DRAFT

Recommendations of flow and Water Temperature for Chinook Salmon and Steelhead in Clear Creek, California

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Revised Draft

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EXECUTIVE SUMMARY

The 35-mile long Clear Creek, a tributary to the Sacramento River, is located near Redding, California. Clear Creek, historically supported spring-run and fall-run Chinook salmon (*Oncorhynchus tshawytscha*) and steelhead trout (*O. mykiss*), has been severely degraded over the past 150 years, particularly since 1912 when McCormack-Saeltzer Dam was built (the dam was removed in 2000) and 1963 when Whiskeytown Dam was built. These dams not only blocked fish from accessing the upper Clear Creek, but also reduced flow. Operations of Whiskeytown Dam essentially eliminated flow conditions that flush silt from gravel, deposit new spawning gravel, facilitate timely juvenile out-migration, attract adult salmon into the stream, and prevent riparian vegetation encroachment. The operations also caused summer low flows in the creek that resulted in harmful temperatures for spring-run Chinook spawning adult and their eggs, and possibly for immigrating steelhead adult and rearing juveniles. Both spring-run Chinook and steelhead are listed as threatened species under the Endangered Species Act. Spring-run Chinook were extirpated from Clear Creek for 30 years until they were reintroduced in the early 1990s.

Protecting and recovering threatened spring-run Chinook and steelhead from extinction in Clear Creek requires the restoration of populations to sustainable levels. To help achieve the goal, it is necessary to take the following actions: 1) Operating Whiskeytown Reservoir to provide flows mimicking natural flow patterns, which the species have evolved from and adapted to; 2) Providing adequate water temperatures for various life stages of the species; 3) Providing suitable and sufficient physical habitat for spawning and rearing; and 4) Monitoring responses from the actions and making adjustments accordingly.

This report discusses fundamental principles that underpin these proposed actions and describes comprehensive approaches for developing flow regime, managing water temperature, and enhancing physical habitat for all salmonid species in Clear Creek. Although the focus of the report is on Clear Creek, the principles and approaches described in this report are applicable to other watersheds.

Stream flow and temperature are critical factors in the protection and recovery of endangered or threatened anadromous fish species. These factors are interrelated as temperature is often controlled by streamflow, particularly in rivers regulated by reservoirs. Streamflow in Clear Creek has changed considerably as a result of flow regulation by Whiskeytown Dam. The pre- and post-dam streamflow data (from water year 1941 to 2009) in Clear Creek were analyzed to assess flow magnitude, frequency, timing, duration, and change of rates using the Indicator of Hydrological Alteration method. The 1-day median maximum flows decreased from 5000 cfs for the pre-dam period (1941-1960) to 1500 cfs for the post-dam period (1963-2009), and the 7-day median maximum flows decreased from 2800 cfs for the pre-dam period to 700 cfs for the post-dam periods. There were 21 small floods (about 7000 cfs) for the pre-dam period, occurring once a year on average, whereas only 6 small floods for the post-dam period, occurring once every 8 years on average.

The flow regime approach was used to develop flows required for anadromous fish. The approach recognizes that biologically relevant flows can't be defined by flow magnitude alone. The frequency, timing, duration, and rate of change of flows are also biologically important. However, this does not mean to restore the pre-dam hydrology of Clear Creek; rather it is to mimic the pattern of the pre-dam hydrology. The important features of a flow pattern include baseflow and functional flow. The baseflow occurs after a rainfall event or snowmelt period has passed and associated surface runoff from the catchment has subsided. The seasonally varying baseflow in a river imposes a fundamental constraint on river's aquatic communities because it determines the amount of aquatic habitat available for most of the year. Functional flows are required for maintaining or improving important ecosystem functions, for example, migration cues, habitat connectivity and diversity, and stream channel morphology and geometry. Combining baseflows with functional flows provides seasonal and interannual variability, to which anadromous fish species have evolved and adapted.

The flow developed from the flow regime approach is incomplete if water temperature requirements for listed fish species are not considered. Water temperature influences growth and feeding rates, metabolism, development of embryos and alevins, timing of life history events, and the availability of food. For protecting and recovering vulnerable populations, such as endangered or threatened fish species, optimal water temperatures are needed for each of their life stages. These optimal temperatures serve as the base of setting water temperature criteria for the listed species. Flows for sustaining optimal water temperatures are particularly important in warm seasons when flow is low and air temperature is high. Water temperature data in Clear Creek were analyzed and compared with temperature criteria. Statistical methods were used to develop flow-temperature models, which were used to estimate the flow required to meet the established temperature requirements in Clear Creek.

The recommended environmental flow is the integration of the baseflow, the water temperature sustaining flow, and the functional flow. Combination of the baseflow with the temperature sustaining flow is referred to as instream flow. The recommended instream flows in Clear Creek from November through May were based on the 50th, 75th, and 90th percentile baseflows, while the instream flows from June through October were based on the 75th, 90th, and 95th percentiles meeting water temperature requirements.

Functional flows include channel flushing flows and channel maintenance flows. The flushing flow in Clear Creek is recommended to be about 700 cfs with a frequency of 2-3 times a year from January to May. The duration of the flows would be 3-5 days with a rise rate of 200 cfs/day and a fall rate of -100 cfs/day. This flow magnitude would allow the removal and transport of fine sediments from riverbed. The channel maintenance flow in Clear Creek is recommended to be about 6000-7000 cfs with a frequency of once every two or three years between January and March. The duration of the flow may span 10 days with a rise rate of 1700 cfs/day, and a fall rate of -1000 cfs/day.

Potential water costs were evaluated when the recommended environmental flows were implemented in Clear Creek. The annual additional water use was about 23,000 acre feet for the 50th and 90th percentiles and 28,000 acre feet for the 75th percentile. If the additional water allocated to Clear Creek goes directly to powerplants, it could generate, on average, hydropower of 1.54 to 1.85 MW (about 1% of the plant capacity of 180 MW) at Springs Creek Powerplant and 0.22 to 0.26 MW (about 0.2% of the plant capacity of 117 MW) at Keswisk Powerplant.

The effectiveness of the recommended flow releases from Whiskeytown Reservoir in achieving targeted benefits can only be realized through implementing the actions and consistent monitoring. Implementation of actions and monitoring programs should occur in parallel. A monitoring plan should include the collection and management of data for reservoir release, streamflow, water temperature, stream morphology, weather conditions, and fish biology. The information gained from monitoring are used to better refine flow recommendations for the following season and inform recommendations for other management actions.

1 Introduction

Alterations to the natural hydrologic systems in the California's Central Valley include the construction of a number of dams or diversion structures. The upstream dams were generally constructed with no ladders for fish passage, and anadromous fish have been blocked from accessing the upper reaches of streams. As the upper stream reaches, serving as adequate spawning and rearing habitat, are not available, the populations of salmonids including spring-, fall-, and winter-run Chinook salmon and steelhead trout have declined dramatically in the area.

Clear Creek is a tributary to the Sacramento River originating in the Trinity Mountains between the Trinity River and the Sacramento River watersheds. The 35-mile long creek historically supported spring-run and fall-run Chinook salmon and steelhead trout. Clear Creek has been severely degraded over the past 140 years of gold mining, gravel extraction, and dam construction, which adversely changed the stream channel, fish habitat, and riparian conditions. In 1912, McCormack-Saeltzer Dam was built on the creek to supply water for gold mining and, later, agriculture. This dam made it impossible for Chinook salmon and steelhead to migrate upstream for spawning (the dam was removed in 2000).

In 1963, Whiskeytown Dam was built on Clear Creek to provide hydroelectric power, flood control, and municipal, industrial and agricultural water supply. Whiskeytown Dam not only blocked fish from accessing the upper Clear Creek, but also dramatically reduced streamflow downstream of Whiskeytown Reservoir. Operations of Whiskeytown Dam essentially eliminated flow conditions that flush silt from gravel, deposit new spawning gravel, facilitate timely juvenile out-migration, attract adult salmon into the stream, and prevent riparian vegetation encroachment. The operations also caused summer low flows in the creek that resulted in harmful temperatures for spring-run Chinook spawning adult and their eggs, and possibly for immigrating steelhead adult and rearing juveniles. Both spring-run Chinook and steelhead are listed as threatened species under the Endangered Species Act. In fact, spring-run Chinook were extirpated from Clear Creek until they were reintroduced in the early 1990s.

Since 1992, the U.S. Bureau of Reclamation (USBR) and the U.S. Fish and Wildlife Service (FWS) have worked cooperatively to implement the Central Valley Project Improvement Act (CVPIA) in Clear Creek. Implementation of the CVPIA included increased streamflow, removal of McCormick-Saeltzer Dam, gravel placement, erosion control, and channel restoration in the lower reaches. Although these actions have helped increase the abundance of spring-run Chinook salmon in Clear Creek, they are still at an abundance level that makes the population vulnerable to extirpation from demographic stochasticity. As such, the population would fall into the high risk of extinction category (NMFS 2009a).

Protecting and recovering threatened spring-run Chinook and steelhead from extinction in Clear Creek requires the restoration of populations to sustainable levels (*e.g.*, double populations). To help achieve the doubling goal, the following actions should be taken:

- Operating Whiskeytown Reservoir to provide adequate flows, mimicking natural flow patterns, which the species have evolved from and adapted to,
- Providing adequate water temperatures for various life stages of the species,

- Providing suitable and sufficient physical habitat for spawning and rearing, and
- Monitoring responses from the actions and making adjustments accordingly.

This report discusses fundamental principles to support for taking these actions and describes comprehensive approaches for developing flow regime, managing water temperature, and enhancing physical habitat for all three salmonid species in Clear Creek. The report also assesses the feasibility of taking the actions by analyzing water cost and infrastructure constraints. It is important to recognize that mimicking nature flow patterns does not mean restoring historical flows. The principles of the approaches described in this report are applicable to any other watersheds although the focus of the report is on Clear Creek.

2 Clear Creek and Whiskeytown Reservoir

2.1 Clear Creek

2.1.1 Geography

Clear Creek situates in the northwestern portion of the upper Sacramento River basin in California. Clear Creek originates near 6,000 ft elevation in the Trinity Mountains, and flows south between the Trinity River basin to the west and the Sacramento River basin to the east, and into Whiskeytown Reservoir (elevation 1,210 ft) at Oak Bottom. It flows approximately 35 miles, with a drainage area of 238 square miles, until it meets the Sacramento River (**Figure 1**).

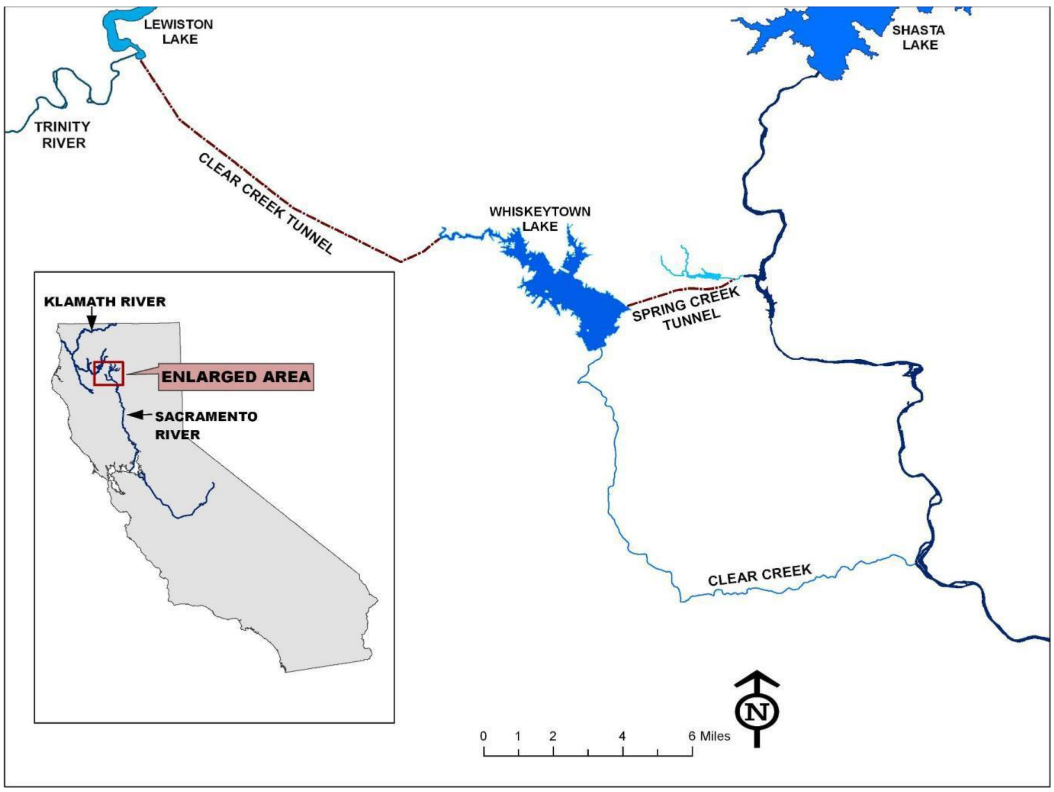


Figure 1. Map showing Clear Creek, Whiskeytown Reservoir, and vicinity (After Matt Brown 2010)

Average annual precipitation in the Clear Creek watershed varies from 20 inches near the confluence with the Sacramento River to more than 60 inches in the upper watershed. Most precipitation falls into this watershed as rainfall. Most of the watershed's rainfall occurs between November and April, with little or none occurring during the summer months. The ambient air temperatures range from approximately 32°F (0ºC) in winter to summer highs in excess of 115°F (46ºC).

The maximum watershed elevation is approximately 6,000 ft, but a majority of the watershed area is below the 4,000 ft snow line, so high flow hydrology is driven by rainfall and rain-on-snow events, which typically occur during the winter months. The unimpaired snowmelt hydrograph is small in magnitude; the snowmelt peak is typically less than 1,500 cfs. Unimpaired summer/fall baseflows were low because the imperviousness of the Klamath Mountains terrain minimizes shallow and deeper groundwater storage to the point where no significant springs exist to maintain high baseflows. This imperviousness, combined with periodic high intensity rainstorms, results in extremely flashy streamflow response to rainfall events.

The lower portion of Clear Creek, starting at Whiskeytown Dam at river mile 18.4, flows south before changing direction and flowing east approximately 8.5 miles upstream of the Sacramento River confluence. The drainage area between Whiskeytown Dam and the confluence with the Sacramento River is 49 square miles (**Figure 1**). “Clear Creek” used in this report refers to the lower portion of the creek downstream of Whiskeytown Dam.

Separated at the Clear Creek Road Bridge, the upper and lower reaches of the creek are geologically distinct (Giovannetti and Brown 2008). The upper reach flows south from Whiskeytown Reservoir almost 10 miles. The stream bedrock in the upper reach is composed primarily of Paleozoic to Mesozoic igneous, metasedimentary, and metamorphic rocks that are largely resistant to erosion. The stream is more constrained by canyon walls and a bedrock channel and has a higher gradient, less spawning gravels, and greater pool depths than the lower portion of Clear Creek.

The lower reach flows in an easterly direction to the Sacramento River for approximately 8.4 miles. The stream bedrock in the lower reach is composed of sedimentary rocks that are much less resistant to erosion. The stream meanders through a less constrained alluvial flood plain, and has a lower gradient, more spawning gravels, and fewer deep pools (Giovannetti and Brown 2008).

2.1.2 Changes of Fish Habitat in Clear Creek

In 1848, gold was first discovered in Shasta County. Through the 1940s, placer, hydraulic, and dredge mining significantly altered the lower Clear Creek watershed. The dredging process significantly disturbed floodplains and terraces and removed all riparian and upland vegetation along the corridor as prospectors searched for gold. Piles of dredger tailings were left along the corridor and are still seen today, confining Clear Creek in some locations, and, in other locations, discouraging the natural recovery of riparian and upland plant species.

Through the mid-1980s, commercial instream gravel mining in the lower reaches of Clear Creek removed most of the gravel (several hundred thousand cubic yards) within a 1.8-mile reach. An additional 1-mile reach was significantly altered by disposal of dredger tailings. Impacts to channel morphology and salmonid habitat were significant. The bankfull channel was destroyed and floodplains removed, leaving wide, shallow channels and interspersed deep pits. Excessive gravel removal exposed a clay hardpan over much of the channel bottom, directly removing salmonid spawning and fry-rearing habitat. Equally important was the lost channel confinement, allowing both adult and juvenile salmonids to stray into adjacent pits and be stranded.

Historically, there were about 347,000 ft^2 of spawning habitat from Whiskeytown Dam to [former] Saeltzer Dam in Clear Creek as surveyed in 1956 (pre-Whiskeytown Dam). It was decreased to about 29,000 ft^2 of habitat in 1971, indicating a 91% reduction. In 2000, the same reaches were revisited and 98% of the gravels that were present in 1956 were gone. Previously classified spawning habitat was replaced by stretches of unproductive coarse sand deposits, due to the reduced gravel carrying capacity of the stream and accelerated erosion and sediment delivery by tributaries (*i.e.*, Paige Boulder Creek and South Fork Clear Creek). Sediment supply from Paige Boulder Creek is primarily decomposed granite as coarse sand, which accounted for 30% in 1971, comparing with 50% in 1997 and 1998 (McBain *et al.* 2001).

2.2 Whiskeytown Reservoir

The reservoir, about 8 miles west of Redding, has a capacity of 241,100 acre feet with a surface area of 3,458 acres. Construction of the earth-fill dam - 263 feet tall - began in 1959 and was completed in 1963. President John F. Kennedy dedicated Whiskeytown Dam on September 28, 1963 before a crowd of over 10,000 people. He spoke briefly of the development of the American West and the significance of Whiskeytown Dam and its relationship to the Central Valley Project. The reservoir is owned and operated by USBR. Its purpose is to provide flood control, irrigation, electricity generation, and fish and wildlife benefits (pursuant to CVPIA). There are also recreational activities available including camping, swimming, boating, water skiing, and fishing.

A majority of the reservoir water comes from Lewiston Reservoir supplied by the Trinity River downstream of Trinity Lake. Before entering Whiskeytown Reservoir through the Clear Creek Tunnel, the water generates hydroelectricity at the 184-MW Judge Francis Carr Powerplant. A large portion of the Reservoir water leaves through the Spring Creek Tunnel, which delivers the water to the 180-MW Spring Creek Powerplant, whose tailrace empties into Keswick Reservoir. The 117-MW Keswick Powerplant at Keswick Dam empties into the Sacramento River (USBR 2004). There is no powerplant on Whiskeytown Dam. The outlet at the bottom discharges water to Clear Creek at rates up to 1200 cfs and also supplies the City of Redding drinking water.

Whiskeytown Dam entirely blocked fish from accessing the upper stream and dramatically altered hydrology downstream of Whiskeytown Reservoir. Furthermore, the reservoir, acting as a sediment (gravel) trap, has starved the lower Clear Creek of its gravel. Combined with years of gravel and gold mining below the dam and channel scouring by high flows, sediment starvation has limited the amount of gravel available to spawning salmonids for building redds. In some areas of the Clear Creek channel only clay hardpan or bedrock remains.

In addition, the 15-foot-tall and 200-foot-long McCormack-Saeltzer Dam was built in 1912 approximately 10 miles downstream of the Whiskeytown Dam to supply water for gold mining and, later, agriculture. Despite efforts to provide access to 10 miles of upstream spawning habitat using fish ladders and fish tunnels, the dam remained essentially impassable to salmonids. To restore fish access to the reach between Saeltzer and Whiskeytown Dams, Saeltzer Dam was removed in 2000 (Ferry and Miller 2003).

3 Anadromous Salmonids in Clear Creek

Clear Creek supported populations of fall-run (including late fall-run) and spring-run Chinook salmon and steelhead trout. Life history adaptations and different spatial distributions allowed these runs to utilize the entire watershed to the fullest extent possible. The cumulative effects of gold mining, dams, gravel extraction, timber harvest, land development, and roads in the Clear Creek watershed have led to loss of suitable habitat and degradation of stream channel and riparian conditions and contributed to the continuous decline of the Clear Creek salmonids. Spring-run Chinook and steelhead have been impacted the most from flow alteration and habitat loss.

3.1 Central Valley Spring-run Chinook

Central Valley Spring-run Chinook salmon were listed as a threatened species on September 16, 1999 and the threatened status was reaffirmed on June 28, 2005. The ESU includes all naturally spawned populations of spring-run Chinook salmon in the Sacramento River and its tributaries in California, including the Feather River, as well as the Feather River Hatchery spring-run Chinook program.

It was estimated that the Central Valley had supported spring-run Chinook salmon as large as 600,000 fish between the late 1880s and 1940s (CDFG 1998). The median population of spring-run from 1986 to 2007 was 10,652, with two thirds from independent populations (NMFS 2009a). Of the 19 independent populations of spring-run that occurred historically, only three remains in Deer, Mill, and Butte Creeks, respectively. The populations in other tributaries (*e.g.*, Battle, Antelope, Big Chico Creeks, and Feather River) and the mainstem Sacramento River below Keswick Dam are considered dependent populations, which rely on the three independent populations for continued existence at this time (NMFS 2009a).

In 1998, the population estimate on Butte Creek was between 18,742 and 20,259 salmon; in 1999 it was between 3,529 and 3,679 (CDFG 2000a). In 1998 at Deer Creek, the population was estimated at 1,879 adult spring-run salmon; in 1999 there were an estimated 1,591 spring-run salmon. Mill Creek had a population estimate of 424 salmon in 1998 and 560 in 1999. Big Chico Creek had a population estimate of 369 salmon in 1998 and 27 in 1999.

After 30 years of extirpation of spring-run Chinook salmon from Clear Creek, 35 spring-run adults reappeared in 1999 in the lower reaches of the creek. The adult counts showed an increasing trend from 1999 to 2008 (Giovannetti and Brown 2009) but decreased after 2008 and plunged to a low level in 2010 (**Figure 2**). Both redd and carcass counts appeared increasing from 2003 to 2008. The juvenile population for spring-run Chinook, ranging from 110,000 to 150,000 (Earley *et al.* 2010), showed a pattern similar to the redd counts (**Figure 2**). The population of spring-run Chinook in Clear Creek is considered at high risk of extinction (NMFS 2009a; NMFS 2009b).

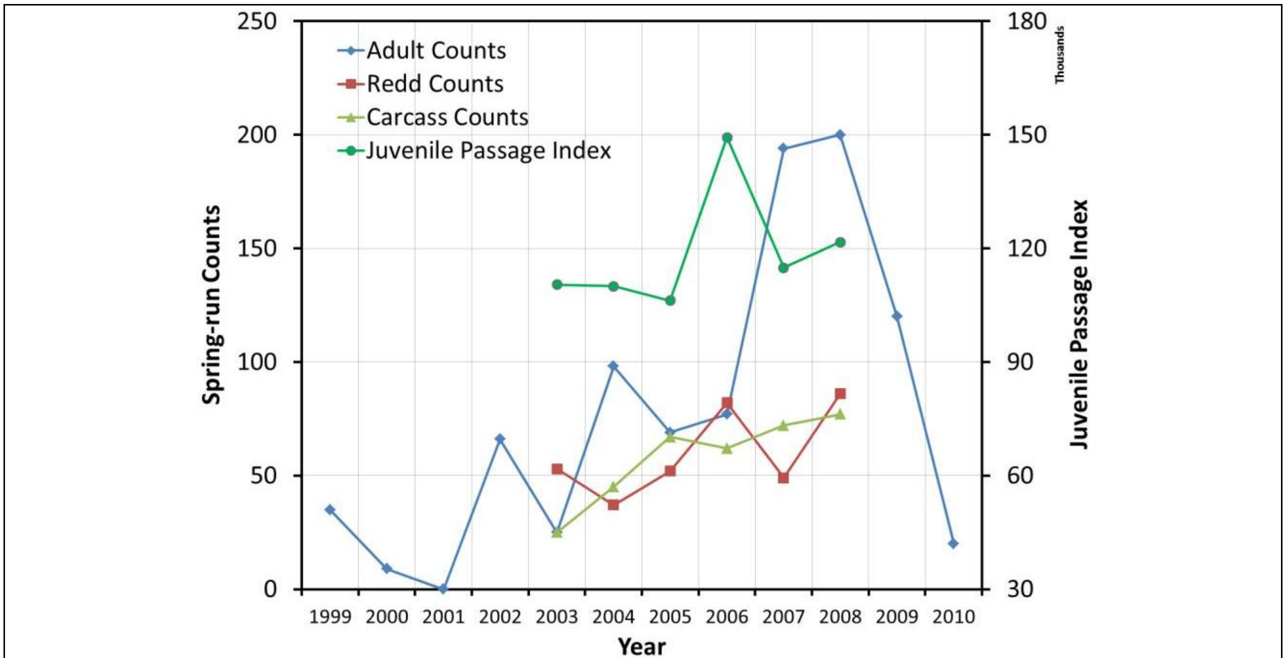


Figure 2. Estimated yearly number of spring-run Chinook adults, carcasses, redds, and juveniles in Clear Creek

Spring-run Chinook salmon exhibits a stream-type life history. Adults enter freshwater in the spring, hold over the summer, spawn in the fall, and juveniles typically spend a year or more in freshwater before emigrating (**Figure 3**). Adult spring-run salmon leaves the ocean to begin their upstream migration in late January and early February and enters the Sacramento River between March and September, primarily in June and July. They utilize mid- to high-elevation streams that provide appropriate temperature, sufficient flow, cover, and pool depth to allow for over-summer holding (from May through September) while conserving energy and allowing their gonadal tissue to mature.

Spring-run salmon spawning occurs between September and October depending on water temperature. Fries emerge on the gravel bed from November to March. The fries seek areas of shallow water and low velocities while finishing absorbing the yolk sac and transition to exogenous feeding. Many also will disperse downstream during high-flow events. Spring-run juveniles may rear over a year in natal streams or the lower reaches of non-natal tributaries in the Sacramento River watershed. As they grow larger, juveniles tend to use microhabitats with heavy cover and deeper water.

The outmigration period for spring-run juveniles extends from November to April. It was observed that up to 70 percent of the young-of-year (YOY) fish emigrated through the lower Sacramento River and Delta during this period. The peak movement of the juveniles at Knights Landing in the Sacramento River occurs in December for yearlings, and in March and April for YOY (NMFS, 2009; (NMFS 2009b).

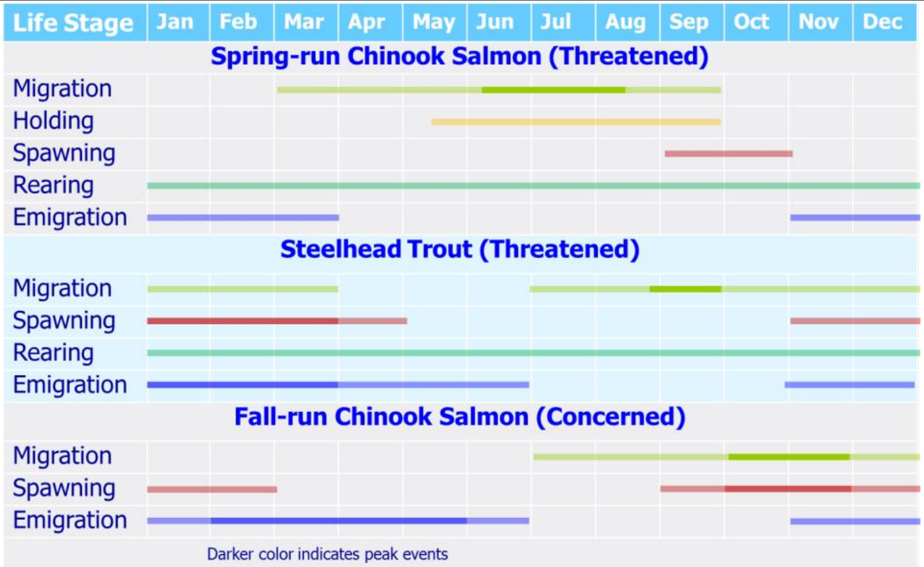


Figure 3. Life history of spring-run and fall-run Chinook and steelhead in Clear Creek

3.2 California Central Valley Steelhead

California Central Valley steelhead were listed as a threatened species on March 19, 1998; threatened status reaffirmed on January 5, 2006. This Distinct Population Segment (DPS) includes all naturally spawned anadromous steelhead populations below natural and manmade impassable barriers in the Sacramento and San Joaquin Rivers and their tributaries, excluding steelhead from San Francisco and San Pablo Bays and their tributaries, as well as two artificial propagation programs: the Coleman National Fish Hatchery and the Feather River Fish Hatchery.

The population numbers returning to the Red Bluff Diversion Dam fish ladders had decreased substantially since 1966. In the late 1960's, roughly 20,000 fish passed through the fish ladders; in 1994, only 2,000 returned. These statistics include hatchery fish from the Coleman National Fish Hatchery. The average estimated spawning population size above the mouth of the Feather River in the Sacramento River system was 20,540 fish in the 1950's. In 1991-1992, the annual run size for the total Sacramento River system was likely less than 10,000 adult fish (Butte County Association of Governments (BCAG) 2011a).

In Clear Creek, steelhead redd counts showed an increasing trend from 2003 to 2010 (Giovannetti and Brown 2010) (**Figure 4**) although there were no adult counts available. Only a few carcasses were identified from 2003 to 2009. The juvenile population seemed increasing from 1999 to 2009 (Earley *et al.* 2010), following a similar pattern to the redd counts.

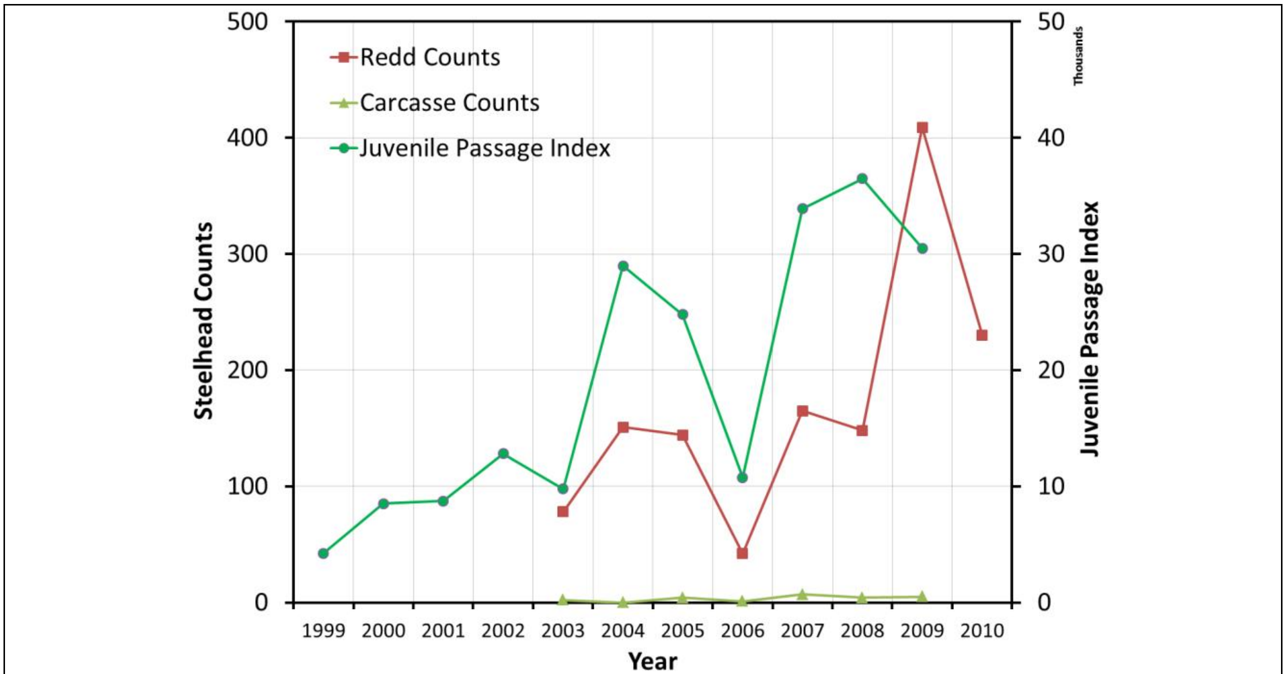


Figure 4. Estimated yearly number of steelhead redds, carcasses, and juveniles in Clear Creek

Adult steelhead enter freshwater in fall and winter (September to December) and spawn in winter (January to March) (Figure 3). After eggs hatch, fry emerge from the gravel in late May to early June. Juvenile steelhead often move to deeper water as they grow and will remain in freshwater for an average of 2 years before migrating to the ocean, which usually occurs from November to April.

The peak migration into the upper Sacramento River above the mouth of the Feather River from 1953 to 1959 was in late September. Adult counts at Clough Dam on Mill Creek for a 10-year period beginning in 1953 indicated that the peak of adult migration into that stream occurred in late October, with a smaller peak about mid-February. Examination of adult steelhead counts at Red Bluff Diversion Dam indicates that run timing on the upper Sacramento River does not appear to have changed appreciably: adult counts from 1969 to 1982 also show this same pattern, as do counts from 1983 to 1986 (McEwan 2001).

Juvenile steelhead migrated downstream during most months of the year, but the peak period of emigration occurred in spring, with a much smaller peak in fall. The emigration period for naturally spawned steelhead juveniles migrating past Knights Landing on the lower Sacramento River in 1998 ranged from late December through early May, and peaked in mid-March. Most naturally-produced Central Valley steelhead rear in freshwater for two years before emigrating to the ocean. Scale analysis indicated that 70% had spent two years in freshwater before emigrating to the ocean, 29% had spent one year, and 1% had spent three years (McEwan 2001).

Photoperiod, stream flow, and temperature appear to influence emigration timing. Juvenile steelhead may spend several weeks in the coastal lagoon or estuary of a stream before entering the ocean. They reside in the ocean for 2 to 3 years before returning to their natal stream to spawn, although in wet years steelhead may return to spawn after only one year in the ocean.

The adults can spawn more than once, although most do not spawn more than twice (McEwan 2001).

3.3 Central Valley Fall-run Chinook

Central Valley fall-run Chinook were classified as a Species of Concern on April 15, 2004 due to specific risk factors. The ESU includes all naturally spawned populations of fall-run Chinook salmon in the Sacramento and San Joaquin River Basins and their tributaries, east of Carquinez Strait, California.

Fall-run Chinook salmon in the Sacramento River have been artificially propagated in hatcheries and released into the rivers and bays since 1872. In the last 50 years, 1.6 billion fall-run fish have been released from hatcheries into Central Valley waterways. State hatcheries on the American and Feather rivers now transport young fish to salt water to avoid mortality in the Delta, but it is thought that this increases straying of adults when they return to spawn (Butte County Association of Governments (BCAG) 2011b).

Fall-run Chinook salmon are the most abundant run in the Central Valley. From 1981 to 2010, in-river (non-hatchery) adult escapement averaged 240,971 per year. Escapement peaked in 2002 (766,668 individuals) and declined to historical lows in 2009 (30,426 individuals). In 2010, in-river adult escapement was estimated at 111,455 fish (Butte County Association of Governments (BCAG) 2011b).

In Clear Creek, fall-run Chinook abundance has fluctuated widely since 1951, from an estimated 10,000 adults in 1963 to fewer than 100 fish in 1978, but has generally been the most abundant run in Clear Creek. Three of the latest five years have exceeded the fall-run Chinook salmon escapement target of 7,100 adults set by the Anadromous Fish Restoration Program (McBain *et al.* 2001). Fall-run redd counts increased from 2000 to 2005 (Gard 2009) while juvenile populations showed no appreciable changes from 1998 to 2008 except for 2000 when the population peaked (about 15 millions) (Earley *et al.* 2010) (**Figure 5**).

Fall-run Chinook enter the Sacramento and San Joaquin Rivers from July through December and spawn from September through February. They generally utilized mainstem habitats for spawning and rearing during fall through spring. Juveniles emigrate as early as in November predominantly as fry and sub-yearlings and remain off the California coast during their ocean migration.

Primarily fall-run Chinook salmon in the Sacramento River have been artificially propagated in hatcheries and released since 1872. In the last 50 years, 1.6 billion fall-run fish have been released in the Central Valley. State hatcheries on the American and Feather rivers now transport young fish to salt water to avoid mortality in the Delta. This increases straying of adults when they return to spawn.

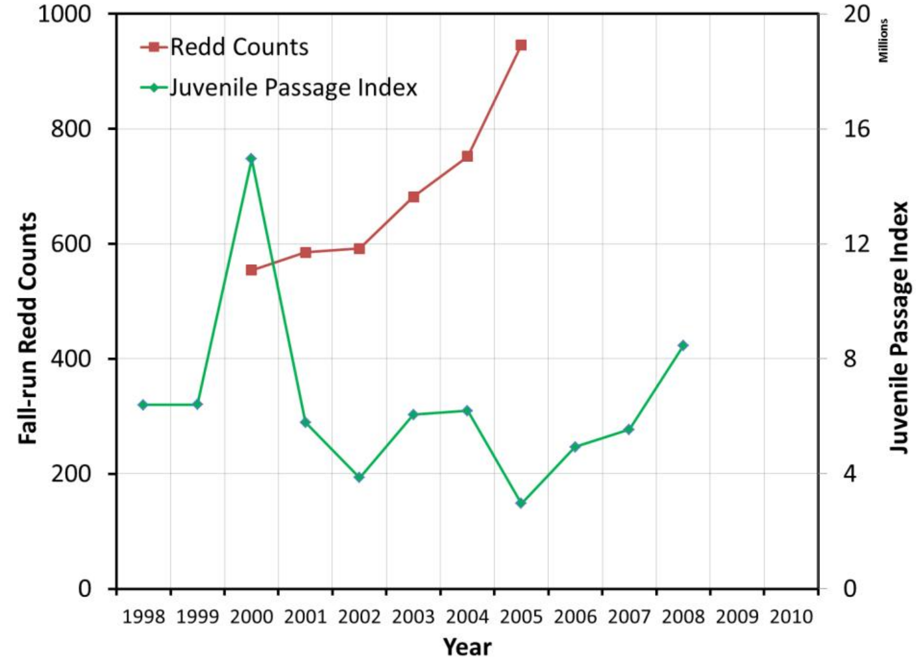


Figure 5. Estimated yearly number of fall-run redds and juveniles in Clear Creek

4 Streamflow for Salmonids

4.1 Hydrologic Alteration and Flow Regime

From the beginning of the 20th century, the construction of dams in the United States were prompted by promises of cheap electrical power and of transforming land into agriculturally fit regions through easy irrigation. There are estimated 75,000 dams, of which are 8,100 major dams[1] in the US in 2005. These dams blocked about 600,000 miles of what had once been free flowing rivers, which is about 17% of rivers in the nation.

There are about 1,400 federal and state jurisdictional dams in California. Approximately 200 of these fall under federal jurisdiction. Ten reservoirs have storage capacity greater than 1 million acre-feet, while 910 reservoirs have capacity less than 1 thousand acre-feet. These dams are owned and operated by a variety of entities including the USBR, US Army Corps of Engineers, California Department of Water Resources (DWR), cities, counties, individual water and power districts, private water and power providers, and individual landowners.

As river flows are depleted or otherwise altered, ecological degradation results and society loses benefits provided by healthy, functioning ecosystems, such as commercial and subsistence fisheries, water purification, flood storage, recreation and esthetic values (Poff *et al.* 1997). Dams block almost every major river system in the West. These dams and the growing diversion of water from rivers and streams posed a grave threat to aquatic life (Poff and Hart 2002; Poff *et al.* 2007). Many of these dams have destroyed important spawning and rearing habitat for salmonids. In some once productive salmon river systems (such as the Sacramento River), less than 5% of their original habitat is now still available to salmon (http://www.pcffa.org/dams.htm). In the Columbia River Basin, once the most productive salmon river system in the world, less than 70 miles of that once great river still remains free flowing (http://www.pcffa.org/dams.htm).

Dam construction and operation changed natural streamflow regimes, which adversely affected water temperature, chemistry, sediment transport, floodplain vegetation communities, downstream estuaries, deltas, and coastal zones. Dams can heavily modify the volume of water flowing downstream, change the timing, frequency, and duration of high and low flows, and alter the natural rates at which rivers rise and fall during runoff events (Richter and Thomas 2007). As summarized by Bunn and Arthington (Bunn and Arthington 2002), the ecological consequences of hydrologic alteration include the following:

(1) Adversely changing physical habitats (*e.g.*, riffles, pools, and bars in rivers and floodplains) that determine the distribution, abundance, composition, and diversity of aquatic communities;
(2) Leading to recruitment failure and loss of biodiversity of native species as aquatic species have evolved life history strategies (*e.g.*, timing of reproduction) in direct response to natural flow regimes;

[1] The National Inventory of Dams defines a major dam as being 50 feet (15 m) tall with a storage capacity of at least 5,000 acre-feet (6.17 million cubic meters or 1.63 billion U.S. gallons), or of any height with a storage capacity of 25,000 acre-feet or more (30.8 million m^3 or 8.15 billion gallons).

(3) Losing longitudinal and lateral connectivity that can lead to isolation of populations, failed recruitment, and local extinction of aquatic species; and
(4) Facilitating the invasion of exotic and introduced species in river systems.

There is broad acceptance that human water demands must be balanced with the needs of rivers themselves - consider rivers (and other freshwater systems) as legitimate ‘‘users’’ of fresh water. It is widely recognized that the full range of natural flow variation – ranging from baseflows to high-flow pulses and floods – play important ecological roles in a river ecosystem. In fact, environmental flow has been defined as ‘‘the quantity, timing, and quality of water flows required to sustain freshwater and estuarine ecosystems and the human livelihoods and wellbeing that depend on these ecosystems’’ (Poff *et al.* 2010). While the science and practice of environmental flow has been studied during the past half century and much progress has been made in environmental flow protection and management, it has proven quite difficult for scientists to describe the full range of flow variability necessary to support a healthy river ecosystem, which can be practically implemented by water managers. And it still remains a daunting challenge to answer these practical questions: ‘‘How much can we change the flow regime of a river before the aquatic ecosystem begins to show decline?” or “how should we manage the daily flows, floods, and interannual patterns of variability to achieve desired ecological outcomes?’’

4.2 Instream Flow Methods

A variety of methods have been developed for prescribing environmental flows; each has its strengths and weaknesses with varying levels of effort and cost. These methods can be grouped into three categories: standard-setting rules, habitat approach, and integrated approach (Annear *et al.* 2004; Petts 2009; Tharme 2003).

4.2.1 Standard-setting Rules

Following the reservoir and water development era of the mid-twentieth century, resource agencies became concerned over the loss of fisheries in the west. Consequently, several states began issuing rules for protecting existing stream resources from future depletions. Many assessment methods appeared during the 1960's and early 1970's based on hydrologic analysis of the water supply coupled with empirical observations of habitat quality and an understanding of riverine fish ecology. These efforts led to a class of flow techniques meant to help reserve water within the channel for the benefit of fish and other aquatic life.

These standard-setting methods include the Tennant method, the aquatic baseflow standard method, and the flow duration curve method. Hydrologic data were used to establish a flow rate that should be met or exceeded, based upon statistical evaluation of historical flows. Using these methods, flow targets can be developed for the year as a whole, or for individual seasons or months. These methods are designed to protect some portion of the overall flow in a river (*e.g.*, 30% of mean or median annual flow). Application of these methods usually resulted in a single 'minimum' flow value for a stream reach, below which water may not be withdrawn for consumptive use. The minimum flow is almost always less than the optimal habitat condition for native fish species. These 'reservations' of water form the basis for issuing water permits in many states. While useful for their ease of application at minimal cost, these methods have been criticized because they do not adequately reflect the full range of variability in flows that is essential for sustaining river-dependent species and ecosystem processes.

4.2.2 Habitat Approach

This group of methods includes the instream flow incremental method (IFIM) and Meso-HABSIM. The IFIM method was developed by the USFWS in the late 1970s (Bovee *et al.* 1998). The method relies on three principles:

(1) the chosen species exhibits preferences within a range of habitat conditions that it can tolerate;
(2) these ranges can be defined for each species; and
(3) the area of stream providing these conditions can be quantified as a function of discharge and channel structure.

The method involves two distinct stages: data collection through field investigation and computer simulation. Field investigations include selecting stream reaches, establishing transects, and measuring the depth and velocity of the river along each transect at high, medium and low flows. These hydrological and physical data are collected for each life stage of a target species (*e.g.*, holding, spawning, or rearing). In the meantime, biological data of the species also need to be collected at each transect.

The data collected from field investigations are used to establish curves relating weighted usable area (WUA) with flow for each life stage of a fish species and this is done with computer simulation. The core of the simulation is the Physical HABitat SIMulation (PHABSIM) that integrates the changing hydraulic conditions with discharge and the habitat preferences of one selected species. Since the IFIM focuses on the needs of a single species and generally on one critical life stage, it does not address the need of high flows that move sands and gravels downstream and prevent seedlings from germinating and taking hold on the river banks (Barinaga 1996). The IFIM requires quality field data and this is often time consuming, expensive, and impossible at times, to obtain. The output is location specific. Some fish population data must be collected on-site to calibrate effective habitat simulations. Also, this method does not address the flow needed for meeting water temperature requirements. This method is most appropriate for in-depth, site-specific analysis of flow needs , but is not intended for prescribing flow standards (Annear *et al.* 2004).

Recently, a meso-scale physical habitat modeling method – Mesohabitat Simulation Model (Meso-HABSIM) offer advantages over PHABSIM, involving modeling of depth and velocity separately (Parasiewicz 2001; Parasiewicz 2007a; Parasiewicz 2007b; Parasiewicz 2008; Parasiewicz and Walker 2007). MesoHABSIM requires smaller expenditure resources and field efforts (*e.g.*, 10 days to collect data from a 20 km-long reach with MesoHABSIM as opposed to 50 days of using PHABSIM). MesoHABSIM modifies the data acquisition technique and analytical approach of similar models by changing the scale of resolution from micro- to meso-scales. The model takes variations in stream morphology along the river into account and is more applicable to large-scale issues.

4.2.3 Integrated Approach

After extensive research and development of environmental flow during the past 15 years, it is now generally accepted by the scientific community that to protect freshwater biodiversity and maintain the valuable goods and services provided by rivers, it is essential to mimic components of natural flow variability, taking into consideration the magnitude, frequency, timing, duration, and rate of change of flow events (Petts 2009; Poff *et al.* 1997; Richter *et al.* 1996; Richter *et al.* 1997). These five critical components of the flow regime regulate ecological processes in river ecosystems. They can be used to characterize the entire range of flows and specific hydrologic phenomena such as floods and low flows. Furthermore, by defining flow regimes in these terms, the ecological consequences of particular human activities that modify one or more components of the flow regime can be considered and addressed explicitly.

For analysis of the five flow regime components, Richter et al. (1996) introduced the Indicators of Hydrologic Alteration (IHA) method. This method analyzes daily streamflow data using 32 hydrologic parameters such as monthly flows, 1-day annual maximum flows, timing of the 1-day annual maximum flow, and frequency and duration of maximum flows. The latest version of the IHA software includes 34 new flow parameters – the environmental flow components (The Nature Conservancy 2009). The software program examines 66 ecologically relevant statistics such as the timing and maximum flow of each year's largest flood or lowest flows, then calculates mean or median and variance of these values over some period of time. The IHA method can be used to quickly pinpoint aspects of a hydrologic regime that needs to be addressed to restore ecosystem integrity (Annear *et al.* 2004). More than 1,000 water resource managers, hydrologists, ecologists, researchers, and policy makers from around the world have used this method to assess how current flow patterns differ from the system's natural flow regime.

Understanding hydrologic alterations between pre- and post-dam is the first step to restore a degraded aquatic system. Richter et al. (1997) extended the IHA method and proposed the Range of Variability Approach (RVA). The RVA employs pre-dam or natural flows to establish IHA target ranges (*i.e.*, 1 standard deviation from the mean or percentiles from the median) for each of the 32 IHA parameters. This allows initial river management decisions to be made when no or limited long-term ecological or biological data are available (Annear *et al.* 2004). This method has been widely used to incorporate regime based flows in water resources and environmental management in the U.S. and other countries (Petts 2009).

Most recently, Poff et al. (Poff *et al.* 2010) developed a new framework for assessing environmental flow needs for many streams and rivers simultaneously to foster development and implementation of environmental flow standards at the regional scale. This framework, the ecological limits of hydrologic alteration (ELOHA), is a synthesis of a number of existing hydrologic techniques and environmental flow methods that are currently being used to various degrees and that can support comprehensive regional flow management.

4.3 Recent Flow Criteria Development in the Central Valley

Based on the same principles and/or concepts as described in sections 4.1 and 4.2.3, both SWRCB and CDFG developed flow criteria for the Sacramento River inflow, San Joaquin River inflow, and Delta outflow (CDFG 2010a; SWRCB 2010).

The SWRCB's Delta flow criteria were developed using information on unimpaired flows, historical impaired inflows that support more desirable ecological conditions, statistical relationships between flow and native species abundance, and/or ecological functions-based analyses for desirable species and ecosystem attributes (Fleenor *et al.* 2010). In an attempt to more closely reflect the variation of the natural hydrograph (*i.e.*, magnitude, frequency, duration, timing, and rate of change of flows) (Delta Environmental Flows Group 2010), the SWRCB recommended that, when possible, the flow criteria be expressed as a percentage of the unimpaired flow. To develop criteria in this way, the unimpaired flow rate for a specified time period (*e.g.* average monthly flow over a range of months) was plotted on an exceedance probability graph along with the flow recommendations and desired return frequencies. The unimpaired flow rates were also plotted such that the associated water year type can be identified and their percent exceedance estimated. A percentage of unimpaired flow was selected by trial and error so that the desired flow rate and exceedance frequency was achieved (SWRCB 2010).

The SWRCB developed Sacramento River inflow criteria with an attempt to mimic the natural hydrograph to protect emigrating juvenile Chinook salmon. While emigration of some runs may occur outside of this period, peak emigration is generally believed to occur between November through June. To achieve the attributes of a natural hydrograph, the criteria were recommended as a percentage of unimpaired flow on a 14-day average, to be provided generally on a proportional basis from the tributaries to the Sacramento River. The 14-day average is intended to better capture the peaks of actual flows compared to a 30-day average time-step, while still allowing for a time-step at which facilities can be operated. The SWRCB concluded that flow criteria equal to 75% of unimpaired flows from November through June, on average, would provide favorable conditions for juvenile Chinook salmon in at least 50% of years (assuming 25,000 cfs flows at Rio Vista) (SWRCB 2010).

4.4 History of Flow Criteria Development in Clear Creek

Streamflow in Clear Creek changed dramatically after the construction of Whiskeytown Dam began in 1959. Water rights permits issued by the SWRCB specified minimum flows from Whiskeytown Dam. The 1960 Memorandum of Agreement with CDFG established minimum flows in Clear Creek (**Table 1**). The 1963 release schedule from Whiskeytown Dam was proposed by FWS. Although this release schedule was never formalized, USBR has operated the dam according to the FWS schedule since May 1963. Historically, streamflows below Whiskeytown Dam were set 50 cfs from January to October and 100 cfs from November to December during an average water year.

Denton (1986) conducted a flow study in Clear Creek from 1982-1983 using the IFIM method. He found that flow should be 300 cfs from mid-May to mid-October and 200-250 cfs from Mid-October to March 31 (Denton 1986) (**Table 1**). However, these flow recommendations were never implemented in Clear Creek.

In response to Central Valley steelhead being listed as threatened on March 19, 1998 under the Endangered Species Act, minimum summer flows were increased to 150 cfs to provide suitable rearing habitat for steelhead pursuant to the AFRP. After spring-run Chinook salmon was listed as threatened on September 16, 1999, flows were further increased to provide suitable spawning habitat for spring-run in September and October. Thus, Whiskeytown releases were set at 150 cfs on June 1, increased to 200 cfs on September 7, increased to 250 cfs on September 10, and decreased to 200 cfs on October 1 (Newton and Brown 2004). This flow schedule was based largely on the Denton study in1986.

Table 1. Evolution of minimum flows in Clear Creek

Period	Flow (cfs)
1960 MOA minimum flows	
January 1 – February 18 (29)	50
March 1 – May 31	30
June 1 – September 30	0
October 1 – October 15	10
October 15 – October 31	30
November 1 – December 31	100
1963 FWS proposed flows (Normal year – Critical year)	
January 1 - October 31	50 - 30
November 1 - December 31	100 - 70
1986 Denton study for steelhead and fall-run	
January 1 – March 31	200
April 1 – May 15	230
May 16 – October 15	300
October 16 – December 31	250
1999 CVPIA AFRP Implementation	
October 1 – June 30	200
July1- September 30	150

The FWS conducted a multiyear study on flow criteria in Clear Creek from 1995 to 2001. They used PHABSIM to generate water surface elevations with a wide range of simulation flows. The PHABSIM generated water surface elevations were used as input to a 2D hydraulic and habitat model (RIVER2D). The 2D model predicts available habitat at various simulations flows. The predicted available habitat was described with optimum depth, velocity, and substrate. From the RIVER2D output, the optimum flow can be identified, at which maximum habitat is achieved. The results of the flow study are summarized in **Table 2**. Flows greater than 600 cfs in the upper canyon reaches are needed from September through December to increase spring-run habitat availability for spawning. At the flow rate of 200 cfs, only 50% of the habitat in the upper reach, and 30% of the habitat in the lower reach (to Clear Creek Road Bridge) are available for spring-run spawning.

Table 2. Recommended flows for Clear Creek based on the 7-year flow investigation conducted by USFWS from 1995 to 2001

Stream Reach	Life Stage	Spring-run	Steelhead	Fall-run	Publication Year
Upper (Whiskeytown Dam to Clear Creek Road Bridge)	Spawning	650-900	350-600		2007
	Rearing	600-900	650-900		2010
Lower (Clear Creek Road Bridge to the Sacramento River	Spawning		300	300	2011
	Rearing		NA	NA	Unpublished

NA = Not Available

5 Water Temperature for Salmonids

5.1 Water Temperature and Salmonids

Water temperature influences the survival of salmonids at all life stages of the life cycle, including growth and feeding rates, metabolism, development of embryos and alevins, timing of life history events such as upstream migration, spawning, freshwater rearing, and seaward migration, and the availability of food (CDFG 2010b; McCullough 1999; Myrick and Cech 2001; US EPA 2003). Further, the stressful impacts of water temperatures on salmonids are cumulative and positively correlated to the duration and severity of exposure. The longer the salmonid is exposed to thermal stress, the less chance it has for long-term survival (Carter 2008; US EPA 2001). It may be possible for healthy fish populations to endure some of these chronic impacts with little appreciable loss in population size. However, for vulnerable fish populations such as the endangered or threatened salmonids of the Central Valley, these sub-lethal effects can reduce the overall health and size of the population, making the survival and eventual recovery of these listed species more uncertain. It is therefore imperative to recognize that optimal water temperatures should be provided to those listed fish species whenever and wherever possible.

The general response of salmonids to temperature is that growth increases as temperature increases to an optimum, at which growth is maximized, followed by a rapid decline in growth as temperatures increase further (**Figure 6**). The optimum temperature for growth is dependent to some degree on the availability of food. At ration levels lower than the maximum (Rmax), the optimal temperature for growth is reduced because of the effects of temperature on metabolic rates and the subsequent maintenance metabolic demands for energy inputs (Brett *et al*. 1982 as cited in US EPA 2001).

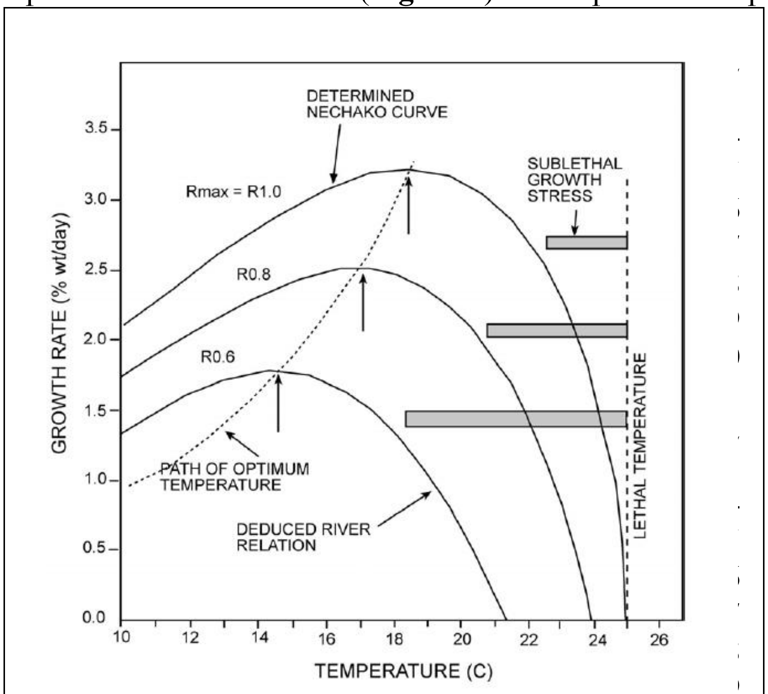


Figure 6. Water temperature vs. growth rate of spring-run Chinook (US EPA 2001). R0.8 = 80% satiation feeding. Rmax = R1.0 = unlimited rations

While considering salmonids' responses, water temperatures may be described as optimal, sublethal, and lethal (**Figure 6**). Physiologically optimal temperatures are those where physiological functions (*e.g.*, growth, swimming, heart performance) are optimized. At optimal temperatures, growth rates, expressed as weight gain per unit of time, are maximal for a life stage. These temperatures are generally determined in laboratory experiments. Ecologically optimal temperatures are those where fish do best in the natural environment, considering food availability, competition, predation, and fluctuating temperatures (US EPA 2003).

Exposure to water temperatures above the optimal range results in increased severity of harmful effects, often referred to as sublethal or chronic effects such as decreasing juvenile growth that results in smaller, more vulnerable fish; increasing susceptibility to disease that can lead to mortality, affecting reproduction, inhibiting smoltification, and decreasing ability to compete and avoid predation (McCullough *et al.* 2009; US EPA 2003). All of these responses, even those not resulting in immediate death, can lead to mortality prior to reproduction or reduced fecundity. These factors result in reduced productivity of a stock and reduced population size. In addition to the seasonal probability of consecutive days of critical maxima, consecutive years with serious cumulative thermal effects over significant portions of a species' range for one or more life stages can lead to dramatic reduction in stock viability (McCullough 1999).

Growth rates at temperatures above the optimum plummet with increasing temperature and rapidly reach zero. As temperatures rise to some point, they become lethal. Lethal temperatures are those that cause direct mortality within an exposure period of less than one week. One of the measures for lethal temperatures is the upper incipient lethal temperature (UILT), at which 50% mortality occurs (McCullough 1999; US EPA 2001).

5.2 Water Temperature Criteria for Salmonids

There were several metrics used for evaluating the effect of temperature on salmonids. The maximum weekly average temperature (MWAT) is a measure of chronic exposure of temperature. MWAT is the mean of multiple, equally spaced, average daily temperatures over a running seven-day consecutive period. The instantaneous maximum temperature is a measure of acute effects to salmonids. The 7-day average of the daily maximum (7DADM) temperatures is the daily maximum temperature over a running seven-day consecutive period. The 7DADM was recommended by the US Environmental Protection Agency (USEPA) in 2003 because it describes the maximum temperatures in a stream, but is not overly influenced by the maximum temperature of a single day. Thus, it reflects an average of maximum temperatures that fish are exposed to a weeklong period. Since this metric is oriented to daily maximum temperatures, it can be used to protect against acute effects, such as lethality and migration blockage conditions. This metric can also be used to protect against sub-lethal or chronic effects (*e.g.*, temperature effects on growth, disease, smoltification, and competition) (USEPA 2003).

Table 3. EPA water temperature criteria (7DADM) for Chinook and steelhead

Designated Uses for Life Stages	°C	°F
Adult Migration	20	68.0
Adult Migration plus Non-core Juvenile Rearing*	18	64.4
Adult holding	16	60.8
Spawning, Egg Incubation, and Fry Emergence	13	55.4
Core Juvenile Rearing**	16	60.8
Steelhead Smoltification	14	57.2

*This use is generally found in the mid and lower part of a river basin, downstream of the Core Juvenile Rearing use.
**This use is generally found in the mid-to-upper reaches of a river basin.

The US EPA water temperature criteria (**Table 3**) have been adopted by the states of Oregon and Washington. The criteria have been used to develop the 303(d) list and/or total maximum daily loads in the North Coast Region (Carter 2008) and Central Valley Region (CVRWQCB 2009). The criteria were supported and used by CDFG to develop flow criteria in the San Joaquin River basin (CDFG 2010a; CDFG 2010b). They were also used to analyze the effects of the long-term operations of the Central Valley Project and State Water Project, and to develop the reasonable and prudent alternative actions to address temperature-related issues in the Stanislaus River (NMFS 2009a).

The use of the US EPA 2003 criteria for listing water temperature impaired water bodies in the Central Valley is scientifically justified. It has been recognized that salmonid stocks do not tend to vary much in their life history thermal needs, regardless of their geographic location. There is not enough significant genetic variation among stocks or among species of salmonids to warrant geographically specific water temperature standards (US EPA 2001). Based upon reviewing a large volume of thermal tolerance literature, McCullough (1999) concluded that there appears to be little justification for assuming large genetic adaptation on a regional basis to temperature regimes. Prior to adoption of the revised water temperature standards for Oregon streams in 1996, there were separate water temperature standards assigned to salmon habitat in the western vs. the eastern portions of the state. Salmon-bearing streams in the western Cascades and Coast Range were assigned a standard of 14.4°C, but salmon-bearing streams in northeastern Oregon had a standard of 20.0°C, largely on the assumption that they would be adapted to the warmer air temperature regimes of the region. The large (5.6°C) difference in adaptation that would be required, however, is not supportable by any known literature (McCullough 1999).

Varying climatic conditions could potentially have led to evolutionary adaptations, resulting in development of subspecies differences in thermal tolerance. However, the literature on genetic variation in thermal effects indicates occasionally significant but very small differences among stocks and increasing differences among subspecies, species, and families of fishes. Many differences that had been attributed in the literature to stock differences are now considered to be statistical problems in analysis, fish behavioral responses under test conditions, or allowing insufficient time for fish to shift from field conditions to test conditions (US EPA 2001).

Although many of the published studies on the responses of Chinook salmon and steelhead to water temperature have been conducted on fish from stocks in Oregon, Washington, and British Columbia, a number of studies were reported for the Central Valley salmonids. Myrick and Cech (2001, 2004) performed a literature review on the temperature effects on Chinook salmon and steelhead, with a focus on Central Valley populations. Summarized in **Table 4** is a comparison of thermal responses between northern and Central Valley stocks.

Table 4. Similar thermal responses of the Central Valley (CV) and northern stocks of salmonids

Parameter	Water Temperature (°C)	Response	Note
Egg incubation	13.9	82% mortality	CV Fall-run
	>12	Increased mortality	CV Fall-run
	>13.3	Increased mortality	CV Winter-run
	13	Optimal	US EPA (2003)
Growth	17-20	Max growth	CV Fall-run, 100% satiation
	19	Max growth	CV Fall-run, 100% satiation
	18.9-20.5	Max growth	Northern Chinook, 100% satiation
	15	Max growth	60% satiation
	16	Optimal	US EPA (2003)
Upper thermal limit	25	50% mortality	7 day exposure, northern Chinook
	24	50% mortality	8 day exposure, CV Chinook

It is evident that the difference in thermal response is minimal in terms of egg incubation, growth, and upper thermal limit. Healey (1979 as cited in Myrick and Cech 2004) concluded that Sacramento River fall-run Chinook salmon eggs did not appear to be any more tolerant of elevated water temperature than eggs from more northern races. Myrick and Cech (2001) concluded that it appears unlikely that there is much variation among races with regard to egg thermal tolerance because data from studies on northern Chinook salmon races generally agree with those from California. They further concluded that fall-run Central Valley and northern Chinook growth rates are similarly affected by water temperature. There was one study on water temperature effects on the Central Valley steelhead growth. When American River steelhead were fed to satiation at water temperatures of 11, 15, and 19 °C, the growth rate was highest at 19 °C (Myrick and Cech 2001; Myrick and Cech 2004; Myrick and Cech 2005). The optimum water temperature for steelhead growth is expected to be lower at lower ration levels (*e.g.*, 60% satiation). Myrick and Cech (2001) also cautioned that the maximum growth rate at 19 °C was based on a single study and clear conclusions would not be possible until large-scale experiments were conducted.

US EPA indicated that these numeric criteria apply to the warmest times of the summer, the warmest years (except for extreme conditions), and the lowest downstream extent of use. Because of the conservative nature of this application, US EPA believes that it is appropriate to recommend numeric criteria near the warmer end of the optimal range for uses intended to protect high quality development and growth of salmonids. Adopting a numeric criterion near the warmer end of the optimal range is likely to result in temperatures near the middle of the optimal range for most of the spring through fall period in the segments where most of the rearing use occurs. If the criterion is met at the summer maximum, then temperatures will be lower than the criterion during most of the year. Because the criterion would apply at the furthest point downstream where the use is designated, temperatures will generally be colder across the full range of the designated use (USEPA 2003).

5.3 Water Temperature Modeling

To address the problem of elevated stream temperatures, it is necessary to incorporating stream temperature objectives in reservoir operations as high stream temperatures can be manipulated by releasing coldwater from an upstream reservoir. This requires the ability of predicting stream temperature with normal reservoir operations. Based on the prediction, decisions can be made to release additional water to improve the stream temperature to a target level appropriate for salmonids. In addition, a water temperature model will help better understand the relationship between physical landscape characteristics, weather conditions, and water temperature.

Numerous models have been developed for stream temperature prediction. Discussed below are two types of the models: Statistical and physical process based models.

5.3.1 Statistical Models

Statistical models use statistical methods including regression and artificial neural network.

5.3.1.1 Regression

Stream temperature generally increases with an increase of air temperature, and the impact is more significant for low streamflows during summer. Most regression models that predict water temperatures use univariate or multivariate regression techniques. Univariate regression models use air temperature as an explanary variable because stream temperature often has a high statistical correlation with ambient air temperature. Multivariate regression models are based on many variables including stream characteristics (*e.g.*, elevation, stream morphology, streamflow, channel aspect, riparian shade), ambient climate conditions (air temperature and solar radiation), and reservoir operations.

A univariate regression model was developed to evaluate the relationship between air temperature and stream temperature at a geographically diverse set of streams. The majority of streams showed an increase in water temperature of about 0.6-0.8°C for every 1°C increase in air temperature, with very few streams displaying a linear 1:1 air-water temperature trend. For most of the streams, a nonlinear model produced a better fit than did a simple linear model (Morrill *et al.* 2005). Modeling maximum daily stream temperatures using regression models to relate air and water temperatures was carried out in Catamaran Brook, a small stream in New Brunswick, Canada. The regression model, a logistic type function, predicted water temperatures on a weekly basis with a root mean square error (RMSE) of 1.93°C (Caissie *et al.* 2001).

A multivariate regression model was developed to predict daily maximum stream temperatures in the Truckee River in California and Nevada for the summer period (Neumann *et al.* 2003). The model used a stepwise linear regression procedure to select significant explanary variables. The stepwise procedure selected daily maximum air temperature at Reno and average daily flow at Farad as the variables to predict maximum daily stream temperature at Reno. The model was validated using three years of historical data. The model can be used to determine the amount of required additional flow to meet a target stream temperature with a desired level of confidence.

Multivariate regression models were developed on the basis of available stream temperature data to predict temperatures for unmeasured periods of time and for unmeasured streams in the Lower Klamath River in northern California (Flint and Flint 2008). The most significant factor in matching measured minimum and maximum stream temperatures was the seasonality of the estimate. Adding minimum and maximum air temperature to the regression model improved the estimate. The addition of simulated solar radiation and vapor saturation deficit to the regression model significantly improved predictions of maximum stream temperature but was not required to predict minimum stream temperature. The average standard error in estimated maximum daily stream temperature for the individual basins was 0.9 ± 0.6°C at the 95% confidence interval. In a similar study, it was found that air temperature was the most important variable in stream temperature prediction; however, the prediction performance efficiency was higher if solar radiation was included (Sahoo *et al.* 2009).

5.3.1.2 Artificial Neural Network

The advantages of artificial neural network (ANN) models in modeling stream temperature lie in their simplicity of use, low data requirement, and good performance, as well as their flexibility in allowing many input and output parameters.

ANN models were developed to estimate water temperatures in small streams using data collected at 148 sites throughout western Oregon from June to September 1999 (Risley *et al.* 2002). The sites were located on 1st-, 2nd-, or 3rd-order streams having undisturbed or minimally disturbed conditions. Data collected at each site for ANN model development included continuous hourly water temperature and description of riparian habitat. Additional data pertaining to the landscape characteristics of the basins upstream of the sites were assembled using geographic information system (GIS) techniques.

Clustering analysis was used to partition 142 sites into 3 groups. Separate ANN models were developed for each group. Critical input variables included riparian shade, site elevation, and percentage of forested area of the basin, and hourly meteorological data. The output variable was the hourly water temperature for the June to September period. Approximately one-third of the data sets were used for ANN training, and the remaining two-thirds were used for ANN testing. Coefficient of determination and RMSE for the models ranged from 0.88 to 0.99 and 0.05 to 0.59 ºC, respectively. The models were validated using historical temperature time series, habitat, and basin landscape data from 6 sites that were separate from the 142 sites that were used to develop the models.

ANNs were used to develop models for predicting both the mean and maximum daily water temperature (Jean-Francois and Daniel 2008). Eight models were investigated using a variety of input parameters. Of these models, four predicted mean daily water temperature and four predicted maximum daily water temperature. The best model for mean daily temperature had eight input parameters: minimum, maximum and mean air temperatures of the current day and those of the preceding day, the day of year and the water level. This model had a RSME of 0.96 °C, a bias of 0.26 °C and a coefficient of determination $R^2 = 0.971$. The model that best predicted maximum daily water temperature was similar to the first model but excluded mean daily air temperature. Good results were obtained for maximum water temperatures with an overall RMSE of 1.18 °C, a bias of 0.15 °C and $R^2 = 0.961$.

ANNs were used to predict stream temperature from solar radiation and air temperature (Sahoo *et al.* 2009). They developed a four-layer back propagation neural network. The optimal model performance was realized when the solar radiation and air temperature data were presented to the model with a 1-day or 3-day time lag.

5.3.2 Physical Process Based Models

Physical process based models predict water temperature using energy-balance equations. Mathematical equations are used to represent the physical processes of heat transfer among the Sun, stream, and surrounding environment. Meteorological data, including solar radiation, air temperature, wind speed and direction, and humidity, are typical inputs to these models. Water temperature models must be coupled with a hydrologic flow model. After a model has been calibrated and validated with measured data, it is possible to use the model to simulate water temperatures under various flow scenarios.

It is imperative to recognize that any model should be evaluated to determine whether a model and its results are of sufficient quality to serve as the basis for a decision (USEPA 2009). The process of evaluating a model should include the theoretical foundation of science underlying a model, the quality and quantity of available data, the degree of correspondence with observed conditions, and the appropriateness of a model for a given application (*e.g.*, temporal and spatial scales). The selected model should simulate system responses at temporal and spatial scales approximately one step lower than desired results. For example, to more effectively represent daily conditions, an hourly model at a minimum should be used. The hourly output data from the model can be used to generate daily maximum, minimum, and average. The opposite approach, disaggregation, requires taking longer interval data and attempting to reduce it to shorter time periods (*e.g.*, reducing monthly values to daily values, or daily values to hourly values). Disaggregation, by its very nature, typically introduces appreciable uncertainty into an analysis (Deas and Lowney 2001), making it worse when no description for disaggregating temperature or flow was provided by the user (Stillwater Sciences 2004).

Chen et al. developed a temperature modeling system for the Upper Grande Ronde watershed in northeast Oregon (Chen *et al.* 1998). The system consisted of a hydrologic simulation model (Hydrologic Simulation Program-FORTRAN (HSPF)) and a riparian shading model (SHADE). Solar radiation, diurnal, seasonal, and longitudinal variations were evaluated to verify the accuracy and reliability of SHADE computations. Simulated maximum stream temperatures, on which the riparian restoration forecasts are based, were accurate to 2.6–3.0°C compared with 8–10°C exceedances over stream temperature goals for salmon habitat restoration under the present riparian vegetation conditions. Hourly simulations have approximately the same accuracy and precision. Stream temperature regimes were simulated for different hydroclimatic conditions and hypothetical restoration scenarios of riparian vegetation. Regardless of natural weather cycles, the restoration of riparian vegetation is needed along many headwater streams to significantly alleviate the lethal and sublethal stream temperatures associated with salmon habitat in the watershed.

The U.S. Geological Survey (USGS), to support Department of Interior implementation of the Truckee-Carson-Pyramid Lake Water Rights Settlement Act of 1990, developed two physically based water quality models for simulating stream temperature and dissolved solids, for the Truckee River (Taylor 1998). The foundation of these water quality models is the physically based USGS daily flow-routing model of the Truckee River using HSPF. The flow-routing model routes streamflow, which transports heat and dissolved solids along 114 miles of the mainstem Truckee River from just downstream of Lake Tahoe, California to Marble Bluff Dam, just upstream from Pyramid Lake, Nevada.

Data to calibrate, validate, and evaluate these models included daily streamflow data; hourly stream temperature and meteorological data; and dissolved solids and specific conductance data covering the period from June 1, 1993 to September 30, 1995. Results of the stream-temperature model were evaluated at three USGS gaging stations along the Truckee River for June 1, 1993 to May 31, 1994 (calibration period) and June 1, 1994 to September 30, 1994 (validation period). The validation period included summer streamflows lower than the calibration period summer flows. In fact, the maximum streamflow for July-September 1994 at the Nixon station was less than the minimum streamflow for the same period in 1993 (38 and 45 cfs respectively).

Statistical comparisons at all three stations of simulated and observed values generally showed that during the calibration period mean absolute errors were less than 1^{o}C and there was generally a small negative bias less than 0.5^{o}C. For the validation period, the mean absolute errors were still generally less than 1^{o}C for daily maximum and minimum values. An exception to this was the error in daily maximum stream temperature simulations at Marble Bluff which increased from 0.8^{o}C during the calibration period to 1.8^{o}C during the validation period. There were also increases in the bias and variability of the errors for the validation period.

Statistical comparisons were also done by simulated streamflow class and showed that there was a tendency for daily maximum/minimum and hourly model simulation errors to decrease with increasing streamflow. The best simulations were found when simulated streamflow was greater than 500 cfs. When simulated streamflows were greater than 500 cfs, daily and hourly mean absolute errors were 0.4^{o}C to 0.8^{o}C for the calibration and validation periods at all three gaging stations (except daily maximum error at Marble Bluff which was 1.5^{o}C with only 5 values to compare in this flow class) (Taylor 1998).

Cox and Bolte developed the WET-Temp (Watershed Evaluation Tool - Temperature) model to estimate stream temperature distribution using spatially explicit data sets. Spatial data sets describing vegetation cover, stream network locations, elevation, stream discharge, and channel characteristics are utilized by WET-Temp to quantify geometric relationships between the sun, stream channel and riparian areas. These relationships are used in conjunction with air temperature, relative humidity and wind speed data to estimate the energy gained or lost by the stream via various heat flux processes (solar and longwave radiation, evaporation, convection and advection). The sum of these processes is expressed as a differential energy balance applied at discrete locations across the stream network. The model describes diurnal stream temperature dynamics at each location and thus temperature distribution across the entire network. WET-Temp was calibrated to McDowell Creek in the western Oregon Cascade Range. Differences between observed and simulated values of maximum daily temperature in McDowell Creek were less than 0.3 °C. The model was then used to estimate temperature distributions in an adjacent watershed, Hamilton Creek, where differences in maximum daily temperature were 1 °C or less. WET-Temp advances the subject of stream temperature modeling through direct incorporation of spatially explicit data sets and treatment of temperature as a network phenomenon (Cox and Bolte 2007).

Two stream temperature models, SNTEMP and CE-QUAL-W2, were applied to the Speed River in Southern Ontario in order to gauge the effectiveness of various stream temperature management options. Calibrated versions of both models performed well (0.2 °C less than mean absolute error of SNTEMP less than 1.8 °C; 0.5 °C less than mean absolute error of CE-QUAL-W2 less than 1.4 °C). However, CE-QUAL-W2 performed more consistently spatially and temporally. Air temperature and relative humidity were found to be the most sensitive parameters in both models. Management alternatives considered in this study included modifying discharge from upstream dams, removal of in-stream impoundments, allowing the growth of adequate riparian vegetation to provide shade, and reducing stream width during low-flow periods. Of the various management practices investigated, model results suggest that the removal of in-stream impoundments would be the most effective management alternative to reduce summer stream temperatures. Management alternatives involving removal of in-stream impoundments were projected to reduce minimum stream temperatures by up to 2.2 °C, while those not involving the removal of in-stream impoundments were projected to reduce minimum temperatures by less than 0.5 °C (Norton and Bradford 2009).

Caissie et al. (2007) applied a deterministic water temperature model to two streams in New Brunswick, Canada. Data from 1992 to 1994 were used to calibrate the model, while data from 1995 to 1999 were used for the model validation. Results showed equally good agreement between observed and predicted water temperatures during the calibration period for both rivers with a RMSE of 1.49 °C for the Little Southwest Miramichi River compared to 1.51 °C for Catamaran brook. During the validation period, RMSEs were calculated to be 1.55 °C for the Little Southwest Miramichi River and 1.61 °C for Catamaran Brook. Poorer model performances were generally observed early in the season (*e.g.*, spring), especially for the Little Southwest Miramichi River due to the influence of snowmelt conditions, while late summer to autumn performances showed among the best results for both rivers. Late autumn performances were more variable in Catamaran Brook and presumably influenced by the groundwater, geothermal conditions and potentially riparian shading. The geothermal aspect was further investigated at Catamaran Brook (using 1998 data) and results revealed that although geothermal fluxes are present, they explained very little of the unexplained variability (less than 0.1 °C). The net solar radiation was shown to be the dominant energy flux, while the evaporative heat flux contributed significantly to cooling rivers during warmer summers (Caissie *et al.* 2007).

In the Central Valley, a 1D stream temperature model (sub-hourly time step, sub-kilometer spatial resolution) has been developed for the Upper Sacramento River (Danner and Pike 2011). The model has the capability to both hindcast and forecast water temperatures. The model uses a physically-based heat budgets to calculate the rate of heat transfer to/from the river. The hydrodynamics of the river (flow velocity and channel geometry) are characterized using densely spaced channel cross-sections and flow data. Water temperatures are calculated by considering the hydrologic and thermal characteristics of the river and solving the advection-diffusion equation for heat transport in a mixed Eulerian-Lagrangian framework.

Modeled hindcasted temperatures for several test periods (May – November 2008, 2009, and 2010) substantially improve upon the existing daily-to-monthly mean temperature standards. Modeled values closely approximate both the magnitude and the phase of measured water temperatures. The model results reveal important longitudinal patterns in diel temperature variation that are unique to regulated rivers, and may be critical to salmon habitat. The model can be used to access the forecast model online, run various scenarios of water discharge and temperature under forecasted weather conditions (3-5 days and seasonal), and inform decisions about water releases to maintain optimal temperatures for fishery health (Danner and Pike 2011).

In the Stanislaus River, a 1D hydrodynamic model has been developed to simulate water temperature downstream of Goodwin Dam (Tetra Tech 2011). The model is based on the Environmental Fluid Dynamics Code (EFDC). The EFDC model simulates water temperature based on meteorological and hydrodynamic conditions and can generate high resolution results in both space and time. The water temperature simulation algorithm in EFDC is based on the physics of heat transport, and user intervention is minimal. The model was calibrated with observed flow and water temperature. The calibration results showed good agreement between the modeled and measured.

The results of 25 simulation scenarios suggest that releasing cool water from the reservoirs greatly improves water temperatures in the lower Stanislaus River. At Knights Ferry there were expected to be no temperature target exceedances when boundary water temperatures 8 ºC, even under the critical dry condition. When water temperature was above 11ºC, exceedances were expected to occur throughout the year. Both boundary water temperature and flow govern the water temperature at Orange Blossom Bridge. Decreasing boundary water temperature and increasing flow can dramatically reduce exceedances at Orange Blossom Bridge. The lowest flow and highest boundary water temperature combination expected to achieve no exceedances at both Knights Ferry and Orange Blossom Bridge is the above average flow condition and 9 ºC boundary water temperature (Tetra Tech 2011).

5.3.3 Water Temperature Modeling in Clear Creek

The Water Resources and Environmental Modeling Group of the University of California at Davis conducted in 1998 a study on water temperature modeling for Whiskeytown Reservoir and Clear Creek, which was funded by the USBR (Orlob *et al.* 1999). Based on RMA2 and RMA11, the Group developed a one-dimensional hydrodynamic model for simulating water temperatures from Whiskeytown Dam to the confluence of the Sacramento River. The model was calibrated and verified with data from four controlled flows: 50, 100, 150, 200 cfs during the time period of August 12 to 30, 1998. Each of the four controlled flows lasted for 4 days. Actual flows were measured at 6 locations on the third day of each controlled flow along the creek. Water temperatures were measured continuously at 9 locations.

The model was calibrated for water temperature at the controlled flow of 50 cfs. The primary adjustment in heat exchange rate was in the rate of evaporative heat flux, *i.e.*, in changing the co0ling effect of evaporation until the mean simulated water temperature and the diurnal fluctuation, in both magnitude and phase, correspond to observed values. Differences between measured and simulated temperatures were found to be as high as 2 ºC, which occurred during the time period of peak temperatures.

The model was then verified for water temperature at the controlled flows of 100, 150, and 200 cfs. Differences between measured and simulated temperatures were found to be as high as 3 ºC, which also occurred during the time period of peak temperatures. While integrating the topographic shading to the model, the difference at peak temperatures decreased, whereas the difference at low temperatures increased.

The application of the model for real-time reservoir operations seems limited as calibration and verification of the model were based on a short period of time and discrepancies between the modeled and observed temperatures were high. However, the study has shed some light on the relationship between flow and water temperature in Clear Creek. Streamflow in Clear Creek should be maintained above 100 cfs to achieve a water temperature target of 16 ºC and above 200 cfs to achieve 13 ºC at Igo under the maximum weather condition. Unfortunately, the report did not specify if the temperature targets were the daily average or daily maximum, and neither define the maximum weather condition.

6 Flow, Temperature, and Weather Data Used in This report

Field observation data used in this report include streamflow, water temperature, reservoir release, and weather (air temperature, solar radiation, wind speed, and precipitation). The daily streamflow data in Clear Creek were obtained from the USGS gage station (USGS 11372000) near Igo (40.5132 N, -122.5229 W, elevation 673 ft) in Shasta County, California (http://waterdata.usgs.gov/nwis/uv?11372000). The station is located under an old highway bridge on Redding-Igo Road, 1.0 mi northeast of Igo, and 10.4 mi upstream from mouth. The flow data are from October 1, 1940 to September 30, 2009. Since the completion of Whiskeytown Dam in May 1963, baseflows were completely regulated by Whiskeytown Reservoir. Transbasin diversion from the Trinity River through Judge Francis Carr Powerplant to Whiskeytown Reservoir began in April 1963. Diversions from Whiskeytown Reservoir to Spring Creek Powerplant began in December 1963.

Water temperature, weather, and reservoir release data were obtained from the California Data Exchange Center (CDEC) (http://cdec.water.ca.gov). The hourly water temperature was available at the USGS Igo station beginning September 6, 1996. There are two weather stations within or near the Clear Creek watershed that provide hourly air temperature, solar radiation, and wind speed and daily precipitation beginning November 26, 2001. One station is at Oak Bottom (40.6510, -122.6060, elevation 1326 ft) operated by the National Park Service and the other is at Redding (40.5165, -122.2910, elevation 500 ft) operated by the US Forest Service. However, the wind speed data were not included for use in model development due to high uncertainty observed in the data. Since the Oak Bottom station exhibited more missing data, the data from the Redding station were used in this report. The data of daily reservoir release to Clear Creek were from the Whiskeytown Dam station, which is operated by the USBR. **Table 5** summarizes the basic information of the data used in this report.

The data compiled from the sources described above were checked to assure data quality. For example, any records of data, which showed apparent recording errors (*e.g.*, spikes on streamflow, temperature, or other parameters) or were marked as “missing”, were excluded from use in the report. Any records of data showing zero reservoir release were not included in the report. Any records of data that showed the ratio of the Whiskeytown Reservoir release to streamflow at Igo less than 0.8 or greater than1.2 were excluded. Since this report focuses on the time period of June through October, any data with daily precipitation greater than0.2 inch were not included to avoid potential influence of rainfall generated runoff. There were a total of 40 days that had a precipitation $\geq$ 0.2 inch between June 1 to October 31 from 2002 to 2009, accounting for 3% of the total record days (1224 days).

The daily maximum (DMax) of water temperature, air temperature, solar radiation was derived from the recorded hourly data. The 7-day average of daily maximum (7DADM) is the average of consecutive 7-day daily maximum including the current day, previous three days, and next three days.

Table 5. Description of the data used in this report

Parameter	Station Location	Time Step	Start Date	End Date
Air temperature	Redding	Hourly	11/26/2001	9/30/2009
Precipitation	Redding	Hourly	11/26/2001	9/30/2009
Solar radiation	Redding	Hourly	11/26/2001	9/30/2009
Reservoir release	Whiskeytown Dam	Daily	4/1/2000	9/30/2009
Streamflow	Igo	Daily	10/1/1940	9/30/2010
Streamflow	Igo	Hourly	9/6/1996	9/30/2009
Water temperature	Igo	Hourly	9/6/1996	9/30/2009

7 Analysis of Flow Regime in Clear Creek

Streamflow in Clear Creek has changed as a result of flow regulation by Whiskeytown Dam. The number of flows greater than 2000 cfs was reduced dramatically after the dam was built in 1963 (**Figure 7**). The number of flows greater than 6000 cfs was 16 from WY1941 to 1962, as opposed to 5 from 1963 to 2009. Detailed flow change analyses are provided below.

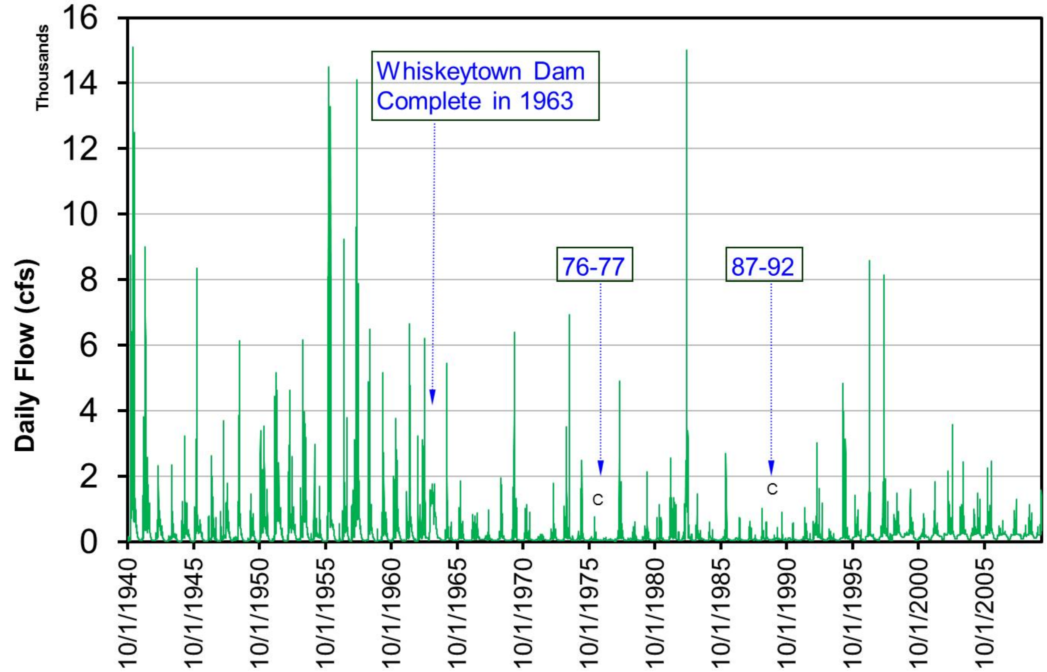


Figure 7. Daily streamflow in Clear Creek from water year 1941-2009. The periods of 1976-1977 and 1987-1992 were critically dry years.

7.1 Flow Alteration Analysis

Flow data at Igo in Clear Creek were analyzed using the IHA method (software program version 7.1) (The Nature Conservancy 2009). The IHA method, developed by Richter and others (Richter *et al.* 1996; Richter *et al.* 1997; Richter *et al.* 1998), characterizes flow variations on the basis of 32 ecologically relevant statistical parameters. These parameters were compared for two time periods: pre-dam (1941 to 1960) and post-dam (1963 to 2009). The data from 1961 and 1962 were not used in the analysis due to possible interference from the dam construction. The IHA parameters were calculated using nonparametric (percentile) statistics because of the skewed (non-normal) nature of the flow data used in this report.

Table 6. Results of hydrologic data analysis and RVA boundaries for Clear Creek (flow rate in cfs)

Hydrologic Parameters	Pre-dam period: 1941-1960			Post-dam period: 1963-2009			RVA Boundaries		Middle HA
	Median	Min	Max	Median	Min	Max	Low	High	
Group #1 Monthly flows (cfs)									
October	36	20	246	56	32	1340	29	47	-0.840
November	65	38	291	121	61	1360	55	111	0.064
December	163	45	1010	144	75	1320	64	351	1.128
January	386	64	2070	159	48	797	164	681	0.117
February	576	132	5010	170	48	1350	320	851	-0.681
March	499	168	1560	185	49	2500	399	670	-0.894
April	393	160	1705	126	49	1535	308	723	-0.905
May	298	80	714	77	48	310	200	391	-0.468
June	131	69	285	63	43	217	109	202	-0.415
July	50	22	126	55	39	155	42	82	0.797
August	29	14	65	53	38	152	23	40	-0.947
September	27	13	49	54	35	1550	20	32	-1.000
Group #2 Magnitude and duration of annual minimum and maximum flows (cfs)									
1-day min	18	9	37	49	30	112	16	22	-1.000
3-day min	18	9	37	49	30	114	17	22	-1.000
7-day min	19	9	38	50	31	114	17	23	-1.000
30-day min	23	12	46	51	38	149	20	33	-1.000
90-day min	35	18	78	54	38	154	29	55	0.383
1-day max	5165	1470	15100	1500	130	15000	3510	6630	-0.681
3-day max	3888	1074	10670	1051	109	13300	2949	5270	-0.734
7-day max	2784	733	7920	695	106	7979	2109	3719	-0.681
30-day max	1506	439	5673	415	99	3477	1047	1958	-0.681
90-day max	1012	324	3379	321	83	1970	670	1165	-0.734
Group #3 Timing of annual 1-day minimum and maximum flows (Julian date)									
Date of min	266	224	290	238	165	305	251	270	-0.734
Date of max	35	7	362	41	1	357	35	55	-0.309
Group #4 Frequency and duration of low and high flow pulses*									
Low pulse count	3	1	5	0	0	8	2	3	-0.905
Low pulse duration (d)	25	6	122	3	1	35	14	53	-0.894
High pulse count	6	1	9	5	0	11	5	7	-0.447
High pulse duration (d)	5	1	167	2	1	13	4	7	-0.734
Group #5 Rate and frequency of flow changes									
Rise rate (cfs/d)	14	4	166	6	2	34	8	18	-0.291
Fall rate (cfs/d)	-9	-22	-5	-3	-12	-1	-12	-8	-0.787
Number of reversals	70	64	87	81	61	104	68	75	-0.622

*The low pulse threshold is 46 cfs. The high pulse threshold is 420 cfs.

The IHA results for Clear Creek are summarized in **Table 6** for both pre- and post-dam time periods. The IHA analysis includes 5 groups of hydrologic parameters: Monthly magnitude, magnitude and duration of annual extremes, timing of annual extremes, frequency and duration of high and low flows, and rate and frequency of flow changes.

From the pre-dam to post-dam period, the median monthly flows from January through June decreased by 50-80%, while the median monthly flows from August through November increased by about 100%. The annual maximum flows decreased by 70%, whereas the annual minimum flows increased by more than 100%.

7.1.1 Magnitude of Monthly Flows

The magnitude of monthly flows provides a general measure of habitat availability for aquatic organisms and influences water temperature, dissolved oxygen, and photosynthesis in water column. The median monthly streamflows are presented in **Figure 8**. For the pre-dam period, the median streamflow in Clear Creek was low in October (36 cfs), increased from December, reached the highest in February (576 cfs), and gradually decreased from March, and reached low again in July through September (27 cfs). For the post-dam period, on the other hand, monthly flows were relatively similar from October through September. Monthly flows increased starting in the late 1990s when the CVPIA AFRP was implemented in Clear Creek.

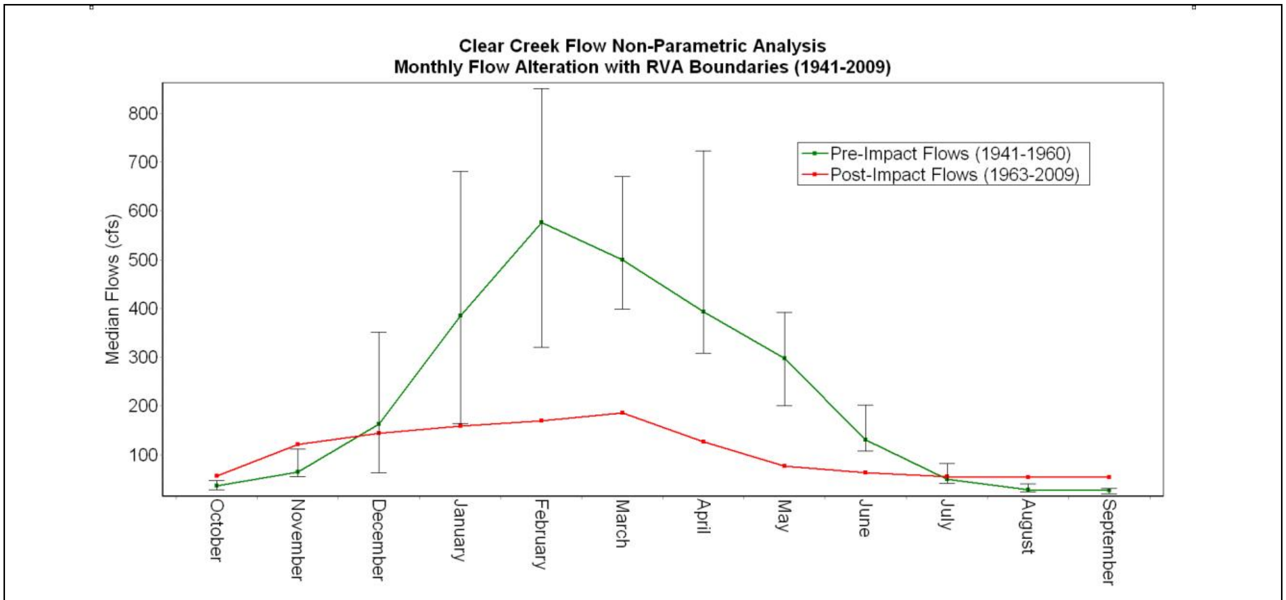


Figure 8. Pre-dam median monthly flow (green line) with error bars indicating the RVA Boundaries - 33rd and 67th percentiles. Also shown is the post-dam median monthly flow (red line).

7.1.2 Magnitude and Duration of Annual Extreme Flows

The magnitude and duration of annual extreme flows affect river channel morphology, riverine vegetation, sediment/gravel transport, and aeration of spawning beds. These extremes may serve as precursors or triggers for migration and reproduction of salmonids. The durations include the 1-day, 3-day, 7-day (weekly), 30-day (Monthly), and 90-day (seasonal) extremes (**Table 6**). The

1-day minimum (or maximum) represents the lowest (or highest) single daily flow occurring during the year. The multi-day minimum (or maximum) represents the lowest (or highest) multi-day average flow occurring during the year. The median minimum flows for the pre-dam period were lower than that for the post-dam periods (**Figure 9**).

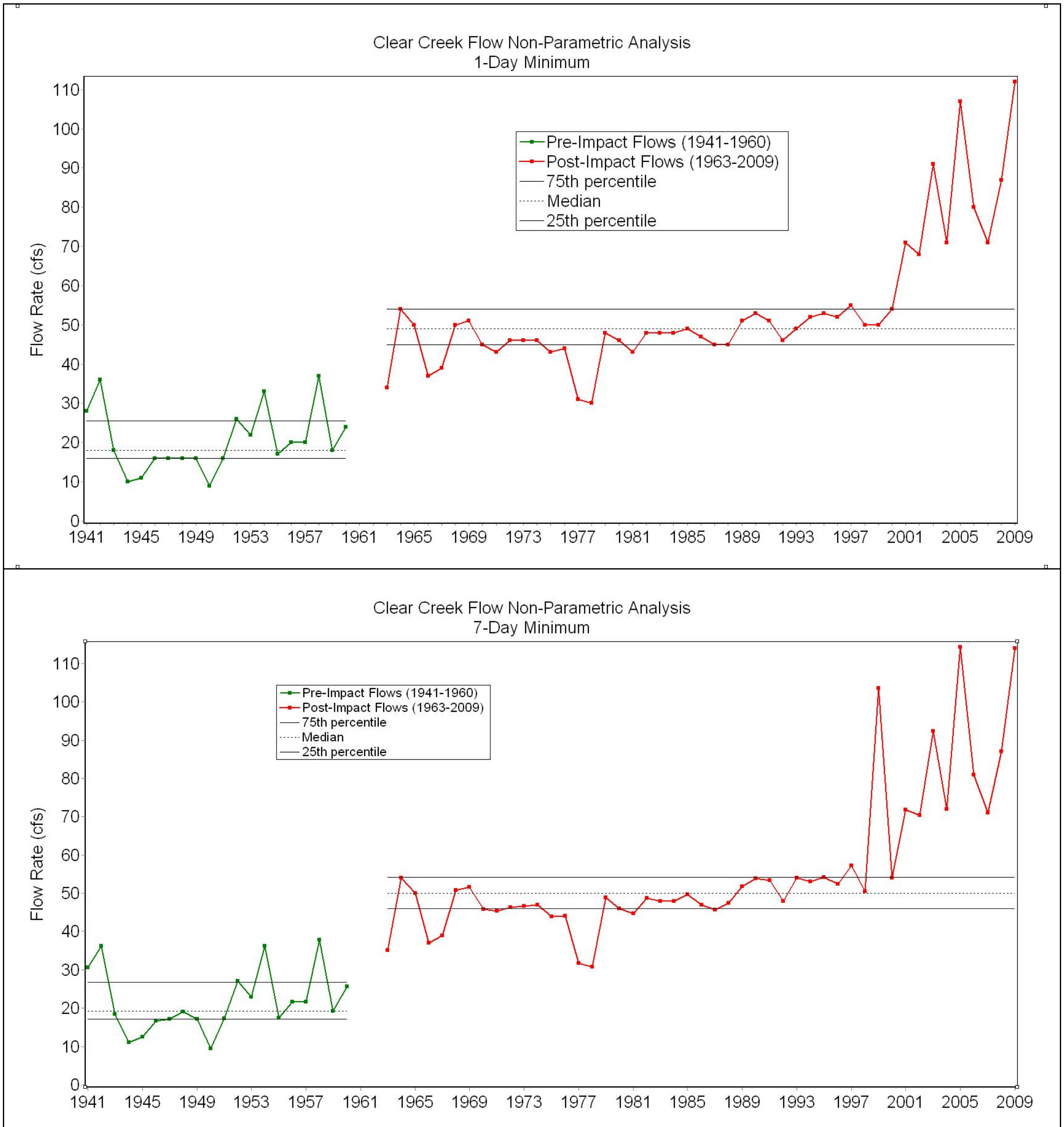


Figure 9. 1-day and 7-day minimum flows in Clear Creek

The 1-day median maximum flows decreased from 5000 cfs for the pre-dam period to 1500 cfs for the post-dam periods, and The 7-day median maximum flows decreased from 2800 cfs for the pre-dam period to 700 cfs for the post-dam periods (**Table 6** and **Figure 10**).

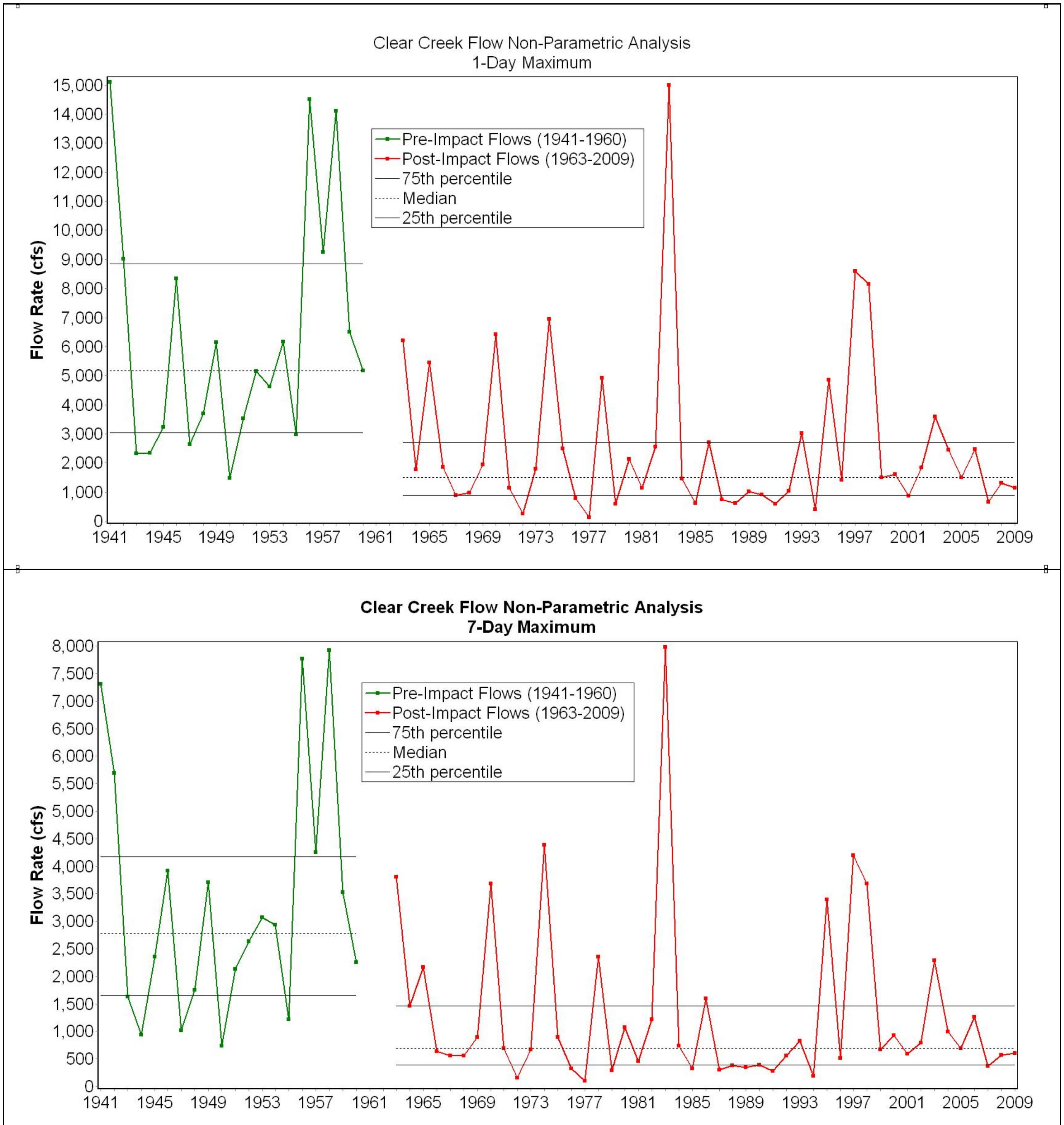


Figure 10. 1-day and 7-day maximum flows in Clear Creek

7.1.3 Timing of Annual Extreme Flows

This is the Julian date of the annual 1-day minimum and 1-day maximum flows. The timing of annual extreme flows provides spawning cues for anadromous fish, compatibility with life cycles of organisms, and access to special habitats during reproduction or to avoid predation. The median Julian date for the 1-day minimum flow was 266 (end of September) for the pre-dam period, comparing to 238 (end of August) for the post-dam period. The inter-annual variation in timing of the 1-day minimum flow was larger for the post-dam period than for the pre-dam period, whereas the timing and inter-annual variation in the 1-day maximum flow were similar between these two periods (**Figure 11**).

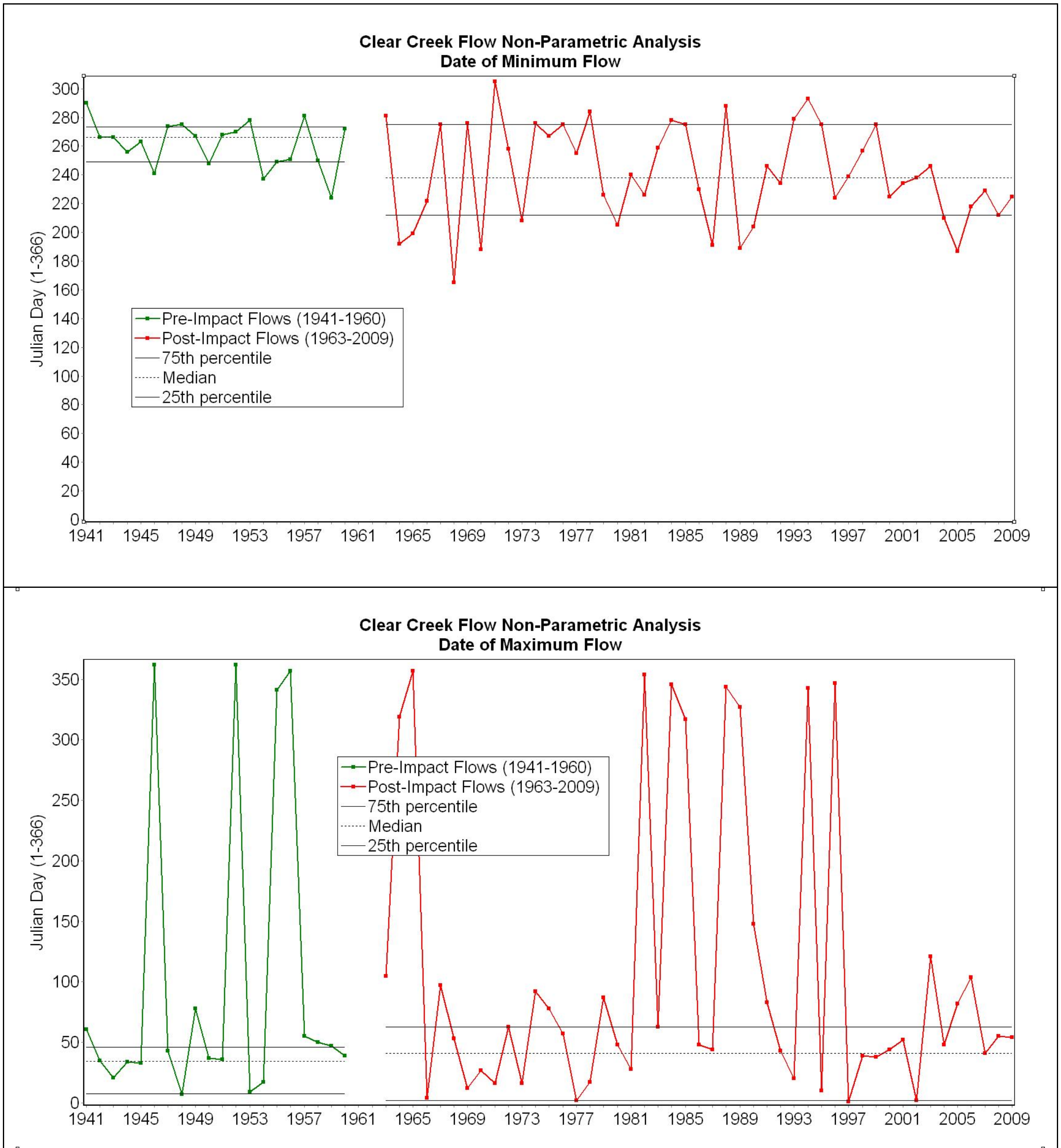
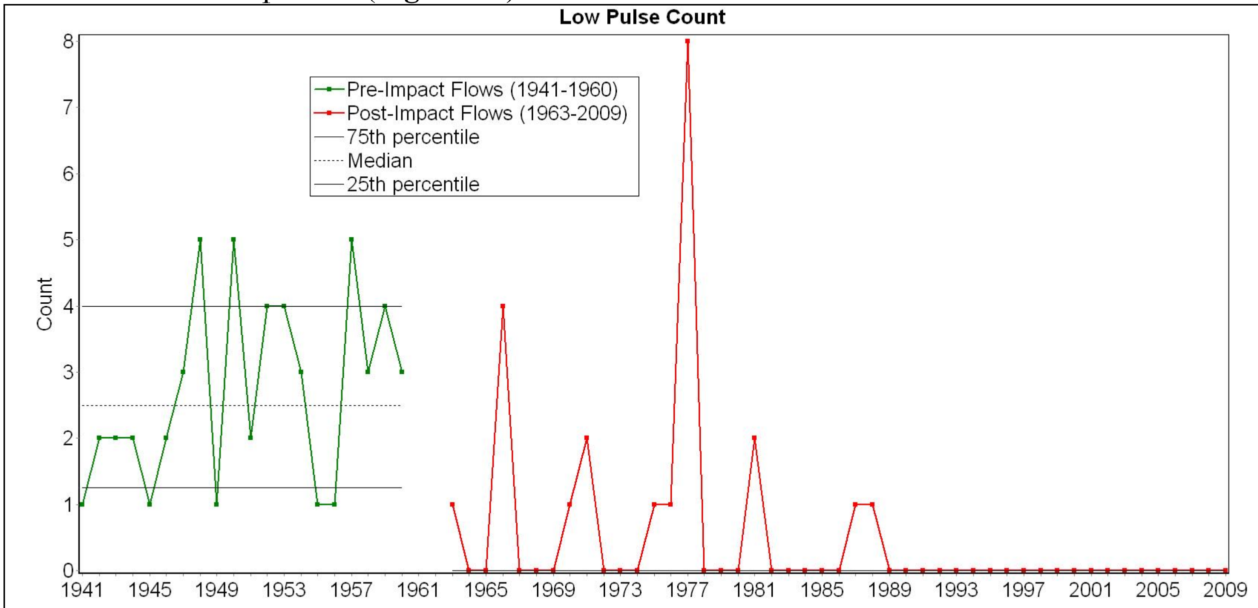


Figure 11. Julian date of annual 1-day minimum and 1-day maximum flows in Clear Creek

7.1.4 Frequency and Duration of High and Low Pulses

The frequency and duration of high and low pulses influence the availability of floodplain habitat, nutrient exchange between river and floodplain, and sediment/gravel transport. Hydrologic pulses in this report are defined as those periods within a year in which the daily average water flow either rises above the 75th percentile (high pulse) or drop below the 25th percentile (low pulse) of all daily values for the pre-dam period. The low pulse threshold was 46 cfs and the high pulse threshold was 420 cfs. The number of low flow pulses decreased from the pre-dam period to the post-dam period, whereas the number of high flow pulses was similar between these two periods (**Figure 12**).



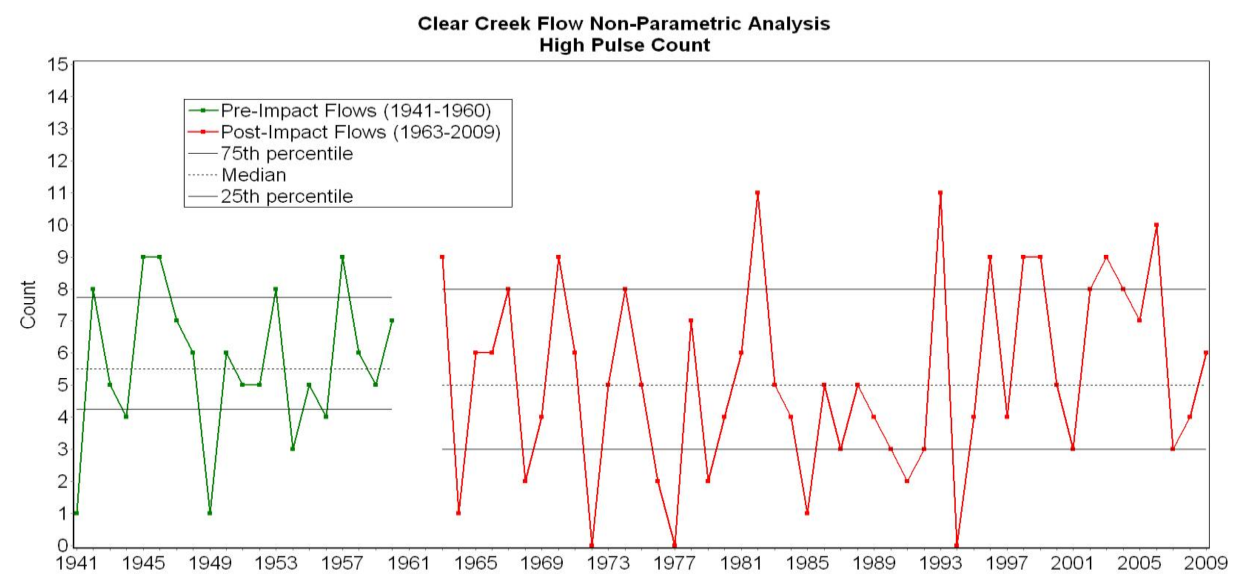


Figure 12. Number of low (upper chart) and high (lower chart) flow pulses within each water year in Clear Creek. The low pulse threshold is 46 cfs and the high pulse threshold is 420 cfs.

The duration of both low and high flow pulses decreased from the pre-dam period to the post-dam period (**Figure 13**).

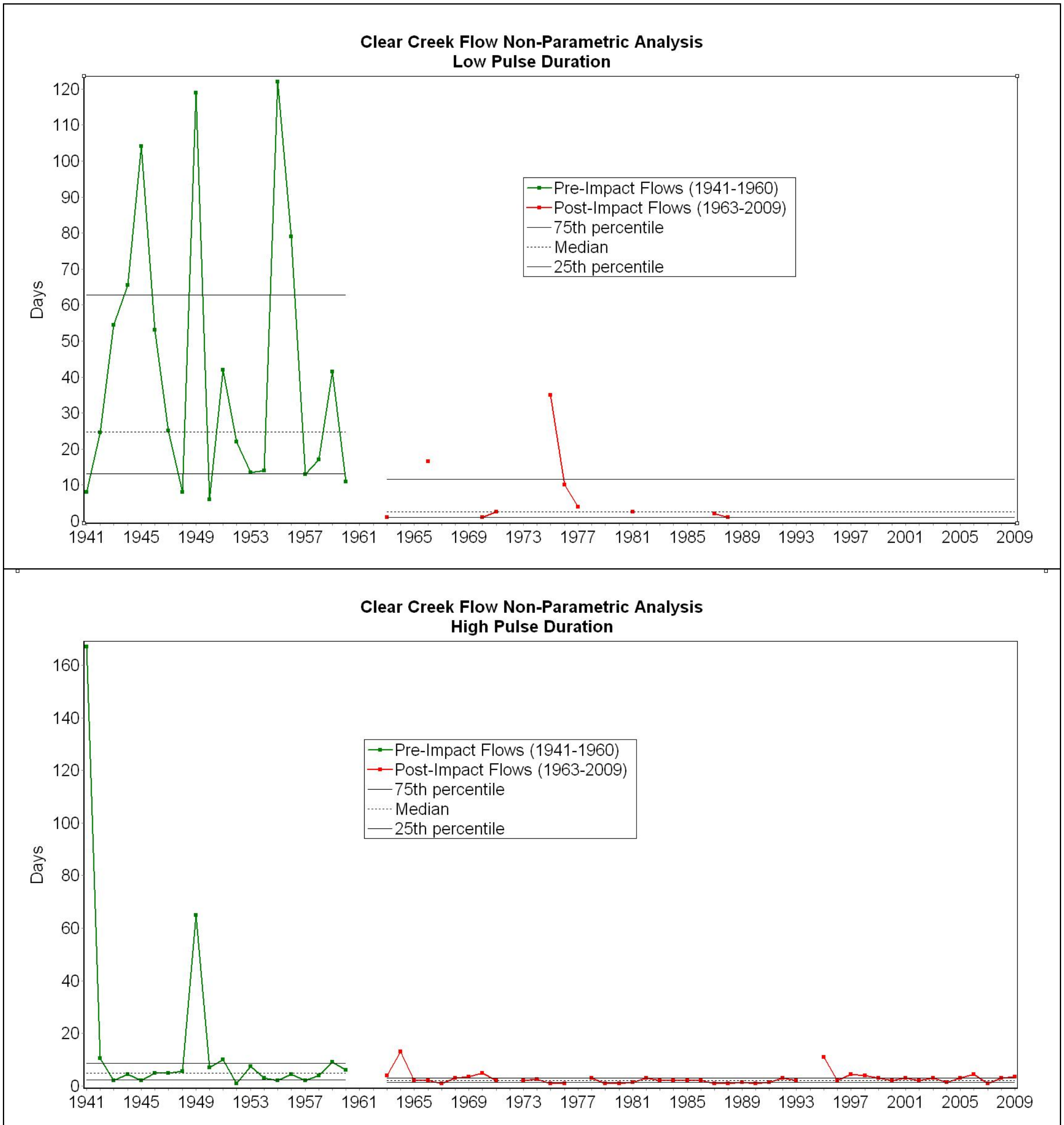


Figure 13. Duration of low (upper chart) and high (lower chart) flow pulses within each water year in Clear Creek. The low pulse threshold is 46 cfs and the high pulse threshold is 420 cfs.

7.1.5 Rate and Frequency of Flow Changes

The rate of flow changes includes the rise rate (positive differences between consecutive daily flows) and fall rate (negative differences between consecutive daily flows). Both the rise and fall rates decreased from the pre-dam period to the post-dam period (**Figure 14**).

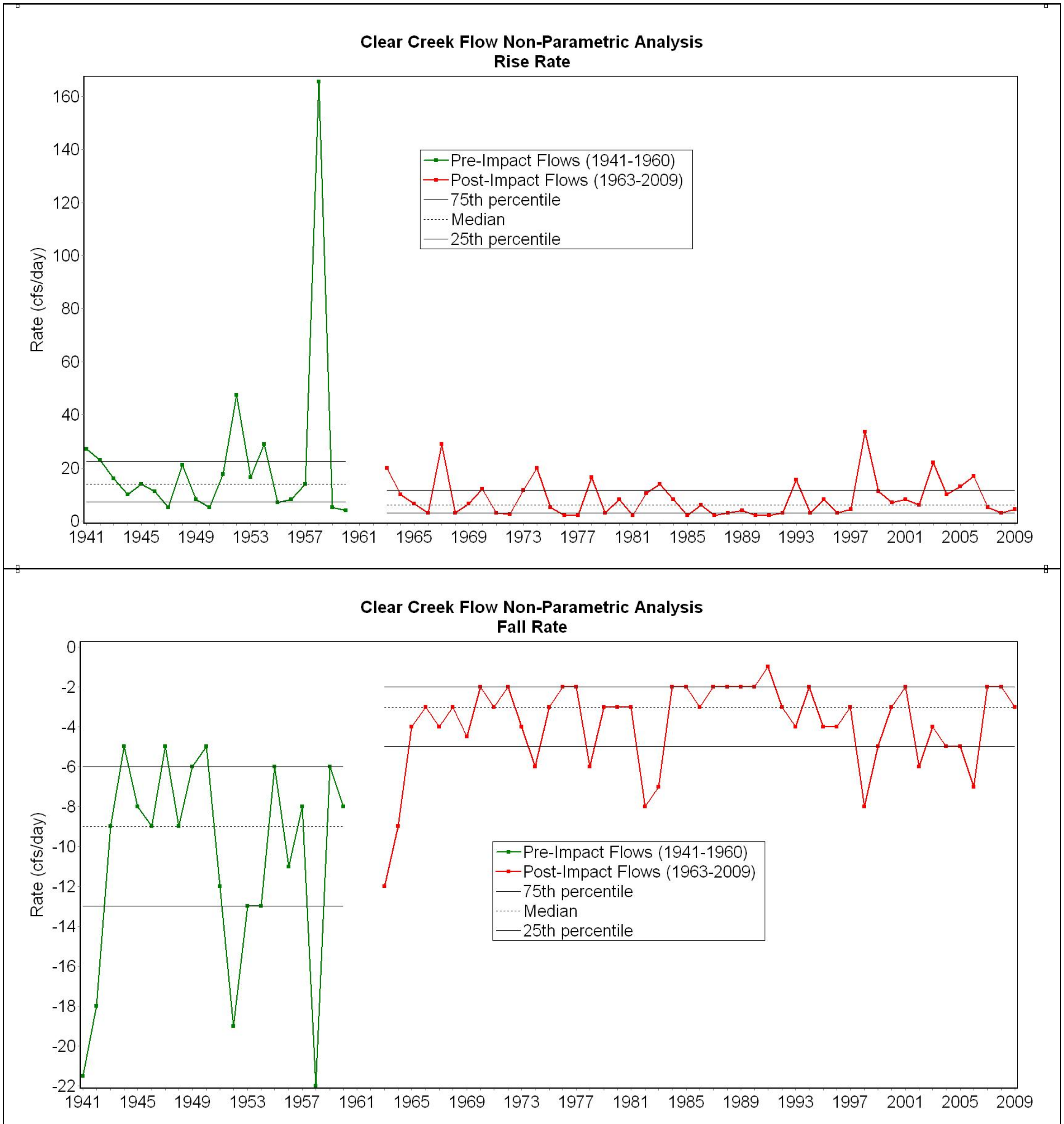


Figure 14. Rates of flow changes (cfs/day) in Clear Creek

Hydrologic reversals are calculated by dividing the hydrologic record into "rising" and "falling" periods, which correspond to periods in which daily changes in flows are either positive or negative, respectively. Note that a rising or falling period is not ended by a pair of days with constant flow, only by a change of sign in the rate of change. The number of reversals is the number of times that flow switches from one type of period to another. The post-dam period showed more hydrologic reversals than the pre-dam period (**Figure 15**).

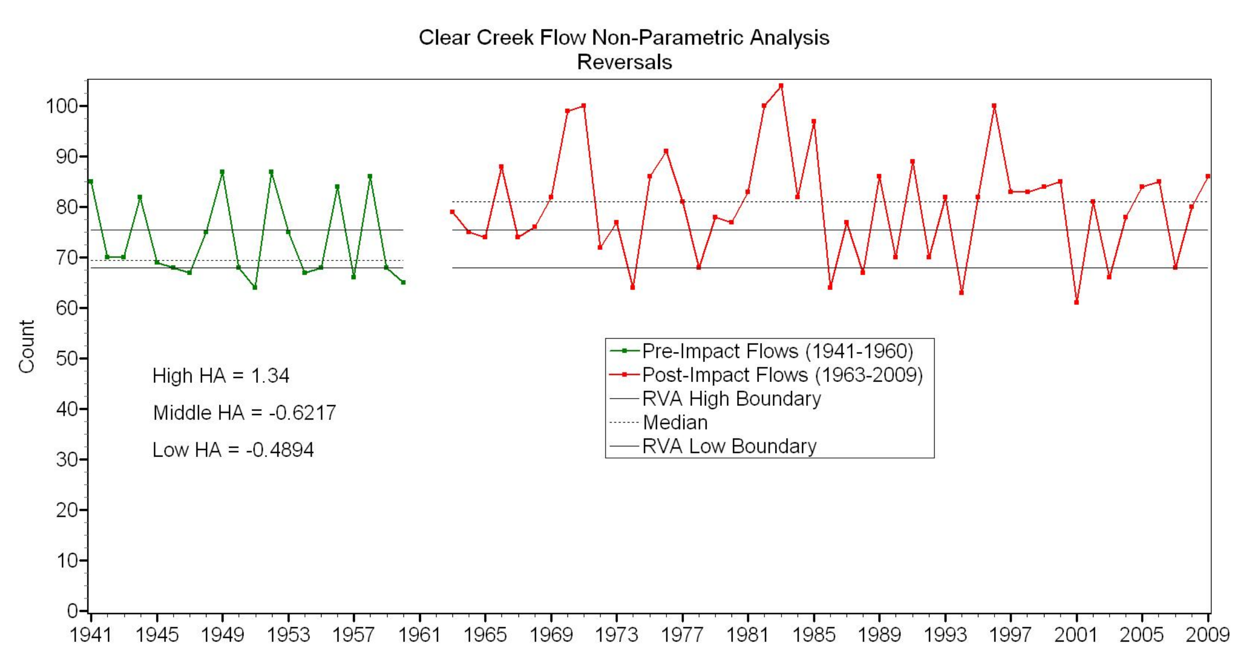


Figure 15. Number of hydrologic reversals in Clear Creek

7.1.6 Range of Variation Approach (RVA) Analysis

The RVA was developed to set initial streamflow-based river ecosystem management targets, which are intended to guide the design of river management strategies (*e.g.*, reservoir operation rules and habitat restoration) (Richter *et al.* 1997). The RVA targets may be redefined as new research on the linkage between hydrological characteristics and aquatic ecosystem integrity becomes available. The management objective is not to have the river attain the targeted range every year, rather, it is to attain the targeted range at the same frequency as occurred in the pre-dam flow regime. For example, attainment of an RVA target range defined by the 33rd and 67th percentiles of a particular parameter would be expected in only 50% of years.

In an RVA analysis (Richter *et al.* 1997; Richter *et al.* 1998), the full range of pre-dam data for each parameter is divided into three different categories. The boundaries between categories are based on percentile values for non-parametric analysis, in which the category boundary is 17 percentiles from the median. This yields an automatic delineation of three categories of equal size:

1) the low category that contains all values less than or equal to the 33rd percentile;
2) the middle category that contains all values falling in the range of the 34th to 67th percentiles, which is the RVA boundaries as given in **Table 6**; and

3) the high category that contains all values greater than the 67th percentile.

The degree, to which the RVA target range is not attained, is a measure of hydrologic alteration. The IHA software program computes the expected frequency, with which the "post-impact" values of the IHA parameters should fall within each category. The expected frequency is equal to the number of values in the category during the pre-dam period multiplied by the ratio of post-dam years to pre-dam years. The program also computes the observed frequency, which is the number of the "post-impact" annual values of IHA parameters fall within each of the three categories. The Hydrologic Alteration (HA) factor is then calculated for each of the three categories as:

HA = (observed frequency – expected frequency) / expected frequency

When the HA values are close to zero, minimum changes occurred from the pre-dam period to post-dam period. A positive HA value means that the frequency of values in the category has increased from the pre-dam to the post-dam period (with a maximum value of infinity), while a negative HA value means that the frequency of values has decreased (a minimum value of -1).

The RVA boundaries and HA values are given in **Table 6** and shown from **Figures 9 to** Figure. For the middle RVA category, most of the 31 parameters showed a decreased frequency from the pre-dam period to the post-dam period (**Figure 16**). Note that even though the monthly flow in August, September, and October increased from the pre-dam to post-dam period, the number of monthly flow values falling within the middle RVA category decreased and the number of monthly flow values falling within the high RVA category increased (**Figure 17**).

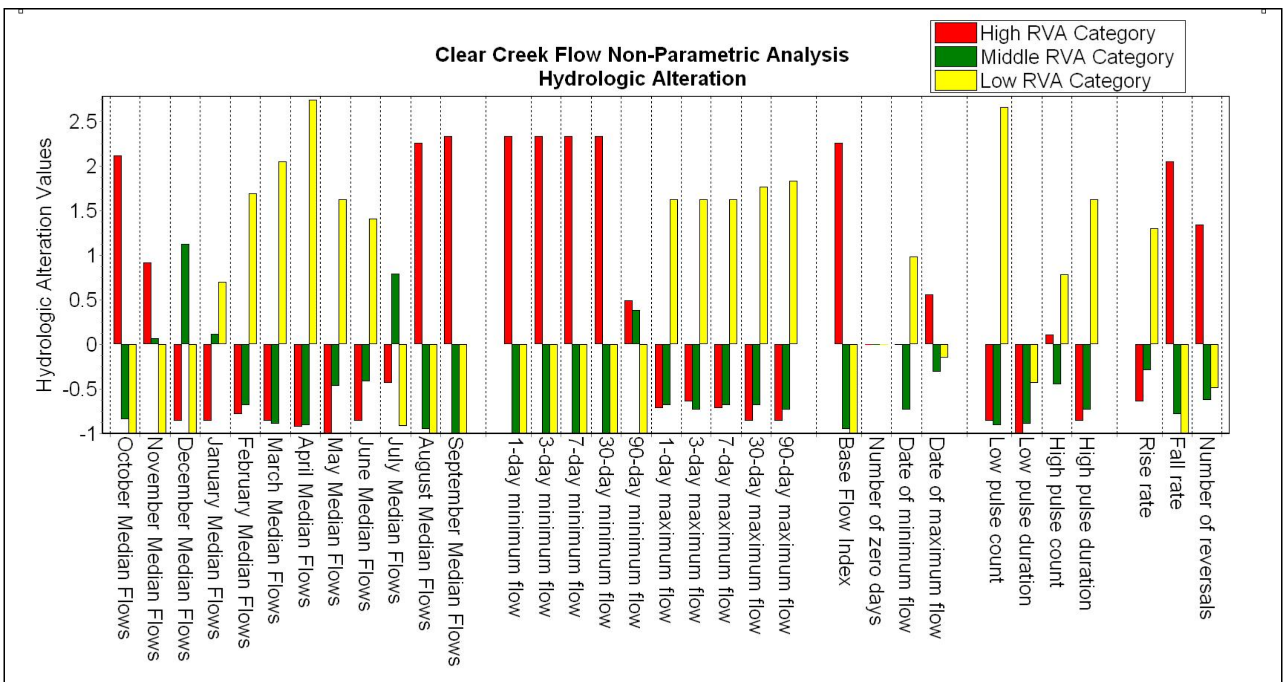


Figure 16. Hydrologic alterations for three RVA categories

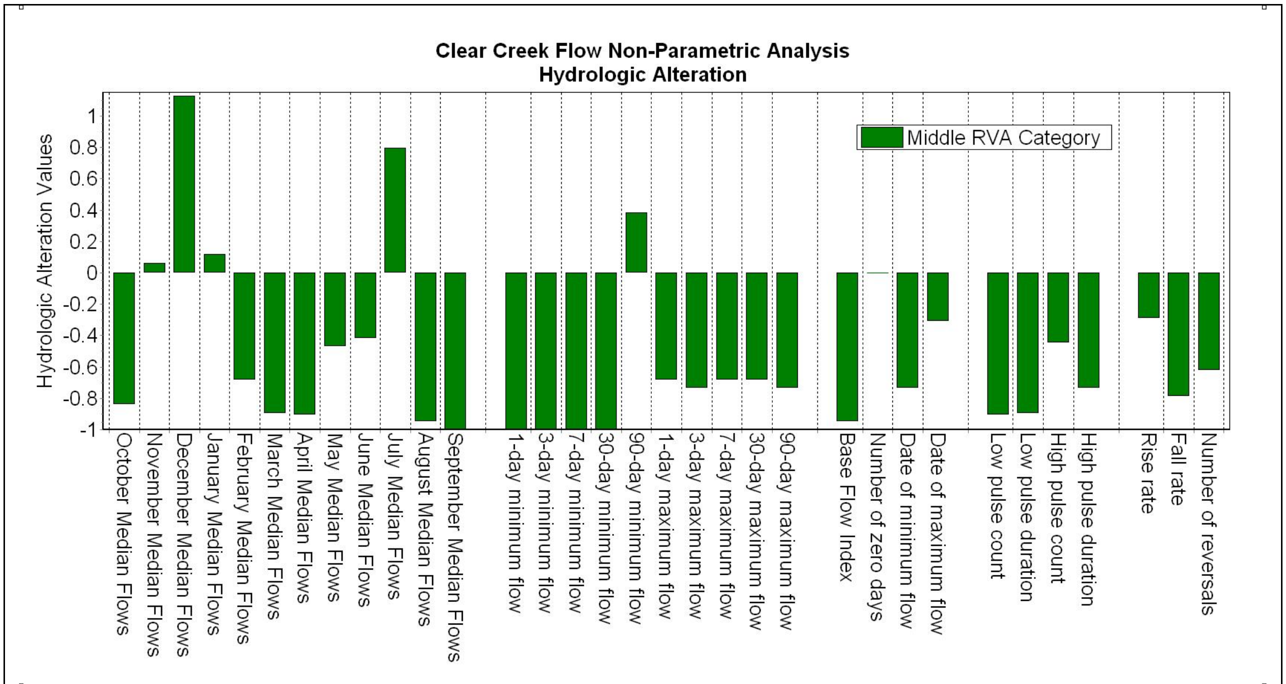


Figure 17. Hydrologic alterations for the middle RVA category (33rd to 67th percentiles)

7.2 Environmentally Relevant Flows in Clear Creek

In addition to the hydrologic parameters, the IHA software program also provides parameters for five types of Environment Flow Components (EFCs): large floods, small floods, high-flow pulses, low flows, and extreme low flows. This delineation of EFCs is based on the realization by ecologists that river hydrographs can be divided into a repeating set of hydrographic patterns that are ecologically relevant (The Nature Conservancy 2009).

In this report, all flows that exceed 75% of daily flows for the period are classified as initial high flows. All flows that are below 50% of daily flows for the period are classified as initial low flows. Between these two flow levels, a high flow will begin when flow increases by more than 25% per day, and will end when flow decreases by less than 10% per day.

A Small Flood event is defined as an initial high flow with a peak greater than the 2-year return interval event. A Large Flood event is defined as an initial high flow with a peak greater than the 10-year return interval event. All other initial high flows not classified as Small Floods or Large Floods will be classified as High-flow Pulses. An Extreme Low Flow is defined as an initial low flow below 10% of daily flows for the period. All other initial low flows not classified as Extreme Low Flows will be classified as Low Flows.

With the criteria described above, the thresholds for the five EFCs were computed for Clear Creek and listed in **Table 7**. The analysis results for environmental flow components are shown in **Figure 18**. Each of the five EFCs is discussed below.

Table 7. Thresholds for the five Environmental Flow Components in Clear Creek

EFC	Threshold (cfs)
Large flood	14,460
Small flood	5,165
High-flow pulse	420
Low flow	133
Extreme low flow	26

7.2.1 Large Flood

Large floods will typically re-arrange the biological, physical, and chemical structure of a river and its floodplain, including transport of significant amounts of sediment, large woody debris and other organic matter, formation of new habitats (*e.g.*, oxbow lakes and floodplain wetlands), changes in water quality conditions, and flushing of many organisms in the main channel and floodplain. There were three (3) large floods in Clear Creek from 1940 to 2009, each on March 1, 1941, December 22, 1956, and March 3, 1983, respectively (**Figure 18**).

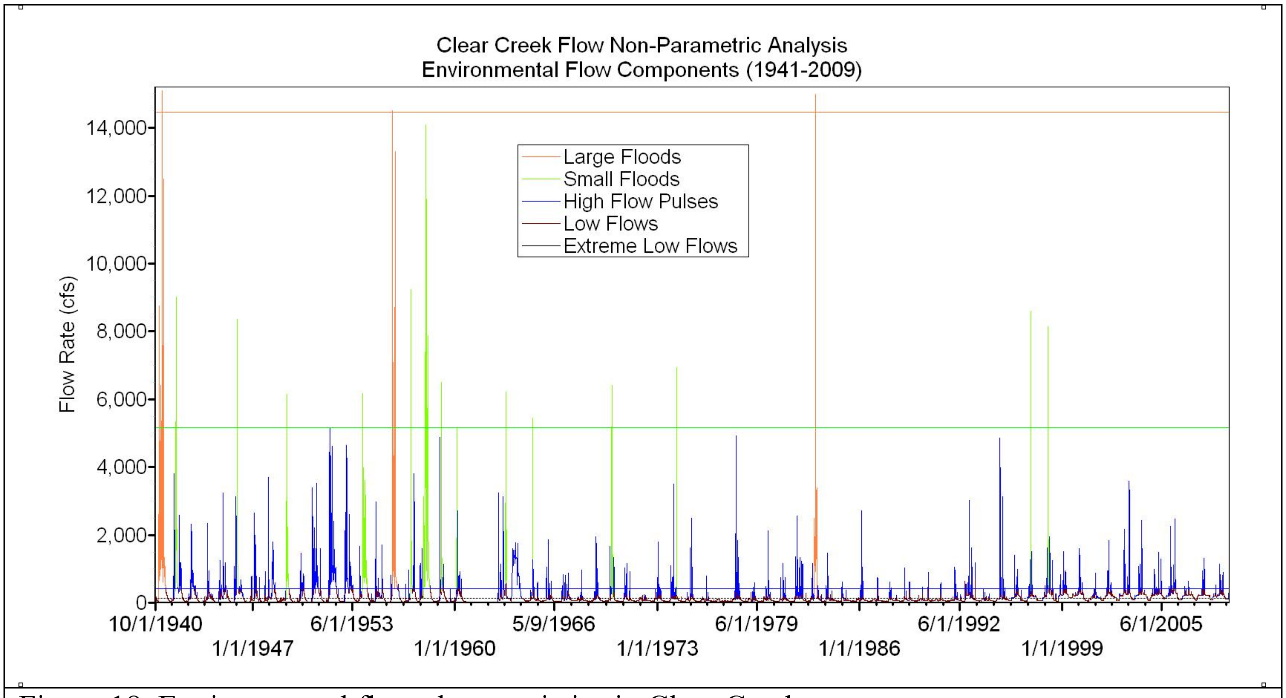


Figure 18. Environmental flow characteristics in Clear Creek

7.2.2 Small Flood

Small floods are river flows that overtop the main channel banks and occur frequently (*e.g.*, every 2 or so years). These floods allow fish and other mobile organisms to access floodplains and habitats, such as secondary channels, backwaters, sloughs, and wetlands. These areas can provide significant food resources allowing for fast growth, offer refuge from high-velocity, lower temperature water in the main channel, or be used for spawning or rearing. There were 21 small floods from 1941 to 1960 (pre-dam period), occurring once a year on average. On the contrary, there were 6 small floods from 1963 to 2009, occurring once every 8 years on average (**Figure 18**). The characteristics of the pre-dam small floods in Clear Creek are summarized in **Table 8**. Most pre-dam small floods occurred between late January and late March. The small flood duration spanned from 5 to 30 days. Their peak flows ranged from 5170 to 14100 cfs. The rise rates ranged from 485 to 8620 cfs/day, while the fall rates ranged from -2705 to -373 cfs/day.

Table 8. Characteristics of pre-dam small floods in Clear Creek

Feature	Median	Middle Range*
Small flood peak (cfs)	7100	6150 - 9240
Small flood duration (day)	11	7 - 14
Small flood timing (Julian day)	47	29 - 77
Small flood rise rate (cfs/day)	1865	1295 - 2745
Small flood fall rate (cfs/day)	-1011	(-1115) – (-696)

*Between 25th and 75th percentiles

7.2.3 High-flow Pulse

High-flow pulses may occur during rainstorms or brief periods of snowmelt. Water levels rise above low-flow levels but do not overtop the channel banks. For many organisms, these short-term changes in flow may provide necessary respite from stressful low-flow conditions. These pulses of freshwater may relieve higher water temperatures and low dissolved oxygen availability typical of low flow conditions, flush wastes, and deliver organic matter that nourishes the aquatic food web. High-flow pulses typically facilitate improved access to upstream or downstream areas for mobile organisms.

The characteristics of high-flow pulses in Clear Creek are presented in **Figures 19-20** and summarized in **Table 9**. High-flow pulses in Clear Creek occurred 3-7 times a year between late December and late February during the pre-dam period. The median duration of the high-flow pulses was 3.5 days with a median peak flow of 800 cfs, a rise rate of 422 cfs/day, and a fall rate of -139 cfs/day.

The magnitude of high-flow pulses decreased from 800 cfs in the pre-dam period to 400 cfs in the post-dam period, while the duration, timing, and change rates of high-flow pulses remained similar between the two periods.

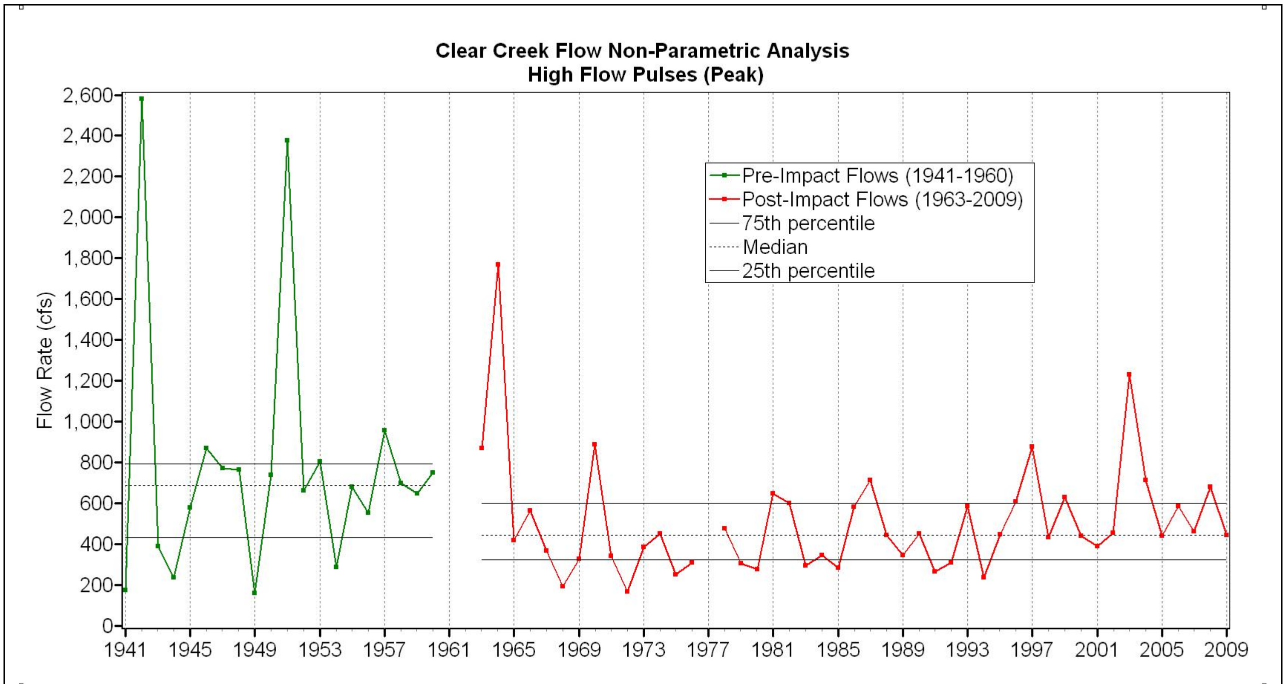


Figure 19. High flow pulse peak (cfs) in Clear Creek

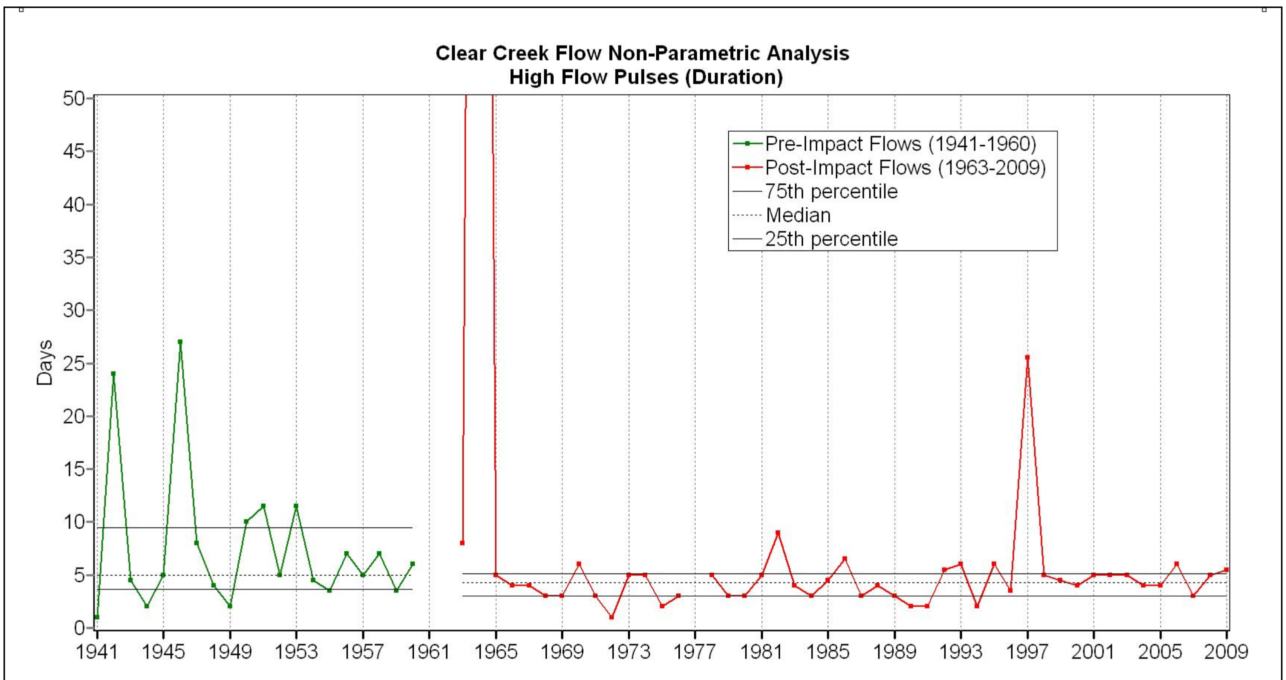


Figure 20. High flow pulse duration (days)) in Clear Creek

Table 9. Characteristics of pre-dam high-flow pulses in Clear Creek

Feature	Median	Middle Range*
High flow pulse peak (cfs)	689	431 - 795
High flow pulse duration (day)	5	3 - 9
High flow pulse timing (Julian day)	9	344 - 52
High flow pulse frequency	6	5 - 8
High flow pulse rise rate (cfs/day)	180	124 - 288
High flow pulse fall rate (cfs/day)	-75	(-99) – (-66)

* Between 25^{th} and 75^{th} percentiles

7.2.4 Low Flow

In natural rivers, after a rainfall event or snowmelt period has passed and associated surface runoff from the catchment has subsided, the river returns to its baseflow level. These low-flow levels are sustained by groundwater discharge into the river. The seasonally varying low flows in a river impose a fundamental constraint on a river's aquatic communities because it determines the amount of aquatic habitat available for most of the year. This has a strong influence on the diversity and number of organisms that can live in the river.

The EFC monthly low flows in Clear Creek are presented in **Table 10.** For the pre-dam period of 1941 to 1960, the lower flows occurred in July, August, and September while the higher flows occurred in March and April. For the post-dam period of 1991 to 2010, the highest flow occurred in March or April, whereas the lowest flow occurred in August. While the magnitude of flows from December to June was comparable between the two periods, the post-dam flows from July through November were higher than the pre-dam flows. The increased flows during the post-dam period reflect the CVPIA AFRP implementation (beginning 1999) for water temperature management in Clear Creek.

Inter-annual variations in monthly low flows are selectively presented in **Figure 21** (October) and **Figure 22** (March). The EFC median monthly low flows were lower than the overall IHA median monthly flows from December through June, while they were similar from July through November (**Figure 23**).

Table 10. Monthly low flows (in cfs) as percentiles in Clear Creek

Month	Pre-dam (1941-1960)			Post-dam (1991-2010)		
	50th	75th	90th	50th	75th	90th
October	39	50	80	207	217	264
November	59	100	176	212	215	237
December	123	214	301	226	269	321
January	158	297	391	272	292	337
February	232	348	370	272	277	330
March	301	377	404	271	299	388
April	291	398	413	247	303	336
May	241	346	380	237	274	296
June	112	220	255	171	207	223
July	50	88	104	108	153	211
August	33	47	57	88	116	205
September	33	42	48	156	198	221

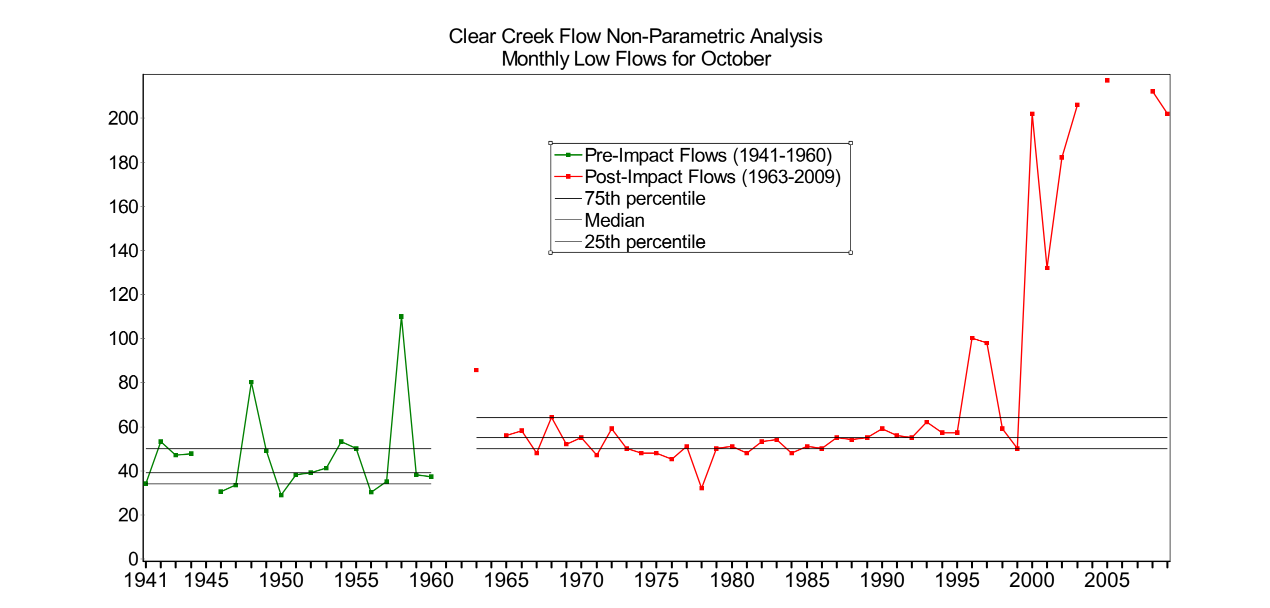


Figure 21. Monthly low flow in October in Clear Creek

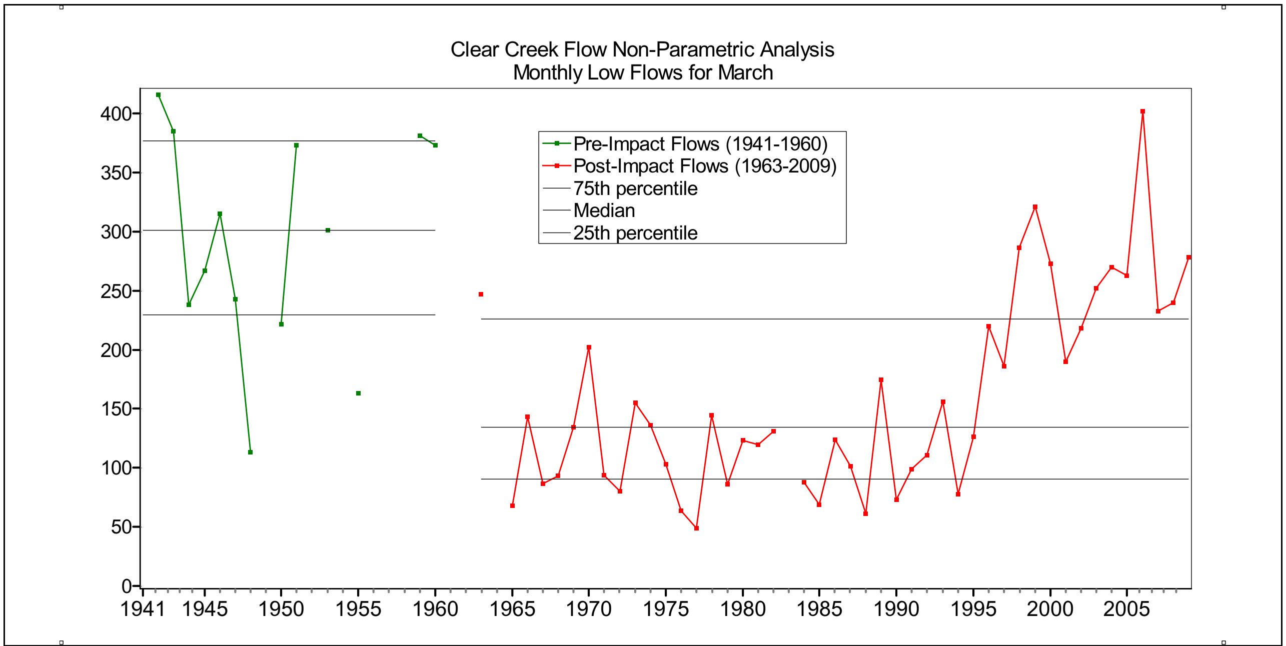


Figure 22. Monthly low flow in March in Clear Creek

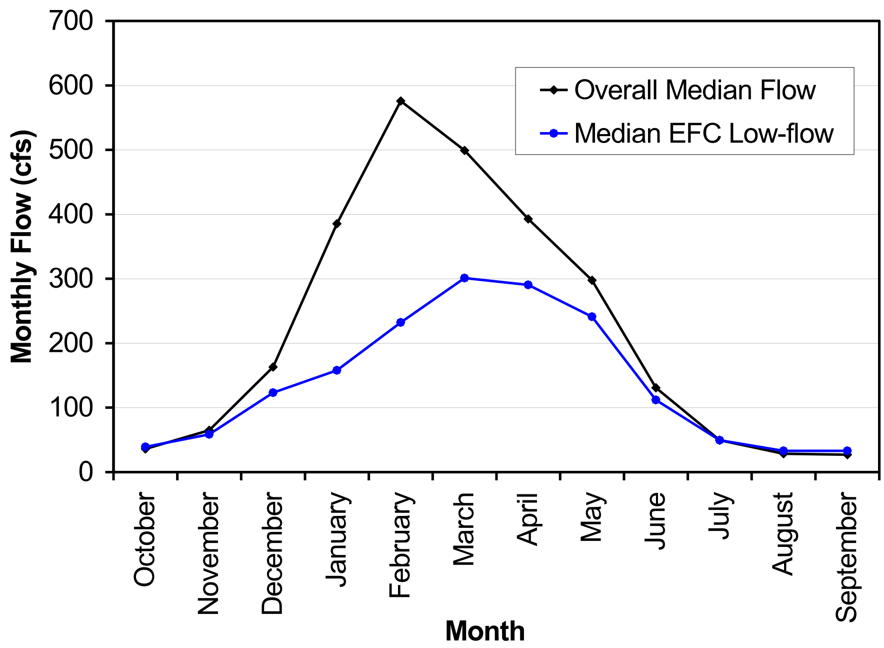


Figure 23. Pre-dam median EFC low-flows comparing with Overall median flows in Clear Creek

7.2.5 Extreme Low Flow

During drought periods, rivers drop to very low levels that can be stressful for many organisms, but may provide necessary conditions for other species. Water chemistry, temperature, and dissolved oxygen availability can become highly stressful to many organisms during extreme low flows, to the point that these conditions can cause considerable mortality. On the other hand, extreme low flows may concentrate aquatic prey for some species, or may be necessary to dry out low-lying floodplain areas and enable certain species of plants to regenerate.

The pre-dam EFC extreme low flows in Clear Creek are presented in **Table 11**. They occurred once a year on average in September, with a median extreme low flow of 21 cfs in Clear Creek. These pre-dam summer low flows may have caused elevated water temperatures in some areas of the creek, but the creek would provide refuge areas with a proper water temperature for fish through mechanisms such as the cover of debris and riparian vegetation and groundwater seepage. During the past 50 years of development, the geomorphology and riparian vegetation of Clear Creek have degraded to such a large degree that higher summer flows have to be maintained to provide adequate water temperatures for spring-run Chinook salmon. The potential adverse effects of these increased baseflows may cause changes in the composition and distribution of riparian vegetation species.

Table 11. Characteristics of pre-dam extreme low flows as percentiles in Clear Creek

Feature	25th	50th	75th
Extreme low flow peak (cfs)	17	21	23
Extreme low flow duration (day)	9	15	33
Extreme low flow timing (Julian day)	248	260	274
Extreme low flow frequency	1	1	3

8 Water Temperature in Clear Creek

8.1 Air and Water Temperature

Stream temperature can be affected by many factors including atmospheric conditions, topography, stream discharge, and streambed (Caissie 2006). Atmospheric conditions are the most important factors that are responsible for heat exchange occurring at the air-water interface, and to a lesser degree at the water-streambed interface. Topography or geographical setting influences atmospheric conditions. Stream discharge influences the heating capacity of a stream.

Air temperature (daily maximum and 7DADM) in Redding and water temperature (daily maximum and 7DADM) from 2002 to 2009 are presented in **Figure 24**. Temperatures were shown from June 1 to October 31as this period of time is identified to be critical for life stages of spring run Chinook salmon in Clear Creek and likely to have water temperature exceedance. Water temperature in Clear Creek changed with changes in air temperature, but water temperature variations were smaller than air temperature variations. When the data were transformed from daily maximum to 7DADM, their variations were further reduced for both water and air temperatures as displayed in whisker-box plots (**Figure 25**). The horizontal line crossing the inner box is the median. The circle below the median is the mean. The top of the inner box is the third quartile (Q3) – 75% of the data values are less than or equal to this value. The bottom of the inner box is the first quartile (Q1) – 25% of the data values are less than or equal to this value. The upper whisker extends to the highest data value within the upper limit:

$$\text{Upper limit} = Q3 + 1.5\,(Q3 - Q1)$$

The lower whisker extends to the lowest value within the lower limit:

$$\text{Lower limit} = Q1 - 1.5\,(Q3 - Q1)$$

Values beyond the whiskers are outliers, represented by “*”.

Cumulative distribution function graphs (**Figure 26**) were used to estimate the percentage of water temperatures that exceed the water temperature objectives: 60 ^{o}F from June 1 to September 15 and 56 ^{o}F from September 16 to October 31. The graphs include an empirical cumulative distribution function (ECDF) of the temperature data, and a fitted normal cumulative distribution function (CDF) (**Figure 26**). The stepped ECDF resembles a cumulative histogram without bars, while the fitted CDF is based on parameters estimated from the temperature data.

The percentage of days exceeding the water temperature objectives were obtained from **Figure 26** and are summarized in **Table 12**. Water temperature in July was the highest and 90% of the days exceeded 60 ^{o}F, which is the temperature protective of adult migration and holding. Ninety six percent of the temperature in September (16th-30th) exceeded 56 ^{o}F, which is the temperature protective of spawning.

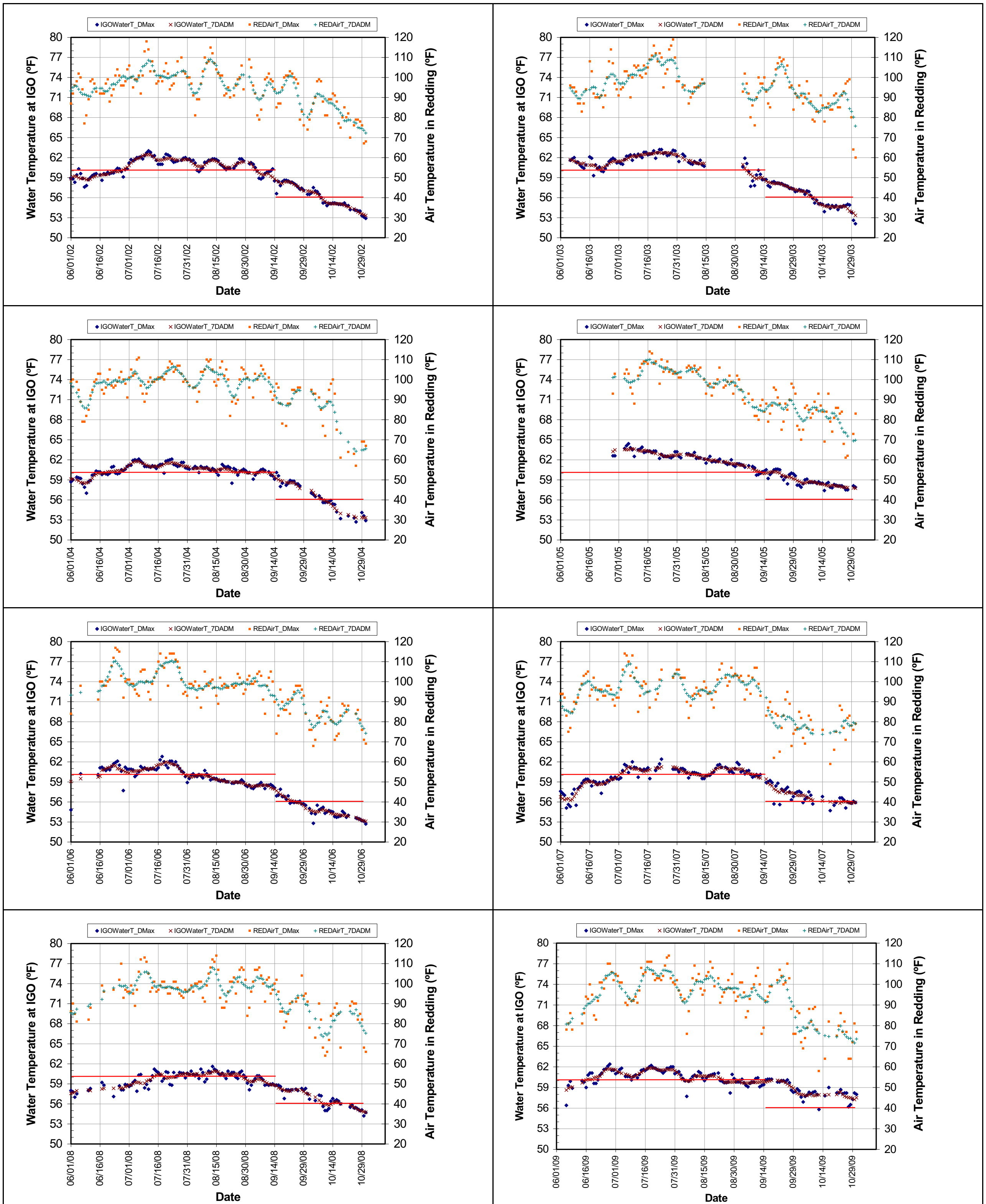


Figure 24. Water temperature at IGO and air temperature in Redding. DMax = Daily maximum. 7DADM = 7-day average of daily maximum. The red lines represent water temperature objectives: 60 ºF and 56 ºF, respectively.

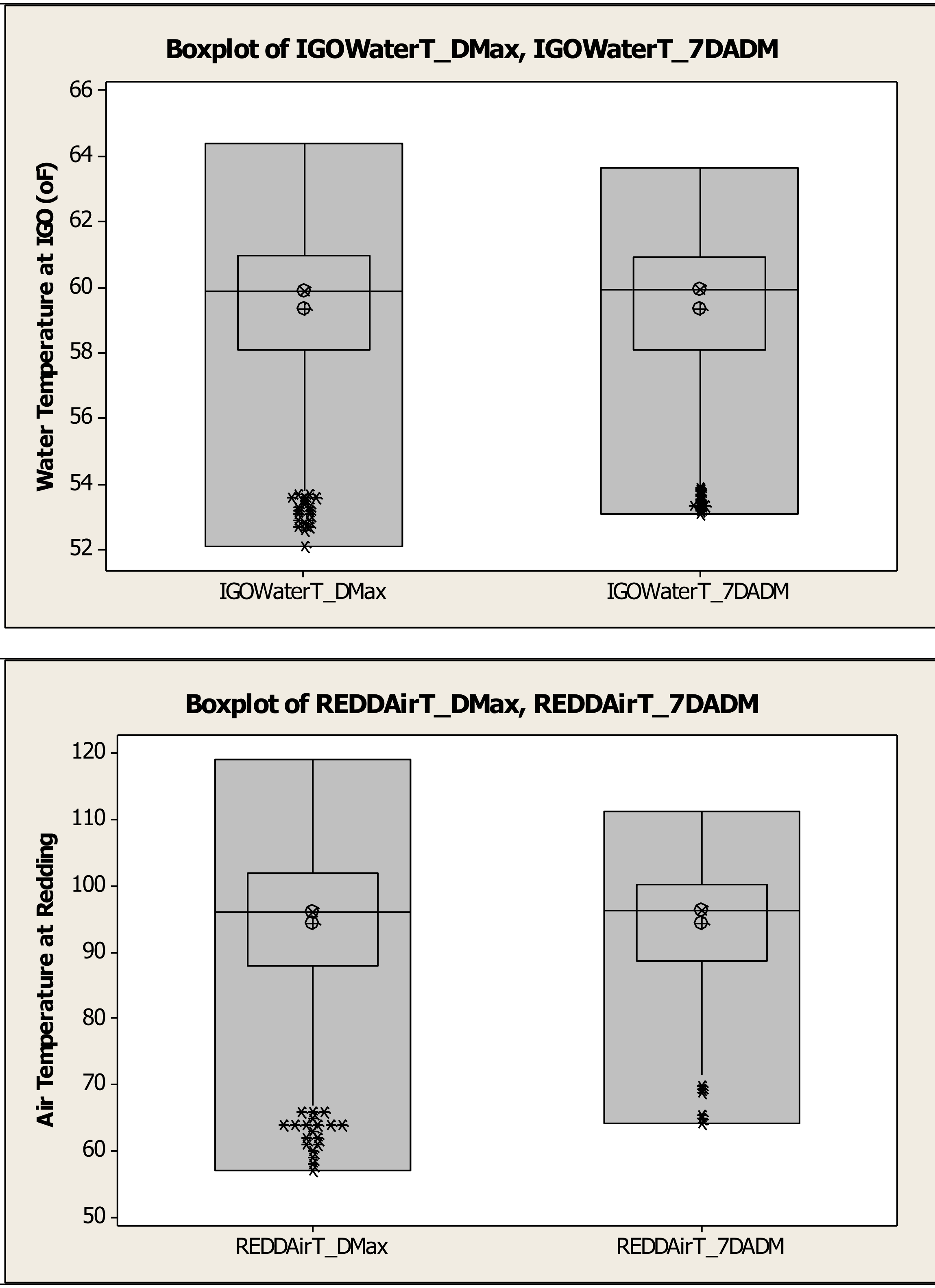


Figure 25. Box plot of water temperature at Igo in Clear Creek and air temperature in Redding

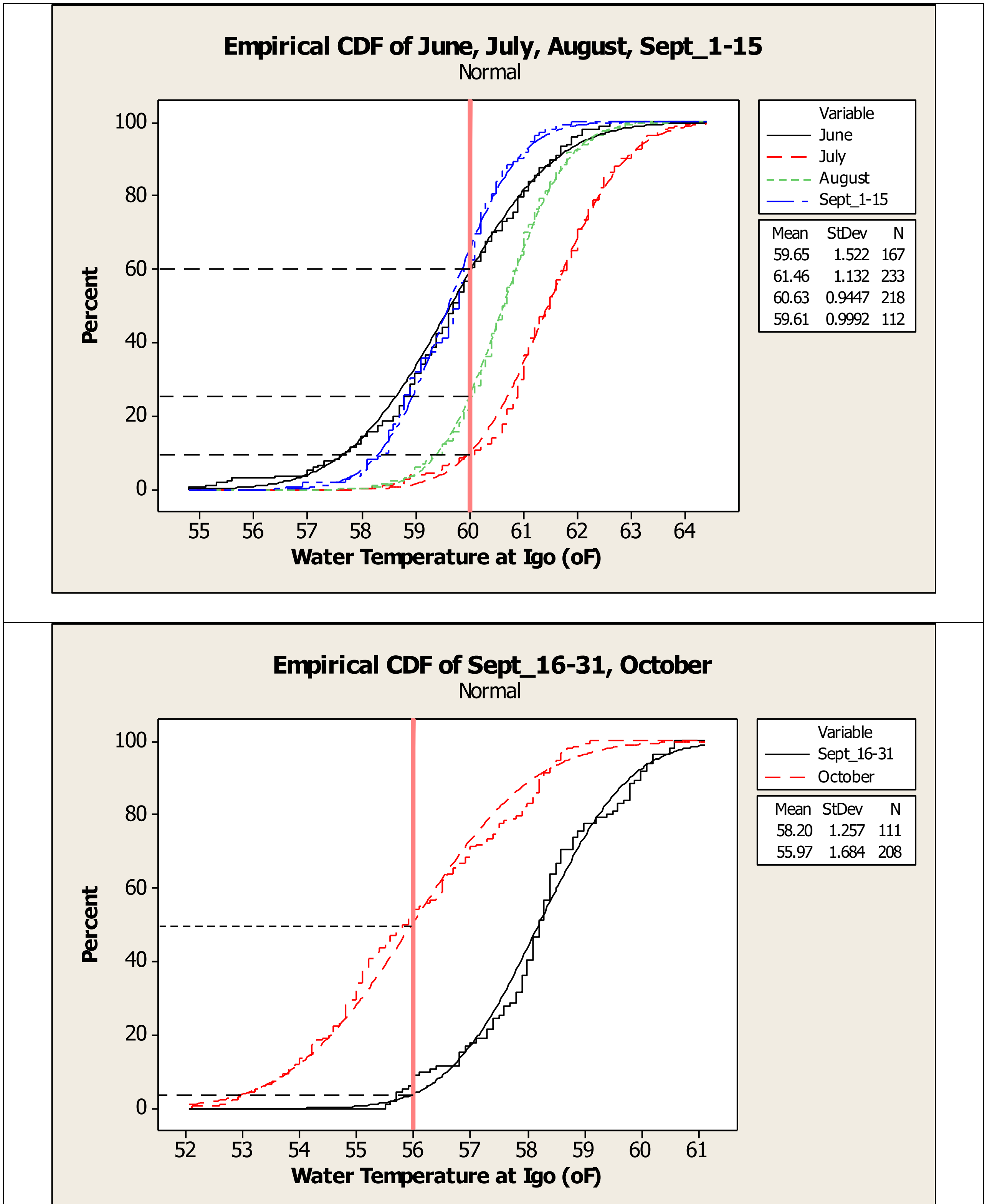


Figure 26. Cumulative distribution of water temperature at Igo in Clear Creek

Table 12. The percentage of days exceeding water temperature objectives in Clear Creek

Month	Days greater than 60 °F (%)	Days greater than 56 °F (%)
June	40	--
July	90	--
August	75	--
September 1-15	35	--
September 16-30	--	96
October	--	50

Dams directly modify river's thermal regimes by releasing water that differs greatly in temperature to that occurring naturally in the river. The magnitude of thermal alteration depends largely on the stratification of a reservoir, and the depth at which water is released from the reservoir. Dams also modify water temperatures indirectly by influencing processes controlling the delivery, distribution, and retention of heat within the river channel. Changes to discharge and the volume of water in a river, for example, affect the rate at which water heats and cools in response to natural diurnal heat exchange at the air-water and streambed-water interfaces (Olden and Naiman 2010).

From June 1 to October 31, streamflow in Clear Creek at Igo was largely controlled by the Whiskeytown reservoir release (**Figure 27**). Streamflow in June and October was generally greater than the reservoir release. This may indicate the contribution of lower evaporation and/or tributary flows downstream of the reservoir.

8.2 Regression Model for Water Temperature

The scatter plots of water temperature at Igo against reservoir release from Whiskeytown Dam and air temperature and solar radiation at Redding are presented in Figures 28-30. There is a negative correlation between water temperature and reservoir release, and a positive correlation of water temperature with air temperature and solar radiation. Higher reservoir release leads to lower water temperature downstream and higher air temperature or solar radiation leads to higher water temperature.

8.2.1 Model Development

Two steps were used to develop a regression model - possibly the best for prediction. The first step is to use all the observed data, which had been examined with the procedures as described in Section 6, to develop a model. The second step is to perform a diagnostic analysis of the established model.

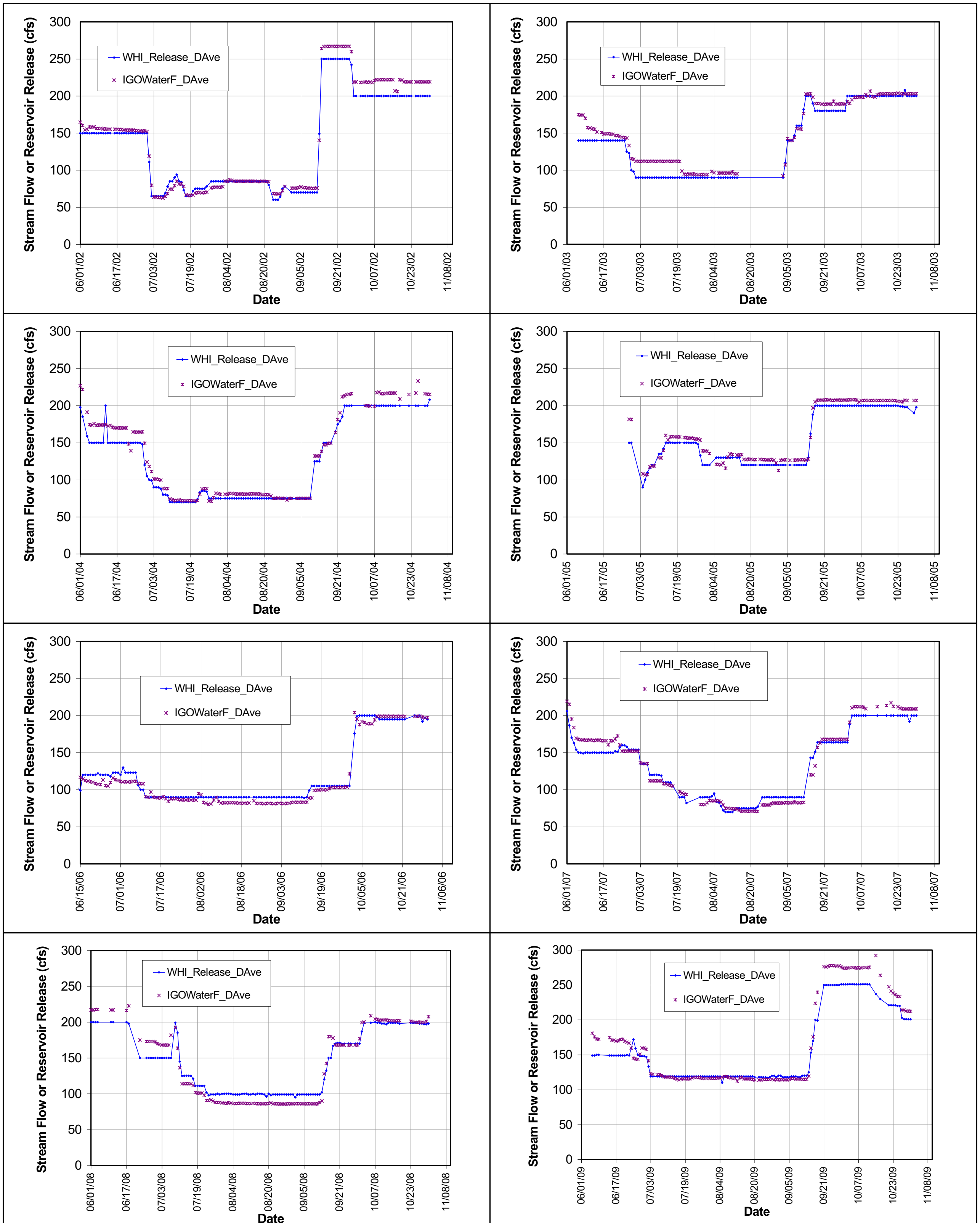


Figure 27. Streamflow at IGO in Clear Creek and reservoir release from Whiskeytown Dam. DAve = Daily average.

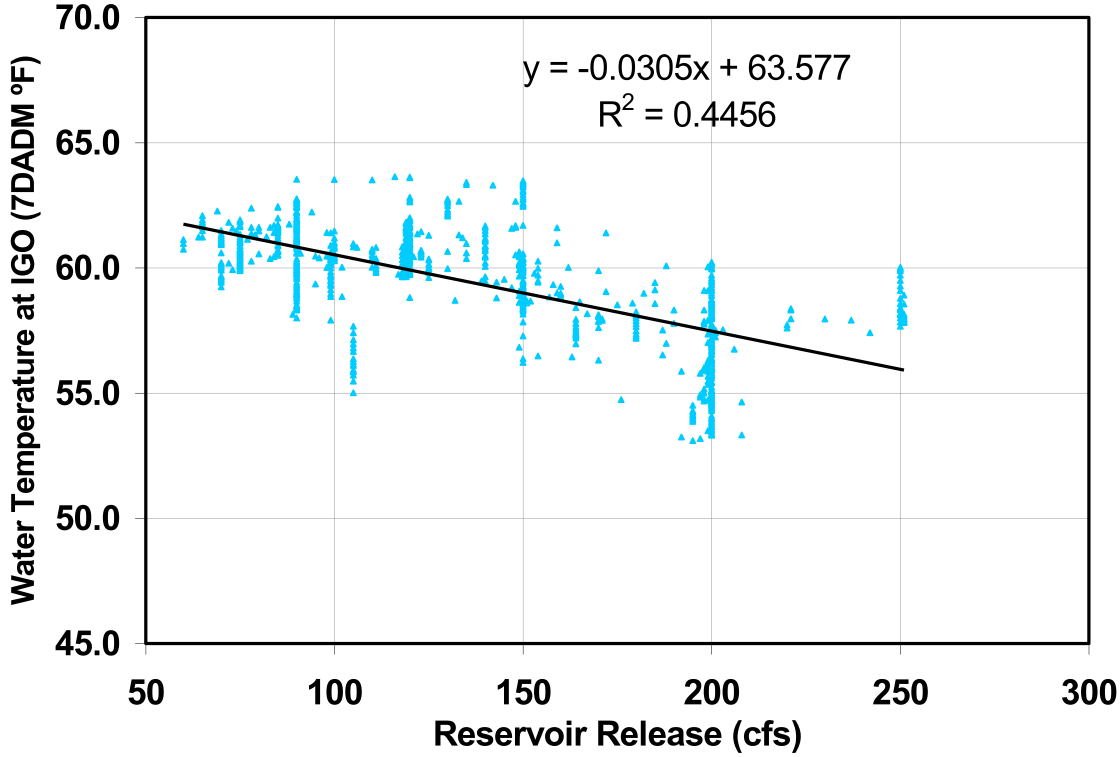


Figure 28. Correlation of water temperature at Igo with reservoir release

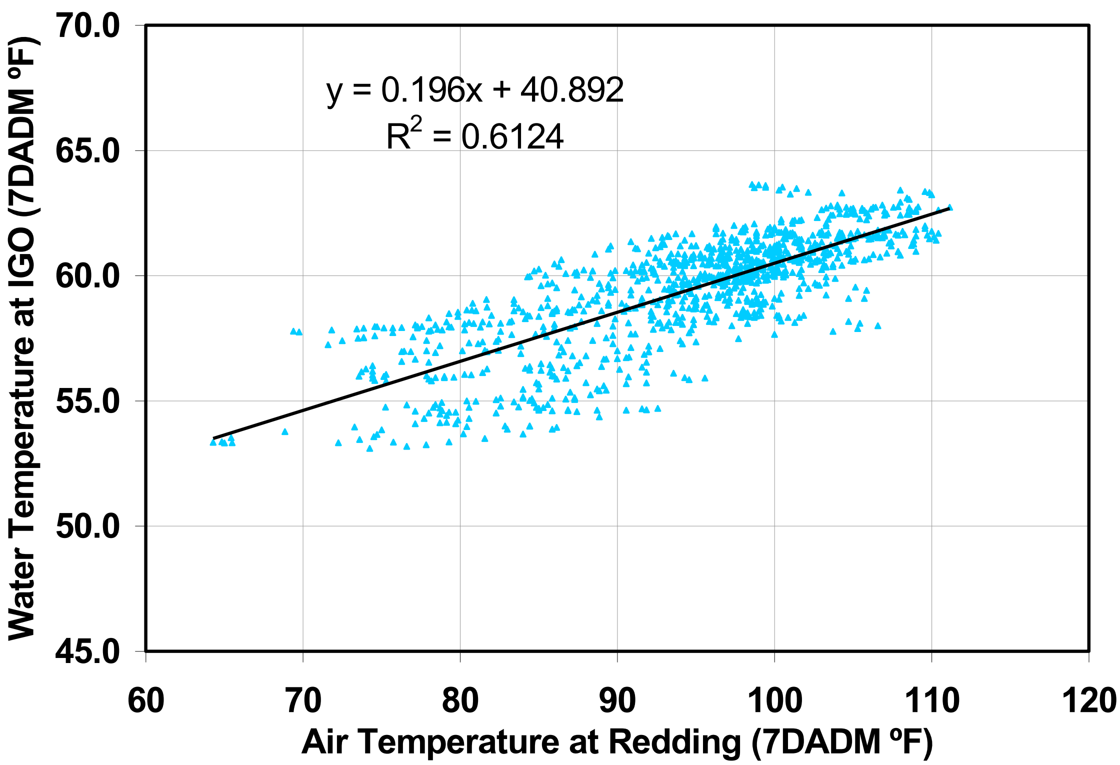


Figure 29. Correlation of water temperature at Igo with air temperature

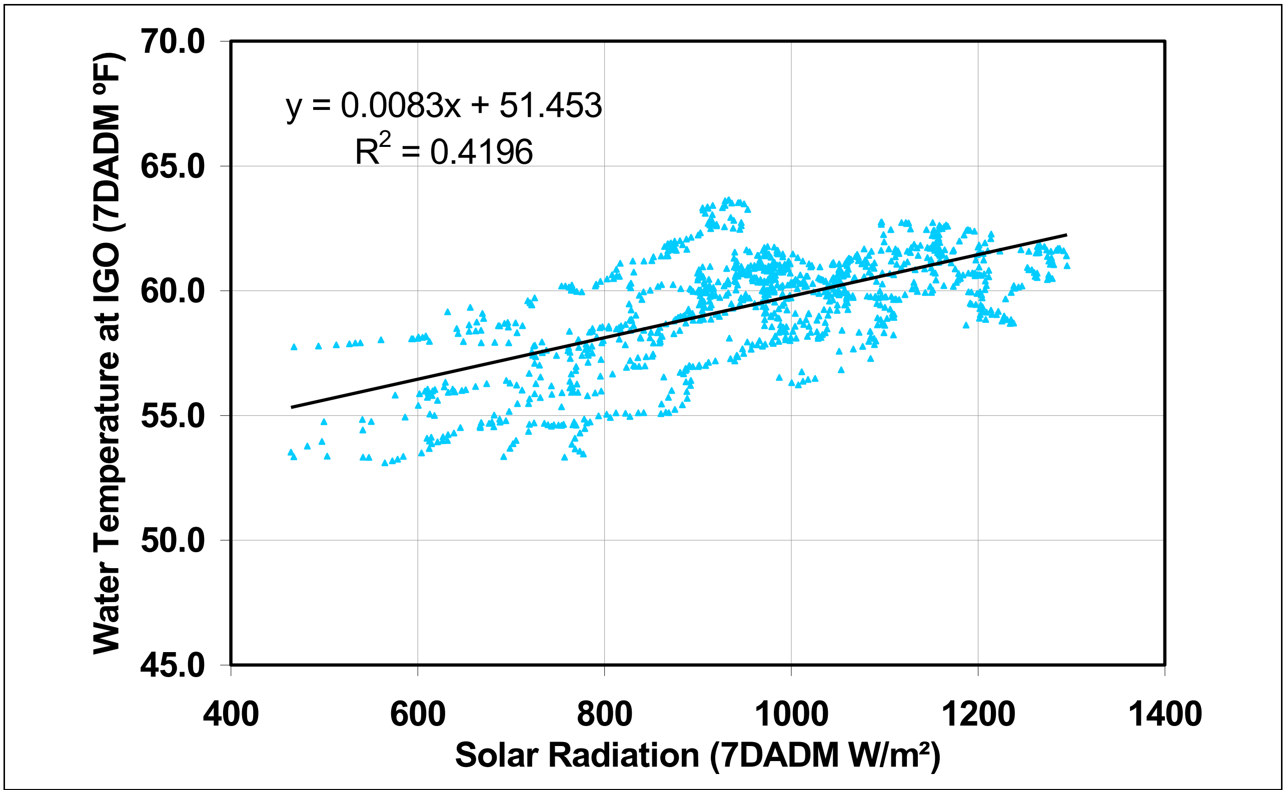


Figure 30. Correlation of water temperature at Igo with solar radiation

The multiple linear regression was used to fit a set of data with the following equation:

$$\hat{T}_w = a_0 + a_1 Q_r + a_2 T_a + a_3 R_s \qquad \text{Eq. 1}$$

where $\hat{T}_w$ = predicted water temperature (°F); Q_r = reservoir release (cfs); T_a = air temperature (°F); R_s = solar radiation; and a_0, a_1, a_2, a_3 = coefficients.

The stepwise regression procedure was used to identify the best subset of predictors from the candidate predictors: reservoir release, air temperature, and solar radiation. The stepwise regression alternates between adding and removing variables, checking significance of individual variables within and outside the model. Variables significant when entering the model will be eliminated if later they test as insignificant. The stepwise regression results are presented in (**Table 13**). The partial t-test was performed by comparing the t statistic for a slope coefficient to a student's t-distribution. For a two-sided test with $\alpha = 0.05$ and sample sizes n of 20 or more, the critical value of t is $|t| \approx 2$. Larger t-statistics (in absolute value) for a slope coefficient indicate significance (Helsel and Hirsch 2002). For all the three predictors in this study, the t-values (absolute values) were greater than 9 and their p-values were less than 0.0005.

Table 13. Multiple linear regression results

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Welcome to Minitab, press F1 for help.
Retrieving project from file: 'H:\01 CLEAR
CREEK\CLEAR_CREEK_TEMP_REGRESSION-7DADM_20100119.MPJ'
```

Stepwise Regression: IGOWaterT_7D versus WHI_Release_, REDAirT_7DAD, ...

```
  Alpha-to-Enter: 0.15  Alpha-to-Remove: 0.15
Response is IGOWaterT_7DADM on 3 predictors, with N = 922

Step                   1        2        3
Constant           40.76    46.32    46.48

REDAirT_7DADM     0.1982   0.1565   0.1243
T-Value            37.91    24.16    17.55
P-Value            0.000    0.000    0.000

WHI_Release_DAve          -0.0118  -0.0108
T-Value                    -10.03    -9.61
P-Value                     0.000    0.000

REDSolarR_7DADM                    0.00288
T-Value                               9.37
P-Value                              0.000

S                   1.44     1.37     1.31
R-Sq               60.97    64.82    67.89
R-Sq(adj)          60.92    64.74    67.79
Mallows Cp         198.0     89.9      4.0
PRESS            1926.82  1740.73  1592.23
R-Sq(pred)         60.78    64.57    67.59
```

Regression Analysis: IGOWaterT_7D versus WHI_Release_, REDAirT_7DAD, ...

```
The regression equation is
IGOWaterT_7DADM = 46.5 - 0.0108 WHI_Release_DAve + 0.124 REDAirT_7DADM
                  + 0.00288 REDSolarR_7DADM

Predictor              Coef    SE Coef      T      P    VIF
Constant            46.4780     0.6946  66.92  0.000
WHI_Release_DAve  -0.010826   0.001126  -9.61  0.000  1.715
REDAirT_7DADM      0.124302   0.007082  17.55  0.000  2.225
REDSolarR_7DADM   0.0028762  0.0003068   9.37  0.000  1.646


S = 1.31092   R-Sq = 67.9%   R-Sq(adj) = 67.8%
PRESS = 1592.23   R-Sq(pred) = 67.59%


Analysis of Variance

Source           DF      SS      MS       F      P
Regression        3  3335.7  1111.9  647.00  0.000
Residual Error  918  1577.6     1.7
Total           921  4913.3

Source            DF  Seq SS
WHI_Release_DAve   1  2087.0
REDAirT_7DADM      1  1097.6
REDSolarR_7DADM    1   151.0
```

8.2.2 Model Diagnostics

Three diagnostic tools were applied to identify the points of high leverage, influence, or outliers (Helsel and Hirsch 2002).

8.2.2.1 Leverage

Leverage measures the distance from an observation's x-value to the average of the x-values for all observations in a data set. The leverage (h_i) may be expressed as:

$$h_i = \frac{1}{n} + \frac{(x_i - \bar{x})^2}{\sum_{i=1}^{n}(x_i - \bar{x})^2} \qquad \text{Eq. 2}$$

where n = number of observations, x_i = the ith x value, and $\bar{x}$ = the mean of all x values.

Leverage values fall between 0 and 1. Observations with large leverage values may exert disproportionate influence on a model and produce misleading results. For example, a significant coefficient may appear to be insignificant. High leverage values are the values greater than 3p/n, where p is the number of coefficients (including the constant) and n is the number of observations. The 3p/n value in this study is 0.013 (p = 4 and n = 922).

By examining the leverage values, any observations with a value greater than 0.01 were excluded from the refined regression analysis described in 8.2.3.

8.2.2.2 DFFITS

DFFITS is a measure of the influence of each observation on the fitted values in a regression. Influential observations have a disproportionate impact on the model and can produce misleading results.

$$DFFTIS_i = \frac{e_{(i)}\sqrt{h_i}}{s_{(i)}} \qquad \text{Eq. 3}$$

where $e_{(i)}$ = prediction residual = $e_i/(1-h_i)$, and

$$s_{(i)} = \sqrt{\frac{(n-p)s^2 - [e_{(i)}^2/(1-h_i)]}{n-p-1}} \qquad \text{Eq. 4}$$

DFFITS represents roughly the number of standard deviations that the fitted value changes when each observation is removed from the data set and the model is refit. An observation is considered to have a high influence if $|DFFITS_i| \geq 2\sqrt{p/n}$, which is 0.13 for this study.

By examining the DFFITS values, any observations with a value greater than 0.12 were excluded from the refined regression analysis described in 8.2.3.

8.2.2.3 Standardized Residuals

The standardized residual equals the value of an actual residual, e_i, divided by an estimate of its standard deviation:

$$e_{si} = \frac{e_i}{s\sqrt{1-h_i}} \qquad \text{Eq. 5}$$

Standardizing residuals is useful because raw residuals can be poor indicators of outliers due to their nonconstant variance: residuals with corresponding x-values that are far from $\bar{x}$ have greater variance than residuals with corresponding x-values closer to $\bar{x}$. Standardizing controls for this nonconstant variance, and all standardized residuals have the same standard deviation. Since about 70% of the observations in 2005 showed standardized residuals greater than 1.5, the data from 2005 were excluded from the refined regression analysis described in 8.2.3.

8.2.3 Refined Regression Model

A new data set of observations (n = 730) was developed by excluding those observations of high leverage, influence, or possible outliers as described above. The multiple regression results are presented in **Table 14**. This refined model has been improved from the previous model. The overall quality of the regression models from the stepwise procedure can be evaluated by three statistics: Mallow's C_p, PRESS, and adjusted R^2 (R_a^2) (Helsel and Hirsch 2002). The Mallow's C_p is defined as

$$C_P = p + \frac{(n-p)(s_p^2 - \hat{\sigma}^2)}{\hat{\sigma}^2} \qquad \text{Eq. 6}$$

where n = the number of observations, p = the number of coefficients (number of explanatory variables plus 1), s_p^2 = the mean square error (MSE) of this p coefficient model, $\hat{\sigma}^2$ = the best estimate of the true error that is the minimum MSE among all possible models. The best model is the one with the lowest C_p value. The Cp values were 649, 284, and 4 for one-, two-, and three-parameter models, respectively (**Table 14**).

The PRESS statistic is the prediction error sum of squares. PRESS uses n−1 observations to develop the equation, then estimates the value of the one left out. It then changes the observation left out, and repeats the process for each observation. The prediction errors are squared and summed. Minimizing PRESS means that the equation produces the least error when making new predictions. In multiple regression it is a very useful estimate of the quality of possible regression models. The PRESS values were 999, 735, and 531 for one-, two-, and three-parameter models, respectively (**Table 14**).

The R_a^2 is the R^2 value adjusted for the number of explanatory variables (or equivalently, the degrees of freedom) in the model. This adjustment is important because the R^2 for any model will always increase when a new term is added. A model with more terms may appear to have a better fit simply because it has more terms. However, some increases in R^2 may be due to chance alone.

Table 14. Refined regression results

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Stepwise Regression: IGOWaterT_7D versus WHI_Release_, REDAirT_7DAD, ...

```
Alpha-to-Enter: 0.15  Alpha-to-Remove: 0.15

Response is IGOWaterT_7DADM on 3 predictors, with N = 730

Step                     1        2         3
Constant             39.56    40.03     48.02

REDAirT_7DADM       0.2098   0.1583    0.0946
T-Value              39.20    28.40     15.59
P-Value              0.000    0.000     0.000

REDSolarR_7DADM            0.00447   0.00456
T-Value                      16.26     19.55
P-Value                      0.000     0.000

WHI_Release_DAve                    -0.01569
T-Value                               -16.80
P-Value                                0.000

S                     1.17     1.00     0.850
R-Sq                 67.85    76.43     83.03
R-Sq(adj)            67.81    76.37     82.96
Mallows Cp           649.0    284.1       4.0
PRESS              999.416  734.557   531.198
R-Sq(pred)           67.65    76.23     82.81
```

Regression Analysis: IGOWaterT_7D versus WHI_Release_, REDAirT_7DAD, ...

```
The regression equation is
IGOWaterT_7DADM = 48.0 - 0.0157 WHI_Release_DAve + 0.0946 REDAirT_7DADM
                  + 0.00456 REDSolarR_7DADM

Predictor                Coef    SE Coef       T      P    VIF
Constant              48.0189     0.6046   79.42  0.000
WHI_Release_DAve   -0.0156865  0.0009339  -16.80  0.000  1.917
REDAirT_7DADM        0.094576   0.006066   15.59  0.000  2.426
REDSolarR_7DADM     0.0045649  0.0002334   19.55  0.000  1.478

S = 0.849931   R-Sq = 83.0%   R-Sq(adj) = 83.0%

PRESS = 531.198   R-Sq(pred) = 82.81%

Analysis of Variance
Source           DF       SS      MS        F      P
Regression        3  2565.38  855.13  1183.76  0.000
Residual Error  726   524.45    0.72
Total           729  3089.83

Source            DF   Seq SS
WHI_Release_DAve   1  1737.70
REDAirT_7DADM      1   551.44
REDSolarR_7DADM    1   276.24
```

The R_a^2 is a useful tool for comparing the explanatory power of models with different numbers of predictors. The R_a^2 will increase only if the new term improves the model more than would be expected by chance. It will decrease when a predictor improves the model less than expected by chance. The model with the highest R_a^2 is identical to the one with the smallest standard error (s) or its square - the mean squared error (MSE). When p is considerably smaller than n, R_a^2 is a less sensitive measure than either PRESS or Cp. PRESS has additional advantage of being a validation criteria. The R_a^2 values in this study are 0.6781, 0.7637, and 0.8296 for one-, two-, and three-parameter models, respectively (**Table 14**).

In addition, the variance inflation factor (VIF) is calculated to diagnose multi-collinearity. Multi-collinearity is the condition where at least one explanatory variable is closely related to one or more other explanatory variables. If there is no collinearity, VIF ≈ 1. Serious problems are indicated when VIF greater than 10 (Helsel and Hirsch 2002). The VIF values in this study are less than 2.4 for all the three parameters (**Table 14**).

Based on the analysis of the statistics provided above, all the three parameters – reservoir release, air temperature, and solar radiation are significantly related to water temperature at Igo. The regression coefficients are $a_0 = 48.02$, $a_1 = -0.0157$, $a_2 = 0.0946$, and $a_3 = 0.00456$. **Figure 31** shows the predicted water temperature at Igo plotted against the observed water temperature.

The prediction interval is the confidence interval for prediction of an estimate of an individual response variable. For example, the 95% prediction interval indicates that 95% of the time the predicted value will be within the interval. Most of the observations are between the upper and lower prediction interval lines (**Figure 31**).

8.2.4 Model Performance Evaluation

The performance of the regression model was evaluated using the following diagnostics: normality and random distribution of the residuals. Linear regression theory assumes residuals are normally distributed and symmetric about the mean. The histogram of the residuals (**Figure 32**) shows that the residuals appear to be normally distributed, and centered on zero. The probability plot of the residuals (**Figure 33**) shows a slight departure from normality.

Plotting residuals against predicted data helps to examine if the variance of residuals are constant. **Figure 34** presents the residuals plot against the predicted water temperature that shows no curvature or changing variance in residuals, whereas residuals appear to change from negative to positive when plotted against the observed water temperature (**Figure 35**).

An empirically developed multiple linear regression model may fit the data used to estimate the regression coefficients very well, but it is unknown if the model predicts new data well. The model was validated using observations not used in fitting the regression to assess the ability of the model to predict future events. **Figure 36** shows the observed and validated water temperatures at Igo for 2008. The model predicts the 2008 data very well ($R^2 = 0.9071$).

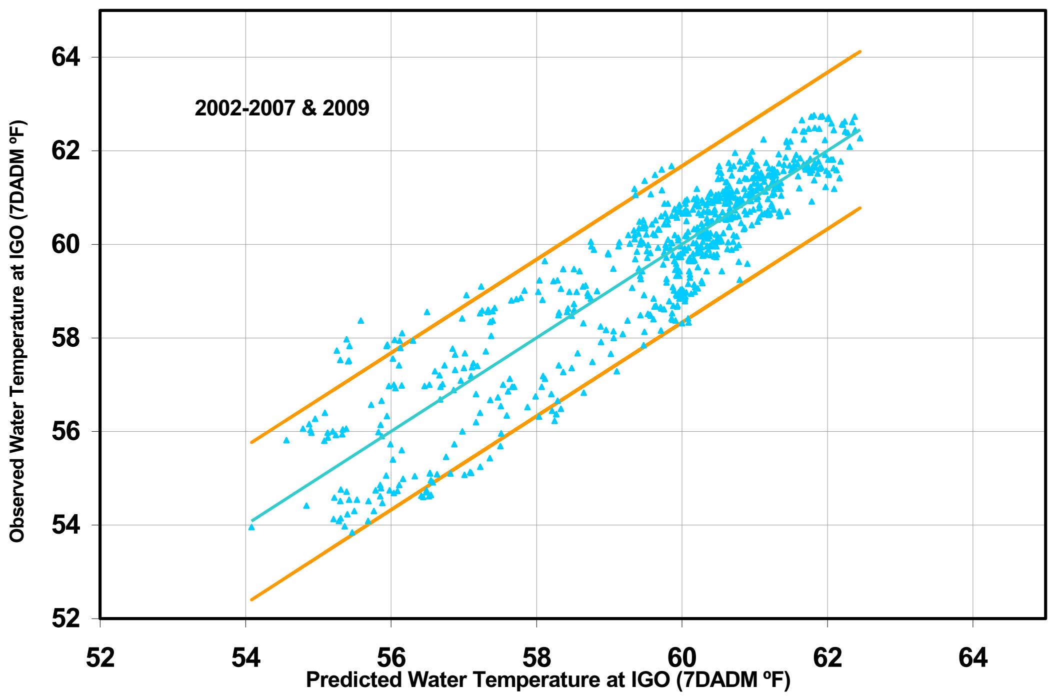

Figure 31. Predicted versus observed water temperature at Igo in Clear Creek. The orange lines represent the 95% prediction interval

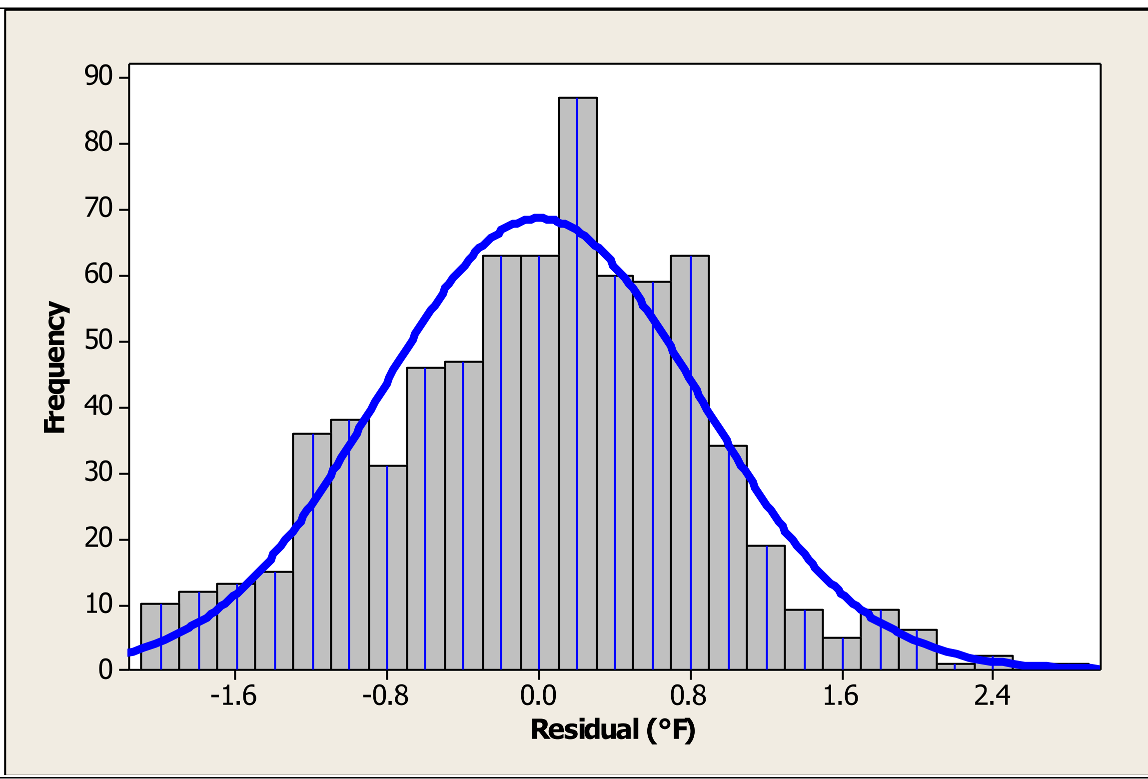

Figure 32. Water temperature prediction residual histogram

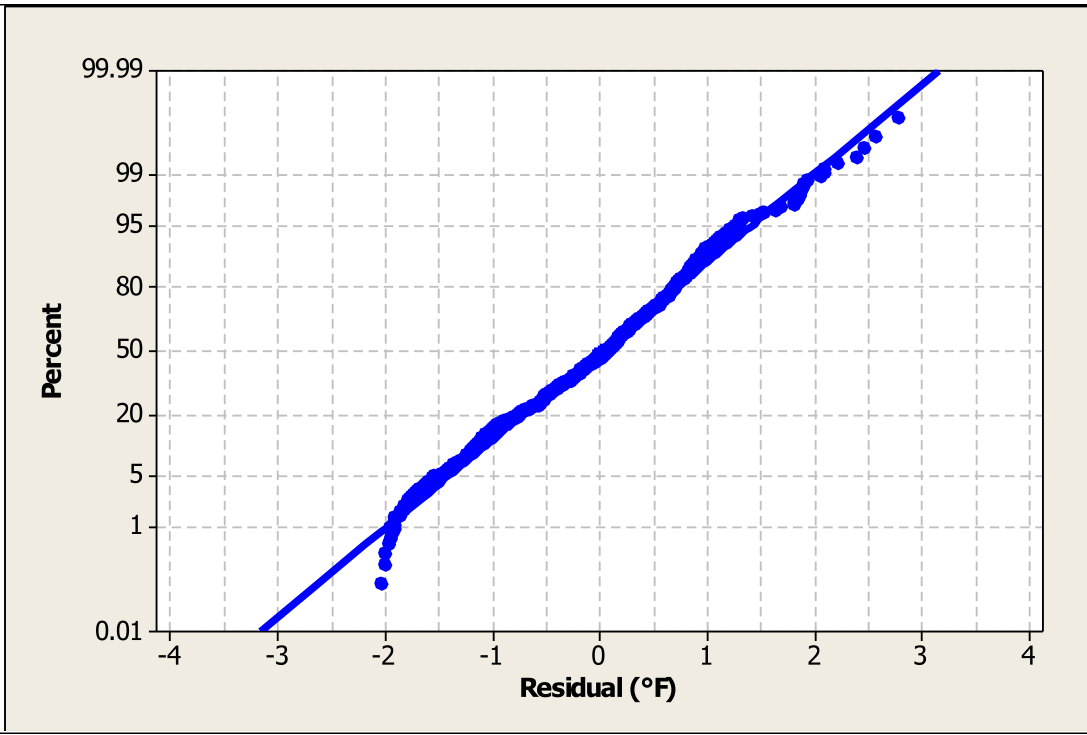

Figure 33. Normal probability plot of water temperature prediction residuals

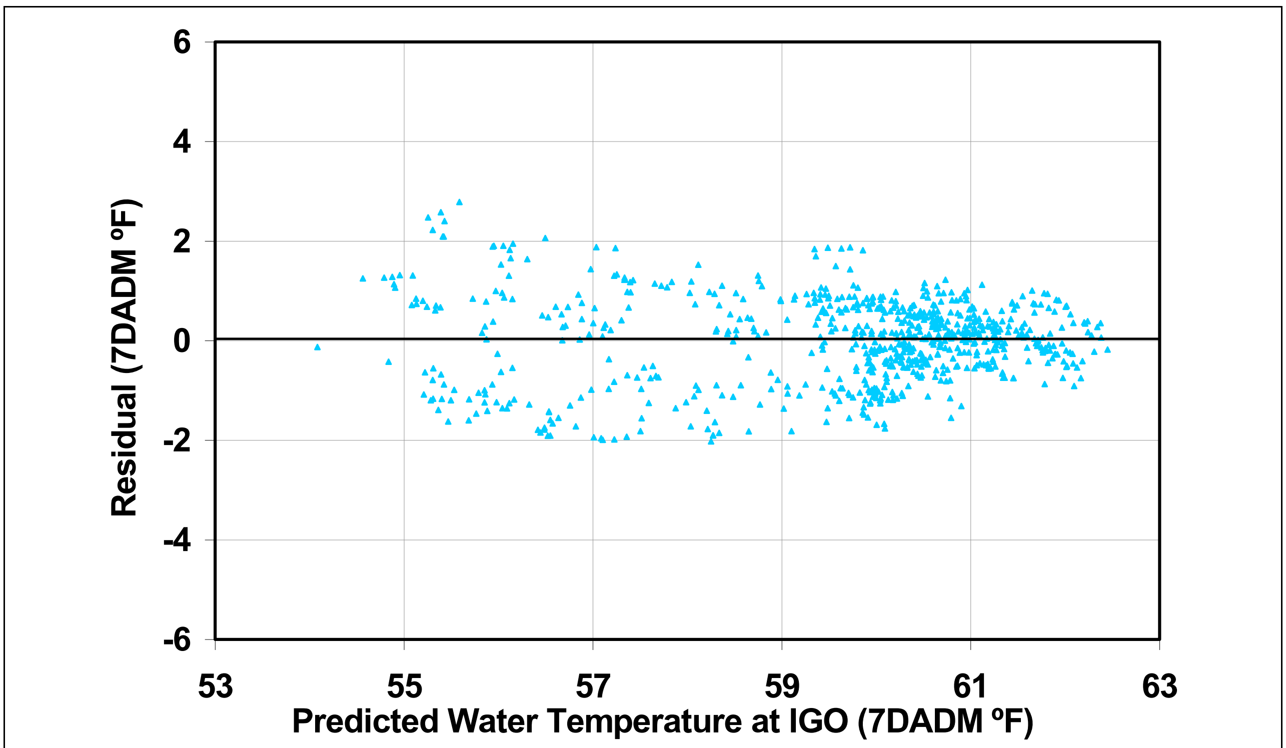

Figure 34. Residual plotted against the predicted water temperature

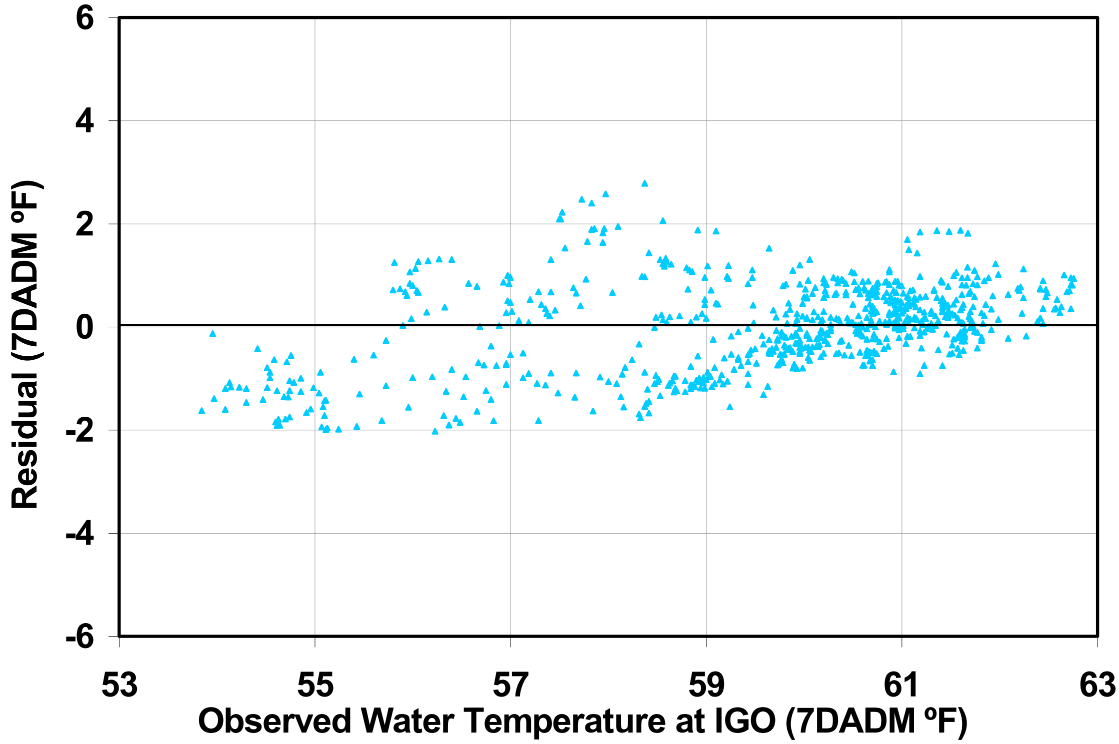

Figure 35. Residual plotted against the observed water temperature

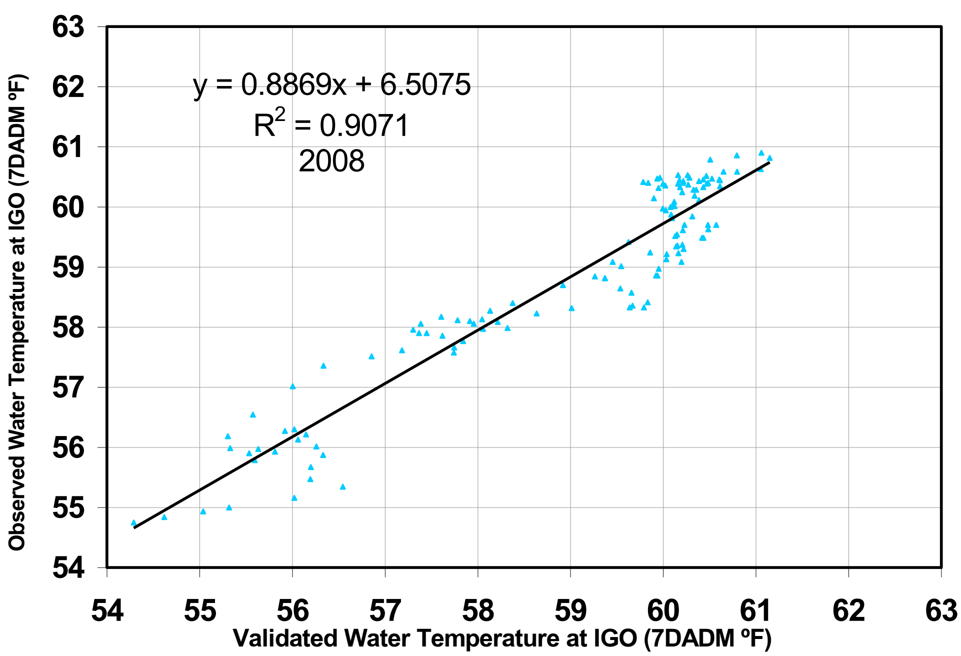

Figure 36. Observed and validated water temperature at Igo in Clear Creek in 2008

8.3 Estimation of Reservoir Release to Meet Temperature Requirements

The regression model to predict water temperature from reservoir release, air temperature, and solar radiation is used to estimate additional water required to lower the water temperature to a specified target.

Table 15. Estimated reservoir release using percentile air temperature and solar radiation. $T_{w(target)} = 60\,^{o}F$

Percentile	Air Temperature (7DADM ^{o}F)	Solar Radiation (7DADM W/m^{2})	Estimated Release (cfs)
June			
50	95.8	1100	134
70	98.58	1159	168
75	99.42	1177	178
80	100.26	1195	188
90	102.59	1244	217
95	104.51	1285	240
99	108.12	1361	284
July			
50	102.37	1091	171
70	104.62	1152	202
75	105.3	1170.5	212
80	105.98	1189	221
90	107.86	1241	247
95	109.42	1283	269
99	112.34	1363	310
August			
50	98.27	1017	125
70	100.36	1076	154
75	100.99	1093	163
80	101.62	1110	172
90	103.37	1158	196
95	104.81	1198	217
99	107.52	1272	254
September 1-15			
50	95.84	951	91
70	97.81	1005	118
75	98.41	1022	127
80	99.01	1039	135
90	100.66	1085	159
95	102.03	1123	178
99	104.6	1194	214

Rearranging Eq. 1 to solve for reservoir release ($\hat{Q}_r$) gives

$$\hat{Q}_r = \frac{T_{w(t\,\mathrm{arg}et)} - a_0 - a_2 T_a - a_3 R_s}{a_1} \qquad \text{Eq. 7}$$

where $T_{w(t\,\mathrm{arg}et)}$= specified water temperature target. Since there was a wide range of T_a and R_s, their percentile values were computed and used to estimate the reservoir release to meet a specified temperature target (**Table 15 and Table 16**).

The estimated reservoir release in cfs is the percentile flow that considers air temperature and solar radiation. For example, the estimated release of 240 cfs from Whiskeytown Dam in June would make 95% of water temperatures (7DADM) at Igo lower than 60 °F in June (Table 15), while the estimated release of 324 cfs from Whiskeytown Dam in the second half of September would make 75% of water temperatures (7DADM) at Igo lower than 56 °F from September 16 to 30 (**Table 16**).

Table 16. Estimated reservoir release using percentile air temperature and solar radiation. $T_{w(t\,\mathrm{arg}et)}$= 56 °F

Percentile	Air Temperature (7DADM °F)	Solar Radiation (7DADM W/m^2)	Estimated Release (cfs)
September 16-30			
50	88.44	870	277
70	91.74	924	313
75	92.73	940	324
80	93.73	957	335
90	96.50	1003	365
95	98.79	1040	389
99	103.08	1111	436
October			
50	82.35	757	208
70	85.16	817	242
75	86.01	835	253
80	86.86	853	263
90	89.22	903	292
95	91.17	945	316
99	94.82	1023	360

8.4 Integrated Flow Estimates

So far, we developed two parts of flows – one is based on the flow regime approach and the other is based on water temperature requirements from June through October. The overall streamflow for protecting listed fish species is the integration of the flow derived from the flow regime approach with the temperature sustaining flow (**Figure 37**). Flows were increased from June through October to meet water temperature requirements. The highest increase in flow was in the second half of September due mainly to high air temperatures in September and a more stringent water temperature criterion (56 ºF) for spawning. The high flow increase in July resulted from high air temperatures.

Flows showing in **Figure 37** may not be appropriate for fish species as they fluctuate considerably during the warm time period. These flows should be adjusted to remove those flow fluctuations. The adjustment procedure is discussed in Section 9.

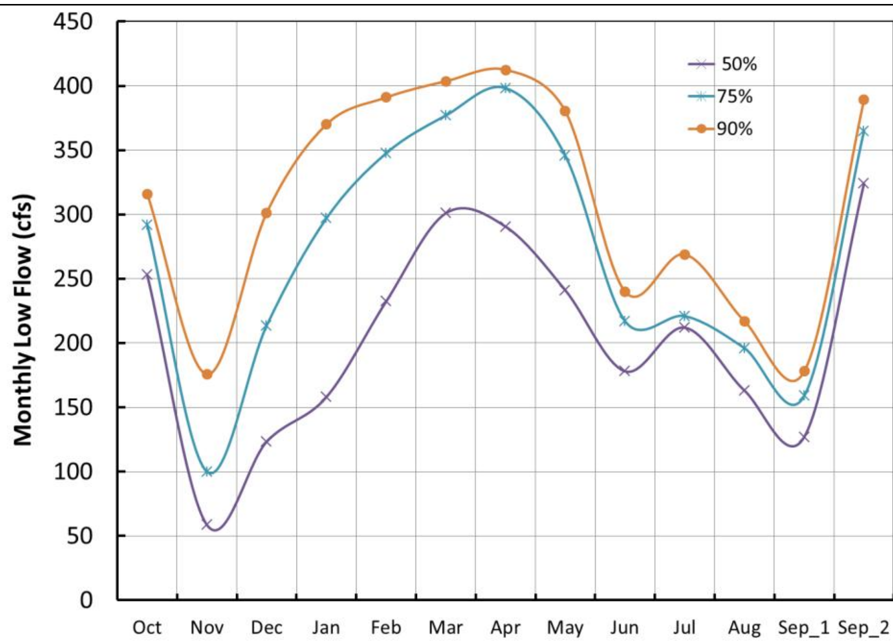

Figure 37. Monthly flow estimates based on the flow regime approach and water temperature requirements in Clear Creek

9 Recommendation of Flows in Clear Creek

Both streamflow and water temperature are imperative to the survival and growth of salmonids in Clear Creek. Streamflow in Clear Creek should be maintained to 1) mimic the natural flow regime with appropriate magnitude, timing, duration, frequency, and rise/fall rate because salmonids have evolved under these natural flow regimes, 2) provide suitable water temperature for holding and spawning adults and rearing juveniles, and 3) provide suitable physical habitat for spawning and rearing.

9.1 Instream Flow

As presented in Section 7, the analysis of flow data provides a flow pattern that mimics the "natural" flow in Clear Creek before the Whiskeytown Dam was built. To meet water temperature requirements, those flows need to be increased in order to lower water temperatures from June through October as discussed in Section 8. The integration of flows from the flow regime approach and flows for meeting water temperature requirements provides instream flows in Clear Creek. The integrated flows were adjusted to remove some considerable fluctuations. The principle of the flow adjustment is to keep higher flows unchanged and to increase lower flows for smoothing out flow patterns (**Figure 38**). For example, the highest flow would occur in March. The high flows in July and the second half of September were not changed. The low flows in June, the first half of September, and November were increased. Flows in other months were adjusted slightly (either increase or decrease) to have a similar pattern for the three flow curves (**Figure 38**).

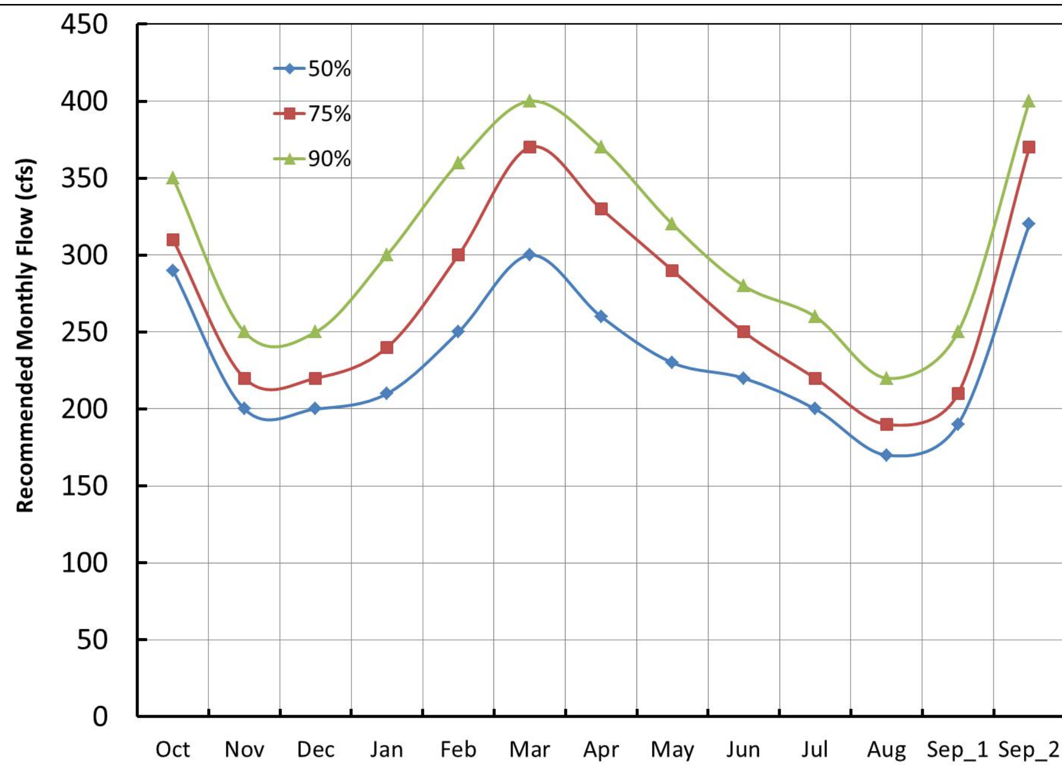

Figure 38. Recommended monthly instream flows in Clear Creek

The instream flows from November through May are based on the 50th, 75th, and 90th percentile baseflows (**Table 10**), while the instream flows from June through October are based on the 75th, 90th, and 95th percentiles meeting water temperature requirements. Even though water temperature criteria would need to apply nearly all the time, US EPA indicated that it is reasonable not to apply the numeric temperature criteria during unusually warm conditions for purposes of determining if a waterbody is attaining criteria. One way to do this is to exclude water temperature data when the air temperature during the warmest week of the year exceeds the 90th percentile for the warmest week of the year based on a historical record (*e.g.*, 10 years or more) at the nearest weather reporting station (USEPA 2003). **Table 17** presents the recommended monthly instream flows in Clear Creek.

Table 17. Recommended monthly instream flows in Clear Creek

Month	Minimal (50th %ile)	Desirable (75th %ile)	Optimal (90th %ile)
October	290	310	350
November	200	220	250
December	200	220	250
January	210	240	300
February	250	300	360
March	290	360	390
April	250	320	370
May	240	290	320
June	220	250	280
July	205	220	260
August	170	190	220
September 1-15	190	210	250
September 16-30	320	370	400

9.2 Channel Flushing Flow

Channel flushing flows are high-flow pulses. They are greater in magnitude than baseflows but lower than small floods. They are usually shorter in duration (a few days) and contained within the channel at or above the half of the bankfull discharge level for most streams. Channel flushing flows are designed to remove fine sediments, organic matter, and detritus from the interstitial voids in channel coarse substrates and depositional areas (Locke *et al.* 2008). These flows also facilitate improved access to upstream or downstream areas for adult immigration and juvenile outmigration.

The magnitude of the flushing flow in Clear Creek should be about 700 cfs with a frequency of 2-3 times a year from January to May. The duration of the flows would be 3-5 days with a rise rate of 200 cfs/day and a fall rate of -100 cfs/day. This flow magnitude would allow the removal and transport of fine sediments from the riverbed. When the streamflow was 646 cfs in Clear

Creek, 95% of particles from bedload measurements were less than 2 mm and the transport rate of the particles was about 20 tons/day (Graham Matthews & Associates 2003).

The combined instream flows and flushing flows are presented in **Figure 39**. Also shown in the figure is the median daily flow in Clear Creek for the pre-dam period. There were a number of high-flow pulses of greater than 600 cfs from late January through early April for the pre-dam period.

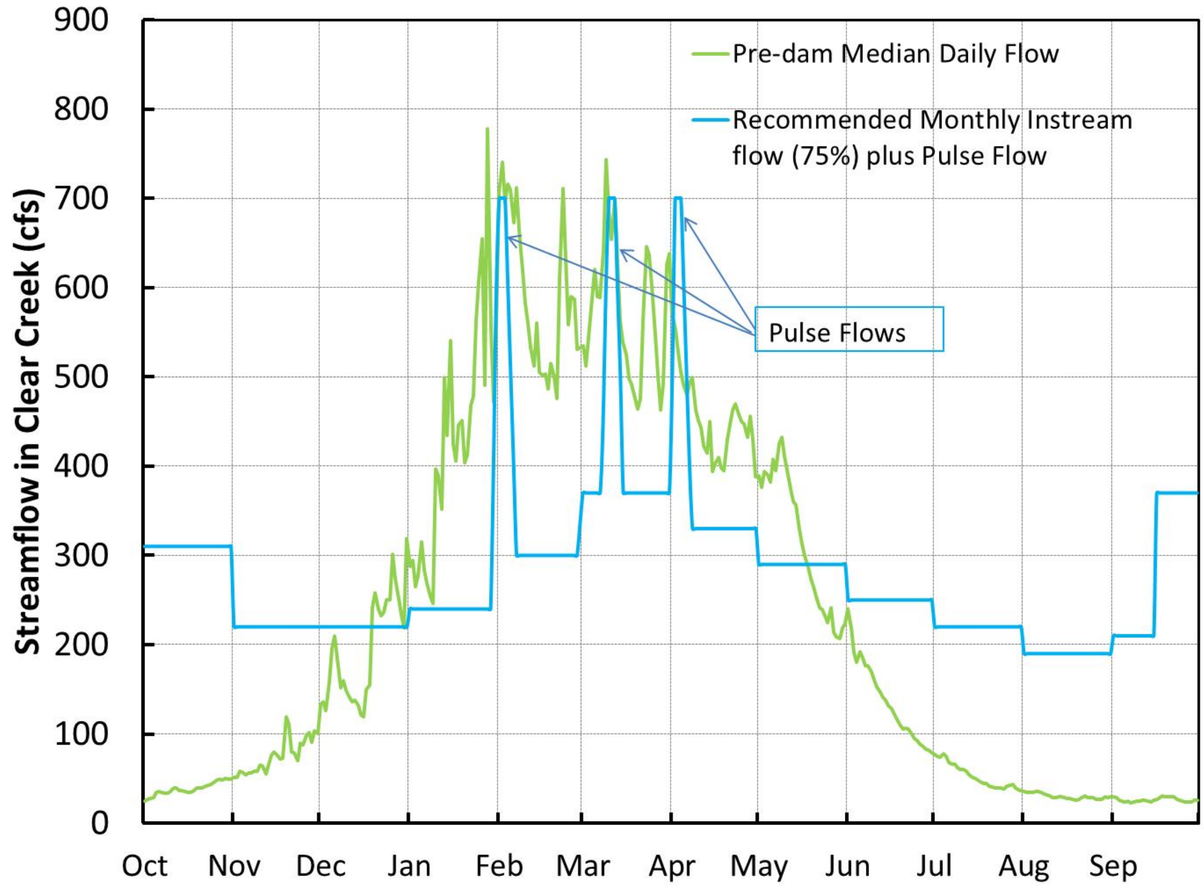

Figure 39. Recommended monthly instream flow plus pulse flows in Clear Creek

The recommended flows should be carefully examined for their potential benefits to and adverse effects on the three anadromous fish species in Clear Creek. The high instream flows from February to May will provide benefits to juvenile emigration of all three species, adult immigration of spring-run Chinook, rearing habitat for spring-run Chinook and steelhead (**Figure 40**).

The enhanced flow in September and October will provide benefits to adult immigration of steelhead and fall-run Chinook and adequate water temperature for spawning of spring-run and possibly fall-run Chinook (**Figure 40**). The only concern relevant to the enhanced flow is that some redds might be adversely impacted if those areas where redds exist dry out when flow decreases from September to October to November. Depending on the degree to which redds are impacted, the enhanced flow could be extended for some time so that eggs in those redds may develop into fry that can move to lower water when flow decreases.

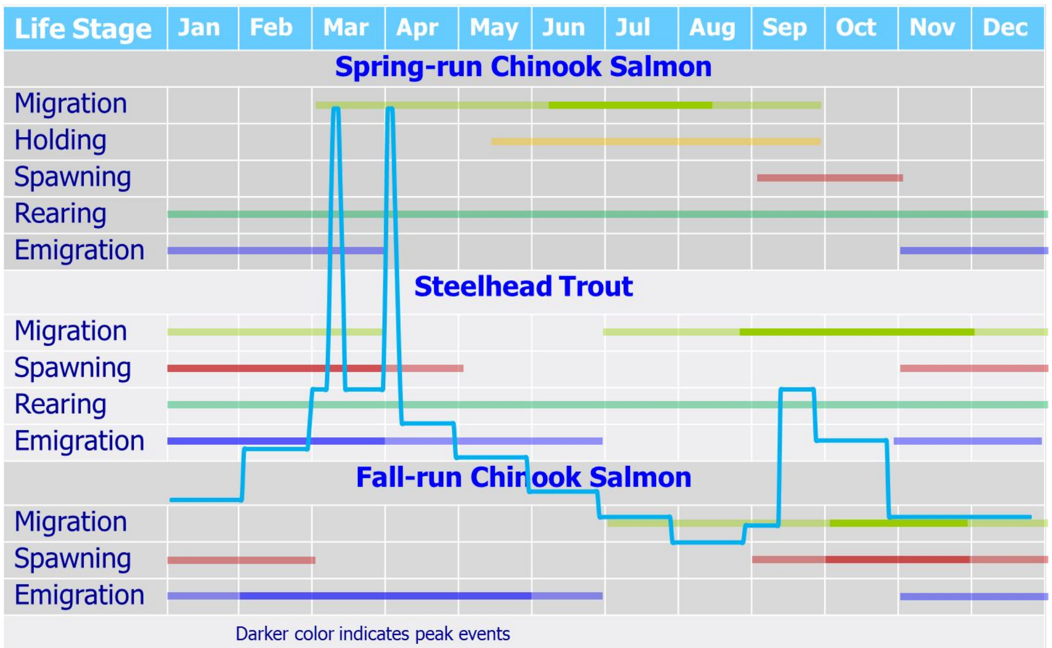

Figure 40. Diagram showing the recommended flows and the life stages of three fish species in Clear Creek

The high-flow pulses not only improve access for spring-run adult migration, but also help remove fine sediments for spawning and facilitate juvenile emigration of all three fish species in Clear Creek. When the high-flow pulses occur during the peak spawning of steelhead, they may adversely impact the redds of steelhead in Clear Creek. The adverse impact can be avoided if pulse flows occur after eggs have developed into fry, for example, in April and May instead of in March and April.

9.3 Channel Maintenance Flow

Channel maintenance flows usually cover the stream banks. They maintain a long-term sediment balance, maintain streamside vegetation and structural stability of streambanks, and prevent vegetation encroachment in the channel (Locke *et al.* 2008). The magnitude of the channel maintenance flow should be about 6000-7000 cfs with a frequency of once every two or three years between January and March. The duration of the flow may span 10 days with a rise rate of 1700 cfs/day, and a fall rate of -1000 cfs/day. Restoring the channel maintenance flow will help to create and maintain several different types of habitats in Clear Creek, including:

- Habitat for spawning: Recruit and distribute augmented gravels that are placed on the banks and floodplains. A flow magnitude of about 6,000 cfs is required to achieve sediment transport (Stillwater Sciences 2004). Since 1996, local, state, and federal partners have augmented the creek’s gravel supply. The gravels are placed on the banks and floodplain and recruited into the channel and distributed by high flow events, rather than being placed directly in the channel. It was estimated that 103,371 tons of gravel have been injected at specific locations on the creek, resulting in a steady increase in spawning habitat. This added gravel has recharged spawning gravel within about 3 miles of creek below the dam. Securing a long-term gravel supply is critical for reestablishing sediment transport processes that create and maintain fish habitat (USBR and USFWS 2008).

- Habitat for juvenile rearing: Increase the margin habitat area of low velocity for feeding and rearing, increase the frequency and availability of temporary backwater channels to provide rearing habitat and refuge, increase instream habitat complexity and cover by recruiting large woody debris to provide juvenile rearing habitat, and incorporate complex channel habitats including floodplain ponds and scour channels to support juvenile salmonid rearing.

- Habitat for adult holding: Increase and maintain deep, coldwater pools to provide holding habitat for adult spring-run Chinook salmon.

The recommended channel maintenance flow would allow fluvial processes to reshape and maintain a new dynamic river channel and provide favorable water temperature and physical habitat conditions for juvenile and adult salmonids in Clear Creek. There is potential for the channel maintenance flow to scour steelhead redds. This effect can be minimized if the flow is scheduled in April to avoid the peak spawning period of steelhead.

Reservoir operation for the channel maintenance flow may be a concern because Whiskeytown Dam has a maximum discharge of 1200 cfs to Clear Creek through the Clear Creek outlet. However, the release of 6000-7000 cfs can be achieved through the use of Glory Hole. The Glory Hole discharge depends on a few factors including the rate of inflow and the hydraulic properties of the Glory Hole spillway. The Glory Hole has a crest elevation of 1210 feet and reaches its maximum design discharge capacity of 28,000 cfs at a water surface elevation of 1220.5 feet (Stillwater Sciences 2004). Note that the crest of the Whiskeytown Dam itself is 1228 feet.

10 Water Cost Analysis

It is our interest to determine the difference in water use between the recommended flows and the current flows. The additional water allocated to Clear Creek may be used to generate hydroelectricity at Spring Creek Powerplant and Keswick Powerplant.

The calculation of the current flows was based on the flow data from water year 2001 to 2010. For this time period, a new flow scheme was implemented: 200 cfs from October through June and 150 cfs from July through September. These flows were higher than those for the time period of 1963 to 2000. Monthly low flows from 2001 to 2010 are presented in **Figure 41**. The median (50th percentile) flow during this time period was higher than 200 cfs from October through May, about 100 cfs in July and August, and about 150 cfs in September.

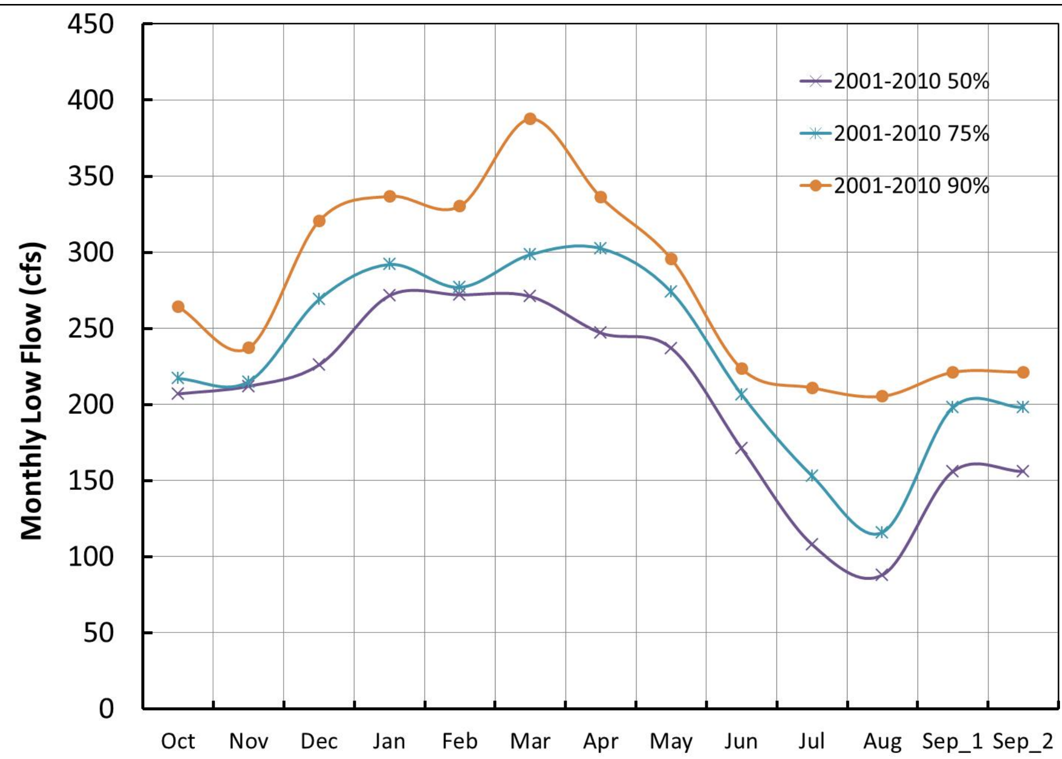

Figure 41. Monthly low flows in Clear Creek from WY 2001 to 2010

Implementing the recommended flows requires higher flows than the current flows from March through October (except for the 50th percentile flow in May), lower flows in December and January, and lower flows for the 50th percentile and higher flows for the 75th and 90th percentiles in November and February (**Figure 42**). The highest increases in flow were in the second half of September and October when a more stringent water temperature criterion (56 ^{o}F) was applied, followed by July and August when air temperatures were high and existing flows were low (Figure 42). On average, increases from the recommended flows to the current flows were 32.1, 38.7 and 32.2 cfs for 50th, 75th, and 90th percentiles, respectively.

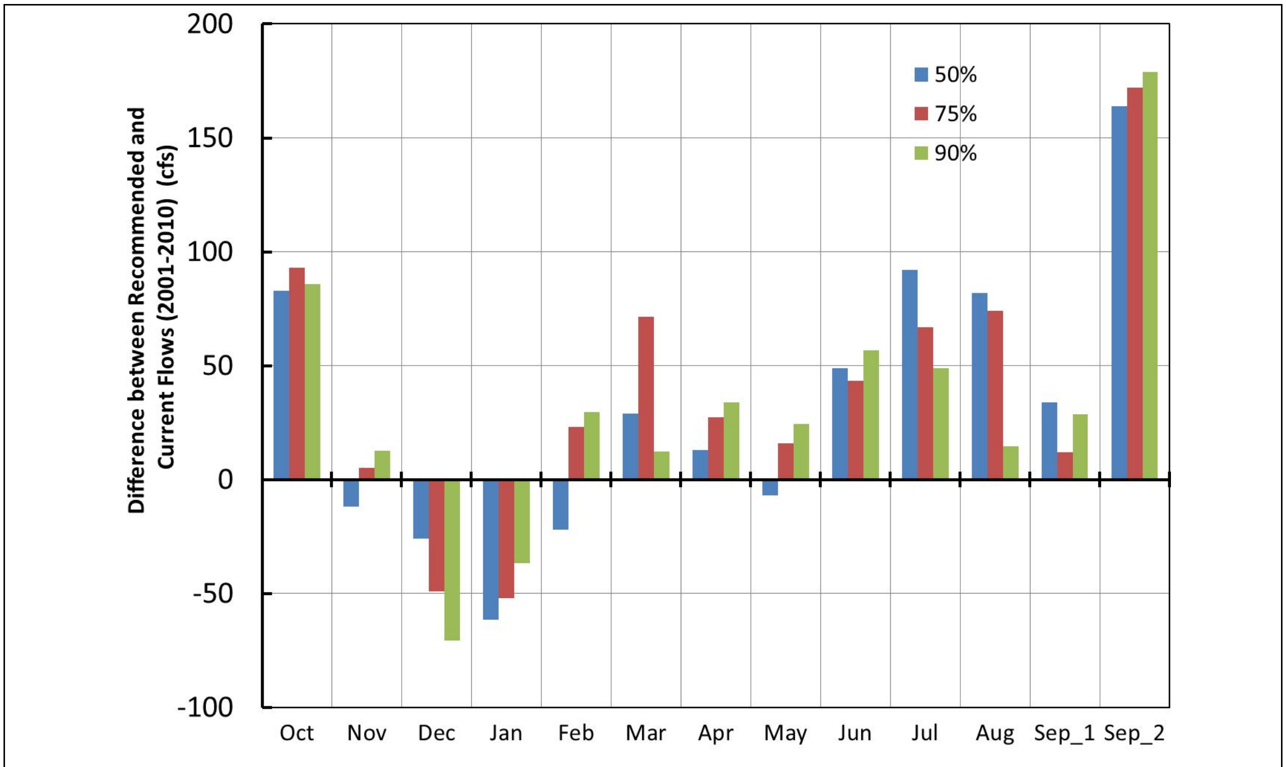

Figure 42. Flow difference between the recommended and current flows in Clear Creek

Differences in water use can be readily calculated from a flow and the time period for the flow and are presented in **Figure 43**. As expected, water use differences follow the same pattern as flow differences. Higher water uses were required from March through October (except for the 50^{th} percentile flow in May), lower water uses in December and January. The highest increases in water use were in the second half of September and October, followed by July and August. The annual additional water use was about 23,000 acre feet for the 50^{th} and 90^{th} percentiles and 28,000 acre feet for the 75^{th} percentile. The higher additional water use for the 75^{th} percentile than the 90^{th} percentile was attributed to much higher current flows for the 90^{th} percentile than the 75^{th} percentile (**Figure 43**).

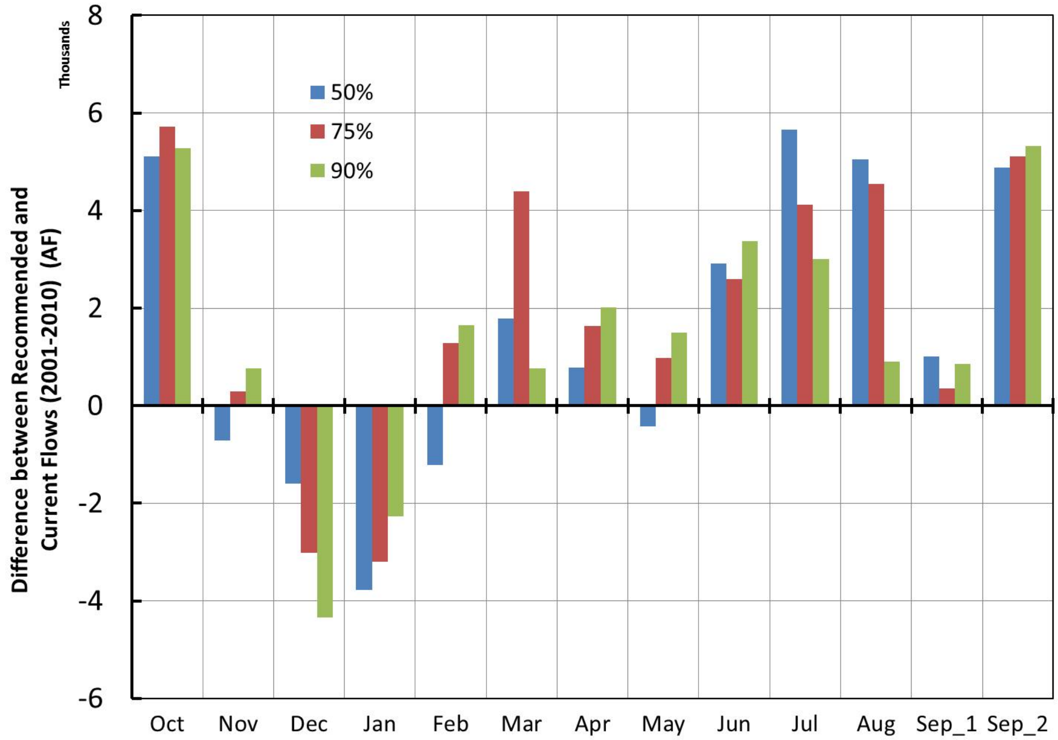

Figure 43. Water use difference between the recommended and current flows in Clear Creek

If we assume that the additional water allocated to Clear Creek is equal to the reduction of flows to Spring Creek Powerplant and then Keswick Powerplant, the hydropower generated from the powerplants would reduce accordingly. The reduced power generation may be calculated by the following equation (assuming the efficiency is 100%):

$$P = 10^4\, hQ$$

where P is power in watt, h is the head of water in meter, and Q is the flow rate in m^3/second.

The water head is 169.5 meters for Spring Creek Powerplant and 23.8 meters for Keswick Powerplant. If the additional water allocated to Clear Creek goes directly to the powerplants, it could generate, on average, hydropower of 1.54 to 1.85 MW (about 1% of the plant capacity of 180 MW) at Springs Creek Powerplant and 0.22 to 0.26 MW (about 0.2% of the plant capacity of 117 MW) at Keswisk Powerplant.

11 Implementation and Monitoring

The effectiveness of the recommended flow releases from Whiskeytown Reservoir in achieving the targeted benefits will be realized through implementing the actions and consistent monitoring. Implementation of actions and monitoring programs should occur in parallel. However, before an action is implemented initial conditions should be clearly documented so that a baseline is established. Described in this section is an adaptive management framework, which is derived from the Delta Plan (Delta Stewardship Council 2011)

11.1 Establish Goals, Objectives, and Performance Measures

The management goal is to increase the populations of spring-run Chinook and steelhead in Clear Creek to a level that is sustainable, *e.g.*, doubling their populations. Achieving this goal requires taking imperative actions to improve flow, water temperature, and gravel bed for spawning and rearing. To help achieve the doubling goal, the following actions should be taken:

- Operating Whiskeytown Reservoir to provide instream flows with appropriate magnitude, timing, duration, frequency, and rise/fall rates. Specifically, meet the prescribed flow regime within 10% variance and 90% days each year.

- Operating Whiskeytown Reservoir to provide adequate water temperatures for different life stages. Specifically, meet temperature requirements within the variance of 0.25 ^{o}C and at least 90% of the days for each prescribed time period.

- Improving physical habitat for spawning and rearing through continuous gravel supply to the streambed.

11.2 Develop Implementation and Monitoring Plans

The design of implementation and monitoring should clearly describe specific activities that will occur under those actions, including a plan for both implementation of the actions and monitoring responses from the actions. It is clear that alternative operations of Whiskeytown Reservoir is needed and should be implemented in order to achieve the stated goal and objectives. These alternatives must be described in the implementation plan, which should include specific flows released from the reservoir to Clear Creek.

A monitoring plan should include the collection and management of data for reservoir release at the dam, streamflow flow and water temperature at Igo, and weather condition in Redding. The monitoring of the parameters has already been in place and make sure they will continue. Monitoring activities must also include fish biology and stream morphology. The FWS, CDFG, and their contractors have been conducting these types of monitoring for the past decade. It is crucial to continue these efforts.

11.3 Analyze, Synthesize and Evaluate Actions and Monitoring

Analysis, synthesis, and evaluation of the actions and monitoring are critical for improving current understanding. Analysis and synthesis should be informative of how conditions have changed, both expected and unexpected, as a result of the implementation of the actions. The evaluation should examine whether or not one or more of the performance measures have been met as a result of the implemented actions and why. If a performance measure is not met, an explanation of the potential reasons why this measurement has not been met should be clearly identified and communicated. The following questions should be addressed:

- Has Whiskeytown Reservoir been operated to meet the prescribed flows within 10% variation and 90% days?
- Has the reservoir been operated to meet the temperature requirements within 0.25 ºC variation and 90% days?
- Has the reservoir been operated to improve geomorphic processes? (*e.g.*, moving gravels, accessing floodplains, increasing pool depth)
- Have the changes in water temperature and stream structure had the expected effects on fish habitat? (*e.g.*, have we increased the habitat quantity and quality for adult holding, spawning, and juvenile rearing?)
- Have the improved flow, water temperature, and physical habitat showed positive biological effects on salmonid adults and juveniles?

It is apparent that, further down the above list, the questions become more difficult resolving by monitoring. For example, daily flow data or hourly water temperature data within a few years can be used to evaluate to what degree the alternative reservoir releases improved flow and water temperature, while evaluating changes in spring-run and steelhead populations may take decades. In addition, both the uncertainty, with which parameters can be measured, and the impact of confounding factors increases. For practicality, the focus of monitoring will be on short-term effects that may indicate longer-term trends of spring-run and steelhead in Clear Creek.

As a baseline assessment, monitoring should begin prior to the implementation of the proposed actions. Results from the first year of the implementation will be used to initiate the adaptive management process. For instance, water temperature data at Igo will be used to assess if the alternative releases are sufficient to meet water temperature requirements. Information gained regarding downstream transport of introduced gravels, changes in gravel quality, or changes to channel morphology should help determine the degree of success of that year’s flow management and gravel introduction. This information, in combination with other projects, will then be used to better refine flow recommendations for the following season, and also inform recommendations for other management actions such as gravel augmentation.

12 References

Annear, T., I. Chisholm, H. Beacher, A. Locke, P. Aarrestad, C. Coomer, C. Estes, J. Hunt, R. Jacobson, G. Jobsis, J. Kauffman, J. Marshall, K. Mayes, G. Smith, R. Wentworth, and C. Stalnake. 2004. Instream Flows for Riverine Resource Stewardship. Revised Edition. Instream Flow Council, Cheyenne, WY. 268 p.

Barinaga, M. 1996. A recipe for river recovery? Science 273:1648-1650.

Bovee, K.D., B.L. Lamb, J.M. Bartholow, C.B. Stalnaker, J. Taylor, and J. Henriksen. 1998. Stream habitat analysis using the instream flow incremental methodology: U.S. Geological Survey Information and Technology Report 1998-0004. 130 p.

Bunn, S.E., and A.H. Arthington. 2002. Basic Principles and Ecological Consequences of Altered Flow Regimes for Aquatic Biodiversity. Environmental Management 30:492-507.

Butte County Association of Governments (BCAG). 2011a. Butte Regional Conservation Plan - Administrative Draft. Appendix A.17 Central Valley Steelhead.http://www.buttehcp.com/documents/1st_Admin_Draft_BRCP_Docs/Appendicies/Appendix_A/A17_Central_Valley_Steelhead.pdf. Accessed July 13, 2011.

Butte County Association of Governments (BCAG). 2011b. Butte Regional Conservation Plan - Administrative Draft. Appendix A.20 Central Valley Fall-/Late Fall-Run Chinook Salmon. http://www.buttehcp.com/documents/1st_Admin_Draft_BRCP_Docs/Appendicies/Appendix_A/A20_CV__Fall-Run_Chinook_Salmon.pdf. Accessed July 13, 2011.f. Accessed July 13, 2011.

Caissie, D. 2006. The thermal regime of rivers: a review. Freshwater Biology 51:1389-1406.

Caissie, D., N. El-Jabi, and M.G. Satish. 2001. Modeling of maximum daily water temperatures in a small stream using air temperatures. Journal of Hydrology 251:14-28.

Caissie, D., M.G. Satish, and N. El-Jabi. 2007. Predicting water temperatures using a deterministic model: Application on Miramichi River catchments (New Brunswick, Canada). Journal of Hydrology 336:303-315.

Carter, K. 2008. Effects of Temperature, Dissolved Oxygen/Total Dissolved Gas, Ammonia, and pH on Salmonids: Implications for California's North Coast TMDLs. California Regional Water Quality Control Board, North Coast Region. January 2008. 47 p.

CDFG. 1998. A status review of the spring-run Chinook salmon (*Oncorhynchus tshawyscha*) in the Sacramento River drainage. Department of Fish and Game, Sacramento, California. June 1998. .

CDFG. 2010a. Quantifiable Biological Objectives and Flow Criteria for Aquatic and Terrestrial Species of Concern Dependent on the Delta. Prepared pursuant to the Sacramento-San Joaquin Delta Reform Act of 2009. November 23, 2010. 169 p.

CDFG. 2010b. Effects of Water Temperature on Anadromous Salmonids in the San Joaquin River Basin. California Department of Fish and Game, Central Region, 1234 E. Shaw Avenue, Fresno, CA 93710. February 2010.

Chen, Y.D., C.M. Steven, J.N. Douglas, and L.N. Wade. 1998. Stream Temperature Simulation of Forested Riparian Areas: II. Model Application. Journal of Environmental Engineering 124:316-328.

Cox, M.M., and J.P. Bolte. 2007. A spatially explicit network-based model for estimating stream temperature distribution. Environmental Modeling & Software 22:502-514.

CVRWQCB. 2009. Clean Water Act Sections 305(b) and 303(d) Integrated Report for the Central Valley Region. Draft Final Staff Report, May 2009. 16 p.

Danner, E., and A. Pike. 2011. Modeling river water temperatures for salmon: the importance of spatial and temporal scales. PowerPoint presentation to the NMFS Central Valley Office. June 22, 2011.

Deas, M.L., and C.L. Lowney. 2001. Water Temperature Modeling Review. Central Valley. September 2000. 113 p.

Delta Environmental Flows Group. 2010. Key Points on Delta Environmental Flows for the State Water Resources Control Board. February 2010. 9 p.

Delta Stewardship Council. 2011. Fourth Staff Draft Delta Plan. June 13, 2011. 180 p.

Denton, D.N. 1986. Clear Creek fishery study. Department of Water Resources, 1986. 70 p.

Earley, J.T., D.J. Colby, and M.R. Brown. 2010. Juvenile salmonid monitoring in Clear Creek, California, from October 2008 through September 2009. U.S. Fish and Wildlife Service, Red Bluff Fish and Wildlife Office, Red Bluff, California. September 2010. 80 p.

Ferry, M., and P. Miller. 2003. The removal of Saeltzer Dam on Clear Creek: an update. University of California, Berkeley. 21 p.

Fleenor, W., W. Bennett, P. Moyle, and J. Lund. 2010. On developing prescriptions for freshwater flows to sustain desirable fishes in the Sacramento-San Joaquin Delta. Submitted to the State Water Resources Control Board regarding flow criteria for the Delta necessary to protect public trust resources. 43 p.

Flint, L.E., and A.L. Flint. 2008. A basin-scale approach to estimating stream temperatures of tributaries to the Lower Klamath River, California. J. Environ. Qual. 37:57-68.

Gard, M. 2009. CVPIA Instream Flow Investigations - Clear Creek Fall-Run Chinook Salmon and Steelhead/Rainbow Trout Spawning. 111 p.

Giovannetti, S., and M.R. Brown. 2008. Adult Spring Chinook Salmon Monitoring in Clear Creek, California: 2007 Annual Report. USFWS Report. U.S. Fish and Wildlife Service, Red Bluff Fish and Wildlife Office, Red Bluff, California 96080. September 2008. 45 p.

Giovannetti, S., and M.R. Brown. 2009. Adult Spring Chinook Salmon Monitoring in Clear Creek, California: 2008 Annual Report. USFWS Report. U.S. Fish and Wildlife Service, Red Bluff Fish and Wildlife Office, Red Bluff, California 96080. December 2009. 38 p.

Giovannetti, S., and M.R. Brown. 2010. Adult steelhead and late-fall Chinook salmon monitoring on Clear Creek, California. 2009 Annual report. U.S. Fish and Wildlife Service, Red Bluff Fish and Wildlife Office, Red Bluff, California. November 2010. 41 p.

Graham Matthews & Associates. 2003. Clear Creek Floodplain Rehabilitation Project, Shasta County, California: WY 2003 Geomorphic Monitoring Report. Prepared for Western Shasta Resource Conservation District, Anderson Clear Creek Floodplain Rehabilitation Project, Shasta County, California: WY 2003 Geomorphic Monitoring Report. Prepared for Western Shasta Resource Conservation District, Anderson CA by Graham Matthews & Associates, Weaverville, CA. July 2003. 124 p.

Helsel, D.R., and R.M. Hirsch. 2002. Statistical Methods in Water Resources, Techniques of Water Resources Investigations, Book 4, chapter A3. U.S. Geological Survey. 522 p.

Jean-Francois, C., and C. Daniel. 2008. Stream temperature modeling using artificial neural networks: application on Catamaran Brook, New Brunswick, Canada. Hydrological Processes 22:3361-3372.

Locke, A., C. Stalnaker, S. Zellmer, K. Williams, H. Beecher, T. Richards, C. Robertson, A. Wald, A. Paul, and T. Annear. 2008. Integrated Approaches to Riverine Resource Management: Case Studies, Science, Law, People, and Policy. Instream Flow Council, Cheyenne, WY. 430 p.

McBain, S., D. Mierau, J. Bair, B. Trush, T. Koehler, F. Meyer, G. Matthews, M. Hampton, and Y. Cui. 2001. Geomorphic Evaluation of Lower Clear Creek Downstream of Whiskeytown Dam, California. Final Report. McBain and Trush, P.O. Box 663, CA 95518. November 2001. 257 p.

McCullough, D.A. 1999. A Review and Synthesis of Effects of Alterations to the Water Temperature Regime on Freshwater Life Stages of Salmonids, With Special Reference to Chinook Salmon. Prepared for the U.S. Environmental Protection Agency Region 10, Seattle, Washington. Published as EPA 910-R-99-010, July 1999. 291 p.

McCullough, D.A., J.M. Bartholow, H.I. Jager, R.L. Beschta, E.F. Cheslak, M.L. Deas, J.L. Ebersole, J.S. Foott, S.L. Johnson, K.R. Marine, M.G. Mesa, J.H. Petersen, Y. Souchon, K.F. Tiffan, and W.A. Wurtsbaugh. 2009. Research in Thermal Biology: Burning Questions for Coldwater Stream Fishes. Reviews in Fisheries Science 17:90-115.

McEwan, D.R. 2001. Central Valley Steelhead. Fish Bulletin 179: Volume One.

Morrill, J.C., R.C. Bales, and M.H. Conklin. 2005. Estimating stream temperature from air temperature: Implications for future water quality. Journal of Environmental Engineering 131:139-146.

Myrick, C.A., and J.J. Cech. 2001. Temperature Effects on Chinook Salmon and Steelhead: a Review Focusing on California's Central Valley Populations. Bay-Delta Modeling Forum. Technical Publication 01-1. 57 p.

Myrick, C.A., and J.J. Cech. 2004. Temperature effects on juvenile anadromous salmonids in California's central valley: what don't we know? Reviews in Fish Biology and Fisheries 14:113-123.

Myrick, C.A., and J.J. Cech. 2005. Effects of temperature on the growth, food consumption, and thermal tolerance of age-0 Nimbus-strain steelhead. North American Journal of Aquaculture 67:324-330.

Neumann, D.W., B. Rajagopalan, and E.A. Zagona. 2003. Regression model for daily maximum stream temperature. Journal of Environmental Engineering 129.

Newton, J.M., and M.R. Brown. 2004. Adult spring Chinook salmon monitoring in Clear Creek, California, 1999-2002. US Fish and Wildlife Service, Red Bluff. July 2004. 71 p.

NMFS. 2009a. Biological opinion and conference opinion on the long-term operations of the Central Valley Project and State Water Project. 844 p.

NMFS. 2009b. Public Draft Recovery Plan for the Evolutionarily Significant Units of Sacramento River Winter-run Chinook Salmon and Central Valley Spring-run Chinook Salmon and the Distinct Population Segment of Central Valley Steelhead. Sacramento Protected Resources Division. October 2009. 273 p.

Norton, G.E., and A. Bradford. 2009. Comparison of two stream temperature models and evaluation of potential management alternatives for the Speed River, Southern Ontario. Journal of Environmental Management 90:866-878.

Olden, J.D., and R.J. Naiman. 2010. Incorporating thermal regimes into environmental flows assessments: modifying dam operations to restore freshwater ecosystem integrity. Freshwater Biology 55:86-107.

Orlob, G.T., M. Jensen, X. Wang, and J. Fellos. 1999. Temperature regulation through Whiskeytown Reservoir. Report 00-5.Water Resources and Environmental Modeling Group, Center for Water Resources and Environmental Modeling Group, Department of Civil and Environmental Engineering, University of California, Davis. .

Parasiewicz, P. 2001. MesoHABSIM:A concept for application of instream flow models in river restoration planning. Fisheries 26:6-13.

Parasiewicz, P. 2007a. Using MesoHABSIM to develop reference habitat template and ecological management scenarios. River Research and Applications 23:924-932.

Parasiewicz, P. 2007b. The MesoHABSIM model revisited. River Research and Applications 23:893-903.

Parasiewicz, P. 2008. Application of MesoHABSIM and Target Fish Community approaches to restoration of the Quinebaug River, Connecticut and Massachusetts, USA. River Research and Applications 24:459-471.

Parasiewicz, P., and J.D. Walker. 2007. Comparison of MesoHABSIM with two microhabitat models (PHABSIM and HARPHA). River Research and Applications 23:904-923.

Petts, G.E. 2009. Instream Flow Science For Sustainable River Management. Journal of the American Water Resources Association 45:1071-1086.

Poff, N.L., and D.D. Hart. 2002. How dams vary and why it matters for the emerging science of dam removal. Bioscience 52:659-668.

Poff, N.L., J.D. Olden, D.M. Merritt, and D.M. Pepin. 2007. Homogenization of regional river dynamics by dams and global biodiversity implications. Proceedings of the National Academy of Sciences of the United States of America 104:5732-5737.

Poff, N.L., J.D. Allan, M.B. Bain, J.R. Karr, K.L. Prestegaard, B.D. Richter, R.E. Sparks, and J.C. Stromberg. 1997. The natural flow regime. Bioscience 47:769-784.

Poff, N.L., B.D. Richter, A.H. Arthington, S.E. Bunn, R.J. Naiman, E. Kendy, M. Acreman, C. Apse, B.P. Bledsoe, M.C. Freeman, J. Henriksen, R.B. Jacobson, J.G. Kennen, D.M. Merritt, J.H. O'Keeffe, J.D. Olden, K. Rogers, R.E. Tharme, and A. Warner. 2010. The ecological limits of hydrologic alteration (ELOHA): a new framework for developing regional environmental flow standards. Freshwater Biology 55:147-170.

Richter, B.D., and G.A. Thomas. 2007. Restoring environmental flows by modifying dam operations. Ecology and Society 12.

Richter, B.D., J.V. Baumgartner, J. Powell, and D.P. Braun. 1996. A method for assessing hydrologic alteration within ecosystems. Conservation Biology 10:1163-1174.

Richter, B.D., J.V. Baumgartner, R. Wigington, and D.P. Braun. 1997. How much water does a river need? Freshwater Biology 37:231-249.

Richter, B.D., J.V. Baumgartner, D.P. Braun, and J. Powell. 1998. A spatial assessment of hydrologic alteration within a river network. Regulated Rivers-Research & Management 14:329-340.

Risley, J.C., J. Edwin A. Roehl, and P.A. Conrads. 2002. Estimating water temperatures in small streams in western Oregon using neural network models. WRIR 02-4218. .

Sahoo, G.B., S.G. Schladow, and J.E. Reuter. 2009. Forecasting stream water temperature using regression analysis, artificial neural network, and chaotic nonlinear dynamic models. Journal of Hydrology In Press, Accepted Manuscript.

Stillwater Sciences. 2004. Environmental Water Program - Draft Conceptual Proposal for Flow Acquisition on Lower Clear Creek. 2855 Telegraph Avenue, Suite 400, Berkeley, CA 94705. May 2004.

SWRCB. 2010. Development of Flow Criteria for the Sacramento-San Joaquin Delta Ecosystem. Prepared Pursuant to the Sacramento-San Joaquin Delta Reform Act of 2009. August 3, 2010. 190 p.
Taylor, R.L. 1998. Simulation of hourly stream temperature and daily dissolved solids for the Truckee River, California and Nevada: U.S. Geological Survey Water-Resources Investigations Report 98-4064, 70 p. .
Tetra Tech. 2011. Lower Stanislaus River Water Temperature Model. Final report prepared for NOAA National Marine Fisheries Service. March 2011. 464 p.
Tharme, R.E. 2003. A global perspective on environmental flow assessment: Emerging trends in the development and application of environmental flow methodologies for rivers. River Research and Applications 19:397-441.
The Nature Conservancy. 2009. Indicators of Hydrologic Alteration (version 7.1) User's Manual.
US EPA. 2001. Summary of Technical Literature Examining the Physiological Effects of temperature on Salmonids, Issue Paper 5, prepared by Dale McCullough, Shelley Spalding, Debra Sturdevant, and Mark Hicks as Part of EPA Region 10 Temperature Water Quality Criteria Guidance Development Project. EPA-910-D-01-005. May 2001. 114 p.
US EPA. 2003. EPA Region 10 Guidance for Pacific Northwest State and Tribal Temperature Water Quality Standards. EPA 910-B-03-002. Region 10 Office of Water, Seattle, WA.
USBR. 2004. Long-term Central Valley Project Operations Criteria and Plan. Sacramento, California. June 30, 2004. , Sacramento, California.
USBR, and USFWS. 2008. Central Valley Project Improvement Act. Public Law 102-575. Annual Report. Fiscal Year 2008.
USEPA. 2003. EPA Region 10 Guidance for Pacific Northwest State and Tribal Temperature Water Quality Standards. EPA 910-B-03-002. Region 10 Office of Water, Seattle, WA.
USEPA. 2009. Guidance on the Development, Evaluation, and Application of Environmental Models. EPA/100/K-09/003. March 2009. 90 p.