SECTION 5.0 STATUS OF LISTED SPECIES AND CRITICAL HABITAT

The USACE completed a BA in 2013 regarding the O&M of existing fish passage facilities at Daguerre Point Dam on the Yuba River (USACE 2013a), and the RMT issued their M&E Program Interim Report (RMT 2013a). Much of the following information regarding the status of ESA-listed fish species and their critical habitats is taken from those two documents. In addition to information provided by USACE (2013a) and RMT (2013a), more recent information and data that has become available since the release of these two documents has been incorporated into this section. Moreover, some of the analyses conducted in RMT (2013a) have been updated with new data, to the extent that new data were available at the time of preparation of this Applicant-Prepared Draft BA.

5.1 Central Valley Spring-run Chinook Salmon ESU

5.1.1 ESA Listing Status

On September 16, 1999, NMFS listed the Central Valley ESU of spring-run Chinook salmon as a "threatened" species (64 FR 50394). On June 14, 2004, following a 5-year species status review, NMFS proposed that the Central Valley spring-run Chinook salmon remain listed as a threatened species based on the Biological Review Team strong majority opinion that the Central Valley spring-run Chinook ESU is ''*likely to become endangered within the foreseeable future*'' due to the greatly reduced distribution of Central Valley spring-run Chinook salmon and hatchery influences on the natural population. On June 28, 2005, NMFS reaffirmed the threatened status of the Central Valley spring-run Chinook salmon ESU, and included the FRFH spring-run Chinook salmon population as part of the Central Valley spring-run Chinook salmon ESU (70 FR 37160).

Section 4(c)(2) of the ESA requires that NMFS review the status of listed species under its authority at least every 5 years and determine whether any species should be removed from the list or have its listing status changed. In April 2016, NMFS completed a third 5-year status review of the Central Valley spring-run Chinook salmon ESU. Prior to making a determination on whether the listing status of the ESU should be "uplisted" (i.e., threatened to endangered), "downlisted", or remain unchanged, NMFS: 1) considered new and substantial scientific information that had become available since the previous status review, and used this information to produce an updated biological status summary report (Williams et al. 2016, which is referred to as the "viability report"); 2) considered whether the five ESA listing factors (threats) changed substantially since the previous status review, based upon the following:

(a) The 5-year Status Review Report for CV spring-run Chinook salmon published in 2011 (NMFS 2011a);

(b) Central Valley Salmon and Steelhead Recovery Plan (NMFS 2014);

(c) Discussions with Cal Fish and Wildlife and USFWS on watershed assessments and recovery action implementation status;

(d) Implementation of the reasonable and prudent alternative for the Biological Opinion on the Long-term Operations of the Central Valley Project and State Water Project (NMFS 2009b);

(e) Grandtab (CDFW 2015);

(f) Framework for assessing viability of threatened and endangered Chinook salmon and steelhead in the Sacramento-San Joaquin Basin (Lindley et al. 2007);

3) considered the current threats to the species; 4) considered recovery action implementation; and 5) considered relevant ongoing and future conservation measures and programs.

Based on a review of the available information, NMFS (2016) recommended that the Central Valley spring-run Chinook salmon ESU remain classified as a threatened species. NMFS' review also indicates that the biological status of the ESU has probably improved since the previous status review in 2010/2011 and that the ESU's extinction risk may have decreased. However, the ESU is still facing significant risks, and those risks are likely to increase over at least the next few years as the full effects of the recent drought occur (Williams et al. 2016). In addition to the low adult returns observed during 2015, juveniles hatched during the drought years of 2013 through 2015 are expected to produce low adult returns in 2016 through 2018. Monitoring environmental and biological conditions and management actions for these drought impacted year classes will be extremely important (NMFS 2016b). As part of the 5-year review, NMFS also re-evaluated the status of the FRFH stock and concluded that it should remain part of the Central Valley spring-run Chinook salmon ESU.

In addition to federal regulations, the California Endangered Species Act (CESA, Fish and Game Code § 2050 to § 2089) establishes various requirements and protections regarding species listed as threatened or endangered under state law. California's Fish and Game Commission is responsible for maintaining lists of threatened and endangered species under CESA. Spring-run Chinook salmon in the Sacramento River Basin, including the lower Yuba River, was listed as a threatened species under CESA on February 2, 1999.

5.1.2 Critical Habitat Designation

Critical habitat was designated for the Central Valley spring-run Chinook salmon ESU on September 2, 2005 (70 FR 52488), and includes stream reaches of the Feather and Yuba rivers, Big Chico, Butte, Deer, Mill, Battle, Antelope, and Clear creeks, the Sacramento River, and portions of the northern Delta (NMFS 2009b). On the Yuba River, critical habitat is designated from the confluence with the Feather River upstream to Englebright Dam. This critical habitat includes the stream channels in the designated stream reaches and their lateral extents, as defined by the ordinary high-water line. In areas where the ordinary high-water line has not been defined, the lateral extent will be defined by the bankfull elevation (defined as the level at which water begins to leave the channel and move into the floodplain; it is reached at a discharge that generally has a recurrence interval of 1 to 2 years on the annual flood series) (Bain and Stevenson 1999; 70 FR 52488, September 2, 2005).

In designating critical habitat, NMFS (2009b) considered the following requirements of the species: 1) space for individual and population growth, and for normal behavior; 2) food, water, air, light, minerals, or other nutritional or physiological requirements; 3) cover or shelter; 4) sites for breeding, reproduction, or rearing offspring; and, generally; 5) habitats that are protected from disturbance or are representative of the historic geographical and ecological distributions of a species [see 50 C.F.R. 424.12(b)]. In addition to these factors, NMFS also focused on the key physical and biological features within the designated area that are essential to the conservation of the species and that may require special management considerations or protection. Specifically, the 2005 designation of critical habitat for spring-run Chinook salmon used the term "primary constituent elements" (PCEs) of critical habitat to identify those physical and biological features essential to the conservation of a species for which its designated critical habitat was based on. During 2016, the regulations were revised to remove the terms ‘‘principal biological or physical constituent elements’’ and ‘‘primary constituent elements’’ from 50 C.F.R. 424.12(b). These concepts were replaced by the statutory term ‘‘physical or biological features" (PBFs) (81 FR 7432, February 11, 2016).

NMFS (81 FR 7432, February 11, 2016) define ‘‘physical or biological features’’ as *‘‘the features that support the lifehistory needs of the species, including but not limited to water characteristics, soil type, geological features, sites, prey, vegetation, symbiotic species, or other features. A feature may be a single habitat characteristic, or a more complex combination of habitat characteristics. Features may include habitat characteristics that support ephemeral or dynamic habitat conditions. Features may also be expressed in terms relating to principles of conservation biology, such as patch size, distribution distances, and connectivity.’’* The definition of ‘‘physical or biological features,’’ encompasses similar habitat characteristics as previously described in 50 C.F.R. 424.12(b), including "…*nesting grounds, spawning sites, feeding sites, seasonal wetland or dryland, water quality or quantity, …geological formation, vegetation type, tide, and specific soil types*" (81 FR 7432, February 11, 2016).

As described in the Final Rule (81 FR 7432, February 11, 2016), "…*removing the phrase ‘‘primary constituent elements’’ is not intended to substantively alter anything about the designation of critical habitat, but to eliminate redundancy in how we* [NMFS] *describe the physical or biological features. The phrase ‘‘primary constituent element’’ is not found in the Act and the regulations have never been clear as to how primary constituent elements relate to or are distinct from physical or biological features essential to the conservation of the species, which is the phrase used in the Act. In fact, the removal of the phrase ‘‘primary constituent elements’’ will alleviate the tension caused by trying to understand the relationship between the phrases. The specificity of the primary constituent elements that has been discussed in previous designations will now be discussed in the descriptions of the physical or biological features essential to the conservation of the species*".

As also described in NMFS (2016b; 2016c), this is a shift in terminology only and does not change the categories of such features (i.e., freshwater rearing habitat or freshwater migration corridors) or the approach used in conducting an effects analysis, which is the same regardless of whether the original designation identified primary constituent elements, physical or biological features, or essential features. Therefore, in this Applicant-Prepared Draft BA, the term PBF is used to mean PCE or essential feature, as appropriate for the specific critical habitat.

5.1.2.1 Physical or Biological Features

A central component of NMFS assessment of critical habitat is the basic premise that the value of critical habitat for the conservation of a listed species is the sum of the values of the components that comprise the habitat (NMFS 2016a). In turn, the value of critical habitat for the conservation of a listed species is the sum of the value of the PBFs that make up the area. PBFs are specific areas or functions, such as spawning or rearing habitat, that support different life history stages or requirements of the species (NMFS 2016a). The value of critical habitat for the conservation of a listed species of the PBFs is the sum of the quantity, quality, and availability of the physical or biological features of those PBFs. Physical or biological features are the specific processes, variables, or elements that comprise a PBF. Thus, an example of a PBF would be spawning habitat and the physical or biological features of that PBF are conditions such as clean spawning gravels, appropriate timing and duration of certain water temperatures, and water quality free of pollutants (NMFS 2016a). Therefore, reductions in the quantity, quality, or availability of one or more physical or biological feature reduce the value of the PBF, which in turn reduces the function of the sub-area (e.g., watershed), which in turn reduces the function of the overall critical habitat designation. In the strictest interpretation, reductions to any one PBF would equate to a reduction in the value of the whole (NMFS 2016a). NMFS analysis will identify the species and critical habitats that are likely to occur in the same space and at the same time as potential stressors. Consideration will then be given to try and estimate the nature of that co-occurrence ("exposure analyses"). In this step of the analyses, NMFS will try to identify the number and age (or lifestage) of individuals that are likely to be exposed to an action's effects, the population that those individuals represent, and the specific areas and physical or biological features (PBFs) of critical habitat that are likely to be exposed (NMFS 2016a).

Within the range of the spring-run Chinook salmon ESU, the PBFs of the designated critical habitat include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, estuarine areas, and nearshore and offshore marine areas. The following summary descriptions of the current conditions of the freshwater PBFs for the Central Valley spring-run Chinook salmon ESU were taken from NMFS (2009b), with the exception of new or updated information regarding current habitat conditions. The definitions and descriptions of PBFs of designated critical habitat in the Sacramento River and its primary tributaries would also apply to designated critical habitat in the Yuba River.

5.1.2.1.1 Freshwater Spawning Habitat

Freshwater spawning sites are areas with appropriate water quantity, water quality and substrate for successful spawning, egg incubation, and larval development. Spring-run Chinook salmon have been reported to spawn in the mainstem Sacramento River between Red Bluff Diversion Dam (RBDD) and Keswick Dam, although little spawning activity has been reported in recent years. Spring-run Chinook salmon primarily spawn in Sacramento River tributaries such as Mill, Deer, and Butte creeks. Operations of Shasta and Keswick dams on the mainstem Sacramento

River are constrained by the need to provide water of suitable temperature for adult winter-run Chinook salmon migration, holding, spawning and incubation, as well as for spring-run Chinook salmon embryo incubation in the mainstem Sacramento River.

5.1.2.1.2 Freshwater Rearing Habitat

Freshwater rearing sites are areas with: 1) water quantity and floodplain connectivity to form and maintain physical habitat conditions and support juvenile growth and mobility; 2) water quality and forage supporting juvenile development; and 3) habitat complexity characterized by natural cover such as shade, submerged and overhanging large woody material (LWM), log jams and beaver dams, aquatic vegetation, large rocks and boulders, side channels, and undercut banks. Both spawning areas and migratory corridors comprise rearing habitat for juveniles, which feed and grow before and during their outmigration. Rearing habitat condition is strongly affected by habitat complexity, food supply, and the presence of predators of juvenile salmonids. The channelized, leveed, and rip-rapped river reaches and sloughs that are common in the Sacramento River system typically have low habitat complexity, relatively low production of food organisms, and offer little protection from either fish or avian predators. However, some complex, productive habitats with floodplains remain in the system (e.g., Sacramento River reaches with setback levees (i.e., primarily located upstream of the City of Colusa) and flood bypasses (i.e., Yolo and Sutter bypasses). Juvenile lifestages of salmonids are dependent on the function of this habitat for successful survival and recruitment.

5.1.2.1.3 Freshwater Migration Corridors

Freshwater migration corridors provide upstream passage for adults to upstream spawning areas, and downstream passage of outmigrant juveniles to estuarine and marine areas. Migratory corridors are downstream of the spawning areas and include the lower reaches of the spawning tributaries, the mainstem of the Sacramento River and the Delta.

Migratory habitat condition is strongly affected by the presence of barriers, which can include dams (i.e., hydropower, flood control, and irrigation flashboard dams), unscreened or poorly screened diversions, degraded water quality, or behavioral impediments to migration. RBDD, completed in 1964, features a series of 11 gates that, when lowered, provided for gravity diversion of irrigation water from the Sacramento River into the Tehama-Colusa and Corning Canals for potential delivery to the Sacramento Valley National Wildlife Refuge and to approximately 140,000 ac of irrigable lands along the Interstate 5 corridor between Red Bluff and Dunnigan, California (Reclamation 2008b). The RBDD was a serious impediment to upstream and downstream fish migration, and a significant portion of the Sacramento River spawning habitat for Chinook salmon and steelhead occurred upstream of the dam. Until recently, the RBDD created an upstream migratory barrier in the mainstem Sacramento River during its May 15 through September 15 "gates in" configuration. In response to the NMFS (2009b) BO, the RBDD gates were permanently raised in September 2011 and thus, fish passage conditions have likely improved at the RBDD. The Red Bluff Fish Passage Improvement Project, which included construction of a pumping plant to allow for diversion of water from the Sacramento River without closing the RBDD gates, was completed in 2012 (Tehama-Colusa Canal Authority 2012).

Sacramento River flow, along with many juvenile spring-run Chinook salmon, enters the Delta Cross Channel (DCC) (when the gates are open) and Georgiana Slough, and subsequently the central Delta, especially during periods of increased water export pumping from the Delta. Mortality of juvenile salmon entering the central Delta is higher than for those continuing downstream in the Sacramento River. This difference in mortality could be caused by a combination of factors, including: 1) the longer migration route through the central Delta to the western Delta; 2) exposure to higher water temperatures; 3) higher predation rates; 4) exposure to seasonal agricultural diversions; 5) water quality impairments due to agricultural and municipal discharges; and 6) a more complex channel configuration that makes it more difficult for salmon to successfully migrate to the western Delta and the ocean. In addition, the State and federal pumps and associated fish facilities increase mortality of juvenile spring-run Chinook salmon through various means, including entrainment into the State and federal canals, and salvage operations.

5.1.2.1.4 Estuarine Habitat Areas

The current condition of the estuarine habitat in the Delta has been substantially degraded from historic conditions. Over 90 percent of the fresh, brackish, and salt marshes have been lost due to human activities. This loss of the fringing marshes reduces the availability of forage species and eliminates the cycling of nutrients from the marsh vegetation into the water column of the adjoining waterways.

The channels of the Delta have been modified by the raising of levees and armoring of the levee banks with riprap, which has decreased habitat complexity by reducing the incorporation of woody material and vegetative material into the nearshore area, minimizing and reducing local variations in water depth and velocities, and simplifying the community structure of the nearshore environment.

Heavy urbanization and industrial actions have lowered water quality and introduced persistent contaminants to the sediments surrounding points of discharge (e.g., refineries in Suisun and San Pablo bays and creosote factories in Stockton).

Delta hydraulics have been modified as a result of CVP and SWP actions. Within the central and southern Delta, net water movement is towards the pumping facilities, altering the migratory cues for emigrating fish in these regions. Spring-run Chinook salmon smolts are drawn to the central and south Delta as they outmigrate, and are subjected to the indirect effects (e.g., predation, contaminants) and direct effects (e.g., salvage, loss) in the Delta and the CVP and SWP fish facilities.

The area of salinity transition, referred to as the low salinity zone (LSZ), is an area of high productivity. This zone fluctuates in its location in relation to the outflow of water from the Delta and moves westwards with high Delta inflow (i.e., floods and spring runoff) and eastwards with reduced summer and fall flows. This variability in the salinity transition zone has been substantially reduced by the operations of the CVP/SWP. The CVP/SWP long-term water diversions also have contributed to reductions in the phytoplankton and zooplankton populations in the Delta, as well as to alterations in nutrient cycling within the Delta ecosystem.

5.1.2.1.5 Nearshore Coastal Marine and Offshore Marine Areas

Spring-run Chinook salmon reside in the Pacific Ocean from 1 to 4 years. The first few months of a salmon's ocean life have been identified as the period of critical climatic influences on survival which, in turn, suggests that coastal and estuarine environments are key areas of biophysical interaction (NMFS 2014). Juvenile salmon grow rapidly as they feed in the highly productive currents along the continental shelf (Barnhart 1986).

Most climate factors affect the entire West Coast complex of salmonids. This is particularly true in their marine phase, because the California populations are believed to range fairly broadly along the coast and intermingle, and climate impacts in the ocean occur over large spatial scales (Schwing 2009, as cited in NMFS 2014). Salmon and steelhead residing in coastal areas where upwelling is the dominant process are more sensitive to climate-driven changes in the strength and timing of upwelling (NMFS 2014).

Oceanic and climate conditions such as sea surface temperatures, air temperatures, strength of upwelling, El Niño events, salinity, ocean currents, wind speed, and primary and secondary productivity affect all facets of the physical, biological and chemical processes in the marine environment. Some of the conditions associated with El Niño events include warmer water temperatures, weak upwelling, low primary productivity (which leads to decreased zooplankton biomass), decreased southward transport of subarctic water, and increased sea levels (Pearcy 1997 as cited in NMFS 2014). Strong upwelling is probably beneficial because it causes greater transport of smolts offshore, beyond major concentrations of inshore predators (Pearcy 1997 as cited in NMFS 2014).

The California Current Ecosystem (CCE) is designated by NMFS as one of eight large marine ecosystems within the United States Exclusive Economic Zone. The California Current begins at the northern tip of Vancouver Island, Canada and ends somewhere between Punta Eugenia and the tip of Baja California, Mexico (NMFS 2009, as cited in NMFS 2014). The northern end of the current is dominated by strong seasonal variability in winds, temperature, upwelling, plankton production and the spawning times of many fishes, whereas the southern end of the current has much less seasonal variability (NMFS 2014). The primary issue for the CCE is the onset and length of the upwelling season, that is when upwelling begins and ends (i.e., the "spring" and "fall" transitions). The biological transition date provides an estimate of when seasonal cycles of significant plankton and euphausiid production are initiated (NMFS 2014).

5.1.3 Historical Abundance and Distribution

Spring-run Chinook salmon were once the most abundant run of salmon in the Central Valley (Campbell and Moyle 1991, as cited in Yoshiyama et al. 1998) and were found in both the Sacramento and San Joaquin drainages. The Central Valley drainage as a whole is estimated to have supported annual runs of spring-run Chinook salmon as large as 600,000 fish between the late 1880s and 1940s (CDFG 1998). More than 500,000 spring-run Chinook salmon were reportedly caught in the Sacramento-San Joaquin commercial fishery in 1883 alone (Yoshiyama et al. 1998). Before the construction of Friant Dam (completed in 1942), nearly 50,000 adults were counted in the San Joaquin River (Fry 1961). The San Joaquin populations were essentially extirpated by the 1940s, with only small remnants of the run that persisted through the 1950s in the Merced River (Hallock and Van Woert 1959; Yoshiyama et al. 1998).

Annual run sizes of spring-run Chinook salmon are reported in *GrandTab*, a database administered by Cal Fish and Wildlife for the Central Valley that includes reported run size estimates from 1960 through 2012, although mainstem Sacramento River estimates are not available for years before 1969 (CDFW 2013). The Central Valley spring-run Chinook salmon ESU has displayed broad fluctuations in adult abundance. Estimates of spring-run Chinook salmon in the Sacramento River and its tributaries (not including the lower Yuba and Feather rivers because GrandTab does not distinguish between fall-run and spring-run Chinook salmon in-river spawners, and not including the FRFH) have ranged from 1,404 in 1993 to 25,890 in 1982.

The average abundance for the Sacramento River and its tributaries (excluding the lower Yuba and Feather rivers – see above) was 11,646 for the period extending from 1970 through 1979, 14,240 for the period 1980 through 1989, 5,825 for the period 1990 through 1999, and 14,055 for the period 2000 through 2009. Since 1995, spring-run Chinook salmon annual run size estimates typically have been dominated by Butte Creek returns. Since carcass survey estimates have been available in Butte Creek in 2001 through 2015, Butte Creek returns have averaged 10,226 fish. The estimated spring-run Chinook salmon run size on Butte Creek was 16,317 for 2012. The highest reported spring-run Chinook salmon run size was 16,782 in 2013. Since then, the annual sizes have been lower, with 5,083 and 569 in 2014 and 2015, respectively.

Historically, spring-run Chinook salmon occurred in the headwaters of all major river systems in the Central Valley where natural barriers to migration were absent, and occupied the middle and upper elevation reaches (1,000 to 6,000 ft) of most streams and rivers with sufficient habitat for over-summering adults (Clark 1929). Excluding the lower stream reaches that were used as adult migration corridors (and, to a lesser degree, for juvenile rearing), it has been estimated that at least 72 percent of the original Chinook salmon spawning and holding habitat in the Central Valley drainage is no longer available due to the construction of non-passable dams (Yoshiyama et al. 2001). Adult migrations to the upper reaches of the Sacramento, Feather, and Yuba rivers were eliminated with the construction of major dams during the 1940s, 1950s and 1960s. Naturally spawning populations of spring-run Chinook salmon have been reported to be restricted to accessible reaches of the upper Sacramento River, Antelope Creek, Battle Creek, Beegum Creek, Big Chico Creek, Butte Creek, Clear Creek, Deer Creek, Mill Creek, Feather River, and the Yuba River (CDFG 1998).

Historically, the Yuba River Basin reportedly was one of the most productive habitats for runs of Chinook salmon and steelhead (Yoshiyama et al. 1996). Although it is not possible to estimate the numbers of spawning fish from historical data, CDFG (1993) suggested that the Yuba River "*historically supported up to 15% of the annual run of fall-run Chinook salmon in the Sacramento River system*" (Yoshiyama et al. 1996).

By the late 1800s, anadromous fish populations were experiencing significant declines, primarily because of mining activities and resultant extreme sedimentation following flood events (McEwan 2001; Yoshiyama et al. 2001). As an example, the flood of 1861–1862 buried much of the bottomlands along the lower Yuba River under sand deposits averaging 2 to 7 ft deep (Kelley 1989). By 1876, the channel of the lower Yuba River reportedly had become completely filled, and what remained of the adjoining agricultural lands was covered with sand and gravel (Kelley 1989; CDFG 1993) — a marked deterioration of the river as salmon habitat (Yoshiyama et al. 2001).

To control flooding and the downstream movement of sediment, construction of several man-made instream structures on the Yuba River occurred during the early 1900s. A structure referred to as Barrier No. 1, built in 1904–1905, was located 1 mi below Parks Bar Bridge near Smartsville and was destroyed by flood waters in March 1907 (Sumner and Smith 1939). This barrier probably hindered salmon upstream movement (Sumner and Smith 1939). In 1906, the California Debris Commission, a partnership between the Federal Government and the State of California, constructed Daguerre Point Dam, specifically to hold back mining debris. In 1910, the Yuba River was diverted over the new dam. This approximately 24-ft high dam retained the debris, but made it difficult for spawning fish to migrate upstream, although salmon reportedly did surmount the dam in occasional years because they were reportedly observed in large numbers in the North Yuba River at Bullards Bar during the early 1920s (Yoshiyama et al. 2001). Two fishways, one for low water and the other for high water, were constructed at Daguerre Point Dam prior to the floods of 1927-1928 (Clark 1929), when the fish ladders were destroyed, and were not replaced until 1938, leaving a 10-year period when upstream fish passage at Daguerre Point Dam was blocked (CDFG 1991a). A fish ladder was constructed at the south end of Daguerre Point Dam in 1938 and was generally ineffective (CDFG 1991a), but during the fall of 1938, *"several salmon were reported seen below the Colgate Head Dam on the North Fork of the Yuba, 35 miles above Daguerre Point Dam."* (Sumner and Smith 1939).

Upstream of Daguerre Point Dam, the USACE's 260-ft high Englebright Dam was authorized in 1935 to hold back hydraulic mining debris, and was constructed in 1941 by the California Debris Commission. Englebright Dam was not authorized to provide fish passage, therefore it has no fish ladders and blocks anadromous fish access to all areas upstream of the dam (Eilers 2008; PG&E 2008; DWR 2009b). The dam restricts anadromous fish to the lower 24 mi of the Yuba River.

There is limited information on the historical population size of spring-run Chinook salmon in the Yuba River. Historical accounts indicate that "large numbers" of Chinook salmon may have been present as far upstream as Downieville on the North Fork Yuba River (Yoshiyama et al. 1996). Due to their presence high in the watershed, Yoshiyama et al. (1996) concluded that these fish were spring-run Chinook salmon.

For the Middle Fork Yuba River, Yoshiyama et al. (2001) concluded that direct information was lacking on historic abundance and distribution of salmon, and they conservatively considered the 10-ft falls located 1.5 mi above the mouth of the Middle Fork Yuba River was the upstream limit of salmon distribution.

Yoshiyama et al. (2001) report that little is known of the original distribution of salmon in the South Fork Yuba River where the Chinook salmon population was severely depressed and upstream access was obstructed by dams when Cal Fish and Wildlife began surveys in the 1930s.

Sumner and Smith (1939) stated that the "*South Fork of the Yuba is not considered an angling stream in its 24 miles below the mouth of Poorman Creek, where slickens* (pulverized rock) from the Spanish Mine turns the river a muddy grey*." They also reported that in "*Poorman Creek, cyanide poisoning may have done more harm than the slickens… It was evident that some strong poison was entering the stream with the tailings. An occasional heavy dose of cyanide would kill off fish and fish food…*" Yoshiyama et al. (2001) consider the cascade, with at least a 12-ft drop, located 0.5-mi below the juncture of Humbug Creek, as essentially the historical upstream limit of salmon during most years of natural streamflows.

Clark (1929) reported that the salmon spawning grounds extended from the mouth of the Yuba River upstream to the town of Smartsville, but that very few salmon (evidently spring-run) went farther upstream past that point. Sumner and Smith (1940) report that salmon ascended in considerable numbers up to Bullard's Bar Dam on the North Fork Yuba River while it was being constructed (1921-1924). In their 1938 survey of Yuba River salmon populations, Sumner and Smith (1940) stated that the height of the dams in the Yuba River blocked all potential salmon and steelhead runs upstream of the barriers (Sumner and Smith 1940). However, Sumner and Smith (1940) describe the ladders as "*a rather ineffectual fishway... That few fish have been able to use it... is testified to by the almost universal belief among local residents that at present no fish ever come above the dam*." In addition, the fall-run Chinook salmon run was reportedly destroyed at least temporarily, and many miles of streams rendered unfit for trout (Sumner and Smith 1939).

In 1951, two functional fish ladders were installed at Daguerre Point Dam by the State of California and it was stated that "*With ladders at both ends, the fish have no difficulty negotiating this barrier at any water stage.*" (CDFG 1953).

CDFG (1991a) reports that a small spring-run Chinook salmon population historically occurred in the lower Yuba River, but the run virtually disappeared by 1959, presumably due to the effects of water diversion and hydraulic developments on the river (Fry 1961). As of 1991, a remnant spring-run Chinook salmon population reportedly persisted in the lower Yuba River downstream of Englebright Dam, maintained by fish produced in the lower Yuba River, fish straying from the Feather River, or fish previously and infrequently stocked from the FRFH (CDFG 1991a).

In the 1990s, relatively small numbers of Chinook salmon that exhibit spring-run phenotypic characteristics were observed in the lower Yuba River (CDFG 1998). Although precise escapement estimates are not available, the USFWS testified at the 1992 SWRCB lower Yuba River hearing that "…*a population of about 1,000 adult spring-run Chinook salmon now exists in the lower Yuba River*" (San Francisco Bay RWQCB 2006 as cited in NMFS 2009a).

5.1.4 Summary of Past and Ongoing Fisheries Studies on the Lower Yuba River

As stated in YCWA (2010), the Yuba River downstream of Englebright Dam is one of the more thoroughly studied rivers in the Central Valley of California. A description of existing information regarding salmonid populations in the Yuba River downstream of Englebright Dam is contained in Attachment 7-8A to Technical Memorandum 7-8, *ESA/CESA-Listed Salmonids Downstream of Englebright Dam*, which can be found on FERC's eLibrary as referenced by the FERC accession number provided in Table E6-2 of Appendix E6, of YCWA's Amended FLA. In Appendix E6 of YCWA's Amended FLA, Technical Memorandum 7-8 summarizes the available literature for spring-run Chinook salmon where specifically identified, Chinook salmon in general where runs are not specifically identified, and *O. mykiss*. Much of the referenced information discusses both runs of Chinook salmon and *O. mykiss*, and therefore is presented in its entirety in Technical Memorandum 7-8. The technical memorandum describes available field studies and data collection reports, other relevant documents, and ongoing data collection, monitoring and evaluation activities including the Yuba Accord M&E Program and other data collection and monitoring programs. Technical Memorandum 7-8 summarily describes 21 available field studies and data collection reports, 20 other relevant documents (e.g., plans, policies, historical accounts and regulatory compliance), 14 ongoing data collection, monitoring and evaluation activities for the M&E Program, and 4 other data collection and monitoring programs.

5.1.5 General Life History and Habitat Requirements

This section presents a general overview of lifestage-specific information (e.g., adult immigration and holding, adult spawning, embryo incubation, juvenile rearing and outmigration) for the Central Valley spring-run Chinook salmon ESU. Then, this section specifically focuses and provides information on lifestage-specific temporal and spatial distributions for spring-run Chinook salmon in the lower Yuba River.

Four distinct runs of Chinook salmon spawn in the Sacramento-San Joaquin River system, with each run named for the season when the majority of the run enters freshwater as adults. The primary characteristic distinguishing spring-run Chinook salmon from the other runs of Chinook salmon is that adult spring-run Chinook salmon enters their natal streams during the spring, and hold in areas downstream of spawning grounds during the summer months until their eggs fully develop and become ready for spawning.

The RMT developed representative temporal distributions for specific spring-run Chinook salmon lifestages in the lower Yuba River through review of previously conducted studies, as well as recent and currently ongoing data collection activities of the M&E Program (Table 5.1-1). The resultant lifestage periodicities encompass the majority of activity for a particular lifestage, and are not intended to be inclusive of every individual in the population (RMT 2010b; RMT 2013a).

5.1.5.1 Adult Immigration and Holding

Adult spring-run Chinook salmon immigration and holding in California's Central Valley has been reported to occur from mid-February through September (CDFG 1998; Lindley et al. 2004). Spring-run Chinook salmon are known to use the Sacramento River primarily as a migratory corridor to holding and spawning areas located in upstream tributaries. For the mainstem Sacramento River, all of the potential spring-run Chinook salmon holding habitat is located upstream from the RBDD and downstream of Keswick Dam (CDFG 1998).

Table 5.1-1. Lifestage-specific periodicities for spring-run Chinook salmon in the lower Yuba River.

Lifestage	Jan		Feb		Mar		Apr		May		Jun		Jul		Aug		Sep		Oct		Nov		Dec	
SPRING-RUN CHINOOK SALMON																								
Adult Immigration & Holding							■	■	■	■	■	■	■	■	■	■	■	■						
Spawning																	■	■	■					
Embryo Incubation																	■	■	■	■	■	■	■	■
Fry Rearing	■	■	■																			■	■	■
Juvenile Rearing	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■
Juvenile Downstream Movement	■	■	■	■	■	■	■	■	■	■	■	■										■	■	■
Smolt (Yearling+) Emigration	■	■	■	■	■	■	■	■	■										■	■	■	■	■	■

Source: RMT 2013.

Suitable water temperatures for adult upstream migration reportedly range between 57ºF and 67ºF (NMFS 1997). In addition to suitable water temperatures, adequate flows are required to provide migrating adults with olfactory and other cues needed to locate their spawning reaches (CDFG 1998). The primary characteristic distinguishing spring-run Chinook salmon from the other runs of Chinook salmon is that adult spring-run Chinook salmon hold in areas downstream of spawning grounds during the summer months until their eggs fully develop and become ready for spawning. NMFS (1997) states, "*Generally, the maximum temperature for adults holding, while eggs are maturing, is about 59-60°F, but adults holding at 55-56°F have substantially better egg viability*."

For the lower Yuba River, adult spring-run Chinook salmon immigration and holding has previously been reported to primarily occur from March through October (Vogel and Marine 1991; YCWA et al. 2007), with upstream migration generally peaking in May (SWRI 2002). The RMT's examination of preliminary data obtained since the VAKI Riverwatcher™ infrared and videographic sampling system has been operated (2003–present) found variable temporal modalities of Chinook salmon ascending the fish ladders at Daguerre Point Dam. RMT (2013a) identified the spring-run Chinook salmon adult immigration and holding period as extending from April through September.

Previously, it has been reported that spring-run Chinook salmon in the lower Yuba River hold over during the summer in the deep pools and cool water downstream of the Narrows 1 and Narrows 2 powerhouses, or further downstream in the Narrows Pool (CDFG 1991a; SWRCB 2003), where water depths can exceed 40 ft (YCWA et al. 2007). Congregations of adult Chinook salmon (approximately 30 to 100 fish) have been observed in the outlet pool at the base of the Narrows 2 Powerhouse, generally during late August or September. During this time period, the pool becomes clear enough to see the fish (M. Tucker, NMFS, pers. comm. 2003; S. Onken, YCWA, pers. comm. 2004). While it is difficult to visually distinguish spring-run from fall-run Chinook salmon in this situation, the fact that these fish are congregated this far up the river at this time of year indicates that some of them are likely to be spring-run Chinook salmon (NMFS 2007).

Past characterizations of spring-run Chinook salmon distributions from available literature on the lower Yuba River have provided some anecdotal references to behavioral run details (e.g., migration timing and areas of holding and spawning), but the referenced information has not provided or referenced the basis for these descriptions. Spring-run Chinook salmon have been reported to migrate immediately to areas upstream of the Highway 20 Bridge after entering the Yuba River from March through October (Vogel and Marine 1991; YCWA et al. 2007), and then over-summer in deep pools located downstream of the Narrows 1 and 2 powerhouses, or further downstream in the Narrows Reach through the reported spawning period of September through November (CDFG 1991a; SWRCB 2003).

The RMT's (2013a) examination of preliminary data obtained since the VAKI Riverwatcher™ infrared and videographic sampling system has been operated (2003 – present) found variable temporal modalities of Chinook salmon ascending the fish ladders at Daguerre Point Dam. The RMT's 3-year acoustic telemetry study of adult spring-run Chinook salmon tagged downstream of Daguerre Point Dam during the phenotypic adult upstream migration period has provided new information to better understand adult spring-run Chinook salmon temporal and spatial distributions in the lower Yuba River. The results from the VAKI Riverwatcher™ monitoring, and particularly from the acoustic telemetry study found past characterizations of temporal and spatial distributions to be largely unsupported, as phenotypic adult spring-run Chinook salmon were observed to exhibit a much more diverse pattern of movement, and holding locations in the lower Yuba River were more expansive than has been previously reported (RMT 2013a).

Although some of the acoustically-tagged spring-run Chinook salmon were observed to adhere to other previously reported characterizations, observations from the telemetry study also identified that a large longitudinal extent of the Yuba River was occupied by the tagged phenotypic adult spring-run Chinook salmon during immigration and holding periods (Figure 5.1-1). Figure 5.1-1 displays all individual fish detections obtained during the RMT's mobile acoustic tracking surveys conducted from May 2009 until November 2011 (RMT 2013a).

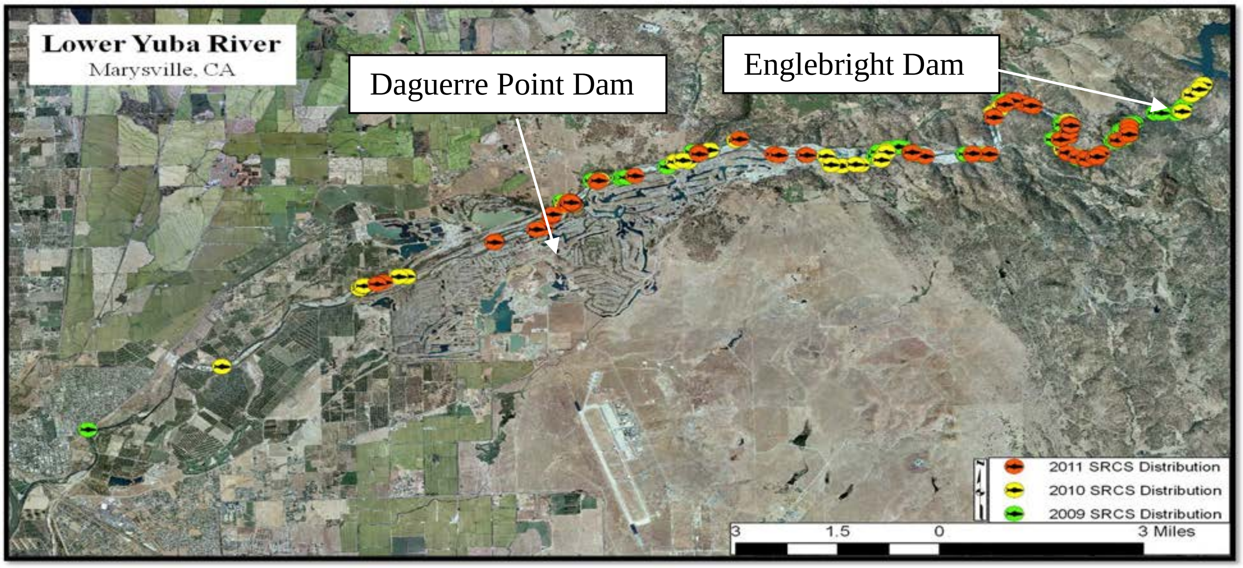

Figure 5.1-1. Spatial distribution of all individual acoustically-tagged adult phenotypic spring-run Chinook salmon (SRCS) detections obtained from the mobile tracking surveys conducted during 2009, 2010 and 2011.
(Source: RMT 2013a)

Also, temporal migrations to areas upstream of Daguerre Point Dam occurred over an extended period of time (Figure 5.1-2). The tagged phenotypic adult spring-run Chinook salmon in the lower Yuba River actually migrated upstream of Daguerre Point Dam from May through September, and utilized a broad expanse of the lower Yuba River during the summer holding period, including areas as far downstream as Simpson Lane Bridge (i.e., ~RM 3.2), and as far upstream as the area just below Englebright Dam. A longitudinal analysis of acoustic tag detection data indicated that distributions were non-random, and that the tagged spring-run Chinook salmon were selecting locations for holding.

The area of the river between Daguerre Point Dam and the Highway 20 Bridge was largely used as a migratory corridor by the tagged adult spring-run Chinook salmon during all 3 years of the study (RMT 2013a). Telemetry data in this area demonstrated relatively brief periods of occupation, characterized by sequential upstream detections as individually-tagged fish migrated through this area. By contrast, frequent and sustained detections were observed from the Highway 20 Bridge upstream to Englebright Dam (RMT 2013a).

Examination of individual detection data indicated that tagged phenotypic adult spring-run Chinook salmon that moved upstream of Daguerre Point Dam had generally passed through the Daguerre Point Dam fish ladders by the end of September during all 3 years (RMT 2013a). Acoustic tag detection data were used to discern tagged spring-run Chinook salmon residing in holding areas during June, July and August, and shifting to spawning areas during September into early October.

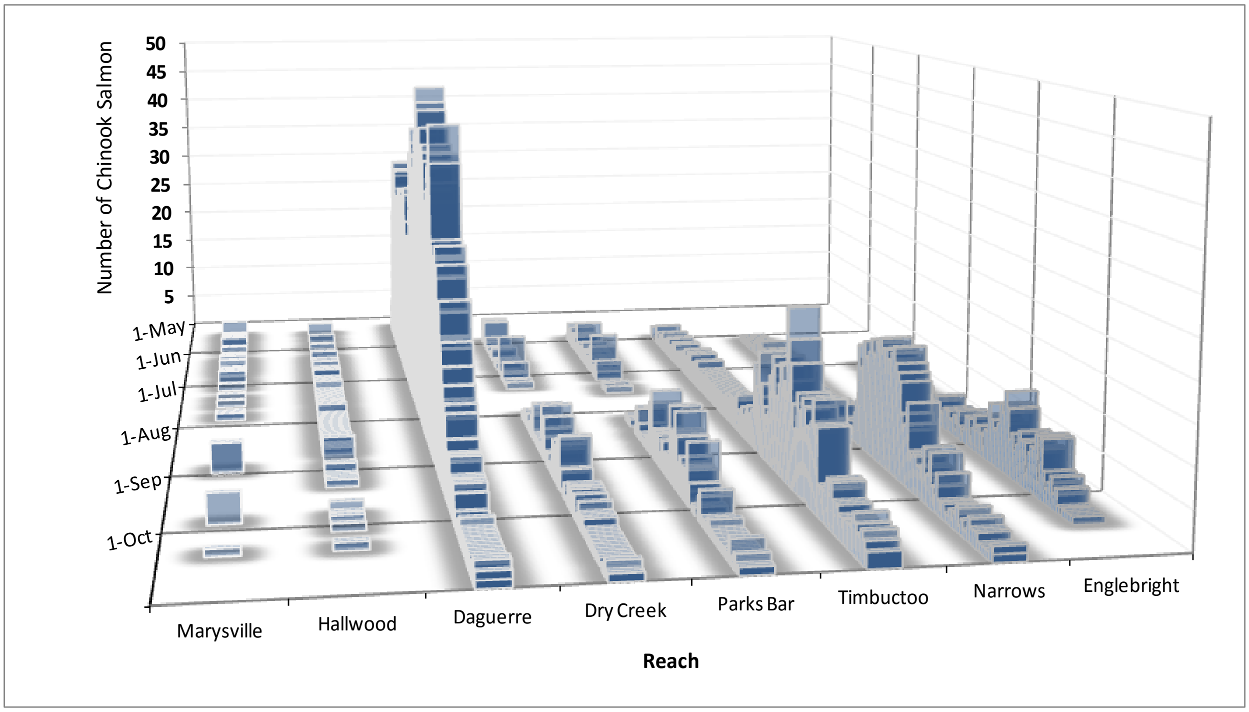

Figure 5.1-2. Spatial and temporal distribution of all individual acoustically-tagged adult phenotypic spring-run Chinook salmon detected from the mobile tracking surveys conducted during 2009, 2010 and 2011 in the lower Yuba River.
(Source: RMT 2013a)

This observation was repeated during all 3 years of the study, and in all occupied reaches. Telemetry data demonstrated that the majority of tagged phenotypic adult spring-run Chinook salmon that ascended the ladders at Daguerre Point Dam also continued to move farther upstream to the Timbuctoo, Narrows, and Englebright Dam reaches during September, coincident with the initiation of spawning activity (RMT 2013a).

YCWA (2013) used the RMT's 2009-2011 acoustic tagging study data to evaluate movements of the individual acoustically-tagged spring-run Chinook salmon and potential relationships between changes in flow. Visual examination of the time series plots of daily locations of individual acoustically-tagged Chinook salmon and mean daily flows at the Smartsville gage showed highly variable behavior among individuals on a daily basis within and among years. However, several general patterns of fish movement in relationship to flow are apparent.

- Upstream movement coinciding with an increase in flow
- Upstream movement coinciding with a decrease in flow
- Downstream movement coinciding with a decrease in flow
- Upstream movement occurring after an increase in flow

YCWA (2013) found that most of the individual movements of acoustically-tagged spring-run Chinook salmon potentially associated with a change in Smartsville flow were abrupt upstream movements occurring concurrently with, and in the days following, a noticeable decrease in flow.

Observed movements of individual spring-run Chinook salmon identified during 2009 generally occurred within the time period from about mid-May to early September, and generally occurred over a period ranging from 1 to 9 days. Most of the observed movements identified during 2010 occurred during early to mid-June, with a few movements occurring during August, and generally occurred over a period ranging from about 1 to 7 days. The identified movements during 2011 generally occurred during late August into early September, and generally occurred over a period ranging from about 1 to 5 days. Because spring-running Chinook salmon immigrated into the lower Yuba River later in 2011 than during 2009 and 2010, and were not captured and acoustically-tagged until July, no potential relationships between fish movement and flow reductions during the spring months could be evaluated for 2011.

More than half (40 out of 60) of the identified movements of Chinook salmon over the 3 years that were potentially associated with a concurrent change in flow consisted of upstream movements coinciding with a large decrease in flow (measured at the Smartsville gage). Most of the identified upstream movements occurring coincident to a decrease in flow occurred when flow decreased substantially during a 1 to 2 week period in late August to early September and/or during a 1 to 2 week period during May or June, depending on the year. In other words, the most common potential relationship identified between spring-run Chinook salmon movement and flow was an abrupt and continued movement upstream to the upper reaches during a large reduction in mean daily Smartsville flow (38 percent to 68 percent reduction in flow) occurring over about 1 to 2 weeks.

5.1.5.2 Adult Spawning

In the Central Valley, spawning has been reported to primarily occur from September to November, with spawning peaking in mid-September (DWR 2004; Moyle 2002; Vogel and Marine 1991). Within the ESU, spring-run Chinook salmon spawn in accessible reaches of the upper Sacramento River, Antelope Creek, Battle Creek, Beegum Creek, Big Chico Creek, Butte Creek, Clear Creek, Deer Creek, Mill Creek, Feather River, and the Yuba River (CDFG 1998).

All of the potential spring-run Chinook salmon spawning habitat in the mainstem Sacramento River is located upstream from the RBDD and downstream of Keswick Dam (CDFG 1998). It has been reported that in some years high water temperatures could prevent spring-run Chinook salmon egg and embryo survival (USFWS 1990 as cited in CDFG 1998).

In general, Central Valley spring-run Chinook salmon has been reported to spawn at the tails of holding pools (Moyle 2002; NMFS 2007). Redd sites are apparently chosen in part by the presence of subsurface flow. Chinook salmon usually seek a mixture of gravel and small cobbles with low silt content to build their redds. Characteristics of spawning habitats that are directly related to flow include water depth and velocity. Chinook salmon spawning reportedly occurs in water velocities ranging from 1.2 feet per second (ft/s) to 3.5 ft/s, and spawning typically occurs at water depths greater than 0.5 ft (YCWA et al. 2007).

For the lower Yuba River, the spring-run Chinook salmon spawning period has previously been reported to extend from September through November (CDFG 1991a; YCWA et al. 2007). Limited reconnaissance-level redd surveys conducted by Cal Fish and Wildlife since 2000 during late August and September have detected spawning activities beginning during the first or second week of September. They have not detected a bimodal distribution of spawning activities (i.e., a distinct spring-run spawning period followed by a distinct fall-run Chinook salmon spawning period), and instead have detected a slow build-up of spawning activities starting in early September and transitioning into the main fall-run spawning period.

The RMT's (2013a) examination of the 2009, 2010 and 2011 acoustically-tagged spring-run Chinook salmon data revealed a consistent pattern in fish movement. In general, acoustically-tagged spring-run Chinook salmon exhibited an extended holding period, followed by a rapid movement into upstream areas (upper Timbuctoo Reach, Narrows Reach, and Englebright Reach) during September. Then, a period encompassing approximately one week was observed when fish held at one specific location, followed by rapid downstream movement. The approximate 1-week period appeared to be indicative of spawning events, which ended by the first week in October. These observations, combined with early redd detections and initial carcasses appearing in the carcass surveys (see below), suggest that the spring-run Chinook salmon spawning period in the lower Yuba River may be of shorter duration than previously reported, extending from September 1 through mid-October (RMT 2013a).

The earliest spawning (presumed to be spring-run Chinook salmon) generally occurs in the upper reaches of the highest quality spawning habitat (i.e., below the Narrows pool) and progressively moves downstream throughout the fall-run Chinook salmon spawning season (NMFS 2007). Spring-run Chinook salmon spawning in the lower Yuba River is believed to occur upstream of Daguerre Point Dam. USFWS (2007) collected data from 168 Chinook salmon redds in the lower Yuba River on September 16-17, 2002 and September 23-26, 2002, considered to be spring-run Chinook salmon redds. The redds were all located above Daguerre Point Dam. During the pilot redd survey conducted from the fall of 2008 through spring of 2009, the RMT (2010c) report that the vast majority (96%) of fresh Chinook salmon redds constructed by the first week of October 2008, potentially representing spring-run Chinook salmon, were observed upstream of Daguerre Point Dam. Similar distributions were observed during the 2010 and 2011 redd surveys, when weekly redd surveys were conducted. About 97 and 96 percent of the fresh Chinook salmon redds constructed by the first week of October were observed upstream of Daguerre Point Dam during 2009 and 2010, respectively (RMT 2013a).

5.1.5.3 Embryo Incubation

The spring-run Chinook salmon embryo incubation period encompasses the time period from egg deposition through hatching, as well as the additional time while alevins remain in the gravel while absorbing their yolk sacs prior to emergence.

The length of time for spring-run Chinook salmon embryos to develop depends largely on water temperatures. In well-oxygenated intragravel environs where water temperatures range from about 41°F to 55.4°F embryos hatch in 40 to 60 days and remain in the gravel as alevins for another 4 to 6 weeks, usually after the yolk sac is fully absorbed (NMFS 2009a). In Butte and

Big Chico creeks, emergence occurs from November through January, and in the colder waters of Mill and Deer creeks, emergence typically occurs from January through as late as May (Moyle 2002).

In the lower Yuba River, the RMT (2013a) concluded that spring-run Chinook salmon embryo incubation period generally extends from September through December.

5.1.5.4 Juvenile Rearing and Outmigration

After emerging, Chinook salmon fry tend to seek shallow, nearshore habitat with slow water velocities and move to progressively deeper, faster water as they grow. However, fry may disperse downstream, especially if high-flow events correspond with emergence (Moyle 2002). Spring-run juveniles may emigrate as fry soon after emergence, rear in their natal streams for several months prior to emigration as young-of-year (YOY), or remain in their natal streams for extended periods and emigrate as yearlings. Information regarding the duration of rearing and timing of emigration of spring-run Chinook salmon in the Central Valley is summarized in NMFS (2009a), much of which is presented herein.

Upon emergence from the gravel, juvenile spring-run Chinook salmon may reside in freshwater for 12 to 16 months, but some migrate to the ocean as YOY fish in the winter or spring months within 8 months of hatching (CALFED 2000a). The average size of fry migrants (approximately 40 millimeters (mm) between December and April in Mill, Butte and Deer creeks) reflects a prolonged emergence of fry from the gravel (Lindley et al. 2004).

The timing of juvenile emigration from the spawning and rearing grounds varies among the tributaries of origin, and can occur during the period extending from October through April (Vogel and Marine 1991). Studies in Butte Creek (Ward et al. 2003) found the majority of spring-run migrants to be fry, moving downstream primarily during December, January and February, and that these movements appeared to be influenced by flow. Small numbers of spring-run juveniles remained in Butte Creek to rear and migrate later in the spring. Some juveniles continue to rear in Butte Creek through the summer and emigrate as yearlings from October to February, with peak yearling emigration occurring in November and December (CDFG 1998). Juvenile emigration patterns in Mill and Deer creeks are very similar to patterns observed in Butte Creek, with the exception that Mill and Deer creeks juveniles typically exhibit a later YOY migration and an earlier yearling migration (Lindley et al. 2004). In contrast, data collected on the Feather River suggests that the bulk of juvenile emigration occurs during November and December (Painter et al. 1977). Seesholtz et al. (2003) speculate that because juvenile rearing habitat in the Low Flow Channel of the Feather River is limited, juveniles may be forced to emigrate from the area early due to competition for resources.

In general, juvenile Chinook salmon have been collected by electrofishing and observed by snorkeling throughout the lower Yuba River, but with higher abundances above Daguerre Point Dam (Beak 1989; CDFG 1991a; Kozlowski 2004). This may be due to larger numbers of spawners, greater amounts of more complex, high-quality cover, and lower densities of predators such as striped bass (*Morone saxatilis*) and American shad (*Alosa sapidissima*), which reportedly are restricted to areas below the dam (YCWA et al. 2007). During juvenile rearing and outmigration, salmonids prefer stream margin habitats with sufficient depths and velocities to provide suitable cover and foraging opportunities. Juvenile Chinook salmon reportedly utilize river channel depths ranging from 0.9 ft to 2.0 ft, and most frequently are in water with velocities ranging from 0 ft/s to 1.3 ft/s (Raleigh et al. 1986).

To better understand juvenile habitat use in the lower Yuba River, the RMT (2013) conducted a series of habitat use surveys employing snorkel methods to assess juvenile fish communities, the timing of juvenile fish presence in the river, and the areas used by juvenile fishes. Snorkel surveys were performed in seven reaches along the lower Yuba River (Figure 5.1-3) during January, February, March, June, and September of 2012.

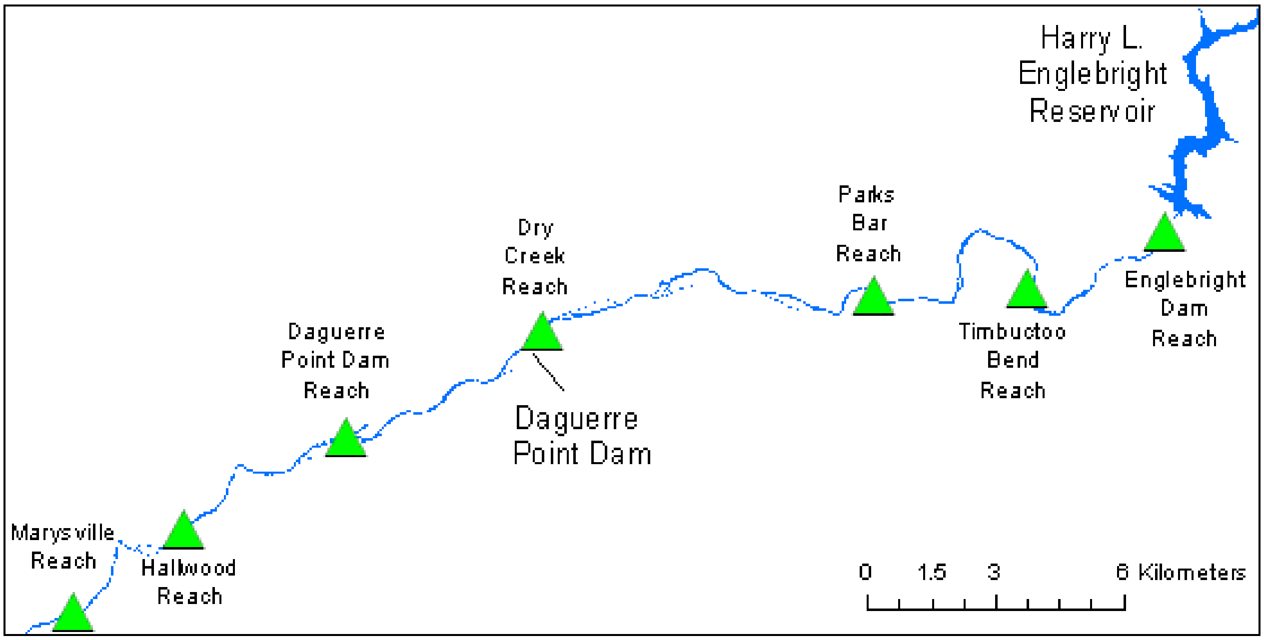

Figure 5.1-3. RMT juvenile snorkeling survey site locations on the lower Yuba River.
Source: RMT 2013a

The RMT's juvenile fish snorkel survey observed a total of eight fish species among all of the survey periods throughout the surveyed reaches. Chinook salmon were the most frequently observed positively identified species, followed by Sacramento pikeminnow, and Sacramento sucker. Juvenile fishes occupied a number of different morphological units throughout the survey. Juvenile Chinook salmon occurred primarily in lateral bar, slackwater, slow glide, and riffle transition morphological units (MUs) (RMT 2013a).

To assess major trends in spatial patterns of juvenile Chinook salmon abundances in the lower Yuba River, the RMT (2013a) calculated the average density observed for each surveyed reach. The density of juvenile Chinook salmon was highly variable throughout the lower Yuba River. Observations indicated that, with the exception of the upstream-most survey reach (i.e., Englebright Dam Reach) the density of juvenile Chinook salmon generally was higher in the survey reaches located upstream rather than downstream of Daguerre Point Dam. However, there was no statistically significant difference in the mean density of observed juvenile Chinook salmon among reaches (ANOVA, $F_{6,25} = 1.09$, $P = 0.398$, Figure 5.1-4). Lower densities were observed in the Englebright Dam and Daguerre Point Dam reaches, and higher densities were observed in the Timbuctoo Bend and Dry Creek reaches (RMT 2013a).

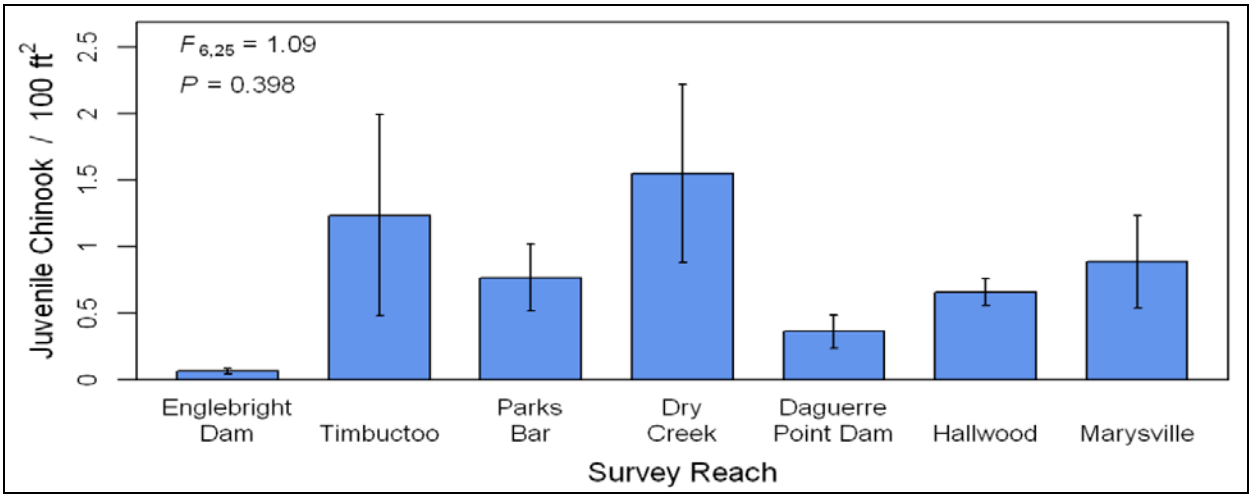

Figure 5.1-4. Observed densities of juvenile Chinook salmon across all survey reaches.
Source: RMT 2013a

The densities of Chinook salmon observations by survey month are shown in Figure 5.1-5, with significantly higher density during March than during January, June, and September (ANOVA, $F_{4,30} = 7.87$, $P < 0.001$) (RMT 2013a). A peak in juvenile Chinook salmon abundance was observed during March of 2012. This observation is supported in part from RST surveys in the lower Yuba River from 1999-2009, which identified peak emigration timing for juvenile Chinook salmon to occur from January through March. Fewer juvenile Chinook salmon were observed during January and February, with the lowest densities recorded during the June and September surveys. Emigration from the lower Yuba River may account for the decline in observed abundance of juvenile Chinook salmon as the survey months progressed (RMT 2013a).

When compared across sample reaches, juvenile Chinook salmon were observed further from shore in the Marysville survey reach than in other reaches (ANOVA $F_{6,4864} = 70.57$, $P < 0.001$) (RMT 2013a). When compared across sample months, juvenile Chinook salmon were generally located further from shore as the year progressed (ANVOA $F_{4,4866} = 24.39$, $P < 0.001$). Chinook salmon juveniles exhibited a similar pattern of observations farther from shore as they grew in size, although individuals in the 50-70 mm size class were observed closer to shore than smaller or larger size classes (ANOVA $F_{3,4867} = 60.69$, $P < 0.001$) (RMT 2013a).

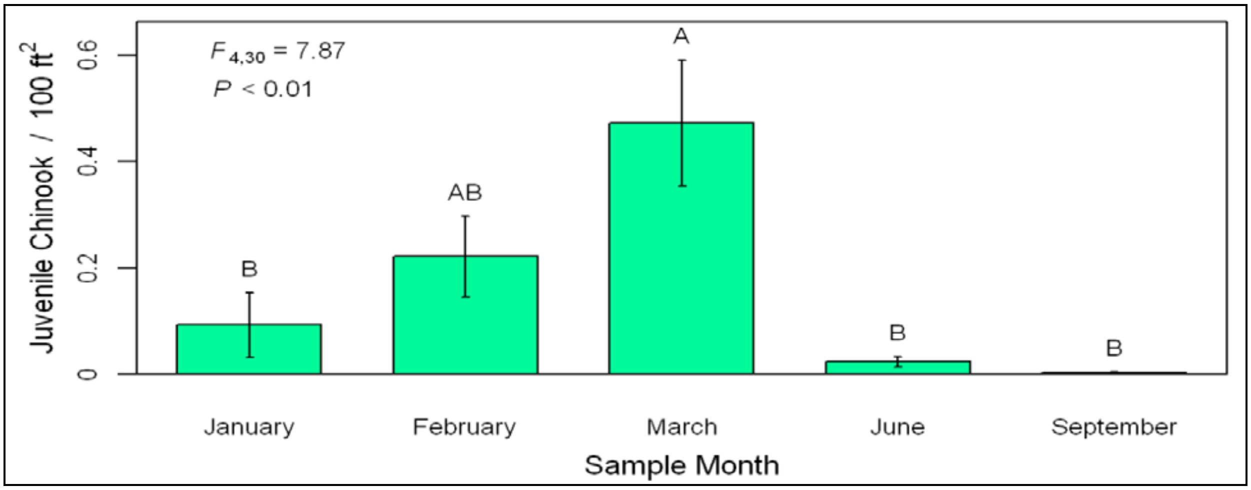

Figure 5.1-5. Observed densities of juvenile Chinook salmon during each survey month. Months sharing the same letter(s) indicate that densities were not significantly different.
Source: RMT 2013a

Juvenile Chinook salmon appeared to occupy areas in close proximity to the shore during most survey months and in most survey reaches. However, in the Marysville reach, juveniles were distributed considerably further from shore relative to the other reaches. The Marysville reach has an extended shallow sandy bar on the north bank on which large woody debris collects, which may provide refuge to juveniles away from the shoreline (RMT 2013a).

When compared across months, juvenile Chinook salmon remain within 10 ft of shore until June, and stayed relatively close to shore until September. Similarly, smaller juveniles tended to remain closer to shore than larger juvenile Chinook salmon. Both of these findings are consistent with observations of juvenile Chinook salmon occupying areas further from shore as they age (e.g., Allen 2000). However, juveniles in the 30-50 mm size class actually occupied a mean distance further from shore than individuals in the 50-70 mm size class (RMT 2013a).

To evaluate potential relationships between juvenile Chinook salmon observations and mesohabitat characteristics, mean column water velocity at 60 percent depth (or at an average of the 80 percent and 20 percent depth) and the total measured stream depth at each Chinook salmon observation were overlaid with the mesohabitat characterization plot by Wyrick and Pasternack (2012). In addition, potential relationships were evaluated separately between juvenile Chinook salmon observations by 20 mm size class and: 1) measured total stream depth; 2) the vertical position of the fish in the water column relative to total depth (depth of fish/total depth); and 3) the mean water velocity at Chinook salmon observation locations (RMT 2013a).

The general trends in mesohabitat occupation (as contrasted with morphological units) occupied by juvenile Chinook salmon throughout the survey are shown in Figure 5.1-6. As shown in the figure, juvenile Chinook salmon occupied primarily slackwater and slow glide mesohabitats, and were rarely encountered in water depths greater than 4.5 ft or velocities greater than 2 ft/s (RMT 2013a).

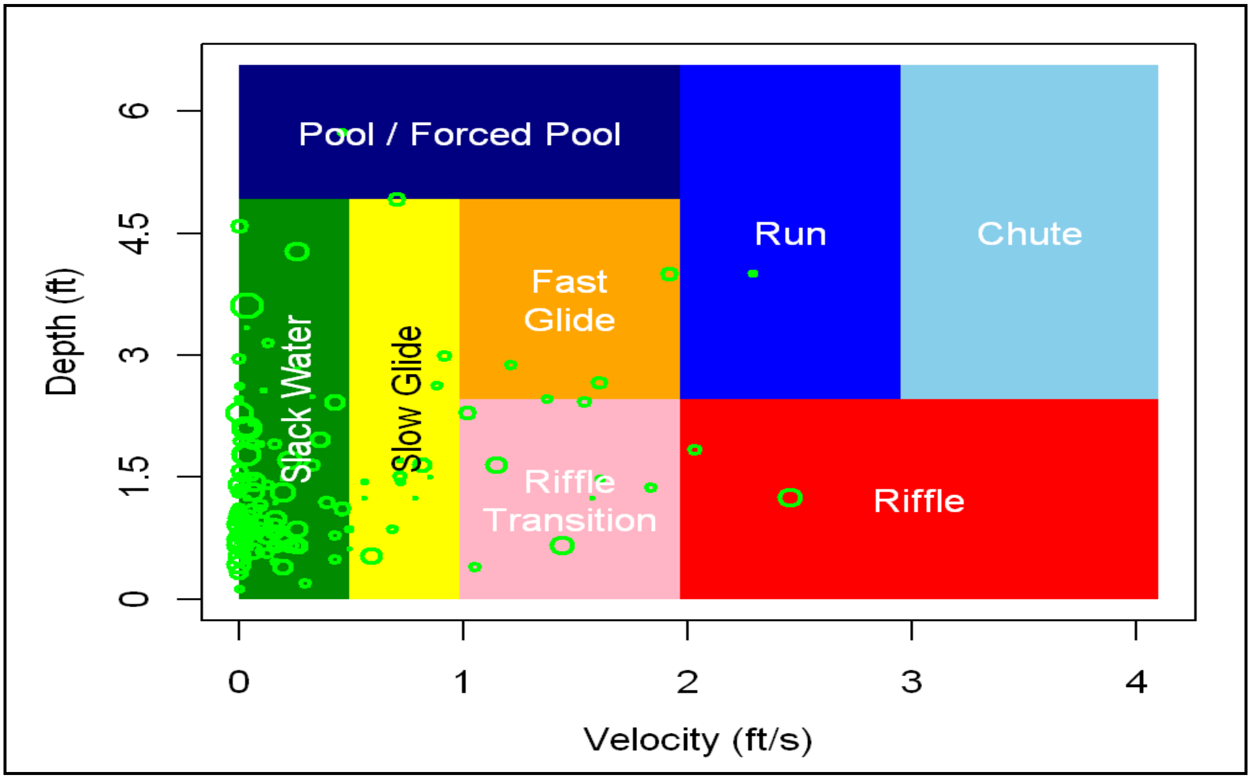

Figure 5.1-6. Overlay of the total measured stream depths and mean column water velocities at which juvenile Chinook salmon were encountered with mesohabitat characterizations. The size of the circle indicates the $\log_{10}$ transformed number of juveniles occurring at the total measured stream depth and mean column water velocity.
Source: RMT 2013a

To investigate potential relationships between different ages (using size) of juvenile Chinook salmon observations and mesohabitat characteristics, juvenile salmon were grouped by size class. Size classes of 90-110 mm and 110 mm+ were combined in this analysis due to their small sample sizes. Larger individuals tended to occur in deeper water than the individuals in the smallest size class (ANOVA, $F_{3,5538} = 28.94$, $P < 0.001$). The proportional depth (i.e., vertical position in the water column) where juvenile salmon were observed indicates a trend of increasingly deeper water utilization as the individuals grow, until a size larger than 90 mm is reached, at which time the larger juveniles were observed again closest to the shore (ANOVA, $F_{3,5538} = 62.35$, $P < 0.001$). Chinook salmon generally occurred in the lower half of the water column, regardless of actual water depth, and occurred in progressively faster water as they grew (ANOVA, $F_{3,5538} = 99.18$, $P < 0.001$) (RMT 2013a).

In summary, the vast majority of observations of juvenile Chinook salmon in the lower Yuba River occurred in water velocities and depths indicative of slackwater and slow glide mesohabitats (RMT 2013a). Juvenile Chinook salmon are known to prefer slower water habitats than many other members of *Oncorhynchus* (Quinn 2005), and have been previously reported to actively seek out slow backwaters, pools, or floodplain habitat for rearing (Sommer et al. 2001; Jeffres et al. 2008). The snorkeling data collected by the RMT during 2012 are generally consistent with other data available for multiple rivers (Bjornn and Reiser 1991). Juvenile Chinook salmon in the 30-50 mm size class tended to occupy shallower habitats than larger (and presumably older) individuals, which is consistent with other observations of salmonids (e.g., Bjornn and Reiser 1991). Similarly, juvenile Chinook salmon showed a clear preference for faster water (up to an average of about 1.8 ft/s) as they grew, consistent with trends found with salmonids in other rivers (Bjornn and Reiser 1991). Juveniles also preferred a station deeper in the water column as they aged, with the exception of fishes larger than 90 mm Fork Length (FL). Whether this reflects bias from a small sample size of juvenile Chinook salmon greater than 90 mm FL (n = 19), behavioral avoidance of snorkelers, or actual habitat preference is not known. The overall findings from this survey indicate that juvenile Chinook salmon in the lower Yuba River initially prefer slower, shallower habitat, and move into faster and deeper water as they grow (RMT 2013a). For additional detail regarding the RMT juvenile fish snorkeling studies, refer to RMT (2013a).

Based upon review of available information, the RMT (2010b) recently identified the spring-run Chinook salmon fry rearing period as extending from mid-November through March, the juvenile rearing period extending year-round, and the YOY emigration period extending from November through mid-July. Associated with the previously described shortened duration of spring-run Chinook salmon spawning, the fry rearing period is estimated to extend from mid-November through mid-February (RMT 2013a; 2013b). Updated characterization of the juvenile YOY emigration (i.e., downstream movement) period extends from mid-November through June (RMT 2013a).

In the lower Yuba River, Cal Fish and Wildlife has conducted juvenile salmonid outmigration monitoring by operating rotary screw traps (RSTs) near Hallwood Boulevard, located approximately 6 RM upstream from the city of Marysville. Cal Fish and Wildlife's RST monitoring efforts generally extended from fall (October or November) through winter, and either into spring (June) or through the summer (September) annually from 1999 to 2006. The RMT took over operation of the year-round RST effort in the fall of 2006, and continued operations through August 2009 (RMT 2013a).

Analyses of Cal Fish and Wildlife RST data indicate that most Chinook salmon juveniles move downstream past the Hallwood Boulevard location prior to May of each year. For the 5 years of data included in the analyses, 97.5 to 99.2 percent of the total numbers of juvenile Chinook salmon were captured by May 1 of each year. The percentage of the total juvenile Chinook salmon catch moving downstream past the Hallwood Boulevard location each year ranged from 0.4 to 1.3 percent during May, and 0 to 1.2 percent during June (YCWA et al. 2007). During the 2007/2008 sampling period, 95 percent of all juvenile Chinook salmon were captured by June 2, 2008 (Campos and Massa 2010a). Analysis of the fitted distribution of weekly juvenile Chinook salmon catch at the Hallwood Boulevard RST site from survey year 1999 through 2008 revealed that most emigration occurred from late-December through late-April in each survey year (RMT 2013a). Approximately 95 percent of the observed catch across all years based on the fitted distribution occurred by April 30 (RMT 2013a).

Overall, most (about 84 percent) of the juvenile Chinook salmon were captured at the Hallwood Boulevard RSTs soon after emergence from November through February, with relatively small numbers continuing to be captured through June. Although not numerous, captures of (oversummer) holdover juvenile Chinook salmon ranging from about 70 to 140 mm FL, primarily occurred from October through January with a few individuals captured into March (Massa 2005; Massa and McKibbin 2005). These fish likely reared in the river over the previous summer, representing an extended juvenile rearing strategy characteristic of spring-run Chinook salmon. During the 2007/2008 sampling period, 33 Chinook salmon within this size range were observed at the Hallwood Boulevard RST site from mid-December through January. Juvenile Chinook salmon captured during the fall and early winter (October-January) larger than 70 mm are likely exhibiting an extended rearing strategy in the lower Yuba River (Campos and Massa 2010a).

For the sampling periods extending from 2001 to 2005, Cal Fish and Wildlife identified specific runs based on sub-samples of lengths of all juvenile Chinook salmon captured in the RSTs by using the length-at-time tables developed by Fisher (1992, as cited in DWR 2003a), as modified by S. Greene (DWR 2003a). Although the veracity of utilization of the length-at-time tables for determining the run type of Chinook salmon in the Yuba River has not been ascertained, based on the examination of run-specific determinations, in the lower Yuba River the vast majority (approximately 94 percent) of spring-run Chinook salmon were captured as post-emergent fry during November and December, with a relatively small percentage (nearly 6 percent) of individuals remaining in the lower Yuba River and captured as YOY from January through March. Only 0.6 percent of the juvenile Chinook salmon identified as spring-run was captured during April, and only 0.1 percent during May, and none were captured during June (YCWA et al. 2007). The above summary of juvenile Chinook salmon emigration monitoring studies in the Yuba River is most consistent with the temporal trends of spring-run Chinook salmon outmigration reported for Butte and Big Chico creeks (YCWA et al. 2007).

5.1.5.5 Smolt Emigration

For the Central Valley, it has been reported that while some spring-run Chinook salmon emigrate from natal streams soon after emergence during the winter and early-spring (NMFS 2004b), some may spend as long as 18 months in freshwater and move downstream as smolts during the first high flows of the winter, which typically occur from November through January (CDFG 1998; USFWS 1995a). In the Sacramento River drainage, spring-run Chinook salmon smolt emigration reportedly occurs from October through March (CDFG 1998). In Butte Creek, some juvenile spring-run Chinook salmon rear through the summer and emigrate as yearlings from October to February, with peak yearling emigration occurring in November and December (CDFG 1998). In the Feather River, some spring-run Chinook salmon smolts reportedly emigrate from the Feather River system from October through June (B. Cavallo, DWR, pers. comm. 2004).

Although it has been previously suggested that spring-run Chinook salmon emigrate as smolts from November through June in the Yuba River (CALFED and YCWA 2005; CDFG 1998; SWRI 2002), more recent RST monitoring data indicate that the vast majority of spring-run Chinook salmon emigrate as post-emergent fry during November and December.

Based upon review of available information, the RMT (2013a) recently identified the spring-run Chinook salmon smolt (yearling+) outmigration period as extending from October through mid-May.

5.1.6 Limiting Factors, Threats and Stressors

Limiting factors and threats supporting the listing of the Central Valley spring-run Chinook salmon ESU are presented in two documents. The first is titled *Factors for Decline: A Supplement to the Notice of Determination for West Coast Steelhead* (NMFS 1996a). That report concluded that all of the factors identified in § 4(a)(1) of the ESA have played roles in the decline of steelhead and other salmonids, including Chinook salmon. The report identifies destruction and modification of habitat, overutilization of fish for commercial and recreational purposes, and natural and human-made factors as being the primary reasons for the declines of west coast steelhead and other salmonids including Chinook salmon. The second document is a supplement to the document referred to above. This document is titled *Factors Contributing to the Decline of West Coast Chinook Salmon: An Addendum to the 1996 West Coast Steelhead Factors for Decline Report* (NMFS 1998).

At the ESU level, more recent descriptions of limiting factors, threats and stressors are provided in the BA for the CVP/SWP OCAP (Reclamation 2008a), the CVP/SWP OCAP BO (NMFS 2009b), and the *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* (Recovery Plan) (NMFS 2014). In addition to the ESU-level discussions, limiting factors, threats and stressors specifically addressing spring-run Chinook salmon in the lower Yuba River are discussed in the Recovery Plan (NMFS 2014). These documents are incorporated by reference into this Applicant-Prepared Draft BA, and brief summaries of limiting factors, threats and stressors to spring-run Chinook salmon at the ESU level, and in the lower Yuba River specifically, are provided below. These brief summaries provide additional detail, explanation or clarification of limiting factors, threats and stressors in the lower Yuba River.

5.1.6.1 ESU

According to NMFS' Recovery Plan (NMFS 2014), threats to Central Valley spring-run Chinook salmon are in three broad categories: 1) loss of historical spawning habitat; 2) degradation of remaining habitat; and 3) threats to the genetic integrity of the wild spawning populations from the FRFH spring-run Chinook salmon production program. As stated in the NMFS (2014a), the Central Valley spring-run Chinook salmon ESU continues to be threatened by habitat loss, degradation and modification, small hydropower dams and water diversions that reduce or eliminate instream flows during migration, unscreened or inadequately screened water diversions, excessively high water temperatures, and predation by non-native species. The potential effects of long-term climate change also may adversely affect spring-run Chinook salmon and their recovery. The 2009 NMFS OCAP BO (2009b), summarized below, identified the factors that have lead to the current status of the species to be habitat blockage, water development and diversion dams, water conveyance and flood control, land use activities, water quality, hatchery operations and practices, over-utilization (e.g., ocean commercial and sport harvest, inland sport harvest), disease and predation, environmental variation (e.g., natural environmental cycles, ocean productivity, global climate change), and non-native invasive species.

5.1.6.1.1 Habitat Blockage

Hydropower, flood control, and water supply dams of the CVP, SWP, and other municipal and private entities (and in the Yuba River, debris dams created by federal and State partnerships) have permanently blocked or hindered salmonid access to historical spawning and rearing grounds. As a result of migrational barriers, spring-run Chinook salmon (as well as winter-run Chinook salmon and steelhead) populations have been confined to lower elevation mainstems that historically only were used by these species for migration and rearing. Population abundances have declined in these streams due to decreased quantity, quality, and spatial distribution of spawning and rearing habitat (Lindley et al. 2009). Higher temperatures at these lower elevations during late-summer and fall are also a major stressor to adult and juvenile salmonids, although dams upstream with sufficient storage of cold water may provide cool temperature releases throughout the late-summer and fall.

Juvenile downstream migration patterns have been altered by the presence of dams. Juvenile spring-run Chinook salmon (as well as winter-run) on the mainstem Sacramento River generally outmigrate earlier than they did historically because they are hatched considerably farther downstream and now have less distance to travel. Therefore, smolts in the Sacramento River under present conditions must rear for a longer period of time in order to reach sizes comparable to those of smolts that historically reared in upstream reaches above the dams. However, for several months of the year, habitat conditions in the mainstem Sacramento River do not provide the necessary features in amounts necessary for rearing of listed anadromous fish species, especially for an extended period of time.

5.1.6.1.2 Water Development

The diversion and storage of natural flows by dams and diversion structures on Central Valley waterways have altered the natural hydrologic cycles which juvenile and adult salmonids historically based their migration patterns upon (NMFS 2009b). As much as 60 percent of the natural historical inflow to Central Valley watersheds and the Delta has been diverted for human uses. Dams have often contributed to lower flows, higher water temperatures, lower dissolved oxygen (DO) levels, and decreased recruitment of gravel and LWM. More uniform flows year round have resulted in diminished natural channel formation, altered food web processes, and slower regeneration of riparian vegetation.

Water diversions for irrigated agriculture, municipal and industrial use, and managed wetlands exist throughout the Central Valley. Thousands of small and medium-size water diversions exist along the Sacramento River, its tributaries and the Delta. Although efforts have been made in recent years to screen some of these diversions, many remain unscreened. Depending on the size, location, and season of operation, these unscreened diversions have the potential to entrain many lifestages of aquatic species, including juvenile salmonids.

The Anderson-Cottonwood Irrigation District (ACID) operates a diversion dam across the Sacramento River about 5 mi downstream of Keswick Dam, which is one of the three largest diversions on the Sacramento River. Operated from April through October, the installation and removal of the diversion dam flashboards requires close coordination between Reclamation and ACID. Because substantial reductions (limited to 15 percent in a 24-hour period and 2.5percent in any 1 hour) in Keswick Dam releases are necessary to install or remove the flashboards, the ACID diversion dam operations have the potential to impact various lifestages of Chinook salmon (e.g., redd dewatering, juvenile stranding and exposure to elevated water temperatures). Redd dewatering primarily affects spring- and fall-run Chinook salmon during October. Although flow reductions are usually of a short-term duration (i.e., lasting less than 8 hours), these short-term flow reductions may cause mortality through desiccation of incubating eggs and loss of stranded juveniles, if the reduction in water surface elevation is sufficiently large relative to the depth at which incubating eggs were constructed or the depth of off-channel habitats where juveniles may be located.

Located 59 mi downstream of Keswick Dam, RBDD is owned and operated by Reclamation. Historically, RBDD impeded adult salmonid passage throughout its May 15 through September 15 “gates in” period. Although there were fish ladders at the right and left banks, and a temporary ladder in the middle of the dam, they were not very efficient at passing fish because it was difficult for fish to locate the entrances to the ladders. Water released from RBDD flowed through a small opening under each of the 11 gates in the dam and caused turbulent flows that confused fish and kept them from finding the ladders. The effects resulting from upstream migrational delays at RBDD ranged from delayed but eventually successful spawning, to pre-spawn mortality and the complete loss of spawning potential in that fraction of the population. The fish ladders were not designed to allow a sufficient amount of flow through them to attract adult salmonids, and previous studies have shown that salmon could be delayed up to 20 days in passing the dam. These delays had the potential to reduce the fitness of adults that expend their energy reserves fighting the flows beneath the gates, and increase the chance of pre-spawn mortality. Passage delays of a few days up to a week were believed to prevent timely movement of adult spring-run Chinook salmon upstream to enter the lower reaches of Sacramento River tributaries (e.g., Cottonwood Creek, Cow Creek) above the RBDD, which dry up or warm up during the spring. These passage delays prevented adult spring-run Chinook salmon from accessing summer holding pools in the upper reaches of these tributaries. As previously discussed, the RBDD gates were permanently raised in September 2011 and, thus, many of the historical migration-related stressors associated with this location have likely been eliminated due to the improved fish passage conditions.

Outmigrant juvenile salmonids in the Delta have been subjected to adverse environmental conditions created by water export operations at the CVP and SWP facilities. Specifically, juvenile salmonid survival has been reduced by: 1) water diversions from the mainstem Sacramento River into the Central Delta through the DCC; 2) upstream or reverse flows of water in the lower San Joaquin River and southern Delta waterways; 3) entrainment at the CVP/SWP export facilities and associated problems at Clifton Court Forebay; and 4) increased exposure to introduced, non-native predators such as striped bass, largemouth bass (*Micropterus salmoides*), and sunfishes (*Centrarchidae spp.*) within the waterways of the Delta.

5.1.6.1.3 Water Conveyance and Flood Control

More than 1,600 mi of levee construction in the Central Valley has constricted river channels, disconnected floodplains from active river channels, reduced riparian habitat, and reduced natural channel function, particularly in lower reaches of the Sacramento River and the Delta (NMFS 2009b). The development of the water conveyance system in the Delta also has resulted in the construction of armored, rip-rapped levees on more than 1,100 mi of channels and diversions to increase channel elevations and flow capacity of the channels (Mount 1995 as cited in NMFS 2009b).

Levee development in the Central Valley has affected anadromous salmonid spawning habitat, freshwater rearing habitat, freshwater migration corridors, and estuarine habitats. Many of the levees use large angular rock (riprap) to armor the banks from erosive forces. The effects of channelization and rip-rapping include the alteration of river hydraulics and vegetative cover along the banks as a result of changes in bank configuration and structural features (Stillwater Sciences 2006 as cited in NMFS 2009b). These changes affect the quantity and quality of nearshore habitat for juvenile salmonids and have been thoroughly studied (USFWS 2000; Schmetterling et al. 2001 as cited in NMFS 2009b; Garland et al. 2002). Simple slopes protected with rock revetment generally create nearshore hydraulic conditions characterized by greater depths and faster, more homogeneous water velocities than those that occur along natural banks. Higher water velocities typically inhibit deposition and retention of sediment and woody debris. These changes generally reduce the range of habitat conditions typically found along natural shorelines, especially by eliminating the shallow, slow-velocity river margins used by juvenile fish as refuge and to escape from fast currents, deep water, and predators (Stillwater Sciences 2006 as cited in NMFS 2009b). In addition, the armoring and revetment of stream banks tend to narrow rivers, reducing the amount of habitat per unit channel length (Sweeney et al. 2004). As a result of river narrowing, benthic habitat decreases and the number of macroinvertebrates (e.g., stoneflies, mayflies) per unit channel length decreases, affecting salmonid food supply.

LWM is a functionally important component of many streams (NMFS 1996b). LWM influences stream morphology by affecting channel pattern, position, and geometry, as well as pool formation (Keller and Swanson 1979; Bilby 1984; Robison and Beschta 1990). Reduction of wood in the stream channel, either from past or present activities, generally reduces pool quantity and quality, alters stream shading which can affect water temperature regimes and nutrient input, and can eliminate critical stream habitat needed for both vertebrate and invertebrate populations. Removal of vegetation also can destabilize marginally stable slopes by increasing the subsurface water load, lowering root strength, and altering water flow patterns in the slope. During the 1960s and early 1970s, it was common practice among California fishery management agencies to remove LWM thought to be a barrier to fish migration (NMFS 1996b). However, it is now recognized that too much LWM was removed from streams in past decades, resulting in a loss of salmonid habitat. The large scale removal of LWM prior to 1980 is believed to have had major, long-term adverse effects on juvenile salmonid rearing habitat in northern California (NMFS 1996b). Aquatic habitat areas that were subjected to the removal of LWM are still limited in the recovery of salmonid stocks, and NMFS (2009b) expects that this limitation could persist for 50 to 100 years.

5.1.6.1.4 Land Use Activities

Land use activities continue to have large-scale impacts on salmonid habitat in the Central Valley. According to Lindley et al. (2009), "*Degradation and simplification of freshwater and estuary habitats over a century and a half of development have changed the Central Valley Chinook salmon complex from a highly diverse collection of numerous wild populations to one dominated by fall Chinook salmon from four large hatcheries*."

Until about 150 years ago, the Sacramento River was bordered by up to 500,000 ac of riparian forest, with bands of vegetation extending outward for 4 or 5 mi (California Resources Agency 1989). Starting with the gold rush, vast riparian forests were cleared for building materials, fuel, and to open land for farming along the banks of the river. The clearing of the riparian forests also removed a vital source of snags and driftwood in the Sacramento River Basin. The removal of in-river snags and obstructions for navigational safety has further reduced the presence of LWM in the Sacramento River and the Delta (see LWM discussion above). The degradation and fragmentation of riparian habitat continued with extensive flood control and bank protection projects, together with the conversion of the fertile riparian lands to agriculture. By 1979, riparian habitat along the Sacramento River diminished to about 2 percent (i.e., 11,000 to 12,000 ac) of historic levels (McGill and Price 1987).

Land use activities associated with road construction, urban development, logging, mining, agriculture, and recreation have significantly altered fish habitat quantity and quality through the alteration of streambank and channel morphology, alteration of ambient water temperatures, degradation of water quality, elimination of spawning and rearing habitat, fragmentation of available habitats, elimination of downstream recruitment of LWM, and removal of riparian vegetation, resulting in increased streambank erosion (Meehan 1991 as cited in NMFS 2009b). Urban stormwater and agricultural runoff may be contaminated with herbicides and pesticides, petroleum products, sediment, etc. Agricultural practices in the Central Valley have eliminated large trees and logs and other woody debris that would otherwise be recruited into the stream channel (NMFS 1998).

Increased sedimentation resulting from agricultural and urban practices is one of the primary causes of salmonid habitat degradation in the Central Valley (NMFS 1996a). Sedimentation can adversely affect salmonids during all freshwater lifestages by clogging or abrading gill surfaces, adhering to eggs, hampering fry emergence (Phillips and Campbell 1961 as cited in NMFS 2009b), burying eggs or alevins, scouring and filling in pools and riffles, reducing primary productivity and photosynthesis activity (Cordone and Kelley 1961), and affecting intergravel permeability and DO levels. Excessive sedimentation over time can cause substrates to become embedded, which reduces successful salmonid spawning and egg and fry survival (Waters 1995 as cited in NMFS 2009b).

River channel dredging to enhance inland maritime trade and to provide raw material for levee construction also has altered the natural hydrology and function of the Central Valley rivers. Since the mid-1800s, USACE and others have straightened and artificially deepened river channels to enhance shipping commerce, consequently reducing the natural river meander and the formation of pool and riffle segments. In the early 1900s, the Sacramento Flood Control

Project ushered in large scale USACE actions for reclamation and flood control purposes along the Sacramento River and in the Delta. The creation of levees and the deep shipping channels reduced the natural tendency of the Sacramento River to create floodplains along its banks during seasonal inundation periods (e.g., spring snow melt). The annual inundations provided necessary juvenile rearing and foraging habitat that became available in conjunction with seasonal flooding processes. The armored riprapped levee banks and active maintenance actions of Reclamation Districts precluded the establishment of ecologically important riparian vegetation, introduction of valuable LWM from these riparian corridors, and the productive intertidal mudflats characteristic of the undisturbed Delta habitat.

Since the 1850s, reclamation of wetlands for urban and agricultural development has resulted in the cumulative loss of tidal marsh habitat downstream (79 percent) and upstream (94 percent) of Chipps Island (Conomos et al. 1985; Nichols et al. 1986; Wright and Phillips 1988 as cited in NMFS 2009b; Monroe et al. 1992 as cited in NMFS 2009b; Goals Project 1999). Little of the extensive tracts of wetland marshes that existed prior to 1850 along the Central Valley river systems and within the natural flood basins exist today. Most wetland and marsh areas have been "reclaimed" for agricultural purposes, leaving only small remnant patches of available habitat. In the Delta, juvenile salmonids are exposed to increased water temperatures during the late spring and summer due to the loss of riparian shading and thermal inputs from municipal, industrial, and agricultural discharges. Studies by DWR on water quality in the Delta over the last 30 years show a steady decline in food resources available for juvenile salmonids, as well as an increase in the clarity of the water due to a reduction in phytoplankton and zooplankton. These conditions are believed to have contributed to increased juvenile Chinook salmon and steelhead mortality as fish move through the Delta.

5.1.6.1.5 Water Quality

Over the past 150 years, the water quality of the Delta has been adversely affected by increased water temperatures, decreased DO levels, and increased turbidity and contaminant loads, which have degraded the quality of the aquatic habitat for the rearing and migration of salmonids. Historic and ongoing point and nonpoint source discharges impact surface waters, and portions of major rivers and the Delta are impaired, to some degree, by discharges from agriculture, mines, urban areas and industries (California RWQCB 1998). Pollutants include effluents from wastewater treatment plants and chemical discharges (e.g., dioxin from San Francisco Bay petroleum refineries) (McEwan and Jackson 1996). Agricultural drain water, another possible source of contaminants, can contribute up to 30 percent of the total inflow into the Sacramento River during drier conditions (Reclamation 2008a).

According to NMFS (2009b), the California Regional Water Quality Control Board (California RWQCB) (1998, 2001) has identified the Delta as an impaired waterbody having elevated levels of chlorpyrifos, dichlorodiphenyltrichlor (i.e., DDT), diazinon, mercury, Group A pesticides (e.g., aldrin, dieldrin, chlordane, endrin, heptachlor, heptachlor epoxide, hexachlorocyclohexanes (including lindane), endosulfan and toxaphene), organic enrichment, as well as low DO. In general, water degradation or contamination can lead to either acute toxicity, resulting in death when concentrations are sufficiently elevated, or more typically, when concentrations are lower, to chronic or sublethal effects that reduce the physical health of the organism, and lessens its survival over an extended period of time. Mortality may become a secondary effect due to compromised physiology or behavioral changes that lessen the organism's ability to carry out its normal activities. For listed species, these effects may occur directly to the listed fish or to its prey base, which reduces the forage base available to the listed species.

In the aquatic environment, most anthropogenic chemicals and waste materials, including toxic organic and inorganic chemicals eventually accumulate in sediment (Ingersoll 1995 as cited in NMFS 2009b). Direct exposure to contaminated sediments may cause deleterious effects if a fish swims through a plume of the re-suspended sediments or rests on contaminated substrate and absorbs the toxic compounds via dermal contact, ingestion, or uptake across the gills. Although sediment contaminant levels can be significantly higher than the overlying water column concentrations (EPA 1994), the more likely means of exposure is through the food chain when fish feed on organisms that are contaminated with toxic compounds. Prey species become contaminated either by feeding on the detritus associated with the sediments or dwelling in the sediment itself. Therefore, the degree of exposure to the salmonids depends on their trophic level and the amount of contaminated forage base consumed. Salmonid biological responses to contaminated sediments are similar to those resulting from waterborne exposures once a contaminant has entered the body of the fish.

5.1.6.1.6 Hatchery Operations and Practices

Cal Fish and Wildlife is currently operating 10 salmon and steelhead hatchery facilities in California. Eight of these facilities (i.e., Iron Gate, Trinity River, Warm Springs, Feather River, Nimbus, Mokelumne River, and Merced River Hatcheries and the Coyote Valley Fish Facility) were constructed below dams on major rivers as mitigation for loss of access to anadromous fish habitat upstream of the dams. In addition to the Cal Fish and Wildlife hatcheries, USFWS operates the Coleman National Fish Hatchery located on Battle Creek, and the Livingston Stone National Fish Hatchery located on the upper Sacramento River, which includes an artificial propagation program for Sacramento River winter-run Chinook salmon.

Five hatcheries currently produce Chinook salmon in the Central Valley, and four of these also produce steelhead. Releasing large numbers of hatchery fish can pose a threat to wild Chinook salmon and steelhead stocks through genetic impacts, competition for food and other resources between hatchery and wild fish, predation of hatchery fish on wild fish, and increased fishing pressure on wild stocks as a result of hatchery production (Waples 1991). The genetic impacts of artificial propagation programs in the Central Valley are primarily caused by straying of hatchery fish and the subsequent interbreeding of hatchery fish with wild fish. In the Central Valley, practices such as transferring eggs between hatcheries and trucking smolts to distant sites for release contribute to elevated straying levels (USDOI 1999 as cited in NMFS 2009b).

Hatchery practices as well as spatial and temporal overlaps of habitat use and spawning activity between spring- and fall-run Chinook salmon have led to the hybridization and homogenization of some subpopulations (CDFG 1998). As early as the 1960s, Slater (1963) observed that spring-run and early fall-run were competing for spawning sites in the Sacramento River below Keswick Dam, and speculated that the two runs may have hybridized. Spring-run Chinook salmon from the FRFH have been documented as straying throughout the Central Valley for many years (CDFG 1998), and may have contributed to hybridization. In the Feather River, the lack of physical separation has led to hybridization of spring- and fall-run Chinook salmon.

The relatively low number of spawners needed to sustain a hatchery population can result in high harvest-to-escapements ratios in waters where fishing regulations are set according to hatchery population. This can lead to over-exploitation and reduction in the size of wild populations existing in the same system as hatchery populations due to incidental by-catch (McEwan 2001).

Hatcheries also can have some positive effects on salmonid populations. Spring-run Chinook salmon produced in the FRFH are considered part of the spring-run Chinook salmon ESU. Artificial propagation has been shown to be effective in bolstering the numbers of naturally spawning fish in the short term under specific scenarios. Artificial propagation programs can also aid in conserving genetic resources and guarding against catastrophic loss of naturally spawned populations at critically low abundance levels (NMFS 2004b).

5.1.6.1.7 Overutilization

Ocean Commercial and Sport Harvest

Extensive ocean recreational and commercial troll fisheries for Chinook salmon exist along the Northern and Central California coast, and an inland recreational fishery exists in the Central Valley for Chinook salmon and steelhead. The Central Valley Index (CVI) is an annual index of abundance of all Central Valley Chinook salmon stocks combined, and is defined as the calendar year sum of ocean fishery Chinook harvests in the area south of Point Arena, California (where 85% of Central Valley Chinook salmon are caught), plus the Central Valley adult Chinook spawning escapement (Lindley et al. 2009). Since 1991, the Pacific Fishery Management Council's (PFMC) Salmon Technical Team (comprised of scientists from NMFS, USFWS, and state fisheries agencies from Oregon, Washington and California) has used a linear regression of the CVI on the previous year's Central Valley age-2 return to forecast the CVI (BDCP 2009). The CVI harvest rate index is an annual index of the ocean harvest rate on all Central Valley Chinook stocks combined, and is defined as the ocean harvest landed south of Point Arena, California, divided by the CVI (Lindley et al. 2009).

There are no Pacific Coast Salmon Fishery Management Plan (FMP) objectives in place specifically regulating the harvest of spring-run Chinook salmon, except that the FMP will manage ocean fisheries consistent with NMFS ESA consultation standards (BDCP 2009). The current FMP harvest constraints on winter-run Chinook salmon serve as a proxy for Central Valley spring-run Chinook salmon (BDCP 2009). Spring-run Chinook salmon CVI harvest rate index ranged from 0.55 to nearly 0.80 between 1970 and 1995, when harvest rates were adjusted for the protection of winter-run Chinook salmon (NMFS 2003). The decline in the CVI harvest rate index to 0.27 in 2001 as a result of high fall-run Chinook salmon escapement also resulted in reductions to the authorized harvest of spring-run Chinook salmon (NMFS 2003).

FRFH spring-run Chinook salmon provide indices of harvest of natural spring-run. Maturing age-3 and age-4 spring-run Chinook salmon are vulnerable to the early portion of the recreational and commercial season, whereas fall-run Chinook salmon are exposed to an entire harvest season

(BDCP 2009). Inferences drawn from coded-wire tag recoveries indicate that 44 percent of the spring-run Chinook salmon are taken prior to May 1, the start of the commercial fishing season (BDCP 2009). Ocean fisheries have affected the age structure of spring-run Chinook salmon through targeting large fish for many years and reducing the numbers of 4- and 5-year-old fish (CDFG 1998). As a result of very low returns to the Central Valley in 2007, there was a complete closure of the commercial and recreational ocean Chinook salmon fishery in 2008 and 2009. Due to improved ocean salmon numbers, a severely restricted commercial season and short recreational season opened in 2010 (Bacher 2011). On April 13, 2011, the PFMC adopted a set of ocean salmon seasons that provides both recreational and commercial opportunities during the 2011 fishing season. PFMC (2011a) reports that "*Greatly improved abundance of Sacramento River fall-run Chinook salmon will fuel the first substantial ocean salmon fisheries off California and Oregon since 2007. Fisheries south of Cape Falcon are supported by Sacramento River fall Chinook. In 2008 and 2009, poor Sacramento returns led to the largest ocean salmon fishery closure on record. The abundance forecast of Sacramento River fall Chinook in 2011 is 730,000, far above the number needed for optimum spawning this fall (122,000-180,000 fish).*"

Inland Sport Harvest

Historically in California, almost half of the river sport fishing effort has occurred in the Sacramento-San Joaquin River system, particularly upstream from the city of Sacramento (Emmett et al. 1991). In-river recreational fisheries historically have taken spring-run Chinook salmon throughout the species' range. During the summer, adult spring-run Chinook salmon are targeted by anglers when the fish congregate and hold in large pools. Poaching also occurs at fish ladders, and other areas where adults congregate. However, the significance of poaching on the adult population is unknown (NMFS 2009b). Specific regulations for the protection of spring-run Chinook salmon in Mill, Deer, Butte, and Big Chico creeks and the lower Yuba River have been added to the Cal Fish and Wildlife regulations.

5.1.6.1.8 Disease and Predation

Salmonids are exposed to numerous bacterial, protozoan, viral, and parasitic organisms in spawning and rearing areas, hatcheries, migratory routes, and the marine environment (NMFS 1996a, 1996b, 1998), and infectious disease is one of many factors that influence adult and juvenile salmonid survival. Specific diseases such as bacterial kidney disease, Ceratomyxosis shasta, columnaris, furunculosis, infectious hematopoietic necrosis, redmouth and black spot disease, whirling disease, and erythrocytic inclusion body syndrome are known, among others, to affect Chinook salmon and steelhead (NMFS 1996a; 1996b; 1998). Little current or historical information exists to quantify changes in infection levels and mortality rates attributable to these diseases; however, studies have shown that wild fish tend to be less susceptible to pathogens than are hatchery-reared fish (NMFS 2009b). Nevertheless, wild salmonids may contract diseases that are spread through the water column (i.e., waterborne pathogens) as well as through interbreeding with infected hatchery fish. The stress of being released into the wild from a controlled hatchery environment frequently causes latent infections to convert into a more pathological state, and increases the potential of transmission from hatchery reared fish to wild stocks within the same waters.

As described in NMFS (2005a), accelerated predation is also a significant factor affecting critical habitat for spring-run Chinook salmon. Although predation is a natural component of spring-run Chinook salmon life ecology, the rate of predation likely has greatly increased through the introduction of non-native predatory species such as striped bass and largemouth bass (*Micrapterus salmaides*), and through the alteration of natural flow regimes and the development of structures that attract predators, including dams, bank revetment, bridges, diversions, piers, and wharfs (Stevens 1961; Vogel et al. 1988 as cited in NMFS 2014; Garcia 1989 as cited in Reclamation 2008a; Decoto 1978 as cited in Reclamation 2008a). The USFWS found that more predatory fish were found at rock revetment bank protection sites between Chico Landing and Red Bluff than at sites with naturally eroding banks (Michny and Hampton 1984). On the mainstem Sacramento River, high rates of predation have been reported to occur at diversion facilities associated with RBDD, ACID, Glenn-Colusa Irrigation District (GCID), and at south Delta water diversion structures (CDFG 1998). However, the permanent raising of the RBDD gates has likely reduced the potential for predation in the vicinity of the RBDD. From October 1976 to November 1993, Cal Fish and Wildlife conducted ten mark/recapture experiments at the SWP's Clifton Court Forebay to estimate prescreen losses using hatchery-reared juvenile Chinook salmon. Pre-screen losses ranged from 69 to 99 percent. Predation from striped bass is thought to be the primary cause of the loss (CDFG 1998; Gingras 1997).

Predation on juvenile salmonids has increased as a result of water development activities, which have created ideal habitats for predators and non-native invasive species. As juvenile salmonids pass the Sacramento River system dams, fish are subject to conditions that can disorient them, making them highly susceptible to predation by fish or birds. Striped bass and Sacramento pikeminnow (*Ptychocheilus grandis*), a species native to the Sacramento River Basin that co-evolved with anadromous salmonids, congregate below dams and prey on juvenile salmon in the tail waters. Tucker et al. (1998) reported that: 1) striped bass exhibit a strong preference for juvenile salmonids; 2) during the summer months, juvenile salmonids increased to 66 percent of the total weight of Sacramento pikeminnow stomach contents; and 3) the percent frequency of occurrence for juvenile salmonids nearly equaled other fish species in the stomach contents of the predatory fish. Additionally, Tucker et al. (2003) showed the temporal distribution for these two predatory species in the RBDD area were directly related to RBDD operations (i.e., predators congregated when the dam gates were in, and dispersed when the dam gates were removed). Due to the permanent raising of the RBDD gates, predation in the vicinity of the RBDD has likely been reduced.

Other locations in the Central Valley where predation is of concern include flood bypasses, post-release sites for salmonids salvaged at the CVP and SWP Fish Facilities, and the Suisun Marsh Salinity Control Gates (SMSCG). The dominant predator species at the SMSCG was striped bass, and the remains of juvenile Chinook salmon were identified in their stomach contents (Edwards et al. 1996; Tillman et al. 1996; NMFS 1997). Striped bass and Sacramento pikeminnow predation on salmon at salvage release sites in the Delta and lower Sacramento River has been documented (Orsi 1967; Pickard et al. 1982). However, accurate predation rates at these sites are difficult to determine. More recent studies by DWR (2008) have verified this level of predation also exists for steelhead smolts within Clifton Court Forebay, indicating that these predators were efficient at removing salmonids over a wide range of body sizes.

Avian predation on fish contributes to the loss of migrating juvenile salmonids (NMFS 2009b). Fish-eating birds (e.g., great blue herons, black-crowned night herons, gulls, osprey) in the Central Valley have high metabolic rates and require large quantities of food relative to their body size.

Mammals can also be an important source of predation on salmonids within the California Central Valley. These animals, especially river otters, are capable of removing large numbers of salmon and trout from the aquatic habitat (Dolloff 1993 as cited in NMFS 2009b). Mammals have the potential to consume large numbers of salmonids, but generally scavenge post-spawned salmon. In the marine environment, Southern Resident killer whales target Chinook salmon as their preferred prey (Ford and Ellis 2006).

5.1.6.1.9 Environmental Variation

The scientific basis for understanding the processes and sources of climate variability has grown significantly in recent years, and the ability to forecast human and natural contributions to climate change has improved dramatically. With consensus on the reality of climate variability now established (Oreskes 2004; IPCC 2007), the scientific, political, and public priorities are evolving toward determining its ecosystem impacts, and developing strategies for adapting to those impacts. Global climate change is playing an increasingly important role in scientific and policy debates related to effective water management. The most considerable impacts of climate change on water resources in the United States are believed to occur in the mid-latitudes of the West, where the runoff cycle is largely determined by snow accumulation and subsequent melt patterns. Evidence is continuing to accumulate to indicate global climate change will have a marked effect on water resources in California. Numerous peer-reviewed scientific articles on climate and water issues in California have been published to date, with many more in preparation, addressing a range of considerations from proposed improvements in the downscaling of general circulation models to understanding how reservoir operations might be adapted to new conditions (Kiparsky and Gleick 2003).

NMFS (2014a) states that the potential effects of long-term climate change may adversely affect spring-run Chinook salmon and steelhead, and the recovery of both species. Current climate change information suggests that the Central Valley climate will become warmer, a challenging prospect for Chinook salmon and steelhead – both of which are coldwater fish at the southern end of their distribution. According to NMFS (2009b), early marine survival for juvenile salmon is a critical phase in their survival and development into adults. The correlation between various environmental indices that track ocean conditions and salmon productivity in the Pacific Ocean, both on a broad and local scale, provides an indication of how climate-related factors influence salmon survival in the ocean. Consistent with the approach taken in recent NMFS BOs (NMFS 2011b; NMFS 2010a; NMFS 2010b; NMFS 2010c), the discussion below describes the potential climate-related threats anticipated to affect the status of listed species, including inter-annual climatic variations (e.g., El Niño and La Niña), the Wells Ocean Productivity Index, and longer term cycles in ocean conditions pertinent to salmonid survival (e.g., Pacific Decadal Oscillation (PDO)).

Natural Environmental Cycles

Natural climate variability in freshwater and marine environments has the potential to substantially affect salmonid abundance, particularly during early lifestages (NMFS 2008a). Sources of variability include inter-annual climatic variations (e.g., El Niño and La Niña), longer-term cycles in ocean conditions (e.g., PDO, Mantua et al. 1997), and ongoing global climate change. Climate variability can affect ocean productivity in the marine environment, as well as water storage (e.g., snow pack) and in-stream flow in the freshwater environment. Early lifestage growth and survival of salmon can be negatively affected when climate variability results in conditions that hinder ocean productivity (e.g., Scheuerell and Williams 2005) and water storage (e.g., Independent Scientific Advisory Board 2007) in marine and freshwater systems, respectively.

Fisheries scientists have shown that ocean climate varies strongly at decadal scales (e.g., Beamish 1993; Beamish and Bouillon 1993; Graham 1994; Miller et al. 1994; Hare and Francis 1995; Mantua et al. 1997; Mueter et al. 2002). In particular, the identification of the PDO (Mantua et al. 1997) has led to the belief that decadal-scale variation may be cyclical, and thus predictable (Lindley et al. 2007). Evidence also suggests that marine survival among salmonids fluctuates in response to 20- to 30-year cycles of climatic conditions and ocean productivity (Hare et al. 1999 as cited in NMFS 2009b; Mantua and Hare 2002). In addition, large-scale climatic regime shifts, such as the El Niño condition, appear to change productivity levels over large expanses of the Pacific Ocean. A further confounding effect is the fluctuation between drought and wet conditions in the basins of the American west. During the first part of the 1990s, much of the Pacific Coast was subject to a series of very dry years, which reduced inflows to watersheds up and down the west coast.

"El Niño" is an environmental condition often cited as a cause for the decline of West Coast salmonids (NMFS 1996a). El Niño is an unusual warming of the Pacific Ocean off South America and is caused by atmospheric changes in the tropical Pacific Ocean (El Niño Southern Oscillation [ENSO]) resulting in reductions or reversals of the normal trade wind circulation patterns. El Niño ocean conditions are characterized by anomalous warm sea surface temperatures and changes to coastal currents and upwelling patterns. Principal ecosystem alterations include decreased primary and secondary productivity in affected regions and changes in prey and predator species distributions. Cold-water species are displaced towards higher latitudes or move into deeper, cooler water, and their habitat niches are occupied by species tolerant of warmer water that move upwards from the lower latitudes with the warm water tongue.

A key factor affecting many West Coast stocks has been a general 30-year decline in ocean productivity. The mechanism whereby stocks are affected is not well understood, partially because the pattern of response to these changing ocean conditions has differed among stocks, presumably due to differences in their ocean timing and distribution. It is presumed that survival of Chinook salmon in the ocean is driven largely by events occurring between ocean entry and recruitment to a sub-adult lifestage. The freshwater life history traits and habitat requirements of juvenile winter-run and fall-run Chinook salmon are similar. Therefore, the unusual and poor ocean conditions that caused the drastic decline in returning fall-run Chinook salmon populations coast-wide in 2007 (Varanasi and Bartoo 2008) are suspected to have also caused the observed decrease in the winter-run Chinook salmon spawning population in 2007 (Oppenheim 2008 as cited in NMFS 2009b). Lindley et al. (2009) reviewed the possible causes for the decline in Sacramento River fall-run Chinook salmon in 2007 and 2008 for which reliable data were available. They concluded that a broad body of evidence suggested that anomalous conditions in the coastal ocean in 2005 and 2006 resulted in unusually poor survival of the 2004 and 2005 broods of fall-run Chinook salmon. However, Lindley et al. (2009) recognize that the rapid and likely temporary deterioration in ocean conditions acted on top of a long-term, steady degradation of the freshwater and estuarine environment.

As suggested by Rudnick and Davis (2003) and Hsieh et al. (2005), apparent regime shifts need not be cyclical or predictable, but rather may be the expression of a stochastic process. If this interpretation is correct, then we should expect future ocean climate conditions to be different than those observed over the past few decades (Lindley et al. 2007).

Lindley et al. (2007) further state that Central Valley salmonid ESUs and DPSs are capable of surviving the kinds of climate extremes observed over the past few thousand years if they have functional habitats, because these lineages are on order of a thousand years old or older. There is growing concern, however, that the future climate will be unlike that seen before, due to global warming in response to anthropogenic greenhouse gas emissions (Lindley et al. 2007).

Ocean Productivity

The time when juvenile salmonids enter the marine environment marks a critical point in their life history. Studies have shown the greatest rates of growth and energy accumulation for Chinook salmon occur during the first 1 to 3 months after they enter the ocean (Francis and Mantua 2003 as cited in NMFS 2009b; MacFarlane et al. 2008 as cited in NMFS 2009b). Emigration periods and ocean entry can vary substantially among, and even within, runs in the Central Valley. Winter-run Chinook salmon exhibit a peak emigration period in March and April, whereas spring-run Chinook salmon emigration is more variable and can occur in December or January (soon after emergence as fry), or from October through March (after rearing for a year or more in freshwater; Reclamation 2008a). Steelhead ocean entry can span many months. Juvenile steelhead presence at Chipps Island has been documented between at least October and July (Reclamation 2008a). The general timing pattern of ocean entry is commonly attributed to evolutionary adaptations that allow salmonids to take advantage of highly productive ocean conditions that typically occur off the California coast beginning in spring and extending into the fall (MacFarlane et al. 2008 as cited in NMFS 2009b). Therefore, the conditions that juvenile salmonids encounter when they enter the ocean can play an important role in their early marine survival and eventual development into adults.

Variations in salmon marine survival correspond with periods of cold and warm ocean conditions, with cold regimes being generally favorable for salmon survival and warm regimes unfavorable (Behrenfeld et al. 2006; Wells et al. 2006). Peterson et al. (2006) provide evidence that growth and survival rates of salmon in the California Current System (CCS) off the Pacific Northwest can be linked to fluctuations in ocean conditions.

An evaluation of conditions in the CCS since the late 1970s reveals that a generally warm, unproductive regime persisted until the late 1990s. This regime was followed by a period of high variability that began with colder, more productive conditions lasting from 1999 to 2002. In general, salmon populations increased substantially during this period. However, the brief cold cycle was immediately succeeded by a 4-year period of predominantly warm ocean conditions beginning in late 2002, which appeared to negatively impact salmon populations in the CCS (Peterson et al. 2006).

The generally warmer ocean conditions in the CCS that began to prevail in late 2002 have resulted in coastal ocean temperatures remaining 1°C to 2°C (1.8°F to 3.6°F) above normal through 2005. A review of the previously mentioned indicators for 2005 revealed that almost all ecosystem indices were characteristic of poor ocean conditions and reduced salmon survival (NMFS 2009b).

Peterson et al. (2006) shows the transition to colder ocean conditions, which began in 2007 and persisted through 2008. For juvenile salmon that entered the ocean in 2008, ocean indicators suggested a highly favorable marine environment (NMFS 2009b). Because coastal upwelling was initiated early and the larger, energy-rich, coldwater plankton species were present in large numbers during 2007 and 2008, ocean conditions in the broader California Current appear to have been favorable for salmon survival in 2007 and to a greater extent in 2008.

Wells et al. (2008) developed a multivariate environmental index that can be used to assess ocean productivity on a finer scale for the central California region. This index (also referred to as the Wells Ocean Productivity Index) has also tracked the Northern Oscillation Index, which can be used to understand general ocean conditions in the North Pacific Ocean. In addition to its use as an indicator of general ocean productivity, the index may also relate to salmon dynamics due to their heavy reliance on krill and rockfish as prey items during early and later lifestages. Contrary to the poor ocean conditions observed in the spring of 2005 and 2006, the Wells et al. (2008) index parameters indicate spring ocean conditions have been generally favorable for salmon survival off California in 2007 and 2008.

In contrast to the relatively “good” ocean conditions that occurred in the spring, the Wells et al. (2008) index values for the summer of 2007 and 2008 were poor in general, and similar to previous years characterized by extremely low productivity of salmon off the central California coast.

Coastal waters off Oregon and northern California were affected by unusually strong downwelling during winter 2009-2010. Overall, spring 2009 appeared to be relatively good for salmon marine survival but oceanographic conditions appear to have deteriorated for salmon by late summer 2009 (Bjorkstedt et al. 2010). Juvenile salmonids at sea in the northern region of the California Current appear to have fared poorly during the warmer than usual conditions of summer and fall 2009.

In 2008 and 2009, poor Sacramento returns, primarily supported by Sacramento River fall-run Chinook salmon, led to the largest fishery closure on record. In 2009, adult spawning escapement for Sacramento River fall-run Chinook failed to meet the escapement goal (122,000-

180,000 adults) for the third year in a row, leading to the formal declaration of an overfishing concern (although fishing is not considered one of the major causes of the stock's decline). The forecast for the index of ocean abundance in 2010 was 245,500 adults, which provided adequate numbers for limited fisheries (PFMC 2011b).

NMFS (2009b) suggests that early marine survival for juvenile salmon is a critical phase in their survival and development into adults. The correlation between various environmental indices that track ocean conditions and salmon productivity in the Pacific Ocean, both on a broad and local scale, provides an indication of the role they play in salmon survival in the ocean. Moreover, when discussing the potential extinctions of salmon populations, Francis and Mantua (2003) state that climate patterns would not likely be the sole cause but could certainly increase the risk of extinction when combined with other factors, especially in ecosystems under stress from humans. Thus, the efforts to try and gain a greater understanding of the role ocean conditions play in salmon productivity will continue to provide valuable information that can be incorporated into the management of these species and should continue to be pursued. However, the highly variable nature of these environmental factors makes it very difficult, if not impossible, to accurately predict what they will be like in the future. Because the potential for poor ocean conditions exists in any given year, and because there is no way for salmon managers to control these factors, any deleterious effects endured by salmonids in the freshwater environment can only exacerbate the problem of an inhospitable marine environment (NMFS 2009b).

Global Climate Change

Warming over this century is projected to be considerably greater than over the last century (Thomas et al. 2009). Since 1900, the global average temperature has risen by about 1.5°F. By about 2100, it is projected to rise between 2°F and 10.5°F, but could increase up to 11.5°F (Thomas et al. 2009; California Climate Change Center 2006). In the United States, the average temperature has risen by a comparable amount and is very likely to rise more than the global average over this century, with some variation according to location.

Regarding climate change impacts already being observed, the Sierra Nevada Alliance (2008) reports that seven of the largest Sierra glaciers have retreated by 30 to 70 percent in the past 100 years. Changes observed over the past several decades also have shown that the earth is warming, and scientific evidence suggests that increasing greenhouse gas emissions are changing the earth's climate (Moser et al. 2009). Accumulating greenhouse gas concentrations in the earth's atmosphere have been linked to global warming, and projected future trends of increasing atmospheric greenhouse gas concentrations suggest global warming will continue (National Research Council 2001). Several factors will determine future temperature increases. Increases at the lower end of this range are more likely if global heat-trapping gas emissions are substantially reduced. If emissions continue to rise at or near current rates, temperature increases are more likely to be near the upper end of the range (NMFS 2014).

Global climate change has the potential to impact numerous environmental resources in California through potential, though uncertain, impacts related to future air temperatures and precipitation patterns, and the resulting implications to stream runoff rate and timing, water temperatures, reservoir operations, and sea levels. Although current models are broadly consistent in predicting increases in probable global air temperatures and increasing levels of greenhouse gasses resulting from human activities, there are considerable uncertainties about precipitation estimates. For example, many regional modeling analyses conducted for the western United States indicate that overall precipitation will increase, but uncertainties remain due to differences among larger-scale General Circulation Models (GCMs) (Kiparsky and Gleick 2003). Some researchers believe that climate warming might push the storm track on the West Coast further north, which would result in drier conditions in California. At the same time, relatively newer GCMs, including those used in the National Water Assessment, predict increases in California precipitation (DWR 2005a). Similarly, two popular climate models, including HadCM2 developed by the U.K. Hadley Center and the Parallel Climate Model (PCM) developed by the United States National Center for Atmospheric Research, also predict very different future scenarios. The HadCM2 predicts wetter conditions while the PCM predicts drier conditions (Brekke et al. 2004).

While much variation exists in projections related to future precipitation patterns, all available climate models predict a warming trend resulting from the influence of rising levels of greenhouse gasses in the atmosphere (Barnett et al. 2005). The potential effects of a warmer climate on the seasonality of runoff from snowmelt in the Central Valley have been well-studied and results suggest that melt runoff will likely shift from spring and summer to earlier periods in the water year (Vanrheenen et al. 2001). Presently, snow accumulation in the Sierra Nevada acts as a natural reservoir for California by delaying runoff from winter months when precipitation is high (Kiparsky and Gleick 2003). However, compared to present water resources development, Null et al. (2010) report that watersheds in the Northern Sierra Nevada are most vulnerable to decreased mean annual flow, southern-central watersheds are most susceptible to runoff timing changes, and the central portion of the range is most affected by longer periods with low flow conditions. Despite the uncertainties about future changes in precipitation rates, it is generally believed that higher temperatures will lead to changes in snowfall and snowmelt dynamics. Higher atmospheric temperatures will likely increase the ratio of rain to snow, shorten and delay the onset of the snowfall season, and accelerate the rate of spring snowmelt, which would lead to more rapid and earlier seasonal runoff relative to current conditions (Kiparsky and Gleick 2003). Studies suggest that the spring stream flow maximum could occur about one month earlier by 2050 (Barnett et al. 2005).

If air temperatures in California rise significantly, it will become increasingly difficult to maintain appropriate water temperatures in order to manage coldwater fisheries, including salmonids. A reduction in snowmelt and increased evaporation could lead to decreases in reservoir levels and, perhaps more importantly, coldwater pool reserves (California Energy Commission 2003). As a result, increasing air temperatures, particularly during the summer, lead to rising water temperatures in rivers and streams, which increase stress on coldwater fish. Projected temperatures for the 2020s and 2040s under a higher emissions scenario suggest that the habitat for these fish is likely to decrease dramatically (Mote et al. 2008; Salathé 2005; Keleher and Rahel 1996). Reduced summer flows and warmer water temperatures will create less favorable instream habitat conditions for coldwater fish species.

In the Central Valley, by 2100 mean summer temperatures may increase by 2° to 8°C (3.6°F to 14.4°F), precipitation will likely shift to more rain and less snow, with significant declines in total precipitation possible, and hydrographs will likely change, especially in the southern Sierra Nevada mountains (NMFS 2014). Thus, climate change poses an additional risk to the survival of salmonids in the Central Valley. As with their ocean phase, Chinook salmon and steelhead will be more thermally stressed by stream warming at the southern ends of their ranges (e.g., Central Valley Domain). For example, warming at the lower end of the predicted range (about 2°C (3.6°F)) may allow spring-run Chinook salmon to persist in some streams, while making some currently utilized habitat inhospitable (Lindley et al. 2007). At the upper end of the range of predicted warming, very little spring-run Chinook salmon habitat is expected to remain suitable (Lindley et al. 2007).

Under the expected warming of around 5°C (9°F), substantial amounts of habitat would be lost, with significant amounts of habitat remaining primarily in the Feather and Yuba rivers, and remnants of habitat in the upper Sacramento, McCloud, and Pit rivers, Battle and Mill creeks, and the Stanislaus River (Lindley et al. 2007). Under the less likely but still possible scenario of an 8°C (14.4°F) warming, spring-run Chinook salmon habitat would be found only in the upper-most reaches of the north fork Feather River, Battle Creek, and Mill Creek. This simple analysis suggests that Central Valley salmonids are vulnerable to warming, but more research is needed to evaluate the details of how warming would influence individual populations and sub-basins.

As summarized by Lindley et al. (2007), climate change may pose new threats to Central Valley salmonids by reducing the quantity and quality of freshwater habitat. Under the worst case scenario, spring-run Chinook salmon may be driven extinct by warming in this century, while the best-case scenario may allow them to persist in some streams, although prediction of the future status of Central Valley salmonids associated with long-term climate change is fraught with uncertainty.

By contrast to the conditions for other Central Valley floor rivers, climate change may not be likely to have such impacts on salmonids in the Yuba River downstream of Englebright Reservoir (YCWA 2010). Presently, the Yuba River is one of the few Central Valley tributaries that consistently have suitable water temperatures for salmonids throughout the year. Lower Yuba River water temperatures generally remain below 58°F year-round at the Smartsville gage (downstream of Englebright Dam), and below 60°F year-round at Daguerre Point Dam (YCWA et al. 2007). At Marysville, water temperatures generally remain below 60°F from October through May, and below 65°F from June through September (YCWA et al. 2007).

According to YCWA (2010), because of specific physical and hydrologic factors, the lower Yuba River is expected to continue to provide the most suitable water temperature conditions for anadromous salmonids of all Central Valley floor rivers, even if there are long-term climate changes. This is because New Bullards Bar Reservoir is a deep, steep-sloped reservoir with ample coldwater pool reserves. Throughout the period of operations of New Bullards Bar Reservoir (1969 through present), which encompasses the most extreme critically dry year on record (1977), the coldwater pool in New Bullards Bar Reservoir never was depleted. Since 1993, coldwater pool availability in New Bullards Bar Reservoir has been sufficient to accommodate year-round utilization of the reservoir's lower level outlets to provide cold water to the lower Yuba River. Even if climate conditions change, New Bullards Bar Reservoir still will have a very substantial coldwater pool each year that will continue to be available to provide sustained, relatively cold flows of water into the lower Yuba River during the late spring, summer and fall of each year (YCWA 2010).

Ocean Acidification

Ocean acidification has been called a "sister" or co-equal problem to climate change because it is caused by the same human-caused production of large amounts of CO_2. Its impacts are additional to, and may exacerbate, the effects of climate change (Alaska Marine Conservation Council 2011).

Seawater pH is a critical variable in marine systems. Today's surface ocean water is slightly alkaline, with a pH ranging from 7.5 to 8.5 and it is saturated with calcium carbonate, a very important organic molecule for organisms like corals, mollusks and crustaceans that make shells. As CO_2 reacts with the seawater, it lowers the pH and releases hydrogen ions. These ions bind strongly with carbonate, preventing it from forming the important calcium carbonate molecules. If the pH of the global oceans drops 0.4 by the end of the century as predicted, the levels of calcium carbonate available for use by marine organisms will decrease by 50 percent (Alaska Marine Conservation Council 2011).

Ocean acidification is likely to alter the biodiversity of the world's marine ecosystems and may affect the total productivity of the oceans. Previously it was thought that these changes would take centuries, but new findings indicate that an increasingly acidic environment could cause problems in high-latitude marine ecosystems within just a few decades (Alaska Marine Conservation Council 2011).

Currently, the oceans' surface water layers have sufficient amounts of calcium carbonate for organisms to use (known as saturated conditions). This calcium carbonate rich layer is deeper in warmer regions and closer to the surface in colder regions. Because calcium carbonate is less stable in colder waters, marine life in the polar oceans will be affected by calcium carbonate loss first. A study published in Nature by 27 U.S. and international scientists stated, "*Some polar and sub-polar waters will become under-saturated [at twice the pre-industrial level of CO_2, 560 ppm], probably within the next 50 years*" (Orr et al. 2005). Under-saturated refers to conditions in which the seawater has some calcium carbonate remaining, but it does not have enough available for the organisms to build strong shells (Alaska Marine Conservation Council 2011).

Research has shown that lowered ocean pH will affect the processes by which animals such as corals, mollusks and crustaceans make their support structures. Because these organisms depend on calcium carbonate, increasing acidity threatens their survival. At higher levels of acidity (lower pH levels), any organism that forms a shell through calcification — from clams to pteropods — could be adversely affected. These species use the naturally occurring carbonate minerals calcite and aragonite for the calcification process.

Pteropods are small planktonic mollusks that are at the bottom of the food chain and because of their dependence on calcium carbonate, they will be one of the first casualties of increasing acidity in Alaska's marine waters. In recent experiments exposing live pteropods to the conditions predicted by "business-as-usual" carbon emission scenarios – the pteropod shells showed evidence of dissolution and damage within only 48 hours. Pteropods are a key food source for salmon and other species (Alaska Marine Conservation Council 2011). Increased research into ocean acidification caused by the saturation of water with carbon dioxide suggests that a 10 percent decline in pteropod production can lead to a 20 percent reduction in the body weight of mature salmon (Climate Solutions 2011). A decrease in these mineral levels to food web base species like pteropods, also known as sea butterflies, which make up 45 percent of the diet for juvenile pink salmon, can cause cascading waves of disruption up the food chain (Climate Solutions 2011).

5.1.6.1.10 Non-Native Invasive Species

Non-native invasive species are of concern throughout the ESU and DPSs and can result in numerous deleterious effects to native species. For example, introduction of non-native invasive species can alter the natural food webs that existed prior to their introduction, as illustrated by the Asiatic freshwater clams *Corbicula fluminea* and *Potamocorbula amurensis* in the Delta. Cohen and Moyle (2004) report that the arrival of these two clam species disrupted the normal benthic community structure, and depressed phytoplankton levels in the Delta due to the highly efficient filter feeding of the introduced clams. Declines in phytoplankton levels have consequently resulted in reduced populations of zooplankton that feed upon them, thereby reducing the forage base available to salmonids transiting through the Delta and the San Francisco estuary on their ocean migrations. The lack of forage base can adversely affect the health and physiological condition of salmonids as they migrate to the Pacific Ocean.

Attempts to control non-native invasive plant species also can adversely affect the health and habitat suitability of salmonids within affected water systems, through either direct exposure to toxic chemicals or reductions in DO levels associated with the decomposition of vegetative matter in the water. As an example, control programs for the invasive water hyacinth and *Egeria densa* plants in the Delta must balance the toxicity of the herbicides applied to control the plants against the probability of exposure to listed salmonids during herbicide application period.

5.1.6.2 Lower Yuba River

The phenotypic lower Yuba River spring-run Chinook salmon population is exposed and subject to the myriad of limiting factors, threats and stressors described above for the Central Valley ESU. Lower Yuba River phenotypic spring-run Chinook salmon generally spend a few months (with some individuals remaining up to several months, or a year) in the lower Yuba River prior to migrating downstream through the lower Feather River, the lower Sacramento River, the Delta, and San Francisco Bay to the Pacific Ocean, where they spend from two to four years growing and maturing. Following their ocean residency, these fish then undertake an upstream migration through this same system, and are again exposed to the associated limiting factors, threats and stressors, prior to spending a few additional months in the lower Yuba River holding and subsequently spawning.

Three separate efforts have been undertaken over the past few years to identify, characterize and prioritize limiting factors (i.e., "stressors") for anadromous salmonids (including spring-run Chinook salmon) in the lower Yuba River. The Lower Yuba River Fisheries Technical Working Group (LYRFTWG), a multi-party stakeholder group including USACE and YCWA, established a process to rank stressors as part of the *Draft Implementation Plan for Lower Yuba River Anadromous Fish Habitat Restoration* (CALFED and YCWA 2005). The Yuba Accord Technical Team built upon these efforts and utilized a stressor analysis in the development of the Yuba Accord minimum flow requirements (i.e., "flow schedules") (YCWA et al. 2007).

NMFS (2014a) conducted a comprehensive assessment of stressors affecting spring-run Chinook salmon both within the lower Yuba River, and affecting lower Yuba River populations as they migrate downstream (as juveniles) and upstream (as adults) through the lower Feather River, the lower Sacramento River, and the Bay-Delta system.

As stated by NMFS (2014a), stressor matrices, which structured hierarchically related tiers in order to prioritize stressors, were developed. After all of the variables in the matrix were identified and weighted, stressors within the matrices were sorted in descending order (from the highest to the lowest biological impact). Although the resultant sorted matrices provide a pseudo-quantitative means of comparatively ranking individual stressors, to avoid attributing unwarranted specificity to the prioritized stressor list, it was distributed into four separate quartiles ("Very High," "High," "Medium," and "Low"). The ranking and quartile characterization of stressors were organized such that stressors affecting the individual lifestages also could be ascertained.

According to NMFS (2014a), for the lower Yuba River population of spring-run Chinook salmon, the number of stressors according to the categories of "Very High," "High," "Medium," and "Low" that occur in the lower Yuba River or occur out of basin are presented below by lifestage (Table 5.1-2).

Table 5.1-2. The number of stressors according to the categories of "Very High," "High," "Medium," and "Low" that occur in the lower Yuba River, or occur out-of-basin, by lifestage for the lower Yuba River population of spring-run Chinook salmon.

Lifestage	Location	Stressor Categories			
		Very High	High	Medium	Low
Adult Immigration and Holding	Lower Yuba River	2	1	3	1
	Out of Basin	1	5	8	6
Spawning	Lower Yuba River	3	2	0	2
	Out of Basin	N/A*	N/A	N/A	N/A
Embryo Incubation	Lower Yuba River	1	0	4	0
	Out of Basin	N/A	N/A	N/A	N/A
Juvenile Rearing and Outmigration	Lower Yuba River	5	1	1	5
	Out of Basin	12	16	6	9

Source: NMFS 2014
* Not Applicable. These lifestages for this population only occur in the lower Yuba River.

As shown by the numbers in Table 5.1-2, of the total number of 94 stressors affecting all identified lifestages of the lower Yuba River populations of spring-run Chinook salmon, 31 are within the lower Yuba River and 63 are out-of-basin. Because spawning and incubation occurs only in the lower Yuba River, all of the stressors associated with these lifestages occur in the lower Yuba River. Therefore, for the adult immigration and holding, and the juvenile rearing and outmigration lifestages combined, a total of 43 "Very High" and "High" stressors were identified, with 9 of those occurring in the lower Yuba River and 34 occurring out-of-basin.

NMFS (2014a) Recovery Plan states that "*Implementation of the flow schedules specified in the Fisheries Agreement of the Yuba Accord is expected to address the flow-related major stressors including flow-dependent habitat availability, flow-related habitat complexity and diversity, and water temperatures*."

As acknowledged by NMFS in this statement, stressors associated with instream flows and water temperatures in the lower Yuba River have been addressed, to the extent feasible within hydrological constraints, by the Yuba Accord. Stressors on lower Yuba River spring-run Chinook salmon are discussed below, primarily based upon information presented in NMFS (2009a, 2014a).

As part of its analysis of potential effects to critical habitat, NMFS will conduct an exposure analysis to: (1) identify the species and critical habitats that are likely to occur in the same space and at the same time as potential stressors; and (2) identify the number and age (or lifestage) of individuals that are likely to be exposed to an action's effects, the population that those individuals represent, and the specific areas and PBFs of critical habitat that are likely to be exposed (NMFS 2016a). The information provided below is intended to assist NMFS address potential concerns about exposure responses that may be sufficient to reduce the quantity, quality, or availability of PBFs within the Action Area.

PBFs of designated spring-run Chinook salmon critical habitat in the lower Yuba River include freshwater spawning sites, freshwater rearing sites, and freshwater migration corridors. A description of the primary biological features of spring-run Chinook salmon critical habitat that are present within the Action Area, including potential stressors to spring-run Chinook salmon and other factors affecting PBFs, is described below.

5.1.6.2.1 Passage Impediments/Barriers

Englebright Dam was not designed for fish passage and presents an impassable barrier to the upstream migration of anadromous salmonids, and marks the upstream extent of currently accessible spring-run Chinook salmon habitat in the lower Yuba River, whereas Daguerre Point Dam presents a potential impediment to upstream migration.

Englebright Dam, built in 1941 to retain hydraulic mining debris from the Yuba River Basin, blocks upstream migration of fish in the lower Yuba River and, in particular, blocks the migration of steelhead and spring-run Chinook salmon to their historic spawning grounds (NMFS 2002).

Daguerre Point Dam has been reported to be an impediment to upstream migration of adult salmon and steelhead under certain conditions. Factors contributing to impeded adult spring-run Chinook salmon upstream passage have been suggested to include inadequate attraction flows to the ladders, proximity and orientation of the ladder entrances to the spillway, periodic obstruction of the ladders by sediment and woody debris, and other fish ladder physical design issues.

Sheet flow across the dam's spillway, particularly during high-flow periods, may obscure ladder entrances and, thus, makes it difficult for immigrating adult salmonids to find the entrances (NMFS 2007). For example, fall-run Chinook salmon have been observed attempting to leap over the dam, demonstrating that these fish may have difficulty in finding the fish ladder entrances (USACE 2000). This phenomenon may particularly affect spring-run Chinook salmon, because spring-run adult Chinook salmon upstream migration encompasses the relatively high-flow periods of spring through early summer. Since 2001, wooden flashboards have been periodically affixed to the crest of the dam during low flow periods to aid in directing the flows towards the fish ladder entrances. Fish passage monitoring data from 2006 indicates that the installation of the flashboards resulted in an immediate and dramatic increase in the passage of salmon up the ladders, and is thought to have improved the ability of salmon to locate and enter the ladders (NMFS 2007).

Both the north and south fish ladders at Daguerre Point Dam, particularly the north ladder, historically tended to clog with woody debris and sediment, which had the potential to block passage or substantially reduce attraction flows at the ladder entrances. Additionally: 1) the north and south ladders' exits are close to the spillway, potentially resulting in adult fish exiting the ladder being immediately swept by flow back over the dam; 2) sediment accumulates at the upstream exits of the fish ladders, reducing the unimpeded passage from the ladders to the main channel, and may cause potential "fall-back" into the ladders; and 3) fish could jump out of the upper bays of the fishway, resulting in direct mortality. Many of the past issues associated with woody debris accumulation have either been eliminated or minimized since locking metal grates were installed over the unscreened bays on the north and south fish ladders during 2011.

The RMT (2013a) examined passage of adult Chinook upstream of Daguerre Point Dam and corresponding flow data during eight years of available data. Chinook salmon passage was observed over a variety of flow conditions, including ascending or descending flows, as well as during extended periods of stable flows. Flow thresholds prohibiting passage of Chinook salmon through the ladders at Daguerre Point Dam were not apparent in the data (RMT 2013a).

Phenotypic spring-run Chinook salmon (those entering the lower Yuba River during spring months) may remain in the lower Yuba River in areas downstream (and proximate) to Daguerre Point Dam for extended periods of time during the spring and summer. It is uncertain whether, or to what extent, the duration of residency in the large pool located downstream of Daguerre Point Dam is associated with upstream passage impediment and delay, or volitional habitat utilization prior to spawning in upstream areas. However, RMT (2013a) reported that temporal migrations of adult phenotypic spring-run Chinook salmon to areas upstream of Daguerre Point Dam occurred over an extended period of time. The tagged spring-run Chinook salmon in the lower Yuba River actually migrated upstream of Daguerre Point Dam from May through September, and utilized a broad expanse of the lower Yuba River during the phenotypic summer holding period, including areas as far downstream as Simpson Lane Bridge (i.e., ~RM 1.8), and as far upstream as the area just below Englebright Dam. A longitudinal analysis of acoustic tag detection data indicated that distributions were non-random, and that the tagged spring-run Chinook salmon were selecting locations for holding (RMT 2013a).

NMFS (2007) suggested that delays resulting from adult spring-run Chinook salmon adult passage impediments could weaken fish by requiring additional use of fat stores prior to spawning, and potentially could result in reduced spawning success (i.e., production) from reduced resistance to disease, increased pre-spawning mortality, and reduced egg viability. However, these statements suggesting biological effects associated with fish passage issues at Daguerre Point Dam are not supported by studies or referenced literature. For example, the RMT (2010b) included evaluation of water temperatures at Daguerre Point Dam during the spring-run Chinook salmon adult upstream immigration and holding lifestage, which addressed considerations regarding both water temperature effects to pre-spawning adults and egg viability. They concluded that during this lifestage, characterized as extending from April through August, water temperatures [modeled] at Daguerre Point Dam are suitable and remain below the reported optimum water temperature index (WTI) value of 60°F at least 97 percent of the time over all water year types during these months. Thus, it is unlikely that this represents a significant source of mortality to spring-run Chinook salmon. Moreover, actual data monitored since the Yuba Accord has been implemented (October 2006 to July 2016) demonstrate that water temperatures at Daguerre Point Dam nearly always remained at about or below 60°F during the adult immigration and holding period each year, except for about a 3-week (June) and a 4-week (September-October) period during 2014, and during June-October 2015. Although monitored water temperatures at Daguerre Point Dam exceeded the optimum WTI value of 60°F for a few weeks during the extreme summer conditions experienced in 2014, the increase in daily average temperatures was ≤ 1°F in June 2014 and ≤ 1.5°F from September to October in 2014. During 2015, although monitored water temperatures at Daguerre Point Dam exceeded the optimum WTI value of 60°F for a few weeks, water temperatures never reached the upper tolerable WTI value of 65°F for adult holding, nor the upper tolerable WTI value of 68°F for adult spring-run Chinook salmon immigration.

As described in NMFS' *2016 Viability Assessment for Pacific Salmon and Steelhead Listed Under the Endangered Species Act: Southwest* (Williams et al. 2016), "*California has experienced well below average precipitation in each of the past four water years (2012, 2013, 2014, and 2015), record high surface air temperatures the past two water years (2014 and 2015), and record low snowpack in 2015. Some paleoclimate reconstructions suggest that the current four-year drought is the most extreme in the past 500 or perhaps more than 1,000 years. Anomalously high surface temperatures have made this a "hot drought", in which high surface temperatures substantially amplified annual water deficits during the period of below average precipitation.*" NMFS further recognizes that "*four consecutive years of drought (2012−2015) and the past two years (2014−2015) of exceptionally high air, stream, and upper ocean temperatures have together likely had negative impacts for many populations of Chinook salmon*" (Williams et al. 2016).

As shown by the monitoring data, the thermal conditions experienced in the lower Yuba River during the last two years of extreme drought (2014, 2015) exhibited only relatively slight increases in water temperatures at Daguerre Point Dam. Also, water temperatures in the lower

Yuba River above Daguerre Point Dam continued to remain thermally suitable for spring-run Chinook salmon adult immigration and holding.

As reported by NMFS (2007), Daguerre Point Dam may adversely affect outmigration success of juvenile salmon and steelhead. During downstream migration, juvenile Chinook salmon and steelhead may be disoriented or injured as they plunge over the spillway, increasing their exposure and vulnerability to predators in the large pool at the base of the dam (NMFS 2007).

In consideration of all of the potential associated effects of the dams, both Englebright and Daguerre Point Dam represent a high stressor to spring-run Chinook salmon.

5.1.6.2.2 Harvest/Angling Impacts

Fishing for Chinook salmon on the lower Yuba River is regulated by Cal Fish and Wildlife. Although harvest/angler impacts were previously listed as a stressor, the magnitude of this potential stressor has been reduced associated with changes in fishing regulations over time. Angling regulations on the lower Yuba River are intended to protect sensitive species, in particular spring-run Chinook salmon (and wild steelhead). Cal Fish and Wildlife angling regulations (2016-2017) (CDFW 2016a) state that the Yuba River from its confluence with the lower Feather River up to Englebright Dam is closed year-round to salmon fishing and no take or possession of salmon is allowed.

Fishing for hatchery trout or hatchery steelhead is allowed on the lower Yuba River from its confluence with the lower Feather River up to the Highway 20 Bridge year-round. The lower Yuba River, between the Highway 20 Bridge and Englebright Dam, is closed to fishing from September through November to protect spring-run Chinook salmon spawning activity and egg incubation.

Although these regulations are intended to specifically protect spring-run Chinook salmon, anglers can potentially harass, harm and kill listed species (spring-run Chinook salmon and wild steelhead) through incidental actions while targeting non-listed species. Examples of potential angler impacts may include, but are not necessarily limited to, angler harvest, physical disturbance of salmonid redds, hooking and catch-and-release stress or mortality, including that which results from incidental hooking (CALFED and YCWA 2005). Overall, angling impacts represent a relatively low stressor to spring-run Chinook salmon in the lower Yuba River.

5.1.6.2.3 Poaching

"Poaching" is a term used in this Applicant-Prepared Draft BA to represent any illegal fishing activity in the lower Yuba River. Poaching of adult Chinook salmon at the fish ladders and at the base of Daguerre Point Dam has been previously reported in several documents. Poaching has been reported as a "chronic problem" by Falxa (1994 as cited in CALFED and YCWA 2005), as an "ongoing problem" at Daguerre Point Dam by CDFG (1998), and as a "long-standing problem" on the Yuba River, particularly at Daguerre Point Dam, by John Nelson (CDFG, pers. comm., November 2000, as cited in NMFS 2005b). USACE (2001) and NMFS (2009a) also referred to poaching of adult Chinook salmon at the Daguerre Point Dam.

Although these reports referred to poaching within the fish ladders and immediately downstream of Daguerre Point Dam as potential issues of concern, until recently, the only account of documented poaching was provided by Nelson (2009). In his declaration, Nelson (2009) stated that during his tenure at Cal Fish and Wildlife (which extended until 2006) he personally observed people fishing illegally in the ladders, and further observed gear around the ladders used for poaching. The time period to which Nelson (2009) was referring is not clear, although he may have been referring to the period prior to 2000.

While poaching had been previously reported as a stressor, it was unclear whether, or to what extent, poaching affected the spring-run Chinook salmon population in the lower Yuba River. According to Sprague (2011), the amount of poaching from the fish ladders has not been quantified, and there does not appear to be data on the amount of poaching, so the extent of the problem has not been well understood.

On February 18, 2014, HDR's Fisheries Team met with two Cal Fish and Wildlife Wardens, to discuss whether poaching has been observed on the lower Yuba River and specifically at Daguerre Point Dam. The Wardens stated that they regularly observe fishing line gear and other evidence of illegal fishing at Daguerre Point Dam (Figure 5.1-7) and described poaching as a growing problem on the lower Yuba River, specifically in the plunge pool immediately downstream of Daguerre Point Dam, where spring-run Chinook salmon hold during the summer.

Figure 5.1-7. Illegal hooks used for poaching at Daguerre Point Dam.

For example, in August 2013, after observing illegal hooks and blood at Daguerre Point Dam, a Warden captured poachers that were pulling fish out of the pool below Daguerre Point Dam. The poachers were in possession of illegal fishing gear, and salmon they had caught illegally that night. The Warden estimated that during the summer of 2013, prior to being captured in August, the poachers had caught nearly 50 spring-run Chinook salmon. Moreover, the Warden indicated that the poachers were previously cited for poaching two years prior to August of 2013 and that poachers are often repeat offenders that may act as guides to repeatedly bring new groups of people to illegally fish at the Daguerre Point Dam pool.

The Wardens also reported that some fisherman drive boats from the lower Feather River and come up the lower Yuba River to Daguerre Point Dam, where they illegally catch several Chinook salmon quickly and then go back to the lower Feather River, where they can claim the salmon as legally caught in the lower Feather River. These incidents of poaching in the Daguerre Point Dam pool usually occur at dusk. In addition, there has been an anecdotal report of jet skis dragging a net between them in the Daguerre Point Dam pool.

In addition to poaching in the Daguerre Point Dam pool, the Wardens noted observations of people fishing directly in the north fish ladder and off the wing dam into the mouth of the ladder. Nelson (2009) suggested that one measure that could reduce poaching would be to place grates over the top of the ladders to restrict poacher access. However, Sprague (2011) expressed concern regarding the potential for fish injury resulting from multiple sharp edges of the grates and contact with adult migrating fish. He further suggested that solid covers could be used, but consideration should be given to the potential for how to avoid pressurizing the fish ladders during high flow events.

The July 25, 2011, Interim Remedy Order issued by the United States District Court, Eastern District of California in Case 2:06-cv-02845-LKK-JFM ordered USACE to install locking metal grates over all but the lower eight bays of the fish ladders at Daguerre Point Dam by September 14, 2011. This Order was issued in part, to prohibit the potential for poaching in the fish ladders at Daguerre Point Dam. In response to the Interim Remedy Order, during the summer of 2011 USACE proceeded with installation of locking metal grates on all 33 unscreened bays. Due to concerns expressed by both NMFS and Cal Fish and Wildlife, the Court then reconsidered the requirement to put grates over the bays on the lowermost section of the south fish ladder at Daguerre Point Dam. Consequently, grates were not installed over the lower eight bays of the south fish ladder at Daguerre Point Dam, which provides the potential for illegal fishing directly in the ladder, which has been observed by the Wardens.

Illegal fishing in the lower Yuba River is not confined to the vicinity of Daguerre Point Dam. Cal Fish and Wildlife Wardens have written citations for poaching throughout the lower Yuba River. For example, at the Highway 20 Bridge, poachers have been cited for snagging Chinook salmon from redds using frog gigs and spears. In the Hallwood area, poachers have been observed attempting to pull RSTs onto the bank in order to remove juvenile Chinook salmon.

During the 2006 October carcass survey, Pacific States Marine Fisheries Commission (PSMFC) staff found fillets of approximately 13 salmon at Lower Gilt Edge Bar, located downstream of the Highway 20 Bridge. The Wardens also reported catching poachers at Upper Gilt Edge Bar in possession of steelhead ranging 16 to 22 in length, well in excess of the daily bag limit.

Poachers also target downstream-migrating juvenile Chinook salmon, netting them at sandbars and other areas along the river to use as sturgeon bait, and Wardens have ticketed individuals who had caught 25-50 juveniles each. The Wardens also have found buckets with over 200-300 juvenile Chinook salmon that have been caught by poachers.

While poaching is most notable during summer when spring-run Chinook salmon are present in the lower Yuba River, steelhead also are affected. Fishers using illegal worms and hooks are known to target trout and steelhead, particularly in the reach between Parks Bar and Hammonton Road in the Goldfields.

The regular observations of evidence of poaching, as well as the Wardens' accounts of poachers fishing illegally at the Daguerre Point Dam pool and fish ladders, and throughout other sections of the lower Yuba River, suggest that poaching could present a high stressor to spring-run Chinook salmon in the lower Yuba River.

5.1.6.2.4 Hatchery Effects

Although no fish hatcheries are located on the lower Yuba River, and the river continues to support a persistent population of spring-run Chinook salmon that spawn downstream of Englebright Dam, the genetic integrity of the fish expressing the phenotypic characteristics of spring-run Chinook salmon is presently uncertain. CDFG (1998) suggested that spring-run Chinook salmon populations may be hybridized to some degree with fall-run Chinook salmon due to lack of spatial separation of spawning habitat. Also, the observation of adipose fin clips on adult Chinook salmon passing upstream through the VAKI Riverwatcher™ system at Daguerre Point Dam during the spring demonstrates that hatchery straying into the lower Yuba River has and continues to occur, most likely from the FRFH (NMFS 2009a; RMT 2013a).

Feather River Fish Hatchery Genetic Considerations

Spring-run Chinook salmon from the FRFH were planted in the lower Yuba River during 1980 (CDFG 1991a). In addition, it is possible that some hatchery-reared juvenile Chinook salmon from the FRFH may move into the lower Yuba River in search of rearing habitat. Some competition for resources with naturally spawned spring-run Chinook salmon could occur as a result (YCWA et al. 2007). The remainder of this discussion pertains to hatchery effects associated with the straying of adult Chinook salmon into the lower Yuba River.

The FRFH is the only hatchery in the Central Valley that currently produces spring-run Chinook salmon. The FRFH was constructed in 1967 to compensate for anadromous salmonid spawning habitat lost with construction of the Oroville Dam. The FRFH has a goal of releasing 2,000,000 spring-run Chinook salmon smolts annually (DWR 2004).

From 1962 to 1966, spring-run Chinook salmon were trapped and trucked above Oroville Dam. Beginning in 1967, spring-run Chinook salmon were collected for artificial propagation at FRFH as the construction of Oroville Dam was completed. The program is funded by the DWR and managed by Cal Fish and Wildlife (NMFS 2004b).

The program was founded with local native stock collected at the FRFH. Early attempts to over-summer spring-run Chinook salmon at the hatchery resulted in high mortality and the decision to allow the run to hold in the river until September 1. Prior to 2004, FRFH hatchery staff differentiated spring-run Chinook salmon from fall-run Chinook salmon by opening the ladder to the hatchery on September 1 (NMFS 2009a). Those fish ascending the ladder from September 1 through September 15 were assumed to be spring-run Chinook salmon while those ascending the ladder after September 15 were assumed to be fall-run (Kastner 2003 as cited in NMFS 2009a). This practice led to considerable hybridization between spring- and fall-run Chinook salmon (DWR 2004). Since 2004, the FRFH fish ladder remains open during the spring months, closing on June 30, and those fish ascending the ladder are marked with an external floy tag and returned to the river. This practice allows FRFH staff to identify those previously marked fish as spring-run when they re-enter the ladder in September. Only floy-tagged fish are spawned with floy-tagged fish in the month of September. No other fish are spawned during this time, as part of an effort to prevent hybridization with fall-run, and to introduce a temporal separation between stocks in the hatchery. During the FRFH spring-run spawning season, all heads from adipose fin-clipped fish are taken and sent to Cal Fish and Wildlife's laboratory in Santa Rosa for tag extraction and decoding. The tag information will be used to test the hypothesis that early spring-run spawners will produce progeny that maintain that run fidelity.

Regardless of recently improved FRFH practices, previous practices appear to have resulted in hybridization between "spring-run" and "fall-run" Chinook salmon. The following discussion was taken from Garza et al. (2008).

Evaluation of the FRFH "spring-run" stock found that it is genetically most similar to the FRFH fall-run stock, as indicated both by clustering on the phylogeographic trees and by comparison of the [standardized variance in allele frequencies between the sample years] (F_{ST}) values, and is nested within the fall-run group of populations in all analyses (Garza et al. 2008). F_{ST} values between the FRFH "spring-run" and naturally-spawned spring-run are in the low end of the range of values for fall-run populations to spring-run populations, but not the lowest. In addition, they are the essentially the same as those of FRFH fall-run to spring-run populations. This demonstrates convincingly that the FRFH "spring-run" stock is dominated by fall-run ancestry. However, Garza et al. (2008) also found very slight, but significant, differentiation between the two FRFH stocks, which is concordant with the results of Hedgecock et al. (unpublished study as cited in Garza et al. 2008) on these stocks. In addition, Garza et al. (2008) found a strong signal of linkage (gametic phase) disequilbrium, absent in all other population samples, in the FRFH "spring-run" stock. Garza et al. (2008) interpreted this as evidence that the FRFH "spring" run retains remnants of the phenotype and ancestry of the Feather River spring-run Chinook salmon that existed prior to the dam and hatchery (as opposed to representing a hatchery selection-created and maintained phenotypic variant), but that has been heavily introgressed by fall-run Chinook salmon through some combination of hatchery practices and natural hybridization, induced by habitat concentration due to lack of access to spring-run Chinook salmon habitat above the dam. This suggests that it may be possible to preserve some additional component of the ancestral Central Valley spring-run Chinook salmon genomic variation through careful management of this stock that can contribute to the recovery of the ESA-listed Central Valley spring-run Chinook salmon ESU, although it will not be possible to reconstitute a "pure" spring-run stock from these fish (Garza et al. 2008).

The FRFH spring-run Chinook salmon population is part of the Central Valley spring-run Chinook salmon ESU (70 FR 37160). At the time of issuance of the final rule regarding the listing status of the Central Valley ESU of spring-run Chinook salmon, NMFS (70 FR 37160) recognized that naturally spawning spring-run Chinook in the Feather River are genetically similar to the FRFH spring-run Chinook stock, and that the hatchery stock shows evidence of introgression with Central Valley fall-run Chinook salmon. NMFS also stated that FRFH stock should be included in the ESU because the FRFH spring-run Chinook salmon stock may play an important role in the recovery of spring-run Chinook salmon in the Feather River Basin, as efforts progress to restore naturally spawning spring-run populations in the Feather and Yuba Rivers (70 FR 37160).

Although the FRFH spring-run Chinook salmon population is part of the Central Valley spring-run Chinook salmon ESU, concern has been expressed that straying of FRFH fish into the lower

Yuba River may represent an adverse impact due to the potential influence of previous hatchery management practices on the genetic integrity of FRFH spring-run Chinook salmon.

Straying into the Lower Yuba River

The RMT (2013a) reported that substantially higher amounts of straying of adipose fin-clipped Chinook salmon into the lower Yuba River occur than that which was previously believed. Although no quantitative analyses or data were presented, NMFS (2007) stated that some hatchery fish stray into the lower Yuba River and that these fish likely come from the FRFH.

Some information indicating the extent to which adipose-clipped Chinook salmon originating from the FRFH return to the lower Yuba River is available from coded wire tag (CWT) analysis. During the October through December 2010 carcass survey period in the lower Yuba River, the RMT collected heads from fresh Chinook salmon carcasses with adipose fin clips, and sent the heads to the Cal Fish and Wildlife CWT interpretive center. In April of 2011, the results of the interpretation of the CWTs became available. Of the 333 Chinook salmon heads sent to the Cal Fish and Wildlife interpretive center, 11 did not contain a CWT, 8 were fall-run Chinook salmon from the Coleman National Fish Hatchery, 2 were from the RST captured and tagged juveniles in the lower Yuba River, 1 was a naturally-spawned fall-run Chinook salmon from the Feather River, 1 was a fall-run Chinook salmon from the Mokelumne River Hatchery, and 310 were Chinook salmon from the FRFH (234 spring-run and 76 fall-run Chinook salmon). Thus, for all CWT hatchery-origin fish returning to the Yuba River from out-of-basin sources, 97 percent were from the FRFH. However, this information does not indicate the percentage of hatchery contribution from the FRFH to the phenotypic spring-run Chinook salmon run in the lower Yuba River, because, among other reasons, all of these heads were collected during the fall and represent a mixture of phenotypic spring- and fall-run Chinook salmon spawning in the lower Yuba River (RMT 2013a).

Additional information that can be used to assess the amount of straying of FRFH Chinook salmon into the lower Yuba River is provided from VAKI Riverwatcher™ data collected from 2004 through 2015. The estimated numbers of adipose fin-clipped spring-run Chinook salmon that passed upstream of Daguerre Point Dam from 2004 through 2015 that were derived from the VAKI Riverwatcher™ data are an indicator of the minimum number of Chinook salmon of hatchery origin (most likely of FRFH origin) that strayed into the lower Yuba River. The following discussion of adipose fin-clipped spring-run Chinook salmon is updated from RMT (2013a), to include four additional years (March 2012 – February 2016) of VAKI Riverwatcher™ data. Discussion of the procedure utilized by the RMT to first differentiate phenotypic spring-run from phenotypic fall-run Chinook salmon is provided in Section 5.1.7.2.2, below.

Because the VAKI Riverwatcher™ systems located at both the north and south ladder of Daguerre Point Dam can record both silhouettes and electronic images of each fish passage event, the systems were able to differentiate Chinook salmon with adipose fins clipped or absent from Chinook salmon with their adipose fins intact. Thus, annual series of daily counts of Chinook salmon with adipose fins clipped (i.e., ad-clipped fish) and with adipose fins intact (i.e., not ad-clipped fish) that passed upstream of Daguerre Point Dam from March 1, 2004 through

February 28, 2016 were obtained. The estimated numbers of spring-run Chinook salmon of hatchery (i.e., ad-clipped fish) and potentially non-hatchery origin (i.e., not ad-clipped fish) passing upstream of Daguerre Point Dam for the last 12 years of available VAKI Riverwatcher™ data were used to evaluate straying into the lower Yuba River.

Relationships between Spring-run Chinook Salmon Straying into the lower Yuba River and Attraction Flows and Water Temperatures

As reported by RMT (2013a), to evaluate the influence of "attraction" flows and water temperatures on the straying of adipose fin-clipped adult phenotypic spring-run Chinook salmon into the lower Yuba River, variables related to flows and water temperatures in the lower Yuba River and the lower Feather River were developed and statistically related to the weekly proportions of adipose fin-clipped phenotypic spring-run Chinook salmon (relative to all spring-run Chinook salmon) passing upstream of Daguerre Point Dam. Analyses presented in this Applicant-Prepared BA were updated to include one additional year (March 2012 – February 2013) of monitoring data. Details of this analytical evaluation are provided in RMT (2013a).

Results of the RMT (2013a) analysis suggest that there is a moderately strong (R^2=0.72) and highly significant (P < 0.000001) relationship between the percentage of adipose fin-clipped spring-run Chinook salmon contribution to the weekly spring-run Chinook salmon total counts at Daguerre Point Dam and the attraction flow and WTIs four weeks prior. The updated analysis including an additional year (March 2012 – February 2013) provided similar results. Results of the analysis applied to the 9 years of VAKI Riverwatcher™ counts currently available suggest that there also is a moderately strong (R^2=0.65) and highly significant (P < 0.000001) relationship, between the weekly percentage of adipose fin-clipped spring-run Chinook salmon and the attraction flow and water temperature indices six weeks prior, by contrast to four weeks prior in the previous analysis. Figure 5.1-8 displays the 3-D response surface produced by the fitted logistic model for the updated analysis.

The analysis showed that an estimated 65 percent of the variation in the proportion of adipose fin-clipped phenotypic spring-run Chinook salmon passing upstream of Daguerre Point Dam can be accounted for by the ratio of lower Yuba River flow relative to lower Feather River flow, and the ratio of lower Yuba River water temperature relative to lower Feather River water temperature, 6 weeks prior to the time of passage at Daguerre Point Dam. In other words, the higher the Yuba River flows relative to Feather River flows, combined with the lower the Yuba River water temperatures relative to Feather River water temperatures, the higher the percentage of fin-clipped Chinook salmon passing upstream of Daguerre Point Dam 6 weeks later.

As described in RMT (2013a), the acoustically-tagged phenotypic spring-run Chinook salmon spent variable and extended periods of time holding below Daguerre Point Dam after being tagged and prior to passing upstream of Daguerre Point Dam, with a range of 0 to 116 days. Based on all 67 acoustically-tagged spring-run Chinook salmon that passed upstream of Daguerre Point Dam, the average holding time before passing upstream of Daguerre Point Dam was about 50 days.

For the phenotypic acoustically-tagged spring-run Chinook salmon that passed upstream of Daguerre Point Dam by the annual spring-run Chinook salmon demarcation date for each year, the average holding periods before passing upstream of Daguerre Point Dam were approximately 51, 41, and 57 days during 2009, 2010 and 2011, respectively. Therefore, it would be expected that attraction of adipose fin-clipped fish to the lower Yuba River associated with flows and water temperatures in the lower Yuba River relative to the lower Feather River would occur at least several weeks prior to passage of phenotypic spring-run Chinook salmon upstream of Daguerre Point Dam (RMT 2013a).

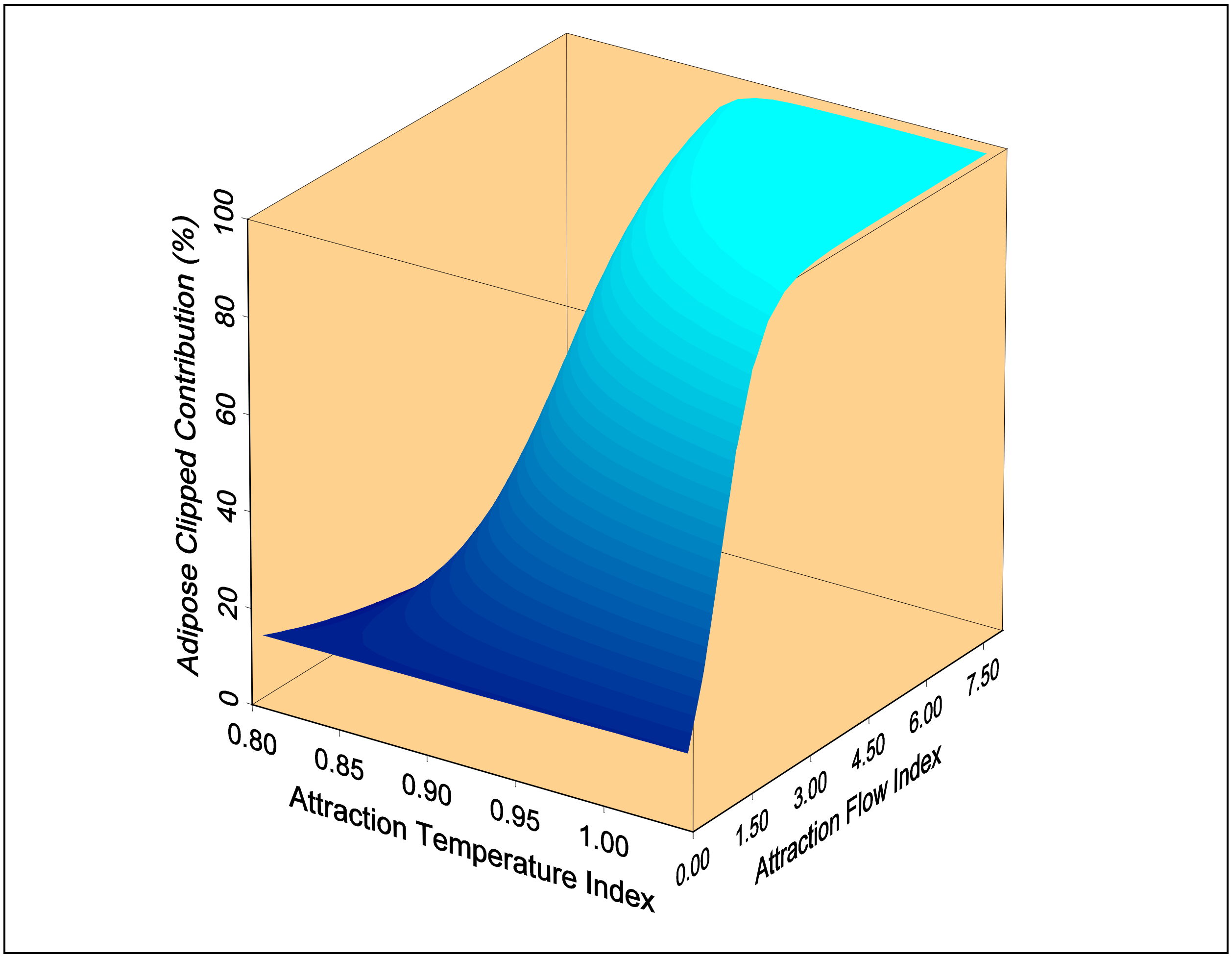

Figure 5.1-8. Relationship of the weekly percentage of adipose fin-clipped contribution to the weekly phenotypic spring-run Chinook salmon count at Daguerre Point Dam as function of the weekly attraction flow and water temperature indices calculated six weeks prior to the week of passage at Daguerre Point Dam.

While the variation in the proportion of adipose fin-clipped phenotypic spring-run Chinook salmon passing Daguerre Point Dam was best explained with ratios of flows and water temperatures in the lower Yuba and Feather rivers 6 weeks prior to passage at Daguerre Point Dam, the acoustically-tagged individuals exhibited a somewhat longer duration of holding on average. However, due to the relatively small sample size of acoustically-tagged spring-run

Chinook salmon passing upstream of Daguerre Point Dam (N=67), the short duration of the study, and based on the highly variable holding duration (i.e., 0-116 days), the average holding time calculated for the acoustically-tagged spring-run Chinook salmon is considered to be a general approximation of holding duration downstream of Daguerre Point Dam (RMT 2013a). Therefore, consideration of holding duration downstream of Daguerre Point Dam supports the observation that the ratios of flows and water temperatures in the lower Yuba River relative to the lower Feather River 6 weeks prior to passage of spring-run Chinook salmon at Daguerre Point Dam may be influencing the attraction of adipose fin-clipped spring-run Chinook salmon of FRFH-origin into the lower Yuba River.

Lower Yuba River Genetic Considerations

Spring-run Chinook salmon historically acquired and maintained genetic integrity through reproductive (spatial-temporal) isolation from other Central Valley Chinook salmon runs. However, construction of dams has prevented access to headwater areas and much of this historical reproductive isolation has been compromised, resulting in intermixed life history traits in many remaining habitats (YCWA 2010).

Between 1900 and 1941, debris dams constructed on the lower Yuba River by the California Debris Commission to retain hydraulic mining debris, now owned and operated by USACE, completely or partially blocked the migration of Chinook salmon and steelhead to historic spawning and rearing habitats (CDFG 1991b; Wooster and Wickwire 1970; Yoshiyama et al. 1996). Englebright Dam (constructed in 1941) completely blocks spawning runs of Chinook salmon and steelhead, and is the upstream limit of fish migration. Fry (1961) reported that a small spring-run Chinook salmon population historically occurred in the lower Yuba River, but the run virtually disappeared by 1959.

Since the completion of New Bullards Bar Reservoir in 1970 by YCWA, higher, colder flows in the lower Yuba River have improved conditions for over-summering and spawning of spring-run Chinook salmon in the lower Yuba River (YCWA et al. 2007). As of 1991, a remnant spring-run Chinook salmon population reportedly persisted in the Yuba River downstream of Englebright Dam maintained by fish produced in the Yuba River, fish straying from the Feather River, or fish previously and infrequently stocked from the FRFH (CDFG 1991a). In the 1990s, relatively small numbers of Chinook salmon exhibiting spring-run phenotypic characteristics were reported to have been observed in the lower Yuba River (CDFG 1998). Although precise escapement estimates are not available, the USFWS testified at the 1992 SWRCB lower Yuba River hearing that "*…a population of about 1,000 adult spring-run Chinook salmon now exists in the lower Yuba River*" (San Francisco Bay RWQCB 2006 as cited in NMFS 2009a).

If spring-run Chinook salmon were extirpated from the lower Yuba River in 1959 (Fry 1961) and, as reported by CDFG (1991), a population of spring-run Chinook salmon became reestablished since the 1970s due to improved habitat conditions and fish straying from the Feather River or stocked and straying from the FRFH, then it is likely that spring-run Chinook salmon on the lower Yuba river do not represent a "pure" ancestral genome.

There also is concern that the existing spring-run Chinook salmon population has interbred with fall-run Chinook salmon and, as a result, it is a hybrid species and not a true spring-run species (USACE 2001). In addition to the effects of hatchery straying, an additional issue regarding the genetic integrity of phenotypic spring-run Chinook salmon in the lower Yuba River pertains to the loss or reduction of reproductive isolation. Spring-run Chinook salmon acquired and maintained genetic integrity through spatial-temporal isolation from other Central Valley Chinook salmon runs. Historically, spring-run Chinook salmon were temporally isolated from winter-run, and largely isolated in both time and space from the fall-run. Much of this historical spatial-temporal integrity has disappeared, resulting in intermixed life history traits in many remaining habitats. Consequently, the present self-sustaining, persistent populations of spring-run Chinook salmon in the upper Sacramento, lower Yuba, and lower Feather rivers may be hybridized to some degree with fall-run Chinook salmon (YCWA et. al 2007).

Englebright Dam is a complete migration barrier to anadromous fish, precluding migration of Chinook salmon to historical holding and spawning areas upstream of the dam. Consequently, both fall-run and spring-run Chinook salmon are restricted to areas below the dam. Because the spawn timing overlaps between the two runs and they potentially interbreed, genetic swamping of the relatively smaller numbers of spring-run Chinook salmon by more abundant fall-run fish could occur (DWR and PG&E 2010).

The presence of Englebright Dam has necessitated that spring-run Chinook salmon spawn in areas that were believed to formerly represent fall-run Chinook salmon spawning areas. Although the lower Yuba River continues to support a persistent population of spring-run Chinook salmon that now are restricted to spawning downstream of Englebright Dam, the genetic integrity of the fish expressing the phenotypic characteristics of spring-run Chinook salmon is presently uncertain. For example, CDFG (1998) suggests that spring-run populations may be hybridized to some degree with fall-run populations due to lack of spatial separation of spawning habitat for the two runs of Chinook salmon in the lower Yuba River.

In the report titled *Salmonid Hatchery Inventory and Effects Evaluation* (NMFS 2004b), through an analysis of Yuba River Chinook salmon tissues, NMFS genetically linked the spring-run and fall-run populations, which exhibits a merged run timing similar to that found in the Feather River.

In conclusion, available information indicates that: 1) the phenotypic spring-run Chinook salmon in the lower Yuba River actually represents hybridization between spring- and fall-run Chinook salmon in the lower Yuba River, and hybridization with Feather River stocks including the FRFH spring-run Chinook salmon stock, which itself represents a hybridization between Feather River fall- and spring-run Chinook salmon populations; and 2) straying from FRFH origin "spring-run" Chinook salmon into the lower Yuba River occurs, and that this rate of straying is associated with the relative proportion of lower Yuba River flows and water temperatures to lower Feather River flows and water temperatures ("attraction flows and water temperatures"); and 3) the FRFH spring-run Chinook salmon is included in the ESU, in part because of the important role this stock may play in the recovery of spring-run Chinook salmon in the Feather River Basin, including the Yuba River (70 FR 37160). Although straying of FRFH "spring-run" Chinook salmon into the Yuba River has oftentimes been suggested to represent an adverse impact on Yuba River spring-run Chinook salmon stocks, it is questionable whether the phenotypic spring-run Chinook salmon in the lower Yuba River represents an independent population. The RMT (2013a) recently reported that data obtained through the course of implementing the RMT's M&E Program demonstrate that phenotypically "spring-running" Chinook salmon in the lower Yuba River do not represent an independent population – rather, they represent an introgressive hybridization of the larger Feather-Yuba river regional population.

Hatchery effects including genetic considerations and straying into the river represent a high stressor to lower Yuba River spring-run Chinook salmon.

5.1.6.2.5 Narrows 2 Operations

During 2013, YCWA conducted an assessment of the relationship between shutdowns of the Narrows 2 Powerhouse Partial Bypass (Partial Bypass) and adult fish stranding. Assessments occurred in proximity to the Narrows 2 Powerhouse. While the study examined stranding of all fish species, it focused on spring-run Chinook salmon, steelhead, and fall-run Chinook salmon. The relationship between shutdowns of the Partial Bypass and fish stranding was assessed by conducting fish stranding surveys to document the occurrence and condition of any fish found stranded after shutdowns of the Partial Bypass, visual observation events (or visual observations) of fish from the Narrows 2 Powerhouse deck before and after shutdowns of the Partial Bypass, summarizing historical and current operations of the Partial Bypass, and summarizing incidental observations.

The Partial Bypass consists of a pipe off the Narrows 2 Powerhouse turbine spiral casing, which discharges water into the Yuba River through a 36-in valve located on the downstream face of the powerhouse above the draft tube outlet. The Partial Bypass does not have a gage to directly measure flow through the bypass. YCWA can estimate 15-minute flow through the Partial Bypass using: 1) data from an acoustic velocity meter (AVM) attached to the Narrows 2 Penstock upstream from a bifurcation in the penstock; 2) operator logs of when the Partial Bypass is opened and closed); and 3) records of Narrows 2 Powerhouse generation. Typically, YCWA does not operate the Partial Bypass when the Narrows 2 Powerhouse turbine is in normal operation, so if the Narrows 2 Powerhouse is not generating and the Narrows 2 Full Bypass (Full Bypass) is not open, flow as measured at the AVM equals the flow through the Partial Bypass.

From October 1, 2006 (the Narrows 2 Full Bypass went into operation in January 2007) to December 15, 2013, the Partial Bypass was used 23 times, ranging from less than 1 day up to 37 days of continual use. The Partial Bypass was used most often in January, February, and September. Discharge from the Partial Bypass was normally less than 230 cfs, but was as high as 612 cfs.

The existing FERC license, and other permits and licenses, do not include any restrictions, including ramping, regarding how YCWA operates the Partial Bypass, as long as instream flow and flow fluctuation requirements of the FERC license are met.

Over the course of the study period, operational changes that led to shutdowns of the Partial Bypass occurred twice, once on September 8, 2013 and again on October 7, 2013. Both involved transfers of flows from the Partial Bypass to the Full Bypass. During both events, the Narrows 1 Powerhouse operated at approximately 680 cfs. The operational conditions at the Partial Bypass varied from about 30 cfs during Event 1, to just over 200 cfs during Event 2. Fish stranding surveys were conducted immediately after operation of the Partial Bypass ceased. Surveys were conducted along the right bank as oriented downstream. No fish carcasses or stranded live fish were observed during the field surveys following operational changes.

During the September 8, 2013 event, visual observations events from the powerhouse deck resulted in a total of 111 fish observations, of which 99 were of Chinook salmon and 12 were of fish that could not be identified. All of the fish observations occurred after the Partial Bypass was shut down. Observations of fish occurred as close as 15 ft and as far as 170 ft from the Narrows 2 Powerhouse, although the majority of observations consisted of fish circling between 50 and 150 ft downstream of the powerhouse. Observations during the October 7, 2013 event resulted in a total of 30 observations consisting of 20 Chinook salmon and 10 fish that could not be identified. All but one of the observations occurred after an operational event.

The majority of fish observations occurred within 50 ft of the Narrows 2 Powerhouse, as fish swam into or out of the Full Bypass Pool, although observations were made as far as 250 ft downstream of the Narrows 2 Powerhouse.

YCWA is aware of five salmon observations that may be related to stranding in the vicinity of the Narrows 2 Development facilities along the lower Yuba River. Four incidental observations of apparent strandings were recorded during data collection activities for YCWA's Study 7.11, *Fish Behavior and Hydraulics Near Narrows 2 Powerhouse*. Two occurred prior to initiation of Study 7.13 and included an observation by YCWA operators on October 23, 2012 of a fish carcass on the bank near the pool at the base of the Full Bypass and an observation by Relicensing Participants on October 25, 2012 of a fish carcass on the bank near the Partial Bypass. The other two incidental observations occurred in 2013. The first observation included an observation of a fish carcass near Narrows 2 Powerhouse on October 7, 2013. The second observation included multiple fish in an isolated pool in the channel near Narrows 2 Powerhouse on October 13, 2013. The fifth observation was made during fish stranding monitoring as part of YCWA's Narrows 2 Facilities Prioritized Operations and Monitoring Plan (Prioritized Operations Plan) and Streambed Monitoring Below Englebright Dam Plan (Streambed Monitoring Plan) in October of 2015. These incidents are further described in Section 6.0. Overall, Narrows 2 operations represent a moderate stressor to spring-run Chinook salmon in the lower Yuba River.

5.1.6.2.6 Spawning Habitat Availability

Studies conducted by the RMT demonstrate that the earlier spawning Chinook salmon, putatively spring-run, primarily spawn in the uppermost areas of the lower Yuba River. Studies have demonstrated that extensive amounts of substrate suitable for spawning, in combination with suitable flow conditions during the September through mid-October spring-run Chinook salmon spawning period, provide ample amounts of spawning habitat for spring-run Chinook salmon in the lower Yuba River. The only exception is the uppermost reach (Englebright Dam Reach) where there is a relative paucity of appropriate spawning substrate. However, since 2007 the USACE has been injecting a mixture of coarse sediment in the gravel (2-64 mm) and cobble (64-256 mm) size ranges into the Englebright Dam Reach, as part of their voluntary conservation measures associated with ESA consultations regarding Daguerre Point Dam (see Section 6.0 for additional detail). Overall, spawning habitat availability represents a low stressor to spring-run Chinook salmon in the lower Yuba River.

5.1.6.2.7 Potential Redd Dewatering

As reported by CALFED and YCWA (2005), direct and indirect mortality of eggs and alevins resulting from redd dewatering caused by flow fluctuations are difficult to accurately assess. The magnitude of the impact depends upon a number of factors, including the magnitude and duration of the flow fluctuation event, the extent of water elevation reduction in specific reaches of the river (as affected by local channel morphology), the percentage of redds affected by the water elevation reduction, the length of time that specific redds are dewatered (if dewatered at all) and intra- and inter-specific differences in sensitivity to short-term redd dewatering (Reiser and White 1981, as cited in CALFED and YCWA 2005). Although the magnitude of the impact is uncertain, mortality of eggs and alevins may occur when redds are completely dewatered, thereby exposing eggs and alevins to air, or when gravel flow-through is substantially reduced, thereby reducing the supply of oxygen to incubating embryos and removal of waste metabolites (CALFED and YCWA 2005).

Flow reductions resulting from normal maintenance and emergency operations of the Narrows 1 and 2 powerhouses have previously been implicated as potentially affecting redd dewatering (CALFED and YCWA 2005). Maintenance activities at Narrows 2 Powerhouse include generator brush replacement, requiring a six-hour shutdown two to three times per year, and annual maintenance, typically requiring a two- to three-week shutdown, but can be longer if major maintenance is required. Since 1991, YCWA has scheduled annual maintenance activities during periods when the potential for redd dewatering and fish stranding is believed to be the lowest (i.e., late August to mid-September), as determined by discussions at Lower Yuba River Fisheries Technical Working Group (Working Group) meetings and redd and fish stranding surveys (CALFED and YCWA 2005; NMFS 2005a). In addition, flow changes are, to some extent, attenuated with increasing distance downstream of Englebright Dam, due to channel configuration, flow hydraulics, tributary inflows and other related factors. The large Narrows Pool, just below Englebright Dam, also naturally attenuates flow fluctuations. Additionally, the majority of the lower Yuba River’s bed and banks are formed by cobble that was washed downstream from hydraulic gold mining in the mid-1800s. These cobble banks have a minor water storage capacity, which releases some water when the river’s water surface elevation drops, and absorbs some water when the river’s water surface elevation increases (CALFED and YCWA 2005).

As stated by NMFS (2005a), construction of the Narrows 2 Full Bypass would “*minimize the possibility that emergencies or other events requiring that Narrows 2 Powerhouse be taken offline cause significant flow fluctuations in the lower Yuba River, and thereby minimize the possibility that such fluctuations would strand juvenile Central Valley spring-run Chinook* *salmon and Central Valley steelhead or dewater redds of those species.*" The Full Bypass became operational in January 2007. Additional discussion regarding redd dewatering for spring-run Chinook salmon (as well as steelhead) is provided in Section 6.0. Overall, redd dewatering represents a low stressor to spring-run Chinook salmon in the lower Yuba River.

5.1.6.2.8 Physical Habitat Alteration (Including Waterway 13)

According to NMFS (2009a), the stressor associated with physical habitat alteration specifically addressed the issue of return flows and attraction of adult anadromous salmonids into the Yuba Goldfields through Waterway 13, and Lake Wildwood operations. Various efforts have been undertaken to prevent anadromous salmonids from entering the Goldfields via Waterway 13. In May 2005, heavy rains and subsequent flooding breached the structure at the east (upstream facing) end. Subsequently, funded by USFWS, the earthen "plug" was replaced with a "leaky-dike" barrier intended to serve as an exclusion device for upstream migrating adult salmonids (AFRP 2010). Although Waterway 13 is located on USACE property, the USACE does not have any O&M responsibilities for the earthen "plug" and Waterway 13, nor has it issued any permits or licenses for it. YCWA also has no O&M responsibilities for Waterway 13. Nonetheless, until a more permanent solution is implemented, ongoing issues associated with attraction of upstream migrating adult salmonids into Waterway 13 are considered to remain a stressor to spring-run Chinook salmon.

In addition to Waterway 13 issues, physical habitat alternation stressors include Lake Wildwood operations and resultant Deer Creek flow fluctuations (according to the SWRCB's Revised Decision 1644, Lake Wildwood is operated by the Lake Wildwood Association – a gated community in Penn Valley, California). This stressor refers to the potential for stranding or isolation events to occur in Deer Creek, near its confluence with the lower Yuba River. Observational evidence suggests that, in the past, adult Chinook salmon entered Deer Creek during relatively high flow periods, presumably for holding or spawning purposes, only to subsequently become stranded in the creek when flows receded due to changes in Lake Wildwood operations. Stranding may delay or prevent adult Chinook salmon from spawning, or cause decreased spawning success due to increased energy expenditure or stress due to delayed spawning (CALFED and YCWA 2005).

From 2011-2013, the Sierra Streams Institute (SSI) implemented three gravel augmentation projects to increase the availability of spawning habitat in Deer Creek, which is located on a tributary to the lower Yuba River (SSI 2015). A total of 500 cubic yards of spawning material was placed into Deer Creek during September 2012 and 2013. Chinook salmon redd surveys were conducted after the initial placement to document the number and characteristics of salmon redds created in Deer Creek during the 2012 spawning season. On November 27, 2012, more than 51 salmon redds were observed in Deer Creek, compared to 15 redds in 2011, and 9 redds in 2003 (SSI 2013). Approximately 75 percent of spawning activity during 2012 occurred in the newly created spawning areas, with the remaining spawning activity occurring in locations where spawning was observed in 2011. Gravel transport also was monitored to understand the effects of higher stream flows on gravel movement, and to evaluate transport of spawning gravels in Deer Creek. Tracer gravel surveys were conducted during February, March, and April 2013. Based on these and other visual observations of substrate deposition in Deer Creek, SSI

(2013) report that it is likely that some of the placed gravels remain in Deer Creek providing spawning habitat, and that some of the gravels were mobilized downstream into the Yuba River to provide habitat for anadromous salmonids. SSI (2015) reports that the gravel augmentation projects in Deer Creek have resulted in over a 500% increase in salmon redds observed in Deer Creek during 2013. SSI is also working with Lake Wildwood on a long-term effort to use the coarse dredged material from the Lake Wildwood reservoir inlet to provide spawning materials to Deer Creek in perpetuity (SSI 2015). Overall, physical habitat alteration as described herein represents a low to moderate stressor to spring-run Chinook salmon in the lower Yuba River.

5.1.6.2.9 Fry and Juvenile Rearing Physical Habitat Structure

Fry and juvenile salmonid rearing physical habitat structure pertains to habitat complexity and diversity. The concepts of habitat complexity and diversity pertinent to the lower Yuba River were described by CALFED and YCWA (2005), as discussed below.

Habitat complexity and diversity refer to the quality of instream physical habitat including, but not necessarily limited to, the following physical habitat characteristics:

- Escape cover
- Feeding cover
- Allochthonous material contribution
- Alternating point-bar sequences
- Pool-to-riffle ratios
- Sinuosity
- Instream object cover
- Overhanging riparian vegetation

The physical structure of rivers plays a significant role in determining the suitability of aquatic habitats for juvenile salmonids, as well as for other organisms upon which salmonids depend for food. These structural elements are created through complex interactions among natural geomorphic features, the power of flowing water, sediment delivery and movement, and riparian vegetation, which provides bank stability and inputs of large woody debris (Spence et al. 1996). The geomorphic conditions caused by hydraulic and dredge mining since the mid-1800s, and the construction of Englebright Dam, which affects the transport of nutrients, fine and coarse sediments and, to a lesser degree, woody material from upstream sources to the lower river, continue to limit habitat complexity and diversity in the lower Yuba River.

LWM creates both micro- and macro-habitat heterogeneity by forming pools, back eddies and side channels and by creating channel sinuosity and hydraulic complexity. This habitat complexity provides juvenile salmonids numerous refugia from predators and water velocity, and provides efficient locations from which to feed. LWM also functions to retain coarse sediments and organic matter in addition to providing substrate for numerous aquatic invertebrates (Spence et al. 1996).

In the lower Yuba River, mature riparian vegetation is scattered intermittently, leaving much of the banks devoid of LWM and unshaded – affecting components that are essential to the health and survival of the freshwater lifestages of salmonids (NMFS 2002). Although the ability of the lower Yuba River to support riparian vegetation has been substantially reduced by the historic impacts from mining activities, the dynamic nature of the river channel results in periodic creation of high-value shaded riverine aquatic (SRA) cover for fish and wildlife (Beak 1989).

Other important components of habitat structure at the micro-scale include large boulders, coarse substrate, undercut banks and overhanging vegetation. These habitat elements offer juvenile salmonids concealment from predators, shelter from fast current, feeding stations and nutrient inputs. At the macro-scale, streams and rivers with high channel sinuosity, multiple channels and sloughs, beaver impoundments or backwaters typically provide high-quality rearing and refugia habitats (Spence et al. 1996). The lower Yuba River can be generally characterized as lacking an abundance of such features. Consequently, juvenile rearing physical habitat structure represents a high stressor to spring-run Chinook salmon in the lower Yuba River.

5.1.6.2.10 Entrainment

According to NMFS (2009a), entrainment of juvenile salmonids remains a stressor in the lower Yuba River. Entrainment represents a suite of potential negative impacts to juvenile fish that may occur while, or after, the fish encounter a diversion facility in operation. For instance, entrainment impacts may include the non-volitional recruitment of juveniles past a diversion facility and/or screening structure, or impingement upon diversion screens and physical damage to fish caused by diversion activities. It has been suggested that as juvenile salmonids pass Daguerre Point Dam, physical injury may occur as they pass over the dam or through its fish ladders (SWRI 2002).

There are three water diversions associated with Daguerre Point Dam, which utilize the elevated head[1] created by the dam, or the influence of the dam in the prevention of additional river channel incision, to gravity-feed their canals. The three diversions are the Hallwood-Cordua diversion, the South Yuba/Brophy diversion, and the BVID diversion. Water diversions in the lower Yuba River generally begin in the early spring and extend through the fall. In 1999, a new state-of-the-art fish screen was installed at the BVID diversion point that meets NMFS and Cal Fish and Wildlife screening criteria, and entrainment is no longer considered to be an issue at that facility. Potential threats to juvenile salmonids occur at the Hallwood-Cordua and South Yuba/Brophy diversions (NMFS 2009a). The relatively recent fish screen constructed at the Hallwood-Cordua diversion is considered a notable improvement over the previous design, and is believed to reduce the amount of fry and juvenile entrainment at the diversion. The new diversion fish screen is believed to reduce loss rates of emigrating fall-run Chinook salmon at this location. However, predation losses of emigrating fry and juvenile fall-run Chinook salmon may remain a limiting factor at this location. In addition, the configuration of the current return pipe and flows though the pipe may also be a limiting factor (CALFED and YCWA 2005).

[1] The "elevated head" at Daguerre Point Dam is created by the hydraulic conditions associated with water being impounded behind (i.e., upstream) of the dam. USACE has no control over the in-river flows, and has no obligation to provide a "head" for local water users in the vicinity of Daguerre Point Dam.

As previously described, the South Yuba/Brophy system diverts water through an excavated channel from the south bank of the lower Yuba River in the vicinity of Daguerre Point Dam. The water is then subsequently diverted through a porous rock dike that is intended to exclude fish. The current design of this rock structure does not meet current NMFS or Cal Fish and Wildlife juvenile fish screen criteria (SWRI 2002), and additional issues regarding predation in the diversion channel and the rate of water bypassing the rock gabion and returning to the lower Yuba River through the diversion channel have been raised as potential stressors. YCWA is in the process of considering a major rehabilitation of the South Canal Diversion, which would restore diversion capacity and reliability, and improve protection of anadromous salmonids in the lower Yuba River at the South Canal Diversion. A Notice of Preparation for the proposed project was issued in early 2016. More recently, environmental review for YCWA's South Canal Diversion Fish Screen Project was approved for funding through Cal Fish and Wildlife's Proposition 1 Watershed Restoration Grant Program, and an EIR for the project is being developed.

Overall, the potential for entrainment and associated effects (i.e., focused predation) at Daguerre Point Dam represent a relatively low stressor to spring-run Chinook salmon, primarily due to the asynchronous timing between diversions and juvenile outmigrant lifestage periodicity.

5.1.6.2.11 Predation

Predation can occur in three forms: 1) natural; 2) predation resulting from a relative increase in predator habitat and opportunity near major structures and diversions; and 3) predation resulting from minimal escape cover and habitat complexity for prey species (CALFED and YCWA 2005). For the purpose of stressor identification in this Applicant-Prepared Draft BA, predation includes the predation associated with increases in predator habitat and predation opportunities for piscivorous species created by major structures and diversions, and predation resulting from limited amounts of prey escape cover in the lower Yuba River.

The extent of predation on juvenile Chinook salmon in the lower Yuba River is not well documented (NMFS 2009a). Although predation is a natural component of salmonid ecology, the rate of predation of salmonids in the lower Yuba River has potentially increased through the introduction of non-native predatory species such as striped bass, largemouth bass and American shad and through the alteration of natural flow regimes and the development of structures that attract predators (NMFS 2009a).

Predatory fish are known to congregate around structures in the water including dams, diversions and bridges, where their foraging efficiency is improved by shadows, turbulence and boundary edges (CDFG 1998). Thus, juvenile salmonids can also be adversely affected by Daguerre Point Dam on their downstream migration. Daguerre Point Dam creates a large plunge pool at its base, which provides ambush habitat for predatory fish in an area where emigrating juvenile salmonids may be disoriented after plunging over the face of the dam into the deep pool below (NMFS 2002). The introduced predatory striped bass and American shad have been observed in this pool (CALFED and YCWA 2005). In addition to introduced predatory species, several native fish species also prey on juvenile salmonids in the lower Yuba River, including Sacramento pikeminnow, hardhead and large juvenile and adult rainbow trout/steelhead (CALFED and

YCWA 2005). It has been suggested that the rate of predation of juvenile salmonids passing over dams in general, and Daguerre Point Dam in particular, may be unnaturally high (NMFS 2007), although specific studies addressing this suggestion have not been conducted.

In addition to the suggestion of increased rates of predation resulting from disorientation of juveniles passing over Daguerre Point Dam into the downstream plunge pool, it also has been suggested that unnaturally high predation rates may also occur in the diversion channel associated with the South Yuba/Brophy diversion (NMFS 2007). Other structure-related predation issues include the potential for increased rates of predation of juvenile salmonids: 1) in the entryway of the Hallwood-Cordua diversion canal upstream of the fish screen; and 2) at the point of return of fish from the bypass pipe of the Hallwood-Cordua diversion canal into the lower Yuba River. Overall, in consideration of potential elevated predation rates at Daguerre Point Dam and particularly in the river downstream of Daguerre Point Dam, predation in the lower Yuba River is considered to be a moderate to high stressor.

5.1.6.2.12 Riparian Habitat and Instream Cover

Riparian Vegetation

As stated in CALFED and YCWA (2005), riparian vegetation, an important habitat component for anadromous fish, is known to provide: 1) bank stabilization and sediment load reduction; 2) shade that results in lower instream water temperatures; 3) overhead cover; 4) streamside habitat for aquatic and terrestrial insects, which are important food sources for rearing juvenile fishes; 5) a source of instream cover in the form of woody material; and 6) allochthonous nutrient input.

SRA cover generally occurs in the lower Yuba River as scattered, short strips of low-growing woody species (e.g., *Salix sp*.) adjacent to the shoreline. Beak (1989) reported that the most extensive and continuous segments of SRA cover occur along bars where [then] recent channel migrations or avulsions had cut new channels through relatively large, dense stands of riparian vegetation. SRA cover consists of instream object cover and overhanging cover. Instream object cover provides structure, which promotes hydraulic complexity, diversity and microhabitats for juvenile salmonids, as well as escape cover from predators. The extent and quality of suitable rearing habitat and cover, including SRA, generally has a strong effect on juvenile salmonid production in rivers (Healey 1991 as cited in CALFED and YCWA 2005).

Since completion of New Bullards Bar Reservoir, the riparian community (in the lower Yuba River) has expanded under summer and fall streamflow conditions that have generally been higher than those that previously occurred (SWRCB 2003). However, the riparian habitat is not pristine. NMFS (2005a) reports …"*The deposition of hydraulic mining debris, subsequent dredge mining, and loss/confinement of the active river corridor and floodplain of the lower Yuba River which started in the mid-1800's and continues to a lesser extent today, has eliminated much of the riparian vegetation along the lower Yuba River. In addition, the large quantities of cobble and gravel that remained generally provided poor conditions for re-establishment and growth of riparian vegetation. Construction of Englebright Dam also inhibited regeneration of riparian vegetation by preventing the transport of any new fine sediment, woody debris, and nutrients from upstream sources to the lower river. Subsequently,* *mature riparian vegetation is sparse and intermittent along the lower Yuba River, leaving much of the bank areas unshaded and lacking in large woody debris. This loss of riparian cover has greatly diminished the value of the habitat in this area*."

Where hydrologic conditions are supportive, riparian and wetland vegetative communities are found adjacent to the lower Yuba River and on the river sides of retaining levees. These communities are dynamic and have changed over the years as the river meanders. The plant communities along the river are a combination of remnant Central Valley riparian forests, foothill oak/pine woodlands, agricultural grasslands, and orchards (Beak 1989).

According to CALFED and YCWA (2005), the lower Yuba River, especially in the vicinity of Daguerre Point Dam and the Yuba Goldfields, is largely devoid of sufficient riparian vegetation to derive the benefits (to anadromous salmonids) discussed above (Figure 5.1-9).

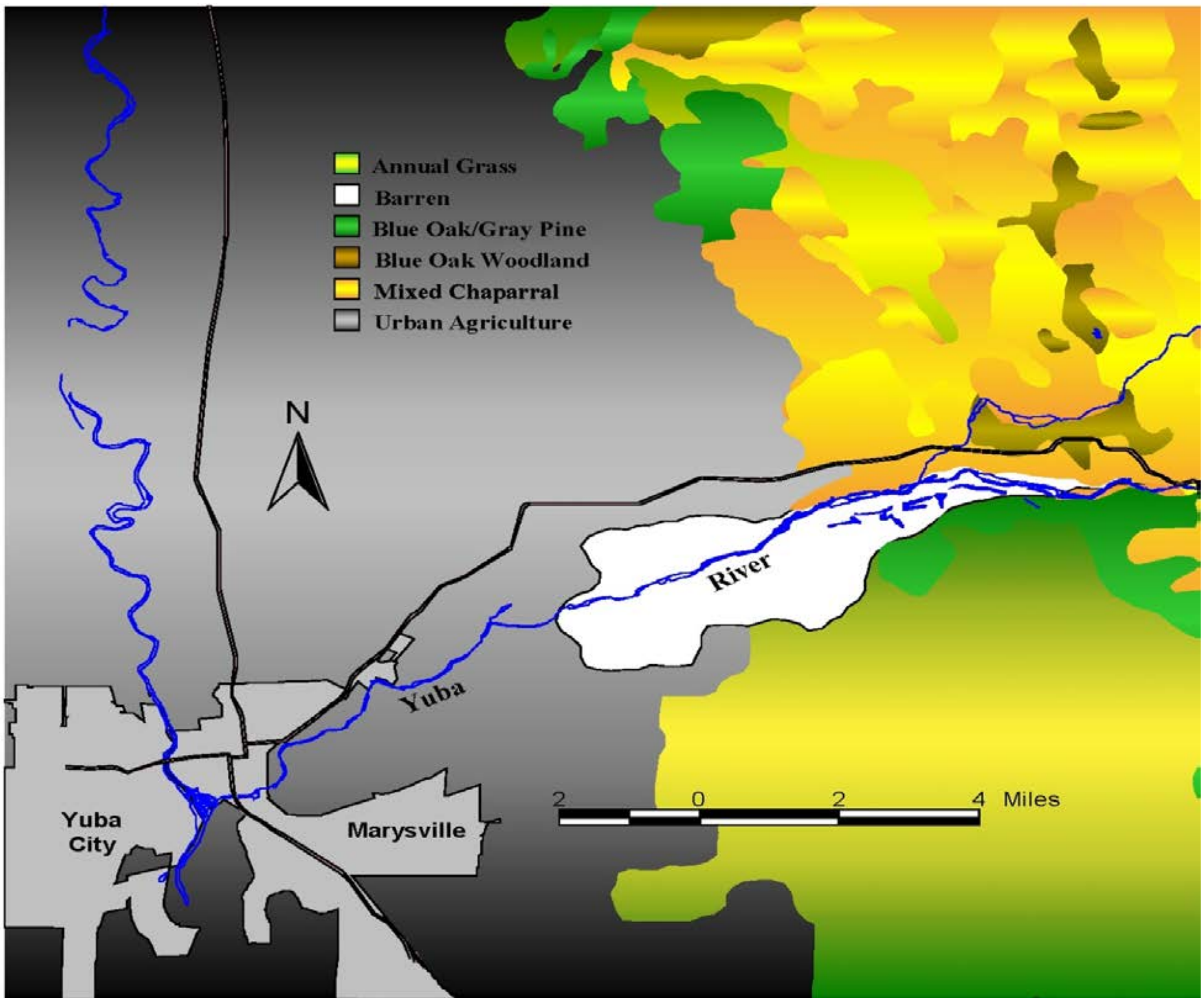

Figure 5.1-9. Vegetation communities in the lower Yuba River vicinity.
Source: CALFED and YCWA 2005

In 2012, YCWA conducted a riparian habitat study in the Yuba River from Englebright Dam to the confluence with the Feather River (see Technical Memorandum 6-2, *Riparian Habitat Downstream of Englebright Dam*, which can be found on FERC's eLibrary as referenced by the FERC accession number provided in Table E6-2 of Appendix E6, of YCWA's Amended FLA). Field efforts included descriptive observations of woody and riparian vegetation, cottonwood inventory and coring, and an LWM survey. The study was performed by establishing eight LWM study sites and seven riparian habitat study sites. One LWM study site was established within each of eight distinct reaches (i.e., Marysville, Hallwood, Daguerre Point Dam, Dry Creek, Parks Bar, Timbuctoo Bend, Narrows, and Englebright Dam). Riparian habitat sites were established in the same locations as the LWM study sites, with the exception of the Marysville study site. Riparian information regarding the Marysville Reach was developed, but no analysis was performed because of backwater effects of the Feather River.

The RMT contracted Watershed Sciences Inc. to use existing LiDAR to produce a map of riparian vegetation stands by type. The resulting data was subject to a field validation and briefly summarized in WSI (2010) and the data were also utilized in YCWA's Technical Memorandum 6-2.

Based on field observations, Technical Memorandum 6-2 reported that all reaches supported woody species in various lifestages – mature trees, recruits, and seedlings were observed within all reaches. Where individuals or groups of trees were less vigorous, beaver (*Castor canadensis*) activity was the main cause, although some trees in the Marysville Reach appeared to be damaged by human camping.

The structure and composition of riparian vegetation was largely associated with four landforms. Cobble-dominated banks primarily supported bands of willow shrubs with scattered hardwood trees. Areas with saturated soils or sands supported the most complex riparian areas and tended to be associated with backwater ponds. Scarps and levees supported lines of mature cottonwood and other hardwood species, typically with a simple understory of Himalayan blackberry or blue elderberry shrubs. Bedrock dominated reaches had limited riparian complexity and supported mostly willow shrubs and cottonwoods.

Based on analysis of the mapping data, RMT (2013a) reported that the majority of the woody species present in the river valley include, in order of most to least number of individuals: various willow species (*Salix* sp. and *Cephalanthus occidentalis*); Fremont cottonwood (*Populus fremontii*) (i.e., cottonwoods); blue elderberry (*Sambucus nigra* ssp. *caerulea*); black walnut (*Juglans hindsii*); Western sycamore (*Platanus racemosa*); Oregon ash (*Fraxinus latifolia*); white alder (*Alnus rhombifolia*); tree of heaven (*Ailanthus altissima*); and grey pine (*Pinus sabiniana*). Willow species could not be differentiated by species using remote sensing information. Willow on the lower Yuba River are dominated by dusky sandbar willow (*Salix melanopsis*) and narrow leaf willow (*Salix exigua*), and relative dominance of the two species shifts respectively in the downstream direction (WSI 2010). Other species occurring are arundo willow (*Salix lasiolepsis*), Goodings willow (*Salix goodingii*) and red willow (*Salix laevigata*). Goodings and red willow comprise 6.4 percent of the willow according to a limited field validation survey (WSI 2010).

Cottonwoods are one of the most abundant woody species in the study area of Technical Memorandum 6-2, and the most likely source of locally-derived large instream woody material (IWM) due to rapid growth rates and size of individual stems commonly exceeding 2 ft in diameter and 50 ft in length. Cottonwoods exist in all lifestages including as mature trees, recruits, or saplings, and as seedlings. Cottonwoods are more abundant in downstream areas of the study area relative to upstream. Cottonwoods are distributed laterally across the valley floor. Of the estimated 18,540 cottonwood individuals/stands, 12 percent are within the bankfull channel (flows of 5,000 cfs or less), and 39 percent are within the floodway inundation zone (flows between 5,000 and 21,100 cfs). However, recruitment patterns of cottonwood have not been analyzed with respect to time or with any more detail regarding channel location (see Technical Memorandum 6-2).

A total of 97 cottonwood trees were cored to estimate age. Age estimates ranged from 11 to 87 years. The cottonwood tree age analysis resulted in age estimates that place the year of establishment for trees in a range of years from ±7 to 16 years, which is too wide to allow for linking the establishment of trees to any year's specific hydrologic conditions (YCWA 2013).

YCWA conducted a historical aerial photograph analysis to describe changes over time to total vegetation delineated within the valley walls, riparian vegetation delineated within 50 ft of the active river channel,[2] and channel alignment (Technical Memorandum 6-2). To determine the cumulative change over time[3] in total vegetative cover and riparian vegetation cover for the Marysville, Timbuctoo Bend, Narrows, and Englebright Dam study sites, YCWA compared the aerial photographs from 1937 and 2010.

Cumulative changes in the Englebright Dam and Narrows study sites showed an overall decrease in vegetative cover. For the remaining study sites, including Marysville, Hallwood, Daguerre Point Dam, Dry Creek, Parks Bar, and Timbuctoo Bend study sites, the cumulative change in vegetative cover increased. The least amount of vegetation change over time was observed in the Englebright Dam, Narrows and Marysville sites. The Dry Creek, Daguerre Point Dam and Hallwood sites had the greatest vegetated area, and YCWA identified those sites as the most dynamic (i.e., both decreased in vegetative cover through 1970 and then increased through 2010).

Cumulative changes in riparian vegetation cover in the Englebright Dam and Narrows study sites decreased with very little detectable change for the Narrows study site. For the remaining study sites, the cumulative change in riparian vegetation cover increased. The observed changes for the Englebright Dam, Narrows and Marysville study sites were very small. For the Dry Creek and Parks Bar study sites, the greatest changes were observed, with dramatic increases in riparian vegetation cover. The magnitude of change of riparian vegetation cover between photoset years (in a stepwise comparison) was greater than that seen in the cumulative total riparian vegetation cover change over the entire period examined.

[2] Total vegetation is inclusive of riparian vegetation.
[3] Cumulative change describes the changes to observable area for either total vegetation or riparian vegetation from the earliest photo date to the most recent photo date.

Instream Woody Material

IWM provides escape cover and relief from high current velocities for juvenile salmonids and other fishes. LWM also contributes to the contribution of invertebrate food sources, and micro-habitat complexity for juvenile salmonids (NMFS 2007). Snorkeling observations in the lower Yuba River have indicated that juvenile Chinook salmon had a strong preference for near-shore habitats with IWM (Jones & Stokes 1992).

There is currently a lack of consensus regarding the amount of IWM occurring in the lower Yuba River (USACE 2012b). It has been suggested (CALFED and YCWA 2005) that the presence of Englebright Dam has resulted in decreased recruitment of LWM to the lower Yuba River, although no surveys or studies were cited to support these statements. Some woody material may not reach the lower Yuba River due to collecting on the shoreline and sinking in Englebright Reservoir (USACE 2012b). However, Englebright Dam does not functionally block woody material from reaching the lower Yuba River because there is no woody material removal program implemented for Englebright Reservoir, and accumulated woody material therefore spills over the dam during uncontrolled flood events (R. Olsen, USACE, pers. comm. 2011, as cited in USACE 2012b).

About 8.7 mi of the Yuba River downstream of Englebright Dam, distributed among study sites per reach, were surveyed and evaluated for pieces of wood (YCWA 2013). The number of pieces of wood was relatively similar above and below Daguerre Point Dam (i.e., about 5,100 and 5,750 pieces, respectively). Woody material was generally found in bands of willow shrubs near the wetted edge, dispersed across open cobble bars, and stranded above normal high-flow indicators. Most of the woody material was diffuse and located on floodplains and high floodplains, with only about a quarter of the material in heavy concentrations (YCWA 2013).

Most (77-96%) pieces of wood found in each reach were smaller than 25 ft in length and smaller than 24 in in diameter, which is the definition of LWM in Technical Memorandum 6-2. These pieces would be typically floated by flood flows and trapped within willows and alders above the 21,100 cfs line, which is defined as the flow delineating the floodway boundary (YCWA 2013).

IWM was not evenly distributed throughout the reaches. For the smaller size classes (i.e., shorter than 50 ft, less than 24 in in diameter), the greatest abundance of pieces was found in the Hallwood or Daguerre Point Dam reaches, with lower abundances above and below these reaches (YCWA 2013).

The largest size classes of LWM (i.e., longer than 50 ft and greater than 24 in in diameter) were rare or uncommon (i.e., fewer than 20 pieces total) with no discernible distribution. Pieces of this larger size class were counted as “key pieces”, as were any pieces exceeding 25 in in diameter and 25 ft in length and showing any morphological influence (e.g., trapping sediment or altering flow patterns). A total of 15 key pieces of LWM were found in all study sites, including six in the Marysville study site. Few of the key pieces were found in the active channel or exhibiting channel forming processes (YCWA 2013).

Overall, the relative abundance of riparian vegetation and LWM in the lower Yuba River is considered to be a moderate to high stressor in the lower Yuba River.

5.1.6.2.13 Natural River Morphology and Function

According to NMFS (2014a), "Loss of Natural River Morphology and Function" is the result of river channelization and confinement, which leads to a decrease in riverine habitat complexity, and thus, a decrease in the quantity and quality of juvenile rearing habitat. Additionally, this primary stressor category includes the effect that dams have on the aquatic invertebrate species composition and distribution, which may have an effect on the quality and quantity of food resources available to juvenile salmonids.

According to NMFS (2014a), attenuated peak flows and controlled flow regimes have altered the lower Yuba River's geomorphology and have affected the natural meandering of the river downstream of Englebright Dam. However, alteration of river morphology and function has been very substantively affected by hydraulic mining legacy and confinement of the river channel from dredger tailings and gravel berm deposits.

As reported by RMT (2013a), preliminary evaluation of available data collected to date related to Yuba River fluvial geomorphology indicates that the Yuba River downstream of Englebright Dam has complex river morphological characteristics. Evaluation of the morphological units (MUs) in the Yuba River as part of the spatial structure analyses indicates that, in general, the sequence and organization of MUs is non-random, indicating that the channel has been self-sustaining of sufficient duration to establish an ordered spatial structure (RMT 2013a).

The Yuba River downstream of Englebright Dam exhibits lateral variability in its form-process associations (RMT 2013a). In the Yuba River, MU organization highlights the complexity of the channel geomorphology, as well as the complex and diverse suite of MUs. The complexity in the landforms creates diversity in the flow hydraulics which, in turn, contributes to a diversity of habitat types available for all riverine lifestages of anadromous salmonids in the Yuba River downstream of Englebright Dam (RMT 2013a).

In the lower Yuba River, anadromous salmonids spawn in mean substrate sizes ranging from about 50 to 150 mm, and most of the lower Yuba River from Englebright Dam to the confluence with the Feather River is characterized by average substrate particle sizes within this size range (RMT 2013a). The exceptions are sand/silt areas near the confluence with the Feather River, and the boulder/bedrock regions in the upper sections of Timbuctoo Bend and most of the Englebright Dam Reach. However, gravel augmentation funded by USACE in the Englebright Dam Reach over the past several years has spurred spawning activity and Chinook salmon redd construction in this reach. The net result is an increase in the spatial distribution of spawning habitat availability in the river, particularly for early spawning (presumably spring-run) Chinook salmon (RMT 2013a).

The loss of natural river morphology and function represents a high stressor to spring-run Chinook salmon in the lower Yuba River.

5.1.6.2.14 Floodplain Habitat

NMFS (2014a) listed the loss of floodplain habitat in the lower Yuba River as one of the key stressors affecting anadromous salmonids (including spring-run Chinook salmon). NMFS (2009a) stated …"*Historically, the Yuba River was connected to vast floodplains and included a complex network of channels, backwaters and woody material. The legacy of hydraulic and dredger mining is still evident on the lower Yuba River where, for much of the river, dredger piles confine the river to an unnaturally narrow channel. The consequences of this unusual and artificial geomorphic condition include reduced floodplain and riparian habitat and resultant limitations in fish habitat, particularly for rearing juvenile salmonids.*"

NMFS (2014a) further stated that in the lower Yuba River, controlled flows and decreases in peak flows has reduced the frequency of floodplain inundation resulting in a separation of the river channel from its natural floodplain. Within the Yuba Goldfields area (RM 8–14), confinement of the river by massive deposits of cobble and gravel derived from hydraulic and dredge mining activities resulted in a relatively simple river corridor dominated by a single main channel and large cobble-dominated bars, with little riparian and floodplain habitat (DWR and PG&E 2010).

Loss of off-channel habitats such as floodplains, riparian, and wetland habitats has substantially reduced the productive capacity of the Central Valley for many native fish and wildlife species, and evidence is growing that such habitats were once of major importance for the growth and survival of juvenile salmon (Moyle 2002). Observations on the lower Yuba River indicate that remnant side channels and associated riparian vegetation play a similar role by providing flood refugia, protection from predators, and abundant food for young salmonids and other native fishes. These habitats also promote extended rearing and expression of the stream-type rearing characteristic of spring-run Chinook salmon (DWR and PG&E 2010).

As reported by RMT (2013a), despite some flow regulation, the channel and floodplain in the lower Yuba River are highly connected, with floods spilling out onto the floodplain more frequently than commonly occurs for unregulated semiarid rivers. Although some locations exhibit overbank flow below 5,000 cfs while others require somewhat more than that, 5,000 cfs generally represents bankfull flow in the lower Yuba River. In any given year, there is an 82 percent chance the river will spill out of its bankfull channel and a 40 percent chance that the floodway will be fully inundated. These results demonstrate that floodplain inundation occurs with a relatively high frequency in the lower Yuba River compared to other Central Valley streams which, in turn, contributes to a diversity in habitats available for anadromous salmonids (RMT 2013a).

RMT (2013a) conducted a flood-frequency analysis of the annual peak discharges recorded at the USGS stream gage near Marysville (11421000) that showed average annual return periods of 1.25 years and 2.5 years for the bankfull and flood discharges, respectively. Bankfull flows for similar rivers are generally assumed to occur with return periods of 1.5-2 years. The fact that the lower Yuba River is less than this implies that the channel is naturally undersized relative to generalized expectations and flows spill into the floodplain at a more frequent rate (RMT 2013a).

Massive deposits of cobble and gravel derived from hydraulic and dredge mining activities, and dredger gravel deposits and berms and levee construction resulting in a relatively simple river corridor dominated by a single main channel reflect a loss of floodplain habitat, and represent a relatively high stressor to spring-run Chinook salmon in the lower Yuba River.

5.1.6.2.15 Fry Stranding and Juvenile Isolation

In its 2001 Decision (D)-1644, the SWRCB directed YCWA to submit a plan that described the scope and duration of future flow fluctuation studies to verify that Chinook salmon and steelhead redds are being adequately protected from dewatering with implementation of D-1644 criteria (YCWA 1992).

The studies combined habitat mapping, field surveys, and information on the timing and distribution of fry rearing in the Yuba River to evaluate the effectiveness of D-1644 flow fluctuation and reduction criteria in protecting Chinook salmon and steelhead fry. Two studies were conducted and summarized in the 2007 and 2008 *Lower Yuba River Redd Dewatering* and *Fry Stranding Annual Report* (Jones and Stokes 2008; Jones and Stokes 2009) to the SWRCB, and results from an additional study were reported in a progress report in 2010 (ICF Jones and Stokes 2010).

The first survey was conducted on April 5, 2007 to evaluate bar and off-channel stranding of juvenile salmonids associated with a flow reduction of 400 cfs total (1,300-900 cfs) at Smartsville at a ramping rate of 100 cfs per hour. Bar stranding was again evaluated on June 18, 2007 with a temporary flow reduction of 300 cfs total (1,600-1,300 cfs) at a rate of 100 cfs per hour. Snorkel surveys were conducted between Rose Bar, located approximately 2.5 mi downstream of Englebright Dam, and the Highway 20 Bridge, located approximately 5.7 mi downstream of Englebright Dam.

During the April 5, 2007 drawdown, field crews observed eight stranded salmon fry in the interstitial spaces of substrates on bar slopes (perpendicular to shoreline) ranging from 0.5 to 5.5 percent in slope. No stranded fish were observed during surveys conducted on June 18, 2007. The presence of both juvenile Chinook salmon and *O. mykiss* were confirmed in shallow, near-shore areas adjacent to the study sites, suggesting that the risk of bar stranding is greatly reduced by June. Following the April 5, 2007 flow reductions, juvenile salmon were found in 16 of the 24 disconnected off-channel sites. Most of the fish that had become isolated in off-channel sites were 30-50 mm fry. Out of the 16 sites where isolation of fry was observed, 70 percent of the fish were found in the four largest sites, which accounted for nearly 60 percent of the total wetted area that had become disconnected from the main river. These four sites were unique in that they were all associated with man-made features within or adjacent to the main river channel (e.g., diversion channels, ponds and bridge piers) (B. Mitchell, ICF/JSA, pers. comm. 2012).

Another survey was conducted from May 29, 2008 through June 4, 2008 with a scheduled flow reduction of 250 cfs from approximately 1,400 cfs to 1,150 cfs at a rate of about 100 cfs per hour on June 1, 2008. A total of seven stranded trout fry ranging between 30-35 mm were observed in the interstitial spaces of substrates on bar slopes ranging from 2.0 to 5.7 percent in slope.

Juvenile salmon were found isolated in seven of the 12 off-channel sites that had become disconnected from the main river by the June 1, 2008 event. One site accounted for only about 7 percent of the total wetted area that had been disconnected from the main river, but nearly 80 percent of the total number of juvenile salmon that had been isolated by the June 1, 2008 event. A total of 13 steelhead fry were found isolated in 2 of the 12 off-channel sites that had become disconnected from the main river by the June 1, 2008 event. Nearly all of these fish were 30-50 mm fry that had been isolated in a single backwater pool adjacent to the main river in the Timbuctoo Reach (B. Mitchell, ICF/JSA, pers. comm. 2012).

Jones and Stokes (2008) suggested that the preliminary findings indicated that juvenile *O. mykiss* fry may be less vulnerable to off-channel stranding than juvenile Chinook salmon because of their more restricted distribution and inability to access off-channel areas under late spring flow conditions. Long-term monitoring of several isolated off-channel sites confirmed that some sites can support juvenile salmonids for long periods and even produce favorable summer rearing conditions.

A 2010 study was conducted from June 21, 2010 through July 1, 2010, with a scheduled 800 cfs flow reduction between June 28 and June 30 from approximately 4,000 cfs to 3,200 cfs as measured at the Smartsville gage (ramping rate not reported). As reported by ICF Jones and Stokes (2010), fish stranding surveys were conducted on June 21, 22, and 23 to identify potential stranding areas and document habitat conditions and fish presence before the flow reduction, and were repeated on June 29, June 30, and July 1 to document the incidence of fish stranding and habitat conditions after the flow reduction.

After the June flow reduction, a total of six juvenile salmon and 46 juvenile trout were observed in seven of the 26 off-channel sites that had become fully or nearly disconnected (≤0.1 ft deep in the connection channel) from the main river. Most of the stranded fish were juvenile trout 30-70 mm in length that had become isolated in five off-channel sites above Daguerre Point Dam. Below Daguerre Point Dam, observations of stranded fish were limited to six juvenile salmon and two juvenile trout at two study sites (ICF Jones and Stokes 2010).

Hydrologic and operating conditions in January and February 2011 provided the first opportunity to evaluate the effect of a winter flow reduction on the incidence of bar stranding. A series of three successive flow reductions were evaluated. Following a 3-week period of relatively stable flows, flows were reduced from 3,000-2,600 cfs on January 31, 2,600-2,200 cfs on February 7, and 2,200-2,000 cfs on February 11.

The first event was a 400-cfs flow reduction (3,000–2,600 cfs) conducted from 8:00 AM to 10:00 AM at a target rate of 200 cfs per hour on January 31, 2011. This event resulted in a 2.1–2.5 in drop in water surface elevation and a rate of change of 0.6–0.8 in per hour at the three study sites. Field crews searched a total of 764 square feet (ft^2) of dewatered shoreline and found a total of 20 stranded salmon fry (30-40 mm long) and six stranded steelhead (50-90 mm long) (B. Mitchell, ICF/JSA, pers. comm. 2012).

During the second event on February 7, 2011, flows were again reduced by 400 cfs (2,600–2,200 cfs) from 8:00 AM to 10:00 AM, but at a target rate of 100 cfs per hour. This event resulted in a

1.8–2.1 inch drop in water surface elevation and a rate of change of 0.4–0.5 inch per hour at the three study sites. Field crews searched a total of 560 ft^2 of dewatered shoreline and found a total of 10 stranded salmon fry (30-40 mm long) and no steelhead (B. Mitchell, ICF/JSA, pers. comm. 2012).

During the third event on February 11, 2011, flows were reduced by 200 cfs (2,200–2,000 cfs) from 2:00 AM to 4:00 AM at a target rate of 100 cfs per hour. This event resulted in a 0.8–1.3 inch drop in water surface elevation and a rate of change of 0.4–0.7 inch per hour at the three study sites. Field crews searched a total of 248 sq ft of dewatered shoreline and found a total of four stranded salmon fry (30-40 mm long) and no steelhead (B. Mitchell, ICF/JSA, pers. comm. 2012).

Potential fry stranding and juvenile isolation represent a moderate stressor to spring-run Chinook salmon in the lower Yuba River. Refer to Section 6.0 for additional information regarding the potential for juvenile stranding in the Yuba River downstream of Englebright Dam.

5.1.6.2.16 Water Temperature

During November 2010, the RMT prepared a technical memorandum (RMT 2010b) to review the appropriateness of the water temperature regime associated with implementation of the Yuba Accord using previously available data and information. An update to the water temperature suitability evaluation in RMT (2010b) was presented by the RMT in their M&E Program Interim Report (RMT 2013a). Lifestage-specific WTI values were used as evaluation guidelines to assess the suitability of water temperatures in the lower Yuba River for all lifestages of spring-run Chinook salmon.

Water temperature monitoring data in the lower Yuba River for the period extending from October 2006 through October 2012, during which time operations have complied with the Yuba Accord, were evaluated by RMT (2013a). Monitored water temperatures at the representative locations of Smartsville, above Daguerre Point Dam, and Marysville during the period evaluated were always below the upper tolerance WTI values for yearling+ smolt outmigration, juvenile rearing and outmigration, and adult immigration and holding. The upper tolerance spawning and embryo incubation WTI value was never exceeded at Smartsville, which was the only location evaluated for spring-run Chinook salmon spawning and embryo incubation.

The RMT (2013a) concluded that implementation of the Yuba Accord provides a suitable thermal regime for target species (including spring-run Chinook salmon) in the lower Yuba River, and did not recommend water temperature-related operational or infrastructure modifications at that time. Consequently, for this Applicant-Prepared Draft BA water temperature is considered to be a low stressor for spring-run Chinook salmon in the lower Yuba River. Water temperature suitability in the Yuba River downstream of Englebright Dam is further described in Section 6.0 of this Applicant-Prepared Draft BA.

5.1.7 Viability of Central Valley Spring-run Chinook Salmon

The "Viable Salmonid Population" (VSP) concept was developed by McElhany et al. (2000) to facilitate establishment of ESU-level delisting goals and to assist in recovery planning by identifying key parameters related to population viability. Four key parameters were identified by McElhany et al. (2000) as the key to evaluating population viability status: 1) abundance; 2) productivity; 3) diversity; and 4) spatial structure. McElhany et al. (2000) interchangeably use the term population growth rate (i.e., productivity over the entire life cycle) and productivity. Good et al. (2007) used the term productivity when describing this VSP parameter, which also is the term used for this parameter in this Applicant-Prepared Draft BA. The following discussion regarding the four population viability population parameters was taken from NMFS (2009a).

Abundance is an important determinant of risk, both by itself and in relationship to other factors (McElhany et al. 2000). Small populations are at a greater risk for extinction than larger populations because risks that affect the population dynamics operate differently on small populations than in large populations. A variety of risks are associated with the dynamics of small populations, including directional effects (i.e., density dependence – compensatory and depensatory), and random effects (i.e., demographic stochasticity, environmental stochasticity, and catastrophic events).

The parameter of productivity and factors that affect productivity provide information on how well a population is "performing" in the habitats it occupies during the life cycle (McElhany et al. 2000). Productivity and related attributes are indicators of a population's performance in response to its environment and environmental change and variability. Intrinsic productivity (the maximum production expected for a population sufficiently small relative to its resource supply not to experience density dependence), the intensity of density dependence, and stage-specific productivity (productivity realized over a particular part of the life cycle) are useful in assessing productivity of a population.

Diversity refers to the distribution of traits within and among populations, and these traits range in scale from deoxyribonucleic acid (DNA) sequence variation at single genes to complex life-history traits (McElhany et al. 2000). Traits can be completely genetic or vary due to a combination of genetics and environmental factors. Diversity in traits is an important parameter because: 1) diversity allows a species to use a wide array of environments; 2) diversity protects a species against short-term spatial and temporal changes in its environment; and 3) genetic diversity provides the raw material for surviving long-term environmental changes (McElhany et al. 2000). Some of the varying traits include run timing, spawning timing, age structure, outmigration timing, etc. Straying and gene flow strongly influence patterns of diversity within and among populations (McElhany et al. 2000).

Spatial structure reflects how abundance is distributed among available or potentially available habitats, and how it can affect overall extinction risk and evolutionary processes that may alter a population's ability to respond to environmental change. A population's spatial structure encompasses the geographic distribution of that population, as well as the processes that generate or affect that distribution (McElhany et al. 2000). A population's spatial structure depends fundamentally on habitat quality, spatial configuration, and dynamics as well as the dispersal characteristics of individuals in the population. Potentially suitable but unused habitat is an indication of the potential for population growth.

5.1.7.1 ESU

To determine the current viability of the spring-run Chinook salmon ESU, NMFS (2009b) used the historical population structure of spring-run Chinook salmon presented in Lindley et al. (2007) and the concept of VSP for evaluating populations described by McElhany et al. (2000). Lindley et al. (2004) identified 26 historical populations within the spring-run ESU; 19 were independent populations, and 7 were dependent populations. Of the 19 independent populations of spring-run that occurred historically, only three remain, in Deer, Mill, and Butte creeks. Extant dependent populations occur in Battle, Antelope, Big Chico, Clear, Beegum, and Thomes creeks, as well as in the Yuba River, the Feather River below Oroville Dam, and in the mainstem Sacramento River below Keswick Dam (NMFS 2009b).

Lindley et al. (2007) provide criteria to assess the level of risk of extinction of Pacific salmonids based on population size, recent population decline, occurrences of catastrophes within the last 10 years that could cause sudden shifts from a low risk state to a higher one, and the impacts of hatchery influence. Although these criteria were developed for application to specific populations, insight to the viability of the spring-run Chinook salmon ESU can be obtained by examining population trends within the context of these criteria.

5.1.7.1.1 Viable Salmonid Population (VSP) Parameters and Application

Abundance

According to NMFS (2009b), spring-run Chinook salmon in the Central Valley declined drastically in the mid- to late 1980s before stabilizing at very low levels in the early to mid-1990s. Since the late 1990s, there does not appear to be a trend in basin-wide abundance (NMFS 2009b). Since NMFS presented these data, additional abundance estimates are available for the spring-run Chinook salmon ESU.

Central Valley-wide spring-run Chinook salmon abundance estimates are available through GrandTab (CDFW 2016b). Since 1983, in-river estimates for the lower Feather River have not been included in the system-wide estimates, although FRFH estimates are provided separately. Additionally, spring-run Chinook salmon are not estimated in GrandTab for the lower Yuba River, and all lower Yuba River Chinook salmon escapement estimates are reported as fall-run Chinook salmon. For the Sacramento River system (not including the FRFH or the lower Yuba River) since 1983, spring-run Chinook salmon run size estimates have ranged from a high of 24,903 in 1998 to a low of 1,195 in 2015. For the past 5 years (2011 - 2015), the abundance of in-river spawning Central Valley spring-run Chinook salmon has steadily declined from a high of 19,402 in 2013 to a low of 1,195 in 2015.

Overall, most Central Valley spring-run Chinook salmon escapement numbers have increased slightly in recent years (2012-2014), however, numbers dropped dramatically in 2015 (NMFS 2016a). Sacramento River tributary populations in Mill, Deer, and Butte creeks are likely the best trend indicators for the Central Valley spring-run Chinook salmon ESU as a whole because these streams contain the majority of the abundance and are the only independent populations within the ESU. Some other tributaries to the Sacramento River, such as Clear Creek and Battle Creek, have seen population gains in the years from 2001 to 2009, but the overall abundance numbers have remained low. Data suggest that 2012 appeared to be a good return year for most of the tributaries with some, such as Battle Creek, having the highest return on record (799). Additionally, 2013 escapement numbers increased in most tributary populations, which resulted in the second highest number of Central Valley spring-run Chinook salmon returning to the tributaries since 1960. The 2014 data indicate an overall large decline in spring-run Chinook salmon populations in the Sacramento River Basin in comparison to 2012 and 2013, possibly as a result of the current drought. The full effects of the drought (2010-2016) have yet to be seen in the returning Central Valley spring-run Chinook salmon (NMFS 2016a).

Productivity

The spring-run Chinook salmon run size estimate for the Sacramento River system (not including the FRFH or the lower Yuba River) over the past three consecutive years totaled 27,722 fish, thereby exceeding both the minimum total escapement value of 2,500 (Lindley et al. 2007), as well as the mean value of 833 fish per year identified by NMFS (2011a).

From 1983 through 2015, the annual contribution of spring-run Chinook salmon from the FRFH to the total annual run size in the Sacramento River system has ranged from a high of 78.8 percent (4,440 fish) in 2015 to a low of 5.6 percent (1,433 fish) in 1986. As an indicator of the FRFH influence on spring-run Chinook salmon in the Sacramento River system, the average annual percent contribution of FRFH spring-run Chinook salmon relative to the total annual run in the Sacramento River system was 33.6 percent over the entire 33-year period (1983-2015), and was 28.1 percent over the last 10 years (2006-2015). The percent contribution of FRFH to the total population of Central Valley spring-run Chinook salmon does not represent straying *per se*. The guidelines presented in Figure 1 in Lindley et al. (2007) present extinction risk levels corresponding to different amount, duration and source of hatchery strays, taking into consideration whether hatchery strays are from within the ESU, the diversity group, and from a "best management practices" hatchery. These criteria indicate a high extinction risk if hatchery straying represents more than 20 percent hatchery contribution for one generation or more than 10 percent for four generations from a hatchery within a given diversity group, or more than 50 percent hatchery contribution for one generation or more than 15 percent for four generations from a best management practices hatchery within a given diversity group. Although not technically representing straying, the average percentage contribution of spring-run Chinook salmon from the FRFH to the total annual run size in the Sacramento River system has been 29.3 percent over the most recent generation, 25.5 percent over the two most recent generations, 22.6 percent over the three most recent generations, and 20.1 percent over the four most recent generations assuming a three-year life cycle. According to NMFS (2016); recent anomalous conditions in the coastal ocean, along with consecutive dry years affecting inland freshwater conditions, have contributed to statewide escapement declines. Four consecutive years of drought (2012−2015) and the past two years (2014−2015) of exceptionally high air, stream, and upper ocean temperatures have together likely had negative impacts on the freshwater, estuary, and marine phases for many populations of Chinook salmon (Williams et al. 2016).

Spatial Structure

Lindley et al. (2007) indicated that of the 19 independent populations of spring-run that occurred historically, only three (Butte, Mill, and Deer creeks) remain, and their current distribution makes the spring-run ESU vulnerable to catastrophic disturbance (e.g., disease outbreaks, toxic spills, or volcanic eruptions). Butte, Mill, and Deer creeks all occur in the same biogeographic region (diversity group), whereas historically, independent spring-run populations were distributed throughout the Central Valley among at least three diversity groups (i.e., the Basalt and Porous Lava Diversity Group, the Northern Sierra Nevada Diversity Group, and the Southern Sierra Nevada Diversity Group). In addition, dependent spring-run populations historically persisted in the Northwestern California Diversity Group (Lindley et al. 2004). Currently, there are dependent populations of spring-run Chinook salmon in the Big Chico, Antelope, Clear, Thomes, Battle, and Beegum creeks, and in the Sacramento, Feather, and Yuba rivers (Lindley et al. 2007).

Spring-run Chinook salmon have been reported more frequently in several upper Central Valley creeks, but the sustainability of these runs is still unknown (NMFS 2004c). In 2004, NMFS reported that Butte Creek spring-run cohorts had recently utilized all available habitat in the creek, so the population cannot expand further. It is unknown if individuals have opportunistically migrated to other systems. The spatial structure of the Central Valley spring-run Chinook salmon ESU has been reduced with the extirpation of all San Joaquin River Basin spring-run populations (NMFS 2004c). Recently, implementation of the spring-run Chinook salmon reintroduction plan into the San Joaquin River began in 2014, which if successful, will benefit the spatial structure of the ESU. The reintroduced fish have been designated as a 10(j) nonessential experimental population when within the defined boundary in the San Joaquin River (78 FR 79622). While the San Joaquin River Restoration Program is managed to imprint Central Valley spring-run Chinook salmon to the mainstem San Joaquin River, NMFS (2016) anticipates that some of the reintroduced spring-run Chinook salmon are likely to stray into the San Joaquin tributaries, which will increase the likelihood for spring-run Chinook salmon to repopulate other Southern Sierra Nevada diversity group rivers where suitable conditions exist.

Diversity

As discussed in NMFS (2009b), diversity, both genetic and behavioral, provides a species the opportunity to track environmental changes. As a species' abundance decreases, and spatial structure of the ESU is reduced, a species has less flexibility to track changes in the environment. Spring-run Chinook salmon reserve some genetic and behavioral variation in that in any given year, at least two cohorts are in the marine environment and, therefore, are not exposed to the same environmental stressors as their freshwater cohorts (NMFS 2009b).

Genetic analysis of natural and hatchery spring-run Chinook salmon stocks in the Central Valley reveal that the southern Cascades spring-run population complex has retained its genetic integrity (NMFS 2004a). However, although spring-run produced at the FRFH are part of the spring-run Chinook salmon ESU (70 FR 37160, June 28, 2005), they compromise the genetic diversity of naturally-spawned spring-run Chinook salmon (NMFS 2009b). The spring-run hatchery stock introgressed with the fall-run hatchery stock and both are genetically linked with the natural populations in the Feather River (NMFS 2004b). The FRFH program has affected the diversity of the Central Valley spring-run Chinook salmon and, together with the loss of the San Joaquin River Basin spring-run populations, the diversity of the Central Valley spring-run Chinook salmon ESU has been reduced (NMFS 2004b).

Concerns remain with the spring-run Chinook salmon hatchery that is part of the ESU, as there has been and continues to be some introgression with other Central Valley spring-run Chinook salmon populations, as well as with fall-run Chinook salmon (NMFS 2016b). The majority of the FRFH spring-run Chinook salmon broodstock and in-river spawning population on the Feather River are first generation hatchery-produced fish (Kormos et al. 2012; Palmer-Zwahlen and Kormos 2013). The proportion of natural-origin fish in the broodstock was estimated to be 18 percent and 6 percent during 2010 and 2011, respectively (Kormos et al. 2012, Palmer-Zwahlen and Kormos 2013). Thus, the minimum criteria of greater than 10 percent of natural-origin fish in the broodstock is not being met annually (California HSRG 2012). The proportion of hatchery-origin spring- or fall-run Chinook salmon contributing to the natural spawning spring-run Chinook salmon population on the Feather River remains unknown due to overlap in the spawn timing of spring-run and fall-run Chinook salmon, and lack of physical separation. However, the hatchery component is likely to be high (NMFS 2016b). As an example, 78 percent and 90 percent of spawners in the 2010/2011 spring-/fall- run Chinook salmon carcass survey were estimated to be from the FRFH, respectively (Kormos et al. 2012; Palmer-Zwahlen and Kormos 2013).

According to NMFS (2016), FRFH-origin spring-run Chinook salmon adults have been recovered in other Central Valley spring- and fall-run Chinook salmon populations outside of the Feather River. Up until 2015, at least half of the FRFH spring-run Chinook salmon production was trucked to release sites such as the San Francisco Bay, which led to the returns straying to other watersheds at a relatively high rate, posing genetic risk to those other Central Valley salmon populations (Kormos et al. 2012; Palmer-Zwahlen and Kormos 2013). The annual spawning run size of Central Valley spring-run Chinook salmon on the Yuba River followed the annual abundance trend of the FRFH spring-run Chinook salmon population. On Battle Creek, as high as 29 percent of spring-run Chinook salmon in 2010 were estimated to have originated from the FRFH (USFWS 2014). A significant number of FRFH spring-run Chinook salmon strays have been observed in the Keswick Dam fish trap, with a high in 2015 of 114 fish. This indicates a likelihood that these fish could be interbreeding with natural-origin Central Valley spring- or fall-run Chinook salmon in the Sacramento River (Rueth 2015). A prolonged influx of FRFH spring-run Chinook salmon strays to other Central Valley spring-run Chinook salmon populations, even at levels of less than 1 percent is undesirable, and can cause the receiving population to shift to a moderate risk after four generations of such impact (Lindley et al. 2007). According to NMFS (2016), more information on the incidence of FRFH spring-run straying is desirable to more accurately estimate the extent to which spawning and introgression is occurring between fall-run and spring-run Chinook salmon populations outside of the Feather River.

Summary of the Viability of the Central Valley Spring-run Chinook Salmon ESU

According to NMFS (2005a), threats from hatchery production, climatic variation, predation, and water diversions persist. Because the Central Valley spring-run Chinook salmon ESU is confined to relatively few remaining streams and continues to display broad fluctuations in abundance, high quality critical habitat containing spawning sites with adequate water and substrate conditions, or rearing sites with adequate floodplain connectivity, cover, and water conditions (i.e., key primary constituent elements of critical habitat that contribute to its conservation value) is considered to be limited and the population is at a moderate risk of extinction.

According to NMFS (2014a), spring-run Chinook salmon fail the representation and redundancy rule for ESU viability, because the current distribution of independent populations has been severely constricted to only one of their former geographic diversity groups. NMFS (2009a) concluded that the Central Valley spring-run Chinook salmon ESU is at moderate risk of extinction in 100 years.

In 2016, NMFS completed a 5-year status review of the Central Valley spring-run Chinook salmon ESU. According to NMFS (2016), new information for the Central Valley spring-run Chinook salmon ESU suggests an overall improvement since the 2010 status review, through 2014, with two (Mill and Deer) of the three extant independent populations improving from high extinction risks to moderate extinction risks. The third independent population (Butte Creek) has remained at low risk, and all viability metrics had been trending in a positive direction, up until 2015. NMFS (2016) states that most dependent spring-run populations have been experiencing continued and somewhat drastic declines. The Central Valley spring-run Chinook salmon ESU has experienced two drought periods over the past decade. From 2007 to 2009, and 2012 to 2015, the Central Valley experienced drought conditions and low river and stream discharges, which are generally associated with lower survival of Chinook salmon (Michel et al. 2015). The impacts of the recent drought years and warm ocean conditions on the juvenile lifestage will not be fully realized by the viability metrics until they manifest in potential low run size returns in 2015 through 2018 (Williams et al. 2016). This is already being realized with very low returns in 2015 (NMFS 2016b). Overall, the recent declines have been significant but not severe enough to qualify as a catastrophe under the criteria of Lindley et al. (2007). On the positive side, spring-run Chinook salmon appear to be repopulating Battle Creek, home to a historical independent population in the Basalt and Porous Lava diversity group that was extirpated for many decades. Similarly, the spring-run Chinook salmon population in Clear Creek has been increasing, although Lindley et al. (2004) classified this population as a dependent population, and thus it is not expected to exceed the low-risk population size threshold of 2,500 fish (i.e., annual spawning run size of about 833 fish). NMFS (2016) reports that spring-run Chinook salmon in both Battle Creek and Clear Creek continue to repopulate those watersheds, and now fall into the moderate extinction risk category for abundance.

The status of the Central Valley spring-run Chinook salmon ESU has probably improved since the 2010 status review (NMFS 2016b) and Lindley et al.'s (2007) assessment. The largest improvements are due to extensive habitat restoration, and increases in spatial structure, with historically extirpated populations trending in the positive direction. Improvements, evident in the moderate and low risk of extinction of the three independent populations, however, are not enough to warrant the delisting of the ESU according to NMFS (2016). The recent declines of many of the dependent populations, high pre-spawn and egg mortality during the 2012 to 2015 drought, uncertain juvenile survival during the drought, and ocean conditions, as well as the level of straying of FRFH spring-run Chinook salmon to other Central Valley spring-run Chinook salmon populations, are all causes for concern for the long-term viability of the Central Valley spring-run Chinook salmon ESU (NMFS 2016b).

In summary, NMFS (2016) states that, with a few exceptions, Central Valley spring-run Chinook salmon populations have increased through 2014 returns since the last status review (2010/2011), which has moved the Mill and Deer creek populations from the high extinction risk category, to moderate, and the Butte Creek population has remained in the low risk of extinction category. Additionally, the Battle Creek and Clear Creek populations have continued to show stable or increasing numbers the last five years, putting them at moderate risk of extinction based on abundance. Overall, NMFS concluded in their viability report that the status of Central Valley spring-run Chinook salmon (through 2014) has probably improved since the 2010/2011 status review and that the ESU's extinction risk may have decreased, however the ESU is still facing significant extinction risk, and that risk is likely to increase over at least the next few years as the full effects of the recent drought are realized (Williams et al. 2016). According to NMFS (2016), there are potentially significant conservation measures to restore or expand habitat that are in early stages of implementation, but the potential benefits from these actions will not be realized for several years or more and the degrees to which they will help benefit Central Valley spring-run Chinook salmon and their habitat are uncertain.

5.1.7.2 Lower Yuba River

As previously discussed, the VSP concept was developed by McElhany et al. (2000) in order to facilitate establishment of ESU-level delisting goals and to assist in recovery planning by identifying key parameters related to population viability. The four parameters established by McElhany et al. (2000) included abundance, productivity, spatial structure and genetic and life-history diversity, although McElhany et al. (2000) did not provide quantitative criteria that would allow assessment of whether particular populations or ESUs/DPSs are viable.

Lindley et al. (2007) characterized the spring-run Chinook salmon population in the lower Yuba River as data deficient, and therefore did not characterize its viability. In 2007, there was limited information on the current population size of spring-run Chinook salmon in the lower Yuba River, although NMFS (2009a) stated that ongoing monitoring is providing additional information. According to NMFS (2016), new data based on VAKI Riverwatcher™ counts for the lower Yuba River suggest that the spring-run Chinook salmon population's size meets the low extinction risk criteria for abundance (ranging from a few hundred to a few thousand fish). However, the population is likely at high extinction risk due to hatchery influence (NMFS 2016b).

5.1.7.2.1 Abundance and Productivity

Run Differentiation (Spring-run vs. Fall-run Chinook Salmon)

Prior to application of VSP performance indicators or the extinction risk criteria, it is necessary to differentiate between annually returning spring-run and fall-run Chinook salmon in the lower Yuba River. However, as reported by RMT (2013a), there is no discernible genetic differentiation available to determine spring-run Chinook salmon, only phenotypic differentiation. The phenotypic expression is often obscure, requiring application of advanced statistical techniques to VAKI Riverwatcher™ and other datasets in order to identify the phenotypic differences in run timing. The following discussion of differentiating phenotypic spring-run from phenotypic fall-run Chinook salmon in the lower Yuba River is generally taken from RMT (2013a), but is updated based on the availability of more recent VAKI Riverwatcher™ data that has been collected since RMT (2013a) was released. The RMT (2013a) spring-run and fall-run Chinook salmon analyses extended from biological year (i.e., March 1 through February 28) 2004 through 2011. The analyses presented below incorporate four additional biological years extending from March 1, 2012 through February 28, 2016.

Infrared-imaging technology has been used to monitor fish passage at Daguerre Point Dam in the lower Yuba River since 2003 using VAKI Riverwatcher™ systems to document specific observations used to address VSP parameters of adult abundance and diversity. The VAKI Riverwatcher™ infrared systems produced by VAKI Aquaculture Systems Ltd., of Iceland, provided a tool for monitoring fish passage year-round. The VAKI Riverwatcher™ system records both silhouettes and electronic images of each fish passage event in both of the Daguerre Point Dam fish ladders. By capturing silhouettes and images, fish passage can be accurately monitored even under turbid conditions.

The VAKI Riverwatcher™ systems located at both the north and south ladder of Daguerre Point Dam were able to record and identify the timing and magnitude of passage for Chinook salmon at Daguerre Point Dam during most temporal periods of a given year. Prior to applying any analysis of temporal modalities to the 12 annual time series of Chinook salmon daily VAKI counts, the annual daily count series at each ladder were adjusted to account for days when the VAKI Riverwatcher™ systems were not fully operational. The procedure used to obtain complete annual daily count series of Chinook salmon migrating upstream of Daguerre Point Dam is provided in RMT (2013a).

The daily time series of Chinook salmon moving upstream of Daguerre Point Dam resulting from the previous step were further analyzed and temporal modalities were explored to differentiate spring-run from fall-run Chinook salmon each year. For a full description of the run differentiation process, see RMT (2013a).

Figures 5.1-10 through Figure 5.1-13 display the daily number of Chinook salmon that passed upstream of Daguerre Point Dam during the 2004/2005 through 2015/2016 biological years (March 1 through February 28) and the fitted generalized logistic functions describing the distributions of spring-run and fall-run Chinook salmon resulting from the application of the annually variable temporal demarcation procedure.

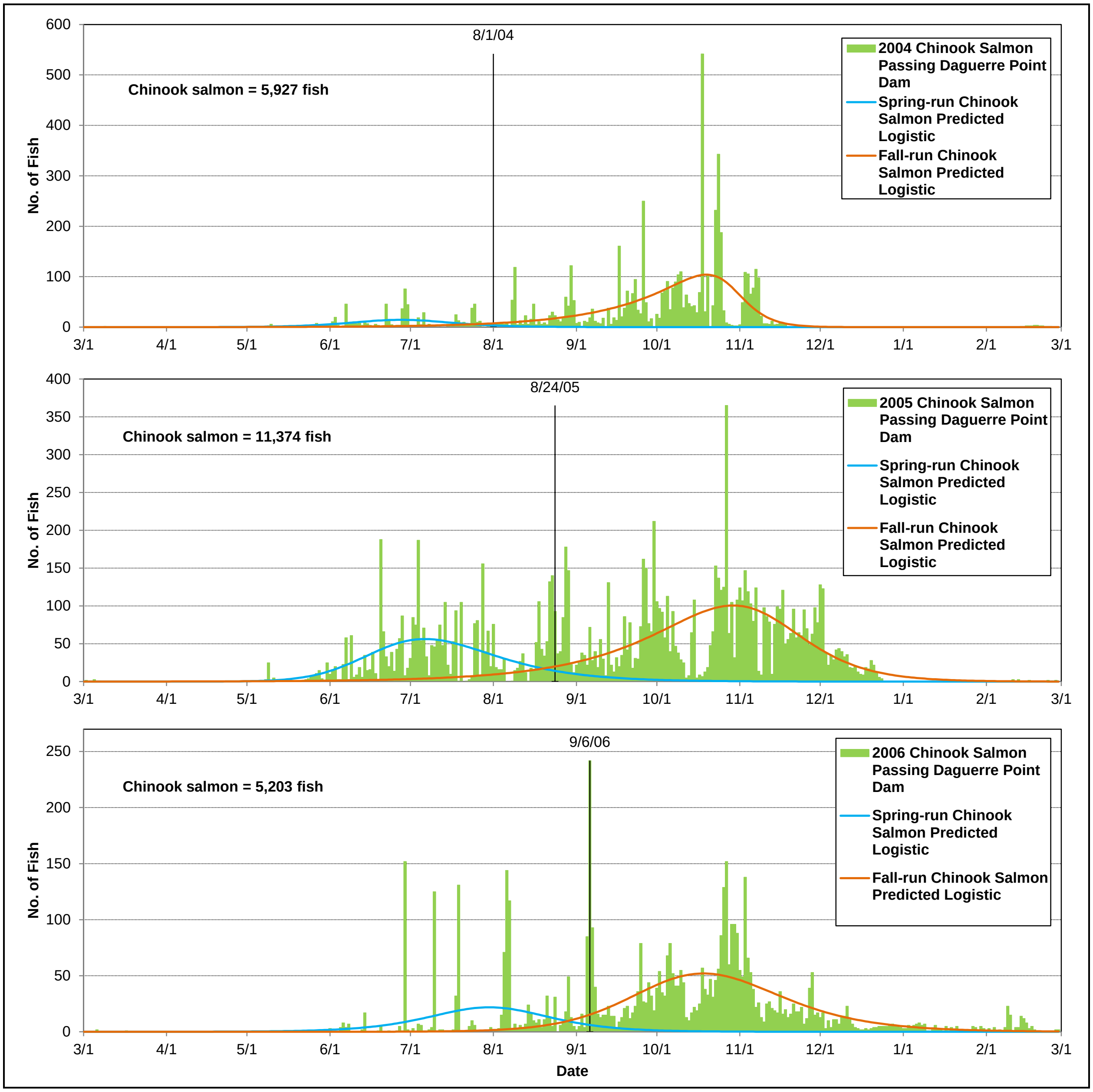

Figure 5.1-10. Daily number of Chinook salmon passing upstream of Daguerre Point Dam during the 2004/2005 to 2006/2007 biological years.[1]

[1] Bars indicate the VAKI Riverwatcher™ daily counts and lines indicate the predicted daily distributions of spring-run (blue line) and fall-run (orange line) Chinook salmon based on the fitting of two generalized logistic functions to the data. The demarcation date differentiating the two runs of Chinook salmon is indicated for each year.

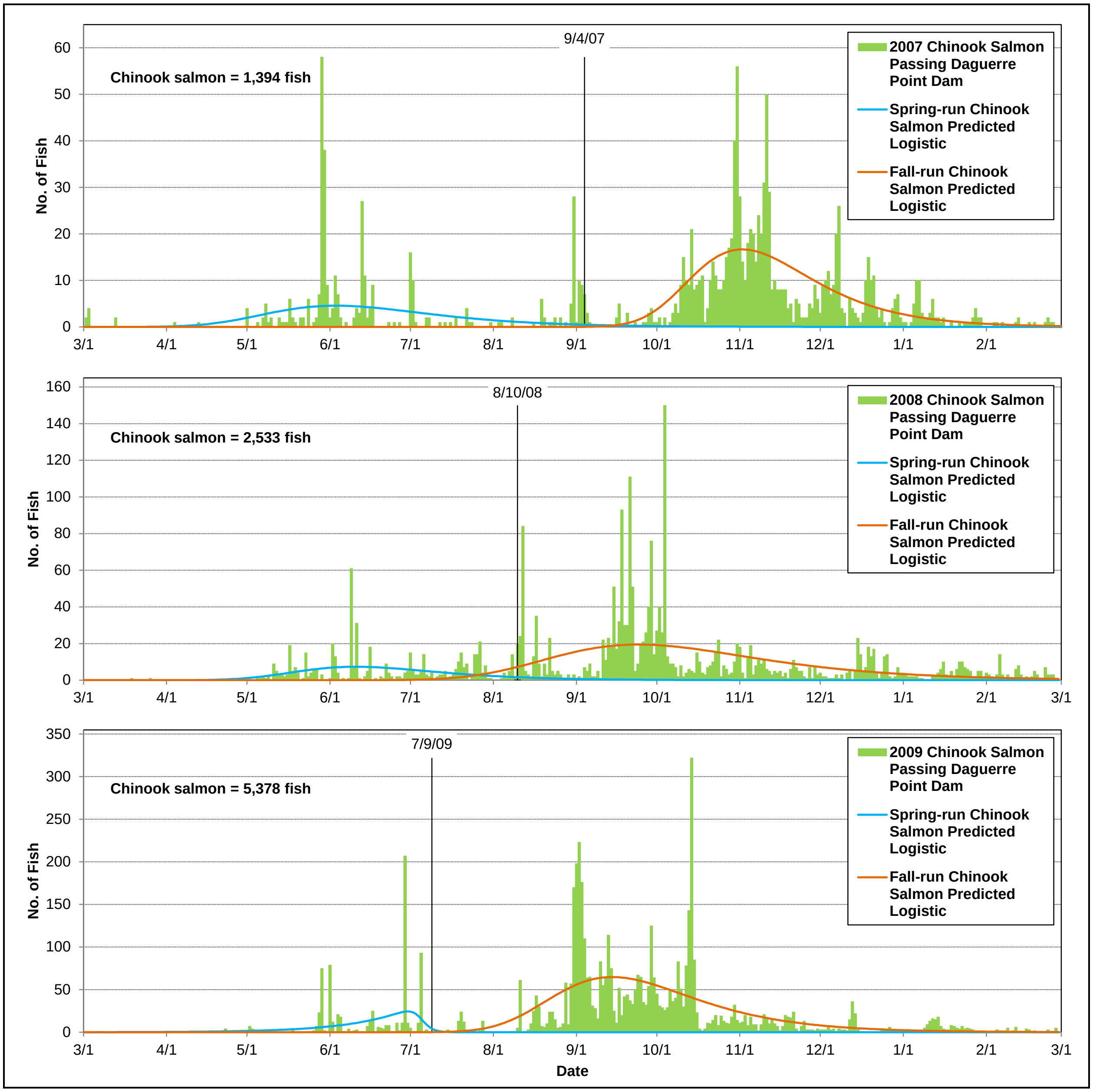

Figure 5.1-11. Daily number of Chinook salmon passing upstream of Daguerre Point Dam during the 2007/2008 to 2009/2010 biological years.[1]

[1] Bars indicate the VAKI Riverwatcher™ daily counts and lines indicate the predicted daily distributions of spring-run (blue line) and fall-run (orange line) Chinook salmon based on the fitting of two generalized logistic functions to the data. The demarcation date differentiating the two runs of Chinook salmon is indicated for each year.

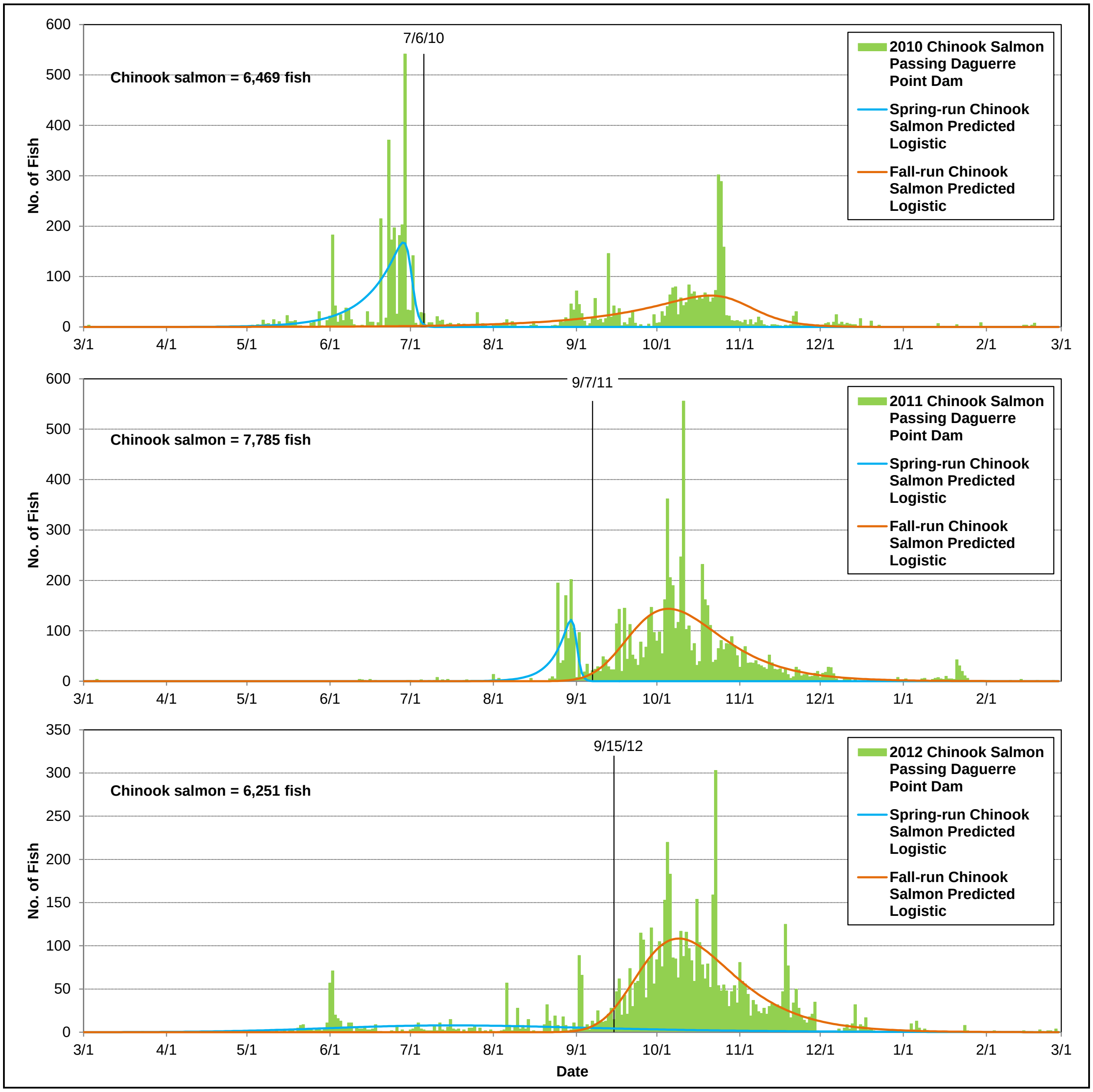

Figure 5.1-12. Daily number of Chinook salmon passing upstream of Daguerre Point Dam during the 2010/2011 to 2012/2013 biological years.[1]

[1] Bars indicate the VAKI Riverwatcher™ daily counts and lines indicate the predicted daily distributions of spring-run (blue line) and fall-run (orange line) Chinook salmon based on the fitting of two generalized logistic functions to the data. The demarcation date differentiating the two runs of Chinook salmon is indicated for each year.

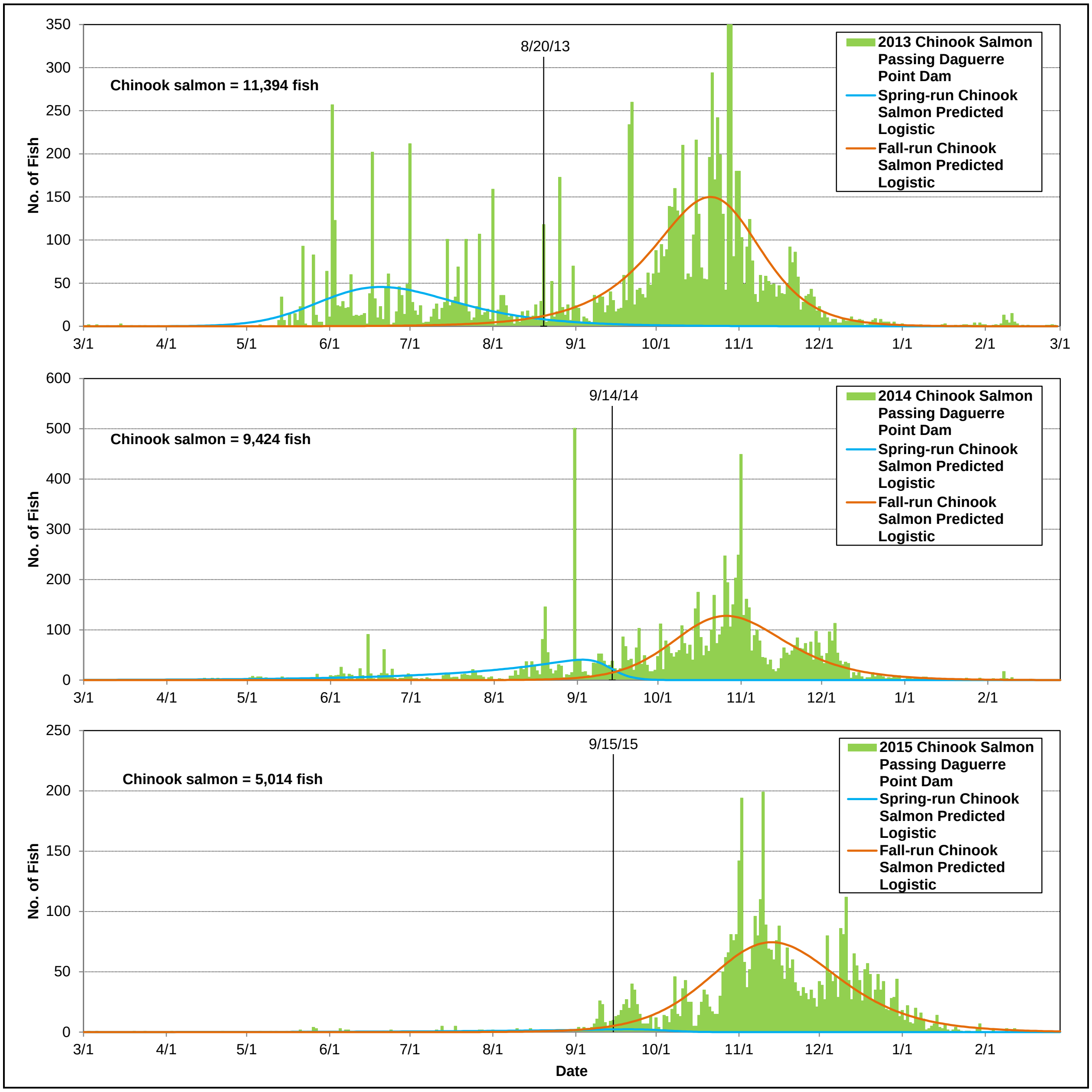

Figure 5.1-13. Daily number of Chinook salmon passing upstream of Daguerre Point Dam during the 2013/2014 to 2015/2016 biological years.[1]

[1] Bars indicate the VAKI Riverwatcher™ daily counts and lines indicate the predicted daily distributions of spring-run (blue line) and fall-run (orange line) Chinook salmon based on the fitting of two generalized logistic functions to the data. The demarcation date differentiating the two runs of Chinook salmon is indicated for each year.

Table 5.1-3 summarizes the total number of spring-run and fall-run Chinook salmon estimated to have passed upstream of Daguerre Point Dam annually, and the estimated annual percentage of spring-run Chinook salmon relative to all Chinook salmon each year.

Table 5.1-3. Annual number of spring-run and fall-run Chinook salmon estimated to have passed upstream of Daguerre Point Dam, and the estimated annual percentage of spring-run Chinook salmon relative to all Chinook salmon each year.

Run	Biological Year											
	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015
Spring-run	738	3,592	1,326	372	521	723	2,886	1,159	1,046	3,130	2,336	184
Chinook Salmon	12.5%	31.6%	25.5%	26.7%	20.6%	13.4%	44.6%	14.9%	16.7%	27.5%	24.8%	3.7%
Fall-run	5,189	7,782	3,877	1,022	2,012	4,655	3,583	6,626	5,205	8,264	7,088	4,830
Chinook Salmon	87.5%	68.4%	74.5%	73.3%	79.4%	86.6%	55.4%	85.1%	83.3%	72.5%	75.2%	96.3%

5.1.7.2.2 Annual Abundance of Spring-run Chinook Salmon

For the biological years 2004/2005 through 2015/2016, during which VAKI Riverwatcher™ data are available, the annual number of spring-run Chinook salmon estimated to have passed upstream of Daguerre Point Dam ranged from 184 in 2015 to 3,592 in 2005, with an average of 1,501. With the exception of 2015, the abundance of spring-run Chinook salmon during the past 6 years (2010-2015) has been substantially higher than the three years prior (2007-2009).

As previously described by NMFS (2016), populations with a low risk of extinction (less than 5 percent chance of extinction in 100 years) are those with a minimum total escapement of 2,500 spawners in 3 consecutive years (mean of 833 fish per year). For the last 3 consecutive years, an estimated total of 5,650 spring-run Chinook salmon have passed upstream of Daguerre Point Dam, with an average of 1,883 fish per year. However, as further discussed below, the annual abundances of phenotypic spring-run Chinook salmon in the lower Yuba River are strongly influenced by hatchery fish.

Trends in the Annual Abundance of Spring-run Chinook Salmon

The statistical approach recommended by Lindley et al. (2007) was followed by RMT (2013a) and by YCWA in this Applicant-Prepared Draft BA to examine whether the abundance of lower Yuba River spring-run Chinook salmon exhibited a statistically significant linear trend over time during the 12 most recent years for which VAKI Riverwatcher™ data are available. The natural logarithms of the abundance estimates of lower Yuba River spring-run Chinook salmon for the 12 most recent years (20042005 – 2015/2016) were linearly regressed against time (year) using a simple least-squares approach. The estimated slope of the resulting line is a measure of the average rate of change of the abundance in the population over time.

Figure 5.1-14 displays the antilogarithmic transformation of the estimated annual number of spring-run Chinook salmon passing upstream of Daguerre Point Dam from 2004/2005-2015/2016. Figure 5.1-14 demonstrates that the abundance of spring-run Chinook salmon in the lower Yuba River has exhibited a very slight increase over the 12 years examined. However, the coefficient of determination is very weak (r^2 = 0.001) and the slope, statistically, is not significantly different from zero (P = 0.973), indicating that the positive trend is not significant. The relationship indicates that the phenotypic spring-run Chinook salmon annual abundance over this time period is stable, and is not exhibiting a significant declining trend. These abundance and trend considerations would correspond to low extinction risk according to NMFS criteria (Lindley et al. 2007). However, the RMT (2013a) questions the applicability of any of these criteria addressing extinction risk, because they presumably apply to independent populations and, as previously discussed, lower Yuba River anadromous salmonids represent introgressive hybridization of larger Feather-Yuba river populations, with substantial contributions of hatchery-origin fish to the annual runs. As previously mentioned, the annual abundances of phenotypic spring-run Chinook salmon in the lower Yuba River are strongly influenced by hatchery fish, as discussed below.

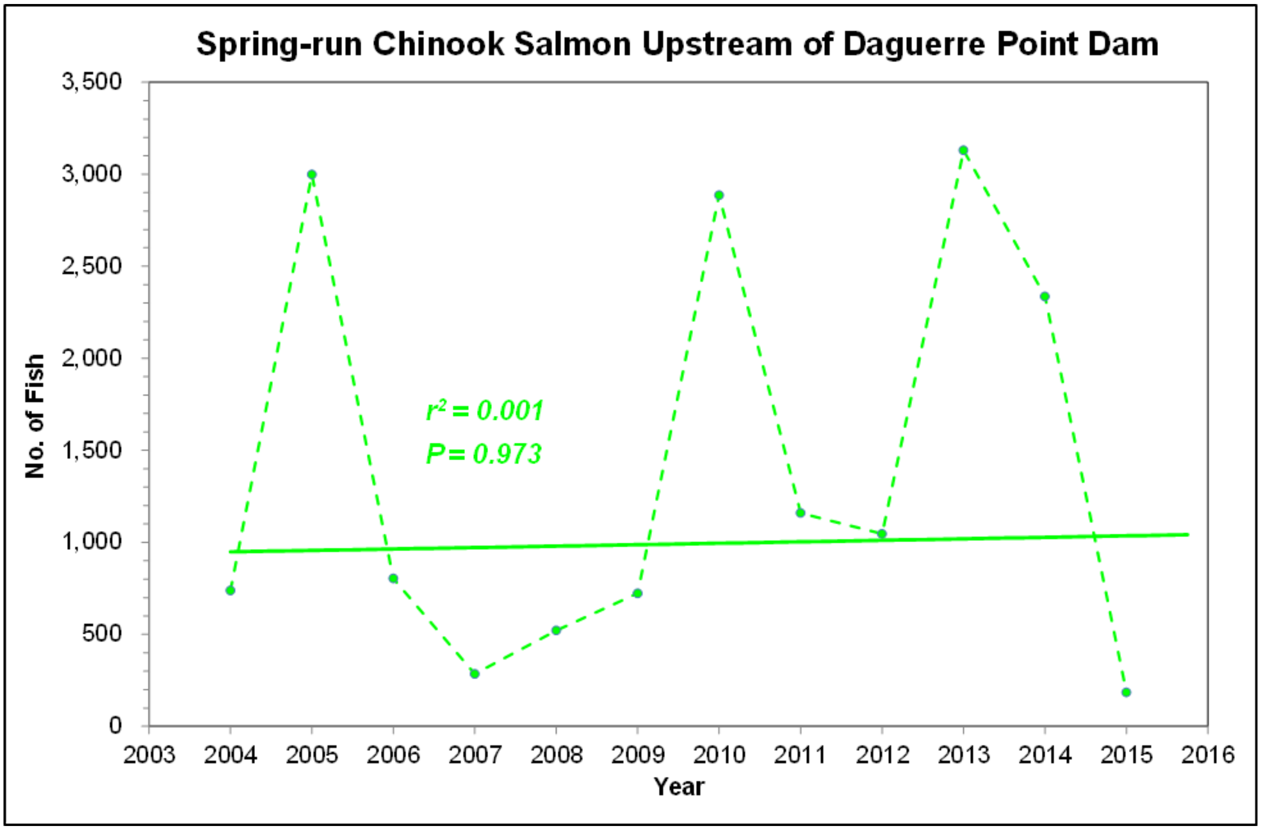

Figure 5.1-14. Temporal trend and estimated annual number of phenotypic adult spring-run Chinook salmon passing upstream of Daguerre Point Dam from 2004 through 2015.

Annual Abundance of Adipose Fin-clipped and Non Adipose Fin-Clipped Spring-run Chinook Salmon

Because the VAKI Riverwatcher™ systems located at both the north and south ladder of Daguerre Point Dam can record both silhouettes and electronic images of each fish passage event, the systems were able to differentiate Chinook salmon with adipose fins clipped or absent from Chinook salmon with their adipose fins intact. Thus, annual series of daily counts of Chinook salmon with adipose fins clipped (i.e., ad-clipped fish) and with adipose fins intact (i.e., not ad-clipped fish) that passed upstream of Daguerre Point Dam from March 1, 2004 through February 28, 2016 were obtained from the RMT.

The estimated numbers of spring-run Chinook salmon of hatchery (i.e., ad-clipped fish) and potentially non-hatchery origin (i.e., not ad-clipped fish) passing upstream of Daguerre Point

Dam from 2004 through 2015 of available VAKI Riverwatcher™ data are presented in Table 5.1-4. Examination of Table 5.1-4 demonstrates a sharp increase in the annual percent contribution of ad-clipped phenotypic spring-run Chinook salmon to the total estimated annual run beginning in 2009, reaching a peak in 2010, and returning to similar contribution rates that were observed in 2009 during 2011 and 2014. These results may be due, in part, to the fact that FRFH-origin spring-run Chinook salmon were fractionally marked prior to 2005 and 100 percent marked thereafter. These fish would have returned as age-3 fish during 2008. Also, fractional marking of fall-run hatchery fish at the FRFH started during 2006, and these fish may return, to some extent, as phenotypic spring-run Chinook salmon. Age-3 fish would have returned during 2009. The first full year (age-3 and age-4) of recovery data from the constant fractional marking (CFM) program occurred during 2010.

Table 5.1-4. Estimated numbers of Chinook salmon, ad-clipped and non ad-clipped phenotypic spring-run Chinook salmon that passed upstream of Daguerre Point Dam annually from 2004 through 2015.

Year	Demarcation Date	Chinook Salmon Passage Upstream of Daguerre Point Dam				
		All Chinook Salmon	Spring-run Chinook Salmon			
			Total	Ad-Clipped	Not Ad-Clipped	% Ad-Clipped
2004	8/1/04	5,927	738	72	666	10
2005	8/24/05	11,374	3,592	676	2,916	19
2006	9/6/06	5,203	1,326	81	1,245	6
2007	9/4/07	1,394	372	38	334	10
2008	8/10/08	2,533	521	15	506	3
2009	7/9/09	5,378	723	213	510	29
2010	7/6/10	6,469	2,886	1,774	1,112	61
2011	9/7/11	7,785	1,159	323	836	28
2012	9/15/12	6,251	1,046	297	749	28
2013	8/20/13	11,394	3,130	137	2,993	4
2014	9/14/14	9,424	2,336	218	2,118	9
2015	9/15/15	5,014	184	14	170	8

Although it was not possible to differentiate between phenotypic spring- and fall-run Chinook salmon in the lower Yuba River carcass surveys, evaluation of the Yuba River carcass survey data and recovery of coded wire-tags indicated that hatchery-origin Chinook salmon comprised an estimated 71 percent of the total 2010 Chinook salmon run in the entire Yuba River downstream of Englebright Dam (Kormos et al. 2012 as cited in RMT 2013a). Carcass survey data and recovery of coded-wire tags from 2011 indicate that approximately 34 percent of all Chinook salmon that spawned downstream of Daguerre Point Dam were of hatchery origin (Palmer-Zwahlen and Kormos 2013). VAKI Riverwatcher™ data, in conjunction with a biosample of 107 heads recovered during 2011 upstream of Daguerre Point Dam, indicate that approximately 65 percent of all spawning Chinook salmon upstream of Daguerre Point Dam were of hatchery origin (Palmer-Zwahlen and Kormos 2013).

The majority of hatchery-origin Chinook salmon identified throughout the Yuba River based on coded-wire tag data during 2011 originated from the FRFH (about 75%), the vast majority of which were net pen releases[4], followed by Coleman National Fish Hatchery net pen releases (~15%), Nimbus Fish Hatchery in-basin and net pen releases (about 7%), Mokelumne River

[4] "Net pen releases" refer to hatchery fish that were transported to and held in net pens before being released into San Pablo Bay.

Hatchery trucked[5] and net pen releases (about4%), and the Merced Fish Hatchery (0.2%) (Palmer-Zwahlen and Kormos 2013).

The average contribution of adipose fin-clipped phenotypic spring-run Chinook salmon to the total annual run size in the lower Yuba River, as inferred by the percentage of adipose fin-clipped fish passing upstream of Daguerre Point Dam during the annual defined phenotypic period, has been 17.9 percent over the 12 years of available data. The RMT (2013a) recognized that there are limitations to simply using percent adipose fin-clipped spring-run Chinook salmon passing through the VAKI Riverwatcher™ systems as an estimate of total hatchery influence, and that resulting estimates should be considered as minimum estimates. It is important to note that the adipose fin-clipped phenotypic spring-run Chinook salmon abundance represents a minimum indicator of hatchery-origin individuals due to fractional marking of spring-run hatchery fish prior to 2005, and CFM of fall-run hatchery fish at the FRFH since 2006 which may return as phenotypic spring-run Chinook. While not run-specific, the proportion of hatchery-origin Chinook salmon in the Yuba River in 2010 and 2011 estimated based on coded-wire tag recovery data suggest that the adipose fin-clipped counts of Chinook salmon may underestimate the proportion of hatchery-origin Chinook salmon in the Yuba River. The coded-wire tag recovery data for 2011 in the Yuba River also suggest that the number of hatchery fish that are released from net pens in San Pablo Bay also may influence the annual proportion of hatchery-origin Chinook salmon entering the Yuba River in future years.

The RMT (2013a) also recognized that the hatchery influence criterion presumably is applicable to an independent, genetically distinct population. However, as previously discussed, the phenotypic spring-run Chinook salmon in the lower Yuba River actually represent hybridization between spring- and fall-run Chinook salmon in the lower Yuba River, and hybridization with Feather River stocks including the FRFH spring-run Chinook salmon stock, which itself represents a hybridization between Feather River fall- and spring-run Chinook salmon populations.

5.1.7.2.3 Applicability of Additional VSP Parameters and Extinction Risk Criteria

The M&E Program framework developed by the RMT (2010) utilized VSP performance indicators that were identified based on the precept that the lower Yuba River anadromous salmonid populations represented independent populations. However, the RMT has identified a substantial amount of reproductive interaction between lower Yuba River and lower Feather River anadromous salmonid stocks. As described in RMT (2013a), phenotypic spring-run Chinook salmon in the lower Yuba River likely represents hybridization between spring- and fall-run Chinook salmon in the lower Yuba River, hybridization with Feather River fall- and spring-run Chinook salmon stocks, and hybridization with the FRFH spring-run Chinook salmon stock, which itself represents hybridization between Feather River fall- and spring-run Chinook salmon populations. Additionally, it is likely that anadromous *O. mykiss* stocks are similarly hybridized, with fluid intermixing of lower Feather River and lower Yuba River fish.

[5] "Trucked" refers to hatchery fish that were trucked from the hatchery and released into the waters of the Carquinez Strait, without any net pen acclimation.

The recognition of the extent of hybridization and lack of reproductive isolation of lower Yuba River and lower Feather River anadromous salmonid stocks logically constrains the manner in which the VSP concept can be applied to the lower Yuba River, because many of the VSP metrics are designed to evaluate the viability of discrete, independent populations. Even the simplified approach suggested by Lindley et al. (2007) to evaluate 'extinction risk' is of limited applicability in the evaluation of highly introgressed populations whose evaluation metrics are directly influenced by other stocks, and out-of-basin factors.

Lindley et al. (2007) provide criteria to assess the level of risk of extinction of Pacific salmonids based on population size, recent population decline, occurrences of catastrophes within the last 10 years that could cause sudden shifts from a low risk state to a higher one, and the impacts of hatchery influence. Populations with a low risk of extinction (less than 5% chance of extinction in 100 years) are those with a minimum total escapement of 2,500 spawners in 3 consecutive years (mean of 833 fish per year), no apparent decline in escapement, no catastrophic declines within the last 10 years, and a low hatchery influence (NMFS 2011a). The overall estimated risk of extinction for the population is determined by the highest risk score for any category Lindley et al. (2007). While more detailed population viability assessment (PVA) models could be constructed to assess Chinook salmon populations, Lindley et al. (2007) suggest any PVA results should be compared with the results of applying their simpler criteria to estimate status (NMFS 2011a).

Only some of the VSP performance indicators identified in the RMT (2010a) M&E Program framework and some of the extinction risk criteria provided by Lindley et al. (2007) are appropriate for application specifically to lower Yuba River anadromous salmonids. VSP performance indicators regarding spatial structure are applicable to the habitat conditions in the lower Yuba River. Similarly, the catastrophe occurrence extinction risk criterion also is applicable to the lower Yuba River. The extinction risk criteria including abundance and trends in abundance are of limited applicability and serve as illustrative comparative measures in consideration of the apparent non-independent salmonid populations in the lower Yuba River. Application of the hatchery risk extinction criterion also is confounding regarding the apparent non-independent lower Yuba River salmonid populations. Considerations regarding each of these applicabilities are discussed below.

Spatial Structure

According to McElhany et al. (2000), spatial structure reflects how abundance is distributed among available or potentially available habitats, and how it can affect overall extinction risk and evolutionary processes that may alter a population's ability to respond to environmental change. A population's spatial structure depends fundamentally on habitat quality, spatial configuration, and dynamics, as well as on the dispersal characteristics of individuals in the population.

Performance indicators and analytics addressing spatial structure include spatial organization of MUs (e.g., lateral variability/diversity, adjacency, randomness, and abundance), persistence of MUs through time, and the quality, number, size and distribution of MUs available for spawning Chinook salmon. Additional considerations include floodplain connectivity, entrenchment, channel sinuosity, substrate size, changes in topographic depth, scour and fill processes, bankfull and flood flow recurrence interval, and maintenance of watershed processes to maintain suitable habitat for anadromous salmonid lifestages.

As stated in the M&E Plan (RMT 2010a), the spatial structure evaluation includes examination of maintenance of watershed processes and regulatory management practices to create and maintain suitable habitat for all freshwater lifestages of spring-run and fall-run Chinook salmon, and steelhead. As discussed in RMT (2013a), one of the performance indicators preliminarily evaluated by Wyrick and Pasternack (2012) is whether the sequence of MUs in the lower Yuba River is non-random. Highly disturbed systems often degrade into homogeneity or randomness.

Of the 12 major near-bankfull MUs, the most uniformly distributed (i.e., randomly located) units are slackwater, slow glide, and lateral bar. As an example of non-uniform distribution, pool units were predominantly found in the upstream reaches (i.e., Englebright and Timbuctoo Bend) and the downstream reach (i.e., Marysville), but were less abundant in the middle, wider reaches (i.e., Daguerre Point Dam and Dry Creek). Consequently, evaluation of the morphological units in the lower Yuba River as part of the spatial structure analyses indicates that, in general, the sequence of MUs is non-random, indicating that the channel has been self-sustaining of sufficient duration to establish an ordered spatial structure (refer to RMT 2013a for additional discussion).

Another method for analyzing the morphological unit organization that Wyrick and Pasternack (2012) developed is an adjacency probability analysis, which evaluates the frequency at which each MU is adjacent to every other unit, and compares that against random adjacency expectations. Results of this analysis indicate that the in-channel units near the thalweg typically exhibit low adjacency probabilities to the bar units, although they do exhibit higher-than-random adjacency probabilities to other in-channel units.

Wide, diverse rivers should also exhibit lateral variability in their form-process associations. In the lower Yuba River, MU organization highlights the complexity of the channel geomorphology, as well as the complex and diverse suite of potential habitat at any given location in the Yuba River. The above summary (described in more detail in RMT 2013a) illustrates that spatial structure of MUs in the lower Yuba River is complex, diverse, and persistent.

Catastrophe Occurrence

According to Lindley et al. (2007), the catastrophe criteria trace back to Mace and Lande (1991), and the underlying theory is further developed by Lande (1993). The following discussion was taken from Lindley et al. (2007). The overall goal of the catastrophe criteria is to capture a sudden shift from a low risk state to a higher one. Catastrophes are defined as instantaneous declines in population size due to events that occur randomly in time, in contrast to regular environmental variation, which occurs constantly and can have both positive and negative effects on the population. Lindley et al. (2007) view catastrophes as singular events with an identifiable cause and only negative immediate consequences, as opposed to normal environmental variation which can produce very good as well as very bad conditions. Some examples of catastrophes include disease outbreaks, toxic spills, or volcanic eruptions. A high risk situation is created by a 90 percent decline in population size over one generation. A moderate risk event is one that is smaller but biologically significant, such as a year-class failure.

5.1.7.2.4 Extinction Risk Criteria and Application

Lindley et al. (2007) characterized the spring-run Chinook salmon population in the lower Yuba River as data deficient, and therefore did not characterize its viability. In 2007, there was limited information on the current population size of spring-run Chinook salmon in the lower Yuba River. NMFS' *5 Year Review: Summary and Evaluation of Central Valley Spring-run Chinook Salmon Evoluationarily Significant Unit* (NMFS 2016b) reported that the annual spawning run size of spring-run Chinook salmon in the lower Yuba River generally ranges from a few hundred to a few thousand fish with the annual trend closely following the annual abundance trend of the FRFH spring-run Chinook salmon population. NMFS (2016) concluded that the Yuba River spring-run Chinook salmon population satisfies the low extinction risk criteria for abundance, but likely falls into the high risk category for hatchery influence.

Criteria to assess extinction risk of Pacific salmonids are based on population size, recent population decline, occurrences of catastrophes within the last 10 years, and the impacts of hatchery influence (Lindley et al. 2007). As previously discussed, for the last 3 consecutive years, an estimated total of 5,650 phenotypic spring-run Chinook salmon have passed upstream of Daguerre Point Dam, with an average of 1,883 fish per year. Catastrophes have not occurred in the Yuba River Basin, nor have catastrophic declines been observed within the phenotypic spring-run Chinook salmon abundance estimates within the last 10 years. The abundance of phenotypic spring-run Chinook salmon in the lower Yuba River has exhibited a very slight increase over the 12 years examined, although the positive trend is not statistically significant. These abundance and trend considerations would correspond to low extinction risk according to NMFS criteria (Lindley et al. 2007). However, the estimated number of spring-run Chinook salmon passing Daguerre Point Dam during 2015 was the lowest of all 12 years of recorded. Moreover, the RMT (2013a) questions the applicability of any of these criteria addressing extinction risk, because they presumably apply to independent populations and, as previously discussed, lower Yuba River anadromous salmonids represent introgressive hybridization of larger Feather-Yuba river populations, with substantial contributions of hatchery-origin fish to the annual runs. For additional discussion, see RMT (2013a).

Although straying of FRFH-origin Chinook salmon into the lower Yuba River occurs, available information indicates that: 1) the FRFH spring-run Chinook salmon is included in the ESU, in part because of the important role this stock may play in the recovery of spring-run Chinook salmon in the Feather River Basin, including the Yuba River (70 FR 37160); 2) the spring-run Chinook program at FRFH is an Integrated Recovery Program which seeks to aid in the recovery and conservation of Central Valley spring-run Chinook salmon (DWR 2009b); and 3) fish produced at FRFH are intended to spawn in the wild or be genetically integrated with the targeted natural population as FRFH broodstock (DWR 2009a).

5.1.8 NMFS Recovery Plan Considerations

According to NMFS (2005c) *Recommendations for the Contents of Biological Assessments and Biological Evaluations* pertaining to status of the species in the action area, a BA should:

- Identify any recovery plan implementation that is occurring in the action area, especially priority one action items from recovery plans.

The NMFS (2014a) Recovery Plan establishes three population levels to help guide recovery efforts for existing populations, referred to as Core 1, 2, and 3 populations. The NMFS Recovery Plan (pg. 76) identifies lower Yuba River spring-run Chinook salmon (and steelhead) populations below Englebright Dam as Core 2 populations. Core 2 populations meet, or have the potential to meet, the biological recovery standard for moderate risk of extinction. Core 2 populations provide increased life history diversity to the ESU and are likely to provide a buffering effect against local catastrophic occurrences that could affect other nearby populations, especially in geographic areas where the number of Core 1 populations is lowest (NMFS 2014).

Currently unoccupied areas in the Yuba River Basin upstream of Englebright Dam that are classified by NMFS (2014a) as "primary", or of top priority for reintroduction for spring-run Chinook salmon include the North Yuba and Middle Yuba rivers.

To meet recovery objectives, NMFS (2014a) recovery effort for the spring-run Chinook salmon ESU (and the steelhead DPS) includes: 1) securing extant populations by addressing stressors; and 2) reintroducing populations into historically occupied or other suitable areas (Lindley et al. 2007).

The NMFS (2014a) Recovery Plan states that Yuba River Recovery Actions involving habitat restoration actions include the following:

- Modify Daguerre Point Dam to provide unobstructed volitional upstream passage of adult steelhead and Chinook salmon (and sturgeon) and to minimize predation of juveniles moving downstream.
- Improve spawning habitat in the Englebright Dam Reach (Englebright Dam [RM 24] downstream to the Deer Creek confluence [RM 23]) through habitat rehabilitation and a long-term gravel injection program (Pasternack 2009).
- Develop programs and implement projects that promote natural river processes, including projects that add riparian habitat and instream cover.
- Federal, State, and local Agencies should use their authorities to develop and implement programs and projects that focus on retaining, restoring and creating river riparian corridors within their jurisdiction in the Yuba River Basin.
- Develop and implement a large woody material restoration program along the lower Yuba River utilizing sources of wood that enter upstream reservoirs.
- Implement flow fluctuation and ramping rates found to be protective of embryos and juveniles.

- Increase floodplain habitat availability in the lower Yuba River.
- Create and restore side channel habitats to increase the quantity and quality of off-channel rearing and spawning areas in the Yuba River.
- Implement programs and measures designed to minimize predation by non-native fish in the Yuba River, including harvest management techniques and programs for non-native predators (e.g., striped bass, largemouth bass, and smallmouth bass).
- Utilize biotechnical techniques that integrate riparian restoration for river bank stabilization instead of conventional rip rap in the Yuba River.

The NMFS (2014a) Recovery Plan includes Priority 1, Priority 2 and Priority 3 recovery actions, which are characterized as follows.

> Priority 1 – An action that addresses the most important threats within an area (e.g., Pacific Ocean or Delta) or watershed
>
> Priority 2 – An action that addresses threats of moderate importance
>
> Priority 3 – All other actions of lower importance to implement[6]

The NMFS (2014a) Recovery Plan (pg. 253) identifies the following proposed action as a Priority 1 recovery action for the Yuba River:

> **Recovery Action YUR-1.1** Develop and implement a program to reintroduce spring-run Chinook salmon and steelhead to historic habitats upstream Englebright Dam. The program should include:
>
> - Feasibility studies
> - Habitat evaluations
> - Fish passage design studies
> - A pilot reintroduction phase prior to implementation of the long-term reintroduction program
> - Implement long-term fish passage program

NMFS (2014a) prioritized the upper Yuba River (upstream of Englebright Dam) as a primary area to re-establish viable populations of spring-run Chinook salmon and steelhead based on four reasons, which are as follows.

[6] In NMFS' Public Draft Recovery Plan (2009a) for the Evolutionarily Significant Units of Sacramento Winter-run Chinook Salmon and Central Valley Spring-run Chinook Salmon and the Distinct Population Segment of Central Valley Steelhead, October 2009, Appendix C, NMFS described how they applied the recovery action priorities 1-3 described in NMFS recovery planning guidelines (55 FR 24296; June 15, 1990), which are also described in NMFS' Recovery Planning Guidance (NMFS 2010b), in developing recovery actions for each species addressed in the recovery plan. The recovery actions priorities 1-3 described in the final recovery plan are based on grouping the recovery actions for all three listed species addressed in the recovery plan by area or watershed and prioritizing those actions.

> "*First, spring-run Chinook salmon and steelhead historically occurred there (Lindley et al. 2004; Yoshiyama et al. 1996) and studies suggest that multiple areas in the upper river would currently still support those species (DWR 2007; Stillwater Sciences 2012). Second, evidence suggests that significant amounts of summer holding habitat in the upper Yuba River are expected to remain thermally suitable for spring-run Chinook salmon throughout the 21st century even if the climate warms by as much as 5°C (Lindley et al. 2007). That expectation of thermally suitable habitat in the upper Yuba River watershed in the face of climate change is based on a simple analysis of air temperatures and did not account for the presence of New Bullard's Bar Reservoir, a deep, steep-sloped reservoir with ample coldwater pool reserves that could be used to provide suitable flows and water temperatures in the upper watershed downstream of the reservoir in perpetuity. The coldwater pool in New Bullards Bar Reservoir has never been depleted, even during the most extreme critically dry year on record (1977) (YCWA 2010). Third, there is considerable distance between the Yuba River watershed and the cluster of watersheds in the diversity group that currently support wild spring-run Chinook salmon. This spatial isolation is important because if one or more spring-run Chinook salmon populations were established in the upper Yuba River watershed, those populations would not be at risk if there was a volcanic eruption at Mt. Lassen, a volcano that the USGS views as highly dangerous. In contrast, all three extant independent populations (Mill, Deer, and Butte creeks) of spring-run Chinook salmon are in basins whose headwaters occur within the debris and pyroclastic flow radii of Mt. Lassen. Even wildfires, which are of much smaller scale than large volcanic eruptions, pose a significant threat to the spring-run Chinook salmon ESU in its current configuration. A fire large enough to burn the headwaters of Mill, Deer and Butte creeks simultaneously, has roughly a 10% chance of occurring somewhere in the Central Valley each year (Lindley et al. 2007). Lastly, the Yuba River watershed has an ample supply of water to support spring-run Chinook salmon and steelhead with one of the highest annual discharges (~2,300,000 acre-feet/year) in the Central Valley (Lindley et al. 2004).*"

The NMFS (2014a) Recovery Plan (pg. 253-254) identifies the following other proposed actions as a Priority 1 recovery action for the Yuba River:

> **Recovery Action YUR-1.2** Improve spawning habitat in the Englebright Dam Reach (Englebright Dam [RM 24] downstream to the Deer Creek confluence[RM 23]) through habitat rehabilitation and a long-term gravel injection Program (Pasternack 2009).

> **Recovery Action YUR-1.3.** Develop programs and implement projects that promote natural river processes, Including projects that add riparian habitat and instream cover.

> **Recovery Action YUR-1.4.** Modify Daguerre Point Dam to provide unobstructed Volitional upstream passage of adult steelhead and Chinook salmon (and sturgeon) and to minimize predation of juveniles moving downstream.

Also, lower Yuba River large woody material and floodplain habitat availability considerations are discussed as Priority 2 actions on pgs. 254 to 256 in NMFS (2014a).

Of the proposed recovery actions regarding juvenile rearing, the actions that would be most beneficial and cost-effective, and the actions that would yield the most immediate benefits, are the creation of new side-channel habitats associated with existing stands of riparian vegetation that are not presently hydraulically connected to the river channel (YCWA 2010). Specifically, new side-channel habitats would: 1) increase and maintain existing riparian vegetation; 2) provide instream object and overhanging object cover; 3) provide new SRA, and associated allochthonous food sources for rearing juveniles; 4) increase aquatic habitat complexity and diversity; 5) provide habitats more consistent with those previously available in the upper watershed; and 6) provide predator escape cover, and overall increased survival of juvenile spring-run Chinook salmon and steelhead.

NMFS (2014a) Recovery Plan (pg. iv) states "*As this Recovery Plan is implemented over time, additional information will become available to help determine the degree to which the threats have been abated, to further develop understanding of the linkages between threats and population responses, to identify any additional threats and to evaluate the viability of Chinook salmon and steelhead in the Central Valley.*"

The NMFS (2014a) Recovery Plan (pg. 362) states that it may not be necessary to reintroduce fish to all of the listed river and creek systems to meet the recovery criteria for Central Valley spring-run Chinook salmon [and steelhead]. "*…it is important to note that it is not necessary to reestablish populations in all of these watersheds to meet the recovery criteria for CV spring-run Chinook salmon or CV steelhead. In fact, successful reintroductions into just a few areas will allow the recovery criteria to be met*." NMFS (2014a, pg. 86) further states "*Primary areas for spring-run Chinook salmon re-introduction into historic habitat include upstream of Shasta Dam in the Basalt diversity group and the Yuba River above Englebright Dam in the Northern Sierra Nevada*."

5.2 Central Valley Steelhead DPS

5.2.1 ESA Listing Status

On March 19, 1998 (63 FR 13347) NMFS listed the California Central Valley steelhead ESU as "threatened", concluding that the risks to Central Valley steelhead had diminished since the completion of the 1996 status review based on a review of existing and recently implemented state conservation efforts and federal management programs (e.g., Central Valley Project Improvement Act (CVPIA), Anadromous Fish Restoration Program (AFRP), CALFED Bay-Delta Program (CALFED)) that address key factors for the decline of this species. The

California Central Valley steelhead ESU included all naturally spawned populations of steelhead in the Sacramento and San Joaquin rivers and their tributaries, but excluded steelhead from the tributaries of San Francisco and San Pablo bays (NMFS 2004d).

On June 14, 2004, NMFS proposed listing determinations for 27 ESUs of West Coast salmon and *O. mykiss*, including the California Central Valley steelhead ESU. In the proposed rule, NMFS concluded that steelhead were not in danger of extinction, but were likely to become endangered within the foreseeable future throughout all or a significant portion of their range and, thus, proposed that steelhead remain listed as threatened under the ESA. Steelhead from the Coleman National Fish Hatchery and the FRFH, as well as resident populations of *O. mykiss* (rainbow trout) below impassible barriers that co-occur with anadromous populations, were included in the California Central Valley steelhead ESU and, therefore, also were included in the proposed listing.

During the 2004 comment period on the proposed listings, the USFWS provided comments that the USFWS does not use NMFS' ESU policy in any USFWS ESA listing decisions. As a result of the comments received, NMFS re-opened the comment period to receive comments on a proposed alternative approach to delineating ''species'' of West Coast *O. mykiss* (70 FR 67130). NMFS proposed to depart from past practice of applying the ESU Policy to *O. mykiss* stocks, and instead proposed to apply the DPS Policy in determining ''species'' of *O. mykiss* for listing consideration. NMFS noted that within a discrete group of *O. mykiss* populations, the resident and anadromous life forms of *O. mykiss* remain ''markedly separated'' as a consequence of physical, physiological, ecological, and behavioral factors, and may therefore warrant delineation as separate DPSs (71 FR 834).

NMFS issued a policy for delineating DPSs of Pacific salmon in 1991 (56 FR 58612; November 20, 1991). Under this policy, a group of Pacific salmon populations is considered an ''ESU'' if it is substantially reproductively isolated from other conspecific populations, and it represents an important component in the evolutionary legacy of the biological species. Further, an ESU is considered to be a ''DPS'' (and thus a ''species'') under the ESA. In 1996, NMFS and USFWS adopted a joint policy for recognizing DPSs under the ESA (DPS Policy; 61 FR 4722; February 7, 1996). The DPS Policy adopted criteria similar to, but somewhat different from, those in the ESU Policy for determining when a group of vertebrates constitutes a DPS – The group must be discrete from other populations, and it must be significant to its taxon. A group of organisms is discrete if it is ''*markedly separated from other populations of the same taxon as a consequence of physical, physiological, ecological, and behavioral factors*.'' Significance is measured with respect to the taxon (species or subspecies) as opposed to the full species (71 FR 834). Although the ESU Policy did not by its terms apply to steelhead, the DPS Policy stated that NMFS will continue to implement the ESU Policy with respect to ''Pacific salmonids'' (which included *O. mykiss*). In a previous instance of shared jurisdiction over a species (Atlantic salmon), NMFS and USFWS used the DPS Policy in their determination to list the Gulf of Maine DPS of Atlantic salmon as endangered (65 FR 69459; November 17, 2000).

Given NMFS and USFWS shared jurisdiction over *O. mykiss*, and consistent with joint NMFS and USFWS approaches for Atlantic salmon, it was concluded that application of the joint DPS policy was logical, reasonable, and appropriate for identifying DPSs of *O. mykiss* (71 FR 834).

Moreover, NMFS determined that use of the ESU policy – originally intended for Pacific salmon – should not continue to be extended to *O. mykiss*, a type of salmonid with characteristics not typically exhibited by Pacific salmon (71 FR 834).

On January 5, 2006 NMFS issued a final decision that defined Central Valley steelhead as a DPS rather than an ESU, and retained the status of Central Valley steelhead as threatened (71 FR 834). The DPS includes all naturally spawned anadromous *O. mykiss* (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 (63 FR 13347). Steelhead in two artificial propagation programs – the Coleman National Fish Hatchery, and FRFH steelhead hatchery programs are considered to be part of the DPS. NMFS determined that these artificially propagated stocks are no more divergent relative to the local natural population(s) than what would be expected between closely related natural populations within the DPS (71 FR 834).

As previously discussed, the ESA requires that NMFS review the status of listed species under its authority at least every 5 years and determine whether any species should be removed from the list or have its listing status changed. In May 2016, NMFS completed a 5-year status review of the Central Valley steelhead DPS. Based upon a review of available information, NMFS (2016a) recommended that the Central Valley steelhead DPS remain classified as a threatened species. However, NMFS (2016a) also indicated that the biological status of the DPS has declined since the previous status review in 2011. According to NMFS (2016a), there are indications that natural production of steelhead continues to decline and is now at a very low level. Their continued low numbers in most hatcheries, domination by hatchery fish, and relatively sparse monitoring makes the continued existence of naturally reproduced steelhead a concern. Due to this declining trend, NMFS (2016a) suggests that the DPS is likely to become endangered within the foreseeable future throughout all or a significant portion of its range.

Based on new genetic evidence described by Pearse and Garza (2015), NMFS recommend that steelhead originating from the Mokelumne River Hatchery be added to the Central Valley steelhead DPS in the same manner as FRFH fish are considered to be a native Central Valley stock and are listed as part of the DPS. NMFS (2016a) also recommended that the status of the DPS should be monitored and Hatchery and Genetic Management Plans should mandate that all Central Valley steelhead hatcheries collect a full set of biological data, including scale samples, length, weight, sex, origin, and state of maturity, from a subset of all returning fish. Hatcheries also should be required to conduct studies of smolt survival using modern tagging methods such as PIT tags and/or acoustic tags.

5.2.2 Critical Habitat Designation

On February 16, 2000 (65 FR 7764), NMFS published a final rule designating critical habitat for Central Valley steelhead. This critical habitat includes all river reaches accessible to listed steelhead in the Sacramento and San Joaquin rivers and their tributaries in California, including the lower Yuba River upstream to Englebright Dam. NMFS proposed new critical habitat for spring-run Chinook salmon and Central Valley steelhead on December 10, 2004 (69 FR 71880) and published a final rule designating critical habitat for these species on September 2, 2005.

This critical habitat includes the Yuba River (70 FR 52488) from the confluence with the lower Feather River upstream to Englebright Dam.

5.2.2.1 Physical or Biological Features

The 2005 critical habitat designation (70 FR 52488) lists PCEs, which are physical or biological elements essential for the conservation of the listed species. The PCEs include sites essential to support one or more lifestages of the DPS (sites for spawning, rearing, migration, and foraging). The specific PCEs include:

- Freshwater spawning sites
- Freshwater rearing sites
- Freshwater migration corridors
- Estuarine areas
- Nearshore marine areas
- Offshore marine areas

The most recent discussion of PCEs in the Central Valley is in the CVP/SWP OCAP BO (NMFS 2009b). The following summary descriptions of the current conditions of the PCEs for the Central Valley steelhead DPS were taken from NMFS (2009b).

As previously discussed, the regulations regarding critical habitat were recently revised to remove the terms ‘‘principal biological or physical constituent elements’’ and ‘‘primary constituent elements’’ from 50 C.F.R. 424.12(b). These concepts were replaced by the statutory term ‘‘physical or biological features” (PBFs) (81 FR 7432, February 11, 2016). As described by NMFS (2016b; 2016c), this is a shift in terminology only and does not change the categories of such features (i.e., freshwater rearing habitat or freshwater migration corridors) or the approach used in conducting an effects analysis, which is the same regardless of whether the original designation identified primary constituent elements, physical or biological features, or essential features. Therefore, in this Applicant-Prepared Draft BA, the term PBF is used to mean PCE or essential feature, as appropriate for the specific critical habitat.

5.2.2.1.1 Freshwater Spawning Habitat

According to NMFS (2009b), steelhead in the Sacramento River spawn primarily between Keswick Dam and RBDD during the winter and spring. The highest density spawning area is likely in the upstream portion of this area in the vicinity of the city of Redding, although detailed surveys of steelhead spawning in the mainstem Sacramento River are not available. Most Sacramento River steelhead probably spawn in the tributary streams. Steelhead spawn in Clear Creek mostly within a couple miles of Whiskeytown Dam but spawning extends for about 10 mi downstream of the dam (M. Brown, pers. comm. as cited in Reclamation 2008a). Steelhead spawn in the Feather River from the Fish Barrier Dam downstream to Gridley with nearly 50 percent of all spawning occurring within the upper mile of the low flow channel (DWR 2003b).

5.2.2.1.2 Freshwater Rearing Habitat

Juvenile steelhead reside in freshwater for a year or more, so they are more dependent on freshwater rearing habitat than are the ocean type Chinook salmon in the Central Valley. Steelhead rearing occurs primarily in the upstream reaches of the rivers where channel gradients tend to be higher and, during the warm weather months, where temperatures are maintained at more suitable levels by cool water dam releases. The Sacramento River contains a long reach of suitable water temperatures even during the heat of the summer. Steelhead rearing in the Sacramento River occurs mostly between Keswick Dam (RM 302) and Butte City (RM 169) with the highest densities likely to be upstream of RBDD. Steelhead rearing in Clear Creek is concentrated in the upper river higher gradient areas but probably occurs down to the mouth. Steelhead rearing in the Feather River is concentrated in the low flow channel where temperatures are most suitable (DWR 2004).

5.2.2.1.3 Freshwater Migration Corridors

Steelhead migrate during the winter and spring of the year, as juveniles, from the rearing areas described above downstream through the rivers and the Delta to the ocean. The habitat conditions they encounter during migration from the upstream reaches of the rivers downstream to the Delta generally become less suitable as fish move away from their natal streams until they reach the ocean. The generally non-turbulent flows and sand substrates found in the lower river reaches are not preferred types of habitat, so steelhead do not likely reside for extended periods in these areas except when food supplies, such as smaller young fish, are abundant and temperatures are suitable. Predatory fishes such as striped bass tend to be more abundant in the lower rivers and the Delta.

Adult steelhead migrate upstream from the ocean to their spawning grounds near the terminal dams primarily during the fall and winter months. Flows are generally lower during the upstream migrations than during the outmigration period. Areas where their upstream progress can be affected are the DCC Gates and ACID Diversion Dam.

5.2.2.1.4 Estuarine Habitat Areas

Steelhead use the San Francisco estuary as a rearing area and migration corridor between their upstream rearing habitat and the ocean. The San Francisco Bay estuarine system includes the waters of San Francisco Bay, San Pablo Bay, Grizzly Bay, Suisun Bay, Honker Bay, and can extend as far upstream as Sherman Island during dry periods. At times, steelhead likely remain for extended periods in areas of suitable habitat quality where food such as young herring, salmon and other fish and invertebrates is available.

5.2.2.1.5 Nearshore Coastal Marine and Offshore Marine Areas

The most recent discussion of PBFs for the Central Valley steelhead DPS (NMFS 2009b) did not include the PBFs of nearshore coastal marine and offshore marine areas. Although relatively little is known about steelhead utilization of nearshore coastal marine and offshore marine areas, it is reasonable to assume that the discussion of these PBFs previously provided for spring-run

Chinook salmon in Section 5.1 of this Applicant-Prepared Draft BA generally is applicable to steelhead.

5.2.3 Historical Distribution and Abundance

According to NMFS (2014a), steelhead historically occurred naturally throughout the Sacramento and San Joaquin River basins, although stocks have been extirpated from large areas in both basins. The California Advisory Committee on Salmon and Steelhead (CDFG 1988a) reported a reduction in Central Valley steelhead habitat from 6,000 mi historically to 300 mi of rivers and streams.

NMFS (2014a) reported that prior to dam construction, water development and watershed perturbations, Central Valley steelhead were distributed throughout the Sacramento and San Joaquin rivers (Busby et al. 1996; McEwan 2001). Steelhead were found from the upper Sacramento and Pit rivers (now inaccessible due to Shasta and Keswick dams) south to the Kings and possibly the Kern River systems, and in both east- and west-side Sacramento River tributaries (Yoshiyama et al. 1996). Lindley et al. (2006) estimated that historically there were at least 81 independent Central Valley steelhead populations distributed primarily throughout the eastern tributaries of the Sacramento and San Joaquin rivers. Presently, impassable dams block access to 80 percent of historically available habitat, and block access to all historical spawning habitats for about 38 percent of historical populations (Lindley et al. 2006). Existing wild steelhead stocks in the Central Valley are mostly confined to the upper Sacramento River and its tributaries, including Antelope, Deer, and Mill creeks, and the Yuba River. Populations may exist in Big Chico and Butte creeks, and a few wild steelhead are produced in the American and Feather rivers (McEwan 2001).

Steelhead were previously thought to be extirpated from the San Joaquin River system. However, monitoring activities have detected small self-sustaining populations of steelhead in the Stanislaus, Mokelumne, Merced, and Calaveras rivers, and other streams previously thought to be devoid of steelhead (McEwan 2001; NMFS 2014).

It is possible that naturally spawning populations exist in many other streams but are undetected due to lack of monitoring programs (IEP Steelhead Project Work Team 1999 as cited in NMFS 2009a). Incidental catches and observations of steelhead juveniles also have occurred on the Tuolumne and Merced rivers during fall-run Chinook salmon monitoring activities, indicating that steelhead are widespread, throughout accessible streams and rivers in the Central Valley (Good et al. 2005). Naturally spawning populations of steelhead also occur in the Feather, Yuba, American, and Mokelumne rivers, but these populations have had substantial hatchery influence and their ancestries are not clear (Busby et al. 1996). Steelhead runs in the Feather and American rivers are sustained largely by the FRFH and Nimbus Hatchery, respectively (McEwan and Jackson 1996).

Historical Central Valley steelhead run sizes are difficult to estimate because of the lack of data, but McEwan (2001) suggested that steelhead run sizes may have approached one to two million adults annually. McEwan and Jackson (1996) suggested that by the early 1960s, the steelhead run size had declined to about 40,000. Over the last 30 years the steelhead populations in the upper Sacramento River have declined substantially (NMFS 2009a). In 1996, NMFS estimated the Central Valley total run size based on dam counts, hatchery returns, and past spawning surveys was probably fewer than 10,000 fish. Both natural and hatchery runs have declined since the 1960s. Counts at RBDD averaged 1,400 fish from 1991 to 1996, compared to counts in excess of 10,000 fish in the late 1960s (McEwan and Jackson 1996).

Specific information regarding steelhead spawning within the mainstem Sacramento River is limited due to lack of monitoring (NMFS 2004c). Currently, the number of steelhead spawning in the Sacramento River is unknown because redds cannot be distinguished from a large resident rainbow trout population that has developed as a result of managing the upper Sacramento River for coldwater species.

The lack of sustained monitoring programs for steelhead throughout most of the Central Valley persists to the present time. There is a paucity of reliable data to estimate run sizes of steelhead in the Central Valley, particularly wild stocks. However, some steelhead escapement monitoring surveys have been initiated in upper Sacramento River tributaries (e.g., Beegum, Deer, and Antelope Creeks) using snorkel methods similar to spring-run Chinook escapement surveys (NMFS 2009b).

There is a general lack of steelhead population monitoring in most of the Central Valley (NMFS 2009b). Lindley et al. (2007) stated that there are almost no data with which to assess the status of any of the Central Valley steelhead populations. They further stated that Central Valley steelhead populations are classified as data deficient, with the exceptions restricted to streams with long-running hatchery programs including Battle Creek and the Feather, American and Mokelumne rivers. According to NMFS (2007), in the *Updated Status Review of West Coast Salmon and Steelhead* (Good et al. 2005), the Biological Review Team made the following conclusion based on steelhead Chipps Island trawl data:

> If we make the fairly generous assumptions (in the sense of generating large estimates of spawners) that average fecundity is 5,000 eggs per female, 1 percent of eggs survive to reach Chipps Island, and 181,000 smolts are produced (the 1998-2000 average), about 3,628 female steelhead spawn naturally in the entire Central Valley.

In classifying the Central Valley steelhead DPS as threatened in 1998 (63 FR 13347), NMFS referenced the lack of monitoring data for most populations in the DPS as a cause for concern. In response to these concerns, the Cal Fish and Wildlife, with funding from the Reclamation and DWR, has written a detailed monitoring plan for Central Valley steelhead, with a focus on estimating adult escapement in the Sacramento River and its major tributaries (Fortier et al. 2014). Cal Fish and Wildlife began capturing and tagging adult steelhead in the Sacramento River during the fall of 2015. When fully implemented, this monitoring plan will provide steelhead abundance data for several watersheds in the Central Valley, and eventually allow for the long-term tracking of populations in a way that currently exists for the three species of Chinook salmon in the Central Valley (NMFS 2016c).

In the Yuba River, definitive historic population estimates do not exist for steelhead, but it is likely that the river supported large steelhead runs in the 1800s (USFWS 1995b). McEwan and Jackson (1996) reported that the Yuba River historically supported the largest, naturally reproducing, persistent population of steelhead in the Central Valley.

Prior to construction of Englebright Dam in 1941, Cal Fish and Wildlife fisheries biologists stated that they observed large numbers of steelhead spawning in the uppermost reaches of the Yuba River and its tributaries (CDFG 1998; Yoshiyama et al. 1996). After construction of Englebright Dam in 1941, Cal Fish and Wildlife estimated that only approximately 200 steelhead spawned in the lower Yuba River annually before New Bullards Bar Reservoir was completed in 1969. From 1970 to 1979, Cal Fish and Wildlife annually stocked 27,270–217,378 fingerlings, yearlings, and sub-catchables from Coleman National Fish Hatchery into the lower Yuba River (CDFG 1991b). Cal Fish and Wildlife stopped stocking steelhead into the lower Yuba River in 1979. Based on angling data, Cal Fish and Wildlife estimated a run size of 2,000 steelhead in the lower Yuba River in 1975 (CDFG 1991b). McEwan and Jackson (1996) reported that, as of 1996, the status of the lower Yuba River steelhead population was unknown, but it appeared to be stable and able to support a significant sport fishery. Cal Fish and Wildlife currently manages the river to protect natural steelhead through strict "catch-and-release" fishing regulations.

5.2.4 General Life History and Habitat Requirements

Steelhead exhibits perhaps the most complex suite of life-history traits of any species of Pacific salmonid. Members of this species can be anadromous or freshwater residents and, under some circumstances, members of one form can apparently yield offspring of another form (YCWA 2010).

“Steelhead” is the name commonly applied to the anadromous form of the biological species *O. mykiss*. The physical appearance of *O. mykiss* adults and the presence of seasonal runs and year-round residents indicate that both anadromous (steelhead) and resident rainbow trout exist in the lower Yuba River downstream of Englebright Dam, although no definitive visual characteristics have been identified to distinguish young steelhead from resident trout (SWRI et al. 2000). Zimmerman et al. (2009) analyzed otolith strontium:calcium (Sr:Ca) ratios in 964 otolith samples comprised of YOY, age-1, age-2, age-3, and age-4+ fish to determine maternal origin and migratory history (anadromous vs. non-anadromous) of *O. mykiss* collected in Central Valley rivers between 2001 and 2007, including the lower Yuba River.

The proportion of steelhead progeny in the lower Yuba River (about 13%) was intermediate to the other rivers examined (Sacramento, Deer Creek, Calaveras, Stanislaus, Tuolumne, and Merced), which ranged from about 4 percent in the Merced River to 74 percent in Deer Creek (Zimmerman et al. 2009). Results from Mitchell (2010) indicate *O. mykiss* in the lower Yuba River are exhibiting a predominately residential life history pattern. He found that 14 percent of scale samples gathered from 71 *O. mykiss* moving upstream and trapped in the fish ladder at Daguerre Point Dam from November 1, 2000, through March 28, 2001, exhibited an anadromous life history. Thus, it is recognized that both anadromous and resident life history strategies of *O. mykiss* have been and continue to be present in the lower Yuba River**.**

The RMT (2013a) developed representative temporal distributions for specific steelhead lifestages in the lower Yuba River through review of previously conducted studies, as well as recent and currently ongoing data collection activities of the M&E Program. As with spring-run Chinook salmon, the resultant lifestage periodicities are intended to encompass the majority of activity for a particular lifestage, and are not intended to be inclusive of every individual in the population. The lifestage-specific periodicities for steelhead in the lower Yuba River are summarized in Table 5.2-1, and are discussed below.

Table 5.2-1. Lifestage-specific periodicities for steelhead in the lower Yuba River.

Lifestage	Jan		Feb		Mar		Apr		May		Jun		Jul		Aug		Sep		Oct		Nov		Dec	
Adult Immigration & Holding	■	■	■	■	■	■									■	■	■	■	■	■	■	■	■	■
Spawning	■	■	■	■	■	■	■	■																
Embryo Incubation	■	■	■	■	■	■	■	■	■	■														
Fry Rearing							■	■	■	■	■	■	■	■										
Juvenile Rearing	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■
Juvenile Downstream Movement							■	■	■	■	■	■	■	■	■	■	■	■						
Smolt (Yearling+) Emigration	■	■	■	■	■	■	■												■	■	■	■	■	■

Source: RMT 2013a

5.2.4.1 Adult Immigration and Holding

Adult migration from the ocean to spawning grounds occurs during much of the year, with peak migration occurring in the fall or early winter. Central Valley steelhead are known to use the Sacramento River as a migration corridor to spawning areas in upstream tributaries. Historically, steelhead likely did not utilize the mainstem Sacramento River downstream from the present location of Shasta Dam, except as a migration corridor to and from headwater streams (NMFS 2014).

Migration through the Sacramento River mainstem begins in July, peaks at the end of September, and continues through February or March (Bailey 1954; Hallock et al. 1961 both as cited in McEwan and Jackson 1996). Counts made at RBDD from 1969 through 1982 (Hallock 1989 as cited in McEwan and Jackson 1996) and on the Feather River (Painter et al. 1977) follow the above pattern, although some fish were counted as late as April and May. Weekly counts at Clough Dam on Mill Creek during a 10-year period from 1953 to 1963 showed a similar migration pattern as well, with a peak in migration during mid-November and another peak during February (NMFS 2009b). This second peak is not reflected in counts made in the Sacramento River mainstem (Bailey 1954; Hallock et al. 1961; both as cited in McEwan and Jackson 1996) or at RBDD (Hallock 1989 as cited in McEwan and Jackson 1996).

According to NMFS (2009b), Central Valley steelhead are mostly 'winter steelhead' and may contain some 'summer steelhead' (the naming convention refers to the seasonal period of adult upstream migration). Winter steelhead mature in the ocean and arrive on the spawning grounds nearly ready to spawn, whereas summer steelhead enter freshwater with immature gonads and typically spend several months in freshwater before spawning. The reported minimum depth for successful passage is about 7 in (Reiser and Bjornn 1979 as cited in McEwan and Jackson 1996). Excessive water velocity (>10 to 13 ft/s) and obstacles may prevent access to upstream spawning grounds (NMFS 2009b).

The optimal temperature range during adult upstream migration is unknown for Central Valley steelhead stocks (NMFS 2009b). Prolonged exposure to water temperatures above 73°F is reported to be lethal to adult steelhead (Moyle 2002). Based on northern stocks, the optimal temperature range for migrating adult steelhead is 46 to 52°F (Bovee 1978; Reiser and Bjornn 1979; Bell 1986; all as cited in McEwan and Jackson 1996).

The immigration of adult steelhead in the lower Yuba River has been reported to occur from August through March, with peak immigration from October through February (CALFED and YCWA 2005; McEwan and Jackson 1996). Cal Fish and Wildlife (1984) reported that during the drought years of 1976-1977, two steelhead immigration peaks were observed – one in October and one in February. CDFG (1991b) reported that steelhead enter the lower Yuba River as early as August, migration peaks in October through February, and may extend through March. In addition, they report that a run of “half-pounder” steelhead occurred from late-June through the winter months.

RMT (2010b) examined preliminary data and identified variable annual timing of *O. mykiss* ascending the fish ladders at Daguerre Point Dam since the VAKI Riverwatcher™ infrared and videographic sampling system began operations in 2003. For example, Massa et al. (2010) state that peak passage of steelhead at Daguerre Point Dam occurred from April through June during 2007. They also suggest that the apparent disparity between the preliminary data and other reports of steelhead adult immigration periodicity may be explained by the previously reported (Zimmerman et al. 2009; Mitchell 2010) relatively high proportion of resident (vs. anadromous) *O. mykiss* occurring in the lower Yuba River, because the VAKI Riverwatcher™ system did document larger (>40.6 centimeters (cm)) *O. mykiss* ascending the fish ladders at Daguerre Point Dam during the winter months (December through February). The observed timing of larger *O. mykiss* ascending the fish ladders at Daguerre Point Dam more closely corresponds with previously reported adult steelhead immigration periodicities. RMT (2010b; 2013a) identified the period extending from August through March as encompassing the majority of the upstream migration and holding of adult steelhead in the lower Yuba River.

5.2.4.2 Adult Spawning

Central Valley adult steelhead generally begin spawning in late December and spawning extends through March, but also can range from November through April (CDFG 1986). Steelhead adults typically spawn from December through April with peaks from January through March in small streams and tributaries where cool, well oxygenated water is available year-round (Hallock et al. 1961; McEwan 2001). Based on all available information collected to date, the RMT (2013a) identified the steelhead spawning period as extending from January through April.

Central Valley steelhead spawn downstream of dams on every major tributary within the Sacramento and San Joaquin River systems. Due to water development projects, most spawning is now confined to lower stream reaches below dams. In a few streams, such as Mill and Deer creeks, steelhead still have access to historical spawning areas (NMFS 2009b).

The female steelhead selects a site with good intergravel flow, digs a redd with her tail, usually in the coarse gravel of the tail of a pool or in a riffle, and deposits eggs while an attendant male fertilizes them (NMFS 2009a). Spawning occurs mainly in gravel substrates (particle size range of about 0.2−4.0 in). Sand-gravel and gravel-cobble substrates are also used, but these must be highly permeable and contain less than 5 percent sand and silt for the water to be able to provide sufficient oxygen to the incubating eggs. Adults tend to spawn in shallow areas (6−24 in deep) with moderate water velocities (about 1 to 3.6 ft/s) (Bovee 1978 as cited in McEwan and Jackson 1996; Hannon and Deason 2008 as cited in Reclamation 2008a). The optimal temperature range for spawning has been reported to range from 39° to 52°F (Bovee 1978; Reiser and Bjornn 1979; Bell 1986 all as cited in McEwan and Jackson 1996). Egg mortality begins to occur at 56°F (McEwan and Jackson 1996).

Unlike Chinook salmon, Central Valley steelhead may not die after spawning (McEwan and Jackson 1996). Some may return to the ocean and repeat the spawning cycle for 2 or 3 years. The percentage of adults surviving spawning is generally thought to be low for Central Valley steelhead, but varies annually and between stocks. Acoustic tagging of Central Valley steelhead kelts from the Coleman Hatchery indicates survival rates can be high, especially for Central Valley steelhead reconditioned by holding and feeding at the hatchery prior to release. Some return immediately to the ocean and some remain and rear in the Sacramento River (NMFS 2009b).

Steelhead spawning has been reported to generally extend from January through April in the lower Yuba River (CALFED and YCWA 2005; CDFG 1991b; YCWA et al. 2007). The RMT conducted a pilot redd survey from September 2008 through April 2009 (RMT 2010c). Surveys were not conducted during March, which is a known time for steelhead spawning in other Central Valley rivers, due to high flows and turbidity. An extensive area redd survey was conducted by surveyors kayaking from the downstream end of the Narrows pool to the Simpson Lane Bridge. During the extensive area redd survey, redds that were categorized as steelhead based on redd size criteria were reportedly observed from October through April. However, some of those redds categorized as steelhead, particularly during October, may actually have been small Chinook salmon redds because the size criteria used to identify steelhead redds was found to be 53 percent accurate for identifying steelhead redds in the American River (USFWS 2010).

Campos and Massa (2011) synthesized results of near-census redd surveys conducted on the lower Yuba River during the 2009 and 2010 survey periods. During both annual survey efforts, a substantial proportion of the weekly strata in the January through April time periods were not sampled due to elevated flows and associated turbidity levels. Nonetheless, RMT (2013a) demonstrated that based upon cumulative temporal distribution curves, the steelhead spawning period in the lower Yuba River is generally characterized to extend from January through April.

Steelhead spawning has been reported to primarily occur in the lower Yuba River upstream of Daguerre Point Dam (SWRI et al. 2000; YCWA et al. 2007). Kozlowski (2004) states that field observations during winter and spring 2000 (YCWA unpublished data) indicated that the majority of steelhead spawning in the lower Yuba River occurred from Long Bar upstream to the Narrows, with the highest concentration of redds observed upstream of the Highway 20 Bridge. USFWS (2007) data were collected on *O. mykiss* redds in the lower Yuba River during 2002, 2003, and 2004, with approximately 98 percent of redds located upstream of Daguerre Point Dam. Near-census redd surveys were conducted on the lower Yuba River during the 2009 and 2010 survey periods, although a substantial proportion of the weekly strata in the January through April time periods were not sampled due to elevated flows and associated turbidity levels. The numbers of redds counted each year were drastically different, although the proportions of redds in each of the survey reaches was quite similar between years. The most consistent and reliable steelhead survey year was 2010, when over 94 percent of all steelhead redds were observed upstream of Daguerre Point Dam. Female steelhead construct redds within a range of depths and velocities in suitable gravels, oftentimes in pool tailouts and heads of riffles. In the lower Yuba River, steelhead have also been observed to spawn in side channel areas (YCWA unpublished data).

5.2.4.3 Embryo Incubation

California Central Valley adult steelhead eggs incubate within the gravel and hatch from approximately 19 to 80 days at water temperatures ranging from 60°F to 40°F, respectively (NMFS 2014). After hatching, the young fish (alevins) remain in the gravel for an extra 2 to 6 weeks before emerging from the gravel and taking up residence in the shallow margins of the stream.

Steelhead embryo incubation generally occurs from December through June in the Central Valley. The RMT (2013a) identified the period of January through May as encompassing the majority of the steelhead embryo incubation period in the lower Yuba River. Following deposition of fertilized eggs in the redd, they are covered with loose gravel. Central Valley steelhead eggs can reportedly survive at water temperature ranges of 35.6°F to 59°F (Myrick and Cech 2001). Steelhead eggs reportedly have the highest survival rates at water temperature ranges of 44.6°F to 50.0°F (Myrick and Cech 2001). Studies conducted at or near 54.0°F report high survival and normal development of steelhead incubating embryos, a relatively low mortality of incubating steelhead embryos is reported to occur at 57.2°F, and a sharp decrease in survival has been reported for *O. mykiss* embryos incubated above 57.2°F (RMT 2010b).

Steelhead eggs hatch in three to four weeks at 50°F to 59°F, and fry emerge from the gravel 4 to 6 weeks later (Shapovalov and Taft 1954). Steelhead embryo development requires a constant supply of well oxygenated water. This implies a loose gravel substrate allowing high permeability, with little silt or sand deposition during the development time period. Merz et al. (2004) showed that spawning substrate quality influenced a number of physical parameters affecting egg survival including temperature, dissolved oxygen, and substrate permeability.

The entire egg incubation lifestage encompasses the time when adult steelhead spawn through the time when emergent fry exit the gravel (CALFED and YCWA 2005). In the lower Yuba River, steelhead embryo incubation generally occurs from January through May (CALFED and YCWA 2005; SWRI 2002).

5.2.4.4 Juvenile Rearing and Outmigration

As reported in NMFS (2014a), juvenile Central Valley steelhead may migrate to the ocean after spending 1 to 3 years in freshwater (McEwan and Jackson 1996). Upon emergence from the gravel, the fry move to shallow protected areas associated with the stream margin (Royal 1972; Barnhart 1986; both as cited in McEwan and Jackson 1996). Steelhead fry tend to inhabit areas with cobble-rubble substrate, a depth less than 14 in, and temperature ranging from 45° to 60°F (Bovee 1978 as cited in McEwan and Jackson 1996). Myrick (1998 as cited in Reclamation 2008a) found steelhead from the Feather and Mokelumne rivers preferred temperatures between 62.5° and 68°F.

In general, it has been reported that after emergence steelhead fry move to shallow-water, low velocity habitats, such as stream margins and low gradient riffles, and will forage in open areas lacking instream cover (Hartman 1965; Everest et al. 1986). As fry increase in size and their swimming abilities improve in late summer and fall, juvenile steelhead have been reported to increasingly use areas with cover and show a preference for higher velocity, deeper mid-channel areas near the thalweg (Hartman 1965; Everest and Chapman 1972).

Juvenile steelhead have been reported to occupy a wide range of habitats, preferring deep pools as well as higher velocity rapid and cascade habitats (Bisson et al. 1982; 1988). During the winter period of inactivity, steelhead prefer low velocity pool habitats with large rocky substrate or woody debris for cover (Hartman 1965; Swales et al. 1986; Raleigh et al. 1984). During periods of low temperatures and high flows associated with the winter months, juvenile steelhead seeks refuge in interstitial spaces in cobble and boulder substrates (Bustard and Narver 1975; Everest et al. 1986).

Older juveniles use riffles and larger juveniles may also use pools and deeper runs (Barnhart 1986 as cited in McEwan and Jackson 1996). However, specific depths and habitats used by juvenile rainbow trout can be affected by predation risk (Brown and Brasher 1995). Central Valley steelhead can show mortality at constant temperatures of 77°F although they can tolerate 85°F for short periods (Myrick and Cech 2001). Juvenile steelhead in northern California rivers reportedly exhibited increased physiological stress, increased agonistic activity, and a decrease in forage activity after ambient stream temperatures exceeded 71.6°F (Nielsen et al. 1994). Hatchery reared steelhead in thermal gradients selected temperatures of 64-66°F while wild caught steelhead selected temperatures around 63°F (Myrick and Cech 2001). An upper water temperature limit of 65°F is preferred for growth and development of Sacramento River and American River juvenile steelhead (NMFS 2002).

In the lower Yuba River, juvenile steelhead exhibit variable durations of rearing. RMT (2010b) distinguished fry, juvenile, and yearling+ lifestages through evaluation of bi-weekly length-frequency distributions of *O. mykiss* captured in RSTs in the lower Yuba River, and other studies that report length-frequency estimates (Mitchell 2010; CDFG 1984). Some juvenile *O. mykiss* may rear in the lower Yuba River for short periods (up to a few months) and others may spend from one to three years rearing in the river.

Some age-0 *O. mykiss* disperse downstream soon after emerging and continue throughout the year (Kozlowski 2004). Thus, the steelhead fry (individuals less than about 45 millimeters (mm)) lifestage generally extends from the time of initial emergence (based upon accumulated thermal units from the time of egg deposition through hatching and alevin incubation) until 3 months following the end of the spawning period. YCWA (2010) identified the fry rearing lifestage as generally extending from mid-March through July, and identified the juvenile rearing lifestage as extending year-round. Based on all information collected to date, the RMT (2013a) identified the steelhead fry rearing period as extending from April through July.

Juvenile steelhead have been reported to rear in the lower Yuba River for up to 1 year or more (SWRI 2002). CDFG (1991a) reported that juvenile steelhead rear throughout the year in the lower Yuba River, and may spend from 1 to 3 years rearing in the river. Scale analysis conducted by Mitchell (2010) indicates the presence of at least four age categories for *O. mykiss* in the lower Yuba River that spent 1, 2, or 3 years in freshwater and 1 year at sea before returning to the lower Yuba River to spawn.

Based on the combined results from electrofishing and snorkeling surveys conducted during the late 1980s, CDFG (1991a) reported that juvenile steelhead were observed in all river reaches downstream of the Englebright Dam and, in addition to Chinook salmon, were the only fish species observed in the Narrows Reach. They also indicated that most juvenile steelhead rearing occurred above Daguerre Point Dam. SWRI et al. (2000) summarized data collection in the lower Yuba River obtained from 1992 through 2000. Since 1992, Jones and Stokes (JSA) biologists conducted fish population surveys in the lower Yuba River using snorkel surveys to determine annual and seasonal patterns of abundance and distribution of juvenile *O. mykiss* and Chinook salmon during the spring and summer rearing periods. The primary rearing habitat for juvenile *O. mykiss* is upstream of Daguerre Point Dam. In 1993 and 1994, snorkeling surveys indicated that the population densities and overall abundance of juvenile *O. mykiss* (age 0 and 1+) were substantially higher upstream of Daguerre Point Dam, with decreasing abundance downstream of Daguerre Point Dam.

Similarly, Kozlowski (2004) found higher abundances of juvenile *O. mykiss* above Daguerre Point Dam, relative to downstream of Daguerre Point Dam. Kozlowski (2004) observed age-0 *O. mykiss* throughout the entire study area, with highest densities in upstream habitats and declining densities with increasing distance from the Narrows. Approximately 82 percent of juvenile *O. mykiss* were observed upstream of Daguerre Point Dam. Kozlowski (2004) suggested that the distribution of age-0 *O. mykiss* appeared to be related to the distribution of spawning adults. SWRI et al. (2000) suggested that higher abundances of juvenile *O. mykiss* above Daguerre Point Dam may have been due to larger numbers of spawners, greater amounts of more complex, high quality cover, and lower densities of predators such as striped bass and American shad, which reportedly were restricted to areas below Daguerre Point Dam.

In the lower Yuba River, Kozlowski (2004) reports that juvenile *O. mykiss* were observed in greater numbers in pool habitats than in run habitats. He suggests that results of his study indicated a relatively higher degree of habitat complexity, suitable for various lifestages, in the reaches just below the Narrows compared to farther downstream. The Narrows reach includes greater occurrence of pool-type microhabitat suitable for juvenile *O. mykiss* rearing, as well as small boulders and cobbles preferred by the age-0 emerging lifestage (Kozlowski 2004).

Juvenile *O. mykiss* apparently demonstrate a proclivity for near-bank areas, rather than open-channel habitats, in the lower Yuba River. USFWS (2008) reports 258 observations of juvenile *O. mykiss* and 244 observations of juvenile Chinook salmon, all but 8 of them made near the river banks in the lower Yuba River.

A broad range of *O. mykiss* size classes have been observed in the lower Yuba River during spring and summer snorkeling, electrofishing, and angling surveys (SWRI et al. 2000). Juvenile *O. mykiss* ranging in size from 40-150 mm were commonly observed upstream of Daguerre Point Dam. Numerous larger juveniles and resident trout up to 18 in long were also commonly observed in the mainstem upstream and downstream of Daguerre Point Dam (SWRI et al. 2000). Age 0 YOY *O. mykiss* were clearly shown by the distinct mode in lengths of fish caught by electrofishing (40-100 mm FL). A preliminary examination of scales indicated that most yearling (age 1+) and older *O. mykiss* were represented by fish greater than 110 mm long, including most if not all of the fish caught by hook and line. The sizes of age 0 and 1+ *O. mykiss* indicated substantial annual growth of *O. mykiss* in the lower Yuba River. Seasonal growth of age 0 *O. mykiss* was evident from repeated sampling in 1992 and 1999, but actual growth rates could not be estimated because of continued recruitment of fry (newly emerged juveniles) or insufficient sample sizes (SWRI et al. 2000).

Mitchell (2010) reports that analysis of scale growth patterns of juvenile *O. mykiss* in the lower Yuba River indicates a period of accelerated growth during the spring peaking during the summer months, followed by decelerated growth during the fall and winter. Following the second winter, juvenile *O. mykiss* in the lower Yuba River exhibit reduced annual growth in length with continued growth in mass until reaching reproductive age. Additionally, more rapid juvenile and adult *O. mykiss* growth occurred in the lower Yuba River compared to the lower Sacramento River and Klamath River *O. mykiss*, with comparable growth rates to *O. mykiss* in the upper Sacramento River (Mitchell 2010).

CDFG (1991a) reports that juvenile steelhead in the lower Yuba River rear throughout the year, and may spend from one to three years in the river before emigrating primarily from March to June. Salvage data at the Hallwood-Cordua fish screen suggest that most juvenile fish initiated their downstream movements immediately preceding and following a new moon, indicating the presence of lunar periodicity in the timing or outmigration patterns in the lower Yuba River (Kozlowski 2004).

Based on all information collected to date, the RMT (2013a) identified the steelhead juvenile rearing period as extending year-round, and the steelhead juvenile downstream movement period as extending from April through September.

In the lower Yuba River, some YOY *O. mykiss* are captured in RSTs located downstream of Daguerre Point Dam during late-spring and summer, indicating movement downstream. However, at least some of this downstream movement may be associated with the pattern of flows in the river. Water transfer monitoring in 2001, 2002, and 2004 (YCWA and SWRCB

2001; YCWA 2003a; YCWA 2005), generally from about mid-June through September, indicated that the character of the initiation of the water transfers could potentially affect juvenile *O. mykiss* downstream movement. Based upon the substantial differences in juvenile *O. mykiss* downstream movements (RST catch data) noted between the 2001 study, and the 2002 and 2004 studies, it was apparent that the increases in juvenile *O. mykiss* downstream movement associated with the initiation of the 2001 water transfers were avoided due to a more gradual ramping-up of flows that occurred in 2002 and 2004 (YCWA et al. 2007).

Numerous studies have been conducted regarding temperature preference, mortality, and water temperature growth-related relationships for *O. mykiss.* As previously described, some steelhead may rear in freshwater for up to 3 years before emigrating as yearling+ smolts, whereas other individuals move downstream shortly after emergence as post-emergent fry, or rear in the river for several months and move downstream as juveniles without exhibiting the ontogenetic characteristics of smolts. Presumably, these individuals continue to rear and grow in downstream areas (e.g., lower Feather River, Sacramento River, and Upper Delta) and undergo the smoltification process prior to entry into saline environments. Thus, fry and juvenile rearing occur concurrently with post-emergent fry and juvenile downstream movement.

5.2.4.5 Smolt Emigration

Most juvenile steelhead spend 1 to 3 years in fresh water before emigrating to the ocean as smolts (Shapovalov and Taft 1954). During their downstream migration, juvenile steelhead undergo a process referred to as smoltification, which is a physiologic transformation and osmoregulatory pre-adaptation to residence in saline environs. Physiologic expressions of smoltification include increased gill ATPase and thyroxin levels, and more slender body form which are silvery in appearance. The primary period of steelhead smolt outmigration from rivers and creeks to the ocean generally occurs from January to June (NMFS 2014).

In the Sacramento River, juvenile steelhead migrate to the ocean in spring and early summer at 1 to 3 years of age with peak migration through the Delta in March and April (Reynolds et al. 1993 as cited in NMFS 2014). Hallock et al. (1961) found that juvenile steelhead in the Sacramento River Basin migrate downstream during most months of the year, but the peak emigration period occurred in the spring, with a much smaller peak in the fall (NMFS 2014).

According to NMFS (2009b), steelhead are present at Chipps Island between at least October and July, according to catch data from the USFWS Chipps Island Trawl. It appears that adipose fin-clipped steelhead have a different emigration pattern than unclipped steelhead. Adipose fin-clipped steelhead showed distinct peaks in catch between January and March corresponding with time of release, whereas unclipped steelhead were more evenly distributed over a period of 6 months or more. These differences are likely an artifact of the method and timing of hatchery releases (NMFS 2009b).

Steelhead successfully smolt at water temperatures in the 43.7°F to 52.3°F range (Myrick and Cech 2001). The optimum water temperature range for successful smoltification in young steelhead has been reported as 44.0°F to 52.3°F (Rich 1987 as cited in NMFS 2014). Wagner (1974) reported smolting ceased rather abruptly when water temperatures increased to 57°F-

64°F. NMFS (2009b) reported that water temperatures under 57°F are considered best for smolting.

In the lower Yuba River, the steelhead smolt emigration period has been reported to extend from October through May (CALFED and YCWA 2005; SWRI 2002; YCWA et al. 2007). RMT (2010b; 2013a) review of all available data indicates that 1 year old and older (yearling+) steelhead smolt emigration may extend from October through mid-April.

For the purposes of impact assessment, RMT (2010b) developed separate water temperature index values for the yearling+ smolt emigration lifestages distinct from values for juvenile steelhead rearing and/or outmigration as juveniles from the lower Yuba River. They assumed that juvenile steelhead that exhibit extended rearing in the lower Yuba River undergo the smoltification process and volitionally emigrate from the river as yearling+ individuals.

5.2.5 Limiting Factors, Threats and Stressors

As stated by NMFS (2005a), the factors affecting the survival and recovery of Central Valley steelhead and their habitat are similar to those affecting spring-run Chinook salmon and are primarily associated with habitat loss (McEwan 2001). McEwan and Jackson (1996) attribute this habitat loss and other impacts to steelhead habitat primarily to water development resulting in inadequate flows, flow fluctuations, blockages, and entrainment into diversions. Other effects on critical habitat related to land use practices and urbanization have also contributed to steelhead declines (Busby et al. 1996). Although many of the factors affecting spring-run Chinook salmon habitat are common to steelhead, some stressors, especially summer water temperatures, cause greater effects to steelhead because juvenile steelhead rear in freshwater for more than 1 year. Because most suitable habitat has been lost to dam construction, juvenile steelhead rearing is generally confined to lower elevation stream reaches, where water temperatures during late summer and early fall can be sub-optimal (NMFS 2005a).

Many of the improvements to critical habitat that have benefited spring-run Chinook salmon, including water management through the CVPIA § 3406(b)(2) water supply and the CALFED Environmental Water Account, improved screening conditions at water diversions, and changes in inland fishing regulations (there is no ocean steelhead fishery) also benefit Central Valley steelhead (NMFS 2005a). However, many dams and reservoirs in the Central Valley do not have water storage capacity or release mechanisms necessary to maintain suitable water temperatures for steelhead rearing through the critical summer and fall periods, especially during critically dry years (McEwan 2001).

5.2.5.1 DPS

According to the NMFS (2014a) Recovery Plan, threats to Central Valley steelhead are similar to those for spring-run Chinook salmon and fall into three broad categories: 1) loss of historical spawning habitat; 2) degradation of remaining habitat; and 3) threats to the genetic integrity of the wild spawning populations from hatchery steelhead production programs in the Central Valley. Also, as for spring-run Chinook salmon, the potential effects of long-term climate change also may adversely affect steelhead and their recovery.

In 1998, NMFS concluded that the risks to Central Valley steelhead had diminished, based on a review of existing and recently implemented state conservation efforts and federal management programs (e.g., CVPIA, AFRP, CALFED) that address key factors for the decline of this species (NMFS 2009a). NMFS stated that Central Valley steelhead were benefiting from two major conservation initiatives, being simultaneously implemented: 1) the CVPIA, which was passed by Congress in 1992; and 2) the CALFED Program, a joint state/federal effort implemented in 1995. The following discussion of these two programs was taken from NMFS (2014a).

The CVPIA is specifically intended to remedy habitat and other problems associated with the construction and operation of the CVP. The CVPIA has two key features related to steelhead. First, it directs the Secretary of the Interior to develop and implement a program that makes all reasonable efforts to double natural production of anadromous fish in Central Valley streams (§ 3406(b)(1)) by the year 2002. The AFRP was initially drafted in 1995 and subsequently revised in 1997. Funding has been appropriated since 1995 to implement restoration projects identified in the AFRP planning process. Second, the CVPIA dedicates up to 800,000 ac-ft of water annually for fish, wildlife, and habitat restoration purposes (§ 3406(b)(2)) and provides for the acquisition of additional water to supplement the 800,000 ac-ft (§ 3406(b)(3)). USFWS, in consultation with other federal and state agencies, has directed the use of this dedicated water yield since 1993.

The CALFED Program, which began in June 1995, was charged with the responsibility of developing a long-term Bay-Delta solution. A major element of the CALFED Program is the Ecosystem Restoration Program (ERP), which was intended to provide the foundation for long-term ecosystem and water quality restoration and protection throughout the region. Among the non-flow factors causing decline that have been targeted by the program are unscreened diversions, waste discharges and water pollution, impacts due to poaching, land derived salts, exotic species, fish barriers, channel alterations, loss of riparian wetlands, and other causes of estuarine habitat degradation. The level of risk faced by the Central Valley steelhead DPS may have diminished since the 1996 listing proposal as a result of habitat restoration and other measures that have recently been implemented through the CALFED and CVPIA programs. Although most restoration measures designed to recover Chinook salmon stocks can benefit steelhead, focusing restoration solely on Chinook salmon may lead to inadequate measures to restore steelhead because of their different life histories and resource requirements, particularly for rearing juveniles (McEwan 2001). Additional actions that benefit Central Valley steelhead include efforts to enhance fisheries monitoring, such as the *Central Valley Steelhead Monitoring Plan*, and conservation actions to address artificial propagation.

In spite of the benefits derived from implementation of these two programs, NMFS (2014a) identified several major stressors presently applicable to the entire Central Valley steelhead DPS. Many of the most important stressors specific to the steelhead DPS correspond to the stressors described for the spring-run Chinook salmon ESU. As previously stated, the 2009 NMFS OCAP BO (2009b) identified factors leading to the current status of the spring-run Chinook salmon ESU, which also are applicable to the steelhead DPS, including habitat blockage, water development and diversion dams, water conveyance and flood control, land use activities, water quality, hatchery operations and practices, over-utilization (e.g., ocean commercial and sport harvest, inland sport harvest), disease and predation, environmental variation (e.g., natural environmental cycles, ocean productivity, climate change), and non-native invasive species. The previous discussions in this Applicant-Prepared Draft BA addressing limiting factors and threats for the spring-run Chinook salmon ESU and their specific geographic influences, including the Sacramento River and the Delta, are not repeated in this section of this Applicant-Prepared Draft BA. Stressors that are unique to the steelhead DPS, or substantially differ in the severity from the stressor for the previously described spring-run Chinook salmon ESU, are described below.

Threats and stressors for the Central Valley steelhead DPS identified in Appendix B (Threats Assessment) of the NMFS (2014a) Recovery Plan include: 1) destruction, modification, or curtailment of habitat or range; 2) overutilization for commercial, recreational, scientific or education purposes; 3) disease or predation; 4) inadequacy of existing regulatory mechanisms, including federal and non-federal efforts; 5) other natural and man-made factors affecting its continued existence; and 6) non-lifestage specific threats and stressors including artificial propagation programs, small population size, genetic integrity and long-term climate change. The following summarization of threats and stressors for the Central Valley steelhead DPS is taken directly from Appendix B (Threats Assessment) of the NMFS Recovery Plan (NMFS 2014).

5.2.5.1.1 Destruction, Modification, or Curtailment of Habitat or Range

The spawning habitat for Central Valley steelhead has been greatly reduced from its historical range (NMFS 2014). The vast majority of historical spawning habitat for Central Valley steelhead has been eliminated by fish passage impediments associated with water storage, withdrawal, conveyance, and diversions for agriculture, flood control, and domestic and hydropower purposes (NMFS 2014). Modification of natural flow regimes has resulted in increased water temperatures, changes in fish community structures, depleted flow necessary for migration, spawning, rearing, and flushing of sediments from spawning gravels. These changes in flow regimes may be driving a shift in the frequencies of various life history strategies, especially a decline in the proportion of the population migrating to the ocean. Land use activities, such as those associated with agriculture and urban development, have altered steelhead habitat quantity and quality. Although many historically harmful practices have been halted, much of the historical damage to habitats limiting steelhead remains to be addressed, and the necessary restoration activities will likely require decades.

5.2.5.1.2 Overutilization for Commercial, Recreational, Scientific or Educational Purposes (Inland Sport Harvest)

Steelhead has been, and continues to be, an important recreational fishery throughout their range. Although there are no commercial fisheries for steelhead in the ocean, inland steelhead fisheries include tribal and recreational fisheries. In the Central Valley, recreational fishing for steelhead is popular, yet harvest is restricted to only the visibly marked hatchery-origin fish, which reduces the likelihood of retaining naturally spawned wild fish. The permits NMFS issues for scientific or educational purposes stipulate specific conditions to minimize take of steelhead individuals during permitted activities. There are currently 11 active permits in the Central Valley that may affect steelhead. These permitted studies provide information about Central Valley steelhead that is useful to the management and conservation of the DPS. Additional information regarding inland sport harvest of steelhead in the Central Valley contained in Reclamation (2008a) is provided below.

Inland Sport Harvest

Historically in California, almost half of the river sport fishing effort has occurred in the Sacramento-San Joaquin River system, particularly upstream from the city of Sacramento (Emmett et al. 1991). There is little information on steelhead harvest rates in California. Hallock et al. (1961) estimated that harvest rates for Sacramento River steelhead from the 1953/1954 through 1958/1959 seasons ranged from 25.1 percent to 45.6 percent assuming a 20 percent non-return rate of tags. The average annual harvest rate of adult steelhead above RBDD for the 3-year period from 1991/1992 through 1993/1994 was 16 percent (McEwan and Jackson 1996). Since 1998, all hatchery steelhead have been marked with an adipose fin clip allowing anglers to distinguish hatchery and wild steelhead. Current regulations restrict anglers from keeping unmarked steelhead in Central Valley streams. Overall, this regulation has greatly increased protection of naturally produced adult steelhead (Reclamation 2008a). However, the total number of steelhead contacted might be a significant fraction of basin-wide escapement, and even low catch-and-release mortality may pose a problem for wild populations (Good et al. 2005).

Recent drought conditions have affected some steelhead fishing opportunities for the Central Valley steelhead DPS. As an example, the California Fish and Game Commission imposed an emergency fishery closure on the American River during February of 2014. The closure ended in April of that year (NMFS 2016c). The regulation changes reviewed above for steelhead fishing in the Central Valley suggest that there is the potential for a change in harvest dynamic over the past several years. The overall trend has been to incrementally increase the opportunity for harvest of hatchery-origin steelhead by increasing the daily bag and possession limits. The rationale behind encouraging more harvest of hatchery-origin steelhead is to minimize potential negative behavioral and genetic interactions with natural-origin steelhead. In addition, retention of hatchery-origin steelhead in the Central Valley is typically very low. The purpose of the hatchery programs is to provide a harvestable fishery resource, and CDFW would like to see more of that resource utilized for its intended consumptive purpose (NMFS 2016c).

5.2.5.1.3 Disease or Predation

Steelhead are exposed to bacterial, protozoan, viral, and parasitic organisms in spawning and rearing areas, hatcheries, migratory routes, and the marine environment. Very little current or historical information exists to quantify changes in infection levels and mortality rates attributable to these diseases for steelhead. Naturally spawned fish tend to be less susceptible to pathogens than hatchery-reared fish. Introduction of non-native species and modification of habitat have resulted in increased predatory populations and salmonid predation in river systems. In general, predation rates on steelhead are considered to be an insignificant contribution to the large declines observed in West Coast steelhead populations. In some local populations, however, predation may significantly influence salmonid abundance when other prey species are not present and habitat conditions lead to the concentration of adults and/or juveniles.

5.2.5.1.4 Inadequacy of Existing Regulatory Mechanisms (Federal Efforts, Non-Federal Efforts)

Federal Efforts

There have been several federal actions attempting to reduce threats to the Central Valley steelhead DPS. The BOs for the CVP and SWP and other federal projects involving irrigation and water diversion and fish passage, for example, have improved or minimized adverse impacts to steelhead in the Central Valley. There have also been several habitat restoration efforts implemented under CVPIA and CALFED programs that have led to several projects involving fish passage improvements, fish screens, floodplain management, habitat restoration, watershed planning, and other projects that have contributed to improvement of steelhead habitat. However, despite federal actions to reduce threats to the Central Valley steelhead DPS, the existing protective efforts are inadequate to ensure the DPS is no longer in danger of extinction. There remain high risks to the abundance, productivity, and spatial structure of the steelhead DPS.

Non-Federal Efforts

Measures to protect steelhead throughout the State of California have been in place since 1998. The State's Natural Communities Conservation Planning (NCCP) program involves long-term planning with several stakeholders. A wide range of measures have been implemented, including 100 percent marking of all hatchery steelhead, zero bag limits for unmarked steelhead, gear restrictions, closures, and size limits designed to protect smolts. NMFS and Cal Fish and Wildlife are working to improve inland fishing regulations to better protect both anadromous and resident forms of *O. mykiss* populations. A proposal to develop a comprehensive status and trends monitoring plan for Central Valley steelhead was submitted for funding consideration to the CALFED ERP in 2005. The proposal, drafted by Cal Fish and Wildlife and the interagency Central Valley Steelhead Project Work Team, was selected by the ERP Implementing Agency Managers, and is to receive funding as a directed action. Long-term funding for implementation of the monitoring plan, once it is developed, still needs to be secured (NMFS 2014).

There are many sub-watershed groups, landowners, environmental groups, and non-profit organizations that are conducting habitat restoration and planning efforts that may contribute to the conservation of steelhead. However, despite federal and non-federal efforts to promote the conservation of the Central Valley steelhead DPS, few efforts address conservation needs at scales sufficient to protect the entire steelhead DPS. The lack of status and trend monitoring and research is one of the critical limiting factors to this DPS.

5.2.5.1.5 Other Natural and Man-made Factors Affecting the Continued Existence of the DPS

NMFS and the Biological Review Team (BRT) are concerned that the proportion of naturally produced fish is declining. Two artificial propagation programs for steelhead in the Central Valley – Coleman National Fish Hatchery and FRFH – may decrease risk to the DPS to some degree by contributing increased abundance to the DPS. Potential threats to natural steelhead posed by hatchery programs include: 1) mortality of natural steelhead in fisheries targeting hatchery-origin steelhead; 2) competition for prey and habitat; 3) predation by hatchery-origin fish on younger natural fish; 4) genetic introgression by hatchery-origin fish that spawn naturally and interbreed with local natural populations; and 5) disease transmission. Changes in climatic events and global climate, such as El Niño ocean conditions and prolonged drought conditions, can threaten the survival of steelhead populations already reduced to low abundance levels as the result of the loss and degradation of freshwater and estuarine habitats. Floods and persistent drought conditions have reduced already limited spawning, rearing, and migration habitats. Unscreened water diversions and CVP and SWP pumping plants entrain outmigrating juvenile steelhead and fry, leading to fish mortality.

5.2.5.1.6 Non-lifestage Specific Threats and Stressors for the DPS (Artificial Propagation Programs, Small Population Size, Genetic Integrity and Long-term Climate Change)

Potential threats to the Central Valley steelhead population that are not specific to a particular lifestage include the potential negative impacts of the current artificial propagation program utilizing several hatcheries in the Sacramento-San Joaquin drainage, the small wild population size, the genetic integrity of the population due to both hatchery influence and small population size, and the potential effects of long-term climate change (NMFS 2014). Each of these potential threats is discussed in the following sections.

Artificial Propagation Program

Research has indicated that approximately 63 to 92 percent of steelhead smolt production is of hatchery-origin (NMFS 2003). These data suggest that the relative proportion of wild to hatchery smolt production is decreasing (NMFS 2003). All California hatchery steelhead programs began 100 percent adipose fin-clipping in 1998 to differentiate hatchery steelhead from natural steelhead (NMFS 2014).

Propagation of steelhead at the Coleman National Fish Hatchery has been occurring for over 50 years. Hatchery-origin and natural-origin steelhead have been managed as a single stock; mixing of hatchery and natural origin population components occurred through spawning at the hatchery and intermingling with natural spawners in Battle Creek. Niemela et al. (2008, as cited in NMFS 2014) used genetic pedigree analysis to evaluate relative reproductive success and fitness among hatchery-origin and natural origin population components based on multi-locus DNA microsatellite genotypes. Preliminary results suggest that hatchery origin spawners experienced low relative reproductive success, producing significantly fewer adult offspring in comparison to natural origin spawners. Additionally, repeat spawning was more prevalent in the natural origin component of the population.

Population Size

In the technical memorandum titled *Updated Status of Federally Listed ESUs of West Coast Salmon and Steelhead* (Good et al. 2005), NMFS estimated the abundance of natural spawners for the steelhead DPS (then classified as an ESU), which was reported as the geometric mean (and range) of the most recent data available at that time, consistent with previous coast-wide status reviews of the species (Weitkamp et al. 1995; Busby et al. 1996; Gustafson et al. 1997; Johnson et al. 1997; Myers et al. 1998). Geometric means were calculated to represent the abundance of natural spawners for each population or quasi-population. Geometric means were calculated for the most recent 5 years of steelhead data, to correspond with modal age at maturity (Good et al. 2005). Where possible, the BRTs obtained population or ESU-level estimates of the fraction of hatchery-origin spawners or calculated estimates from information using scale analyses, fin clips, etc. (Good et al. 2005).

The Central Valley steelhead DPS mean annual escapement of natural spawners was estimated at 1,952 based on a 5-year period ending in 1993 (Good et al. 2005). During that time period a minimum escapement of 1,425 and a maximum escapement of 12,320 were observed (Good et al. 2005). A long-term trend analysis indicated that the population was declining (Good et al. 2005). In the *Updated Status of Federally Listed ESUs of West Coast Salmon and Steelhead* (Good et al. 2005), NMFS suggests that there has been no significant status change since the 1993 data and the Central Valley steelhead population continues to decline (Good et al. 2005). Good et al. (2005) also suggested that hatchery production is large relative to natural production. As an example, the steelhead run in the lower Feather River has been increasing over the past several years; however, over 99 percent of the run is of direct hatchery-origin (DWR 2002). Williams et al. (2011) reported that the viability of this steelhead DPS had worsened since the 2005 review when Good et al. (2005) concluded that the DPS was in danger of extinction.

Since 2011, Williams et al. (2016) report that the USFWS' Chipps Island midwater trawl dataset indicated that the production of natural-origin steelhead remains very low relative to hatchery production. Updated through 2013, the trawl data indicate that catch-per-unit-effort has fluctuated but remained level over the past decade, but the proportion of the catch that is adipose fin-clipped (100% of hatchery steelhead production have been adipose fin-clipped starting in 1998) has risen steadily, exceeding 90 percent in recent years, reaching 95 percent in 2010, and remaining very high through 2013. Because hatchery releases have been fairly constant, this implies that natural production of juvenile steelhead has been falling (Williams et al. 2016).

Genetic Integrity

There is still significant local genetic structure to Central Valley steelhead populations, although fish from the San Joaquin and Sacramento basins cannot be distinguished genetically (Nielsen et al. 2003). Hatchery effects appear to be localized – for example, Feather River and FRFH steelhead are closely related as are American River and Nimbus Hatchery fish (DWR 2002). Leary et al. (1995) report that hatchery straying has increased gene flow among steelhead populations in the Central Valley and that a smaller amount of genetic divergence is observed among Central Valley populations compared to wild British Columbia populations largely uninfluenced by hatcheries. Natural annual production of steelhead smolts in the Central Valley is estimated at 181,000 and hatchery production is 1,340,000 for a ratio of 0.148 (Good et al. 2005). Monitoring by hydroacoustic tracking has revealed that Mokelumne River/Hatchery steelhead (FRFH source stock) are straying into the American River (J. Smith, EBMUD, pers. comm. as cited in NMFS 2014).

There has been significant transfer of genetic material among hatcheries within the Central Valley as well as some transfer from systems outside the Central Valley. There have occasionally been transfers of steelhead from the FRFH to the Mokelumne Hatchery. For example, eyed eggs from the Nimbus Hatchery were transferred to the FRFH several times in the late 1960s and early 1970s (DWR 2002). Also, Nimbus Hatchery steelhead eggs have often been transferred to the Mokelumne Hatchery. Additionally, an Eel River strain of steelhead was used as the founding broodstock for the Nimbus Hatchery (CDFG 1991b). In the late 1970s, a strain of steelhead was brought in from Washington State for the FRFH (DWR 2002).

As discussed in Williams et al. (2016), new genetic analysis show that the steelhead stock currently propagated in the Mokelumne Hatchery is genetically similar to the steelhead broodstock in the FRFH (Pearse and Garza 2015), consistent with documentation on the recent transfers of eggs from the FRFH for broodstock at the Mokelumne Hatchery. Nimbus Hatchery steelhead remain genetically divergent from the Central Valley DPS lineages, consistent with their founding from coastal steelhead stocks, and remain excluded from the DPS (Pearse and Garza 2015, as cited in Williams et al 2016). Thus, NMFS recommends a change in boundary delineation, the boundary of the Central Valley DPS should be modified to include steelhead from the Mokelumne Hatchery (Williams et at. 2016).

5.2.5.1.7 Long-Term Climate Change

Because steelhead normally spend a longer time in freshwater as juveniles than other anadromous salmonids, any negative effects of climate change may be more profound on steelhead populations. As previously mentioned, if more precipitation falls as rain instead of snow, reservoirs may have less coldwater pool available to maintain instream flows and suitable water temperatures during the summer and fall months. In addition, if more precipitation falls as rain earlier in the season, the potential increase in "rain on snow" events may increase mortality of steelhead incubating embryos during the spring.

5.2.5.1.8 Hatchery Operations and Practices

In addition to the immediately previous discussion taken from Appendix B (Threats Assessment) of the NMFS (2014a) Recovery Plan, an additional discussion regarding the impacts of hatcheries on the Central Valley steelhead DPS is provided below.

Hatcheries have come under scrutiny for their potential effects on wild salmonid populations (Bisson et al. 2002; Araki et al. 2007). The concern with hatchery operations is two-fold. First, they may result in unintentional, but maladaptive genetic changes in wild steelhead stocks (McEwan and Jackson 1996). Cal Fish and Wildlife believes its hatcheries take eggs and sperm from enough individuals to avoid loss of genetic diversity through inbreeding depression and genetic drift. However, artificial selection for traits that improve hatchery success (e.g., fast growth, tolerance of crowding) are not avoidable and may reduce genetic diversity and population fitness (Araki et al. 2007). Past and present hatchery practices represent the major threat to the genetic integrity of Central Valley steelhead (NMFS 2014). Overlap of spawning hatchery and natural fish within the steelhead DPS exists, resulting in genetic introgression. Also, a substantial problem with straying of hatchery fish exists within this DPS (Hallock 1989).

Habitat fragmentation and population declines resulting in small, isolated populations also pose genetic risk from inbreeding, loss of rare alleles, and genetic drift (NMFS 2014).

The second concern with hatchery operations revolves around the potential for undesirable competitive interactions between hatchery and wild stocks. Intraspecific competition between wild and artificially produced stocks can result in wild fish declines (McMichael et al. 1997; 1999). Although wild fish are presumably more adept at foraging for natural foods than hatchery-reared fish, this advantage can be negated by density-dependent effects resulting from large numbers of hatchery fish released at a specific locale, as well as the larger size and more aggressive behavior of the hatchery fish (Reclamation 2008a).

Currently, four hatcheries in the Central Valley produce steelhead to supplement the Central Valley wild steelhead population. These four Central Valley steelhead hatcheries (Mokelumne River, FRFH, Coleman, and Nimbus hatcheries) collectively produce approximately 3.4 million steelhead yearlings annually when all four hatcheries reach production goals (CDFW 2014). The hatchery steelhead programs originated as mitigation for the habitat lost by construction of dams. Steelhead are released at downstream locations in January and February at about four fish per pound, generally corresponding to the initiation of the peak of outmigration (Reclamation 2008a). In the Central Valley, practices such as transferring eggs between hatcheries and trucking smolts to distant sites for release contribute to elevated straying levels (USDOI 1999, as cited in NMFS 2009b).

According to Reclamation (2008a), the hatchery runs in the American and Mokelumne rivers are probably highly introgressed mixtures of many exotic stocks introduced in the early days of the hatcheries (McEwan and Jackson 1996; NMFS 1998). Beginning in 1962, steelhead eggs were imported into Nimbus Hatchery from the Eel, Mad, upper Sacramento, and Russian rivers and from the Washougal and Siletz rivers in Washington and Oregon, respectively (McEwan and Nelson 1991, as cited in McEwan and Jackson 1996). Egg importation has also occurred at other Central Valley hatcheries (McEwan and Jackson 1996).

Reclamation (2008a) further states that stock introductions began at the FRFH in 1967, when steelhead eggs were imported from Nimbus Hatchery to be raised as broodstock. In 1971, the first release of Nimbus origin fish occurred. From 1975 to 1982, steelhead eggs or juveniles were imported from the American, Mad, and Klamath rivers and the Washougal River in Washington. The last year that Nimbus-origin fish were released into the Feather River was 1988. Based on preliminary genetic assessments of Central Valley steelhead, NMFS (1998) concluded the FRFH steelhead were part of the Central Valley DPS despite an egg importation history similar to the Nimbus Hatchery stock, which NMFS did not consider part of the Central Valley DPS.

The increase in Central Valley hatchery production has reversed the composition of the steelhead population, from 88 percent naturally-produced fish in the 1950s (McEwan 2001) to an estimated 23 to 37 percent naturally-produced fish (Nobriga and Cadrett 2003). The increase in hatchery steelhead production proportionate to the wild population has reduced the viability of the wild steelhead populations, increased the use of out-of-basin stocks for hatchery production, and increased straying (NMFS and CDFG 2001). Thus, the ability of natural populations to successfully reproduce and continue their genetic integrity likely has been diminished (Reclamation 2008a).

In addition, harvest impacts associated with hatchery-wild population interactions have been identified as a stressor to wild Central Valley steelhead stocks (NMFS 2009a). The relatively low number of spawners needed to sustain a hatchery population can result in high harvest-to-escapements ratios in waters where fishing regulations are set according to hatchery population. This can lead to over-exploitation and reduction in the size of wild populations existing in the same system as hatchery populations due to incidental bycatch (McEwan 2001). According to Cal Fish and Wildlife creel census surveys, the majority (93 percent) of steelhead catches occur on the American and Feather rivers, sites of steelhead hatcheries (CDFG 2001d, as cited in NMFS 2009a). Creel census surveys conducted during 2000 indicated that 1,800 steelhead were retained, and 14,300 were caught and released. The total number of steelhead contacted might be a significant fraction of basin-wide escapement, so even low catch-and-release mortality may pose a problem for wild populations. Additionally, NMFS (2005b) asserted that steelhead fisheries on some tributaries and the mainstem Sacramento River may affect some steelhead juveniles.

5.2.5.2 Lower Yuba River

The lower Yuba River steelhead population is exposed and subject to the myriad of limiting factors, threats and stressors described above for the DPS. Concurrently with the effort conducted for spring-run Chinook salmon, NMFS (2014a) conducted a comprehensive assessment of stressors affecting both steelhead within the lower Yuba River, and lower Yuba River steelhead populations as they migrate downstream (as juveniles) and upstream (as adults) through the lower Feather River, the lower Sacramento River, and the Bay-Delta system. For the lower Yuba River population of steelhead, the number of stressors according to the categories of “Very High”, “High”, “Medium”, and “Low” that occur in the lower Yuba River or occur out of basin are presented below by lifestage (Table 5.2-2).

Table 5.2-2. The number of stressors according to the categories of “Very High”, “High”, “Medium”, and “Low” that occur in the lower Yuba River, or occur out-of-basin, by lifestage for the lower Yuba River population of steelhead.

Location	Stressor Categories			
	Very High	High	Medium	Low
ADULT IMMIGRATION AND HOLDING				
Lower Yuba River	2	1	3	1
Out of Basin	1	5	10	4
SPAWNING				
Lower Yuba River	3	2	0	2
Out of Basin	N/A*	N/A	N/A	N/A
EMBRYO INCUBATION				
Lower Yuba River	1	0	4	0
Out of Basin	N/A	N/A	N/A	N/A
JUVENILE REARING AND OUTMIGRATION				
Lower Yuba River	5	1	1	5
Out of Basin	12	16	6	9

Source: NMFS 2014
* N/A – Not Applicable.

As shown by the numbers in Table 5.2-2, of the total number of 94 stressors affecting all identified lifestages of lower Yuba River populations or steelhead, 31 are within the lower Yuba River and 63 are out-of-basin. Because spawning and incubation occurs only in the lower Yuba River, all of the stressors associated with these lifestages occur in the lower Yuba River. For the adult immigration and holding, and the juvenile rearing and outmigration lifestages combined, a total of 43 "Very High" and "High" stressors were identified, with 9 of those occurring in the lower Yuba River and 34 occurring out-of-basin.

NMFS (2014a) Recovery Plan states that "*Implementation of the flow schedules specified in the Fisheries Agreement of the Yuba Accord is expected to address the flow-related major stressors including flow-dependent habitat availability, flow-related habitat complexity and diversity, and water temperatures*."

NMFS will conduct an exposure analysis as part of its assessment of potential effects to steelhead critical habitat that will include consideration of PBFs of critical habitat that are likely to be exposed (NMFS 2016a). PBFs of designated steelhead critical habitat in the lower Yuba River include freshwater spawning sites, freshwater rearing sites, and freshwater migration corridors. The information provided below is intended to assist NMFS address potential concerns about exposure responses that may be sufficient to reduce the quantity, quality, or availability of PBFs within the Action Area. A description of the primary biological features of steelhead critical habitat that are present within the Action Area, including potential stressors to steelhead and other factors affecting PBFs, is described below.

Many of the most important stressors specific to steelhead in the lower Yuba River correspond to the stressors described for spring-run Chinook salmon in the lower Yuba River, which included passage impediments and barriers, poaching, hatchery effects, fry and juvenile rearing physical habitat structure, predation, loss of riparian habitat and instream cover (e.g., riparian vegetation, instream woody material), loss of natural river morphology and function, and loss of floodplain habitat. The previous discussions in this Applicant-Prepared Draft BA addressing limiting factors and threats for the spring-run Chinook salmon population in the lower Yuba River that are pertinent to the steelhead population in the lower Yuba River are not repeated in this section of the Applicant-Prepared Draft BA. Stressors that are unique to steelhead or notably differ from spring-run Chinook salmon in the mechanism of effect in the lower Yuba River, or stressors that substantially differ in severity for steelhead are described below.

5.2.5.2.1 Harvest/Angling Impacts

Fishing for steelhead on the lower Yuba River is regulated by Cal Fish and Wildlife. Angling regulations on the lower Yuba River are intended to protect sensitive species, including wild steelhead. Cal Fish and Wildlife angling regulations (2016/2017) permit fishing for steelhead from the mouth of the Yuba River to the Highway 20 Bridge with only artificial lures with barbless hooks all year-round (CDFW 2016a). The regulations include a daily bag limit of two hatchery trout or hatchery steelhead (identified by an adipose fin clip), and a possession limit of four hatchery trout or hatchery steelhead. From the Highway 20 Bridge to Englebright Dam, fishing for steelhead is permitted from December 1 through August 31 only, with only artificial lures with barbless hooks. For this time period, the regulations include a daily bag limit of two hatchery trout or hatchery steelhead (identified by an adipose fin clip), and a possession limit of four hatchery trout or hatchery steelhead. Angling effects are considered to be a low stressor to steelhead in the lower Yuba River.

5.2.5.2.2 Poaching

While poaching is most notable during summer when spring-run Chinook salmon are present in the lower Yuba River, steelhead also are affected. According to Cal Fish and Game Wardens, fishers using illegal worms and hooks are known to target trout and steelhead in the Yuba River, particularly in the reach between Parks Bar and Hammonton Road in the Goldfields, suggesting that poaching could present a moderate stressor to steelhead in the lower Yuba River.

5.2.5.2.3 Hatchery Effects

The previous discussion in this Applicant-Prepared Draft BA addressing limiting factors, threats and stressors resulting from straying and other hatchery effects on the steelhead DPS that are pertinent to steelhead in the lower Yuba River are not repeated in this section of the Applicant-Prepared Draft BA. Hatchery-related stressors that are unique to steelhead in the lower Yuba River, or substantially differ in severity for Yuba River steelhead, are described below.

Although it has been oft-repeated that hatcheries historically have not been located on the Yuba River, that does not appear to be the case. According to a document titled *A History of California's Fish Hatcheries 1870–1960* (Leitritz 1970), an experimental fish hatchery station (i.e., the Yuba River Hatchery) was established in 1928 by the California Department of Natural Resources, Division of Fish and Game. The site was on Fiddle Creek, a tributary of the North Fork Yuba River about 34 miles north of Nevada City, near Camptonville. Fish rearing began at the station in 1929. Over the years, improvements were made to the hatchery. No reference could be found regarding salmon, but the hatchery was reported to hatch and rear trout, including steelhead (CDNR 1931). The hatchery continued operations until storms during November 1950 caused such extensive damage that repairs could not be made and it was permanently closed (Leitritz 1970).

Since that time, no fish hatcheries have been located on the Yuba River, and the river continues to support a persistent population of steelhead. According to the NMFS (2014a) Recovery Plan, the major threat to the genetic integrity of Central Valley steelhead results from past and present hatchery practices. These practices include the planting of non-natal fish, overlap of spawning hatchery and natural fish, and straying of hatchery fish.

Genetic Considerations

From 1970 to 1979, Cal Fish and Wildlife annually stocked 27,270–217,378 fingerlings, yearlings, and sub-catchable steelhead from Coleman National Fish Hatchery into the lower Yuba River (CDFG 1991b). Cal Fish and Wildlife stopped stocking steelhead into the lower Yuba River in 1979. In addition, it is possible that some hatchery-reared juvenile steelhead from the FRFH may move into the lower Yuba River in search of rearing habitat. Some competition for resources with naturally spawned steelhead could occur as a result.

Previous genetic work on population structure of steelhead in California has relied primarily on analyses of mitochondrial DNA (e.g., Berg and Gall 1988; Nielsen et al. 1997), which is the genetic material contained in a single cellular organelle that contains only 37 genes. Because mitochondria are inherited primarily from the maternal line, the information obtained from this type of analysis is often not reflective of population history or true relationships (Chan and Levin 2005). However, microsatellites, also known as simple sequence repeat loci, have been used in numerous studies of salmonids and have proven to be a valuable tool for elucidating population genetic structure. Work on *O. mykiss* in California using microsatellite loci has demonstrated that genetic structure can be identified with such data, both at larger scales (Aguilar and Garza 2006) and at relatively fine ones (Deiner et al. 2007; Pearse et al. 2007). The following discussion was taken from Garza and Pearse (2008).

Garza and Pearse (2008) studied populations of *O. mykiss* in the Central Valley using molecular genetic techniques to provide insight into population structure in the region. Data were collected from 18 nuclear microsatellite loci and variation analyzed to trace ancestry and evaluate genetic distinction among populations. The goals of the study were to use population genetic analyses of the data to assess origins and ancestry of *O. mykiss* populations above and below dams in Central Valley tributary rivers, to better understand the relationship of these populations to others in California, and to provide information on genetic diversity and population structure of these populations. Genotypes were collected from over 1,600 individual fish from 17 population samples and five hatchery rainbow trout strains. Fish populations from rivers and creeks that flow to both the Sacramento and San Joaquin rivers were evaluated, including the McCloud River, Battle Creek, Deer Creek, Butte Creek, Feather River, Yuba River, American River, Calaveras River, Stanislaus River and Tuolumne River sub-basins. Analyses included comparing genetics of fish collected both above and below barriers to anadromy in some of the study basins (Garza and Pearse 2008).

Phylogeographic trees were used to visually and quantitatively evaluate genetic relationships of Central Valley *O. mykiss* populations both with each other and with other California populations. Genetic diversity was relatively similar throughout the Central Valley. Above-barrier populations clustered with one another and below-barrier populations are most closely related to populations in far northern California, specifically the genetic groups that include the Eel and Klamath rivers. Since Eel River origin broodstock were used for many years at Nimbus Hatchery on the American River, it is likely that Eel River genes persist there and have also spread to other basins by migration, and that this is responsible for the clustering of the below-barrier populations with northern California ones. This suggests that the below-barrier populations in this region appear to have been widely introgressed with hatchery fish from out-of-basin broodstock sources. In phylogeographic analyses, above-barrier populations are more similar to San Francisco Bay *O. mykiss* populations than the below-barrier populations in the Central Valley. Because this relationship is expected for steelhead, given their extraordinary historic dependence on short distance migration events (Pearse and Garza 2007), they may represent relatively non-introgressed historic population genetic structure for the region. Other possible explanations for this pattern that rely on complicated, widespread patterns of introgression with hatchery fish are not entirely ruled out, but are highly improbable given that the above-barrier populations also group with moderate consistency into geographically-consistent clusters (e.g. Yuba-Upper and Feather-Upper) in all analyses and also because of the low apparent reproductive success of hatchery trout in streams throughout California (Garza and Pearse 2008).

The analyses also identified possible heterogeneity between samples from different tributaries of the upper Yuba and Feather rivers, although linkage disequilibrium was lower in these populations. Linkage disequilibrium can be caused by physical linkage of loci, sampling of related individuals/family structure, and by the sampling of more than one genetically distinct group within a population sample (Garza and Pearse 2008).

In general, although structure was found, all naturally-spawned *O. mykiss* populations within the Central Valley Basin were closely related, regardless of whether they were sampled above or below a known barrier to anadromy (Garza and Pearse 2008). This is due to some combination of pre-impoundment historic shared ancestry, downstream migration and, possibly, limited anthropogenic upstream migration. However, lower genetic diversity in above-barrier populations indicates a lack of substantial genetic input upstream and highlights lower effective population sizes for above-barrier populations. The consistent clustering of the above-barrier populations with one another, and their position in the California-wide trees, indicate that they are likely to most accurately represent the ancestral population genetic structure of steelhead in the Central Valley (Garza and Pearse 2008).

According to NMFS (2014a), there currently there is still concern about the ecological and genetic impacts of steelhead hatchery management in the Central Valley (California Hatchery Scientific Review Group 2012). These concerns continue to be related to the proportion of hatchery fish relative to naturally produced fish, the predominance of Eel River steelhead genetics in the Nimbus Hatchery steelhead program, and straying of hatchery produced steelhead.

Straying into the Lower Yuba River

The observation of adipose fin clips on adult steelhead passing upstream through the VAKI Riverwatcher™ system at Daguerre Point Dam demonstrates that hatchery straying into the lower Yuba River has, and continues, to occur. Although no information is presently available regarding the origin of adipose-clipped steelhead observed at the VAKI Riverwatcher™ systems at Daguerre Point Dam, it is reasonable to surmise that they most likely originate from the FRFH (RMT 2013a). The remainder of this discussion pertains to hatchery effects associated with the straying of adult steelhead into the lower Yuba River.

If hatchery-origin steelhead stray into the lower Yuba River and interbreed with naturally-spawning Yuba River steelhead, then such interbreeding has been suggested to represent a threat to the genetic diversity and integrity of the naturally-spawning steelhead population in the lower Yuba River. No previously conducted quantitative analyses or data addressing the extent of hatchery-origin steelhead straying into the lower Yuba River are available for presentation in this Applicant-Prepared Draft BA. However, some information is presently available to assess the amount of straying of hatchery-origin (adipose fin-clipped) steelhead into the lower Yuba River from recent VAKI Riverwatcher™ data.

In the lower Yuba River, attempts were made to differentiate adult steelhead from other *O. mykiss* (i.e., juvenile steelhead and resident rainbow trout) recorded passing Daguerre Point Dam utilizing daily VAKI Riverwatcher™ data. However, only 6 years of data (biological years[7] 2010/2011 through 2015/2016 (the steelhead 2015/2016 Biological Year was only evaluated through June 13, 2016, corresponding to the period of data availability) are currently available identifying adipose fin-clipped *O. mykiss* passing through the VAKI Riverwatcher™ system, during which extensive inoperable periods did not occur during the adult steelhead upstream migration period. Data reduction, limitations and applications are described in Section 5.2.7 (Viability) of this Applicant-Prepared Draft BA, below.

Analysis of the VAKI Riverwatcher™ data indicates that the percent contribution of hatchery-origin adult upstream migrating fish (represented by the percentage of adipose fin-clipped adult steelhead relative to the total number of adult upstream migrating steelhead, because 100 percent of FRFH-origin steelhead have been marked since 1996) was approximately 42 percent for the 2010/2011 biological year, about 62 percent for the 2011/2012 biological year, about 38 percent for the 2012/2013 biological year, about 55 percent for the 2013/2014 biological year, about 42 percent for the 2014/2015 biological year, and about 40 percent for the currently available data (i.e., August 2015 through June 2016) of the 2015/2016 biological year.

Hatchery effects including genetic considerations and straying of hatchery fish into the river represent a high stressor to lower Yuba River steelhead.

5.2.5.2.4 Potential Redd Dewatering

The potential for steelhead redd and egg pocket dewatering is very different for steelhead than for spring-run Chinook salmon. Potential steelhead redd and egg pocket dewatering is much higher for steelhead relative to spring-run Chinook salmon. However, the increased potential for steelhead redd dewatering is due to high flow events (storm flows) occurring during their spawning and incubation period (i.e., January through May), which exceed the combined flow capacity at the Narrows 1 and Narrows 2 facilities (4,130 cfs). Potential redd dewatering represents a moderate to high stressor to lower Yuba River steelhead.

5.2.5.2.5 Entrainment

The potential for juvenile steelhead entrainment at Daguerre Point Dam is greater than for juvenile spring-run Chinook salmon. This is because diversion rates are increased from spring through summer, and the timing of juvenile steelhead outmigration from the lower Yuba River more closely corresponds to the time of increased diversion. Consequently, the potential for entrainment at Daguerre Point Dam represents a moderate stressor to steelhead.

[7] For the purposes of the adult steelhead VAKI analyses presented in this Applicant-Prepared Draft BA, a steelhead biological year extends from August 1 through the end of July of the following year.

5.2.6 Viability of the Central Valley Steelhead DPS

The VSP concept (McElhany et al. 2000) previously described in Section 5.1.7 of this Applicant-Prepared Draft BA for the spring-run Chinook salmon ESU also is used to address and describe the viability of the Central Valley steelhead DPS.

5.2.6.1 DPS

As described by NMFS (2014a), there are few data with which to assess the status of Central Valley steelhead populations. Lindley et al. (2007) stated that, with the few exceptions of streams with long-running hatchery programs such as Battle Creek and the Feather, American and Mokelumne rivers, Central Valley steelhead populations are classified as data deficient. In all cases, hatchery-origin fish likely comprise the majority of the natural spawning run, placing the natural populations at high risk of extinction (Lindley et al. 2007). As of 2016, NMFS reinforced the conclusion that the Central Valley steelhead DPS is data deficient, with the exception of these hatchery programs (Williams et al. 2016).

From 1967-1993, steelhead run-size estimates were generated from fish counts in the fish ladder at RBDD (CDFG 2010a). From these counts, estimates of the natural spawner escapement upstream of RBDD were generated. Because RBDD impacted winter-run Chinook salmon by delaying their upstream migration, dam operations were changed in 1993 so that dam gates were raised earlier in the season, which eliminated the need for fish to navigate fish ladders, but also eliminated the ability to generate accurate run-size estimates for the upper Sacramento River Basin (CDFG 2010a).

Presently, little information is available regarding the abundance of steelhead in the Central Valley (CDFG 2010a). Currently there is virtually no coordinated, comprehensive, or consistent monitoring of steelhead in the Central Valley. In 2004, the Interagency Ecological Program Steelhead Project Work Team developed a proposal to develop a comprehensive monitoring plan for Central Valley steelhead. In 2007, development of this steelhead monitoring plan was funded by the CALFED Ecosystem Restoration Program. In 2010, a document titled *A Comprehensive Monitoring Plan for Steelhead in the California Central Valley* was completed by Cal Fish and Wildlife (2010), which recommended steelhead monitoring activities in the Central Valley. The objectives of the plan include: 1) estimate steelhead population abundance with levels of precision; 2) examine trends in steelhead abundance; and 3) identify the spatial distribution of steelhead in the Central Valley to assess their current range and observe changes in their range that may occur over time. However, for the most part, recommendations in the plan remain to be implemented.

According to NMFS (2014a), data are lacking to suggest that the Central Valley steelhead DPS is at low risk of extinction, or that there are viable populations of steelhead anywhere in the DPS. Conversely, there is evidence to suggest that the Central Valley steelhead DPS is at high risk of extinction (Good et al. 2005; Williams et al. 2011; Williams et al. 2016). Most of the historical habitat once available to steelhead has been lost (Yoshiyama et al. 1996; McEwan 2001; Lindley et al. 2006). Furthermore, the observation that anadromous *O. mykiss* are becoming rare in areas where they were probably once abundant indicates that an important component of life history diversity is being suppressed or lost (NMFS 2009a). Habitat fragmentation, degradation, and loss are likely having a strong negative impact on many resident, as well as anadromous, *O. mykiss* populations (Hopelain 2003).

5.2.6.1.1 Viable Salmonid Population (VSP) Parameters and Application

Abundance and Productivity

According to NMFS (2009b) and Cal Fish and Wildlife (2010), there is still a paucity of steelhead monitoring in the Central Valley. Therefore, data are lacking regarding abundance estimates for the steelhead DPS, or for specific steelhead populations in the Central Valley (NMFS 2009b). Recognizing these data limitations, NMFS (2009a) suggested that natural steelhead escapement in the upper Sacramento River declined substantially from 1967 through 1993, and that the little data that do exist indicate that the steelhead population continues to decline. Also, according to Lindley et al. (2007), even if there were adequate data on the distribution and abundance of steelhead in the Central Valley, their approaches for assessing steelhead population and DPS viability might be problematical because the effect of resident *O. mykiss* on the viability of steelhead populations and the DPS is unknown.

Spatial Structure

For the Central Valley steelhead DPS, Lindley et al. (2006) identified historical independent populations based on a model that identifies discrete habitat and interconnected habitat patches isolated from one another by downstream regions of thermally unsuitable habitat. They hypothesized that historically 81 independent populations of steelhead were dispersed throughout the Central Valley domain.

About 80 percent of the habitat that was historically available to steelhead is now behind impassable dams, and 38 percent of the populations have lost all of their habitats (NMFS 2009b). Although much of the habitat has been blocked, or degraded, by impassable dams, small populations of steelhead are still found throughout habitat available in the Sacramento River and many of the tributaries, and some of the tributaries to the San Joaquin River. The current distribution of steelhead is less well understood, but the DPS is composed of at least four diversity groups and at least 26 populations (NMFS 2014).

Remnant steelhead populations are presently distributed through the mainstem of the Sacramento and San Joaquin rivers, as well as many of the major tributaries of these rivers (NMFS 2009a). Steelhead presence in highly variable “flashy” streams and creeks in the Central Valley depend primarily on flow and water temperature, which can change drastically from year to year (McEwan and Jackson 1996). Spawner surveys of small Sacramento River tributaries (Mill, Deer, Antelope, Clear, and Beegum creeks) and incidental captures of juvenile steelhead during Chinook salmon monitoring (Calaveras, Cosumnes, Stanislaus, Tuolumne, and Merced rivers) confirmed that steelhead are widespread, if not abundant, throughout accessible streams and rivers (Good et al. 2005).

Diversity

Steelhead naturally experience the most diverse life history strategies of the listed Central Valley anadromous salmonid species (NMFS 2009b). However, steelhead has less flexibility to track changes in the environment as the species' abundance decreases and spatial structure of the DPS is reduced (NMFS 2009b).

The posited historical existence of 81 independent steelhead populations is likely to be an underestimate because large watersheds that span a variety of hydrological and environmental conditions, such as the Pit River, probably contained multiple populations (Lindley et al. 2006). Regardless, the distribution of many discrete populations across a wide variety of environmental conditions implies that the Central Valley steelhead DPS contained biologically significant amounts of spatially structured genetic diversity (Lindley et al. 2006). However, it appears that much of the historical diversity within Central Valley *O. mykiss* has been lost or is threatened by dams, which have heavily altered the distribution and population structure of steelhead in the Central Valley (Lindley et al. 2006).

Although historically two different runs of steelhead (summer-run and winter-run) occurred in the Central Valley (McEwan and Jackson 1996), the summer run has been largely extirpated due to a lack of suitable holding and staging habitat, such as coldwater pools in the headwaters of Central Valley streams, presently located above impassible dams (Lindley et al. 2006).

Throughout the Central Valley (and in particular the Merced River, Tuolumne River, and upper Sacramento River) it is difficult to discriminate between adult anadromous and resident forms of *O. mykiss,* as well as their progeny (McEwan 2001), further complicating resource management agencies' understanding of steelhead distribution in the Central Valley (CDFG 2008).

The genetic diversity of steelhead also is compromised by hatchery-origin fish. According to Reclamation (2008a), estimates of straying rates only exist for Chinook salmon produced at the FRFH. However, general principles and the potential effects of straying are also applicable for steelhead.

Summary of the Viability of the Central Valley Steelhead DPS

NMFS states that there is evidence to suggest that the Central Valley steelhead DPS is at a high risk of extinction (Williams et al 2016). Steelhead have been extirpated from most of their historical range throughout the Central Valley domain, and most of the historical habitat once available to steelhead is largely inaccessible. Anadromous forms of *O. mykiss* are becoming less abundant or rare in areas where they were probably once abundant, and habitat fragmentation, degradation, and loss are likely having a strong negative impact on many resident as well as anadromous *O. mykiss* populations. In addition, widespread hatchery steelhead production within this DPS also raises concerns about the potential ecological interactions between introduced stocks and native stocks (USACE 2007).

As previously discussed, NMFS completed a 5-year status review of the Central Valley steelhead DPS during May 2016, which reaffirmed much of the information presented in the previous status review. Good et al. (2005) previously found that Central Valley steelhead were in danger of extinction, with a minority of the NMFS BRT viewing the DPS as likely to become endangered. The NMFS BRT's primary concerns for the DPS included the low abundance of naturally-produced anadromous fish at the DPS level, the lack of population-level abundance data, and the lack of information to suggest that the monotonic decline in steelhead abundance evident from 1967-1993 dam counts has stopped (NMFS 2011c).

Steelhead population trend data remain extremely limited (Williams et al. 2011; Williams et al. 2016; NMFS 2016c). The Chipps Island midwater trawl dataset of USFWS provides information on the trend in abundance for the Central Valley steelhead DPS as a whole. Updated through 2014, the trawl data indicate that the natural production of steelhead has continued to be very low since the 2011 status review (NMFS 2016c). Catch-per-unit-effort has fluctuated but remained level over the past decade, but the proportion of the catch that is ad-clipped (100 percent of hatchery steelhead production have been ad-clipped starting in 1998) has risen, exceeding 90 percent in some years and reaching a high of 95 percent in 2010 (NMFS 2011c). Because hatchery releases have been fairly constant, this implies that natural production of juvenile steelhead has been declining (NMFS 2016c).According to NMFS (2016c), steelhead returns to the FRFH were low during 2009 and 2010, with only 312 and 86 fish returning, respectively. Because almost all of the returning fish were of hatchery origin and stocking levels have remained fairly constant over the years, data suggest that adverse freshwater and/or ocean survival conditions caused or at least contributed to these declining hatchery returns (NMFS 2011c). Since then, the numbers have rebounded, with a high of 1,797 in 2013, and have averaged over 1,100 fish over the last five years. According to NMFS (2016c), escapement at the FRFH seems to be quite variable over the years, despite the fact that stocking levels have remained fairly constant and that the vast majority of fish are of hatchery origin. Currently, nearly all the steelhead that return to the FRFH are hatchery-origin fish, indicating that spawning or rearing habitat for steelhead in the Feather River is very poor and natural production is limited (NMFS 2016c). California experienced well below average precipitation in each of the past 4 water years (2012, 2013, 2014 and 2015), record high surface air temperatures the past 2 water years (2014 and 2015), and record low snowpack in 2015. The combination of low precipitation and high temperatures favored elevated stream temperatures, some of which have been documented to be extreme in some watersheds. These conditions would likely have impacted parr and smolt growth and survival. Additionally, much of the northeast Pacific Ocean, including parts typically used by California steelhead, experienced exceptionally high upper ocean temperatures beginning early in 2014 and areas of extremely high ocean temperatures continue to cover most of the northeast Pacific Ocean. According to NMFS (2016a), adult steelhead returns for the next 2 to 3 years (depending on ocean residence times, maturing in 2015, 2016, 2017 and 2018) have likely been and will be negatively impacted by poor stream and ocean conditions.

The steelhead DPS includes two hatchery populations – the FRFH and Coleman National Fish Hatchery. Two additional hatchery populations (i.e., Nimbus and Mokelumne River hatcheries) also are present in the Central Valley, but they were founded from out-of-DPS broodstock and were not considered part of the DPS during the 2011 status review (NMFS 2011c). Genetic information suggests that below-dam populations of *O. mykiss* are similar genetically throughout the Central Valley and that genetic diversity and population structure may have been lost over time. Garza and Pearse (2008) analyzed the genetic relationships among Central Valley *O. mykiss* populations and found that all below-barrier populations were generally closely related, and that there was a high level of genetic similarity to Eel River and Klamath River steelhead in all below-barrier population samples. These findings raise an issue about whether or not the steelhead stocks propagated at the Nimbus and Mokelumne River hatcheries should be excluded from the Central Valley steelhead DPS. These two stocks were excluded from the DPS in 2006 because they originated from the Eel River which is not from within the DPS. Because the Eel River strain appears to be widely introgressed in many Central Valley steelhead populations, NMFS (2011c) stated that it may be appropriate to re-evaluate whether or not these stocks should be in the DPS based upon the new genetic information. Recent genetic work (Pearse and Garza 2015) now shows that steelhead from the Mokelumne Hatchery are nearly identical genetically to steelhead from the FRFH, and are descended from steelhead that are currently part of the DPS. Based on recommendations from the Hatchery Scientific Review Group and an RPA required by the 2009 OCAP BO, planning has begun for the eventual replacement of the out-of-basin broodstock currently used at the Nimbus Hatchery with a more suitable broodstock native to the DPS.

Using data through 2005, Lindley et al. (2007) found the data were insufficient to determine the status of any of the naturally-spawning populations of Central Valley steelhead, except for those spawning in rivers adjacent to hatcheries. These hatchery influenced populations were likely to be at high risk of extinction due to extensive spawning of hatchery-origin fish in natural areas (NMFS 2011c). Continued decline in the ratio between naturally produced juvenile steelhead to hatchery juvenile steelhead in fish monitoring efforts indicates that the wild population abundance is declining. NMFS (2016a) states that it is unclear whether the impacts of hatchery programs have changed in severity since the previous status review. However, new information clearly suggests a loss of genetic diversity and population structure over time. Consequently, impacts from hatcheries continue to be an ongoing threat to the DPS.

Overall, the status of the Central Valley steelhead DPS appears to have worsened since the 2011 status review when the DPS was considered to be in danger of extinction (Good et al. 2005). The general lack of data on the status of wild populations remains a concern. There are some encouraging signs, as several hatcheries in the Central Valley have experienced increased returns of steelhead over the last few years. There has also been a slight increase in the percentage of wild steelhead in salvage at the south Delta fish facilities, and the percentage of wild fish in those data remains much higher than at Chipps Island. The new video counts at Ward Dam show that Mill Creek likely supports one of the best wild steelhead populations in the Central Valley, although at much reduced levels from the 1950's and 60's. Restoration and dam removal efforts in Clear Creek continue to benefit CCV steelhead. However, the catch of unmarked (wild) steelhead at Chipps Island is still less than 5 percent of the total smolt catch, which indicates that natural production of steelhead throughout the Central Valley remains at very low levels. Despite the positive trend on Clear Creek and encouraging signs from Mill Creek, all other concerns raised in the previous status review remain (NMFS 2016c).

One continuing area of strength for the Central Valley steelhead DPS is its widespread spatial distribution throughout most watersheds in the Central Valley. The widespread distribution of wild steelhead in the Central Valley provides the spatial structure necessary for the DPS to survive and avoid localized catastrophes. All of the factors originally identified as being responsible for the decline of this DPS are still present, although in some cases they have been reduced by regulatory actions (e.g., NMFS CVP/SWP OCAP BO in 2009, actions required by CVPIA). Important conservation efforts have been implemented including the 2009 CVP/SWP BO, CVPIA restoration efforts, and continued efforts to implement the Battle Creek Restoration Project that will eventually open up 42 mi of high quality habitat to steelhead (NMFS 2011c). Although these efforts have provided benefits to steelhead and its habitat in the Central Valley, threats from lost habitat and degraded habitat continue to be important factors affecting the status of this DPS.

In summary**,** the most recent biological information suggests that the extinction risk of this DPS has increased since the last status review and that several of the listing factors have contributed to the decline, including recent years of drought and poor ocean conditions (Williams et al. 2016). According to NMFS, there continue to be ongoing threats to the genetic integrity of naturally-spawning steelhead from Central Valley steelhead hatchery programs, but it is unclear if or how this factor has influenced the overall viability of the DPS (Williams et al. 2016; NMFS 2016c). The best available information on the biological status of the DPS and continuing and new threats to the DPS indicate that its ESA status as a threatened species is appropriate (Williams et al. 2016; NMFS 2016c).

5.2.6.2 Lower Yuba River

As with all naturally-spawning populations of steelhead in the Central Valley, Lindley et al. (2007) characterized the steelhead population in the lower Yuba River as data deficient, and therefore did not characterize its viability. Data limitations, particularly regarding abundance and productivity, continue to render problematic quantitative estimation procedures to assess the viability of the steelhead population in the lower Yuba River. Continued monitoring of adult steelhead in the lower Yuba River is providing additional information that is needed to assess extinction risk based on Lindley et al. (2007) criteria regarding population size, recent population decline, occurrences of catastrophes within the last 10 years that could cause sudden shifts from a low risk state to a higher one, and the impacts of hatchery influence. The VSP parameters of abundance, productivity, spatial structure and diversity for the steelhead population in the lower Yuba River are discussed below.

5.2.6.2.1 Abundance and Productivity

VAKI Riverwatcher™ Data

Ongoing monitoring of the adult steelhead population in the lower Yuba River has been conducted since 2003 with VAKI Riverwatcher™ systems at Daguerre Point Dam. By contrast to Chinook salmon, escapement surveys involving carcass mark-recovery experiments are not performed on steelhead/*O. mykiss*.

In the lower Yuba River, silhouettes and corresponding photographs were examined for species identification and categorization using methodology similar to that which is described for spring-run Chinook salmon. However, the accurate identification of *O. mykiss* in the VAKI Riverwatcher™ is more difficult than it is for Chinook salmon.

By contrast to the identification of Chinook salmon which may be conducted with a single attribute, the identification of steelhead becomes more problematic with the absence of a defining silhouette or a clear digital photograph. Additionally, the silhouettes of steelhead cannot reliably be differentiated from resident rainbow trout, and photo documentation of an individual is problematic because adult steelhead typically immigrate during periods of high flow and associated high turbidity and low visibility. The VAKI Riverwatcher™ systems cannot differentiate an individual as a resident form of the species (i.e., rainbow trout) or as anadromous (i.e., steelhead). Additionally, the VAKI Riverwatcher™ systems cannot directly distinguish between an adult or juvenile *O. mykiss* (RMT 2013a).

The following sections present analyses of steelhead VAKI Riverwatcher™ data conducted by the RMT (2013a) for steelhead biological years 2003/2004 through 2011/2012, updated by YCWA for this Applicant-Prepared Draft BA with additional data available from August 2012 through June 13, 2016, corresponding to the period of data availability. For more detailed explanation of the methodologies discussed in this section, refer to RMT (2013a).

Differentiation of Adult Steelhead VAKI Riverwatcher™ Counts

The silhouettes and/or electronic images of each fish passage event that was identified as an *O. mykiss* fish passage event allow the VAKI Riverwatcher™ systems to calculate an approximate length (in centimeters) for the observed fish.

As an initial step in the differentiation of adult steelhead passing upstream of Daguerre Point Dam from resident rainbow trout or juvenile *O. mykiss*, the length distribution of all fish identified as *O. mykiss* passing through both the north and south ladders at Daguerre Point Dam for the period extending from January 1, 2004 through March 31, 2015was plotted and visually examined (Figure 5.2-1). This figure indicates the possible presence of several potential length groups. These groups represent the potential combination of juvenile and adult anadromous *O. mykiss* (steelhead), as well as juvenile and adult resident *O. mykiss* (rainbow trout). However, this length-frequency distribution does not provide information necessary to differentiate between steelhead and rainbow trout.

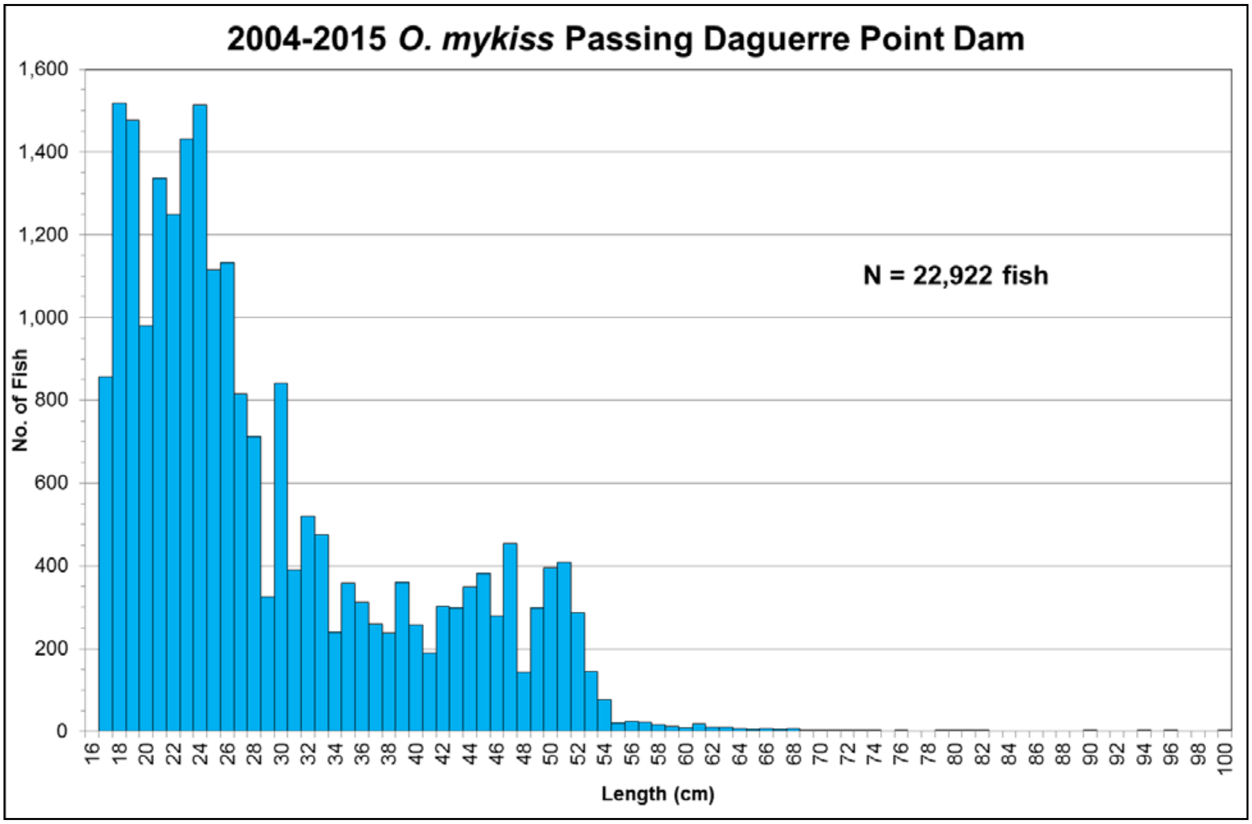

Figure 5.2-1. Length-frequency distribution of all fish identified by the VAKI Riverwatcher™ systems as *O. mykiss* passing upstream through the north and south ladders of Daguerre Point Dam from January 1, 2004 through March 31, 2015.

Prior to March 1, 2009, digital photographic or video imagery was not available for consistent use with the VAKI Riverwatcher™. However, beginning on March 1, 2009, VAKI Riverwatcher™ fish identified as *O. mykiss* also were classified as fish with or without clipped adipose fins, based on the inspection of the fish silhouette and photogrammetric representation (digital photographs and/or video imagery). The analysis of the length-frequency distribution of all adipose fin-clipped *O. mykiss* provides a means of differentiating adult hatchery steelhead passing upstream of Daguerre Point Dam from all other *O. mykiss*, because all adipose fin-clipped *O. mykiss* are steelhead that were released by a Central Valley hatchery.

The lengths of all fish passing upstream at Daguerre Point Dam that were identified as *O. mykiss* with clipped adipose fins (i.e., all hatchery steelhead) between March 1, 2009 through March 31, 2015 are presented in Figure 5.2-2. Visual examination of the observed length-frequency distribution indicates the possible presence of several possible groups of fish. An initial demarcation mode appeared to have occurred at approximately 11.4 inches (29 cm) or 12.2 inches (31 cm).

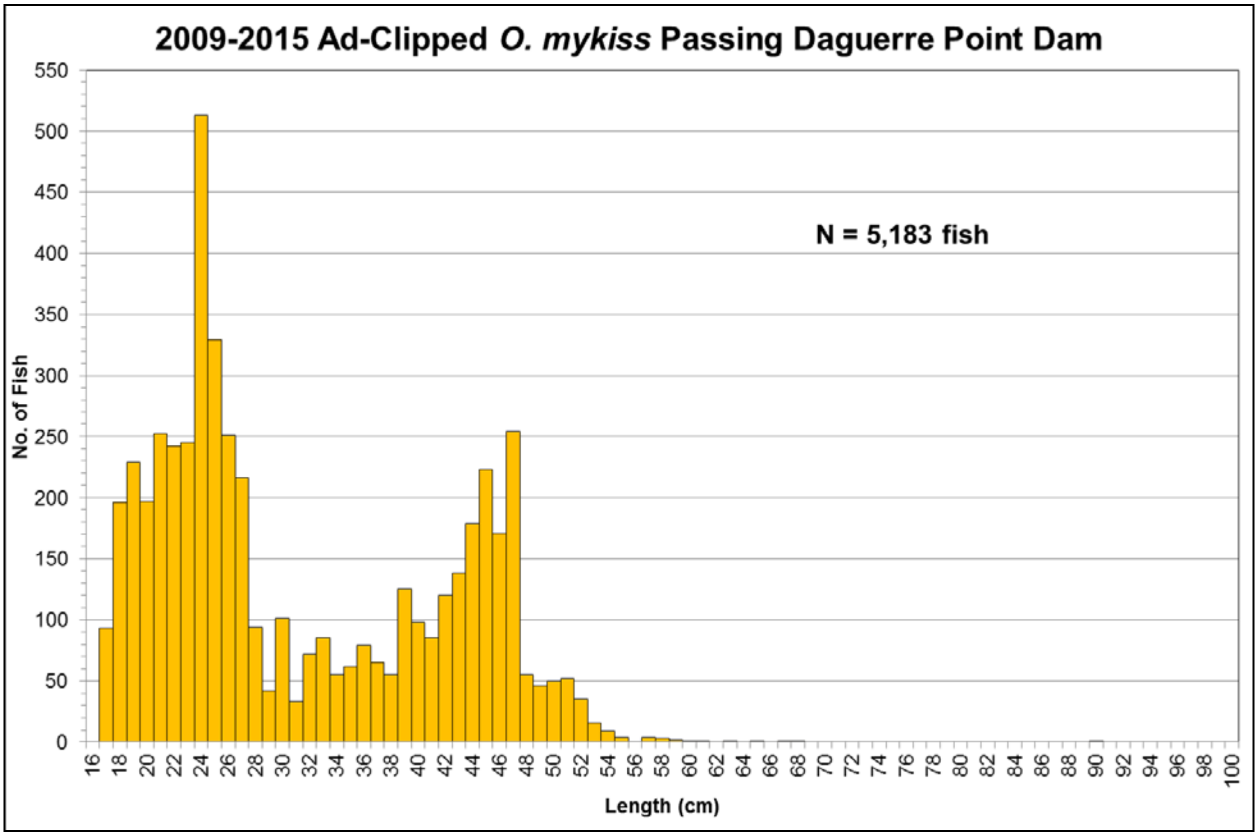

Figure 5.2-2. Length-frequency distribution of all fish identified by the VAKI Riverwatcher™ systems as adipose clipped *O. mykiss* passing upstream through the north and south ladders of Daguerre Point Dam from March 1, 2009 through March 31, 2015.

According to Cal Fish and Wildlife and USFWS (2010), the normal FRFH release schedule includes the release of steelhead yearlings, from January to February, released in the Feather River near Gridley at four fish per pound. Although not readily available from Cal Fish and Wildlife, other sources indicate that steelhead smolts averaging four to five fish per pound range in length from approximately 8-9 in (20-23 cm) (IDFG 1992). The presence of small, adipose fin-clipped steelhead in the lower Yuba River as displayed in Figure 5.2-2 may be related to releases of yearling FRFH-produced steelhead on the Feather River. The following analyses and discussion was initially presented in RMT (2013a).

Since 2007, the FRFH has been releasing only steelhead yearlings at various sites along the Feather River, as well as in the Sacramento River at Sutter Slough, and in Butte Creek (Table 5.2-3). To determine whether fish planted in the lower Feather River may have been detected in the lower Yuba River, an examination of the VAKI Riverwatcher™ data was conducted for adipose fin-clipped steelhead consistent with the observed potential length-mode demarcation length of 11.4 in (29 cm) (RMT 2013a).

Table 5.2-3. Releases of hatchery steelhead by the Feather River Fish Hatchery.

Release Dates		Brood Year	Numbers Released		Release Stage[2]	Study Type[3]	Release Location	Agency	
Start	End		Tagged[1] Ad clipped	Untagged Ad clipped				Reporting	Release
01/08/07	02/05/07	2006	0	10,036	Y	E	Feather River Thermalito Bypass	CDFW	DWR
02/05/07	02/21/07	2006	0	488,043	Y	E	Feather River	CDFW	DWR
05/29/07	05/29/07	2006	0	1,643	Y	E	Feather River	CDFW	DWR
02/01/08	02/14/08	2007	0	307,986	Y	P	Feather River Boyds Pump Ramp	CDFW	DWR
05/30/08	05/30/08	2007	0	1,109	Y	E	Feather River	CDFW	DWR
02/03/09	02/03/09	2008	0	2,750	Y	P	Feather River at Live Oak	CDFW	CDFW
02/03/09	02/03/09	2008	0	398,148	Y	P	Feather River Boyds Pump Ramp	CDFW	CDFW
02/01/10	02/11/10	2009	0	272,798	Y	P	Feather River Boyds Pump Ramp	CDFW	CDFW
02/02/11	02/15/11	2010	0	49,800	Y	P	Feather River Boyds Pump Ramp	CDFW	CDFW

Source: Regional Mark Information System (RMIS) of the Regional Mark Processing Center; RMT 2013a
1 Tagged releases refer to releases with coded wire tags.
2 Release stage Y indicates yearling releases.
3 Study type E stands for experimental releases, and study type P indicates a production releases.

From February 1, 2010 to February 2, 2011 (i.e., the starting date for the last reported release of adipose fin-clipped juvenile steelhead from the FRFH), 104 adipose fin-clipped juvenile steelhead with lengths less than or equal to 11.4 in (29 cm)were recorded passing upstream of Daguerre Point Dam. Most of these individuals were observed in the VAKI Riverwatcher™ system during February through April of 2010. Additionally, from February 2, 2011 through January 31, 2012, a total of 1,702 adipose fin-clipped steelhead with lengths less than or equal to 11.4 in (29 cm) were recorded passing upstream of Daguerre Point Dam. While these individuals were observed in the VAKI Riverwatcher™ system throughout calendar year 2011, they were most frequently observed during April and May of 2011. In other words, most of the observed adipose fin-clipped juvenile steelhead less than or equal to 11.4 in (29 cm) passing upstream of Daguerre Point Dam occurred within a few months after plantings of juvenile steelhead in the Feather River from the FRFH. Additionally, between February 2011 and January 2012, approximately 676 adipose fin-clipped steelhead with lengths less than or equal to 11.4 in (29 cm) were recorded passing downstream of Daguerre Point Dam, with the majority of these individuals passing downstream during April through June. Therefore, approximately one-third of the presumed FRFH steelhead that migrated upstream of Daguerre Point Dam during 2011 apparently turned around and migrated back downstream of Daguerre Point Dam shortly after passing upstream of Daguerre Point Dam (RMT 2013a).

If the observation of adipose fin-clipped juvenile steelhead passing upstream at Daguerre Point Dam is associated with the release of yearling steelhead from the FRFH into the lower Feather River, then it logically follows that the planted FRFH yearling steelhead would have had to swim 6 mi upstream from the planting location at Boyds Pump Ramp to the mouth of the lower Yuba River, and then an additional nearly 12 mi upstream in the lower Yuba River to reach Daguerre

Point Dam. Although this phenomenon may seem illogical, it has been reported elsewhere (Steiner Environmental Consulting 1987, as cited in RMT 2013a) and is an explanation for the observation of adipose fin-clipped juvenile steelhead passing upstream at Daguerre Point Dam, because no marked juvenile steelhead have been reported to be released over this time frame into the lower Yuba River.

The length-frequency distribution of all adipose fin-clipped steelhead observed at Daguerre Point Dam from March 1, 2009 through March 31, 2015 was used to differentiate between "juvenile" *O. mykiss* and "adult" steelhead. Modeled length-frequency distributions were fit to the observed data to determine a threshold length to separate both fish groups. Consequently, the recorded lengths of fish identified as *O. mykiss* passing through Daguerre Point Dam from January 1, 2004 through March 31, 2015 were classified as adult steelhead if the recorded length was 16 inches or higher. If the recorded lengths of fish identified as *O. mykiss* passing through Daguerre Point Dam were less than 16 inches, then the fish were considered to be "other" *O. mykiss* (e.g., juvenile or adult rainbow trout, or juvenile steelhead). A detailed description of the analytical processes is provided in RMT (2013a).

Unlike the methodology employed by the RMT (2013a) for Chinook salmon, the daily counts of adult steelhead passing upstream of Daguerre Point Dam were not corrected for days when the VAKI Riverwatcher™ systems were not fully operational. The RMT determined it would be inappropriate to attempt to correct the adult steelhead counts due to: 1) the relatively low numbers of adult steelhead recorded during most of the steelhead biological years; and 2) the frequently extended durations when the VAKI Riverwatcher™ systems were not fully operational during the steelhead immigration season. Instead, the daily counts of adult steelhead passing upstream at Daguerre Point Dam were used to represent the abundance of steelhead, with the understanding that the resultant estimates are minimum numbers, and most of the survey years considerably underestimate the potential number of steelhead because the annual estimates do not include periods of VAKI Riverwatcher™ system non-operation, and do not consider the fact that not all steelhead migrate past Daguerre Point Dam, due to some spawning occurring downstream of Daguerre Point Dam (RMT 2013a).

Assessment of Available VAKI Riverwatcher™ Data

For assessment purposes, a "steelhead biological year" was identified as extending from August 1 through July 31 each year, because: 1) preliminary review of the VAKI Riverwatcher™ data indicated a general paucity of upstream migrant *O. mykiss* during early summer; 2) the immigration of adult steelhead in the lower Yuba River has been reported to occur beginning during August (CALFED and YCWA 2005; McEwan and Jackson 1996); and 3) RMT (2010b) identified the steelhead upstream migration period as beginning during August in the lower Yuba River (RMT 2013a).

Annual Time Series of Steelhead Passing Upstream of Daguerre Point Dam

Figures 5.2-3 through 5.2-15 illustrate the daily counts of adult steelhead passing upstream at Daguerre Point Dam through both the North and South ladders combined, and the percentage of the daily number of hours when the VAKI Riverwatcher™ systems were operational at both ladders, during the 13 steelhead biological years (data are only available for August 1, 2015 through June 13, 2016 for biological year 2015/2016).

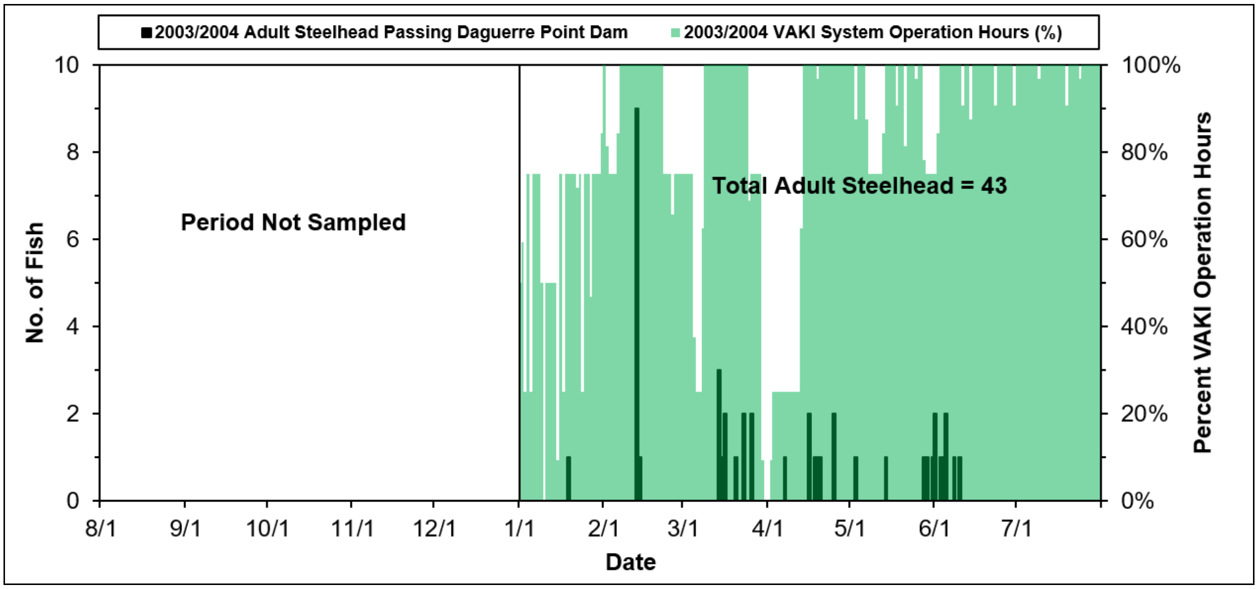

Figure 5.2-3. Daily counts of adult steelhead passing upstream of Daguerre Point Dam (dark green bars), and daily number of hours when the VAKI Riverwatcher™ systems were operational (light green bars), during the 2003/2004 Steelhead Biological Year (August 1 through July 31).

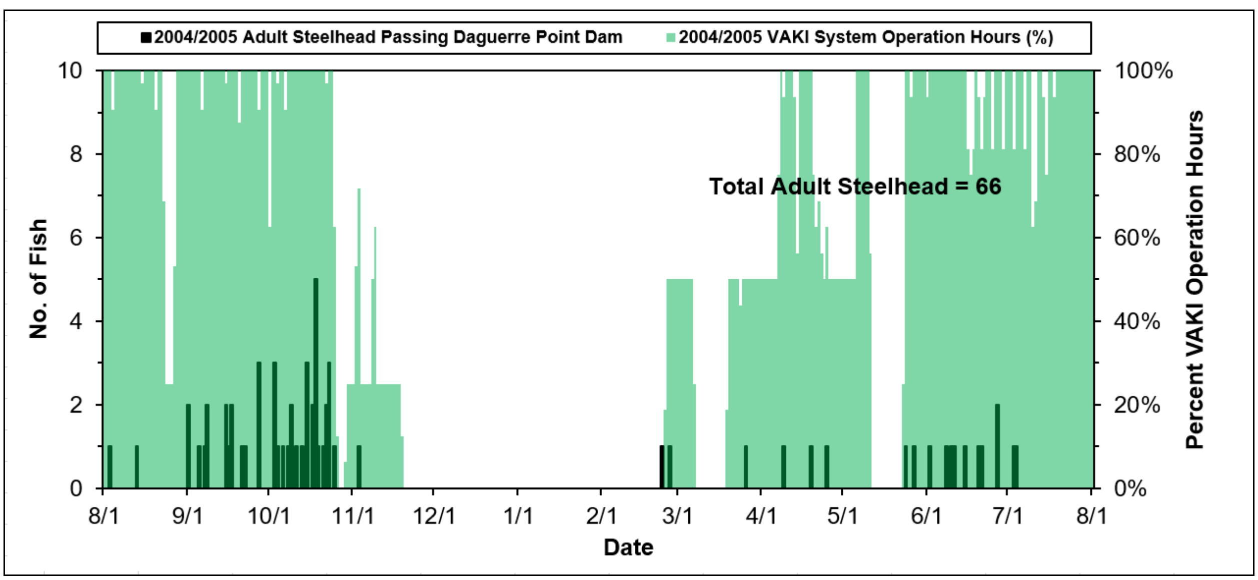

Figure 5.2-4. Daily counts of adult steelhead passing upstream of Daguerre Point Dam (dark green bars), and daily number of hours when the VAKI Riverwatcher™ systems were operational (light green bars), during the 2004/2005 Steelhead Biological Year (August 1 through July 31).

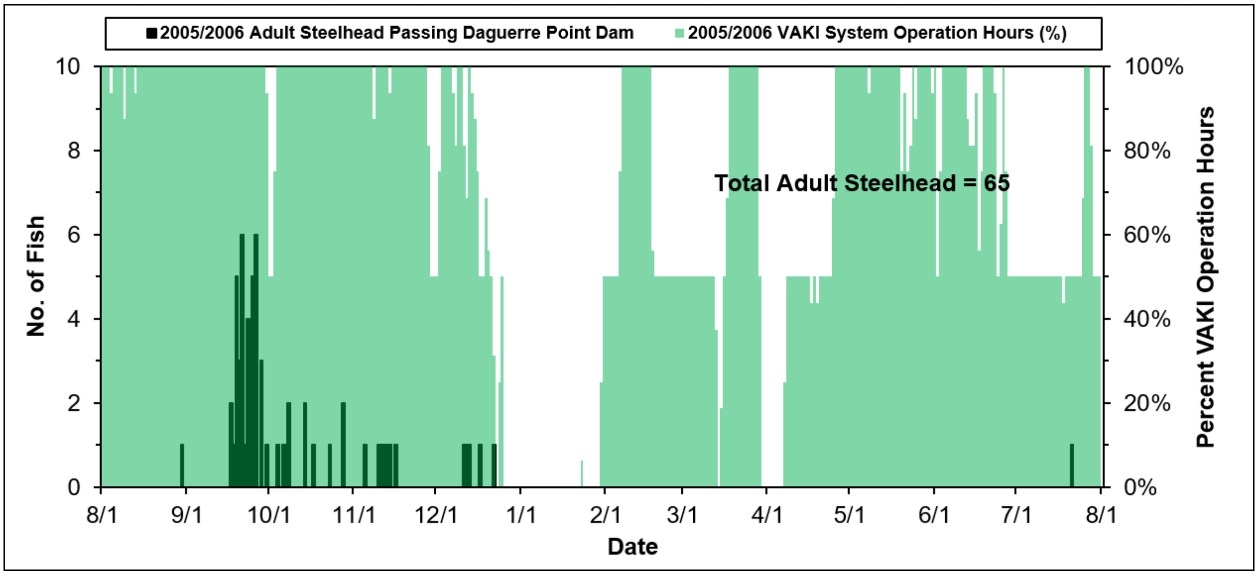

Figure 5.2-5. Daily counts of adult steelhead passing upstream Daguerre Point Dam (dark green bars), and daily number of hours when the VAKI Riverwatcher™ systems were operational (light green bars), during the 2005/2006 Steelhead Biological Year (August 1 through July 31).

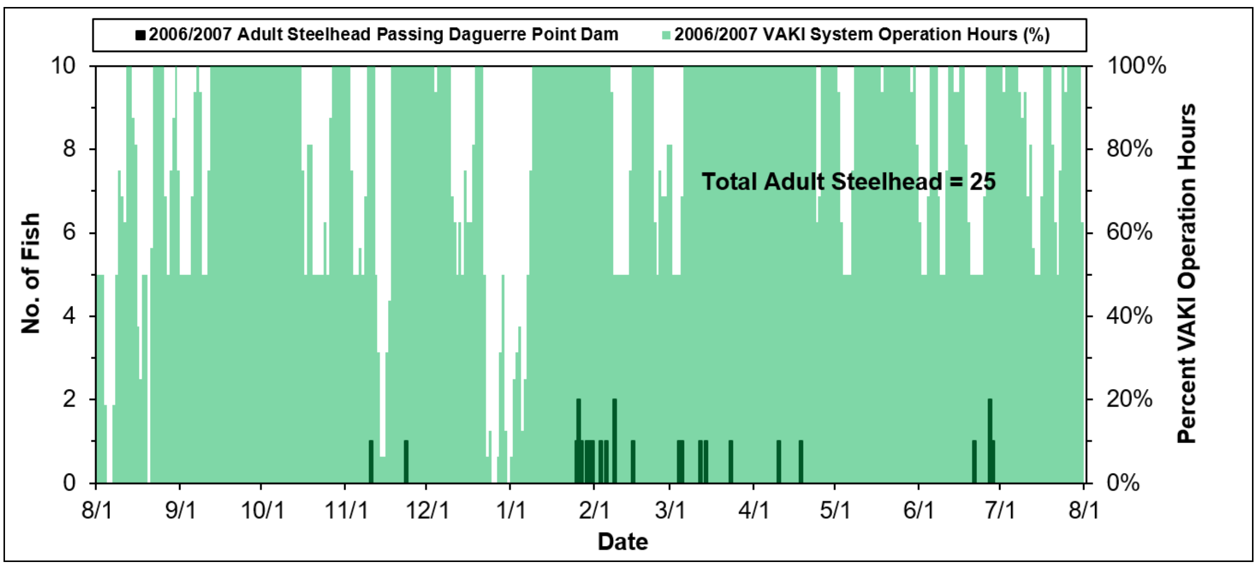

Figure 5.2-6. Daily counts of adult steelhead passing upstream Daguerre Point Dam (dark green bars), and daily number of hours when the VAKI Riverwatcher™ systems were operational (light green bars), during the 2006/2007 Steelhead Biological Year (August 1 through July 31).

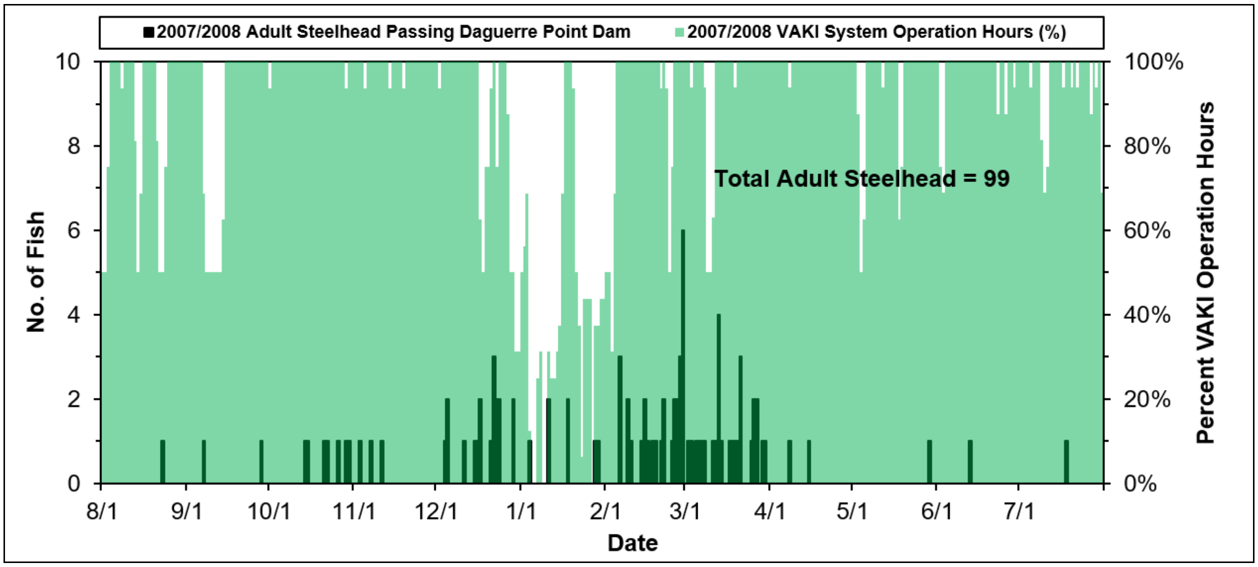

Figure 5.2-7. Daily counts of adult steelhead passing upstream of Daguerre Point Dam (dark green bars), and daily number of hours when the VAKI Riverwatcher™ systems were operational (light green bars), during the 2007/2008 Steelhead Biological Year (August 1 through July 31).

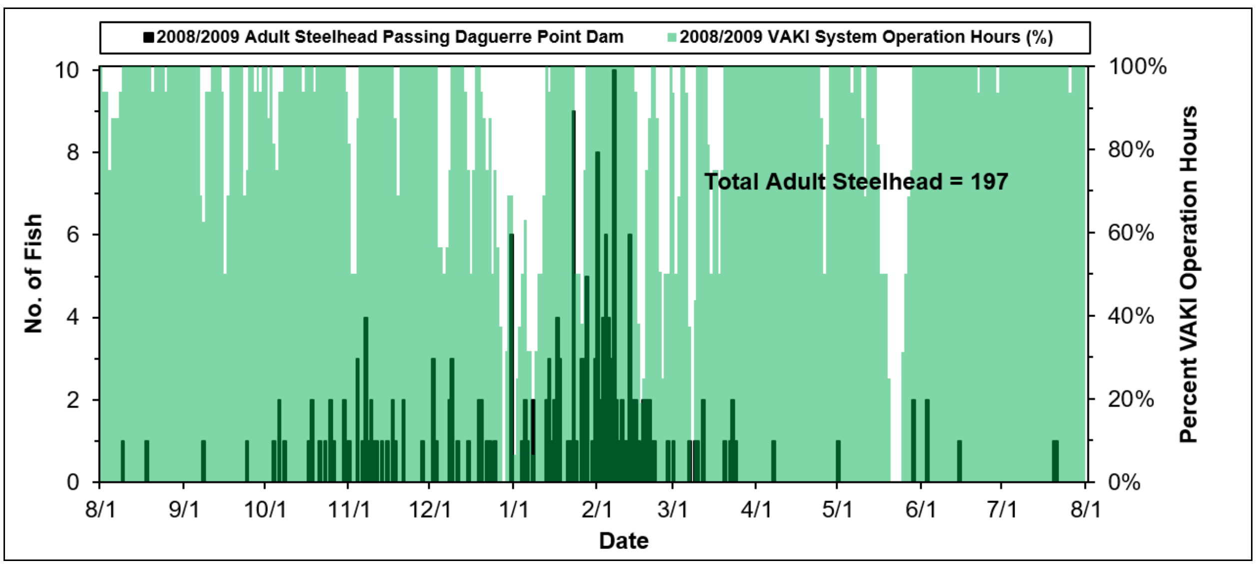

Figure 5.2-8. Daily counts of adult steelhead passing upstream of Daguerre Point Dam (dark green bars), and daily number of hours when the VAKI Riverwatcher™ systems were operational (light green bars), during the 2008/2009 Steelhead Biological Year (August 1 through July 31).

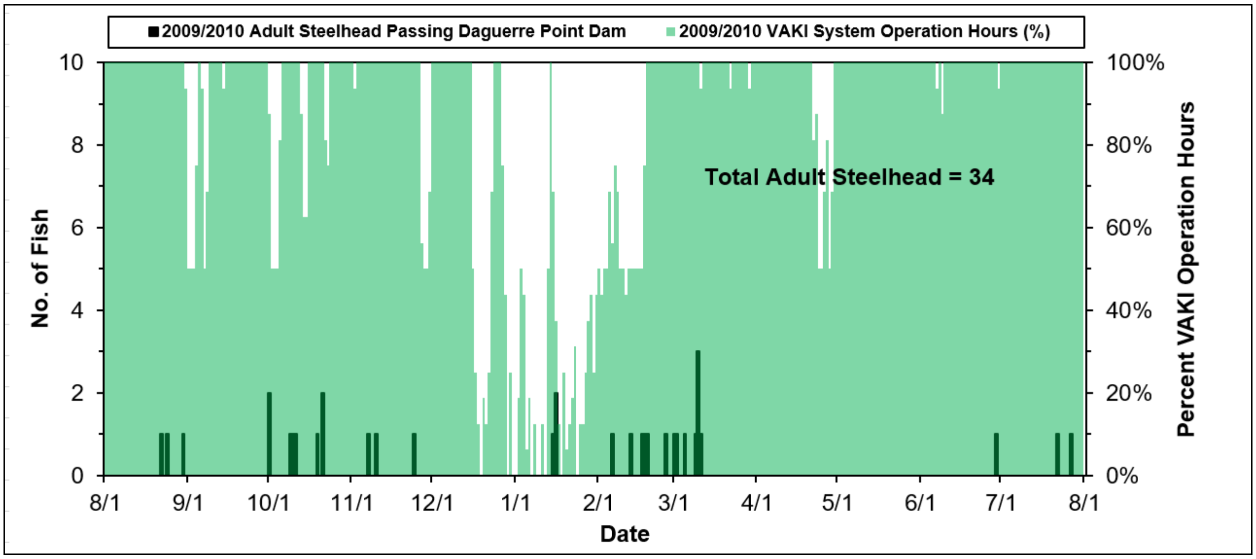

Figure 5.2-9. Daily counts of adult steelhead passing upstream of Daguerre Point Dam (dark green bars), and daily number of hours when the VAKI Riverwatcher™ systems were operational (light green bars), during the 2009/2010 Steelhead Biological Year (August 1 through July 31).

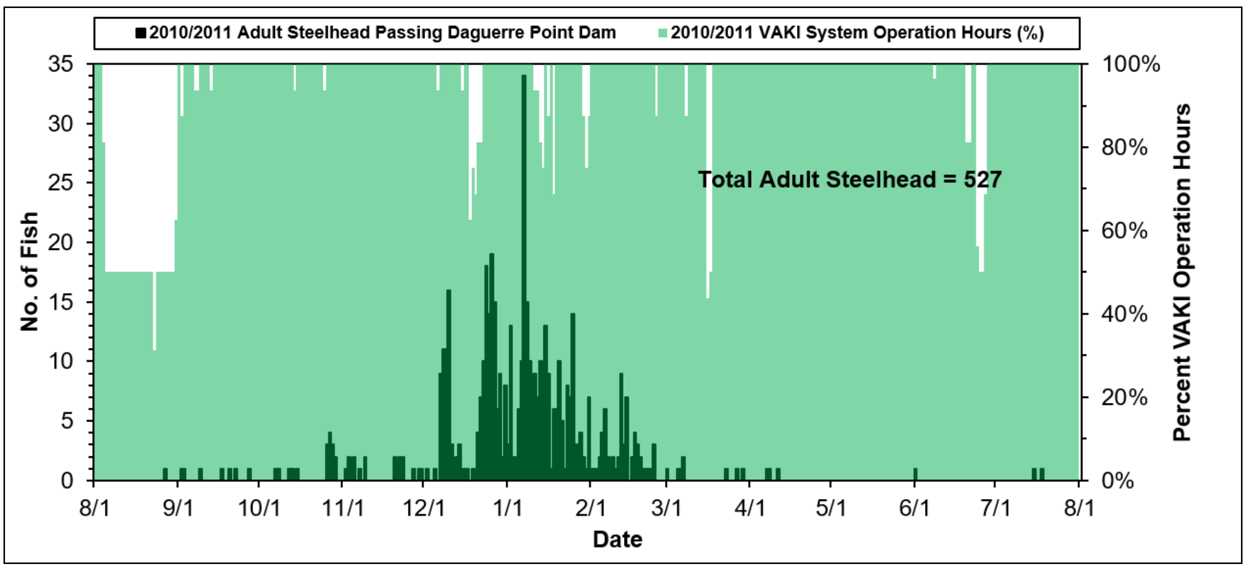

Figure 5.2-10. Daily counts of adult steelhead passing upstream of Daguerre Point Dam (dark green bars), and daily number of hours when the VAKI Riverwatcher™ systems were operational (light green bars), during the 2010/2011 Steelhead Biological Year (August 1 through July 31).

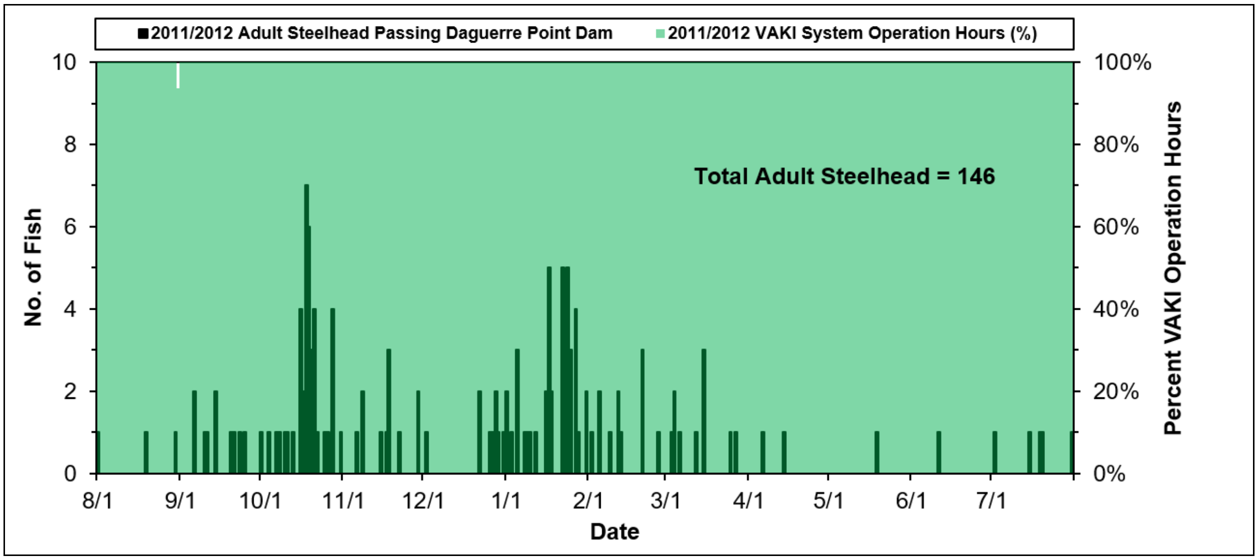

Figure 5.2-11. Daily counts of adult steelhead passing upstream of Daguerre Point Dam (dark green bars), and daily number of hours when the VAKI Riverwatcher™ systems were operational (light green bars), during the 2011/2012 Steelhead Biological Year (August 1 through July 31).

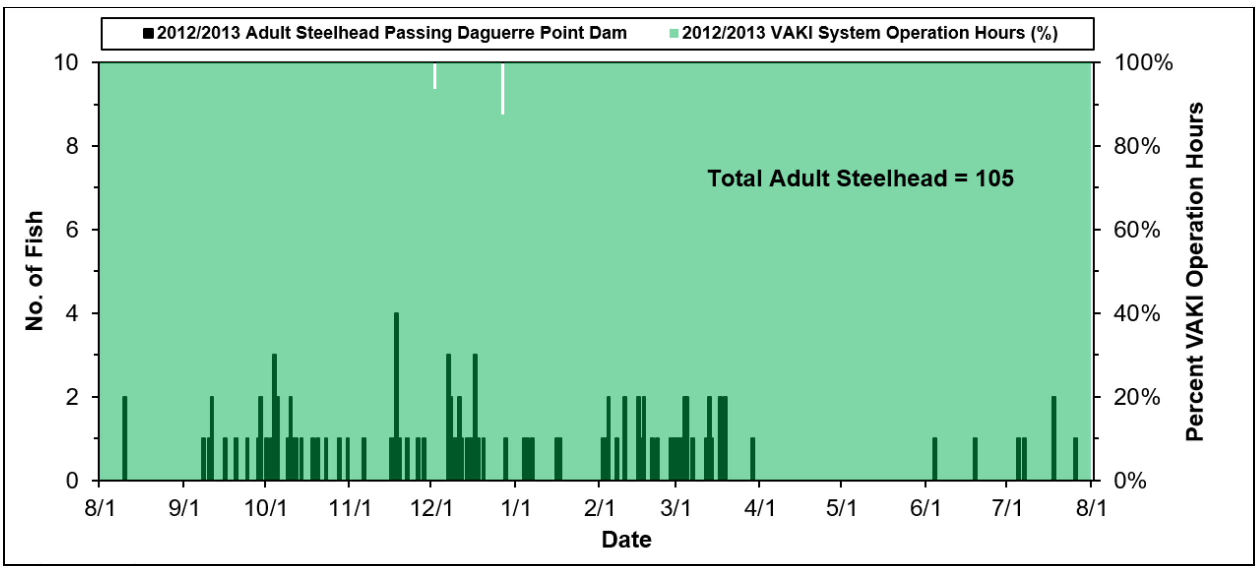

Figure 5.2-12. Daily counts of adult steelhead passing upstream of Daguerre Point Dam (dark green bars), and daily number of hours when the VAKI Riverwatcher™ systems were operational (light green bars), during the 2012/2013 Steelhead Biological Year (August 1 through July 31).

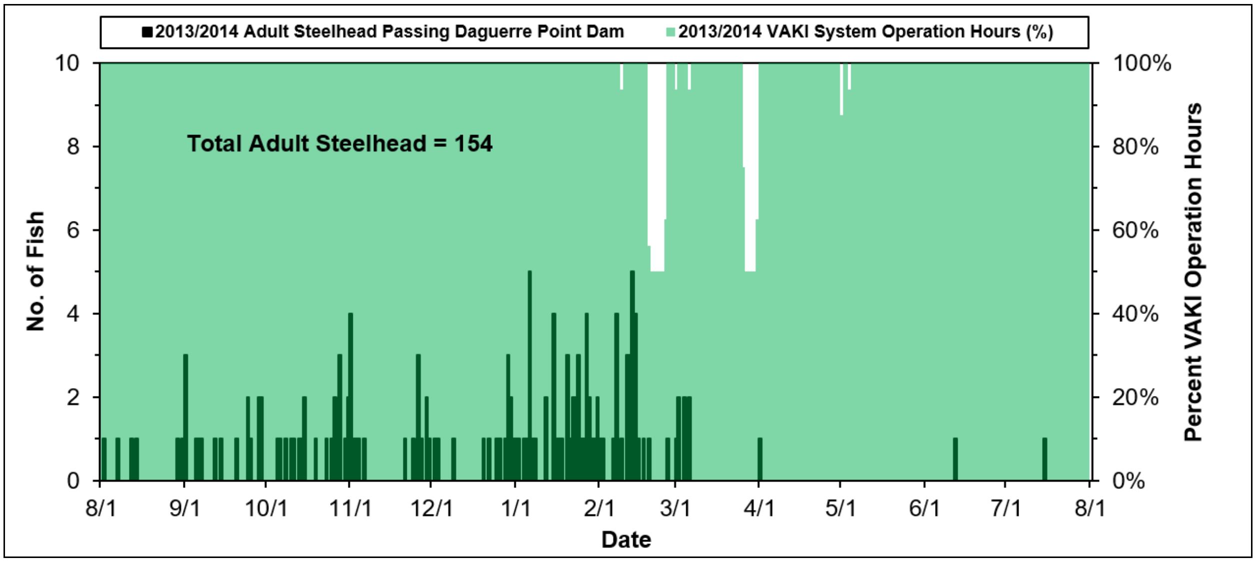

Figure 5.2-13. Daily counts of adult steelhead passing upstream of Daguerre Point Dam (dark green bars), and daily number of hours when the VAKI Riverwatcher™ systems were operational (light green bars), during the 2013/2014 Steelhead Biological Year (August 1 through July 31).

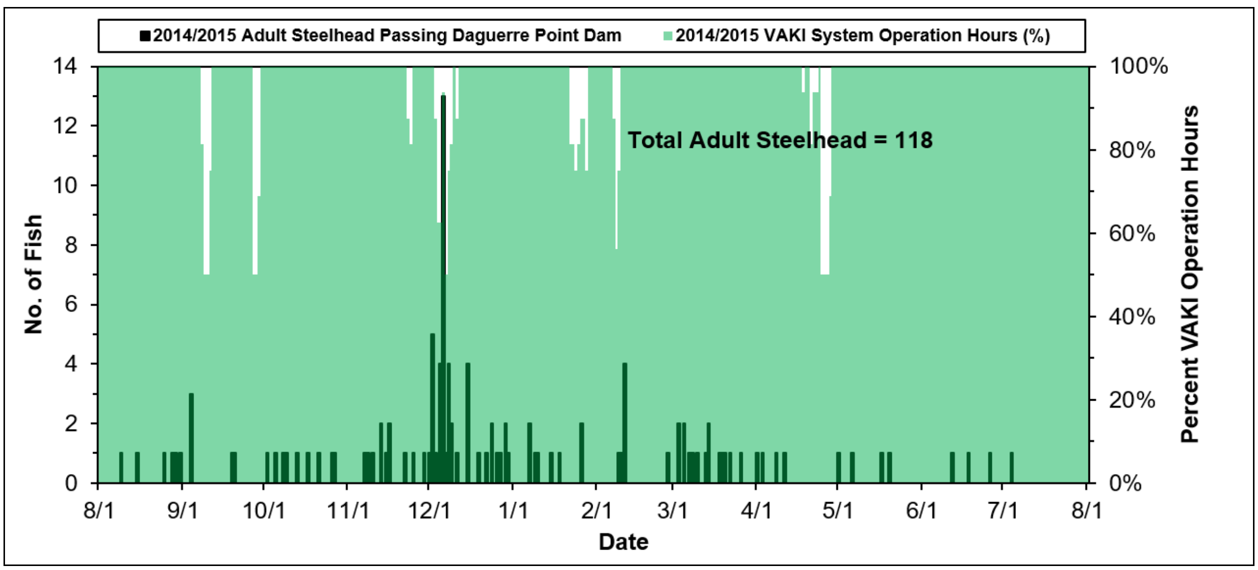

Figure 5.2-14. Daily counts of adult steelhead passing upstream of Daguerre Point Dam (dark green bars), and daily number of hours when the VAKI Riverwatcher™ systems were operational (light green bars), during the portion of the 2014/2015 Steelhead Biological Year (August 1 through July 31).

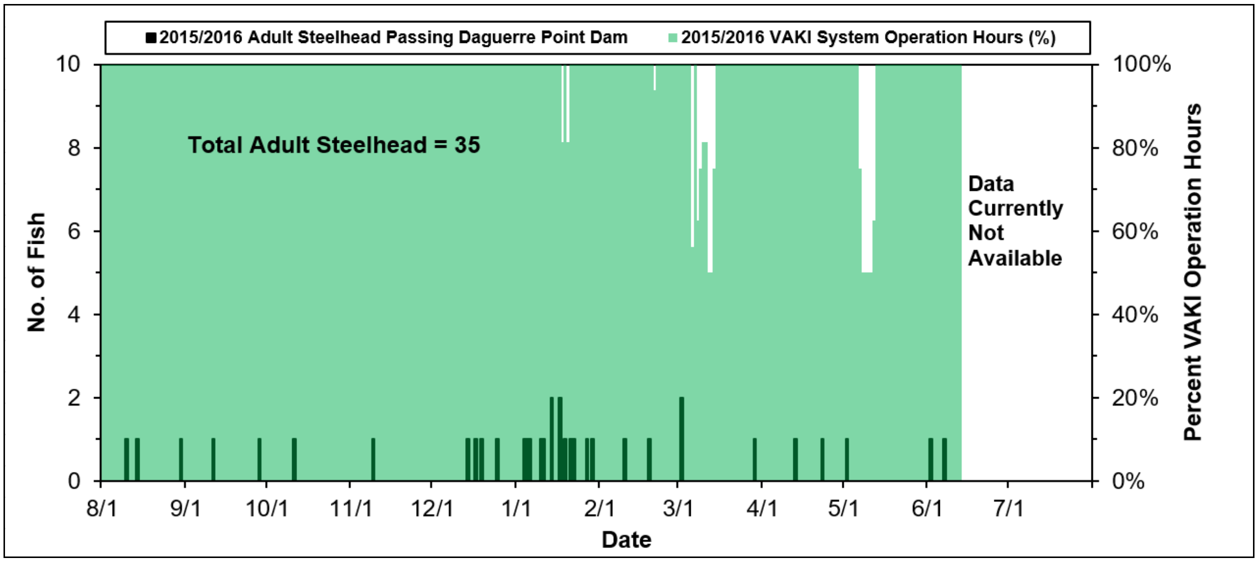

Figure 5.2-15. Daily counts of adult steelhead passing upstream of Daguerre Point Dam (dark green bars), and daily number of hours when the VAKI Riverwatcher™ systems were operational (light green bars), during the portion of the 2015/2016 Steelhead Biological Year that is currently available (August 1 through June 13).

Examination of Figures 5.2-3 through 5.2-15 demonstrates that although the VAKI Riverwatcher™ systems have been in place since June of 2003, reliable estimates of the number of adult steelhead passing upstream at Daguerre Point Dam are essentially restricted to the most recent nearly six years of available data (2010/2011, 2011/2012, 2012/2013, 2013/2014, 2014/2015 and the partial biological year of 2015/2016).

Due to system failures, including equipment malfunctions and operationally detrimental environmental conditions (heavy overcast and foggy conditions resulting in lack of photovoltaic charging of the system), the VAKI Riverwatcher™ systems were partially operational or completely non-operational during several months each year of sampling. Additionally, high flows and turbidities reduced the ability of the system to identify, or prevented the system from identifying, adult steelhead frequently even when the systems were operational. Although improvements to the system have been made over time, it was not until the most recent system improvements were implemented during the 2010/2011 sampling season that the system began demonstrating sustained reliability in the documentation of steelhead passing upstream of Daguerre Point Dam, over a range of environmental conditions.

Since June 2003, numerous improvements have been implemented to improve the reliability of the VAKI Riverwatcher™ systems, and particularly their ability to document passage during the steelhead upstream migration season. A chronology of the VAKI Riverwatcher™ system improvements that have occurred over time are described in RMT (2013a). As a result, it is not reasonable to consider data gathered prior to 2010/2011 to be reliable estimates of the annual number of adult steelhead passing upstream of Daguerre Point Dam (RMT 2013a).

As stated approximately 11 years ago by Lindley et al. (2006), there are almost no data with which to assess the status of any of the Central Valley steelhead populations, with the exceptions of the hatchery programs on Battle Creek and the Feather, American and Mokelumne rivers. Therefore, they classified Central Valley steelhead populations as data deficient. As of 2010, CDFG (2010) stated that steelhead monitoring programs in the Central Valley lack statistical power, are not standardized and in many cases lack dedicated funding.

The relatively short time period encompassed by the reporting of reliable abundance estimates, and in consideration that steelhead may have returned to the lower Yuba River but remained and spawned in the river downstream of Daguerre Point Dam, currently render problematic the determination of abundance or trends in the productivity of the steelhead over recent years (RMT 2013a). Continued implementation of the improved VAKI Riverwatcher™ systems at Daguerre Point Dam is likely to obtain some of the data necessary to allow abundance estimation and productivity evaluation of steelhead in the lower Yuba River. However, as was previously stated by RMT (2013a), presently the lack of multi-year abundance data precludes the provision of quantitative values associated with extinction risk assessment, addressing abundance and productivity.

Spatial Structure

Spatial structure and considerations regarding anadromous salmonid viability was presented for spring-run Chinook salmon previously in this Applicant-Prepared Draft BA. The spatial structure considerations, as one of the four VSP parameters, for steelhead are analogous to those for spring-run Chinook salmon previously presented. Namely, spatial structure of morphological units in the lower Yuba River is complex, diverse, and persistent.

Diversity

<u>Phenotypic Considerations</u>

O. mykiss in the lower Yuba River exhibit a high amount of diversity in phenotypic expression and life history strategy. As demonstrated in Figures 5.2-3 - 5.2-15, *O. mykiss* categorized as adult steelhead exhibit a broad temporal distribution in passing upstream of Daguerre Point Dam. *O. mykiss* (including steelhead) exhibit highly diverse spatial and temporal distributions in patterns of spawning, and juvenile outmigration (RMT 2013a). Moreover, *O. mykiss* in the lower Yuba River exhibit polyphenism, or the occurrence of several phenotypes in a population which may not be due to different genetic types, including expressions of anadromy or residency. A thorough discussion of anadromy vs. residency of *O. mykiss* in the lower Yuba River is provided in RMT (2013a). A polymorphic *O. mykiss* population structure may be necessary for the long-term persistence in highly variable environments such as the Central Valley (McEwan 2001). Resident fish may reduce extinction risk through the production of anadromous individuals that can enhance weak steelhead populations (Lindley et al. 2007). Such considerations may be applicable to the *O. mykiss* populations in the lower Yuba River.

Genetic Considerations

Although no fish hatcheries have been located on the Yuba River since 1950, and the lower Yuba River continues to support a persistent population of steelhead, the genetic integrity of these fish is presently uncertain. According to the NMFS (2014a) Recovery Plan, the major threat to the genetic integrity of Central Valley steelhead results from past and present hatchery practices. These practices include the planting of non-natal fish, overlap of spawning hatchery and natural fish, and straying of hatchery fish.

The observation of adipose fin clips on adult steelhead passing upstream through the VAKI Riverwatcher™ system at Daguerre Point Dam demonstrates that hatchery straying into the lower Yuba River occurs. Although no information is presently available regarding the origin of adipose-clipped steelhead observed at the VAKI Riverwatcher™ system at Daguerre Point Dam, it is reasonable to surmise that they most likely originate from the FRFH.

As previously stated, analysis of the VAKI Riverwatcher™ data indicates that the percent contribution of hatchery-origin adult upstream migrating fish (represented by the percentage of adipose fin-clipped adult steelhead relative to the total number of adult upstream migrating steelhead) was approximately 42 percent for the 2010/2011 biological year, about 62 percent for the 2011/2012 biological year, about 38 percent for the 2012/2013 biological year, about 55 percent for the 2013/2014 biological year, about 42 percent for the 2014/2015 biological year, and about 40 percent for the currently available data (i.e., August 2015 through June 2016) of the 2015/2016 biological year. If hatchery-origin steelhead stray into the lower Yuba River and interbreed with naturally-spawning Yuba River steelhead, then such interbreeding has been suggested to represent a threat to the genetic diversity and integrity of the naturally-spawning steelhead population in the lower Yuba River. Nonetheless, the question remains regarding the implication of straying of hatchery-origin adult steelhead into the lower Yuba River, given past management practices. From 1970 to 1979, Cal Fish and Wildlife annually stocked 27,270–217,378 fingerlings, yearlings, and sub-catchable steelhead from Coleman National Fish Hatchery into the lower Yuba River (CDFG 1991b). Cal Fish and Wildlife stopped stocking steelhead into the lower Yuba River in 1979. In addition, as previously discussed, it is possible that some hatchery-reared juvenile steelhead from the FRFH may move into the lower Yuba River in search of rearing habitat. Some competition for resources with naturally spawned steelhead could occur as a result.

As previously discussed, Garza and Pearse (2008) studied populations of *O. mykiss* in the Central Valley using molecular genetic techniques to provide insight into population structure in the region. Their analyses suggest that the below-barrier populations in this region appear to have been widely introgressed with hatchery fish from out-of-basin broodstock sources. In phylogeographic analyses, above-barrier populations are more similar to San Francisco Bay *O. mykiss* populations than the below-barrier populations in the Central Valley. The analyses also identified possible heterogeneity between samples from different tributaries of the upper Yuba and Feather Rivers, although linkage disequilibrium was lower in these populations. Linkage disequilibrium can be caused by physical linkage of loci, sampling of related individuals/family structure, and by the sampling of more than one genetically distinct group within a population sample (Garza and Pearse 2008).

In general, although genetic structure was found, all naturally-spawned *O. mykiss* populations within the Central Valley Basin were closely related, regardless of whether they were sampled above or below a known barrier to anadromy (Garza and Pearse 2008). This is due to some combination of pre-impoundment historic shared ancestry, downstream migration and, possibly, limited anthropogenic upstream migration. However, lower genetic diversity in above-barrier populations indicates a lack of substantial genetic input upstream and highlights lower effective population sizes for above-barrier populations. The consistent clustering of the above-barrier populations with one another, and their position in the California-wide trees, indicate that they are likely to most accurately represent the ancestral population genetic structure of steelhead in the Central Valley (Garza and Pearse 2008).

The above discussions indicating that below-barrier populations of steelhead in the Central Valley, including the lower Yuba River (particularly in consideration of historic plantings and documented straying) likely do not accurately represent the ancestral population genetic structure. In other words, the current steelhead population in the lower Yuba River likely does not represent a “pure” ancestral genome (RMT 2013a).

5.2.6.2.2 Extinction Risk

As stated approximately 11 years ago by Lindley et al. (2006), there are almost no data with which to assess the status of any of the Central Valley steelhead populations, with the exceptions of the hatchery programs on Battle Creek and the Feather, American and Mokelumne rivers. Therefore, they classified Central Valley steelhead populations, including the lower Yuba River, as data deficient.

According to NMFS (2014a), data are lacking to suggest that the Central Valley steelhead DPS is at low risk of extinction, or that there are viable populations of steelhead anywhere in the DPS. Lindley et al. (2007) stated that even if there were adequate data on the distribution and abundance of steelhead in the Central Valley, approaches for assessing steelhead populations and DPS viability might be problematic because the effect of resident *O. mykiss* on the viability of steelhead populations and the DPS is unknown.

Recently, NMFS determined that the viability of the Central Valley steelhead DPS appears to have slightly improved since the 2010/2011 assessment, when it was concluded that the DPS was in danger of extinction (Williams et al. 2016). This modest improvement is driven by the increase in adult returns to hatcheries from their recent lows, but the state of naturally produced fish remains poor. Improvements to the total population sizes of the three previously evaluated steelhead populations (Battle Creek, Coleman National Fish Hatchery, and FRFH), does not warrant a downgrading of the ESU extinction risk. In fact, the lack of improved natural production as estimated by samples taken at Chipps Island, and low abundances coupled with large hatchery influence in the Southern Sierra Nevada Diversity group is cause for concern. As in the previous assessments (Good et al. 2005; Williams et al. 2011), the Central Valley steelhead DPS continues be at a high risk of extinction.

For the lower Yuba River, the data limitations previously discussed preclude extended multi-year abundance and trend analyses. In consideration of the available data, estimated abundance, trends and percentage of hatchery contribution would indicate the lower Yuba River steelhead population to be at a high extinction risk. However, continued implementation of the improved VAKI Riverwatcher™ systems at Daguerre Point Dam is likely to obtain some of the data necessary to allow further abundance estimation and productivity evaluation of steelhead in the lower Yuba River. Moreover, the previous discussion regarding the limited applicability of VSP parameters and extinction risk criteria for spring-run Chinook salmon also pertain to steelhead in the lower Yuba River, in consideration of non-independent populations.

5.2.7 NMFS Recovery Plan Considerations

The NMFS (2014a) Recovery Plan (pg. 77) identifies the existing lower Yuba River steelhead population below Englebright Dam as a Core 2 population. Currently unoccupied areas in the Yuba River Basin upstream of Englebright Dam that are classified by NMFS (2014a) as "primary", or of top priority for reintroduction for steelhead include the North, Middle, and South Yuba rivers.

The discussion regarding recovery plan implementation provided for spring-run Chinook salmon also directly pertains to steelhead in the Yuba River Basin. Therefore, it is not repeated in this section of this Applicant-Prepared Draft BA.

5.3 North American Green Sturgeon Southern DPS

The green sturgeon is the most widely distributed member of the sturgeon family *Acipenseridae* (70 FR 17386). North American green sturgeon are found in rivers from British Columbia south to the Sacramento River, California, and their ocean range is from the Bering Sea to Ensenada, Mexico. In assessing North American green sturgeon status, NMFS determined that two DPSs exist. The northern DPS is made up of known North American green sturgeon spawning (or single stock populations) in the Rogue, Klamath and Eel rivers. In 2005, the southern DPS was believed to contain only a single spawning population in the Sacramento River (70 FR 17386). However, four fertilized green sturgeon eggs collected in 2011 near the Thermalito Afterbay Outlet provide the first documentation of at least some successful spawning in the Feather River (A. Seesholtz, DWR, pers. comm., June 16, 2011).

5.3.1 ESA Listing Status

The Southern DPS of North American green sturgeon (*Acipenser medirostrus*) was listed as a federally threatened species on April 7, 2006 (71 FR 17757) and includes the green sturgeon population spawning in the Sacramento River and utilizing the Sacramento-San Joaquin River Delta, and San Francisco Estuary. NMFS (2009c) *Draft Environmental Assessment for the Proposed Application of Protective Regulations Under Section 4(D) of the Endangered Species Act for the Threatened Southern Distinct Population Segment of North American Green Sturgeon* indicated that the Southern DPS of North American green sturgeon faces several threats to its survival, including the loss of spawning habitat in the upper Sacramento River, and potentially in the Feather and Yuba rivers, due to migration barriers and instream alterations.

Section 4(c)(2) of the ESA requires that NMFS review the status of listed species under its authority at least every 5 years and determine whether any species should be removed from the list or have its listing status changed. In October 2012, NMFS noticed the initiation of the 5-year status review of the Southern DPS of North American green sturgeon (77 FR 64959). The purpose of the 5-year review was to ensure the accuracy of the listing classification for the Southern DPS of North American green sturgeon. A 5-year review is based on the best scientific and commercial data available; therefore, NMFS requested submission of any such information on the Southern DPS that has become available since the listing determination in 2006. To ensure that the 5-year review was complete and based on the best available scientific and commercial information, NMFS solicited new information from the public, governmental agencies, Tribes, the scientific community, industry, environmental entities, and any other interested parties concerning the status of the Southern DPS since the listing determination in 2006 (77 FR 64959). Eleven responses to NMFS' Federal Register notice were received from 11 different agencies or individuals, and included information on population abundance, reviews of recent literature, lists of agency reports summarizing fieldwork, fisheries data, salvage, and academic scientific studies (NMFS 2015a).

In August 2015, NMFS completed the 5-year status review of the Southern DPS of the North American green sturgeon. According to NMFS (2015), evaluation of new information generated since the last status review does not suggest a significant change in the status of Southern DPS green sturgeon. Because many of the threats cited in the original listing still exist, NMFS (2015) concluded that the "threatened" status continues to be applicable.

5.3.2 Critical Habitat Designation

On October 9, 2009, NMFS (74 FR 52300) designated critical habitat for the Southern DPS of North American green sturgeon. This designated critical habitat includes most of the DPS' occupied range, including: 1) coastal marine waters from Monterey Bay to the Washington/Canada border; 2) coastal bays and estuaries in California, Oregon, and Washington; and 3) freshwater rivers in the Central Valley, California. In the Central Valley, designated critical habitat for green sturgeon includes the Sacramento River, lower Feather River, lower Yuba River, the Sacramento-San Joaquin River Delta, and San Francisco Estuary. NMFS (74 FR 52300) defined specific habitat areas in the Sacramento, Feather, and Yuba rivers in California to include riverine habitat from each river mouth upstream to and including the furthest known site of historic and/or current sighting or capture of North American green sturgeon, as long as the site is still accessible. Critical habitat in the lower Yuba River includes the river channel to the ordinary high water line extending from the confluence with the mainstem Feather River upstream to Daguerre Point Dam.

The essential physical and biological habitat features identified for the Southern DPS of North American green sturgeon include food resources (e.g., benthic invertebrates and small fish), substrate types (i.e., appropriate spawning substrates within freshwater rivers), water flow (particularly in freshwater rivers), water quality, water depth, migratory corridors, and sediment quality. The following summary descriptions of the current conditions of the freshwater PBFs for the Southern DPS of North American green sturgeon were taken from the 2009 NMFS OCAP BO (NMFS 2009b) and the 2009 NMFS *Draft Biological and Conference Opinion for the*

Federal Energy Regulatory Commission's (FERC) Relicensing of the California Department of Water Resources Oroville Facilities (FERC Project No. 2100-134) (NMFS 2009c).

5.3.2.1 Physical or Biological Features

PBFs for the Southern DPS of North American green sturgeon critical habitat include specific features of freshwater riverine systems, estuarine habitats, and nearshore coastal marine waters (74 FR 52300, October 9, 2009). PBFs for green sturgeon critical habitat identified as being present in the lower Yuba River include water depth, flow, passage and water quality (74 FR 52328, October 9, 2009).

5.3.2.1.1 Freshwater Riverine Systems

Freshwater riverine systems are used by green sturgeon for spawning and for adult holding after spawning. The green sturgeon eggs hatch in freshwater, and the larvae spend their initial days and weeks in freshwater, migrating to estuarine areas in a relatively short time (NMFS 2016a). Following is a discussion of PBFs for green sturgeon critical habitat in freshwater riverine systems.

Food Resources

Abundant food items for larval, juvenile, sub-adult, and adult lifestages should be present in sufficient amounts to sustain growth (larvae, juveniles, and sub-adults) or support basic metabolism (adults). Although specific data are lacking on food resources for green sturgeon within freshwater riverine systems, nutritional studies on white sturgeon suggest that juvenile green sturgeon most likely feed on benthic macroinvertebrates, which can include plecoptera (stoneflies), ephemeroptera (mayflies), trichoptera (caddis flies), chironomid (dipteran fly larvae), oligochaetes (tubifex worms) or decapods (crayfish). These food resources are important for juvenile foraging, growth, and development during their downstream migration to the Delta and bays. In addition, sub-adult and adult green sturgeon may forage during their downstream post-spawning migration or on non-spawning migrations within freshwater rivers. Sub-adult and adult green sturgeon in freshwater rivers most likely feed on benthic invertebrates similar to those fed on in bays and estuaries, including freshwater shrimp and amphipods. Many of these different invertebrate groups are endemic to and readily available in the Sacramento River from Keswick Dam downstream to the Delta. Heavy hatches of mayflies, caddis flies, and chironomids occur in the upper Sacramento River, indicating that these groups of invertebrates are present in the river system. NMFS anticipates that the aquatic lifestages of these insects (nymphs, larvae) would provide adequate nutritional resources for green sturgeon rearing in the river.

Substrate Type or Size

Suitable freshwater riverine system habitat includes substrates suitable for egg deposition and development (e.g., cobble, gravel, or bedrock sills and shelves with interstices or irregular surfaces to "collect" eggs and provide protection from predators, and free of excessive silt and debris that could smother eggs during incubation), larval development (e.g., substrates with interstices or voids providing refuge from predators and from high flow conditions), and sub-adults and adult lifestages (e.g., substrates for holding and spawning). Stream surveys by USFWS and United States Department of Interior, Bureau of Reclamation (Reclamation) biologists have identified approximately 54 suitable holes and pools between Keswick Dam and the GCID diversion that would support spawning or holding activities for green sturgeon, based on identified physical criteria. Many of these locations are at the confluences of tributaries with the mainstem Sacramento River or at bend pools. Observations of channel type and substrate compositions during these surveys indicate that appropriate substrate is available in the Sacramento River between Keswick Dam and the GCID diversion. Ongoing surveys are anticipated to further identify river reaches in the upper river with suitable substrate characteristics and their utilization by green sturgeon.

Water Flow

An adequate flow regime (i.e., magnitude, frequency, duration, seasonality, and rate-of-change of fresh water discharge over time) is necessary for normal behavior, growth, and survival of all lifestages in the upper Sacramento River. Such a flow regime should include stable and sufficient water flow rates in spawning and rearing reaches to maintain water temperatures within the optimal range for egg, larval, and juvenile survival and development (11-19°C; ~52-66°F (Cech et al. 2000; Mayfield and Cech 2004; Van Eenennaam et al. 2005; Allen et al. 2006). Sufficient flow is also needed to reduce the incidence of fungal infestations of the eggs, and to flush silt and debris from cobble, gravel, and other substrate surfaces to prevent crevices from being filled in and to maintain surfaces for feeding. Successful migration of adult green sturgeon to and from spawning grounds is also dependent on sufficient water flow. Spawning success is more associated with water flow and water temperature than compared with other variables. Spawning in the Sacramento River is believed to be triggered by increases in water flow to about 14,000 cfs (Brown 2007). Post-spawning downstream migrations are triggered by increased flows, ranging from 6,150-14,725 cfs in the late summer (Vogel 2005) and greater than 3,550 cfs in the winter (Erickson et al. 2002; Benson et al. 2007). The current suitability of these flow requirements is almost entirely dependent on releases from Shasta Dam. High winter flows associated with the natural hydrograph do not occur within the section of the river utilized by green sturgeon with the frequency and duration that occurred during pre-dam conditions.

Water Quality

Adequate water quality, including temperature, salinity, oxygen content, and other chemical characteristics necessary for normal behavior, growth, and viability of all green sturgeon lifestages, is required for the proper functioning of the freshwater habitat. Suitable water temperatures include: 1) stable water temperatures within spawning reaches (wide fluctuations could increase egg mortality or deformities in developing embryos); 2) water temperatures within 51.8-62.6°F (optimal range = 57.2-60.8°F) in spawning reaches for egg incubation (March-August) (Van Eenennaam et al. 2005); 3) water temperatures below 68°F for larval development (Werner et al. 2007 as cited in NMFS 2009b); and 4) water temperatures below 75.2°F for juveniles (Mayfield and Cech 2004; Allen et al. 2006). Due to the temperature management of the releases from Keswick Dam for winter-run Chinook salmon in the upper Sacramento River, water temperatures in the river reaches utilized currently by green sturgeon appear to be suitable for proper egg development and larval and juvenile rearing. Suitable salinity levels range from fresh water [<3 parts per thousand (ppt)] for larvae and early juveniles [to about 100 days post hatch (dph)] to brackish water (10 ppt) for juveniles prior to their transition to salt water. Prolonged exposure to higher salinities may result in decreased growth and activity levels and even mortality (Allen and Cech 2007). Salinity levels are suitable for green sturgeon in the Sacramento River and freshwater portions of the Delta for early lifestages. Adequate levels of DO are needed to support oxygen consumption by early lifestages (Allen and Cech 2007). Current DO levels in the mainstem Sacramento River are suitable to support the growth and migration of green sturgeon. Suitable water quality also includes water free of contaminants (i.e., pesticides, organochlorines, elevated levels of heavy metals, etc.) that may disrupt normal development of embryonic, larval, and juvenile lifestages of green sturgeon. Legacy contaminants such as mercury still persist in the watershed and pulses of pesticides have been identified in winter storm discharges throughout the Sacramento River Basin.

Water Depth

Pools equal to or greater than 5 m deep are critical for adult green sturgeon spawning and for summer holding within the Sacramento River. Summer aggregations of green sturgeon are observed in these pools in the upper Sacramento River upstream of the GCID diversion. The significance and purpose of these aggregations are unknown at the present time, although it is likely that they are the result of an intrinsic behavioral characteristic of green sturgeon. Adult green sturgeon in the Klamath and Rogue rivers also occupy deep holding pools for extended periods of time, presumably for feeding, energy conservation, and/or refuge from high water temperatures (Erickson et al. 2002; Benson et al. 2007). As described above, approximately 54 pools with adequate depth have been identified in the Sacramento River upstream of the GCID diversion.

Migration Corridor

Unobstructed migratory pathways are necessary for passage within riverine habitats and between riverine and estuarine habitats (e.g., an unobstructed river or dammed river that still allows for passage). Unobstructed migratory pathways are necessary for adult green sturgeon to migrate to and from spawning habitats, and for larval and juvenile green sturgeon to migrate downstream from spawning/rearing habitats within freshwater rivers to rearing habitats within the estuaries. Unobstructed passage throughout the Sacramento River up to Keswick Dam (RM 302) is important, because optimal spawning habitats for green sturgeon are believed to be located upstream of the RBDD (RM 242).

Green sturgeon adults that migrate upstream during April, May, and June are completely blocked by the ACID diversion dam. Therefore, 5 mi of spawning habitat are inaccessible upstream of the diversion dam. It is unknown if spawning is occurring in this area. Adults that pass upstream of ACID dam before April are forced to wait 6 months until the stop logs are pulled before returning downstream to the ocean. Upstream blockage at the ACID diversion dam forces sturgeon to spawn in approximately 12 percent less habitat between Keswick Dam and RBDD. Newly emerged green sturgeon larvae that hatch upstream of the ACID diversion dam are forced to hold for 6 months upstream of the dam or pass over it and be subjected to higher velocities and turbulent flow below the dam, thus rendering the larvae and juvenile green sturgeon more susceptible to predation.

Closure of the gates at RBDD from May 15 through September 15 previously precluded all access to spawning grounds above the dam during that time period. However, as previously discussed, the RBDD gates were permanently raised in September 2011.

Juvenile green sturgeon first appear in USFWS sampling efforts at RBDD during May, June, and July. Juvenile green sturgeon are likely subjected to the same predation and turbulence stressors caused by RBDD as the juvenile anadromous salmonids, leading to diminished survival through the structure and waters immediately downstream.

Sediment Quality

Sediment should be of the appropriate quality and characteristics necessary for normal behavior, growth, and viability of all lifestages. This includes sediments free of contaminants (e.g., elevated levels of heavy metals such as mercury, copper, zinc, cadmium, and chromium), polycyclic aromatic hydrocarbons, and organochlorine pesticides) that can result in negative effects on any lifestages of green sturgeon. Based on studies of white sturgeon, bioaccumulation of contaminants from feeding on benthic species may negatively affect the growth, reproductive development, and reproductive success of green sturgeon. The Sacramento River and its tributaries have a long history of contaminant exposure from abandoned mines, separation of gold ore from mine tailings using mercury, and agricultural practices with pesticides and fertilizers which result in deposition of these materials in the sediment horizons in the river channel. Disturbance of these sediment horizons by natural or anthropogenic actions can liberate the sequestered contaminants into the river. This is a continuing concern throughout the watershed.

5.3.2.1.2 Estuarine Habitat Areas

Food Resources

Abundant food items within estuarine habitats and substrates for adult, sub-adult and juvenile lifestages are required for the proper functioning of this PBF for green sturgeon. Prey species for green sturgeon within bays and estuaries primarily consist of benthic invertebrates and fish, including crangonid shrimp, callianassid shrimp, burrowing thalassinidean shrimp, amphipods, isopods, clams, annelid worms, crabs, sand lances, and anchovies. These prey species are critical for the rearing, foraging, growth, and development of juvenile, sub-adult, and adult green sturgeon within the bays and estuaries. Currently, the estuary provides these food resources, although annual fluctuations in the population levels of these food resources may diminish the contribution of one group to the diet of green sturgeon relative to another food source. The recent spread of the Asian overbite clam (*Corbula amurensis*) has shifted the diet profile of white sturgeon (*Acipenser transmontanus*) to this invasive species. The overbite clam now makes up a substantial proportion of the white sturgeon's diet in the estuary. NMFS assumes that green sturgeon have also altered their diet to include this new food source, because of its increased prevalence in the benthic invertebrate community.

Water Flow

Within bays and estuaries adjacent to the Sacramento River (i.e., the Sacramento-San Joaquin Delta and the Suisun, San Pablo, and San Francisco bays), sufficient inflow into the bay and estuary to allow adults to successfully orient to the incoming flow and migrate upstream to spawning grounds is required. Sufficient flows are needed to attract adult green sturgeon to the Sacramento River from the bay and to initiate the upstream spawning migration into the upper river. Currently, flows provide the necessary attraction to green sturgeon to enter the Sacramento River. Nevertheless, these flows are substantially less than those that historically occurred and stimulated the spawning migration.

Water Quality

Adequate water quality, including temperature, salinity, oxygen content, and other chemical characteristics, is necessary for normal behavior, growth, and viability of all lifestages. Suitable water temperatures for juvenile green sturgeon should be below 75°F. At temperatures above 75.2°F, juvenile green sturgeon exhibit decreased swimming performance (Mayfield and Cech 2004) and increased cellular stress (Allen et al. 2006). Suitable salinities in the estuary range from brackish water (10 ppt) to salt water (33 ppt). Juveniles transitioning from brackish to salt water can tolerate prolonged exposure to salt water salinities, but may exhibit decreased growth and activity levels (Allen and Cech 2007), whereas sub-adults and adults tolerate a wide range of salinities (Kelly et al. 2007 as cited in Reclamation 2008a). Sub-adult and adult green sturgeon occupy a wide range of DO levels, but may need a minimum DO level of at least 6.54 milligrams per liter (mg/L) (Kelly et al. 2007 as cited in Reclamation 2008a; Moser and Lindley 2007 as cited in Reclamation 2008a). Suitable water quality also includes water free of contaminants, as described above. In general, water quality in the Delta and estuary meets these criteria, but local areas of the Delta and downstream bays have been identified as having deficiencies. Water quality in the areas such as the Stockton turning basin and Port of Stockton routinely have depletions of DO and episodes of first flush contaminants from the surrounding industrial and urban watershed. Discharges of agricultural drain water have also been implicated in local elevations of pesticides and other related agricultural compounds within the Delta and the tributaries and sloughs feeding into the Delta. Discharges from petroleum refineries in Suisun and San Pablo Bay have been identified as sources of selenium to the local aquatic ecosystem (Linville et al. 2002)

Water Depth

A diversity of depths is necessary for shelter, foraging, and migration of juvenile, sub-adult, and adult lifestages. Sub-adult and adult green sturgeon occupy deep ($\geq$ 5 m) holding pools within bays and estuaries as well as within freshwater rivers. These deep holding pools may be important for feeding and energy conservation, and may serve as thermal refugia for sub-adult and adult green sturgeon (Benson et al. 2007). Tagged adults and sub-adults within the San Francisco Bay estuary primarily occupied waters with depths of less than 10 m, either swimming near the surface or foraging along the bottom (Kelly et al. 2007 as cited in Reclamation 2008a). In a study of juvenile green sturgeon in the Delta, relatively large numbers of juveniles were captured primarily in shallow waters from 3 to 8 ft deep, indicating juveniles may require shallower depths for rearing and foraging (Radtke 1966). Thus, a diversity of depths is important to support different lifestages and habitat uses for green sturgeon within estuarine areas.

Currently, there is a diversity of water depths found throughout the San Francisco Bay estuary and Delta waterways. Most of the deeper waters, however, are comprised of artificially maintained shipping channels, which do not migrate or fluctuate in response to the hydrology in the estuary in a natural manner. The channels are simplified trapezoidal shapes with little topographical variation along the channel alignment. Shallow waters occur throughout the Delta and San Francisco Bay. Extensive "flats" occur in the lower reaches of the Sacramento and San Joaquin River systems as they leave the Delta region and are even more extensive in Suisun and San Pablo bays. In most of the region, variations in water depth in these shallow water areas occur due to natural processes, with only localized navigation channels being dredged (e.g., the Napa River and Petaluma River channels in San Pablo Bay).

Migration Corridor

Within the waterways comprising the Delta and bays downstream of the Sacramento River, unobstructed passage is needed for juvenile green sturgeon during the rearing phase of their life cycle. Rearing fish need the ability to freely migrate from the river through the estuarine waterways of the Delta and bays and eventually out into the ocean. Passage within the bays and the Delta is also critical for adults and sub-adults for feeding and summer holding, as well as to access the Sacramento River for their upstream spawning migrations and to make their outmigration back into the ocean. Within bays and estuaries outside of the Delta and the areas comprised by Suisun, San Pablo, and San Francisco bays, unobstructed passage is necessary for adult and sub-adult green sturgeon to access feeding areas, holding areas, and thermal refugia, and to ensure passage back out into the ocean. Currently, unobstructed passage has been diminished by human actions in the Delta and bays. The CVP and SWP water projects alter flow patterns in the Delta due to export pumping and create entrainment issues in the Delta at the pumping and fish facilities.

Power generation facilities in Suisun Bay create risks of entrainment and thermal barriers through their cooling water diversions and discharges. Installation of seasonal barriers in the South Delta and operations of the radial gates in the DCC facilities alter migration corridors available to green sturgeon. Actions such as the hydraulic dredging of ship channels and operations of large ocean going vessels create additional sources of risk to green sturgeon within the estuary. Hydraulic dredging can result in the entrainment of fish into the dredger's hydraulic cutterhead intake. Commercial shipping traffic can result in the loss of fish, particularly adult fish, through ship and propeller strikes.

Sediment Quality

Sediment quality (i.e., chemical characteristics) is necessary for normal behavior, growth, and viability of all lifestages. This includes sediments free of contaminants (e.g., elevated levels of selenium, polycyclic aromatic hydrocarbons [PAHs], and organochlorine pesticides) that can cause negative effects on all lifestages of green sturgeon (see description of sediment quality for riverine habitats above).

5.3.3 Historical Distribution and Abundance

Green sturgeon are widely distributed along the Pacific Coast, have been documented offshore from Ensenada, Mexico, to the Bering Sea, and are found in rivers from British Columbia to the Sacramento River (Moyle 2002). As is the case for most sturgeon, the Southern DPS of North American green sturgeon are anadromous; however, they are the most marine-oriented of the sturgeon species (Moyle 2002).

The historical distribution of green sturgeon in the Sacramento-San Joaquin river basins is poorly documented, but Adams et al. (2007) summarizes information that suggests that green sturgeon may have been distributed above the locations of present-day dams on the Sacramento and Feather rivers (Mora et al. 2009). Historical records from the 1930s indicate that green sturgeon were not listed as either “known to occur” or “presumed to occur” in the Yuba or American rivers (Sumner and Smith 1939; Evermann and Clark 1931).

According to NMFS (2009b), spawning populations of green sturgeon in North America are currently found in only three river systems: the Sacramento and Klamath rivers in California and the Rogue River in southern Oregon. Data from commercial trawl fisheries and tagging studies indicate that the green sturgeon occupy ocean waters down to the 110 m contour (Erickson and Hightower 2007). During the late summer and early fall, sub-adults and non-spawning adult green sturgeon frequently can be found aggregating in estuaries along the Pacific Coast (Emmett et al. 1991; Moser and Lindley 2007 as cited in Reclamation 2008a). Particularly large concentrations of green sturgeon from both the northern and southern populations occur in the Columbia River estuary, Willapa Bay, Grays Harbor and Winchester Bay, with smaller aggregations in Humboldt Bay, Tillamook Bay, Nehalem Bay, and San Francisco and San Pablo bays (Emmett et al 1991; Moyle et al. 1992 as cited in Reclamation 2008a; Beamesderfer et al. 2007). Lindley et al. (2008) reported that green sturgeon make seasonal migratory movements along the west coast of North America, overwintering north of Vancouver Island and south of Cape Spencer, Alaska. Individual fish from the Southern DPS of green sturgeon have been detected in these seasonal aggregations. Information regarding the migration and habitat use of green sturgeon has recently emerged. Lindley (2006 as cited in NMFS 2009b) presented preliminary results of large-scale green sturgeon migration studies, and verified past population structure delineations based on genetic work and found frequent large-scale migrations of green sturgeon along the Pacific Coast. This work was further expanded by tagging studies of green sturgeon conducted by Erickson and Hightower (2007) and Lindley et al. (2008). The data indicate that green sturgeon are migrating considerable distances up the Pacific Coast into other estuaries, particularly the Columbia River estuary. This information also agrees with the results of previous green sturgeon tagging studies (CDFG 2002), where Cal Fish and Wildlife tagged a total of 233 green sturgeon in the San Pablo Bay estuary between 1954 and 2001. A total of 17 tagged fish were recovered: 3 in the Sacramento-San Joaquin Estuary, 2 in the Pacific Ocean off California, and 12 from commercial fisheries off of the Oregon and Washington coasts. Eight of the 12 commercial fisheries recoveries were in the Columbia River estuary (CDFG 2002).

In the lower Feather River, green sturgeon have intermittently been observed (Beamesderfer et al. 2007). NMFS (2008b) states that the presence of adult, and possibly sub-adult, green sturgeon within the lower Feather River has been confirmed by photographs, anglers' descriptions of fish catches (P. Foley, pers. comm. cited in CDFG 2002), incidental sightings (DWR 2005a), and occasional catches of green sturgeon reported by fishing guides (Beamesderfer et al. 2004).

In the mid-1970s, green sturgeon were caught each year on the Feather River, with the majority of catches occurring from March to May and a few additional catches occurring in July and August (USFWS 1995b). In 1993, seven adult green sturgeon were captured at the Thermalito Afterbay Outlet, ranging in size from 60.9 to more than 73.2 in (USFWS 1995b). In a broad scale survey from 1999 to 2001, green sturgeon were infrequently observed within the area downstream of the Thermalito Afterbay Outlet and none observed upstream (DWR 2003a). In 2006, four green sturgeon were positively identified by a DWR biologist near the Thermalito Afterbay Outlet. Eight additional sturgeon were also observed in the same area but could not be positively identified as green sturgeon (DWR 2007a as cited in Reclamation 2008a).

More recently, studies in the Feather River have documented spawning by Southern DPS green sturgeon (Seesholtz et al. 2014). Seesholtz and Manuel (2012) performed DIDSON surveys in the river and estimated 21-28 sturgeon in-river during 2011 and at least 3 to 4 sturgeon in-river during the 2012 spawning season. Visual information confirmed that these counts include green sturgeon (NMFS 2015a). The reason that fewer sturgeon were observed in 2012 is possibly due to a lack of high flow events upstream in the Feather River in that year (pers. comm. with A. Seesholtz, DWR, 2013, as cited in NMFS 2015a). The breach of Shanghai Bench on the Feather River in early 2012 likely eliminated this naturally formed passage barrier (flow dependent) in the lower Feather River (pers. comm. with A. Seesholtz, DWR, 2013, as cited in NMFS 2015a). Tagged green sturgeon were recorded as making upstream and downstream forays from the breached area (DWR 2013, as cited in NMFS 2015a).

Although adult green sturgeon occurrence in the Feather River has been previously documented, larval and juvenile green sturgeon have not been collected despite attempts to collect larval and juvenile sturgeon during early spring through summer using rotary screw traps, artificial substrates, and larval nets deployed at multiple locations (Seesholtz et al. 2003). Moreover, unspecific past reports of green sturgeon spawning (Wang 1986; USFWS 1995a; CDFG 2002) have not been corroborated by observations of young fish or significant numbers of adults in focused sampling efforts (Niggemeyer and Duster 2003; Seesholz et al. 2003; Beamesderfer et al. 2004). Based on these results, in 2006, NMFS concluded that an effective population of spawning green sturgeon did not exist in the lower Feather River (71 FR 17757). However, four fertilized green sturgeon eggs were collected near the Thermalito Afterbay Outlet on June 14, 2011, thus providing the first documentation of at least some successful spawning in the Feather River (A. Seesholtz, DWR, pers. comm., June 16, 2011).

Historical accounts of sturgeon in the Yuba River have been reported by anglers, but these accounts do not specify whether the fish were white or green sturgeon (Beamesderfer et al. 2004). Since the 1970s, numerous surveys of the lower Yuba River downstream of Englebright Dam have been conducted, including annual salmon carcass surveys, snorkel surveys, beach seining, electrofishing, rotary screw trapping, redd surveys, and other monitoring and evaluation activities. Over the many years of these surveys and monitoring of the lower Yuba River, only one confirmed observation of an adult green sturgeon has occurred prior to 2011. The NMFS September 2008 *Draft Biological Report, Proposed Designation of Critical Habitat for the Southern Distinct Population Segment of North American Green Sturgeon* (NMFS 2008b) states that of the three adult or sub-adult sturgeon observed in the Yuba River below Daguerre Point Dam during 2006, only one was confirmed to be a green sturgeon, and that "*Spawning is possible in the river, but has not been confirmed and is less likely to occur in the Yuba River than in the Feather River. No green sturgeon juveniles, larvae, or eggs have been observed in the lower Yuba River to date.*"

As part of ongoing sturgeon monitoring efforts in the Feather River Basin under the AFRP, Cramer Fish Sciences conducted roving underwater video surveys in the lower Feather and lower Yuba rivers using a drop-down camera suspended from a motorized boat. On May 24, 25 and 26, 2011, underwater videographic monitoring was conducted in the lower Yuba River downstream of Daguerre Point Dam. In a memorandum dated June 7, 2011 Cramer Fish Sciences (2011) stated that they observed what they believed were 4-5 green sturgeon near the center of the channel at the edge of the bubble curtain below Daguerre Point Dam. The sturgeon were observed either on a gravel bar approximately 1.5 m deep, or in a pool approximately 4 m deep immediately adjacent to the gravel bar. Photographs taken by Cramer Fish Sciences (2011) were forwarded to green sturgeon experts. Olaf P. Langness, Sturgeon and Smelt Projects, Washington Department of Fish and Wildlife Region 5, expressed the opinion that the photographs were of green (rather than white) sturgeon. Also, David Woodbury, NMFS Sturgeon Recovery Coordinator, expressed his opinion that the fish in the photographs were green sturgeon.

During 2012 and 2013, underwater videography also was used in an attempt to document the presence of green sturgeon downstream of Daguerre Point Dam, but no observations of green sturgeon were made. During 2016, CDFW personnel conducted some observational snorkeling in the plunge pool located immediately below Daguerre Point Dam. During the course of the snorkeling, observations of a few green sturgeon were made in the plunge pool and video documented.

YCWA (2013) examined the potential occurrence of green sturgeon in the lowermost 24 mi of the Yuba River based on detections of acoustically-tagged green sturgeon in the Yuba River. The examination included coordination with agencies and organizations involved with green sturgeon research in the Central Valley, and collection of available information and data regarding the presence and use of the Yuba River by green sturgeon. YCWA collaborated with DWR's Feather River Program, CFTC, and Cal Fish and Wildlife's Heritage and Wild Trout and Steelhead Management and Recovery Programs to examine whether any acoustically-tagged green sturgeon were found in the lower Yuba River. The CFTC is tracking 217 green sturgeon acoustically tagged in the Central Valley, and DWR's Feather River Program has acoustically tagged 2 green sturgeon in the lower Feather River.

None of the 217 green sturgeon acoustically-tagged in the Central Valley were detected in the Yuba River, with the exception of one fish tagged by DWR in the Feather River. This individual fish was detected once on September 6, 2011 in the Yuba River by the Cal Fish and Wildlife's lowermost acoustic receiver located at the confluence of the Yuba and Feather rivers. That fish also was detected upstream in the Feather River earlier on the same day and downstream in the Sacramento River on the evening of September 6, 2011. Therefore, the fish apparently only entered the mouth of the lower Yuba River for a very brief period of time before continuing its downstream migration in the Feather and Sacramento rivers.

5.3.4 General Life History and Habitat Requirements

Limited information regarding green sturgeon distribution, movement and behavioral patterns, as well as lifestage-specific habitat utilization preferences, is available for the Sacramento and Feather rivers.

A general timeline of green sturgeon development is provided in Table 5.3-1 (NMFS 2016a). Developmental stage is given by size, which is a common practice in fisheries biology to infer lifestage through the measured length of the fish. As indicated in Table 5.3-1, there is considerable variability across categories, such as size or age at maturity (NMFS 2016a).

Table 5.3-1. General Timeline of Southern DPS of North American Green Sturgeon Life History, From Egg to Adult, With Length-Lifestage Information Provided (NMFS 2016a).

Timeline	Lifestage, Length-Age Relationship
Fertilization of eggs (spawning)	Spawning occurs primarily in deepwater (>5m) pools[1] at very few select sites[2], predominantly in the Sacramento River, predominantly mid-April to mid-June[3].
144–192 hours (6-8 days) after fertilization of eggs	Newly hatched larvae emerge. **Larvae are 12.6–14.5 mm** long[4].
6 days post hatch	**Nocuturnal swim up, hide-by-day behavior observed4.**
10 days post hatch (dph)	**Exogenous feeding begins around 10 dph4. Larvae begin to disperse downstream.**
2 weeks old (approx)	Larvae appear in USFWS rotary screw traps at RBDD at lengths of 24–31 mm.
45 days post hatch	Larval to juvenile metamorphosis complete. Begin juvenile lifestage. **Juveniles are 63–94 mm** long.
45 days to 1.5 years	Juveniles migrate downstream and into the Delta or the estuary and rear to the subadult phase. **Juveniles range in size from around 70 mm to 90 cm**. Little information available about this lifestage.
1.5 to 4 years	Sometime between the age of 1.5 to 4 years, juvenile green sturgeon migrate to sea for the first time, thereby entering the subadult phase. **Subadults are 107 cm to 174[5] cm**.
1.5 years to 15-17 years	After green sturgeon enter the ocean for the first time, they grow and develop, reaching maturity between 15–17 years old.*
15 to 17 years*	Green sturgeon reach sexual maturity and become adults, with **males maturing around 120 cm and females maturing around 145 cm[6]** (based on Nakamoto's Klamath River studies).
15 to 50+ years	Green sturgeon have a lifespan that can reach 50 or more years and can grow to a total length of over 2 meters.

References
1. Thomas et al. (2013); 2. Mora (unpub, UC Davis, as cited in NMFS 2016a); 3. Poytress et al. (2013); 4. Deng et al. (2002); 5. Heppell (2007); 6. Nakamoto et al. (1995) found that green sturgeon in the Klamath River might reach sexual maturity as early as 13 years for females and 9 years for males. More research is needed to determine the typical age and size of green sturgeon at maturity (NMFS 2016a).

5.3.4.1 Adult Immigration, Holding and Emigration

Green sturgeon in the Sacramento River have been documented and studied more widely than they have in either the Feather or the Yuba rivers. Green sturgeon adults in the Sacramento River are reported to begin their upstream spawning migrations into freshwater during late February, before spawning between March and July, with peak spawning believed to occur between April and June (Adams et al. 2002). NMFS (2009b) reports that, based on data gathered from acoustically tagged adult green sturgeon, these fish migrate upstream during May as far as the mouth of Cow Creek, near Bend Bridge on the Sacramento River.

For the Sacramento River, NMFS (2009b) reports that adult green sturgeon prefer deep holes (≥ 5 m depth) at the mouths of tributary streams, where they spawn and rest on the bottom. After spawning, the adults hold over in the upper Sacramento River between RBDD and the GCID diversion until November (Klimley 2007). Heublein et al. (2006, 2008) reported the presence of adults in the Sacramento River during the spring through the fall into the early winter months, holding in upstream locations before their emigration from the system later in the year. Green sturgeon downstream migration appears to be triggered by increased flows and decreasing water temperatures, and occur rapidly once initiated (NMFS 2009b). Some adult green sturgeon rapidly leave the system following their suspected spawning activity and re-enter the ocean in early summer (Heublein 2006). NMFS (2009b) states that green sturgeon larvae and juveniles are routinely observed in rotary screw traps at RBDD and the GCID diversion, indicating that spawning occurs upstream of both these sites.

Before the studies conducted by the University of California, Davis (UC Davis), there were few empirical observations of green sturgeon movement in the Sacramento River (Heublein et al. 2008). The study by Heublein et al. (2008) is reportedly the first to describe the characteristics of the adult green sturgeon migration in the Sacramento River, and to identify putative regions of spawning habitat, based on the recorded movements of free-swimming adults.

The Sacramento River adjacent to the GCID diversion routinely contains a large aggregation of green sturgeon during summer and fall months, although the GCID aggregation site is atypical of over-summering habitats in other systems, being an area of high water velocity (Heublein et al. 2008). The GCID site is over five meters deep, with structural current refuges and eddy formations. It is possible that green sturgeon occupy lower-velocity subsections of the site, although observations of green sturgeon capture, and manual tracking estimates, indicate that green sturgeon are found in, or in very close proximity to, high velocity areas (Heublein et al. 2008).

5.3.4.2 Adult Spawning

Adult green sturgeon are believed to spawn every 2 to 5 years (Beamesderfer et al. 2007). Upon maturation of their gonadal tissue, but prior to ovulation or spermiation, the adult fish enter freshwater and migrate upriver to their spawning grounds (NMFS 2009b). Heublein et al. (2008) observed that green sturgeon enter San Francisco Bay in March and April and migrate rapidly up the Sacramento River to the region between GCID and Cow Creek. The fish lingered at these regions at the apex of their migration for 14 to 51 days, presumably engaged in spawning behavior, before moving back downriver (Heublein et al. 2008).

To investigate adult immigration, spawning or juvenile nursery habits of green sturgeon in the upper Sacramento River, Brown (2007) developed a study to identify green sturgeon spawning locations and dates in the upper Sacramento River. Using a depth finder, study sites were selected at locations upstream of deeper holes in higher velocity water in the Sacramento River (Brown 2007). The study was originally designed in 1997 using the prevalent methodology at the time (e.g., artificial substrate mats) for the capture of eggs and larvae of white sturgeon. Brown (2007) reports that later findings from artificial spawning and larval rearing of green sturgeon (Van Eenennaam et al. 2001) indicate that green sturgeon eggs may be less adhesive than eggs from other acipenserids, possibly reducing the effectiveness of artificial substrate sampling. Brown (2007) suggested that spawning in the Sacramento River may occur from April to June, and that the potential spawning period may extend from late April through July, as indicated by the rotary screw trap data at the RBDD from 1994 to 2000.

Heublein et al. (2008) stated that, in contrast to the behavior of green sturgeon observed during 2004–2005, the majority of out-migrants detected in 2006 displayed an entirely different movement strategy. Nine of the ten tagged fish detected that year exited the system with no extended hold-over period and with no apparent relation to flow increases, eight leaving before July 4 and the last on August 22. Heublein et al. (2008) suggested that the rapid out-migration of green sturgeon in 2006, and the reduced aggregation period at the GCID site could be a result of consistently higher flows and lower temperatures than in previous study years. Alternatively, this could be an unusual behavior, related to unknown cues, that has not been documented in green sturgeon before this study (Heublein et al. 2008).

The apex detections of individual fish indicate reaches and dates when spawning might have occurred during the study conducted by Heublein et al. (2008). They reported that spawning may have occurred between May and July, and that high water velocities and extensive bedrock habitat were found in all of the apex detection reaches. Furthermore, water temperatures did not exceed 62.6°F in these reaches during this study, which would have permitted normal green sturgeon larval development (Van Eenennaam et al. 2005 as cited in Heublein et al. 2009).

The Sacramento and Feather rivers currently host the only known spawning populations of the Southern DPS of North American green sturgeon (Poytress et al. 2010; Seezholtz et al. 2014). During 2009, four spawning sites of green sturgeon were confirmed in the upper Sacramento River (Poytress et al. 2010). Three confirmed sites from 2008 surveys were reconfirmed and one of three newly sampled sites in 2009 was confirmed by the presence of green sturgeon eggs on artificial substrate mats.

During 2010, five spawning sites of green sturgeon were confirmed within a 60 river kilometer reach of the upper Sacramento River, California (Poytress et al. 2011). As stated by Poytress et al. (2010), spawning events occurred several river kilometers upstream and downstream of the RBDD before and after the June 15 seasonal dam gate closure. Spawning occurred directly below RBDD within 2 weeks after the gate closure. The temporal distribution pattern suggested by 2009 sampling results indicates spawning of Sacramento River green sturgeon occurs from early April through late June (Poytress et al. 2010). Sampling conducted during 2010 suggested that spawning of Sacramento River green sturgeon occurs from early April through mid-June (Poytress et al. 2011). During 2010 sampling, depths for eggs collected from all of the sites combined ranged from 2.4 to 10.9 m (7.9 to 35.8 ft) with an average of 6.9 m (22.6 ft). Sacramento River flows and water temperatures at sites located above RBDD during the estimated spawning period ranged from 166 to 459 m^3s^{-1} (5,862 cfs to 16,209 cfs), with an average of 293 m^3s^{-1} (10,347 cfs), and 52.0°F to 57.9°F during the estimated spawning period. Sacramento River flows and temperatures at sites located below RBDD during the estimated spawning period ranged from 268 to 509 m^3s^{-1} (9,464 cfs to 17,975 cfs), with an average of 349 m^3s^{-1} (12,324 cfs), and 52.9°F to 60.1°F during the estimated spawning period (Poytress et al. 2011).

Seesholtz et al. (2014) described egg mat studies that collected 13 fertilized green sturgeon eggs in June of 2011, indicating that Southern DPS green sturgeon are using the Feather River for spawning. Developmental stages of the eggs ranged from early gastrulation (Stage 15) to post-neurulation (Stage 27), which led Seesholtz et al. (2014) to estimate that four independent spawning events occurred between June 12 and June 19, 2011. Egg mats were set in water depths that ranged from 1.1 to 11.0 m (3.6 to 36.1 ft) in depth at the Thermalito Afterbay Outlet. Flows at the Thermalito Afterbay Outlet ranged from 99 to 340 m^3s^{-1} (3,496 to 12,007 cfs). Eggs were collected from the mats at the Thermalito Afterbay Outlet when flows ranged from 172 to 312 m^3s^{-1} (6,074 to 11,018 cfs). Eggs were collected from the mats at depths between 5.2 ft and 18 ft when water temperatures were 60.8 to 62.6°F (Seesholtz et al. 2014).

The habitat requirements of green sturgeon are not well known. Eggs are likely broadcast and externally fertilized in relatively fast water and probably in depths greater than three meters (Moyle 2002). Preferred spawning substrate is likely large cobble where eggs settle into cracks, but spawning substrate can range from clean sand to bedrock (Moyle 2002). Spawning is believed to occur over substrates ranging from clean sand to bedrock, with preferences for cobble (Emmett et al. 1991; Moyle et al. 1995). Eggs likely adhere to substrates, or settle into crevices between substrates (Van Eenennaam et al. 2001; Deng et al. 2002). Both embryos and larvae exhibited a strong affinity for benthic structure during laboratory studies (Van Eenennaam et al. 2001; Deng et al. 2002; Kynard et al. 2005), and may seek refuge within crevices, but use flat-surfaced substrates for foraging (Nguyen and Crocker 2007 as cited in NMFS 2009b).

5.3.4.3 Embryo Incubation

Green sturgeon larvae hatch from fertilized eggs after approximately 169 hours of incubation at a water temperature of 59°F (Van Eenennaam et al. 2001; Deng et al. 2002), which is similar to the sympatric white sturgeon development rate (176 hours). Van Eenennaam et al. (2005) indicated that an optimum range of water temperatures for egg development was between 57.2°F and 62.6°F. Water temperatures over 73.4°F resulted in 100 percent mortality of fertilized eggs before hatching. Water temperatures above 68°F are reportedly lethal to green sturgeon embryos (Cech et al. 2000; Beamesderfer and Webb 2002).

5.3.4.4 Larval and Juvenile Rearing

Information about larval green sturgeon in the wild is very limited (NMFS 2016a). Newly hatched green sturgeon are approximately 12.5 to 14.5 mm long. After approximately 10 days, larvae begin feeding and growing rapidly. Under laboratory conditions, green sturgeon larvae cling to the bottom during the day, and move into the water column at night (Van Eenennaam et al. 2001). Exogenous feeding starts at approximately 14 days (23 to 25 mm) (Van Eenennaam et al. 2001).

Green sturgeon larvae do not exhibit the initial pelagic swim–up behavior characteristic of other *Acipenseridae*. They are strongly oriented to the bottom and exhibit nocturnal activity patterns (NMFS 2009b). After 6 days, the larvae exhibit nocturnal swim-up activity (Deng et al. 2002) and nocturnal downstream migrational movements (Kynard et al. 2005). Juvenile fish continue to exhibit nocturnal behavior beyond the metamorphosis from larvae to juvenile stages (NMFS 2009b). Kynard et al. (2005) laboratory studies indicated that juvenile fish continued to migrate downstream at night for the first six months of life. Observations made during nocturnal sampling in the Sacramento River indicate a possible preference of larvae for mid-channel environments or swift water velocity areas (Poytress et al. 2010). When ambient water temperatures reached 8^{o}C (46.4^{o}F), downstream migrational behavior diminished and holding behavior increased (Kynard et al. 2005). These data suggest that 9 to 10 month old fish would hold over in their natal rivers during the ensuing winter following hatching, but at a location downstream of their spawning grounds (NMFS 2009b).

Post-migrant larvae are benthic, foraging up- and downstream diurnally with a nocturnal activity peak (NMFS 2009b). Foraging larvae select open habitat, not structure habitat, but continue to use cover during the day (NMFS 2009b).

As reported in USACE (2007), metamorphosis to the juvenile stage is complete at 45 days, and juveniles continue to grow rapidly, reaching 300 mm in one year. Juveniles spend from 1 to 4 years in fresh and estuarine waters and disperse into salt water at lengths of 300 to 750 mm (USACE 2007).

The primary diet for juvenile green sturgeon reportedly consists of small crustaceans, such as amphipods and opossum shrimp (CDFG 2001). As juvenile green sturgeon develop, they reportedly eat a wider variety of benthic invertebrates, including clams, crabs, and shrimp (CDFG 2001).

Green sturgeon juveniles tested under laboratory conditions had optimal bioenergetic performance (i.e., growth, food conversion, swimming ability) between 59^{o}F and 66.2^{o}F under either full or reduced rations (Mayfield and Cech 2004).

Larvae and juvenile green sturgeon appear to be nocturnal (Cech et al. 2000), which may protect them from downstream displacement (LCFRB 2004). Green sturgeon larvae and juveniles (up to day 84) forage day and night, but activity is reported to peak at night. At day 110 to 118, juvenile green sturgeon move downstream at night and habitat preference suggests that juveniles prefer deep pools with low light and some rock structure (Kynard et. al. 2005).

Wintering juveniles forage actively at night between dusk and dawn and are inactive during the day, seeking the darkest available habitat (Kynard et al. 2005).

Rearing habitat preferences of green sturgeon larvae and juveniles in the Sacramento River are poorly understood (Stillwater Sciences 2007). However, additional information about habitat use is available for white sturgeon populations, which has been used as a proxy for green sturgeon. The seemingly random foraging patterns used by young sturgeon are probably a result of their poor ability to use visual cues to locate and capture food. Juveniles of other species of sturgeon have been shown to be non-visual feeders (Sbikin 1974, as cited in Utter et al. 1985), and it is generally assumed that most sturgeon use other senses than vision when feeding (Buddington and Christofferson 1985). This means that the success sturgeon have with mobile prey could be dependent on the amount of light available for prey to detect their approach (Utter et al. 1985). A non-visual predatory strategy would be an advantage to sturgeon when feeding on large populations of visually oriented prey species in habitats that are often turbid (Miller 1978, as cited in Utter et al. 1985). A dependence on sensory systems other than vision would also be advantageous when foraging at night or in areas too deep for light penetration. A random searching pattern is characteristic of all ages of juvenile sturgeon that were observed in laboratory and hatchery settings (Utter et al. 1985).

Olfactory cues are important for sturgeon when feeding on odorous food types. Sturgeon have large olfactory rosettes with both ciliated and microvillus receptors (Hara 1982, as cited in Utter et al. 1985), and Utter et al. (1985) observed that sturgeon behavior is instantaneously affected by contact with food odors. Sturgeon will often stop after detecting an odor and begin circling the general area in an attempt to contact the food item (Utter et al. 1985).

Tagged adult and subadult green sturgon in the San Francisco Bay estuary primarily occupied waters over shallow depths of less than 10 m, either swimming near the surface or foraging along the bottom (Kelly et al. 2007 as cited in Reclamation 2008a). In a study of juvenile green sturgeon in the Delta, relatively large numbers of juveniles were captured primarily in shallow waters from 1–3 m deep, indicating juveniles may require shallower depths for rearing and foraging (Radtke 1966).

5.3.4.5 Juvenile Emigration

Juvenile green sturgeon migrate downstream and feed mainly at night. Juvenile green sturgeon are taken in traps at the RBDD and the GCID diversion in Hamilton City, primarily in the months of May through August. Peak counts occur in the months of June and July (68 FR 4433). Juvenile emigration may reportedly extend through September (Environmental Protection Information Center et al. 2001).

Juvenile green sturgeon have been salvaged at the Harvey O. Banks Pumping Plant and the John E. Skinner Fish Collection Facility in the South Delta, and captured in trawling studies by Cal Fish and Wildlife during all months of the year (CDFG 2002). The majority of these fish were between 200 and 500 mm long, indicating they were from 2 to 3 years of age based on Klamath River age distribution work by Nakamoto et al. (1995). The lack of a significant proportion of juveniles shorter than approximately 200 mm in Delta captures indicates that juvenile green sturgeon likely hold in the mainstem Sacramento River, as suggested by Kynard et al. (2005).

5.3.4.6 Ocean Rearing

Once green sturgeon juveniles make their first entry into sea, they enter the subadult phase and spend a number of years migrating up and down the coast. Subadult green sturgeon mature in coastal marine environments and in bays and estuaries until they are at least 9-17 years old before returning to their natal freshwater river to spawn. While green sturgeon may enter river mouths and coastal bays throughout their years in the subadult phase, they do not return to their natal freshwater environments before they are mature (NMFS 2016a). During the summer months, multiple rivers and estuaries throughout the Southern DPS' range are visited by dense aggregations of green sturgeon (Moser and Lindley 2007; Lindley et al. 2011). Some of these aggregations are mixtures of both Southern DPS and Northern DPS green sturgeon, and there is considerable overlap in their ranges. However, Northern DPS green sturgeon do not appear to migrate into San Francisco Bay. Genetic studies on green sturgeon stocks indicate that the green sturgeon in the San Francisco Bay ecosystem belong to the Southern DPS (Israel et al. 2009). Capture of green sturgeon as well as tag detections in tagging studies have shown that green sturgeon are present in San Pablo Bay and San Francisco Bay at all months of the year (Kelly et al. 2006; Heublein et al. 2008; Lindley et al. 2011). An increasing amount of information is becoming available regarding green sturgeon habitat use in estuaries and coastal ocean and why they aggregate episodically (Lindley et al. 2008; Lindley et al. 2011).

5.3.5 Limiting Factors, Threats and Stressors

5.3.5.1 DPS

Limiting factors and threats to the Southern DPS of North American green sturgeon, both natural and anthropogenic, are presented according to the following five ESA listing factors.

5.3.5.1.1 Present or Threatened Destruction, Modification, or Curtailment of Habitat or Range (Reduction in Spawning Habitat, Alteration of Habitat)

Reduction in Spawning Habitat

Access to historical spawning habitat has been reduced by construction of migration barriers, such as major dams, that block or impede access to the spawning habitat. The principal factor for the decline of green sturgeon reportedly comes from the reduction of green sturgeon spawning habitat to a limited area of the Sacramento River (70 FR 17391). Although existing water storage dams only block access to about 9 percent of historically available green sturgeon habitat, Mora et al. (2009) suggest that the blocked areas historically contained relatively high amounts of spawning habitat because of their upstream position in the river system. Adams et al. (2007) hypothesized that significant amounts of historically-utilized spawning habitat may be blocked by Shasta Dam and Oroville Dam on the Feather River, reducing the productive capacity and simplifying the spatial structure of the Sacramento River green sturgeon population.

Keswick Dam is an impassible barrier blocking green sturgeon access to what are thought to have been historic spawning grounds upstream (70 FR 17386). Spawning currently appears to be limited to the upper portion of the mainstem Sacramento River downstream of Keswick Dam. In addition, a substantial amount of what may have been historical spawning and rearing habitat in the Feather River upstream of Oroville Dam has also been lost (70 FR 17386). According to NMFS (2016b), even if fish passage were provided past the Oroville Facilities, loss of access to historical spawning and rearing habitats upstream of the Oroville Facilities would probably continue somewhat into the foreseeable future due to the significant number of upstream hydroelectric projects that start at the upstream extent of Oroville Reservoir and extend into the upper watersheds of all main forks of the Feather River and their tributaries.

Alteration of Habitat

Green sturgeon habitat in the mainstem Sacramento River and the Delta has been greatly modified since the mid-1800s. Based on NMFS (2010d), the following examples illustrate relationships between threats to green sturgeon and specific types of habitat alteration:

- Hydraulic gold mining resulted in the removal of gravel and the deposition of mercury-laced fine sediment within streams, rivers, and the Bay/Delta estuary.
- Agricultural practices have converted tidal and seasonal marshlands and continue to release contaminants into Central Valley waterways.
- Levees have been created extensively along the Sacramento River and the Delta, resulting in the removal of riparian vegetation and the reduction of channel complexity.
- Historical reclamation of wetlands and islands, channelization and hardening of levees with riprap have reduced and degraded in- and off-channel intertidal and sub-tidal rearing habitat for green sturgeon.
- The hydrographs of the Sacramento River and its tributaries have been substantially altered from unimpaired conditions, and may no longer favorably correspond with green sturgeon lifestage periodicities.
- In-river water diversions alter flow and potentially entrain larval/juvenile green sturgeon.
- Introduced and invasive species have likely modified trophic relationships in both freshwater and estuarine habitats, which may have resulted in increased predation on young green sturgeon, as well as reduced growth and fitness as a result of feeding on non-optimal prey resources.

<u>Flows</u>

NMFS (2005d) and USFWS (1995b) found a strong correlation between mean daily freshwater outflow (April to July) and white sturgeon year class strength in the Sacramento-San Joaquin Estuary (these studies primarily involve the more abundant white sturgeon; however, the threats to green sturgeon are thought to be similar), indicating that insufficient flow rates are likely to pose a significant threat to green sturgeon (71 FR 17757). Low flow rates affect adult migration and may cause fish to stop their upstream migration or may delay access to spawning habitats. Also, it was posited that low flow rates could dampen survival by hampering the dispersal of larvae to areas of greater food availability, hampering the dispersal of larvae to all available habitat, delaying the transportation of larvae downstream of water diversions in the Delta, or decreasing nutrient supply to the nursery, thus stifling productivity (NMFS 2005d). Very little information is available on the habitat requirements and utilization patterns for early lifestages of green sturgeon (Mora et al. 2009).

Stranding due to flow reduction also may pose a threat to green sturgeon in the Sacramento River system. Green sturgeon that are attracted by high flows in the Yolo Bypass move onto the floodplain and eventually concentrate behind Fremont Weir, where they are blocked from further upstream migration (DWR 2005a). As the Yolo Bypass recedes, these sturgeon become stranded behind the flashboards of the weir and can be subjected to heavy illegal fishing pressure. Sturgeon can also be attracted to small pulse flows and trapped during the descending hydrograph (Harrell and Sommer 2003).

Water Temperatures

The installation of the Shasta Dam temperature control device in 1997 is thought to have reduced the previous problems related to high water temperatures in the upper Sacramento River, although Shasta Dam has a limited storage capacity and cold water reserves could be depleted in long droughts (NMFS 2007). Mayfield and Cech (2004) report a green sturgeon egg and larvae optimum range for growth and survival of 59 to 66°F (). Summer water temperatures in the upper Sacramento River have typically been below this range (NMFS 2007; NMFS 2015a). However, the compliance point has not been maintained in the Sacramento River during periods of 2014 and 2015 due to the historic drought. This recent change in temperature management has increased water temperatures throughout the green sturgeon spawning range in the Sacramento River (NMFS 2015a).

The operation of Oroville Dam and associated facilities produce complicated effects upon water temperature in the Feather River below Oroville Dam (NMFS 2016a). A variety of temperature control devices have been included in the Oroville Facilities, allowing DWR to adjust river temperatures to better suit the needs of listed fish species. According to NMFS (2016b), water temperatures during the green sturgeon spawning and early juvenile development period are one of the most significant stressors affecting green sturgeon individuals in the lower Feather River. Water temperatures within potential spawning areas are within optimal ranges during a majority of the spawning and early rearing period from March through May, but are warmer in June, exceeding optimal levels that may result in egg and early juvenile mortalities or abnormalities (NMFS 2016a). Although the range of optimal water temperatures varies depending on month and water year type, NMFS determined that there appears to be at least as much suitable spawning habitat now as under pre-dam conditions, and water temperatures appear adequate to support reproduction, especially during wet and above normal water years when green sturgeon production is known to be highest (NMFS 2016a). However, post-Oroville Dam water temperatures are cooler than historic river temperatures during the summer months when early lifestages are likely to be present in the lower Feather River (DWR 2005a in Reclamation 2008a). Prior to the construction of the Oroville Dam, water temperatures in the Feather River at Oroville averaged 65-71°F from June through August for the period of 1958-1968 (DWR 2004). After Oroville Dam construction, water temperatures in the Feather River at the Thermalito Afterbay averaged 60-65°F from June through August for the period of 1993-2002 (DWR 2004).

It is likely that high water temperatures (greater than 63°F) may deleteriously affect sturgeon egg and larval development, especially for late-spawning fish in drier water years (70 FR 17386).

5.3.5.1.2 Delayed or Blocked Migration

It has been suggested that the primary effect of construction of large water-storage reservoirs in the Sacramento–San Joaquin river basin has been to curtail the distribution of green sturgeon within the DPS (Mora et al. 2009). For example, water storage dams are hypothesized to be a major factor in the decline of green sturgeon in the Sacramento River (Adams et al. 2007). The existence and ongoing effects of these dams may have reduced the amount and altered the spatial distribution of spawning, rearing and holding habitat available and by restriction to the mainstem Sacramento River, resulting in green sturgeon becoming more vulnerable to environmental catastrophes (Mora et al. 2009).

Other potential adult migration barriers to green sturgeon have been reported to include the Sacramento Deep Water Ship Channel locks, Fremont Weir, Sutter Bypass, and the DCC Gates on the Sacramento River, and Shanghai Bench and Sunset Pumps on the Feather River (71 FR 17757).

DWR (2005) reported that the lock connecting the Sacramento River Deep Water Ship Channel with the Sacramento River blocks the migration of all fish from the deep water ship channel back to the Sacramento River. Thus, if green sturgeon enter the Sacramento River Deep Water Ship Channel, they will be unable to continue their migration upstream in the Sacramento River.

Green sturgeon are attracted by high floodwater flows into the Yolo Bypass, but are restricted from entering the Sacramento River by the Fremont Weir (DWR 2005a). Sturgeon also may be attracted to small pulse flows into the Yolo Bypass, and isolated during the descending hydrograph (Harrell and Sommer 2003).

Green sturgeon can become entrained in the Sutter Bypass during storm flow events. During April 2011, several sturgeon (green and white) were stranded behind the Tisdale Weir on the Sutter Bypass when storm flows receded. Cal Fish and Wildlife, in collaboration with UC Davis, organized a fish rescue operation and returned the sturgeon to the Sacramento River.

According to NMFS (2010c), the DCC, located near Walnut Grove, California, was constructed in 1951 to facilitate the transfer of fresh water from the Sacramento River to the federal and state pumps located in the south Delta. Flow from the Sacramento River into the DCC is controlled by two radial arm gates that can be opened or closed depending on water quality, flood protection, and fish protection requirements. When the gates are open, Sacramento River water is diverted into the Mokelumne and San Joaquin rivers. The gates are closed in fall to protect migrating salmonids, and then are opened the following spring. Thirty-percent of the tagged adult green sturgeon migrating down the Sacramento River after spawning entered the DCC (Israel et al. 2010). Most of these fish were able to successfully negotiate their way through the Delta and reach the Pacific Ocean. However, four fish were detected in the south Delta, with only one surviving to reach the Pacific Ocean. Juvenile green sturgeon may also be entrained into the interior delta during the summer when the DCC is open. Further studies are necessary to investigate the threat this alternative route through the Delta poses for these fish (NMFS 2010d).

NMFS (2009d) stated that potential physical barriers to adult green sturgeon migration in the Feather River are located at Shanghai Bench (RM 25) and at the Sutter Extension Water District's (SEWD) Sunset Pumps (RM 39). The breach of Shanghai Bench on the Feather River in early 2012 likely eliminated this naturally formed passage barrier (flow dependent) in the lower Feather River (pers. comm. with A. Seesholtz, DWR, May 13, 2013, as cited in NMFS 2015a). To raise the surface elevation of the river to allow SEWD's pumping facility (Sunset Pumps) to function properly, the SEWD maintains a boulder weir that stretches across the river. This structure blocks, or partially blocks, fish passage at low to moderate flows. The structure therefore prevents green sturgeon from accessing upriver spawning habitat until flows are sufficient for green sturgeon to pass over and above this impediment (NMFS 2016a). Impediments to migration may cause fish to stop their natural upstream migration or may delay access to spawning habitats (Moser and Ross 1995). Man-made (Sunset Pumps) impediments to upstream movements in the Feather River during low flow years might also limit significant spawning activities of green sturgeon above such obstacles to wet, high flow water years when they are most likely to be able to pass these obstacles (Beamesderfer et al. 2004).

5.3.5.1.3 Impaired Water Quality

Exposure of green sturgeon to toxics has been identified as a factor that can lower reproductive success, decrease early lifestage survival, and cause abnormal development, even at low concentrations (USFWS 1995a). Contamination of the Sacramento River increased substantially in the mid-1970s when application of rice pesticides increased (70 FR 17386). Additionally, water discharges containing metals from Iron Mountain Mine, located adjacent to the Sacramento River, have been identified as a factor affecting survival of sturgeon downstream of Keswick Dam. However, treatment processes and improved drainage management in recent years have reduced the toxicity of runoff from Iron Mountain Mine to acceptable levels. It has been reported that white sturgeon may accumulate polychlorinated biphenyls (PCBs) and selenium (White et al. 1989 as cited in Reclamation 2008a). While green sturgeon spend more time in the marine environment than white sturgeon and, therefore, may have less exposure, the NMFS BRT for North American green sturgeon concluded that contaminants also pose some risk for green sturgeon. However, this risk has not been quantified or estimated (NMFS 2007).

Additionally, events such as toxic oil or chemical spills in the upper Sacramento River could result in the loss of both spawning adults and their progeny, and lead to year-class failure (BRT 2005).

5.3.5.1.4 Dredging and Ship Traffic

Hydraulic suction dredging is conducted in the Sacramento and San Joaquin rivers, navigation channels within the Delta, and Suisun, San Pablo, and San Francisco bays. Juvenile green sturgeon residing within the Delta and the San Francisco Bay Estuary may be entrained during hydraulic suction dredging, which is conducted to maintain adequate depth within navigation areas or to mine sand for commercial use (NMFS 2010d). Additionally, the disposal of dredged material at aquatic sites within the estuary might bury green sturgeon or their prey, and expose green sturgeon to elevated levels of contaminated sediments (NMFS 2010d).

5.3.5.1.5 Ocean Energy Projects

According to NMFS (2010d), projects that harness the ocean's energy are currently being considered along the entire west coast. Potential concerns for green sturgeon include, but are not limited to, exposure to electromagnetic field (EMF) emissions, blade strikes, turbine entrainment, and ocean energy facilities functioning as fish aggregation devices. One of the primary concerns involves the exposure of green sturgeon to EMF generated from Project cables, turbine structures, and junction boxes, because green sturgeon use electroreceptors for feeding and perhaps migration, and these activities may be affected by EMF.

NMFS (2010d) suggested that the proposed installation and operation of energy-generating turbines at the mouths of several estuaries, including San Francisco Bay, may lead to injury and mortality as a result of potential blade strikes in association with turbine operation. Additionally, wave buoy and tidal turbine arrays may act as artificial reefs (e.g., DuPont 2008) or fish aggregation devices for marine mammals, fish, and invertebrates. If so, related changes to the local marine community, predator-prey interactions (i.e., increased presence of sea lions), or the distribution and abundance of marine species around ocean energy installation sites are also possible, and these sites are within the migratory corridors of green sturgeon (NMFS 2010d).

5.3.5.1.6 Commercial, Recreational, Scientific, or Educational Overutilization

While this factor was not considered the primary factor causing the decline of the Southern DPS of North American green sturgeon, it is believed that past and present commercial and recreational fishing is likely to pose a threat to green sturgeon (71 FR 17757).

Commercial, tribal, and recreational fishing probably had negative impacts on green sturgeon in the past. Current fishing regulations in Washington, Oregon, and California prohibit retention of green sturgeon in all commercial and recreational fisheries, although a small number of tribes still retain green sturgeon captured in some coastal bays and estuaries (NMFS 2010d).

Coastal groundfish trawl fisheries have been substantially reduced since the 1990s due to increasingly restrictive management measures (NMFS 2010d). These include reduced trip limits, increased gear restrictions, and a vessel buyback program, all of which are expected to reduce green sturgeon bycatch. Recent modifications to existing fishing regulations have almost certainly reduced overall green sturgeon take, but the impact of discard mortality and sublethal effects of capture remain unknown (NMFS 2010d).

As a long-lived, late maturing fish with relatively low fecundity and only periodic spawning, the green sturgeon is particularly susceptible to threats from overfishing (Musick 1999 as cited in Reclamation 2008a). Green sturgeon are vulnerable to recreational sport fishing with the Bay-Delta estuary and Sacramento River. Green sturgeon are primarily captured incidentally in California by sport fishermen targeting the more desirable white sturgeon, particularly in San Pablo and Suisun bays (Emmett et al. 1991). Since the listing of the Southern DPS of green sturgeon, new federal and state regulations, including the June 2, 2010 NMFS take prohibition (75 FR 30714), mandate that no green sturgeon can be taken or possessed in California (CDFG 2007a). If green sturgeon are caught incidentally and released during fishing for white sturgeon, the event must be reported to Cal Fish and Wildlife. The level of hooking mortality that results following release of green sturgeon by anglers is unknown. Cal Fish and Wildlife (2002) indicates that sturgeon are highly vulnerable to the fishery in areas where sturgeon are concentrated, such as the Delta and Suisun and San Pablo Bays in late winter and the upper Sacramento River during spawning migration. In March 2010, Cal Fish and Wildlife prohibited fishing for either white or green sturgeon within the upper mainstem Sacramento River between Keswick Dam and Butte Bridge (Hwy 162) in an effort to protect adult green sturgeon during their spawning runs (NMFS 2010d).

The demand for sturgeon caviar continues to increase both nationally and globally, and enforcement to protect sturgeon from poaching within the Central Valley is a high priority (CDFG 2002), as indicated by the number of sturgeon poaching operations that have been discovered there in recent years (NMFS 2010d). However, the degree to which poaching of green sturgeon occurs is largely unknown.

Poaching (illegal harvest) of sturgeon is known to occur in the Sacramento River, particularly in areas where sturgeon have been stranded (e.g., Fremont Weir), as well as throughout the Bay-Delta. Catches of sturgeon are thought to occur during all years, especially during wet years. The small population of green sturgeon inhabiting the San Joaquin River experiences heavy fishing pressure, particularly from illegal fishing (USFWS 1995a). Areas just downstream of Thermalito Afterbay Outlet, Cox’s Spillway, and several barriers impeding migration on the Feather River may be areas of high adult mortality from increased fishing efforts and poaching.

Poaching pressure is expected to remain high because of the increasing demand for caviar, coupled with the decline of other sturgeon species around the world, primarily the beluga sturgeon (71 FR 17757). Presently, however, poaching rates in the rivers and estuary and the impact of poaching on green sturgeon abundance and population dynamics are unknown.

The amount of green sturgeon take associated with scientific research has recently become a concern. NMFS (2010d) suggested that any Project (or suite of projects) that allows green sturgeon to be taken be carefully reviewed and evaluated.

In summary, NMFS (2015) concluded that the level of lethal take of Southern DPS of North American green sturgeon is not expected to have increased since 2006, but has decreased because of state and federal regulations that prohibit the retention of green sturgeon in almost all fisheries. Lethal take still occurs as a result of bycatch mortality and a limited number of permitted activities. The impact of lethal take on the overall population abundance of Southern DPS is still unknown, and no estimate of an annual rate of mortality due to poaching has become available since the last status review (NMFS 2015b).

5.3.5.1.7 Disease and Predation

A number of viral and bacterial infections have been reported for sturgeon in general (Mims et al. 2002), however specific issues related to diseases of green sturgeon have not been studied or reported. Therefore, it is not known if disease has played a role in the decline of the Southern DPS of green sturgeon.

The significance of predation on each lifestage of green sturgeon has not been determined. There has been an increasing prevalence of nonnative species in the Sacramento and San Joaquin rivers and the Delta (CDFG 2002) and this may pose a significant threat (NMFS 2010d). Striped bass, an introduced species, may affect the population viability of Chinook salmon (Lindley et al. 2004), and probably preys on other species, such as sturgeon (Blackwell and Juanes 1998). It is likely that sea lions consume green sturgeon in the San Francisco Bay estuary, but the extent to which this occurs is unknown (NMFS 2010d).

5.3.5.1.8 Inadequacy of Existing Regulatory Mechanisms

Inadequacy of existing regulatory mechanisms has contributed significantly to the decline of green sturgeon and to the severity of threats they currently face (NMFS 2010d; NMFS 2015a). During the process of developing the 4(d) rule for the Southern DPS of green sturgeon (70 FR 17386), NMFS noted several federal, State, and local regulatory programs that have been implemented to help reduce historical risk, including the AFRP of the CVPIA and the CALFED ERP. However, growing conflicts between the protection of other species (e.g., Sacramento River winter-run Chinook salmon and sea lions) may prove problematic for green sturgeon (NMFS 2010d). Although some effort has been made to improve habitat conditions across the range of the Southern DPS of green sturgeon, less progress has been accomplished through regulatory mechanisms to reduce threats posed by water diversions or blocked passage to spawning habitat (NMFS 2010d).

5.3.5.1.9 Other Natural or Man-Made Factors Affecting the Species' Continued Existence (Non-Native Invasive Species, Entrainment)

Non-Native Invasive Species

This factor was not considered a primary factor in the decline of the Southern DPS of green sturgeon. However, non-native species are an ongoing problem in the Sacramento and San Joaquin rivers and the Delta (CDFG 2002). One risk for green sturgeon associated with the introduction of non-native species involves the replacement of relatively uncontaminated food items with those that may be contaminated (70 FR 17386). Sturgeon regularly consume overbite and Asian clams, which is of particular concern because of the high bioaccumulation rates of these clams (Doroshov 2006 in BDCP 2010). The significance of this threat to green sturgeon is unclear (NMFS 2007). Green sturgeon also are likely to experience predation by introduced species including striped bass, but the actual impacts of predation have yet to be estimated (70 FR 17392). Introductions of non-native invasive plant species such as water hyacinth and Brazilian waterweed have altered habitat and have affected local assemblages of fish within the

Bay-Delta estuary (Nobriga et al. 2005), and may also affect green sturgeon through habitat alteration and potential increased predation rates on juveniles.

Entrainment

Larval and juvenile green sturgeon entrainment or impingement from screened and unscreened agricultural, municipal, and industrial water diversions along the Sacramento River and within the Delta is still considered an important threat (71 FR 17757). The threat of screened and unscreened agricultural, municipal, and industrial water diversions in the Sacramento River and Delta to green sturgeon is largely unknown because juvenile sturgeon are often not identified and current Cal Fish and Wildlife and NMFS screen criteria do not address sturgeon. Based on the temporal occurrence of juvenile green sturgeon and the high density of water diversion structures along rearing and migration routes, NMFS (2005a) found the potential threat of these diversions to be serious and in need of study.

In 1997, NMFS and Cal Fish and Wildlife developed screening criteria designed to prevent entrainment and impingement of juvenile salmonids. Similar criteria for larval and juvenile green sturgeon have not been developed and, although discussions regarding their development are occurring, there has been no timeline created for when guidelines will be available (NMFS 2010d).

The largest diversions within the Delta are the SWP and CVP export facilities, located in the southern Delta. Juvenile and sub-adult green sturgeon are recovered year-round at the CVP/SWP facilities, and have higher levels of salvage during the months of July and August compared to the other months of the year. The reason for this distribution is unknown. Based on salvage data, it appears that green sturgeon juveniles are present in the Clifton Court Forebay year round, but in varying numbers. NMFS (2009a) expects that predation on green sturgeon during their stays in the forebay is minimal, given their size and protective scutes, but this has never been verified.

In summary, NMFS (2015) concluded that no new information is available regarding the threats posed by non-native species. While efforts have been made to screen some large diversions, entrainment still poses a threat to the Southern DPS of North American green sturgeon. No changes in NMFS or CDFW screen criteria have been made since the last status review (NMFS 2015a).

5.3.5.2 Lower Yuba River

Given the extremely infrequent sightings of green sturgeon in the lower Yuba River, and the lack of green sturgeon life history information for the lower Yuba River, the foregoing discussion regarding threats and stressors for the DPS is assumed to be generally applicable to the lower Yuba River.

Moreover, according to NMFS (2008b), the lower Yuba River downstream of Daguerre Point Dam is subject to the same management considerations as the lower Feather River, which include operation of dams and water diversion operations resulting in the alteration of water flow and reduced water quality, in-water construction or alterations (e.g., bridge repairs, gravel augmentation, bank stabilization), and National Pollution Discharge Elimination System (NPDES) activities and other activities resulting in non-point source pollution (e.g., agricultural pesticide application, agricultural runoff and outfalls).

5.3.6 Summary of the Current Viability of the Southern DPS of North American Green Sturgeon

Although McElhany et al. (2000) specifically addresses viable populations of salmonids, NMFS (2009b) suggested that the concepts and viability parameters in McElhany et al. (2000) also could be applied to the Southern DPS of green sturgeon. Therefore, NMFS (2009b) applied the concept of VSP and reviewed population size, abundance, spatial distribution and diversity in the 2009 NMFS OCAP BO, and also applied the VSP concepts to green sturgeon in the 2009 Oroville FERC Relicensing NMFS BO (2009d; 2016b).

5.3.6.1 DPS

5.3.6.1.1 Abundance

Historically, abundance and population trends of the Southern DPS of North American green sturgeon have been inferred in two ways - by analyzing salvage numbers at the State and Federal pumping facilities, and by incidental catch of green sturgeon by the CDFW's white sturgeon sampling and tagging program. Both methods of estimating green sturgeon abundance are problematic because biases in the data are evident. As an example, a decrease in green sturgeon abundance has been inferred from a decrease in the amount of take observed at the south Delta pumping facilities (Skinner Delta Fish Protection Facility, Tracy Fish Collection Facility). NMFS (2016b) suggests that these data should be interpreted with some caution because operations and practices at the facilities have changed over the decades.

Currently, there are no reliable data on population sizes and population trends are lacking. However, beginning in 2010, more robust estimates of green sturgeon have been generated, and recent studies provide more reliable indices such as a minimum effective spawner population size found in Israel et al. (2009) or the Sacramento River Dual Frequency Identification Sonar (DIDSON) counts, which provide annual total spawner estimates (NMFS 2015b). In 2013, researchers at UC Davis began to release research findings on the population dynamics of breeding adult green sturgeon in the Sacramento River, including abundance estimates. Results of these surveys indicate an average annual spawning run size of 364 fish, with a variance of 246 (Klimley et al. 2015). The estimates in Klimley et al. (2015) do not include the number of spawning adults in the lower Feather River, where green sturgeon spawning was recently confirmed (NMFS 2016a). Estimates of adult green sturgeon in the Sacramento, Feather and Yuba rivers from 2010 - 2014 are provided in Table 5.3-2. As reported in NMFS (2016b), the numbers listed are likely "*unique individuals, although this is unverifiable given the survey methods used to collect the data*."

Table 5.3-2. Estimates of Adult Green Sturgeon Presence and Abundance (NMFS 2016a).

Year	River		
	Sacramento[1]	Feather[2]	Yuba[3]
2010	164	Data Unavailable	Data Unavailable
2011	220	25	4-5
2012	329	Data Unavailable	Presumed to be zero, but data unavailable
2013	338	Data Unavailable	Presumed to be zero, but data unavailable
2014	526	Data Unavailable	Presumed to be zero, but data unavailable

[1] NMFS 2015b
[2, 3] Cramer Fish Sciences 2011

Green sturgeon in the Sacramento River have been documented and studied more widely than those in either the Feather River or the Yuba River. In general, sturgeon year class strength appears to be episodic with overall abundance and dependent on a few successful spawning events. Genetic techniques were used to estimate the number of green sturgeon spawners contributing to juvenile production between 2002 and 2006 in the upper segment of spawning habitat above RBDD. Based upon these techniques, it was estimated that between 10 and 28 individuals contributed to juvenile production (Israel and May 2010). The study was also conducted prior to the decommissioning of RBDD (2011) when upstream access to spawning habitat by green sturgeon was limited (NMFS 2015a). Because populations appear to be not in equilibrium, conclusions regarding equilibrium dynamics are uncertain, given the lack of information (NMFS 2010d).

Since 2010, DIDSON surveys of aggregating sites in the upper Sacramento River are providing the first data on the number of spawning adult green sturgeon in the Southern DPS population (Table 5.3-2). DIDSON surveys of green sturgeon spawning sites have been conducted along the Sacramento River and have identified numerous spawning areas across a 75-mile stretch of the river (E. Mora, UC Davis, pers. comm., as cited in Bergman et al. 2016). Based on these results and estimates of mean spawning periodicity, the total number of adults in the Southern DPS population is estimated to be 1,348 ± 524 (E. Mora, pers. comm. UC Davis, May 6, 2015, as cited in NMFS 2015a).

Green sturgeon occasionally range into the Feather River, but numbers are low. NMFS (71 FR 17757) concluded that an effective population of spawning green sturgeon does not exist in the Feather River at the present time.

In summary, recent studies are providing preliminary information on the population abundance of Southern DPS of North American green sturgeon. This new information allows preliminary calculation of baseline information on spawning adult population abundance, although uncertainties exist because of the preliminary nature of the data (NMFS 2015a). Additionally, because the current time series is temporally limited, there is no basis for examining trends over time. Future surveys and abundance estimates will provide a basis for understanding the population trajectory of the Southern DPS (NMFS 2015a).

5.3.6.1.2 Productivity

There is insufficient information to evaluate the productivity of green sturgeon (NMFS 2009d). Recruitment data for green sturgeon are essentially nonexistent (NMFS 2009b). Incidental catches of larval green sturgeon in the mainstem Sacramento River and juvenile fish at the CVP and SWP pumping facilities in the South Delta suggest that green sturgeon are successful at spawning, but that annual year class strength may be highly variable (Beamesderfer et al. 2007; Adams et al. 2002). In general, green sturgeon year class strength appears to be episodic with overall abundance dependent upon a few successful spawning events (NMFS 2010d). Other indicators of productivity, such as cohort replacement ratios and spawner abundance trends, require data sets that simply do not exist for the Southern DPS of North American green sturgeon (NMFS 2016a). However, green sturgeon are iteroparous and long-lived, so that spawning failure in any one year may be rectified in a succeeding spawning year (NMFS 2009b). The long lifespan and the long age to maturity makes trend detection dependent upon data sets spanning decades, which is something that is currently lacking. NMFS (2015) found that the relationship between altered flows and temperatures in spawning and rearing habitat, and Southern DPS green sturgeon population productivity is uncertain. However, the studies being conducted on the Sacramento River and on the Feather River may eventually produce enough data to gain statistically robust insights into productivity (NMFS 2016a).

5.3.6.1.3 Spatial Structure

Historical green sturgeon spawning habitat may have extended up into the three major branches of the upper Sacramento River above the current location of Shasta Dam – the Little Sacramento River, the Pit River, and the McCloud River (NMFS 2009b; NMFS 2009e). Additional spawning habitat is believed to have once existed above the current location of Oroville Dam on the Feather River (NMFS 2009b). Current scientific information indicates that the Southern DPS of green sturgeon population has been relegated to a single, independent population, which principally spawning in the mainstem Sacramento River, and green sturgeon also breed opporunitistically in the Feather River (NMFS 2016a) which is, for the most part, outside of their historical spawning area.

According to NMFS (2009b; 2016b), the reduction of green sturgeon spawning habitat into one reach on the Sacramento River between Keswick Dam and Hamilton City has increased the vulnerability of this spawning population to catastrophic events. One spill of toxic materials into this reach of river, similar to the Cantara Loop spill of herbicides on the upper Sacramento River, could remove a significant proportion of the adult spawning broodstock from the population, as well as reduce the recruitment of the exposed year class of juvenile fish. Concentration of adults into a very few select spawning locations also makes the species highly vulnerable to poaching (NMFS 2016a). Additionally, extended drought conditions could imperil the spawning success for green sturgeon (NMFS 2009b).

5.3.6.1.4 Diversity

Diversity, both genetic and behavior, provides a species the opportunity to track and adapt to environmental changes. While it is recognized that diversity is crucial to the viability of a species in general, it is not well understood how well the Southern DPS of green sturgeon display these diversity traits and if there is sufficient diversity to buffer against long-term extinction risk (NMFS 2016a). The reduction of the Southern DPS of green sturgeon population to one extant spawning population has reduced the potential variation of life history expression and genetic diversity within this population (NMFS 2009e). In addition, the historical closed gate configuration at RBDD from mid-May to September may have altered the genetic diversity of the population by separating the population into upstream and downstream spawning groups based on run timing (NMFS 2009b).

Green sturgeon stocks from the northern and southern DPSs are genetically differentiated (Israel et al. 2004; Israel et al. 2009). Genetic differentiation is moderate and statistically similar between the southern and northern DPSs (NMFS 2010d). However, the genetic diversity of the Southern DPS is not well understood (NMFS 2009e). NMFS (2016b) suggests that the diversity of the Southern DPS of green sturgeon is probably low, given available abundance estimates. Also, because human alteration of the environment is so pervasive in the Central Valley, basic diversity principles such as run timing and behavior are likely adversely influenced through mechanisms such as diminished springtime flow rates as water is impounded behind dams (NMFS 2016a).

5.3.6.1.5 Summary of the Current Viability of the Southern DPS of North American Green Sturgeon

The Southern DPS of green sturgeon is at substantial risk of future population declines (Adams et al. 2007). The principal threat to green sturgeon in the Southern DPS is the reduction in available spawning habitat due to the construction of barriers on Central Valley rivers (NMFS 2009e). According to NMFS (2009b; 2016b), the potential threats faced by the green sturgeon include enhanced vulnerability due to the reduction of spawning habitat into one concentrated area on the Sacramento River, lack of good empirical population data, vulnerability of long-term cold water supply for egg incubation and larval survival, loss of juvenile green sturgeon due to entrainment at the Project fish collection facilities in the South Delta and agricultural diversions within the Sacramento River and the Delta, alterations of food resources due to changes in the Sacramento River and Delta habitats, and exposure to various sources of contaminants throughout the basin to juvenile, sub-adult, and adult lifestages. In summary, NMFS (2016b) concluded that the risk of extinction for the Southern DPS of green sturgeon is moderate, because, although threats due to habitat alteration are thought to be high and indirect evidence suggests a decline in abundance, there is much uncertainty regarding the scope of threats and the viability of population abundance indices.

A study (Thomas et al. 2013) provided additional analysis regarding population-level impacts due to stranding of green sturgeon. During April 2011, 24 green sturgeon were rescued that had been stranded behind two weirs (Fremont and Tisdale) along the Sacramento River. Those 24 green sturgeon were acoustically tagged and their survival and migration success to their spawning grounds was analyzed. Additionally, population viability modeling and analysis was conducted to show the potential impacts of stranding and the benefits of conducting rescues at the population level. Population viability analyses of rescue predicted a 7 percent decrease below the population baseline model over 50 years as opposed to 33 percent without rescue (Thomas et al. 2013).

5.3.6.2 Lower Yuba River

As previously discussed, very few observations of green sturgeon have occurred in the Yuba River historically or in recent years. The few occasions when confirmed observations have occurred were downstream of Daguerre Point Dam and consisted of adult green sturgeon. Green sturgeon acoustic tag detections do not indicate substantive use of the Yuba River (YCWA 2013).

Monitoring and studies of green sturgeon in the Delta, the Sacramento River and its tributaries continue to be undertaken by a variety of agencies implementing numerous different programs. The CFTC continues to monitor acoustically tagged green sturgeon throughout the system, and fixed-station acoustic monitors and roving hydrophonic surveys continue to be conducted on the lower Yuba River by both the RMT and Cal Fish and Wildlife's Heritage and Wild Trout and the Steelhead Management and Recovery Programs. The AFRP is continuing to fund ongoing sturgeon videographic monitoring efforts in the Feather River Basin, including the lower Yuba River. Additionally, the Sturgeon Interagency Ecological Program (IEP) Project Work Team coordinates green sturgeon research, disseminates information and is overseeing the development of a green sturgeon population model, and the USACE's Long-term Management Strategy (LTMS) for the Placement of Dredged Material in the San Francisco Bay Region Program includes green sturgeon tracking, evaluation of susceptibility to suction dredging and development of entrainment models. Available results from these and other programs may provide additional information regarding green sturgeon in the Central Valley and lower Yuba River. However, despite the contribution resulting from these and other studies conducted to date, knowledge of the population biology and dynamics of green sturgeon remains limited.

Limited information regarding green sturgeon abundance, distribution, movement and behavioral patterns, as well as lifestage-specific habitat utilization preferences, is available for the Sacramento and Feather rivers. According to NMFS (2009b), the current population status of the Southern DPS of North American green sturgeon is unknown. Currently, there is little reliable data on population sizes, and population trends are lacking (NMFS 2015a; 2016b). There is insufficient information to evaluate the productivity of green sturgeon (NMFS 2015a), and recruitment data for green sturgeon are essentially nonexistent (NMFS 2009b). Essentially no information regarding these topics is available for the lower Yuba River.

Hence, it is not practicable to attempt to apply the VSP concepts developed for salmonids to green sturgeon in the lower Yuba River. Moreover, the lack of information pertaining to abundance, productivity, habitat utilization, life history and behavioral patterns in the lower Yuba River, due to infrequent sightings over the past several decades, does not provide the opportunity for reliable alternative methods of viability assessment of green sturgeon in the lower Yuba River. Data limitations preclude application of the extinction risk criteria to green sturgeon in the lower Yuba River. Consequently, green sturgeon in the lower Yuba River cannot be concluded to be stable or at a specific risk of extinction.

5.3.7 Recovery Considerations

In November 2009, NMFS (74 FR 58245) announced its intent to develop a recovery plan for the Southern DPS of North American green sturgeon. NMFS is required by the ESA to develop and implement recovery plans for the conservation and survival of ESA-listed species. As part of the process, NMFS will be coordinating with state, Federal, tribal, and local entities in California, Oregon, Washington, Canada, and Alaska to develop the recovery plan.

Presently, NMFS is in the process of preparing the draft recovery plan, and has prepared an outline of the plan (NMFS 2010d). As stated in the outline, the goal is to set out a plan to conserve and recover green sturgeon by identifying actions that may improve its potential for recovery. These include, but are not limited to, the following:

- Improve existing research and initiate novel research and monitoring on distribution, status, trends, and lifestage survival of the Southern DPS of green sturgeon at the population level.
- Establish better inter- and intra-agency coordination regarding scientific research conducted on green sturgeon under ESA sections 7, 10, and 4(d).
- Evaluate the significance of green sturgeon bycatch in commercial fisheries through the implementation of directed surveys.
- NMFS Office of Law Enforcement (OLE) should monitor and collaborate with state enforcement agencies along the west coast related to illegal retention of green sturgeon in recreational fisheries.
- NMFS OLE should collaborate with Cal Fish and Wildlife wardens to address sturgeon poaching in the Central Valley.
- Assess the potential for establishing independent spawning populations in areas outside of the mainstem Sacramento River (e.g., Feather, Yuba, Russian rivers, as well as tributaries of San Joaquin River).
- Address the need to develop a multiple species water flow and temperature management plan for Shasta, Keswick, Oroville and Englebright dams.
- Address the application of pesticides (Carbaryl and others) and herbicides applied to control burrowing shrimp and non-native plants in estuaries.
- Identify and prioritize potential contaminants of concern in the Central Valley.
- Ensure that screens are placed on water diversions on the upper mainstem Sacramento River below Keswick Dam and that they are designed to be protective of larval and juvenile green sturgeon. Research on screening criteria should be initiated as soon as feasible.
- Continue to support the removal of the Red Bluff Diversion Dam.
- Monitor hydraulic suction dredges for potential entrainment of juvenile green sturgeon.
- Determine the impact of non-native species.

- Determine if electromagnetic fields produced by offshore energy projects alter green sturgeon migration patterns.

The draft recovery plan outline (NMFS 2010d) further states that recovery actions will be refined in the recovery plan, and will be specific to several regions, including the Sacramento River, the Delta/Estuary, and coastal marine areas, which include several estuaries/bays. Actions specific to lifestages in each region will be identified to address more localized factors that currently suppress potential for recovery for green sturgeon (NMFS 2010d).

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