Appendix 9B

Aquatic Species Life History Accounts

This appendix provides additional information on the life history characteristics of the target aquatic species assessed in the Remanded Biological Opinions on the Coordinated Long-Term Operation of the Central Valley Project (CVP) and State Water Project (SWP) Environmental Impact Statement (EIS). This information is intended to provide a more holistic understanding of how these species use the water bodies influenced by operation of the CVP and SWP and to help clarify relationships that provide the logical foundation for conclusions regarding the potential environmental consequences associated with changes in operation.

This appendix addresses the following species:

- River Lamprey
- Pacific Lamprey
- Green Sturgeon
- White Sturgeon
- Chinook Salmon
  - Winter-run Chinook Salmon
  - Central Valley Spring-run Chinook Salmon
  - Central Valley Fall-run and Late Fall-run Chinook Salmon
  - Upper Klamath and Trinity Rivers Spring-run Chinook Salmon
- Central Valley Steelhead
- Klamath Mountains Province Steelhead
- Sacramento Splittail
- Longfin Smelt
- American Shad
- Eulachon
- Striped Bass
- Southern Resident Killer Whale

9B.1 River Lamprey (*Lampetra ayresii*)

9B.1.1 Legal Status

Federal: None
State: Species of Special Concern

River Lamprey was petitioned for listing by a number of conservation groups in 2003, along with three other lamprey species (Klamath-Siskiyou Wildlands Center et al. 2003). The petition was declined by the U.S. Fish and Wildlife Service (USFWS) in 2004 because of insufficient evidence that listing was warranted.

9B.1.2 Distribution

River Lamprey are found in large coastal streams from just north of Juneau, Alaska, to the San Francisco Bay (Vladykov and Follett 1958, Wydoski and Whitney 1979). The Sacramento and San Joaquin basins are at the southern edge of their range (Moyle et al. 2009). Little is known regarding their abundance and distribution within California; they seem to be primarily associated with the lower portions of certain large river systems, and most records for the state are from the lower Sacramento-San Joaquin system, especially the Stanislaus and Tuolumne rivers (Moyle et al. 1989, Moyle 2002). In the Sacramento River, they have been documented upstream to at least Red Bluff Diversion Dam (RBDD) (Hanni et al. 2006, Moyle et al. 2009). River Lamprey have also been collected in the Feather River, American River, Mill and Cache creeks (Vladykov and Follett 1958, Hanni et al. 2006, Moyle et al. 2009). River Lamprey have not been documented during rotary screw trapping efforts in Clear, Battle, and Deer creeks, or in the Yuba River (Hanni et al. 2006). Other streams where they have been found in California outside of the Central Valley include the Napa and Russian rivers, and Alameda, Sonoma, and Salmon creeks (DWR et al. 2013).

9B.1.3 Life History and Habitat Requirements

River Lamprey are a small parasitic anadromous species. Most studies of their biology have been conducted in British Columbia; relatively little is known regarding their life history and habitat requirements in California (Moyle 2002).

Adult River Lamprey migrate from the ocean into spawning areas in the fall. Adults of both sexes construct nests in gravel at the upstream end of riffles (Wydoski and Whitney 1979, Beamish and Youson 1987, Moyle 2002). Eggs are deposited and fertilized in these depressions, after which the adults typically die, similar to other species of lampreys. In the Sacramento-San Joaquin basin of California, most spawning is believed to occur in April and May (Vladykov and Follett 1958; Scott and Crossman 1973) at temperatures of about 55 to 56 degrees Fahrenheit (°F) (Wang 1986). Two females in Cache Creek were reported to have 11,400 and 37,300 eggs each (Vladykov and Follett 1958).

After hatching, young ammocoetes (the larval stage of lamprey) drift downstream to settle in the silt-sand substrates of backwaters, eddies, and pools, where they remain burrowed for approximately 3 to 5 years (Moyle 2002). At this stage, they are filter feeders, with a diet consisting of algae (primarily diatoms) and other organic detritus and microorganisms (Wydoski and Whitney 1979). Good water quality and temperatures not exceeding 77°F are believed to be necessary for their survival (Moyle 2002). Their metamorphosis into adults begins in July when they reach about 12 centimeters (cm) (4.7 in) (Beamish 1980), and is not complete for about 9 to 10 months until around April the following spring, when the esophagus opens and adults are able to osmoregulate (Beamish and Youson 1987, Moyle 2002). This is a more extended period of metamorphosis than observed in other lamprey species. During this time, they are believed to live in deep waters of the river channel. Just prior to the completion of metamorphosis, the juvenile lampreys (macropthalmia) congregate immediately upstream of salt water and enter the estuary or ocean from May to July (Beamish and Youson 1987).

Adults spend 3 to 4 months in salt water, remaining close to shore and growing to lengths of about 25 to 31 cm. In the estuary or ocean, River Lamprey are obligate parasites, typically killing their host in the process of feeding. They most commonly parasitize fishes 10 to 30 cm long, feeding near the surface on smelt, herring, and mid-size salmonids (Beamish 1980, Roos et al. 1973, Beamish and Neville 1995). In Canada, they have been documented to be an important source of mortality on salmon (Beamish and Neville 1995). In the fall, adults migrate back upstream into spawning areas and cease to feed. Fidelity to the streams in which they were spawned remains unknown.

The species is expected to use Delta habitats primarily as a migration corridor (DWR et al. 2013), and have been collected in Suisun Bay, Montezuma Slough, and Delta sloughs during California Department of Fish and Wildlife (DFW) plankton sampling efforts. CVP and SWP salvage data indicate that they are found in the salvage primarily from December through March (DWR et al. 2013). Juveniles are weak swimmers, frequently becoming entrained in water diversions or turbine intakes of hydroelectric projects or becoming impinged on screens meant to bypass juvenile salmonids or other fish (USFWS 2007).

Very little is known regarding the distribution, habitat use, and life history of this species in the action area. Numerous adults (less than 200 millimeters [mm]), presumably of spawning age, have been captured in rotary screw traps at RBDD from March through June (Hanni et al. 2006). Individuals smaller than most adults (greater than 200 mm), likely outmigrating macropthalmia, have been captured at RBDD and Feather River rotary screw traps from late September through early June (Hanni et al. 2006). Factors limiting River Lamprey populations in the Sacramento River are likely similar to those limiting salmonids (Moyle et al. 2009). Quantitative data on populations are extremely limited, but loss and degradation of historical habitats suggest populations have likely declined (Moyle et al. 2009).

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9B.2 Pacific Lamprey (*Entosphenus tridentatus*)

9B.2.1 Legal Status

Federal: None
State: None

The Pacific Lamprey was petitioned for listing by 12 conservation groups in 2003, along with three other lamprey species (Klamath-Siskiyou Wildlands Center et al. 2003). The petition was declined by USFWS in 2004 because of insufficient evidence that listing was warranted (USFWS 2004).

9B.2.2 Distribution

The Pacific Lamprey is a widely distributed anadromous species found in river systems along the northern margin of the Pacific Ocean from central Baja California north along the west coast of North America to the Bering Sea in Alaska (Ruiz-Campos and Gonzales-Guzman 1996, Lin et al. 2008). Historically, Pacific Lamprey were generally distributed wherever salmon and steelhead occurred and sometimes upstream of waterfalls that are impassable to anadromous salmonids. In California, they were historically found along the entire coast and far inland (Moyle et al. 2009). However, recent data and anecdotal accounts indicate that distribution of the Pacific Lamprey has been reduced in many river systems, including the Sacramento-San Joaquin (Moyle et al. 2009). Although widely distributed in the Sacramento-San Joaquin basin, the species is absent from as much as 80 percent of its historical spawning habitats, primarily due to migratory barriers (Moyle et al. 2009).

9B.2.3 Life History and Habitat Requirements

9B.2.3.1 Adult Migration

Pacific Lamprey are anadromous, rearing in freshwater before outmigrating to the ocean, where they grow to full size prior to returning to their natal streams to spawn. Pacific Lamprey are thought to remain in the ocean for approximately 18 to 40 months before returning to freshwater as sexually immature adults, typically from late winter until early summer (Kan 1975, Beamish 1980). After entering freshwater from the ocean, adult Pacific Lamprey typically spend approximately 1 year in freshwater prior to spawning (Robinson and Bayer 2005, Clemens et al. 2009, Stillwater Sciences 2010, Lampman 2011). The adult freshwater residence period can be divided into three distinct stages: (1) Initial migration from the ocean to holding areas, (2) pre-spawning holding, and (3) secondary migration to spawn (Robinson and Bayer 2005; Clemens et al. 2010, 2012).

The initial migration from the ocean to upstream holding areas occurs from approximately January until early August (Stillwater Sciences 2010, McCovey 2011, Clemens et al. 2012). In the Eel River and the nearby Klamath River, where ample information exists, entry into freshwater from the ocean generally begins in January and ends by June (Petersen-Lewis 2009, McCovey 2010, Stillwater Sciences 2010). Most individuals cease upstream migration by mid-July, although some individuals continue moving into August (McCovey 2010). Data from mid-water trawls in Suisun Bay and the lower Sacramento and San Joaquin rivers indicate that adults likely migrate into the Sacramento-San Joaquin Basin from late winter through early summer (Hanni and Blalock-Herod 2006).

The pre-spawning holding stage begins when individuals cease upstream movement in the summer, and continues until fish began their secondary migration to spawn, generally in late winter or early spring (Robinson and Bayer 2005, McCovey 2010). During this holding period, most fish remain stationary throughout the summer and fall, but some individuals undergo additional upstream movements in the winter following high flow events (Robinson and Bayer 2005, McCovey 2010). In the Sacramento River, adults, likely either in the holding or spawning stage, have been detected at Glenn-Colusa Irrigation District (GCID) from December through July and nearly year-round at RBDD (Hanni and Blalock-Herod 2006). It is expected that adult Pacific Lamprey with varying levels of sexual maturity are present in the Sacramento-San Joaquin Basin throughout the year.

After the pre-spawning holding period, individuals undergo a secondary migration from holding areas to spawning areas. This migration generally begins in late winter and continues through July, by which time most individuals have spawned and died (Robinson and Bayer 2005, Stillwater Sciences 2010, Lampman 2011). During this secondary migration, movement to spawning areas can be both upstream and downstream (Robinson and Bayer 2005, Lampman 2011).

Unlike Pacific salmon and steelhead (and like the Great Lakes Sea Lamprey; Bergstedt and Seelye 1995), Pacific Lamprey do not necessarily home to natal spawning streams (Moyle et al. 2009). Instead, migratory lampreys may select spawning locations based on the presence of a pheromone-like substance secreted by ammocoetes (Bjerselius et al. 2000, Vrieze and Sorensen 2001, Yun et al. 2011). Results of recent genetics research supports lack of homing by the Pacific Lamprey. A study of Pacific Lamprey population structure found few genetic differences among individuals sampled at widely dispersed sites across their range, indicating substantial genetic exchange among populations from different streams (Goodman et al. 2006).

9B.2.3.2 Spawning

Spawning typically takes place from March through July depending on water temperature and local conditions such as seasonal flow regimes (Kan 1975, Brumo et al. 2009, Gunckel et al. 2009). Evidence from the Santa Clara River in southern California suggests that individuals in the southern portion of the species' range can spawn as early as January, with peak spawning from February to April (Chase 2001), whereas inland and northern populations initiate spawning considerably later in the spring (Kan 1975, Beamish 1980, Brumo et al. 2009). Hannon and Deason (2007) have documented Pacific Lamprey spawning in the American River between early January and late May, with peak spawning typically occurring in early April. Spawning occurs in both the mainstem of medium-sized rivers and smaller tributaries (Luzier et al. 2006, Brumo et al. 2009, Gunckel et al. 2009), and generally takes place in pool and run tailouts and low gradient riffles. Both males and females build redds that are approximately 40-by-40 cm in area and are constructed in gravel and cobble substrate (Brumo 2006, Gunckel et al. 2009). Spawning substrate size typically ranges from approximately 25 to 90 mm (1.0 to 3.5 inches), with a median of 48 mm (1.9 inches) (Gunckel et al. 2009). Water velocity above redds ranges from 0.2 to 1.0 meters per second (m/s) (median 0.6 m/s), and depth varies from approximately 0.2 to 1.1 m (0.7 to 3.6 feet [ft]) (Gunckel et al. 2009). Depending on their size, females lay between 30,000 and 240,000 eggs (Kan 1975), which are approximately 1.4 mm (0.06 inch) in diameter (Meeuwig et al. 2004). In comparison, Chinook Salmon generally lay approximately 4,000 to 12,000 eggs (Jasper and Evensen 2006). During spawning, eggs are released in clutches of about 500 every 2 to 5 minutes (Pletcher 1963). Upon fertilization, eggs adhere to sandy substrate in the gravel redd (Pletcher 1963).

Depending on water temperature, hatching occurs in approximately 2 to 3 weeks, and yolk-sac larvae known as prolarvae remain in redd gravels for approximately 2 to 3 more weeks before emerging at night as 8-to-9-mm larvae, and drift downstream to rear in depositional areas (Meeuwig et al. 2005, Brumo 2006). Pacific Lamprey typically die soon after spawning (Kan 1975; Brumo 2006), although there is some anecdotal evidence that this is not always the case (Moyle 2002; Michael 1980; Michael 1984).

9B.2.3.3 Juvenile Rearing and Outmigration

After larvae emerge from redds drifting downstream, the eyeless, toothless larvae known as ammocoetes settle out of the water column and burrow into fine silt and sand substrate in low-velocity, depositional areas such as pools, alcoves, and side channels (Moore and Mallatt 1980, Torgensen and Close 2004, Stone and Barndt 2005). Ammocoete presence has also been shown to be associated with presence of woody debris (Roni 2003, Graham and Brun 2006). Rearing Pacific Lamprey ammocoetes appear to prefer rearing temperatures below 68°F (20 degrees Celsius [°C]) (BioAnalysts, Inc. 2000); and temperatures above 82.4°F (28°C) result in mortality of ammocoetes (van de Wetering and Ewing 1999). Depending on factors influencing their growth rates, they remain in this habitat from 4 to 10 years, filter-feeding on algae and detrital matter prior to metamorphosing into an adult form (Pletcher 1963, Moore and Mallatt 1980, Beamish and Levings 1991, van de Wetering 1998). During the ammocoete stage, individuals may periodically move and relocate in response to changing water levels, channel adjustments, or substrate movements (ULEP 1998). These factors generally result in a gradual downstream movement that may lead to higher densities in downstream reaches (Richards 1980). During metamorphosis, individuals develop eyes, a suctoral disc, sharp teeth, and more-defined fins (McGree et al. 2008). After metamorphosis, smolt-like individuals known as macropthalmia migrate to the ocean—typically in conjunction with high-flow events between fall and spring (van de Wetering 1998). Data from rotary screw trapping at sites in the Sacramento-San Joaquin Basin indicate that emigration of Pacific Lamprey macropthalmia peaks from early winter through early summer; however, some outmigration has been observed year-round in the mainstem Sacramento River at both RBDD and GCID (Hanni and Blalock-Herod 2006). When abundant, outmigrating Pacific Lamprey may act to buffer predation on juvenile and smolt salmon because they are easier to capture than salmonids (Close et al. 2002).

9B.2.3.4 Ocean Residence

In the ocean, adult Pacific Lamprey feed parasitically on a variety of marine and anadromous fishes such as salmon, flatfish, rockfish, and pollock. Pacific Lamprey are preyed upon by sharks, sea lions, and other marine animals (Richards and Beamish 1981, Beamish and Levings 1991, Close et al. 2002), and have been captured in depths from 300 to 2,600 ft and as far as 62 miles off the coast (USFWS 2007).

9B.2.4 Population Trends

In recent years, state, federal, and tribal agencies have expressed concern at the apparent decline of lamprey populations in the Northwestern United States (Close et al. 2002; Moser and Close 2003; CRBLTW 2005). Widespread anecdotal accounts of decreased Pacific Lamprey spawning and carcasses have been supported by a substantial reduction in counts of migrating individuals at dams since the late 1960s (Moser and Close 2003, Klamath-Siskiyou Wildlands Center et al. 2003). Very few data on Pacific Lamprey populations are available to assess status in the Sacramento-San Joaquin Basin; however, loss of access to historical habitat throughout California indicates that populations are greatly suppressed compared with historical levels (Moyle et al. 2009).

Factors limiting Pacific Lamprey populations are numerous and interrelated (Moser and Close 2003, Moyle et al. 2009). Although very little data or published studies are available for Pacific Lamprey in the region, parallels in their life cycle with salmon and steelhead suggest that these species are adversely affected by many of the same factors. Lack of access to historical spawning habitats because of dams, entrainment by water diversions, agricultural practices, urban development, harvesting, mining, transportation, estuary modification, prey abundance, and nonnative invasive species have all been cited as important anthropogenic factors limiting the viability of Pacific Lamprey populations in California (Moyle et al. 2009). In the Delta, the impacts of agricultural practices, development, estuary modification, and predation by nonnative species are expected to be particularly pronounced.

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9B.3 Green Sturgeon (*Acipenser medirostris*)

9B.3.1 Legal Status

Federal: Threatened, Designated Critical Habitat
State: Species of Special Concern

The National Marine Fisheries Service (NMFS) has divided North American Green Sturgeon into two Distinct Population Segments (DPSs) using the Eel River in California as the line of demarcation (Adams et al. 2002). The Southern DPS of North American Green Sturgeon includes all coastal and Central Valley populations south of the Eel River, including the Sacramento River basin (NMFS 2006). Although the Southern DPS is considered a separate population from the Northern DPS based on genetic data and spawning locations, their ranges outside the spawning season overlap (DFG 2002, Israel et al. 2004, Moser and Lindley 2007).

After a status review was completed in 2002 (Adams et al. 2002), NMFS determined that the Southern DPS did not warrant listing as threatened or endangered but should be identified as a Species of Concern. This determination was challenged in April 2003, and NMFS was asked to consider new information on the species. NMFS updated its status review in February 2005 and determined that the Southern DPS should be listed as threatened under the Federal Endangered Species Act (ESA) (NMFS 2005a). NMFS published a final rule (NMFS 2006) in April 2006 that listed the Southern DPS as threatened; the rule took effect on June 6, 2006.

NMFS made a final critical habitat designation for the Southern DPS in October 2009 (74 *Federal Register* [FR] 52300). Designated critical habitat in California includes the Sacramento, lower Feather, and lower Yuba rivers; the Delta; and Suisun, San Pablo, and San Francisco bays (NMFS 2014). NMFS published a final 4(d) rule to apply ESA take prohibitions to the Southern DPS in July 2010 (75 FR 30714). In California, Green Sturgeon is a Class 1 Species of Special Concern (qualifying as threatened under the California Endangered Species Act).

9B.3.2 Distribution

North American Green Sturgeon are the most wide-ranging sturgeon species, with ocean migrations ranging between northern Mexico and southern Alaska (Adams et al. 2002). Ocean abundance and densities of Green Sturgeon increase north of the Golden Gate because both the Southern DPS and Northern DPS generally migrate northward along the coast when at sea (NMFS 2005b), as confirmed by radio telemetry studies conducted on Sacramento River Green Sturgeon (DFG 2002). Subadult and adult Green Sturgeon migrate thousands of miles along the western coast of the United States, often venturing into coastal estuaries like Willapa Bay and Grays Harbor in Washington, where they concentrate during summer (Adams et al. 2002). Two adults tagged in Willapa Bay have been detected by radio telemetry stations in the Sacramento River (Heublein et al. 2009), indicating that Green Sturgeon from the Sacramento River migrate as far north as Washington before returning to the Sacramento River to spawn. Concentrations of Green Sturgeon have also been detected near Vancouver Island in Canada (NMFS 2005b).

Though Green Sturgeon migrate thousands of miles through rivers, estuaries, and ocean, they do not readily establish new spawning populations; they are known from only three river systems: the Sacramento, Rogue, and Klamath. However, data suggest there may be spawning populations in both the Eel River and the Umpqua River in Oregon (NMFS 2005b), which could indicate previously undetected relict populations or the seeds of new subpopulations. The population that spawns in the Sacramento River constitutes the only known spawning population in the Southern DPS. Populations may have formerly spawned in the San Joaquin and South Fork Trinity rivers, but have since been extirpated (Israel and Klimley 2008).

Green Sturgeon juveniles, subadults, and adults are widely distributed in the Sacramento-San Joaquin Delta and estuary areas including San Pablo Bay (Beamesderfer et al. 2004). The Sacramento-San Joaquin Delta serves as a migratory corridor, feeding area, and juvenile rearing area for North American Green Sturgeon in the Southern DPS.

9B.3.2.1 Current Distribution in Sacramento River

Within the Sacramento River, data only support an approximation of spawning locations. Larval Green Sturgeon have been captured routinely, but in small numbers in the RBDD rotary screw traps (River Mile [RM] 243.5) and the GCID fish facility (RM 206), suggesting that spawning generally occurs upstream of Hamilton City (RM 199), though spawning may occur as far downstream as Chico Landing (RM 194) (Heublein et al. 2009). Adult Green Sturgeon have been observed congregating below RBDD during late spring and early summer when the gates are down (Beamesderfer et al. 2004), suggesting that these may be ripe adults trying to migrate upstream to spawn. Spawning may occur in reaches upstream of RBDD (DFG 2002), but the upstream extent of spawning is unknown. In 1999, USFWS placed egg mats in the Sacramento River from Anderson Cottonwood Irrigation District (ACID) Dam (RM 298.4) to 10 miles downstream of RBDD to identify Green Sturgeon spawning sites; however, only two eggs were captured, both at mats downstream of RBDD, so the study did not clarify the location of specific spawning sites or the upstream extent of spawning (Beamesderfer et al. 2004). A radio telemetry study detected two adult Green Sturgeon migrating past a remote monitoring station above RBDD, suggesting possible spawning migration upstream (Heublein et al. 2009).

9B.3.2.2 Historical Distribution in Sacramento River

The location and character of spawning sites in the Rogue and Klamath rivers suggest that Green Sturgeon spawned in the Sacramento River above Keswick Dam (RM 302), including in the Pit, McCloud, and Little Sacramento rivers (Nakamoto et al. 1995, NMFS 2005b). The timing of upstream migration (February through July) corresponds with winter base and high flows and spring snowmelt. Adult Green Sturgeon likely entered the Sacramento River during winter, holding in pools in the middle and upper Sacramento River until high-flow events triggered upstream migration; high flows would have allowed adults to navigate through areas that might otherwise act as passage barriers at lower flows, providing them with access to steeper reaches with higher-velocity flows and coarser substrates for broadcast spawning. Such areas may have resulted in higher egg survival—crevices between substrate particles would provide the Green Sturgeon's relatively non-adhesive eggs to settle in areas less accessible to egg predators.

The location and characteristics of preferred Green Sturgeon spawning habitats in the Rogue and Klamath rivers suggest that most of the historical spawning habitat in the Sacramento River likely occurred upstream of Keswick Dam (RM 302), with dam construction in the 1940s creating a permanent barrier that eliminated access to the majority of spawning habitat. Upstream passage may have been impeded even earlier by the seasonal operation of the ACID Dam, which began in 1916. Later-arriving adults would have even less access to spawning habitat because of the operation of RBDD, which blocked upstream passage when the gates were lowered in mid-May. Beginning in the late 1800s, those adults that successfully spawned upstream might have had their larvae entrained by water diversions such as the GCID diversion near Hamilton City.

9B.3.3 Life History and Habitat Requirements

Sturgeon live 40 to 50 years, delay maturation to large sizes (125 cm total length), and spawn multiple times over their lifespan. This life history strategy has been successful through normal environmental variation in the large river habitats where spawning occurs. Their long lifespan, repeat spawning in multiple years, and high fecundity allow them to persist through periodic droughts and environmental catastrophes. The high fecundity associated with large size allows them to produce large numbers of offspring when suitable spawning conditions occur and compensate for years of poor reproductive and juvenile rearing conditions. Adult Green Sturgeon do not spawn every year, and only a fraction of the population enters fresh water where they might be at risk of a catastrophic event (Beamesderfer et al. 2007). Though there are general descriptions of preferred habitat conditions for Green Sturgeon, much of this information is derived from Rogue River and Klamath River data, and little is known about specific spawning, rearing, or holding locations in the Sacramento River.

9B.3.3.1 Adult Migration

Though Green Sturgeon spend most of their life in marine and estuarine environments, they periodically migrate into freshwater streams to spawn, spending up to 6 months in fresh water during their spawning migration. Upstream migration generally begins in February and may last until late July (Adams et al. 2002). In the Rogue River, telemetry studies have shown that adult Green Sturgeon hold in low-velocity, deep-water habitats prior to migrating upstream to spawn (Erickson et al. 2002). The adults move around in the pools and may stray short distances, but the scope of their movement is limited. In the Sacramento River, adult Green Sturgeon begin their upstream spawning migrations into the San Francisco Bay in March and reach Knights Landing on the Sacramento River during April (Heublein et al. 2006).

9B.3.3.2 Spawning

Spawning occurs between March and July, peaking between mid-April and mid-June (Emmett et al. 1991). Based on the distribution of sturgeon eggs, larvae, and juveniles in the Sacramento River, DFG (2002) indicated that Green Sturgeon spawn in late spring and early summer above Hamilton City, possibly up to Keswick Dam (Brown 2007). Israel and Klimley (2008) state that Green Sturgeon spawn in the mainstem from the confluence of Battle Creek (river kilometer 438) to the area upstream of Molinos, but may also spawn below RBDD closer to GCID in some years. Adults spawn within about a week, and females appear to spawn regardless of habitat conditions (Beamesderfer et al. 2007).

Green Sturgeon prefer areas of fast, deep, turbulent water in mainstem channels for spawning (Moyle 2002). They spawn in a variety of substrates, from clean sand to bedrock, but prefer bed surfaces composed of coarse cobble (Moyle 2002). In the Rogue River, suspected spawning sites (inferred from the movement of radio-tagged Green Sturgeon) have beds composed of cobbles and boulders, with water depths greater than 10 to 15 feet (3 to 4.6 meters) and turbulent water over slope breaks in the channel (Wildlife Conservation Society 2005). The interstitial spaces between large particles may provide eggs with cover from predation (Moyle 2002). Eggs and larvae require cool water temperatures and high dissolved oxygen concentrations while digesting their yolk sac (Van Eenennaam et al. 2005).

Female Green Sturgeon produce 59,000 to 242,000 eggs, about 4.34 mm in diameter (Van Eenennaam et al. 2001, 2006). Green Sturgeon eggs have the largest mean diameter of any sturgeon species (Cech et al. 2000), but they lay fewer eggs. The larger eggs may allow embryos to grow larger before hatching and emerging from cover, increasing their survival relative to other sturgeon species. Fecundity peaks at around age 24 years (Beamesderfer et al. 2007).

9B.3.3.3 Juvenile Rearing

Hatchling Green Sturgeon embryos seek nearby cover and remain under rocks (Deng et al. 2002). After about 6 to 9 days, the hatchings develop into larvae and initiate exogenous foraging on the benthos (Deng et al. 2002, Kynard et al. 2005). After a day or so, larvae disperse downstream for 1 to 2 weeks. Movements and foraging activity during this period are nocturnal (Cech et al. 2000, Kynard et al. 2005). Larval Green Sturgeon are regularly captured during this dispersal stage at about 2 weeks old (24- to 34-mm fork length) in rotary screw traps at RBDD (DFG 2002, USFWS 2002) and 3 weeks old when captured farther downstream at the GCID fish facility (Van Eenennaam et al. 2001). Following emergence in early summer, larval Green Sturgeon migrating downstream with snowmelt flows between May and July, growing quickly and becoming more tolerant of increasing water temperatures and salinities. The upper thermal limit for optimal development and hatching is between 17 to 18°C; temperatures higher than this may affect development and hatching success, and complete mortality occurs at temperatures above 23°C (Van Eenennaam et al. 2005).

Young Green Sturgeon appear to rear for the first 1 to 2 months in the Sacramento River between Keswick Dam and Hamilton City (DFG 2002). Larvae and post-larvae are present in the lower Sacramento River and North Delta between May and October, primarily in June and July (DFG 2002). Little is known of distribution and movements of young-of-the-year and riverine juveniles, but observations suggest they may be distributed primarily in the mainstem Sacramento River downstream of Anderson and in the brackish portions of the north and interior Delta (Israel and Klimley 2008). Juvenile Green Sturgeon have been captured in the Delta during all months of the year (Borthwick et al. 1999, DFG 2002). Catches of 1- and 2-year-old Southern DPS Green Sturgeon on the shoals in the lower San Joaquin River, at the CVP/SWP fish salvage facilities, and in Suisun and San Pablo bays indicate that some fish rear in the estuary for at least

2 years (DFG 2002). Larger juvenile and subadult Green Sturgeon occur throughout the estuary, possibly temporarily, after spending time in the ocean (DFG 2002, Kelly et al. 2007).

The rearing habitat preferences of Green Sturgeon larvae and juveniles in the Sacramento River are not well understood. Laboratory research has identified water temperature thresholds for larval Green Sturgeon. Water temperatures above 68°F (20°C) were found to be lethal to Green Sturgeon embryos by Cech et al. (2000), and temperatures above 63 to 64°F (17 to 18°C) were found to be stressful by Van Eenennaam et al. (2005). Cech et al. (2000) found that optimal growth of larvae occurred at 59°F (15°C), with growth slowing at temperatures below 52°F (11°C) and above 62°F (19°C).

Several studies suggest that juvenile Green Sturgeon rear in fresh water for 1 to 4 years, acclimating gradually to brackish environments before migrating to the ocean (Beamesderfer and Webb 2002, Nakamoto et al. 1995). Larval Green Sturgeon are captured at RBDD and the GCID fish facility between May and August, with peak capture at RBDD in June and July and at the GCID fish facility in July (Adams et al. 2002). Green Sturgeon larvae trapped at RBDD average 1.1 inches (2.9 cm) in length, while larvae trapped at the GCID fish facility average 1.4 inches (3.6 cm) (Adams et al. 2002), suggesting that larvae move downstream soon after hatching; however, it is not clear how long larval and juvenile Green Sturgeon remain in the middle Sacramento River. Larval Green Sturgeon grow quickly, reaching 2.9 inches (74 mm) by the time they become juveniles at around 45 days posthatching (Deng 2000). Klamath River studies indicate that juvenile Green Sturgeon can grow to 12 inches (30 cm) in their first year and 24 inches (60 cm) within 2 to 3 years (Nakamoto et al. 1995). The small size of salvaged juvenile Green Sturgeon at the CVP and SWP fish facilities indicates that they move downstream to rear in the Bay-Delta estuary (Adams et al. 2002), though it is unclear how long they remain before migrating to the ocean.

While in the riverine environment, juveniles occupy low-light habitat and are active at night (Kynard et al. 2005). Older juveniles may be adapted to move through habitats with variable gradients of salinity, temperature, and dissolved oxygen (Kelly et al. 2007, Moser and Lindley 2007). Their diet during their Sacramento River residence is unknown, but likely consists of drifting and benthic aquatic macroinvertebrates (Israel and Klimley 2008).

Stomach contents from adult and juvenile Green Sturgeon captured in the Sacramento-San Joaquin Delta included shrimp, mollusks, amphipods, and small fish (Radtke 1966, Houston 1988, Moyle et al. 1992). Stomachs of Green Sturgeon caught in Suisun Bay contained *Corophium* sp. (amphipod), *Cragon franciscorum* (bay shrimp), *Neomysis awatchensis* (Opossum shrimp: synonymous with *Neomysis mercedis*), and annelid worms (Ganssle 1966). Stomachs of Green Sturgeon caught in San Pablo Bay contained *C. franciscorum*, *Macoma* sp. (clam), *Photis californica* (amphipod), *Corophium* sp., *Synidotea laticauda* (isopod), and unidentified crab and fish (Ganssle 1966). Stomachs of Green Sturgeon caught in the Delta contained *Corophium* sp. and *N. awatchensis*

(Radtke 1966). As a result of recent changes in the species composition of macroinvertebrates inhabiting the Bay-Delta estuary due to nonnative species introductions, the current diet of Green Sturgeon is likely to differ from that reported in the 1960s.

In the Rogue River, adults hold in deep pools after spawning until late fall or early winter, when they emigrate to downstream estuaries or the ocean, perhaps cued by winter freshets that cause water temperatures to drop (Erickson et al. 2002). Erickson et al. (2002) noted that adult downstream migration appeared correlated with water temperatures below 50°F (10°C).

9B.3.3.4 Ocean Residence

Green Sturgeon from the Southern DPS pass through the San Francisco Bay to the ocean where they commingle with other sturgeon populations (DFG 2002). Subadult and adult sturgeon tagged in San Pablo Bay oversummer in bays and estuaries along the coast of California, Oregon, and Washington, between Monterey Bay and Willapa Bay, before moving farther north in the fall to overwinter north of Vancouver Island. Individual Southern DPS Green Sturgeon tagged by DFW in the San Francisco estuary have been recaptured off Santa Cruz, California; in Winchester Bay on the southern Oregon coast; at the mouth of the Columbia River; and in Grays Harbor, Washington (USFWS 1993, Moyle 2002). Most Southern DPS Green Sturgeon tagged in the San Francisco estuary have been returned from outside that estuary (Moyle 2002).

Subadult and adult Green Sturgeon generally migrate north along the coast once they reach the ocean, concentrating in coastal estuaries like Willapa Bay, Grays Harbor, and the Columbia River estuary during summer (Adams et al. 2002). The strategy underlying summer visits to coastal estuaries is unclear because sampling indicates they have relatively empty stomachs, suggesting they may not be entering the estuaries to feed (Beamesderfer 2000). Females reach sexual maturity after about 17 years and males after about 15 years (Adams et al. 2002). Spawning was believed to occur every 3 to 5 years (Tracy 1990), but may occur as frequently as every 2 years (NMFS 2005a).

9B.3.4 Population Trends

Empirical estimates of Green Sturgeon abundance are not available for any west coast population including the Sacramento River population. Interpretations of available time series of abundance index data for Green Sturgeon are confounded by small sample sizes, intermittent reporting, fishery-dependent data, lack of directed sampling, subsamples representing only a portion of the population, and potential confusion with White Sturgeon (Adams et al. 2002). Musick et al. (2000) noted that the North American Green Sturgeon population has declined by 88 percent throughout much of its range. The current population status of Southern DPS Green Sturgeon is unknown (Beamesderfer et al. 2007, Adams et al. 2007). Based on captures of Green Sturgeon during surveys for White Sturgeon in San Francisco Bay (USFWS 1995), the population is believed to range from several hundred to a few thousand adults.

Population estimates of Green Sturgeon in the Sacramento River have been derived from data collected by monitoring programs that generally focus on other species because few monitoring programs specifically address Green Sturgeon in the Sacramento River. Green Sturgeon larvae are captured annually in the RBDD rotary screw traps, the GCID fish screen, and the CVP/SWP fish salvage facilities in the South Delta. DFW conducts annual trammel net surveys in San Pablo Bay to track the White Sturgeon population, and Green Sturgeon often form part of the incidental catch. Eggs, larvae, and post-larval Green Sturgeon are now commonly reported in sampling directed at Green Sturgeon and other species (Beamesderfer et al. 2004, Brown 2007). Young-of-the-year Green Sturgeon have been observed annually since the late 1980s in fish sampling efforts at RBDD and the Glenn-Colusa Canal (Beamesderfer et al. 2004). Green Sturgeon in the Sacramento River are believed to have declined over the last 2 decades, with fewer than 50 spawning adults observed annually in the best spawning habitat along the middle section of the Sacramento River (Israel and Klimley 2008).

Similar to other anadromous fish, Green Sturgeon in the Sacramento River likely exhibit seasonal behavioral patterns in response to changes in flows, water temperature, or other environmental cues affected by flows, but it is not clear if anthropogenically induced changes in the flow regime have contributed to the apparent decline in Green Sturgeon spawners. Researchers have hypothesized that high spring flows, or the turbidity associated with them, may act as an upstream migration cue. The annual catch of larval sturgeon at the RBDD and GCID fish screens suggests that spawning occurs in the Sacramento River in most years, regardless of water year type; however, it is unclear how many adults return to spawn each year and whether there is a relationship between flows and the number of adult spawners in any given year. The relationship between flow and water temperature in the Sacramento River may influence Green Sturgeon through controlling the amount of suitable rearing habitat available for larvae and juveniles (Adams et al. 2002).

The most consistent sample data for Sacramento Green Sturgeon are for subadults captured in San Pablo Bay during periodic White Sturgeon assessments since 1948. The California Department of Fish and Game (now DFW) measured and identified 15,901 sturgeon of both species between 1954 and 1991 (USFWS 1996). Catches of subadult and adult North American Green Sturgeon by the Interagency Ecological Program between 1996 and 2004 ranged from 1 to 212 Green Sturgeon per year, with the highest catch in 2001. Various attempts have been made to infer Green Sturgeon abundance based on White Sturgeon mark-recapture estimates and relative numbers of White and Green Sturgeon in the catch (USFWS 1996, Moyle 2002). However, low catches of Green Sturgeon preclude estimates or indices of Green Sturgeon abundance from these data (Schaffter and Kohlhorst 1999, Gingras 2005). It is unclear if the high annual variability in length distributions in these samples reflects variable recruitment and abundance or is an artifact of small sample sizes, pooling of sample years, or variable distribution patterns between freshwater and ocean portions of the population.

Anecdotal information is also available on young-of-the-year Green Sturgeon from juvenile fish monitoring efforts at RBDD and the GCID pumping facility on the upper Sacramento River. Fish traps at these facilities captured between 0 and 2,068 juvenile Green Sturgeon per year (Adams et al. 2002), which suggests that at least some Green Sturgeon reproduction occurred during the 1990s.

Approximately 3,000 juvenile Green Sturgeon have been observed in rotary screw traps operated for juvenile salmon at RBDD from 1994 to 2000. Annual catches have declined from 1995 through 2000 although the relationship of these catches to actual abundance is unknown. Recent data indicate that little production occurred in 2007 and 2008 (13 and 3 larvae, respectively, were captured in the rotary screw traps at RBDD) (Poytress et al. 2009). Larger production occurred in 2009, 2010, and 2011 (45, 122, and 643 larvae, respectively, were captured using a benthic D-net), and no larvae were captured in 2012 (Poytress et al. 2010, 2011, 2012, 2013).

More than 2,000 juvenile Green Sturgeon have been collected in fyke and rotary screw traps operated at the GCID diversion from 1986 to 2003. Operation of the screw trap at the GCID site began in 1991 and has continued year-round with the exception of 1998. Juvenile Green Sturgeon at the GCID site were consistently larger in average size, but the number captured varied widely with no apparent patterns in abundance between the two sites. Abundance of juveniles peaked during June and July with a slightly earlier peak at RBDD (Adams et al. 2002).

Variable numbers of juvenile Green Sturgeon are observed each year from two south Delta water diversion facilities (DFG 2002). When water is exported through the CVP/SWP export facilities, fish become entrained into the diversion. Since 1957, Reclamation has salvaged fish at the CVP Tracy Fish Collection Facility. DFW's Fish Facilities Unit, in cooperation with DWR, began salvaging fish at the SWP Skinner Delta Fish Protective Facility in 1968. The salvaged fish are trucked daily and released at several sites in the western Delta. Salvage of fish at both facilities is conducted 24 hours a day, 7 days a week, at regular intervals. Entrained fish are subsampled for species composition and numbers. Numbers of Green Sturgeon observed at these fish facilities have declined since the 1980s, which contributed to NMFS' decision to list the Southern DPS as a threatened species. From the SWP Skinner Fish Facility, Green Sturgeon counts averaged 87 individuals per year between 1981 and 2000 and 20 individuals per year from 2001 through 2007. From the CVP Tracy Fish Collection Facility, Green Sturgeon counts averaged 246 individuals per year between 1981 and 2000 and 53 individuals per year from 2001 through 2007 (Reclamation 2008). Patterns were similar between total numbers per year and numbers adjusted for water export volumes, which increased during the 1970s and 1980s. Annual counts of Green Sturgeon from the SWP and CVP fish facilities are not significantly correlated (Beamesderfer 2005).

USFWS (1996) reported substantial uncertainty in the interpretation of salvage data for Green Sturgeon because of poor quality control on both counts and species identification, expansions from small sample sizes, variability in sturgeon dispersal patterns and collection vulnerability in response to complex changes in

Delta flow dynamics, and changes in configuration and operations over time. Estimated sturgeon salvage numbers are expanded from subsamples, and actual numbers of Green Sturgeon observed are substantially smaller. Historical expansions were based on variable expansion rates (subsample duration) ranging from 15 seconds per 2 hours when fish numbers were high to 100 percent counting during periods when fish numbers were low. Under current conditions, NMFS (2004) requires sampling of fish salvage at both the SWP and CVP facilities at intervals of no less than 10 minutes every 2 hours. Green Sturgeon salvage estimates reported for years before 1993 may be in error because of uncertainty whether smaller sturgeon were correctly identified (USFWS 1996, DFG 2002). Reclamation and DWR recommended that only more recent (from 1993 and later) CVP and SWP salvage data be used to analyze the effects of water project operations on Green Sturgeon and other anadromous fishes.

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9B.4 White Sturgeon (*Acipenser transmontanus*)

9B.4.1 Legal Status

Federal: None
State: None

9B.4.2 Distribution

White Sturgeon have a marine distribution spanning from the Gulf of Alaska south to Mexico, but a spawning distribution ranging only from the Sacramento River northward. Currently, self-sustaining spawning populations are only known to occur in the Sacramento, Fraser, and Columbia rivers.

In California, the largest numbers are in the San Francisco Bay estuary, with spawning occurring mainly in the Sacramento and Feather rivers. White Sturgeon historically ranged into upper portions of the Sacramento system including the Pit River, and a substantial number were trapped in and above Lake Shasta when Shasta Dam was closed in 1944 and successfully reproduced until the early 1960s (State Water Contractors 2004). They may have occurred historically in the San Joaquin River based on habitat similarities with these other watersheds.

Adult sturgeon were caught in the sport fishery industry in the San Joaquin River between Mossdale and the confluence with the Merced River in late winter and early spring, suggesting this was a spawning run (Kohlhorst 1976). Kohlhorst et al. (1991) estimated that approximately 10 percent of the Sacramento River system spawning population migrated up the San Joaquin River. Spawning may occur in the San Joaquin River when flows and water quality permit; however, no evidence of spawning is present (Kohlhorst1976, Kohlhorst et al. 1991).

Landlocked populations are located above major dams in the Columbia River basin, and residual non-reproducing fish above the Shasta Dam and Friant Dam have been occasionally found.

Adult White Sturgeon are occasionally noted in the San Joaquin River during DFW fall midwater trawls, DFW summer townet surveys, and University of California Davis Suisun Marsh fisheries monitoring. White Sturgeon spawning has recently been confirmed in the lower San Joaquin River (Jackson and Van Eenennaam 2013), and the U.S. Geological Survey (USGS) is currently mapping and characterizing White Sturgeon spawning habitat in the lower portion of the river (USGS 2015).

9B.4.3 Life History and Habitat Requirements

White Sturgeon are long-lived and have a high fecundity, which coupled with successful management has led to a relatively stable population within the Sacramento-San Joaquin Estuary (Moyle 2002). Because White Sturgeon require a long time to mature, however, large year classes are typically associated with years of high outflow (Kohlhorst et al. 1991, Schaffter and Kohlhorst 1999), and population size can fluctuate to extremes (Schaffter and Kohlhorst 1999).

Reports of maximum size and age of White Sturgeon are as great as 6 meters fork length (FL) (820 kilograms) and greater than 100 years, although they generally do not exceed 2 meters FL or 27 years of age. Males mature in 10 to 12 years (75 to 105 centimeters FL) and females in 12 to 16 years (95 to 135 centimeters FL). Maturation depends largely on temperature and photoperiod.

9B.4.3.1 Adult Migrations and Spawning

White Sturgeon migrate upstream in late winter. Upstream migration is usually initiated by a large pulse flow (Schaffter 1997), and not all adults will spawn each year. Because of this, successful year classes tend to occur at irregular intervals, and therefore numbers of adult fish within a population can fluctuate significantly. Although males may spawn each year, females usually spawn once every 2 to 4 years. White Sturgeon have high fecundities, and typical females may have as many as 200,000 eggs. Spawning occurs over deep gravel riffles or in deep pools with swift currents and rock bottoms between late February and early June when temperatures are between 8°C and 19°C. Eggs become adhesive subsequent to fertilization, and adhere to the substrate until they hatch 4 to 12 days later, depending on temperature. Once the eggs have been deposited, the adults move back downstream to the estuary. Larvae hatch in 1 to 2 weeks, depending on temperature. Once the yolk sac is absorbed (approximately 1 week after hatching), the larvae can begin to actively forage along the benthos.

In the Sacramento River, most White Sturgeon spawn downstream of the Glenn-Colusa Irrigation Dam.

9B.4.3.2 Juvenile Rearing

White Sturgeon are benthic feeders, and adults may move into food-rich areas to forage. Juveniles consume mainly crustaceans, especially amphipods and opossum shrimp. Adult diets include invertebrates (mainly clams, crabs, and shrimp), as well as fish, especially herring, anchovy, Striped Bass, and smelt. White Sturgeon are opportunistic predators and may feed on many introduced species.

Juvenile sturgeon are often found in upper reaches of estuaries in comparison to adults, which suggests that there is a correlation between size and salinity tolerance.

9B.4.3.3 Estuary and Ocean Residence

White Sturgeon primarily live in brackish portions of estuaries where they tend to concentrate in deep sections having soft substrate. They move according to salinity changes, and may swim into intertidal zones to feed at high tide.

Recent stomach content analysis of White Sturgeon from the San Francisco Bay estuary indicates that the invasive overbite clam, *Corbula amurensis*, may now be a major component of the White Sturgeon diet and possibly Green Sturgeon diet, and unopened clams were often observed throughout the alimentary canal (Kogut 2008). Kogut's study found that at least 91 percent of clams that passed through sturgeon digestive tracts were alive. Green Sturgeon could be affected in a similar manner. This suggests sturgeon are potential vehicles for transport of adult overbite clams and also raise concern about the effect of this invasive clam on sturgeon nutrition and contaminant exposure.

In the ocean, White Sturgeon have been known to migrate long distances, but spend most of their life in brackish portions of large river estuaries.

9B.4.4 Population Trends

Peak catches of both Green and White Sturgeon in the Sacramento River prior to 1985 were generally correlated with high flows. NMFS (2005) noted the relationships between flow and apparent White Sturgeon spawning success and inferred that low flow rates might affect Green Sturgeon in a similar manner. Periodic high flows in the 1990s produced small increases in White Sturgeon salvage catches, but salvage numbers were much lower than prior to 1985.

USFWS (1996) in the *Sacramento/San Joaquin Delta Native Fishes Recovery Plan* also reported that juvenile sturgeon are probably more vulnerable to entrainment at the SWP and CVP at low to intermediate flows during those years when river and Delta inflow are normal or below normal.

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9B.5 Chinook Salmon (*Oncorhynchus tshawytscha*)

9B.5.1 Introduction

The Sacramento-San Joaquin Delta functions as a migration corridor and potential rearing area for adult and juvenile Chinook Salmon in the Sacramento and San Joaquin River basins. The Sacramento River basin supports four runs of Chinook Salmon: winter-run, spring-run, fall-run, and late fall-run. The San Joaquin River basin currently supports fall-run (and possibly late fall-run) Chinook Salmon in its lower tributaries: the Merced, Tuolumne, and Stanislaus rivers. The winter-run consists of a single population spawning in the Sacramento River mainstem below Keswick Dam. The other runs consist of populations that spawn in multiple tributaries. Three ESUs of Chinook Salmon are represented in the combined basins: Sacramento River winter-run (federally listed as endangered), Sacramento River spring-run (federally listed as threatened), and Central Valley fall-run and late fall-run (species of concern). Each of these runs exhibits a variety of different life-history strategies.

9B.5.2 Chinook Salmon Habitat Requirements

The Sacramento River basin is the largest watershed in California (about 27,000 mi^2) and empties into the largest estuary on the west coast of the United States. This diverse basin is unique in that it supports four runs of Chinook Salmon, including the winter-run, which only occurs in the Sacramento River basin. Because the four runs exhibit a variety of different life-history strategies, anthropogenic activities in the basin have affected each of the runs differently. The habitat requirements and the life-history strategies of the four runs are discussed below.

9B.5.2.1 Upstream Migration and Holding

Adult Chinook Salmon require water deeper than 0.8 ft (24 cm) and water velocities less than 8 ft/s (2.4 m/s) for successful upstream migration (Thompson 1972). Adult Chinook Salmon appear to be less capable of negotiating fish ladders, culverts, and waterfalls during upstream migration than Coho Salmon or steelhead (Nicholas and Hankin 1989), due in part to slower swimming speeds and inferior jumping ability compared to steelhead (Reiser and Peacock 1985, Bell 1986). The maximum jumping height for Chinook Salmon has been calculated to be approximately 7.9 ft (2.4 m) (Bjornn and Reiser 1991).

Both winter-run and spring-run Chinook Salmon return to the Sacramento River when reproductively immature, typically holding for a few months in deep pools near spawning areas until spawning. Adult winter-run and spring-run Chinook Salmon require large, deep pools with flowing water for summer holding, tending to hold in pools with depths greater than 4.9 ft (greater than 1.5 m) that contain cover from undercut banks, overhanging vegetation, boulders, or woody debris (Lindsay et al. 1986), and have water velocities ranging from 0.5 to 1.2 ft/s (15 to 37 cm/s) (Marcotte 1984). Water temperatures for adult Chinook holding are reportedly best when less than 60.8°F (less than 16°C), and lethal when greater than 80.6°F (greater than 27°C) (Moyle et al. 1995). Spring-run Chinook Salmon in the Sacramento River system typically hold in pools below 69.8 to 77°F (21 to 25°C).

In general, adult Chinook Salmon appear capable of migrating upstream under a wide range of temperatures. Bell (1986) reported that salmon and steelhead migrate upstream in water temperatures that range from 3 to 20ºC (37 to 68°F). Bell (1986) reports that temperatures ranging from 3 to 13ºC (37 to 55°F) are suitable for upstream migration of spring-run Chinook Salmon, and 10 to 19ºC (50 to 66°F) is suitable for upstream migration of fall-run Chinook Salmon. In a review of available literature, Marine (1992) reported a water temperature range of 6 to 14ºC (43 to 57°F) as optimal for pre-spawning broodstock survival, maturation, and spawning for adult Chinook Salmon.

9B.5.2.2 Spawning

Most Chinook Salmon spawn in larger rivers or tributaries, although spawning has been observed in streams as small as 7 to 10 ft (2 to 3 m) wide (Vronskiy 1972). Chinook Salmon typically spawn in low- to moderate-gradient reaches of streams, but can navigate shorter reaches with steeper gradients to access suitable spawning areas. Armantrout (ULEP 1998) concluded that Chinook Salmon seldom inhabit streams with gradients greater than 3 percent after examining extensive inventory data from Oregon. The upper extent of Chinook Salmon distribution in the Umpqua River basin in Oregon appears to occur where gradients are less than 3 percent (ULEP 1998).

Upon arrival at the spawning grounds, adult females dig shallow depressions or pits (redds) in suitably sized gravels (discussed in further detail below), deposit eggs in the bottom during the act of spawning, and cover them with additional gravel. Over a period of one to several days, the female gradually enlarges the redd by digging additional pits in an upstream direction (Burner 1951). Redd areas vary considerably depending on female size, substrate size, and water velocities, and can range from 5.4 (Neilson and Banford 1983) to 482 ft^2 (0.5 to 44.8 m^2) (Chapman et al. 1986).

Chinook Salmon tend to seek spawning sites with high rates of intergravel flow. Upwelling, which is associated with a concave bed profile, may be an important feature selected by spawning Chinook Salmon (Vaux 1968).

Chinook Salmon are capable of spawning within a wide range of water depths and velocities, provided that intergravel flow is adequate for delivering sufficient oxygen to eggs and alevins (Healey 1991). Depths most often recorded for Chinook Salmon redds range from 4 to 80 inches (10 to 200 cm) (Burner 1951, Chambers et al. 1955, Vronskiy 1972), and velocities range from 0.5 to 3.3 ft/s (15 to 100 cm/s) (Burner 1951, Chambers et al. 1955, Thompson 1972, Vronskiy 1972, Smith 1973), although values may vary between races and stream basins. Fall-run Chinook Salmon, for instance, are able to spawn in deeper water with higher velocities such as the mainstem Sacramento River because of their larger size (Hallock et al. 1957).

Substrate particle size composition has been shown to have a significant influence on intragravel flow dynamics (Platts et al. 1979). Chinook Salmon may therefore have evolved to select redd sites with specific particle size criteria that will ensure adequate delivery of dissolved oxygen to their incubating eggs and developing alevins. In addition, salmon are limited by the size of substrate that they can physically move during the redd building process. Substrates selected likely reflect a balance between water depth and velocity, substrate composition and angularity, and fish size. As depth, velocity, and fish size increase, Chinook Salmon are able to displace larger substrate particles. D50 values (the median diameter of substrate particles found within a redd) for spring-run Chinook have been found to range from 10.8 to 78.0 mm (0.43 to 3.12 inches) (Platts et al. 1979; Chambers et al. 1954, 1955).

In 1997, USFWS researchers collected data on substrate particle size, velocity, and depth at hundreds of Chinook Salmon redds in the Sacramento River between Keswick Dam and Battle Creek to develop habitat suitability criteria for use in models that can aid in determining instream flows beneficial for anadromous salmonids. Redds in both shallow and deep areas were sampled. Table 9B.1 summarizes habitat suitability criteria data collected in this study for three of the four runs (too few spring-run redds were found from which to collect data). Much more detail on the methods used and results can be found in USFWS (2003).

Table 9B.1 Range of Suitable Habitat Values for Chinook Salmon Spawning in the Sacramento River (USFWS 2003)

Run	Range of Suitable Values Velocity ft/s	Range of Suitable Values Velocity m/s	Range of Suitable Values Depth ft	Range of Suitable Values Depth m	Range of Suitable Values Substrate in	Range of Suitable Values Substrate cm
Fall	0.93 to 2.66	0.28 to 0.81	1–14	0.3–4	1–3 to 3–5	3–8 to 8–13
Late fall	0.90 to 2.82	0.27 to 0.86	1–14	0.3–4	1–3 to 4–5	3–8 to 10–13
Winter	1.54 to 4.10	0.47 to 1.25	3–16	0.9–5	1–3 to 3–5	3–8 to 8–13

9B.5.2.3 Egg Incubation and Alevin Development

Once redd construction is completed, a key determinant of survival from egg incubation through fry emergence is the amount of fine sediment in the gravel (McCuddin 1977; Reiser and White 1988). High concentrations of fine sediment in (or on) a streambed can reduce permeability and intergravel flow within the redd. This can result in reduced delivery rate of oxygen and increasingly elevated metabolic waste levels around incubating eggs, larvae, and sac-fry as they develop within egg pockets (Kondolf 2000), which can in turn lead to high mortality. Several studies have correlated reduced dissolved oxygen levels with mortality, impaired or abnormal development, delayed hatching and emergence, and reduced fry size at emergence in anadromous salmonids (Wickett 1954, Alderdice et al. 1958, Coble 1961, Silver et al. 1963, McNeil 1964a, Cooper

1965, Shumway et al. 1964, Koski 1981). Silver et al. (1963) found that low dissolved oxygen concentrations are related to mortality and reduced size in Chinook Salmon and steelhead embryos. Fine sediments in the gravel interstices can also physically impede fry emergence, trapping (or entombing) them within the redd (Phillips et al. 1975, Hausle and Coble 1976).

The effects of high fine sediment concentrations may be counteracted to a certain extent by the redd construction process itself. As adult salmon build redds, they displace fine material downstream and coarsen the substrate locally (Kondolf et al. 1993, Peterson and Foote 2000, Moore et al. 2004). However, the effects of sediment reduction during redd construction may be rapidly reversed by infiltration of fine sediment into the redds during the incubation period (Kondolf et al. 1993).

Suitable water temperatures are required for proper embryo development and emergence. Incubating Chinook Salmon eggs can withstand constant temperatures between 35.1 (Combs and Burrows 1957) and 62.1°F (1.7 and 16.7°C) (USFWS 1999); however, substantial mortality may occur at the extremes. Myrick and Cech (2004) conclude that temperatures between 43 and 54°F (6 and 12°C) are best for ensuring egg and alevin survival. Sublethal stress and/or mortality of incubating eggs resulting from elevated temperatures would be expected to begin at temperatures of about 58°F (14.4°C) for constant exposures (Combs and Burrows 1957, Combs 1965, Healey 1979).

Some have suggested that the eggs and fry of winter-run Chinook Salmon may be slightly more tolerant of warm water temperatures than those of fall-run Chinook Salmon. One study by USFWS (1999) showed fall-run Chinook Salmon egg mortality increasing at lower temperatures (53.6°F [12°C]) than winter-run (56.0°F [13.3°C]). Greater tolerance to temperature was also observed in the post-hatching period, as was also found by Healey (1979). According to Myrick and Cech (2001), however, temperature tolerances of winter-run eggs and fry generally agree with those found for populations in more northern regions, and there does not appear to be much variation, if any, with regard to egg thermal tolerances between runs of Chinook Salmon (Healey 1979, Myrick and Cech 2001).

9B.5.2.4 Fry Rearing

Following emergence, fry occupy low-velocity, shallow areas near stream margins, including backwater eddies and areas associated with bank cover such as large woody debris (Lister and Genoe 1970, Everest and Chapman 1972, McCain 1992). As the fry grow, they tend to move into deeper and faster water further from banks (Hillman et al. 1987, Everest and Chapman 1972, Lister and Genoe 1970). Everest and Chapman (1972) suggests that habitat with water velocities less than 0.5 ft/s (15 cm/s) and depths less than 24 inches (60 cm) are suitable for newly emerged fry.

Although fry typically drift downstream following emergence (Healey 1991), movement upstream or into cooler tributaries following emergence has also been observed in some systems (Lindsay et al. 1986, Taylor and Larkin 1986). On the

Sacramento River, juvenile Chinook Salmon are more commonly found in association with natural banks and shaded riparian cover than banks stabilized with riprap (DFG 1983; Michny and Hampton 1984; Michny and Deibel 1986; Michny 1987, 1988, 1989; Fris and DeHaven 1993). DeHaven (1989) found this association to be weaker at lower water temperatures than at temperatures over 70°F (21°C).

9B.5.2.5 Juvenile Rearing

Little is known regarding habitat selection of juvenile Chinook Salmon in the Sacramento River system specifically. Habitat preferences of Chinook Salmon may vary depending on channel confinement, substrate and bank characteristics, abundance of small and large wood, presence of other salmonids (particularly Coho Salmon), and whether the Chinook display an ocean- or stream-type life history. Juvenile habitat use may also change seasonally, diurnally, or as a function of growth, with larger juveniles tending to occupy habitats with higher water velocities.

Several researchers have shown relationships between velocity and juvenile Chinook Salmon habitat use, with juveniles generally occupying areas with water velocities less than 15 to 30 cm/s (Thompson 1972, Hillman et al. 1987, Steward and Bjornn 1987, Murphy et al. 1989, Beechie et al. 2005), as well as a preference for areas with cover provided by brush, large wood, or undercut banks (Hillman et al. 1987, Johnson et al. 1992, Beechie et al. 2005). Lister and Genoe (1970) found that juvenile Chinook Salmon preferred "slow water adjacent to faster water (40 cm/s)," and Shirvell (1994) suggested that preferred habitat locations vary by activity. For feeding, they are likely to select positions with optimal velocity conditions, whereas for predator avoidance, optimal light conditions are more likely to be important (Shirvell 1994). At night, juvenile Chinook Salmon appear to move to quiet water or pools and settle to the bottom, returning the next day to the riffle and glide habitats they had occupied the previous day (Edmundson et al. 1968, Chelan County Public Utility District 1989).

Although some researchers have found juvenile Chinook Salmon to reside primarily in pools, they may also use glides and runs as well as riffles. Chinook Salmon may prefer deeper pools with low water velocities during spring and summer as well as during winter (Lister and Genoe 1970, Everest and Chapman 1972, Swales et al. 1986, Hillman et al. 1987). In the Elk River in Oregon, Burnett and Reeves (2001) found most juvenile ocean-type Chinook Salmon (in sympatry with Coho Salmon and steelhead) in valley segments with deeper pools, larger volume pools, and pools with greater densities of large wood. In Elk River tributaries, the juveniles were observed almost exclusively in pools. Roper et al. (1994) also found age-0+ Chinook to be strongly associated with pools in the South Umpqua River basin in Oregon. In the Sacramento and American rivers, CDFG (1997) found juvenile Chinook Salmon densities to be highest in runs, closely followed by pools, with fish also occupying riffles and glides.

9B.5.2.6 Summer Rearing

Juvenile growth rates are an important influence on survival because juvenile salmon are gape-limited predators that are themselves subject to gape-limited predation by larger fish. Thus, faster growth both increases the range of food items available to them and decreases their vulnerability to predation (Myrick and Cech 2004). Temperatures have a significant effect on juvenile Chinook Salmon growth rates. On maximum daily rations, growth rate increases with temperature to a certain point and then declines with further increases. Reduced rations can also result in reduced growth rates; therefore, declines in juvenile salmonid growth rates are a function of both temperature and food availability. Laboratory studies indicate that juvenile Chinook Salmon growth rates are highest at rearing temperatures from 65 to 70°F (18.3 to 21.1°C) in the presence of unlimited food (Clarke and Shelbourn 1985, Banks et al. 1971, Brett et al. 1982, Rich 1987), but decrease at higher temperatures. Myrick and Cech (2004) note that two studies have been published on the relationship between temperature and growth of Central Valley Chinook Salmon—one by Marine and Cech (2004) on Sacramento River fall-run Chinook Salmon, and one by Myrick and Cech (2002) on American River fall-run Chinook Salmon. Provided that food is not limited, these studies showed that optimum temperatures for growth were between 63 and 68°F (17 and 20°C). Under natural conditions, it is unlikely that Chinook Salmon will feed at 100 percent rations, and disease, competition, and predation are also factors that may affect survival. To determine temperatures that might be optimal for growth of juvenile Chinook under natural conditions, Brett et al. (1982) used a value of 60 percent rations, based on field studies that suggested fish in the wild fed at roughly 60 percent of their physiological maximum. When used in a model developed for sockeye salmon, Brett determined that juvenile Chinook Salmon would reach their optimal growth at a temperature of about 59°F (15°C) (Brett et al. 1982). Nicholas and Hankin (1989) suggest that the duration of freshwater rearing is tied to water temperatures, with juveniles remaining longer in rivers with cool water temperatures.

Temperatures of greater than 74°F (23.3°C) are considered potentially lethal to juvenile Chinook Salmon (State Water Contractors 1990). Myrick and Cech (2004) summarized available information on juvenile Chinook Salmon temperature tolerances. Incipient upper lethal temperature (IULT) studies, which may be the most biologically relevant for studying juvenile temperature tolerances, are lacking for Central Valley Chinook Salmon. Sacramento River fall-run Chinook Salmon were reared at temperatures between 70 and 75°F (21 and 24°C) by Marine and Cech (2004) without significant mortality; however, Rich (1987) observed significant mortality after only 8 days of rearing at 75°F (24°C) (Myrick and Cech 2004). Myrick and Cech (2004) suggests that, until IULT studies are conducted on Central Valley Chinook Salmon, managers use Brett's (1952) and Brett et al.'s (1982) data on more northern Chinook Salmon, which determined that the IULT is in the range of 24 to 25°C (75 to 77°F). More detail on temperature tolerances of various Chinook life stages can be found in Myrick and Cech (2001, 2004).

Chronic exposure to high temperatures may result in greater vulnerability to predation. Marine (1997) found that Sacramento River fall-run Chinook Salmon reared at the highest temperatures (21 to 24°C [70 to 75°F]) were preyed upon by Striped Bass more often than those reared at low or moderate temperatures. Consumption rates of piscivorous fish such as Sacramento pikeminnow, Striped Bass, and largemouth bass increase with temperature, which may compound the effects of high temperature on juvenile and smolt predation mortality.

9B.5.2.7 Winter Rearing

Juvenile Chinook Salmon rearing in tributaries may disperse downstream into mainstem reaches in the fall and take up residence in deep pools with LWD, in interstitial habitat provided by boulder and rubble substrates, or along river margins (Swales et al. 1986, Healey 1991, Levings and Lauzier 1991). During high flow events, juveniles have been observed to move to deeper areas in pools, and they may also move laterally in search of slow water (Shirvell 1994, Steward and Bjornn 1987). Hillman et al. (1987) found that individuals remaining in tributaries to overwinter chose areas with cover and low water velocities, such as areas along well-vegetated, undercut banks. There is very little information available on Chinook Salmon use of floodplains and off-channel habitats such as sloughs and oxbows compared to Coho Salmon. However, studies in the Sacramento and Cosumnes rivers have shown that shallow, seasonally inundated floodplains can provide suitable rearing habitat for Chinook Salmon.

In winter, juvenile Chinook Salmon may make use of the interstitial spaces between coarse substrates as cover (Bjornn 1971, Hillman et al. 1987). Hillman et al. (1987) found that the addition of cobble substrate to heavily sedimented glides in the fall substantially increased winter rearing densities, with juvenile Chinook Salmon using the interstitial spaces between the cobbles as cover. Fine sediment can act to reduce the value of gravel and cobble substrate as winter cover by filling interstitial spaces between substrate particles. This may cause juveniles to avoid these embedded areas and move elsewhere in search of suitable winter cover (Stuehrenberg 1975, Hillman et al. 1987).

Over much of the Chinook Salmon’s range, winter temperatures are too cold to allow for much growth in the winter. The low-temperature threshold for positive growth in juvenile Chinook Salmon is believed to be about 40.1°F (4.5°C), with 39.4°F (4.1°C) being the lower limit for zero net growth in a juvenile Chinook Salmon population (Armour 1990). In the Sacramento River, water temperatures rarely fall below 43°F (6°C), however, allowing for growth throughout the winter.

Within the action area, where juvenile Chinook Salmon are rearing in mainstem channels downstream of reservoirs, water temperatures rarely fall below 43°F (6°C), allowing for growth throughout the winter months. Under these conditions, habitat shifts are less related to seasonal temperature changes and more strongly affected by growth (i.e., as individuals grow, they can take advantage of habitats with stronger flow and are better able to escape predation).

In the Sacramento/San Joaquin system, some juvenile Chinook Salmon rear on seasonally inundated floodplains in the winter. Sommer et al. (2001) found higher growth and survival rates of juveniles that reared on the Yolo Bypass floodplain than in the mainstem Sacramento River, and Moyle (2000) observed similar results on the Cosumnes River floodplain. On the Yolo Bypass, bioenergetic modeling suggested that increased prey availability on the floodplain was sufficient to offset increased metabolic demands from higher water temperatures (9°F [5°C] higher than mainstem). The Yolo Bypass has a relatively smooth topography with few pits and depressions, which possibly enhances its value as floodplain rearing habitat by reducing stranding mortality as floodwaters recede and juvenile salmon return to the main stem (Sommer et al. 2001).

9B.5.2.8 Smoltification and Outmigration

Juveniles of all four runs of Chinook Salmon in the Central Valley must pass through the Sacramento-San Joaquin Delta and San Francisco Bay Estuary on their way to the ocean, and many rear there for varying periods prior to ocean entry. Williams (2012) found evidence that many naturally produced fall-run Chinook Salmon that survived to return as adults had left freshwater at lengths greater than 55 mm, while juvenile Chinook Salmon from other Central Valley runs were older and larger upon entering the estuary and likely passed through it more quickly (Williams 2012).

In many systems within the species' distribution, juvenile Chinook Salmon spend up to several months in estuaries feeding and growing before entering the ocean (Healey 1991); in productive estuaries, this strategy can result in ocean entry at a larger size with a higher chance of survival, presumably by reducing predation at this critical juncture. Although wetlands and floodplains may have been extensive enough in the Delta under historical conditions (Atwater et al. 1979) to support high juvenile production in an environment where there were fewer predators, Delta marsh habitats and native fish communities have undergone such extreme changes from historical conditions (Kimmerer et al. 2008) that few locations in the eastern and central Delta currently provide suitable habitat for rearing Chinook Salmon. For example, substantial numbers of fry may be found in the Delta from January through March, but relatively few were found in the remaining months of the year during sampling from 1977 to 1997 (Brandes and McLain 2001). The annual abundance of fry (defined as less than 2.8 inches [70 mm] fork length) in the Delta during this period appears related to flow, with the highest numbers observed in wet years (Brandes and McLain 2001).

Although growth rates of juvenile Chinook Salmon may be high at temperatures approaching 66°F (19°C), cooler temperatures may be required for Chinook Salmon to successfully complete the physiological transformation from parr to smolt. Smoltification in juvenile Sacramento River fall-run Chinook Salmon was studied by Marine (1997), who found that juveniles reared under a high temperature regime of 70 to 75°F (21 to 24°C) exhibited altered and impaired smoltification patterns relative to those reared at low 55 to 61°F (13 to 16°C) and moderate 63 to 68°F (17 to 20°C) temperatures. Some alteration and impairment of smoltification was also seen in the juveniles reared at moderate temperatures.

9B.5.3 Winter-Run Chinook Salmon

9B.5.3.1 Legal Status

Federal: Endangered, Designated Critical Habitat
State: Endangered

Although Chinook Salmon range from California's Central Valley to Alaska and the Kamchatka Peninsula in Asia, winter-run Chinook Salmon are only found in the Sacramento River. Chinook Salmon of this race are unique because they spawn during the summer months when air temperatures usually approach their yearly maximum. As a consequence, winter-run Chinook Salmon require stream reaches with cold water sources that will protect embryos and juveniles from the warm ambient conditions in the summer. Historically, high-elevation reaches of tributaries to the upper Sacramento River (e.g., McCloud River) provided the cold water reaches that supported summer spawning by winter-run Chinook Salmon. Currently, hypolimnetic releases from Shasta Lake provide the cold water temperatures that allow winter-run Chinook Salmon to persist downstream of the dam, despite the complete loss of historical spawning habitat, access to which was cut off upon completion of Shasta Dam (1963).

The California-Nevada chapter of the American Fisheries Society petitioned NMFS to list the run as a threatened species in 1985 (AFS 1985) and, following a dangerously low year-class in 1989, NMFS issued an emergency listing for Sacramento River winter-run Chinook Salmon as a threatened species (NMFS 1989); the California Fish and Game Commission listed the winter run as endangered in the same year. After several years of low escapements in the early 1990s, the status of winter-run was changed from threatened to endangered by NMFS in 1994, which was reaffirmed in 2005 and 2011 (NMFS 1994, 2005, 2011).

The ESU includes fish that are propagated as part of a conservation hatchery program managed by the USFWS at Livingston Stone National Fish Hatchery (LSNFH). Since 2000, the proportion of the ESU spawning in the Sacramento River that are of hatchery origin has generally ranged from 5 to 10 percent of the total population, but reached a high of 20 percent in 2005 (NMFS 2011). USFWS's goal is to manage the LSNFH program such that hatchery origin fish are less than 20 percent of total in-river escapement. Hatchery fish were estimated to be 12 percent of the total in-river spawners in 2010, based on carcass surveys (DFG 2010). Over the last 10 years, hatchery returns have averaged 8 percent of total escapement (NMFS 2011).

Critical habitat was designated as the Sacramento River from Keswick Dam at river mile (RM) 302 to Chipps Island (RM 0) at the westward margin of the Delta; all waters from Chipps Island westward to the Carquinez Bridge, including Honker Bay, Grizzly Bay, Suisun Bay, and the Carquinez Strait; all waters of San Pablo Bay westward of the Carquinez Bridge; and all waters of San Francisco Bay (north of the San Francisco-Oakland Bay Bridge) to the Golden Gate Bridge (NMFS 1993).

9B.5.3.1.1 Distribution

Winter-run Chinook Salmon are found only in the Sacramento River basin. The distribution of winter-run Chinook Salmon spawning has shifted over time in response to changes in upstream passage caused by water supply development and operations. Prior to construction of Shasta Dam in the 1940s, winter-run Chinook Salmon spawned in the upper Sacramento River system (in the Little Sacramento, McCloud, and possibly Pit and Fall rivers) and in nearby Battle Creek (Yoshiyama et al. 1998). Since the construction of Shasta Dam, winter-run Chinook Salmon have been limited to the mainstem Sacramento River below Keswick Dam (RM 302), although a few adults occasionally stray into tributaries (e.g., Battle and Mill creeks) to spawn (Harvey-Arrison 2001). The distribution of spawning likely shifted again in 1966, when the construction and operation of RBDD (RM 243.5) impeded access to upstream reaches, forcing more winter-run adults to spawn downstream of the diversion dam. A radio-tag survey of winter-run adults between 1979 and 1981 indicated that adults were delayed at RBDD between 1 and 40 days, with an average delay of 18 days (Hallock and Fisher 1985). The dam also forced winter-run adults to spawn downstream of Red Bluff, where summer water temperatures were frequently too high to support successful egg incubation and emergence. Beginning in 1986, the Bureau of Reclamation (Reclamation) began raising RBDD gates during the winter to facilitate upstream passage of winter-run Chinook (Reclamation 2004), which precipitated an upstream shift in the distribution of winter-run spawning. In 2012, the RBDD gates were opened to allow year-round passage.

Until 2001, most winter-run spawning occurred downstream of ACID Dam (RM 298.4); however, an improvement of this dam's fish passage facilities in 2001 allowed another upstream shift in the distribution of spawning (DFG 2002a, 2004).

9B.5.3.1.2 Life History and Habitat Requirements

General habitat requirements for Chinook Salmon are described above; the following describes life history strategies and habitat requirements unique to the winter-run or of primary importance to its life history. The winter-run Chinook Salmon's life history is unique to the Sacramento River because it provides the thermal conditions that allow for the success of this strategy. Because winter-run Chinook Salmon spawn in late spring and early summer, they require access to stream reaches with summer water temperatures cool enough to allow egg incubation. The spawning reaches and reaches downstream have sufficiently warm water temperatures to support growth throughout the winter, allowing juveniles to grow large enough to smolt and outmigrate before water temperatures become too high the following spring and summer. This life-history strategy reduces competition for spawning habitat with other runs. However, it also makes the run reliant on year-round coldwater sources, which limits the potential for expanding the range of the run in the Sacramento River basin.

Table 9B.2 illustrates life history timing for winter-run Chinook Salmon in the Sacramento River basin. Winter-run Chinook Salmon display a life history that is intermediate between ocean-type and stream-type. They spend between 5 and 10 months rearing in fresh water before migrating to sea, which is longer than for typical ocean-type Chinook Salmon, but shorter than for other stream-type Chinook Salmon (Healey 1991).

Table 9B.2 Life History Timing of Winter-run Chinook Salmon in the Sacramento River Basin

Life Stage	Jan		Feb		Mar		Apr		May		Jun		Jul		Aug		Sept		Oct		Nov		Dec	
Adult entry into San Francisco Bay[a]	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate									Moderate	Moderate	Moderate	Moderate
Migration past RBDD[b]	Moderate	Moderate	Moderate	Moderate	Peak	Peak	Moderate	Moderate	Moderate	Moderate	Low	Low	Low	Low									Low	Low
Spawning[c]								Moderate	Peak	Peak	Peak	Peak	Moderate	Moderate	Moderate									
Incubation[c]								Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate					
Fry emergence[c]												Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate					
Rearing (age 0+)	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate				Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate
Presence at CVP/SWP salvage facilities[c]	Peak	Peak	Moderate	Moderate	Moderate	Moderate	Low	Low															Low	Low
Outmigration toward and through the Delta[c]	Moderate	Moderate	Peak	Peak	Peak	Peak	Peak	Peak	Moderate	Moderate									Moderate	Moderate	Moderate	Moderate	Peak	Peak

Notes:
a. Van Woert 1958; Hallock et al. 1957
b. Hallock and Fisher 1985
c. NMFS 2012 (unpubl. data)

- Low = Period of Low Activity
- Moderate = Period of Moderate Activity
- Peak = Period of Peak Activity

9B.5.3.1.3 Adult Upstream Migration and Spawning

Adult winter-run Chinook Salmon enter San Francisco Bay from November through June (Van Woert 1958, Hallock et al. 1957). Migration past RBDD begins in mid-December and can continue into early August, but the majority of winter-run adults migrate past RBDD between January and May, with a peak in mid-March (Hallock and Fisher 1985). In recent years, upstream passage of winter-run adults at RBDD was addressed by raising the gates between September 15 and May 15, which encompasses the vast majority of the upstream migration period for winter-run Chinook Salmon. As of 2012, the gates at RBDD are open year-round to allow for upstream passage.

Like spring-run Chinook Salmon, winter-run Chinook Salmon enter spawning streams while still reproductively immature. Adults hold for a few months in deep pools near spawning areas, which provides time for gonadal development. Spawning occurs from mid-April to mid-August, peaking in May and June, in the Sacramento River reach between Keswick Dam and RBDD (Reclamation 1991). With the changes in RBDD gate operations, volitional spawning below RBDD is negligible in most years. Since fish passage improvements were completed at the ACID Dam in 2001, winter-run Chinook Salmon spawning has shifted upstream. The majority of winter-run Chinook Salmon in recent years (i.e., more than 50 percent since 2007) spawn in the area from Keswick Dam to the ACID Dam (approximately 5 miles) (NMFS 2009).

9B.5.3.1.4 Juvenile Rearing and Outmigration

Winter-run fry emerge from the spawning gravels from mid-June through mid-October (NMFS 1997). Because spawning is concentrated upstream in the reaches below Keswick Dam, the entire Sacramento River can serve as a nursery area for juveniles as they migrate downstream. 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 (Reclamation 1991, NMFS 1997). Many juveniles apparently rear in the Sacramento River below RBDD for several months before they reach the Delta (Williams 2006). From 1995 to 1999, all Sacramento River winter-run Chinook Salmon outmigrating as fry passed the RBDD by October, and all outmigrating presmolts 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, although the overall timing may extend from September to early May (NMFS 2012). The timing of migration varies somewhat because of 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 are between 5 and 10 months of age, after reaching a fork length of approximately 118 mm. Distinct emigration pulses from the Delta appear to coincide with periods of high precipitation and increased turbidity (Del Rosario et al. 2013).

The entire population of the Sacramento River winter-run Chinook Salmon passes through the Delta as migrating adults and emigrating juveniles. Because winter-run Chinook Salmon use only the Sacramento River system for spawning, adults are likely to migrate upstream primarily along the western edge of the Delta through the Sacramento River corridor. Juveniles likely use a wider area within the Delta for migration and rearing than adults; juvenile winter-run salmon have been collected at various locations in the Delta, including the SWP and CVP south Delta export facilities. 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/CVP export operations) on salmon migration through the Delta (Perry et al. 2010, 2012; Michel et al. 2013).

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.

9B.5.3.1.5 Population Trends

There is little historical data available to characterize winter-run Chinook Salmon escapements prior to the construction of Shasta Dam; indeed, the agencies did not recognize winter-run Chinook Salmon as a distinct run until the 1940s (Needham et al. 1943). In the late 1930s, the pending construction of Shasta Dam prompted the agencies to commission a study of potential salmon salvage options. As part of this investigation, researchers placed a counting weir at ACID Dam between 1937 and 1939 to estimate the size of the salmon run in the Sacramento River (Hatton 1940). The counting weir enabled scientists to estimate the run size of the fall-run Chinook Salmon populations; however, the removal of flashboards from the ACID Dam during winter prevented observations of winter-run Chinook Salmon during their period of upstream migration (December–May).

There were no direct observations of winter-run Chinook Salmon spawning in the mainstem Sacramento River between 1943 and 1946—the first years when the construction of Shasta Dam blocked upstream passage. Nevertheless, incidental observations of winter-run salmon during trap-and-haul operations for spring-run salmon, coupled with poor environmental conditions in the Sacramento River and Deer Creek, led Slater to conclude that "the winter-run populations were small" in the years when Shasta Dam was being constructed (1963).

Slater (1963) hypothesized that the winter-run salmon population began to rebound in 1947, and that "this initial recovery seems to have been both substantial and rapid" from the "low point of 1943–1946." He cites an angling survey conducted by Smith (1950), which evaluated the 1947–1948 and 1949–1950 sport fishery in the upper Sacramento River. "Increased catches of winter-run Chinook Salmon in January and February 1949" (Slater 1963) led Smith (1950) to conclude that a "sizable" winter-run population existed. Similarly, Slater cited an increase in the number of winter-run salmon that were harvested by Coleman National Fish Hatchery between 1949 and 1956 (as part of the fall-run salmon propagation program) (Azevedo and Parkhurst 1958) as evidence that winter-run salmon escapements increased in the late 1940s and early 1950s. Although these qualitative assessments do not permit a detailed tracking of winter-run salmon abundance, they do suggest a positive trend in the population in the years after Shasta Dam was completed.

This positive trend seems to have continued through the 1950s, because Hallock estimated that 11,000 winter-run adults were harvested from the Sacramento River by anglers in the winter of the 1961–1962 fishing season (Slater 1963). Hallock's estimate of the percentage of winter-run Chinook Salmon caught in the in-river recreational harvest suggests that total winter-run escapements in the winter of 1961–1962 numbered in the tens of thousands. In June 1963, Slater personally observed winter-run Chinook Salmon spawning in the vicinity of Redding in numbers that approached the fall-run population that spawned in the same sites (Slater 1963). For context, the four years before Slater's observation of winter-run spawning in 1963 (1959–1962) had fall-run salmon escapement estimates ranging from 115,500 to 250,000 salmon. Although Slater observed spawning in only a small portion of the habitat available to both winter-run and fall-run salmon in the Sacramento River, his observation suggests that the winter-run salmon population had increased substantially from the few hundred fish captured during the trap-and-haul salvage operation in 1943 and 1945. His observation also suggests that the winter-run salmon population had recovered from a probable year-class failure in 1943 and a partial year-class failure in 1944.

Beginning in 1967, agency biologists began estimating annual winter-run escapements by monitoring adults migrating through the fish passage facilities of RBDD. Although the dam facilitated a more accurate account of the winter-run population, gate operations interfered with upstream passage. Gate operations were modified beginning in winter 1986 to facilitate the upstream passage of winter-run Chinook Salmon. However, raising the dam gates rendered winter-run escapement estimates less reliable, because migrating salmon could bypass the dam's fish counting facilities.

The RBDD counts permitted agency biologists to track the decline in winter-run Chinook abundance beginning in the 1970s. The drought of 1976–1977 caused a precipitous decline in abundance between 1978 and 1979, when escapements fell below 2,500 fish. Population abundance remained very low through the mid-1990s, with adult abundance in some years less than 500 fish (DFW 2014).

Beginning in the mid-1990s and continuing through 2006, adult escapement showed a trend of increasing abundance, approaching 20,000 fish in 2005 and 2006. However, recent population estimates of winter-run Chinook Salmon spawning upstream of the RBDD have declined since the 2006 peak. The escapement estimate for 2007 through 2014 has ranged from a low of 738 adults in 2011 to a high of 5,959 adults in 2013. The escapement estimate of 738 adults in 2011 was the lowest total escapement estimate since the all-time low escapement estimate of 144 adults in 1994. Poor ocean productivity (Lindley et al. 2009), drought conditions from 2007 to 2009, and low in-river survival (National Marine Fisheries Service 2011) are suspected to have contributed to the recent decline in escapement of adult winter-run Chinook Salmon. Table 9B.3 shows winter-run Chinook Salmon natural and hatchery escapement subsequent to 2004.

1 **Table 9B.3 Recent Winter-run Chinook Salmon Natural and Hatchery Escapement**

Year	Sacramento River above RBDD	Sacramento River below RBDD	Subtotal	CNFH Transfers	LSNFH Transfers	Battle Creek	Total
Dec 1990-Aug 1991	177	0	177	33	–	–	211
Dec 1991-Aug 1992	1,159	44	1,203	34	–	–	–
Dec 1992-Aug 1993	369	9	378	–	–	–	–
Dec 1993-Aug 1994	144	0	144	42	–	–	–
Dec 1994-Aug 1995	1,159	7	1,166	43	–	88	–
Dec 1995-Aug 1996	1,012	0	1,012	–	–	325	–
Dec 1996-Aug 1997	836	0	836	–	–	44	–
Dec 1997-Aug 1998	2,831	62	2,893	–	99	–	–
Dec 1998-Aug 1999	3,264	0	3,264	–	24	–	–
Dec 1999-Aug 2000	1,261	0	1,261	–	89	2	–
Dec 2000-Aug 2001	8,085	35	8,120	–	104	–	–
Dec 2001-Aug 2002	7,325	12	7,337	–	104	–	–
Dec 2002-Aug 2003	8,105	28	8,133	–	85	–	–
Dec 2003-Aug 2004	7,784	0	7,784	–	85	–	–
Dec 2004-Aug 2005	15,730	0	15,730	36	109	0	15,875
Dec 2005-Aug 2006	17,157	48	17,205	5	93	6	17,304
Dec 2006-Aug 2007	2,487	0	2,487	1	54	0	2,542
Dec 2007-Aug 2008	2,725	0	2,725	0	105	0	2,830

Year	Sacramento River above RBDD	Sacramento River below RBDD	Subtotal	CNFH Transfers	LSNFH Transfers	Battle Creek	Total
Dec 2008-Aug 2009	4,537	0	4,537	0	121	0	4,658
Dec 2009-Aug 2010	1,533	0	1,533	0	63	0	1,596
Dec 2010-Aug 2011	738	0	738	2	86	1	827
Dec 2011-Aug 2012	2,578	0	2,578	0	93	–	2,671
Dec 2012-Aug 2013	5,920	0	5,920	0	164	–	6,084
Dec 2013-Aug 2014	2,627	0	2,627	0	388	–	3,015

Source: DFW 2014

Note:
CNFH = Coleman National Fish Hatchery

Winter-run Chinook Salmon escapement to the Sacramento River in 2011 was 827 fish, which is the smallest number since 1994 and only 10 percent of the 40-year-average of approximately 8,000 fish (Azat 2012). Unusual ocean conditions appear to have been affecting the ESU in the past 5 years, along with other Central Valley Chinook Salmon stocks (NMFS 2011). Climate change and future variations in ocean conditions, along with the many factors affecting survival during freshwater life stages, may pose a serious risk to the ESU (NMFS 2011).

9B.5.4 Central Valley Spring-Run Chinook Salmon

9B.5.4.1 Legal Status

Federal: Threatened, Designated Critical Habitat
State: Threatened

Spring-run Chinook Salmon were probably the most abundant salmonid in the Central Valley under historical conditions (Mills and Fisher 1994); however, large dams eliminated access to vast amounts of historical habitat, and the spring run has exhibited the severest declines of any of the four Chinook Salmon runs in the Sacramento River basin (Fisher 1994).

The Central Valley spring-run Chinook Salmon ESU was federally listed as threatened in 1999, and the listing was reaffirmed in 2005 when critical habitat was also designated (NMFS 1999a, 2005). Spring-run Chinook Salmon was listed as a threatened species under the California Endangered Species Act (CESA) in February 1999. 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. Feather River Hatchery spring-run Chinook Salmon are also included in the ESU. This ESU largely consists of three self-sustaining wild populations (i.e., Mill, Deer, and Butte creeks). Fish in these streams spawn outside of the action area but pass through it on their upstream and downstream migrations. Spring-run Chinook Salmon in the Feather River and Clear Creek spawn within the action area.

Designated critical habitat for Central Valley spring-run Chinook Salmon includes stream reaches of the American, Feather, Yuba, and Bear rivers; tributaries of the Sacramento River, including Big Chico, Butte, Deer, Mill, Battle, Antelope, and Clear creeks; and the main stem of the Sacramento River from Keswick Dam through the Delta. Designated critical habitat in the Delta includes portions of the Delta Cross Channel, Yolo Bypass, and portions of the network of channels in the northern Delta. Critical habitat for spring-run Chinook Salmon was not designated for the Stanislaus or San Joaquin rivers.

9B.5.4.2 Distribution

Prior to the construction of dams in the Sacramento and San Joaquin basins, spring-run Chinook Salmon migrated during the spring snowmelt flows to access coldwater holding and spawning habitat higher up in the basins. These steeper, higher-elevation reaches are often characterized by falls and cascades that may be obstacles to upstream movement of salmonids at lower flows. By migrating during the high spring snowmelt flows, spring-run Chinook Salmon can also access areas above reaches that become too warm for salmon in the summer and fall, isolating them from the fall run. Thus, under historical conditions, the spring- and fall-run Chinook Salmon were geographically isolated in terms of where they spawned in the basin, which maintained their genetic integrity.

Spring-run Chinook Salmon once occupied all major river systems in California where there was access to cool reaches that would support oversummering adults. Historically, they were widely distributed in streams of the Sacramento-San Joaquin basin, spawning and rearing over extensive areas in the upper and middle reaches (elevations ranging from 1,400 to 5,200 ft [450 to 1,600 m]) of the San Joaquin, American, Yuba, Feather, Sacramento, McCloud, and Pit rivers (Myers et al. 1998). Spring Chinook Salmon runs in the San Joaquin River were extirpated in the mid- to late 1940s following the closure of Friant Dam and diversion of water for agricultural purposes to the San Joaquin Valley.

In the Sacramento River, the closure of Shasta Dam in 1945 cut off access to the spring run's major historical spawning grounds in the McCloud, Pit, and upper Sacramento rivers. This represented a loss of 70 percent of spring-run spawning habitat in the Sacramento River basin (Yoshiyama et al. 2001). Populations of spawning spring-run Chinook Salmon in the Sacramento River basin are more common in east-side tributaries to the Sacramento River upstream of the mouth of the American River. The most important spawning populations are in Deer, Mill, and Butte creeks because of their relative lack of past hatchery influence, as well as relatively stable numbers. Some spawning also takes place in Big Chico, Antelope, Cottonwood, Beegum, Clear, and Battle Creeks, and in the mainstem Sacramento River downstream of Keswick Dam and upstream of RBDD (Association of California Water Agencies and California Urban Water Agencies 1997; DFG 1998, 2002b, 2012 [GrandTab data]). A spring run in the Feather River basin is maintained by hatchery production; however, the stock is believed to have been hybridized with the fall run to a great extent (Lindley et al. 2004).

9B.5.4.2.1 Changes in Distribution and Hybridization with Fall Chinook Salmon

Dams have reduced or eliminated spatial segregation between spawning spring- and fall-run Chinook Salmon in some areas, particularly in the mainstem Sacramento River, leading to increased potential for hybridization on the spawning grounds. The completion of Keswick and Shasta dams in the mid-1940s blocked spring-run Chinook Salmon access to habitat in the McCloud, Pit, and Little Sacramento rivers. After construction of the dams, spring-run Chinook Salmon were forced to spawn in the mainstem Sacramento River below Keswick Dam. Historically, water temperatures would have been too high in the mainstem Sacramento River for spring-run Chinook Salmon to hold in this area during the summer. But because of hypolimnetic releases from Shasta Lake, this reach provides temperatures during the summer that are now suitable for spring-run Chinook Salmon holding and spawning, where before they were only suitable for fall-run spawning once temperatures cooled in the fall. However, coldwater releases from Shasta Dam can warm relatively rapidly during the very hot days typical of the Sacramento Valley in summer and early fall. As a result, both the fall and spring runs must spawn in close enough proximity to Keswick Dam to benefit from these releases. The elimination of the spatial segregation that had existed between the fall and spring runs results in competition between the runs for the limited spawning habitat. Since fall-run Chinook Salmon spawn slightly later than spring-run, spring-run redds may also be superimposed by spawning fall-run fish. This may have contributed to the loss of the spring-run population, along with hybridization between the two runs, as described below.

The majority of spring-run Chinook Salmon used to spawn upstream in tributaries rather than in the mainstem Sacramento River; however, the completion and operation of Shasta Dam reduced water temperatures in the main stem downstream of Keswick Dam, which permitted spring-run Chinook Salmon to spawn there, resulting in hybridization with fall-run stocks. Although spring-run Chinook Salmon spawn earlier than fall-run, the timing of spawning of the two runs overlaps enough that hybridization can occur where they share the same spawning areas. Where the spring run is now forced to share spawning grounds in the mainstem Sacramento River with the fall run, fall-run Chinook Salmon may dominate because of their longer growth period in the ocean, slightly larger size, and less time spent holding in the stream prior to spawning. Hybridization between the two runs has tended to be to the detriment of the spring run life history.

Because of this hybridization with fall-run Chinook Salmon in the mainstem channel, there are considered to be only three “pure” self-sustaining populations of wild spring-run Chinook Salmon remaining in Deer, Mill, and Butte creeks.

Similar patterns have been observed in the Feather River, where the spring run historically spawned upstream of the location of Oroville Dam, and where they are now forced to spawn in the same area as the fall run, as well as in the Yuba and American rivers, where forced sympatry on the spawning grounds and subsequent hybridization following dam construction led to DFW concluding that the spring run was “extinct” in those rivers.

9B.5.4.3 Life History and Habitat Requirements

General habitat requirements for Chinook Salmon are described above; the following describes life history strategies and habitat requirements unique to the spring run or of primary importance to its life history. Spring-run Chinook Salmon display a stream-type life history strategy—adults migrate upstream while sexually immature, hold in deep cold pools over the summer, and spawn in late summer and early fall. Juvenile outmigration is highly variable, with some juveniles outmigrating in winter and spring, and others oversummering and then emigrating as yearlings. Table 9 illustrates life-history timing for spring-run Chinook Salmon in the Sacramento River basin. The table illustrates some of the changes in timing that have been observed for the run over the years, particularly with regard to upstream migration and spawning.

Table 9B.4 Life History Timing of Spring-run Chinook Salmon in the Sacramento River Basin

Life Stage	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sept	Oct	Nov	Dec
Adult entry into Sacramento-San Joaquin Delta Estuary												
"Historical" adult migration past Red Bluff Diversion Dam[a]												
"Recent" adult migration past Red Bluff Diversion Dam[b]												
Entry into spawning tributaries (current)[c]												
Adult holding												
Historical spawning in Sacramento River basin[d]												
Spawning (Deer, Mill, Butte creeks[e])												
Spawning (mainstem Sacramento River[f])												
Incubation												
Fry emergence												
Fry/juvenile outmigration from tributaries[g]												
Subyearling/Yearling outmigration from tributaries[g, h]												
Presence at CVP/SWP salvage facilities[i]												
Outmigration toward and through the Delta[i]												
Ocean entry (yearlings)												

Sources*:* Fisher 1994; Myers et al. 1998; Hill and Weber 1999; Ward and McReynolds 2001; USFWS 2005

Notes:

a. As observed in the 1970s (Association of California Water Agencies and California Urban Water Agencies 1997)

b. As observed in the 1980s (Association of California Water Agencies and California Urban Water Agencies 1997)

c. Association of California Water Agencies and California Urban Water Agencies (1997), Hill and Webber (1999)

d. Rutter (1908), Parker and Hanson (1944)

e. Harvey (1995), Moyle et al. (1995)

f. Association of California Water Agencies and California Urban Water Agencies (1997)

g. Some spring run disperse downstream soon after emergence as fry in March and April, with others smolting after several months of rearing, and still others remaining to oversummer and emigrate as yearlings (USFWS 1995).

h. Based on outmigrant trapping in Butte Creek in 1999 and 2000, up to 69% of age 0+ juveniles outmigrate through the lower Sacramento River and Sacramento-San Joaquin Delta between mid-November and mid-February, with a peak in December and January (DFG 1998, Hill and Weber 1999, Ward and McReynolds 2001). A smaller number remain in Butte Creek and outmigrate in late spring or early summer; and in both Butte and Mill creeks, some of these oversummer and outmigrate as yearlings from October to March, with a peak in November (Association of California Water Agencies and California Urban Water Agencies 1997, Hill and Webber 1999)

i. NMFS 2012 (unpublished data)

Period of activity

Period of peak activity

9B.5.4.3.1 Adult Upstream Migration and Spawning

Adult spring-run Chinook Salmon may return between the ages of 2 to 5 years. Historically, adults of this run are believed to have returned predominantly at ages 4 and 5 years at a large size. Most spring-run Chinook Salmon now return at age 3, although some portion returns at age 4 (Fisher 1994, McReynolds et al. 2005) probably because of intense ocean harvest (which removes the largest fish from the population and selects for fish that spend fewer years at sea). In 2003, an estimated 69 percent of the spring run in Butte Creek returned at age 4 (Ward et al. 2004); however, in most years, the proportion of age 4 adults is much smaller.

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 February and September, primarily in May and June (DFG 1998, Myers et al. 1998). 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. Adults enter Deer and Mill creeks beginning in March, peaking in May, and concluding in June (Vogel 1987a, 1987b; Association of California Water Agencies and California Urban Water Agencies 1997). Their upstream migration is timed to take advantage of spring snowmelt flows, which allow them access to upstream holding areas where temperatures are cool enough to hold over the summer prior to the spawning season (NMFS 1999a). In the Sacramento River, upstream migration of spring-run Chinook Salmon overlaps to a certain extent with that of winter-run Chinook Salmon; and adults from particular runs are not generally distinguishable from one another by physical appearance alone, making it difficult to pinpoint migration timing with precision (Healey 1991).

Adults require large, deep pools with moderate flows for holding over the summer prior to spawning in the fall. Marcotte (1984) reported that suitability of pools declines at depths less than 7.9 ft (2.4 m) and that optimal water velocities range from 0.5 to 1.2 ft/s (15 to 37 cm/s). In the John Day River in Oregon, spring-run adults usually hold in pools deeper than 4.9 ft (1.5 m) that contain cover from undercut banks, overhanging vegetation, boulders, or woody debris (Lindsay et al. 1986).

In Sacramento River tributaries, adults will pack densely in the limited holding pool habitat that is available. Some fish remain to spawn at the tails of the holding pools, while most move upstream to the upper watersheds to spawn, and still others move back downstream to spawn. Although there are several deep pools in the upper Sacramento River that may provide holding habitat for adult spring-run Chinook Salmon, it is not clear which pools are heavily used. As a result of cold water releases from Shasta Reservoir and natural channel characteristics, numerous deep pools with suitable holding habitat are located between Keswick Dam and Red Bluff (Northern California Water Association and Sacramento Valley Water Users 2011).

Water temperatures for adult spring-run Chinook Salmon holding and spawning are reportedly best when less than 60.8°F (16°C), and are lethal when greater than 80.6°F (27°C) (Hinze 1959, Boles et al. 1988, DFG 1998). Spring Chinook Salmon in the Sacramento River typically hold in pools below 69.8 to 77°F (21 to 25°C). Adults may be particularly sensitive to temperatures during July and August, when energy reserves are low and adults are preparing to spawn. There is evidence that spring-run Chinook Salmon in the San Joaquin River were exposed to high temperatures during migration and holding under historical conditions (Clark 1943, Yoshiyama et al. 2001). It is possible that Central Valley spring-run Chinook Salmon are adapted to tolerate warmer temperatures than other Chinook Salmon stocks; however, there is no experimental evidence to confirm this hypothesis, and short-term exposure to temperatures as high as 25 to 27°C (77 to 80.6°F) is known to be tolerated by adult Chinook Salmon (Boles et al. 1988).

Habitat suitability studies conducted by USFWS (2004) indicate that suitable spawning velocities for spring-run Chinook Salmon in Butte Creek range from 0.80 to 3.22 ft/s (24.4 to 98 cm/s), and suitable substrate size ranges from 1 to 5 inches (2.5 to 12.7 cm) in diameter. Adult Chinook have been observed spawning in water greater than 0.8 foot deep and in water velocities of 1.2 to 3.5 ft/s (DFG 1998).

The timing of spring run spawning in the mainstem Sacramento River has shifted later in the year, which is believed to be a result of genetic introgression with the fall run (Association of California Water Agencies and California Urban Water Agencies 1997). Populations in Deer and Mill creeks, which do not appear to have significantly hybridized with the fall run, generally spawn earlier than those in the main stem (Lindley et al. 2004). Rutter (1908) noted that most spawning in the late 1800s/early 1900s in the Sacramento River basin occurred in August. Parker and Hanson (1944) observed intensive spawning of spring-run Chinook Salmon from the first week of September through the end of October in 1941. Redd counts have indicated that spring-run Chinook Salmon spawning typically begins in late August, peaks in September, and concludes in October in both Deer and Mill creeks (Harvey 1995, Moyle et al. 1995, NMFS 2004a).

In the Feather River, the time of river entry for spring-run Chinook Salmon has apparently shifted to later in the season, and is now intermediate between timing of entry of spring run into other tributaries and timing of entry of the fall run. Whereas wild-type spring-run Chinook Salmon enter Deer and Mill creeks primarily in mid-April to mid-June, coded-wire tag data and anecdotal information from anglers indicate that Feather River fish do not enter fresh water until June or July (Association of California Water Agencies and California Urban Water Agencies 1997).

9B.5.4.3.2 Egg Incubation and Alevin Development

In the Sacramento River and its tributaries, egg incubation for spring-run Chinook Salmon extends from August to March (Fisher 1994, Ward and McReynolds 2001). Egg incubation generally lasts between 40 and 90 days at water temperatures of 42.8 to 53.6°F (6 to 12°C) (Vernier 1969, Bams 1970, Heming

1982). At temperatures of 37°F (2.7°C), time to 50 percent hatching can take up to 159 days (Alderdice and Velsen 1978). Alevins remain in the gravel for 2 to 3 weeks after hatching while absorbing their yolk sacs. Emergence from the gravels occurs from November to March in the Sacramento River basin (Fisher 1994, Ward and McReynolds 2001). Once fry emerge from the gravel, they initially seek areas of shallow water and low velocities while they finish absorbing the yolk sac (Moyle 2002). 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). USFWS catches of juvenile salmon in the Sacramento River near West Sacramento showed that larger juvenile salmon were captured in the main channel and smaller fry were typically captured along the channel margins (USFWS 1997).

9B.5.4.3.3 Juvenile Rearing and Outmigration

Fry and juvenile rearing takes place in the natal streams, the mainstem of the Sacramento River, inundated floodplains (including the Sutter and Yolo bypasses), and the Delta. During the winter, some spring-run juveniles have been found rearing in the lower portions of non-natal tributaries and intermittent streams (Maslin et al. 1997, Snider et al. 2001).

The rearing and outmigration patterns exhibited by spring-run Chinook Salmon are highly variable, with fish rearing anywhere from 3 to 15 months before outmigrating to the ocean (Fisher 1994). Variation in length of juvenile residence may be observed both within and among streams (e.g., Butte versus Mill creeks, [USFWS 1996]). Some may disperse downstream soon after emergence as fry in March and April, with others smolting after several months of rearing, and still others remaining to oversummer and emigrate as yearlings (USFWS 1996). Scale analysis indicates that most returning adults have emigrated as subyearlings (Myers et al. 1998). Calkins et al. (1940) conducted an analysis of scales of returning adults, and estimated that more than 90 percent had emigrated as subyearlings, at about 3.5 inches (88 mm).

The term “yearling” is generally applied to any juveniles that remain to oversummer in their natal stream. Yearling outmigrants are common in Deer and Mill creeks, but rare in Butte Creek (Association of California Water Agencies and California Urban Water Agencies 1997). Extensive outmigrant trapping in Butte Creek has shown that spring-run Chinook Salmon outmigrate primarily as juvenile (age 0+) fish from November through June, with a small proportion remaining to emigrate as yearlings beginning in mid-September and extending through March, with a peak in November (Association of California Water Agencies and California Urban Water Agencies 1997, Hill and Webber 1999, Ward et al. 2004). Peak movement of yearling Central Valley spring-run Chinook Salmon in the Sacramento River at Knights Landing occurs in December, and young-of-the-year juveniles occur in March and April; however, juveniles were also observed between November and the end of May (Snider and Titus 2000).

Coded-wire-tag studies conducted on Butte Creek spring-run Chinook Salmon have shown that juveniles use the Sutter Bypass as a rearing area until it begins to drain in the late winter or spring (Hill and Webber 1999). Few juvenile Chinook Salmon are observed in the bypass after mid-May. Five recaptures indicate that juveniles leaving the Sutter Bypass migrate downstream rapidly and do not use the mainstem Sacramento River as rearing habitat (Hill and Webber 1999).

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). Very little information is available on the estuarine rearing of spring-run Chinook Salmon (NMFS 2004a). NMFS (2004a) postulates that, because spring-run Chinook Salmon yearling outmigrants are larger than fall-run Chinook Salmon smolts, and are ready to smolt upon entering the Delta, they may spend little time rearing in the estuary. Most have presumably left the estuary by mid-May (DFG 1995). Once in the ocean, spring-run Chinook Salmon perform extensive offshore migrations before returning to their natal streams to spawn.

9B.5.4.4 Population Trends

At one time, spring-run Chinook Salmon may have been the most abundant race in the Central Valley, with escapement in the hundreds of thousands (Mills and Fisher 1994). Spring-run Chinook Salmon have since declined to remnant populations totaling a few thousand fish, sometimes approaching 30,000 to 40,000 in good years (Mills and Fisher 1994, NMFS 1999a). Loss of access to upstream spawning and rearing areas due to the construction of dams in the Sacramento and San Joaquin rivers is believed to have been a major cause of the decline of the spring run.

Under historical conditions, it is doubtful that spring-run Chinook Salmon spawned in the mainstem Sacramento in significant numbers (Lindley et al. 2004). After the closure of Shasta and Keswick dams, spring-run Chinook Salmon began to spawn in the mainstem Sacramento River when changes in temperatures made this a viable life-history strategy. Throughout the 1970s and 1980s, thousands of spring-run Chinook Salmon passed RBDD en route to spawning grounds farther upstream. By the 1990s, escapements had declined; however, changes in the RBDD gate operations beginning in 1986 complicated the process of estimating spring-run Chinook Salmon abundance. Identification of the spring run at RBDD is also complicated by their low escapements and the difficulty of distinguishing fish of this run from those of the fall run. The two runs cannot be distinguished reliably by physical characteristics or run timing (Healey 1991)because of the naturally protracted run timing of the abundant fall run, and the apparent shift to later upstream migration timing by the spring run, which results in the runs being more temporally overlapped than they were historically.

Populations of spring-run Chinook Salmon in Butte Creek increased after the 1990s, and Butte Creek currently has the largest naturally spawning spring-run population (DFW 2014, GrandTab data). A few naturally spawning fish are also present in Battle, Clear, Cottonwood, Antelope, Mill, Deer, and Big Chico creeks (DFW 2014, GrandTab data). In general, spring-run Chinook Salmon that are most genetically similar to the runs that occurred historically in the Sacramento basin are currently confined to spawning primarily in Deer, Mill, and Butte creeks, with perhaps a few spawning in the mainstem Sacramento River.

Restrictions on ocean harvest to protect winter-run Chinook Salmon, as well as improved ocean conditions, have likely had a positive impact on spring-run Chinook Salmon adult returns to the Central Valley. In 2008, abundance in key indicator streams (e.g., Mill, Deer, and Butte Creeks) was at historical levels; however, between 2008 and 2011, spring-run populations in these same streams dropped closer to historical lows (as based on preliminary DFW 2014, GrandTab data). Spring-run Chinook Salmon populations generally increased from 1990 through 2006, but then returned to very low levels by 2008 and remained low through 2011. The preliminary total spring-run Chinook Salmon escapement count for 2013 was 23,697 adults, which was the highest count since 2003 (30,697 adults) and over three times that of 2011 (7,408 adults) (DFW 2014) (Table 9B.5).

Table 9B.5 Recent Spring-run Chinook Salmon Natural and Hatchery Escapement

YEAR	Sacramento River Mainstem	Battle Ck[a]	Clear Ck	Cottonwood Ck	Antelope Ck	Mill Ck	Deer Ck	Big Chico Ck	Butte Ck Snorkel	Butte Ck Carcass	Feather River Hatchery[b]	TOTAL SPRING RUN
1990	4,198	2	–	–	–	844	496	–	250	–	1,893	7,683
1991	825	–	–	–	–	319	479	–	–	–	4,303	5,926
1992	371	–	–	–	0	237	209	–	730	–	1,497	3,044
1993	391	–	1	1	3	61	259	38	650	–	4,672	6,076
1994	862	–	0	–	0	723	485	2	474	–	3,641	6,187
1995	426	66	2	8	7	320	1,295	200	7,500	–	5,414	15,238
1996	378	35	–	6	1	253	614	2	1,413	–	6,381	9,083
1997	128	107	–	0	0	202	466	2	635	–	3,653	5,193
1998	1,115	178	47	477	154	424	1,879	369	20,259	–	6,746	31,649
1999	262	73	35	102	40	560	1,591	27	3,679	–	3,731	10,100
2000	43	78	9	122	9	544	637	27	4,118	–	3,657	9,244
2001	621	111	0	245	8	1,104	1,622	39	9,605	18,670	4,135	26,663
2002	195	222	66	125	46	1,594	2,195	0	8,785	16,409	4,189	25,043
2003	0	221	25	73	46	1,426	2,759	81	4,398	17,404	8,662	30,697
2004	370	90	98	17	3	998	804	0	7,390	10,558	4,212	17,150
2005	30	73	69	47	82	1,150	2,239	37	10,625	17,592	1,774	23,093
2006	0	221	77	55	102	1,002	2,432	299	4,579	6,537	2,181	12,906
2007	248	291	194	34	26	920	644	0	4,943	6,871	2,635	11,144

YEAR	Sacramento River Mainstem	Battle Ck[a]	Clear Ck	Cottonwood Ck	Antelope Ck	Mill Ck	Deer Ck	Big Chico Ck	Butte Ck Snorkel	Butte Ck Carcass	Feather River Hatchery[b]	TOTAL SPRING RUN
2008	52	105	200	0	3	381	140	0	3,935	11,046	1,460	13,387
[2009]	0	194	120	0	0	220	213	6	2,059	2,763	989	4,505
[2010]	0	172	21	15	17	482	262	2	1,160	1,991	1,661	4,623
[2011]	0	157	8	2	6	366	271	124	2,130	4,505	1,969	7,408
[2012]	0	799	68	1	1	768	734	0	8,615	16,140	3,738	22,249
[2013]	0	608	659	1	0	644	708	0	11,470	16,783	4,294	23,697
[2014]	0	429	95	2	7	679	830	0	3,616	5,083	2,776	9,901

Source: DFW 2014, GrandTab data.

Notes:
Data for years in brackets are preliminary.

a. In 2009, USFWS conducted a comprehensive analysis of Battle Creek coded wire tag data from 2000-2008 to estimate numbers of fall- and late fall-run Chinook Salmon returning to Battle Creek. Previously, a cutoff date of December 1 was used to assign run. This changed some Battle Creek estimates.

b. Feather River Hatchery implemented a methodology change in 2005 for distinguishing spring- from fall-run. Fish arriving prior to the spring-run spawning period were tagged and returned to the river. The spring-run escapement was the number of these tagged fish that subsequently returned to the hatchery during the spring-run spawning period.

9B.5.5 Central Valley Fall-run and Late Fall-run Chinook Salmon

9B.5.5.1 Legal Status

Federal: Species of Concern
State: Central Valley fall-run – None; Central Valley late fall-run – Species of Special Concern

Fall-run populations occur throughout the range of Chinook Salmon and are currently the most abundant and widespread of the salmon runs in California and the Central Valley, largely because the construction of dams was not as damaging in terms of loss of historical habitat compared to the runs that spawned at higher elevations. Fall-run abundance is also a function of hatchery supplementation, because fall-run Chinook Salmon have been the primary focus of hatchery production at Central Valley hatcheries for several decades. As the most abundant salmonid species in the Central Valley, fall-run Chinook Salmon constitute an important component of the commercial and recreational salmon fishery in California. NMFS designated the Central Valley Fall (and Late fall) Chinook Salmon ESU as a Species of Concern in 2004 (NMFS 2004b).

NMFS classifies late fall-run Chinook Salmon as part of the Central Valley fall-run and late fall-run Chinook Salmon ESU, reasoning that the late fall-run population represents a life-history variation of the fall-run salmon population rather than a distinct run (NMFS 2004b). However, agencies generally treat late fall-run salmon in the Sacramento River basin as a distinct run, conducting separate carcass and redd surveys for them, and publishing separate reports to address the fall-run and late fall-run populations. Agencies also manage the hatchery propagation of late fall-run separately from fall-run Chinook Salmon. Except for hatchery propagation, there are relatively few restoration and management activities that focus specifically on late fall-run Chinook Salmon in the Sacramento River, as compared to the other runs of Chinook Salmon in the basin (USFWS 1996).

9B.5.5.2 Distribution

9B.5.5.2.1 Fall-run Chinook Salmon

Within the range of the Central Valley ESU, large populations of fall-run Chinook Salmon are found in the Sacramento River and its major tributaries. Fall-run Chinook Salmon are the most widely distributed salmonid in the Sacramento River basin, with significant spawning populations documented as far north as the upstream limit of anadromy in the upper Sacramento River (Keswick Dam at RM 302) and as far south as the American River near Sacramento. Sizeable spawning populations occur in other tributaries to the Sacramento River—Clear Creek, Battle Creek, Butte Creek, and Feather River—with more modest spawning populations in numerous smaller tributaries (e.g., Deer, Mill, Cow, and Antelope creeks). The San Joaquin River system once supported large runs of both spring-run and fall-run Chinook Salmon. Fall-run Chinook Salmon historically spawned in the mainstem San Joaquin River upstream of the Merced

River confluence and in the mainstem channels of the major tributaries—the Merced, Tuolumne, and Stanislaus rivers. Dam construction and water diversion dewatered much of the mainstem San Joaquin River, limiting fall-run Chinook to the three major tributaries where they currently spawn and rear downstream of mainstem dams.

9B.5.5.2.2 Late Fall-run Chinook Salmon

Little is known about the historical distribution of late fall-run salmon in the Sacramento River valley. Late fall-run Chinook Salmon currently spawn primarily in the mainstem Sacramento River between Red Bluff (RM 243.5) and Keswick Dam (RM 302). DFW conducts aerial redd surveys that target the late fall-run spawning period, and an analysis of the surveys suggests that adults generally spawn upstream of RBDD (RM 243.5). Yoshiyama et al. (1996) gleaned incidental references to late fall-run fish from historical documents to suggest that late fall-run Chinook Salmon historically spawned in the mainstem reaches of the upper Sacramento River and tributaries such as the Little Sacramento, Pit, and McCloud rivers. Because a significant fraction of juvenile late fall-run Chinook Salmon oversummer in natal streams before emigrating, mainstem reaches close to coldwater sources were likely the most important historical spawning areas for late fall-run Chinook Salmon. Unfortunately, there is little historical data on water temperatures in the upper Sacramento River basin to analyze the stream reaches that may have been important spawning and rearing areas for the late fall-run. Yoshiyama et al. (1996) also suggested the presence of historical spawning populations of late fall-run Chinook Salmon in the American and San Joaquin rivers prior to the era of large dam construction.

9B.5.5.3 Life History and Habitat Requirements

General habitat requirements for Chinook Salmon were described previously. Only habitat requirements specific to fall-run and late fall-run Chinook Salmon are described here.

Historically, the summer water temperature regime in the Sacramento River was a key variable that influenced the life history timing and strategy of the different salmonids that occur in the basin. Fall-run Chinook Salmon avoid stressful summer conditions by migrating upstream in the fall (September–November) when both air and water temperatures begin to cool. Because they arrive at spawning grounds with fully developed gonads, adult fall-run can spawn immediately (October–November), which allows their progeny to emerge in time to emigrate from the Sacramento River as fry in the subsequent spring (February–May) before water temperatures become too high.

Because fall-run Chinook Salmon adults migrate upstream during periods of low fall baseflows, spawning is generally limited to the alluvial reaches of mainstem rivers below flow-related obstacles. There is relatively little oversummering habitat in these lower mainstem reaches to support a yearling life history strategy, so the majority of fall-run juveniles emigrate as fry before spring water temperatures become lethal. Historically, warming spring water temperatures may have imposed a lethal penalty on the progeny of any late-arriving fall-run adults.

Yoshiyama et al. (1996) suggested that spawning populations of late fall-run salmon occurred in the Sacramento River prior to the construction of Shasta Dam, citing what are mostly incidental references to late fall-run salmon in several historical documents. Although these historical accounts indicate the occurrence of salmon migrating upstream and spawning in December or later on several different Central Valley tributaries, it is not clear whether such migration and spawning activity occurred consistently or in substantial numbers. These historical references to late fall-run fish may document fall-run stragglers whose progeny perished the subsequent spring and contributed little to the population, or they may indicate passage barriers that delayed the upstream migration and spawning of fall-run fish en masse.

Late fall-run salmon in the Sacramento River have been a collateral beneficiary of the operation of the Shasta and Trinity divisions of the CVP, which maintain suitable water conditions for endangered winter-run Chinook Salmon. Since 1994, coldwater releases designed to protect winter-run eggs incubating through the summer months have likely expanded suitable oversummering habitat for late fall-run juveniles downstream. Fall-run juveniles could continue to emigrate as fry or spend a summer growing in the river before emigrating as subyearlings.

The late fall-run Chinook Salmon strategy is successful because a substantial fraction of juveniles oversummer in the Sacramento River before emigrating, which allows them to avoid predation through both their larger size and greater swimming ability (larger juvenile salmon can evade a certain amount of predation through size alone). One implication of this life history strategy is that rearing habitat is most likely the limiting factor for late fall-run Chinook Salmon, especially if availability of cool water determines the downstream extent of spawning habitat for late fall-run salmon.

Table 9B.6 and 9B.7 display the life-history timing of fall-run and late fall-run Chinook Salmon in the action area.

Table 9B.6 Life History Timing of Central Valley Fall-run Chinook Salmon

Life Stage	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sept	Oct	Nov	Dec
Adult migration past Red Bluff Diversion Dam												
Spawning												
Incubation												
Fry emergence[a]												
Rearing in mainstem Sacramento River[b]												
Outmigration past Red Bluff Diversion Dam												
Presence at CVP/SWP salvage facilities												
Emigration toward and through the Delta[c]												

Notes:
a. Northern California Water Association and Sacramento Valley Water Users (2011) shows emergence ending in February; Williams (2006) shows emergence ending in April.
b. A few fall-run Chinook Salmon remain upstream of RBDD location to rear to a yearling life stage.
c. NMFS (2012, unpublished data)

Period of light activity
Period of moderate activity
Period of peak activity

Table 9B.7 Life History Timing of Central Valley Late Fall-run Chinook Salmon

Life Stage	Jan		Feb		Mar		Apr		May		Jun		Jul		Aug		Sept		Oct		Nov		Dec	
Adult entry into mainstem Sacramento River[a, b]	Peak	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate													Moderate	Moderate	Moderate	Peak	Peak
Migration past Red Bluff Diversion Dam[a, b, c]	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate													Moderate	Moderate	Moderate	Peak	Peak
Adult holding[d]																				Moderate	Moderate	Moderate	Moderate	Moderate
Spawning[a, b, c, e, f, g]	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate																Moderate	Moderate
Incubation	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate												
Fry emergence[a, c]							Peak	Peak	Peak	Peak	Moderate	Moderate												
Stream residency[a, c]	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate
Fry outmigration past Red Bluff Diversion Dam[b]							Peak	Peak	Peak	Peak	Moderate	Moderate												
Smolt outmigration past Red Bluff Diversion Dam[b]													Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Peak	Peak	Moderate	Moderate		
Presence at CVP/SWP salvage facilities	Peak	Peak	Moderate	Moderate	Light	Light	Light	Light	Light	Light	Light	Light			Light	Light	Light	Light			Moderate	Moderate	Moderate	Moderate
Emigration toward and through the Delta[c]	Peak	Peak	Light	Light			Moderate	Moderate	Moderate	Moderate			Light	Light	Light	Light	Light	Light	Light	Light	Peak	Peak	Peak	Peak
Smolt outmigration[a]	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate											Moderate	Moderate	Moderate	Moderate
Ocean entry[c]	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate									Moderate	Moderate	Moderate	Moderate	Moderate	Moderate

Sources:
a. Yoshiyama et al. 1998
b. Association of California Water Agencies and California Urban Water Agencies
c. Fisher 1994
d. Moyle 2002
e. Snider et al. 1998, 1999, 2000
f. Northern California Water Association and Sacramento Valley Water Users 2011
g. Williams 2006

Period of light activity
Period of moderate activity
Period of peak activity

9B.5.5.3.1 Adult Upstream Migration and Spawning

Adult fall-run Chinook Salmon migrate into the Sacramento River and its tributaries from June through December in mature condition, with upstream migration peaking in September and October. Fall-run Chinook Salmon in the San Joaquin system typically enter spawning streams from September through November. Adults spawn soon after arriving at their spawning grounds between late September and December, with peak spawning activity in late October and early November.

Adult late fall-run Chinook Salmon migrate up the Sacramento River between mid-October and mid-April, with peak migration occurring in December (Reclamation 1991) (Table 9B.7). Adults spawn soon after reaching spawning areas between January and April. Fisher reports that peak spawning in the Sacramento River occurs in early February (1994), but carcass surveys conducted in the late 1990s suggest that peak spawning may occur in January (Snider et al 1998, 1999, 2000).

Fall-run and late fall-run Chinook Salmon are generally able to spawn in deeper water with higher velocities than Chinook Salmon in other runs because of their larger size (Healey 1991). Late fall-run salmon tend to be the largest individuals of the Chinook Salmon species that occur in the Sacramento River basin (USFWS 1996).

Fry emergence occurs from December through March, and fry rear in freshwater for only a few months before migrating downstream to the ocean as smolts between March and July (Yoshiyama et al. 1998). Late fall-run fry emerge from redds between April and June (Vogel and Marine 1991).

9B.5.5.3.2 Juvenile Rearing and Outmigration

Fall-run Chinook Salmon in the Sacramento River generally exhibit two rearing strategies: migrating to the lower reaches of the river or Delta as fry, or remaining to rear in the gravel-bedded reach for about 3 months and then smolting and outmigrating. The highest abundances of fry in the Delta are observed in wet years (Brandes and McLain 2001). Fall-run Chinook Salmon fry rear during a time and in a location where floodplain inundation is most likely to occur, thereby expanding the amount of rearing habitat available. Relative survival of fry appears to be higher in the upper Sacramento River than in the Delta or bay, especially in wet years (Brandes and McClain 2001).

One potential disadvantage of early emergence and emigration and rearing in mainstem channels and the estuary is the possibility of higher predation mortality because of the relatively small size of emigrants. However, fall-run Chinook Salmon fry exhibit several characteristics to combat predation mortality. Predators often occupy deep pools in mainstem channels, so fry generally use shallow water habitat found along channel margins or in runs and riffles to avoid predators. Because rearing habitat is not limiting for fall-run Chinook Salmon fry, they do not exhibit territorial behavior, which allows them to rear, smolt, and outmigrate in higher densities. By emigrating synchronously in schools rather than as individuals, fall-run Chinook Salmon fry and smolts can swamp potential predators to avoid significant losses to predation; and by emigrating in late spring, they have the advantage of higher discharge fueled by early snowmelt, which can reduce their exposure to predation.

Fall-run Chinook Salmon juvenile smolt during early spring, prior to increases in water temperatures. Juvenile Chinook Salmon feed and grow as they move downstream in spring and summer; larger individuals are more likely to move downstream earlier than smaller juveniles (Nicholas and Hankin 1989, Beckman et al. 1998), and it appears that in some systems juveniles that do not reach a critical size threshold will not outmigrate, but will remain to oversummer (Bradford et al. 2001). Bell (1958) suggests that the timing of yearling smolt outmigration corresponds to increasing spring discharges and temperatures. Kjelson et al. (1981) observed that peak seine catches of Chinook Salmon fry in the Sacramento-San Joaquin Delta correlated with increases in flow associated with storm runoff. Flow accounted for approximately 30 percent of the variability in the fry catch.

As fall-run Chinook Salmon fry and parr migrate downstream, they also use the lower reaches of non-natal tributaries as rearing habitat (Maslin et al. 1997). During periods of high winter and spring runoff, fall-run Chinook Salmon juveniles are also diverted into the bypasses that border the Sacramento River, where growing conditions are generally better than mainstem rearing habitats, which can facilitate higher rates of juvenile survival (Sommer et al. 2001). Natural floodplain or riparian areas that become inundated during high flows may also provide good habitat for juvenile Chinook Salmon and prevent them from being displaced downstream (The Nature Conservancy 2003).

Research conducted in the Central Valley suggests that seasonally inundated, shallow water habitats may provide superior rearing habitat for juvenile salmonids than mainstem channels (Sommer et al. 2001). Juvenile fall-run salmon migrate downstream between January and June when floodplains and bypasses are periodically flooded during wet water years. By promoting faster growth, prolonged floodplain inundation likely helps the fall-run population by increasing juvenile salmon survival.

As described above, the timing of late fall-run spawning in January through March means that fry emerge between April and June. Water temperatures in the lower Sacramento River are often too high in May and June to support fry survival, so later-emerging fry that migrate downstream likely suffer high rates of mortality and contribute little to the population. This suggests that a significant fraction of late fall-run juveniles rear in the upper Sacramento River throughout the summer before emigrating in the following fall and early winter as large subyearlings (Fisher 1994). Summer rearing is made possible by the cold water releases from the Shasta-Trinity divisions of the CVP. Late fall-run juveniles generally leave the Sacramento River by December (Vogel and Marine 1991), with peak emigration of smolts in October.

Although growth rates of juvenile Chinook Salmon may be high at temperatures approaching 19°C (66°F), cooler temperatures may be required to successfully complete the physiological transformation from parr to smolt. Smoltification in juvenile Sacramento River fall-run Chinook Salmon was studied by Marine (1997), who found that juveniles reared under a high temperature regime of 21 to 24°C (70 to 75°F) exhibited altered and impaired smoltification patterns relative to those reared at low 55 to 61°F (13 to 16°C) and moderate 17 to 20°C (63 to 68°F) temperatures. Some alteration and impairment of smoltification was also seen in the juveniles reared at the moderate temperatures.

Chronic exposure to high temperatures may also result in greater vulnerability to predation. In this same study by Marine (1997), Sacramento River fall-run Chinook Salmon reared at the highest temperatures (21 to 24°C [70 to 75°F]) were preyed upon by Striped Bass more often than those reared at low or moderate temperatures. Consumption rates of piscivorous fish such as Sacramento pikeminnow, Striped Bass, and largemouth bass increase with temperature, which may compound the effects of high temperature on juvenile and smolt predation mortality. Juvenile growth rates are an important influence on survival; faster growth thus both increases the range of food items available to them and decreases their vulnerability to predation (Myrick and Cech 2004).

9B.5.5.3.3 Ocean Residence

When fall-run Chinook Salmon produced from the Sacramento-San Joaquin system enter the ocean, they appear to head north to inhabit the northern California-southern Oregon coast (Oregon Department of Fish and Wildlife 1987). They typically have a greater tendency to remain along the continental shelf than do stream-type Chinook Salmon (Healey 1983). The age of returning Chinook Salmon adults in California ranges from 2 to 5 years.

9B.5.5.4 Population Trends

Although NMFS considers fall-run and late fall-run Chinook Salmon as part of the same ESU in the Central Valley, most resource agencies have tracked the two runs separately. For example, DFW has conducted aerial redd surveys specifically targeting late fall-run salmon, and the Anadromous Fish Restoration Program (AFRP) has tracked late fall-run salmon escapements as a separate population. However, reports on fall-run escapement estimates vary because some include late fall-run in the estimates, while others do not. Because the older reports often fail to clarify which runs are being enumerated in the escapement estimate, care must be exercised when using fall-run escapement estimates, especially from different sources.

9B.5.5.4.1 Fall-run Chinook Salmon

Fall-run Chinook Salmon estimates are available from 1940; however, systematic counts of Chinook Salmon in the San Joaquin Basin began in 1953, long after construction of large dams on the major San Joaquin basin rivers. Comparable estimates of population size before 1940 are not available. Since population estimates began, the number of fall-run Chinook returning to the San Joaquin

Basin annually has fluctuated widely. Escapement in the Tuolumne River dropped from a high of 40,300 in 1985 to a low of about 100 resulting from the 1987 to 1992 dry period (TID/MID 1997). With increased precipitation and improved flow conditions, escapement increased to 3,300 in 1996 (TID/MID 1997). From 1971 to 2007, hatchery production is estimated to have composed about 29 percent of the returning adult fall-run Chinook Salmon in the San Joaquin basin (PFMC 2008). Table 9B.8 provides a summary of estimated escapement from 1990 to 2013 in the Sacramento and San Joaquin River systems.

Table 9B.8 Recent Fall-run Chinook Salmon Natural and Hatchery Escapement

	Sacramento River System				San Joaquin River System			Sacramento and San Joaquin Combined		
Year	Hatch.	Main.	Trib.	Total	Hatch.	Trib.	Total	Hatch.	In-River	Total
1990	25,611	48,284	12,803	86,698	114	1,041	1,155	25,725	62,128	87,853
1991	28,528	30,631	72,296	131,455	83	917	1,000	28,611	103,844	132,455
1992	30,171	32,229	44,995	107,395	1,078	1,940	3,018	31,249	79,164	110,413
1993	30,234	46,231	82,975	159,440	2,573	3,410	5,983	32,807	132,616	165,423
1994	42,760	58,546	111,078	212,384	2,862	5,421	8,283	45,622	175,045	220,667
1995	45,324	63,934	211,025	320,283	3,925	5,960	9,885	49,249	280,919	330,168
1996	36,936	84,086	213,646	334,668	5,024	11,859	16,883	41,960	309,591	351,551
1997	71,448	119,296	185,484	376,228	7,440	19,129	26,569	78,888	323,909	402,797
1998	75,028	6,318	141,079	222,425	3,890	19,711	23,601	78,918	167,108	246,026
1999	49,657	161,192	180,501	391,350	4,787	18,122	22,909	54,444	359,815	414,259
2000	50,965	96,688	290,698	438,351	7,396	39,934	47,330	58,361	427,320	485,681
2001	61,318	75,296	453,323	589,937	7,391	27,303	34,694	68,709	555,922	624,631
2002	96,248	65,690	672,962	834,900	9,753	28,016	37,769	106,001	766,668	872,669
2003	118,097	89,229	362,161	569,487	8,666	12,839	21,505	126,763	464,229	590,992
2004	116,869	43,604	202,904	363,377	11,406	12,065	23,471	128,275	258,573	386,848
2005	187,427	57,012	172,457	416,896	5,984	14,813	20,797	193,411	244,282	437,693
2006	80,594	55,468	146,427	282,489	4,289	6,176	10,465	84,883	208,071	292,954
2007	22,511	17,061	54,767	94,339	1,130	1,699	2,829	23,641	73,527	97,168
2008	18,785	24,743	25,618	69,146	315	1,830	2,145	19,100	52,191	71,291
[2009]	20,904	5,827	22,842	49,573	1,799	1,757	3,556	22,703	30,426	53,129

Year	Sacramento River System				San Joaquin River System			Sacramento and San Joaquin Combined		
	Hatch.	Main.	Trib.	Total	Hatch.	Trib.	Total	Hatch.	In-River	Total
[2010]	46,306	16,372	90,154	152,832	5,421	4,937	10,358	51,727	111,463	163,190
[2011]	87,679	11,957	105,460	205,096	16,293	6,500	22,793	103,972	123,917	227,889
[2012]	136,710	28,701	155,450	320,861	7,620	13,342	20,962	144,330	197,493	341,823
[2013]	107,001	40,084	279,871	426,956	6,279	14,668	20,947	113,280	334,623	447,903
[2014]	50,713	34,876	152,587	238,176	9,627	8,094	17,721	60,340	195,557	255,897

Source: DFW 2014

Note:
Data for years in brackets are preliminary.

9B.5.5.4.2 Late Fall-run Chinook Salmon

There is little information to evaluate the historical abundance of late fall-run salmon in the Sacramento River basin. In fact, late fall-run salmon were first recognized by fishery agencies as a distinct run only after the construction of RBDD in 1966, which permitted more accurate counting of upstream migrants and the timing of upstream migration (USFWS 1996). Between 1967 and 1976, late fall-run salmon escapements averaged 22,000 adults (USFWS 1996); however, between 1977 and 1985, escapements averaged only about 9,900 adults (DFW 2014). Population estimates of late fall-run salmon after 1985 are complicated by changes in RBDD gate operations, when Reclamation began raising the dam gates during winter months to facilitate the upstream migration of winter-run Chinook Salmon. Because the upstream migration of late fall-run salmon overlaps with that of winter-run Chinook Salmon, late fall-run benefited from improved upstream access, but the accuracy of escapement estimates suffered (USFWS 1996). RBDD gate operations were revised again in 1994 so that gates were raised between September 15 and May 15, encompassing the entire upstream migration period of late fall-run salmon and further compromising the calculation of escapements. Post-1985 escapement estimates are cruder because of the change in RBDD gate operations. Table 9B.9 provides a summary of estimated escapement from 1970 to 2013 in the mainstem Sacramento River, Battle Creek, and Clear Creek.

Table 9B.9 Recent Late Fall-run Chinook Salmon Natural and Hatchery Escapement

Year	Sacramento River above RBDD	CNFH Transfers	Total above RBDD	Sacramento River below RBDD	Battle Creek	Battle Creek CNFH	Battle Creek Total	Clear Creek	Total
Nov 1990-Apr 1991	6,493	118	6,611	1,491	–	161	161	–	8,263
Nov 1991-Apr 1992	8,958	398	9,356	431	–	344	344	–	10,131
Nov 1992-Apr 1993	339	400	739	–	–	528	528	–	1,267
Nov 1993-Apr 1994	137	154	291	–	–	598	598	–	889
Nov 1994-Apr 1995	–	166	166	–	–	323	323	–	489
Nov 1995-Apr 1996	–	48	48	–	–	1,337	1,337	–	1,385
Nov 1996-Apr 1997	–	–	–	–	–	4,578	4,578	–	4,578
Nov 1997-Apr 1998	38,239	–	38,239	1,101	–	3,079	3,079	–	42,419
Nov 1998-Apr 1999	8,683	–	8,683		–	7,075	7,075	–	15,758
Nov 1999-Apr 2000	8,580	–	8,580	122	0	4,181	4,181	–	12,883
Nov 2000-Apr 2001	18,351	–	18,351	925	98	2,439	2,537	–	21,813
Nov 2001-Apr 2002	36,004	–	36,004	0	216	4,186	4,402	–	40,406
Nov 2002-Apr 2003	5,346	38	5,384	148	57	3,183	3,240	110	8,882
Nov 2003-Apr 2004	8,824	60	8,884	0	40	5,166	5,206	60	14,150
Nov 2004-Apr 2005	9,493	79	9,572	1,031	23	5,562	5,585	94	16,282
Nov 2005-Apr 2006	7,678	12	7,690	2,485	50	4,822	4,872	42	15,089
Nov 2006-Apr 2007	13,798	66	13,864	1,477	72	3,361	3,433	69	18,843
Nov 2007-Apr 2008	3,673	0	3,673	291	19	6,334	6,353	55	10,372

Year	Sacramento River above RBDD	CNFH Transfers	Total above RBDD	Sacramento River below RBDD	Battle Creek	Battle Creek CNFH	Battle Creek Total	Clear Creek	Total
Nov 2008-Apr 2009	3,271	58	3,329	63	32	6,436	6,468	336	10,196
[Nov 2009-Apr 2010]	3,843	81	3,924	439	27	5,505	5,532	91	9,986
[Nov 2010-Apr 2011]	3,686	39	3,725	0	28	4,635	4,663	58	8,446
[Nov 2011-Apr 2012]	2,811	47	2,858	11	19	3,031	3,050	50	5,969
[Nov 2012-Apr 2013]	4,918	43	4,961	309	42	3,577	3,619	77	8,966
[Nov 2013-Apr 2014]	7,227	39	7,266	723	120	4,869	4,989	72	13,050

Source: DFW 2014

Note:
Data for years in brackets are preliminary.

9B.5.5.4.3 Hybridization

Historically, spring-run Chinook Salmon and fall-run Chinook Salmon both spawned during the fall, but they were separated spatially because spring-run Chinook Salmon spawned in upper tributaries that the fall-run Chinook Salmon could not access. Under current conditions, the Keswick and Shasta dams have prevented spring-run Chinook Salmon from accessing upper tributaries, and instead they spawn in the mainstem Sacramento River where the fall run spawns. The elimination of spatial segregation of fall-run Chinook Salmon and spring-run Chinook Salmon spawning contributed to hybridization on the spawning grounds (Yoshiyama et al. 1998). Also, hatchery practices have likely mixed fall-run and spring-run Chinook Salmon stocks, causing even greater hybridization. By hybridizing with spring-run Chinook Salmon, the peak spawning activity of fall-run Chinook Salmon has likely shifted to occur earlier than it did historically.

9B.5.5.5 Hatchery Influence

Fall-run Chinook Salmon have long been a focus of hatchery production in the Central Valley, and the artificial propagation of the fall run supports the commercial and recreational harvest of salmon in California. Within the Sacramento River basin, Coleman National Fish Hatchery on Battle Creek produces substantial numbers of fall-run salmon for release in the Sacramento River and Bay-Delta estuary. Using a mixed-stock model to estimate the contribution of wild fish from the Central Valley to the fall-run Chinook Salmon ocean fishery, Barnett-Johnson et al. (2007) found that the contribution of wild fish was about 10 percent, which suggests that hatchery supplementation is a substantial contributor to the population.

Late fall-run salmon have been artificially propagated at the Coleman National Fish Hatchery on Battle Creek for more than two decades. USFWS releases between 200,000 and 2.5 million late fall-run juveniles in the Sacramento basin each year, primarily in Battle Creek. Although hatchery strays likely compose a portion of the spawning population of late fall-run salmon in the Sacramento River, it is unclear what proportion of escapements that hatchery-origin fish constitutes. It is also unclear whether hatchery juveniles that are released in Battle Creek compete with naturally spawned juveniles for oversummering habitat in the mainstem Sacramento River.

9B.5.6 Upper Klamath and Trinity Rivers Spring-Run Chinook Salmon

9B.5.6.1 Legal Status

Federal: Not warranted
State: Species of Special Concern

Two Chinook Salmon ESUs are found in the Klamath basin, the Southern Oregon and Coastal (SOCC) ESU and the Upper Klamath and Trinity Rivers ESU. The former are fall-run fish that spawn in the mainstem of the lower Klamath River. The Upper Klamath and Trinity Rivers ESU contains fall-run, late fall-run, and spring-run fish that spawn in the Klamath and Trinity rivers upstream of the Trinity River's confluence with the Klamath. Although wild spring-run Chinook Salmon in the Klamath River system differ from fall-run Chinook Salmon genetically, as well as in terms of life history and habitat requirements (NRC 2004), all are included within this ESU (Myers et al. 1998). The following profile pertains only to the spring-run, and focuses on the South Fork Trinity River (SFTR), which is within the action area and supports one of the few remaining stocks of wild spring-run Chinook Salmon within the greater Klamath Basin (Van Kirk and Naman 2008). The SFTR is the largest undammed river remaining in California.

A status review in 1999 concluded that neither ESU warranted listing (NMFS 1999b). A petition to list the Upper Klamath and Trinity Rivers ESU was submitted to NMFS in January 2011 (CBD et al. 2011); in April 2011, NMFS announced that listing was not warranted. Of primary importance in their decision was their conclusion that the spring-run and fall-run Chinook Salmon in the basin constitute a single ESU (NMFS 2012). The genetic structure of Chinook Salmon populations in coastal basins (as opposed to the Central Valley) indicates that the spring- and fall-run life histories have evolved multiple times in different watersheds (Myers et al. 1998, Waples et al. 2004). Three hatchery stocks from the Iron Gate and Trinity River hatcheries are considered part of the ESU because they were founded using native, local stock in the watershed where fish are released (NMFS 2012).

9B.5.6.2 Distribution

The Upper Klamath and Trinity Rivers ESU includes all naturally spawned and hatchery populations of spring, fall, and late-fall runs of Chinook Salmon in the Klamath and Trinity rivers upstream of the confluence of the Klamath and Trinity rivers. Iron Gate Dam currently blocks upstream migration to historical spawning habitat on the Klamath River, and Lewiston Dam is likewise a barrier to upstream migration on the Trinity River.

9B.5.6.3 Life History and Habitat Requirements

General habitat requirements for Chinook Salmon are described earlier; the following describes life-history strategies and habitat requirements unique to the spring-run Chinook or of primary importance to its life history. Spring-run Chinook Salmon display a stream-type life-history strategy—adults migrate upstream while sexually immature, hold in deep cold pools over the summer, and spawn in late summer and early fall. Juvenile outmigration is highly variable, with some age 0+ juveniles outmigrating in their first spring, but others oversummering and then emigrating as yearlings the following spring.

Table 9B.10 illustrates life-history timing for spring-run Chinook Salmon in the South Fork Trinity River basin.

Table 9B.10 Life History Timing of Spring-run Chinook Salmon in the South Fork Trinity River

Life Stage	Jan		Feb		Mar		Apr		May		Jun		Jul		Aug		Sept		Oct		Nov		Dec	
Adult upstream migration in Klamath River[a]																								
Spawning in SFTR[b]																								
Incubation and alevin development																								
Fry emergence[c]																								
Age 0+ outmigration in SFTR[d, e]																								
Age 1+ outmigration in SFTR[d, f]						?	?	?	?	?	?													
Ocean entry (yearlings)																								

Sources:
a. Snyder 1931; Strange 2008
b. State Coastal Conservancy 2009
c. West et al. 1990
d. Dean 1994, 1995
e. It is not possible to differentiate between fall-run and spring-run juveniles; therefore, exact timing for the spring run is unknown and may differ from the fall run.
f. Occurs in the spring after spawning; exact timing unknown.

Period of activity
Period of peak activity

9B.5.6.3.1 Adult Upstream Migration, Holding, and Spawning

Adults spawn from September through early November in the South Fork Trinity River (State Coastal Conservancy 2009).

Within the SFTR watershed, spring-run Chinook Salmon spawning takes place primarily between Hitchcock Creek and the East Fork of the SFTR on the mainstem SFTR, in Plummer Creek, in the mainstem of Hayfork Creek and the lower reaches of Salt and Tule creeks (USFS 2001a, Reclamation 1994), and possibly Big Creek (Chilcote et al. 2012). The East Fork of Hayfork Creek is used as summer holding habitat by adults, according to USFS (2001b), and adults have been observed during August in the lower SFTR below Surprise Creek and below Mule Bridge (USFS 2011).

9B.5.6.3.2 Egg Incubation and Alevin Development

Emergence takes place from March until early June (West et al. 1990).

9B.5.6.3.3 Juvenile Rearing and Outmigration

Rearing in the SFTR basin takes place in the mainstem SFTR between Hitchcock Creek and the East Fork of the SFTR (USFS 2001a). This area was noted to be an oversummering area by USFS (2001a). Rearing also takes place in Plummer Creek (USFS 2001a).

Juvenile spring-run Chinook Salmon of the Upper Klamath and Trinity Rivers ESU generally remain in fresh water for a year or more. On the South Fork Trinity River, outmigration occurs in late April and May with a peak in May (Dean 1994, 1995); however, it is not possible to differentiate between spring and fall juveniles, so spring-run outmigration timing may differ somewhat from the fall run. Age-1 juveniles (Type III) have been found to outmigrate from the South Fork Trinity River during the following spring (Dean 1994, 1995).

9B.5.6.4 Population Trends

A review by Williams et al. (2011) of Myers et al. (1998) and DFG (1965) estimates historical abundance of the entire ESU (both spring and fall runs) at approximately 130,000 adults for 1912, evenly split between the Klamath and Trinity rivers (NMFS 2012). Since the review by Myers et al. (1998) was published, there apparently has been little change in abundance, population trends, or population growth rates (Williams et al. 2011), except for two of the three spring-run populations that were evaluated, one of which was the South Fork Trinity River, where abundance is low relative to historical estimates (NMFS 2012). The spring run likely dominated numbers of Chinook Salmon in the South Fork Trinity River historically (Reclamation 1994). Declines in the SFTR basin have been attributed to increased sediment delivery and destruction of riparian vegetation from a history of logging and road-building in the characteristically unstable soils found there (USFS 1996; Trinity County Resource Conservation District 2003), effects of the 1964 flood (Reclamation 1994), major wildfire events (e.g., 1987, 2008), mining, and livestock grazing (Chilcote et al. 2012), as well as water withdrawals and clearing of large woody debris from stream channels (USFS 1994). Water withdrawals for domestic and agricultural uses appear to be a major factor influencing fish production in Hayfork Creek (Reclamation 1994), a major tributary to the SFTR that is located in more stable soils. Temperatures in the SFTR and Hayfork Creek are believed to be limiting spring-run populations in the SFTR and Hayfork Creek (Chilcote et al. 2012), thus climate change could result in future declines (Van Kirk and Naman 2008). NMFS suspects that dams on the mainstem Klamath and Trinity rivers caused as much as 90 percent of the spring-run Chinook Salmon decline (USFS 2001b). These dams may affect Chinook Salmon populations by altering natural seasonal flow patterns and temperatures, which affects habitat as well as behavioral cues for life-history transitions (USFS 1999). Escapement of spring-run Chinook Salmon to the Trinity River is shown in Figure 9B.1.

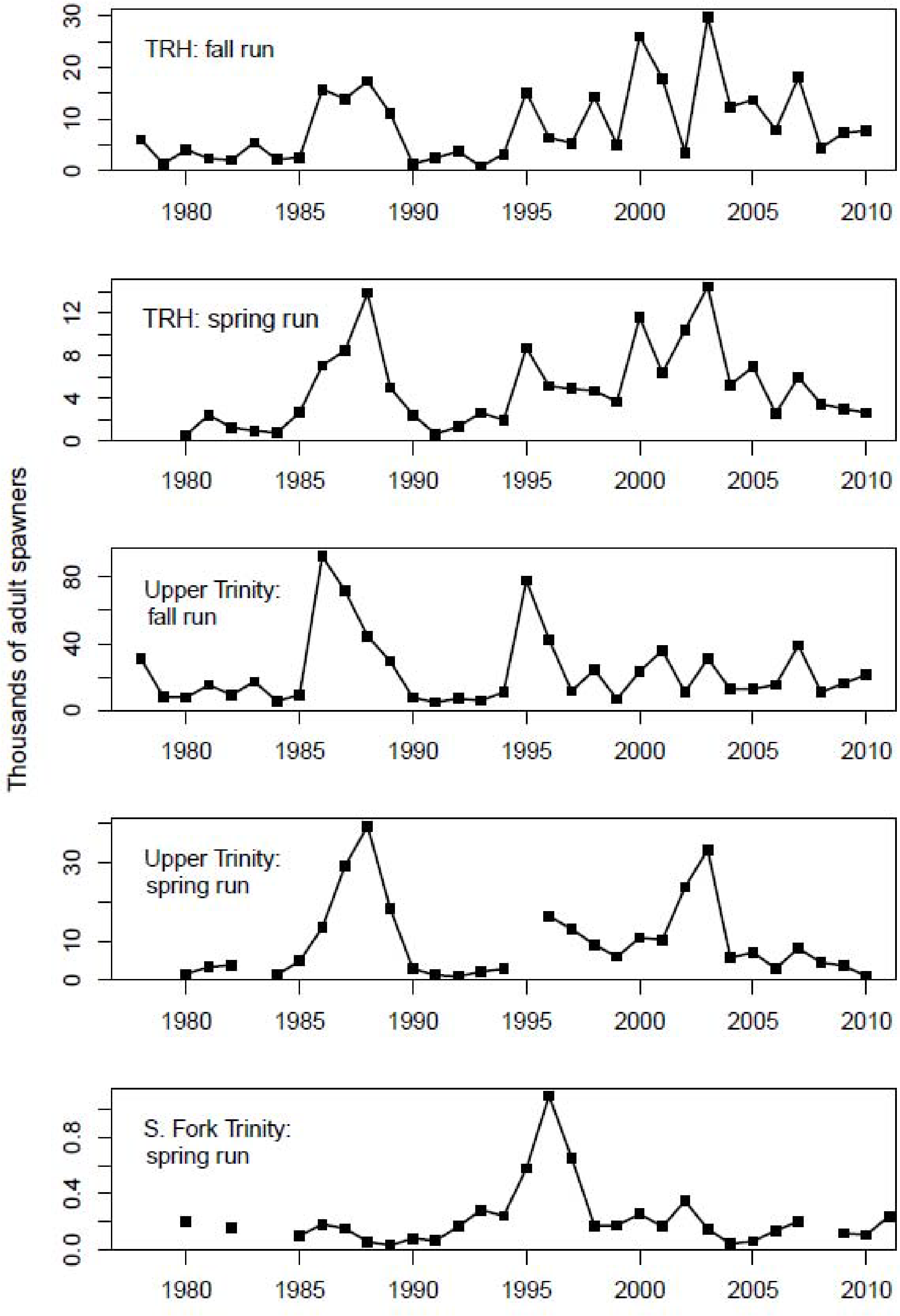

Figure 9B.1 Spring-run Chinook Salmon Escapement in the Trinity River, 1980–2010 (from Williams et al. 2011)

9B.5.6.5 Hatchery Influences

Hatchery stocking using native Chinook Salmon began in 1917 and includes both fall- and spring-run fish. There are two hatcheries in the basin: Iron Gate Hatchery on the Klamath River and Trinity River Hatchery on the Trinity River. Chinook Salmon released from Iron Gate Hatchery are all fall-run fish (NRC 2004), while the Trinity River Hatchery produces both spring- and fall-run Chinook Salmon. Approximately 10.3 million fingerling and yearling Chinook Salmon are released annually from these two hatcheries (NMFS 2012). The stocks from these hatcheries were founded from local, native fish and are genetically similar to local, natural populations; they are considered part of the same ESU by NMFS (NMFS 2012).

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9B.6 Central Valley Steelhead (*Oncorhynchus mykiss*)

9B.6.1 Legal Status

Federal: Threatened; Designated Critical Habitat
State: None

NMFS listed the Central Valley Steelhead ESU as threatened under the Federal ESA in 1998 (NMFS 1998). In 2004, NMFS proposed that all west coast steelhead ESUs be reclassified to DPSs and proposed to retain Central Valley Steelhead as threatened. In January 2006, after a status review (Good et al. 2005), NMFS issued its final decision to retain the status of Central Valley Steelhead as threatened (NMFS 2006).

Designated critical habitat for Central Valley Steelhead includes stream reaches of the American, Feather, Yuba, and Bear rivers and their tributaries and tributaries of the Sacramento River including Deer, Mill, Battle, Antelope, and Clear creeks in the Sacramento River basin; the Mokelumne, Calaveras, Stanislaus, Tuolumne, and Merced rivers in the San Joaquin River basin; and portions of the Sacramento and San Joaquin rivers. Designated critical habitat in the Delta includes portions of the Delta Cross Channel Yolo Bypass, Ulatis Creek, and portions of the network of channels in the Sacramento River portion of the Delta as well as portions of the San Joaquin, Cosumnes, and Mokelumne rivers and portions of the network of channels in the San Joaquin portion of the Delta.

The DPS includes naturally spawned anadromous *O. mykiss* (steelhead) populations below natural and manmade impassable barriers in the Sacramento and San Joaquin rivers and their tributaries, excluding steelhead from San Francisco and San Pablo bays and their tributaries and those from two artificial propagation programs: the Coleman Nimbus Fish Hatchery and Feather River Hatchery steelhead hatchery programs.

NMFS considered including resident *O. mykiss* in listed steelhead DPSs in certain instances, including (1) where resident *O. mykiss* have the opportunity to interbreed with anadromous fish below natural or artificial barriers, or (2) where resident fish of native lineage once had the ability to interbreed with anadromous fish but no longer do because they are above artificial barriers and are considered essential for the recovery of the DPS (NMFS 1998). However, USFWS, which under the ESA has authority over resident fish, concluded that behavioral forms of *O. mykiss* can be regarded as separate DPSs and that lacking evidence that resident Rainbow Trout need ESA protection, only anadromous forms should be included in the DPS and listed under the ESA (NMFS 1998). USFWS also did not believe that steelhead recovery would rely on the intermittent exchange of genetic material between resident and anadromous forms. In the final rule, the listing includes only the anadromous form of *O. mykiss*.

However, NMFS considers all *O. mykiss* that have access to the ocean (including resident Rainbow Trout) to potentially be steelhead and will treat these fish as steelhead because (1) resident fish can produce anadromous offspring, and (2) it is difficult or impossible to distinguish between juveniles of the different forms. Adult resident Rainbow Trout in Central Valley streams are often larger than Central Valley Steelhead. Several sources indicate that resident trout in the Central Valley commonly exceed 16 inches (406 mm) in length. Cramer et al. (1995) reported that resident Rainbow Trout in Central Valley rivers grow longer than 20 inches (508 mm). Hallock et al. (1961) observed resident trout in the upper Sacramento River upstream of the Feather River that were 14 to 20 inches (356 to 508 mm) in length. Also, at Coleman National Fish Hatchery, USFWS found about 15 percent overlap in size distribution between resident and anadromous *O. mykiss* at a length of 22.8 inches (579 mm) (Cramer et al. 1995). Steelhead, therefore, have significant size overlap with resident Rainbow Trout in Central Valley rivers, and many resident adult trout will be considered by NMFS to be steelhead.

The following profiles focus on the anadromous form of the species because these are the most likely to be affected by the proposed action, and several have special status under the ESA.

9B.6.2 Distribution

Central Valley Steelhead are widely distributed throughout their range but are low in abundance, particularly in the San Joaquin River basin, and they continue to decline (NMFS 2003). Microchemical analyses of otoliths taken from *O. mykiss* in the San Joaquin River basin have verified that the anadromous form of this species occurs in low numbers in the San Joaquin River basin (Zimmerman et al. 2009).

9B.6.2.1 Historical Distribution

O. mykiss once occurred throughout the Central Valley, spawning in the upper reaches of tributaries to the Sacramento and San Joaquin rivers. Lindley et al. (2006) conducted geographic information system (GIS) habitat modeling to estimate the amount of suitable habitat to support *O. mykiss* populations in the Central Valley, and their results suggest that steelhead were widely distributed throughout the Sacramento River basin, but relatively less abundant in the San Joaquin River basin due to natural barriers to migration. Yoshiyama et al. (1996) conducted a review of historical sources to document the historical distribution of Chinook Salmon in the Central Valley, which can be used to infer historical distribution of steelhead. The assumption that steelhead distribution in the Sacramento River basin overlapped with, and was likely more extensive than, spring-run Chinook distribution under historical conditions has been supported by studies conducted in the Klamath-Trinity River basin (Bureau of Indian Affairs 1985, Voight and Gale 1998). Yoshiyama et al. (1996) concluded that, because steelhead upstream migration occurs during high flows, their leaping abilities are superior to those of Chinook Salmon, and they have less restrictive spawning gravel criteria. Steelhead in the Sacramento River basin "could have used at least hundreds of miles of smaller tributaries not accessible to the earlier-spawning salmon." The model created by Lindley et al. (2006) estimates that 80 percent of historically accessible habitat for Central Valley Steelhead is now behind impassable dams; this estimate is supported by other research into steelhead and Chinook Salmon habitat loss in the Central Valley (Clark 1929; Yoshiyama et al. 1996, 2001).

9B.6.2.2 Current Distribution

Steelhead distribution in Central Valley drainages has been greatly reduced (McEwan and Jackson 1996). Steelhead are now primarily restricted to a few remaining free-flowing tributaries and to stream reaches below large dams, although a few steelhead may also spawn in intermittent streams during wet years. Naturally spawning steelhead populations have been found in the upper Sacramento River and tributaries below Keswick Dam; Mill, Deer, and Butte creeks; and the Feather, Yuba, American, and Mokelumne rivers (CMARP 1998). However, the records of naturally spawning populations depend on fish monitoring programs. Recent implementation of monitoring programs has found steelhead in additional streams, such as Auburn Ravine, Dry Creek, and the Stanislaus River. It is possible that naturally spawning populations exist in many other streams but are undetected because of the lack of monitoring or research programs. Although impassable dams prevent resident Rainbow Trout from emigrating, populations with steelhead ancestry may still exist above some dams (Reclamation 2008).

In the Sacramento River basin, populations of *O. mykiss* are known to spawn in the upper Sacramento, Yuba, Feather, and American rivers and in Deer, Mill, and Butte creeks. Saeltzer Dam was removed from Clear Creek in 2000, granting easier access to habitats in the higher-elevation canyon reaches. Though improved access may have opened up suitable spawning and rearing habitat for steelhead, it is not clear if steelhead have colonized Clear Creek since removal of the dam. A summary of recent distribution information for steelhead in Sacramento River tributaries in Good et al. (2005) shows that steelhead are widespread in accessible streams, if not abundant.

Research and monitoring on steelhead are limited in comparison with Chinook Salmon, so there is little specific information about the status and trend of the species and how adults and juveniles use habitats in the mainstem river and the Bay-Delta estuary. Though the upper reaches of the Sacramento River support a spawning population of resident Rainbow Trout, the mainstem river habitat used by the species is atypical for steelhead, which usually spawn in higher elevation, steeper, and narrower channels. Management of the species is also complicated by its polymorphism, with individuals being capable of exhibiting either a resident (Rainbow Trout) or an anadromous (steelhead) life history.

9B.6.3 Life History and Habitat Requirements

Steelhead generally exhibit a more flexible life history strategy than Chinook Salmon, and the habitat requirements of juvenile steelhead differ from those of juvenile Chinook Salmon. Unlike Chinook Salmon, steelhead can be iteroparous—that is, they can survive spawning, return to the ocean, and migrate into fresh water to spawn again. Post-spawning adults are known as kelts. In general, there are two types of steelhead: winter steelhead and summer steelhead. Winter steelhead are of the ocean-maturing reproductive ecotype, becoming sexually mature during their ocean phase and spawning soon after their arrival at the spawning grounds. Adult summer steelhead are of the stream-maturing type, which enter their natal streams and spend several months holding and maturing in fresh water before spawning. Central Valley Steelhead are predominantly winter steelhead, and this section describes the life history and habitat requirements of winter steelhead.

Table 9B.11 illustrates aspects of the life-history timing of Central Valley Steelhead.

Table 9B.11 Life-History Timing of Central Valley Steelhead

Life Stage	Jan		Feb		Mar		Apr		May		Jun		Jul		Aug		Sept		Oct		Nov		Dec	
Adult Upstream Migration[a]	▒	▒	▒	▒	▒	▒	▒	▒	▒	▒	▒	▒	▒	▒	▓	▓	▓	█	▓	▓	▓	▓	▒	▒
Spawning in Mainstem Sacramento River Downstream of Keswick Dam[b]	▓	▓	▓	▓	▓	▓	?																?	?
Incubation and Alevin Development[c]	▒	▒	▒	▒	▒	▒	▒	▒																
Fry Emergence[c]			▒	▒	▒	▒	▒	▒	▒	▒	▒													
Age 0+ Outmigration from Upper Sacramento River[b]	▓	▓	▓	▓	▓	▓	▒	▒	▒	▒	▒	▒									▒	▒	▒	▒
Age 1+ Outmigration through the Delta[d]	▒	▒	▓	▓	▓	▓	▓	▓	▒	▒	▒	▒									▒	▒	▒	▒

Notes:
a. Bailey 1954, Hallock et al. 1961, McEwan 2001
b. Reclamation 2004
c. Based on timing of spawning
d. Based on fish facility salvage data (Reclamation 2004)

▒ Period of activity
▓ Period of peak activity

9B.6.3.1 Adult Migration and Spawning

Central Valley Steelhead generally leave the ocean and migrate upstream from August through March (Busby et al. 1996), In the Sacramento River, steelhead migrate upstream nearly every month of the year, with the bulk of migration from August through November and the peak in late September (Bailey 1954, Hallock et al. 1961, McEwan 2001). Spawning in the upper Sacramento River generally occurs from December through April (Newton and Stafford 2011).

The majority of steelhead in the mainstem Sacramento River spawn downstream of Keswick Dam (RM 302), with peak spawning from January through March when water temperatures throughout much of the Sacramento River are suitable to support egg incubation and emergence. The highest-density spawning within the mainstem is likely in the upstream portion of this area near Redding; however, the downstream extent of spawning is likely determined by the location of suitable water temperatures to support summer rearing of 0+ juveniles, which lack the swimming ability to move significant distances upstream to follow the upstream retreat of cold water in summer. Most Sacramento River steelhead are believed to spawn in the tributary streams. The progeny of adults that construct redds downstream of locations with suitable water temperatures in summer likely suffer high rates of mortality and contribute little to the population.

Steelhead migrate and spawn during high flows when observations and sampling are difficult (McEwan 2001). They may have a spawning distribution similar to late fall-run Chinook Salmon in that the juveniles of both species oversummer at least once before outmigration, so redds must be located where summer water temperatures can support summer rearing. The downstream extent of late fall-run Chinook Salmon spawning is generally near Ball’s Ferry Bridge (RM 276) in most years. Steelhead generally have higher thermal tolerances than Chinook Salmon (Moyle 2002), so steelhead spawning may extend slightly farther downstream.

Under historical conditions, steelhead likely spawned in much higher-gradient reaches in the Sacramento River and its tributaries, as do steelhead in other portions of their range. Steelhead are common in reaches with gradients of less than 6 percent (Burnett 2001, Harvey et al. 2002, Hicks and Hall 2003) and occur in some systems in reaches of up to 12 percent and more (Engle 2002). Though steelhead will spawn in mainstem river channels, it is unlikely that they spawned in the reach of the mainstem Sacramento River below Keswick Dam where they currently spawn because summer water temperatures in this reach were likely too high to support oversummering by juveniles.

As with Chinook Salmon, steelhead spawn in areas with suitable gravel and hydraulics. Work by Bovee (1978) found that steelhead prefer water depths of 14 inches (36 cm) for spawning, with a range between 6 and 24 inches (15 and 61 cm), and water velocities of 2 feet/second (61 cm/second), with a range of 1 to 3.6 feet/second (30 to 110 cm/second), which is similar to the hydraulic conditions preferred by Chinook Salmon in the Central Valley. Steelhead generally prefer to spawn in gravels, with optimal grain sizes ranging between

0.6 and 10 cm (6 and 102 mm) (Bjornn and Reiser 1991). For comparison, grain sizes used by spawning Chinook range from a D_{50} of 0.43 inch (10.8 mm) (Platts et al. 1979) to a D_{50} of 3.1 inches (78.0 mm) (Chambers et al. 1954, 1955).

Research in more northerly populations suggests that optimal spawning temperatures range from 39 to 52°F (4 to 11°C), with egg mortality at water temperatures above 56°F (13°C) (Hooper 1973, Bovee 1978, Reiser and Bjornn 1979, Bell 1986). More research is needed to understand the specific temperature tolerances of steelhead in the Central Valley and southern portions of their range. There is evidence that different strains of *O. mykiss* may have different thermal tolerances at the egg and embryo stage (Myrick and Cech 2001).

As stated above, steelhead can survive spawning, return to the ocean, and migrate into fresh water to spawn again. Although some kelts have been documented in the Sacramento River, there are probably few repeat spawners in the Sacramento River population (Reclamation 2004).

9B.6.3.2 Fry and Juvenile Rearing

Fry emergence is influenced by water temperature, but hatching generally requires 4 weeks, with another 4 to 6 weeks in the gravels before emergence. After emerging, steelhead fry typically disperse to shallow (<14 inches [36 cm]), low-velocity near-shore areas such as stream margins and low-gradient riffles and will forage in open areas lacking instream cover (Hartman 1965, Everest et al. 1986, Fontaine 1988). Everest and Chapman (1972) found that juvenile steelhead of all sizes most often chose territories over large-sized substrates. As they increase in size in late summer and fall, they increasingly use areas with cover and show a preference for higher-velocity, deeper mid-channel areas near the thalweg (Hartman 1965, Everest and Chapman 1972, Fontaine 1988). Bovee (1978) reports that fry prefer water depths ranging between 10 inches (25 cm) and 20 inches (51 cm) and water temperatures ranging between 45°F (7°C) and 60°F (16°C). Age 0+ steelhead have been relatively abundant in backwater pools and often live in the downstream ends of pools in late summer (Bisson et al. 1988, Fontaine 1988).

Steelhead fry may establish and defend territories soon after emerging (Shapovalov and Taft 1954). Fry and juvenile steelhead that are unsuccessful in establishing a territory may suffer density-dependent mortality or be displaced downstream where they may suffer higher rates of mortality from predation, entrainment, or elevated water temperatures (Dambacher 1991, Peven et al. 1994, Reedy 1995). Keeley (2001) found that increased competition between juvenile steelhead, caused by higher fish densities or lower food densities, caused increased mortality, lower or more variable growth rates, and emigration of smaller fish. Downstream dispersal due to density dependence or high flows in rearing habitat does not necessarily increase mortality where there is suitable habitat downstream (Kahler et al. 2001). Downstream dispersal to larger stream reaches for further rearing prior to smolting appears common in many systems (Bjornn 1978, Loch et al. 1985, Leider et al. 1986, Dambacher 1991).

9B.6.3.3 Summer Rearing

Summer habitat can generally be assumed to be more limiting for age 1+ and 2+ juvenile steelhead than for age 0+ in many streams. Older age classes of juvenile steelhead (ages 1+ and 2+) prefer deeper water in summer than fry and show a stronger preference for pool habitats, especially deep pools near the thalweg with ample cover, as well as higher-velocity rapid and cascade habitats (Bisson et al. 1982, 1988; Dambacher 1991). Dambacher (1991) observed that most 1+ steelhead in the Steamboat Creek watershed of the North Umpqua River in Oregon were concentrated in mainstem reaches with relatively deep riffles and large substrates. Age 1+ fish typically feed in pools, especially scour and plunge pools (Fontaine 1988, Bisson et al. 1988). Age 1+ steelhead appear to avoid secondary channel and dammed pools, glides, and low-gradient riffles with mean depths less than 7.8 inches (20 cm) (Fontaine 1988, Bisson et al. 1988, Dambacher 1991). Beecher et al. (1993) reported that juvenile steelhead longer than 3 inches (75 mm) avoided areas less than 6 inches (15 cm) deep. Reedy (1995) indicates that age 1+ steelhead especially prefer high-velocity pool heads, where food resources are abundant, and pool tails, which provide optimal feeding conditions in summer due to lower energy expenditure requirements than the more turbulent pool heads. Fast, deep water, in addition to optimizing feeding versus energy expenditure, provides greater protection from avian and terrestrial predators (Everest and Chapman 1972).

9B.6.3.4 Winter Rearing

For juvenile steelhead to survive winter, they must avoid predation and high flows. The higher-gradient reaches typically used for spawning by steelhead (generally >3 percent) are often confined and characterized by coarse substrate that is immobile at all but the highest flows. Juvenile steelhead often use the interstitial spaces between cobbles and boulders as cover from high water velocity and presumably to avoid predation (Bjornn 1971, Hartman 1965, Bustard and Narver 1975, Swales et al. 1986, Everest et al. 1986, Grunbaum 1996). Age 0+ steelhead can use shallower habitats and can find interstitial cover in gravel-size substrates, while age 1+ or 2+ steelhead, because of their larger size, need coarser cobble/boulder substrate for cover (Bustard and Narver 1975; Bisson et al. 1982, 1988; Fontaine 1988; Dambacher 1991). Bustard and Narver (1975) reported that 1+ steelhead prefer water deeper than 17.5 inches (45 cm) in winter, while age 0+ steelhead often occupy water less than 5.8 inches (15 cm) deep and are rarely found at depths over about 23.4 inches (60 cm). In winter, age 1+ steelhead typically stay within the area of streambed that remains inundated at summer low flows, while age 0+ fish frequently overwinter beyond the summer low flow perimeter along the stream margins (Everest et al. 1986). Consequently, winter rearing habitat for age 1+ and 2+ juvenile steelhead is assumed to be more limiting than for age 0+ juveniles.

9B.6.3.5 Length of Stream Residence

Juvenile steelhead typically rear in fresh water from 1 to 3 years before outmigrating (McEwan and Jackson 1996). The majority of returning adult steelhead in the Central Valley have spent 2 years in fresh water before emigrating to the ocean (McEwan 2001). A scale analysis conducted by Hallock et al. (1961) indicated that 70 percent emigrated after 2 years, 29 percent after 1 year, and 1 percent after 3 years in fresh water. Juvenile emigration from the upper Sacramento River occurs between November and late June, with a peak between early January and late March (Reclamation 2004).

9B.6.3.6 Bay-Delta Residence

The Delta serves as an adult and juvenile migration corridor, connecting inland habitat to the ocean. The Delta may also serve as a nursery area for juvenile steelhead (McEwan and Jackson 1996); however, much is unknown regarding historical and current role of the Delta as steelhead nursery habitat. In coastal populations of winter steelhead, it is common for juvenile steelhead to migrate downstream at age 1+ and rear in the estuary for an additional year before smolting. Based on fish facility salvage data, most steelhead move through the Delta from November through June, with the peak salvage during February, March, and April. The majority of steelhead salvaged range from 175 to 325 mm, with the most common size ranging from 226 to 250 mm. Some of the age 1+ steelhead captured in rotary screw traps at RBDD, GCID, and Knights Landing may continue rearing for another year before entering the ocean. There may be some areas of the Bay-Delta estuary where summer water temperatures are moderated by tidal action so that steelhead 1+ migrants are able to rear throughout summer (Reclamation 2008).

9B.6.4 Population Trends

Construction of large dams in the Central Valley had great impact on *O. mykiss* populations because it eliminated access to nearly 80 percent of historical spawning and rearing habitat (Lindley et al. 2006). Construction of Shasta and Keswick dams eliminated access to many upstream tributaries (e.g., McCloud River, Pit River, and Sacramento River) that provided the cold water temperatures required for year-round rearing by steelhead. Dam construction also landlocked potentially anadromous *O. mykiss* populations in the upper watershed, forcing them to adopt a resident life history strategy (McEwan 2001).

In general, the majority of Central Valley Steelhead are confined to nonhistorical spawning and rearing habitat below impassable dams, but the existing spawning and rearing habitat can sustain steelhead at current population levels. In addition, monitoring data indicate that much of the anadromous form of the species is hatchery supported. Also, a strong resident component to the population (Rainbow Trout) interacts with and produces both resident and anadromous offspring.

In general, steelhead stocks throughout California have declined substantially. McEwan and Jackson (1996) reported that the adult population of steelhead in California was approximately 250,000, less than half the population that existed in the 1960s (McEwan and Jackson 1996). In the Central Valley, approximately 1 to 2 million adult steelhead may have returned annually prior to 1850, as based on historical Chinook Salmon abundance (McEwan 2001, NMFS 2006). In the Sacramento River basin, the average run size of steelhead in the 1950s was estimated to be approximately 20,540 adults (McEwan and Jackson 1996). In contrast, escapement estimates in 1991 and 1992 were less than 10,000 adults, less than half of the run size in the 1950s (McEwan and Jackson 1996). Similarly, counts of wild steelhead at RBDD declined from an average annual run size of 12,900 in the late 1960s to 1,100 adults in the 1993–94 season (McEwan and Jackson 1996). The most recent 5-year average for steelhead spawning upstream of RBDD is less than 2,000 adults (Good et al. 2005). NMFS (2006) notes that escapement estimates have not been made for the area upstream of RBDD since the mid-1990s and that estimates of abundance are derived from extrapolation of incidental catch of outmigrating juvenile steelhead captured as part of the midwater-trawl sampling for juvenile Chinook Salmon at Chipps Island, downstream of the confluence of the Sacramento and San Joaquin rivers.

Populations of naturally spawned Central Valley Steelhead have declined and are composed predominantly of hatchery fish. The California Fish and Wildlife Plan of 1965 estimated the combined annual run size for Central Valley and San Francisco Bay tributaries to be about 40,000 during the 1950s (DFG 1965). The spawning population during the mid-1960s for the Central Valley basin was estimated at about 27,000 (DFG 1965). These numbers likely consisted of both hatchery and wild steelhead. McEwan and Jackson (1996) estimated the annual run size for the Central Valley basin to be less than 10,000 adults by the early 1990s. Much of the abundance data since the mid-1960s were obtained by visual fish counts at the RBDD fish ladders when gates were closed during much of the steelhead migration season. Current abundance estimates are not available for naturally spawned fish since RBDD gate operations were changed, so the extent to which populations have changed following the 1987−94 drought is unknown. NMFS' (2003) status review estimated the Central Valley Steelhead population at less than 3,000 adults.

9B.6.5 Hatchery Influence

Reclamation funds the operation of Coleman Hatchery, Livingston Stone Hatchery, Nimbus Hatchery, and Trinity River Hatchery. DWR funds the operation of the Feather River Hatchery. USFWS operates Coleman and Livingston Stone hatcheries, and DFW operates Feather River, Nimbus, and Trinity hatcheries. These hatcheries are operated to mitigate for the anadromous salmonids that would be produced by the habitat if not for the dams on each respective river. Reclamation and DWR have discretion over how the hatcheries are operated, but generally leave operational decisions on how to meet mitigation goals to the operating agency (Reclamation 2008).

Hatchery production of steelhead is large compared to natural production, based on the Chipps Island trawl data (Good et al. 2005). The bulk of hatchery releases in the Central Valley occurs in the Sacramento River basin. An analysis of steelhead captures from trawl data by Nobriga and Cadrett (2001) indicated that hatchery steelhead composed 63 to 77 percent of the steelhead catch. Steelhead stocks at the Mokelumne River Hatchery and Nimbus Hatchery on the American River are not part of the Central Valley Steelhead DPS because of the source of broodstock used and genetic similarities to Eel River stocks (Good et al. 2005).

Genetic analysis indicated steelhead from the American River (collected from both the Nimbus Hatchery and the American River) are genetically more similar to Eel River steelhead (Northern California ESU) than other Central Valley Steelhead stocks. Eel River steelhead were used to found the Nimbus Hatchery stock. Mokelumne River Rainbow Trout (hatchery produced and naturally spawned) are genetically most similar to Mount Shasta Hatchery trout, but also show genetic similarity to the Northern California ESU (Nielsen 1997). Nielsen et al. (2005) found American River steelhead to be genetically different from other Central Valley stocks.

9B.6.6 References

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9B.7 Klamath Mountains Province Steelhead (*Oncorhynchus mykiss*)

9B.7.1 Legal Status

Federal: Not warranted
State: Species of Special Concern

A status review in 2001 (NMFS 2001) concluded that the Klamath Mountains Province Steelhead DPS was not in danger of extinction or likely to become so in the foreseeable future; therefore, it was not warranted for listing as threatened or endangered. This conclusion was based on population estimates and a finding that the genetic risk from naturally spawning hatchery fish was lower than estimated in previous reviews, as well as consideration of ongoing and proposed conservation efforts for anadromous salmonids in the basin (NMFS 2001).

The Klamath Mountains Province Steelhead DPS contains both summer and winter runs. Moyle (2002) describes steelhead in the Klamath Basin as having a summer run and a winter run. Some divide the winter run into fall and winter runs (Barnhart 1994, Hopelain 1998, USFWS 1998, Papa et al. 2007). In this section, winter steelhead refers to steelhead returning from fall through winter, except in cases when the distinction is pertinent to the discussion. The following summary focuses on steelhead in the Trinity River, which is within the area potentially affected by the proposed action, and on the mainstem Klamath in terms of potential effects on its role as a migration corridor for the steelhead runs.

9B.7.2 Distribution

Based on escapement data, approximately 55 percent of the summer run spawn in the Trinity River and other lower-elevation tributaries to the Klamath River. The Trinity, Scott, Shasta, and Salmon rivers are important spawning streams for the winter run.

Historically, steelhead probably ascended Clear Creek past the French Gulch area, but access to the upper basin was blocked by Whiskeytown Dam in 1964 (Yoshiyama et al. 1996). Operation of Whiskeytown Dam can produce suitable cold-water habitat downstream to Placer Road Bridge depending on flow releases (DFG 1998). McCormick-Saeltzer Dam, which limited steelhead migrations through ineffective fish ladders, was removed in 2000, allowing steelhead potential access to good habitat up to Whiskeytown Dam. USFWS has conducted snorkel surveys targeting spring-run Chinook (May through September) since 1999. Steelhead/rainbow are enumerated and separated into small, medium, and large (>22 inches) during these surveys, but because the majority of the steelhead run is unsurveyed, no spawner abundance estimates have been attempted (Reclamation 2008). Redd counts conducted during the 2001-02 run found that most spawning occurred upstream, near Whiskeytown Dam. Because of the large resident rainbow population, no steelhead population estimate could be made (Reclamation 2008). A remnant "landlocked" population of Rainbow Trout with steelhead ancestry may exist in Clear Creek above Whiskeytown Dam (Reclamation 2008).

9B.7.3 Life History and Habitat Requirements

General habitat requirements for steelhead are described in the Central Valley Steelhead profile; the following describes life history strategies and habitat requirements unique to steelhead of the Upper Klamath Mountains Province DPS or of primary importance to its life history. Both winter and summer runs of steelhead are included in the DPS. Winter steelhead become sexually mature during their ocean phase and spawn soon after arriving at their spawning grounds. Adult summer steelhead enter their natal streams and spend several months holding and maturing in fresh water before spawning. Throughout the entire year, at least one of the diverse life stages can be found present in the river (Israel 2003). As with the Central Valley DPS, this DPS is composed predominantly of winter steelhead.

9B.7.3.1 Winter Run

Winter steelhead adults generally enter the Klamath River from July through October (fall run) and from November through March (winter run) (USFWS 1998). Winter steelhead primarily spawn in tributaries from January through April (USFWS 1998), with peak spawn timing in February and March (ranging from January to April) (NRC 2004). Adults may repeat spawning in subsequent years after returning to the ocean. Half-pounders typically use the mainstem Klamath River until leaving the following March (NRC 2004), although they also use larger tributaries such as the Trinity River (Dean 1994, 1995).

Fry emerge in spring (NRC 2004), with fry observed in outmigrant traps in Bogus Creek and Shasta River from March through mid-June (Dean 1994). Age-0+ and 1+ juveniles have been captured in outmigrant traps in spring and summer in tributaries to the Klamath River above Seiad Creek (DFG 1990a, 1990b). These fish are likely rearing in the mainstem or non-natal tributaries before leaving as age-2+ outmigrants.

Juvenile outmigration primarily occurs between May and September with peaks between April and June, although smolts are captured in the estuary as early as March and as late as October (Wallace 2004). Most adult returns (86 percent) originate from fish that smolt at age 2+, in comparison with only 10 percent for age-1 juveniles and 4 percent for age 3+ juveniles (Hopelain 1998).

Similar limiting factors listed for summer steelhead also affect winter steelhead populations, including degraded habitats, decreased habitat access, fish passage, predation, and competition (for more species information see USFWS 1998, NRC 2004, and Wallace 2004).

9B.7.3.2 Summer Run

Summer steelhead adults enter and migrate up the Klamath River from March through June while sexually immature (Hopelain 1998), then hold in cooler tributary habitat until spawning begins in December (USFWS 1998).

Juvenile summer steelhead in the Klamath Basin may rear in fresh water for up to 3 years before outmigrating. Although many juveniles migrate downstream at age 1+ (Scheiff et al. 2001), those that outmigrate to the ocean at age 2+ appear to have the highest survival (Hopelain 1998). Juveniles outmigrating from tributaries at age 0+ and age 1+ may rear in the mainstem or in non-natal tributaries (particularly during periods of poor water quality) for 1 or more years before reaching an appropriate size for smolting. Age-0 juvenile steelhead have been observed migrating upstream into tributaries, off-channel ponds, and other winter refuge habitat in the lower Klamath River. Juvenile outmigration can occur from spring through fall. Smolts are captured in the mainstem and estuary throughout fall and winter (Wallace 2004), but peak smolt outmigration normally occurs from April through June, based on estuary captures (Wallace 2004). Temperatures in the mainstem are generally suitable for juvenile steelhead, except during summer, especially upstream of Seiad Valley.

9B.7.4 Population Trends

Long-term data are not available to evaluate Klamath River steelhead population trends. DFG (1965) estimated a basinwide annual run size of 283,000 adult steelhead (spawning escapement + harvest). Busby et al. (1994) reported winter steelhead runs in the basin to be 222,000 during the 1960s. Steelhead spawning surveys on tributaries to the mainstem Trinity River were conducted in 1964, 1971, 1972, and 1974 to monitor the effect of Lewiston Dam on steelhead populations. Hopelain (2001) used creel and gill net harvest data to estimate the winter-run steelhead population at 10,000 to 30,000 adults annually in the early

1980s. Spawning surveys were also conducted in South Fork Trinity River tributaries from 1989 to 1995 under DFW's Trinity River Project (Garrison 2000).

Population estimates of summer steelhead showed a steep decline during the 1990s (Reclamation 2008), but Koch (2001) reported increasing runs on the Klamath and Trinity rivers following the late 1990s.

9B.7.5 Hatchery Influence

Reclamation funds the operation of Coleman Hatchery, Livingston Stone Hatchery, Nimbus Hatchery, and Trinity River Hatchery. DWR funds the operation of the Feather River Hatchery. USFWS operates Coleman and Livingston Stone hatcheries, and DFW operates Feather River, Nimbus, and Trinity hatcheries. These hatcheries are operated to mitigate for the anadromous salmonids that would be produced by the habitat if not for the dams on each respective river. Reclamation and DWR have discretion over how the hatcheries are operated, but generally leave operational decisions on how to meet mitigation goals to the operating agency (Reclamation 2008).

NMFS (2001) reported that the Trinity River population is thought to contain a large percentage of hatchery origin spawners of mostly fall-run fish (20-70 percent).

9B.7.6 References

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9B.8 Southern Oregon/Northern California Coast Coho Salmon ESU (*Oncorhynchus kisutch*)

9B.8.1 Legal Status

Federal: Threatened
State: Threatened

Coho Salmon (*Oncorhynchus kisutch*) in the Trinity River are in the Southern Oregon/Northern California Coast Coho Salmon ESU and were listed as threatened under the ESA in 1997 (NMFS 1997) and threatened under the California Endangered Species Act in 2002. This ESU includes naturally spawning populations between Punta Gorda, California, and Cape Blanco, Oregon, which encompasses the Trinity and Klamath basins (NMFS 1997). Three artificial propagation programs are considered to be part of the ESU: the Cole Rivers Hatchery, Trinity River Hatchery, and Iron Gate Hatchery Coho Salmon programs. NMFS has determined that these artificially propagated stocks are no more than moderately diverged from the local natural populations. In addition, Coho Salmon in the Klamath Basin have been listed by the California Fish and Game Commission as threatened under the California Endangered Species Act (DFG 2002).

9B.8.2 Life History and Habitat Requirements

Coho Salmon exhibit a 3-year life cycle in the Trinity River and depend on freshwater habitat conditions year-round because they spend a full year residing in fresh water. Most Coho Salmon enter rivers between August and January, with some more northerly populations entering as early as June. Coho Salmon river entry timing is influenced by such factors as genetics, stage of maturity, river discharge, and access past the river mouth. Spawning is concentrated in riffles or in gravel deposits at the downstream end of pools with suitable water depth, velocity, and substrate size. Spawning in the Trinity River occurs mostly in

November and December. Coho eggs incubate from 35 to more than 100 days depending on water temperature and emerge from the gravel 2 to 7 weeks after hatching. Coho eggs hatch after an accumulation of 400 to 500 temperature units measured in degrees Celsius and emerge from the gravel after 700 to 800 temperature units. After emergence, fry move into areas out of the main current. As Coho grow, they spread out from the areas where they were spawned. During summer, juvenile Coho prefer pools and riffles with adequate cover such as large woody debris with smaller branches, undercut banks, and overhanging vegetation and roots.

Juvenile Coho Salmon overwinter in large mainstem pools, beaver ponds, backwater areas, and off-channel pools with cover such as woody debris and undercut banks. Most juvenile Coho Salmon spend a year in fresh water, with northerly populations spending 2 full years in fresh water. Coho in the Trinity River are thought be exclusively 3-year-life-cycle fish (1 year in fresh water). Because juvenile Coho remain in their spawning stream for a full year after emerging from the gravel, they are exposed to the full range of freshwater conditions. Most smolts migrate to the ocean between March and June, with most leaving in April and May. Coho Salmon typically spend about 16 to 18 months in the ocean before returning to their natal streams to spawn as 3- or 4-year-olds, age 1.2 or 2.2. Trinity River Coho are mostly 3-year-olds. Some precocious males, called jacks, return to spawn after only 6 months in the ocean.

Juvenile Coho Salmon in the Trinity River spend up to a full year in fresh water before migrating to the ocean. Their habitat preferences change throughout the year and are highly influenced by water temperature. During summer, when Coho are most actively feeding and growing, they spend more time closer to main channel habitats. Coho use slower water than steelhead or Chinook Salmon. Coho juveniles are more oriented to submerged objects, such as woody debris, while Chinook and steelhead select habitats in summer based largely on water movement and velocities, although the species are often intermixed in the same habitat. Juvenile Coho use the same habitats as pikeminnows, a possible reason that Coho are not present in Central Valley watersheds. Juvenile Coho would be vulnerable to predation from larger pikeminnows during warm-water periods. Pikeminnow do not occur in Southern Oregon/Northern California Coast coho streams. When the water cools in fall, juvenile Coho move farther into backwater areas or into off-channel areas and beaver ponds if available. There is often no water velocity in the areas inhabited by Coho during winter. These same off-channel habitats are often dry or unsuitable during summer because temperatures get too high.

Lewiston Dam blocks access to 109 miles of upstream habitat. Trinity River Hatchery produces Coho Salmon with a production goal of 500,000 yearlings to mitigate for the upstream habitat loss. Habitat in the Trinity River has changed since flow regulation with the encroachment of riparian vegetation restricting channel movement and limiting fry rearing habitat (Trush et al. 2000). According to the Trinity River Restoration Plan, higher peak flows are needed to restore attributes of a more alluvial river such as alternate bar features and more off-channel habitats. These are projected in the restoration plan to provide better rearing habitat for Coho Salmon than the dense riparian vegetation currently present. A number of restoration actions have been completed. A new flow schedule has provided higher spring releases to geomorphically maintain habitat. Physical habitat manipulations have been implemented providing better juvenile rearing in selected sites along the river.

9B.8.3 Population Trends

Coho Salmon were not likely the dominant species of salmon in the Trinity River before dam construction. However, Coho were widespread in the Trinity Basin ranging as far upstream as Stuarts Fork above Trinity Dam. Wild Coho in the Trinity Basin today are not abundant, and the majority of the fish returning to the river are of hatchery origin. An estimated 2 percent (200 fish) of the total Coho Salmon run in the Trinity River were composed of naturally produced Coho from 1991 through 1995 at a point in the river near Willow Creek (USFWS 1998). This, in part, prompted the threatened status listing in 1997. These estimates included a combination of hatchery produced and wild Coho. About 10 percent of the Coho were naturally produced since 1995.

9B.8.4 Hatchery Influences

The Trinity River portion of the Southern Oregon/Northern California Coast Coho Salmon ESU is predominately of hatchery origin. Termination of hatchery production of Coho Salmon at the Mad River and Rowdy Creek facilities has eliminated further potential adverse risks associated with hatchery releases from these facilities. Likewise, restrictions on recreational and commercial harvest of Coho Salmon since 1994 likely have had a positive impact on Coho Salmon adult returns.

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9B.9 Sacramento Splittail (*Pogonichthys macrolepidotus*)

9B.9.1 Legal Status

Federal: None
State: Species of Special Concern

USFWS listed Sacramento Splittail as a threatened species on March 10, 1999, because of the reduction in its historical range and because of the large population decline during the 1987-93 drought (USFWS 1996, 1999). On June 23, 2000, the Federal Eastern District Court of California found the final rule to be unlawful and on September 22, 2000, remanded the determination back to USFWS for a reevaluation of the final decision. After a thorough review, USFWS removed the Sacramento Splittail from the list of threatened species (USFWS 2003) and reaffirmed this decision in 2010 (USFWS 2010).

9B.9.2 Distribution

Sacramento Splittail are endemic to the Sacramento and San Joaquin River systems of California, including the Delta and the San Francisco Bay. Historically, splittail were found in the Sacramento River as far upstream as Redding, in the Feather River to Oroville, and in the American River upstream to Folsom. In the San Joaquin River, they were once documented as far upstream as Friant (Rutter 1908). Splittail are thought to have originally ranged throughout the San Francisco estuary, with catches reported by Snyder (1905) from southern San Francisco Bay and at the mouth of Coyote Creek.

In wet years, Sacramento Splittail have been found in the San Joaquin River as far upstream as Salt Slough (Saiki 1984, Baxter 1999, Brown and Moyle 1993, Baxter 2000) and in the Tuolumne River as far upstream as Modesto (Moyle 2002), where the presence of both adults and juveniles during wet years in the 1980s and 1990s indicated successful spawning.

When spawning, splittail can be found in the lower reaches of rivers and flooded areas. Otherwise they are primarily confined to the Delta, Suisun Bay, Suisun Marsh, the lower Napa River, the lower Petaluma River, and other parts of the San Francisco estuary (Meng et al. 1994, Meng and Moyle 1995). In general, splittail are most abundant in Suisun Marsh, especially in drier years (Meng and Moyle 1995), and reportedly rare in southern San Francisco Bay (Leidy 1984). Splittail abundance appears to be highest in the northern and western Delta when population levels are low, and they are more evenly distributed throughout the Delta during successful year classes (Sommer et al. 1997, Moyle 2002).

Splittail are largely absent from the upper river reaches where they formerly occurred, residing primarily in the lower parts of the Sacramento and San Joaquin rivers and tributaries and in Central Valley lakes and sloughs (Moyle 2002, Moyle et al. 2004). In wet years, however, they have been known to ascend the Sacramento River as far as RBDD and into the lower Feather and American rivers (Baxter et al. 1996; Sommer et al. 1997; Baxter 1999, 2000). The Sutter and Yolo bypasses along the lower Sacramento River appear to be important splittail spawning areas (Sommer et al. 1997). Splittail now migrate into the San Joaquin River only during wet years, and use of the Sacramento River and its tributaries is likely more important (Moyle 2002).

9B.9.3 Life History and Habitat Requirements

9B.9.3.1 Non-Breeding

Non-reproductive adult splittail are most abundant in moderately shallow, brackish areas, but can also be found in freshwater areas with tidal or riverine flow (Moyle et al. 2004). Non-breeding splittail are found in temperatures ranging from 5 to 24°C, depending on the season, and acclimated fish can survive temperatures up to 33°C for short periods (Young and Cech 1996). Juveniles and adult splittail demonstrate optimal growth at 20ºC and signs of physiological distress only above 29ºC (Young and Cech 1995).

Because splittail are adapted for living in brackish waters with fluctuating conditions, they are tolerant of high salinities and low dissolved oxygen (DO) levels. Splittail are often found in salinities of 10 to 18 parts per thousand (ppt), although lower salinities may be preferred (Meng and Moyle 1995) and can survive low DO levels (0. 6 to 1.2 milligrams per liter for young-of-the-year, juveniles, and subadults) (Young and Cech 1995, 1996). Because splittail have a high tolerance for variable environmental conditions (Young and Cech 1996) and are generally opportunistic feeders (prey includes mysid shrimp, clams, copepods, amphipods, and terrestrial invertebrates), reduced prey abundance will not likely have major population-level impacts. Year class success appears dependent on access and availability of floodplain spawning and rearing habitats, high outflow, and wet years (Sommer et al. 1997).

9B.9.3.2 Spawning

Adults typically migrate upstream from brackish areas in January and February and spawn in fresh water on inundated floodplains in March and April (Moyle et al. 2004). Foraging in flooded areas along the main rivers, bypasses, and tidal freshwater marsh areas of Montezuma and Suisun sloughs and San Pablo Bay before the onset of spawning may contribute to spawning success and survival of adults after spawning (Moyle et al. 2004). Splittail are adapted to the wet-dry climatic cycles of Northern California and thus concentrate their reproductive effort in wet years when potential success is enhanced by the availability of inundated floodplain (Meng and Moyle 1995, Sommer et al. 1997). Splittail are thought to be fractional spawners, with individuals spawning over a protracted period—often as long as several months (Wang 1995). Older fish are believed to begin spawning first (Caywood 1974).

Splittail eggs are deposited in flooded areas among submerged vegetation, to which they adhere until hatching. Rising flows appear to be the major trigger for splittail spawning, but increases in water temperature and day length may also be factors (Moyle et al. 2004). Spawning typically occurs on inundated floodplains from February through June, with peak spawning in March and April.

Information indicates that splittail spawn in open areas with moving, turbid water less than 5 feet (1.5 m) deep, among dense annual vegetation and where water temperatures are below 15°C (Moyle et al. 2004). Perhaps the most important spawning habitat in the eastern Delta is the Cosumnes River floodplain, where ripe splittail have been observed in flooded fields with cool temperatures below 15°C, turbid water, and submerged terrestrial vegetation (Crain et al. 2004).

Females are typically highly fecund, with the largest individuals potentially producing 100,000 or more eggs (Daniels and Moyle 1983, Feyrer and Baxter 1998). Fecundity has been found to be variable, however, and may be influenced by food supplies in the year before spawning (Moyle et al. 2004). The adhesive eggs are released by the female, fertilized by one or more attendant males, and adhere to vegetation until hatching (Moyle 2002). Splittail eggs, which are 0.4 to 0.6 inch (1.0 to 1.6 mm) in diameter (Wang 1986, Feyrer and Baxter 1998), begin to hatch within 3 to 7 days, depending on temperature (Bailey 1994). Eggs laid in clumps hatch more quickly than individual eggs (Moyle et al. 2004). Within 5 to 7 days after hatching, swim bladder inflation occurs, and larvae begin active swimming and feeding (Moyle 2002). Little is known regarding the tolerance of splittail eggs and developing larvae to DO, temperature, pH, or other water quality parameters, or to other factors such as physical disturbance or desiccation.

9B.9.3.3 Larvae

Juveniles are strong swimmers and are usually found in shallow (less than 6.6 feet [2 m] deep), turbid water (Young and Cech 1996). As their swimming ability increases, juveniles move away from the shallow areas near spawning sites into faster, deeper water (Moyle 2002). Floodplain habitat offers high food quality and production and low predator densities to increase juvenile growth.

After emergence, most larval splittail remain in flooded riparian areas for 10 to 14 days, most likely feeding among submerged vegetation before moving off floodplains into deeper water as they become stronger swimmers (Sommer et al. 1997, Wang 1986). Although juvenile splittail rear in upstream areas for a year or more (Baxter 1999), most move to tidal waters after only a few weeks, often in response to flow pulses (Moyle et al. 2004). The majority of juveniles move downstream into shallow, productive bay and estuarine waters from April to August (Meng and Moyle 1995). Growth likely depends on the availability of high-quality food, especially in the first year of life (Moyle et al. 2004).

9B.9.4 Population Trends

A variety of surveys have compiled splittail abundance data. None of these, however, was specifically designed to systematically sample splittail abundance, and definitive conclusions are therefore not possible (Moyle et al. 2004). Combined, the survey data indicate that successful reproduction occurs on a yearly basis, but large numbers of juvenile splittail are produced only when outflow is relatively high. Thus, the majority of adult fish in the population probably result from spawning in wet years (Moyle et al. 2004). The stock-recruitment relationship in splittail is apparently weak, indicating that given the right environmental conditions, a small number of large females can produce many young (Sommer et al. 1997, Meng and Moyle 1995).

Accounts of early fisheries suggested that splittail had large seasonal migrations (Walford 1931). Splittail migration now appears closely tied to river outflow. In wet years with increased river flow, adult splittail will still move long distances upstream to spawn, allowing juvenile rearing in upstream habitats. The upstream migration is smaller during dry years, although larvae and juveniles are often found upstream of Sacramento to Colusa or Ord Bend on the Sacramento River (Moyle et al. 2004). The tidal upper estuary, including Suisun Bay, provides most juvenile rearing habitat, although young-of-the-year may rear over a broader area, including the lower Sacramento River. Brackish water provides optimal rearing habitat for splittail.

DFW estimates that splittail during most years are only 35 to 60 percent as abundant as they were in 1940 (DFG 1992). DFW midwater trawl data indicate considerable fluctuations in splittail numbers since the mid-1960s, with abundance often tracking river and Delta outflow conditions. The overall trends include a decline from the mid-1960s to the late 1970s, somewhat of a resurgence through the mid-1980s, and another decline from the mid-1980s through 1994 (Moyle 2002). In 1995 and 1998, the population increased dramatically, demonstrating the extreme short- and long-term variability of splittail recruitment success and the apparent correlation with river outflow (Sommer et al. 1997). In 2006, when spring outflows were the highest since 1998, beach seine surveys conducted by USFWS in the lower portion of the estuary recorded the highest number of 0+ fish individuals since the surveys began in 1992 (Greiner et al. 2007). Surveys in the upper portions of the estuary showed a decline in catches of splittail and many other Delta fish. These declines were coupled with declines in zooplankton, which are the primary food source for splittail (Hieb et al. 2004). Pesticide use in the Central Valley may be responsible for the decline in zooplankton, which is causing the widespread pelagic organism decline in the Delta (Oros and Werner 2005).

Splittail may also be negatively affected by the introduction of the overbite clam (*Potamocorbula amurensis*) in the 1980s, which resulted in a collapse of opossum shrimp (*Neomysis mercedis*) populations, which were a primary source of food for splittail. The recent introduction of the Siberian prawn may similarly pose a threat to splittail food sources, as the Siberian prawns prey on mysid shrimp, which make up a large portion of spittail diets (Moyle et al. 2004). River outflow in February through May can explain between 55 and 69 percent of the variability in abundance of splittail young, depending on the abundance measure. Age -0 abundance of splittail declined in the estuary during most dry years, particularly in the drought that began in 1987 (Sommer et al. 1997). However, not all wet years result in high splittail recruitment because recruitment success largely depends on the availability of flooded spawning habitat. In 1996, for example, most high river flows occurred in December and January, before the onset of the splittail spawning season (Moyle 2002).

9B.9.5 References

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9B.10 Delta Smelt (*Hypomesus transpacificus*)

9B.10.1 Legal Status

Federal: Threatened, Designated Critical Habitat
State: Endangered

The USFWS listed the Delta Smelt as threatened in March 1993 (USFWS 1993), and critical habitat for this species was designated in 1994 (USFWS 1994). The Delta Smelt was one of eight fish species addressed in the Recovery Plan for the Sacramento–San Joaquin Delta Native Fishes (USFWS 1996). This recovery plan is currently under revision. The 2004 status review affirmed the need to retain the Delta Smelt as a threatened species (USFWS 2004). A 12-month finding on a petition to reclassify the Delta Smelt was completed in April 2010 and the USFWS determined that re-classifying the Delta Smelt from a threatened to an endangered species was warranted, but precluded by other higher-priority listing actions (USFWS 2010).

9B.10.2 Distribution

Delta Smelt are endemic to and resident in the Delta and San Francisco Bay, typically downstream of Isleton on the Sacramento River and downstream of Mossdale on the San Joaquin River, and are seasonally distributed in Suisun Bay (Moyle 2002). Delta Smelt abundance and geographic distribution are dependent upon freshwater outflows and the salinity of the Bay and Delta (Herbold et al. 1992). There is a close association between Delta Smelt abundance and surface salinity of 0–18 practical salinity units (psu) (psu are roughly equivalent to ppt), suggesting that their distribution is determined largely by the interaction with salinity conditions as determined by tidal currents, freshwater outflow, and diffusion, rather than by geography (Bennett 2000, 2005; Moyle 2002). For instance, water clarity and salinity were found to be the most reliable abiotic predictors of Delta Smelt abundance during the summer and fall (Feyrer et al. 2007, Nobriga et al. 2008). In addition, geographic distribution for particular life stages can vary dramatically between dry and wet years. Thus, in low outflow years, Delta Smelt occur primarily in the lower Sacramento River, with the area near Decker Island consistently exhibiting greatest catch over time. In years of very high outflow, however, their distribution extends into San Pablo Bay and the Napa River (Bennett 2000).

9B.10.3 Life History and Habitat Requirements

Overall, the Delta Smelt life cycle is completed in the brackish and tidal freshwater reaches of the upper San Francisco Estuary. However, salinity requirements vary by life stage. Apart from spawning and egg-embryo development, the distribution and movements of all life stages are influenced by transport processes associated with water flows in the estuary, which also affect the quality and location of suitable open water habitat (Dege and Brown 2004; Feyrer et al. 2007; Nobriga et al. 2008).

9B.10.3.1 Spawning

Delta Smelt have an annual, 1-year lifecycle. They typically require low-salinity, shallow openwater habitat in the estuary (Moyle 2002). They are found at 0-18 psu surface salinity (Baxter et al. 1999), although most are caught at salinities less than 6.0 psu, with older juveniles and adults being found at the higher end of that gradient (Bennett 2005). Delta Smelt feed primarily on planktonic copepods, cladocerans, and amphipods (Baxter et al. 2008). In recent years, a small to moderate number of Delta Smelt have been observed in the Deep Water Ship Channel during the late fall. The Deep Water Ship Channel can provide suitable water temperatures for Delta Smelt year-round (Sommer and Mejia 2013), which likely promotes freshwater residence in Delta Smelt in this region of the Delta (Sommer and Mejia 2013).

Delta Smelt are weakly anadromous and undergo a spawning migration from the low salinity zone to freshwater in most years (Grimaldo et al. 2009; Sommer et al. 2011). Spawning migrations occur between late December and late February, typically during “first flush” periods when inflow and turbidity increase on the Sacramento and San Joaquin Rivers (Grimaldo et al. 2009, Sommer et al. 2011). Notably, spawning movements are not always upstream. Under high outflow conditions, when total outflow exceeds 100,000 cubic feet per second (cfs), adult smelt tend to concentrate and spawn in Suisun Bay, Cache Slough Complex, and Napa River (Hobbs et al. 2007; Sommer et al. 2011). During drier years, when total outflow is less than 20,000 cfs, smelt tend to concentrate and spawn in the Cache Slough Complex and western Delta.

Adequate flows and suitable water quality are needed to attract migrating adults in the Sacramento and San Joaquin River channels and their associated tributaries, including Cache and Montezuma sloughs and their tributaries (USFWS 1996). Adult smelt do not spawn immediately after migration to freshwater, but appear to stage in upstream habitats (Sommer et al. 2011). Spawning typically commences when water temperatures reach 12°C, which typically occurs in early March. Spawning can continue into July (Wang 1986, Sweetnam and Stevens 1993), although most spawning takes place from early April to mid-May (Moyle 2002).

Delta Smelt are believed to spawn in shallow water along edges of rivers and sloughs subject to tidal influence (USFWS 2001). Based upon the occurrence of ripe females and yolk-sac larvae, spawning areas during dry and typical years are found in the north Delta reaches of the Sacramento River (Moyle 2002). Spawning locations in the Delta have not been identified and are inferred from larval catches (Bennett 2005). Larval fish have been observed in Montezuma Slough (Wang 1986), Suisun Slough in Suisun Marsh (Moyle 2002), the Napa River estuary (Stillwater Sciences 2006), the Sacramento River above Rio Vista, and Cache, Lindsey, Georgiana, Prospect, Beaver, Hog, Sycamore, and Barker sloughs (USFWS 1996). During wet years, Delta Smelt can be found spawning throughout most of the Delta, Suisun Marsh, and west to the Napa River (Herbold et al. 1992).

Although spawned eggs have not been found in the field, it is theorized that spawning occurs on hard substrates such as rocks, gravel, and tree roots (Herbold et al. 1992, Bennett 2000, Moyle 2002) in relatively low velocity currents (Swanson et al. 1998). Although smelt can be found within a wide salinity range, from 0 to 18.4 ppt (Swanson et al. 2000), spawning probably occurs within a narrow range of salinity—likely from 2–7 ppt. Spawning apparently can occur at temperatures ranging from 45-72°F (7-22°C) (Moyle 2002), but most often takes place between 45 and 59°F (7 and 15°C) (Wang 1986).

Spawning is thought to occur at night during new or full moons when the tide is low (Moyle 2002). Females (2.3-2.8 in [59-70 mm] SL) typically lay between 1,200 and 2,600 eggs (Moyle et al. 1992) and the relationship between female size (FL) and fecundity has been determined to be: Number of eggs = $0.266FL^{2.089}$ (Mager 1996). Most adults die after spawning, although a small number remain in the population for a second year (Moyle 2002) and may contribute disproportionately to the egg supply because of their increased size (3.5-4.7 in [90-120 mm] SL) (Moyle 2002).

9B.10.3.2 Hatching and Larval Distribution

No data are available on optimal temperature for survival of embryos, though some data suggest that high temperatures correspond to low hatching success and low embryo survival (R. Mager, unpubl. data; as cited in Winternitz and Wadsworth 1997). According to Moyle (2002), "it is likely that survival decreases as temperature increases beyond 18°C [64°F]." At temperatures between 59 and 62°F (14.8 and 16.5°C), embryonic development is reported to take approximately 9-13 days (Mager 1996). Although hatching has been detected from late February to June, peak hatching typically occurs in April.

Newly hatched smelt begin feeding on rotifers and other microscopic prey approximately 4-5 days after hatching, maintaining a position just above the bottom with the help of a large oil globule that makes them semi-buoyant (Mager 1996). The swim bladder and fins are fully developed several weeks later, and larvae rise up into the water column (Moyle 2002). During high outflow periods, larvae are distributed more widely as the spawning range extends further west when Delta outflows are high (Hobbs et al. 2007). Dege and Brown (2004) found that larvae less than 20 mm rear 5 to 20 km upstream of X2 (Dege and Brown 2004; Sommer and Mejia 2013). As larvae grow and water temperatures increase in the Delta (to approximately 23°C), their distribution shifts towards the low salinity zone (Dege and Brown 2004; Nobriga et al. 2008), where they circulate with the abundant zooplankton (Moyle 2002). By fall, the centroid of Delta Smelt distribution is tightly coupled with X2 (Sommer et al. 2011; Sommer and Mejia 2013).

Sommer and Mejia (2013) conducted a General Additive Model (GAM) analysis of Delta Smelt catch data from the 20-mm survey to determine suitable habitat parameters. They found larval Delta Smelt are more frequently captured in turbid and low salinity water. The analysis also showed that larval smelt presence in the survey peaked when water temperatures reach 20°C with low capture probability below 10°C and above 25°C.

The abundance of suitable rearing habitat for larvae varies from year to year, depending upon when peak spawning occurs. Peak larval density may occur as late as July or August. Base flows and pulse flows that transport and provide behavioral cues for Delta Smelt larvae and juveniles from February through June may not be adequate if larval peaks occur in July or August.

9B.10.3.3 Juvenile Rearing and Growth

The specific geographic area critical to the maintenance of suitable rearing habitat for Delta Smelt extends eastward from Carquinez Strait, up the Sacramento River to its confluence with Three Mile Slough (at RM 9), and south along the San Joaquin River including Big Break (USFWS 1996). Within this area, Delta Smelt typically rear in shallow (less than 10 ft [3 m]), open estuarine waters (Moyle 2002), in salinities ranging from 2-7 ppt (Swanson and Cech 1995) where "fresh and brackish water mix and hydrodynamics are complex as a result of the meeting of tidal and riverine currents" (Moyle 2002). These conditions are typically most common in Suisun Bay, which provides vital nursery habitat for Delta Smelt. When the mixing zone is located in Suisun Bay, it provides optimal conditions for algal and zooplankton growth, an important food source for Delta Smelt (Moyle 2002). When freshwater outflow is low, the mixing zone moves further up into the deeper, narrow channels of the Delta and Sacramento River, reducing food availability and total area available to the smelt (Moyle 2002).

Water quality preferences and thresholds for Delta Smelt are not well documented. Winternitz and Wadsworth (1997) observed that fewer Delta Smelt were collected in areas of higher temperatures than in areas of lower temperatures. Because other factors were not controlled, it is not clear whether temperature or other factors were driving Delta Smelt distribution. Nobriga et al. (2000) reported that Delta Smelt tolerated slightly higher water temperatures at a salinity of 4 ppt than in fresh water, but noted that further study is needed of these potentially interacting factors. Similar to larvae, a GAM analysis of the tow net survey data shows that suitable smelt habitat is best defined by water clarity, specific conductance (salinity), water temperature (Nobriga et al. 2008). As previously noted, some juvenile smelt will remain in the Sacramento Deep Water

Ship Channel during the summer and fall months. The channel is deep, turbid, and offers some temperature refuge, which may explain why smelt remain in this freshwater habitat when most other smelt at this life stage are in found in the low salinity zone.

Planktonic copepods, cladocerans, amphipods, and, to a lesser extent, insect larvae, are the primary prey items for Delta Smelt (Moyle 2002). Delta Smelt larvae have more specific prey-size requirements for first feeding. In a study conducted in the northern estuary and Delta, Lott (1998) found that smaller size classes of Delta Smelt tended to consume more nauplii and juvenile copepods, while larger size classes consumed more adult copepods. It appears that food availability after yolk-sac absorption is critical in determining success of Delta Smelt (Nobriga 1998). However, it is not known if a limited food supply contributes to reduced year-class success and therefore has population-level implications.

Juvenile Delta Smelt grow rapidly, typically reaching 1.6-2 inches (40-50 mm) FL by early August (Radtke 1966, Moyle et al. 1992). Growth rate appears to be dependent on the quality and abundance of food (Moyle 2002). Adult length (2.2-2.8 inches [55-70 mm] SL) is typically reached by September, or approximately 7-9 months after hatching (Moyle 2002). By fall, Delta Smelt are fully capable of altering their distribution to suitable habitat. Using a GAM approach, Feyrer et al. (2007) showed that Delta Smelt habitat is best defined by turbidity and specific conductance (salinity). Unlike the other analyses, Feyrer et al. (2011) converted the GAM model results to a habitat index for Delta Smelt, showing that habitat improves and expands for Delta Smelt when X2 is in Suisun Bay compared to when X2 is located at or above the confluence. The relationship between the habitat index and X2 is asymptotic, whereby the index does not increase when X2 is greater than 74 km or decrease when X2 is below 81 km. Feyrer et al. (2007) was also able to demonstrate that when the habitat index is higher (i.e., X2 is west of the confluence), it has a positive effect on subsequent juvenile abundance of Delta Smelt.

Larvae and young juveniles are affected by entrainment during the spring and early summer. As Delta Smelt become adults, they migrate downstream to brackish water areas in the fall and winter and are considered less vulnerable to diversion effects. Pre-spawning adults migrating back into freshwater to spawn in the late winter and early spring become vulnerable to entrainment effects once again.

The quantity and suitability of Delta Smelt habitat increases with higher outflow (Bennett 2005). When the near-bottom mixing zone is contained within Suisun Bay and when adequate outflow from both the Sacramento and San Joaquin rivers have allowed downstream movement, young Delta Smelt are dispersed more widely throughout a large expanse of shallow-water and marsh habitat than when the isohaline is upstream in the narrower, deeper Delta sloughs and channels. If smelt use this habitat and their distribution is wider and shifted downstream, subsequent entrainment in the winter will be reduced. Habitat conditions suitable for transport of larvae and juveniles are needed as early as February 1 and as late as August 31, because the spawning season varies from year to year and starts as early as December and extends until July (USFWS 1996). Adequate river flow is necessary to provide this transport to Suisun Bay and to maintain rearing habitat (USFWS 1996).

Spawning adults become vulnerable to entrainment effects during the winter and spring (Kimmerer 2008). Combined particle tracking models and 20 mm survey distributions suggest Delta Smelt population losses from entrainment at the Banks and Jones pumping plants are directly correlated with X2 position and might reach an estimated 20-40 percent when X2 moves landward of 37 mi (60 km). Maintaining X2 in a favorable location (i.e., away from Central and South Delta) during the spawning period of Delta Smelt reduces their exposure to the effects of reverse flow in the southern Delta channels (California Resources Agency 2007). Larvae and young juveniles typically follow the direction of spring flows downstream into the estuary. Reverse flows have been shown to direct larvae and young juvenile smelt toward the pumps and salvage of adult Delta Smelt is very low or zero during years when Old and Middle River flows are positive (i.e., away from the export facilities) (California Resources Agency 2007). A favorable location for X2 during this period is defined as seaward of 40 mi (65 km) from the Golden Gate Bridge based on a 14-day running average (California Resources Agency 2007).

The abundance of many local estuarine taxa has tended to increase in years when flows into the estuary are high and the X2 location is pushed seaward (Jassby et al. 1995), implying that over the range of historical experience the quantity or suitability of estuarine habitat increases when outflows are high. Feyrer et al. (2007) reported that fall environmental quality has declined over the long-term in the core range of Delta Smelt, including Suisun Bay and the Delta. This decline was largely due to changes in salinity in Suisun Bay and the western Delta, and changes in water clarity within the Delta. Baxter et al. (2008) reported the long-term environmental quality declines for Delta Smelt and Striped Bass are defined by a lowered probability of occurrence in samples based on changes in specific conductance and Secchi depth.

Planktonic copepods, cladocerans, amphipods, and, to a lesser extent, insect larvae, are the primary prey items for Delta Smelt (Moyle 2002). Delta Smelt larvae have more specific prey-size requirements for first feeding. In a study conducted in the northern estuary and Delta, Lott (1998) found that smaller size classes of Delta Smelt tended to consume more nauplii and juvenile copepods, while larger size classes consumed more adult copepods. It appears that food availability after yolk-sac absorption is critical in determining success of Delta Smelt (Nobriga 1998). However, it is not known if a limited food supply contributes to reduced year-class success and therefore has population-level implications.

The overbite clam has been associated with large changes in phytoplankton abundance in San Francisco Bay and the western Delta (Carlton et al. 1990), causing a decrease in abundance of other species that depend on phytoplankton (zooplankton) for food. Due in part to its efficiency in filtering water, the clarity of Suisun Bay and delta waters has increased. This has affected Delta Smelt by reducing food supply and increasing its susceptibility to predation.

9B.10.4 Population Trends

California Department of Fish and Wildlife has conducted several long-term monitoring surveys that have been used to index the relative abundance of Delta Smelt. The 20-mm Survey has been conducted every year since 1995. This survey targets late-stage Delta Smelt larvae. Most sampling has occurred from April to June. The Summer Townet Survey (TNS) has been conducted nearly every year since 1959. This survey targets 38-mm Striped Bass, but collects similar-sized juvenile Delta Smelt. Most sampling has occurred from June to August. The Fall Midwater Trawl Survey (FMWT has been conducted nearly every year since 1967. This survey also targets age-0 Striped Bass, but collects Delta Smelt longer than 40 mm. The FMWT samples monthly from September to December. These abundance index time series document the long-term decline of the Delta Smelt.

Early statistical assessments of Delta Smelt population dynamics concluded that the relative abundance of the adult Delta Smelt population had only a very weak influence on subsequent juvenile abundance (Sweetnam and Stevens 1993). Thus, early attempts to looked for environmental variables that were directly correlated with interannual abundance variation (e.g., Stevens and Miller 1983; Moyle et al. 1992; Sweetnam and Stevens 1993; Jassby et al. 1995). Because these analyses did not find strong support for an outflow-abundance linkage, the prevailing conceptual model was that multiple interacting factors had caused the Delta Smelt decline (Moyle et al. 1992; Bennett and Moyle 1995; Bennett 2005). It has also recently been noted that Delta Smelt's FMWT index is partly influenced by concurrent environmental conditions (Feyrer et al. 2007; 2011).

It is now recognized that Delta Smelt abundance plays an important role in subsequent smelt abundance. Bennett (2005) examined (1) the influence of adult stock (FMWT) on the next generation of juveniles (TNS); (2) the influence of the juvenile stock (TNS) on the subsequent adult stock (FMWT); (3) the influence of the FMWT on the following year's FMWT and on the FMWT two years later, and (4) the influence of the TNS abundance on the following year's TNS and on the TNS 2 years later. His conclusions were that (1) 2-year-old Delta Smelt might play an important role in Delta Smelt population dynamics, (2) it was not clear whether juvenile production was a density-independent or density dependent function of adult abundance, and (3) adult production was a density-dependent function of juvenile abundance and the carrying capacity of the estuary to support this life-stage transition had declined over time. These conclusions are also supported by Maunder and Deriso (2011).

Delta Smelt were historically one of the most common species in the San Francisco Estuary, but exhibited significant declines during the 1980s (DFG 2000). Kimmerer (2002) and Thomson et al. (2010) reported a Delta Smelt step-decline during 1981-1982. Prior to this decline, the stock-recruit data are consistent with "Ricker" type density-dependence where increasing adult abundance resulted in decreased juvenile abundance. Since the decline, recruitment has been positively and essentially linearly related to prior adult abundance, suggesting that reproduction has been basically density-independent for about the past 30 years. In contrast to the transition among generations, the weight of scientific evidence strongly supports the hypothesis that, at least over the history of IEP fish monitoring, Delta Smelt has experienced density-dependence during the juvenile stage of its life cycle (i.e., between the summer and fall) (Bennett 2005; Maunder and Deriso 2011). The most relevant aspect of this juvenile density dependence is that the carrying capacity of the estuary for Delta Smelt has likely declined (Bennett 2005).

Therefore, it is now thought that the Delta Smelt population decline has occurred for two basic reasons. First, the compensatory density-dependence that historically enabled juvenile abundance to rebound from low adult numbers stopped happening. This change had occurred by the early 1980s as described above. The reason is still not known, but the consequence of the change is that for the past several decades, adult abundance has driven juvenile production in a largely density-independent manner. Thus, if numbers of adults or adult fecundity decline, juvenile production will also decline (Kimmerer 2011). Second, because juvenile carrying capacity has declined, juvenile production hits a 'ceiling' at a lower abundance than it once did. This limits adult abundance and possibly per capita fecundity, which cycles around and limits the abundance of the next generation of juveniles. The mechanism causing carrying capacity to decline is likely due to the long-term accumulation of adverse changes in both physical and biological aspects of habitat during the summer to fall (Bennett et al. 2008; Feyrer et al. 2007; 2011; Maunder and Deriso 2011).

9B.10.5 References

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9B.11 Longfin Smelt (*Spirinchus thaleichthys*)

9B.11.1 Legal Status

Federal: Candidate for listing as Endangered
State: Threatened

Longfin Smelt is a state-listed threatened species throughout its range in California (DFG 2009). USFWS denied a petition for Federal listing because the population in California (and specifically the San Francisco Bay) was not believed to be sufficiently genetically isolated from other populations (USFWS 2009). The Center for Biological Diversity challenged the merits of this determination. In 2011, USFWS entered into a settlement agreement with the Center for Biological Diversity and agreed to conduct a rangewide status review and prepare a 12-month finding to be published by September 30, 2011. The 12-month finding on the petition to list the San Francisco Bay-Delta population of the Longfin Smelt as endangered or threatened was completed in March 2012. USFWS determined that listing the Longfin Smelt rangewide was not warranted at the time, but that listing the Bay-Delta DPS of Longfin Smelt was warranted but precluded by other higher priority listing actions (USFWS 2012).

9B.11.2 Distribution

Populations of the Longfin Smelt have been found in estuaries along the Pacific coast from Prince William Sound, Alaska, to the Sacramento-San Joaquin estuary (USFWS 2012). The largest population occupies the Sacramento-San Joaquin estuary, with a smaller population in Humboldt Bay and the Eel River (Moyle 2002). They may occur throughout the year in the estuary and lowest reaches of the Klamath River, but little is known of this population.

9B.11.3 Life History and Habitat Requirements

Longfin Smelt typically live in bays and estuaries and make seasonal migrations. During winter, they congregate for spawning in the upper reaches of the bays and lower reaches of the river deltas. Juvenile and adult Longfin Smelt have been found throughout the year in salinities ranging from pure fresh water to pure seawater, although once past the juvenile stage, they are typically collected in waters with salinities ranging from 14 to 28 ppt (Baxter 1999). Within the Delta, adult Longfin Smelt occupy water at temperatures from 16 to 20°C (61 to 68°F) and spawn in water with temperatures from 5.6 to 14.5°C (41 to 58°F) (Wang 1986).

Longfin Smelt have been observed in their winter and spring spawning period as far upstream as Isleton in the Sacramento River, Santa Clara shoal in the San Joaquin system, Hog Slough off the South-Fork Mokelumne River, and Old River south of Indian Slough (DFG 2009). Exact spawning locations in the Delta are unknown and may vary from year to year, depending on environmental conditions. However, it seems likely that spawning locations consist of the overlap of appropriate conditions of flow, temperature, and salinity with appropriate substrate (Rosenfield 2010). Most individuals die after spawning, but occasionally a female may live to spawn a second time.

Longfin Smelt congregate in deep waters near the low salinity zone near X2 during the spawning period, and they likely make short runs upstream, possibly at night, to spawn from these locations (DFG 2009, Rosenfield 2010). Longfin Smelt in the Delta may spawn as early as November and as late as June, although spawning typically occurs from January to April (DFG 2009, Moyle 2002). The adhesive eggs are deposited on rocks or aquatic plants in the freshwater sections of bays and river deltas. Baxter et al. (2010) found that female Longfin Smelt produced between 1,900 and 18,000 eggs, with fecundity greater in fish with greater lengths.

Larval Longfin Smelt less than 12 mm (0.5 inch) in length are buoyant because they have not yet developed an air bladder; as a result, they occupy the upper one-third of the water column. Longfin Smelt develop an air bladder at approximately 12 to 15 mm (0.5 to 0.6 inch) in length and are able to migrate vertically in the water column. At this time, they shift habitat and live in the bottom two-thirds of the water column (DFG 2009). Longfin Smelt are dispersed broadly in the Delta by high flows and currents, which facilitate transport of larvae and juveniles long distances. Longfin Smelt larvae are dispersed farther downstream during high freshwater flows (Dege and Brown 2004). They spend approximately 21 months of their 24-month life cycle in brackish or marine waters (Baxter 1999, Dege and Brown 2004). In the Bay-Delta, most Longfin Smelt spend their first year in Suisun Bay and Marsh. The remainder of their life is spent in the San Francisco Bay or the Gulf of Farallones (Moyle 2008). Based on monthly survey results, Rosenfield and Baxter (2007) inferred that the majority of Longfin Smelt from the Bay-Delta migrate out of the estuary after the first winter of their life cycle and return during late fall to winter of their second year. They noted that migration out of the estuary into nearby coastal waters is consistent with captures of Longfin Smelt in the coastal waters of the Gulf of Farallones and hypothesized that the movement is a behavioral response to warm water temperatures during summer and early fall in the shallows of south San Francisco Bay and San Pablo Bay. Some Longfin Smelt may stay in the ocean and not re-enter fresh water to spawn until the end of their third year.

In the Bay-Delta, calanoid copepods such as *Pseudodiatomus forbesi* and *Eurytemora* sp., as well as the cyclopoid copepod *Acanthocyclops vernali*, are the primary prey of Longfin Smelt during the first few months of their lives (approximately January through May) (Slater 2008). The Longfin Smelt’s diet shifts to include mysids such as opossum shrimp (*Neomysis mercedis*) and other small crustaceans (*Acanthomysis* sp.) as soon as they are large enough (20 to 30 mm [0.78 to 1.18 inches]) to consume these larger prey items (DFG 2009).

Longfin Smelt numbers in the Bay-Delta have declined significantly since the 1980s (Rosenfield and Baxter 2007, Baxter et. al. 2010). Rosenfield and Baxter (2007) confirmed the positive correlation between Longfin Smelt abundance and freshwater flow that had been previously documented by others (Stevens and Miller 1983, Baxter 1999, Kimmerer 2002), noting that abundances of both adults and juveniles were significantly lower during the 1987–94 drought than during either the pre- or post-drought periods. Abundance of Longfin Smelt has remained low since 2000, even though freshwater flows increased during several of these years (Baxter et al. 2010). Abundance indices derived from the FMWT, Bay Study Midwater Trawl, and Bay Study Otter Trawl show marked declines in Longfin Smelt populations from 2002 to 2009. Longfin Smelt abundance over the last decade is the lowest recorded in the 40-year history of DFG’s FMWT monitoring surveys (USFWS 2012).

Research on declines of Longfin Smelt and other pelagic fish species in the Bay-Delta since 2002 (referred to as pelagic organism decline) have most recently been summarized in the Interagency Ecological Program 2010 Pelagic Organism Decline Work Plan and Synthesis of Results (Baxter et al. 2010). Although there is substantial uncertainty about the causal mechanisms underlying the pelagic organism decline, reduced Delta freshwater flows have been identified as one of several key factors believed to contribute to recent declines in the abundance of Longfin Smelt (Baxter et al. 2010).

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9B.12 Eulachon (*Thaleichthys pacificus*)

9B.12.1 Legal Status

Federal: Threatened
State: Species of Special Concern

9B.12.2 Summary

Eulachon are anadromous fish that occur in the lower portions of certain rivers draining into the northeastern Pacific Ocean, ranging from northern California to the southeastern Bering Sea in Bristol Bay, Alaska (Scott and Crossman 1973, Willson et al. 2006).

The southern population of Pacific Eulachon consists of populations spawning in rivers south of the Nass River in British Columbia, Canada, to and including the Mad River in California (NMFS 2009). On March 18, 2010, NMFS listed the southern DPS of Pacific Eulachon as threatened under the ESA (NMFS 2010); critical habitat was designated in 2011 (NMFS 2011). The Klamath River is near the southern limit of the range of Eulachon (Eulachon BRT 2010).

Spawning occurs in gravel riffles, with hatching about a month later. The larvae generally move downstream to the estuary following hatching.

Large spawning aggregations of Pacific Eulachon used to regularly occur in the Klamath River (Fry 1979), migrating in March and April to spawn, but they rarely moved more than 8 miles inland (NRC 2004). DFW sampled in the Klamath River from 1989 to 2003 with no Pacific Eulachon captures (USDI and DFG 2011). The Yurok Tribe sampled extensively for Pacific Eulachon in early 2011, and although tribal fishermen did not capture Pacific Eulachon from the Klamath River itself, they did recover Pacific Eulachon from the surf zone at the mouth of the river (USDI and DFG 2011).

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9B.13 Striped Bass (*Morone saxatilis*)

9B.13.1 Legal Status

Federal: None
State: None

Striped Bass are native to the Atlantic Coast of North America and were introduced to California in 1879. Striped Bass are a large (>1 meter), long-lived (>10 years) species. They are widespread in the San Francisco Estuary watershed as juveniles and adults. Striped Bass move regularly from salt to fresh water.

They require a large body of water for foraging on fish (usually estuaries or large reservoirs) and large cool rivers for spawning. Striped Bass spend most of their lives in estuaries.

9B.13.2 Distribution in Affected Area

Adult Striped Bass are distributed mainly in the lower bays and ocean during the summer, and in the Delta during fall and winter. Spawning takes place in the spring (April–June), at which time Striped Bass swim upstream to spawning grounds. In the Sacramento River, most spawning takes place between RM 77.7 and RM 121.2 (Moyle 2002). After spawning, adults move downstream into the Delta and bays (Blunt 1962).

9B.13.3 Life History and Habitat Requirements

Female Striped Bass mature at between 4 and 6 years of age and can spawn every year. In the Delta and Sacramento and San Joaquin rivers, spawning occurs from April to June at temperatures between 14°C and 21°C. Eggs are free-floating and negatively buoyant, and hatch in about two days as they drift downstream, with larvae occurring in shallow and open waters of the lower reaches of the Sacramento and San Joaquin rivers, the Delta, Suisun Bay, Montezuma Slough, and Carquinez Strait. Location of spawning varies based on temperature, flow, and salinity (Turner 1972). In the Yolo Bypass, Harrell and Sommer (2003) observed that flow pulses immediately preceding floodplain inundation triggered upstream movement of Striped Bass, resulting in successful spawning. During low flow years, spawning occurs within the Delta itself.

Newly hatched Striped Bass feed off their yolk sac for up to 8 days (Wang 1986), after which they start feeding on zooplankton. Larvae in the Sacramento River migrate into the water column from April to mid-June (Stevens 1966). In the Sacramento River, embryos and larvae are carried into the Delta and Suisun Bay (Moyle 2002). In the San Joaquin River, embryos remain in the same general area where spawning took place, as freshwater outflow is balanced by tidal currents (Moyle 2002). When larval bass from both rivers begin to feed, they are concentrated in the most productive part of the estuary—where freshwater and salt water meet or near X2 (Moyle 2002).

Striped Bass are tolerant of a wide range of environmental conditions, surviving temperatures up to 25°C (77ºF) (and up to 34°C [93ºF] for shorter periods), rapid temperature swings, low oxygen levels between 3 and 5 milligrams per liter (mg/L), and high turbidity (Moyle 2002). Hassler (1988), in a summary of environmental tolerance studies, reported that Striped Bass could tolerate dissolved oxygen concentrations ranging from 3 to 20 mg/L, and a pH range of 6 to 10, although the optimum level ranged from 6 to 12 mg/L and 7 to 9, respectively. The information compiled by Hassler (1988) suggested juveniles preferred rearing temperatures of 24 to 26°C (60.8 to 66.2°F). As Striped Bass grow, their temperature preference shifts towards cooler water (Hill et al. 1989). Adult Striped Bass appear to prefer water temperatures ranging from 20 to 24°C (68 to 75.2°F) (Emmett et al. 1991).

Typical of an anadromous species, salinity tolerance of Striped Bass also changes with age (Lal et al. 1977, Hill et al. 1989). Eggs and larvae reportedly thrive at salinities less than 3 practical salinity units (psu) (Mansueti 1958, Dovel 197l), and can tolerate salinities of 8 to 9 psu without ill effects (Morgan and Rasin 1973). Adults can apparently tolerate salinities from 0 to 34 psu or more (Rogers and Westin 1978), with a range of 10 to 20 psu reported as optimal for larger juveniles (Bogdanov et al. 1967).

9B.13.4 Biotic Interactions

Striped Bass are pelagic, opportunistic predators, feeding on invertebrates and fishes. They tend to exhibit a roving school foraging strategy (Pickard et al. 1982). Larval and juvenile Striped Bass feed on invertebrates such as copepods or opossum shrimp. In the San Francisco Bay area, juvenile bass form small schools or feeding groups (Skinner 1962) with specific prey varying with fish size, habitat, and season (Hill et al. 1989).

Striped Bass are a top predator in the Delta and are considered major predators on fish (Thomas 1967). Fish become important in the diet of juveniles when they reach a FL of 130 to 350 mm, especially late in the summer when young-of-the-year Striped Bass and shad become available (Moyle 2002). Striped Bass are primarily piscivorous as subadults, when they reach 250 to 470 mm FL (approximately age 2+). Stevens (1966) found that the importance of fish in the diet of subadult (260 to 470 mm FL) and adult (>380 mm FL) Striped Bass in the Sacramento-San Joaquin estuary varied seasonally. Fish were most prevalent in the diet of subadults in fall, and occurred most frequently in the diet of adults in fall and winter. Adult Striped Bass feed primarily on smaller Striped Bass, threadfin shad, and juvenile salmonids, as well as pelagic ocean fishes (Moyle 2002). Striped Bass can successfully switch to feeding on novel prey (Moyle 2002). Striped Bass are considered important predators on juvenile salmon in the Sacramento River (Tucker et al. 1998, Moyle 2002). Average populations of 1.7 million adults during the late 1960s to early 1970s, and 1.25 million adults during 1967-1991 (USFWS 1995), likely exerted considerable predation pressure on outmigrating juvenile salmon (Yoshiyama et al. 1998). The impact of Striped Bass on Delta Smelt and Sacramento Splittail is not known (Moyle 2002). Delta Smelt were occasional prey fish for Striped Bass in the early 1960s (Turner and Kelley 1966) but went undetected in a recent study of predator stomach contents (Nobriga and Feyrer 2007). Striped Bass are likely the primary predator of juvenile and adult Delta Smelt given their spatial overlap in pelagic habitats (NMFS 2009).

Though Striped Bass may commonly exhibit a roving school foraging strategy (Pickard et al. 1982), they appear to take advantage of prey that is concentrated at screened diversions or pumps, and may be partially responsible for the decline of some native fishes, including salmon, thicktail chub, and Sacramento perch (Tucker et al. 1998). Striped Bass are considered to be a primary cause of juvenile salmon mortality at the state water-export facility in the south Delta (USFWS 1995). Tucker et al. (1998) observed Striped Bass preying heavily on juvenile Chinook Salmon that passed through the diversion facilities at Red Bluff

Diversion Dam on the Sacramento River. Juvenile Chinook Salmon were found by Thomas (1967) to be a major food item in the diet of Striped Bass in the spring and early summer during smolt outmigration through the Sacramento and San Joaquin rivers and Delta.

The introduction of the overbite clam in the 1980s has been associated with large decreases in zooplankton and phytoplankton densities in San Francisco Bay and the western Delta (Carlton et al.1990), which has decreased the amount of food available for larval and juvenile Striped Bass. The population responses of juvenile Striped Bass to winter-spring outflows changed after the overbite clam invasion as young Striped Bass relative abundance stopped responding to outflow altogether (Sommer et al. 2007). In addition to decreased copepod densities, the principal historic copepod food source, *Eurytemora affinis*, for larval and juvenile Striped Bass has largely been replaced by alien copepod species that may be energetically less desirable (Meng and Orsi 1991).

Within the Delta, adult Striped Bass feed primarily on Threadfin Shad and juvenile Striped Bass. Thus, when shortages of alternate prey exist, survival rates of juvenile bass may decrease as they become increasingly important to adult diets, resulting in an unusually high response to decreased productivity in the Delta (Moyle 2002).

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9B.14 Southern Resident Killer Whale (*Orcinus orca*)

9B.14.1 Legal Status

Federal: Endangered
State: None

Three distinct forms of Killer Whales, termed residents, transients, and offshores, are recognized in the northeastern Pacific Ocean. Resident Killer Whales in U.S. waters are distributed from Alaska to California, with four distinct communities recognized: Southern, Northern, Southern Alaska, and Western Alaska (Krahn et al. 2002, 2004). Resident Killer Whales are fish eaters and live in stable matrilineal pods. Of these, only the Southern Resident Distinct Population Segment (DPS) is listed as endangered.

The designated critical habitat does not overlap with the action area for this consultation, nor are there any discernible changes to the physical environment that occur within designated critical that could be correlated to project operations. The only potential effects of project operations on the identified physical or biological features essential to conservation would be to prey quantity, quality, and availability. Project operations have the potential to affect only a portion of juvenile salmon originating in California's Central Valley streams. As discussed earlier, salmon originating in California streams are estimated to contribute between 3 and 5 percent of the salmon population off the Washington coast based on analysis of troll catches. These estimates were made based on data collected during the time of year when the Southern Residents are present. As discussed above, the majority of the fish attributed to California streams that are affected by the project are expected to be hatchery fish.

9B.14.2 Distribution

The Southern Resident Killer Whale DPS is designated as endangered under the ESA (NMFS 2005). This DPS primarily occurs in the inland waters of Washington state and southern Vancouver Island, particularly during the spring, summer, and fall, but members of the population have been observed off coastal California in Monterey Bay, near the Farallon Islands, and off Point Reyes (Heimlich-Boran 1988, Felleman et al. 1991, Olson 1998, Osborne 1999, NMFS 2005). The action area is outside of the DPS's designated Critical Habitat, which is in Washington state (NMFS 2006a).

9B.14.3 Life History and Habitat Requirements

Southern Resident Killer Whales spend a significant portion of the year in the inland waterways of the Strait of Georgia, Strait of Juan de Fuca, and Puget Sound, particularly during the spring, summer, and fall, when all three pods are regularly present in the Georgia Basin (defined as the Georgia Strait, San Juan Islands, and Strait of Juan de Fuca) (Heimlich-Boran 1988, Felleman et al. 1991, Olson 1998, Osborne 1999). The Southern Resident population consists of three pods, identified as J, K, and L pods. Typically, K and L pods arrive in May or June and spend most of their time in this core area until departing in October or November. During this time, both pods also make frequent trips lasting a few days to the outer coasts of Washington and southern Vancouver Island (Ford et al. 2000). J pod continues to spend intermittent periods of time in the Georgia Basin and Puget Sound during late fall, winter, and early spring.

While the Southern Residents are in inland waters during the warmer months, all of the pods concentrate their activities in Haro Strait, Boundary Passage, the southern Gulf Islands, the eastern end of the Strait of Juan de Fuca, and several localities in the southern Georgia Strait (Heimlich-Boran 1988, Felleman et al. 1991, Olson 1998, Ford et al. 2000). In general, they spend less time elsewhere, including other sections of the Georgia Strait, Strait of Juan de Fuca, and San Juan Islands, Admiralty Inlet west of Whidbey Island, and Puget Sound. Individual pods are similar in their preferred areas of use (Olson 1998), although there are some seasonal and temporal differences in certain areas visited by each pod (Hauser 2006). For example, J pod visits Rosario Strait more frequently than K or L pods (Hauser 2006). The movements of Southern Resident Killer Whales relate to those of their preferred prey—salmon. Pods commonly seek out and forage in areas where salmon occur, especially those associated with migrating salmon (Heimlich-Boran 1986, 1988; Nichol and Shackleton 1996). Notable locations of particularly high use include Haro Strait and Boundary Passage, the southern tip of Vancouver Island, Swanson Channel off North Pender Island, and the mouth of the Fraser River delta, which is visited by all three pods in September and October (Felleman et al. 1991, Ford et al. 2000). These sites are major corridors for migrating salmon.

Wild female Southern Resident Killer Whales give birth to their first surviving calf between the ages of 12 and 16 years (mean = about 14.9 years) (Olesiuk et al. 1990, Matkin et al. 2003). Females produce an average of 5.4 surviving calves during a reproductive life span lasting about 25 years (Olesiuk et al. 1990). Males become sexually mature at body lengths ranging from 5.2 to 6.4 meters, which corresponds to between the ages of 10 and 17.5 years (mean = about 15 years) (Christensen 1984, Perrin and Reilly 1984, Duffield and Miller 1988, Olesiuk et al. 1990), and are presumed to remain sexually active throughout their adult lives (Olesiuk et al. 1990).

Southern Resident Killer Whales are known to consume 22 species of fish and one species of squid (Scheffer and Slipp 1948; Ford et al. 1998, 2000; Ford and Ellis 2005; Saulitis et al. 2000). Ford and Ellis (2005) found that salmon represent over 96 percent of the prey consumed during the spring, summer, and fall. Chinook Salmon were selected over other species, comprising over 70 percent of the identified salmonids taken. This preference occurred despite the much lower abundance of Chinook in the study area in comparison to other salmonids and is probably related to the species' large size, high fat and energy content, and year-round occurrence in the area. Other salmonids eaten in smaller amounts include chum (22 percent of the diet), pink (3 percent), coho (2 percent), sockeye (less than 1 percent), and steelhead (less than 1 percent) (Ford and Ellis 2005). This work suggested an overall preference of these whales for Chinook during the summer and fall, but also revealed extensive feeding on chum salmon in the fall.

Southern Resident Killer Whale survival and fecundity are correlated with Chinook Salmon abundance (Ward et al. 2009, Ford et al. 2009). Southern Resident Killer Whales could potentially be affected by changes in salmon populations caused by the Proposed Action, because their survival and fecundity appear dependent on the abundance of Chinook Salmon (Ward et al. 2009, Ford et al. 2009).

Chinook Salmon originating from the Fraser River are the dominant prey of resident Killer Whales in the summer months when they are usually in inland marine waters (Hanson et al. 2010). Less is known of their diet during the remainder of the year (September through May), when they spend much of their time in outer coastal waters, and may range from central California to northern British Columbia (Hanson et al. 2010). However, it is believed likely that they preferentially feed on Chinook Salmon when available, and roughly in proportion to their relative abundance (Hanson et al. 2010). Hanson et al. (2010) found Southern Resident stomachs to contain several different ESUs of salmon, including Central Valley fall-run Chinook Salmon.

NMFS (2008) estimated the biological requirements of Southern Resident Killer Whales including the diet composition and number of salmon the population requires in their coastal range. NMFS estimated that the current population of Southern Residents at the time (87) would be required to consume between 392,555 and 470,288 salmon based on diet compositions and bioenergetic needs in their coastal range. These estimates were based on Chinook Salmon comprising 70 to 88 percent of their diet.

Salmon originating in California streams are estimated to contribute 3 percent of the salmon population off the Washington coast based on genetic stock identification (GSI) of Washington troll catch in May of 1981 and 1982 (Utter et al. 1983). Research in the mid-1970s estimated California's contribution at 5 percent (Wright 1976). More recent data from Collaborative Research on Oregon Ocean Salmon using GSI estimate that 59 percent of salmon analyzed from the Oregon commercial harvest (June–October 2006) were Central Valley fall-run or spring-run Chinook Salmon (https://fp.pacificfishtrax.org/portal/). It is important to note that these percentages could vary during different years or seasons.

Reclamation funds the operation and maintenance of the Coleman, Livingstone, and Nimbus hatcheries. These hatcheries have a combined yearly production goal of 17,200,000 Chinook Salmon smolts. DWR funds the operation of the Feather River hatcheries for production of approximately 8 million Chinook Salmon smolts annually (yearly production goal).

Analysis of Chinook Salmon otoliths in 1999 and 2002 found that the contribution of hatchery-produced fish (from the Sacramento and San Joaquin river system) made up approximately 90 percent of the ocean fishery off the central California coast from Bodega Bay to Monterey Bay (Barnett-Johnson et al. 2007). Similar studies have not been completed to assess the percentage that Central Valley hatcheries contribute to the salmon originating from California off the Oregon and Washington coasts, but it suggests that hatchery fish would likely be the majority.

Based on observations of captive Killer Whales, studies have extrapolated the energy requirements of wild Killer Whales and estimate an average size value for the five salmon species combined. Osborne (1999) estimated that adult Killer Whales would consume 28 to 34 adult salmon per day, and that younger Killer Whales (less than 13 years of age) would consume about 15 to 17 salmon per day to meet their daily energy requirements. Extrapolating these results, the Southern Resident population (approximately 90 individuals) would consume about 750,000 to 850,000 adult salmon per year.

9B.14.4 Population Trends

Some evidence suggests that until the mid- to late-1800s, the Southern Resident Killer Whale population may have numbered more than 200 animals (Krahn et al. 2002). This estimate was based, in part, on a recent genetic analysis of microsatellite DNA, which found that the genetic diversity of the Southern Resident population resembles that of the Northern Residents (Barrett-Lennard 2000, Barrett-Lennard and Ellis 2001), and concluded that the two populations were possibly once similar in size. Recent efforts to assess the Killer Whale population during the past century have been hindered by an absence of empirical information prior to 1974 (NMFS 2006b). For example, a report by Scheffer and Slipp (1948) is the only pre-1974 account of Southern Resident abundance in the area, and it merely noted that the species was “frequently seen” during the 1940s in the Strait of Juan de Fuca, northern Puget Sound, and off the coast of the Olympic Peninsula, with smaller numbers along Washington’s outer coast. Olesiuk et al. (1990) estimated the Southern Resident population size in 1967 to be 96 animals. At about this time, marine mammals became popular attractions in zoos and marine parks, which increased the demand for interesting and exotic display animals. Between 1967 and 1973, it is estimated that 47 Killer Whales, mostly immature, were taken from the Southern Resident population for public display. The rapid removal of individual whales caused an immediate decline in numbers (Ford et al. 2000). By 1971, the level of removal decreased the population by about 30 percent, to approximately 67 whales (Olesiuk et al. 1990). In 1993, two decades after the live capture of Killer Whales ended, the three Southern Resident pods—J, K, and L—totaled 96 animals (Ford et al. 2000).

Over the past decade, the Southern Resident population has fluctuated. For example, the population appeared to experience a period of recovery by increasing to 99 whales in 1995, but then declined by 20 percent to 79 whales in 2001 (-3.3 percent per year) before another slight increase to 83 whales in 2003 (Ford et al. 2000, Carretta et al. 2004). NMFS (2008) estimated the 2007 population to be 87 whales. The population estimate in 2006 was approximately 90 animals (+3.5 percent per year since 2001); the decline in the 1990s, unstable population status, and population structure (e.g., few reproductive age males and non-calving adult females) continue to be causes for concern. Moreover, it is unclear whether the recent increasing trend will continue because these observations may represent an anomaly in the general pattern of survival or a longer-term shift in the survival pattern.

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