Appendix A

Covered Species Accounts

APPENDIX A1. CENTRAL VALLEY STEELHEAD (*ONCORHYNCHUS MYKISS*)

A1.1 LEGAL STATUS

The Central Valley steelhead Evolutionarily Significant Unit (ESU) was listed as a threatened species under the Federal Endangered Species Act (ESA) on March 19, 1998, and includes all naturally spawned populations of steelhead in the Sacramento and San Joaquin rivers and their tributaries, including the Bay-Delta (63 FR 13347). Steelhead from San Francisco and San Pablo Bays and their tributaries are excluded from this listing but are included in the Central California Coast DPS, which is also listed as threatened under the ESA. On June 14, 2004, the National Marine Fisheries Service (NMFS) proposed that all West Coast steelhead be reclassified from ESUs to Distinct Population Segments (DPS) and proposed to retain Central Valley steelhead as threatened (69 FR 33102). On January 5, 2006, after reviewing the best available scientific and commercial information, NMFS issued its final decision to retain the status of Central Valley steelhead as a threatened DPS (71 FR 834). This decision included the Coleman National Fish Hatchery and Feather River Hatchery steelhead populations. These populations were previously included in the ESU but were not deemed essential for conservation and thus not part of the listed steelhead population.

Central Valley steelhead are not listed under the California Endangered Species Act but are designated as a California Species of Special Concern.

A1.2 SPECIES DISTRIBUTION AND STATUS

Information on the status and geographic distribution of Central Valley steelhead is extremely limited (The Nature Conservancy et al. 2008). Adult steelhead typically migrate upstream and spawn during the winter months when river flows are high and water clarity is low. Unlike Chinook salmon, adult steelhead do not necessarily die after spawning and can return to coastal waters. Juvenile steelhead cannot be differentiated from resident rainbow trout based on visual characteristics. In addition, steelhead frequently inhabit streams and rivers that are difficult to access and survey. As a result of these and other factors, information on the trends in steelhead abundance within the Central Valley has primarily been limited to observations at fish ladders and weirs (e.g., Red Bluff Diversion Dam [RBDD] when the gates were closed, Woodbridge Irrigation District dam and fish ladders on the Mokelumne River, etc.) and returns to Central Valley fish hatcheries. Juvenile steelhead are collected incidentally in various fishery surveys (e.g., Mossdale and Chipps Island trawls). However, as a result of their relatively large size and good swimming performance, juvenile steelhead are able to avoid capture in most fishery surveys. Therefore, information on the distribution, abundance, habitat use, and behavior of steelhead within the Plan Area is very limited.

A1.2.1 Range and Status

Central Valley steelhead were widely distributed historically throughout the Sacramento and San Joaquin rivers (see Figure A-1a) (Busby et al. 1996, McEwan 2001). Steelhead inhabited waterways from the upper Sacramento and Pit River river systems (now inaccessable due to Shasta and Keswick Dams) south to the Kings River 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 there were historically at least 81 independent Central Valley steelhead populations distributed primarily throughout the eastern tributaries of the Sacramento and San Joaquin rivers.

The geographic distribution of spawning and juvenile rearing habitat for Central Valley steelhead has been greatly reduced by the construction of dams (McEwan and Jackson 1996, McEwan 2001). Presently, impassable dams block access to 80 percent of historically available habitat, and all spawning habitat for approximately 38 percent of historic populations (Lindley et al. 2006). Existing wild steelhead stocks in the Central Valley inhabit 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 and Jackson 1996).

Historical Central Valley steelhead run sizes are difficult to estimate given the paucity of data, but may have approached one to two million adults annually (McEwan 2001). By the early 1960s, steelhead run size had declined to approximately 40,000 adults (McEwan 2001). Over the past 30 years, naturally-spawned steelhead populations in the upper Sacramento River have declined substantially (see Figure A-1b). Until recently, Central Valley steelhead were thought to be extirpated from the San Joaquin River system. However, recent monitoring has detected small self-sustaining populations of steelhead in the Stanislaus, Mokelumne, and Calaveras rivers, and other streams previously thought to be devoid of steelhead (McEwan 2001). 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). Some of these fish, however, may have been resident rainbow trout, which are the same species but are not anadromous.

A1.2.2 Distribution and Status in the Plan Area

The entire population of the Central Valley steelhead DPS must pass through the Plan Area as adults migrating upstream to spawning areas and juveniles emigrating downstream to rearing areas and the ocean. Furthermore, juvenile steelhead likely use the Delta as well as Suisun Marsh and the Yolo Bypass for rearing. Adult Central Valley steelhead migrating into the San Joaquin River and its tributaries use the central, southern, and eastern edge of the Delta, whereas adults entering the Sacramento River system to spawn use the northern, western, and central Delta as a migration pathway

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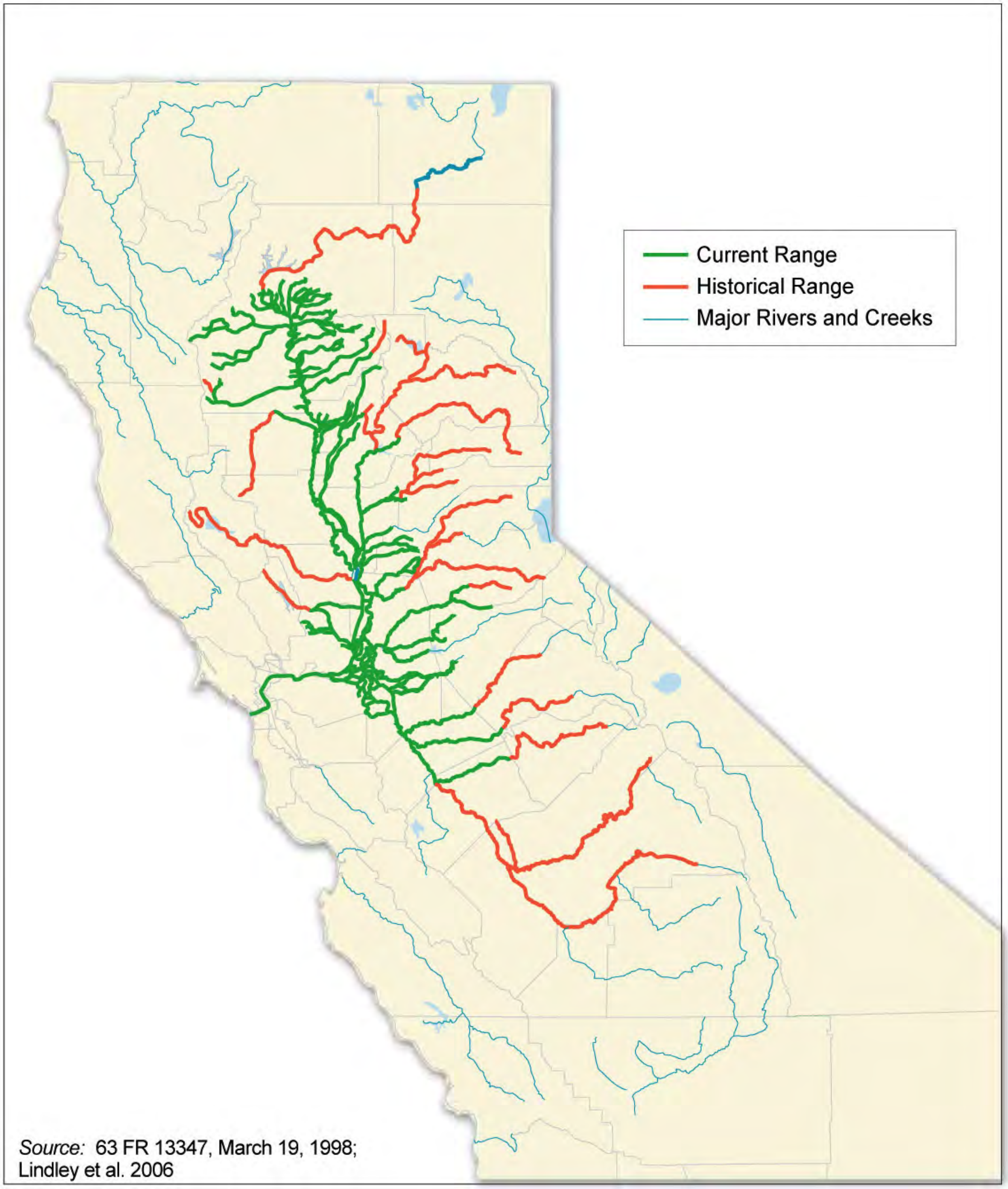
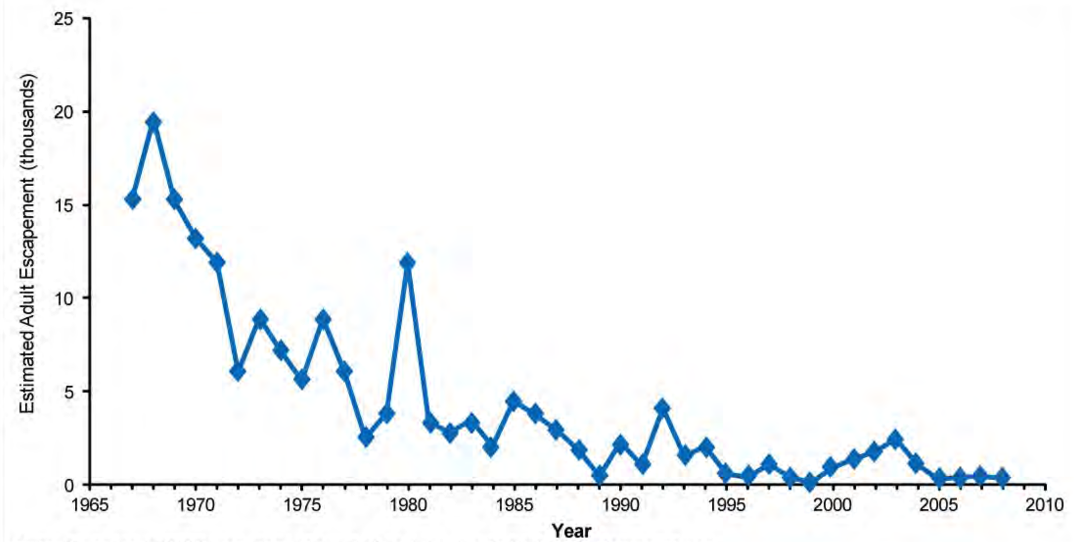


Figure A-1a. Central Valley Steelhead Inland Range in California



Note: Trapping in 2008 was discontinued after June due to high water temperatures.

Source: 1967-1991, Mills and Fisher (1994); 1992-2008, CDFG, Red Bluff (unpubl. data).

Figure A-1b. Estimated Historical Spawner Escapement of Wild Central Valley Steelhead in the Upper Sacramento River Upstream of the Red Bluff Diversion Dam (1967-2008)

A1.3 HABITAT REQUIREMENTS AND SPECIAL CONSIDERATIONS

Critical habitat for the Central Valley steelhead DPS was designated by NMFS on September 2, 2005 (70 FR 52488) with an effective date of January 2, 2006 and includes 2,308 miles of stream habitat in the Central Valley and an additional 254 square miles of estuarine habitat in the San Francisco-San Pablo-Suisun Bay complex (see Figure A-1c). Critical habitat for Central Valley steelhead includes stream reaches such as those of the Sacramento, Feather, and Yuba rivers; Deer, Mill, Battle, and Antelope creeks in the Sacramento River basin; the San Joaquin River and its tributaries; and the Delta. Critical habitat includes stream channels in the designated stream reaches and the lateral extent as defined by the ordinary high-water line. In areas where the ordinary high-water line has not been defined, the lateral extent of critical habitat is defined by the bankful 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) (70 FR 52488). Critical habitat for Central Valley steelhead is defined as specific areas that contain the Primary Constituent Elements (PCEs) and physical habitat elements or biological features essential to the conservation of a species for which its designated or proposed critical habitat is based on" (USFWS 2004). The following are the habitat types used as PCEs for Central Valley steelhead.

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Figure A-1c. Central Valley Steelhead Critical Inland Habitat in California

A1.3.1 Spawning Habitat

Freshwater spawning sites are those with water quantity and quality conditions and substrate supporting spawning, egg incubation, and larval development. Spawning habitat for Central Valley steelhead primarily occurs in mid to upper elevation reaches or immediately downstream of dams located throughout the Central Valley, which contain suitable environmental conditions (e.g., seasonal water temperatures, substrate, dissolved oxygen, etc.) for spawning and egg incubation. Spawning habitat has a high conservation value as its function directly affects the spawning success and reproductive potential of steelhead.

A1.3.2 Freshwater Rearing Habitat

Freshwater steelhead rearing sites are those with suitable instream flows, water quantity (e.g., water temperatures) and floodplain connectivity to form and maintain physical habitat conditions that support juvenile growth and mobility, provide forage species and include cover such as shade, submerged and overhanging large wood, log jams, beaver dams, aquatic vegetation, large rocks and boulders, side channels, and undercut banks. Spawning areas and migratory corridors may also function as rearing habitat for juveniles, which feed and grow before and during their outmigration. Rearing habitat quality is strongly affected by habitat complexity, food supply, and the presence of predators. Some of these more complex and productive habitats with floodplain connectivity are still present in the Central Valley (e.g., the lower Cosumnes River, Sacramento River reaches with set-back levees [i.e., primarily located upstream of the City of Colusa]). The channeled, leveed, and riprapped river reaches and sloughs common in the lower Sacramento and San Joaquin rivers and throughout the Delta, however, typically have low habitat complexity, low abundance of food organisms, and offer little protection from predation by fish and birds. Freshwater rearing habitat has a high conservation value because juvenile steelhead are dependent on the function of this habitat for successful survival and recruitment to the adult population.

A1.3.3 Freshwater Migration Corridors

Optimal freshwater steelhead migration corridors (including river channels, channels through the Delta, and the Bay-Delta estuary) support mobility, survival, and food supply for juveniles and adults. Migration corridors should be free from obstructions (passage barriers and impediments to migration), provide favorable water quantity (instream flows) and quality conditions (seasonal water temperatures), and contain natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, side channels, and undercut banks. Migratory corridors are typically downstream of the spawning area and include the lower Sacramento and San Joaquin rivers, the Delta, and the San Francisco Bay complex extending to coastal marine waters. These corridors allow the upstream passage of adults and the downstream emigration of juvenile steelhead. Migratory corridor conditions are strongly affected by the presence of passage barriers, which can include dams, unscreened or poorly screened diversions, and degraded water quality. For freshwater migration corridors to function properly, they must provide adequate passage, provide suitable migration cues, reduce false attraction, avoid areas where vulnerability to predation is increased, and avoid impediments and delays in both upstream and downstream migration. For this reason, freshwater migration corridors are considered to have a high conservation value.

A1.3.4 Estuarine Areas

Estuarine migration and juvenile rearing habitats should be free of obstructions (i.e., dams and other barriers) and provide suitable water quality, water quantity, and salinity conditions to support juvenile and adult physiological transitions between fresh and salt water. Natural cover, such as submerged and overhanging large wood, aquatic vegetation, and side channels, provide juvenile and adult foraging. Estuarine areas contain a high conservation value as they function to support juvenile steelhead growth, smolting, avoidance of predators, and provide a transition to the ocean environment.

A1.3.5 Ocean Habitats

Although ocean habitats have not been designated as critical habitat for Central Valley steelhead, biologically productive coastal waters are an important habitat component. Juvenile steelhead rear within coastal marine waters for a period of approximately one to three years before returning to the Central Valley rivers as adults to spawn. During their marine residence, steelhead forage on krill and other marine organisms. Offshore marine areas with water quality conditions and food, including squid, crustaceans, and fish (fish become a larger component in the steelhead diet later in life [Moyle 2002]), to support growth and maturation are important habitat elements. These features are essential for conservation because, without them, juveniles cannot forage and grow to adulthood.

Results of oceanographic studies have shown the variation in ocean productivity off the West Coast within and among years. Changes in ocean currents and upwelling have been identified as significant factors affecting nutrient availability, phytoplankton and zooplankton production in near-shore surface waters. Although the effects of ocean conditions on steelhead growth and survival have not been investigated, recent observations since 2007 have shown a significant decline in the abundance of adult Chinook salmon and coho salmon returning to California rivers and streams. This decline has been hypothesized to be the result of declines in ocean productivity and associated high mortality rates during the period when these fish were rearing in near-shore coastal waters (MacFarlane et al. 2008). The importance of changes in ocean conditions on growth, survival, and population abundance of Central Valley steelhead, although potentially similar to that of Chinook salmon, is largely unknown.

A1.4 LIFE HISTORY

Steelhead can be divided into two life history types based on their state of sexual maturity at the time of river entry and the duration of their spawning migration: stream-maturing and ocean-maturing. Stream-maturing steelhead enter freshwater in a sexually immature condition and require several months to mature prior to spawning, whereas ocean-maturing steelhead enter freshwater with well-developed gonads and spawn shortly after river entry. These two life history types are more commonly referred to by their season of freshwater entry (i.e., summer [stream-maturing] and winter [ocean-maturing] steelhead). Only winter steelhead currently are present in Central Valley rivers and streams (McEwan and Jackson 1996). There are, however, indications that summer steelhead were present in the Sacramento River system prior to the commencement of large-scale dam construction in the 1940s (Interagency Ecological Program (IEP) Steelhead Project Work Team 1999, McEwan 2001). At present, summer steelhead are found only in North Coast drainages, mostly in tributaries of the Eel, Klamath, and Trinity river systems (McEwan and Jackson 1996).

There is high polymorphism among steelhead/rainbow trout populations with respect to a continuum from anadromy to permanent freshwater residency (Behnke 1992 as cited in McEwan 2001). Furthermore, there is plasticity in an individual from a specific life history form to assume a different life history strategy if conditions necessitate it (McEwan 2001). For example, if environmental conditions, such as water temperature and flow, allow for year-round residence in freshwater, an individual may choose not to emigrate to the ocean. This polymorphic life history structure provides the flexibility for steelhead to remain persistent within highly variable conditions, particularly near the edges of their range (McEwan 2001).

Central Valley steelhead generally leave the ocean and migrate upstream from August through April (Busby et al. 1996), and spawn from December through April. Peak spawning typically occurs from January though March in small streams and tributaries where cool, well-oxygenated water is available year-round (see Table A-1a; Hallock et al. 1961, McEwan and Jackson 1996). Timing of upstream migration is correlated with higher flow events such as freshets and associated lower water temperatures and increased turbidity. Before the occurrence of large-scale changes to the hydrology of the Delta system, the peak period of adult immigration appears to have been during fall months with a smaller component of immigrants in the winter (as reviewed in McEwan 2001). Unlike Pacific salmon, steelhead are iteroparous, or capable of spawning more than once before death (Busby et al. 1996). It is, however, rare for steelhead to spawn more than twice before dying; most individuals that do spawn more than twice are females (Busby et al. 1996). Iteroparity is more common among southern steelhead populations than northern populations (Busby et al. 1996). Although one-time spawners are the great majority, Shapolov and Taft (1954) reported that repeat spawners are relatively numerous (17.2 percent) in California streams.

After reaching a suitable spawning area, the female steelhead selects a site with good intergravel flow, digs a redd, and deposits eggs while an attendant male fertilizes them. Eggs in the redd are covered with gravel dislodged just upstream by similar redd building actions. The length of time it takes for eggs to hatch varies in response to water temperature. Hatching of steelhead eggs in hatcheries takes about 30 days at 51 °F (10.6 °C). Fry generally emerge from the gravel four to six weeks after hatching, but factors such as redd depth, gravel size, siltation, and water temperature can speed or retard the time to emergence (Shapovalov and Taft 1954, as cited in

Table A-1a. Temporal occurrence of (a) adult and (b) juvenile Central Valley steelhead in the Central Valley. Darker shades indicate months of greatest relative abundance.

(a) Adult

Location	*Jan*		*Feb*		*Mar*		*Apr*		*May*		*Jun*		*Jul*		*Aug*		*Sep*		*Oct*		*Nov*		*Dec*	
[1,3]Sacramento (Sac) River (R.)	L	L	L	L	L	L						L	L	L	L	M	M	H	H	M	L	L	L	L
[2,3]Sac R. at Red Bluff	L	L	L	L	L	L	L	L	L	L	L	L	L	L	L	L	M	M	H	M	L	L	L	L
[4]Mill, Deer creeks			H	M	M	L	L	L	L	L	L	L								H	H	M	M	M
[5]Sac R. at Fremont Weir	L	L	L	L											M	M	H	H	M	M	L	L	L	L
[6]San Joaquin River	M	M	M	M	M	M							L	L	L	L	M	M	M	M	M	M	H	H

(b) Juvenile

Location	*Jan*		*Feb*		*Mar*		*Apr*		*May*		*Jun*		*Jul*		*Aug*		*Sep*		*Oct*		*Nov*		*Dec*	
[1,2]Sac R.	L	L	L	L	M	M	M	M	M	M	M	M	L	L	L	L	L	L	M	M	M	M	L	L
[2,7]Sac R. at Knights Landing (KL)	L	L	L	L	H	H	L	L	L															L
[8]Sac R. at KL	M	M	M	M	H	H	H	H	M	M											L	L	L	L
[9]Chipps Island (wild)	L	L	L	L	H	H	H	H	M	M	M	M	L	L					L	L	L	L	L	L
[7]Mossdale							H	H	H	H														
[10]Woodbridge Dam	M	M	M	M	M	M	M	M	M	M	M	M	M	M										
[11]Stanislaus R. at Caswell	L	L	M	M	H	H	M	M	M	M	L	L												
[12]Sac R. at Hood	L	L	H	H	H	H	H	H	H	H											L	L	L	L
Relative Abundance:	H	= High							M	= Medium							L	= Low						

Sources: [1]Hallock et al. 1961; [2]McEwan 2001; [3]USFWS unpublished data; [4]DFG 1995; [5]Hallock et al. 1957 [6]Based on limited unpublished data from DFG Steelhead Report Card; [7]DFG unpublished data; [8]Snider and Titus 2000; [9]Nobriga and Cadrett 2003; [10]Jones & Stokes Associates, Inc., 2002; [11]S.P. Cramer and Associates, Inc. 2000 and 2001; [12]Schaffter 1980

McEwan and Jackson 1996). Newly emerged fry move to shallow, protected areas with lower water velocities associated with the stream margin, and soon establish feeding locations within the juvenile rearing habitat (Shapovalov and Taft 1954, as cited in McEwan and Jackson 1996).

Steelhead rearing during the summer takes place primarily in higher velocity areas in pools, although young-of-the-year also are abundant in glides and riffles. Productive steelhead habitat is characterized by habitat complexity, primarily in the form of large and small woody debris. Cover is an important habitat component for juvenile steelhead both as velocity refugia and as a means of avoiding predation (Meehan and Bjornn 1991, as cited in McEwan and Jackson 1996).

Juvenile steelhead emigrate episodically from natal streams during fall, winter, and spring high flows. Emigrating Central Valley steelhead use the lower reaches of the Sacramento and San Joaquin rivers and the Delta for rearing and as a migration corridor to the ocean. Juvenile Central Valley steelhead feed mostly on drifting aquatic organisms and terrestrial insects and will also take active bottom invertebrates (Moyle 2002).

Some juvenile steelhead may use tidal marsh areas, non-tidal freshwater marshes, and other shallow water areas in the Delta and estuary as rearing areas for short periods prior to their emigration to the ocean. 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. Nobriga and Cadrett (2003) verified these temporal findings based on analysis of captures in USFWS salmon monitoring conducted near Chipps Island. Diversity and richness of habitat and food sources in the estuary allow juveniles to attain a larger size before entry into the ocean, thereby increasing their chances for survival in the marine environment.

Central Valley steelhead spend from several months to 3 years (with a maximum of 6 years) in the Pacific Ocean before returning to freshwater. The age composition of the steelhead population in the Pacific Ocean is dominated by 1-year (61.9 percent) and 2-year (31.4 percent) fish (Burgner et al. 1992). Ocean migration and distribution of Central Valley steelhead stocks is unknown.

Steelhead experience most of their marine phase mortality soon after they enter the Pacific Ocean (Pearcy 1992). Ocean mortality is poorly understood, however, because few studies have been conducted to evaluate the importance of various factors including predation mortality, changes in ocean currents, water temperatures, and coastal upwelling, on steelhead survival. Possible causes of ocean mortality include predation, competition, starvation, osmotic stress, unauthorized high seas driftnet fisheries, disease, advective losses and other poor environmental conditions (Wooster 1983, Cooper and Johnson 1992, Pearcy 1992). Competition between steelhead and other species for limited food resources in the Pacific Ocean may be a contributing factor to declines in steelhead populations, particularly during years of low productivity (Cooper and Johnson 1992).

Ocean 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 1992). For juvenile steelhead, warmer water and weak upwelling are possibly the most important of the ocean conditions associated with El Niño. Because of the weakened upwelling during an El Niño year, juvenile California steelhead must migrate more actively offshore through possibly stressful warm waters with numerous inshore predators. Strong upwelling is probably beneficial because of the greater transport of smolts offshore, beyond major concentrations of inshore predators (Pearcy 1992). Investigations are currently underway to examine decadal oscillations in coastal marine environmental conditions and the associated biological changes that may affect the survival, growth, and recruitment of steelhead to the adult population.

A1.5 THREATS AND STRESSORS

The following have been identified as important threats and stressors to Central Valley steelhead (without priority).

Reduced staging and spawning habitat. Adult steelhead historically migrated upstream into higher gradient reaches of rivers and tributaries where water temperatures were cooler, turbidity was lower, and gravel substrate size was suitable for spawning and egg incubation (McEwan 2001). Steelhead are known to migrate upstream into higher gradient and elevation reaches of the rivers and streams than fall-run Chinook salmon, which predominantly spawn at lower elevations within the valley floor. The majority of historical adult staging/holding and spawning habitat for Central Valley steelhead is no longer accessible to upstream migrating steelhead or has been eliminated or degraded by man-made structures (e.g., dams and weirs) associated with water storage and conveyance, diversions, flood control, municipal, industrial, agricultural, and hydropower purposes (see Figure A-1a) (McEwan and Jackson 1996, McEwan 2001, USBR 2004, Lindley et al. 2006, NMFS 2007). Due to construction of these impediments and barriers to upstream passage, steelhead are currently limited in their geographic distribution within the Central Valley to lower elevation habitats.

Steelhead in the Central Valley migrate upstream into the mainstem Sacramento River and major tributaries (e.g., American, Feather rivers; Clear, Battle creeks and others), and are also known to occur within tributaries to the San Joaquin River (e.g., Mokelumne, Cosumnes, Stanislaus, Merced, Tuolumne rivers), where they spawn and rear. Steelhead do not currently spawn in the mainstem San Joaquin River. The majority of current steelhead spawning habitat exists upstream of the RBDD on the Sacramento River and its tributaries. Although the overall effect of operations of the RBDD on the Central Valley steelhead populations is not well understood, concerns have been expressed regarding the effect of gate operations on upstream and downstream migration by steelhead. Additional concerns include the potential for increased vulnerability of juvenile steelhead to predation by Sacramento pikeminnow, striped bass, and other predators that pass through the RBDD gates or fish ladder.

Reduced flows from dams and upstream water diversions can lower attraction cues for adult spawners, causing straying and delays in spawning or the inability to spawn (California Department of Water Resources [DWR] 2005). Adult steelhead migration delays can reduce fecundity and egg viability and increase susceptibility to disease and harvest.

Reduced rearing and out-migration habitat. Juvenile steelhead prefer to utilize natural stream banks, floodplains, marshes, and shallow water habitats for rearing during out-migration. Modification of natural flow regimes from upstream reservoir operations has resulted in dampening of the hydrograph in most Central Valley rivers, reducing the extent and duration of inundation of floodplains and other flow-dependent habitat used by migrating juvenile steelhead (DWR 2005, 70 FR 52488). Changes in river hydrology that have impacted floodplain inundation may have impacted areas thought to provide significant growth benefits to rearing fish (Sommer et al. 2001). Reductions in flow rates have also resulted in increased water temperature and residence time, and reductions in dissolved oxygen levels in localized areas of the Delta (e.g., Stockton Deep Water Ship Channel) which impact the quality of rearing and migration habitat. Reduced dissolved oxygen levels in the lower San Joaquin River during late summer and early fall have been identified as a barrier and/or impediment to migration for some salmonids (Regional Water Resources Control Board 2003), including Central Valley steelhead (Jassby and Van Nieuwenhuyse 2005). .Much of the Delta has been leveed, channelized, and fortified with riprap for flood protection, reducing and degrading the quality and availability of natural habitat for use by steelhead during migration (McEwan 2001). Furthermore, impacts to the quality, quantity, and availability of suitable habitat is likely to reduce fitness and increase susceptibility to entrainment, disease, exposure to contaminants, and predation.

Predation by non-native species. In general, the effect of non-native predation on the Central Valley steelhead distinct population segment (DPS) is unknown. However, non-native predation is likely an important threat to Central Valley steelhead in areas with high densities of non-native fish (e.g., small and large mouth bass, striped bass, and catfish) are thought to prey on out-migrating juvenile steelhead. Predation risk may covary with increased temperatures. Metabolic rates of non-native, predatory fish increase with increasing water temperatures based on bioenergetic studies (Loboschefsky et al. 2009, Miranda et al. 2010). Upstream gravel pits and flooded ponds such as those that occur on the San Joaquin River and its tributaries, attract non-native predators because of their depth and lack of cover for juvenile steelhead (DWR 2005). Non-native aquatic vegetation, such as Brazilian waterweed and water hyacinth, provide suitable habitat for non-native predators (Brown and Michniuk 2007). The low spatial complexity of channelized waterways (e.g., riprap-lined levee that provide virtually no cover protection from predators) and general low habitat diversity elsewhere in the Delta reduces refuge cover and protection of steelhead from predators (Raleigh et al. 1984, Missildine et al. 2001, 70 FR 52488). A major concern is the potential invasion of the Delta by the highly predatory northern pike. The pike, recently present in Lake Davis on the Feather River, is currently the target of a major eradication effort (California Department of Fish and Game [DFG] 2007a). If eradication fails and pike were to escape downstream to the Delta, they would likely be present in areas frequently inhabited by Central Valley steelhead.

Predation by native species such as the Sacramento pikeminnow in the Sacramento River at locations such as the RBDD has also been identified as a potentially significant source of mortality on juvenile steelhead.

Harvest. Steelhead have been, and continue to be, an important recreational fishery within inland rivers throughout the Central Valley. Although there are no commercial fisheries for steelhead, inland steelhead fisheries include tribal and recreational fisheries. In the Central Valley, recreational fishing for steelhead of hatchery origin is popular, but harvest is restricted to only visibly marked fish of hatchery origin (adipose fin clipped). Unmarked steelhead (adipose fin intact) must be released, reducing the take of naturally spawned wild fish. The impacts of illegal harvest occurring in the Delta and tributary rivers is thought to be relatively minor for Chinook salmon and steelhead. The effects of recreational fishing and the unknown level of illegal harvest on the abundance and population dynamics of wild Central Valley steelhead have not been quantified.

Reduced genetic diversity/integrity. Artificial propagation programs for steelhead in Central Valley hatcheries present multiple threats to the wild steelhead population, including mortality of natural steelhead in fisheries targeting hatchery origin steelhead, competition for prey and habitat, predation by hatchery origin fish on younger natural fish, disease transmission, and impediments to fish passage imposed by hatchery facilities. It is now recognized that Central Valley hatcheries are a significant and persistent threat to wild Chinook salmon and steelhead populations and fisheries (NMFS 2009a). One major concern with hatchery operations is the genetic introgression by hatchery origin fish that spawn naturally and interbreed with local natural populations (USFWS 2001, USBR 2004, Goodman 2005). Such introgression introduces maladaptive genetic changes to the wild steelhead stocks (McEwan and Jackson 1996, Myers et al. 2004). Impacts to fitness in Chinook salmon have been detected due to hatchery operations (Araki et al. 2007). Taking eggs and sperm from a large pool of individuals is one method for ameliorating genetic introgression, but artificial selection for traits that assure individual success in a hatchery setting (e.g., rapid growth and tolerance to crowding) are unavoidable (USBR 2004).

Entrainment. Juvenile steelhead migrating downstream through the Delta are vulnerable to entrainment and salvage at the SWP and CVP export facilities, primarily between March and May (see Table A-1a). There are also multiple factors that can influence the vulnerability of juvenile steelhead to entrainment by State Water Project (SWP) and Central Valley Project (CVP) export facilities, including the geographic distribution of steelhead within the Delta and hydrodynamic factors such as reverse flows in Old and Middle rivers, which are a function of export operations relative to San Joaquin River inflows, and southward flows of Sacramento River water towards pumps through an open Delta Cross Channel and Georgiana Slough. SWC and CVP exports have been shown to affect the tidal hydrodynamics (e.g., water current velocities and direction). The magnitude of these hydrodynamic effects varies in response to a variety of factors including tidal stage and magnitude of ebb and flood tides, the rate of SWP and CVP exports, operation of the Clifton Court Forebay (CCF) radial gate opening, and inflow from upstream tributaries. Steelhead respond behaviorally to hydraulic cues (e.g., water currents) during both upstream adult and downstream juvenile migration through the Delta. Changes in these hydraulic cues as a result of SWP and/or CVP export operations during the period when steelhead are migrating through Delta channels may contribute to attraction to false migration pathways, delays in migration, or increased movement of migrating steelhead toward the export facilities where there is an increase in the risk fish will be entrained into the salvage facilities. DWR and U.S. Bureau of Reclamation (USBR) (1999) found significant relationships between total monthly exports in January through May and monthly steelhead salvage at SWP and CVP facilities, suggesting the risk of steelhead entrainment is related, in part, to export rates. During the past several years, additional investigations have been designed using radio or acoustically-tagged juvenile and adult (post spawning adults) steelhead to monitor their migration behavior through the Delta channels and to assess the effects of changes in hydraulic cues and SWP and CVP export operations on migration (Holbrook et al. 2009, Perry et al. 2010, San Joaquin River Group Authority 2010). These studies are ongoing. Studies have also been recently conducted to assess the potential losses of juvenile steelhead, primarily as a result of predation by adult striped bass, during passage through CCF (Clark et al. 2008). Results of these studies have estimated that pre-screen losses of juvenile steelhead within CCFare greater than 80 percent.

In addition to SWP and CVP export facilities, there are over 2,200 small water diversions within the Delta, of which the majority are unscreened (Herren and Kawasaki 2001). The risk of entrainment is a function of the size of juvenile fish and the slot opening of the screen mesh (Tomljanovich et al. 1978, Schneeberger and Jude 1981, Zeitoun et al. 1981, Weisberg et al. 1987, C. Hanson unpubl. data). Although entrainment/salvage of steelhead at the SWP and CVP export facilities is well documented, it is unclear how many juvenile steelhead are entrained at other unscreened Delta diversions. Because steelhead are moderately large (greater than 200 mm fork length) and relatively strong swimmers when out-migrating, the effects of small in-Delta agricultural water diversions are thought to be lower than those of other Central Valley salmonids. In addition, many of the juvenile steelhead migrate downstream through the Delta during the late winter or early spring before many of the agricultural irrigation diversions are operating. Power plants within the Plan Area have the ability to impinge juvenile steelhead on the existing intake screens. However, use of cooling water is currently low with the retirement of older units. Furthermore, newer units are equipped with a closed cycle cooling system that virtually eliminates the risk of impingement of juvenile steelhead.

Exposure to toxins. Toxic chemicals are widespread throughout the Delta and may occur on a more localized scale in response to episodic events (e.g., stormwater runoff, point source discharges, etc.). These toxic substances include mercury, selenium, copper, pyrethroids, and endocrine disruptors with the potential to impact fish health and condition, and negatively impact steelhead distribution and abundance directly or indirectly. Some loads of toxics, such as selenium, are much higher in the San Joaquin River than the Sacramento River because they are naturally occurring in the alluvial soils and have been leached by irrigation water and concentrated by evapotranspiration (Nichols et al. 1986). This may indicate that the potential effects of chronic exposure could be greater for steelhead of San Joaquin River origin. Additionally, agricultural return flows that may contain toxic chemicals are widely distributed throughout the Sacramento and San Joaquin rivers and the Delta, although dilution flows from the rivers may reduce chemical concentrations to sublethal levels. Sublethal concentrations of toxics may interact with other stressors on salmonids, such as increasing their vulnerability to predation or disease (Werner 2007). For example, Clifford et al. (2005) found in a laboratory setting that juvenile fall-run Chinook salmon exposed to sublethal levels of a common pyrethroid, esfenvalerate, were more susceptible to infectious hematopoietic necrosis virus than those not exposed to esfenvalerate. Although not tested on steelhead, a similar response is likely, however juvenile steelhead generally migrate through the Delta in a comparatively shorter time period to Chinook salmon. The short duration may decrease juvenile steelhead exposure and susceptibility to toxic substances in the Delta. Adult migrating steelhead may be less affected by toxins in the Delta because they are not feeding, and thus not bioaccumulating toxic exposure, and they are moving rapidly through the system.

Iron Mountain Mine, located adjacent to the upper Sacramento River, has been a source of trace elements that are known to adversely affect aquatic organisms (Upper Sacramento River Fisheries and Riparian Habitat Advisory Council 1989). Storage limitations and limited availability of dilution flows have caused downstream copper and zinc levels to exceed salmonid tolerances and resulted in documented fish kills in the 1960s and 1970s (USBR 2004). The U.S. Environmental Protection Agency's Iron Mountain Mine remediation program has removed toxic metals in acidic mine drainage from the Spring Creek watershed with a state-of-the-art lime neutralization plant. Contaminant loading into the Sacramento River from Iron Mountain Mine has shown measurable reductions since the early 1990s.

Increased water temperature. Water temperature is among the physical factors that affect quality of habitat for salmonid adult holding, spawning and egg incubation, juvenile rearing, and migration. Adverse sublethal and lethal effects can result from exposure to elevated water temperatures at sensitive lifestages, such as during incubation or rearing. Water temperature criteria for various lifestages of salmonids in the Central Valley have been developed by NMFS (2009). The tolerance of steelhead water temperatures depends on life stage, acclimation history, food availability, duration of exposure, health of the individual, and other factors such as predator avoidance (Myrick and Cech 2004, USBR 2004). Higher water temperatures can lead to physiological stress, reduced growth rate, reduced spawning success, and increased mortality of steelhead (Myrick and Cech 2001). Temperature can also indirectly influence disease incidence and predation (Waples et al. 2008). Exposure to seasonally elevated water temperatures may occur as a result of reductions in flow as a result of upstream reservoir operations, reductions in riparian vegetation, channel shading, local climate and solar radiation. The installation of the Shasta Temperature Control Device in 1998, in combination with reservoir management to maintain the cold water pool, has reduced many of the temperature issues on the Sacramento River. During dry years, however, the release of cold water from Shasta Dam is still limited. As the river flows further downstream, particularly during the warm spring, summer, and early fall months, water temperatures continue to increase until they reach thermal equilibrium with atmospheric conditions. As a result of the longitudinal gradient of seasonal water temperatures, the coldest water and, therefore, the best areas for steelhead spawning and rearing are typically located immediately downstream of the dam.

Increased temperature can also arise from a reduction in shade over rivers by tree removal (Watanabe et al. 2005). Because river water is typically in thermal equilibrium with atmospheric conditions by the time it enters the Delta (C. Hanson, unpubl. data), this issue is caused primarily from actions upstream of the Delta. As a result of the relatively wide channels that occur within the Delta, the effects of additional riparian vegetation on reducing water temperatures.

A1.6 Relevant Conservation Efforts

Because steelhead biology is similar to that of Chinook salmon, few conservation actions are specific to steelhead. Efforts by DFG to restore Central Valley steelhead are described in the "Steelhead Restoration and Management Plan for California" (McEwan and Jackson 1996). Measures to protect steelhead throughout the State of California have been in place since 1998 and 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 rearing juveniles and smolts. The Central Valley Steelhead Project

Work Team, an interagency technical working group led by DFG, drafted a proposal to develop a comprehensive steelhead monitoring plan that was selected by the CALFED Ecosystem Restoration Program (ERP) Implementing Agency Managers for directed action funding. Long-term funding for implementation of the monitoring plan still needs to be secured.

Biological opinions for SWP and CVP operations (e.g., NMFS 2009b) and other federal projects involving irrigation and water diversion and fish passage, for example, have improved or minimized adverse impacts on steelhead in the Central Valley. In 1992, an amendment to the authority of the CVP through the Central Valley Project Improvement Act (CVPIA) was enacted to give protection of fish and wildlife equal priority with other Central Valley Project objectives. From this Act arose several programs that have benefited listed salmonids. The USFWS's Anadromous Fish Restoration Program (AFRP) is engaged in monitoring, education, and restoration projects designed to contribute toward doubling the natural populations of select anadromous fish species residing in the Central Valley. Restoration projects funded through the AFRP include fish passage, fish screening, riparian easement and land acquisition, development of watershed planning groups, instream and riparian habitat improvement, and gravel replenishment. The AFRP combines federal funding with State and private funds to prioritize and construct fish screens on major water diversions mainly in the upper Sacramento River. The goal of the Water Acquisition Program is to acquire water supplies to meet the habitat restoration and enhancement goals of the CVPIA, and to improve the ability of the U.S. Department of the Interior to meet regulatory water quality requirements. Water has been used to improve fish habitat for Central Valley steelhead by maintaining or increasing instream flows on Butte and Mill creeks and the San Joaquin River at critical times. Additionally, salmonid entrainment at the SWP and CVP export facilities is decreased through reducing seasonal diversion rates during periods when protected fish species are vulnerable to export related losses.

Two programs included under CALFED, the Ecosystem Restoration Program (ERP) and the Environmental Water Account, were created to improve conditions for fish, including steelhead, in the Central Valley. Restoration actions implemented by the ERP include the installation of fish screens, modification of barriers to improve fish passage, habitat acquisition, and instream habitat restoration. The majority of these actions address key factors affecting listed salmonids and emphasis has been placed in tributary drainages with high potential for Central Valley steelhead and spring-run Chinook salmon production. Additional ongoing actions include efforts to enhance fishery monitoring and directly support salmonid production through hatchery releases. The Environmental Water Account has been under scrutiny recently as to its success in meeting its original goal.

A major CALFED ERP action currently underway is the Battle Creek Salmon and Steelhead Restoration Project. The project will restore 77 km (48 miles) of habitat in Battle Creek to support steelhead and Chinook salmon spawning and juvenile rearing at a cost of over $90 million. The project includes removal of five small hydropower diversion dams, construction of new fish screens and ladders on another three dams, and construction of several hydropower facility modifications to ensure the continued hydropower operations. It is thought that this restoration effort is the largest cold water restoration project to date in North America.

The Delta Regional Ecosystem Restoration Implementation Plan (DRERIP) was formed to guide the implementation of CALFED Ecosystem Restoration Plan elements within the Delta (DFG 2007b). The DRERIP team has created a suite of ecosystem and species conceptual models, including steelhead, that document existing scientific knowledge of Delta ecosystems. The DRERIP Team has used these conceptual models to assess the suitability of actions proposed in the Ecosystem Restoration Plan for implementation. DRERIP conceptual models were used in the analysis of proposed BDCP conservation measures.

Oroville Dam Federal Energy Regulatory Commission (FERC) relicensing efforts on the Feather River have considered instream flows and temperature management for steelhead spawning and juvenile rearing downstream of the dam.

Multiple fish passage projects have been recently implemented for steelhead and other salmonids in the Sacramento and San Joaquin Watersheds. Multiple large diversions on the Sacramento River (e.g., Glenn-Colusa Irrigation District, RD108, RD1004, Sutter Mutual, and Wilkins Slough) have been equipped with positive barrier fish screens to reduce entrainment of steelhead and other salmonids. The Woodbridge Irrigation District Dam on the Mokelumne River was designed to improve upstream and downstream passage of steelhead and other salmonids by installing fish screens and fish ladders at the dam.

Mitigation under the Delta Fish Agreement has increased the number of wardens enforcing harvest regulations for steelhead and other fish in the Bay-Delta and upstream tributaries by creating the Delta Bay Enhanced Enforcement Program (DBEEP). Initiated in 1994, DBEEP currently consists of nine wardens and a supervisor.

Many smaller tributaries to the Sacramento and San Joaquin rivers have local watershed conservancies with master plans to contribute to conservation and recovery of steelhead and other salmonids.

A1.7 RECOVERY GOALS

The Public Draft Recovery Plan for Central Valley salmonids, including steelhead, was released by NMFS on October 19, 2009. Although not final, the overarching goal in the public draft is the removal of, among other listed salmonids, the Central Valley steelhead DPS from the Federal List of Endangered and Threatened Wildlife (NMFS 2009a). Several objectives and related criteria represent the components of the recovery goal, including the establishment of at least two viable populations within each historical diversity group, as well as other measurable biological criteria.

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APPENDIX A2. SACRAMENTO RIVER WINTER-RUN CHINOOK SALMON (*ONCORHYNCHUS TSHAWYTSCHA*)

A2.1 LEGAL STATUS

The Sacramento River winter-run Chinook salmon Evolutionarily Significant Unit (ESU) was originally listed as a threatened species in August 1989, under emergency provisions of the Federal Endangered Species Act (ESA), and was formally listed as threatened in November 1990 (55 FR 46515). The ESU consists of only one population confined to the upper Sacramento River in California's Central Valley. The ESU was reclassified as endangered under the federal ESA on January 4, 1994 (59 FR 440), due to increased variability of run sizes, expected weak returns as a result of two small year classes in 1991 and 1993, and a 99 percent decline between 1966 and 1991. The Sacramento River winter-run Chinook salmon ESU includes all naturally spawned winter-run Chinook salmon in the Sacramento River and its tributaries as well as two artificial propagation programs: winter-run Chinook salmon produced from the Livingston Stone National Fish Hatchery and released as juveniles into the Sacramento River and winter-run Chinook salmon held in a captive broodstock program maintained at Livingston Stone National Fish Hatchery (70 FR 37160, June 28, 2005) (see Figure A-2a).

The National Marine Fisheries Service (NMFS) reaffirmed the listing of Sacramento River winter-run Chinook salmon as endangered on June 28, 2005 (70 FR 37160) and included the Livingston Stone National Fish Hatchery population within the listed population.

Winter-run Chinook salmon was listed as endangered under the California Endangered Species Act on September 22, 1989.

A2.2 SPECIES DISTRIBUTION AND STATUS

A2.2.1 Range and Status

The distribution of winter-run Chinook salmon spawning and rearing was limited historically to the upper Sacramento River and tributaries, where cool spring-fed streams supported successful adult holding, spawning, egg incubation, and juvenile rearing (Slater 1963, Yoshiyama et al. 1998). The headwaters of the McCloud, Pit, and Little Sacramento Rivers, Hat and Battle creeks , provided clean, loose gravel, cold, well-oxygenated water, and year-round flow in riffle habitats for spawning and incubation (see Figure A-2a). These areas also provided the cold, productive waters necessary for egg and fry survival, and juvenile rearing over summer. Construction of Shasta Dam in 1943 and Keswick Dam in 1950 blocked access to all of these upstream waters except Battle Creek, which is blocked by a weir at the Coleman National Fish Hatchery and other small hydroelectric facilities (Moyle et al. 1989, NMFS 1997).

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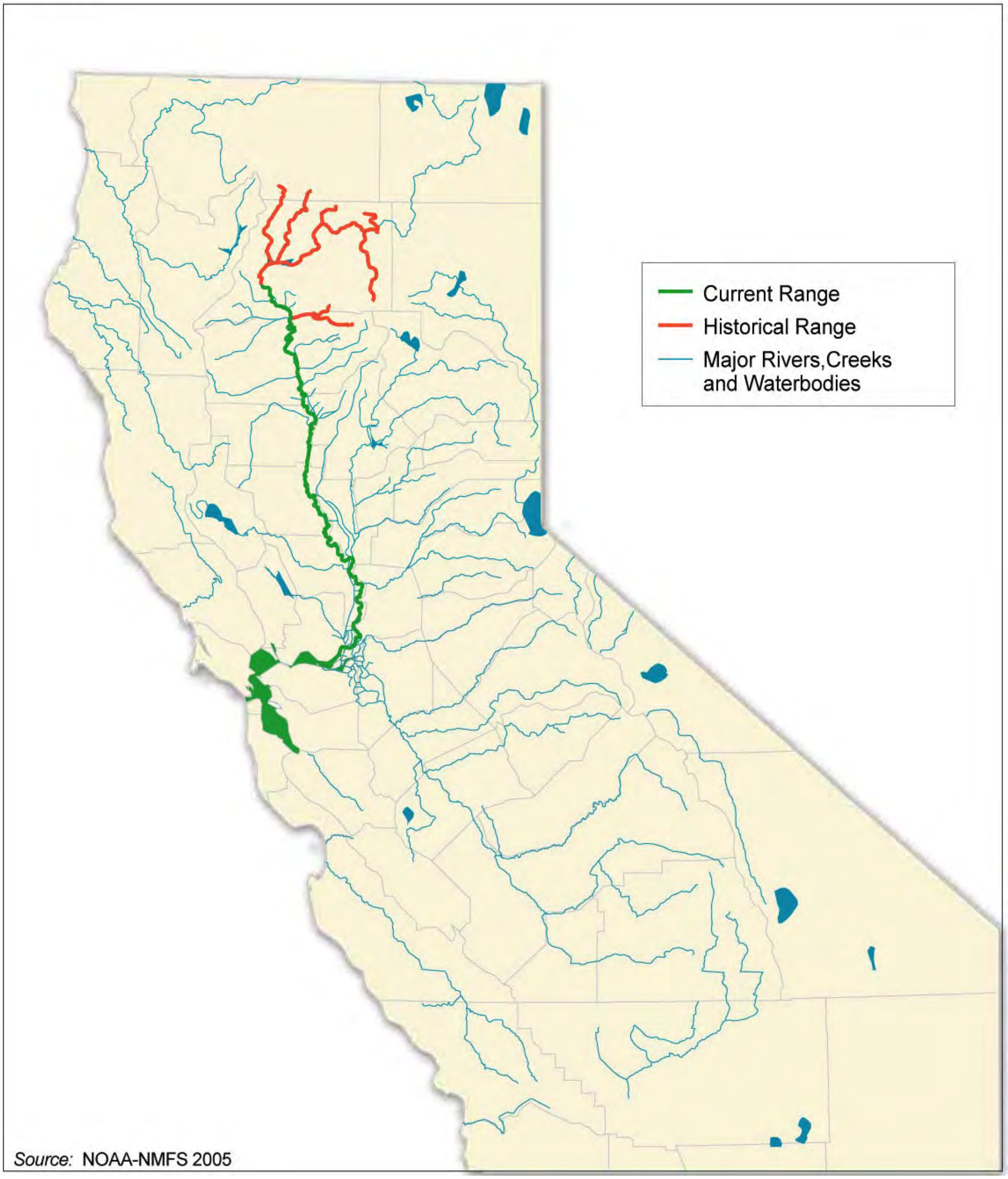

Figure A-2a. Sacramento River Winter-Run Chinook Salmon Inland Range in California

Primary spawning and rearing habitats for winter-run Chinook salmon are now confined to the cold water areas between Keswick Dam and Red Bluff Diversion Dam (RBDD)(see Figure A-2a). The lower reaches of the Sacramento River, Delta, and San Francisco Bay serve as migration corridors for the upstream migration of adult and downstream migration of juvenile winter-run Chinook salmon.

Estimates of the Sacramento River winter-run Chinook salmon population (including both male and female salmon) reached nearly 100,000 fish in the 1960s before declining to under 200 fish in the 1990s (Good et al. 2005). Abundance of returning adult spawners generally increased between the mid-1990s and 2006 (see Figure A-2b). However, recent population estimates of winter-run Chinook salmon spawning upstream of the RBDD have dropped off since the 2006 peak (DFG 2010).

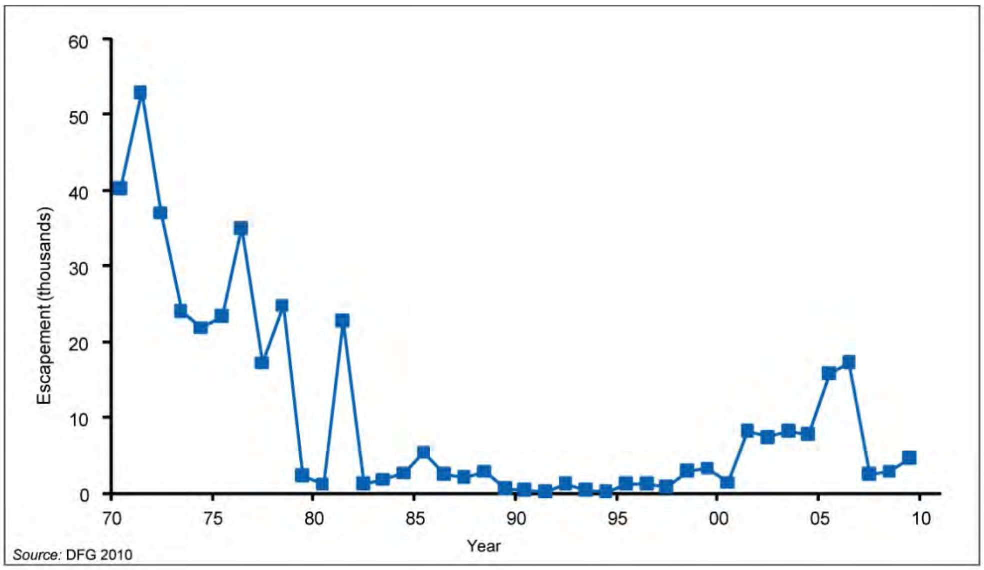

Figure A-2b. Estimate Historical Spawner Escapement of Sacramento River Winter-Run Chinook Salmon (1970-2009)

Two methods are currently used to estimate the juvenile production of Sacramento River winter-run Chinook salmon: the juvenile production index method (using rotary screw traps) and the juvenile production estimate method (using carcass surveys). Average juvenile population of Sacramento River winter-run Chinook salmon inhabiting the upper Sacramento River at the RBDD is 4,230,378 juveniles per year using the juvenile production index method between 1995 and 2007 (excluding 2000 and 2001 when rotary screw trapping was not conducted) (Poytress and Carillo 2010). Using the juvenile production estimate method, average production is estimated to be 5,034,921 juveniles exiting the upper Sacramento River at the RBDD in all years between 1996 and 2007 (Poytress and Carillo 2010).

Although the abundance of the Sacramento River winter-run Chinook salmon population has on average been growing since the 1990s (despite recent declines since 2007), there is only one population and it depends heavily on cold-water releases from Shasta Dam (Good et al. 2005).

Lindley et al. (2007) considers the Sacramento River winter-run Chinook salmon population at a moderate risk of extinction primarily due to the risks associated with only one existing population. The viability of an ESU that is represented by a single population is vulnerable to changes in the environment through a lack of spatial geographic diversity and genetic diversity that result from having only one population. A single catastrophic event with effects persisting for four or more years could extirpate the entire Sacramento River winter-run Chinook salmon ESU, which puts the population at a high risk of extinction over the long-term (Lindley et al. 2007). Such potential catastrophes include volcanic eruption of Mt. Lassen; prolonged drought, which depletes the cold water pool in Lake Shasta or some related failure to manage cold water storage; a spill of toxic materials with effects that persist for four years; regional declines in upwelling and productivity of near-shore coastal marine waters resulting in reduced food supplies for juvenile and sub-adult salmon, reduced growth, and/or increased mortality; or a disease outbreak. Another vulnerability to an ESU that is represented by a single population is the limitation in life history and genetic diversity that would otherwise increase the ability of individuals in the population to withstand environmental variation.

Although NMFS recently proposed that this ESU be downgraded from endangered to threatened status, NMFS decided in its Final Listing Determination (June 28, 2005, 70 FR 37160) to continue to list the Sacramento River winter-run Chinook salmon ESU as endangered because the population remains below the draft recovery goals established for the run (NMFS 1997) and the naturally-spawned component of the ESU is dependent on one extant population in the Sacramento River.

A2.2.2 Distribution and Status in the Plan Area

The entire population of the Sacramento River winter-run Chinook salmon must pass through the Plan Area as migrating adults and emigrating juveniles. Because winter-run Chinook salmon use only the Sacramento River system for spawning, it has been hypothesized that adults are attracted to, and migrate upstream primarily along, the western edge of the Delta through the Sacramento River corridor. Because juvenile winter-run salmon have been collected at various locations within the Delta (including the State Water Project [SWP] and the Central Valley Project [CVP] south Delta export facilities), it has been hypothesized that juveniles likely use a wider range of the Delta for migration and rearing than adults. Studies using acoustically tagged juvenile and adult Chinook salmon are ongoing to further investigate the migration routes, migration rates, reach-specific mortality rates, and the effects of hydrologic conditions (including the effects of SWP and CVP export operations) on salmon migration through the Delta (Lindley et al. 2008, MacFarlane et al. 2008a, Michel et al. 2008, Perry et al. 2008). Juvenile winter-run Chinook salmon likely inhabit Suisun Marsh for rearing and may inhabit the Yolo Bypass when flooded, although use of these two areas is not well understood.

Results of fishery monitoring using a combination of adult counts at the RBDD fish ladder and carcass surveys have been used to estimate annual adult escapement of winter-run Chinook salmon on the mainstem Sacramento River. The estimated annual adult escapement over the period from 1970 through 2010 is shown in Figure A-2b. During the late 1960s and throughout the 1970s, winter-run Chinook salmon abundance declined significantly from a high of approximately 120,000 adults to several thousand adults. Population abundance remained low through the mid-1990s, with adult abundance in some years less than 500 fish. Beginning in the mid-1990s and continuing to date, adult escapement has shown a trend of increasing abundance, approaching 20,000 fish in 2006. A variety of factors have been identified that are thought to have contributed to the recent increasing trend in adult abundance. These factors, include but are not limited to, improved water temperatures and temperature management in the Shasta Reservoir and the mainstem river downstream of Keswick Dam, improvements in the operations of the RBDD (keeping holding gates open for a longer period), favorable hydrological and ocean rearing conditions, habitat enhancements, reductions in loading of toxic chemicals, improved fish screens on major water diversions, and changes in ocean commercial and recreational angling to reduce harvest mortality.

Adult winter-run Chinook salmon escapement to the Sacramento River declined substantially in 2007, with an estimated 2,542 adults retuning to spawn (see Figure A-2b). As discussed below, it has been hypothesized that the substantial decline in adult winter-run Chinook salmon escapement was the result of reduced productivity of near-shore coastal waters and reduced prey availability resulting in poor juvenile salmon growth and high mortality during the juvenile ocean rearing phase (MacFarlane et al. 2008b). A similar substantial decline in abundance of retuning fall-run Chinook salmon (and other salmon populations in California) was observed in 2007. Adult escapement remained low during 2008 and 2009. In response to the low numbers of adult Chinook salmon returning to the Central Valley, commercial and recreational fishing for salmon has been curtailed since 2007.

A2.3 HABITAT REQUIREMENTS AND SPECIAL CONSIDERATIONS

Critical habitat for the winter-run Chinook ESU was designated under the ESA on June 16, 1993 (58 FR 33212). Designated critical habitat includes: the Sacramento River from Keswick Dam (RM 302) to Chipps Island (RM 0) at the westward margin of the Sacramento-San Joaquin Delta, all waters from Chipps Island westward to Carquinez Bridge, including Honker, Grizzly, and Suisun bays, and Carquinez Strait, all waters of San Pablo Bay westward of San Pablo Bay westward of the Carquinez Bridge, and all waters of San Francisco Bay (north of the San Francisco/Oakland Bay Bridge) from San Pablo Bay to the Golden Gate Bridge (59 FR 440, January 4, 1994) (see Figure A-2c). In the Sacramento River, critical habitat includes the river water column, river bottom, and adjacent riparian zone used by fry and juveniles for rearing. In the areas westward of Chipps Island, critical habitat includes the estuarine water column and essential foraging habitat and food resources used by Sacramento River winter-run Chinook salmon as part of their juvenile emigration or adult spawning migration.

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Figure A-2c. Sacramento River Winter-Run Chinook Salmon Inland Designated Critical Habitat in California

Habitat of Sacramento River winter-run Chinook salmon is also protected under the Magnuson-Stevens Fishery Conservation and Management Act as Essential Fish Habitat (EFH). Those waters and substrate necessary to support Sacramento River winter-run Chinook salmon spawning, breeding, feeding, or growth are included as EFH (see Figure A-2d). Critical Habitat and EFH are managed differently from a regulatory standpoint, but are biologically equivalent with regard to conservation.

The PCEs considered essential for the conservation of Sacramento River winter-run Chinook salmon are: (1) freshwater spawning sites, (2) freshwater rearing sites, (3) freshwater migration corridors, (4) estuarine areas, (5) nearshore marine areas, and (6) offshore marine areas.

A2.3.1 Spawning Habitat

Spawning habitat for Sacramento River winter-run Chinook salmon is restricted to the Sacramento River primarily between RBDD and Keswick Dam.

Spawning sites for Sacramento River winter-run Chinook salmon include those stream reaches with water movement, velocity, depth, temperature, and substrate composition that support spawning, egg incubation, and larval development. Water velocity and substrate conditions are more critical to the viability of spawning habitat than depth. Incubating eggs and embryos buried in gravel require sufficient water flow through the gravel to supply oxygen and removal of metabolic wastes (Resources Agency et al. 1998). Spawning occurs in gravel substrate in relatively fast-moving, moderately shallow riffles or along banks with relatively high water velocities. The gravel must be clean and loose, yet stable for the duration of egg incubation and the larval development.

Substrate composition has other key implications to spawning success. The embryos and alevins (newly hatched fish with the yolk sac still attached) require adequate water movement through the substrate; however, this movement can be inhibited by the accumulation of fines and sand. Generally, the redd should contain less than 5 percent fines (Resources Agency et al. 1998).

Water velocity in Chinook salmon spawning areas typically ranges from 1.0 to 3.5 feet per second and optimum velocity is 1.5 feet per second (Resources Agency et al. 1998). Spawning occurs at depths between 1 to 5 feet with a maximum depth observed of 20 feet. A depth of less than 6 inches can be restrictive to Chinook salmon movement.

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Figure A-2d. Sacramento River Winter-Run Chinook Salmon Inland Essential Fish Habitat in California

A2.3.2 Freshwater Rearing Habitat

Freshwater salmon rearing sites are those with sufficient water quantity and floodplain connectivity to form and maintain physical habitat conditions that support juvenile growth and mobility; suitable water quality; availability of suitable forage species that support juvenile salmon growth and development; and cover such as shade, submerged and overhanging large wood, log jams, beaver dams, aquatic vegetation, large rocks and boulders, side channels, andundercut banks. Both spawning areas and migratory corridors also function as rearing habitat for juveniles, which feed and grow before and during their outmigration. Non-natal, intermittent tributaries also may be used for juvenile rearing. Rearing habitat quality is strongly affected by habitat diversity and complexity, food supply, and fish and avian predators. Some of these more complex and productive habitats with floodplains are still found in the system (e.g., the lower Cosumnes River, Sacramento River reaches with set-back levees [i.e., primarily located upstream of the City of Colusa]). The channeled, leveed, and rip-rapped river reaches and sloughs common along the Sacramento River and throughout the Delta, however, typically have low habitat complexity, low abundance of food organisms, and offer little protection from predation by fish and birds. Freshwater rearing habitat has a high conservation value as the juvenile life stage of salmonids is dependent on the function of this habitat for successful survival and recruitment into the adult population.

A2.3.3 Freshwater Migration Corridors

Freshwater migration corridors for winter-run Chinook salmon, including river channels, channels through the Delta, and the Bay-Delta estuary, support mobility, survival, and food supplies for juveniles and adults. Migration corridors should be free from obstructions (passage barriers and impediments to migration), provide favorable water quantity (instream flows) and quality conditions (seasonal water temperatures), and contain natural cover such as submerged and overhanging large wood, native aquatic vegetation, large woody debris, rocks and boulders, side channels, and undercut banks. Migratory corridors for winter-run Chinook salmon are located downstream of the spawning areas and include the lower Sacramento River, the Delta, and the San Francisco Bay complex extending to coastal marine waters. These corridors allow the upstream passage of adults and the downstream emigration of juvenile salmon. Migratory corridor conditions are strongly affected by the presence of passage barriers, which can include dams, unscreened or poorly screened diversions, and degraded water quality. For freshwater migration corridors to function properly, they must provide adequate passage, provide suitable migration cues, limit false attraction, provide low vulnerability to predation, and not contain impediments and delays in both upstream and downstream migration.

Results of mark-recapture studies conducted using juvenile Chinook salmon (typically hatchery-reared late fall-run Chinook salmon that are considered to be representative of juvenile winter-run salmon) released into the Sacramento River have shown high mortality during passage downstream through the rivers and Delta (Brandes and McLain 2001, Newman and Rice 2002, Hanson 2008). Mortality is typically greater in years when spring flows are reduced and water temperatures are increased. Results of survival studies have shown that closing the Delta Cross Channel gates to reduce the movement of juvenile salmon into the Delta, contributes to improved survival of emigrating juvenile Chinook salmon (Brandes and McLain 2001, Manly 2004, Low and White undated). Observations at the SWP and CVP fish salvage facilities have shown that very few of the marked salmon (typically less than 1 percent [Hanson 2008]) are entrained and salvaged at the export facilities. Results of estimating incidental take of juvenile winter-run Chinook salmon at the SWP and CVP fish salvage facilities based on comparison of the juvenile production estimates for winter-run emigrating from the upper Sacramento River rearing areas (e.g., estimated based on results of spawning carcass surveys and environmental conditions and/or fishery monitoring at RBDD) show a similar low magnitude to direct losses of juvenile winter-run Chinook salmon at the fish salvage facilities. Although the factors contributing to the high juvenile mortality have not been quantified, results of acoustic tagging experiments and anecdotal observations suggest that exposure to adverse water quality conditions leading to mortality (e.g., elevated water temperatures, potentially toxic chemicals) and vulnerability to predation mortality are two of the factors contributing to the high juvenile mortality observed in the Sacramento River and Delta.

A2.3.4 Estuarine Areas

Estuarine migration and juvenile rearing habitats should be free of obstructions (i.e., dams and other barriers) and provide suitable water quality, water quantity (river and tidal flows), and salinity conditions to support juvenile and adult physiological transitions between fresh and salt water. Natural cover, such as submerged and overhanging large wood, native aquatic vegetation, and side channels, provide juvenile foraging habitat and cover from predators. Tidal wetlands and seasonally inundated floodplains have also been identified as high value foraging and rearing habitats for juvenile salmon migrating downstream through the estuary. Estuarine areas contain a high conservation value because they function to support juvenile Chinook salmon growth, smolting, avoidance of predators, and provide a transition to the ocean environment.

A2.3.5 Ocean Habitats

Although ocean habitats are not part of the critical habitat listings for Sacramento River winter-run Chinook salmon, biologically productive coastal waters are an important habitat component for the species. Juvenile Chinook salmon inhabit near-shore coastal marine waters for a period of typically two to four years before adults return to Central Valley rivers to spawn. During their marine residence, Chinook salmon forage on krill, squid, and other marine invertebrates as well as a variety of fish such as northern anchovy and Pacific herring. These features are essential for conservation because, without them, juveniles cannot forage and grow to adulthood.

The variation in ocean productivity off the West Coast can be high both within and among years. Changes in ocean currents and upwelling have been identified as significant factors affecting nutrient availability, phytoplankton, and zooplankton production and the availability of other forage species in near-shore surface waters. Ocean conditions during a salmon's ocean residency period can be important, as indicated by the effect of the 1983 El Niño on the size and fecundity of Central Valley fall-run Chinook salmon (Wells et al. 2006). Although the effects of ocean conditions on Chinook salmon growth and survival have not been investigated extensively, recent observations since in 2007 have shown a significant decline in the abundance of adult Chinook salmon and coho salmon returning to California rivers and streams (fall-run adult returns to the Sacramento and San Joaquin rivers were the lowest on record) (Pacific Fishery Management Council 2008) that has been hypothesized to be the result of declines in ocean productivity and associated high mortality rates during the period when these fish were rearing in near-shore coastal waters (MacFarlane et al. 2008b). The importance of changes in ocean conditions on growth, survival, and population abundance of Central Valley Chinook salmon is currently undergoing further investigation.

A2.4 LIFE HISTORY

Chinook salmon exhibit two generalized freshwater life history types (Healey 1991). Stream-type adults enter freshwater months before spawning and juveniles reside in freshwater for a year or more following emergence, whereas ocean-type adults spawn soon after entering freshwater and juveniles migrate to the ocean as fry or parr within their first year. Winter-run Chinook salmon are somewhat anomalous in that they have characteristics of both stream- and ocean-type races (Healey 1991). Adults enter freshwater in winter or early spring, and delay spawning until spring or early summer (stream-type). However, juvenile winter-run Chinook salmon migrate to sea after only 4 to 7 months of river life (ocean-type). Adequate instream flows and cool water temperatures are more critical for the survival of Chinook salmon exhibiting a stream-type life history due to over-summering by adults and/or juveniles.

Sacramento River winter-run Chinook salmon adults enter the Sacramento River basin between December and July; the peak occurring in March (see Table A-2a) (Yoshiyama et al. 1998, Moyle 2002). Spawning occurs from mid-April to mid-August, peaking in May and June, in the Sacramento River reach between Keswick Dam and RBDD (Vogel and Marine 1991). The majority of Sacramento River winter-run Chinook salmon spawners are three years old. Adult winter-run Chinook salmon tend to enter freshwater as sexually immature fish, migrate far upriver, and delay spawning for weeks or months. Pre-spawning activity requires an area of 200 to 650 square feet. The female digs a nest, called a redd, with an average size of 165 square feet, in which she buries her eggs after they are fertilized by the male (Resources Agency et al. 1998).

Table A-2a. Temporal occurrence of (a) Adult and (b) Juvenile Sacramento River Winter-Run Chinook salmon in the Sacramento River and Delta. Darker shades indicate months of greatest relative abundance.

a) Adult

Location	*Jan*	*Feb*	*Mar*	*Apr*	*May*	*Jun*	*Jul*	*Aug*	*Sep*	*Oct*	*Nov*	*Dec*
Sac. River basin[1]	Medium	Medium	High	Medium	Medium	Medium	Medium					Medium
Sac. River[2]	Medium	Medium	Medium	Medium	Medium	Medium					Medium	Medium

b) Juvenile

Location	*Jan*	*Feb*	*Mar*	*Apr*	*May*	*Jun*	*Jul*	*Aug*	*Sep*	*Oct*	*Nov*	*Dec*
Sac. River @ Red Bluff[3]	Low						Low / Medium	Medium	Medium	Medium	Medium	Medium
Sac. River @ Red Bluff[2]	Medium	Medium					Low	High	High	High	Medium	Medium
Sac. River @ Knights L.[4]	Medium	Medium	Medium	Low				Low	Low	Low	High	High
Lower Sac. River (seine)[5]	High	High	Medium	Low / –					Low	Low	Medium	High
West Sac. River (trawl)[5]	Medium	High	High	High	Low / –						Low	Medium
Chipps Island (trawl)[5]	Medium	Medium / High	High	High	Low							Low
Relative Abundance:	= High				= Medium				= Low			

Sources: [1]Yoshiyama *et al.* 1998; Moyle 2002; [2]Myers *et al.* 1998; [3]Martin *et al.* 2001; [4]Snider and Titus 2000; [5]USFWS 2006

Sacramento River winter-run Chinook salmon fry begin to emerge from the gravel in late June to early July and continue through October (Fisher 1994), with emergence generally occurring at night. Fry then seek lower velocity nearshore habitats with riparian vegetation and associated substrates important for providing aquatic and terrestrial invertebrates, predator avoidance, and slower velocities for resting (NMFS 1996). Emigrating juvenile Sacramento River winter-run Chinook salmon pass the RBDD beginning as early as mid-July, typically peaking in September, and can continue through March in dry years (Vogel and Marine 1991, NMFS 1997). From 1995 to 1999, all Sacramento River winter-run Chinook salmon outmigrating as fry passed the RBDD by October, and all outmigrating pre-smolts and smolts passed the RBDD by March (Martin et al. 2001).

Juvenile Sacramento River winter-run Chinook salmon occur in the Delta primarily from November through early May based on data collected from trawls in the Sacramento River at West Sacramento (RM 55; USFWS 2006). The timing of migration varies somewhat due to changes in river flows, dam operations, seasonal water temperatures, and hydrologic conditions (water year type). Winter-run Chinook salmon juveniles remain in the Delta until they reach a fork length of approximately 118 mm and are between five and 10 months of age. It has been hypothesized that changes in habitat conditions within the Delta over the past century have resulted in a reduction in extended juvenile salmon rearing when compared to periods when habitat for juvenile salmon rearing was more suitable. Emigration to the ocean begins as early as November and continues through May (Fisher 1994, Myers et al. 1998). The importance of the Delta in the life history of Sacramento River winter-run Chinook salmon is not well understood.

Data from the Pacific States Marine Fisheries Commission Regional Mark Information System database indicate that Sacramento River winter-run Chinook salmon adults are not as broadly distributed along the Pacific Coast as other Central Valley Chinook salmon runs and concentrate in the region between San Francisco and Monterey. This localized distribution may indicate a unique life history strategy related to the fact that Sacramento River winter-run Chinook salmon also mature at a relatively young age (Myers et al. 1998). Sacramento River winter-run Chinook salmon remain in the ocean environment for two to four years.

A2.5 THREATS AND STRESSORS

The following have been identified as important threats and stressors to winter-run Chinook salmon (without priority).

Reduced staging and spawning habitat. Access to much of the historical upstream spawning habitat for winter-run Chinook salmon (see Figure A-2a) has been eliminated or degraded by man-made structures (e.g., dams and weirs) associated with water storage and conveyance, flood control, and diversions and exports for municipal, industrial, agricultural, and hydropower purposes (Yoshiyama et al. 1998). The construction and operation of Shasta Dam reduced the winter-run Chinook salmon ESU from four independent populations to just one. The remaining available habitat for natural spawners is currently maintained with cool water releases from Shasta and Keswick dams, thereby significantly limiting spatial distribution of this ESU within the reach of the mainstem Sacramento River immediately downstream of the dam.

Upstream diversions and dams have decreased downstream flows and altered seasonal hydrologic patterns, which have been identified as factors resulting in delayed upstream migration by adults and increased mortality of out-migrating juveniles (Yoshiyama et al. 1998, DWR 2005). Dams and reservoir impoundments and associated reductions in peak flows have blocked gravel recruitment and reduced flushing of sediments from existing gravel beds, reducing and degrading natal spawning grounds. Furthermore, reduced flows can lower attraction cues for adult spawners, causing straying and delays in spawning (DWR 2005). Adult salmon migration delays can reduce fecundity and increase susceptibility to disease and harvest (McCullough 1999).

The RBDD, located on the Sacramento River, has been identified as a barrier and impediment to adult winter-run Chinook salmon upstream migration. Although the RBDD is equipped with fish ladders, migration delays occur when the dam gates are closed. Mortality as a result of increased predation by Sacramento pikeminnow on juvenile salmon passing downstream through the fish ladder has also been identified as a factor affecting abundance of salmon produced on the Sacramento River (Hallock 1991). The construction and operation of the RBDD has been identified as one of the primary factors contributing to the decline in winter-run Chinook salmon abundance that lead to listing of the species under the ESA.

Reduced rearing and out-migration habitat. Juvenile winter-run Chinook salmon prefer natural stream banks, floodplains, marshes, and shallow water habitats to utilize as rearing habitat during out-migration. Channel margins throughout the Delta have been leveed, channelized, and fortified with riprap for flood protection and island reclamation, reducing and degrading the quality of natural habitat available for juvenile Chinook salmon rearing (Brandes and McLain 2001). Man-made barriers further reduce and degrade rearing and migration habitat and delay juvenile out-migration. Juvenile out-migration delays can reduce fitness and increase susceptibility to diversion screen impingement, entrainment, disease, and predation. Modification of natural flow regimes from upstream reservoir operations has resulted in dampening and altering the seasonal timing of the hydrograph, reducing the extent and duration of seasonal floodplain inundation and other flow-dependent habitat used by migrating juvenile Chinook salmon (70 FR 52488, Sommer et al. 2001, DWR 2005). Recovery of floodplain habitat in the Central Valley has been found to contribute to increased production in fall-run Chinook salmon (Sommer et al. 2001), but little is known about the potential benefits of recovered floodplains during the migration period for winter-run. Reductions in flow rates have resulted in increased seasonal water temperature. The potential adverse effects of dam operations and reductions in seasonal river flows, such as delays in juvenile emigration and exposure to a higher proportion of agricultural return flows, have all been identified as factors that could affect the survival and success of winter-run Chinook salmon inhabiting the Sacramento River in the future.

Predation by non-native species. Predation on juvenile salmon by non-native fish has been identified as an important threat to winter-run Chinook salmon in areas with high densities of non-native fish (e.g., small and large mouth bass, striped bass, and catfish) that prey on out-migrating juveniles (Lindley and Mohr 2003). Water temperatures are generally lower during out-migration of winter-run compared to other salmonids, and may ameliorate predation pressures that can increase with increasing water temperature. In addition, non-native aquatic vegetation, such as Egeria and water hyacinth, provide suitable habitat for non-native predators (Nobriga et al. 2005, Brown and Michniuk 2007). Predation risk may covary with increased temperatures. Metabolic rates of non-native, predatory fish increase with increasing water temperatures based on bioenergetic studies (Loboschefsky et al. 2009, Miranda et al. 2010). The low spatial complexity and reduced habitat diversity (e.g., lack of cover) of channelized waterways within the Sacramento River and Delta reduces refuge space of salmon from predators (Raleigh et al. 1984, Missildine et al. 2001, 70 FR 52488). A major concern is the potential invasion of the Delta by the highly predatory northern pike. The pike, recently present in Lake Davis on the Feather River, was the target of a major eradication effort (DFG 2007a). If eradication fails and pike escape downstream to the Delta, they would likely be present in areas inhabited by juvenile winter-run Chinook salmon.

Increased predation mortality by native fish species, such as Sacramento pikeminnow at the RBDD, has also been identified as a factor affecting the survival of juvenile salmon within the Sacramento River and Delta.

Harvest. Commercial and recreational harvest of winter-run Chinook salmon in the ocean and inland fisheries has been a subject of management actions by the California Fish and Game Commission and the Pacific Fishery Management Council. The primary concerns focus on the effects of harvest on wild Chinook salmon produced in the Central Valley, as well as the incidental harvest of winter-run Chinook salmon as part of the fall-run and late fall-run salmon fisheries. Naturally reproducing winter-run Chinook salmon are less able to withstand high harvest rates when compared to hatchery-based stocks because of differences in survival rates for incubating eggs and rearing and emigrating juvenile salmon produced in streams and rivers (relatively low survival rates) compared to Central Valley salmon hatcheries (relatively high survival rates) (Knudsen et al. 1999). As a result of recent changes in fishing regulations and restrictions on harvest, commercial and recreational fishing does not appear to have a significant impact on winter-run Chinook salmon populations, but continued assessment is warranted. Commercial fishing for salmon in West Coast ocean waters is managed by the Fishery Management Council and is constrained by time and area closures to meet the Sacramento River winter-run ESA consultation standard and restrictions requiring minimum size limits and use of circle hooks for anglers. Ocean harvest restrictions since 1995 have led to reduced ocean harvest of winter-run Chinook salmon (i.e., Central Valley Chinook salmon ocean harvest index, ranged from 0.55 to nearly 0.80 from 1970 to 1995, and was reduced to 0.27 in 2001). Major restrictions in the commercial fishing industry in California and Oregon during the past two years were enforced to protect Klamath River coho salmon stocks. Because the fishery is mixed, these restrictions have likely reduced harvest of winter-run Chinook salmon, as well. The DFG, NMFS, and Pacific Fishery Management Council continually monitor and assess the effects of harvest of winter-run Chinook salmon, such that regulations can be refined and modified as new information becomes available.

Because adult winter-run Chinook salmon hold in the mainstem Sacramento River until spawning during the summer months, they are particularly vulnerable to illegal (poaching) harvest. Various watershed groups have established public outreach and educational programs in an effort to reduce poaching. In addition, DFG wardens have increased enforcement against illegal harvest of winter-run Chinook salmon. The level and effect of illegal harvest on adult winter-run Chinook salmon abundance and population reproduction is unknown.

Reduced genetic diversity/integrity. Artificial propagation programs conducted for winter-run Chinook salmon conservation purposes (i.e., Livingston Stone National Fish Hatchery) were developed to increase the abundance and diversity of winter-run Chinook salmon and to protect the species from extinction in the event of a catastrophic failure of the wild population. It is unclear what the effects of the hatchery propagation program are on the productivity and spatial structure of the winter-run Chinook salmon ESU (i.e., genetic fitness and productivity). One of the primary concerns with hatchery operations is the genetic introgression by hatchery origin fish that spawn naturally and interbreed with local natural populations (USFWS 2001, USBR 2004, Goodman 2005). It is now recognized that Central Valley hatcheries are a significant and persistent threat to wild Chinook salmon and steelhead populations and fisheries (NMFS 2009a). Such introgression introduces maladaptive genetic changes to the wild winter-run stocks and may reduce overall fitness (Myers et al. 2004, Araki et al. 2007). Taking egg and sperm from a large number of individuals is one method to ameliorate genetic introgression, but artificial selection for traits that assure individual success in a hatchery setting (e.g., rapid growth and tolerance to crowding) are unavoidable (USBR 2004). Investigations are continuing to evaluate the genetic characteristics of winter-run Chinook salmon, improve genetic management of the artificial propagation program, evaluate the minimum viable population size that would maintain genetic integrity within the population, and explore methods for establishing additional independent winter-run Chinook salmon populations as part of recovery planning and conservation of the species.

Entrainment. The vulnerability of juvenile winter-run Chinook salmon to entrainment and salvage at SWP and CVP export facilities varies in response to multiple factors, including the seasonal and geographic distribution of juvenile salmon within the Delta, operation of Delta Cross Channel gates, hydrodynamic conditions occurring within the central and southern regions of the Delta (e.g., Old and Middle rivers), and export rates. The loss of fish to entrainment mortality has been identified as an impact to Chinook salmon populations (Kjelson and Brandes 1989). Juvenile winter-run Chinook salmon tend to be distributed within the central and southern Delta where they have an increased risk of entrainment/salvage between February and April (see Table A.X.1). The effect of changing hydrodynamics within Delta channels, such as reversed flows in Old and Middle rivers resulting from SWP and CVP export operations, has the potential to increase attraction of emigrating juveniles into false migration pathways, delay emigration through the Delta, and directly or indirectly increase vulnerability to entrainment at unscreened diversions. In addition, there is an increase the risk of predation and duration of exposure to seasonally elevated water temperatures and other water quality conditions. SWP and CVP exports have been shown to affect the tidal hydrodynamics (e.g., water current velocities and direction). The magnitude of these hydrodynamic effects vary in response to a variety of factors including tidal stage and magnitude of ebb and flood tides, the rate of SWP and CVP exports, operation of the Clifton Court Forebay (CCF) radial gate opening, and inflow from the upstream tributaries. Chinook salmon behaviorally respond to hydraulic cues (e.g., water currents) during both upstream adult and downstream juvenile migration through the Delta. Changes in these hydraulic cues as a result of SWP and/or CVP export operations during the period that salmon are migrating through Delta channels may contribute to use of false migration pathways, delays in migration, or increased movement of migrating salmon toward the export facilities leading to an increase in entrainment risk. During the past several years, additional investigations have been designed using radio or acoustically-tagged juvenile Chinook salmon to monitor migration behavior through the Delta channels and to assess the effects of changes in hydraulic cues and SWP and CVP export operations on migration (Holbrook et al. 2009, Perry et al. 2010, San Joaquin River Group Authority 2010). These studies are ongoing.

Incidental take of juvenile winter-run Chinook salmon at the SWP and CVP export fish salvage facilities is routinely monitored and reported as part of export operations. Salvage monitoring and the protocol for identifying juvenile winter-run Chinook salmon from other Central Valley Chinook salmon have been refined over the past decade. Run identification was originally determined based on the length of each fish and the date when it was collected. Subsequent genetic testing has been used to refine species identification. Methods for estimating juvenile winter-run Chinook salmon production each year (year class strength) have been developed that take into account the number of adults spawning in the river from carcass surveys, hatching success based on a consideration of water temperatures and other factors, and estimated juvenile survival. Authorized incidental take can then be adjusted each year (1-2 percent of juvenile production) to reflect the relative effect of take at a population level rather than based on a predetermined level that does not reflect year-to-year variation in juvenile production within the Sacramento River.

In addition to SWP and CVP exports, there are more than 2,200 small water diversions throughout the Delta, including unscreened diversions located on the tributary rivers (Herren and Kawasaki 2001). The risk of entrainment is a function of the size of juvenile fish and the slot opening of the screen mesh (Tomljanovich et al. 1978, Schneeberger and Jude 1981, Zeitoun et al. 1981, Weisberg et al. 1987, C. Hanson unpubl. data). Many juvenile winter-run Chinook salmon migrate downstream through the Delta during the late winter or early spring when many of the agricultural irrigation diversions are not operating or are only operating at low levels. Juvenile winter-run Chinook salmon also migrate primarily in the upper part of the water column, reducing their vulnerability to unscreened diversions located near the channel bottom. No quantitative estimates have been developed to assess the potential magnitude of entrainment losses for juveniles migrating through the rivers and Delta, or the effects of these losses on the overall population abundance of returning adult Chinook salmon. The effect of entrainment mortality on the population dynamics and overall adult abundance of winter-run Chinook salmon is not well understood.

Power plants within the Plan Area have the ability to impinge and entrain juvenile Chinook salmon on the existing cooling water system intake screens. However, use of cooling water is currently low with the retirement of older units. Furthermore, newer units are being equipped with a closed cycle cooling system that virtually eliminates the risk of impingement of juvenile salmon.

Besides direct mortality, salmon fitness may be affected by delays in out-migration of smolts caused by reduced or reverse flows. Delays in migration due to water management related to the SWP and CVP operations can make juvenile salmonids more susceptible to many of the threats and stressors discussed in this section, such as predation, entrainment, angling, exposure to poor water quality, and disease. The quantitative relationships among changes in Delta hydrodynamics, the behavioral and physiological response of juvenile salmon, and the increase or decrease in risk associated with other threats is unknown, but currently the subject of a number of investigations and analyses.

Exposure to toxins. Inputs of toxics into the Delta watershed include agricultural drainage and return flows, municipal wastewater treatment facilities, and other point and non-point discharges (Moyle 2002). These toxic substances include mercury, selenium, copper, pyrethroids, and endocrine disruptors with the potential to impact fish health and condition, and adversely impact salmon distribution and abundance. Toxic chemicals have the potential to be widespread throughout the Sacramento River and Delta, or may occur on a more localized scale in response to episodic events (e.g., stormwater runoff, point source discharges, etc.). Agricultural return flows are widely distributed throughout the Sacramento River and the Delta, although dilution flows from the rivers may reduce chemical concentrations to sublethal levels. Toxic algae (e.g., Microcystis) have also been identified as a potential factor adversely affecting salmon and other fish. Exposure to these toxic materials has the potential to directly and indirectly adversely impact salmon distribution and abundance. Concern regarding exposure to toxic substances for Chinook salmon includes both waterborne chronic and acute exposure, but also bioaccumulation and chronic dietary exposure. For example, selenium is a naturally occurring constituent in agricultural drainage water return flows from the San Joaquin River that is then dispersed downstream into the Delta (Nichols et al. 1986). Exposure to selenium in the diet of juvenile Chinook salmon has been shown to result in toxic effects (Saiki 1986, Saiki and Lowe 1987, Hamilton et al. 1986, 1990, Hamilton and Buhl 1990). Selenium exposure has been associated with agricultural and natural drainage within the San Joaquin River basin and refining operations adjacent to San Pablo and San Francisco bays. Other contaminants of concern for Chinook salmon include, but are not limited to, mercury, copper, oil and grease, pesticides, herbicides, ammonia, and localized areas of depressed dissolved oxygen (e.g., Stockton Deep Water Ship Channel, return flows from managed freshwater wetlands, etc.). As a result of the extensive agricultural development within the Central Valley, exposure to pesticides and herbicides has been identified as a significant concern for salmon and other fish species within the Plan Area (Bennett et al. 2001). In recent years, changes have been made in the composition of herbicides and pesticides used on agricultural crops in an effort to reduce potential toxicity to aquatic and terrestrial species. Modifications have also been made to water system operations and discharges related to agricultural wastewater discharges (e.g., agricultural drainage water system lock-up and holding prior to discharge) and municipal wastewater treatment and discharges. Concerns remain, however, regarding the toxicity of contaminants such as pyrethroids that adsorb to sediments and other chemicals (e.g., including selenium and mercury, as well as other contaminants) on salmon.

Mercury and other metals such as copper have also been identified as contaminants of concern for salmon and other fish as a result of direct toxicity and impacts such as those related to acid mine runoff from sites such as Iron Mountain Mine (EPA 2006). The potential problems include tissue bioaccumulation that may adversely impact the fish, but also represents a human health concern (Gassel et al. 2008). These materials originate from a variety of sources including mining operations, municipal wastewater treatment, agricultural drainage within the tributary rivers and Delta, non-point runoff, natural runoff and drainage within the Central Valley, agricultural spraying, and a number of other sources.

The State Water Resources Control Board (SWRCB), Central Valley Regional Water Quality Control Board (CVRWQCB), U.S. Environmental Protection Agency (EPA), U.S. Geological Survey (USGS), DWR, and others have ongoing monitoring programs designed to characterize water quality conditions and identify potential toxicants and contaminant exposure to Chinook salmon and other aquatic resources within the Plan Area. Programs are in place to regulate point source discharges as part of the National Pollutant Discharge Elimination System (NPDES) program as well as programs to establish and reduce total daily maximum loads of various constituents entering the Delta. Changes in regulations have also been made to help reduce chemical exposure and reduce the adverse impacts to aquatic resources and habitat conditions within the Plan Area. These monitoring and regulatory programs are ongoing. Regulations and changes in monitoring and management of agricultural pesticide and herbicide chemicals and their application, education on the effects of urban runoff and chemical discharges, and refined treatment processes have been adopted over the past several decades in an effort to reduce the adverse effects of chemical pollutants on salmon and other aquatic species.

In the final listing determination of the ESU, acid mine runoff from Iron Mountain Mine, located adjacent to the upper Sacramento River, was identified as one of the main threats to winter-run Chinook salmon (Upper Sacramento River Fisheries and Riparian Habitat Advisory Council 1989). Acid mine drainage, including elevated concentrations of metals, produced from the abandoned mine degraded spawning habitat of winter-run Chinook salmon and resulted in high mortality. Storage limitations and limited availability of dilution flows have caused downstream copper and zinc levels to exceed salmonid tolerances and resulted in documented fish kills in the 1960s and 1970s (USBR 2004). The EPA's Iron Mountain Mine remediation program and 2002 restoration plan has removed toxic metals in acidic mine drainage from the Spring Creek watershed with a state-of-the-art lime neutralization plant. Contaminant loading into the Sacramento River from Iron Mountain Mine has shown measurable reductions since the early 1990s. Pollution from Iron Mountain Mine is no longer considered to be a main factor threatening the winter-run Chinook salmon ESU.

Concern has been expressed regarding the potential to resuspend toxic materials into the water column where they may adversely affect salmon through seasonal floodplain inundation, habitat construction projects, channel and harbor maintenance dredging, and other means. For example, mercury deposits exist at a number of locations within the Central Valley and Delta, including the Yolo Bypass. Seasonal inundation of floodplain areas, such as within the Yolo Bypass, has the potential to create anaerobic conditions that contribute to the methylation of mercury, which increases toxicity. Additionally, there are problems with scour and erosion of these mercury deposits by increased seasonal flows. Similar concerns exist regarding creating aquatic habitat by flooding Delta islands or disturbance created by levee setback construction or other habitat enhancement measures. The potential to increase toxicity as a result of habitat modifications designed to benefit aquatic species is one of the factors that needs to be considered when evaluating the feasibility of habitat enhancement projects within the Central Valley.

Sublethal concentrations of toxics may interact with other stressors on salmonids, such as increasing their vulnerability to mortality as a result of exposure to seasonally elevated water temperatures, predation or disease (Werner 2007). For example, Clifford et al. (2005) found in a laboratory setting that juvenile fall-run Chinook salmon exposed to sublethal levels of a common pyrethroid, esfenvalerate, were more susceptible to infectious hematopoietic necrosis virus than those not exposed to esfenvalerate. Although not tested on winter-run Chinook salmon, a similar response is likely.

Increased water temperature. Water temperature is among the physical factors that affect quality of habitat for salmonid adult holding, spawning and egg incubation, juvenile rearing, and migration. Adverse sublethal and lethal effects can result from exposure to elevated water temperatures at sensitive lifestages, such as during incubation or rearing. The Central Valley is the southern limit of Chinook salmon geographic distribution and increased water temperatures are often recognized as an important stressor to California populations. Water temperature criteria for various lifestages of salmonids in the Central Valley have been developed by NMFS (2009). The tolerance of winter-run Chinook salmon to water temperatures depends on life stage, acclimation history, food availability, duration of exposure, health of the individual, and other factors, such as predator avoidance (Myrick and Cech 2004, USBR 2004). Higher water temperatures can lead to physiological stress, reduced growth rates, pre-spawning mortality, reduced spawning success, and increased mortality of salmon (Myrick and Cech 2001). Temperature can also indirectly influence disease incidence and predation (Waples et al. 2008). Exposure to seasonally elevated water temperatures may occur as a result of reductions in flow, as a result of upstream reservoir operations, reductions in riparian vegetation, channel shading, local climate and solar radiation. The installation of the Shasta Temperature Control Device in 1998, in combination with reservoir management to maintain the cold water pool within Shasta Reservoir, has reduced many of the temperature issues on the Sacramento River. Water temperature management on the Sacramento River has been specified in the NMFS biological opinion and has been identified as one of the factors contributing to the observed increase in adult winter-run Chinook salmon abundance in recent years. During dry years, however, the release of cold water from Shasta Dam is still limited. As the river flows further downstream, particularly during the warm spring, summer, and early fall months, water temperatures continue to increase until they reach thermal equilibrium with atmospheric conditions. As a result of the longitudinal gradient of seasonal water temperatures, the coldest temperatures and best areas for winter-run Chinook salmon spawning and rearing are typically located immediately downstream of Keswick Dam.

Increased temperature can also arise from a reduction in shade over rivers by tree removal (Watanabe et al. 2005). Because river water is typically in thermal equilibrium with atmospheric conditions by the time it enters the Delta, this issue is caused primarily from actions upstream of the Delta. As a result of the relatively wide channels that occur within the Delta, the effects of additional riparian vegetation on reducing water temperatures within the Delta are minimal.

The effects of climate change and global warming patterns, in combination with changes in precipitation and seasonal hydrology in the future, have been identified as important factors that may adversely affect the health and long-term viability of Sacramento River winter-run Chinook salmon (Crozier et al. 2008). The rate and magnitude of these potential future environmental changes, and their effect of habitat quality and availability for winter-run Chinook salmon, however, are subject to a high degree of uncertainty.

A2.6 RELEVANT CONSERVATION EFFORTS

Since the listing of Sacramento River winter-run Chinook salmon, several habitat and harvest-related problems that were identified as factors contributing to the decline of the species have been addressed and improved through restoration and conservation actions. The impetus for initiating restoration actions stems primarily from the following: (1) ESA section 7 consultation Reasonable and Prudent Alternatives on temperature, flow, and operations of the CVP and SWP (NMFS 2009b); (2) Regional Water Quality Control Board decisions requiring compliance with Sacramento River water temperature objectives which resulted in the installation of the Shasta Temperature Control Device in 1998; (3) a 1992 amendment to the authority of the CVP through the Central Valley Improvement Act to give fish and wildlife equal priority with other CVP objectives; (4) fiscal support of habitat improvement projects from the California Bay Delta Authority (CALFED) Bay-Delta Program (e.g., installation of a fish screen on the Glenn-Colusa Irrigation District diversion); (5) establishment of the CALFED Environmental Water Account ; (6) EPA actions to control acid mine runoff from Iron Mountain Mine; and, (7) ocean harvest restrictions implemented in 1995.

Results of monitoring at the CVP and SWP fish salvage facility and extensive experimentation over the past several decades have lead to the identification of a number of management actions designed to reduce or avoid the potentially adverse impacts of SWP and CVP export operations on salmon. Many of these actions have been implemented through SWRCB water quality permits (D-1485, D-1641), biological opinions issued on project export operations by NMFS, USFWS, and DFG, as part of CALFED programs (e.g., Environmental Water Account), and as part of Central Valley Project Improvement Act actions. As a result of these requirements, multiple conservation efforts exist to enhance habitat and reduce entrainment of Chinook salmon by the SWP and CVP export facilities.

The artificial propagation program for winter-run Chinook salmon at Livingston Stone National Fish Hatchery, located on the mainstem of the Sacramento River, has operated for conservation purposes since the early 1990s. The increased natural escapement over the last several years has led to the termination of both captive broodstock programs located at University of California at Davis' Bodega Marine Laboratory and Livingston Stone National Fish Hatchery.

Biological opinions for SWP and CVP operations (e.g., NMFS 2009b) and other federal projects involving irrigation and water diversion and fish passage, for example, have improved or minimized adverse impacts to salmon in the Central Valley. In 1992, an amendment through the Central Valley Project Improvement Act gave protection of fish and wildlife equal priority with other CVP objectives. From this act arose several programs that have benefited listed salmonids. The Anadromous Fish Restoration Program is engaged in monitoring, education, and restoration projects designed to contribute toward doubling the natural populations of select anadromous fish species residing in the Central Valley. Restoration projects funded through the Anadromous Fish Restoration Program include fish passage, fish screening, riparian easement and land acquisition, development of watershed planning groups, instream and riparian habitat improvement, and gravel replenishment. The Anadromous Fish Screen Program combines federal funding with state and private funds to prioritize and construct fish screens on major water diversions mainly in the upper Sacramento River. The goal of the Water Acquisition Program is to acquire water supplies to meet the habitat restoration and enhancement goals of the Central Valley Project Improvement Act, and to improve the ability of the U.S. Department of the Interior to meet regulatory water quality requirements. Water has been used to improve fish habitat for Central Valley salmon, with the primary focus on listed Chinook salmon and steelhead, including winter-run Chinook salmon, by maintaining or increasing instream flows (e.g., Environmental Water Account) on the Sacramento River at critical times, and to reduce salmonid entrainment at the SWP and CVP export facilities through reducing seasonal diversion rates during periods when protected fish species are vulnerable to export related losses.

Two programs included under CALFED, the Ecosystem Restoration Program (ERP) and the Environmental Water Account, were created to improve conditions for fish, including winter-run Chinook salmon, in the Central Valley. As part of developing the ERP, a series of conceptual models (DRERIP) have been constructed to provide a framework for identifying and assessing the potential benefits and/or consequences of potential restoration actions. The DRERIP models are being used to evaluate potential BDCP conservation measures, as well as restoration actions as part of the ERP. Restoration actions implemented by the ERP include the installation of fish screens, modification of barriers to improve fish passage, habitat acquisition, and instream habitat restoration. The majority of these actions address key factors and stressors affecting listed salmonids. Additional ongoing actions include efforts to enhance fishery monitoring, and improvements to hatchery management to support salmonid production through hatchery releases.

A major CALFED ERP action currently underway is the Battle Creek Salmon and Steelhead Restoration Project. Although winter-run Chinook salmon do not currently inhabit Battle Creek, they occurred there historically. CALFED is funding the establishment of a second independent population of winter-run Chinook salmon in the upper Battle Creek watershed using the artificial propagation program as a source of fish. The project will restore 77 km (48 miles) of habitat in Battle Creek to support steelhead and Chinook salmon spawning and juvenile rearing at a cost of over $90 million. The project includes removal of five small hydropower diversion dams, construction of new fish screens and ladders on another three dams, and construction of several hydropower facility modifications to ensure the continued hydropower operations. It is thought that this restoration effort is the largest cold water restoration project to date in North America.

As part of CALFED and CVPIA programs, many of the largest water diversions located on the Sacramento River and Delta (e.g., Glenn Colusa Irrigation District, Reclamation District 1001 Princeton diversion, RD 108 Wilkins Slough pumping plant, Sutter Mutual Water Company Tisdale pumping plant, Contra Costa Water District’s Old River and Alternative Intake Project intake, and others) have been equipped with positive barrier fish screens, although the majority of smaller water diversions located on the Sacramento River and Delta remain unscreened. Reclamation District 108 has also designed and constructed a new fish screen and pumping plant (Poundstone Pumping Plant) located on the Sacramento River that consolidates and eliminates three currently existing unscreened water diversions. These fish screening projects are specifically intended to reduce and avoid entrainment losses of juvenile winter-run Chinook salmon and other fish inhabiting the river.

The Delta Regional Ecosystem Restoration Implementation Plan (DRERIP) was formed to guide the implementation of CALFED Ecosystem Restoration Plan elements within the Delta (DFG 2007b). The DRERIP team has created a suite of ecosystem and species conceptual models, including winter-run Chinook salmon, that document existing scientific knowledge of Delta ecosystems. The DRERIP Team has used these conceptual models to assess the suitability of actions proposed in the Ecosystem Restoration Plan for implementation. DRERIP conceptual models were used in the analysis of proposed BDCP conservation measures.

The Central Valley Salmonid Project Work Team, an interagency technical working group led by DFG, drafted a proposal to develop a Chinook salmon escapement monitoring plan that was selected by the CALFED ERP Implementing Agency Managers for directed action funding. Long-term funding for implementation of the monitoring plan still needs to be secured.

Recent habitat restoration initiatives sponsored and funded primarily by the CALFED ERP have funded 29 projects (approximately $24 million) designed to restore ecological function to 9,543 acres (8,091 acres within the Bay Region and the remaining acres located in the Delta and Eastside Tributaries Regions of the CALFED action area) of shallow-water tidal and marsh habitats within the Bay-Delta. Restoration of these areas primarily involves flooding lands previously used for agriculture, thereby creating additional rearing habitat for juvenile salmonids. Similar habitat restoration is imminent adjacent to Suisun Marsh (i.e., at the confluence of Montezuma Slough and the Sacramento River) as part of the Montezuma Wetlands project, which is intended to provide for commercial disposal of material dredged from San Francisco Estuary in conjunction with tidal wetland restoration.

The U.S. EPA's Iron Mountain Mine remediation involves the removal of toxic metals in acidic mine drainage from the Spring Creek Watershed with a state-of-the-art lime neutralization plant. Contaminant loading into the Sacramento River from Iron Mountain Mine, and other mining operations, has shown measurable reductions since the early 1990s. Decreasing the heavy metal contaminants that enter the Sacramento River should increase the survival of salmonid eggs and juveniles. However, during periods of heavy rainfall upstream of the Iron Mountain Mine, Reclamation substantially increases Sacramento River flows to dilute heavy metal contaminants being spilled from the Spring Creek debris dam. This rapid change in flows can cause juvenile salmonids to become stranded or isolated in side channels below Keswick Dam.

In 2001, a new fish screen was constructed at the Anderson Cottonwood Irrigation District Diversion Dam and a state-of-the-art fish ladder was installed to address the threats caused by the diversion dam. As described in the final listing determination for the ESU (70 FR 37160), the flashboard gates and inadequate fish ladders at the diversion dam blocked passage for upstream migrant winter-run Chinook salmon. The seasonal operation of the dam created unsuitable habitat upstream of the dam by reducing flow velocity over the incubating eggs, reducing egg survival. Evaluation of the fish ladder is ongoing.

To help reduce the effects of the RBDD operation on migration of adult and juvenile salmonids and other species, management has changed in recent years to maintain the dam gates in the open position for a longer period of time and thereby facilitate greater upstream and downstream migration. Changes in dam operations have benefited both upstream and downstream migration by salmon and have contributed to a reduction in juvenile predation mortality. In 2009, USBR received funding for the Fish Passage Improvement Project at the RBDD to build a pumping facility to provide reliable water supply for high-valued crops in Tehama, Glenn, Colusa, and northern Yolo counties while providing year-round unimpeded fish passage. This project, which is expected to be completed in late 2012, will eliminate passage issues for winter-run Chinook salmon and other migratory species.

DWR's Delta Fish Agreement Program has approved approximately $49 million for projects that benefit salmon and steelhead production in the Sacramento-San Joaquin basins and Delta since the agreements inception in 1986. Delta Fish Agreement projects that benefit Sacramento River winter-run Chinook salmon include enhanced law enforcement efforts from San Francisco Estuary upstream into the Sacramento River, spawning gravel augmentations, and habitat enhancement projects. Through the Delta-Bay Enhanced Enforcement Program (DBEEP), initiated in 1994, a team of 10 wardens focus their enforcement efforts on salmon, steelhead, and other species of concern from the San Francisco Estuary upstream into the Sacramento and San Joaquin River basins. Enhanced enforcement programs are believed to have had significant benefits to Chinook salmon attributed to DFG, although results have not been quantified.

Harvest protective measures for Sacramento River winter-run Chinook salmon include seasonal constraints on sport and commercial fisheries south of Point Arena in an effort to reduce harvest of winter-run Chinook salmon. Ocean harvest restrictions since 1995 have led to reduced ocean harvest of winter-run Chinook salmon (i.e., Central Valley Chinook salmon ocean harvest index ranged from 0.55 to nearly 0.80 from 1970 to 1995, and was reduced to 0.27 in 2001). The state of California has established specific in-river fishing regulations and no-retention prohibitions designed to protect Sacramento River winter-run Chinook salmon. DFG has implemented enhanced enforcement efforts to reduce illegal harvests.

A2.7 RECOVERY GOALS

The Public Draft Recovery Plan for Central Valley salmonids, including Sacramento River winter-run Chinook salmon, was released by NMFS on October 19, 2009. Although not final, the overarching goal in the public draft is the removal of, among other listed salmonids, Sacramento River winter-run Chinook salmon from the federal list of Endangered and Threatened Wildlife (NMFS 2009a). Several objectives and related criteria represent the components of the recovery goal, including the establishment of at least two viable populations within each historical diversity group, as well as other measurable biological criteria.

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APPENDIX A3. CENTRAL VALLEY SPRING-RUN CHINOOK SALMON (*ONCORHYNCHUS TSHAWYTSCHA*)

A3.1 LEGAL STATUS

The Central Valley spring-run Chinook salmon evolutionarily significant unit (ESU) is listed as a threatened species under the federal Endangered Species Act (ESA). The ESU includes all naturally spawned populations of spring-run Chinook salmon in the Sacramento River and its tributaries in California, including the Feather River (see Figure A-3a). The ESU was listed as threatened on September 16, 1999 (64 FR 50394).

In June 2004, the National Marine Fisheries Service (NMFS) proposed that Central Valley spring-run Chinook salmon remain listed as threatened (69 FR 33102). This proposal was based on the recognition that, although Central Valley spring-run Chinook salmon productivity trends were positive, the ESU continued to face risks from having a limited number of remaining populations (i.e., three existing populations from an estimated 17 historical populations), a limited geographic distribution, and potential hybridization with Feather River Hatchery spring-run Chinook salmon. Until recently, Feather River Hatchery spring-run Chinook salmon were not included in the ESU, yet these fish are genetically distinct from other populations in Mill, Deer, and Butte creeks.

On June 28, 2005, NMFS issued its final decision to retain the status of Central Valley spring-run Chinook salmon as threatened (70 FR 37160). This decision also included the Feather River Hatchery spring-run Chinook salmon population as part of the Central Valley spring-run Chinook salmon ESU.

Spring-run Chinook salmon was listed as a threatened species under the California ESA on February 5, 1999.

A3.2 SPECIES DISTRIBUTION AND STATUS

A3.2.1 Range and Status

Historically, spring-run Chinook salmon were predominant throughout the Central Valley occupying the upper and middle reaches (1,000 to 6,000 feet) of the San Joaquin, American, Yuba, Feather, Sacramento, McCloud and Pit rivers, with smaller populations in most tributaries with sufficient habitat for adult salmon holding over the summer months (see Figure A-3a) (Stone 1874, Rutter 1904, Clark 1929). Completion of Friant Dam extirpated the native spring-run Chinook salmon population from the San Joaquin River and its tributaries. Naturally-spawning populations of Central Valley spring-run Chinook salmon with consistent spawning returns are currently restricted to Butte Creek, Deer Creek, and Mill Creek (Good et al. 2005).

Figure A-3a. Central Valley Spring-Run Chinook Salmon Inland Range in California

There is a small spawning population that has been documented in Clear Creek (Newton and Brown 2004). In addition, the upper Sacramento River and Yuba River support small populations, but their status is not well documented. The Feather River Hatchery produces spring-run Chinook salmon on the Feather River.

Central Valley spring-run Chinook salmon were once the most abundant run of salmon in the Central Valley (Campbell and Moyle 1992). The Central Valley drainage as a whole is estimated to have supported spring-run Chinook salmon runs as large as 600,000 fish between the late 1880s and 1940s (California Department of Fish and Game [DFG] 1998). More than 500,000 Central Valley spring-run Chinook salmon were caught in the Sacramento-San Joaquin commercial fishery in 1883 (Yoshiyama et al. 1998). Population estimates of returning spring-run Chinook salmon for the years immediately preceding and after the closure of Friant Dam in February 1944 are: 35,000 in 1943, 5,000 in 1944, 56,000 in 1945, 30,000 in 1946, 6,000 in 1947, and 2,000 in 1948 (Fry 1961, Yoshiyama et al. 1998). There were occasional records of returning spring-run Chinook salmon during the 1950s and 1960s in wet years. The San Joaquin River population was essentially extirpated by the late 1940s. Populations in the upper Sacramento, Feather, and Yuba rivers were eliminated with the construction of major dams during the 1950s and 1960s.

The Central Valley spring-run Chinook salmon ESU has displayed broad fluctuations in adult abundance between 1961 and 2009 (see Figure A-3b). Adult spring-run salmon escapement to the Sacramento River system in 2009 was 3,802 fish. Sacramento River tributary populations in Mill, Deer, and Butte creeks are probably the best trend indicators for the Central Valley spring-run Chinook ESU as a whole because these streams contain the primary independent populations within the ESU. Generally, there was a positive trend in escapement in these waterways between 1992 and 2005, at which time there was a steep decline (see Figure A-3c). Estimated adult spring-run salmon escapement to Mill, Deer, and Butte creeks in 2009 was only 2,492 fish. Escapement numbers are dominated by Butte Creek returns, which represented nearly 75 percent of fish returning to these three creeks since 2000. Adult spring-run salmon escapement to Butte Creek in 2009 was approximately 2,059 fish, or 83 percent of escapement to these three creeks. During the period between 1992 and 2009, there have been significant habitat improvements in these watersheds, including the removal of several small dams and increases in summer flows, as well as reduced ocean salmon harvest and a favorable terrestrial and marine climate. The significant recent declines in adult fall-run Chinook salmon escapement has resulted in significant curtailment of the commercial and recreational salmon fisheries, which is expected to also increase the level of protection and benefit the Central Valley spring-run Chinook salmon population.

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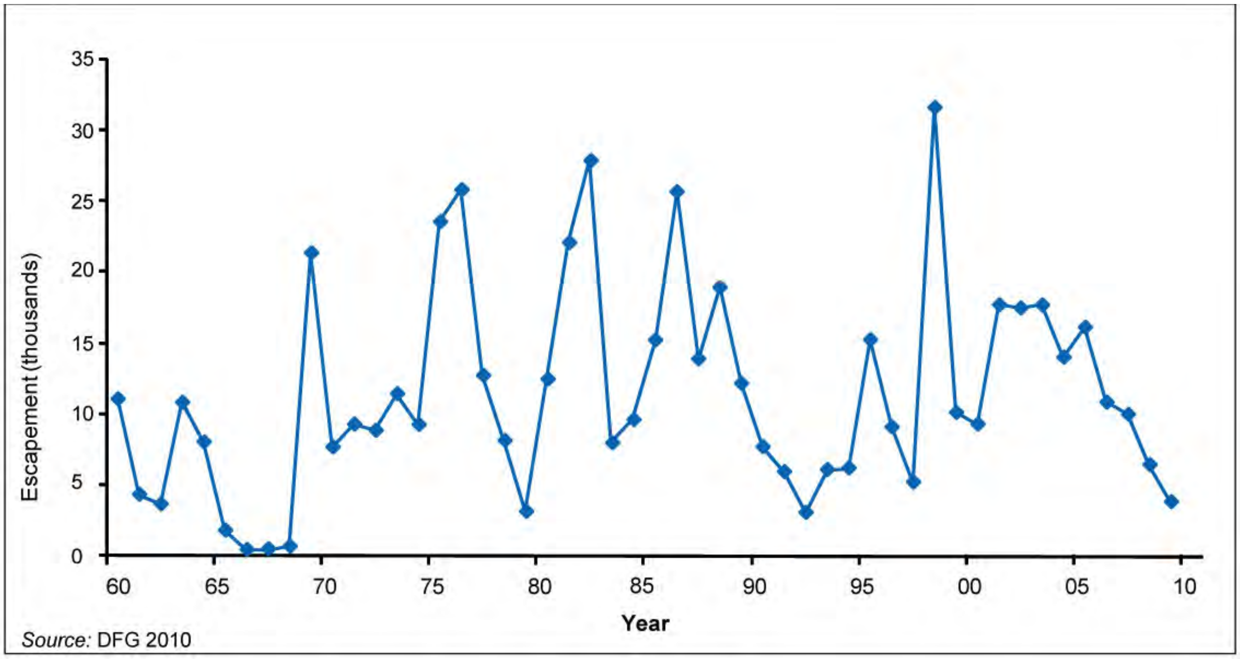

Figure A-3b. Estimate Historical Spawner Escapement of Spring-Run Chinook Salmon throughout the Central Valley (1960-2009)

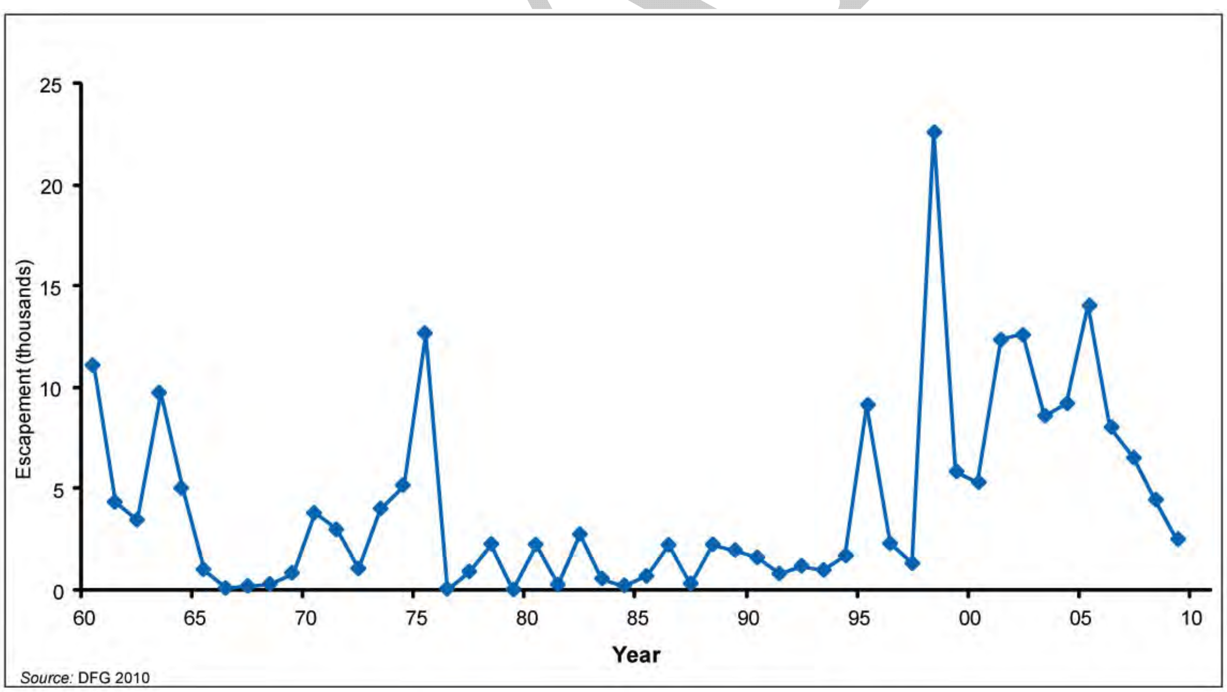

Figure A-3c. Estimated Historical Spawner Escapement of Spring-Run Chinook Salmon in Mill, Deer, and Butte Creeks (1960-2009)

On the Feather River, significant numbers of spring-run Chinook salmon, as identified by run timing, return to the Feather River Hatchery. However, coded-wire tag information from these hatchery returns and results of genetic testing indicate that substantial introgression has occurred between fall-run and spring-run Chinook salmon populations within the Feather River due to hatchery practices and the geographic and temporal overlap with spawning fall-run in the river.

Although recent Central Valley spring-run Chinook salmon population trends are negative, annual abundance estimates display a high level of variation. The overall number of Central Valley spring-run Chinook salmon remains well below estimates of historical abundance. Central Valley spring-run Chinook salmon have some of the highest population growth rates in the Central Valley, but other than Butte Creek and the hatchery-influenced Feather River, population sizes are very small relative to fall-run Chinook salmon populations (Good et al. 2005).

The viability of an ESU that is essentially represented by three populations located within the same ecoregion is vulnerable to changes in the environment through a lack of spatial geographic diversity. The current geographic distribution of viable populations makes the Central Valley spring-run Chinook salmon ESU vulnerable to catastrophic disturbance (Lindley et al. 2007). Such potential catastrophes include volcanic eruption of Mt. Lassen, prolonged drought conditions reducing coldwater pool adult holding habitat, and a large wildfire (approximately 30 km maximum diameter) encompassing the Deer, Mill and Butte creek watersheds. The Central Valley spring-run Chinook salmon ESU remains at a moderate to high risk of extinction because: (1) the ESU is spatially confined to relatively few remaining streams within its historical range; (2) the population continues to display broad fluctuations in abundance; and, (3) a large proportion of the population (i.e., in Butte Creek) faces the risk of high mortality rates due to high water temperatures during the adult holding period.

A3.2.2 Distribution and Status in the Plan Area

The entire population of the Central Valley spring-run Chinook salmon ESU must pass through the Plan Area as migrating adults and emigrating juveniles. Adult Central Valley spring-run Chinook salmon migrate primarily along the western edge of the Delta through the Sacramento River corridor, and juvenile spring-run Chinook salmon use the Delta, Suisun Marsh, and Yolo Bypass for migration and rearing.

A3.3 HABITAT REQUIREMENTS AND SPECIAL CONSIDERATIONS

Critical habitat for spring run Chinook salmon ESU was updated on September 2, 2005 with an effective date of January 2, 2006 (70 FR 52488). Designated critical habitat includes 1,158 miles of stream habitat in the Sacramento River basin and 254 square miles of estuarine habitat in the San Francisco-San Pablo-Suisun Bay complex (70 FR 52488, Figure A-3d). Critical habitat includes stream reaches such as those of the Feather and Yuba rivers, Big Chico, Butte, Deer, Mill, Battle, Antelope, and Clear creeks, and the Sacramento River and Delta.

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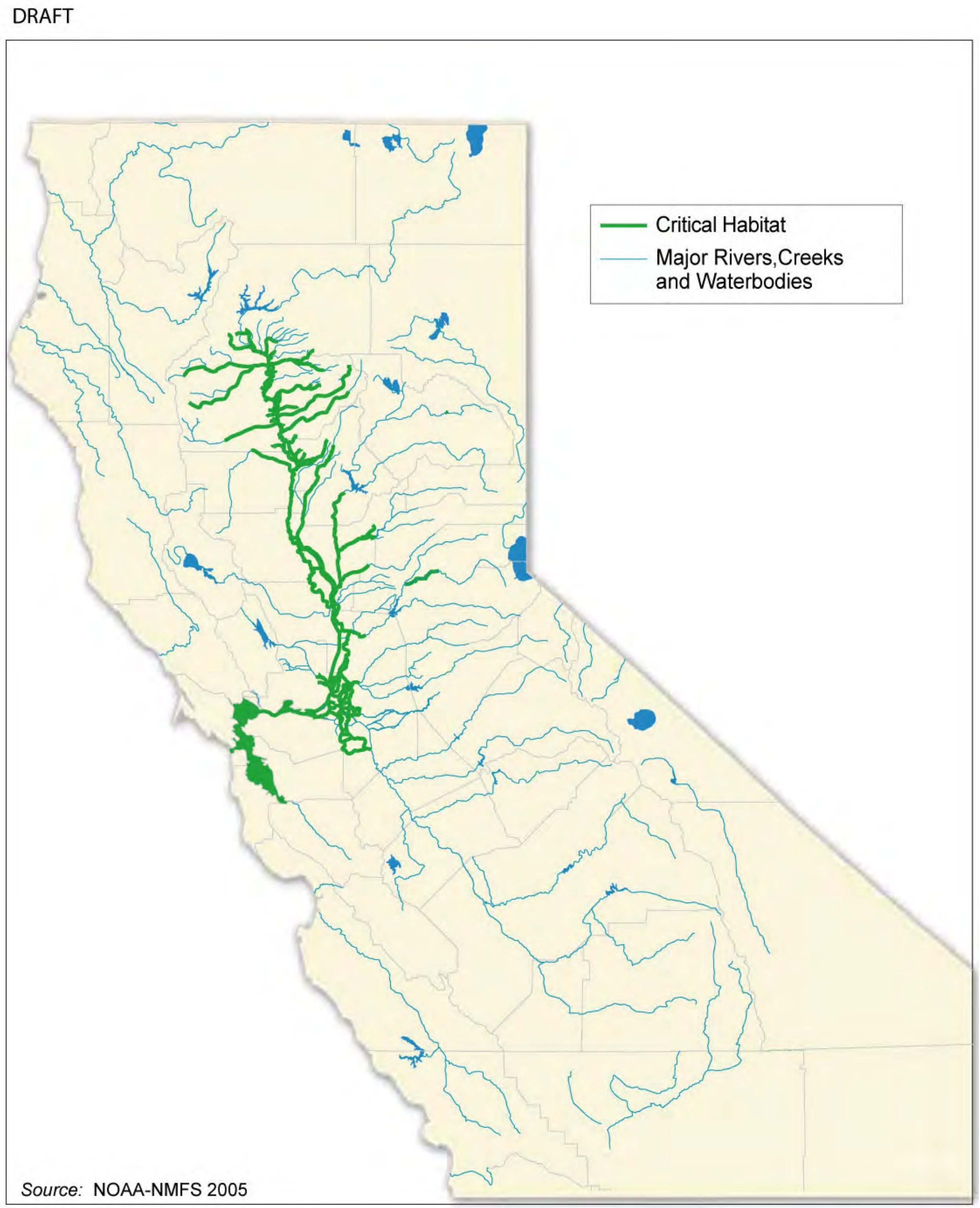

Figure A-3d. Central Valley Spring-Run Chinook Salmon Inland Designated Critical Habitat in California

This habitat is comprised of physical and biological features considered essential to the conservation of the species, including space for individual and population growth and for normal behavior; cover; sites for breeding, reproduction, and rearing of offspring; and habitats protected from disturbance or are representative of the historical geographical and ecological distribution of the species.

Central Valley spring-run Chinook salmon habitats are also protected under the Magnuson-Stevens Fishery Conservation and Management Act as Essential Fish Habitat (EFH). Those waters and substrate necessary to spring-run Chinook salmon for spawning, breeding, feeding, or growth to maturity are included as EFH and are presented in Figure A-3e. Critical Habitat and EFH are managed differently from a regulatory standpoint, but are biologically equal for the conservation of Central Valley spring-run Chinook salmon.

The Primary Constituent Elements (PCEs) considered essential for conservation are: (1) freshwater spawning sites, (2) freshwater rearing sites, (3) freshwater migration corridors, (4) estuarine rearing and migration areas, (5) nearshore marine areas, and (6) offshore marine areas.

A3.3.1 Freshwater Spawning Habitat

Freshwater spawning sites are those stream reaches with water quantity (instream flows) and quality conditions (e.g., water temperature and dissolved oxygen) and substrate suitable to support spawning, egg incubation, and larval development. Most spawning habitat in the Central Valley for spring-run Chinook salmon is located in areas directly downstream of dams containing suitable environmental conditions for spawning and incubation. Historically, spring-run Chinook salmon migrated upstream into high elevation steep gradient reaches of the rivers and tributaries for spawning. Access to the majority of these historical spawning areas has been blocked by construction of major Central Valley dams and reservoirs. Currently, Central Valley spring-run Chinook salmon spawn on the mainstem Sacramento River between the Red Bluff Diversion Dam (RBDD) and Keswick Dam, and in tributaries such as the Feather River and Mill, Deer, and Butte creeks. There is currently an effort underway to re-establish a self-sustaining population of spring-run Chinook salmon on the San Joaquin River downstream of Friant Dam. Spawning habitat has a high conservation value as its function directly affects the spawning success and reproductive potential of listed salmonids.

A3.3.2 Freshwater Rearing Habitat

Freshwater rearing sites are those with water quantity and floodplain connectivity to form and maintain physical habitat conditions and support juvenile growth and mobility; suitable water quality; availability of suitable prey and forage to support juvenile growth and development; and natural cover such as shade, submerged and overhanging large wood, log jams, beaver dams, aquatic vegetation, large woody debris, 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.

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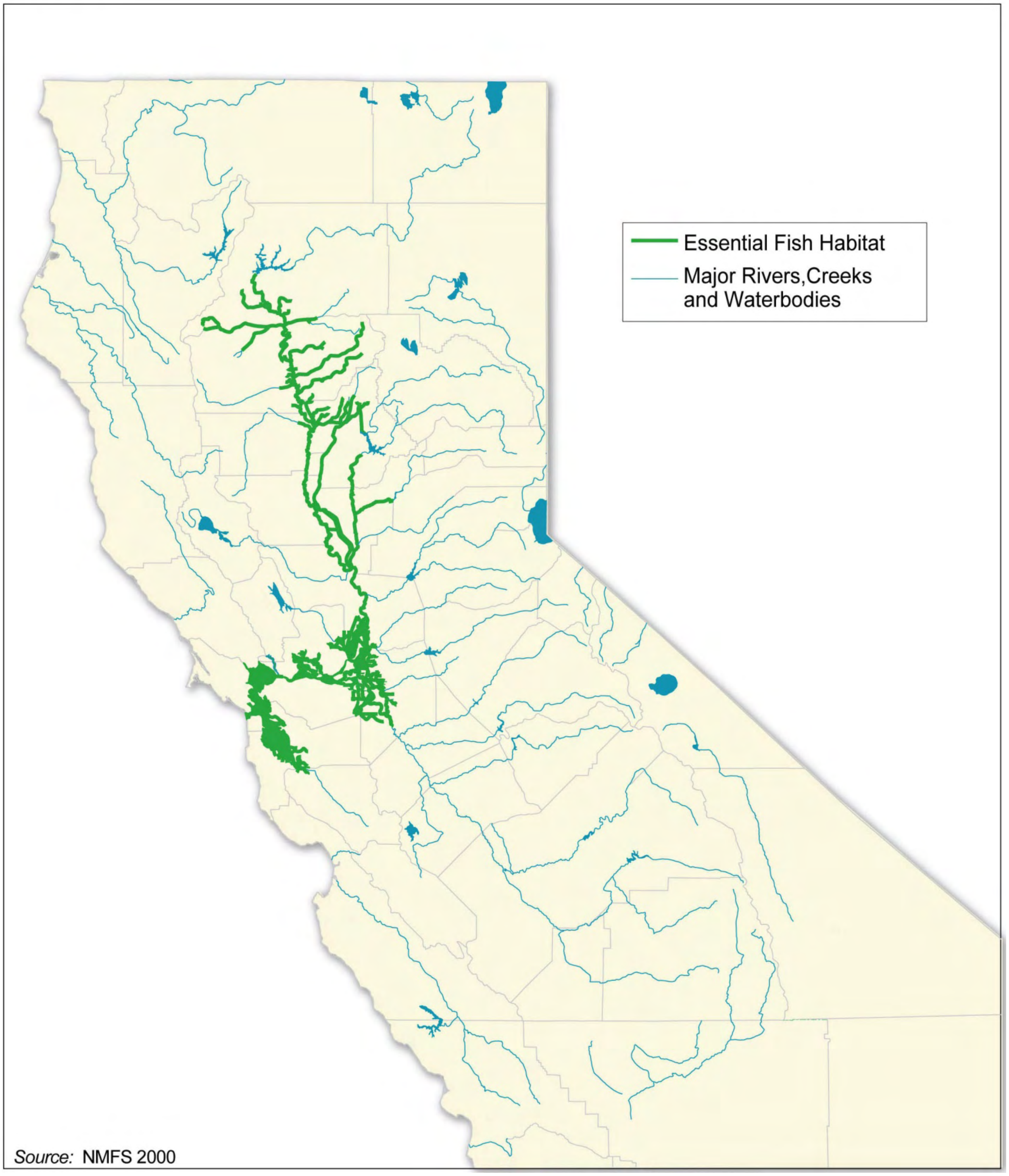

Figure A-3e. Central Valley Spring-Run Chinook Salmon Inland Essential Fish Habitat in California

Non-natal, intermittent tributaries are also used for juvenile rearing. Rearing habitat condition is strongly affected by habitat diversity and complexity, food supply, and presence of predators. Some of these more complex, productive habitats with floodplain connectivity are still present in limited amounts within the Central Valley (e.g., the lower Cosumnes River, Sacramento River reaches with set-back levees [i.e., primarily located upstream of the City of Colusa]). However, the channeled, leveed, and riprapped river reaches and sloughs that are common along the Sacramento and San Joaquin rivers and throughout the Delta typically have low habitat complexity, low abundance of food organisms, and offer little protection from predatory fish and birds. Freshwater rearing habitat also has a high conservation value, as the juvenile life stage of salmonids is dependent on the function of this habitat for successful survival and recruitment to the adult population.

A3.3.3 Freshwater Migration Corridors

Freshwater migration corridors for spring-run Chinook salmon, including river channels, channels through the Delta, and the Bay-Delta estuary, support mobility, survival, and food supplies for juveniles and adults. Migration corridors should be free from obstructions (passage barriers and impediments to migration), have favorable water quantity (instream flows) and quality conditions (seasonal water temperatures), and contain natural cover such as submerged and overhanging large wood, native aquatic vegetation, large rocks and boulders, side channels, and undercut banks. Migratory corridors for spring-run Chinook salmon are located downstream of the spawning areas and include the lower Sacramento River, lower Feather River, tributaries providing suitable adult holding and spawning habitat, the Delta, and the San Francisco Bay complex extending to coastal marine waters. Efforts are currently underway to re-establish a spring-run salmon population on the San Joaquin River downstream of Friant Dam that would use the lower river and Delta as part of the migration corridor. These corridors allow the upstream passage of adults and the downstream emigration of juvenile salmon. Migratory corridor conditions are strongly affected by the presence of passage barriers, which can include dams, unscreened or poorly screened diversions, and degraded water quality. For freshwater migration corridors to function properly, they must provide adequate passage, provide suitable migration cues, reduce false attraction, avoid areas where vulnerability to predation is increased, and avoid impediments and delays in both upstream and downstream migration. For this reason, freshwater migration corridors are considered to have a high conservation value.

Results of mark-recapture studies conducted using juvenile Chinook salmon (typically fall-run or late fall-run Chinook salmon, which are considered to be representative of juvenile spring-run salmon) released into both the Sacramento and San Joaquin rivers have shown high mortality during passage downstream through the rivers and Delta (Brandes and McLain 2001, Newman and Rice 2002, Manly 2004, San Joaquin River Group Authority 2007, Hanson 2008, Low and White undated). Mortality for juvenile salmon is typically greater in the San Joaquin River than in the Sacramento River (Brandes and McLain 2001). In both rivers, mortality is typically greater in years when spring flows are reduced and water temperatures are increased. Results of survival studies have shown that closing the Delta Cross Channel gates and installation of the Head of Old River Barrier to reduce the movement of juvenile salmon into the Delta contribute

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to improved survival of emigrating juvenile Chinook salmon (Brandes and McLain 2001, Manly 2004, San Joaquin River Group Authority 2010, Low and White undated). Observations at the State Water Project (SWP) and Central Valley Project (CVP) fish salvage facilities have shown that very few of the marked salmon (typically fewer than 1 percent) are entrained and salvaged at the export facilities (San Joaquin River Group Authority 2007, Hanson 2008, California Department of Water Resources [DWR] and U.S. Bureau of Reclamation [USBR] unpubl. data). Although the factors contributing to high juvenile mortality have not been quantified, results of acoustic tagging experiments and anecdotal observations suggest that exposure to adverse water quality conditions (e.g., elevated water temperatures, toxic chemicals) and vulnerability to predation are two of the factors contributing to the high juvenile mortality observed in the rivers and Delta (San Joaquin River Group Authority 2007). Additional acoustic tagging experiments are currently underway to better assess factors affecting migration pathways, migration rates, effects of SWP and CVP exports on migration, and reach-specific survival rates for emigrating juvenile Chinook salmon (Lindley et al. 2008, MacFarlane et al. 2008a, Michel et al. 2008, Perry et al. 2008).

A3.3.4 Estuarine Areas

Estuarine migration and juvenile rearing habitats should be free of obstructions (i.e., dams and other barriers) and provide suitable water quality, water quantity (river and tidal flows), and salinity conditions to support juvenile and adult physiological transitions between fresh and salt water. Natural cover, such as submerged and overhanging large wood, native aquatic vegetation, and side channels, provide juvenile foraging habitat and cover from predators. Tidal wetlands and seasonally inundated floodplains have also been identified as high value foraging and rearing habitats for juvenile salmon migrating downstream through the estuary. Estuarine areas contain a high conservation value as they function to support juvenile Chinook salmon growth, smolting, avoidance of predators, and provide a transition to the ocean environment.

A3.3.5 Ocean Habitats

Although ocean habitats are not part of the critical habitat listing for Central Valley spring-run Chinook salmon, biologically productive coastal waters are an important habitat component for the ESU. Juvenile Chinook salmon inhabit near-shore coastal marine waters for a period of typically two to four years before adults return to Central Valley rivers to spawn. During their marine residence Chinook salmon forage on krill, squid, and other marine invertebrates as well as a variety of fish such as northern anchovy and Pacific herring. These features are essential for conservation because, without them, juveniles cannot forage and grow to adulthood.

Results of oceanographic studies have shown the variation in ocean productivity off the West Coast within and among years. Changes in ocean currents and upwelling have been identified as significant factors affecting nutrient availability, phytoplankton and zooplankton production and the availability of other forage species in near-shore surface waters. Ocean conditions during the salmon’s ocean residency period can be important, as indicated by the effect of the 1983 El Niño

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on the size and fecundity of Central Valley fall-run Chinook salmon (Wells et al. 2006). Although the effects of ocean conditions on Chinook salmon growth and survival have not been investigated extensively, recent observations since 2007 have shown a significant decline in the abundance of adult Chinook salmon and coho salmon returning to California rivers and streams (fall-run adult returns to the Sacramento and San Joaquin rivers were the lowest on record; Pacific Fishery Management Council 2008) that is thought to be the result of declines in ocean productivity and associated high mortality rates during the period when these fish were rearing in near-shore coastal waters (MacFarlane et al. 2008b). The importance of changes in ocean conditions on growth, survival, and population abundance of Central Valley Chinook salmon is currently undergoing further investigation.

A3.4 LIFE HISTORY

Chinook salmon typically mature between two and six years of age (Myers et al. 1998). Freshwater entry and spawning timing generally are thought to be related to local water temperature and flow regimes. Runs are designated on the basis of adult migration timing; however, distinct runs also differ in the degree of maturation at the time of river entry, thermal regime and flow characteristics of their spawning site, and the actual time of spawning (Myers et al. 1998). Spring-run Chinook salmon tend to enter freshwater as immature fish, migrate far upriver, hold in cool water pools for a period of months during the spring and summer, and delay spawning until the early fall.

Information on the migration rates of Central Valley spring-run Chinook salmon in freshwater is scant, but a general description of migration rates can be found in Appendix A4, *Central Valley fall-/late fall-run Chinook salmon.*

Adult Central Valley spring-run Chinook salmon begin their upstream migration in late January and early February (DFG 1998) and enter the Sacramento River between March and September, primarily in May and June (Table A-3a) (Yoshiyama et al. 1998, Moyle 2002). Lindley et al. (2006) reported that adult Central Valley spring-run Chinook salmon enter native tributaries from the Sacramento River primarily between mid-April and mid-June. Typically, spring-run Chinook salmon utilize mid- to high-elevation streams that provide appropriate seasonal water temperatures and sufficient flow, cover, and pool depth to allow over-summering while conserving energy and allowing their gonadal tissue to mature (Yoshiyama et al. 1998).

Chinook salmon spawn in clean, loose gravel in swift, relatively shallow riffles or along the margins of deeper reaches where suitable water temperature, depth, and velocity favor redd construction and adequate oxygenation of incubating eggs. Chinook salmon spawning typically occurs in gravel beds located at the tails of holding pools (U.S. Fish and Wildlife Service [USFWS] 1995). Fry emergence generally occurs at night. Upon emergence, fry swim or are displaced downstream (Healey 1991). The daily migration of juvenile spring-run Chinook salmon passing RBDD is highest in the four hour period prior to sunrise (Martin et al. 2001).

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Table A-3a. Temporal Occurrence of (a) Adult and (b) Juvenile Central Valley Spring-Run Chinook Salmon in the Sacramento River. Darker shades indicate months of greatest relative abundance.

(a) Adult

Location	*Jan*	*Feb*	*Mar*	*Apr*	*May*	*Jun*	*Jul*	*Aug*	*Sep*	*Oct*	*Nov*	*Dec*
[1,2]Sac. River basin												
[3]Sac. River												
[4]Mill Creek												
[4]Deer Creek												
[4]Butte Creek												

(b) Juvenile

Location	*Jan*	*Feb*	*Mar*	*Apr*	*May*	*Jun*	*Jul*	*Aug*	*Sep*	*Oct*	*Nov*	*Dec*
[5]Sac. River Tribs												
[6]Upper Butte Creek												
[4]Mill, Deer, Butte creeks												
[3]Sac. River at RBDD												
[7]Sac. River at KL												
[8*]Chipps Island (trawl)												
Relative Abundance:	= High				= Medium				= Low			

* By the time spring-run Chinook salmon yearlings reach Chipps Island they cannot be distinguished with confidence from fall-run Chinook salmon yearlings.
Sources: [1]Yoshiyama et al. 1998; [2]Moyle 2002; [3]Myers et al. 1998; [4]Lindley et al. 2006; [5]DFG 1998; [6]McReynolds et al. 2005; Ward et al. 2002, 2003; [7]Snider and Titus 2000,[8]USFWS 2001

Fry may continue downstream to the estuary and rear, or may take up residence in the stream for a period from weeks to a year (Healey 1991). Fry seek streamside habitats containing beneficial characteristics such as riparian vegetation and associated substrates that provide aquatic and terrestrial invertebrates, predator avoidance cover, and slower water velocities for resting (NMFS 1996).

Spring-run Chinook salmon fry emerge from the gravel from November to March (Moyle 2002) and the emigration timing is highly variable, as they may migrate downstream as young-of-the year or as juveniles or yearlings. The modal size of fry migrants at approximately 40 mm between December and April in Mill, Butte, and Deer creeks reflects a prolonged emergence of fry from the gravel (Lindley et al. 2006). Studies in Butte Creek found that the majority of Central Valley spring-run Chinook salmon migrants are fry occurring primarily during December, January, and February, and that fry movements appeared to be influenced by flow (Ward et al. 2002, 2003, McReynolds et al. 2005). Small numbers of Central Valley spring-run Chinook salmon remained in Butte Creek to rear and migrated as yearlings later in the spring. Juvenile emigration patterns in Mill and Deer creeks are very similar to patterns observed in Butte Creek, with the exception that juveniles from Mill and Deer creeks typically exhibit a later young-of-the year migration and an earlier yearling migration (Lindley et al. 2006).

Once juveniles emerge from the gravel they initially seek areas of shallow water and low velocities while they finish absorbing the yolk sac (Moyle 2002). Many also disperse downstream during high-flow events. As is the case with other salmonids, there is a shift in microhabitat use by juveniles to deeper faster water as they grow. Microhabitat use can be

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influenced by the presence of predators, which can force juvenile salmon to select areas of heavy cover and suppress foraging in open areas (Moyle 2002). Peak movement of juvenile Central Valley spring-run Chinook salmon in the Sacramento River at Knights Landing occurs in December, and again in March and April; however, juveniles were also observed between November and the end of May (Snider and Titus 2000).

As juvenile Chinook salmon grow, they move into deeper water with higher current velocities, but still seek shelter and velocity refugia to minimize energy expenditures (Healey 1991). Catches of juvenile salmon in the Sacramento River near West Sacramento by the USFWS (1997) showed that larger juvenile salmon were captured in the main channel and smaller sized fry were typically captured along the channel margins. When the channel of the river is greater than 9 to 10 feet in depth, juvenile salmon tend to inhabit surface waters (Healey 1980). Stream flow changes and/or turbidity increases in the upper Sacramento River watershed are thought to stimulate juvenile emigration (Kjelson et al. 1982, Brandes and McLain 2001).

Within the Delta, juvenile Chinook salmon forage in shallow areas with protective cover, such as tidally influenced sandy beaches and shallow water areas with emergent aquatic vegetation (Meyer 1979, Healey 1980). Cladocerans, copepods, amphipods, and larval dipterans, as well as small arachnids and ants are common prey items (Kjelson et al. 1982, Sommer et al. 2001a, MacFarlane and Norton 2002). Though the bulk of production in Butte and Big Chico creeks emigrate as fry, yearlings can enter the Delta as early as February and as late as June (DFG 1998). Yearling-sized spring-run Chinook salmon migrants appear at Chipps Island (entrance to Suisun Bay) between October and December (Brandes and McLain 2001, USFWS 2001). It has been hypothesized that changes in habitat conditions within the Delta over the past century may have resulted in a reduction in extended juvenile salmon rearing when compared to periods when habitat for juvenile salmon rearing was more suitable.

Central Valley spring-run Chinook salmon begin their ocean life in the coastal marine waters of the Gulf of the Farallones. Upon reaching the ocean, juveniles feed on larval and juvenile fishes, plankton, and terrestrial insects (Healey 1991, MacFarlane and Norton 2002). Juveniles grow rapidly in the ocean environment with growth rates dependent on water temperatures and food availability (Healey 1991). The first year of ocean life is considered a critical period of high mortality for Chinook salmon that largely determines survival to harvest or spawning (Beamish and Mahnken 2001, Quinn 2005).

A3.5 THREATS AND STRESSORS

The following have been identified as important threats and stressors to spring-run Chinook salmon (without priority).

Reduced staging and spawning habitat. Access to most of the historical upstream spawning habitat for spring-run Chinook salmon (see Figure A-3a) has been eliminated or degraded by man-made structures (e.g., dams and weirs) associated with water storage and conveyance, flood

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control, and diversions and exports for municipal, industrial, agricultural, and hydropower purposes (Yoshiyama et al. 1998). Current spawning and juvenile rearing habitat is restricted to the mainstem and a few tributaries to the Sacramento River. Suitable summer water temperatures for adult and juvenile spring-run Chinook salmon holding and rearing are thought to occur at elevations over 492-1,640 feet (150-500 m), most of which are now blocked by impassible dams. Habitat loss has resulted in a reduction in the number of natural spawning populations from an estimated 17 to 3 (Good et al. 2005).

Upstream diversions and dams have decreased downstream flows and altered the seasonal hydrologic patterns. These factors have been identified as resulting in delayed upstream migration by adults, increased mortality of out-migrating juveniles, and are responsible for making some streams uninhabitable by spring-run salmon (Yoshiyama et al. 1998, DWR 2005). Dams and reservoir impoundments and associated reductions in peak flows have blocked gravel recruitment and reduced flushing of sediments from existing gravel beds, thereby reducing and degrading natal spawning grounds. Further, reduced flows may decrease attraction cues for adult spawners, causing migration delays and increases in straying (DWR 2005). Adult salmon migration delays can reduce fecundity and increase susceptibility to disease and harvest (McCullough 1999).

Dams and other passage barriers also limit the geographic locations where spring-run Chinook salmon can spawn. Within areas such as the Sacramento and Feather rivers, restrictions to upstream movement and spawning site selection for spring-run salmon may increase the risk of hybridization with fall-run salmon, as co-occurrence contributes to an increased risk of redd superimposition. In creeks that are not affected by dams, such as Deer and Mill creeks, adult spring-run Chinook salmon have a greater opportunity to migrate upstream into areas where geographic separation from fall-run salmon reduces the risk of hybridization.

The RBDD located on the Sacramento River has been identified as a barrier and impediment to adult spring-run Chinook salmon upstream migration. Although the RBDD is equipped with fish ladders, migration delays were reported when the dam gates are closed. Mortality from increased predation by Sacramento pikeminnow on juvenile salmon passing downstream through the fish ladder also affects abundance of salmon produced on the Sacramento River (Hallock 1991). To help reduce the effects of dam operation on migration of adult and juvenile salmonids and other species, management changes have occurred in recent years to maintain the dam gates in the open position for a longer period of time and thereby facilitate greater upstream and downstream migration. Changes in dam operations have benefited both upstream and downstream migration of salmon and have contributed to a reduction in juvenile predation mortality.

Reduced rearing and out-migration habitat. Juvenile spring-run Chinook salmon prefer natural stream banks, floodplains, marshes, and shallow water habitats to utilize as rearing habitat during out-migration. Channel margins throughout the Delta have been leveed, channelized, and fortified with riprap for flood protection and island reclamation, reducing and

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degrading the quality of natural habitat available for juvenile Chinook salmon rearing (Brandes and McLain 2001). Man-made barriers further reduce and degrade rearing and migration habitat and delay juvenile out-migration. Juvenile out-migration delays can reduce fitness and increase susceptibility to diversion screen impingement, entrainment, disease, and predation. Modification of natural flow regimes from upstream reservoir operations has resulted in dampening and altering the seasonal timing of the hydrograph, reducing the extent and duration of seasonal floodplain inundation and other flow-dependent habitat used by migrating juvenile Chinook salmon (70 FR 52488, Sommer et al. 2001a, DWR 2005). Recovery of floodplain habitat in the Central Valley has been found to contribute to increases in production in Chinook salmon (Sommer et al. 2001b), but little is known about the potential benefit available to migrating spring-run. Reductions in flow rates have resulted in increased seasonal water temperature. The potential adverse effects of dam operations and reductions in seasonal river flows, delays in juvenile emigration, exposure to a higher proportion of agricultural return flows, and exposure to reduced dissolved oxygen concentrations (e.g., Stockton Deep Water Ship Channel) have all been identified as factors that could affect the survival and success of re-establishing spring-run Chinook salmon on the San Joaquin River in the future (Regional Water Resources Control Board 2003).

Predation by non-native species. Predation on juvenile salmon by non-native fish has been identified as an important threat to spring-run Chinook salmon in areas with high densities of non-native fish (e.g., small and large mouth bass, striped bass, and catfish) that prey on out-migrating juveniles (Lindley and Mohr 2003). Non-native aquatic vegetation, such as Brazilian waterweed and water hyacinth, provide suitable habitat for non-native predators (Nobriga et al. 2005, Brown and Michniuk 2007). Predation risk may covary with increased temperatures. Metabolic rates of non-native, predatory fish increase with increasing water temperatures based on bioenergetic studies (Loboschefsky et al. 2009, Miranda et al. 2010). The low spatial complexity and reduced habitat diversity (e.g., lack of cover) of channelized waterways within the rivers and Delta reduces refuge space of salmon from predators (Raleigh et al. 1984, Missildine et al. 2001, DWR 2005, 70 FR 52488). A major concern among managers is the potential invasion of the Delta by the highly predatory northern pike. The pike, recently present in Lake Davis on the Feather River, was the target of a major eradication effort (DFG 2007a). If pike escape downstream to the Delta, they would likely be present in areas inhabited by spring-run Chinook salmon.

Increased predation mortality by native fish species, such as Sacramento pikeminnow at the RBDD, has also been identified as a factor affecting the survival of juvenile salmon within the rivers and Delta.

Harvest. Commercial and recreational harvest of spring-run Chinook salmon in the ocean and inland fisheries has been a subject of management actions by the California Fish and Game Commission and Pacific Fishery Management Council. The primary concerns focus on the effects of harvest on wild Chinook salmon produced in the Central Valley as well as the incidental harvest of listed salmon as part of the fall-run and late fall-run salmon fisheries. Naturally reproducing spring-run Chinook salmon are less able to withstand high harvest rates

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when compared to hatchery-based stocks. Due to reduced survivorship in incubating eggs and rearing and emigrating individuals wild salmon relative to hatchery-reared individuals, naturally reproducing populations are less able to withstand high harvest rates compared to hatchery based stocks (Knudsen et al. 1999). Because of recent changes in fishing regulations and restrictions on harvest, commercial and recreational fishing does not appear to have a significant impact on spring-run Chinook salmon populations, but continued assessment is warranted. Commercial fishing for salmon in West Coast ocean waters is managed by the Pacific Fishery Management Council, and is constrained by time and area closures to meet the Sacramento River winter-run ESA consultation standard and restrictions requiring minimum size limits and use of circle hooks for anglers. Ocean harvest restrictions since 1995 have led to reduced ocean harvest of spring-run Chinook salmon (i.e., Central Valley Chinook salmon ocean harvest index, ranged from 0.55 to nearly 0.80 from 1970 to 1995, and was reduced to 0.27 in 2001). DFG, NMFS, and Pacific Fishery Management Council are continuing to monitor and assess the effects of harvest of spring-run Chinook salmon, such that regulations can be refined and modified as new information becomes available.

Because adult spring-run Chinook salmon hold in pool habitat within a stream during the summer months they are vulnerable to illegal harvest (poaching). Various watershed groups have established public outreach and educational programs in an effort to reduce poaching. In addition, DFG wardens have increase enforcement against illegal harvest of spring-run Chinook salmon. The level and effect of illegal harvest on adult spring-run Chinook salmon abundance and population reproduction is unknown.

Reduced genetic diversity/integrity. Interbreeding of wild spring-run Chinook salmon with both wild and hatchery fall-run Chinook salmon has the potential to dilute and eventually eliminate the adaptive genetic distinctiveness and diversity of the few remaining naturally reproducing spring-run Chinook salmon populations (DFG 1995, Sommer et al. 2001b, Araki et al. 2007). Central Valley spring- and fall-run Chinook salmon spawning areas were historically isolated in time and space (Yoshiyama et al. 1998). However, the construction of dams has eliminated access to historical upstream spawning areas of spring-run salmon in the upper tributaries and streams of many river systems. Restrictions to upstream access, particularly on the Sacramento and Feather rivers has forced spring-run individuals to spawn in lower elevation areas also used by fall-run individuals, potentially resulting in hybridization of the two races. Hybridization between spring- and fall-run salmon has been identified as a particular concern on the Feather River where both runs co-occur and as a potential concern for restoration of salmon on the San Joaquin River downstream of Friant Dam. Management of the Feather River hatchery and brood stock selection practices have been modified in recent years (e.g., tagging early returning adult salmon showing phenotypic and run timing characteristics of spring-run Chinook salmon for subsequent use as selected brood stock and genetic testing of potential brood stock) in an effort to reduce potential hybridization as a result of hatchery operations. Consideration has also been given to using a physical weir to help segregate and isolate adults showing spring-run characteristics and later arriving fish showing characteristics of fall-run fish to reduce the risk of hybridization and redd superimposition within spawning areas of the river.

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Investigations have been undertaken to assess the potential habitat quality and availability for spring-run Chinook salmon spawning and juvenile rearing within the reaches of the Feather River upstream of Oroville Dam that could potentially be used to expand the geographic range of spring-run salmon using trap and haul techniques. On many of the other Central Valley tributaries, such as Deer and Mill creeks, the risk of hybridization is reduced by the ability of the runs to segregate geographically within the watersheds.

Further, in an effort to improve juvenile survival and the contribution of the Feather River Hatchery to the adult spring-run Chinook salmon population, the spring-run salmon program at the hatchery has released juvenile spring-run salmon far downstream of the hatchery (San Pablo Bay) in the past, which has increased the rate of straying adults migrating back upstream (DFG 2001). Recent changes in hatchery management by DFG, however, have modified juvenile planting with a greater number of juvenile fish released into the Feather River in an effort to improve imprinting and reduce straying, which may reduce potential for hybridization with spring-run salmon in other watersheds (McReynolds et al. 2006). Half of the juvenile spring-run Chinook salmon produced at the hatchery are now released in the Feather River at Live Oak as part of an experimental program designed to improve hatchery management.

Entrainment. The vulnerability of juvenile spring-run Chinook salmon to entrainment and salvage at the SWP and CVP export facilities varies in response to multiple factors, including the seasonal and geographic distribution of juvenile salmon within the Delta, operation of Delta Cross Channel gates, hydrodynamic conditions occurring within the central and southern regions of the Delta (e.g., Old and Middle rivers), and export rates. The losses of fish to entrainment mortality has been identified as an impact to Chinook salmon populations (Kjelson and Brandes 1989). Juvenile spring-run Chinook salmon tend to be distributed within the central and southern Delta where they have an increased risk of entrainment/salvage between February and May (see Table A.-3a). The effect of changing hydrodynamics within Delta channels, such as reversed flows in Old and Middle rivers resulting from SWP and CVP export operations, has the potential to increase attraction of emigrating juveniles into false migration pathways, delay emigration through the Delta, directly or indirectly increase vulnerability to entrainment at unscreened diversions, increase the risk of predation, increase movement of migrating salmon toward the export facilities, increase the risk that these fish will be entrained into the fish salvage facilities, and increase the duration of exposure to seasonally-elevated water temperatures and other depressed water quality conditions. SWP and CVP exports have been shown to affect the tidal hydrodynamics (e.g., water current velocities and direction), and the magnitude of these effects varies in response to a variety of factors, including tidal stage and magnitude of ebb and flood tides, the rate of SWP and CVP exports, operation of the Clifton Court Forebay radial gate opening, and inflow from the upstream tributaries. Chinook salmon behaviorally respond to hydraulic cues (e.g., water currents) during both upstream adult and downstream juvenile migration through the Delta. Over the past several years additional investigations have been designed using radio or acoustically-tagged juvenile Chinook salmon to monitor their migration behavior through the Delta channels and to assess the effects of changes in hydraulic cues and SWP and CVP export operations on migration. These studies are continuing (San Joaquin River

Group Authority 2007, Brandes et al. 2008, Lindley et al. 2008, MacFarlane et al. 2008a, Michel et al. 2008, North Delta Hydrodynamic and Juvenile Salmon Migration Study 2008, Perry et al. 2008).

In addition to SWP and CVP exports, over 2,200 small water diversions exist throughout the Delta, in addition to unscreened diversions located on the tributary rivers (Herren and Kawasaki 2001). The risk of entrainment is a function of the size of juvenile fish and the slot opening of the screen mesh (Tomljanovich et al. 1978, Schneeberger and Jude 1981, Zeitoun et al. 1981, Weisberg et al. 1987, C. Hanson unpubl. data). Many of the juvenile salmon migrate downstream through the Delta during the late winter or early spring when many of the agricultural irrigation diversions are not operating or are only operating at low levels. Juvenile salmon also migrate primarily in the upper part of the water column and therefore their vulnerability to an unscreened diversion located near the channel bottom is reduced. No quantitative estimates have been developed to assess the potential magnitude of entrainment losses for juvenile Chinook salmon migrating through the rivers and Delta, and the effects of these losses on the overall population abundance of returning adult Chinook salmon is unknown. Many of the larger water diversions located within the Central Valley and Delta (e.g., Glenn Colusa Irrigation District, Reclamation District 108 Wilkins Slough, Poundstone, and Sutter Mutual Water Company Tisdale pumping plants, Contra Costa Water District Old River and Alternative Intake Project, and others) have been equipped with positive barrier fish screens to reduce and avoid the loss of juvenile Chinook salmon and other fish species.

Power plants within the Plan Area have the ability to impinge juvenile Chinook salmon on the existing cooling water system intake screens. However, use of cooling water is currently low with the retirement of older units. Further, newer units are being equipped with a closed cycle cooling system that virtually eliminates the risk of impingement of juvenile salmon.

Besides direct mortality, salmon fitness may be affected by entrainment at these diversions and delays in out-migration of smolts caused by reduced or reverse flows. Delays in migration due to water management related to the SWP and CVP operations can make juvenile salmonids more susceptible to many of the threats and stressors discussed in this section, such as predation, entrainment, angling, exposure to poor water quality and toxics, and disease. The quantitative relationships among changes in Delta hydrodynamics, the behavioral and physiological response of juvenile salmon, and the increase or decrease in risk associated with other threats are unknown, but currently the subject of a number of investigations and analyses.

Exposure to toxins. Toxic chemicals have the potential to be widespread throughout the Delta, or may occur on a more localized scale in response to episodic events (i.e., stormwater runoff, point source discharges, etc.). These toxic substances include mercury, selenium, copper, pyrethroids, and endocrine disruptors with the potential to impact fish health and condition, and adversely impact salmon distribution and abundance. Concern regarding exposure to toxic substances for Chinook salmon includes both waterborne chronic and acute exposure, but also bioaccumulation and chronic dietary exposure. For example, selenium is a naturally occurring

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constituent in agricultural drainage water return flows from the San Joaquin River that is then dispersed downstream into the Delta (Nichols et al. 1986). Exposure to selenium in the diet of juvenile Chinook salmon has been shown to result in toxic effects (Saiki 1986, Saiki and Lowe 1987, Hamilton et al. 1986, 1990, Hamilton and Buhl 1990). Selenium exposure has been associated with agricultural and natural drainage within the San Joaquin River basin and refining operations adjacent to San Pablo and San Francisco bays. Other contaminants of concern for Chinook salmon include, but are not limited to, mercury, copper, oil and grease, pesticides, herbicides, ammonia, and localized areas of depressed dissolved oxygen (e.g., Stockton Deep Water Ship Channel, return flows from managed freshwater wetlands, etc.). As a result of the extensive agricultural development within the Central Valley exposure to pesticides and herbicides has been identified as a significant concern for salmon and other fish species within the Plan Area (Bennett et al. 2001). In recent years changes have been made in the composition of herbicides and pesticides used on agricultural crops in an effort to reduce potential toxicity to aquatic and terrestrial species. Modifications have also been made to water system operations and discharges related to agricultural wastewater discharges (e.g., agricultural drainage water system lock-up and holding prior to discharge) and municipal wastewater treatment and discharges. Concerns remain, however, regarding the toxicity of contaminants such as pyrethroids that adsorbed to sediments and other chemicals (e.g., including selenium and mercury, as well as other contaminants) on salmon.

Mercury and other metals such as copper have also been identified as contaminants of concern for salmon and other fish as a result of direct toxicity and impacts such as those related to acid mine runoff from sites such as Iron Mountain Mine (U.S. Environmental Protection Agency [EPA] 2006). There are problems with tissue bioaccumulation that may adversely impact the fish, but also represents a human health concern (Gassel et al. 2008). These materials originate from a variety of sources including mining operations, municipal wastewater treatment, agricultural drainage within the tributary rivers and Delta, non-point runoff, natural runoff and drainage within the Central Valley, agricultural spraying, and a number of other sources. The State Water Resources Control Board (SWRCB), Central Valley Regional Water Quality Control Board (CVRWQCB), U.S. EPA, U.S. Geological Survey (USGS), DWR, and others have ongoing monitoring programs designed to characterize water quality conditions and identify potential toxicants and contaminant exposure to Chinook salmon and other aquatic resources within the Plan Area. Programs are in place to regulate point source discharges as part of the National Pollutant Discharge Elimination System (NPDES) program as well as efforts to establish and reduce total daily maximum loads (TMDL) of various constituents entering the Delta. Changes in regulations have also been made to help reduce chemical exposure and reduce the adverse impacts to aquatic resources and habitat conditions within the Plan Area. These monitoring and regulatory programs are ongoing.

Sublethal concentrations of toxics may interact with other stressors on salmonids, such as increasing their vulnerability to mortality as a result of exposure to seasonally elevated water temperatures, predation, or disease (Werner 2007). For example, Clifford et al. (2005) found in a laboratory setting that juvenile fall-run Chinook salmon exposed to sublethal levels of a common

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pyrethroid, esfenvalerate, were more susceptible to infectious hematopoietic necrosis virus than those not exposed to esfenvalerate. Although not tested on spring-run Chinook salmon, a similar response is likely due to the physiological similarity.

Iron Mountain Mine, located adjacent to the upper Sacramento River, has been a source of trace elements and metals that are known to adversely affect aquatic organisms (Upper Sacramento River Fisheries and Riparian Habitat Advisory Council 1989). Storage limitations and limited availability of dilution flows have caused downstream copper and zinc levels to exceed salmonid tolerances and resulted in documented fish kills in the 1960s and 1970s (USBR 2004). The Environmental Protection Agency's Iron Mountain Mine remediation program has removed toxic metals in acidic mine drainage from the Spring Creek watershed with a state-of-the-art lime neutralization plant. Contaminant loading into the Sacramento River from Iron Mountain Mine has shown measurable reductions since the early 1990s.

Increased water temperature. Water temperature is among the physical factors that affect quality of habitat for salmonid adult holding, spawning and egg incubation, juvenile rearing, and migration. Adverse sublethal and lethal effects can result from exposure to elevated water temperatures at sensitive lifestages, such as during incubation or rearing. The Central Valley is the southern limit of spring-run Chinook salmon geographic distribution, so increased water temperatures are often recognized as an important stressor to California populations. Water temperature criteria for various lifestages of salmonids in the Central Valley have been developed by NMFS (2009). The tolerance of spring-run Chinook salmon to water temperatures depends on life stage, acclimation history, food availability, duration of exposure, health of the individual, and other factors such as predator avoidance (Myrick and Cech 2004, USBR 2004). Higher water temperatures can lead to physiological stress, reduced growth rate, pre-spawning mortality, reduced spawning success, and increased mortality of salmon (Myrick and Cech 2001). Temperature can also indirectly influence disease incidence and predation (Waples et al. 2008). Exposure to seasonally elevated water temperatures may occur as a result of reductions in flow, upstream reservoir operations, reductions in riparian vegetation, channel shading, local climate and solar radiation. The installation of the Shasta Temperature Control Device in 1998, in combination with reservoir management to maintain the cold water pool, has reduced many of the temperature issues on the Sacramento River. During dry years, however, the release of cold water from Shasta Dam is still limited. As the river flows further downstream, particularly during the warm spring, summer, and early fall months, water temperatures continue to increase until they reach thermal equilibrium with atmospheric conditions. As a result of the longitudinal gradient of seasonal water temperatures, the coldest temperatures and best areas for salmon spawning and rearing are typically located immediately downstream of the dam.

Increased temperature can also arise from a reduction in shade over rivers by tree removal (Watanabe et al. 2005). Because river water is typically in thermal equilibrium with atmospheric conditions by the time it enters the Delta, this issue is caused primarily from actions upstream of the Delta. As a result of the relatively wide channels that occur within the Delta, the effects of additional riparian vegetation on reducing water temperatures are minimal.

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Adult and juvenile spring-run Chinook salmon hold and rear within pools at higher elevations within the watershed. On several tributaries, pre-spawning adult mortality has been reported for adults that accumulate in high densities within a pool and are then exposed to elevated summer water temperatures. Flow reductions, resulting from natural hydrologic conditions during the summer, evapotranspiration, or surface and groundwater extractions may all contribute to exposure to elevated temperatures and increased levels of stress or mortality. In some areas groundwater wells have been used to pump cooler water into the stream to reduce oversummer temperatures. Dense riparian vegetation, streams incised into canyons that provide shading, cool water springs, and availability of deep holding pools are factors that affect summer holding and rearing conditions for spring-run Chinook salmon.

The effects of climate change and global warming patterns, in combination with changes in precipitation and seasonal hydrology in the future have been identified as important factors that may adversely affect the health and long-term viability of Central Valley spring-run Chinook salmon (Crozier et al. 2008). The rate and magnitude of these potential future environmental changes, and their effect of habitat quality and availability for spring-run Chinook salmon, however, are subject to a high degree of uncertainty.

A3.6 RELEVANT CONSERVATION EFFORTS

Results of salvage monitoring and extensive experimentation over the past several decades have lead to the identification of a large number of management actions designed to reduce or avoid the potentially adverse impacts of SWP and CVP export operations on salmon. Many of these actions have been implemented through SWRCB water quality permits (D-1485, D-1641), biological opinions issued on project export operations by NMFS, USFWS, and DFG, as part of CALFED programs (e.g., Environmental Water Account [EWA]), and as part of actions associated with Central Valley Project Improvement Act (CVPIA). As a result of these requirements multiple conservation efforts exist to enhance habitat and reduce entrainment of Chinook salmon by the SWP and CVP export facilities.

Several habitat problems that contributed to the decline of Central Valley salmonid species are being addressed and improved through restoration and conservation actions Such actions are related to ESA Section 7 consultation, Reasonable and Prudent Alternatives, addressing temperature, flow, and operations of the Central Valley and State Water Projects; EPA actions to control acid mine runoff from Iron Mountain Mine; and the CVRWB decisions requiring compliance with Sacramento River water temperature objectives. These decisions resulted in the installation of the Shasta Temperature Control Device in 1998.

Biological opinions for SWP and CVP operations (e.g., NMFS 2009a) and other federal projects involving irrigation and water diversion and fish passage, for example, have improved or minimized adverse impacts to salmon in the Central Valley. In 1992, an amendment to the authority of the CVP through the CVPIA was enacted to give protection of fish and wildlife equal priority with other CVP objectives. From this act arose several programs that have

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benefited listed salmonids. The Anadromous Fish Restoration Program (AFRP) is engaged in monitoring, education, and restoration projects designed to contribute toward doubling the natural populations of select anadromous fish species residing in the Central Valley. Restoration projects funded through the AFRP include fish passage, fish screening, riparian easement and land acquisition, development of watershed planning groups, instream and riparian habitat improvement, and gravel replenishment. The Anadromous Fish Screen Program combines federal funding with state and private funds to prioritize and construct fish screens on major water diversions mainly in the upper Sacramento River. The goal of the Water Acquisition Program is to acquire water supplies to meet the habitat restoration and enhancement goals of the Central Valley Improvement Act, and to improve the ability of the U.S. Department of the Interior to meet regulatory water quality requirements. Water has been used to improve fish habitat for Central Valley salmon, with the primary focus on listed Chinook salmon and steelhead, by maintaining or increasing instream flows (EWA) on the Sacramento River at critical times, and to reduce salmonid entrainment at the SWP and CVP export facilities through reducing seasonal diversion rates during periods when protected fish species are vulnerable to export related losses.

Two programs included under CALFED, the Ecosystem Restoration Program (ERP) and the EWA, were created to improve conditions for fish, including spring-run Chinook salmon, in the Central Valley. The ERP Implementing Agency Managers selected a proposal for directed action funding written by the Central Valley Salmonid Project Work Team, an interagency technical working group led by DFG, to develop a spring-run Chinook salmon escapement monitoring plan. Long-term funding for implementation of the monitoring plan must still be secured.

A major CALFED ERP action currently underway is the Battle Creek Salmon and Steelhead Restoration Project. The project will restore 48 miles (77 km) of habitat in Battle Creek to support steelhead and Chinook salmon spawning and juvenile rearing at a cost of over $90 million. The project includes removal of five small hydropower diversion dams, construction of new fish screens and ladders on another three dams, and construction of several hydropower facility modifications to ensure the continued hydropower operations. It is thought that this restoration effort is the largest cold water restoration project to date in North America.

The Delta Regional Ecosystem Restoration Implementation Plan (DRERIP) was formed to guide the implementation of CALFED ERP elements within the Delta (DFG 2007b). The DRERIP team has created a suite of ecosystem and species conceptual models, including spring-run Chinook salmon, that document existing scientific knowledge of Delta ecosystems. The DRERIP team has used these conceptual models to assess the suitability of actions proposed in the ERP for implementation. DRERIP conceptual models were used in the analysis of proposed BDCP conservation measures.

Recent habitat restoration initiatives sponsored and funded primarily by the ERP have resulted in plans to restore ecological function to 9,543 acres of shallow-water tidal and marsh habitats

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within the Delta. Restoration of these areas primarily involves flooding lands previously used for agriculture, thereby creating additional rearing habitat for juvenile salmonids. Similar habitat restoration is adjacent to Suisun Marsh (i.e., at the confluence of Montezuma Slough and the Sacramento River) as part of the Montezuma Wetlands project, which is intended to provide for commercial disposal of material dredged from San Francisco Estuary in conjunction with tidal wetland restoration.

The Vernalis Adaptive Management Program (VAMP) has implemented migration flow augmentation for the San Joaquin River basin to improve juvenile and adult migration for fall-run Chinook salmon (San Joaquin River Group Authority 2007). The VAMP program also includes seasonal reductions in SWP and CVP export rates that may benefit juvenile spring-run Chinook salmon during their emigration period. The program has been designed within the framework of adaptive management to improve the survival of juvenile salmonids migrating from the river through the Delta while providing an experimental framework to quantitatively evaluate the contribution of each action to salmonid survival. The incremental contribution of the VAMP conditions to overall spring-run salmon survival and adult abundance is uncertain. The VAMP experimental design and results of survival testing conducted to date is currently undergoing peer review and will also be the subject of a review conducted by the SWRCB. Based on results and recommendations from these technical reviews, the VAMP experimental design and testing program is expected to be refined.

The U.S. EPA's Iron Mountain Mine remediation involves the removal of toxic metals in acidic mine drainage from the Spring Creek Watershed with a state-of-the-art lime neutralization plant. Contaminant loading into the Sacramento River from Iron Mountain Mine has shown measurable reductions since the early 1990s. Decreasing the heavy metal contaminants that enter the Sacramento River should increase the survival of salmonid eggs and juveniles. However, during periods of heavy rainfall upstream of the Iron Mountain Mine, Reclamation substantially increases Sacramento River flows to dilute heavy metal contaminants being spilled from the Spring Creek debris dam. This rapid change in flows can cause juvenile salmonids to become stranded or isolated in side channels below Keswick Dam.

DWR's Delta Fish Agreement Program has approved approximately $49 million for projects that benefit salmon and steelhead production in the Sacramento-San Joaquin basins and Delta since the agreements inception in 1986. Delta Fish Agreement projects that benefit Central Valley spring-run Chinook salmon include water exchange programs on Mill and Deer creeks; enhanced law enforcement efforts from San Francisco Estuary upstream to the Sacramento and San Joaquin rivers and their tributaries; design and construction of fish screens and ladders on Butte Creek; and, screening of diversions in Suisun Marsh and San Joaquin River tributaries. The Spring-Run Salmon Increased Protection Project provides overtime wages for DFG wardens to focus on reducing illegal take and illegal water diversions on upper Sacramento River tributaries and adult holding areas, where the fish are vulnerable to poaching. This project covers Mill, Deer, Antelope, Butte, Big Chico, Cottonwood, and Battle creeks, and has been in effect since 1996. Through the Delta-Bay Enhanced Enforcement Program (DBEEP), initiated in 1994, a team of 10 wardens focus their enforcement efforts on salmon, steelhead, and other species of concern from the San Francisco Estuary upstream into the Sacramento and San Joaquin River basins. These two enhanced enforcement programs have likely had significant benefits to spring-run Chinook salmon attributed to DFG, although results have not been quantified.

The Mill and Deer Creek Water Exchange projects are designed to provide new wells that enable diverters to bank groundwater in place of stream flow, thus leaving water in the stream during critical migration and oversummering periods. On Mill Creek several agreements between Los Molinos Mutual Water Company, Orange Cove Irrigation District, DFG, and DWR allows DWR to pump groundwater from two wells into the Los Molinos Mutual Water Company canals to pay back Los Molinos Mutual Water Company water rights for surface water released downstream for fish. Although the Mill Creek Water Exchange project was initiated in 1990 and the agreement allows for a well capacity of 25 cfs, only 12 cfs has been developed to date. In addition, it has been determined that a base flow of greater than 25 cfs is needed during the April through June period for upstream passage of adult spring-run Chinook salmon in Mill Creek. In some years, water diversions from the creek are curtailed by amounts sufficient to provide for passage of upstream migrating adult spring-run Chinook salmon and downstream migrating juvenile steelhead and spring-run Chinook salmon.

The Feather River Hatchery is making efforts to segregate spring-run from fall-run Chinook salmon to enhance and restore the genotype of spring-run Chinook salmon in the Feather River (DFG 2001, McReynolds et al. 2006).

To help reduce the effects of the RBDD operation on migration of adult and juvenile salmonids and other species, management has changed in recent years to maintain the dam gates in the open position for a longer period of time and thereby facilitate greater upstream and downstream migration. Changes in dam operations have benefited both upstream and downstream migration by salmon and have contributed to a reduction in juvenile predation mortality. In 2009, USBR received funding for the Fish Passage Improvement Project at the RBDD to build a pumping facility to provide reliable water supply for high-valued crops in Tehama, Glenn, Colusa, and northern Yolo counties while providing year-round unimpeded fish passage. This project, which is expected to be completed in late 2012, will eliminate passage issues for spring-run Chinook salmon and other migratory species.

Seasonal constraints on sport and commercial fisheries south of Point Arena benefit spring-run Chinook salmon. DFG has implemented enhanced enforcement efforts to reduce illegal harvests. Central Valley spring-run Chinook salmon is a state listed fish that is protected by specific in-river fishing regulations.

A3.7 RECOVERY GOALS

The Public Draft Recovery Plan for Central Valley salmonids, including spring-run Chinook salmon, was released by NMFS on October 19, 2009. Although not final, the overarching goal in

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the public draft is the removal of, among other listed salmonids, spring-run Chinook salmon from the federal list of Endangered and Threatened Wildlife (NMFS 2009b). Several objectives and related criteria represent the components of the recovery goal, including the establishment of at least two viable populations within each historical diversity group, as well as other measurable biological criteria.

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69 FR 33102. Proposed Rule: Threatened and Endangered Species; Proposed listing determinations for 27 West Coast salmonid ESUs, and proposed amendments to 4(d) protective regulations for threatened ESUs. Federal Register. 69: 33102.

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70 FR 37160. 2005. Final Rule: Endangered and Threatened Species: Final Listing Determinations for 16 ESUs of West Coast Salmon, and Final 4(d) Protective Regulations for Threatened Salmonid ESUs. Federal Register. 70: 37160.

70 FR 52488. 2005. Final Rule: Endangered and Threatened Species; Designation of Critical Habitat for Seven Evolutionarily Significant Units of Pacific Salmon and Steelhead in California. Federal Register. 70: 52488.

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APPENDIX A4. CENTRAL VALLEY FALL- AND LATE FALL-RUN CHINOOK SALMON (*ONCORHYNCHUS TSHAWYTSCHA*)

A4.1 LEGAL STATUS

The Central Valley fall- and late fall-run Chinook salmon Evolutionarily Significant Unit (ESU) includes all naturally spawned populations of fall- and late fall-run Chinook salmon in the Sacramento and San Joaquin River basins and their tributaries east of Carquinez Strait, California (64 FR 50394) (see Figure A-4a and Figure A-4b, respectively). On September 16, 1999, after reviewing the best available scientific and commercial information, the National Marine Fisheries Service (NMFS) determined that listing Central Valley fall- and late fall-run Chinook salmon was not warranted. On April 15, 2004, the Central Valley fall- and late fall-run Chinook salmon ESU was identified by NMFS as a Species of Concern (69 FR 19975).

The Central Valley fall-/late fall-run Chinook salmon ESU are not listed under the California Endangered Species Act. Fall-run/late fall-run Chinook salmon are identified as a California Species of Special Concern (Moyle et al. 1995).

A4.2 SPECIES DISTRIBUTION AND STATUS

A4.2.1 Range and Status

Central Valley fall-run Chinook salmon historically spawned in all major tributaries, as well as the mainstem of the Sacramento and San Joaquin rivers (see Figure A-4a). The historical geographic distribution of Central Valley late fall-run Chinook salmon is not well understood, but is thought to be less extensive than that of fall-run (see Figure A-4b). A large percentage of fall-run Chinook spawning areas in the Sacramento and San Joaquin rivers historically inhabited the lower gradient reaches of the rivers downstream of sites now occupied by major dams, such as Shasta and Friant dams. As a result of the geographic distribution of spawning and juvenile rearing areas, fall-run Chinook salmon populations in the Central Valley were not as severely affected by early water projects that blocked access to upstream areas as were spring- and winter-runs of Chinook salmon and steelhead that used higher elevation habitat for spawning and rearing (Reynolds et al. 1993, McEwan 2001). Changes in seasonal hydrologic patterns resulting from operation of upstream reservoirs for water supplies, flood control, and hydroelectric power generation have altered instream flows and habitat conditions for fall-run Chinook salmon and other species downstream of the dams (Williams 2006).

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Figure A-4a. Central Valley Fall-Run Chinook Salmon Inland Range in California

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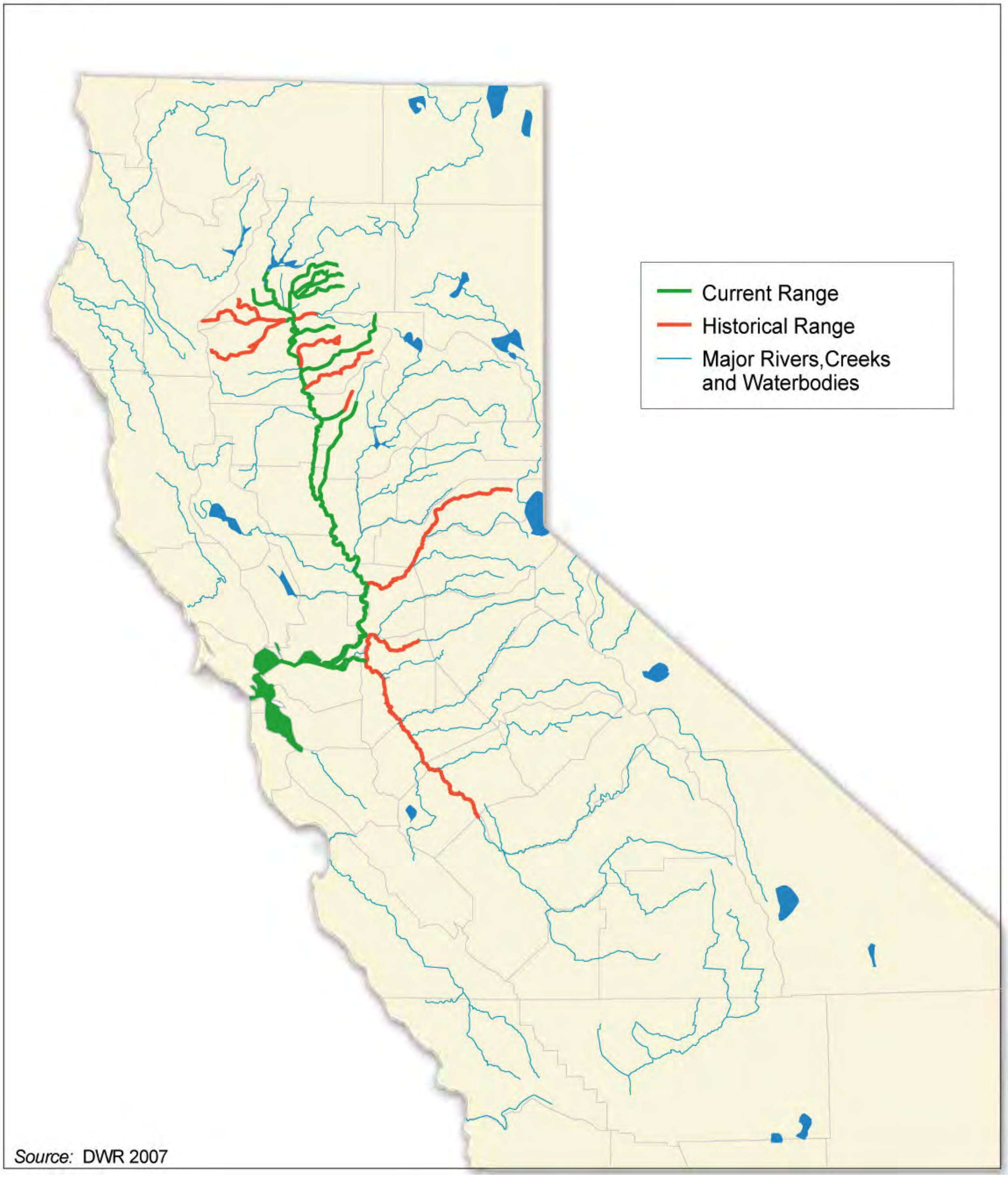

Figure A-4b. Central Valley Late Fall-Run Chinook Salmon Inland Range in California

The abundance of Central Valley fall- and late fall-run Chinook salmon escapement before 1952 is poorly documented. Reynolds et al. (1993) estimated that production of fall- and late fall-run Chinook salmon on the San Joaquin River historically approached 300,000 adults and probably averaged approximately 150,000 adults. Calkins et al. (1940) estimated fall- and late fall-run Chinook salmon abundance at 55,595 adults in the Sacramento River Basin during the period 1931-1939. In the early 1960s, adult fall- and late fall-run Chinook salmon escapement was estimated to be 327,000 fish in the Sacramento River basin (U.S. Department of Fish and Game [DFG] 1965). In the mid-1960s, fall- and late fall-run Chinook salmon escapement to the San Joaquin River Basin was estimated to be about 2,400 fish, which spawned in the San Joaquin River tributaries – the Stanislaus, Tuolumne, and Merced rivers.

Long-term trends in adult fall-run Chinook salmon escapement indicate that abundance in the Sacramento River has been consistently higher than abundance in the San Joaquin River (see Figure A-4c). Escapement on the Sacramento River has been characterized by relatively high interannual variability ranging from approximately 100,000 to over 800,000 fish. Sacramento River escapement showed a marked increase in abundance between 1990 and 2003 followed by a decline in abundance over the period from 2004 through present. In 2009 adult fall-run Chinook salmon returns to the Central Valley rivers showed a substantial decline within both the Sacramento and San Joaquin river systems. Similar declines in adult escapement were also observed for coho salmon and Chinook salmon returning to other river systems in California (MacFarlane et al. 2008).

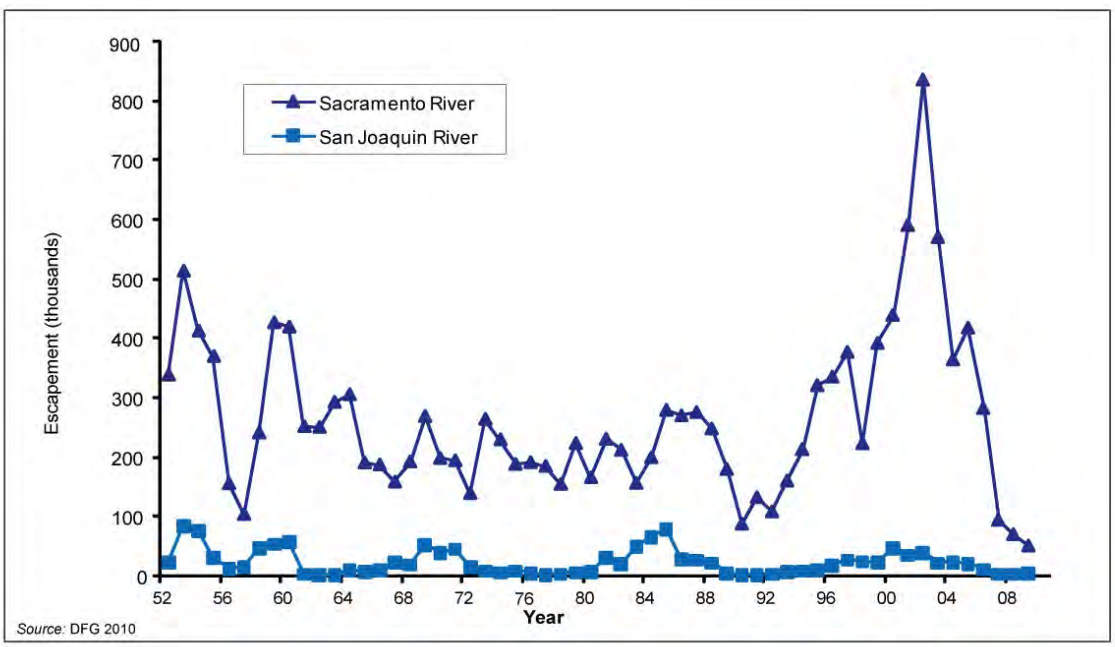

Figure A-4c. Estimated Historical Spawner Escapement of Central Valley Fall-Run Chinook Salmon (1952-2009)

A variety of factors are thought to have influenced adult escapement on both rivers, including hydrological conditions for migration, spawning, and juvenile rearing, ocean conditions, and management actions that have been implemented since the early 1990s to improve seasonal water temperatures, streamflows, modifications to Red Bluff Diversion Dam (RBDD) gate operations, improved fish passage, construction of positive barrier fish screens on larger diversions, and improved habitat conditions.

Trends in adult fall-run Chinook salmon escapement on the San Joaquin River and tributaries has been relatively low since the 1950s, ranging from several hundred adults to approximately 100,000 adults (see Figure A-4c). Results of escapement estimates have shown a relationship between adult escapement in one year and spring flows on the San Joaquin River 2.5 years earlier when the juvenile within the cohort were rearing and migrating downstream through the Delta. Adult escapement appears to be cyclical and may be related to hydrology during juvenile rearing and migration period, among other factors (San Joaquin River Group Authority 2007, DFG 2008).

Population estimates for late fall-run Chinook salmon on the San Joaquin River system are not available, but it is thought that late fall-run Chinook salmon do not regularly spawn in the tributaries of the San Joaquin River (Moyle et al. 1995). Adult escapement estimates for late fall-run Chinook salmon returning to the Sacramento River over the period from 1971 through 2009 have ranged from several hundred adults to over 40,000 adults. Adult escapement showed a general trend of declining abundance between 1971 and 1997 (see Figure A-4d). During the late 1990s and continuing through 2006 escapement has increased substantially but is characterized by high interannual variability. The 2008 and 2009 escapement estimates were lower than the previous 4 years, but were not characterized by the massive decline observed for fall-run Chinook salmon (see Figure A-4c). A number of factors have been identified that may be contributing to the observed trends and patterns in late fall-run Chinook salmon escapement to the upper Sacramento River and its tributaries.

A4.2.2 Distribution and Status in the Plan Area

The entire population of the Central Valley fall-/late fall-run Chinook salmon ESU must pass through Plan Area as adults migrating upstream and as juveniles emigrating downstream. Adult Central Valley fall-/late fall-run Chinook salmon migrating into the Sacramento River and its tributaries primarily use the western and northern portions of the Delta, whereas adults entering the San Joaquin River system to spawn use the western, central, and southern Delta as a migration pathway. Young fall-/late fall-run Chinook salmon must migrate through the Delta towards the Pacific Ocean and use the Delta, Suisun Marsh, and the Yolo Bypass for rearing to varying degrees, depending on their life stage (fry vs. juvenile) and size, river flows, and time of year.

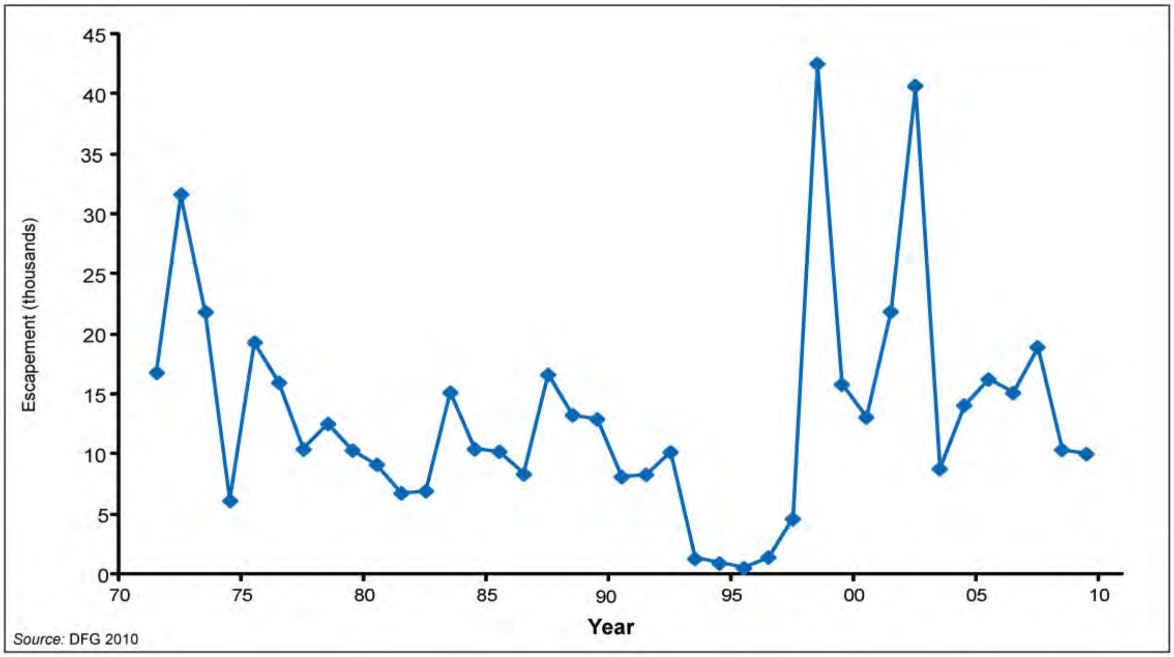

Figure A-4d. Estimated Historical Spawner Escapement of Central Valley Late Fall-Run Chinook Salmon (1971-2009) in the Sacramento River

A4.3 HABITAT REQUIREMENTS AND SPECIAL CONSIDERATIONS

Critical Habitat has not been designated for either fall-run or late fall-run Chinook salmon because the ESU is not listed under the Endangered Species Act. However, Central Valley fall- and late fall-run Chinook salmon habitats are protected under the Magnuson-Stevens Fishery Conservation and Management Act as Essential Fish Habitat. Those waters and substrate that support fall- and late fall-run Chinook salmon growth to maturity are included as Essential Fish Habitat (fall-run: see Figure A-4e; late fall-run: see Figure A-4f).

The Primary Constituent Elements (PCEs) considered essential for the conservation of Central Valley salmonids are: (1) freshwater spawning sites, (2) freshwater rearing sites, (3) freshwater migration corridors, (4) estuarine areas, (5) nearshore marine areas, and (6) offshore marine areas.

A4.3.1 Spawning Habitat

Chinook salmon spawning sites include those stream reaches with instream flows, water quality, and substrate conditions suitable to support spawning, egg incubation, and larval development. Central Valley fall-run Chinook salmon currently spawn downstream of dams on every major tributary within the Sacramento and San Joaquin river systems (with the exception of the San Joaquin River downstream of Friant Dam which is currently the subject of a settlement agreement and salmonid restoration program) in areas containing suitable environmental conditions for spawning and egg incubation.

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Figure A-4e. Central Valley Fall-Run Chinook Salmon Inland Essential Fish Habitat in California

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Figure A-4f. Central Valley Late Fall-Run Chinook Salmon Inland Essential Fish Habitat in California

Late fall-run Chinook salmon spawning is limited to the mainstem and tributaries of the Sacramento River. No Chinook salmon spawning habitat is known to occur within the Plan Area.

A4.3.2 Freshwater Rearing Habitat

Fall-/late fall-run Chinook salmon rear in streams and rivers with sufficient water flow and floodplain connectivity to form and maintain physical habitat conditions that support growth and mobility, provide suitable water quality (e.g., seasonal water temperatures) and forage species that support juvenile salmon growth, and cover such as shade, submerged and overhanging large wood, log jams, beaver dams, aquatic vegetation, large rocks and boulders, side channels, and undercut banks. Both spawning areas and migratory corridors might also function as rearing habitat for juveniles, which feed and grow before and during their outmigration. Non-natal, intermittent tributaries and seasonally inundated flood control bypasses such as the Yolo Bypass also support juvenile rearing (Sommer et al. 2001). Rearing habitat quality is strongly affected by habitat complexity, food supply, and vulnerability to predators. Some of these more complex and productive habitats with floodplains are still present in limited amounts within the Central Valley (e.g., the lower Cosumnes River, Sacramento River reaches with set-back levees [i.e., primarily located upstream of the City of Colusa]). The channeled, leveed, and riprapped river reaches and sloughs common in the Sacramento-San Joaquin River and throughout the Delta typically have low habitat diversity and complexity, low abundance of food organisms, and offer little protection from predation by fish and birds. Freshwater rearing habitat has a high conservation value because the juvenile life stage of salmonids is dependent on the function of this habitat for successful growth, survival, and recruitment to the adult population.

A4.3.3 Freshwater Migration Corridors

Freshwater migration corridors for fall-run and late fall-run Chinook salmon, including river channels, channels through the Delta, and the Bay-Delta estuary, support mobility, survival, and food supply for juveniles and adults. Migration corridors should be free from obstructions (passage barriers and impediments to migration), favorable water quantity (instream flows) and quality conditions (seasonal water temperatures), and contain natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, side channels, and undercut banks. Migratory corridors are typically downstream of the spawning area and include the lower Sacramento and San Joaquin rivers, the Delta, and the San Francisco Bay complex extending to coastal marine waters. These corridors allow the upstream passage of adults and the downstream emigration of juvenile salmon. Migratory corridor conditions are strongly affected by the presence of passage barriers, which can include dams, unscreened or poorly screened diversions, and degraded water quality. For freshwater migration corridors to function properly, they must provide adequate passage, provide suitable migration cues, reduce false attraction, avoid areas where vulnerability to predation is increased, and avoid impediments and delays in both upstream and downstream migration. For this reason, freshwater migration corridors are considered to have a high conservation value. Results of mark-recapture studies conducted using juvenile Chinook salmon released into both the Sacramento and San Joaquin rivers have shown high mortality during passage downstream through the rivers and Delta. Mortality for juvenile salmon is typically greater on the San Joaquin River than for those fish emigrating from the Sacramento River. On both rivers, mortality is typically greater in years when spring flows are reduced and water temperatures are increased. Results of survival studies have shown that closing the Delta Cross Channel gates and installation of the Head of Old River Barrier, to reduce the movement of juvenile salmon into the Delta, contribute to improved survival of emigrating juvenile Chinook salmon. Observations at the State Water Project (SWP) and the Central Valley Project (CVP) fish salvage facilities have shown that very few of the marked salmon are entrained and salvaged at the export facilities. Although factors contributing to the high juvenile mortality have not been quantified, results of anecdotal observations and results of acoustic tagging experiments suggest the exposure to adverse water quality conditions leading to mortality and vulnerability to predation mortality are two of the factors contributing to the high juvenile mortality observed in the rivers and Delta.

A4.3.4 Estuarine Areas

Estuarine migration and juvenile rearing habitats should be free of obstructions (i.e., dams and other barriers) and provide suitable water quality, water quantity (river and tidal flows), and salinity conditions to support juvenile and adult physiological transitions between fresh and salt water. Natural cover, such as submerged and overhanging large wood, aquatic vegetation, and side channels, provide juvenile and adult foraging. Estuarine areas contain a high conservation value as they function to support juvenile Chinook salmon growth, smolting, avoidance of predators, and provide a transition to the ocean environment.

A4.3.5 Ocean Habitats

Biologically productive coastal waters are an important habitat component for Central Valley fall- and late fall-run Chinook salmon. Juvenile fall-run and late fall-run Chinook salmon inhabit near-shore coastal marine waters for a period of typically 2 to 4 years before adults return to Central Valley rivers to spawn . During their marine residence Chinook salmon forage on krill, squid, and other marine invertebrates as well as a variety of fish such as northern anchovy and Pacific herring. These features are essential for conservation because, without them, juveniles cannot forage and grow to adulthood.

Results of oceanographic studies have shown the variation in ocean productivity off the West Coast within and among years. Changes in ocean currents and upwelling have been identified as significant factors affecting nutrient availability, phytoplankton and zooplankton production and the availability of other forage species in near-shore surface waters. Ocean conditions at the end of the salmon’s ocean residency period can be important, as indicated by the effect of the 1983 El Niño on the size and fecundity of Central Valley fall-run Chinook salmon (Wells et al. 2006). Although the effects of ocean conditions on Chinook salmon growth and survival have not been investigated extensively, recent observations since 2007 have shown a significant decline in the abundance of adult Chinook salmon and coho salmon returning to California rivers and streams (fall-run adult returns to the Sacramento and San Joaquin rivers were the lowest on record [Pacific Fishery Management Council 2008]) that has been hypothesized to be the result of declines in ocean productivity and associated high mortality rates during the period when these fish were rearing in near-shore coastal waters (MacFarlane et al. 2008). The importance of changes in ocean conditions on growth, survival, and population abundance of Central Valley Chinook salmon is currently undergoing further investigation.

A4.4 LIFE HISTORY

The following life history information was summarized primarily from the Final Restoration Plan for the Anadromous Fish Restoration Program (2001).

Chinook salmon exhibit two characteristic freshwater life history types (Healey 1991). Stream-type adult Chinook salmon enter freshwater months before spawning, and their offspring reside in freshwater one or more years following emergence. Ocean-type Chinook salmon, in contrast, spend significantly less time in freshwater: spawning soon after entering freshwater as adults and migrating to the ocean as juvenile fry or parr within their first year. Adequate stream flows and cool water temperatures are more critical for the survival of Chinook salmon exhibiting the stream-type life history behaviors due to their residence in freshwater both as adults and juveniles over the warmer summer months.

Central Valley fall-run Chinook salmon exhibit an ocean-type life history. Adult fall-run Chinook salmon migrate through the Delta and into Central Valley rivers from July through December and spawn from October through December (see Table A-4a). Peak spawning activity usually occurs in October and November. The life history characteristics of late fall-run Chinook salmon are not well understood; however, they are thought to exhibit an ocean-type life history. Adult late fall-run Chinook salmon migrate through the Delta and into the Sacramento River from October through April and may wait one to three months before spawning from January through April (Table A-4b). Peak spawning activity occurs in February and March. Chinook salmon typically mature between 2 and 6 years of age (Myers et al. 1998). The majority of Central Valley fall-run Chinook salmon spawn at age 3.

Information on the migration rates of Chinook salmon in freshwater is scant, most of which is taken from the Columbia River basin where migration behavior information is used to assess the effects of dams on salmon travel times and passage (Matter et al. 2003). Adult Chinook salmon upstream migration rates ranged from 29 to 32 km per day in the Snake River, a Columbia River tributary (Matter et al. 2003). Keefer et al. (2004) found migration rates of adult Chinook salmon in the Columbia River ranging between approximately 10 km per day to greater than 35 km per day. Adult Chinook salmon with sonic tags have been tracked throughout the Delta and the lower Sacramento and San Joaquin rivers (CALFED Science Program 2001).

Table A-4a. Temporal Occurrence of (a) Adult and (b) Juvenile Central Valley Fall-Run Chinook Salmon in the Sacramento River and Delta. Darker shades indicate months of greatest relative abundance.

a) Adult

Location	*Jan*	*Feb*	*Mar*	*Apr*	*May*	*Jun*	*Jul*	*Aug*	*Sep*	*Oct*	*Nov*	*Dec*
Delta[1]												
Sacramento (Sac) River Basin[2]												
San Joaquin River[2]												

b) Juvenile

Location	*Jan*	*Feb*	*Mar*	*Apr*	*May*	*Jun*	*Jul*	*Aug*	*Sep*	*Oct*	*Nov*	*Dec*
Sac River @ Red Bluff[3]												
Delta (beach seine)[4]												
Mossdale (trawl)[4]												
West Sac River (trawl)[4]												
Chipps Island (trawl)[4]												
Relative Abundance:	= High				= Medium				= Low			

[1]State Water Project and Federal Water Project fish salvage data 1981-1988.
[2]Yoshiyama et al. 1998, Moyle 2002.
[3]Martin et al. 2001
[4]USFWS 2001.

Table A4-b. Temporal Occurrence of (a) Adult and (b) Juvenile Central Valley Late Fall-Run Chinook Salmon in the Sacramento River and Delta. Darker shades indicate months of greatest relative abundance.

a) Adult

Location	*Jan*	*Feb*	*Mar*	*Apr*	*May*	*Jun*	*Jul*	*Aug*	*Sep*	*Oct*	*Nov*	*Dec*
Delta[1]												
Sacramento (Sac) River Basin[2]												

b) Juvenile

Location	*Jan*	*Feb*	*Mar*	*Apr*	*May*	*Jun*	*Jul*	*Aug*	*Sep*	*Oct*	*Nov*	*Dec*
Sac River @ Red Bluff[3]												
West Sac River (trawl)[4]												
Delta (beach seine)[4]												
Chipps Island (trawl)[4]												
Relative Abundance:	= High				= Medium				= Low			

[1]State Water Project and Federal Water Project fish salvage unpublished data 1981-1988.
[2]Yoshiyama et al. 1998; Moyle 2002.
[3]Martin et al. 2001.
[4]USFWS 2001.

These fish exhibited substantial upstream and downstream movement in a random fashion while migrating upstream several days at a time. Adult salmonids migrating upstream, particularly larger salmon such as Chinook, as described by Hughes (2004), are assumed to make greater use of pool and mid-channel habitat than they are of channel margins (Stillwater Sciences 2004). Adult salmon are thought to exhibit crepuscular behavior during their upstream migrations, primarily migrating during twilight hours (Hallock et al. 1970).

Chinook salmon spawn in clean, loose gravel in swift, relatively shallow riffles; or along the margins of deeper river reaches where suitable water temperatures, depths, and velocities favor redd construction and oxygenation of incubating eggs. Chinook salmon spawning typically occurs in gravel beds located at the tails or downstream ends of holding pools (U.S. Fish and Wildlife Service [USFWS] 1995). Egg incubation for Central Valley fall-run Chinook salmon begins with spawning in October and can extend into March. Egg incubation for late fall-run salmon occurs from January through June.

Fry emergence generally occurs at night. Upon emergence from the gravel, fry swim or are displaced downstream (Healey 1991). Fry seek streamside habitats containing beneficial aspects such as riparian vegetation and associated substrates that provide aquatic and terrestrial invertebrates, predator avoidance cover, and slower water velocities for resting (NMFS 1996). These shallow water habitats have been described as more productive juvenile salmon rearing habitat than the deeper main river channels. Higher juvenile salmon growth rates, partially due to greater prey consumption rates, as well as favorable environmental temperatures have been associated with shallow water habitats (Sommer et al. 2001).

Central Valley fall-run Chinook salmon fry (i.e., juveniles shorter than two inches long) generally emerge from December through March, with peak emergence occurring by the end of January. In general, fall-run Chinook salmon fry abundance in the Delta increases following high winter flows. Most fall-run Chinook salmon fry rear in freshwater from December through June, with emigration as smolts occurring from April through June (see Table A-4a). Smolts that arrive in the estuary after rearing upstream migrate quickly through the Delta and Suisun and San Pablo Bays. A very small number (generally considered less than 5 percent) of fall-run juveniles spend over a year in fresh water and emigrate as yearling smolts the following November through April.

Central Valley late fall-run Chinook salmon fry generally emerge from April through June. Late fall-run fry rear in freshwater from April through the following April and emigrating as smolts from November through April (see Table A-4b). Juvenile fall-run Chinook salmon outmigration through the Delta is thought to be primarily a diurnal activity, whereas outmigration of juvenile late fall-run salmon through the Delta is thought to occur primarily at night (Wilder and Ingram 2006). There are a variety of possible explanations for the difference in diel activity between races, including size of fish, water temperature, flow rate, and water clarity during downstream migration. Once downstream movement has commenced, individuals may continue this movement until reaching the estuary or they may reside in the stream for a time period that varies from a few weeks to a few months (Healey 1991). Juvenile Chinook salmon migration rates vary considerably and likely depend on physiological stage of the fish and hydrologic conditions. Kjelson et al. (1982) found Chinook salmon fry traveled downstream as fast as 30 km per day in the Sacramento River. Sommer et al. (2001) found rates ranging from approximately 1 km to greater than10 km per day in the Yolo Bypass. As juvenile Chinook salmon grow, they move into deeper water with higher current velocities, but still seek shelter and velocity refugia to minimize energy expenditures (Healey 1991). Catches of juvenile salmon in the Sacramento River near West Sacramento by the USFWS (1997) indicate that larger juveniles were captured in the main channel and smaller sized fry along the channel margins. Where the river channel is greater than 9 to 10 feet in depth, juvenile salmon tend to inhabit the surface waters (Healey 1980). Stream flow and/or turbidity increases in the upper Sacramento River basin are thought to stimulate juvenile emigration (Kjelson et al. 1982, Brandes and McLain 2001).

As Chinook salmon begin to smolt (i.e., make the physiological changes necessary for life in salt water), they are found rearing further downstream where ambient salinity reaches 1.5 to 2.5 parts per thousand (Healey 1980, Levy and Northcote 1981, USFWS unpubl. data). Within the Delta, juvenile Chinook salmon forage in shallow areas with protective cover, such as tidally influenced sandy beaches and shallow vegetated zones (Meyer 1979, Healey 1980). Cladocerans, copepods, amphipods, and dipteran larvae dipteral, as well as small arachnids and ants, are common prey items (Kjelson et al. 1982, Sommer et al. 2001).

Juvenile Chinook salmon movements within the estuarine habitat are dictated by the interaction between tidally driven saltwater intrusions through the San Francisco Bay and fresh water outflow from the Sacramento and San Joaquin rivers. Juvenile Chinook salmon follow rising tides into shallow water habitats from the deeper main channels, and return to the main channels when the tides recede (Levy and Northcote 1981, Healey 1991). Juvenile Chinook salmon were found to spend about 40 days migrating through the Delta to the mouth of San Francisco Bay and grew little in length or weight until they reached the Gulf of the Farallones Islands (MacFarlane and Norton 2002). Based on the mainly ocean-type life history observed (i.e., fall-run Chinook salmon), MacFarlane and Norton (2002) concluded that unlike other salmonid populations in the Pacific Northwest, Central Valley Chinook salmon smolts currently show little estuarine dependence and may benefit from expedited ocean entry. However, this may not be the case for emigrating fry that rear for a longer period within the Delta and estuary before emigrating to coastal marine waters. In addition, it has been hypothesized that changes in habitat conditions within the Delta over the past century may have resulted in a reduction in extended juvenile salmon rearing when compared to periods during which habitat for juvenile fall-run and late fall-run salmon rearing was more suitable.

Central Valley Chinook salmon begin their ocean life in the coastal marine waters of the Gulf of the Farallones from where they distribute north and south along the continental shelf primarily between Point Conception and Washington State (Healey 1991). Upon reaching the ocean, juvenile Chinook salmon feed on larval and juvenile fishes, plankton, and terrestrial insects (Healey 1991, MacFarlane and Norton 2002). Chinook salmon grow rapidly in the ocean environment with growth rates dependent on water temperatures and food availability (Healey 1991). The first year of ocean life is considered a critical period of high mortality for Chinook salmon that largely determines survival to harvest or spawning (Beamish and Mahnken 2001, Quinn 2005).

Recovery of coded-wire tagged Chinook salmon from the Feather River Hatchery in the ocean recreational and commercial fisheries (Pacific States Marine Fisheries Commission Regional Mark Information System database) indicates that Central Valley fall-run Chinook salmon adults are broadly distributed along the Pacific Coast from Northern Oregon to Monterey. Recovery of late fall-run coded-wire tagged Chinook salmon from the Coleman Hatchery in the ocean recreational and commercial fisheries (Pacific States Marine Fisheries Commission Regional Mark Information System database) indicates that Central Valley late fall-run Chinook salmon adults are the most broadly distributed along the Pacific Coast of the Central Valley salmon, ranging from British Columbia to Monterey.

Like other ocean-type Chinook salmon, Central Valley fall- and late fall-run Chinook salmon remain near the coast throughout their ocean life (Healey 1983, 1991, Myers et al. 1984). Central Valley fall- and late fall-run Chinook salmon remain in the ocean for 2 to 5 years. Fall-run Chinook salmon mature in the ocean before returning to freshwater to spawn. Late fall-run Chinook salmon may return to freshwater as immature adults as indicated by a 1 to 3 month delay in spawning once reaching the spawning grounds.

A4.5 THREATS AND STRESSORS

The following have been identified as important threats and stressors to fall- and late fall-run Chinook salmon (without priority). Additionally, recent record low numbers of fall-run Chinook salmon adult returns to the Central Valley (Pacific Fishery Management Council 2008) suggest that ocean conditions may be an important stressor to the ESU (MacFarlane et al. 2008), although the mechanisms driving this potential effect are not well understood.

Reduced staging and spawning habitat. Access to the upper extent of the historical upstream spawning habitat for fall- and late fall-run Chinook salmon (see Figures A-4a and A-4b, respectively) has been eliminated or degraded by man-made structures (e.g., dams and weirs) associated with water storage and conveyance, flood control, and diversions and exports for, municipal, industrial, agricultural, and hydropower purposes (Yoshiyama et al. 1998). Because spawning locations of fall- and late fall-run Chinook salmon are typically in the lower reaches of rivers, fall-/late fall-run Chinook salmon have been less affected by dam construction relative to other Central Valley salmonids. Spawning habitat for fall- and late fall-run Chinook salmon is still widely distributed within the Sacramento River basin, but more limited within the San Joaquin River basin.

Upstream diversions and dams have decreased downstream flows and altered the seasonal hydrologic patterns. These factors have been identified as contributing to delays in upstream migration by adults, increased mortality of out-migrating juveniles, and responsible for making some streams uninhabitable by fall-/late fall-run salmon (Yoshiyama et al. 1998, California Department of Water Resources [DWR] 2005). Dams and reservoir impoundments and associated reductions in peak flows have blocked gravel recruitment and reduced flushing of sediments from existing gravel beds, reducing and degrading natal spawning grounds. Further, reduced flows can lower attraction cues for adult spawners, causing straying and delays in spawning (DWR 2005). Adult salmon migration delays can reduce fecundity and increase susceptibility to disease and harvest (McCullough 1999) Because fall-run Chinook salmon spawn shortly after entering freshwater, a delay in migration can have large impacts to pre-spawning mortality and spawning success relative to other races of Chinook salmon.

The RBDD located on the Sacramento River has been identified as a barrier and impediment to adult upstream migration. Although the RBDD is equipped with fish ladders, migration delays were reported when the dam gates are closed. Mortality as a result of increased predation by Sacramento pikeminnow on juvenile salmon passing downstream through the fish ladder has also been identified as a factor affecting abundance of salmon produced on the Sacramento River (Hallock 1991). To help reduce the effects of dam operation on migration of adult and juvenile salmonids and other species, management changes have occurred in recent years to maintain the dam gates in the open position for a longer period of time, facilitating greater upstream and downstream migration. Changes in dam operations have benefited both upstream and downstream migration and have contributed to a reduction in juvenile predation mortality.

Reduced rearing and out-migration habitat. Natural migration corridors for juvenile fall- and late fall-run Chinook salmon consist of a mosaic of complex habitat types, including stream banks, floodplains, marshes, and shallow water areas which are used as rearing habitat during out-migration. Much of the Sacramento and San Joaquin river corridor and Delta have been leveed, channelized, and modified with riprap for flood protection, thereby reducing and degrading the quality and availability of natural habitat for rearing and emigrating juvenile Chinook salmon (Brandes and McLain 2001). Juvenile out-migration delays associated with man-made passage impediments can reduce fitness and increase susceptibility to diversion screen impingement, entrainment, disease, and predation. Modification of natural flow regimes from upstream reservoir operations has resulted in dampening of the hydrograph, reducing the extent and duration of seasonal floodplain inundation and other flow-dependent habitat used by migrating juvenile Chinook salmon (70 FR 52488, Sommer et al. 2001, DWR 2005). Recovery of floodplain habitat in the Central Valley has been found to contribute to increases in production in Chinook salmon (Sommer et al. 2001). Reductions in flow rates have resulted in increased water temperature and residence time, and reduced dissolved oxygen levels in localized areas of the Delta (e.g., Stockton Deep Water Ship Channel). Reduced dissolved oxygen levels in the San Joaquin River during summer and fall have been identified as a water quality barrier to salmon migration (Central Valley Regional Water Quality Control Board 2007).

Predation by non-native species. Predation on juvenile salmon by non-native fish has been identified as an important threat to fall- and late fall-run Chinook salmon in areas with high densities of non-native fish (e.g., small and large mouth bass, striped bass, and catfish) that prey on out-migrating juvenile salmon (Lindley and Mohr 2003). Non-native aquatic vegetation, such as Brazilian waterweed and water hyacinth, provide suitable habitat for non-native predators (Nobriga et al. 2005, Brown and Michniuk 2007). Predation risk may covary with increased temperatures. Metabolic rates of non-native, predatory fish increase with increasing water temperatures based on bioenergetic studies (Loboschefsky et al. 2009, Miranda et al. 2010). Upstream gravel pits and flooded ponds attract non-native predators because of their depth and lack of cover for juvenile salmon (DWR 2005). The low spatial complexity and reduced habitat diversity (e.g., lack of cover) of channelized waterways within the rivers and Delta reduces refuge space of salmon from predators (Raleigh et al. 1984, Missildine et al. 2001, 70 FR 52488). A major concern is the potential invasion of the Delta by the highly predatory northern pike. The pike, recently present in Lake Davis on the Feather River, is currently the target of a major eradication effort (DFG 2007a). If eradication fails and pike escape downstream to the Delta, they would likely be present in areas inhabited by fall- and late fall-run Chinook salmon.

Predation by native species, such as the Sacramento pikeminnow, in the Sacramento River at locations such as the RBDD has also been identified as a potentially significant source of mortality on juvenile salmonids.

Harvest. Commercial or recreational harvest of fall- and late fall-run Chinook salmon populations in the ocean and inland fisheries has been a subject of management actions by the California Fish and Game Commission and the Pacific Fishery Management Council. Coastal marine waters offshore of San Francisco Bay are a mixed stock fishery comprised of both wild and hatchery produced salmon. As a result of differences in survival rates for eggs incubation, rearing, and emigration, juvenile salmon produced in streams and rivers have relatively low survival rates compared to Central Valley salmon hatcheries, which have relatively high survival rates. Therefore, naturally reproducing Chinook salmon populations are less able to withstand high harvest rates compared to hatchery-based stocks (Knudsen et al. 1999). The ocean fishery for fall- and late fall-run Chinook salmon is supplemented by hatchery enhancement programs (USFWS 1999, Williams 2006). The Coleman National Fish Hatchery produces approximately 12 million fall-run and one million late fall-run Chinook salmon juveniles each year to mitigate for habitat loss from construction of Shasta and Keswick dams (Williams 2006). Fall-run Chinook salmon are also produced at hatcheries on the Feather, American, Mokelumne, and Merced rivers (Williams 2006). Harvest as a result of the commercial and recreational fisheries may ultimately be having detrimental effects to wild spawners in this mixed stock fishery, but few data are available. Commercial fishing for salmon is managed by the Pacific Fishery Management Council and is constrained by time and area to meet the Sacramento River winter-run ESA consultation standard and restrictions requiring minimum size limits and use of circle hooks for anglers.

Beginning in 2007, Central Valley hatcheries have implemented a proportional marking program (tagging a set percentage of salmon produced in each hatchery) that is designed to provide improved information on the effects of harvest on various stocks of Chinook salmon. The program also provides information on ocean migration patterns, growth and survival for fish released at various lifestages and locations, the contribution of hatcheries to the adult population, straying among hatcheries and watersheds, the relative contribution of in-river versus hatchery production, and other data that will assist managers in refining harvest regulations. Results of coded wire tag (CWT) mark-recapture studies and data from the proportional marking program are continually being reviewed and analyzed each year and used to modify harvest regulations and Central Valley salmon management.

Reduced genetic diversity/integrity. Artificial propagation programs (hatchery production) for fall- and late fall-run Chinook salmon in the Central Valley present multiple threats to wild (in-river spawning) Chinook salmon populations, including genetic introgression by hatchery origin fish that spawn naturally and interbreed with local wild populations (USFWS 2001, U.S. Bureau of Reclamation [USBR] 2004, Goodman 2005). It is now recognized that Central Valley hatcheries are a significant and persistent threat to wild Chinook salmon and steelhead populations and fisheries (NMFS 2009a). Interbreeding with hatchery fish contributes directly to reduced genetic diversity and introduces maladaptive genetic changes to the wild population (DFG 1995, CALFED 2004, Myers et al. 2004, Araki et al. 2007). In addition, releasing hatchery smolts downstream of hatcheries has resulted in an increase in straying rates, further reducing genetic diversity among populations (Williamson and May 2005). Central Valley hatcheries are currently undergoing a detailed review by NMFS and DFG as part of a comprehensive hatchery master plan process. Various techniques and actions have been identified for reducing the effects of hatchery production on the genetic characteristics of Chinook salmon as part of the hatchery review. These include, but are not limited to, seasonally selecting brood stock for use in the hatchery in proportion to adult escapement to the river, selecting brood stock from various age classes (including grilse) that represents the age structure of the wild population, use of tagging and genetic testing to select brood stock, increasing the number of adults used as brood stock to increase genetic diversity, reduce interbasin transfer of eggs and fry, and imprinting juveniles to reduce straying among watersheds. These and other hatchery management methods (e.g., reduction of the use of antibiotics, juvenile release strategies to reduce impacts to wild rearing juveniles, volitional releases, etc.) are expected to reduce the potential risk of hatchery production on the genetics and success of wild populations; however, artificial selection for traits that assure individual success in a hatchery setting (e.g., rapid growth and tolerance to crowding) are difficult to avoid (USBR 2004).

The potential for inter-breeding between Central Valley spring- and fall-run salmon stocks is generally identified as a genetic concern (Yoshiyama et al. 1998), however some studies indicate no evidence of natural hybridization among Chinook salmon runs despite the spatial and temporal overlap (Banks et al. 2000). Spring- and fall-run Chinook salmon were historically isolated in time and space during spawning; however, the construction of dams and reduction in flows has eliminated access to historical spawning areas of spring-run salmon in the upper tributaries and streams, forcing spring-run salmon to spawn in lower elevation areas also used by fall-run salmon (Yoshiyama et al. 1998). Although hybridization between spring- and fall-run salmon has been identified as a particular concern on the Feather River where both runs co-occur and as a potential concern for future restoration of salmon on the San Joaquin River downstream of Friant Dam, the genotypic proportions in the Butte Creek spring-run cluster farther from the fall-run versus the spring-run from Deer and Mill creeks, not closer as expected under the hybridization hypothesis (Banks et al. 2000). Deer and Mill creeks, as many of the other Central Valley tributaries, has a reduced risk of hybridization by the ability of the runs to segregate geographically within the watersheds.

Entrainment. The vulnerability of fall- and late fall-run Chinook salmon to entrainment and salvage at the SWP and CVP export facilities varies in response to multiple factors, including the seasonal and geographic distribution of juvenile salmon within the Delta, operation of Delta Cross Channel gates and Head of Old River Barrier, hydrodynamic conditions occurring within the central and southern regions of the Delta (e.g., Old and Middle rivers), and export rates. The losses of fish to entrainment mortality has been identified as an impact to Chinook salmon populations (Kjelson and Brandes 1989). Juvenile fall-run Chinook salmon tend to be distributed within the central and southern Delta where they have an increased risk of entrainment/salvage between January and April (see Table A-4a). Juvenile late fall-run Chinook salmon tend to be distributed within the Delta primarily between December and January and again between April and May (see Table A-4b). The effect of changing hydrodynamics within Delta channels, such as reversed flows in Old and Middle rivers resulting from SWP and CVP export operations, has the potential to increase attraction of emigrating juveniles into false migration pathways, delay emigration through the Delta, and directly or indirectly increase vulnerability to entrainment at unscreened diversions, risk of predation, and the duration of exposure to seasonally elevated water temperatures and other water quality conditions. SWP and CVP exports have been shown to affect the tidal hydrodynamics (e.g., water current velocities and direction). The magnitude of these hydrodynamic effects vary in response to a variety of factors that include the tidal stage and magnitude of ebb and flood tides, the rate of SWP and CVP exports, operation of the Clifton Court Forebay radial gate opening, and inflow from the upstream tributaries. Chinook salmon behaviorally respond to hydraulic cues (e.g., water currents) during both upstream adult and downstream juvenile migration through the Delta. During the past several years additional investigations have been designed using radio or acoustically-tagged juvenile Chinook salmon to monitor their migration behavior through the Delta channels and assess the effects of changes in hydraulic cues and SWP and CVP export operations on migration (Holbrook et al. 2009, Perry et al. 2010, San Joaquin River Group Authority 2010). These studies are ongoing

Besides direct mortality, salmon fitness may be affected by entrainment at diversions and delays in out-migration of smolts caused by reduced or reverse flows. Delays in migration due to water operations related to SWP and CVP export facilities can make juvenile salmonids more susceptible to many of the threats and stressors discussed in this section, such as predation, entrainment, harvest, exposure to toxins, etc. The quantitative relationships among changes in Delta hydrodynamics, the behavioral and physiological response of juvenile salmon, and the increase or decrease in risk associated with other threats is unknown, but the subject of a number of current investigations and analyses.

In addition to SWP and CVP exports, over 2,200 small water diversions exist throughout the Delta, in addition to unscreened diversions located on the tributary rivers (Herren and Kawasaki 2001). The risk of entrainment is a function of the size of juvenile fish and the slot opening of the screen mesh (Tomljanovich et al. 1978, Schneeberger and Jude 1981, Zeitoun et al. 1981, Weiserg et al. 1987, C. Hanson unpubl. data). Many of the juvenile salmon migrate downstream through the Delta during the late winter or early spring when many of the agricultural irrigation diversions are not operating or are only operating at low levels. Juvenile salmon also migrate primarily in the upper part of the water column and, as a result, their vulnerability to an unscreened diversion located near the channel bottom is reduced. No quantitative estimates have been developed to assess the potential magnitude of entrainment losses for juvenile Chinook salmon migration through the rivers and Delta, or the effects of these losses on the overall population abundance of returning adult fall-run and late fall-run Chinook salmon. Many of the larger water diversions located within the Central Valley and Delta (e.g., Glenn Colusa Irrigation District, Reclamation District 108 Wilkins Slough and Poundstone pumping plants, Sutter Mutual Water Company Tisdale pumping plant, Contra Costa Water District Old River and Alternaitve Intake Project intake, and others) have been equipped with positive barrier fish screens to reduce and avoid the loss of juvenile Chinook salmon and other fish species.

Power plants within the Plan Area have the ability to impinge juvenile Chinook salmon on the existing cooling water system intake screens. However, use of cooling water is currently low with the retirement of older units. Further, newer units are being equipped with a closed cycle cooling system that virtually eliminates the risk of impingement of juvenile salmon.

Exposure to toxins. Toxic chemicals have the potential to be wide spread throughout the Delta, or may occur on a more localized scale in response to episodic events (stormwater runoff, point source discharges, etc.). These toxic substances include mercury, selenium, copper, pyrethroids, and endocrine disruptors with the potential to impact fish health and condition, and adversely impact salmon distribution and abundance. Concern regarding exposure to toxic substances for Chinook salmon includes waterborne chronic and acute exposure, as well as bioaccumulation and chronic dietary exposure. For example, selenium is a naturally occurring constituent in agricultural drainage water return flows from the San Joaquin River that is subsequently dispersed downstream into the Delta (Nichols et al. 1986). Exposure to selenium in the diet of juvenile Chinook salmon has been shown to result in toxic effects (Saiki 1986, Hamilton et al. 1986, 1990, Saiki and Lowe 1987, Hamilton and Buhl 1990). Selenium exposure has been associated with agricultural and natural drainage within the San Joaquin River basin and refining operations adjacent to San Pablo and San Francisco bays. Other contaminants of concern for Chinook salmon include, but are not limited to: mercury, copper, oil and grease, pesticides, herbicides, and ammonia. As a result of the extensive agricultural development within the Central Valley, exposure to pesticides and herbicides has been identified as a significant concern for salmon and other fish species within the Plan Area (Bennett et al. 2001). Mercury and other metals such as copper have also been identified as contaminants of concern for salmon and other fish as a result of direct toxicity and tissue bioaccumulation adversely impacting fish (U.S. Environmental Protection Agency [EPA] 2006), as well as representing a human health concern

(Gassel et al. 2008). These materials originate from a variety of sources including mining operations, municipal wastewater treatment, agricultural drainage within the tributary rivers and Delta, non-point runoff, natural runoff and drainage within the Central Valley, agricultural spraying, and a number of other sources. The State Water Resources Control Board (SWRCB), Central Valley Regional Water Quality Control Board (CVRWQCB), U.S. EPA, U.S. Geological Survey (USGS), DWR, and others have ongoing monitoring programs designed to characterize water quality conditions and identify potential toxicants and contaminant exposure to Chinook salmon and other aquatic resources within the Plan Area. Programs are in place to regulate point source discharges as part of the National Pollutant Discharge Elimination System (NPDES) as well as programs to establish and reduce total daily maximum loads (TMDL) of various constituents entering the Delta. Changes in regulations have also been made to help reduce chemical exposure and reduce the adverse impacts to aquatic resources and habitat conditions within the Plan Area. These monitoring and regulatory programs are ongoing.

Sublethal concentrations of toxins may interact with other stressors to cause adverse effects to salmonids, such as increasing their vulnerability to mortality as a result of exposure to seasonally elevated water temperatures, predation, or disease (Werner 2007). For example, Clifford et al. (2005) found in a laboratory setting that juvenile fall-run Chinook salmon exposed to sublethal levels of a common parathyroid, esfenvalerate, were more susceptible to infectious hematopoietic necrosis virus than those not exposed to esfenvalerate. Juvenile Chinook salmon have a relatively extended period of Delta and estuarine residence of several months (Quinn 2005), which increases exposure and susceptibility to toxic substances in these areas. Adult migrating Chinook salmon may be less affected by these toxins because they are not feeding, and thus not bioaccumulating toxic exposure, and they are moving rapidly through the system.

Iron Mountain Mine, located adjacent to the upper Sacramento River, has been a source of trace elements and metals that are known to adversely affect aquatic organisms (Upper Sacramento River Fisheries and Riparian Habitat Advisory Council 1989). Storage limitations and limited availability of dilution flows have caused downstream copper and zinc levels to exceed salmonid tolerances and resulted in documented fish kills in the 1960s and 1970s (USBR 2004). The Environmental Protection Agency's Iron Mountain Mine remediation program has removed toxic metals in acidic mine drainage from the Spring Creek watershed with a state-of-the-art lime neutralization plant. Contaminant loading into the Sacramento River from Iron Mountain Mine has shown measurable reductions since the early 1990s.

Increased water temperature. Water temperature is among the physical factors that affect quality of habitat for salmonid adult holding, spawning and egg incubation, juvenile rearing, and migration. Adverse sublethal and lethal effects can result from exposure to elevated water temperatures at sensitive lifestages, such as during incubation or rearing. The Central Valley is the southern limit of Chinook salmon geographic distribution. As a result, increased water temperatures are often recognized as a particularly important stressor to California populations. Water temperature criteria for various lifestages of salmonids in the Central Valley have been developed by NMFS (2009). The tolerance of fall-run and late fall-run Chinook salmon to water temperatures depends on life stage, acclimation history, food availability, duration of exposure, health of the individual, and other factors such as predator avoidance (Myrick and Cech 2004, USBR 2004). Higher water temperatures can lead to physiological stress, reduced growth rate, delayed passage, *in vivo* egg mortality of spawning adults, pre-spawning mortality, reduced spawning success, and increased mortality of salmon (Myrick and Cech 2001). Temperature can also indirectly influence disease incidence and predation (Waples et al. 2008). Exposure to seasonally elevated water temperatures may occur as a result of reductions in flow as a result of upstream reservoir operations, reductions in riparian vegetation, channel shading, local climate and solar radiation. The installation of the Shasta Temperature Control Device in 1998, in combination with reservoir management to maintain the cold water pool, has reduced many of the temperature issues on the Sacramento River. During dry years, however, the release of cold water from Shasta Dam is still limited. As the river flows further downstream, particularly during the warm spring, summer, and early fall months, water temperatures continue to increase until they reach thermal equilibrium with atmospheric conditions. As a result of the longitudinal gradient of seasonal water temperatures, the coldest water and, therefore, the best areas for salmon spawning and rearing are typically located immediately downstream of the dam.

Increased temperature can also arise from a reduction in shade over rivers by tree removal (Watanabe et al. 2005). Because river water is typically in thermal equilibrium with atmospheric conditions by the time it enters the Delta, this issue is caused primarily from actions upstream of the Delta. As a result of the relatively wide channels that occur within the Delta, the effects of additional riparian vegetation on reducing water temperatures are minimal. The effects of climate change and global warming patterns, in combination with changes in precipitation and seasonal hydrology in the future have been identified as important factors that may adversely affect the health and long-term viability of Central Valley spring-run Chinook salmon (Crozier et al. 2008). The rate and magnitude of these potential future environmental changes, and their effect of habitat quality and availability for fall- and late fall-run Chinook salmon, however, are subject to a high degree of uncertainty.

A4.6 RELEVANT CONSERVATION EFFORTS

Results of salvage monitoring and extensive experimentation over the past several decades have led to identification of a large number of management actions designed to reduce or avoid the potentially adverse impacts of SWP and CVP export operations on salmon. Many of these actions have been implemented through SWRCB water quality permits (D-1485, D-1641), biological opinions issued on project export operations by NMFS, USFWS, and DFG, as part of CALFED programs such as the Environmental Water Account (EWA), and as part of CVPIA actions. As a result of these requirements, multiple conservation efforts exist to reduce entrainment of Chinook salmon by the SWP and CVP export facilities.

Several habitat problems that contributed to the decline of Central Valley salmonid species are being addressed and improved through restoration and conservation actions related to Endangered Species Act (ESA) Section 7 consultation, Reasonable and Prudent Alternatives, addressing temperature, flow, and operations of the Central Valley and State Water Projects, the Central Valley Regional Water Board decisions requiring compliance with Sacramento River water temperature objectives that resulted in installation of the Shasta Temperature Control Device in 1998, and EPA actions to control acid mine runoff from Iron Mountain Mine.

Biological opinions for SWP and CVP operations (e.g., NMFS 2009b) and other federal projects involving irrigation and water diversion and fish passage, for example, have improved or minimized adverse impacts to salmon in the Central Valley. In 1992, an amendment to the authority of the CVP through the Central Valley Project Improvement Act was enacted to give protection of fish and wildlife equal priority with other Central Valley Project objectives. From this act arose several programs that have benefited listed salmonids. The Anadromous Fish Restoration Program (AFRP) is engaged in monitoring, education, and restoration projects designed to contribute toward doubling the natural populations of select anadromous fish species residing in the Central Valley. Restoration projects funded through the AFRP include fish passage, fish screening, riparian easement and land acquisition, development of watershed planning groups, instream and riparian habitat improvement, and gravel replenishment. The Anadromous Fish Screen Program combines federal funding with state and private funds to prioritize and construct fish screens on major water diversions mainly in the upper Sacramento River. The goal of the Water Acquisition Program is to acquire water supplies to meet the habitat restoration and enhancement goals of the Central Valley Project Improvement Act (CVPIA), and to improve the ability of the U.S. Department of the Interior to meet regulatory water quality requirements. Water has been used to improve fish habitat for Central Valley salmon, with the primary focus on listed Chinook salmon and steelhead that provide incidental benefits to fall-run and late fall-run Chinook salmon by maintaining or increasing instream flows (Environmental Water Account) on the Sacramento River and the San Joaquin River at critical times, and to reduce salmonid entrainment at the SWP and CVP export facilities through reducing seasonal diversion rates during periods when protected fish species are vulnerable to export related losses.

Two programs included under CALFED, the Ecosystem Restoration Program (ERP) and the Environmental Water Account (EWA), were created to improve conditions for fish, including fall-run and late fall-run Chinook salmon, in the Central Valley. Restoration actions implemented by the ERP include the installation of fish screens, modification of barriers to improve fish passage, habitat acquisition, and instream habitat restoration. The majority of these actions address key factors and stressors affecting listed salmonids that incidentally benefit fall-run and late fall-run Chinook salmon. Additional ongoing actions include efforts to enhance fishery monitoring, and improvements to hatchery management to support salmonid production through hatchery releases.

A major CALFED ERP action currently underway is the Battle Creek Salmon and Steelhead Restoration Project. The project will restore 48 miles (77 km) of habitat in Battle Creek to support steelhead and Chinook salmon spawning and juvenile rearing at a cost of over $90 million. The project includes removal of five small hydropower diversion dams, construction of new fish screens and ladders on another three dams, and construction of several hydropower facility modifications to ensure the continued hydropower operations. It is thought that this restoration effort is the largest cold water restoration project to date in North America.

The Delta Regional Ecosystem Restoration Implementation Plan (DRERIP) was formed to guide the implementation of CALFED ERP elements within the Delta (DFG 2007b). The DRERIP team has created a suite of ecosystem and species conceptual models, including fall-/late fall-run Chinook salmon, that document existing scientific knowledge of Delta ecosystems. The DRERIP team has used these conceptual models to assess the suitability of actions proposed in the ERP for implementation. DRERIP conceptual models were used in the analysis of proposed BDCP conservation measures.

The Vernalis Adaptive Management Program (VAMP) has implemented migration flow augmentation for the San Joaquin River basin to improve juvenile and adult migration for fall-run Chinook salmon (San Joaquin River Group Authority 2010). The VAMP program also includes seasonal reductions in SWP and CVP export rates and installation of the Head of Old River Barrier to further improve the survival of downstream migrating salmon. The program has been designed within the framework of adaptive management to improve the survival of juvenile salmon migrating from the river through the Delta while also providing an experimental framework to quantitatively evaluate the contribution of each action to fall-run Chinook salmon survival. Preliminary results of the VAMP survival studies have shown evidence that juvenile Chinook salmon survival is positively correlated with San Joaquin River flows during the spring emigration period, however no statistically significant relationship between juvenile salmon survival and SWP/CVP exports has been detected. The range of flows and SWP/CVP export rates that can be tested under the VAMP experimental design is relatively small (e.g., river flows from approximately 2000 to 7000 cfs with SWP and CVP export rates ranging from 1500 to 3000 cfs). In addition, during the experimental period installation of the Head of Old River Barrier has been precluded by federal court order to protect delta smelt. As a result of these and other factors, the level of additional protection that the VAMP has provided to naturally produced Chinook salmon during emigration downstream from the San Joaquin River and Delta, and the incremental contribution of the VAMP conditions to overall salmon survival and adult abundance, is uncertain. The VAMP experimental design and results of survival testing conducted to date is currently undergoing peer review and will also be the subject of a review conducted by the SWRCB. Based on results and recommendations from these technical reviews, the VAMP experimental design and testing program, as well as flow management for juvenile salmon migration on the San Joaquin River, is expected to be refined.

A4.7 RECOVERY GOALS

Because fall-run and late fall-run Chinook salmon are not listed for protection under either the federal or California ESA formal recovery goals will not be established. As part of other fishery management programs, such as the CVPIA and the SWRCB salmon doubling goal, goals and objectives have been established for Central Valley Chinook salmon.

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APPENDIX A5. DELTA SMELT *(HYPOMESUS TRANSPACIFICUS)*

A5.1 LEGAL STATUS

The U.S. Fish and Wildlife Service (USFWS) determined that delta smelt warranted listing as a threatened species under the federal Endangered Species Act (ESA) effective April 5, 1993. The listing decision was based on a substantial reduction in delta smelt abundance within the Bay-Delta estuary in a variety of fishery sampling programs, threats to its habitat, and the inadequacy of regulatory mechanisms to protect delta smelt (58 FR 12863). The delta smelt was listed as a threatened species under the California ESA on December 9, 1993. The Sacramento-San Joaquin Delta Native Fishes Recovery Plan, which includes delta smelt, was completed in 1996 (USFWS 1996). The recovery plan identified a number of specific criteria related to delta smelt abundance and geographic distribution that had to be met as a condition for assessing whether the species could be considered to have recovered and be eligible for potential de-listing. During the late 1990s and early 2000s, delta smelt met the criteria set out in the recovery plan. In response to several law suits, the USFWS conducted a five-year status review for delta smelt and, on March 31, 2004, concluded that delta smelt abundance remained relatively low compared to historical levels and that many of the threats to the species identified at the time of listing were still in existence, precluding de-listing of the species (USFWS 2004). Subsequent indices of delta smelt abundance based on results of California Department of Fish and Game (DFG) fishery sampling have shown that the abundance of delta smelt, in addition to other pelagic fish species, has declined substantially in recent years (collectively referred to as pelagic organism decline [POD]), reaching record low levels of abundance. In March 2006, the Center for Biological Diversity, the Bay Institute, and the Natural Resources Defense Council filed an emergency petition with USFWS requesting that the status of delta smelt be elevated from threatened to endangered under the federal ESA (Center for Biological Diversity et al. 2006). The emergency petition was not approved by USFWS. However, the USFWS on July 10, 2008 announced in a 90-day finding that consideration for reclassification of delta smelt was warranted and, after an information collection stage, a status review would be initiated (73 FR 39639). On April 7, 2010, the USFWS ruled that the change in status from threatened to endangered was warranted, but precluded by other higher priority listing actions (75 FR 17667).

An emergency petition was filed in February 2007 to the California Fish and Game Commission to elevate the status of delta smelt from threatened to endangered under the California ESA (the Bay Institute et al. 2007). On March 4, 2009, the California Fish and Game Commission elevated the status of delta smelt to endangered under the California ESA.

A5.2 SPECIES DISTRIBUTION AND STATUS

A5.2.1 Range and Status

Delta smelt are endemic to the Bay-Delta estuary (see Figure A-5a) (Moyle 2002). The geographic distribution of delta smelt occurs primarily downstream of Isleton on the Sacramento River, downstream of Mossdale on the San Joaquin River, and Suisun Bay and Suisun Marsh. Delta smelt have also been collected in the Petaluma and Napa rivers. Delta smelt adults occur primarily in the tidally influenced low salinity region of Suisun Bay and the freshwater regions of the Delta and the Sacramento and San Joaquin rivers (Moyle 2002). The downstream location of the low salinity habitat for delta smelt is typically located in Suisun Bay, extending further to the west in response to high delta outflows and further to the east in response to low delta outflows. Delta smelt have been collected in Carquinez Strait, the Napa River, and even as far downstream as San Pablo Bay in wet years (Moyle 2002). In September or October, adults begin upstream movement towards freshwater sloughs and channels of the western Delta to spawn. Spawning takes place between February and July, but appears to be greatest during mid-April and May (Bennett 2005). Spawning can occur in the Sacramento River as far upstream as Sacramento, the Mokelumne River system, and the Cache Slough region (Moyle 2002). Since 1982, the center of adult delta smelt abundance in the fall has been the northwestern Delta in the channel of the Sacramento River near Decker Island. In any month, two or more life stages (adult, larvae, and juveniles) of delta smelt have the potential to be present in Suisun Bay (Wang 1991, DWR and USBR 1994, Moyle 2002). Delta smelt are also found seasonally in Suisun Marsh.

Results of multiple long-term monitoring programs that include a variety of sampling methods have consistently shown that the abundance of delta smelt inhabiting the Bay-Delta system has been extremely low since 2001 (see Figure A-5b). The observed decline in delta smelt abundance is consistent with declines of other pelagic species in the Delta. The decline in pelagic fish species abundance within the estuary has prompted a large set of investigations funded by the Interagency Ecological Program (IEP), CALFED, and other sources to identify and examine various factors that may be causing these declines (Resources Agency 2007). Indices of delta smelt abundance in the fall, as reflected in the DFG fall mid-water trawl (FMWT) surveys were the lowest on record in 2006 (see Figure A-5b).

A5.2.2 Distribution and Status in the Plan Area

Delta smelt occur throughout the Delta, Suisun Bay and Suisun Marsh, and within the Napa and Petaluma rivers (see Figure A-5a). Multiple permanent sites sampled by DFG and USFWS using many different collection methods intended to sample various life history stages of delta smelt provide a basis for examining trends in abundance of delta smelt under different hydrologic conditions as well as the temporal and geographic distribution of the species within and among years (see Figure A-5c).

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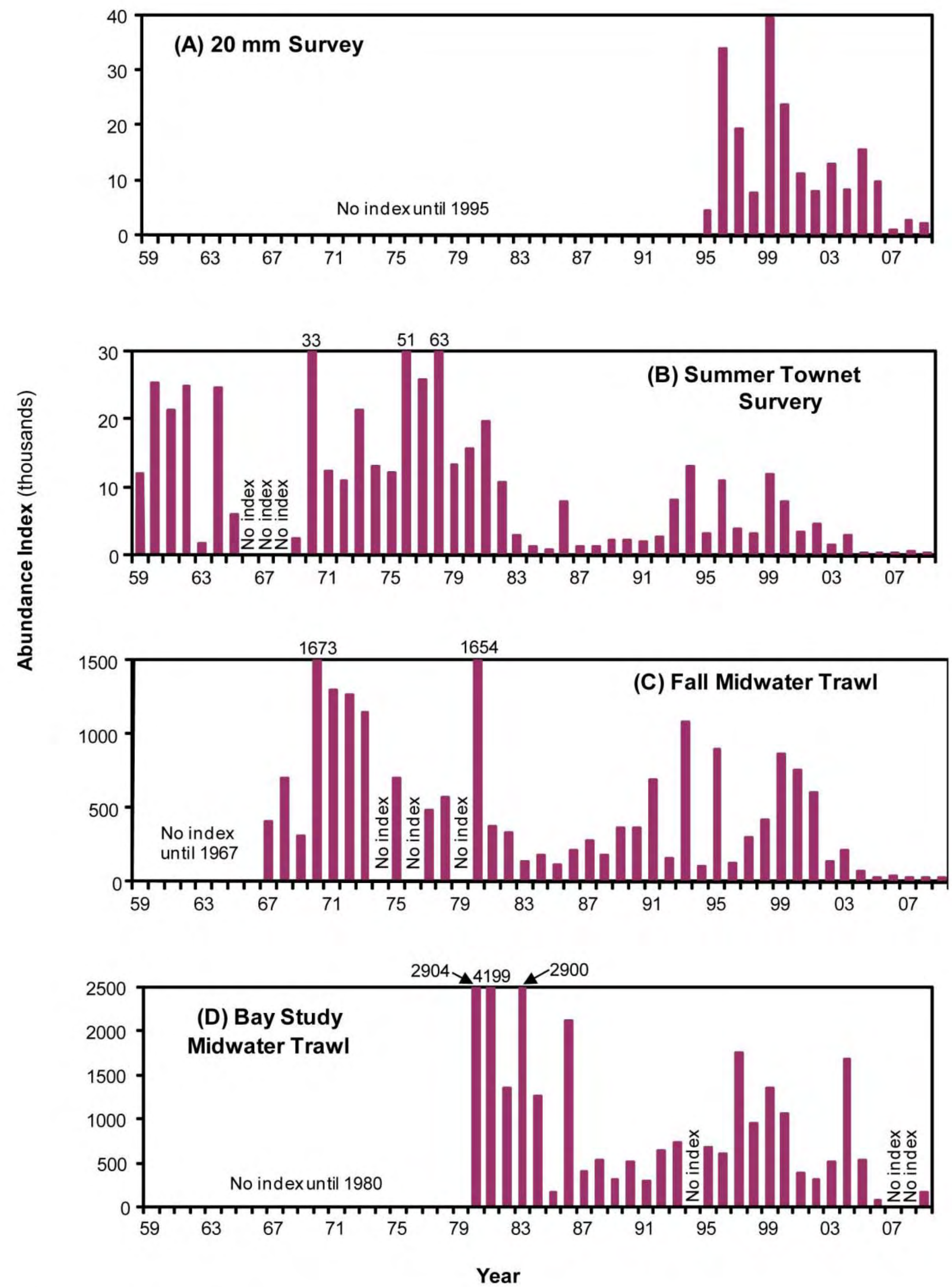
Figure A-5a. Delta Smelt Inland Range in California



Source: DFG, unpubl. data; IEP, unpubl. data.

Figure A-5b. Annual Abundance Indices of Delta Smelt from 1959-2009 in (A) 20-mm Trawl Survey, (B) Summer Townet Survey, (C) Fall Midwater Trawl, and (D) Bay Study Midwater Trawl

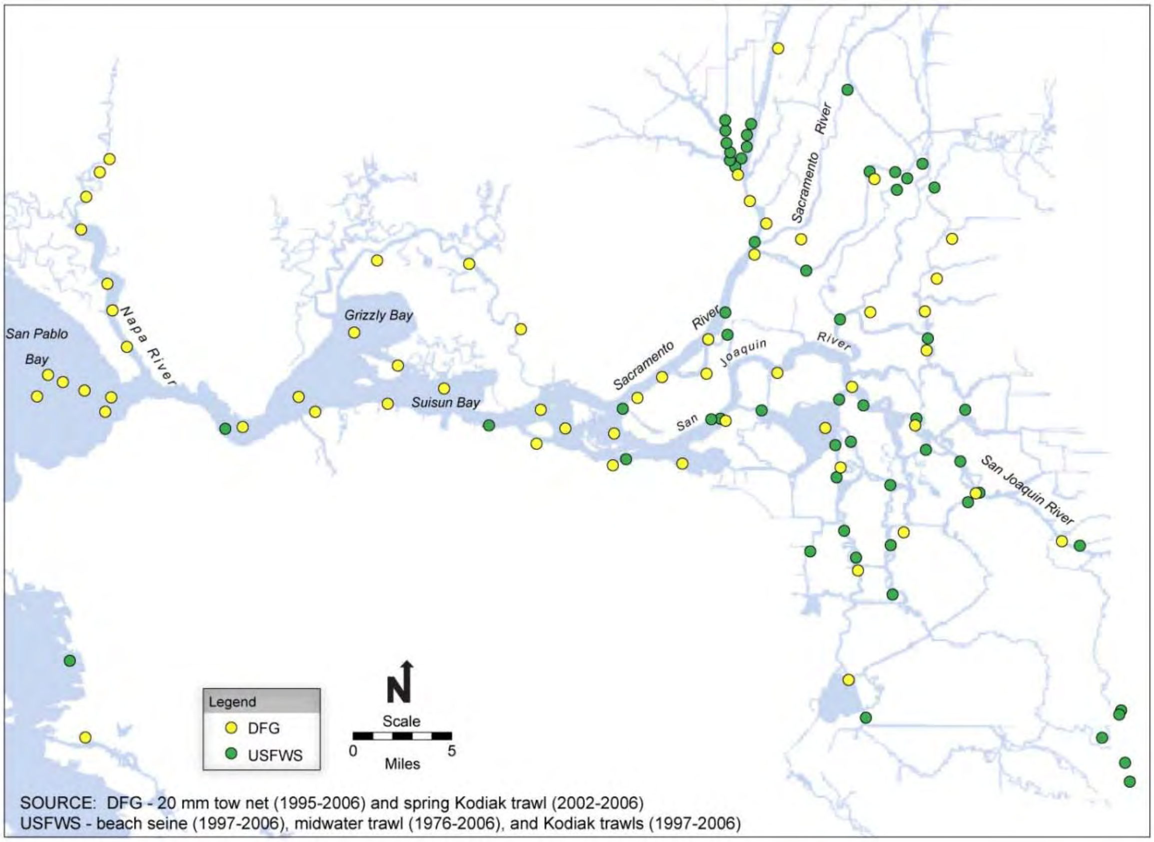


Figure A-5c. Historical Sampling Locations where Delta Smelt Have Been Captured Since 1976

Results of these fishery surveys, in addition to past records and habitat conditions for delta smelt, have been used to determine the current trends in abundance (see Figure A-5b). Trends in all four indices shown in Figure A-5b indicate that delta smelt population size has been at historical lows over the past several years.

Recent evidence suggests that a fairly large proportion of the delta smelt population over-summers in the Cache Slough region (Sommer et al. 2009). It is suspected that turbidity and prey abundance in the region are sufficient to preclude young delta smelt to migrate downstream towards the south and western Delta.

A5.3 Habitat Requirements and Special Considerations

Critical habitat was designated by USFWS for the delta smelt under the federal ESA effective January 18, 1995 (59 FR 65256). The designated critical habitat extends throughout Suisun Bay (including Grizzly and Honker bays), the length of Goodyear, Suisun, Cutoff, first Mallard and Montezuma sloughs, and the contiguous waters of the legal Delta (see Figure A-5d) (59 FR 65256).

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Figure A-5d. Delta Smelt Designated Critical Habitat

Designation of critical habitat for delta smelt was intended to provide additional protection under section 7 of the ESA with regard to activities that require federal agency action.

Delta smelt inhabit the low salinity (brackish water/fresh water interface) waters in the upper Bay-Delta estuary. Bennett (2005) found that over 90 percent of juvenile and pre-adult delta smelt caught in the DFG Summer Townet Survey and DFG Fall Midwater Trawl Survey, respectively, were collected in salinities lower than 6 practical salinity units (psu) (Bennett 2005).

Because the location of the low salinity zone is determined by the interaction of river outflow and tidal inflow of marine water from San Francisco Bay, the daily distribution of adult delta smelt can vary by many kilometers (Bennett 2005) in response to the tidal dynamics of the estuary. The location of the low salinity zone during the late winter and spring (e.g., February-June), commonly referred to as the X2 location (location of the 2 parts per thousand [ppt] bottom salinity isohaline), has been used as an indicator of habitat conditions for delta smelt and other estuarine fish and macroinvertebrates. The location of X2, which varies seasonally and inter-annually, is used as one of the parameters in assessing the effects of changes in freshwater outflow on estuarine habitat conditions.

Delta smelt are most often collected in shallow, low-salinity water upstream of the low salinity zone (about 0.5-6 psu) (Kimmerer 2004), except during spawning (K. Fleming, DFG, pers. comm.). Moyle et al. (1992) reported that delta smelt were collected primarily from waters with a mean salinity of 2 ppt and having a mean temperature of 15 °C (59 °F), but were found in salinities ranging from 0-14 ppt and at temperatures ranging from 6 °C to 23 °C (43 °F to 73 °F). A correlation has also been observed between the geographic distribution and occurrence of sub-adult and adult delta smelt in the State Water Project (SWP) and Central Valley Project (CVP) fish salvage and turbidity within the Delta (D. Fullerton unpubl. data.). Sub-adult and adult delta smelt densities are positively correlated with turbidity. Two hypotheses have been suggested for the observed correlation that include: (1) greater feeding ability; and (2) greater predator avoidance in higher turbidity. Delta smelt larvae require high microzooplankton densities during the spring months to support rapid growth and development (Miller 2007).

A5.4 LIFE HISTORY

Delta smelt inhabit open surface waters of the Delta and Suisun Bay, where they may gather together in loose aggregations. Delta smelt are semi-anadromous, spawning in freshwater areas within the Delta and tributaries (the exact spawning location of delta smelt is unknown) from February to July at water temperatures ranging from approximately 7°C to 22°C (45°F to 72°F; Wang 1986, Bennett 2005). Shortly before spawning, adult smelt migrate upstream to disperse widely into river channels and tidally-influenced backwater sloughs (Radtke 1966, Wang 1991, Moyle 2002). Although the exact location of delta smelt spawning is unknown, sampling of larval smelt in the Delta suggests spawning occurs in the Sacramento River, Barker, Lindsey, Cache, Georgiana, Prospect, Beaver, Hog, and Sycamore sloughs, in the San Joaquin River off Bradford Island including Fisherman's Cut, False River along the shore zone between Frank's and Webb tracts, and possibly other areas (Wang 1991). Recent DFG sampling has suggested that spawning is often centered in Cache Slough and the lower end of the Sacramento Deep-Water Ship Channel (DFG 2007a). Although delta smelt spawning behavior has not been observed in the wild (Moyle et al. 1992), it is thought that their adhesive, demersal eggs attach to substrates such as cattails, tules, tree roots, and submerged branches in shallow waters (Wang 1991, Moyle 2002). Laboratory experiments indicate that delta smelt spawn mainly at night, broadcasting their eggs while swimming against the current. Cultured delta smelt broadcast eggs mainly over gravel, but preferred substrates in the wild are unknown. Eggs incubate from eight to fifteen days, depending upon water temperature (Bennett 2005). Temperatures that are optimal for survival of embryos and larvae have not yet been determined, although survival of newly spawned larvae and older delta smelt appear to decrease at temperatures over 20 °C (68 °F) (Swanson and Cech 1995, Bennett 2005). Delta smelt of all sizes are found in the main channels of the Delta and Suisun Marsh and the open waters of Suisun Bay, where the waters are well oxygenated and temperatures are relatively cool, usually lower than 20-22 °C (68-72 °F) in summer. Although delta smelt tolerate a wide range of temperatures (less than 6 °C [43 °F] to greater than 25 °C [77 °F]), warmer water temperatures restrict their distribution more than colder water temperatures (Swanson and Cech 1995). Over 90 percent of juvenile and pre-adult delta smelt caught in the DFG Summer Townet Survey and DFG FMWT Survey, respectively, were collected at water temperatures lower than 20 °C (68 °F; Bennett 2005). When not spawning, delta smelt tend to concentrate near the low salinity zone, where primary productivity and zooplankton densities are typically greatest (Knutson and Orsi 1983, Orsi and Mecum 1986). Other than newly hatched larvae, all life stages of delta smelt are found in greatest abundance in the top 2 m of the water column and are not usually in close association with the shoreline (Moyle 2002).

Newly hatched delta smelt larvae have a large oil globule that makes them semi-buoyant, allowing the larvae to maintain themselves just off the bottom (R. Mager, unpublished data), where they feed on rotifers and other microscopic prey. Once the swim bladder develops, larvae become more buoyant and rise higher in the water column. It is thought that, at this stage (16-18 mm total length), delta smelt take advantage of tidal flows to move downstream until they reach the low salinity zone or the area immediately upstream of it. Net downstream flows are thought to be important for physical transport of planktonic larval delta smelt towards suitable rearing habitat in the western Delta and Suisun Bay. Prior to 2004, intermediate outflow years tended to produce the greatest abundance of delta smelt, although production was highly variable among years and in response to environmental conditions such as Delta outflow (Moyle 2002, Bennett 2005). It has been hypothesized that very low flows into and through the Delta were insufficient to transport larvae downstream to suitable rearing habitat where sufficient food resources were available to support growth and development. It has also been hypothesized that very high delta outflows may have flushed larval delta smelt downstream into the western region of Suisun Bay or San Pablo Bay where larval and juvenile rearing conditions and habitat suitability are reduced. In recent years, however, low flows appear to provide better habitat conditions for delta smelt than in earlier years (Bennett pers. comm. 1).

Feeding success is highly dependent upon prey densities (Nobriga 2002). Miller and Mongan (2006) have shown a strong correlation between the spatial and temporal co-occurrence of early lifestages of delta smelt and densities of suitable zooplankton for forage and subsequent delta smelt abundance at older lifestages. Growth is rapid and juveniles grow to 40-50 mm total length by early August (Erkkila et al. 1950; Ganssle 1966; Radtke 1966). Delta smelt reach 55-70 mm standard length in 7-9 months (Moyle 2002). Growth during the next 3 months slows down considerably (only 3-9 mm total), presumably because most of the energy ingested is directed towards gonadal development (Erkkila et al. 1950; Radtke 1966).

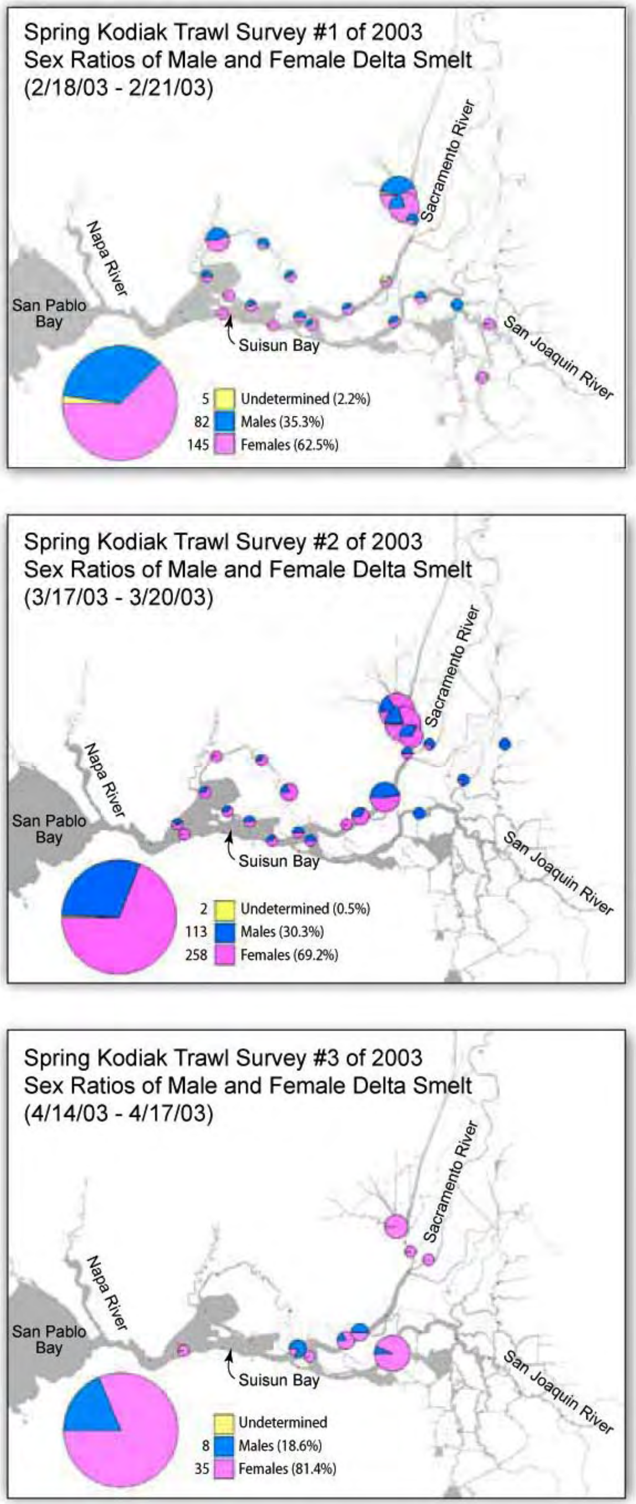
Yearly surveys by DFG (e.g., spring Kodiak trawl and 20 mm survey) provide the ability to track the geographic distribution of delta smelt within the estuary. Kodiak trawls target adult spawning delta smelt in spring months; the 20 mm townet survey targets post-larval and juvenile delta smelt from approximately March-August. Spatial patterns in the abundance of adults from the Spring Kodiak trawl and post-larval/juveniles from the 20 mm survey are shown in Figures A-5e and A-5f, respectively, based on sampling results from surveys conducted in 2003, which was a normal water year. Results of fishery surveys suggest that the geographic distribution of pre-spawning adult delta smelt in the winter and early spring does not vary substantially in response to seasonal and inter-annual variation in inflows to the Delta. Instead, it has been hypothesized that the distribution of pre-spawning delta smelt is a response to staging and foraging prior to spawning and associations with suitable habitat conditions, such as substrate, for spawning. The geographic distribution of larval and early juvenile lifestages of delta smelt appears to be influenced by freshwater inflows to the Delta during the late winter and spring. It has been hypothesized that higher Delta inflows result in faster larval planktonic transport rates from the upstream spawning habitat to the downstream estuarine portions of the Delta. In addition, when Delta inflows are high, the location of the low salinity zone is further west (downstream) and larval and early juvenile delta smelt are frequently observed further downstream within Suisun Bay. Fecundity of delta smelt is relatively low. Mager (1996) reported a length/fecundity range spanning 1,196 eggs for a 56-mm female to 1,856 eggs for a 66-mm female. Captive-reared females may be more fecund than the same size wild female; however, the variability in the length-fecundity relationship also appears to be greater for captive females (B. Baskerville-Bridges, pers. comm. as cited in Bennett 2005). The abrupt change from a single-age, adult cohort during spawning in spring to a population dominated by juveniles in summer suggests strongly that most adults die after they spawn (Radtke 1966, Moyle 2002). However, a small unknown fraction of the adult delta smelt population may survive to become two-year-old fish and spawn in the subsequent year (Moyle 2002). It has been hypothesized that because of their larger size and increased fecundity, two-year-old adults may exert a small or intermittent, but important, reproductive influence in years following poor recruitment (Bennett 2005).

In a near-annual fish like delta smelt, maximizing recruitment success is vital to the long-term persistence of the population. However, investigations using the Beverton-Holt model have found that stock-recruitment relationships accounted for only approximately one quarter of the variability observed in recruitment (Sweetnam and Stevens 1993, Bennett 2005).

Mid-February

Mid-March

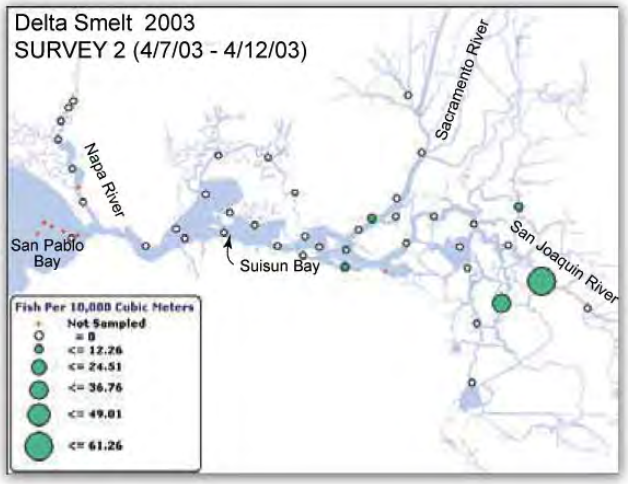
Mid-April



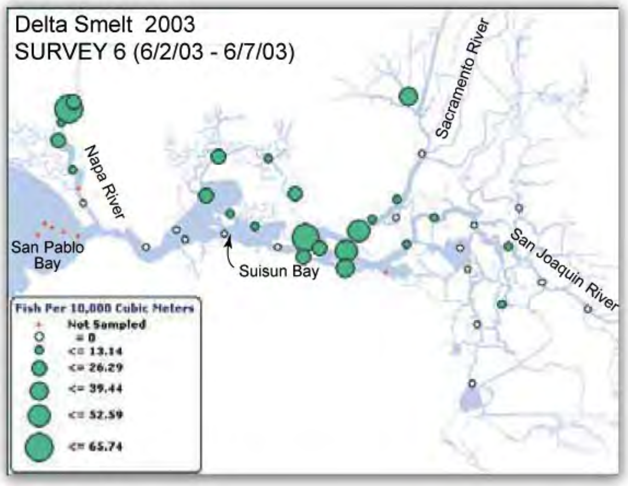
Source: DFG 2003, 20 mm Spring Kodiak trawl. This annual survey monitors adult delta smelt distribution and relative abundance, as well as sex ratio and gonadal stage, during spring throughout their historical range using a variable mesh Kodiak trawl fished in the top 6 feet (1.8 m) of water column. For more information see http://www.delta.dfg.ca.gov/data/skt/

Figure A-5e. Example of Distribution of Adult Delta Smelt in Spring-Summer of a Representative Above Normal Water Year

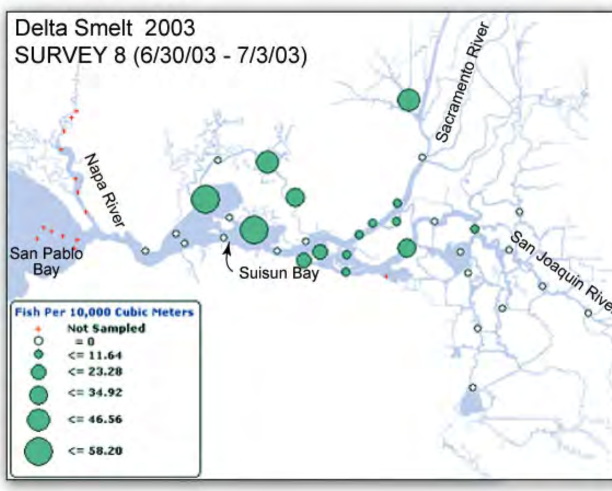
Early April



Early June



Early July



Source: DFG 2003, 20 mm delta smelt survey. This annual survey monitors post-larval and juvenile delta smelt distribution and relative abundance every two weeks during spring and early summer throughout their historical spring range using a 1600 µm egg and larval net. For more information see http://www.delta.dfg.ca.gov/data/20mm/description.asp

Figure A-5f. Example of Distribution of Post-Larval/Juvenile Delta Smelt in Spring-Summer of a Representative Above Normal Water Year

The weak stock-recruitment relationship does indicate, however, that factors affecting the numbers of spawning adults (e.g., entrainment, toxics, and predation) can influence delta smelt abundance (cohort strength) the following year.

Results of more recent evaluations suggest that density-dependent mortality may influence the number of juveniles that reach pre-adulthood in some years, even at the current low level of abundance (Bennett 2005).

Delta smelt feed primarily on planktonic copepods, cladocerans, amphipods, and, to a lesser extent, on insect larvae. Larger delta smelt may also feed on the mysid shrimp, *Neomysis*. The most important food organism for all sizes of delta smelt appears to be the euryhaline copepod, *Eurytemora*, although the non-native *Pseudodiaptomus* has become a major part of the diet since its introduction (Moyle et al. 1992, Nobriga 2002). Nobriga (2002) noted that decades-long declines in the abundance of zooplankton prey organisms such as *Eurytemora* and *Pseudodiaptomus* indicate that delta smelt prey densities have also declined. Recent unpublished analyses by Miller and Mongan (2006) suggest that a decline in the co-occurrence of juvenile delta smelt and their prey may have contributed to their decline in abundance.

A5.5 THREATS AND STRESSORS

There are multiple threats and stressors to delta smelt that appear to act in complicated and synergistic ways to influence the distribution and abundance of delta smelt (Moyle 2002). Individual stressors affect delta smelt at different times based on environmental conditions. Delta smelt are particularly vulnerable to these threats and stressors because of their short life span, low fecundity, low current abundance, and limited geographic range.

Reduced food availability. Reduced food availability in the Bay-Delta estuary has been identified as a major stressor to delta smelt (Resources Agency 2007). The co-occurrence of suitable food supplies (zooplankton) and various life stages of delta smelt (e.g., larval and juvenile life stages) as appears to be an important factor affecting delta smelt survival and abundance (Feyrer et al. 2007, Miller 2007). Histological examination of liver and other tissue collected from delta smelt has shown evidence of necrosis and pathology related to reduced foraging and depleted energy reserves. Furthermore, the size of delta smelt has declined through time. There are at least seven mechanisms briefly described here (without priority) that potentially contribute to the observed reduction in zooplankton prey densities.

First, levee construction, island reclamation, and channelization within the Delta has resulted in a substantial reduction in intertidal and shallow-water subtidal wetland/emergent marshes and open water habitat throughout the Delta. Historically, Delta wetlands and shallow-water habitat was expansive and provided large areas of estuarine and freshwater habitat that was highly productive. The significant reduction in tidal and shallow-water subtidal habitat, and an associated reduction in emergent vegetation, nutrient cycling, and the production of phytoplankton, zooplankton, macroinvertebrates, and other aquatic organisms that provide food resources for delta smelt has been identified as a major factor affecting habitat conditions within the Delta for species such as delta smelt.

Second, much of the seasonally inundated floodplain habitat in the Delta and tributary rivers has been eliminated by levees and reclamation. As a result of levee construction, flood control, and increased reservoir storage, the frequency of inundation on floodplains that still exist has been reduced (Resources Agency 2007). Floodplains are highly productive due to their shallow, warm, low velocity water (Sommer et al. 2001a, b) and input of organic material and nutrients from the terrestrial community (Booth et al. 2006). Floodplains are a key source of nutrients and organic material for the Bay-Delta estuary (Sommer et al. 2001a, Harrell and Sommer 2003).

Third, hydraulic residence time in the Delta has declined as a result of increased channelization and passage of Sacramento River water through the Delta Cross Channel into the central and southern Delta to meet water quality standards and supplies for the SWP and CVP exports. The decreased hydraulic residence time reduces the time available for production of phytoplankton and zooplankton that provide food for delta smelt and other aquatic species and for bacteria to use nutrients and organic carbon (Jassby et al. 2002, Kimmerer 2002a, 2004, Resources Agency 2007).

Fourth, the presence of non-native species has reduced the abundance of food available to delta smelt. The efficient filter feeding and high abundance of the overbite clam has dramatically reduced phytoplankton and zooplankton abundance in Suisun Bay, the western Delta, and Suisun Marsh since its introduction in the mid 1980s (Kimmerer and Orsi 1996). The Asian clam has also reduced phytoplankton abundance in the Delta, which likely reduced zooplankton abundance (Jassby et al. 2002, Thompson 2007). Other non-native zooplanktivores that likely compete for limited available food resources with delta smelt include threadfin shad, inland silverside, and wakasagi.

Fifth, the zooplankton community inhabiting the Bay-Delta estuary has changed multiple times since the 1970s as non-native species have established and outcompeted other zooplankton species (Resources Agency 2007, Sommer 2007). These changes in the zooplankton species composition have affected the quality of food resources available to delta smelt because some of the non-native species do not appear to be as suitable a food resource as the native species (Resources Agency 2007). Most recently, *Limnoithona*, a non-native cyclopoid copepod, invaded the Delta. This copepod is smaller than preferred forage species, *Pseudodiaptomus* and *Eurytemora* (Lott 1998), may be protected from predators by spines (Orsi and Ohtsuka 1999), and, therefore, is thought to be lower quality food for delta smelt (Resources Agency 2007). Preliminary laboratory work by Sullivan et al. (2007, 2008) indicates that larval delta smelt consume these copepods according their size. Also, adult delta smelt, which are larger and need greater amounts of food, may prefer the larger *Pseudodiaptomus* and *Eurytemora* over *Limnoithona*. A decrease in foraging efficiency and/or the availability of suitable prey for various lifestages of delta smelt would result in reduced growth, survival, and reproductive success contributing to reduced population abundance.

Sixth, SWP and CVP exports and the over 2,200 in-Delta agricultural diversions (Herren and Kawasaki 2001) export phytoplankton, zooplankton, nutrients, and organic material that would otherwise support the base of the food web in the Delta, thus reducing food availability for delta smelt (Jassby and Cloern 2000, Resources Agency 2007).

Seventh, it has been hypothesized that exposure of phytoplankton and zooplankton to toxics (e.g., pesticides, herbicides) that enter the Delta from point and non-point sources contribute to the observed low abundance of zooplankton prey species for delta smelt and other species inhabiting the Bay-Delta (Weston et al. 2004, Luoma 2007, Werner 2007). In addition to direct impacts of toxics on delta smelt, such as liver damage and other pathologic symptoms, the indirect effect of toxics on reducing zooplankton and phytoplankton abundance is thought to result in reduced availability of food resources for delta smelt.

Municipal wastewater treatment plants, particularly the Sacramento Regional County Sanitation District's wastewater treatment plant, discharges high loads of ammonia directly into the Sacramento River in the North Delta (Jassby 2008). Results of a recent investigations suggest that high concentrations of ammonium, the ionized form of ammonia, can inhibit diatom production in San Francisco and San Pablo bays (Wilkerson et al. 2006, Dugdale et al. 2007), which could disrupt the foodweb, ultimately resulting in reduced food for delta smelt and other Delta fish species. The source of this ammonium in the bays is unknown, but could come from wastewater treatment plants. Recent preliminary investigations have also examined the role of ammonium in inhibiting freshwater diatom production in the Delta, although results are inconclusive (Parker and Dugdale 2008).

Reduced rearing habitat. Availability of rearing habitat for delta smelt is less dependent on physical substrate and more dependent on water column conditions. There is evidence that the availability and suitability of delta smelt rearing habitat varies with location of the low salinity zone, measured as X2 (Moyle et al. 1992) , although there is no long-term trend indicating that X2 has changed nor is there strong evidence that X2 predicts delta smelt abundance (Armor et al. 2006). Guerin et al. (2006) reported correlations between reductions in fall delta outflow, salinity changes, and indices of delta smelt abundance. State Water Board Decision 1641 (D1641, State Water Resources Control Board [SWRCB] 2000) establishes criteria for maintenance of X2 at specific locations within Suisun Bay and the western Delta by month and water year type. The maintenance of this standard substantially affects when water can be exported from the Delta at the CVP and SWP pumps. SWP and CVP exports were reduced to protect aquatic resources in the spring with a resulting increase in export operations during the summer and fall.

The location of the low salinity zone within Suisun Bay and the western Delta varies in response to the magnitude of freshwater outflow from the Delta and saltwater intrusion from San Francisco and San Pablo bays. The low salinity zone, when positioned over shallow water shoal areas in Suisun Bay in response to Delta outflows, is thought to result in highly productive conditions (Moyle et al. 1992, Bennett et al. 2002, Kimmerer 2004, Bennett 2005). Aasen

(1999) observed higher abundance of delta smelt in shallow water shoal habitat compared to adjacent deep water channels and Hobbs et al. (2006) found evidence that the health and survival of delta smelt were greater in habitats associated with shallow water. When located upstream of the confluence of the Sacramento and San Joaquin rivers, however, the low salinity zone is confined to the deep river channels, resulting in a smaller total surface area, few shoal areas, swifter, more turbulent water currents, and lower zooplankton productivity (USFWS 2004).

Feyrer et al. (2007) developed an index of fall habitat quality for delta smelt based on statistical regression models of delta smelt catches in DFG fishery sampling at specific sampling locations throughout the Delta and Suisun Bay and water quality parameters (salinity, turbidity, and water temperature) measured for each sample. There is substantial debate over whether this index is meaningful to the delta smelt population.

As the geographic distribution of delta smelt shifts upstream with X2, individuals may become more vulnerable to entrainment by the SWP and CVP export facilities and other diversions within the interior of the Delta.

The Suisun Marsh Salinity Control Gates function to decrease salinity in managed wetlands of Suisun Marsh to support crops that attract waterfowl to duck clubs located throughout the marsh. When in operation, generally from October through May, the control gates near Collinsville divert up to 2,500 cubic feet per second (cfs) of freshwater from upstream flows into the marsh. Because the minimum outflow standard during fall months is 5,000 cfs, a significant proportion of total Delta outflow (up to 50 percent) does not flow through the eastern Suisun Bay region. This diversion moves X2 upstream resulting in a measurable increase in salinity in eastern Suisun Bay, which may correspond to a decrease in low salinity habitat for delta smelt (Fullerton 2007)

Elevated water temperatures. Delta smelt are sensitive to exposure to elevated water temperatures (Swanson and Cech 1995). During the late spring, summer, and early fall months water temperatures within the central and southern regions of the Delta typically exceed 25 °C (77 °F), which has been found to be close to the incipient lethal temperature for delta smelt. During these warmer periods of the year, results of fishery sampling have shown that delta smelt avoid inhabiting the central and southern regions of the Delta and are typically located downstream in Suisun Bay and Suisun Marsh. It is generally thought that water temperatures within the Delta vary in response to seasonal air temperatures and solar radiation, and are largely independent of freshwater inflow to the Delta during the summer months (flow rate has been identified as a factor affecting water temperatures in the rivers; however, the influence decreases with distance downstream in the lower rivers and Delta). As a result of coastal fog and marine conditions along the coast and within San Francisco Bay, water temperatures during the summer typically decrease within Suisun Bay when compared to conditions further upstream within the Delta. Although water temperatures are cooler within Suisun Bay during the summer months, water temperatures in excess of 20 °C (68 °F) are typical. Under these warmer summer conditions, delta smelt rearing in Suisun Bay and Suisun Marsh would be stressed by exposure to elevated water temperatures and experience higher metabolic demands and a greater demand for food supplies to maintain individual health and a positive growth rate. Stresses experienced by rearing delta smelt during the warmer summer months, which include the synergistic effects of salinity and seasonally elevated water temperatures, have been hypothesized to be a potentially significant factor affecting delta smelt survival, abundance, and subsequent reproductive success within the Bay-Delta estuary.

Reduced turbidity. Delta smelt appear to have specific turbidity requirements that can influence predation risk and foraging efficiency. Turbidity is a significant predictor of delta smelt occurrence in the Delta (Feyrer et al. 2007, Resources Agency 2007, Nobriga et al. 2008, Grimaldo et al. 2009). That is, delta smelt occurrence increases with higher turbidity. Fullerton (unpubl. data) has demonstrated a correlation between the occurrence of delta smelt and elevated levels of turbidity (above approximately 12 nephelometric turbidity units). It is thought that delta smelt require turbidity for both successful foraging (Baskerville-Bridges et al. 2004) and predator escape (Feyrer et al. 2007), and that turbidity is an important cue for delta smelt spawning migrations (Grimaldo et al 2009). Results of laboratory studies indicate that, in low turbidity waters, delta smelt move to the edge of aquaria and stop swimming, presumably to reduce vulnerability to predation (D. Fullerton unpubl. data). Results of laboratory studies have also demonstrated that delta smelt reduce or stop foraging when turbidity is low and actively forage when turbidity is increased. It was hypothesized that increased turbidity levels provide a more favorable background and contrast that allows delta smelt a better opportunity to detect and effectively capture zooplankton prey.

Turbidity levels have declined in the Bay-Delta estuary since the 1970s (Kimmerer 2004, Wright and Schoellhamer 2004, Feyrer et al. 2007, Fullerton 2007). This trend can be attributed to multiple causes. First, upstream sediment inputs have been reduced due to a range of anthropogenic actions (Jassby et al. 2002, Kimmerer 2004), including depletion of erodable sediments from hydraulic mining in the 1800s, river bank protection, trapping of sediments by dams and reservoirs, levee construction that has reduced floodplain inundation and channel meanders, and changes in land use (Wright and Shoellhamer 2004). Wright and Shoellhamer (2004) estimated that the yield of suspended sediments from the Sacramento River declined by approximately one half from 1957 to 2001.

Second, the distribution and abundance of non-native aquatic plant species, particularly Brazilian waterweed (*Egeria*), have increased dramatically over the past 20 years (Nobriga et al. 2005, Brown and Michniuk 2007). Brazilian waterweed can reduce turbidity by reducing local water velocities and trapping fine suspended sediments (Grimaldo and Hymanson 1999, Nestor et al. 2003, Hobbs et al. 2006).

Third, the high filtering efficiency of the overbite clam has dramatically reduced phytoplankton and zooplankton abundance in the western Delta and Suisun Bay since its introduction (Kimmerer and Orsi 1996, Jassby et al. 2002, Kimmerer 2002b, 2004). The reduction in phytoplankton in the water column may contribute to increased water clarity and reduced turbidity in the Delta.

Fourth, hydraulic residence time in the Delta has declined as a result of increased channelization and the movement of water from the Sacramento River into the interior Delta channels to improve water quality and provide increased supplies to the SWP and CVP exports. SWP and CVP export operations have also directly resulted in changes in the hydrodynamics within Delta channels such as Old and Middle rivers which affect hydraulic residence time. Reduced hydraulic residence time reduces the ability of phytoplankton and bacteria to incorporate nutrients and carbon, ultimately reducing the abundance of these organisms in the water column (Jassby et al. 2002, Kimmerer 2002a, 2004, Resources Agency 2007). This reduction in phytoplankton and zooplankton abundance reduces the turbidity within the Bay-Delta estuary.

The observed reduction in Bay-Delta turbidity has the potential, in combination with other factors such as the effects of non-native species, to fundamentally alter the trophic dynamics of the estuary for species such as delta smelt.

Reduced spawning habitat. Although delta smelt spawning has not been observed within the Bay-Delta estuary, it is generally thought that spawning occurs in shallow, low-salinity upstream areas with sand or gravel substrate on which to deposit adhesive egg sacs (Moyle et al. 2004). Such habitat could occur in Cache Slough or in shallow shoals located in the Deep Water Ship Channel (Bennett 2007). The primary causes of reduced spawning habitat are believed to be reclamation, channelization, and riprapping of historical intertidal and shallow subtidal wetlands.

Non-native species. Predation by introduced species has been identified as a potential stressor on smelt populations (Sommer et al. 2007, Baxter et al. 2008), but the importance of predation on delta smelt abundance is thought to be low (Stevens 1966 as cited in Nobriga and Feyrer 2008, Feyrer et al. 2003, Nobriga and Feyrer 2007, 2008, Nobriga 2009a, Hanson 2009). There are several potential non-native fish predators of delta smelt that have been introduced into the Delta (Bennett 2005). Delta smelt have historically been a minor prey item of juvenile and subadult striped bass in the Delta (Stevens 1966, Bennett 2005), although predation does occur (M. Nobriga, pers. comm.). More recent studies indicate that delta smelt are rarely found in the stomachs of striped bass, largemouth bass or other nearshore predators (Feyrer and Nobriga 2008, Nobriga and Feyrer 2008). Delta smelt have also been reported from the stomach contents of white catfish and black crappie in the Delta (Turner and Kelley 1966). Threadfin shad and inland silversides, both planktivores, possibly eat delta smelt eggs, larvae, and small juveniles. Dense aggregations of silversides occur in shoreline habitats where delta smelt are thought to spawn and may consume delta smelt eggs and larvae (Bennett 2005). The largest single source of predation on delta smelt is thought to occur at or near the SWP and CVP south Delta pumping facilities (Sommer et al. 2007), and especially at Clifton Court Forebay. This predation is related to the number of smelt that are drawn to this area because of export-related changes in hydrology (Grimaldo et al. 2009, Kimmerer et al. 2009).

Competition with inland silversides could have a potentially large impact on delta smelt. Silversides are highly abundant throughout the delta smelt geographic range, their diet range encompasses that of delta smelt, and they spawn repeatedly throughout late spring, summer, and fall, thus providing silversides with a competitive advantage over delta smelt (Bennett 2005).

Wakasagi can occur in the delta smelt geographic range and have similar life requirements. Thus, they likely compete for food and spawning sites. Wakasagi have a higher tolerance to salinity and temperature and a wider geographic range than delta smelt, suggesting that they have a competitive advantage over delta smelt. Furthermore, the introduction of wakasagi has created the potential for a loss of genetic integrity of delta smelt, although the probability that hybridization could be successful is low (Moyle 2002). The two species are not closely related genetically and, although first generation hybrids have been collected, all of them have been sterile (Stanley et al. 1995, Trenham et al. 1998). If wakasagi abundance in delta smelt habitat were to increase dramatically, the risk of genetic introgression would be enhanced (Bennett 2005), although this does not appear to be a large concern at this time (K. Fisch pers. comm.). The recent decline in delta smelt abundance has likely made the species vulnerable to inbreeding and genetic drift, leading to decreased genetic variation and reduced evolutionary fitness (Center for Biological Diversity et al. 2006). However, no estimates currently exist for the minimum viable population size of delta smelt, nor have studies been conducted to evaluate changes in genetic diversity.

It has been hypothesized that the greatest impact of a non-native species on delta smelt is that resulting from colonization of the Bay-Delta estuary by the overbite clam. The clam has been identified as one of the major causes of the dramatic changes observed in the composition and abundance of the delta smelt zooplankton prey base. Because of its high filtration efficiency and dense populations in the western Delta and Suisun Bay, the clam has reduced phytoplankton and zooplankton abundance throughout the region (Kimmerer and Orsi 1996). The euryhaline copepod, *Eurytemora*, was historically the primary prey for all life stages of delta smelt. After the introduction of the overbite clam, the abundance of *Eurytemora* declined sharply, being replaced over much of its range by *Pseudodiaptomus* (Kimmerer and Orsi 1996, Bennett 2005). Although *Eurytemora* is still abundant during early spring, the population is replaced by *Pseudodiaptomus* in later spring, creating a period of low copepod abundance. Low food abundance can cause poor feeding success by larval and juvenile delta smelt, leading to slow growth, liver abnormalities associated with starvation, and, ultimately, reduced survival for cohorts that begin feeding during the period of low copepod abundance (Bennett 2005). In addition to the low copepod abundance period, *Pseudodiaptomus* are faster swimmers than *Eurytemora* and may lead to lower foraging efficiency, starvation, and reduced growth rates for delta smelt (Moyle 2002). Recent evidence suggests that the overbite clam may further negatively impact delta smelt by reducing their foraging efficiency by filtering large quantities of phytoplankton from the water column and increasing water clarity, potentially leading to the inability of delta smelt to forage effectively (Herbold, pers. comm.). The increase in water clarity may also increase the vulnerability of delta smelt to visual predators.

Brazilian waterweed and water hyacinth are fast growing and abundant aquatic plants that have had detrimental effects to the Bay-Delta aquatic ecosystem (Grimaldo and Hymanson 1999, Brown and Michniuk 2007, Feyrer et al. 2007). These non-native plant species grow in dense aggregations can indirectly affect delta smelt by reducing dissolved oxygen levels and reducing nearby flow rates, resulting in local reductions in suspended sediment concentrations and turbidity within the water column. Furthermore, because of the three dimensional structure and shade they provide, these aquatic plants likely create excellent habitat for non-native predators of delta smelt, primarily centrarchids (Nobriga et al. 2005). Because Brazilian waterweed has recently spread by as much as 10 percent per year in areal coverage (Ustin et al. 2008), its negative impacts on delta smelt may increase in future years.

Entrainment. Despite the number of delta smelt that have been entrained by the SWP and CVP export facilities and over 2,200 smaller diversions in the Delta (Herren and Kawasaki 2001), the direct impacts of water diversions on the overall population dynamics of delta smelt is not well understood and there is disagreement among experts about the magnitude of these impacts (Bennett 2005).

Several studies have been conducted that show correlative relationships between SWP and CVP exports and indices of delta smelt abundance, suggesting that entrainment may negatively impact delta smelt abundance (USFWS 2008). These relationships do not establish causality, but they are an indicator that salvage should be considered. Kimmerer (2008) reported results of an analysis of the potential effects of SWP and CVP entrainment losses on larval and adult delta smelt. Results of these analyses suggest that losses of adult delta smelt had a median value of 15 percent (range 1-50 percent) while the seasonal losses for juvenile delta smelt had a median value of 13 percent (range of 0-25 percent). Kimmerer (2008) concluded that the effect of these losses on population abundance of delta smelt was obscured by a 50-fold variation in the overall survival of delta smelt survival between summer and fall.

Guerin et al. (in review) found significant correlations between SWP winter salvage of adult smelt and subsequent FMWT index of delta smelt with 1 and 2 year lags over the past 12 years. More recent work shows that SWP winter salvage of adult delta smelt normalized to the prior FMWT correlates strongly with subsequent FMWT for delta smelt over a longer record.

Bennett (unpubl. data) found a significant negative correlation between winter and early spring salvage and delta smelt survival estimates (Brown and Kimmerer 2001). Swanson (2005) found that winter exports (December through March) were significantly negatively correlated with both the juvenile delta smelt abundance index from DFG summer townet surveys and the sub-adult and adult delta smelt abundance index from DFG FMWT surveys, although SWP and CVP exports explained only 15.5 and 2.4 percent of variation in juvenile and sub-adult/adult abundance indices, respectively. Herbold et al. (2005) reported that delta smelt salvage density relative to apparent abundance has increased markedly since 2002, concurrent with the POD. Manly and Chotkowski (unpubl. data) found a statistically significant correlation between delta smelt abundance and total exports, but the relationship explained only a small proportion of overall variation in delta smelt abundance (Miller 2007). As a result, Manly and Chotkowski assert that exports do not appear to play a large role in controlling delta smelt population abundance relative to other stressors (e.g., reduced food availability).

The risk of entrainment to delta smelt varies seasonally and among years. The most important entrainment risk has been hypothesized to occur during winter when pre-spawning adults migrate into the Delta in preparation for spawning (Moyle 2002, Bureau of Reclamation 2004). Patterns in SWP and CVP salvage data support this hypothesis (DFG, unpubl. data). Bennett (2005) has hypothesized that larger female delta smelt spawn earlier in the winter and are, therefore, more vulnerable to entrainment by export facilities. Larger females are more fecund, spawn repeatedly, and can produce more offspring with higher fitness than smaller females. As a result, Bennett hypothesized that entrainment during winter months may have a disproportionately large impact on the overall population dynamics of delta smelt than entrainment during other periods of the year.

Analyses conducted by P. Smith (unpubl. data) and J. Johns (unpubl. data) present results of an analysis of the relationship between the magnitude of reverse flows in Old and Middle rivers during the winter (January-February) and salvage of pre-spawning delta smelt at the SWP and CVP export facilities. Smith found a linear relationship between reverse flows and delta smelt salvage for January and February combined. Johns found a non-linear relationship between reverse flows and delta smelt salvage separately by month. Results of the non-linear model were statistically significant and showed that delta smelt salvage remained relatively low when reverse flows in Old and Middle rivers were below approximately -5,000 cfs. As reverse flows increased to greater than 5,000 cfs, delta smelt salvage increased substantially. Results of these analyses were used as the basis for a 2007 federal court decision regarding interim operational restrictions on SWP and CVP exports (Wanger decision) and the December 2008 delta smelt biological opinion for SWP and CVP operations by the USFWS (2008).

Entrainment risk for delta smelt has largely been based on analyses of SWP and CVP fish salvage. The fish salvage operation, however, only identifies and counts those individual fish greater than 20 mm in length. As a result, larval delta smelt smaller than 20 mm are not included in fish salvage estimates. Results of several preliminary estimates of the potential magnitude of larval delta smelt entrainment at the SWP and CVP export facilities have been made, as well as estimates of the population size of delta smelt that are intended to put entrainment losses into an population-level framework for evaluation (Hanson unpublished data). Estimates of entrainment losses for larval delta smelt and estimates of population abundance have been based on extrapolations from results of the DFG 20 mm delta smelt survey. These preliminary estimates have been criticized on the basis of a number of assumptions that are required to make the population and entrainment loss estimates that have not been tested or validated. Recognizing that larval delta smelt are vulnerable to SWP and CVP entrainment losses that may vary in magnitude and potential effect on the population among years, the federal district court ordered that a study be conducted beginning in 2008 to monitor the densities of larval delta smelt vulnerable to SWP and CVP entrainment losses for use in the future in determining whether or not additional protective measures would be required to reduce potentially adverse impacts associated with larval delta smelt entrainment.

Nobriga and Matica (2000) and Nobriga et al. (2004) found low and inconsistent entrainment of juvenile delta smelt by small agricultural diversions near Sherman Island; the low entrainment rates were hypothesized to be the result of juvenile delta smelt occurring offshore of the intake location and in the upper portions of the water column. Cook and Buffaloe (1998) also reported that unscreened agricultural diversions entrained low numbers of delta smelt. However, many agricultural diversions are located within primary delta smelt habitat and could potentially entrain delta smelt for a large proportion of the year. It has been hypothesized that, although juvenile and adult delta smelt may avoid entertainment at unscreened water diversions, planktonic larvae are expected to be distributed within the water column and have weak swimming performance. Therefore, larvae may be vulnerable to higher entrainment losses than predicted by results of investigations of juvenile and adult smelt. Therefore, the combined effect of location, abundance, and duration of agricultural diversions on delta smelt survival could be high.

Power plants located within the Plan Area at Pittsburg and Antioch have the potential to entrain large numbers of fish, including delta smelt and other covered fish species, particularly because these species may be located near these facilities for much of the year (Matica and Sommer 2005, C. Hanson unpubl. data). However, use of cooling water is currently low with the retirement of older units. According to recent regulations by the SWRCB, units at these two plants must be equipped with a closed cycle cooling system by 2017that eliminates fish entrainment.

Exposure to toxins. Exposure of delta smelt to toxic substances can result from point and non-point sources associated with agricultural, urban, and industrial land uses. The Delta serves as the receiving waters for a wide variety and large volume of toxic substances, including agricultural pesticides, herbicides, endocrine disruptors, heavy metals, and other agricultural and urban products (Thompson et al. 2000, Moyle 2002). Kuivila and Moon (2004) sampled pesticide concentrations within the Delta and west to Chipps Island for 3 years (1998-2000) during April-June. Their water samples contained multiple pesticides, but at individual concentrations well below lethal 96-hr LC50 concentrations for fishes. A reported toxic event in the winter of 2007 (toxicity was demonstrated using water samples collected from the Delta under laboratory conditions; no tests were performed using delta smelt) coincided temporally and spatially with delta smelt spawning in the Cache Slough region of the Delta and was also detected further downstream in the lower Sacramento River near Sherman Island (DWR, unpubl. data). Indications of toxicity also were detected within Suisun Bay during the summer of 2007 (S. Ford pers comm.). Although no specific causal link has been established, these toxic events coincided with low abundance indices of larval and juvenile delta smelt observed in the 2007 DFG 20 mm townet and summer townet surveys. Bioassay studies conducted as part of the POD studies found two instances of significant larval delta smelt mortality in samples collected from the Sacramento River in June and July 2007 that had relatively low turbidity and salinity and moderate levels of ammonia (Werner unpublished data, as cited in Baxter et al. 2008) . There have been multiple studies indicating that toxics have little direct effect on delta smelt (Resources Agency 2007, Werner et al. 2007, Bennett unpubl. data).

The short life span (1-2 years) and location of their food source in the food web (zooplankton are primary consumers) reduce the ability of toxic chemicals to bioaccumulate in the tissue of delta smelt (Moyle 2002). Their location in the upper portion of the water column may further reduce the probability of some toxic impacts by those chemicals that are sequestered quickly by sediments (i.e., pyrethroids; B. Herbold pers. comm).

Ammonia discharged from municipal wastewater treatment plants may contribute to localized toxicity in delta smelt, although results are highly variable. Werner et al. (2008) found that water samples near the Sacramento Regional County Sanitation District's wastewater treatment plant effluent reduced 4-day survival of larval delta smelt in 2006, but did not affect survival even after 7 days in 2007. Furthermore, there were two instances of significant larval delta smelt mortality from POD bioassays collected from the Sacramento River in June and July 2007 that had relatively low turbidity and salinity and moderate levels of ammonia (Werner unpubl. data, as cited in Baxter et al. 2008). The form and toxicity of ammonia/um changes based on pH and it has been hypothesized that changes in pH of the Delta receiving waters may change in response to algal growth, discharges from managed wetlands and duck clubs, and agricultural return flows that result in ammonia toxicity. These potential water quality interactions and the effects of discharging ammonia from a number of wastewater treatment plants located throughout the Sacramento and San Joaquin rivers, Delta, Suisun Bay and Suisun Marsh on the health and survival of delta smelt and other aquatic species are under investigation.

Consistent evidence of direct toxicity of contaminants to smelt within the Delta is lacking (Werner et al. 2008); however, there is growing evidence that toxics may have indirect effects on delta smelt. For example, invertebrate prey of delta smelt are affected by toxics (Weston et al. 2004, Luoma 2007, Werner 2007), reducing food availability. Additionally, the nitrate uptake by and production of phytoplankton, the base of the food web that supports delta smelt, may be inhibited by ammonia concentrations in the North Delta as has been demonstrated for phytoplankton in San Francisco and San Pablo bays (Dugdale et al. 2007). There is also evidence that toxics may cause sublethal impacts to delta smelt that make them more vulnerable to other sources of mortality (Werner 2007). Most, if not all, pyrethroids are potent neurotoxicants (Bradbury and Coats 1989, Shafer and Meyer 2004) and have immunosuppressive effects (Madsen et al. 1996, Clifford et al. 2005). In addition, these compounds and their breakdown products can act as endocrine disrupting compounds by disrupting hormone-related functions (Go et al. 1999, Tyler et al. 2000, Perry et al. 2006, Sun et al. 2007). Esfenvalerate, a common pyrethroid insecticide, has been shown to increase the susceptibility of juvenile fall-run Chinook salmon to infectious hematopoietic necrosis virus (Clifford et al. 2005), and reduce swimming ability and increase susceptibility to predation in larval fathead minnows (Floyd et al. 2008). In delta smelt, exposure to environmentally relevant pyrethroid concentrations resulted in significant swimming abnormalities, which were strongly linked with downregulation of genes involved in neuromuscular activity (Connon et al. 2009).

Exposure to copper contamination can also result in significant sublethal effects on Delta fish species, with implications for their vulnerability to other stressors. Environmentally relevant copper concentrations are shown to result in significant immunosuppressive effects (Hetrick et al. 1979) and impair olfactory function and eliminate the predator avoidance response in fish (Sandahl et al. 2006; Werner et al. in press). Swimming abnormalities have been observed after exposure to copper concentrations as low as one quarter of the chemical's LC50 values (Little and Finger 1990; Oros and Werner 2005). Dissolved copper causes acute toxicity to the calanoid copepod, *Eurytemora affinis*, in the north and south Delta (Teh 2009) and impairs the sensory function of juvenile salmonids (Hecht et al. 2007), specifically related to predator avoidance behavior. Moreover, specific concentrations of dissolved copper correspond to sublethal endpoints such as primary production and salmonid growth (Hecht et al. 2007). Delta smelt may be affected in a similar way. Additionally, negative synergistic effects have been documented such that the presence of copper in combination with ammonia is more toxic to aquatic organisms than either toxicant individually (Herbert and Vandyke 1964).

A5.6 RELEVANT CONSERVATION EFFORTS

Pursuant to the CALFED objective of ecosystem restoration, the CALFED agencies developed the Ecosystem Restoration Plan (ERP) and the Environmental Water Account (EWA) for the purpose of restoring habitat and recovering at-risk populations like delta smelt in the Bay-Delta estuary (CALFED 2000). The ERP was intended to improve aquatic and terrestrial habitats and natural processes to support stable, self-sustaining populations through an adaptive management process, and the EWA was intended to provide increased water supply reliability while assuring the availability of sufficient water to meet fishery protection and restoration and recovery needs, as part of the overall ERP. Additional enhancement and protective actions are also being identified as part of mitigation programs for various projects, biological opinions, and regional conservation planning efforts.

The Delta Regional Ecosystem Restoration Implementation Plan (DRERIP) was formed to guide the implementation of CALFED ERP elements within the Delta (DFG 2007b). The DRERIP team has created a suite of ecosystem and species conceptual models, including a conceptual model for delta smelt, that document existing scientific knowledge of Delta ecosystems. The DRERIP team has used these conceptual models to assess the suitability of actions proposed in the ERP for implementation. DRERIP conceptual models have been used in the analysis of proposed BDCP conservation measures.

Various projects exist to benefit delta smelt and several other native fish species. For example, in 2007, Westlands Water District acquired land located in the southern end of the Yolo Bypass that is thought to be high value habitat for delta smelt. Designs for potential habitat enhancement projects within the Yolo Bypass are being developed and evaluated. Objectives of these potential actions include enhancing the frequency and duration of access to seasonally inundated floodplain habitat to benefit species such as juvenile Chinook salmon and splittail, as well as to increase nutrient cycling and food production for delta smelt and other species. Wildlands, Inc. has established a mitigation bank to offset site-specific impacts to fish species, such as delta smelt, near Kimball Island in the western Delta. Access to aquatic habitats, such as the waters adjacent to Kimball Island, may provide direct benefits to delta smelt, as well as indirect benefits associated with increased nutrient cycling, phytoplankton and zooplankton production, and an associated increase in food supplies for delta smelt and other aquatic resources. Furthermore, work funded by CALFED and implemented by the Natural Heritage Institute and DWR is intended to improve and protect habitat in Dutch Slough for delta smelt, splittail, and juvenile salmonids.

The Delta Smelt Working Group is a group of scientists under the auspices of the Interagency Ecological Program, Bureau of Reclamation, and DWR that makes recommendations on water operations for the protection of delta smelt. The group uses a delta smelt risk assessment matrix, which consists of month by month criteria that, when exceeded, will trigger a meeting of the group and possible management recommendations.

In January 2005, the Interagency Ecological Program established the Pelagic Organism Decline (POD) work team to investigate the causes of the observed rapid decline in populations of pelagic organisms, including delta smelt, in the upper San Francisco Bay estuary (Baxter et. al. 2008). Since then, numerous studies have been conducted to determine the cause of the POD. Based on results of these studies and relevant studies undertaken by others, the work team has developed conceptual models to discern an understanding of the factors causing POD and to provide a basis from which to identify actions to address POD. Resources Agency has also prepared a Pelagic Fish Action Plan in March 2007 to address POD (Resources Agency 2007). The action plan identifies 17 actions that are being implemented or that are under active evaluation to help stabilize the Delta ecosystem and improve conditions for pelagic fish.

In 2007, the Federal District Court, Eastern District of California, Fresno Division (Judge Wanger) issued a court order for interim actions to protect delta smelt pending completion of a new biological opinion by the USFWS on SWP and CVP operations. The court ruling remained in effect until the new biological opinion was approved in December 2008. In the interim period, export operations of the SWP and CVP during the winter and spring months were restricted based on the magnitude of reverse flows in Old and Middle rivers and the geographic distribution and risk to delta smelt of entrainment at the export facilities. During the winter and spring months, SWP and CVP exports were limited to reverse flows of not greater than -5,000 cfs and may be reduced to as low as -2,000 cfs when delta smelt are at high risk of being entrained. These operating restrictions were intended to provide protection to delta smelt, reduce the potential risk of entrainment losses to the overall abundance, and reduce the effects of operations on population viability of delta smelt. The operating restrictions, in combination with other export limitations (e.g., SWRCB D-1641, operations to reduce the incidental take of winter-run and spring-run Chinook salmon and steelhead) have resulted in reductions in water supply deliveries to SWP and CVP contractors.

In December 2008, the USFWS released a biological opinion on the proposed operations of DWR and USBR, indicating that "coordinated operations of CVP and SWP diversion facilities, as proposed, are likely to jeopardize the continued existence of delta smelt" (USFWS 2008). The new biological opinion supplanted the 2007 Judge Wanger court decision when approved. The biological opinion details reasonable and prudent alternative actions to reduce the likelihood of jeopardy that include improvements to flow conditions, restoration of tidal marsh and associated subtidal habitat in the Delta and Suisun Marsh, and a comprehensive monitoring plan.

A5.7 RECOVERY GOALS

The USFWS recovery strategy for delta smelt is contained in the Sacramento-San Joaquin Delta Native Fishes Recovery Plan (USFWS 1996), which also includes the longfin smelt, Sacramento splittail, green sturgeon, Sacramento perch, and three races of Chinook salmon. The objective of the Delta Native Fishes Recovery Plan for delta smelt is to remove delta smelt from the federal list of threatened species through restoration of its abundance and geographic distribution. The basic strategy for recovery is to manage the estuary in such a way that it provides better habitat for native fish in general and delta smelt in particular. The Recovery Plan defines restoration as a return of the population to pre-decline levels.

Based on the available information at the time, the 1996 recovery plan outlined a number of measurable criteria that could be used to evaluate the status of delta smelt. Delta smelt were to be considered restored when its population dynamics and distribution pattern within the estuary were similar to those that existed in the pre-decline 1967-1981 period. Restoration was to be assessed when the species satisfied both distributional and abundance criteria. The abundance criteria outlined in the 1996 recovery plan for delta smelt were met and the USFWS conducted a status review of the species in compliance with the terms of a settlement agreement. After reviewing the available information, the USFWS concluded that significant threats to the population recovery of delta smelt remain and that delta smelt should continue to be listed as a threatened species under the federal ESA.

Since 1996, new significant findings regarding the status and biology of and threats to delta smelt have emerged. The USFWS has the responsibility to review and update the recovery plan for these species. To accomplish this task, USFWS has formed the Delta Native Fishes Recovery Team to assist in the preparation of this updated recovery plan. An updated recovery plan is currently expected to be released in the near future.

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A5.8.2 Federal Register Notices Cited

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A5.8.3 Personal Communications

Bill Bennett [1] (Professor of Biology, UC Davis) phone conversation with Rick Wilder and Pete Rawlings about factors influencing the distribution and abundance of delta smelt. July 17, 2007.

Bill Bennett [2] (Professor of Biology, UC Davis) discussion with Rick Wilder, Pete Rawlings, and Chuck Hanson about causes of reduced rearing habitat to delta smelt. January 23, 2008.

Kathleen Fisch (PhD Student, UC Davis) email to Rick Wilder about delta smelt-wakasagi hybrid incidence. October 4, 2010.

Kevin Fleming (CDFG) email to Victoria Poage (USFWS) about habitat of delta smelt. April 13, 2007.

Steve Ford (DWR) communication with Victoria Poage (USFWS) about toxicity to delta smelt. 2008.

Bruce Herbold (EPA) presentation to Water Education Foundation Delta Tour about the Pelagic Organism Decline, Clarksburg, CA. June 6, 2007.

Matt Nobriga (Biologist, DFG). Phone conversation with Rick Wilder about non-native species in the Delta. April 14, 2008.

APPENDIX A6. LONGFIN SMELT *(SPIRINCHUS THALEICHTHYS)*

A6.1 LEGAL STATUS

Longfin smelt is not currently listed under the federal Endangered Species Act (ESA). However, the species was recently listed as threatened under the California ESA. The Bay-Delta population of longfin smelt was petitioned for threatened status under the Federal ESA in 1992. However, the petition was denied because the population was surviving well in areas outside the Bay-Delta estuary. The population was deemed insignificant to the entire species and was not deemed sufficiently reproductively isolated to warrant ESA listing (59 FR 869). More recent evidence from electrophoretic analysis has shown minor differences in allele frequencies between longfin smelt populations inhabiting Lake Washington in Washington state and those in the San Francisco Bay-Delta estuary, but gene frequencies differed enough to suggest that current gene flow between these two populations is restricted (Stanley et al. 1995). The Bay-Delta population appears to be more geographically isolated from other West Coast longfin smelt populations than previously thought (Moyle 2002). In 2007, the Bay Institute, Center for Biological Diversity, and Natural Resources Defense Council (2007a, b) petitioned to have the Bay-Delta longfin smelt population listed as a threatened species under both the California and Federal ESAs. On May 6, 2008, the U.S. Fish and Wildlife Service (USFWS) ruled that a status review for longfin smelt was warranted (73 FR 24911). On April 9, 2009, the USFWS determined that the Bay-Delta population did not meet the legal criteria for protection as a species subpopulation under the ESA (74 FR 16169).

In December 2007, the California Department of Fish and Game (DFG) completed a preliminary review of the longfin smelt petition (DFG 2007a) and concluded that there was sufficient information to warrant further consideration by the California Fish and Game Commission. On February 7, 2008 the California Fish and Game Commission designated the longfin smelt as a candidate for potential listing under the California ESA. On June 26, 2009, the California Fish and Game Commission ruled to list the status of longfin smelt as threatened under the California ESA.

A6.2 SPECIES DISTRIBUTION AND STATUS

A6.2.1 Range and Status

Populations of longfin smelt occur along the Pacific Coast of North America, from Hinchinbrook Island, Prince William Sound, Alaska to the San Francisco estuary (Lee et al. 1980). Although individual longfin smelt have been caught in Monterey Bay (Moyle 2002), there is no evidence of a spawning population south of the Golden Gate. The Bay-Delta population is the southernmost, and also the largest, spawning population in California (see Figure A-6a).

DRAFT

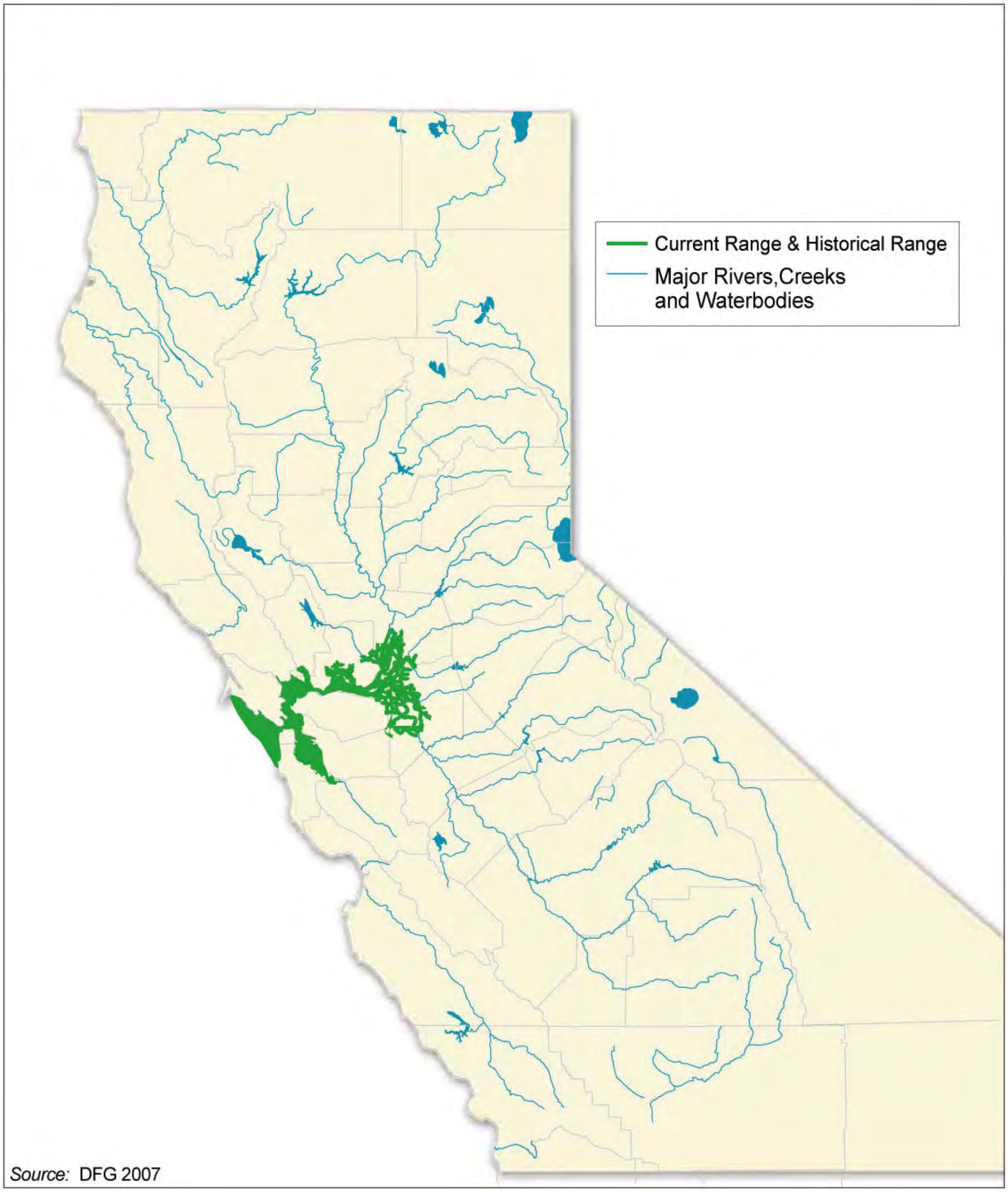


Figure A-6a. Longfin Smelt Range in California

Small and perhaps ephemeral longfin smelt spawning populations have been documented or suspected to exist in Humboldt Bay, the Eel River estuary, the Klamath River estuary, the Eel River drainage, and the Russian River (Moyle 2002, Pinnix et al. 2004).

Longfin smelt seasonally inhabit the entire Bay-Delta estuary, typically in the lower Sacramento River downstream of Rio Vista and the lower San Joaquin River downstream of Medford Island, Suisun Bay, San Pablo Bay, and San Francisco Bay including South San Francisco Bay (see Figure A-6b). During non-spawning periods, individuals are most often concentrated in Suisun, San Pablo and north San Francisco bays (Baxter 1999, Moyle 2002). The species is also common in nearshore coastal marine waters outside the Golden Gate Bridge in late summer and fall (Baxter 1999, Sakuma, pers. comm. 2003). Longfin smelt are periodically caught in nearshore ocean surveys (City of San Francisco, unpublished data, Sakuma pers. comm.), suggesting that some individuals emigrate from or immigrate into the estuary.

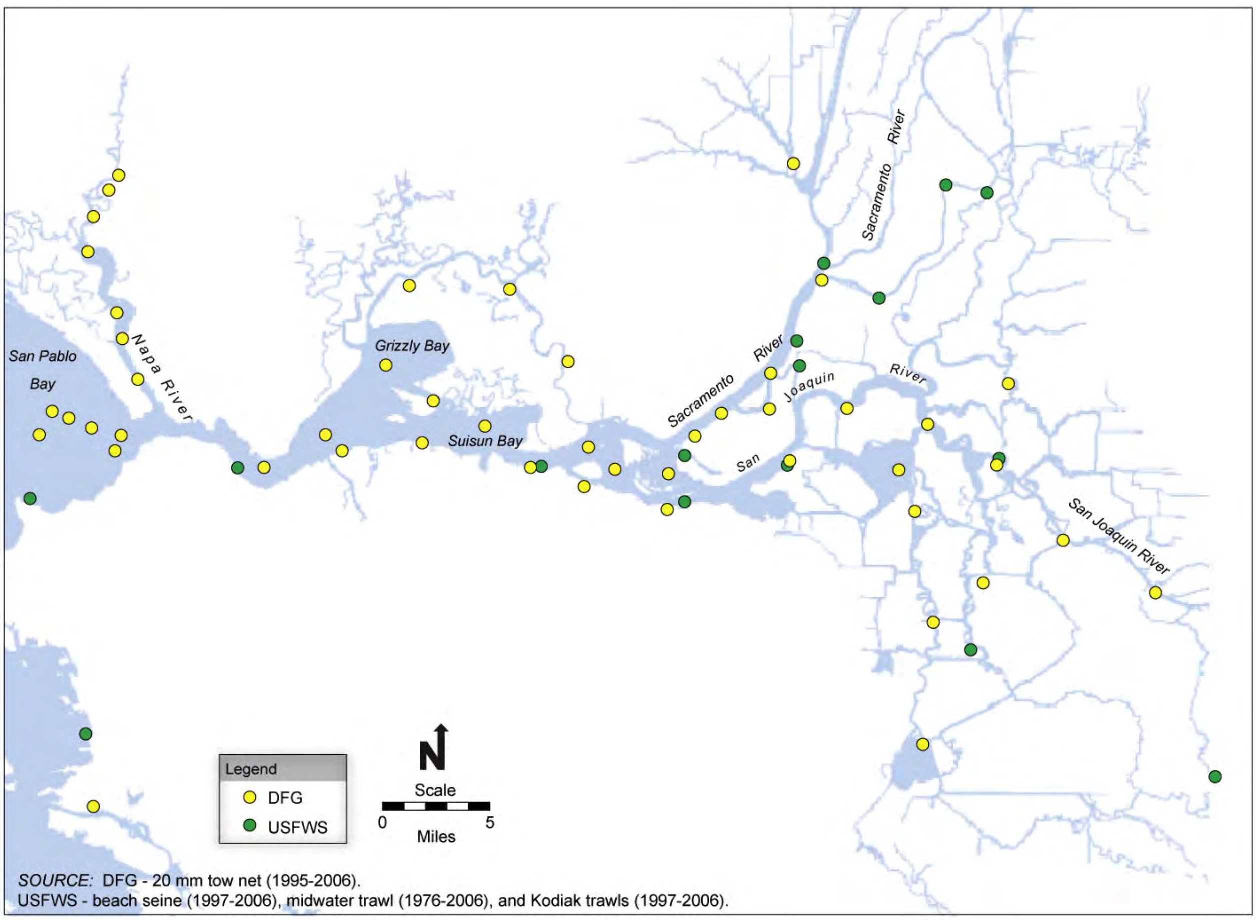


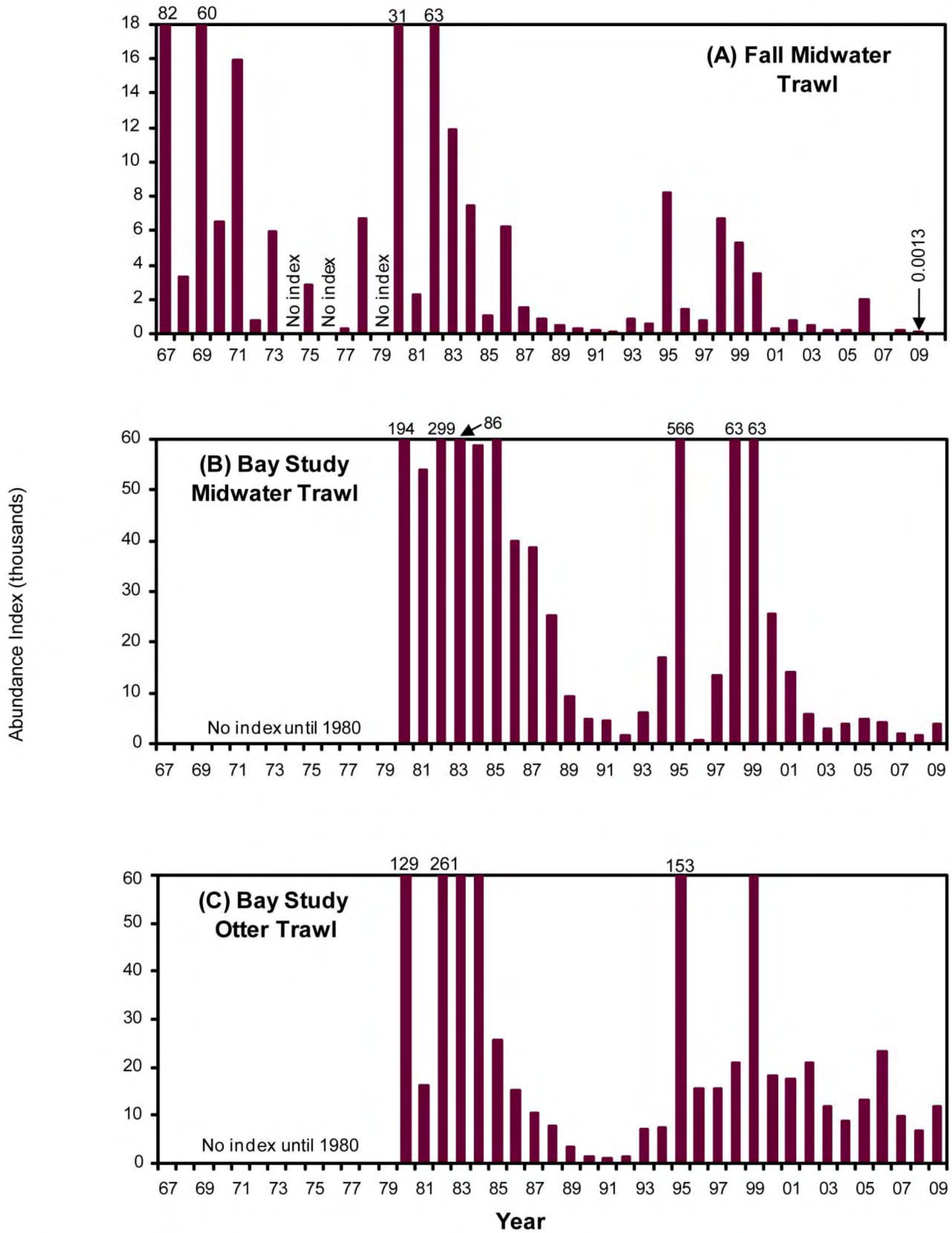
Figure A-6b. Historical Sampling Locations Where Longfin Smelt Have Been Captured Since 1976

Longfin smelt abundance within the Bay-Delta estuary has been highly variable as reflected in the DFG fall midwater trawl surveys and Bay study surveys (see Figure A-6c). The DFG fall midwater trawl samples approximately 100 locations throughout the Bay-Delta system during the period from September through December each year. The survey has been conducted since 1967 and is considered to represent the best long-term record of the index of longfin smelt abundance in the Bay-Delta estuary. Additional information on trends in abundance of longfin smelt inhabiting the estuary is available from the DFG Bay fishery surveys that have sampled monthly since 1980 at a wide range of locations using both an otter trawl and midwater trawl. Since the fall midwater trawl surveys and Bay fishery surveys show similar trends in abundance of longfin smelt (Hieb et al. 2005), the following description of trends in the status of longfin smelt is based on results of the long-term DFG fall midwater trawl surveys.

Indices of longfin smelt abundance (see Figure A-6c) are characterized by high variability among years. Abundance indices were greatest in 1967 and 1969 followed by a second peak in abundance in 1980 and 1982. High abundance indices have generally been associated with years when spring Delta outflow has been high. Abundance indices have typically been low in years when Delta outflow in the spring is low, such as the drought conditions that occurred in 1976 and 1977 and during the early 1990s drought. The trends in longfin abundance also show a general pattern of declining abundance over the 1967 through 2009 survey period. In recent years, longfin smelt abundance was greatest in 1995 followed by a general decline in abundance between 1998 and 2009. The abundance index based on the DFG fall midwater trawl survey conducted in 2007 was the lowest on record over the 1967 to 2009 survey period. Fall midwater trawl abundance indices suggest that abundance of longfin smelt within the Bay-Delta estuary has declined by over 95 percent since the survey began.

Correlations between longfin smelt abundance indices and various environmental parameters suggest that freshwater outflow from the Delta during the longfin smelt larval and early juvenile period (January-June) has a strong influence on longfin smelt abundance (see Figure A-6d) (Moyle 2002).

Although there was a four-fold decline in longfin smelt abundance after the 1987 invasion of the overbite clam, there was no change in the slope of the relationship between freshwater outflow and longfin smelt abundance (see Figure A-6d) Kimmerer 2002a, Sommer et al. 2007). Furthermore, although Delta outflow conditions were relatively high in 2003, 2005, and 2006, reflecting wet and above normal hydrologic conditions, longfin smelt abundance did not increase (as would be expected based on the 1987 to 2000 relationship; Sommer et al. 2007). There appears to be yet a further reduction in the height of the abundance-flow relationship since 2001, although the slope of the relationship remains unchanged (see Figure A-6d). This finding suggests that an additional factor or factors may now be limiting the Bay-Delta population response. When longfin smelt abundance is low, it becomes more difficult to accurately assess their geographic distribution and abundance.



Source: DFG, unpubl. data; IEP, unpubl. data.

Figure A-6c. Annual Abundance Indices of Longfin Smelt from 1967-2009 in (A) Fall Midwater Trawl, (B) Bay Study Midwater Trawl, and (C) Bay Study Otter Trawl

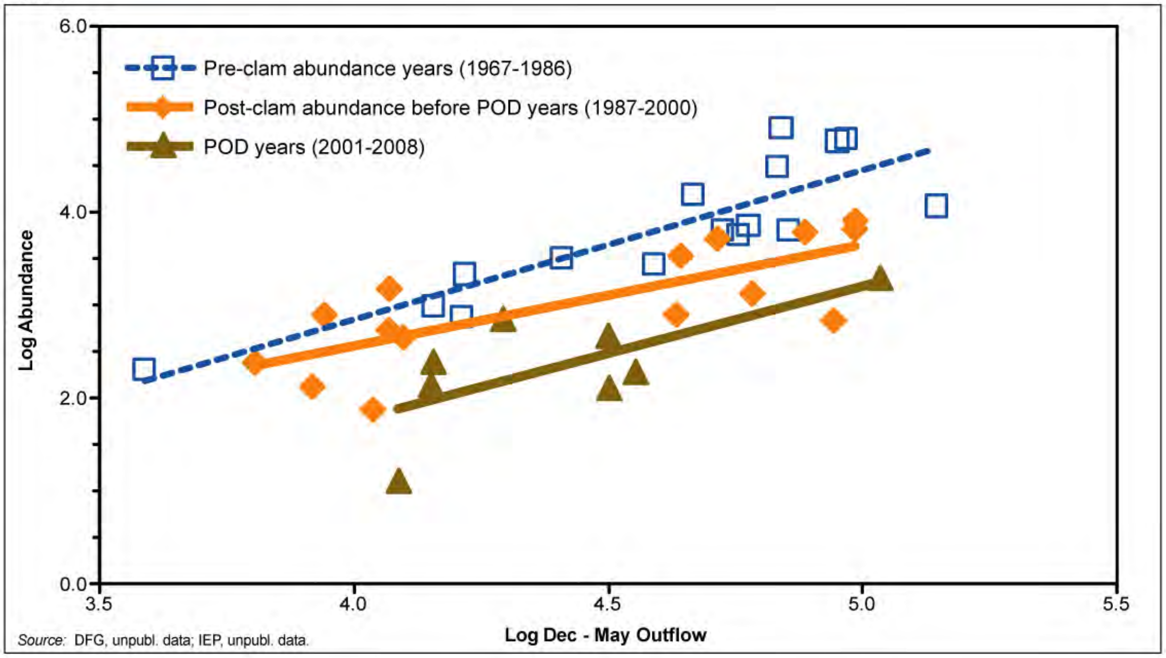


Figure A-6d. Longfin Smelt Abundance ($\log_{10}$) from DFG Fall Midwater Trawl Survey as a Function of Mean Delta Outflow from December through May ($\log_{10}$)

Longfin smelt migration patterns and geographic distribution patterns within the Bay-Delta estuary have remained the same throughout the period of record (1967-present, fall midwater trawl; 1980-present, Bay Study trawls). Results of fishery surveys suggest that the geographic distribution of pre-spawning adult longfin smelt in the winter and early spring does not vary substantially in response to seasonal and inter-annual variation in inflows to the Delta. It has been hypothesized that the pre-spawning longfin smelt distribution is determined by staging and foraging prior to spawning and associations with suitable habitat conditions for spawning. The geographic distribution of larval and early juvenile lifestages of longfin smelt may be influenced by freshwater inflows to the Delta during the late winter and spring, possibly influencing larval planktonic transport rates from the upstream spawning habitat to the downstream estuarine portions of the Delta. In addition, when Delta inflows are high the location of the low salinity zone is further west (downstream) and larval and early juvenile delta smelt are frequently observed further downstream within Suisun Bay.

No spawning habitats have been specifically identified for longfin smelt, but based on the collection of larvae, most spawning is believed to take place in the Sacramento River near or downstream of Rio Vista, and downstream of Medford Island on the San Joaquin River (Wang 1986). Historically, spawning longfin smelt were also common in Suisun Marsh; in recent years, very few adult, spawning-age longfin smelt have been collected in Suisun Marsh (DFG, unpubl. data). Larval longfin smelt have been found concentrated off the mouth of Coyote Creek, indicating that spawning can take place in tributaries of South San Francisco Bay during periods when freshwater runoff and Delta outflow are high, such as conditions that occurred in 1982 and

1983 (Baxter 1999). Collection of small larvae in the Interagency Ecological Program (IEP) 20 mm tow net surveys suggests spawning regularly occurs in the Napa River.

Larval longfin smelt are typically collected in the region of the estuary extending from the western Delta into San Pablo Bay, but their distribution shifts upstream or downstream in response to Delta outflow (Baxter 1999, Dege and Brown 2004). In years when winter-spring Delta outflow is low, few larvae are transported to San Pablo Bay. In years when winter-spring Delta outflow is high, few larvae remain in the western Delta, but are abundant in San Pablo Bay and may reach northern San Francisco Bay (Baxter 1999, Dege and Brown 2004). Longfin smelt larvae are distributed broadly into all open water habitats and into marsh sloughs (Baxter 1999, Meng and Matern 2001).

The initial distribution of young juveniles correlates positively with that of larvae, both vertically within the water column and geographically. During their first year, juveniles disperse broadly downstream, eventually inhabiting Suisun, San Pablo, and Central and South San Francisco bays and moving into near shore coastal marine habitats in most years (see Figure A-6e) (Baxter 1999, Dege and Brown 2004, Hieb and Baxter 1993, Moyle 2002). Juveniles move from offshore shoals into channels during summer and fall (Rosenfield and Baxter 2007).

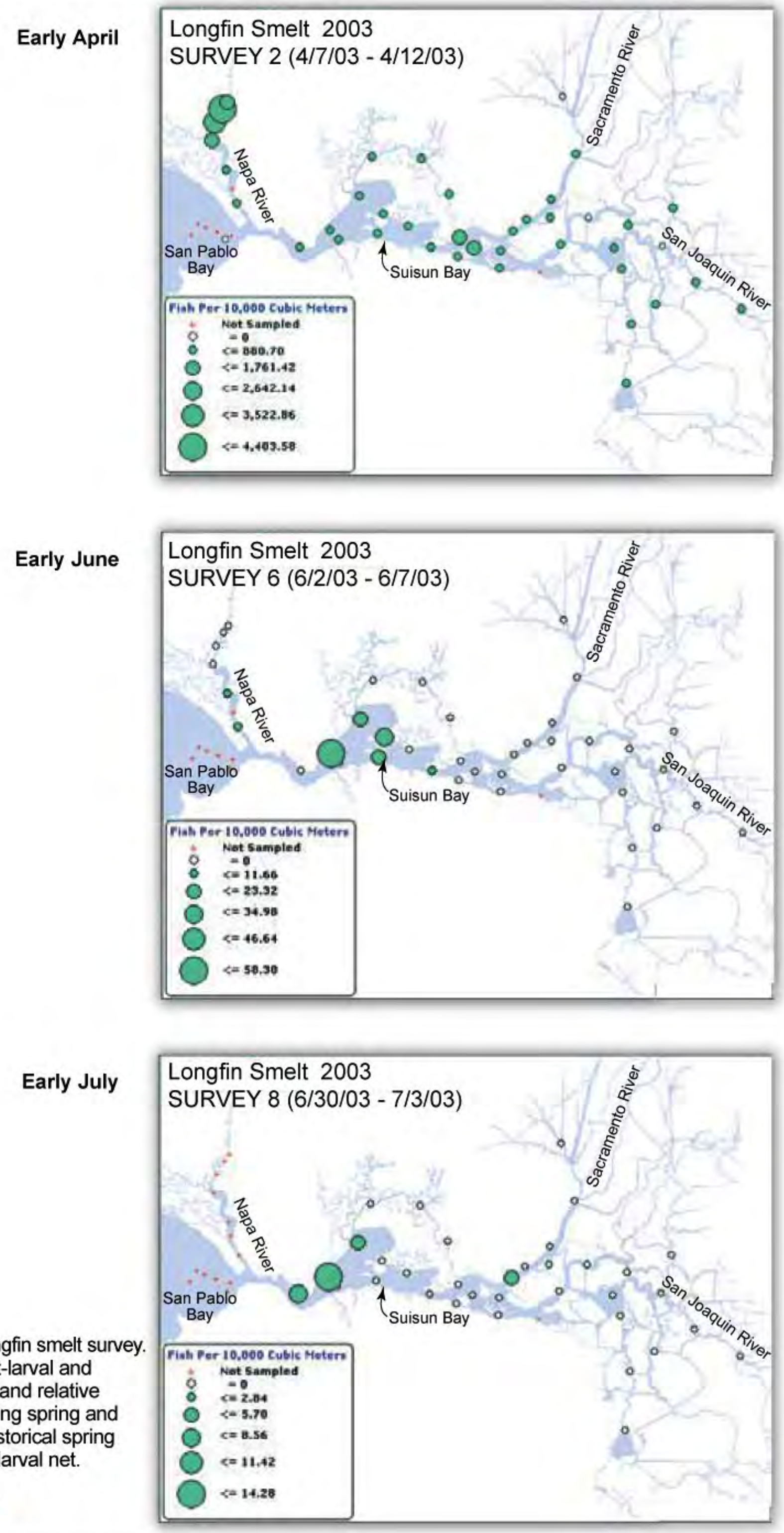
Longfin smelt in their second year of life (age 1) are typically distributed from the western Delta through South San Francisco Bay during January through March. Their distribution then moves toward the Central San Francisco Bay, such that by August and September few, if any, are collected outside of Central San Francisco Bay (Baxter 1999). During the summer longfin smelt are also common in nearshore coastal waters (City of San Francisco unpubl. data, Sakuma pers. comm.). As longfin smelt begin to mature in the fall, they re-inhabit the entire estuary and begin migrating upstream toward freshwater (Baxter 1999, Rosenfield and Baxter 2007).

A6.2.2 Distribution and Status in the Plan Area

Longfin smelt occur primarily in the lower Sacramento River (downstream of Rio Vista), lower San Joaquin River, and western Delta and Suisun Bay within the Plan Area (see Figure A-6a). Longfin smelt occur in relatively low abundance in the south Delta as reflected in results of DFG fishery sampling and fish salvage monitoring at the State Water Project (SWP) and Central Valley Project (CVP) export facilities. The typical distribution of juvenile and adult longfin smelt (brackish water and coastal marine waters of San Pablo and San Francisco bays) is downstream of the Plan Area.

A6.3 Habitat Requirements and Special Considerations

Longfin smelt inhabiting the Bay-Delta estuary are thought to spawn in freshwater or slightly brackish water over sandy or gravel substrates at temperatures ranging from 7 to 14.5 °C (44.6 to 58.1 °F) (Moyle 2002).



Source: CDFG 2003, 20 mm Longfin smelt survey. This annual survey monitors post-larval and juvenile longfin smelt distribution and relative abundance every two weeks during spring and early summer throughout their historical spring range using a 1600 µm egg and larval net. For more information see http://www.delta.dfg.ca.gov/data/20mm/description.asp

Figure A-6e. Example of Distribution of Post-Larval and Juvenile Longfin Smelt in Spring-Summer of a Representative Above Normal Water Year

Other populations of longfin smelt inhabiting West Coast waters are present in coastal estuaries or may complete their entire life cycle in freshwater (Dryfoos 1965, Moulton 1974), indicating that there is no lower limit to salinity tolerance for any life stage. Collections of larval and juvenile longfin smelt smaller than 50 mm fork length (FL) within the Bay-Delta showed that 90 percent of the individuals inhabited areas with salinities lower than 18 parts per thousand (ppt) (Baxter 1999). Healthy individuals 20 mm FL and larger have been captured in salinities of 32 ppt (ocean water) and along the open coast, suggesting that high salinity may be limiting the geographic distribution for only a small portion of their lifecycle, if at all (Baxter, unpubl. data).

Spawning in the Bay-Delta estuary occurs immediately before, and for several months after, the coldest water temperatures of the year (Baxter 1999, Orsi 1999). Movement patterns based on catches in DFG fishery sampling suggest that longfin smelt actively avoid water temperatures greater than 22 °C (72 °F) (see Figure A-6d). Longfin smelt do not occupy areas with temperatures greater than 22 °C (72 °F) in combination with salinities greater than 26 ppt. These conditions occur between August and September almost annually in South San Francisco Bay and periodically in shallower portions of San Pablo Bay. Spawning is thought to occur in freshwater over sand, gravel, rocks, or aquatic vegetation (Wang 1991).

The effect of turbidity on longfin smelt geographic distribution or habitat preferences is unknown. However, longfin smelt larvae hatch coincident with annual peak Delta outflows, which typically coincide with high turbidity. Also, larval and older life stages of longfin smelt possess a well developed olfactory system. The development of the olfactory system at an early lifestage suggests its use in food acquisition (S. Foott, pers. comm.).

A6.4 LIFE HISTORY

An unknown fraction of the longfin smelt population migrates to the marine environment during both their first and second years of life; some may remain in the marine environment from their first year until they return to the estuary to spawn near the end of their second year (rarely their third). It is unknown whether marine residence is necessary for proper egg development, but the extremely limited number of age1 smelt captured upstream of Central San Francisco Bay during fall suggests that salinity or, more likely, higher temperatures may be a factor affecting the seasonal distribution of smelt within the estuary.

Upon hatching from adhesive eggs (primarily January-April), buoyant longfin smelt larvae rise toward the surface and are transported downstream by surface currents resulting from both river flow and tidal mixing of fresh and marine waters. Flow rates are positively related to downstream transport of the planktonic larvae (Hieb and Baxter 1993, Baxter 1999, Dege and Brown 2004). Larval longfin smelt remain in the upper part of the water column until they reach 10 to 15 mm, after which they move to the middle and bottom parts of the water column (Hieb and Baxter 1993, Bennett et al. 2002, Moyle 2002).

Larval and small juvenile longfin smelt feed on copepods and other small crustaceans (Moyle 2002). In Suisun Bay, at low salinities, the non-native copepods, *Pseudodiaptomus* and *Acanthocyclops,* dominate the diet of small juvenile smelt in summer (Hobbs et al. 2006). Mysid shrimp become important in the diets of larger juvenile and adult longfin smelt (Moyle 2002, Feyrer et al. 2003, Rosenfield and Baxter 2007). Since the decline of *Neomysis* following the invasion of the overbite clam in the late 1980s, subadult and adult longfin smelt have fed on a broader variety of organisms, but mysids remain their primary food item (Moyle 2002, Feyrer et al. 2003). In fall 2006, a high outflow year, longfin smelt fed predominantly on the introduced mysid, *Acanthomysis*, but consumed other mysids, as well as the copepod *Pseudodiaptomus* and amphipod, *Corophium* (DFG, unpubl. data).

During their first year, juvenile longfin smelt disperse broadly downstream, eventually inhabiting Suisun, San Pablo, and Central and South San Francisco bays, as well as nearshore coastal marine habitat in most years (Hieb and Baxter 1993, Baxter 1999, Moyle 2002, Dege and Brown 2004). Juveniles move from offshore shoals into channels during the summer and fall (Rosenfield and Baxter 2007). This movement may be a response to increasing water temperatures (greater than 20 °C [68 °F]), as does the late summer emigration from South San Francisco Bay (Baxter 1999).

A6.5 Threats and Stressors

Reduced spawning habitat. A primary mechanism responsible for the reduction in spawning habitat for longfin smelt has been the reclamation, channelization, and riprapping of historical freshwater intertidal and shallow subtidal wetlands. Furthermore, reductions of winter and spring Delta outflows over the past several decades as a result of water exports, reservoir storage, and upstream diversions have repositioned the low salinity region of the estuary (X2 location) farther upstream, possibly forcing spawning adult longfin smelt to migrate farther upstream to reach suitable spawning habitat.

Reduced access to rearing habitat. Access to suitable rearing habitat, which is centered in the low salinity zone (Dege and Brown 2004), has likely declined as a result of reductions in Delta outflow over the past 40 years. Reduced access to rearing habitat can result from low net downstream flows slowing the transport of planktonic larval longfin smelt downstream towards suitable rearing habitat in the western Delta and Suisun Bay. The documented correlation between the abundance of longfin smelt in the FMWT and the location of X2 in the winter and spring months (Dec-May; Kimmerer 2002, Kimmerer et al. 2009) is hypothesized to relate to the transport of larval longfin smelt out of the Delta to rearing habitats downstream.

The low salinity zone, when positioned over shallow shoal areas in Suisun Bay in response to high Delta outflows, is thought to be highly productive (Moyle et al. 1992, Bennett et al. 2002). When located upstream, the low salinity zone is confined to the deep river channels, is smaller in total surface area, contains very few shoal areas, may have swifter, more turbulent water currents, and may lack high zooplankton productivity. Hobbs et al. (2006) found evidence that the health and survival of longfin smelt were greater in habitats associated with shallow water. Furthermore, as the distribution of longfin smelt shifts upstream, individuals may become more vulnerable to entrainment by the SWP and CVP export facilities and other diversions within the interior of the Delta.

The Suisun Marsh Salinity Control Gates function to decrease salinity in managed wetlands of Suisun Marsh to support crops that attract game birds for the many duck clubs located throughout the marsh. When in operation, generally from October through May, the control gates near Collinsville divert up to 2,500 cubic feet per second (cfs) of freshwater from upstream flows into the marsh. Because the minimum outflow standard during fall months is 5,000 cfs, a significant proportion of total Delta outflow (up to 50 percent) does not flow through the eastern Suisun Bay region. This diversion has resulted in a measurable increase in salinity in eastern Suisun Bay, which may correspond to a decrease in low salinity habitat for longfin smelt (Fullerton 2007).

Reduced food availability. Reduced food availability for longfin smelt can result from at least seven impact mechanisms.

First, the presence of non-native species has reduced the abundance of food available to longfin smelt. Efficient filter feeding and high abundance of the overbite clam have dramatically reduced phytoplankton and zooplankton abundance in Suisun Bay, the western Delta, and Suisun Marsh since its introduction in the mid 1980s (Kimmerer and Orsi 1996). The Asian clam has also reduced phytoplankton abundance in the Delta, which likely reduced zooplankton abundance (Jassby et al. 2002, Thompson 2007). Other non-native zooplanktivores that may compete for limited available food resources with longfin smelt include threadfin shad, inland silversides, and wakasagi.

Second, much of the floodplain habitat in the Delta and tributary rivers has been eliminated by levees and reclamation. As a result of levee construction, flood control, and increased reservoir storage the frequency of inundation on floodplains that still exist has been reduced (Resources Agency 2007). Floodplains are highly productive due to their shallow, warm, low-velocity water (Sommer et al. 2001a, b) and input of organic material and nutrients from the terrestrial community (Booth et al. 2006). Floodplains are a key source of nutrients and organic material for the Bay-Delta estuary (Sommer et al. 2001a, Harrell and Sommer 2003).

Third, levee construction, island reclamation, and channelization within the Delta have resulted in a substantial reduction in intertidal and shallow-water subtidal wetland/emergent marshes and open water habitat throughout the Delta. Historically, Delta wetlands and shallow-water habitat was expansive and provided large areas of estuarine and freshwater habitat that was highly productive. The significant reduction in tidal and shallow-water subtidal habitat, and an associated reduction in emergent vegetation, nutrient cycling, and the production of phytoplankton, zooplankton, macroinvertebrates, and other aquatic organisms that provide food resources for delta smelt, have been identified as a major factor affecting habitat conditions within for Delta species, such as longfin smelt.

Fourth, SWP and CVP exports and the over 2,200 in-Delta agricultural diversions (Herren and Kawasaki 2001) export zooplankton, nutrients, and organic material that would otherwise support the base of the food web in the Delta, thus reducing food availability for the longfin smelt (Jassby and Cloern 2000, Resources Agency 2007).

Fifth, hydraulic residence time in the Delta has declined as a result of increased channelization and passage of Sacramento River water through the Delta Cross Channel into the central and southern Delta to meet water quality standards and supplies for in-Delta exports. The decreased hydraulic residence time reduces the time available for bacteria to use nutrients and organic carbon and for production of phytoplankton and zooplankton that provide food for longfin smelt and other aquatic species (Jassby et al. 2002, Kimmerer 2002a, 2004, Resources Agency 2007).

Sixth, exposure of phytoplankton and zooplankton to toxics (e.g., pesticides, herbicides) that enter the Delta from point and non-point sources may contribute to the observed low abundance of zooplankton prey species for longfin smelt and other species inhabiting the Bay-Delta (Weston et al. 2004, Luoma 2007, Werner 2007). Although direct impacts of toxics on longfin smelt have not been extensively studied, the indirect effect of toxics on reducing zooplankton and phytoplankton abundance is thought to result in reduced availability of food resources to longfin smelt (Johnson et al. 2010, Werner et al. in press).

Seventh, in addition to the discharge of toxic contaminants, municipal wastewater treatment plants, particularly the Sacramento Regional County Sanitation District's Wastewater Treatment Plant, discharge high loads of ammonia directly into the Sacramento River in the North Delta (Jassby 2008). High concentrations of ammonium, the ionized form of ammonia, may inhibit phytoplankton production in the Sacramento and San Joaquin rivers, as has been found downstream in the Suisun, San Pablo and Central bays (Wilkerson et al 2006, Dugdale et al. 2007), which could result in reduced food production for longfin smelt. Additional research is ongoing to determine if and to what extent, ammonia/um may affect phytoplankton production.

Non-native species. The effect of non-native predators, such as inland silversides, largemouth bass, striped bass, and other fish on the longfin smelt population is largely unknown, but may be important (Bennett and Moyle 1996, Moyle 2002). The establishment of the highly invasive and fast growing aquatic plants, such as the Brazilian waterweed and water hyacinth, has provided habitat for non-native predatory fish, such as centrarchids and striped bass, although no population level effects on longfin smelt have been detected or quantified (Nobriga et al. 2005). These aquatic plants may have had other potentially detrimental impacts to longfin smelt, including competition with native vegetation and reducing dissolved oxygen concentrations and turbidity within their immediate vicinity (Grimaldo and Hymanson 1999, Brown and Michniuk 2007, Feyrer et al. 2007).

The overbite clam has caused dramatic changes to the composition and abundance of phytoplankton and zooplankton communities in the aquatic food web since its introduction into the Bay-Delta estuary (Kimmerer and Orsi 1996). Kimmerer (2002a) asserted that these changes likely reduced food availability for a large assemblage of organisms, leading to reduced recruitment success of longfin smelt and a four-fold reduction in the abundance of longfin smelt (Rosenfield and Baxter 2007).

Reduced turbidity. The observed change in Bay-Delta turbidity has the potential, in combination with other factors, such as non-native species, to fundamentally alter the trophic dynamics of the estuary for species such as longfin smelt. Based on the similarities in life history, seasonal and geographic distribution, pelagic foraging and diet, it has been hypothesized that longfin smelt may have a similar relationship to turbidity as that observed for delta smelt (S. Foot unpubl. data, R. Baxter pers. comm.). Enlarged olfactory organs in longfin smelt suggest that they are well adapted to high turbidity conditions during foraging. As a result, longfin smelt may lose their competitive advantage in foraging to other zooplanktivores when turbidity is low.

Turbidity has decreased over the past several decades in the Delta as a result of a variety of factors (Kimmerer 2004, Wright and Shoellhamer 2004, Feyrer et al. 2007, Fullerton 2007). First, upstream sediment inputs have been reduced due to a range of anthropogenic actions (Jassby et al. 2002, Kimmerer 2004), including depletion of erodible sediments from hydraulic mining in the 1800s, river bank protection, trapping of sediments by dams and reservoirs, levee construction that reduced flood plain inundation and channel meanders, and changes in land use (Wright and Shoellhamer 2004). Wright and Shoellhamer (2004) estimated that the yield of suspended sediments from the Sacramento River declined by approximately one half from 1957 to 2001.

Second, the distribution and abundance of non-native aquatic plant species, particularly *Egeria* and water hyacinth, has increased dramatically over the past 20 years (Nobriga et al. 2005, Brown and Michniuk 2007). Both plants can reduce turbidity by reducing water velocity and trapping fine suspended sediments (Grimaldo and Hymanson 1999, Jassby et al. 2002, Nestor et al. 2003, Hobbs et al. 2006).

Third, the high filtering efficiency of the overbite clam has dramatically reduced phytoplankton and zooplankton abundance in the western Delta and Suisun Bay since its introduction (Kimmerer and Orsi 1996, Jassby et al. 2002, Kimmerer 2002b, 2004). The reduction in phytoplankton in the water column may contribute to increased water clarity and reduced turbidity in the Delta.

Fourth, hydraulic residence in the Delta has declined as a result of increased channelization and the movement of water from the Sacramento River into the interior Delta channels to improve water quality and provide increased supplies to in-Delta exports. SWP and CVP export operations have also directly resulted in changes in the hydrodynamics within Delta channels such as Old and Middle rivers which affect hydraulic residence time. Reduced hydraulic residence time reduces the ability of phytoplankton and bacteria to incorporate nutrients and carbon (Jassby et al. 2002, Kimmerer 2002a, 2004, Resources Agency 2007). This reduction in phytoplankton and zooplankton production contributes directly to reduced turbidity within the Bay-Delta estuary.

Reduced food quality. The zooplankton community inhabiting the Bay-Delta estuary has changed multiple times in response to multiple introductions of non-native species. These changes in the zooplankton species composition have affected the quality of food resources available to longfin smelt because some of the non-native species do not appear to be as suitable of a food resource as the native species (Resources Agency 2007, Sommer 2007). For example, the non-native copepod *Limnoithona* (Orsi and Ohtsuka 1999) is described as lower quality prey for longfin smelt because they are small and have sufficient swimming ability to avoid capture (Orsi and Ohtsuka 1999, B. Herbold pers. comm.). As a result, foraging efficiency of longfin smelt may have decreased (Resources Agency 2007). A decrease in foraging efficiency and/or the availability of suitable prey for various life stages of longfin smelt may have resulted in reduced growth, survival, and reproductive success contributing to reduced population abundance.

Entrainment. The effect of entrainment on the population dynamics and abundance of longfin smelt remains largely unquantified. Because longfin smelt tend to be mostly estuarine, they likely spend most of their life (approximately 1.5 years) downstream of the influences of the SWP/CVP facilities (see Figure A-6e). However, entrainment during winter months when spawners move upstream may be higher and particularly detrimental to the population because it results in mortality of not only pre-spawning and spawning adults, but also their potential progeny. Guerin et al (2008, in review) found significant correlations for longfin smelt as reported earlier for delta smelt between SWP winter salvage of adult smelt and subsequent fall mid-water trawl (FMWT) index of smelt with 1 and 2 year lags over the past 12 years. More recent work shows that SWP winter salvage of adult smelt normalized to the prior FMWT correlates strongly with subsequent FMWT for delta smelt over a longer record. These relationships do not establish causality, but they are an indicator that winter salvage at the SWP may be a factor to be considered.

The relationship between Delta outflow during the late winter and spring and the DFG fall midwater trawl longfin smelt index (see Figure A-6d) may be partially explained by entrainment vulnerability relative to the geographic distribution of the longfin smelt population. In high outflow years, salvage rates are lower, suggesting that longfin smelt may not be vulnerable to the SWP and CVP exports when the population is located farther west in Suisun Bay and further downstream.

There are over 2,200 small agricultural diversions in the Delta (Herren and Kawasaki 2001). Although these diversions generally take water near the bottom, the intakes may entrain water near the surface at low tide; therefore, the vulnerability of a pelagic species such as juvenile and adult longfin smelt may be reduced. Planktonic larval longfin smelt may have a greater vulnerability to entrainment into diversions than older life stages that have greater swimming ability and may inhabit areas further offshore and in the upper portions of the water column. It has been hypothesized that, although juvenile and adult longfin smelt may avoid entertainment at unscreened water diversions, planktonic longfin smelt larvae are expected to be distributed within the water column and have weak swimming performance and, therefore, may be vulnerable to entrainment losses in larger numbers than suggested by results of investigations of juvenile and adult smelt. Many agricultural diversions are located within longfin smelt spawning and larval rearing habitat. The impact of entrainment mortality at these diversions on the longfin smelt population abundance has not been quantified.

Power plants in Antioch and Pittsburg have the ability to entrain large numbers of longfin smelt, particularly because longfin smelt tend to be located near these facilities for most of the year (Matica and Sommer 2005, C. Hanson unpubl. data). However, use of cooling water is currently low with the retirement of older units. According to recent regulations by the State Water Resources Control Board, units at these two plants must be equipped with a closed cycle cooling system by 2017 that eliminates fish entrainment.

Exposure to toxins. Exposure of longfin smelt to toxic substances can result from point and non-point sources associated with agricultural, urban, and industrial land uses. Longfin smelt can potentially be exposed to these toxic materials, including pesticides, herbicides, endocrine disrupting compounds, and metals, during their period of residence within the Bay-Delta. There are no known studies that directly link mortality of longfin smelt with exposure to toxic chemicals within the Bay-Delta estuary (S. Foott unpubl. data, R. Baxter pers. comm., Resources Agency 2007). However, longfin smelt spawn during winter months when non-point runoff of pesticides tends to be the greatest. The pesticide diazinon is known to reduce growth and increase spinal deformities in Sacramento splittail (Teh et al. 2004), but effects of diazinon on longfin smelt have not been investigated. Reports during January 1997 indicated that flooding along the Feather River dispersed fuel and agricultural chemicals into the water column during a period when longfin smelt larvae were hatching in high numbers; the subsequent 1997 year class was low given the high winter outflow, although a direct cause and effect linkage with exposure to toxics was not documented. Kuivila and Moon (2004) sampled pesticide concentrations within the Delta and west to Chipps Island for 3 years (1998-2000) during April-June. Their water samples contained multiple pesticides, but at individual concentrations well below lethal 96-hr LC50 concentrations for fishes. In 1999 and 2000, sizable, but uncalculated fractions of the longfin smelt population overlapped their pesticide sampling area (see http://www.delta.dfg.ca.gov/data/20mm/CPUE_map.asp), although no known direct link between chemical concentration and larval mortality was established.

The short life span (less than 3 years) and location of their food source in the food web (zooplankton are primary consumers) reduce the ability of toxic chemicals to bioaccumulate in the tissue of longfin smelt (Moyle 2002). Their location in the water column may further reduce the probability of some toxic impacts by those chemicals that are sequestered quickly by sediments (i.e., pyrethroids; B. Herbold pers. comm). Additional research is needed to investigate the potential risk of exposure to toxic chemicals at concentrations and exposure durations typical of Bay-Delta conditions on various life stages of longfin smelt. To date, no formal risk assessment has been performed on the potential lethal and sublethal effects of toxics to longfin smelt population dynamics, although investigations of the toxicity of contaminants to larval delta smelt (Werner unpubl. data, as cited in Baxter et al. 2008) are being undertaken as part of the Interagency Ecological Program's studies of pelagic organism decline.

Ammonia discharged from municipal wastewater treatment plants may contribute to localized toxicity in longfin smelt. Werner et al. (2008) found that water samples near the Sacramento Regional County Sanitation District's wastewater treatment plant effluent reduced 4-day survival of larval delta smelt in 2006, but did not affect survival even after 7 days in 2007. Furthermore, there were two instances of significant larval delta smelt mortality from POD bioassays collected from the Sacramento River in June and July 2007 that had relatively low turbidity and salinity and moderate levels of ammonia (Werner unpubl. data, as cited in Baxter et al. 2008). Exposure to ammonia may have similar effects on longfin smelt. The form and toxicity of ammonia/um changes based on pH and it has been hypothesized that changes in pH of the Delta receiving waters may change in response to algal growth, discharges from managed wetlands and duck clubs, and agricultural return flows that result in ammonia toxicity. These potential water quality interactions and the effects of discharging ammonia from a number of wastewater treatment plants located throughout the Sacramento and San Joaquin rivers, Delta, Suisun Bay and Suisun Marsh on the health and survival of delta smelt and other aquatic species are under investigation.

Consistent evidence of direct toxicity of contaminants to smelt within the Delta is lacking (Werner et al. 2008); however, there is growing evidence that toxics may have indirect effects on longfin smelt. For example, invertebrate prey of longfin smelt are affected by toxics (Weston et al. 2004, Luoma 2007, Werner 2007), reducing food availability of longfin smelt. Additionally, the nitrate uptake by and production of phytoplankton, the base of the food web that supports longfin smelt, may be inhibited by ammonia concentrations in the North Delta as has been demonstrated for phytoplankton in San Francisco and San Pablo bays (Dugdale et al. 2007). There is also evidence that toxics may cause sublethal impacts to longfin smelt that make them more vulnerable to other sources of mortality (Werner 2007). Most, if not all, pyrethroids are potent neurotoxicants (Bradbury and Coats 1989, Shafer and Meyer 2004) and have immunosuppressive effects (Madsen et al. 1996, Clifford et al. 2005). In addition, these compounds and their breakdown products can act as endocrine disrupting compounds by disrupting hormone-related functions (Go et al. 1999, Tyler et al. 2000, Perry et al. 2006, Sun et al. 2007). Esfenvalerate, a common pyrethroid insecticide, has been shown to increase the susceptibility of juvenile fall-run Chinook salmon to infectious hematopoietic necrosis virus (Clifford et al. 2005), reduce swimming ability, and increase susceptibility to predation in larval fathead minnows (Floyd et al. 2008). In delta smelt, exposure to environmentally relevant pyrethroid concentrations resulted in significant swimming abnormalities, which were strongly linked with downregulation of genes involved in neuromuscular activity (Connon et al. 2009).

Exposure to copper contamination can also result in significant sublethal effects on Delta fish species, with implications for their vulnerability to other stressors. Environmentally relevant copper concentrations are shown to result in significant immunosuppressive effects (Hetrick et al. 1979) and impair olfactory function and eliminate the predator avoidance response in fish (Sandahl et al. 2006; Werner et al. in press). Swimming abnormalities have been observed after exposure to copper concentrations as low as 0.25 of the chemical's LC50 values (Little and Finger 1990; Oros and Werner 2005). Dissolved copper causes acute toxicity to the calanoid copepod, *Eurytemora affinis,* in the north and south Delta (Teh 2009) and impairs the sensory function of juvenile salmonids (Hecht et al. 2007), specifically related to predator avoidance behavior. Moreover, specific concentrations of dissolved copper correspond to sublethal endpoints such as primary production and salmonid growth (Hecht et al. 2007). Longfin smelt may be affected in a similar manner. Additionally, negative synergistic effects have been documented such that the presence of copper in combination with ammonia is more toxic to aquatic organisms than either toxicant individually (Herbert and Vandyke 1964).

Predation. Predation by introduced predators, such as inland silversides, striped bass, and centrarchids, has been identified as a potential stressor on smelt populations (Sommer et al. 2007, Rosenfield 2010), but the importance of predation for longfin smelt abundance is thought to be low (Nobriga and Feyrer 2008, Rosenfield 2010). Information regarding the impact of predation on longfin smelt is limited; however, inland silversides are believed to prey on larval longfin smelt, and predation by striped bass adults likely results in mortality for the juvenile and adult lifestages (Rosenfield 2010). Larval longfin smelt are not strong swimmers, and are thus particularly vulnerable to predation (Wang 1986). Various factors such as turbidity, outflows, and exposure to contaminants are likely to influence the susceptibility of longfin smelt to predation (Rosenfield 2010).

Predation has been implicated as an important factor affecting production of juvenile longfin smelt, in part due to the correspondence between freshwater flows, the volume of turbid habitat, and the young-of-year class size for longfin smelt (Rosenfield 2010). The coincidence of the increase in inland silverside abundance and decline in longfin smelt abundance also provides evidence of the potential importance of predation as a stressor to longfin smelt. However, increases in predation are not believed to be responsible for the most recent decline in the longfin smelt population. Although striped bass are likely to be major predators of longfin smelt, their populations have declined substantially in recent years and any impact they have on longfin smelt populations is also expected to have declined (Rosenfield 2010). In addition, inland silversides are predatory, but they prefer shallow-water habitats where juvenile and sub-adult longfin smelt are rare. Consequently, their impact as predators of juvenile longfin smelt is likely limited (Rosenfield 2010).

Elevated water temperature. Temperature affects the metabolic requirements and physiological processes of longfin smelt. Beyond a certain threshold, temperature increases are expected to cause increases in longfin smelt mortality. The temperature limitations and sublethal impacts of temperature variation on longfin smelt are unknown. Given the northerly distribution of longfin smelt and their probable derivation from a marine ancestor, it is possible that longfin smelt distribution and abundance in the Estuary are limited by high temperatures, particularly during summer months. Rosenfield and Baxter (2007) noted several aspects of their distribution patterns that would be consistent with temperature limitation. Temperatures below the longfin smelt minimum temperature threshold are not likely to occur in this estuary.

Low dissolved oxygen. In order to respire, longfin smelt require sufficient dissolved oxygen concentrations. Below a certain threshold, longfin smelt mortality would be expected to increase rapidly with decreasing dissolved oxygen levels (or increased time of exposure to low dissolved oxygen levels). No studies on the dissolved oxygen requirements of longfin smelt are available. Given the species' range (and limited historical exposure to high temperature/low dissolved oxygen conditions) and distribution within this ecosystem, this fish may be expected to have fairly high requirements for dissolved oxygen concentrations. For example, longfin smelt requirements for dissolved oxygen are expected to equal or exceed those of Delta smelt because the latter species specializes in the warmer habitats with lower dissolved oxygen. Given their pelagic distribution, regular exposure to low dissolved oxygen conditions is unlikely. Low dissolved oxygen conditions may limit longfin smelt use of the lower San Joaquin as DO levels are frequently extremely low in that area.

A6.6 RELEVANT CONSERVATION EFFORTS

The CALFED Multi-Species Conservation Strategy (CALFED 2000) designates longfin smelt as an "R" species and states that the goal is to "achieve recovery objectives identified for longfin smelt in the recovery plan for the Sacramento/San Joaquin Delta native fishes" (USFWS 1996). However, no conservation efforts in the recovery plan specifically target longfin smelt; all are referenced to delta smelt.

The Delta Regional Ecosystem Restoration Implementation Plan (DRERIP) was formed to guide the implementation of CALFED Ecosystem Restoration Plan elements within the Delta (DFG 2007b). The DRERIP team has created a suite of ecosystem and species conceptual models, including longfin smelt that document existing scientific knowledge of Delta ecosystems. The DRERIP Team has used these conceptual models to assess the suitability of actions proposed in the Ecosystem Restoration Plan for implementation. DRERIP conceptual models have been used in the analysis of proposed BDCP conservation measures. Additional enhancement and protective actions are also being identified as part of mitigation programs for various projects, biological opinions, and regional conservation planning efforts.

Modifications in the seasonal timing of SWP and CVP export operations on the longfin smelt population are currently being evaluated.

The Environmental Water Account (EWA) is intended to contribute to the protection, restoration, and recovery needs of fish, including longfin smelt, while still providing water supply reliability. However, analysis of the biological response of longfin smelt and other fish species to EWA actions in the past have failed to demonstrate significant protection or benefits to the overall populations of longfin smelt and other fish species.

In January 2005, the Interagency Ecological Program established a new Pelagic Organism Decline (POD) work team to investigate the causes of the recently observed rapid decline in populations of pelagic organisms, including longfin smelt, in the upper San Francisco Bay estuary (Baxter et al. 2008). Since that time, numerous studies have been conducted to determine the cause of the POD. Based on results of these studies and relevant studies undertaken by others, the work team has developed conceptual models to further the understanding POD. The Resources Agency prepared a Pelagic Fish Action Plan in March 2007 to address POD (Resources Agency 2007). The action plan identifies 17 actions that are being implemented or that are under active evaluation to help stabilize the Delta ecosystem and improve conditions for pelagic fish.

A6.7 Recovery Goals

Longfin smelt is included in the 1996 Sacramento-San Joaquin Delta Native Fishes Recovery Plan, which also includes the delta smelt, Sacramento splittail, green sturgeon, Sacramento perch, and three races of Chinook salmon. The USFWS has the responsibility to review and update the recovery plan for these species. To accomplish this task, the USFWS has formed the Delta Native Fishes Recovery Team to assist in the preparation of this updated recovery plan. An updated recovery plan is expected to be completed in the near future.

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A6.8.3 Personal Communications

Keith Sakuma (Fishery Biologist, National Marine Fisheries Service) email to Randall Baxter (Fishery Biologist, California Department of Fish and Game) providing a print out of Osmerid catches 1980-1993 from NOAA May-June trawling in the Gulf of the Farallones. September 23, 2003.

Scott Foott (Pathologist, US Fish and Wildlife Service, Fish Pathology Lab, Anderson, California) phone call to Randall Baxter (Fishery Biologist, California Department of Fish and Game) describing histopathological examinations of larval and juvenile longfin smelt. December 2006.

Bruce Herbold (Fisheries Biologist, EPA) email to Rick Wilder about potential effects of pyrethroids on delta smelt. June 22, 2007.

Randy Baxter (Fishery Biologist, California Department of Fish and Game) phone conversation with Rick Wilder and Pete Rawlings about turbidity effects to longfin smelt. July 7, 2007.

APPENDIX A7. SACRAMENTO SPLITTAIL (*POGONICHTHYS MACROLEPIDOTUS*)

A7.1 LEGAL STATUS

The Sacramento splittail was listed as threatened under the federal Endangered Species Act (ESA) on February 8, 1999 (64 FR 5963). This ruling was challenged by two lawsuits (San Luis & Delta-Mendota Water Authority v. Anne Badgley et al. and State Water Contractors et al. v. Michael Spear et al.). On June 23, 2000, the Federal Eastern District Court of California found the ruling to be unlawful and on September 22 of that same year remanded the determination back to the U.S Fish and Wildlife Serviuce (USFWS) for re-evaluation of their original listing decision. Upon further evaluation, splittail was removed from the ESA on September 22, 2003 (68 FR 55139). On August 13, 2009, the Center for Biological Diversity (2009) challenged the 2003 decision to remove splittail from the ESA. However, on October 7, 2010, the USFWS found that listing of splittail was not warranted (75 FR 62070).

The splittail is designated as a species of special concern by the California Department of Fish and Game (DFG).

A7.2 SPECIES DISTRIBUTION AND STATUS

A7.2.1 Range and Status

The Sacramento splittail is endemic to the San Francisco Estuary and watershed. Splittail regularly inhabit the Sacramento River upstream to the Red Bluff Diversion Dam at river mile (rm) 243 and the San Joaquin River upstream to the mouth of Mud Slough at rm 125 (plus an additional 10.5 miles into Mud Slough) (see Figure A-7a). Splittail also inhabit the Napa and Petaluma River drainages (upper documented range: rm 18 and 17, respectively) and marshes. Splittail inhabiting these drainages have been found to be genetically distinct from splittail inhabiting the Sacramento and San Joaquin rivers (Baerwald et al. 2007). Splittail from the Petaluma River exhibited a higher degree of differentiation from the Sacramento-San Joaquin population than did Napa River splittail, suggesting high salinities in San Pablo Bay and Carquinez Strait isolated these populations to differing degrees from the larger Sacramento-San Joaquin population. Spawning occurs in the Petaluma and Napa rivers, but spawning locations within these rivers remain unknown (Moyle et al. 2004, Feyrer et al. 2005). No populations of splittail exist outside of the Central Valley rivers and the Bay-Delta estuary. Splittail range and selected observations in the lower portions of Sacramento River and tributaries include: the American River to rm 12, in the Feather River to rm 58 and from just below the Thermalito Afterbay outlet (B. Oppenheim pers. comm., A. Seesholtz pers. comm.), and in Butte Creek/ Sutter Bypass to vicinity of Colusa State Park.

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Figure A-7a. Sacramento Splittail Inland Range in California

Long-term beach seine sampling data for age 0 splittail (less than or equal to 50 mm fork length) in the Sacramento River spanning 32 years (1976-2008) indicates that the farthest location upstream where juvenile splittail have been collected was 144 to 184 miles upstream of the confluence of the Sacramento and San Joaquin rivers (USFWS, unpublished data). The consistency in the upstream range of juvenile splittail found in these long-term studies supports a finding that there has been no decrease in distribution during this period (Feyrer et al. 2005).

Splittail range in other rivers includes:

- San Joaquin River – into Salt Slough (rm 135; Moyle 2002) and Mud Slough at the highway 140 bridge (R. Tibstra, pers. comm.);
- Cosumnes River – just above the confluence with the Mokelumne River (Crain et al. 2004);
- Mokelumne River – observed above Woodbridge Diversion Dam to rm 60;
- Stanislaus River – no confirmed sightings, but, based on observations from other tributaries, splittail probably inhabit low gradient portions of the lower river;
- Tuolumne River – rm 17 (Legion Park, Modesto, T. Ford pers. comm.) and several annually at rm 5 during 1999-2002 (T. Heyne, pers. comm.); and
- Merced River – rm 13 several annually 1999-2001 (1 mile upstream of Hagaman Park, M. Horvath pers. comm.; T. Heyne pers. comm.).

Near Mud and Salt Sloughs, splittail can access historical valley floodplains and apparently use them for spawning in wet years (e.g., 1995 and 1998; Baxter 1999, Moyle et al. 2004). Splittail occasionally extend their range farther southward into central and southern San Francisco Bays using freshwater and low salinity habitats created during high outflow years (DFG unpubl. data; Moyle et al. 2004). After high outflow years in the early 1980s and mid-1990s, splittail were captured in the estuary of Coyote Creek, South San Francisco Bay (M. Stevenson, pers. comm.). There is no recent information on the status or persistence of these south Bay populations.

The abundance of juvenile splittail (young-of-the-year) is highly variable from one year to the next and positively correlated with hydrologic conditions within the rivers and Delta during the late-winter and spring spawning period and the magnitude and duration of floodplain inundation (Sommer et al. 1997). Because splittail are a long-lived species (5 to 7 years; Moyle 2002), the abundance of juveniles in a given year may not be a good predictor of adult splittail abundance. Results of DFG fall midwater trawl surveys indicate a marked decline in overall splittail abundance and consistently low population levels since 2002 (see Figure A-7b). In addition, Bay study indices were extremely low.

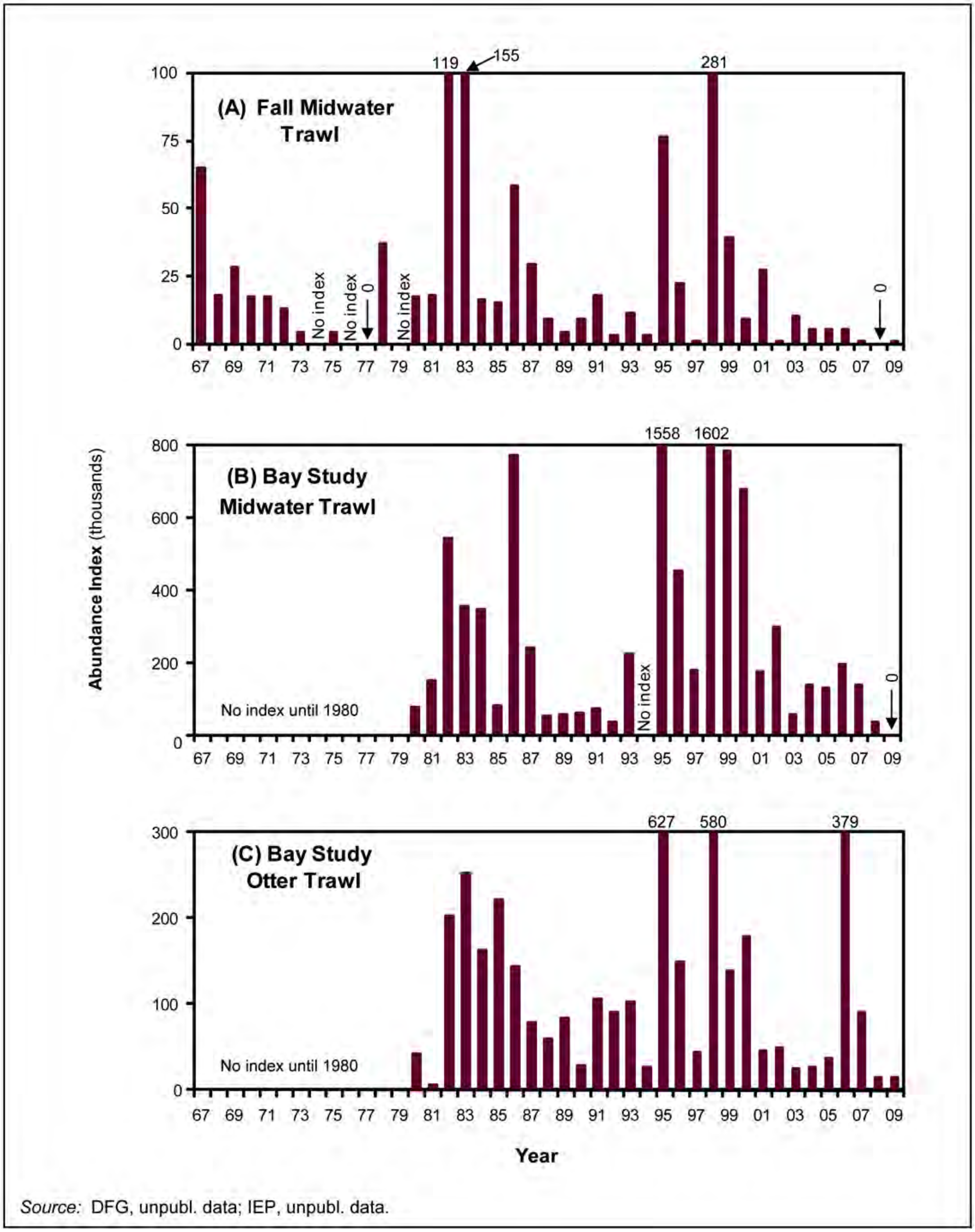


Figure A-7b. Annual Abundance Indices of Splittail from 1967-2009 in (A) Fall Midwater Trawl, (B) Bay Study Midwater Trawl, and (C) Bay Study Otter Trawl

A7.2.2 Distribution and Status in the Plan Area

Adult splittail spawn within the mainstem rivers and major tributaries to the Delta upstream of the Plan Area. Adult splittail spawn in the Plan Area on inundated floodplains of the Yolo Bypass and Cosumnes River. Collection of larvae and young juveniles indicates that inundation of terrestrial habitat within the levees of the San Joaquin River also provides suitable spawning habitat (Moyle et al. 2004). Larvae and young juveniles begin their migration downstream through the Delta with rising water temperatures during the spring; such migrations often occur in late-April, May, or even June of high flow years (Moyle et al. 2004). In low flow years, juvenile splittail are most abundant in the northern and western regions of the Delta; in high flow years, their distribution is more even throughout the Delta (Sommer et al. 1997). Most late stage juveniles and non-reproductive adults inhabit moderately shallow (less than 4 m) brackish and freshwater tidal sloughs and shoals, such as those found in Suisun Marsh and the margins of the lower Sacramento River (Moyle et al. 2004, Feyrer et al. 2005). Figure A-7c indicates the geographic distribution of splittail over the past 34 years throughout the Delta region and Figure A-7d indicates seasonal variation in the abundance of post-larval and juvenile splittail throughout their range.

No population level estimates currently exist for Sacramento splittail. However, because much of the overall distribution of splittail occurs in the Plan Area, population status and trends in the Plan Area are expected to be very similar to overall population status and trends.

A7.3 HABITAT REQUIREMENTS AND SPECIAL CONSIDERATIONS

Spawning and Early Rearing. Splittail spawning is known to take place from February to July in freshwater on inundated floodplains in the Yolo and Sutter Bypasses and along the Cosumnes River (Sommer et al. 1997, 2001, 2002, Crain et al. 2004, Moyle et al. 2004). Limited collections of ripe adults and early stage larvae indicate splittail spawn in shallow water (less than 2 m deep) over flooded vegetated habitat (cockle burr, other annual terrestrial vegetation, and perennial vegetation like willow) with a detectable water flow (Moyle et al. 2004). Turbidity is typically high under these conditions, but decreases rapidly as flows diminish. On floodplains, complex topography slows water velocities, creating eddies, and increasing hydraulic residence time. Increased hydraulic residence time promotes phytoplankton and zooplankton production on seasonally inundated floodplains. Copepods are an important first food for larval splittail (Kurth and Nobriga 2001). Floodplain inundation initiates egg development of an aquatic fly (chironomids) that, as late stage larva or pupa, is an important food of late stage larval splittail (Kurth and Nobriga 2001). Relatively warm temperatures and an abundance of food allow young splittail to grow and develop rapidly on floodplains so that they are physically prepared to leave floodplains when water levels recede. Increased water temperatures and reduced water levels may cue floodplain emigration of juvenile splittail. Many of these ecosystem benefits are dependent upon the frequency, duration, and timing of the floodplain inundation.

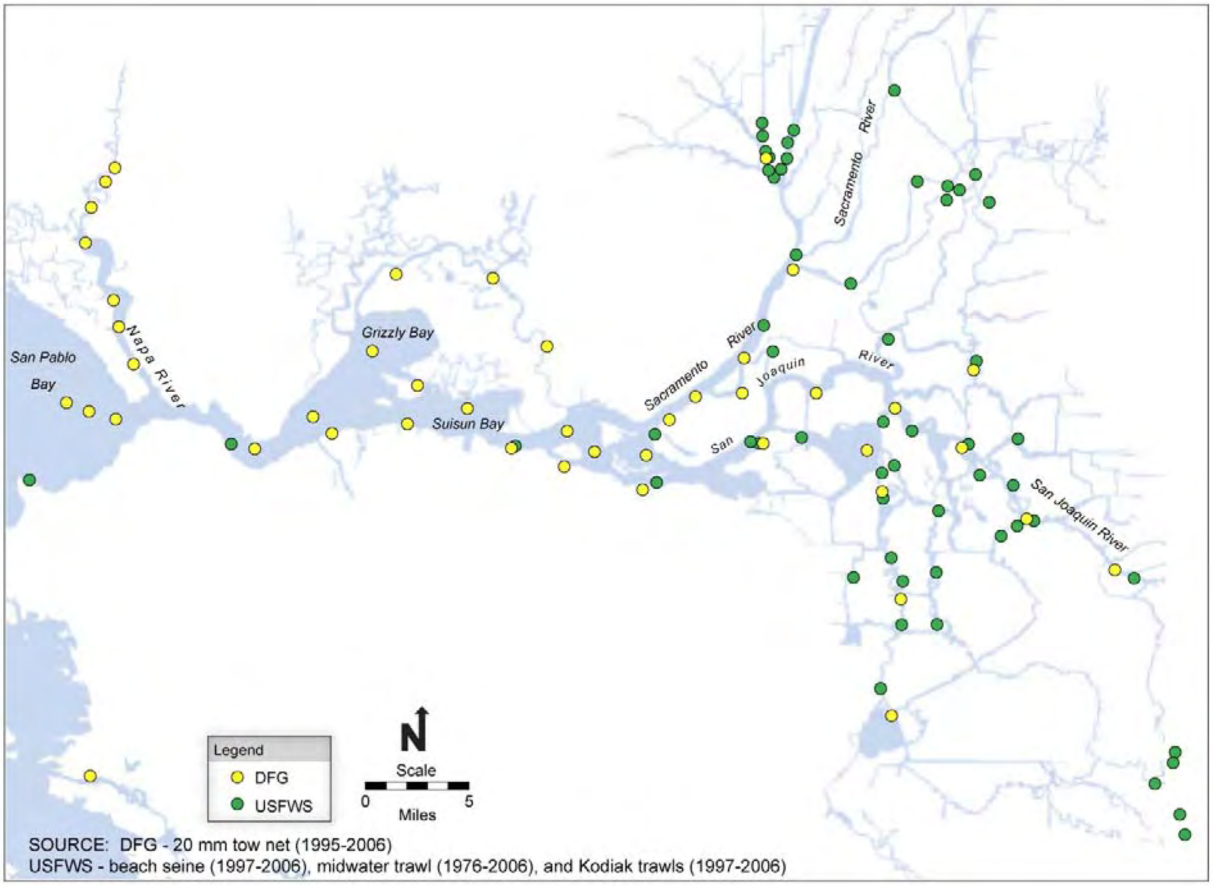
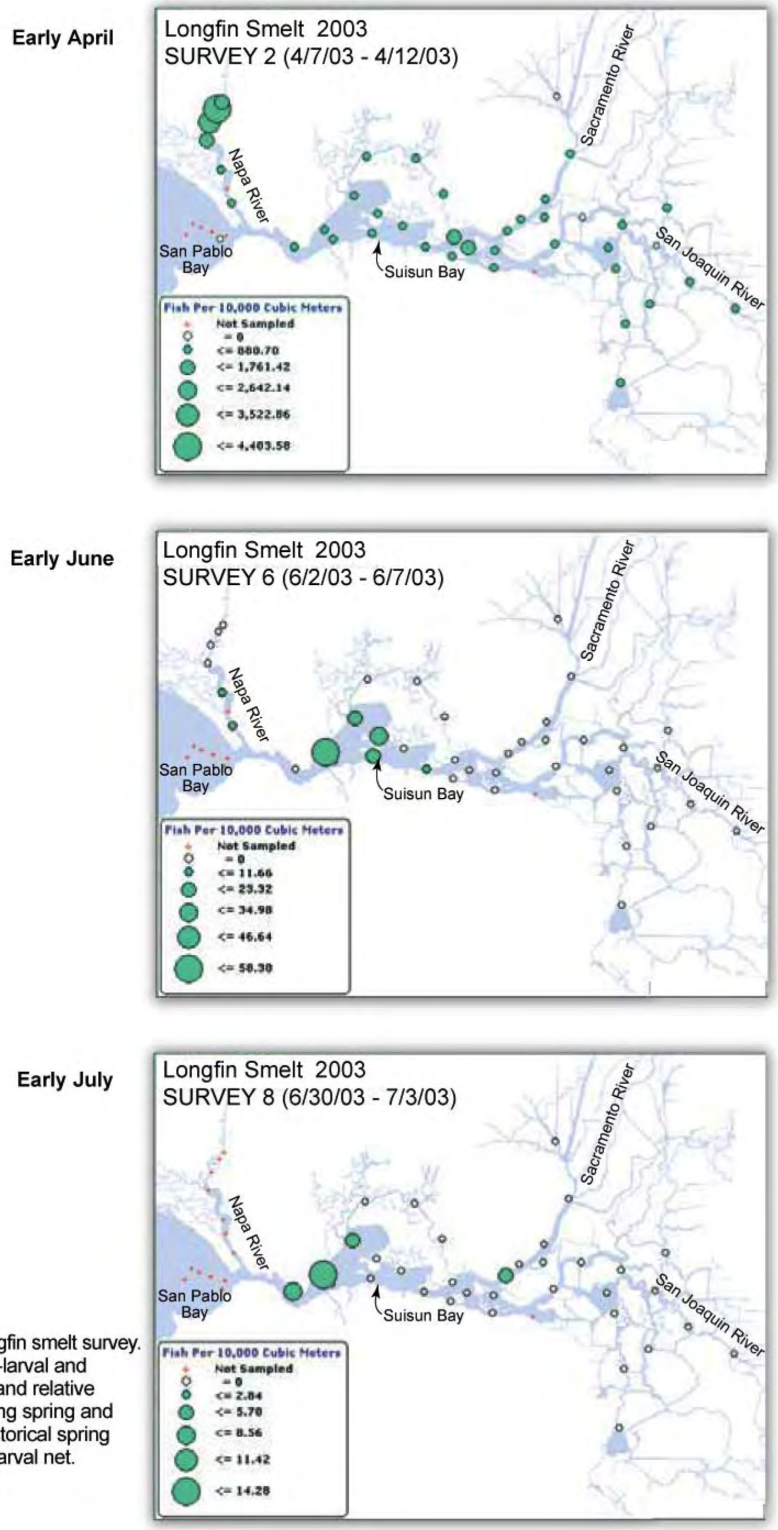


Figure A-7c. Historical Sampling Locations where Splittail have been Captured Since 1976

When floodplain inundation does not occur in the Yolo or Sutter Bypasses, adult splittail migrate farther upstream to suitable habitat along channel margins or flood terraces; spawning in such locations occurs in all water year types (Feyrer et al. 2005). Although evidence from DFG fishery surveys demonstrates that splittail spawn in all years, spawning success, as reflected in juvenile abundance, is typically greatest in wet years.

Juveniles and Adults. Although some larval and juvenile splittail are swept off floodplains and downstream by flood currents (Baxter et al. 1996), many splittail larvae and juveniles remain in riparian or annual vegetation along shallow edges on floodplains as long as water temperatures remain cool (Sommer et al. 2002, Moyle et al. 2004). Juvenile and subadult splittail commonly inhabit regions of the estuary characterized by salinities of 10 to 18 parts per thousand (ppt) (Meng and Moyle 1995; Sommer et al. 1997). Salinity tolerance increases with size (and age) such that adult splittail can survive salinities up to 29 ppt for brief periods of time (Young and Cech 1996). Splittail inhabit a broad range of temperatures, 5 to 24°C (41 to 75.2 °F) depending upon season, and acclimated fish can tolerate 29 to 33°C (84.2 to 91.4 °F) for short periods (Young and Cech 1996).



Source: CDFG 2003, 20 mm Longfin smelt survey. This annual survey monitors post-larval and juvenile longfin smelt distribution and relative abundance every two weeks during spring and early summer throughout their historical spring range using a 1600 µm egg and larval net. For more information see http://www.delta.dfg.ca.gov/data/20mm/description.asp

Figure A-7d. Example of Distribution of Post-Larval and Juvenile Longfin Smelt in Spring-Summer of a Representative Above Normal Water Year

Complementing their temperature and salinity tolerances, splittail of all sizes can tolerate low dissolved oxygen levels (less than 1 mg O_2 L^{-1}, Moyle et al. 2004), making them well suited to slow moving sections of sloughs and rivers. In Suisun Marsh during summer, splittail commonly inhabit areas with salinities of 6 to 10 ppt and temperatures of 15 to 23 °C (59 to 73.4 °F; Meng and Moyle 1995). Juveniles are most abundant in shallow (less than 2 m), turbid water with a current, and are often m) with incoming tide to feed. Napa and Petaluma River stocks may possess a higher salinity tolerance than the Central Valley stock (Baerwald et al. 2007).

Two early life history strategies occur in juvenile splittail produced in the Sacramento River system: the dominant strategy is characterized by juveniles migrating downstream in late spring and early summer to the Delta, Suisun Bay, and Suisun Marsh; a less well studied strategy is to remain upstream through the summer into the next fall or spring and migrate downstream as a subadult (Baxter 1999, Moyle et al. 2004). This latter strategy occurs in Butte Creek and the mainstem Sacramento River. As water recedes further, juveniles remaining in upstream riverine habitats and congregate in large eddies for feeding (R. Baxter, unpubl.data).

Channel margin and backwater habitats can be critical to the survival of young-of-year (YOY) splittail, as well as the population as a whole (Moyle et al. 2004, Feyrer et al. 2005). Such habitats provide refugia from predatory fishes and feeding sites as fish grow in upstream regions before and during downstream migration. Many backwater habitats are associated with the complex topography of remnant riparian habitats and are ephemerally created in response to increases in river stage (water surface elevation); others are synthetic creations such as cut channels, boat ramps, or agricultural pump intakes. This contrasts with major floodplain inundation typically associated with large splittail year classes (Meng and Moyle 1995, Baxter et al. 1996, Sommer et al. 1997), which may require an 8 to 10 m increase in river stage (typically associated with flood flow events). In the Sacramento River, levees constrain river meander from rm 194 at Chico Landing downstream to Collinsville (rm 0) and restrict the riparian zone accessible via the river channel.

Levee configuration differs through three reaches downstream of Chico Landing and has important implications in terms of splittail spawning and rearing habitat: (1) the river reach from Chico Landing to Colusa (rm 144) is characterized by setback levees enclosing remnant floodplain (flood terraces) and a narrowly meandering river channel; (2) the reach from Colusa to Verona (rm 80) is tightly leveed and contains fewer and much narrower flood terraces, many of which are actively eroding and targeted for rip-rap; and (3) the reach from Verona to Collinsville (rm 0) is also tightly leveed and contains extensive, narrow flood terraces between Verona and Sacramento, but is almost completely rip-rapped from Sacramento to Collinsville (Feyrer et al. 2005).

Maintaining and increasing this seasonally inundated floodplain habitat suitable for splittail spawning and juvenile rearing throughout the species range has been identified as a factor that will help maintain successful reproduction and increase juvenile abundance and genetic diversity during prolonged drought events and avoid a genetic “bottleneck.”

A7.4 LIFE HISTORY

Phenology. Adult splittail begin a gradual upstream migration towards spawning areas sometime between late November and late January. The relationship between migrations and river flows is poorly understood, but it is likely that splittail have a positive behavioral response to increases in flows. Feeding in flooded riparian areas in the weeks just prior to spawning may be important for later success of spawning and for post-spawning survival. Not all splittail make significant movements prior to spawning, as indicated by evidence of spawning in Suisun Marsh (Meng and Matern 2001) and the Petaluma River. The upstream movement of splittail is closely linked with flow events during February-April that inundate floodplains and riparian areas (Garman and Baxter 1999, Harrell and Sommer 2003). Seasonal inundation of shallow floodplains provides both spawning and foraging habitat for splittail (Caywood 1974, Daniels and Moyle 1983, Baxter et al. 1996, Sommer et al. 1997). Evidence of splittail spawning on floodplains has been found on both the San Joaquin and Sacramento rivers. In the San Joaquin River drainage, spawning has apparently taken place in wet years in the region where the San Joaquin River is joined by the Tuolumne and Merced rivers (T. Ford, pers. comm.). Spawning has also been documented on flooded areas along the lower Cosumnes River (Crain et al. 2004). Spawning may take place elsewhere in the Delta (e.g., on mid-channel islands) but it has not been documented. In the Sacramento River drainage, the most important spawning areas appear to be the Yolo and Sutter Bypasses, which are extensively flooded during wet years (Sommer et al. 1997, 2001). However, some spawning takes place every year along the river edges and backwaters created by small increases in flow.

In the eastern Delta, the floodplain along the lower Cosumnes River appears to be important as spawning habitat. Ripe splittail have been observed in areas flooded by levee breaches, in association with cool temperatures (less than 15 °C [59 °F]), turbid water, and flooded terrestrial vegetation (P. Moyle, unpubl. data).

Life Cycle. Splittail spawning occurs between late February and early July (Wang 1986). Fecundity is highly variable; females lay between 5,000 to 100,000 eggs. Egg incubation lasts for 3 to 7 days depending on water temperature (Moyle 2002). Newly hatched larvae are typically 6.5 to 8 mm (fork length) (Wang 1986). Larvae remain in shallow weedy areas near spawning areas for 10 to 14 days (Meng and Moyle 1995). When juveniles reach a length of approximately 29 mm (fork length), they move into deeper habitats (Sommer et al. 2002). Splittail grow to a typical length of 110 mm (standard length) during their first year, 170 mm during their second year, 250 mm during their third year, and 35 mm/year during remaining years (Moyle 2002). Maturity is typically reached at the end of their second year (Daniels and Moyle 1983).

Diet. The diet of splittail larvae up to 15 mm in length is dominated by zooplankton, primarily cladocerans with some copepods, chironomids, and rotifers present in small amounts; chironomids become important after splittail reach 15 mm in length (Kurth and Nobriga 2001, Moyle 2002). For age 1+ splittail, detritus is the dominant item found in fish collected from the estuary; various macroinvertebrates, including amphipods, clams, and mysid shrimp, are the most common non-detrital items (Caywood 1974, Daniels and Moyle 1983, Feyrer et al. 2003). During upstream migration to spawning areas, adult splittail captured near inundated shorelines along the Sacramento River were found to contain oligochaetes (earth worms) as well as smaller amounts of dipterans and cladocerans (Caywood 1974).

A7.5 THREATS AND STRESSORS

The following have been identified as important threats and stressors to Sacramento splittail (not in order of priority).

Reduced juvenile/adult rearing habitat. Reclamation of Delta islands and wetlands during the 19th and early 20th centuries removed or degraded large areas of high quality juvenile/adult rearing habitat. This habitat consisted of shallow, low velocity areas throughout the Delta, and particularly in the western Delta and Suisun Marsh (Moyle et al. 2004). In the 1960s and 1970s, the U.S. Army Corps of Engineers (USACE) increased downstream water conveyance and reinforced levees by clearing and rip-rapping levees along the lower Sacramento River. These actions further reduced or eliminated suitable rearing habitat for splittail from the city of Sacramento downstream by removing large areas of shallow channel margins. Current efforts are under way to improve flood protection for communities along much of the lower Sacramento River and several other valley rivers. Actions being proposed and conducted include removal of trees and riparian vegetation and armoring with riprap. USACE's current policy is for removal of all large trees and brush from levees to improve detection of weak points and potential levee failures.

Reduced spawning/larval rearing habitat. Reclamation and levee construction along the majority of Delta waterways and upstream riverine habitats has degraded or eliminated large areas of seasonally inundated floodplains that once served as spawning and larval rearing habitat for splittail. Although some spawning occurs on shallow margins of the main channels every year, floodplains are highly productive and, when inundated, are used more heavily by splittail than channel margin habitat for spawning and larval rearing.

Changes in river stage resulting from upstream diversions and reservoir storage has not been well studied, but, during low and moderate runoff years, water management may affect access of splittail to floodplains and their ability to emigrate successfully after spawning and early rearing (Moyle et al. 2004). Reservoir operations are designed to reduce peak flows during winter and spring months that historically would have resulted in seasonal inundation of floodplains.

Reduced food. There are multiple mechanisms that may cause reductions in food supplies for juvenile and adult splittail, including competition with non-native species and reductions in primary and secondary productivity (Jassby et al. 2002, Resources Agency 2007). The overbite clam is a highly efficient filter feeder that has reduced zooplankton populations in the Delta and Suisun Bay (Kimmerer and Orsi 1996). Zooplankton, particularly mysid shrimp, were the principal component of the splittail diet prior to the invasion of the estuary by the overbite clam, and the introduction of the clam has reduced the availability of mysids to splittail (Feyrer et al. 2003). However, the effect of the overbite clam on food availability to splittail is mixed because splittail also consume the clams (Feyrer et al. 2003).

Reductions in productivity within the estuary have been attributed to changes in hydrology associated with upstream reservoir operations, in-Delta water diversions, and reduced hydraulic residence time in the Delta. Upstream reservoir operations have reduced seasonal variability in Delta and river hydrology, resulting in fewer and shorter high flow events and, therefore, reduced frequency and duration of floodplain inundation (Sommer et al. 1997, 2002, Meng and Matern 2001, Feyrer et al. 2005, 2006). Floodplains are highly productive and are a source of large amounts of organic carbon (Schemel et al. 1996, Sommer et al. 2001, Schemel et al. 2004, Lehman et al. 2008).

The State Water Project (SWP) and Central Valley Project (CVP) export facilities and the over 2,200 in-Delta agricultural diversions (Herren and Kawasaki 2001) export nutrients, organic material, phytoplankton, and zooplankton from the Delta that would otherwise support the base of the food web (Jassby et al. 2002, Resources Agency 2007).

Reductions in hydraulic residence time in the central Delta have resulted, in part, from the need to maintain high water quality in the Delta for agricultural uses and SWP/CVP exports. Higher quality water from the Sacramento River is conveyed southward through the Delta via the Delta Cross Channel, creating a hydraulic barrier against salt water that may otherwise enter the Delta from the west. As a result, water movement has increased and hydraulic residence time has declined in the central Delta. Reduced hydrologic residence time is thought to reduce productivity in the Delta because nutrients and organics are transported downstream and out of the Delta before stimulating phytoplankton or zooplankton production (Jassby et al. 2002, Kimmerer 2002a,b, Resources Agency 2007). Increased hydraulic residence time allows more opportunity for bacterial activity and phytoplankton and zooplankton production.

Exposure to toxins. Although there is strong support from laboratory studies that toxics can be lethal to splittail (Teh et al. 2002, 2004a,b, 2005), there is little information about the chronic or acute toxicity of contaminants within the Delta (e.g., Greenfield et al. 2008). The longevity of splittail relative to most other covered fish species (5 to 7 years, Moyle 2002) enables their tissue to bioaccumulate toxicants to higher concentrations than these other species. This makes splittail particularly vulnerable to heavy metals, such as mercury, and other fat-soluble chemicals. Perhaps the greatest concern among the impacts of contaminants on splittail relates to selenium. Tissues of splittail collected in Suisun Bay had sufficiently high selenium concentrations to potentially cause physiological impacts, in particular reproductive abnormalities (Stewart et al. 2004). Adult splittail feed on the overbite clam, which bioaccumulates and transfers selenium in high concentrations (Luoma and Presser 2000). With the decline of the mysid shrimp, *Neomysis,* in the estuary, juvenile and adult splittail have increased foraging on benthic macroinvertebrates such as clams (Feyrer et al. 2003). Teh et al. (2004b) found that young splittail that were fed a diet high in selenium grew significantly slower and had higher liver and muscle selenium concentrations after nine months of testing.

Kuivila and Moon (2004) documented dissolved pesticides in the Sacramento-San Joaquin Delta during April-June (1998-2000) when young, growing splittail were migrating into the Delta and estuary. The use of pyrethroid pesticides has increased substantially in the Central Valley since the early 1990s (Oros and Werner 2005). Though relatively non-toxic to mammals, these chemicals are highly toxic to aquatic organisms, including fishes. Also, pesticide use on row crops (including rice) commonly grown in the Yolo and Sutter Bypasses and their proclivity to adhere to sediment particles suspended in water and deposited on the bottom provide a dietary pathway to splittail ingestion along with detritus during feeding (see Diet section above) (Werner 2007). Exposure to pesticides and other chemical contaminants may occur while splittail forage on inundated floodplains or in the estuary after the pesticides have entered Delta channels through agricultural drainage and have been transported to and settled in the Delta.

Non-native species. Splittail have persisted in the estuary through numerous invasions of non-native fish and invertebrates. Some, such as the invasion of the mysid shrimp, *Acanthomysis*, may have been beneficial to splittail, as the native mysid, *Neomysis*, was already on a steep decline in abundance. Both mysid species are eaten by splittail. The invasive overbite clam also became a food item, but with potential detrimental effects, such as bioaccumulation of selenium and reduction in overall phytoplankton and zooplankton abundance (see above).

Major non-native predatory fish introduced into the Bay-Delta estuary, such as striped bass and largemouth bass, have resided in the Delta for over a century (Dill and Cordone 1997), and splittail have persisted. However, reduced turbidity in the Delta combined with increased habitat for non-native predatory species provided by Brazilian waterweed and water hyacinth has enhanced both largemouth bass abundance and their ability to visually forage, thus increasing predation risk to splittail (Toft et al. 2003, Brown and Michniuk 2006).

A major concern is the potential invasion of the Delta by the highly predatory northern pike. The pike, recently present in Lake Davis on the Feather River, is currently the target of a major eradication effort (DFG 2007a). If eradication fails and pike escape downstream to the Delta, they would likely to become abundant in many of the same habitats as splittail (Moyle 2002).

Entrainment. Splittail are salvaged year-round in the SWP and CVP fish salvage facilities, with the greatest occurrence during May-July. The majority of splittail observed in fish salvage monitoring are early juveniles. Although juvenile splittail are collected in SWP and CVP fish salvage, there is no evidence that juvenile entrainment mortality has a significant population level effect on splittail (Sommer et al. 1997). Splittail salvage rates at the SWP and CVP facilities are high when splittail populations are at high levels. Young-of-the-year splittail have critical swimming velocities that are similar to water velocities occurring at the SWP and CVP diversions and are entrained at these facilities (Young and Cech 1996). Because salvage rates are high when splittail abundance is high, the effect of entrainment at the SWP and CVP export facilities on the overall population of splittail is not expected to be great. However, prolonged drought and subsequent reduction in adult splittail abundance could eventually cause a proportionally large effect on the population, particularly if the geographic distribution of the splittail population were to occur near the export facilities (Sommer et al. 1997).

Increases in export rates during the winter and total water exports from the south Delta have been associated with increased salvage of a wide variety of upper estuary fishes since 2000 (Herbold et al. 2005). The majority of splittail salvage at the SWP and CVP export facilities is composed of age 0 fish in May-July during years with high outflows that persist into the March-April splittail spawning period (Sommer et al. 1997). For example, splittail salvage increased substantially in both 2005 and 2006, corresponding to high levels of juvenile production within the system, reaching a record high of over 5 million fish at the CVP Tracy Fish Collection Facility (Gartz 2007).

There are no studies that quantitatively examine splittail mortality during the SWP or CVP fish salvage process, but it is thought to be high. Mortality to young splittail may occur as a result of overcrowding within transport tanks and predation at release locations within the Delta. Furthermore, adults that are salvaged are returned to an area downstream of the export facilities, which is expected to increase the energy expenditure needed to reach their upstream spawning sites and could reduce their ability to spawn successfully (Moyle et al. 2004).

In addition to SWP and CVP export facilities, there are over 2200 small water diversions within the Plan Area, the majority of which are unscreened (Herren and Kawasaki 2001). Results of surveys at unscreened diversions (Nobriga et al. 2004) have shown that a variety of fish species (e.g., threadfin shad, silversides, striped bass, etc.), primarily larval and juvenile lifestages, are vulnerable to entrainment. Based on results of this and similar studies conducted on unscreened diversions, it has been hypothesized that early juvenile splittail would be vulnerable to entrainment from these smaller diversions. The available information, however, is insufficient to fully assess the potential magnitude of the entrainment risk, how the risk varies among areas and seasonally, and the cumulative effect of entrainment losses on the population dynamics of splittail. Water velocities at these relatively small agricultural pumps and siphons are low enough that larger fish are able to avoid entrainment. No comprehensive quantitative estimates have been developed for the level of potential entrainment mortality that may occur as a result of diversions from the rivers and Delta.

Power plants within the Plan Area have the ability to entrain large numbers of fish. However, use of cooling water is currently low with the retirement of older units. Furthermore, recent State Water Resources Control Board (SWRCB) regulations require that units at these plants be equipped with a closed cycle cooling system by 2017.

Harvest. The legal fishery for splittail is thought to be substantial, despite poor documentation (Moyle et al. 2004). Subadult and adult splittail are harvested by recreational anglers for consumption, as well as for use as bait by striped bass anglers. There is no evidence that splittail are affected at a population level by the fishery, but there is insufficient evidence to conclude this with confidence.

A7.6 RELEVANT CONSERVATION EFFORTS

The Ecosystem Restoration Program (ERP) (CALFED 2000) lists splittail as "r," contribute to recovery, and includes the following prescription to achieve the species goal:

> *Species recovery objectives will be achieved when 2 of the following 3 criteria are met in at least 4 of every 5 years for a 15 year period: 1) the fall mid-water trawl survey numbers must be 19 or greater for 7 of 15 years. 2) Suisun Marsh catch per trawl must be 3.8 or greater and the catch of young-of-year must exceed 3.1 per trawl for 3 of 15 years, and 3) Bay Study otter trawls must be 18 or greater AND catch of young-of-year must exceed 14 for 3 out of 15 years.*

The CALFED ERP has funded the Yolo Bypass Watershed Restoration Strategy. The purpose of this project is to develop a local implementation strategy for a broad landscape level of restoration and rehabilitation for the Yolo Bypass, which should have direct benefits to splittail.

The ERP has also funded a feasibility study for flood protection and ecosystem restoration at Hamilton City. The feasibility study identified constructing an 11 km (6.8 mile) setback levee with varying heights. To accomplish ecosystem restoration within the project area, the majority of the existing "J" levee would be removed to reconnect the river to the floodplain, allowing overbank flooding and increasing capacity in the Sacramento River. Native vegetation would be restored on all project lands waterside of the new setback levee. Existing orchards in the proposed restoration areas would be removed and native vegetation planted. The native vegetation would be riparian species, scrub, oak savannah, and grassland species.

Connectivity to and restoration of floodplain habitat were achieved along the Cosumnes River through breaching of levees on the Cosumnes River Preserve during the 1990s (Booth et al. 2006). The Cosumnes River Preserve is managed by a coalition of state, federal, and non-profit organizations, such as The Nature Conservancy California. The Cosumnes River floodplain is now thought to be used for spawning by splittail (Crain et al. 2004, Moyle et al. 2004).

There are several conservation activities planned to improve shallow subtidal habitat in the Delta that should provide benefit to splittail. The CALFED ERP Suisun Marsh Land Acquisition and Tidal Marsh Restoration project will restore 500 acres within the Suisun Marsh to tidal wetland. The Suisun Marsh/North San Francisco Bay Ecological Zone Biological Restoration and Monitoring project will restore, maintain, and monitor the biology of at least three major eastern San Pablo Bay and southern Suisun Bay areas within a single CALFED-defined ecological zone (Suisun Bay/North San Francisco Bay), and compare and improve these restoration efforts through an integrated monitoring program. Construction in Ponds 3, 4, and 5 in the Napa-Sonoma Marsh will restore three commercial salt ponds along the Napa River that are expected to provide habitat benefits for splittail and other aquatic species. Restoration of Pond 3 will provide tidal habitat, whereas restoration actions in Ponds 4 and 5 will reduce salinity in preparation for tidal habitat restoration. The overall goal of this project is to restore tidal influence and re-create natural/historic elevations/topography, soil conditions, and plant communities throughout the entire elevational range to restore tidal marsh habitat.

Using ERP funds**,** construction of the Sutter Mutual Water Company Tisdale positive barrier fish screen and pumping plant has been completed. This diversion is located 45 miles north of Sacramento on the Sacramento River and will eliminate entrainment losses while maintaining Sutter Mutual Water Company's diversions.

Construction is ongoing for the Reclamation District 108 Poundstone Intake Consolidation and Positive Barrier Fish Screen Project in Colusa County. This project will construct an 81 foot long positive barrier fish screen at the entrance to a new water diversion site on the Sacramento River (rm 110.5) in Colusa County. The new diversion will consolidate and allow removal of three existing unscreened diversions. Other projects (e.g., Reclamation District 1004 intake screens, RD 108 Wilkins Slough Positive Barrier Fish Screen) have been constructed on the Sacramento River to reduce entrainment of splittail and other fish.

A new integrated monitoring and outreach program to evaluate fish contamination issues has recently been funded by ERP. This project will monitor mercury levels in sport fish and biosentinel indicators for three years throughout the watershed. The monitoring will evaluate spatio-temporal variability and gather information needed for management decisions.

The Delta Regional Ecosystem Restoration Implementation Plan (DRERIP) was formed to guide the implementation of CALFED ERP elements within the Delta (DFG 2007b). The DRERIP team has created a suite of ecosystem and species conceptual models, including splittail, that document existing scientific knowledge of Delta ecosystems. The DRERIP Team has used these conceptual models to assess the suitability of actions proposed in the Ecosystem Restoration Plan for implementation. DRERIP conceptual models were used in the analysis of proposed BDCP habitat restoration actions.

The Sacramento River Conservation Area Forum, DWR, USFWS, DFG, the Department of Parks and Recreation, the Wildlife Conservation Board, nonprofit organizations such as the Nature Conservancy and the Sacramento River Partners, and many other stakeholders conduct conservation and restoration activities in the middle and upper reaches of the Sacramento River. The Sacramento River Conservation Area Forum developed guidelines for all stakeholders to follow in directing their restoration and conservation actions. These guidelines "ensure that riparian habitat management along the river addresses the dynamics of the riparian ecosystem and the reality of the local agricultural economy." Sacramento River Conservation Area Forum goals include preserving remaining riparian habitat and reestablishing a continuous riparian ecosystem along the river. Restoration activities generally fall into one of two categories:

- Actions aimed to protect and maintain existing healthy habitat and natural processes; or
- Actions aimed to restore and recover lost habitat and disrupted processes.

The Sacramento River Conservation Area Forum recommends preserving existing riparian habitat and reestablishing a continuous band of riparian vegetation along the river. Most conservation actions to date embrace this goal and were initiated to offset habitat fragmentation as a significant threat to declining fish and wildlife populations. The most flood-prone land parcels with less productive soils and more rapid bank erosion have been bought from willing sellers and restored first.

On December 10, 2009, the California Fish and Game Commission adopted DFG's proposal to establish fishing regulations on splittail in an effort to reduce the potential effects of harvest on the splittail population. Effective March 1, 2010, there is a year-round two fish daily bag and possession limit.

A7.7 RECOVERY GOALS

Although splittail is not listed, it is included in the USFWS 1996 Sacramento-San Joaquin Delta Native Fishes Recovery Plan, which also includes the delta smelt, longfin smelt, green sturgeon, Sacramento perch, and three races of Chinook salmon (USFWS 1996). The USFWS has the responsibility to review and update the recovery plan for these species. To accomplish this task, the Service has formed a new Delta Native Fishes Recovery Team to assist in the preparation of this updated plan. An updated recovery plan from USFWS is expected to be completed soon.

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APPENDIX A8. WHITE STURGEON (*ACIPENSER TRANSMONTANUS*)

A8.1 LEGAL STATUS

The white sturgeon is not listed under the Federal or California Endangered Species Acts (ESA).

A8.2 SPECIES DISTRIBUTION AND STATUS

A8.2.1 Range and Status

White sturgeon inhabit three major drainages on the west coast of North America including the California Central Valley, Columbia (Washington), and Fraser River (British Columbia) systems. In California, white sturgeon are most abundant in the Sacramento River and San Francisco Bay-Delta estuary (Figure A-8a) (Moyle 2002). White sturgeon have been reported from the San Joaquin River system, particularly in wet years (DFG 2002, Beamesderfer et al. 2004).

Historical spawning range of white sturgeon extended upstream of Shasta Dam before its construction in the 1940s (Figure A-8a). It is thought that white sturgeon also spawned farther upstream on the San Joaquin River before major water diversions existed (Moyle 2002).

In the Central Valley, white sturgeon populations have declined from an estimated 144,000 adults in 1994 to 10,000 adults in 2005 (Bland 2006). The number of adults fluctuates annually and appears to be the result of highly variable juvenile production; the population is dominated by a few strong year classes associated with high spring outflows (Moyle 2002).

A8.2.2 Distribution and Status in the Plan Area

The Delta and Suisun Bay serve as a migratory corridor, feeding area, and juvenile rearing area for white sturgeon. White sturgeon move from coastal marine waters into the Delta and lower Sacramento River during the late fall and winter. Larval and juvenile white sturgeon inhabit the lower reaches of the Sacramento and San Joaquin rivers and the Delta (Stevens and Miller 1970). Adult white sturgeon have also been documented in the Yolo Bypass (Webber et al. 2007; M. Marshall, pers. comm.).

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Figure A-8a. White Sturgeon Inland Range in California

A8.3 HABITAT REQUIREMENTS AND SPECIAL CONSIDERATIONS

As anadromous fish, sturgeon inhabit riverine, estuarine, and marine habitats at various life stages during their long life. White sturgeon spawn preferably in the Sacramento River in the reach between the Red Bluff Diversion Dam (RBDD) and Jelly's Ferry Bridge (River Mile [rm] 267) in areas characterized by swift currents and deep pools with gravel (USFWS 1995, Schaffter 1997, DFG 2002, Moyle 2002). Habitats for migration of white sturgeon are downstream of spawning areas and include the mainstem Sacramento River, Delta, and San Francisco Bay estuary. These corridors allow the upstream passage of adults and the downstream emigration of juveniles. It has been hypothesized that migratory habitat conditions are affected by a variety of factors that may include the presence of barriers and impediments to passage (e.g., dams, gates such as the RBDD gates). Rearing habitat condition and function may be affected by annual and seasonal variation in flow and water temperatures.

A8.4 LIFE HISTORY

White sturgeon spend most of their lives in brackish portions of the estuary, although a small number of individuals move extensively in the ocean (Moyle 2002, Surface Water Resources, Inc. 2004, Welch et al. 2006). Individuals tend to concentrate in deeper areas of the estuary with soft mud and sand substrate (Pacific States Marine Fisheries Council 1996, Moyle 2002). Individuals can live over 100 years and can grow to over 19.7 feet (6 m), but sturgeon greater than 27 years old and over 6.6 feet (2 m) are rare (Moyle 2002).

Male white sturgeon reach sexual maturity at 10 to 12 years old and females reach sexual maturity at 12 to 16 years old (Moyle 2002). Maturation is thought to be a function of both photoperiod and temperature (Birstein et al. 1997). White sturgeon can spawn multiple times throughout their life. Males are believed to spawn every 1 to 2 years, whereas females spawn every 2 to 4 years (Moyle 2002). Females can produce 100,000 to several million eggs (PSMFC 1996), although typical females will produce approximately 200,000 eggs (Moyle 2002). Spawning typically occurs between February and June when temperatures are 46 to 66 °F (8 to 19 °C; Moyle 2002). Maximum spawning occurs at 58 °F (14.4 °C) in the Sacramento River (Kohlhorst 1976). It is thought that adults broadcast spawn in the water column in areas with swift current. Fertilized eggs sink and attach to the gravel bottom, where they hatch. Eggs hatch after four days at 61 °F (16 °C; Beer 1981), but may take up to 2 weeks at lower water temperatures (PSMFC 1996).

Spawning success varies from year to year. Newly hatched larvae are 7.5 to 19.5 mm in length (Kohlhorst 1976) and generally remain in the gravel for 7 to 10 days before emergence into the water column (Moyle 2002). Newly emerged larvae are pelagic for approximately seven to 10 days until their yolk-sac is absorbed, at which time they begin actively feeding on amphipods and other small benthic macroinvertebrates (Wang 1986). Juvenile white sturgeon feed primarily on algae, aquatic insects, small clams, fish eggs, and crustaceans, but their diet becomes more varied with age (Wang 1986, PSMFC 1996, Moyle 2002). Since the invasion by the overbite clam in the western Delta and Suisun Bay during the late 1980s, the clam has become a major component of the diet of juvenile and adult white sturgeon.

A8.5 THREATS AND STRESSORS

The following have been identified as important threats and stressors to white sturgeon inhabiting the Bay-Delta estuary (without priority).

Harvest. As long-lived, late maturing fish, white sturgeon are particularly susceptible to threats from overfishing (Musick 1999). White sturgeon are a popular game species within the Bay-Delta estuary and Sacramento River and support commercial fisheries within estuaries in Oregon and Washington. White sturgeon are also vulnerable to illegal (poaching) harvest. Catches of white sturgeon occur during all years, with the greatest catches typically occurring in wet years. The California Fish and Game Commission has recently adopted more restrictive sport fishing regulations designed to reduce the effects of angler harvest on white sturgeon inhabiting the Bay-Delta estuary.

Due to limits imposed on the sport fishery by DFG (2007a), only white sturgeon between 46 and 66 inches may be retained by sport fisherman with a daily bag limit of one fish in possession. Current regulations, initially implemented by DFG in 2007, require that anglers carry an annual sturgeon report card that limits annual harvest of white sturgeon to three fish per year. DFG (2002) indicates high sturgeon vulnerability to the fishery in areas where sturgeon are concentrated, such as the region between the Delta and San Pablo Bay in late winter and the upper Sacramento River during the spawning migration. In addition, the trophy status of white sturgeon and the consequent incentive for retaining oversized (greater than 66 inches [167.6 cm]) fish is another impetus for active enforcement of sturgeon angling regulations (DFG 2002).

Poaching (illegal harvest) of white sturgeon is known to occur in the Sacramento River, particularly in areas where sturgeon have been stranded (e.g., Fremont Weir) (M. Marshall, pers. comm.), as well as throughout the Bay-Delta estuary. Poaching rates in the estuary, Sacramento River, San Joaquin River and Feather River are unknown.

Furthermore, the effects of legal and illegal harvest on the population dynamics and abundance of white sturgeon within the Bay-Delta estuary are largely unknown. The small population of white sturgeon inhabiting the San Joaquin River experiences heavy fishing pressure, particularly from illegal fishing (U.S. Fish and Wildlife Service [USFWS] 1995). In addition, areas just downstream of Thermalito Afterbay Outlet, Cox's Spillway, and several barriers impeding sturgeon migration on the Feather River, may be areas of high adult mortality from high fishing effort and poaching. Poaching of white sturgeon females for roe has increased over the past 10 years despite increased enforcement efforts by DFG (L. Schwall, pers. comm.). This type of poaching is particularly detrimental to the white sturgeon population because it targets the oldest and largest adults with the highest fecundity, which affects both current and future stocks.

Reduced spawning habitat. Access to historical spawning habitat has been reduced by construction of barriers to upstream migration that block or impede access to spawning and juvenile rearing habitat. Major dams include Keswick Dam on the Sacramento River and Oroville Dam on the Feather River (Lindley et al. 2004, National Marine Fisheries Service [NMFS] 2005). White sturgeon adults have been observed periodically in the Feather River (USFWS 1995, Beamesderfer et al. 2004). Habitat modeling by Mora (2006) suggests there is suitable habitat for sturgeon in the upstream reaches of the Feather River that have been blocked by Oroville Dam. This modeling also suggests that suitable conditions are present in the San Joaquin River upstream of Friant Dam, and in the tributaries such as Stanislaus, Tuolumne, and Merced rivers upstream to their respective dams.

Other potential migration barriers include structures such as the RBDD, Sacramento Deep Water Ship Channel locks, Sutter Bypass, and Delta Cross Channel Gates on the Sacramento River, and Shanghai Bench and Sunset Pumps on the Feather River (70 FR 17386). The RBDD is an important migration barrier for sturgeon on the Sacramento River (USFWS 1995). Adult sturgeon can migrate past the RBDD when gates are raised between mid-September and mid-May to allow passage of winter-run Chinook salmon. However, tagging studies by Heublein et al. (2006) found that, when the gates were closed, a substantial portion of tagged adult green sturgeon failed to use the fish ladders at the dam and were, therefore, unable to access upstream spawning habitats. The same behavioral response may be true for white sturgeon. A set of locks at the end of the Sacramento River Deep Water Ship Channel at the connection with Sacramento River "blocks the migration of all fish from the deep water ship channel back to the Sacramento River" (California Department of Water Resources [DWR] 2005).

The Fremont Weir is located at the upstream end of the Yolo Bypass, a 40 mile (64 km) long basin that functions as a flood control facility on the Sacramento River. When the Yolo Bypass is inundated by flood water, white sturgeon are attracted into the Bypass and become trapped behind Fremont Weir, which acts as a barrier and impediment to upstream migration (DWR 2005). Sturgeon that are trapped by the weir are then subject to heavy legal and illegal fishing pressure, or become stranded behind the flashboards when the flows recede (M. Marshall, pers. comm.). Sturgeon can also be attracted to small pulse flows and trapped during the descending hydrograph (Harrell and Sommer 2003). Methods to reduce stranding and increase passage have been investigated by DWR and DFG (J. Navicky, pers. comm.).

It has been hypothesized that white sturgeon use the same migratory routes as Chinook salmon. Tagging studies have been designed and initiated to track sturgeon movement and migration patterns (P. Klimley, pers. comm.). Delta Cross Channel gate closures occur during the winter and early spring months (February through May) during sturgeon migration. The seasonal closure of the Delta Cross Channel gates is required by the State Water Resources Control Board D-1641 as a measure designed to improve the survival of downstream migrating juvenile Chinook salmon. Upstream migrating adult Chinook salmon are known to use the Delta Cross Channel as a migratory pathway when the gates are open (Hallock et al. 1970). When the gates are open, Sacramento River water flows into the central Delta providing migration cues. It is likely that attraction to flows passing into the central Delta from the Sacramento River cause migration delays and straying of white sturgeon, as it does to Chinook salmon (CALFED Science Program 2001, McLaughlin and McLain 2004). Gate closures completely block juvenile and adult sturgeon migration.

Exact white sturgeon spawning locations in Feather River are unknown; however, based on angler catches, most spawning is believed to occur downstream of Thermalito Afterbay and upstream of Cox's Spillway, just downstream of Gridley Bridge. Potential physical barriers to upstream migration include the rock dam associated with Sutter Extension Water District's sunrise pumps, shallow water caused by a head cut at Shanghai Bend, and several shallow riffles between the confluence of Honcut Creek upstream to the Thermalito Afterbay Outlet (USFWS 1995). These structures are likely to present barriers or impediments during low flow periods that block and or delay upstream sturgeon migration to spawning habitat.

Exposure to toxins. Water quality in the Sacramento and San Joaquin rivers and the Delta is influenced by a variety of point and non-point source pollutants from urban, industrial, and agricultural land uses. Runoff from residential, agricultural, and industrial areas introduces pesticides, oil, grease, heavy metals, other organics, and nutrients that contaminate drainage waters and deteriorate the quality of aquatic habitats necessary for white sturgeon survival (NMFS 1996, California Regional Water Quality Control Board-Central Valley Region 1998). Organic contaminants from agricultural returns, urban and agricultural runoff from storm events, and high concentrations of trace elements, such as boron, selenium, and molybdenum, have been identified as factors that decrease sturgeon early life-stage survival, causing abnormal development and high mortality in yolk-sac fry sturgeon at concentrations of only a few parts per billion (USFWS 1995, California Regional Water Quality Control Board 2004). Principal sources of organic contamination in the Sacramento River are rice field discharges from Butte Slough, Reclamation District 108, Colusa Basin Drain, Sacramento Slough, and Jack Slough (USFWS 1995). In recent years, changes have been made in the composition of herbicides and pesticides used on agricultural crops in an effort to reduce potential toxicity to aquatic and terrestrial species. Modifications have also been made to water system operations and discharges related to agricultural wastewater discharges (e.g., agricultural drainage water system lock-up and holding prior to discharge) and municipal wastewater treatment and discharges. Concerns remain, however, regarding the toxicity to sturgeon of contaminants that adsorb to sediments, such as pyrethroids, and other chemicals including selenium and mercury.

The extent to which toxic pollution has affected the population of white sturgeon is unknown. Sturgeon are a long-lived species that feed on invertebrates, such as clams and shrimp, and are vulnerable to the effects of toxicant bioaccumulation on the health and condition of sub-adult and adult sturgeon and their reproductive success within the estuary. However, sturgeon do not readily concentrate lipid-soluble toxins such as polychlorinated biphenyls (PCBs). Greenfield et al. (2003) found that dichlorodiphenyltrichloroethane (DDT) and chlordane concentrations in white sturgeon tissues have declined since the 1980s while selenium concentrations have remained elevated. High levels of selenium can also be found in some white sturgeon prey

(Johns and Luoma 1988, White et al. 1988), including Corbula (Urquhart and Regalado 1991), as well as in sturgeon muscle, liver, and eggs (White et al. 1987, 1988, 1989, Kroll and Doroshov 1991, Urquhart and Regalado 1991). Doroshov et al. (2007) found selenium incorporation into the plasma vitellogenin and egg yolk proteins after exposing gravid females to a selenium enriched diet. The accumulation of selenium in egg yolk to a level greater than or equal to 15 μg g-1 resulted in severe deformities and mortalities of newly hatched larvae, and the amount of selenium measured in the ovaries of recently caught wild white sturgeon has approached or exceeded these levels (Doroshov et al. 2007). Early life history stages are especially sensitive to contaminant uptake, and Kruse and Scarnecchia (2002) showed moderately increased mortality rates of white sturgeon embryos to concentrations of trace metals and other contaminants found in the Kootenai River environment. The effects on the different life history stages of white sturgeon of contaminants, other than selenium, at concentrations found in the San Francisco Bay Estuary are unknown, as are any additive or synergistic effects of multiple contaminants.

Water quality in the San Joaquin River has degraded significantly since the late 1940s (California Regional Water Quality Control Board 2004). Discharges of agricultural return flows and other point and non-point discharges have resulted in increased loading of various water quality constituents to the river and subsequently the Delta. In an effort to improve water quality, habitat, and reduce stressors on fish species such as sturgeon, water quality monitoring and management programs have been implemented on the San Joaquin River and its tributaries to reduce the loading of salt, selenium, and other water quality contaminants.

Acidic water discharges from Iron Mountain Mine, located adjacent to the upper Sacramento River, have been identified as a factor affecting the survival of fish downstream of Keswick Dam and storage limitations and limited availability of dilution flows cause high levels of downstream copper, cadmium, and zinc (Environmental Protection Agency [EPA] 2007). The EPA's Agency's Iron Mountain Mine remediation has removed toxic metals in acidic mine drainage from the Spring Creek Watershed with a state-of-the-art lime neutralization plant. Contaminant loading into the Sacramento River from Iron Mountain Mine has shown measurable reductions since the early 1990s.

Since the invasion of the overbite clam and its rapid increase in abundance within Suisun Bay, the diet of white sturgeon has shifted such that the clam is now the main component of their diet (State Water Resource Control Board 1999). The overbite clam, due to its high filtration efficiency, accumulates selenium in high concentrations and loses it slowly (Luoma and Presser 2000, Linville et al. 2002, Doroshov et al. 2007). As a result, concentrations of selenium in white sturgeon have been observed at greater than threshold levels at which toxic effects have been observed in other fish species (Lemly 2002). Dietary selenium in high concentrations can adversely affect white sturgeon survival, activity, and growth (Tashjian et al. 2006).

Reduced rearing habitat. Historical reclamation of wetlands and islands has reduced and degraded suitable in- and off-channel rearing habitat for white sturgeon. Furthermore, the channelization and hardening of levees with riprap has reduced in- and off-channel intertidal and subtidal rearing habitat as well as seasonal inundation of floodplains. The resulting changes to river hydraulics, riparian cover, and geomorphology affect important ecoystem functions (Sweeney et al. 2004). Because juvenile and adult white sturgeon feed primarily on benthic organisms such as clams and shrimp, habitat related impacts of reclamation, channelization, and riprapping would be expected to contribute to ecosystem related impacts, such as changes in the availability of food source and altered predator densities. The impacts of channelization and riprapping are thought to affect larval, post-larval, juvenile, and adult stages of sturgeon, as these life stages are dependent on the freshwater and estuarine food webs within the rivers and Bay-Delta estuary.

Increased water temperature. While juvenile and adult white sturgeon are tolerant of higher temperatures, although they appear to show signs of stress at temperatures at and above 68 °F (20 °C) (Cech et al. 1984, Geist et al. 2005). Exposure to water temperatures greater than 63 °F (17.2 °C) has also been shown to increase sturgeon egg and larval mortality (Pacific States Marine Fisheries Commission 1992).

Water temperatures in the upper Sacramento River near the RBDD historically occurred within optimum ranges for sturgeon reproduction; however, temperatures downstream, especially later in the spawning season, were reported to be frequently above 63 °F (17.2 °C; USFWS 1995). Concern regarding exposure to high temperatures in the Sacramento River during the February to June period has been reduced in recent years as temperatures in the upper Sacramento River are actively managed for Sacramento River winter-run Chinook salmon. The Shasta temperature control device, which was installed at Shasta Dam in 1997, cold water pool management within Lake Shasta, and management to maintain higher reservoir storage have all contributed to improving cool water temperature conditions within the upper Sacramento River where white sturgeon spawning and juvenile rearing are thought to occur.

Water temperatures in the lower Feather River may be inadequate for sturgeon spawning and egg incubation as the result of releases of warmed water from Thermalito Afterbay (Surface Water Resources, Inc. 2003). The warmed water may be one reason that neither green nor white sturgeon are found in the river in low-flow years (DFG 2002). Exposure to elevated water temperatures within the Feather River downstream of Thermalito Afterbay is expected to be a factor affecting habitat quality and availability for sturgeon spawning and juvenile rearing on the lower Feather River (DFG 2002).

Reduced flow on the San Joaquin River resulting from dam and diversion operations and agricultural return flows contribute to seasonally elevated water temperatures in the mainstem San Joaquin River, particularly during late summer and fall. Although these effects are difficult to measure, water temperatures in the lower San Joaquin River continually exceed preferred temperatures for sturgeon migration and development during spring months. Temperatures at Stevenson on the San Joaquin River near the Merced River confluence as recorded on May 31 (spawning typically occurs February-June) between 2000 and 2004 ranged from 77 to 82 °F (25 to 27.8 °C; California Data Exchange Center 2007). Juvenile sturgeon are also exposed to increased water temperatures in the Delta during the late spring and summer, as a result in part of the loss of riparian shading, and by thermal inputs from municipal, industrial, and agricultural discharges. Seasonally elevated water temperature on the San Joaquin River and within the Delta has been identified as a factor affecting habitat quality and availability for sturgeon migration, spawning, and juvenile rearing.

Non-native species. White sturgeon have been impacted, both positively and negatively, by the introduction of non-native species into the Bay-Delta estuary. Changes in the species composition of fish and macroinvertebrates have altered trophic interactions and dynamics for juvenile and adult white sturgeon. Many of the recent introductions of invertebrates have greatly affected the benthic fauna in the Delta and Suisun Bay and non-native species such as the overbite clam and Asian clam are now a major component of the diet of white sturgeon (DFG 2002). The overbite clam and other introduced clams are benthic filter feeders that can accumulate various toxic substances , such as selenium, mercury, and other compounds, in their tissue. Sturgeon, which are long-lived species, may bioaccumulate these toxics by foraging on these clams, which may adversely impact the health and survival of sub-adult and adult sturgeon and their reproduction (Doroshov et al. 2007).

DFG (2002) reviewed many of the recent non-native invasive species introductions and the potential consequences to white sturgeon. The most notable species responsible for altering the trophic system of the Sacramento-San Joaquin Delta and Suisun Bay include overbite and Asian clams, and the Chinese mitten crabs. Sturgeon regularly consume both clam species, which is of particular concern because of the high bioaccumulation rates of these clams (Doroshov et al. 2007). Although Chinese mitten crabs may be eaten by adult white sturgeon, it is possible they prey upon sturgeon eggs. The Chinese mitten crab population within the Bay-Delta system has undergone a substantial decline since 2002 and currently occurs in very low abundance (Hieb, pers. comm. 2008) and, therefore, may currently not be a major factor affecting white sturgeon.

Introductions of non-native invasive plant species such as water hyacinth and Brazilian waterweed have altered habitat within the Delta and Suisun Bay and have affected local assemblages of fish within the Bay-Delta estuary (Nobriga et al. 2005). Nobriga et al. (2005) found significant differences in water clarity and fish communities in those areas where submerged aquatic vegetation (SAV) was abundant when compared to open water habitats where SAV was not abundant. The occurrence of dense concentrations of SAV has been hypothesized to result in a number of potential effects on aquatic habitat including raising temperatures, reducing turbidity and dissolved oxygen levels, and inhibiting access to shallow water habitat by fish intolerable to these conditions. The presence of non-native centrarchid species is strongly associated with the occurrence of Brazilian waterweed (Brown and Michniuk 2007). Brazilian waterweed forms thick “walls” along the margins of channels and shallow water habitat in the Delta. This growth may prevent juvenile sturgeon from accessing shallow water habitat along channel edges. Water hyacinth creates dense floating mats that can impede river flows and alter the aquatic environment beneath mats. By reducing water velocities near plants, these species reduce turbidity in the water column, potentially exposing sturgeon to higher predation risk.

Dissolved oxygen levels beneath the mats often drop below suitable levels for fish due to the increased amount of decaying vegetative matter produced from the overlying mat and diel respiration by aquatic plants. Like Brazilian waterweed, water hyacinth is often associated with the margins of the Delta waterways in its initial colonization, but can eventually cover the entire channel if conditions permit. High levels of infestation by non-native aquatic plants may produce barriers to white sturgeon movement within the Delta, although there is no evidence that this occurs.

Dredging. Hydraulic dredging is a common practice in the navigational channels within San Francisco, San Pablo, and Suisun bays, the Delta, and the Sacramento and San Joaquin rivers to allow commercial and recreational vessel traffic. White sturgeon are at risk of entrainment from dredging, with young-of-the-year fish at greatest risk (Boysen and Hoover 2009).. Studies by Buell (1992) reported approximately 2,000 sturgeon entrained in the removal of one million tons of sand from the bottom of the Columbia River at depths of 60 to 80 feet (18 to 24 m). In addition, dredging operations can result in the resuspension of toxics such as ammonia, hydrogen sulfide, and copper as a result of both dredging and dredge spoil disposal, and alter channel bathymetry and current patterns (NMFS 2006).

Reduced turbidity. Turbidity levels in the Delta have decreased over the past few decades (Jassby et al. 2002). This reduction may have had detrimental effects to white sturgeon. Gadomski and Parsley (2005) found that larval white sturgon predation by prickly sculpin was greater with reduced turbidity. However, larval sturgeon are found close to spawning locations generally upstream of the Delta, where turbidity is already lower than the Delta.

The relationship between turbidity and the vulnerablity of various life stages of white sturgeon to predation has not been established within the Delta. As discussed above, the dense colonization of local areas within the Delta by SAV such as the Brazilian waterweed (*Egeria densa*) has been shown to be associated with increased water clarity (e.g., resuting from trapping and settlement of suspended sediments). Increased water clarity may contribute to increased vulnerablity of sturgeon to predation. Juvenile white sturgeon are expected to be less vulnerable to predation than other estuarine fish due to their scutes and protective armoring. In addition, the large size of sub-adult and adult white sturgeon further reduces their vulnerablity to predation. As a result of these factors, the potential increase in vulnerability to predation due to localized reductions in turbidity is expected to be minor relative to other covered fish species.

Stranding. White sturgeon that are attracted to high flows when the Yolo Bypass is inundated by flood waters from the Sacramento River will move onto the floodplain and eventually concentrate behind Fremont Weir, where they are blocked from further upstream migration (DWR 2005). As Bypass flows recede, these sturgeon become stranded behind the weir (Harrell and Sommer 2003) and are then subject to both legal and illegal harvest (M. Marshall, pers. comm.). Methods to reduce stranding and increase sturgeon passage are have been previously developed (J. Navicky, pers. comm.) but have been stalled (Z. Matica, pers. comm.).

Entrainment. There is little evidence that the overall population of white sturgeon is influenced by entrainment. Adults are not likely to be entrained due to their large size and benthic habits. Larval sturgeon are more susceptible to entrainment from water diversion facilities as a result of their migratory behavior within the water column and reduced swimming perfomance capability. Herren and Kawasaki (2001) documented 431 water diversions located on the Sacramento River between Sacramento and Shasta Dam. In the Feather River, there are eight diversions greater than 10 cubic feet per second (cfs) and approximately 60 small diversions between 1 to 10 cfs between the Thermalito Afterbay outlet and the confluence with the Sacramento River (USFWS 1995). White sturgeon have been reported in low numbers in fish salvage at both the SWP and CVP export facilities. White sturgeon observed in fish salvage have predominantly been juvenile and sub-adult life stages. Occasionally, adult white sturgeon have been observed impinged on the trash racks at the CVP intake; it has been hypothesized that these large adults were in weakened conditions or had previously died from stresses associated with spawning, angler mortality, or other causes before being impinged at the export intake. Given the large number of diversions, it is possible that larval white sturgeon are vulnerable to entrainment at these diversions; however, actual entrainment mortality and potential effects on the abundance and population dynamics of white sturgeon are unknown.

A8.6 RELEVANT CONSERVATION EFFORTS

The Central Valley Project Improvement Act's Anadromous Fish Restoration Program (AFRP) have a goal of supporting efforts that lead to doubling the natural production of anadromous fish in the Central Valley at a sustainable, long-term basis, at levels not less than twice the average abundance reported during the period of 1967 to 1991. Though most efforts of the AFRP have focused on Chinook salmon as a result of their listing history and status, sturgeon may receive some unknown incidental amount of benefit from these restoration efforts. For example, the acquisition of water for flow enhancement on tributaries to the Sacramento River, spawning gravel augmentation, fish screening for the protection of Chinook salmon and Central Valley steelhead, or riparian revegetation and instream restoration projects would likely have ancillary benefits to sturgeon.

Many beneficial actions have originated and been funded by the CALFED program including such projects as floodplain and instream restoration, riparian habitat protection, fish screening and passage projects, research regarding non-native invasive species and contaminants, restoration methods, watershed stewardship, education, and outreach programs. Prior Federal Register notices have reviewed the details of Central Valley Project Improvement Act (CVPIA) and CALFED programs and potential benefits for anadromous fish, particularly Chinook salmon and Central Valley steelhead (69 FR 33102). Projects potentially benefiting sturgeon primarily consist of fish screen evaluation and construction projects, restoration evaluation and enhancement activities, contamination studies, and dissolved oxygen investigations related to the San Joaquin River Deep Water Ship Channel.

The Delta Regional Ecosystem Restoration Implementation Plan (DRERIP) was formed to guide the implementation of CALFED Ecosystem Restoration Plan elements within the Delta (DFG 2007b). The DRERIP team has created a suite of ecosystem and species conceptual models, including white sturgeon, that document existing scientific knowledge of Delta ecosystems. The DRERIP Team has used these conceptual models to assess the suitability of actions proposed in the Ecosystem Restoration Plan for implementation. DRERIP conceptual models were used in the analysis of proposed BDCP conservation measures.

New sport fishing regulations adopted over the past several years specifically to protect and reduce harvest of sturgeon and increased law enforcement are expected to further reduce illegal fishing practices, and reduce the effects of harvest of white sturgeon by recreational anglers, throughout the range of the species.

A8.7 Recovery Goals

No recovery plan has been prepared for white sturgeon because the species is not listed under the California or Federal ESA.

A8.8 References

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APPENDIX A9. NORTH AMERICAN GREEN STURGEON *(ACIPENSER MEDIROSTRIS)*

A9.1 LEGAL STATUS

The North American green sturgeon is composed of two Distinct Population Segments (DPS): the northern DPS, which includes all populations in the Eel River and northward; and the southern DPS, which includes all populations south of the Eel River. Only the southern DPS is found in the Plan Area of the Bay Delta Conservation Plan (BDCP).

After a status review was completed in 2002 (Adams et al. 2002), the National Marine Fisheries Service (NMFS) determined that the southern DPS of the North American green sturgeon did not warrant listing as threatened or endangered but should be identified as a Species of Concern. The "not warranted" determination was challenged on April 7, 2003. The National Marine Fisheries Service (NMFS) updated their status review on February 22, 2005, and determined that the southern DPS should be listed as threatened under the Federal Endangered Species Act (Biological Review Team 2005). NMFS published a final rule on April 7, 2006 that listed the southern DPS as threatened, which took effect on June 6, 2006 (71 FR 17757). Included in the listing are the spawning population in the Sacramento River and fish living in the Sacramento River, the Sacramento-San Joaquin Delta, and the San Francisco Estuary.

California Department of Fish and Game has identified green sturgeon as a Species of Special Concern (California Department of Fish and Game [DFG] 2003).

A9.2 SPECIES DISTRIBUTION AND STATUS

A9.2.1 Range and Status

Green sturgeon range from Ensenada, Mexico to the Bering Sea, Alaska (Colway and Stevenson 2007, Moyle 2002). Green sturgeon are currently known to spawn in two California basins: the Sacramento and Klamath rivers. These reproducing populations are genetically distinct and occupy the Southern and Northern DPS, respectively (Adams et al. 2002, Israel et al. 2004). Adult populations in the less-altered Klamath and Rogue rivers are fairly constant with a few hundred spawning adults typically being harvested annually by tribal fisheries. In the Sacramento River, the green sturgeon population is believed to have declined over the last two decades with less than 50 spawning green sturgeon being sighted annually in the best spawning habitat (Richard Corwin, Bureau of Reclamation, pers. comm.). In the Umpqua, Feather, Yuba, and Eel rivers green sturgeon sightings are extremely limited and spawning has not been recently recorded. In the San Joaquin and South Fork Trinity rivers, the green sturgeon population appears extirpated (see Figure A-9a).

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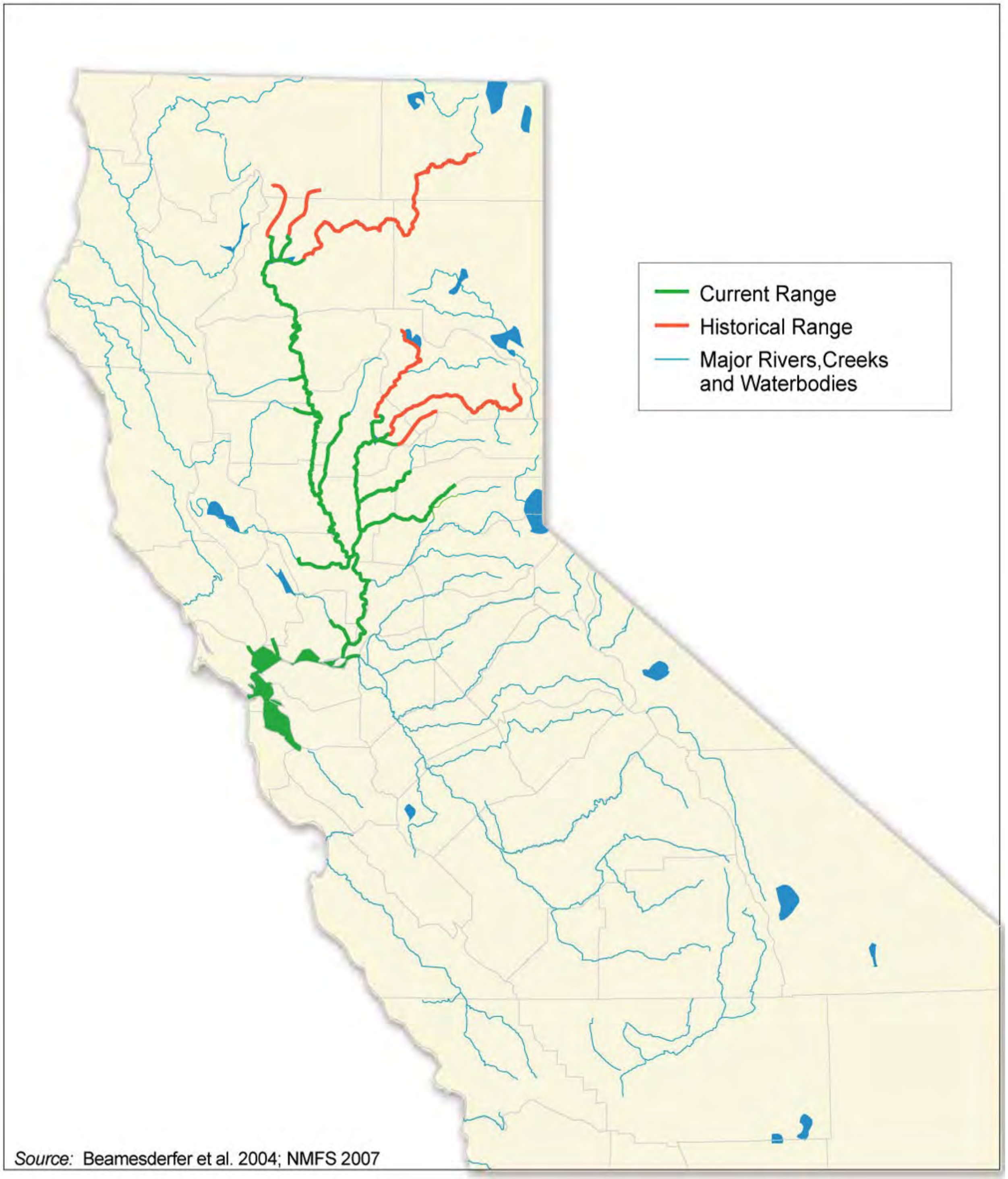

Figure A-9a. Green Sturgeon Inland Range in California

Green sturgeon have been recorded in the Feather River (Beamesderfer et al. 2004), and may spawn there during high flow years (DFG 2002), although no indication of spawning has been documented despite intensive sampling efforts (Niggemyer and Duster 2003). No juvenile green sturgeon have been documented in the San Joaquin River, although Moyle (2002) suggests that reproduction may have taken place in the San Joaquin River because adults have been captured at Santa Clara Shoal and Brannan Island. Additionally, two unidentified juvenile sturgeon have been caught on the Mokelumne River, a tributary to the San Joaquin River (J. Smith, pers. comm.).

Green sturgeon are anadromous and pass through the San Francisco Bay to the ocean where they primarily move northward and commingle with other sturgeon populations, spending much of their lives in the ocean or in Oregon and Washington estuaries (DFG 2002; Kelly et al. 2007). Subadult and adult green sturgeon are thought to potentially migrate thousands of miles along the coasts of northern California and the Pacific Northwest. Relatively large concentrations of sturgeon occur in the Columbia River estuary, Willapa Bay, and Grays Harbor, with smaller aggregations in the San Francisco estuary (Emmett et al. 1991, Moyle et al. 1992, Israel 2006).

Although NMFS indicates that absolute population abundance of green sturgeon is currently not determinable (74 FR 52,300), some information on the population abundance of the southern DPS of North American green sturgeon is available, and is described in NMFS status reviews (Adams et al. 2002, 2007, NMFS 2005). Musick et al. (2000) noted that the abundance of North American green sturgeon populations has declined by 88 percent throughout much of its range. DFG (2002) estimated that green sturgeon abundance within the Bay-Delta estuary ranged from 175 to more than 8,000 adults between 1954 and 2001 with an annual average of 1,509 adults. Fish monitoring efforts at Red Bluff Diversion Dam (RBDD) and the Glenn-Colusa Irrigation District pumping facility on the upper Sacramento River have recorded between zero and 2,068 juvenile North American green sturgeon per year (Adams et al. 2002). Catches of sub-adult and adult North American green sturgeon by the Interagency Ecological Program (IEP) between 1996 and 2004 ranged from one to 212 green sturgeon per year, with the highest catch in 2001 (Samantha Vu, pers. comm.). Because these fish were primarily captured in San Pablo Bay, where both northern and southern DPSs exist, the proportion of fish captured in IEP sampling from the southern DPS is unknown.

Green sturgeon are long-lived (up to 60 to 70 years) and late maturing (sexual maturity is reached at approximately 15 to 20 years) (Moyle 2002). They have a low fecundity rate (59,000 to 242,000 eggs per female) relative to white sturgeon (180,000 to 590,000 eggs per female) and spawn only periodically (Doroshov 1983, Moyle 2002, Van Eenennaam et al. 2006). These characteristics make them particularly susceptible to habitat degradation and overharvest (Musick 1999). With only one population in the Central Valley, the viability of the southern DPS is vulnerable to changes in the environment and catastrophic events through a lack of spatial geographic diversity. As a result of low abundance, the population has limited genetic diversity, which decreases the ability of individuals in the green sturgeon population to withstand environmental variation.

A9.2.2 Distribution and Status in the Plan Area

The Delta serves as a migratory corridor, feeding area, and juvenile rearing habitat for North American green sturgeon in the southern DPS. Adults migrate upstream primarily through the western edge of the Delta into the lower Sacramento River between March and June (Adams et al. 2002). Green sturgeon spawning is thought to occur primarily in the upper reaches of the Sacramento River, although some spawning may also occur in tributaries. Larvae and post-larvae are present in the lower Sacramento and North Delta between May and October, primarily in June and July (DFG 2002). Juvenile green sturgeon have been captured in the Delta during all months of the year (Borthwick et al. 1999, DFG 2002, BDAT 2007). Adult green sturgeon have been documented in the Yolo Bypass (M. Marshall, pers. comm.) and rear in Suisun Bay and marsh.

A9.3 HABITAT REQUIREMENTS AND SPECIAL CONSIDERATIONS

On October 9, 2009, NMFS (74 FR 52,300) designated critical habitat for the green sturgeon Southern DPS throughout most of its occupied range, including: coastal marine waters from Monterey Bay to the Washington/Canada border; coastal bays and estuaries in California, Oregon, and Washington; and fresh water rivers in the Central Valley, California. The essential physical and biological habitat features identified for the Southern DPS include prey resources (benthic invertebrates and small fish), water quality, water flow (particularly in freshwater rivers), water depth, substrate types (i.e., appropriate spawning substrates within freshwater rivers), sediment quality, and migratory corridors (see Figure A-9b). Proposed inland critical habitat in the Sacramento and San Joaquin River basins includes the Sacramento River downstream of Keswick Dam, the Feather River downstream of Oroville Dam, and the Yuba River downstream of Daguerre Dam; portions of Sutter and Yolo Bypasses; the legal Delta, excluding Five Mile Slough, Seven Mile Slough, Snodgrass Slough, Tom Paine Slough and Trapper Slough; and San Francisco, San Pablo, and Suisun bays.

As anadromous fish, North American green sturgeon rely on riverine, estuarine, and marine habitats during their long life. Freshwater habitat of green sturgeon of the southern DPS varies in function, depending on location within the Sacramento River watershed. Spawning areas currently are limited to accessible reaches of the Sacramento River upstream of Hamilton City and downstream of Keswick Dam (see Figure A-9a) (DFG 2002). Preferred spawning habitats are thought to contain large cobble in deep and cool pools with turbulent water (DFG 2002, Moyle 2002, Adams et al. 2002). Sufficient flows are needed to sufficiently oxygenate and limit disease and fungal infection of recently laid eggs (Deng et al. 2002; Parsley et al. 2002). Within the Sacramento River, spawning appears to be triggered by large increases in water flow during spawning (Brown 2007). However, within the Rogue River, Erickson et al. (2002) found that green sturgeon were most often found at depths greater than 5 m with low or no currents during summer and fall months.

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Figure A-9b. Green Sturgeon Inland Critical Habitat in California

In addition, acoustic tagging studies by Erickson et al. (2002) indicate that adult green sturgeon hold for as long as six months in deep (greater than 5 m), low gradient reaches or off-channel sloughs or coves of the river during summer months when water temperatures were between 15 and 23 °C (59 and 73.5 °F). When ambient temperatures in the river dropped in fall and early winter (less than 10 °C [50 °F]) and flows increased, fish moved downstream and into the ocean. Water temperatures in spawning and egg incubation areas are critical; temperatures greater than 19 °C (66.2 °F) are lethal to green sturgeon embryos (Cech et al. 2000, Mayfield and Cech 2004, Van Eenennaam et al. 2005, Allen et al. 2006).

Habitats for migration are downstream of spawning areas and include the mainstem Sacramento River, Delta, and San Francisco Bay estuary. These corridors allow the upstream passage of adults and the downstream emigration of juveniles (NMFS 2006a). Migratory habitat conditions are strongly affected by the presence of barriers and impediments to migration (e.g., dams), unscreened or poorly screened diversions, and degraded water quality. Heublein et al. (2009) found two different patterns of “spawning migration” and out-migration for green sturgeon in the Sacramento River. Results of this study found six individuals potentially spawned, over-summered and moved out of the river with the first fall flow event, nine individuals promptly moved out of the Sacramento River before September 1. While some green sturgeon appeared to be impeded on their upstream movement by closure of the RBDD in mid-May, at least five individuals passed under the dam gates during their downstream migration. Both spawning areas and migratory corridors comprise rearing habitat for juvenile green sturgeon, which feed and grow up to three years in freshwater. Stomach contents from adult and juvenile green sturgeon captured in the Delta point to the importance of habitat that supports shrimp, mollusks, amphipods, and small fish (Radtke 1966, Houston 1988, Moyle et al. 1992). Rearing habitat condition and function may be affected by variation in annual and seasonal flow and water temperatures (NMFS 2006a).

Nearshore marine habitats must provide adequate food resources, suitable water quality conditions, and natural cover for juvenile green sturgeon to successfully forage and grow to adulthood. Offshore marine habitats are also important for supporting growth and maturation of sub-adult green sturgeon.

A9.4 LIFE HISTORY

There is relatively little known about the North American green sturgeon, particularly for those that spawn in the Sacramento River (the Nature Conservancy et al. 2008). Adult North American green sturgeon are believed to spawn every 3 to 5 years, but can spawn as frequently as every 2 years (Lindley and Moser pers. comm., as cited in NMFS 2005) and reach sexual maturity at an age of 15 to 20 years. Adult green sturgeon begin their upstream spawning migrations into the San Francisco Bay in March, reach Knights Landing during April, and spawn between March and July (Heublein et al. 2006). Based on the distribution of sturgeon eggs, larvae, and juveniles in the Sacramento River, DFG (2002) concluded that green sturgeon spawn in late spring and early summer upstream of Hamilton City, and possibly to Keswick Dam.

Peak spawning is believed to occur between April and June (Table A-9a). Adult female green sturgeon produce between 59,000-242,000 eggs, depending on body size, with a mean egg diameter of 4.3 mm (Moyle et al. 1992, Van Eenennaam et al. 2006).

Table A-9a. Temporal Occurrence of (a) Adult, (b) Larval and Post-larval, (c) Juvenile, and (d) Coastal Migrants of the Southern DPS of North American Green Sturgeon. Locations are specific to the Central Valley of California. Darker shades indicate months of greatest relative abundance.

(a) Adult (≥13 years old for females and ≥9 years old for males)

Location	*Jan*	*Feb*	*Mar*	*Apr*	*May*	*Jun*	*Jul*	*Aug*	*Sep*	*Oct*	*Nov*	*Dec*
[1,2,3]Upper Sac River												
[4,8]SF Bay Estuary												

(b) Larval and post-larval (≤10 months old)

Location	*Jan*	*Feb*	*Mar*	*Apr*	*May*	*Jun*	*Jul*	*Aug*	*Sep*	*Oct*	*Nov*	*Dec*
[5]RBDD, Sac River												
[5]GCID, Sac River												

(c) Juvenile (> 10 months old and ≤3 years old)

Location	*Jan*	*Feb*	*Mar*	*Apr*	*May*	*Jun*	*Jul*	*Aug*	*Sep*	*Oct*	*Nov*	*Dec*
[6]South Delta*												
[6]Sac-SJ Delta												
[5]Suisun Bay												

(d) Coastal migrant (3 to 13 years old for females and 3 to 9 years old for males)

Location	*Jan*	*Feb*	*Mar*	*Apr*	*May*	*Jun*	*Jul*	*Aug*	*Sep*	*Oct*	*Nov*	*Dec*
[3,7]Pacific Coast												
Relative Abundance:	= High				= Medium				= Low			

* Fish Facility salvage operations
RBDD – Red Bluff Diversion Dam
GCID – Glenn-Colusa Irrigation District facility
Sources: [1]USFWS 2002, [2]Moyle et al. 1992, [3]Adams et al. 2002 and NMFS 2005, [4]Kelly et al. 2007, [5]DFG 2002, [6]BDAT, fall midwater trawl green sturgeon captures from 1969 to 2003, [7]Nakamoto et al. 1995, [8]Heublein et al. 2006

Newly hatched green sturgeon are approximately 12.5 to 14.5 mm in length. Green sturgeon are strongly oriented to the river bottom and exhibit nocturnal activity patterns (Cech et al. 2000). After six days, the larvae exhibit nocturnal swim-up activity (Deng et al. 2002) and nocturnal downstream migrational movements (Kynard et al. 2005). Juvenile green sturgeon continue to exhibit nocturnal behavior beyond the metamorphosis from larval to juvenile stages. After approximately 10 days, larvae begin feeding and growing rapidly, and young green sturgeon appear to rear for the first one to two months in the Sacramento River between Keswick Dam and Hamilton City (DFG 2002). Length measurements estimate juveniles to be two weeks old (24 to 34 mm fork length) when they are captured at RBDD (DFG 2002, USFWS 2002), and three weeks old when captured further downstream at the Glenn-Colusa facility (DFG, unpubl. data, Van Eenennaam et al. 2001). Growth is rapid as juveniles reach up to 30 cm the first year and over 60 cm in the first 2 to 3 years (Nakamoto et al. 1995).

Juveniles appear to spend 1 to 4 years in freshwater and estuarine habitats before they enter the ocean (Nakamoto et al. 1995). According to Heublein (2006), all adults leave the Sacramento River prior to September. Lindley (2006) found frequent large-scale migrations of green sturgeon along the Pacific coast. Kelly et al. (2007) reported that green sturgeon enter the San Francisco estuary during the spring and remain until fall. Juvenile and adult green sturgeon enter coastal marine waters after making significant long-distance migrations with distinct directionality thought to be related to resource availability.

Little is known about juvenile and adult green sturgeon feeding and diet within the ocean. On entering the highly productive ocean environment, green sturgeon grow at a rate of approximately 7 cm per year until they reach maturity. Male green sturgeon mature at an earlier age and are smaller than females (Van Eenennaam et al. 2006). Green sturgeon spend 3-13 years in the ocean before returning to freshwater to spawn.

A9.5 THREATS AND STRESSORS

The following have been identified as important threats and stressors to the southern DPS of green sturgeon (without priority).

Reduced 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. Major dams include Keswick Dam on the Sacramento River and Oroville Dam on the Feather River (Lindley et al. 2004, NMFS 2005). The Feather River is likely to have supported significant spawning habitat for the green sturgeon population in the Central Valley before dam construction (see Figure A-9a) (DFG 2002). Green sturgeon adults have been observed periodically in the lower Feather River (USFWS 1995, Beamesderfer et al. 2004). Results of habitat modeling by Mora (2006) suggest there is potential habitat on the Feather River upstream of Oroville Dam that would have been suitable for sturgeon spawning and rearing prior to construction of the dam. This modeling also suggests sufficient conditions are present in the San Joaquin River to Friant Dam, and in the tributaries such as Stanislaus, Tuolumne, and Merced rivers upstream to their respective dams, although it is unknown whether green sturgeon ever inhabited the San Joaquin River or its tributaries (Beamesderfer et al. 2004).

NMFS (2006a) reports several potential migration barriers, including structures such as the RBDD, Sacramento Deep Water Ship Channel locks, Sutter Bypass, and Delta Cross Channel gates on the Sacramento River, and Shanghai Bench and Sunset Pumps on the Feather River (). In the Central Valley, approximately 4.6 percent of the total river kilometers have spawning habitat characteristics similar to where Northern DPS green sturgeon spawn, with only 12 percent of this habitat is currently occupied by sturgeon (Neuman et al. 2007). Of the 88 percent that is unoccupied (approx. 4000 km), 44.2 percent is currently inaccessible due to dams Neuman et al. (2007).

The RBDD has been identified as a major barrier and impediment to sturgeon migration on the Sacramento River (USFWS 1995). Adult sturgeon can migrate past RBDD when gates are raised between mid-September and mid-May to allow passage for winter-run Chinook salmon. However, tagging studies by Heublein (2006) found that, when the gates were closed, a substantial portion of tagged adult green sturgeon failed to use fish ladders at the dam and were, therefore, unable to access upstream spawning habitats. A set of locks at the end of the Sacramento River Deep Water Ship Channel at the connection with the Sacramento River "blocks the migration of all fish from the deep water ship channel back to the Sacramento River" (DWR 2005).

The Fremont Weir is located at the upstream end of the Yolo Bypass, a 40 mile (64 km) long basin that functions as a flood control project on the Sacramento River. Green sturgeon are attracted by high floodwater flows into the Yolo Bypass basin and then concentrate behind Fremont Weir, which they cannot effectively pass (DWR 2005). Green sturgeon that concentrate behind the weir are subject to heavy illegal fishing pressure or become stranded behind the flashboards when high flood flows recede (M. Marshall, pers. comm.). Sturgeon can also be attracted to small pulse flows and trapped during the descending hydrograph (Harrell and Sommer 2003). Methods to reduce stranding and increase passage have been investigated by the California Department of Water Resources (DWR) and DFG (DWR 2007, J. Navicky, pers. comm.).

It is thought that adult and juvenile green sturgeon use the same migratory routes as Chinook salmon. Delta Cross Channel gate closures occur during the winter and early spring months sturgeon migration period (February through May) as required by State Water Resources Control Board (SWRCB) D-1641. Upstream migrating adult Chinook salmon are known to use the Delta Cross Channel as a migratory pathway when the gates are open (Hallock et al. 1970). When the gates are open, Sacramento River water flows into the central Delta and the Mokelumne and San Joaquin rivers providing migration cues. It is possible that attraction to water passing from the Sacramento River into the interior Delta causes delays and straying of green sturgeon, as it does to Chinook salmon (CALFED Science Program 2001, McLaughlin and McLain 2004). The Delta Cross Channel completely blocks juvenile and adult sturgeon migration to and from the interior Delta when the gates are closed.

Exposure to toxins. Exposure of green sturgeon to toxics has been identified as a factor that can lower reproductive success, decrease early life stage survival, and cause abnormal development, even at low concentrations (USFWS 1995, Environmental Protection Information Center et al. 2001, Klimley 2002). 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 and storage limitations and limited availability of dilution flows cause downstream copper and zinc levels to exceed salmonid tolerances. Treatment processes and improved drainage management in recent years have reduced the toxicity of runoff from Iron Mountain Mine to acceptable levels. Although the impact of trace elements on green sturgeon reproduction is not completely understood, negative impacts similar to those of salmonids are suspected.

Green sturgeon consume overbite and Asian clams, which are known to bioaccumulate selenium rapidly and lose selenium slowly (Linville et al. 2002, Doroshov 2006). Selenium is transferred to the egg yolk where it can cause mortality of larvae. Although chronic and acute exposure to toxics has been identified as a factor adversely affecting various lifestages of green sturgeon, the severity, frequency, geographic locations, and population level consequences of exposure to toxics have not been quantified (Linville et al. 2002, Doroshov 2006). However, Linville (2006) observed larvae to have increased skeletal deformities and mortality associated with maternal effects of selenium exposure, while smaller quantities (about 20mg/kg) decreased feeding efficiency and larger quantities (greater than 20mg/kg) reduced growth rates after four weeks (Lee et al. 2008a).

Methylmercury is another toxic substance that could potentially affect sturgeon development and survival. Between 2002 and 2006, sediment concentrations of methylmercury was highest in the Central Bay, while shallower parts of San Pablo Bay and Suisun Bay also contained levels greater than 0.2 parts per billion (ppb) (San Francisco Estuary Institute [SFEI] 2007). The amount of methylmercury resulting in the death of juvenile green sturgeon lies between 20 to 40mg/kg, with greater consumption increasing mortality significantly (Lee et al. 2008b).

Harvest. As a long-lived, late maturing fish with relatively low fecundity and periodic spawning, the green sturgeon is particularly susceptible to threats from overfishing (Musick 1999). Commercial harvest for green sturgeon occurs primarily along the Oregon and Washington coasts and within their coastal estuaries, with almost all catch being entirely in bycatch of three fisheries: white sturgeon commercial and sport fisheries, Klamath Tribal salmon gill-net fisheries, and coastal groundfish fisheries trawl fisheries (Adams et al. 2007). Total captures of green sturgeon in the Columbia River Estuary in commercial fisheries between 1985 and 2003 ranged from 46 fish per year to 6,000 (Adams et al. 2007). However, a high proportion of green sturgeon present in the Columbia River, Willapa Bay, and Grays Harbor (as high as 80 percent in the Columbia River) may be from the southern DPS (DFG 2002, Israel 2006).

Green sturgeon are also vulnerable to recreational sport fishing with the Bay-Delta estuary and Sacramento River, as well as other estuaries located in Oregon and Washington. 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 (DFG 2007a). If green sturgeon are caught incidentally and released while fishing for white sturgeon, it must be reported to DFG. The level of hooking mortality that results following release of green sturgeon by anglers is unknown. Sport fishing captures have declined through time; however, it is not known whether this is a result of reduced abundance, changed fishing regulations, or other factors. DFG (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. Because many sturgeon in the Columbia River, Willapa Bay, and Grays Harbor are likely from the southern DPS, additional harvest closures in these areas would likely benefit the southern DPS.

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) (M. Marshall, pers. comm.), as well as throughout the Bay-Delta (L. Schwall, pers. comm.). 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 (U.S. Fish and Wildlife Service [USFWS] 1995). 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 effort and poaching. Poaching rates in the rivers and estuary and the impact of poaching on green sturgeon abundance and population dynamics are unknown.

Reduced rearing habitat. Historical reclamation of wetlands and islands have reduced and degraded the availability of suitable in- and off-channel rearing habitat for green sturgeon. Further, channelization and hardening of levees with riprap has reduced in- and off-channel intertidal and subtidal rearing habitat. The resulting changes to river hydraulics, riparian cover, seasonal floodplain inundation, and geomorphology affect important ecoystem functions (Sweeney et al. 2004). The impacts of channelization and riprapping are thought to affect larval, post-larval, juvenile, and adult stages of sturgeon, as these life stages are dependent on the food web in freshwater and low salinity regions of the Bay-Delta.

Increased water temperature. Exposure to water temperatures greater than 63 ºF (17.2 °F) can increase mortality of sturgeon eggs and larvae (Pacific States Marine Fisheries Commission 1992) and temperatures above 69 °F (20.6 °C) are lethal to embryos (Cech et al. 2000). Temperatures near the RBDD on the Sacramento River historically occur within optimum ranges for sturgeon reproduction; however, temperatures downstream, especially later in the spawning season, were reported to be frequently above 63 °F (17.2 °F; USFWS 1995). High temperatures in the Sacramento River during the February to June period no longer appear to be a major concern for green sturgeon spawning, egg incubation, and juvenile rearing, as temperatures in the upper Sacramento River are actively managed for Sacramento River winter-run Chinook salmon. The Shasta temperature control device, installed at Shasta Dam in 1997, in combination with improved cold water pool management and storage within Lake Shasta, have resulted in improved cool water stream conditions within the upper Sacramento River.

Water temperatures in the Feather River may be inadequate for spawning and egg incubation as the result of releases of warmed water from Thermalito Afterbay (Surface Water Resources, Inc. 2003). Warmed water may be one reason why neither green nor white sturgeon are found in the river during low flow years (DFG 2002). It is not expected that water temperatures will become more favorable in the near future and this temperature problem will continue to be a factor affecting habitat quality for green sturgeon on the lower Feather River (DFG 2002).

The lack of flow in the San Joaquin River from dam and diversion operations and agricultural return flows contribute to higher temperatures in the mainstem San Joaquin River, offering less water to keep temperatures cool for sturgeon, particularly during late summer and fall. Though these effects are difficult to measure, temperatures in the lower San Joaquin River continually exceed preferred temperatures for sturgeon migration and development during spring months. Temperatures at Stevenson on the San Joaquin River near the Merced River confluence recorded on May 31 (spawning typically occurs during April-June; see Table A-9a) between 2000 and 2004 ranged from 77 to 82 °F (25 to 27.8 °C; California Data Exchange Center 2007). Juvenile sturgeon are also exposed to increased water temperatures in the Delta during the late spring and summer due to the loss of riparian shading and by thermal inputs from municipal, industrial, and agricultural discharges.

Non-native species. Green sturgeon have been impacted, both positively and negatively, by non-native species introductions through changes in trophic dynamics within the Delta and Suisun Bay. Many of the recent introductions of invertebrates have greatly affected the benthic fauna in the Delta and Suisun Bay. DFG (2002) reviewed many of the recent non-native invasive species introductions and the potential consequences to green sturgeon. The most notable species responsible for altering the trophic system of the Sacramento-San Joaquin Delta include the overbite clam and the Chinese mitten crab. Sturgeon regularly consume overbite and Asian clams, which is of particular concern because of the high bioaccumulation rates of these clams (Doroshov 2006). Although Chinese mitten crabs may be eaten by adult green sturgeon, it is possible they prey upon sturgeon eggs. However, the Chinese mitten crab population within the Bay-Delta system has undergone a substantial decline since 2002 and currently occurs in very low abundance (K. Hieb, pers. comm.) and, therefore, has not been a major factor affecting green sturgeon during this period.

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). Nobriga et al. (2005) found significant differences in water clarity and fish communities in those areas where submerged aquatic vegetation (SAV) was abundant when compared to open water habitats where SAV was not abundant. The occurrence of dense concentrations of SAV has been hypothesized to result in a number of potential effects on aquatic habitat including raising temperatures, reducing turbidity and dissolved oxygen levels, and inhibiting access to shallow water habitat by fish that cannot tolerate these conditions. The presence of non-native centrarchid species is strongly associated with the occurrence of Brazilian waterweed (Brown and Michniuk 2007). Brazilian waterweed forms thick “walls” along the margins of channels in the Delta. This growth is thought to prevent juvenile sturgeon from accessing shallow water habitat along channel edges. Water hyacinth creates dense floating mats that can impede river flows and alter the aquatic environment beneath the mats. By reducing water velocities near plants, these species reduce turbidity in the water column, potentially exposing young sturgeon to higher predation risk. Several investigators (Abrahams and Kattenfeld 1997, Utne-Palm 2002; cited in Nobriga and Feyrer 2007) observed that various lifestages of estuarine fish may use turbidity as a form of cover from predators. High densities of SAV have been observed to trap suspended sediments and reduce local water velocities resulting in reduced turbidity and increased water clarity associated with SAV. Although there is no direct evidence of a relationship between turbidity and vulnerability of species such as juvenile green sturgeon to predation mortality, Nobriga et al. (2005) found that an inverse relationship between SAV and water clarity, as well as an increase in the occurrence of several predatory fish species in association with SAV within the estuary, which potentially may increase the vulnerability of juvenile sturgeon to predation mortality.

Dissolved oxygen levels beneath mats of aquatic vegetation often drop below suitable levels for fish possibly due to the increased amount of decaying vegetative matter produced from the overlying mat as well as diel respiration by aquatic plants. Like Brazilian waterweed, water hyacinth is often associated with the margins of Delta waterways in its initial colonization, but can eventually cover the entire channel if conditions permit. This level of infestation may produce barriers to green sturgeon migration and access to rearing and foraging habitat within the Delta, although there is no evidence that this occurs.

Dredging. Hydraulic dredging is a common practice in the Sacramento and San Joaquin rivers, navigation channels within the Delta, and Suisun, San Pablo, and San Francisco bays to allow commercial and recreational vessel traffic. Such dredging operations pose risks to bottom oriented fish such as green sturgeon. Studies by Buell (1992) reported approximately 2,000 sturgeon entrained in the removal of one million tons of sand from the bottom of the Columbia River at depths of 60 to 80 feet (18 to 24 m). In addition, dredging operations can decrease the abundance of locally available prey species, and contribute to resuspension of toxics such as ammonia, hydrogen sulfide, and copper during dredging and dredge spoil disposal, and alter bathymetry and water movement patterns (NMFS 2006b).

Reduction in turbidity. Turbidity levels in the Delta have been reduced over the past few decades (Jassby et al. 2002). This reduction may have had detrimental effects to green sturgeon. Although little is known about the effects of reduced turbidity on green sturgeon, Gadomski and Parsley (2005) have found that larval white sturgon predation by prickly sculpin was greater in water with lower turbidity. Because green and white sturgeon larvae may have similar behavior and morphology, this effect likely applies to green sturgeon, as well. However, larval sturgeon are found close to spawning locations generally upstream of the Delta, where turbidity is already lower than the Delta.

Entrainment. Larval sturgeon are susceptible to entrainment from non-project water diversion facilities as a result of their migratory behavior and habitat selection within the rivers and Bay-Delta estuary. The overall impact of entrainment of fish populations is typically unknown (Moyle and Israel 2005), however there is enough descriptive information to predict where green sturgeon may be entrained. Herren and Kawasaki (2001) documented 431 non-project diversions located on the Sacramento River between Sacramento and Shasta Dam. Entrainment information regarding larval and post-larval individuals of the green sturgeon is unreliable because entrainment at these diversions has not been monitored and field identification of green sturgeon larvae is difficult. USFWS staff are working on identification techniques and are optimistic that green sturgeon greater than 40 mm can be identified in the field (Poytress 2006). Sturgeon collected at the Glenn-Colusa Irrigation District diversion located on the upper Sacramento River are not identified to species, but are assumed to primarily consist of green sturgeon because white sturgeon are known to spawn primarily downstream (Schaffter 1997). Although screens at the Glenn-Colusa Irrigation District diversion satisfy both the NMFS and DFG screening criteria for salmonids, the effectiveness of these criteria is unknown for sturgeon. Low numbers of green sturgeon have also been identified and entrained at the Red Bluff Research Pumping Plant (Borthwick et al. 1999).

In the Feather River, there are 8 large diversions greater than 10 cfs and approximately 60 small diversions between 1 and 10 cfs between the Thermalito Afterbay outlet and the confluence with the Sacramento River (USFWS 1995). Based on potential entrainment problems of green sturgeon elsewhere in the Central Valley and the presence of multiple screened and unscreened diversions on the Feather River, it is thought that operation of unscreened water diversions on the Feather River are a possible threat to juvenile green sturgeon.

Presumably, as green sturgeon juveniles grow, they become less susceptible to entrainment as their swimming ability and capacity to escape diversions improves. The majority of North American green sturgeon captured in the Bay-Delta are between 200 and 500 mm in length (DFG 2002). Herren and Kawasaki (2001) inventoried water diversions in the Delta finding a total of 2,209 diversions of various types; only 0.7 percent of which were screened. The majority of these diversions were between 12 and 24 inches in diameter. The vulnerability of juvenile green sturgeon to entrainment at these unscreened diversions is largely unknown. Results of limited entrainment studies at diversions within the Delta suggest that larger juvenile green sturgeon have a lower risk of entrainment mortality. The largest diversions within the Delta are the State Water Project (SWP) and Central Valley Project (CVP) export facilities, located in the southern Delta, where a low number of juvenile green sturgeon have been recorded as part of fish salvage monitoring (DFG 2002). The average number of green sturgeon taken per year at the SWP Skinner Fish Facility was 87 individuals between 1981 and 2000, and 20 individuals from 2001 through 2007 (M. Donnellan, unpublished data). At the CVP Tracy Fish Collection Facility, green sturgeon counts averaged 246 individuals per year between 1981 and 2000, and 53 individuals per year between 2001 and 2007 (M. Donnellan, unpublished data). This reduction in salvage is consistent with a significant reduction in white sturgeon take at the salvage facilities within the same time periods (NMFS 2005).

Stranding. 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 2005). As the Bypass recedes, these sturgeon become stranded behind the flashboards of the weir and can be subjected to heavy illegal fishing pressure (M. Marshall, USFWS, pers. comm.). Sturgeon can also be attracted to small pulse flows and trapped during the descending hydrograph (Harrell and Sommer 2003). Methods to reduce stranding and increase passage have been investigated (J. Navicky, pers. comm.).

A9.6 Relevant Conservation Efforts

The Central Valley Project Improvement Act's Anadromous Fish Restoration Program has a goal of supporting efforts that lead to doubling the natural production of anadromous fish in the Central Valley at a sustainable, long-term basis, at levels not less than twice the average levels attained during the period of 1967 to 1991. Although most efforts of the Anadromous Fish Restoration Program have focused on Chinook salmon as a result of their listing history and status, sturgeon may receive some unknown amount of incidental benefit from these restoration efforts. For example, the acquisition of water for flow enhancement on tributaries to the Sacramento River, fish screening for the protection of Chinook salmon and Central Valley steelhead, spawning gravel augmentation, or riparian revegetation and instream restoration projects would likely have some ancillary benefits to sturgeon. The Anadromous Fish Restoration Program has also invested in a green sturgeon research project that has helped improve our understanding of the life history requirements and temporal patterns of the southern DPS of North American green sturgeon.

Many beneficial actions have originated from and been been funded by the CALFED program including such projects as floodplain and instream restoration, riparian habitat protection, fish screening and passage projects, research on non-native invasive species and contaminants, restoration methods, watershed stewardship, and education and outreach programs. Prior Federal Register notices have reviewed the details of the Central Valley Project Improvement Act (CVPIA) and CALFED programs and potential benefits for anadromous fish, particularly Chinook salmon and Central Valley steelhead (69 FR 33102). Projects potentially benefiting sturgeon primarily consist of fish screen evaluation and construction projects, restoration evaluation and enhancement activities, and contaminant studies. Two evaluation projects specifically addressed green sturgeon while the remaining projects primarily address listed salmonids and fishes of the area in general. The new information developed through these research investigations will be used to enhance the understanding of the risk factors affecting population dynamics and recovery, thereby improving the ability to develop effective management measures.

The Delta Regional Ecosystem Restoration Implementation Plan (DRERIP) was formed to guide the implementation of CALFED Ecosystem Restoration Plan elements within the Delta (DFG 2007b). The DRERIP team has created a suite of ecosystem and species conceptual models, including green sturgeon, that document existing scientific knowledge of Delta ecosystems. The DRERIP team is in the process of using these conceptual models to assess the suitability of actions proposed in the Ecosystem Restoration Plan for implementation. DRERIP conceptual models have been used in the analysis of proposed BDCP conservation measures.

In response to passage impediment concerns to green sturgeon and other migratory species, operations of the RBDD have been modified since its construction in 1964 to reduce the "gates-in" period. In 2009, U.S. Bureau of Reclamation received funding for the Fish Passage Improvement Project at the RBDD to build a pumping facility to provide reliable water supply for high-valued crops in Tehama, Glenn, Colusa, and northern Yolo counties while providing year-round unimpeded fish passage. This project, which is expected to be completed in late 2012, will eliminate passage issues for sturgeon and other migratory species.

The combination of increased law enforcement and new sport fishing regulations adopted over the past several years specifically to protect sturgeon and reduce their harvest is expected further reduce illegal fishing practices as well as the effects of incidental harvest of green sturgeon by recreational anglers throughout the range of the species. Mitigation under the Delta Fish Agreement has increased the number of wardens enforcing harvest regulations for steelhead and other fish in the Bay-Delta and upstream tributaries by creating the Delta Bay Enhanced Enforcement Program (DBEEP).

A9.7 RECOVERY GOALS

On November 12, 2009, NMFS announced its intent to develop a recovery plan for the Southern DPS of North American green sturgeon (*Acipenser medirostris*) and has requested information from the public (74 FR 58245). This plan has not yet been developed.

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APPENDIX A10. PACIFIC LAMPREY (*ENTOSPHENUS TRIDENTATUS*)

A10.1 LEGAL STATUS

The Pacific lamprey is not listed under the California or federal Endangered Species Acts (ESA).

A broad group of West Coast conservation organizations petitioned the U.S. Fish and Wildlife Service (USFWS) on January 27, 2003 to list Pacific lamprey, along with three other lamprey species on the West Coast, as threatened or endangered Klamath-Siskiyou Wildlands Center 2003). However, the petition was declined in a 90-day finding on December 27, 2004, citing insufficient evidence that listing was warranted (69 FR 77158).

A10.2 SPECIES DISTRIBUTION AND STATUS

A10.2.1 Range and Status

The Pacific lamprey is the most widely distributed lamprey species on the west coast of the United States. The species occurs from Hokkaido Island, Japan (Morrow 1980) along the Pacific Rim to Rio Santo Domingo, Baja California, Mexico (Ruiz-Campos and Gonzalez-Guzman 1996). A single individual was caught in 1889 offshore of Clarion Island, Revillagigedo Islands, Mexico, approximately 386 km southwest of Cabo San Lucas (Renaud 2008). Individuals inhabit major river systems, including the Columbia, Fraser-Trinity, Klamath, Eel, and Sacramento-San Joaquin rivers, as well as smaller coastal streams (see Figure A-10a). In general, populations south of San Luis Obispo are scattered and irregular (Swift et al. 1993). Populations may exist in other rivers, but are easily overlooked and have been the subject of few targeted sampling efforts (Moyle 2002). The species is usually absent from highly-altered or polluted streams within its geographic range, although appears to be persistent in currently occupied suitable streams (Moyle 2002).

In the Central Valley, Pacific lamprey occur in both the lower Sacramento and San Joaquin Rivers (Moyle 2002) and many of their tributaries including the Stanislaus, Tuolumne, Merced, and King Rivers (Brown and Moyle 1993, 69 FR 77158).

Population trends are unknown in California, although anecdotal evidence indicates that populations have been in decline (Moyle 2002, 69 FR 77158).

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Figure A-10a. Pacific Lamprey Inland Range in California

A10.2.2 Distribution and Status in the Plan Area

Individuals outmigrating from Sacramento and San Joaquin River watersheds pass through the Plan Area during winter and spring on their way to the Pacific Ocean. Emigrating adults pass through the Plan Area on their way upstream towards spawning grounds primarily between March and June. It is unknown to what extent Pacific lamprey use the Plan Are for purposes other than a migration corridor.

Status and trend data are extremely sparse and unreliable. There are no monitoring programs that target Pacific lamprey in the Delta and those that catch Pacific lamprey do not catch them regularly enough to establish trends through time. In addition, Pacific lamprey are inconspicuous, often overlooked, and ammocoetes can be difficult to distinguish from ammocoetes of the co-occurring river lamprey (H. Webb, pers. comm).

A10.3 HABITAT REQUIREMENTS AND SPECIAL CONDITIONS

The habitat requirements of Pacific lamprey have not been well studied. It is thought that adults need clean, gravelly riffles in permanent streams to spawn successfully and that these requirements are thought to be similar to those of salmonids (Moyle 2002). Ammocoetes live in silty backwaters and eddies with muddy or sandy substrate into which they burrow. Ammocoetes require water temperatures that are lower than 25 °C (76 °F) (Moyle et al. 1995). Meeuwig et al. (2004) found significant death or deformation of eggs and early stage ammocoetes in water greater than 72 °F (22 °C). Lamprey can pass barriers that other fish cannot, although large dams and other habitat modifications remain barriers to migration. Oceanic adults are thought to remain relatively close to the mouths of their home spawning streams where host/prey concentrations may be higher (Moyle 2002).

A10.4 LIFE HISTORY

Pacific lamprey are anadromous, beginning their migration into freshwater towards upstream spawning areas primarily between early March and late June (Moyle 2002). Most upstream migration occurs at night and occurs in pulses. Spawning habitat requirements are thought to be similar to those of salmonids. There is some evidence that Pacific lamprey in larger river systems, such as the Klamath and Eel Rivers, have distinct runs similar to Chinook salmon (Moyle 2002). Both sexes contribute to nest construction by removing larger stones from a gravelly substrate, creating a shallow depression. These simple nests occur in gravelly substrata at a depth of 30-150 cm with moderately swift currents and water temperatures typically of 12-18 °C (53.6 to 64.4 °F) (Moyle 2002). External fertilization of eggs occurs just in front of the nest, after which the fertilized eggs are washed into the nest. Fecundity is unknown, but has been estimated at 98,000 to 238,400 eggs per female (Kan 1975 as cited in Close et al. 2002). Spawning is repeated until both individuals are spent. Adults typically die after spawning.

Eggs hatch into ammocoetes after approximately 19 days at 15 °C, spend a short time in the nest, and then drift downstream to suitable areas in sand or mud (Moyle 2002). Ammocoetes remain in freshwater for approximately 5 to 7 years, where they bury into silt and mud and feed on algae, organic material, and microorganisms. Ammocoetes change locations during this stage.

Ammocoetes begin metamorphosis into macropthalmia (juveniles) when they reach 14-16 cm total length (TL). Individuals develop external features (eyes, oral disc, and color changes) and experience internal and physiological changes that prepare them for their predatory life stage in the ocean (McPhail and Lindsey 1970). Downstream migration begins upon completion of this metamorphosis, generally coinciding with high flow events in winter and spring (Moyle 2002).

Adults spend 3-4 years in the ocean in British Columbia, but this length is thought to be shorter in more southern areas (Moyle 2002). Adult remain close to the mouths of the rivers from which they came, likely because their prey is most abundant in estuaries and other coastal areas (Moyle 2002). Individuals attack a wide variety of fishes, include salmon, Pacific herring, and flatfishes, in the ocean (Beamish 1980). Pacific lamprey are thought to be preyed upon in the ocean by sharks, other fish, otters, seals, and sea lions (Roffe and Mate 1984, Moyle 2002).

A10.5 Threats and Stressors

Evaluation of the threats and stressors to Pacific lamprey has been limited. Therefore, much of the following discussion is based on a recent workshop to identify the state of knowledge, including knowledge gaps, on the biology, population structure, habitat of and threats to Pacific lamprey (Luzier et al. 2009).

Reduced Access (Passage) to Spawning Habitat. Artificial barriers, including dams, culverts, water diversions, tidal gates, and other barriers, can impede or completely block the upstream migration of adults to spawning grounds, resulting in impacts to the distribution and abundance of lamprey (Klamath-Siskiyou Wildlands Center et al. 2003, Luzier et al. 2009). Lamprey adults may have difficulty passing over barriers using ladders and other passage structures designed for salmonids, possibly due to high water velocity, sharp angles, culverts with drop-offs, or insufficient resting areas (Kostow 2002). Hydroelectric projects and water diversions may entrain or impinge weak-swimming macropthalmia (Moursund et al. 2000).

Reduced Access (Passage) to Downstream Habitat. Artificial barriers, including dams, culverts, water diversions, tidal gates, and other barriers, can impede or completely block the downstream migration of ammocoetes and macropthalmia towards the ocean, resulting in impacts to the distribution and abundance of lamprey (Luzier et al. 2009). Lamprey tend to outmigrate deeper in the water column such that traditional spill gates meant to aid migration of salmonids may not be effective on lamprey and may block passage (Moursund et al. 2003). Pacific lamprey populations cannot persist for more than a few years above impassable barriers (Beamish and Northcote 1989).

Stranding. Rapid changes in stream flows as a result of reservoir management can dewater stream beds and strand ammocoetes residing in the substrate. Water diversions and instream construction projects, such as culvert replacements, may also dewater reaches of streams and strand ammocoetes (Streif 2007). Because Pacific lamprey ammocoetes burrow in upstream sediments for 5-7 years in high densities, a dewatering event may affect multiple age classes burrowing together in a single stream reach (Luzier et al. 2009).

Dredging. Dredging associated with channel or irrigation screen maintenance and mining may affect many age classes at once due to their "colonial" nature and long upstream life stage (5-7 years) (Luzier et al. 2009). Beamish and Youson (1987) found that only 3-26 percent of lamprey that pass through a dredge survive. Further, it has been suggested that suction dredge mining was responsible for the decline or even loss of populations in some basins (Kostow 2002).

Chemical Poisoning and Toxins. Ammocoetes spend 5-7 years living in silty areas that are known to accumulate high levels of toxins. As a result, lamprey tend to have high body burdens of toxins relative to other fish species (Haas and Ichikawa 2007, Bettaso and Goodman 2008). Despite this apparent tolerance for high levels of toxins, lamprey are thought to be susceptible to toxins (Kostow 2002).

Ocean Conditions. Reductions the availability of host/prey organisms in the ocean (e.g., salmon and flatfishes) as a result of poor ocean conditions may negatively affect lamprey survival and growth, although very little is known about the oceanic stage of Pacific lamprey (Luzier et al. 2009).

Water Temperature. Elevated water temperature (greater than 22 °C [72 °F]) can cause mortality or significant deformation of eggs and young ammocoetes in laboratory conditions (Meeuwig et al. 1999). Degraded streams with a water temperature greater than 22 °C during early/mid-summer while lamprey spawn and young ammocoetes develop may be common (Luzier et al. 2009).

Disease. Pacific lamprey disease incidence is not well understood, but it is thought that disease may impact lamprey health to the point at which their ability to reproduce and survive is reduced (Luzier et al. 2009).

Overutilization. The extent to which harvest affects the population level effect on Pacific lamprey has not been well studied, but could represent a large proportion of spawning adults. Pacific lamprey adults and ammocoetes are harvested for use as bait to catch other species (Luzier et al. 2009). In addition, the fish is important to tribes in the Pacific Coast for sustenance, medicine, and ceremonial purposes (Close et al. 2002). Pacific lamprey for food and commercial purposes has declined from historical levels and Washington and Oregon have banned harvest for bait. However, harvest has not declined in California, where there are no regulations on lamprey harvest (69 FR 77158).

Predation. Mammals, birds, and other fish species consume lamprey at all life stages (Luzier et al. 2009). Ammocoetes are consumed by terrestrial mammal and birds, fish, and other species. Adult lamprey are consumed by otters, pinnipeds, and sturgeon.

Stream and Floodplain Degradation. The high density and limited mobility of lamprey ammocoetes in streams can potentially make them more vulnerable to channel alterations such as channelization, loss of riffle and side channel habitat, and scouring (Streif 2007, Luzier et al. 2009). Loss or alteration of habitat can also limit spawning if occurring in spawning reaches.

Non-Native Species. Non-native species, including striped bass, centrarchids, and catfish, are believed to consume juvenile and adult lamprey and may pose a threat to population sizes of lamprey (Streif 2007, Luzier et al. 2009). Many of these non-native species have become established within the range of Pacific lamprey in the Central Valley and have populations that are thriving despite recent declines in many native species (Baxter et al. 2008).

Translocation. It is unknown whether migrating adults cue solely on ammocoete pheromones or on other upstream cues to guide them to natal streams to spawn. If an ammocoete pheromone cue does not drive adult migration, translocation of individuals to an area previously extirpated would not affect adult migration cues.

Climate Change. Future climate change is expected to further increase water temperatures and modify the timing of flow-related environmental cues upon which Pacific lamprey rely for life history events (e.g., outmigration, spawning, etc.) (Luzier et al. 2009).

Extirpation. It is unknown whether migrating adults cue solely on ammocoete pheromones or on other upstream cues to guide them to natal streams to spawn. If they cue solely on ammocoete pheromones, extirpation of local populations would have large effects on recolonization of natal streams (Luzier et al. 2009).

A10.6 RELEVANT CONSERVATION EFFORTS

Along with several tribes, state and federal agencies are increasingly incorporating Pacific lamprey into management and monitoring plans to increase the overall body of knowledge and conserve the species.

There have been very few efforts to conserve Pacific lamprey in the Central Valley of California. The CALFED Ecosystem Restoration Program (ERP) designated the entire lamprey family as "Enhance and/or Conserve" (CALFED Bay-Delta Program 2000). This designation indicates that the ERP will undertake actions to conserve and enhance their abundance and distribution and the community diversity in which they live for their long-term stability.

There has been work in the Columbia River basin to modify new or existing ladders and structures to facilitate lamprey passage, such as creating holding areas where lamprey can rest (Columbia River Basin Lamprey Technical Workgroup 2004).

The Pacific Lamprey Conservation Initiative, led by the USFWS, was initiated in 2007 to "facilitate communication and coordination relative to the conservation of Pacific lampreys throughout their range" (USFWS 2007). The goal of the initiative is to restore Pacific lamprey populations and improve their habitat. Anticipated actions from the Initiative include: development of a Pacific Lamprey Conservation Plan, identification of funding for implementation of the Initiative and Plan, development of a network of interested parties, funding immediate conservation actions, and improvements in communication of Pacific lamprey conservation efforts.

A10.7 Recovery Goals

A recovery plan has not been prepared for this species and no recovery goals have been established because the species is not listed under the Federal or California ESA.

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APPENDIX A11. RIVER LAMPREY (*LAMPETRA AYRESII*)

A11.1 LEGAL STATUS

The river lamprey is not listed under the state or federal Endangered Species Acts (ESA).

A broad group of West Coast conservation organizations petitioned the U.S. Fish and Wildlife Service (USFWS) on January 27, 2003 to list river lamprey, along with three other lamprey species on the West Coast, as threatened or endangered (Klamath-Siskiyou Wildlands Center 2003). However, the petition was declined in a 90-day finding on December 27, 2004, citing insufficient evidence that listing was warranted (69 FR 77158).

A11.2 SPECIES DISTRIBUTION AND STATUS

A11.2.1 Range and Status

The river lamprey occurs from near Juneau, Alaska, to San Francisco Bay, California (Moyle 2002). Outside of California, there are widely scattered and isolated populations throughout its range. River lamprey are common in British Columbia, the center of their geographic range. Within California, river lamprey can be found in the Central Valley, Napa River, Sonoma Creek, Alameda Creek, Salmon Creek, and in tributaries of the lower Russian River (see Figure A-11a). In the Central Valley, river lamprey are found in the lower Sacramento and San Joaquin River drainages, including the Stanislaus and Tuolumne Rivers. They may exist in other tributaries of these rivers, but are easily overlooked and have been the subject of few targeted sampling efforts (Moyle 2002). The species appears to be more abundant in the lower Sacramento-San Joaquin River system than in other streams in California.

Population trends are unknown in California, although declines are thought to have occurred synonymously with freshwater habitat degradation (Moyle 2002).

A11.2.2 Distribution and Status in the Plan Area

Individuals outmigrating from Sacramento and San Joaquin River watersheds pass through the Delta on their way to the Pacific Ocean and emigrating adults pass through the Plan Area on their way upstream towards spawning grounds. The extent to which river lamprey use the Plan Area for purposes other than a migration corridor is unknown. However, outmigrating lamprey macropthalmia in the final stages of metamorphosis to adults hold just upstream of salt water until late spring. Depending on the position of X2, this location could be within the Plan Area.

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Figure A-11a. River Lamprey Inland Range in California

Status and trend data are extremely sparse and unreliable. There are no monitoring programs that target river lamprey in the Delta and those that catch river lamprey do not catch them regularly enough to establish trends through time. River lamprey are inconspicuous, often overlooked, and ammocoetes can be difficult to distinguish from ammocoetes of the co-occurring Pacific lamprey (H. Webb, pers. comm).

A11.3 Habitat Requirements and Special Conditions

The habitat requirements of river lamprey have not been well studied. It is thought that adults need clean, gravelly riffles in permanent streams to spawn successfully. These requirements are thought to be similar to those of salmonids. Ammocoetes live in silty backwaters and eddies with muddy or sandy substrate into which they burrow (Moyle et al. 1995). Ammocoetes require water temperatures that are lower than 25 °C (77 °F; Moyle et al. 1995). Lamprey can pass barriers that other fish cannot, although large dams and other habitat modifications remain barriers to migration. Lamprey may have difficulty passing over barriers using ladders and other passage structures designed for salmonids, possibly due to high water velocity, sharp angles, culverts with drop-offs, or insufficient rest areas (Kostow 2002). There has been some work in the Columbia River basin to modify new or existing ladders and structures to facilitate lamprey passage, such as creating holding areas where lamprey can rest (Columbia River Basin Lamprey Technical Workgroup 2004).

Although generally considered anadromous, river lamprey can live in freshwater as adults. For example, the population of river lamprey living in land-locked Sonoma Creek spend their entire life in freshwater.

A11.4 Life History

The biology of the river lamprey has not been well studied in California. As a result, much of this section is derived from information known for river lamprey from British Columbia. The potential exists for dissimilar life histories between fish in these two locations due to differences in physical factors (e.g., temperature, hydrology).

River lamprey are anadromous, beginning their migration into freshwater in the fall towards suitable spawning areas upstream (Moyle et al. 1995, Moyle 2002). Exact spawning locations are not known, although spawning habitat requirements are thought to be similar to those of salmonids. Fidelity to natal streams is also unknown. Spawning occurs from February through May in gravelly riffles in which individuals dig saucer-shaped depressions (Moyle 2002). Adults die after spawning. Fecundity is not well documented, but a study of two females in Cache Creek reported that one female (23 centimeters [cm; 9.1 inches] total length) produced approximately 11,400 eggs and the other (17.5 cm [6.9 inches] total length) produced approximately 37,300 eggs (Vladykov and Follett 1958). The eggs hatch into ammocoetes that remain in freshwater for approximately 3 to 5 years in silty or sandy low-velocity backwaters or stream edges where they bury into the substrate and filter-feed on algae, detritus, and microorganisms (Moyle 2002).

Ammocoetes begin metamorphosis into macropthalmia and then adults during summer at approximately 12 cm total length. This process takes nine to ten months during which individuals may shrink in length by up to 20 percent (Moyle 2002). Prior to entering the ocean, macropthalmia congregate just upstream of salt water until their esophagus opens (Beamish and Youson 1987). Once the esophagus is opened, new adults can properly osmoregulate and can then enter the ocean (Moyle 2002). Adults spend approximately 3 to 4 months in the ocean where they grow rapidly to 25 to 31 cm total length. If the ammocoete stage is 3 to 5 years, the total life span of river lamprey is estimated to be 6 to 7 years (Moyle et al. 1995).

River lamprey adults are parasitic during both freshwater and saltwater phases. Adults feed on a variety of host fish species that are small to intermediate size (4 to 12 inches total length) (Moyle et al 1995), the most common of which are thought to be herring and salmon (Beamish and Youson 1987). In Canada, river lamprey predation is considered to be a significant source of salmon mortality (Beamish and Neville 1995). Individuals feed by attaching to the back of their prey above the lateral line and eating the muscle tissue, even after the host fish dies (Moyle 2002). More than one lamprey can attach to a host salmon (Beamish and Youson 1987).

A11.5 Threats and Stressors

There have been no formal evaluations conducted that assess the threats and stressors to river lamprey. Therefore, much of the following discussion is based on limited resources or has been derived from the co-occurring Pacific lamprey as part of the Pacific Lamprey Conservation Initiative. The primary threat to river lamprey is thought to be loss or degradation of habitat through dams, diversions, toxics, stream channelization, dredging, and urbanization (Moyle et al. 1995, Luzier et al. 2009). Dams have altered flows in channels and limited access to spawning grounds. Toxics may have both lethal and sublethal effects on individuals. Stream channelization, dredging, and diversions have altered flow patterns and rates in channels. Urbanization has degraded habitat by increasing loads of certain toxics, changing runoff patterns, and altering the configuration of some channels. Future climate change is expected to further increase water temperatures and modify the timing of flow-related environmental cues upon which Pacific lamprey rely for life history events (e.g., outmigration, spawning, etc.).

A11.6 Relevant Conservation Efforts

There have been very few efforts to conserve river lamprey in the Central Valley of California. The CALFED Ecosystem Restoration Program (ERP) designated the entire lamprey family as “Enhance and/or Conserve” (CALFED Bay-Delta Program 2000). This designation indicates that the ERP will undertake actions to conserve and enhance their abundance and distribution and the community diversity in which they live for their long-term stability.

River lamprey is currently listed as a covered species under the Butte County Habitat Conservation Plan, but specific conservation measures have not yet been written.

A11.7 Recovery Goals

A recovery plan has not been prepared for this species and no recovery goals have been established because the species is not listed under the Federal or California ESA.

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