Technical Memorandum

Testing the Model: Spawner Abundance

Historical Observations Compared to Simulated Runs

Introduction

This is the first in a series of Technical Memorandums that explains how we applied the Winter-run Chinook Integrated Modeling Framework (Winter-run IMF) to historical data as a test of model accuracy. Our best opportunity, at present, to determine if the Winter-run IMF reasonably simulates population trends is to run the model with historical conditions and see how well the simulated population tracks with observed populations since counts began at Red Bluff Diversion Dam (RBDD). We refer to this exercise as a hindcast simulation. The only difference between a hindcast and a normal simulation with this model is that input values for a hindcast simulation are the actual values observed each year, while input values for a normal simulation are fixed at one value across all years (this is called a deterministic simulation). Input variables that must be supplied each year for a hindcast simulation are the following:

- Percent of Spawning Below RBDD
- Percent Egg Mortality above RBDD
- Freeport Flow (Jan.)
- Export/Inflow Ratio (Dec.- Apr.)
- DCC Gate Position (Dec. – Apr.)
- Ocean survival (smolt-to-age 2)
- Harvest Fraction in Freshwater
- Harvest Fraction in Ocean

We will distribute a series of Technical Memorandums to the Steering Committee and Winter-run Project Work Team that describe how we obtained inputs used in these simulations. This first memorandum explains how we determined the number of winter run spawners for each year of record. These spawners are not the simulated values, but rather the observed values to which our hindcast simulations will be compared. Once the historical estimates of spawning escapement were gathered, they were used to: 1) "seed" the model, and 2) compare to annual spawning escapements simulated by the model. Seeding the model entails using the first four years of observed spawning escapement as the parents that begin a life cycle, and then allowing the model to simulate the number of their offspring that survive to become parents for the next generation. By the fifth year of simulation, the spawners are generated from the offspring of the first 2 years for which spawners were input. For example, the first year of spawners used to seed the model was 1968, and the model simulated the number of offspring that survived to be age-3 spawners in 1971 and age-4 spawners in 1972. Because winter-run spawners are rarely older than age 4, all spawners in the hindcast simulation after 1972 were generated by the simulation that started with only the first four years of observed spawners.

Observed Spawner Escapements

We used observed returns of winter-run Chinook to RBDD as the historical benchmark values to which simulations could be compared. Spawning escapement of winter Chinook in the upper Sacramento has been estimated since 1967 via counts at the fish ladder at RBDD; however, counts since 1987 have only covered a portion of the run. Beginning in 1987, the gates at RBDD have been raised for various durations in late-winter and early spring to assist upstream passage of adult winter-run Chinook. This meant that not all fish passed through the ladder, and thus not all fish were counted. Methods were derived to expand counts at RBDD, based on an expectation of average run timing. The accuracy of those methods was poor, due in part to variation in run timing between years. This extrapolation to assumed run timing can lead to large errors in estimation (43% to 230%; NMFS 1997). Alternative estimates of spawner abundance became possible in 1996, when surveys of spawned-out carcasses were initiated, and continued in subsequent years. Spawning escapement estimates via carcass surveys have been deemed more reliable (USFWS 2001).

Three models are used to estimate population size from the carcass survey data, the Peterson, the Jolly-Seber, and Schaefer models. USFWS recommends using the Peterson estimate for tracking long term trends in the population abundance. The JollySeber model is recommended for use when the most accurate single year estimate is required. The Jolly-Seber method has more rigorous data requirements which may not be met in all years that carcass surveys are conducted (USFWS 2001). Since 1996 when carcass surveys began, the requirements of the Jolly-Seber model could only be met in 2000-2002.

Thus, for the historical run sizes to be compared to IMF simulations, we used RBDD counts from 1968-1995, and carcass-survey estimates of spawners from 1996

2002 (Figure 1). We used the carcass estimate based on the Peterson model during 1996-1999, and based on the Jolly-Seber model for 2000-2002 estimates. These sets of estimates are those recommended by fisheries agencies for use when estimating annual escapement. Estimates by all methods through 2001 are displayed in Table 1.

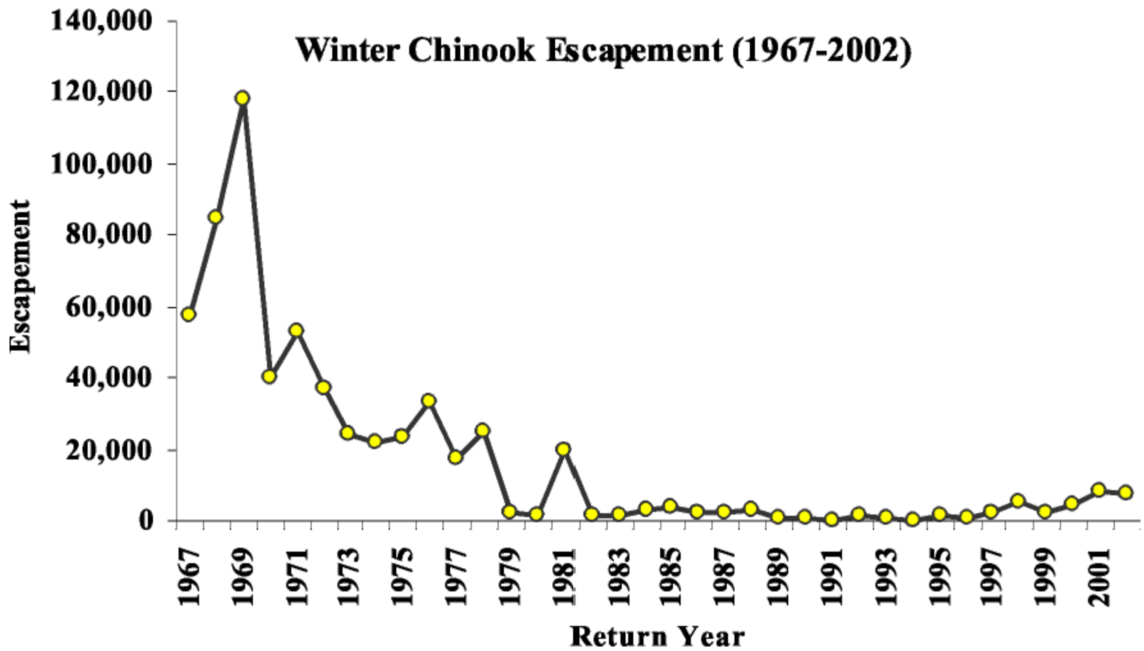

Figure 1. Escapement of winter Chinook in the upper Sacramento River. Data from 1967-1995 is from RBDD ladder counts. Data from 1995-2002 is from spawning ground carcass survey estimates.

Table 1. Estimates of winter Chinook spawner escapement in the upper Sacramento River. Bolded values are those presented in Figure 1 and used as the benchmark for comparison to simulation output. Carcass surveys began in 1996, and Jolly-Seber estimates have only been feasible since 2000.

	RBDD Counts			Carcass Survey: Peterson			Carcass Survey: Jolly-Seber		
Year	Grilse	Adults	Total	Grilse	Adults	Total	Grilse	Adults	Total
1967	24,985	32,321	57,306						
1968	10,299	74,115	84,414						
1969	8,953	108,855	117,808						
1970	8,324	32,085	40,409						
1971	20,864	32,225	53,089						
1972	8,541	28,592	37,133						
1973	4,623	19,456	24,079						
1974	3,788	18,109	21,897						
1975	7,498	15,932	23,430						
1976	8,634	26,462	33,096						
1977	2,186	15,028	17,214						
1978	1,193	23,669	24,862						
1979	113	2,251	2,364						
1980	1,072	84	1,156						
1981	1,744	18,297	20,041						
1982	270	972	1,242						
1983	392	1,439	1,831						
1984	1,869	794	2,663						
1985	329	3,633	3,962						
1986	496	2,101	2,597						
1987	277	1,909	2,186						
1988	1,008	1,878	2,886						
1989	125	571	696						
1990	43	387	430						
1991	19	192	211						
1992	80	1,160	1,240						
1993	137	250	387						
1994	124	62	186						
1995	29	1,268	1,297						
1996	629	708	1,337	156	664	820			
1997	352	528	880	165	1,888	2,053			
1998	924	2,079	3,002	110	5,391	5,501			
1999	2,466	822	3,288	441	1,821	2,262			
2000	789	563	1,352	178	6,492	6,670	116	4,227	4,343
2001	3,827	1,696	5,523	1,216	11,581	12,797	760	7,236	7,996
2002	--	--	--	--	--	--	--	--	7,337

To begin the hindcast simulation, we seeded the model with spawner escapements from 1968-1971. We did not start with the 1967 run because some data needed as inputs to the model were not available for 1967.

RBDD Effects on Passage & Spawning Success

Although we have reasonable estimates for the number of adult winter-run passing RBDD each year, this does not account for all the spawners in the river, and not all spawners were successful. Some fish spawned below RBDD and some of the eggs deposited were killed in some years by exposure to lethal temperatures. In order for the hindcast simulation to predict the number of fish passing RBDD, it must account for the number of returns that spawned below RBDD, and the proportion of eggs that died from high temperatures. The issues of fish spawning below RBDD and eggs dying from exposure to high temperature have largely been resolved by actions in the Recovery Plan, so we have not included cause-effect relationships in the baseline IMF to predict these effects. Therefore, to account for these historical issues in our hindcast simulations, we used best available data to assign values for these effects in each year.

When simulating the historical return of spawners from the ocean, we included a factor to deduct the percentage of the run that was blocked from passing RBDD. Impaired passage at RBDD was a major factor for the decline of winter Chinook populations. Studies in the 1980's found that the RBDD fish ladders were ineffective in attracting adult salmon to migrate past RBDD (Hallock et al. 1982, Vogel and Smith 1986, USFWS 1987, Vogel et al. 1988). Several radio tagging studies estimated that RBDD blocked 43-44% of winter run Chinook that approached the dam. Authors concluded that fish blocked by the dam had difficulty locating the entrance to the fish ladder. Adults obstructed by the dam are forced to spawn below the dam where high temperatures often lead to 100% mortality (NMFS 1997). Although some eggs deposited below RBDD survived in some years, we have assumed in the hindcast that no offspring survive from spawning below RBDD.

Beginning in 1986, gates of RBDD have been raised for varying periods during winter Chinook passage to reduce the percentage of the run blocked by the dam. Initially, gates were opened periodically between mid-December and April. Since then, the "open window" has been expanded to begin in mid-September and end in mid-May (Figure 2). USBR plans continued operation of gates open from mid-September through mid-May in the future (USBR 2003). Run timing data from 1970-1988 indicates that on average, 85% of the winter run Chinook has passed RBDD by mid-May.

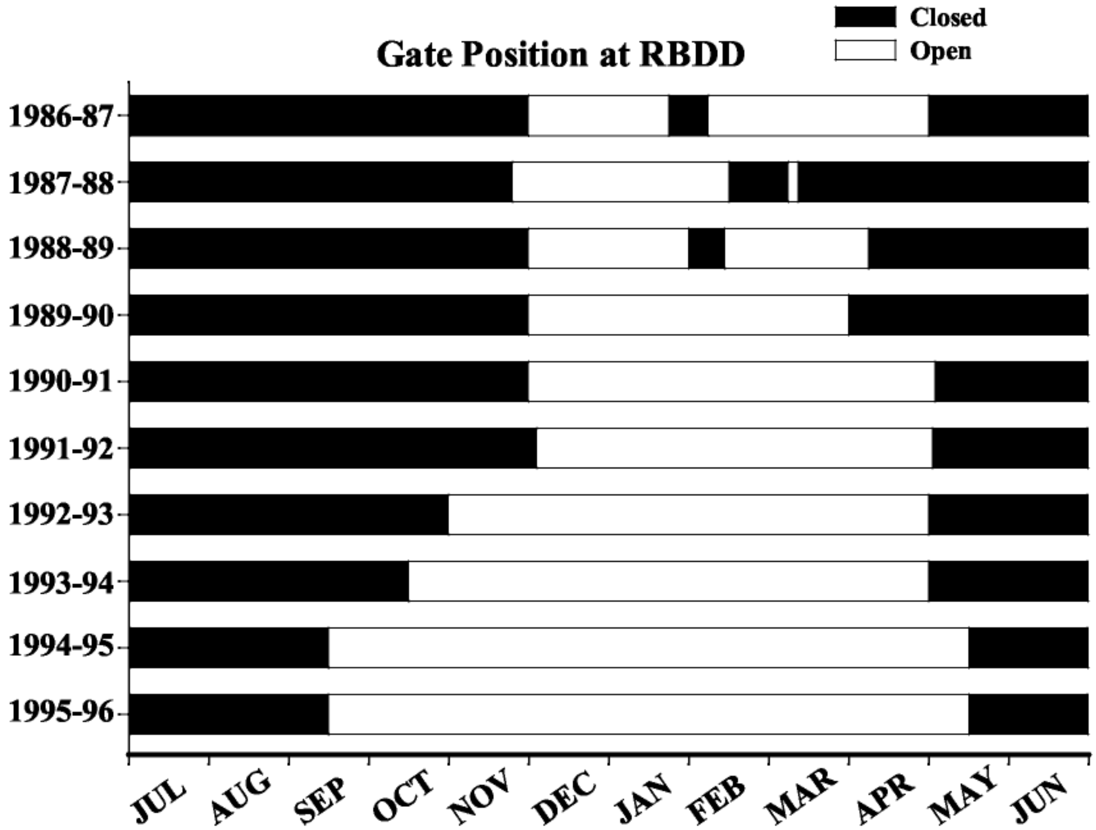

Figure 2. Gate position at RBDD from the 1986-87 to 1995-96 migration years. Figure re-created from NMFS (1997).

Aerial counts of winter run redds were initiated in 1987 by CDFG, and those counts show that more fish spawned above RBDD in years that RBDD gates remained open later in the migration season. In 1987, the gates were closed on April 5 when 60% of the run had passed RBDD. Aerial redd counts showed that 4.4% of winter Chinook redds were located below RBDD in that year. In 1988, the gates were closed earlier on March 12, when only 24% of the run had passed RBDD. That year, 25.7% of redds were located below RBDD. The longer the gates were open, and the earlier the run timing, the higher the proportion of redds above RBDD was. Subsequent data showed that extending the period with the gates open decreased the percentage of redds below RBDD.

Since the RBDD gates were first lifted in the 1987 return year, we assumed that 40% of the run was blocked from passing RBDD prior to that year. This assumption was based on the radio-telemetry survey data cited earlier because the distribution of redds were not surveyed prior to 1987. In each year since 1987, we assumed that the percentage of the run blocked by RBDD was equal to the percentage of all winter Chinook redds that were constructed below the dam (NMFS 1997; Table 2).

Temperature Induced Egg Mortality

Water temperatures in the upper Sacramento River where winter Chinook spawn have reached levels lethal to Chinook eggs in some years, and the hindcast simulation had to account fro these egg losses. Newly spawned and incubating winter-run Chinook eggs and fry are the most sensitive life stages to elevated water temperatures (NMFS 1997). Maximum survival of incubating eggs and pre-emergent fry occurs at water temperatures between 40ºF and 56ºF, and egg mortality increases rapidly at 57.5ºF. At sustained 62ºF, egg mortality is 100% (Seymour 1956, Combs and Burrows 1957, and Hinze 1959; as cited in Boles 1988).

Limited data on river temperatures has hindered the estimation of egg mortality for winter-run Chinook prior to 1989, though losses due to warm temperatures certainly occurred. In 1989, egg mortality was estimated at 4-8%, in 1990 estimates were 2030%, and 5-10% in 1991 (NMFS 1997). Modifications of CVP operations reduced mortalities after 1992 to an estimated 4.2%, and to near zero from 1993-1996 (NMFS 1997). However, modified operations to improved incubation temperatures required that water be released from Shasta Reservoir without power generation. This operation soon proved to be costly, and an alternative method to releasing cool water from the reservoir was sought.

In 1997 a temperature control device was installed in Shasta Dam that allowed operators to release cold water from the bottom of the reservoir while still generating power. The device allows managers the ability to alter the temperature of releases to a certain degree, and thus influence winter Chinook egg incubation temperatures. The device has proven effective at reducing downstream temperatures, though modeling by the USBR indicates that in dry and critically dry water years that incubation temperatures will reach lethal levels for eggs.

For the hindcast model, it was necessary to estimate egg mortalities before operations of Shasta Reservoir improved incubation conditions. Limited temperature data indicated that there were several years where high temperatures resulted in excessive egg mortality. Using estimates of egg mortality from 1989-1996 (NMFS 1997) we were able to correlate estimated egg mortality to average August maximum temperature at Balls Ferry (r^2 = 0.92). The relationship showed that as average August maximum temperature exceeds 58.3°F, estimated egg mortality begins to increase (Figure 3).

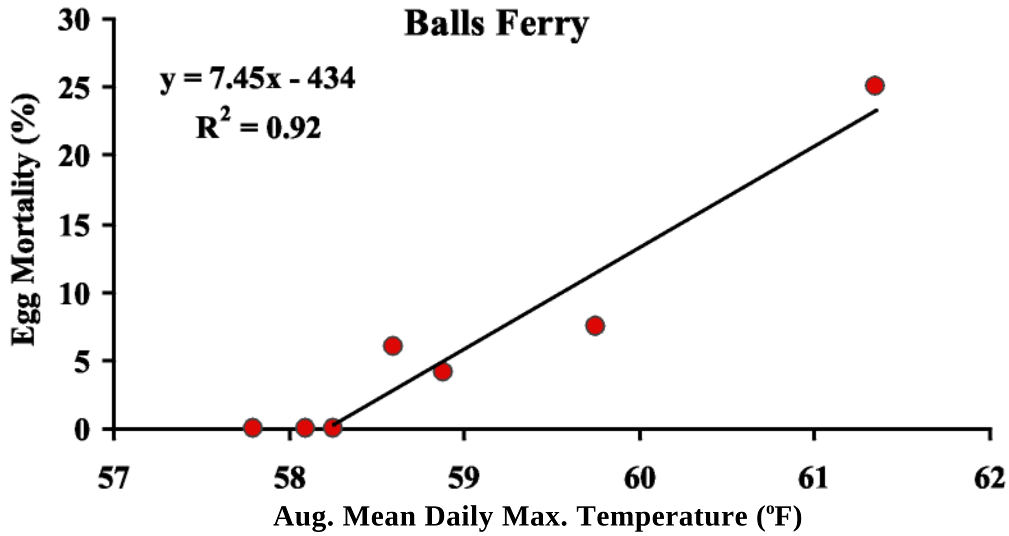

Figure 3. Relationship between August mean maximum daily temperature at Balls Ferry and egg mortality estimates. Egg mortality estimate are from 1989-1994, and 1995-1996 (NMFS 1997). Regression line applies to points of 58.3°F and greater (upper 5 points). Mortality below that point is 0%.

Sporadic temperature data from Keswick Dam, Balls Ferry, Jellys Ferry, Bend Bridge and RBDD from 1970-1989 showed that there were several years where egg mortality would have been significant. Egg mortality would have been greatest in the summers of 1976 and 1977 when the averages of daily maximum temperatures during August at Balls Ferry were 63.5°F and 69°F, respectively. In 1977, daily temperatures were consistently above 60°F for all of July. Even as far upstream as Keswick Dam, August monthly maximum temperatures were 60.8°F in 1976 and 67.3°F in 1977 (Turek 1990). Temperatures from both locations in each year were well above the threshold where mortality begins to occur. Though the temperatures at Balls Ferry were outside the bounds incorporated into the regression, the regression estimated that mortality in 1976 and 1977 were 39% and 80% respectively. Given the extreme temperatures and duration that they were encountered, we believe that these estimates of mortality are conservative.

Other years prior to 1989 where temperatures indicated that significant egg mortality may have occurred included 1985, 1987, and 1988. Temperature data at Balls Ferry for 1987 and 1988 were applied to the regression presented above to estimate mortality in those two years at 8% and 16% respectively. No temperature data from Balls Ferry were available for 1985. Examination of August temperature data at Bend Bridge indicated that water temperatures in 1985 were warmer than those in 1987, and cooler than in 1988. Based on these data we assumed that egg mortality in 1985 was intermediate to those years at 12%.

Annual egg mortality rates used in the hindcast simulation are presented in Figure 4 and Table 2.

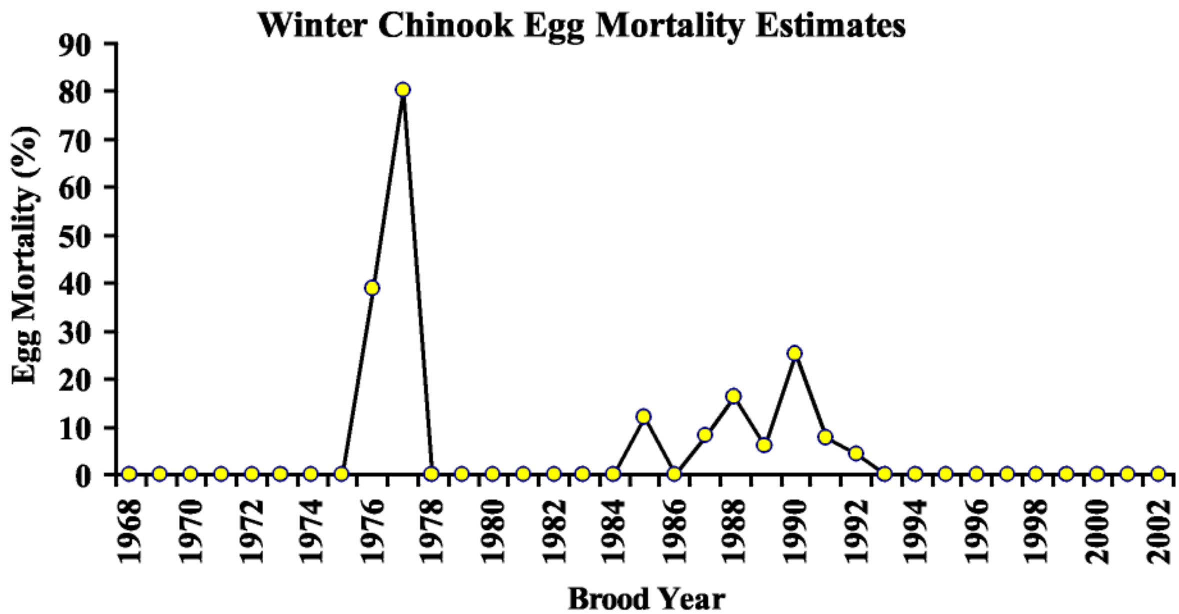

Figure 4. Estimates of egg mortality incorporated into the hindcast model.

Table 2. Annual values used in the hindcast simulation for the % of spawning run blocked by RBDD and the percentage of egg mortality incurred by winter-run Chinook spawning above RBDD. All eggs spawned below RBDD were assumed to die.

Year	% Blocked by	Egg
1968	40.0	0.0
1969	40.0	0.0
1970	40.0	0.0
1971	40.0	0.0
1972	40.0	0.0
1973	40.0	0.0
1974	40.0	0.0
1975	40.0	0.0
1976	40.0	39.0
1977	40.0	80.0
1978	40.0	0.0
1979	40.0	0.0
1980	40.0	0.0
1981	40.0	0.0
1982	40.0	0.0
1983	40.0	0.0
1984	40.0	0.0
1985	40.0	12.0
1986	40.0	0.0
1987	4.4	8.0
1988	25.7	16.0
1989	2.2	6.0
1990	7.2	25.0
1991	0.0	7.5
1992	3.7	4.1
1993	0.8	0.0
1994	0.0	0.0
1995	0.5	0.0
1996	0.0	0.0
1997	0.0	0.0
1998	0.0	0.0
1999	0.0	0.0
2000	0.0	0.0
2001	0.4	0.0
2002	1.0	0.0

References:

Boles, G.L. 1988. Water temperature effects on chinook salmon *(Oncorhynchus tshawytscha)* with emphasis on the Sacramento River: a literature review. Department of Water Resources, Northern District, 42 pp.

Combs, B.D. and R.E. Burrows. 1957. Threshold temperatures for the normal development of chinook salmon eggs. Prog. Fish. Cult. 19:3-6.

Hallock, R.J., D.A. Vogel and R.R. Reisenbichler. 1982. The effect of the Red Bluff Diversion Dam on the migration of adult Chinook salmon, *Oncorhynchus tshawytscha,* as indicated by radio-tagged fish. California Department of Fish and Game, Anadromous Fisheries Branch Report No. 82-8.

Hinze, J.A. 1959. Annual report: Nimbus salmon and steelhead hatchery, fiscal year of 1957-1958. Cal. Dept. Fish and Game, Inland Fisheries Division, Admin. Rept. 59-4.

National Marine Fisheries Service. 1997. NMFS Proposed Recovery Plan for the Sacramento River Winter-run Chinook Salmon. NMFS Southwest Region, Long Beach, CA.

Seymour, A.H. 1956. Effects of temperature upon young chinook salmon. Ph.D. Thesis. University of Washington, Seattle.

Turek, S. M. 1990. Historic Sacramento River temperature data, Keswick to Butte City. Memorandum Report, California Dept. Water Resources, Northern District, Redding.

USBR. 2003. Long-term Central Valley Project OCAP BA CVP-OCAP. USBR MidPacific Region, Sacramento, CA.

USFWS. 1987. Water quality and quantity needs for Chinook salmon production in the upper Sacramento River. Prepared by the U.S. Fish and Wildlife Service and California Department of Fish and Game for the 1987 Hearing Process on the San Francisco Bay/Sacramento-San Joaquin Delta Estuary. USFWS Exhibit No. 29.

USFWS. 2001. Estimating the Abundance of Sacramento River Juvenile Winter Chinook Salmon with Comparisons to Adult Escapement. U.S. Fish and Wildlife Service. Red Bluff Fish and Wildlife Office. Red Bluff, CA.

Vogel, D.A. and J.G. Smith. 1986. Fish passage action program for the Red Bluff Diversion Dam, Interim report on fishery investigations. U.S. Fish and Wildlife Service Report No. FR1-FAO-86-5. 52 p. plus appendices.

Vogel, D.A., K.R. Marine, and J.G. Smith. 1988. Fish Passage Action Program for Red Bluff Diversion Dam. Final Report, U.S. Fish and Wildlife Service Report No. FR1-FAO-88-19. 77 p. plus appendices.