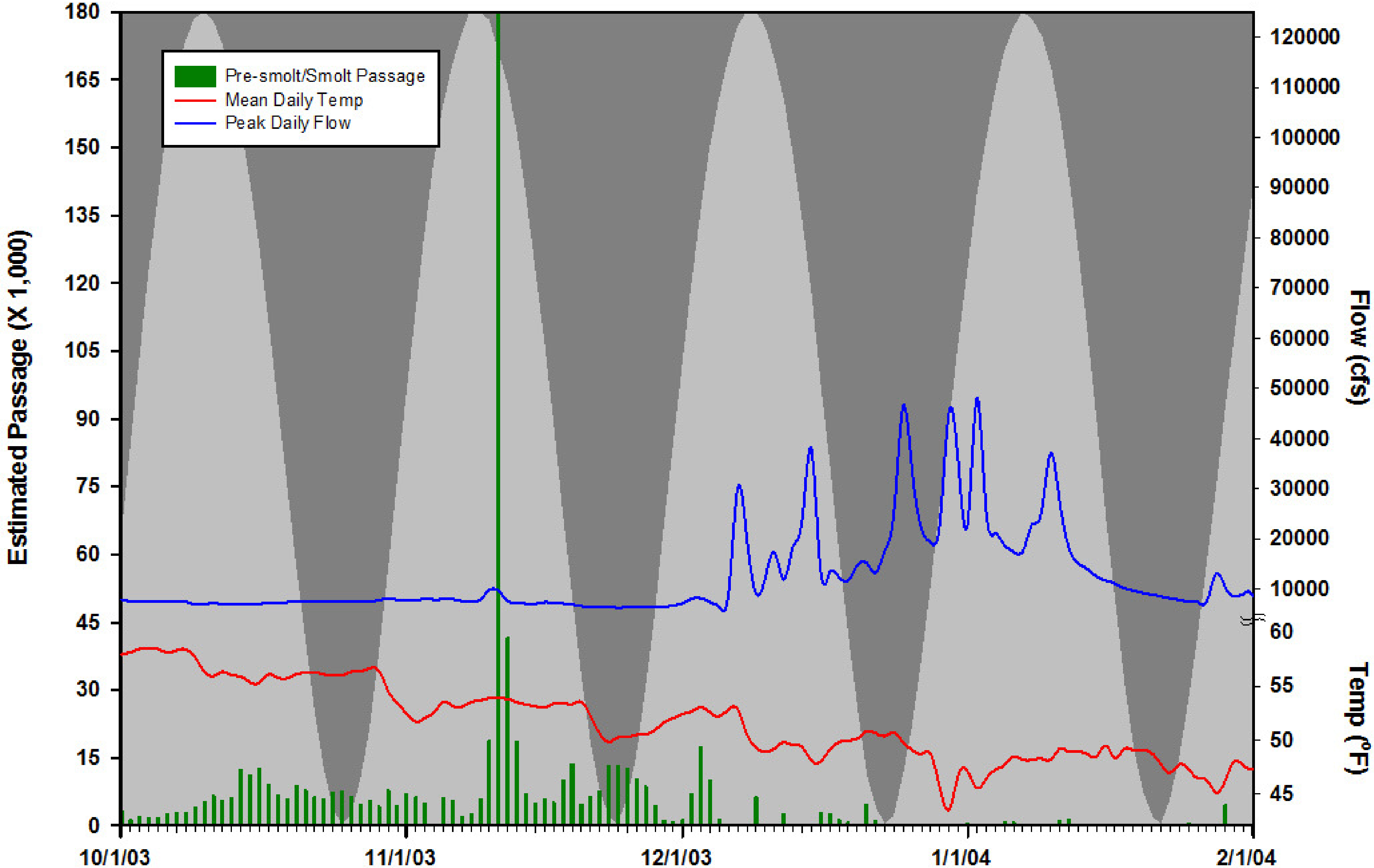

Figure A13. Brood Year 2003 winter Chinook pre-smolt/smolt passage with moon illuminosity indicated by back ground shading (peak of light gray equals full moon), mean daily water temperatures (red), and peak daily flows (blue) at Red Bluff Diversion Dam.

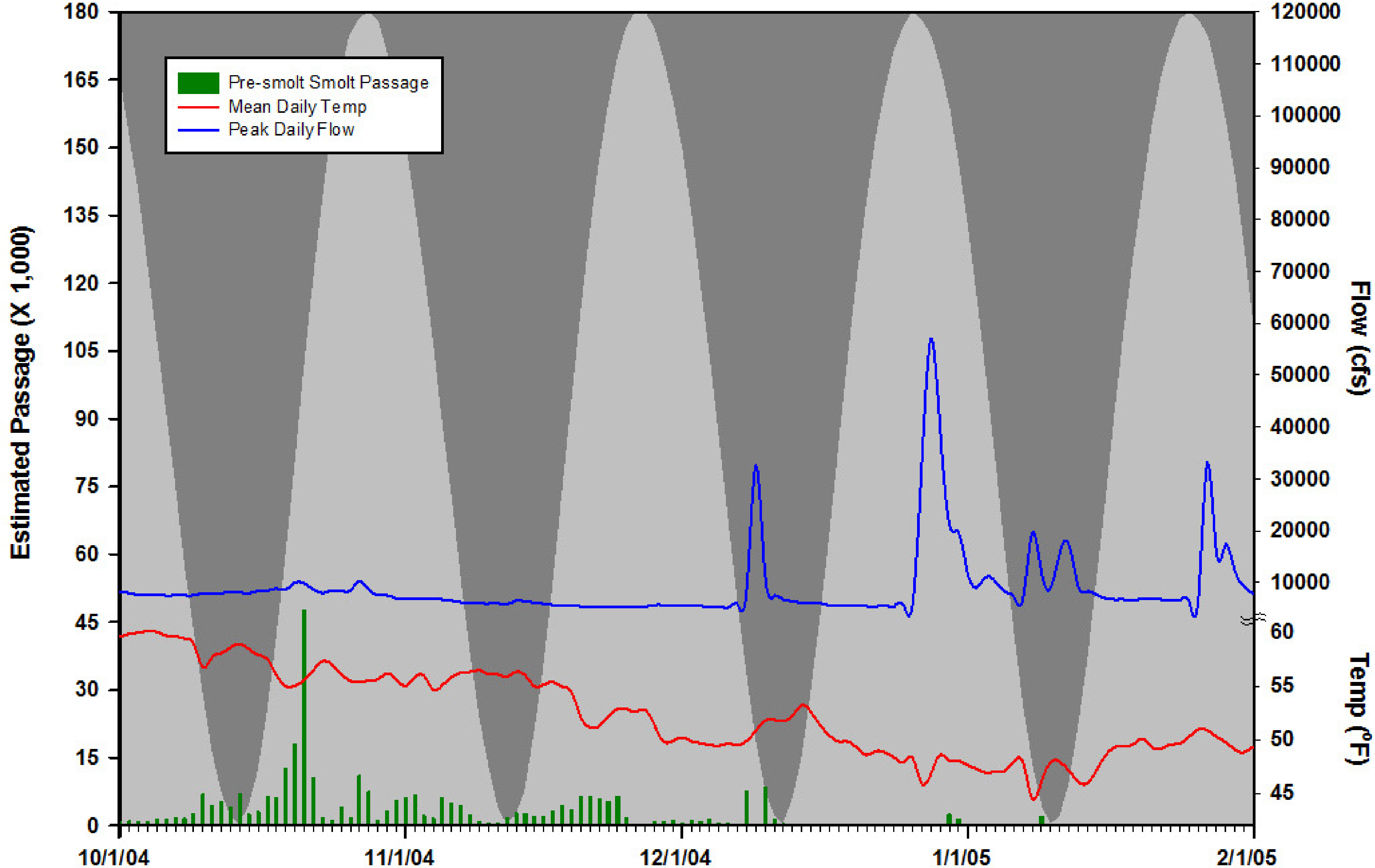

Figure A14. Brood Year 2004 winter Chinook pre-smolt/smolt passage with moon illuminosity indicated by back ground shading (peak of light gray equals full moon), mean daily water temperatures (red), and peak daily flows (blue) at Red Bluff Diversion Dam.

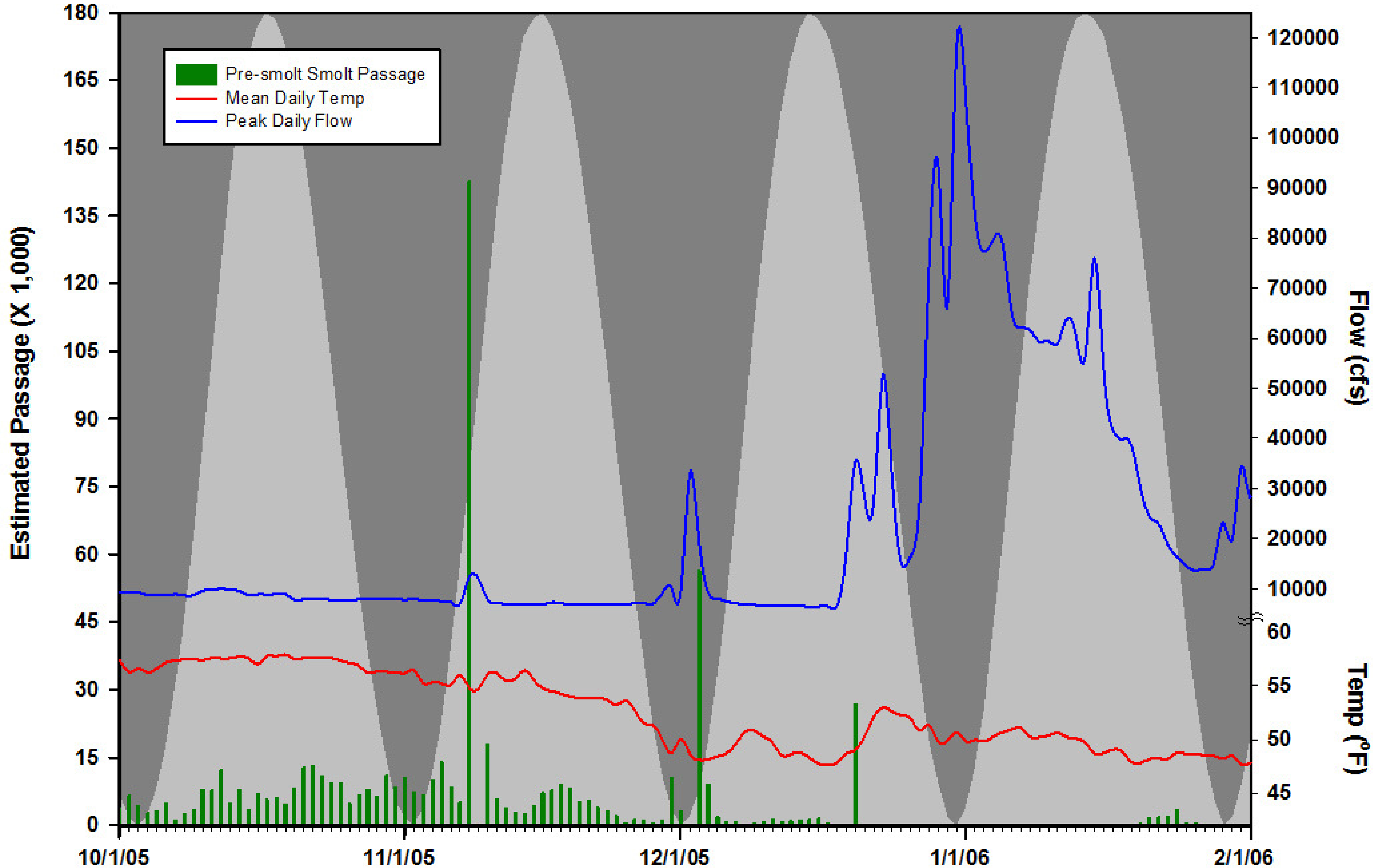

Figure A15. Brood Year 2005 winter Chinook pre-smolt/smolt passage with moon illuminosity indicated by back ground shading (peak of light gray equals full moon), mean daily water temperatures (red), and peak daily flows (blue) at Red Bluff Diversion Dam.

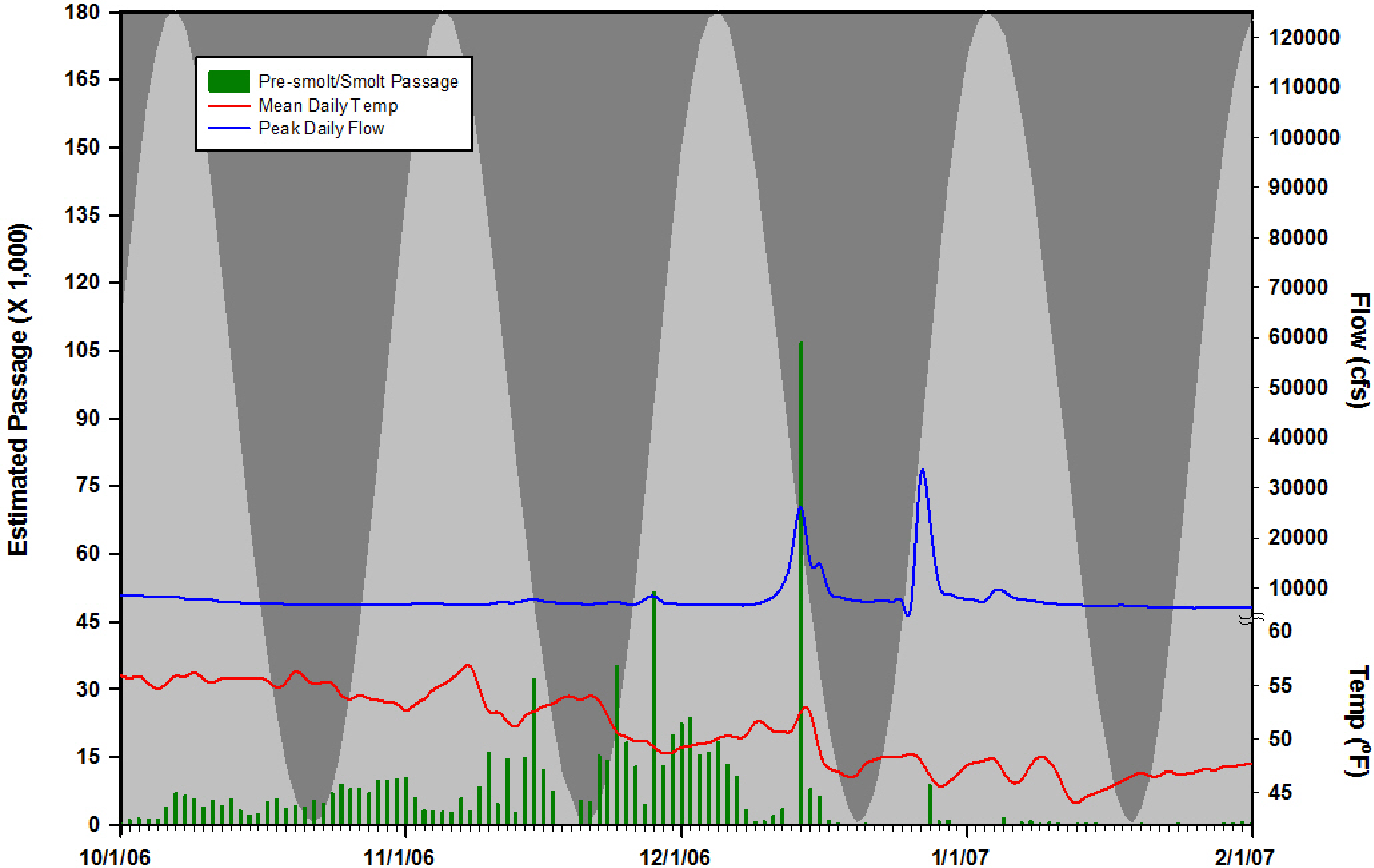

Figure A16. Brood Year 2006 winter Chinook pre-smolt/smolt passage with moon illuminosity indicated by back ground shading (peak of light gray equals full moon), mean daily water temperatures (red), and peak daily flows (blue) at Red Bluff Diversion Dam.

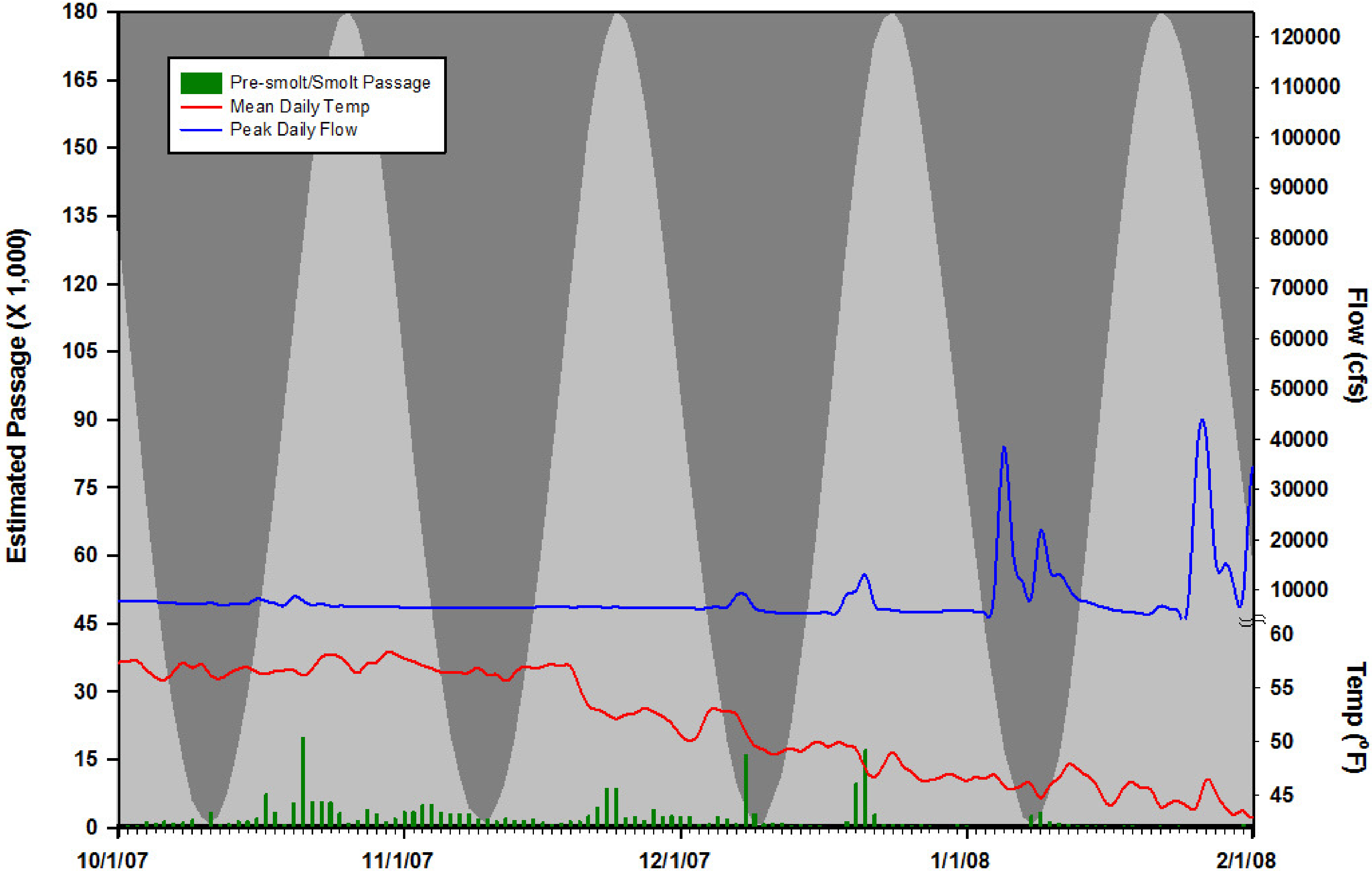

Figure A17. Brood Year 2007 winter Chinook pre-smolt/smolt passage with moon illuminosity indicated by back ground shading (peak of light gray equals full moon), mean daily water temperatures (red), and peak daily flows (blue) at Red Bluff Diversion Dam.

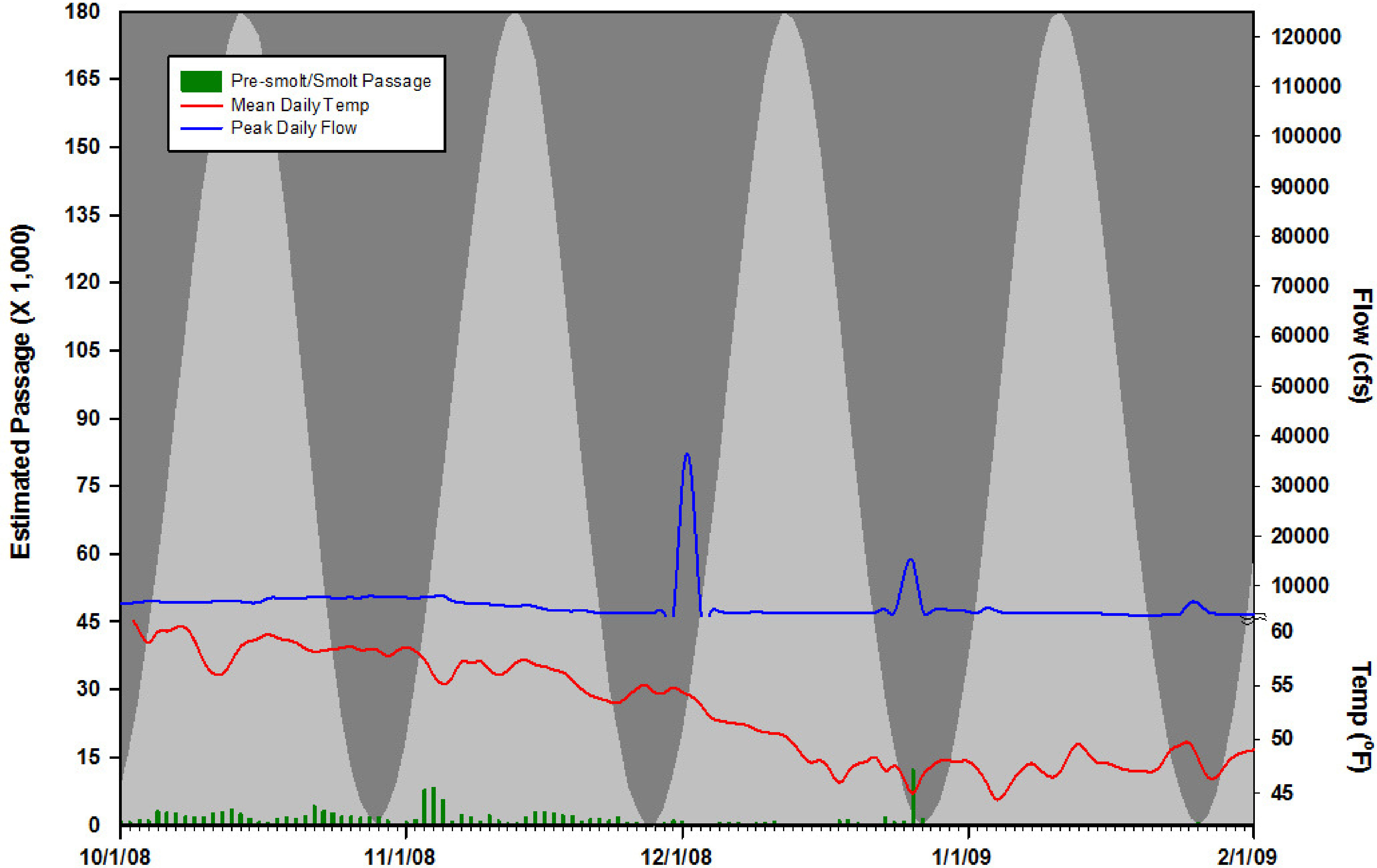

Figure A18. Brood Year 2008 winter Chinook pre-smolt/smolt passage with moon illuminosity indicated by back ground shading (peak of light gray equals full moon), mean daily water temperatures (red), and peak daily flows (blue) at Red Bluff Diversion Dam.

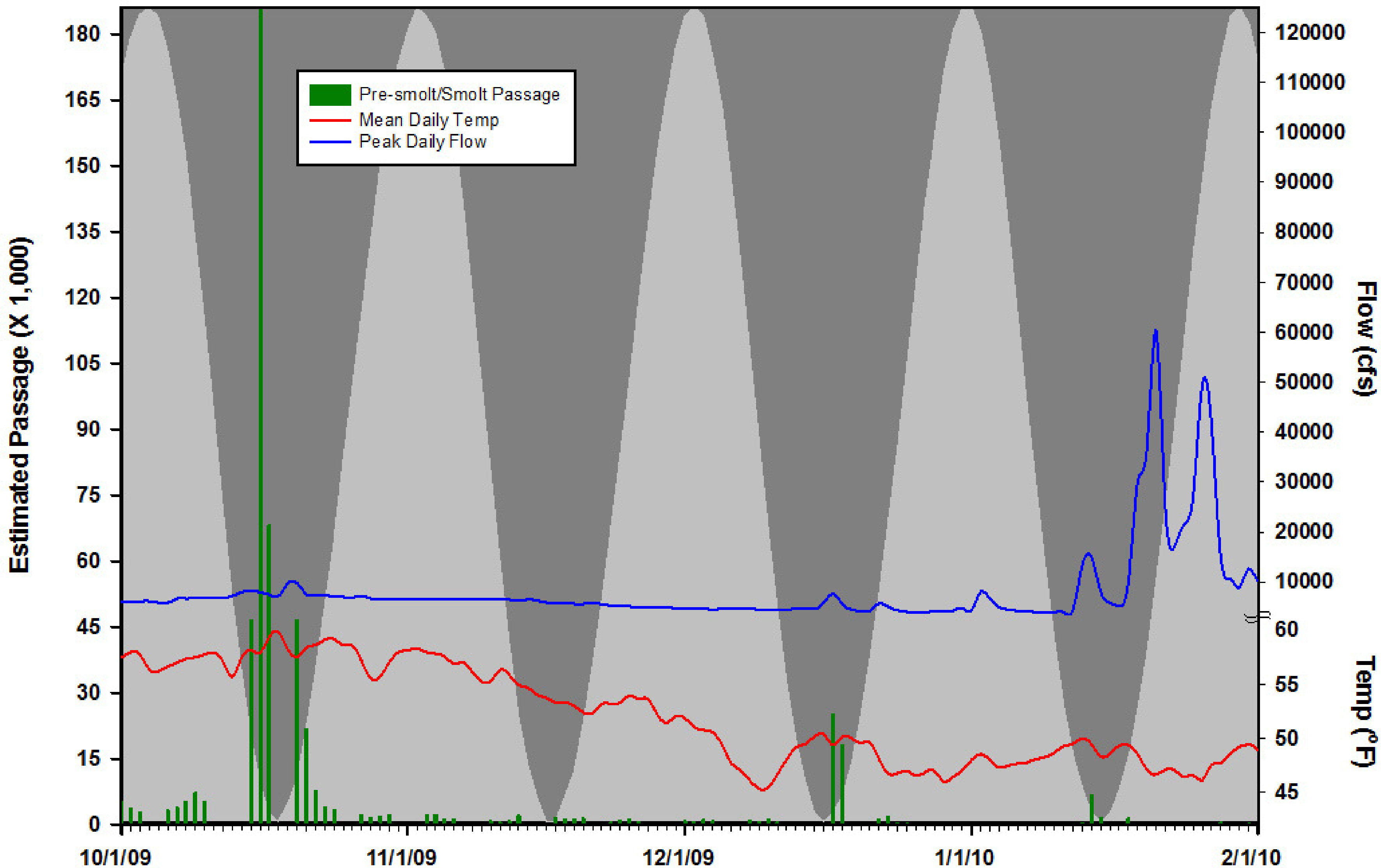

Figure A19. Brood Year 2009 winter Chinook pre-smolt/smolt passage with moon illuminosity indicated by back ground shading (peak of light gray equals full moon), mean daily water temperatures (red), and peak daily flows (blue) at Red Bluff Diversion Dam.

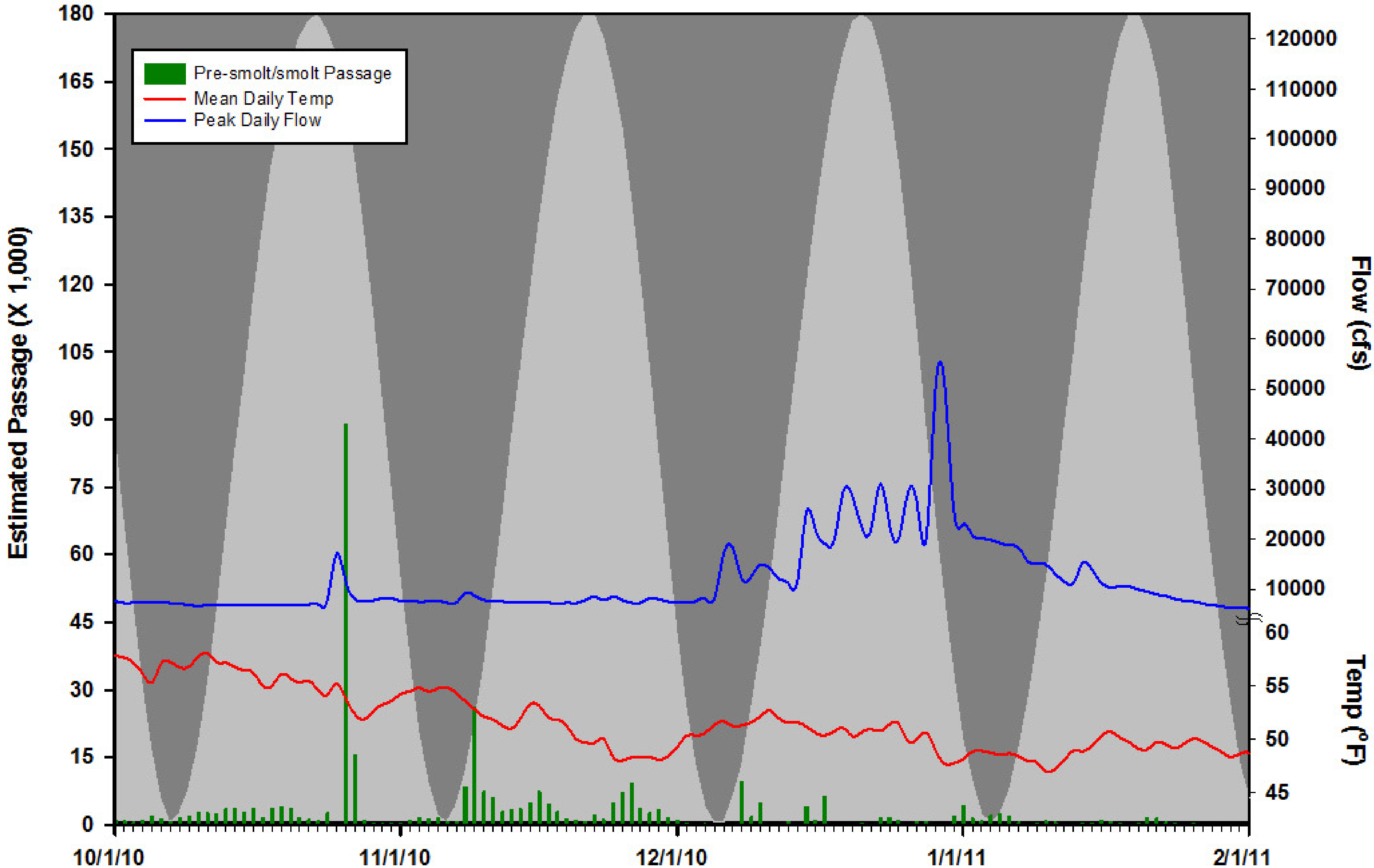

Figure A20. Brood Year 2010 winter Chinook pre-smolt/smolt passage with moon illuminosity indicated by back ground shading (peak of light gray equals full moon), mean daily water temperatures (red), and peak daily flows (blue) at Red Bluff Diversion Dam.

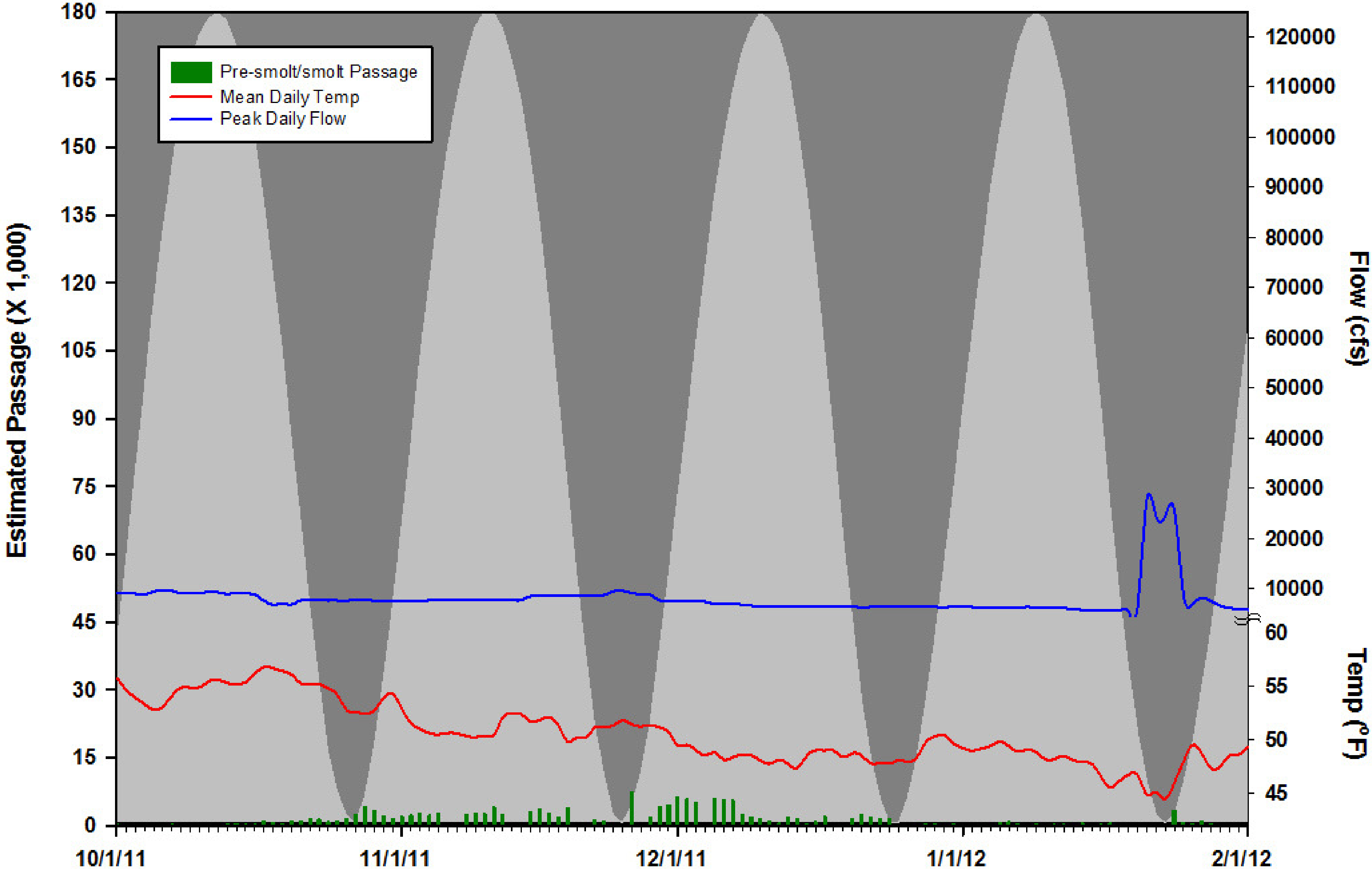

Figure A21. Brood Year 2011 winter Chinook pre-smolt/smolt passage with moon illuminosity indicated by back ground shading (peak of light gray equals full moon), mean daily water temperatures (red), and peak daily flows (blue) at Red Bluff Diversion Dam.

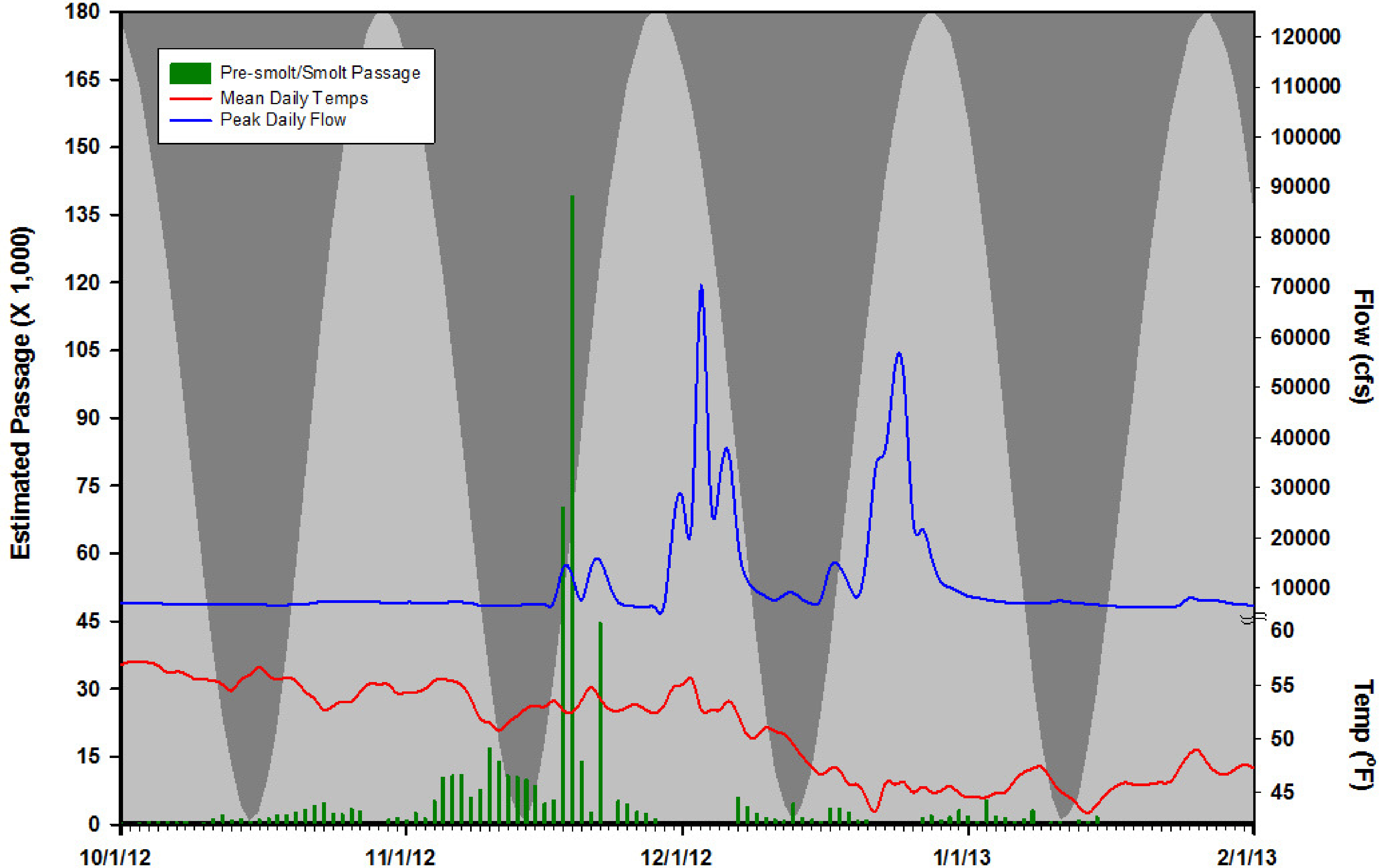

Figure A22. Brood Year 2012 winter Chinook pre-smolt/smolt passage with moon illuminosity indicated by back ground shading (peak of light gray equals full moon), mean daily water temperatures (red), and peak daily flows (blue) at Red Bluff Diversion Dam.