Chapter 11 Aquatic Biological Resources

11.1 Introduction

This chapter describes the environmental setting and environmental impact area for aquatic biological resources, methods of analysis, and analysis of impacts that could result from construction, operation, and maintenance of the Project. Aquatic biological resources discussed in this chapter are state and federally listed species and their critical habitats, as well as other special-status species and species of management concern and their aquatic habitats. The study area for aquatic biological resources is based on the area that could be affected by the Project and includes the following (Figure 11-1):

- Sacramento River from Keswick Dam to the Delta
- Lower American River from Nimbus Dam to its confluence with the Sacramento River
- Lower Feather River from Oroville Dam to its confluence with the Sacramento River
- Funks and Stone Corral Creeks
- CBD
- Flood bypasses (i.e., Butte Basin, Colusa Basin, Sutter Bypass, and Yolo Bypass)
- Delta (includes Suisun Marsh and Bay, San Pablo Bay, and San Francisco Bay)

For the purpose of analysis, the environmental impacts were divided into a construction impacts and operations impacts:

- Construction impacts: include areas where construction activities would be conducted plus buffer zones of varying sizes around those areas, and the areas that would be inundated by under the alternative reservoir sizes.
- Operations impacts: include areas potentially affected by operations of the Project to divert and deliver water and maintenance of the Project facilities.

Figure 11-1
Aquatic Biological Resources Study Area

The analysis used multiple factors to account for the potential effects of the Project on state and federally listed species and their critical habitats, special-status species and their aquatic habitat, and other species of management concern and their aquatic habitats. Where data were available, several analytical tools, including water operations/hydrologic models (e.g., California Water Resources Simulation Model II [CALSIM II], Delta Simulation Model II [DSM2]), were used to screen for potential direct and indirect effects and where the results indicated essentially no difference between the Project and the No Action Alternative (NAA),[1] no effect and no adverse effect were assumed. Areas of no effect included the Trinity River, Trinity Lake, and the San Joaquin River watershed upstream of the Delta. As described in Chapter 2, *Project Description and Alternatives*, the Project would not affect or result in changes in the operation of the CVP Trinity River Division facilities (including Clear Creek) and thus Trinity River resources are not discussed or analyzed further in this chapter. The presence of species and their required habitat in the Project area affected by construction further defined the geographic extent of the study area.

The effects analysis presented in this chapter is based on the comparison of the NAA to the performance of Alternatives 1, 2, and 3 in the models mentioned above. The NAA in these model runs represents 2020 baseline conditions and does not incorporate a climate change scenario. The effects of climate change on the performance of the alternatives are analyzed in Chapter 28, *Climate Change*.

Climate change is likely to alter temperature and hydrologic patterns in the Sacramento Valley. Heat waves are expected to become longer and affect larger areas, with higher daytime and nighttime temperatures and fewer cooling days. The Sacramento Valley will likely see increased precipitation during winter storms, more extreme floods, and greater floodplain vulnerability. On the dry extreme, the region will experience increased dryness and more extreme droughts. As precipitation falls more often as rain rather than snow, streamflow timing will shift from spring to winter in Sacramento Valley (Houlton and Lund 2018). These impacts may result in reduced Delta exports and reservoir carryover storage. Climate change in Critically Dry Water Years may cause storage to decrease in Shasta Lake by about 200 TAF across all months (Chapter 28).

Under climate change scenarios analyzed in Chapter 28, proposed diversions at the RBPP and Hamilton City would increase in January through March of Wet Water Years compared to Project implementation without climate change[2]. This likely reflects more precipitation falling as rain rather than snow. The Project would continue to increase Shasta Lake storage from June to October compared to the NAA with climate change. While not offsetting the entire expected reduction in storage attributable to climate change for the NAA, the increased storage would assist in managing the effects of climate change. In addition, the Project could augment flow through the Delta in October and contribute to Delta outflow during dry conditions, which would help limit increases in Delta salinity.

[1] The term *NAA*, which is identical to the No Project Alternative, is used throughout this chapter and associated aquatic resources appendices in the presentation of modeled results and represents no material difference from the No Project Alternative, as discussed in Chapter 3, *Environmental Analysis*.

[2] The RBDD, which was decommissioned in 2013, and the RBPP are co-located, and the names may be used interchangeably when referring to the geographic location.

Tables 11-1a and 11-1b summarize the CEQA determinations and NEPA conclusions for construction, operation, and maintenance impacts, respectively, between alternatives.

Table 11-1a. Summary of Construction Impacts and Mitigation Measures for Aquatic Biological Resources

Alternative	Level of Significance Before Mitigation	Mitigation Measures	Level of Significance After Mitigation
Impact FISH-1: Construction Effects on Special-Status Fish			
No Project	NI/NE		NI/NE
Alternative 1	S/SA	**Mitigation Measure VEG-2.1**: Conduct Surveys for Sensitive Natural Communities and Oak Woodlands in the Project Area Prior to Construction Activities **Mitigation Measure VEG-2.2**: Avoid and Compensate for Adverse Effects on Sensitive Natural Communities **Mitigation Measure VEG-3.2**: Compensate for Temporary and Permanent Impacts on State- or Federally Protected Wetlands **Mitigation Measure VEG-3.3**: Compensate for Temporary and Permanent Impacts on State- or Federally Protected Non-Wetland Waters	LTSM/NE
Alternative 2	S/SA	Same as Alternative 1	LTSM/NE
Alternative 3	S/SA	Same as Alternative 1	LTSM/NE

Notes:
NI = CEQA no impact
LTS = CEQA less-than-significant impact
S = CEQA significant impact
LTSM = CEQA determination of less than significant with mitigation
NE = NEPA no adverse effect
SA = NEPA substantial adverse effect

Table 11-1b. Summary of Operations and Maintenance Impacts and Mitigation Measures for Aquatic Biological Resources

Alternative	Level of Significance Before Mitigation	Mitigation Measures	Level of Significance After Mitigation
Impact FISH-2: Operations Effects on Winter-Run Chinook Salmon			
No Project	NI/NE	-	NI/NE
Alternative 1	LTS/NE		LTS/NE
Alternative 2	LTS/NE		LTS/NE
Alternative 3	LTS/NE		LTS/NE
Impact FISH-3: Operations Effects on Spring-Run Chinook Salmon			
No Project	NI/NE	-	NI/NE

Alternative	Level of Significance Before Mitigation	Mitigation Measures	Level of Significance After Mitigation
Alternative 1	LTS/NE		LTS/NE
Alternative 2	LTS/NE		LTS/NE
Alternative 3	LTS/NE		LTS/NE
Impact FISH-4: Operations Effects on Fall-Run/Late Fall–Run Chinook Salmon			
No Project	NI/NE	-	NI/NE
Alternative 1	LTS/NE		LTS/NE
Alternative 2	LTS/NE		LTS/NE
Alternative 3	LTS/NE		LTS/NE
Impact FISH-5: Operations Effects on Central Valley Steelhead			
No Project	NI/NE	-	NI/NE
Alternative 1	LTS/NE		LTS/NE
Alternative 2	LTS/NE		LTS/NE
Alternative 3	LTS/NE		LTS/NE
Impact FISH-6: Operations Effects on Green Sturgeon			
No Project	NI/NE	-	NI/NE
Alternative 1	LTS/NE	-	LTS/NE
Alternative 2	LTS/NE	-	LTS/NE
Alternative 3	LTS/NE	-	LTS/NE
Impact FISH-7: Operations Effects on White Sturgeon			
No Project	NI/NE	-	NI/NE
Alternative 1	LTS/NE	-	LTS/NE
Alternative 2	LTS/NE	-	LTS/NE
Alternative 3	LTS/NE	-	LTS/NE
Impact FISH-8: Operations Effects on Delta Smelt			
No Project	NI/NE	-	NI/NE
Alternative 1	S/SA	**Mitigation Measure FISH-8.1**: Prevent Detrimental Dissolved Oxygen and Water Temperature Effects on Fish Associated with Moving Colusa Basin Drain Water Through the Yolo Bypass **Mitigation Measure WQ-2.2**: Prevent Net Detrimental Metal and Pesticide Effects Associated with Moving Colusa Basin Drain Water Through the Yolo Bypass	LTSM/NE
Alternative 2	S/SA	**Mitigation Measure FISH-8.1**: Prevent Detrimental Dissolved Oxygen and Water Temperature Effects on Fish Associated with Moving Colusa Basin Drain Water Through the Yolo Bypass **Mitigation Measure WQ-2.2**: Prevent Net Detrimental Metal and Pesticide	LTSM/NE

Alternative	Level of Significance Before Mitigation	Mitigation Measures	Level of Significance After Mitigation
		Effects Associated with Moving Colusa Basin Drain Water Through the Yolo Bypass	
Alternative 3	S/SA	**Mitigation Measure FISH-8.1**: Prevent Detrimental Dissolved Oxygen and Water Temperature Effects on Fish Associated with Moving Colusa Basin Drain Water Through the Yolo Bypass **Mitigation Measure WQ-2.2**: Prevent Net Detrimental Metal and Pesticide Effects Associated with Moving Colusa Basin Drain Water Through the Yolo Bypass	LTSM/NE
Impact FISH-9: Operations Effects on Longfin Smelt			
No Project	NI/NE	-	NI/NE
Alternative 1	S/SA	**Mitigation Measure FISH-9.1**: Tidal Habitat Restoration for Longfin Smelt	LTSM/NE
Alternative 2	S/SA	**Mitigation Measure FISH-9.1**: Tidal Habitat Restoration for Longfin Smelt	LTSM/NE
Alternative 3	S/SA	**Mitigation Measure FISH-9.1**: Tidal Habitat Restoration for Longfin Smelt	LTSM/NE
Impact FISH-10: Operations Effects on Lampreys			
No Project	NI/NE	-	NI/NE
Alternative 1	LTS/NE	-	LTS/NE
Alternative 2	LTS/NE	-	LTS/NE
Alternative 3	LTS/NE	-	LTS/NE
Impact FISH-11: Operations Effects on Native Minnows (Sacramento Splittail, Sacramento Hitch, Hardhead, and Central California Roach)			
No Project	NI/NE	-	NI/NE
Alternative 1	LTS/NE	-	LTS/NE
Alternative 2	LTS/NE	-	LTS/NE
Alternative 3	LTS/NE	-	LTS/NE
Impact FISH-12: Operations Effects on Starry Flounder and Northern Anchovy			
No Project	NI/NE	-	NI/NE
Alternative 1	LTS/NE	-	LTS/NE
Alternative 2	LTS/NE	-	LTS/NE
Alternative 3	LTS/NE	-	LTS/NE
Impact FISH-13: Operations Effects on Striped Bass			
No Project	NI/NE	-	NI/NE
Alternative 1	LTS/NE	-	LTS/NE
Alternative 2	LTS/NE	-	LTS/NE
Alternative 3	LTS/NE	-	LTS/NE

Alternative	Level of Significance Before Mitigation	Mitigation Measures	Level of Significance After Mitigation
Impact FISH-14: Operations Effects on American Shad			
No Project	NI/NE	-	NI/NE
Alternative 1	LTS/NE	-	LTS/NE
Alternative 2	LTS/NE	-	LTS/NE
Alternative 3	LTS/NE	-	LTS/NE
Impact FISH-15: Operations Effects on Threadfin Shad			
No Project	NI/NE	-	NI/NE
Alternative 1	LTS/NE	-	LTS/NE
Alternative 2	LTS/NE	-	LTS/NE
Alternative 3	LTS/NE	-	LTS/NE
Impact FISH-16: Operations Effects on Black Bass (Largemouth Bass, Smallmouth Bass, and Spotted Bass)			
No Project	NI/NE	-	NI/NE
Alternative 1	LTS/NE	-	LTS/NE
Alternative 2	LTS/NE	-	LTS/NE
Alternative 3	LTS/NE	-	LTS/NE
Impact FISH-17: Operations Effects on California Bay Shrimp			
No Project	NI/NE	-	NI/NE
Alternative 1	LTS/NE	-	LTS/NE
Alternative 2	LTS/NE	-	LTS/NE
Alternative 3	LTS/NE	-	LTS/NE
Impact FISH-18: Operations Effects on Reservoir Fish Species			
No Project	NI/NE	-	NI/NE
Alternative 1	LTS/B	-	LTS/B
Alternative 2	LTS/B	-	LTS/B
Alternative 3	LTS/B	-	LTS/B
Impact FISH-19: Operations Effects on Southern Resident Killer Whale			
No Project	NI/NE	-	NI/NE
Alternative 1	LTS/NE	-	LTS/NE
Alternative 2	LTS/NE	-	LTS/NE
Alternative 3	LTS/NE	-	LTS/NE
Impact FISH-20: Maintenance Effects on Fish and Aquatic Biological Resources			
No Project	NI/NE	-	NI/NE
Alternative 1	LTS/NE	-	LTS/NE
Alternative 2	LTS/NE	-	LTS/NE
Alternative 3	LTS/NE	-	LTS/NE

Notes:
NI = CEQA no impact
LTS = CEQA less-than-significant impact
S = CEQA significant impact
LTSM = CEQA less than significant with mitigation
B = NEPA beneficial effects
NE = NEPA no adverse effect
SA = NEPA substantial adverse effect

11.2 Environmental Setting

11.2.1. Fish and Aquatic Species of Management Concern

Aquatic habitats include riverine and estuarine habitats. As noted above, these include San Francisco Bay, San Pablo Bay, Suisun Marsh, Suisun Bay, and the Delta, and extends upstream within the channels of the Sacramento River to Keswick Dam, the American River to Nimbus Dams, Feather River to Oroville Dam, Funks and Stone Corral Creeks, the CBD, and flood bypasses (i.e., Butte Basin, Sutter Bypass, and Yolo Bypass). Fish species that occur in each of the aquatic habitats are listed in Table 11-2.

Fish and aquatic species were selected for analysis based on their regulatory, commercial, or recreational applicability importance and/or vulnerability and their potential to be affected by construction activities, operations (diversions) associated with the Project, and changes in CVP and SWP system-wide operations that would be expected to be implemented in response to, or in conjunction with, the Project (e.g., water transfers and exchanges, changes to export operations) (Table 11-2). These fish species, referred to herein as the species of management concern, include species listed by state or federal agencies as endangered or threatened or listed on the 2015 California Department of Fish and Wildlife (CDFW) Fish Species of Special Concern in California. Species of management concern also include those of tribal, commercial, or recreational importance. In addition to the species listed in Table 11-2, the Southern Resident killer whale (*Orcinus orca*) distinct population segment (DPS), which is federally listed as endangered, is also considered because of potential effects on its prey, Chinook salmon (*Oncorhynchus tshawytscha*). The species of management concern are listed in Table 11-2 with the general geographic areas where they occur. Species descriptions are provided in Appendix 11A, *Aquatic Species Life Histories*.

Table 11-2. Aquatic Species of Management Concern by Area of Occurrence

Species and *ESU/DPS*	Federal Status	State Status	Tribal, Commercial, or Recreational Importance	Geographic Areas of Occurrence within Region
Winter-run Chinook Salmon *Sacramento River ESU*	Endangered	Endangered	Yes	Sacramento River, Delta, Suisun Bay, San Pablo Bay, San Francisco Bay
Spring-run Chinook Salmon *Central Valley ESU*	Threatened	Threatened	Yes	Sacramento, Feather, and American Rivers, Delta, Suisun Bay, San Pablo Bay, San Francisco Bay
Fall-run/Late Fall–run Chinook Salmon *Central Valley ESU*	Species of Concern	Species of Special Concern	Yes	Sacramento, Feather, and American Rivers, Delta, Suisun Bay, San Pablo Bay, San Francisco Bay
Steelhead *California Central Valley DPS*	Threatened	None	Yes	Sacramento, Feather, and American Rivers, Delta, Suisun Bay, San Pablo Bay, San Francisco Bay

Species and *ESU/DPS*	**Federal Status**	**State Status**	**Tribal, Commercial, or Recreational Importance**	**Geographic Areas of Occurrence within Region**
Delta Smelt	Threatened	Endangered	No	Delta and Suisun Marsh
Longfin Smelt *Bay Delta DPS*	Candidate	Threatened, Species of Special Concern	No	Delta, Suisun Bay, San Pablo Bay, San Francisco Bay
Green Sturgeon *Southern DPS*	Threatened	Species of Special Concern	Yes	Sacramento and Feather Rivers, Delta, Suisun Bay, San Pablo Bay, San Francisco Bay
White Sturgeon	None	Species of Special Concern	Yes	Sacramento, Feather, and American Rivers, Delta, Suisun Bay, San Pablo Bay, San Francisco Bay
Pacific Lamprey	Species of Concern	Species of Special Concern	Yes	Sacramento, Feather, and American Rivers, Delta, Suisun Bay, San Pablo Bay, San Francisco Bay
River Lamprey	None	Species of Special Concern	Yes	Sacramento, Feather, and American Rivers, Delta, Suisun Bay, San Pablo Bay, San Francisco Bay
Sacramento Hitch	None	Species of Special Concern	No	Sacramento, Feather, and American Rivers, Funks Creek, Stone Corral Creek, CBD, and Delta
Sacramento Splittail	None	Species of Special Concern	No	Sacramento, Feather, and American Rivers, Delta, and Suisun Marsh
Hardhead	None	Species of Special Concern	No	Sacramento, Feather, and American Rivers, and Delta
Sacramento Perch	None	Species of Special Concern	No	In pond at Gray Lodge Wildlife Area
Central California Roach	None	Species of Special Concern	No	Sacramento, Feather, and American Rivers, CBD, Stone Corral Creek, and Delta
Starry Flounder	None	None	Yes	Delta, Suisun Bay, San Pablo Bay, San Francisco Bay
Northern Anchovy	None	None	Yes	Delta, Suisun Bay, San Pablo Bay, San Francisco Bay
Striped Bass	None	None	Yes	Sacramento, Feather, and American Rivers, Delta, Suisun Bay, San Pablo Bay, San Francisco Bay
American Shad	None	None	Yes	Sacramento, Feather, and American Rivers, and Delta, Suisun Bay, San Pablo Bay, San Francisco Bay

Species and *ESU/DPS*	Federal Status	State Status	Tribal, Commercial, or Recreational Importance	Geographic Areas of Occurrence within Region
Threadfin Shad	None	None	Yes	Delta and Suisun Marsh
Black Bass (Largemouth, Smallmouth, Spotted)	None	None	Yes	Sacramento, Feather, and American Rivers, Delta and Suisun Marsh, reservoirs
California Bay Shrimp	None	None	Yes	Delta, Suisun Bay, San Pablo Bay, San Francisco Bay

ESU = evolutionarily significant unit; DPS = distinct population segment.

11.2.2. Habitat Conditions and Environmental Stressors

The sections below describe and characterize habitats with properties needed to support the different life stages of the fish species of management concern that rely on the geographic area being evaluated. Environmental stressors are factors that limit a habitat's capacity to support the life stages present. The descriptions focus on stressors that potentially would be affected by the Project. For example, habitat availability for salmonid egg and alevin life stages refers to, among other properties, river bottom substrates with clean suitably sized gravels, suitable and stable water levels, cool water temperatures, and adequate flow to supply dissolved oxygen (DO) to the developing eggs and alevins. Environmental stressors potentially limiting spawning habitat availability could include inadequate flow, warm water temperatures, or insufficient spawning gravels.

11.2.3. Delta and Suisun Bay/Marsh

11.2.3.1. Description of Delta and Suisun Bay/Marsh

Ecologically, the Delta consists of three geographic regions: (1) the north Delta freshwater flood basins composed primarily of freshwater inflow from the Sacramento River system; (2) the south Delta distributary channels composed of predominantly San Joaquin River system inflow; and (3) the central Delta tidal islands landscape wherein the Sacramento, San Joaquin, and eastside tributary flows converge and tidal influences from San Francisco Bay are greater.

Suisun Bay and Marsh are ecologically linked with the western portions of the central Delta, although with different tidal and salinity conditions than are found upstream. Suisun Bay and Marsh are the largest expanse of remaining tidal marsh habitat within the greater San Francisco Estuary ecosystem and include Honker, Suisun, and Grizzly Bays; Montezuma and Suisun Sloughs; and numerous other smaller channels and sloughs.

11.2.3.2. Habitat Conditions and Environmental Stressors in Delta and Suisun Bay/Marsh

A summary of habitat conditions and environmental stressors in the Delta and Suisun Bay/Marsh was recently provided by the California Department of Water Resources (DWR) (2020).

Delta

Aquatic Habitat

Flow management in the Delta has altered the aquatic habitat by: (1) changing aspects of the historical flow regime (e.g., timing, magnitude, duration) that supported life history traits of native species; (2) limiting access to or quality of habitat; (3) contributing to conditions better suited to invasive, nonnative species (e.g., reduced spring flows, increased summer inflows and exports, and low and less-variable interior Delta salinity [Moyle and Bennett 2008]); and (4) causing net reverse flows in channels leading to CVP and SWP south Delta export facilities that can entrain fish (Mount et al. 2012). Native species of the Delta are adapted to and depend on variable flow conditions at multiple scales, which are influenced by the region's dramatic seasonal and interannual climatic variation. In particular, most native fishes evolved reproductive or outmigration timing associated with historical peak flows during spring (Moyle 2002).

Low DO levels have been measured in the San Joaquin River, particularly in the Deep Water Ship Channel from the Port of Stockton 7 miles downstream to Turner Cut (Lee and Jones-Lee 2003). These conditions are the result of increased residence time of water combined with high oxygen demand in the anthropogenically modified channel, which leads to DO depletion, especially near the sediment-water interface (San Joaquin Tributaries Authority 2012). Despite these conditions, adult Chinook salmon and steelhead (*Oncorhynchus mykiss*) migration does not appear to be adversely affected (Pyper et al. 2006). However, Hallock et al. (1970) found that during the 1960s, adult radio-tagged Chinook salmon delayed their upstream migration whenever DO concentrations were less than 5 milligrams per liter (mg/L) at Stockton. It has been shown that low DO conditions in the San Joaquin River can be ameliorated somewhat through installation of the Head of the Old River Barrier, which increases San Joaquin River flows (San Joaquin Tributaries Authority 2012).

Researchers have studied the impacts of water export on Delta flow and velocity using hydrodynamic models. The Salmonid Scoping Team (SST) recently provided a summary of these effects (Salmonid Scoping Team 2017). The SST concluded that the effect of the SWP water exports on Delta flow and velocity varied as a function of distance from the facility as well as a function of export volume, total Delta inflow, and tidal action. While export rates had little effect on distributaries such as Georgiana Slough, a much greater effect exists in the south Delta, particularly in Old River near the CVP and SWP south Delta export facilities.

Water temperatures in the Delta follow a seasonal pattern of winter cold-water conditions and summer warm-water conditions, largely because of the region's Mediterranean climate and its alternating cool/wet and hot/dry seasons. Generally, summer water temperatures in the Delta are largely driven by ambient air temperatures (Wagner et al. 2011), which can approach or exceed the upper thermal tolerances (e.g., 20°C to 25°C) for cold-water fish species such as salmonids and Delta-dependent species such as delta smelt (*Hypomesus transpacificus*) and longfin smelt (*Spirinchus thaleichthys*) (Jeffries et al. 2016). This phenomenon is particularly prevalent in parts of the south Delta and San Joaquin River, and potentially restricts the distribution of these species and precludes the use of previously important rearing areas (National Research Council 2012). Recent research indicates that thermal stratifications can occur in parts of the Delta and downstream bays (Vroom et al. 2017), which suggests that thermal refugia may be available to sensitive fish species.

Landscape-scale changes have resulted from tidal land reclamation for agriculture and flood management infrastructure that began in the mid-nineteenth century, along with flow modifications that have occurred since the construction of flood control and water supply facilities in the late nineteenth and early twentieth century, and have eliminated most of the historical tidal wetlands and floodplain habitat in the Delta and its tributaries, thereby degrading and diminishing Delta habitat for native plant and animal communities (Mount et al. 2012). The reduction of hydrologic variability and landscape complexity, combined with degradation of water quality, has supported invasive aquatic species, which have further degraded conditions for native species. Consequently, over time, the Delta has undergone an ecological regime shift unfavorable to many native species (Baxter et al. 2010), including delta smelt, longfin smelt, Sacramento splittail (*Pogonichthys macrolepidotus*), green sturgeon (*Acipenser medirostris*), white sturgeon (*Acipenser transmontanus*), and Chinook salmon (Jassby et al. 1995; Kimmerer 2002a; Rosenfield and Baxter 2007; Kimmerer et al. 2009; Fish 2010; Perry et al. 2012; Thomson et al. 2010; Feyrer et al. 2010; Loboschefsky et al. 2012; Mount et al. 2012; Heublein et al. 2017).

In response to these landscape conditions, DWR is working with California EcoRestore to advance the restoration of at least 30,000 acres of tidal wetland, floodplain habitat, and riparian habitat throughout the Delta. DWR is the lead agency on 28 of the 30 EcoRestore projects, including but not limited to Decker Island, Bradmoore Island, Lookout Slough Tidal Habitat Restoration and Flood Improvement Project, Winter Island, and the Tule Red Project (California Department of Water Resources 2019a). As these projects are implemented, they would be adaptively managed to improve habitat for delta smelt and other species. DWR is also working with other resource agencies, including CDFW, which is leading the effort, to explore the feasibility of restoring a portion of Franks Tract to reduce invasive weeds and predation while increasing turbidity and fish food production (California Department of Fish and Wildlife 2018). Other public water agencies also are conducting restoration planning activities including the Delta Island Adaptations planning effort that could result in restoration on Bouldin Island, Holland Tract, Webb Tract, and Bacon Island.

Salinity

Salinity is a critical factor influencing the distribution of aquatic communities in the Delta. Although estuarine fish species are generally tolerant of a range of salinities, this tolerance varies by species and life stage. Some species can be highly sensitive to excessively low or high salinity during physiologically vulnerable periods, such as reproductive and early life stages. While most of the Delta contains fresh water year-round due to managed inflows from rivers, the south Delta can have low salinity because of agricultural return water and the west Delta contains a fresh water/salt water prism typical of estuaries. This tidal prism moves east and west in the Delta with the tides and the distance it moves may be influenced by the volume of fresh water inflow from tributary rivers, primarily the Sacramento River.

X2, which represents the location of the 2 parts per thousand (ppt) isohaline and is measured in kilometers as the distance to the Golden Gate Bridge, has been used to determine the location of the low-salinity zone and the extent of habitat for oligohaline fish species and their habitat.

Several researchers (e.g., Feyrer et al. 2007, 2011; Kimmerer 2002a, 2002b; Kimmerer et al. 2009, Jassby et al. 1995) have explored the relationship between X2 and population status of fishes in the Delta. The U.S. Fish and Wildlife Service (USFWS) (2008 and 2019) and the State Water Resources Control Board (State Water Board) have relied on this work to establish the location of X2 as a factor in regulating the effect of the State and federal water projects on habitat quantity and quality for these species. Water Right Decision 1641: Implementation of Water Quality Objectives for the San Francisco Bay/Sacramento–San Joaquin Delta Estuary (D-1641) was developed by the State Water Board and establishes flow, water quality, and monitoring requirements for the state and federal water projects (State Water Resources Control Board 1999).

More recent research (e.g., Hutton et al. 2015, Mac Nally et al. 2010, Murphy and Weiland 2019) has demonstrated the limitations of using a single location for evaluating salinity trends in the estuary, questioned the appropriateness of a single measure (X2) as a surrogate for regulating the complexity between flow, salinity, habitat quality, and population response of Delta pelagic fishes in the estuary. Similarly, Tamburello et al. (2019) reported that the predictive power of the X2-abundance relationships for several species declined as the relationships were tested with new data, suggesting that the relationships are breaking down over time. Nevertheless, X2 remains a regulatory standard for maintenance of habitat for Delta fish species (D-1641).

Nutrients and Foodweb Support

Nutrients are essential components of terrestrial and aquatic environments because they provide a resource base for primary producers. Typically, in freshwater aquatic environments, phosphorus is the primary limiting macronutrient, whereas in marine aquatic environments, nitrogen tends to be limiting. A balanced range of abundant nutrients provides optimal conditions for maximum primary production (phytoplankton), a robust foodweb (zooplankton and macroinvertebrates), and productive fish populations. However, changes in nutrient loadings and forms, excessive amounts of nutrients, and altered nutrient ratios can lead to a suite of problems in aquatic ecosystems, such as low DO concentrations, un-ionized ammonia, excessive growth of toxic forms of cyanobacteria, and changes in components of the foodweb. Nutrient concentrations in the Delta have been well studied (Jassby et al. 2002; Kimmerer 2004; Van Nieuwenhuyse 2007; Glibert et al. 2011, 2014; Saleh and Domagalski 2021).

Estuaries are commonly characterized as highly productive nursery areas for numerous aquatic organisms. Nixon (1988) noted that there is a broad continuum of primary productivity levels in different estuaries, which affects fish production and abundance. Compared to other estuaries, pelagic primary productivity in the upper San Francisco Estuary is relatively poor, and a relatively low fish yield is expected (Wilkerson et al. 2006). In the Delta and Suisun Marsh, this appears to result from turbidity, clam grazing (Jassby et al. 2002), and nitrogen and phosphorus dynamics (Wilkerson et al. 2006; Van Nieuwenhuyse 2007; Glibert et al. 2011, 2014).

A significant long-term decline in phytoplankton biomass (chlorophyll a) and primary productivity to low levels has occurred in the Suisun Bay region and the Delta (Jassby et al. 2002). Shifts in nutrient concentrations, such as high levels of ammonium and nitrogen relative to phosphorus (i.e., the ratios of nitrogen to phosphorus and ammonium to nitrate), may contribute to the phytoplankton reduction and to changes in algal species composition in the San

Francisco Estuary (Wilkerson et al. 2006; Dugdale et al. 2007; Lehman et al. 2005, 2008a, 2010; Glibert et al. 2011, 2014). However, a recent analysis concluded high ammonium loading is not a driver of low productivity in the Delta area (Strong et al. 2021). Low and declining primary productivity in the estuary may be contributing to the long-term pattern of relatively low and declining biomass of pelagic fishes (Jassby et al. 2002), although the statistical analyses by Mac Nally et al. (2010) and Thomson et al. (2010) found limited statistical evidence for a linkage between chlorophyll and pelagic fish.

The introductions of two clams from Asia have led to major alterations in the foodweb in the Delta. Overbite clams (*Potamocorbula amurensis*) are most abundant in the brackish and saline water of Suisun Bay and the west Delta, and Asian clams (*Corbicula fluminea*) are most abundant in the fresh water of the central Delta. These filter feeders significantly reduce the phytoplankton and zooplankton concentrations in the water column, reducing food availability for native fishes such as delta smelt and young Chinook salmon (Feyrer et al. 2007; Kimmerer 2002a; Kimmerer and Thompson 2014). The overbite clam also has been implicated in the reduction of the native opossum shrimp, a preferred food of Delta native fishes such as Sacramento splittail and longfin smelt (Feyrer et al. 2003).

In addition, introduction of the clams led to the decline of native copepods of higher food quality and the establishment of poorer quality nonnative copepods. The clams have been cited for the decline in *Neomysis mercedis* (Orsi and Mecum 1986; Feyrer et al. 2003), the shift in distribution of anchovies (Kimmerer 2006) and young-of-the-year striped bass (*Morone saxatilis*) (Kimmerer et al. 2000; Feyrer et al. 2003; Sommer et al. 2007), as well as the decline in diatoms (Kimmerer 2005) and several zooplankton species (Kimmerer et al. 1994). The impact of the clams on chlorophyll and the Delta ecosystem is also reflected by a shift in many of the original correlations between species abundance and X2, that occurred after the establishment of the clams (Kimmerer 2002b; Sommer et al. 2007).

More recently, the cyclopoid copepod, *Limnoithona*, has rapidly become the most abundant copepod in the Delta since its introduction in 1993 (Hennessy and Enderlein 2013). This species is hypothesized to be a low-quality food source and intraguild competitor of native and nonnative calanoid copepods (Gould and Kimmerer 2010).

Studies on food quantity and quality were historically relatively limited in the San Francisco Estuary, with limited information available regarding long-term trends. Nonetheless, several studies have documented or suggested the food limitations for aquatic species in the estuary, including zooplankton (Mueller-Solger et al. 2002; Kimmerer et al. 2005), delta smelt (Bennett 2005; Bennett et al. 2008), Chinook salmon (Sommer et al. 2001), Sacramento splittail (Greenfield et al. 2008), striped bass (Loboschefsky et al. 2012), and largemouth bass (*Micropterus salmoides*) (Nobriga 2009). Additional recent studies have shed light on factors such as the correlation of Delta outflow with the subsidy (transport) of the delta smelt zooplankton prey *Pseudodiaptomus forbesi* to the low-salinity zone from the freshwater Delta during July–September (Kimmerer et al. 2018), which has implications for species feeding on *P. forbesi* such as delta smelt and longfin smelt.

Turbidity

Turbidity is an important water quality component in the Delta that affects physical habitat through sedimentation and foodweb dynamics by means of attenuation of light in the water column. Light attenuation, in turn, affects the extent of the photic zone where primary production can occur and the ability of predators to locate prey and for prey to escape predation.

Turbidity has been declining in the Delta, as indicated by sediment data collected by the U.S. Geological Survey since the 1950s (Wright and Schoellhamer 2004), and the decline has important implications for foodweb dynamics, nonnative submerged aquatic vegetation growth, and predation. Higher water clarity is at least partially caused by increased water filtration and plankton grazing by highly abundant overbite clams and other benthic organisms (Kimmerer 2004; Greene et al. 2011) and potentially by filtration by high densities of aquatic vegetation (Hestir et al. 2016). High nutrient loads coupled with reduced sediment loads and higher water clarity could contribute to plankton and algal blooms and overall increased eutrophic conditions in some areas (Kimmerer 2004). As noted in Chapter 28, *Climate Change*, recent modeling examining future climate scenarios predicts significant increases in large flow events and sediment loading to the Delta from the Sacramento River over the next century for two representative greenhouse gas concentration pathways, which could increase turbidity (Stern et al. 2020).

Turbidity has been identified as an important component of delta smelt habitat, with turbidity greater than 12 nephelometric turbidity units (NTU) (Sommer and Mejia 2013) or below Secchi depth of 30–40 cm (Hamilton and Murphy 2020) being of higher quality. The first high-flow events of winter create turbid conditions in the Delta, which can be drawn into the south Delta during reverse flow conditions in Old and Middle River, driven by south Delta CVP and SWP exports. Delta smelt may follow turbid waters into the southern Delta, increasing their proximity to CVP and SWP export facilities. This may increase their entrainment risk at the CVP and SWP pumping facilities (U.S. Fish and Wildlife Service 2008). In response to the delta smelt resiliency strategy, DWR assessed the feasibility of adding sediment to increase turbidity in the low-salinity zone of the Delta to improve delta smelt habitat conditions. Computer modeling was performed to assess (1) whether sediment supplementation is feasible, (2) the magnitude of supplement that would be required to affect turbidity, and (3) the spatial and temporal extent of supplementation to affect overall turbidity in the low-salinity zone of the Delta (California Natural Resources Agency 2017). The results of the modeling predicted that 3,350 cubic yards per day of sediment release was needed to increase turbidity by 10 NTU from Emmaton and Mallard Island (Bever and MacWilliams 2018).

Contaminants

Contaminants can change ecosystem functions and productivity through numerous pathways. Trends in contaminant loadings have become better understood in recent years, but their ecological effects are less well understood in the San Francisco Estuary (Johnson et al. 2010; Brooks et al. 2012; Fong et al. 2016). A large body of research has been conducted on contaminant occurrence and effects on aquatic organisms in the Delta. A wide array of contaminants, including pesticides, metals, pharmaceuticals, and personal care products, have been detected in Delta water and sediment. Recent monitoring programs are routinely detecting multiple pesticides in each water sample from the Delta. Fong et al. (2016) report that “For example, 27 pesticides or degradation products were detected in Sacramento River samples, and the average number of pesticides per sample was six. In San Joaquin River samples, 26 pesticides or degradation products were detected, and the average number detected per sample was 9.” Because laboratory analytical protocols generally detect only those pesticides for which the protocol is designed (i.e., laboratory tests can only detect the pesticides they look for), it is likely that the pesticides identified by Fong et al. (2016) are only a subset of the actual number of pesticides and other contaminants in the Delta. The effects of chemical mixtures on aquatic organisms is generally unknown but many chemicals may have additive or synergistic effects. Anthropogenic toxins cause significant disruption to development, reduce growth and recruitment, and increase mortality (Johnson et al. 2010).

In addition to anthropogenic contaminants, natural toxins are associated with blooms of *Microcystis aeruginosa*, a cyanobacterium (blue-green algae) that releases a potent toxin known as microcystin. Toxic microcystins cause foodweb impacts at multiple trophic levels, and histopathological studies of fish liver tissue suggest that fish exposed to elevated concentrations of microcystins have developed liver damage and tumors (Lehman et al. 2005, 2008b, 2010; Acuña et al. 2012a, 2012b). A recent study evaluating 5 years of data collected throughout the Delta and into Suisun Bay indicated that presence of several genera of phytoplankton and cyanobacteria, some of which are potentially toxic, are correlated with *Microcystis* abundance (Lehman et al. 2021), which suggests that *Microcystis* influences ecosystem function in multiple ways (directly through toxin production and indirectly through changes in plankton communities).

There are longstanding concerns related to mercury and selenium in the Sacramento and San Joaquin watersheds, the Delta, and San Francisco Bay. DWR is conducting an additional study to determine imports and exports of mercury and methylmercury from freshwater tidal wetlands in the Delta and Suisun Marsh per the Sacramento-San Joaquin Delta Methylmercury TMDL (total maximum daily load) and Basin Plan Amendment (Lee and Manning 2020; Wood et al. 2010). Current research shows that tidal wetlands do not export mercury or methylmercury in large amounts, although seasonal differences occur and imports and exports are heavily influenced by flow and whether the wetland is associated with a floodplain (Mitchell et al. 2012; Lee and Manning 2020). Methylmercury increases in concentration at each level in the food chain and can cause concern for people and birds that eat piscivorous fish (e.g., striped bass) and benthic fishes such as sturgeon. Studies summarized by Alpers et al. (2008) indicate that mercury in fish has been linked to hormonal and reproductive effects, liver necrosis, and altered behavior in fish. A study by Lee et al. (2011) on dietary methylmercury noted significant abnormalities in the liver and kidneys, lower growth rates, and higher mortality in both green sturgeon and white sturgeon, but particularly in green sturgeon. With regard to selenium, benthic foragers like diving ducks, sturgeon, and Sacramento splittail have the greatest risk of selenium toxicity because of selenium presence in nonnative benthic bivalves and relatively low selenium loss rates in these bivalves relative to other food sources such as copepods (Stewart et al. 2004). Beckon and Maurer (2008) suggest that salmonids are probably among the species that are most sensitive to selenium, while delta smelt are likely to be at low risk of selenium toxicity. The invasion of the nonnative bivalves (e.g., overbite clams) has resulted in increased bioavailability of selenium to benthivores in San Francisco Bay (Linville et al. 2002).

Baxter et al. (2008) prepared a 2007 synthesis of results as part of a Pelagic Organism Decline (POD) Progress Report, including a summary of prior studies of contaminants in the Delta. The summary included studies, which suggested that phytoplankton growth rates may be inhibited by localized high concentrations of herbicides (Edmunds et al. 1999). Toxicity to invertebrates has been noted in water and sediments from the Delta and associated watersheds (Kuivila and Foe 1995; Weston et al. 2004, 2014, 2019). The 2004 Weston study of sediment toxicity recommended additional study of the effects of the pyrethroid insecticides on benthic organisms. Undiluted drainwater from agricultural drains in the San Joaquin River watershed can be acutely toxic (i.e., quickly lethal) to fish (e.g., Chinook salmon and striped bass) and have chronic effects on growth, likely because of high concentrations of major ions (e.g., sodium, sulfates) and trace elements (e.g., chromium, mercury, selenium) (Saiki et al. 1992).

A more recent synthesis of contaminant studies described multiple lines of evidence that contaminants affect species of concern in the Delta (Fong et al. 2016). Fong et al. (2016) reported that many contaminants detected in Delta waters exceed regulatory standards and most water samples contain multiple contaminants. They also summarize the multiple studies that have found sublethal, lethal, chronic, and acute toxicity of Delta water to test species and Delta species of concern, including delta smelt and Chinook salmon.

Fish Passage and Entrainment

With its complex network of channels, low eastern and southern tributary inflows, and reverse flows created by pumping for water exports in the south Delta, the Delta presents a challenge for anadromous and resident fish during upstream and downstream migration. These complex conditions can lead to straying, migration delays, extended exposure to predators, and entrainment during outmigration. Tidal elevations, salinity, turbidity, Delta inflow, meteorological conditions, season, habitat conditions, and south Delta CVP and SWP exports all have the potential to influence fish movement, currents, and ultimately the level of entrainment and fish passage success and survival (see, for example, the review by Salmonid Scoping Team 2017). Many studies have been conducted to identify the effects of CVP and SWP operations on anadromous salmonid routing into the central Delta and entrainment at the CVP and SWP pumping facilities. These studies provide context regarding potential effects of the CVP and SWP facilities and operations but study limitations and uncertainty regarding the applicability to future operations is acknowledged for three main reasons. First, these studies were conducted when previous regulatory requirements were in place (e.g., previous biological opinions) and, consequently, CVP and SWP operations may have been substantially different compared to current operations. Second, these studies were conducted to answer specific questions and should be interpreted in the context of the hypotheses and questions they were designed to address. Lastly, fish population abundance and migration timing is not constant from year to year, and particularly for anadromous species, for which it varies depending on antecedent hydrologic and water temperature conditions in upstream tributaries. Therefore, the uncertainty in the frequency, magnitude, and duration of effects of CVP and SWP operations on Delta hydrodynamic conditions and entrainment at the CVP and SWP facilities is acknowledged.

North Delta Fish Passage and Entrainment

In the north Delta, migrating fish have multiple potential pathways as they move upstream into the Sacramento or San Joaquin River systems. Marston et al. (2012) studied stray rates for immigrating San Joaquin River Basin adult salmon that stray into the Sacramento River Basin. Results indicated that it was unclear whether reduced San Joaquin River pulse flows or elevated south Delta exports caused increased stray rates during the period evaluated (1979–2007). The Delta Cross Channel (DCC), when open, can entrain fish into the interior Delta from the Sacramento River as they emigrate. The opening of the DCC when Chinook salmon are returning to spawn to the Mokelumne and Cosumnes Rivers is believed to lead to increased straying of these fish into the American and Sacramento Rivers because of confusion over olfactory cues associated with Sacramento River water flow (through the DCC) into the lower forks of the Mokelumne River. Experimental DCC closures during the fall-run Chinook salmon migration season for selected days, coupled with pulsed flow releases from reservoirs on the Mokelumne River, have been implemented to reduce straying rates of returning adults. These closures have corresponded with reduced recoveries of Mokelumne River Hatchery fish in the American River system and increased returns to the Mokelumne River Hatchery (East Bay Municipal Utility District 2012).

Migrating adult Chinook salmon and sturgeon may also enter the Yolo Bypass via the Cache Slough Complex and Toe Drain, where they may become stranded. See Section 11.2.4.3, *Yolo Bypass*, for a discussion of the Yolo Bypass.

Water quality can also affect fish passage in the north Delta. Water quality in the mainstem Sacramento River and its distributary sloughs can be poor at times during summer, creating conditions that may stress migrating fish or even impede migration. These conditions include low DO and high water temperatures. For adult Chinook salmon, DO concentration less than 3 to 5 mg/L can impede migration (Hallock et al. 1970), as can mean daily water temperatures of 70°F to 73°F (approximately 21°C to 23°C), depending on whether water temperatures are rising or falling (Strange 2010). DO levels are generally greater than 5 mg/L throughout the Delta, but water temperatures can exceed these thresholds during summer and fall. Contaminants at concentrations that have been detected in the Delta have also been found to impair olfactory responses in many fish, which can lead to straying (Fong et al. 2016; Sandahl et al. 2007; Tierney et al. 2010).

Michel (2010) and Michel et al. (2015) used acoustic telemetry to examine survival of late fall–run Chinook salmon smolts outmigrating from the Sacramento River through the Delta and San Francisco Estuary. Survival was lowest in the tidally influenced freshwater portion (Delta) and the brackish portion of the estuary (bays downstream of the Delta) relative to survival in the non-tidal riverine portion of the migration route. Michel et al. (2015) suggested that the time spent in each region influenced survival (i.e., survival was highest in the channelized riverine reaches because individuals migrate through those reaches relatively quickly rather than rearing for longer periods of time).

Analyses by Perry et al. (2015, 2018) suggests that the mechanisms governing route selection are complex. Their analyses revealed the strong influence of tidal forcing on the probability of fish entrainment into the interior Delta. The probability of entrainment into Georgiana Slough was highest during low Sacramento River flow and associated reverse-flow flood tides, and the probability of fish remaining in the Sacramento River was near zero during flow reversals (Perry et al. 2015). The magnitude and duration of reverse flows at this river junction decrease as flow in the Sacramento River increases. Consequently, reduced Sacramento River inflow increases the frequency of reverse flows at this junction, thereby increasing the proportion of fish that are entrained into the interior Delta, where mortality is higher relative to the mainstem Sacramento River and west Delta (Perry 2010).

Central and South Delta Fish Passage and Entrainment

The south Delta intake facilities include the SWP and CVP south Delta export facilities; local agency intakes, including Contra Costa Water District intakes; and agricultural intakes. Contra Costa Water District intakes include fish screens. However, many agricultural intakes throughout the Delta do not include fish screens. Water flow patterns in the south Delta are influenced by water diversion actions and operations, seasonal temporary barriers, and tides and river inflows to the Delta (Kimmerer and Nobriga 2008). Water from the San Joaquin River mainly moves downstream through the head of Old River and through the channels of the Old and Middle rivers and Grant Line and Fabian-Bell Canals toward the CVP and SWP south Delta intake facilities. Conversely, when water to the north of the diversion points for the two facilities moves southward (upstream), the net flow is negative (toward) the export facilities. When the temporary rock barriers are installed from April through November to protect water quality and stage elevation for local diverters, internal reverse circulation is created within the channels isolated by the barriers from other portions of the south Delta. These conditions are most pronounced during late spring through fall when San Joaquin River inflows are low and water diversion rates are typically high. Drier hydrologic years also reduce the frequency of net downstream flows in the south Delta and mainstem San Joaquin River. While Delta flows are tidal and naturally reverse twice daily, Delta diversions create net reverse flows, which may draw (entrain) fish toward CVP and SWP south Delta export facilities (Arthur et al. 1996; Kimmerer et al. 2008; Grimaldo et al. 2009; Grimaldo et al. 2021).

A portion of fish that enter the Jones Pumping Plant approach channel and the Clifton Court Forebay are salvaged at screening and fish salvage facilities, transported downstream by trucks, and released. The National Marine Fisheries Service (NMFS) (2009) estimated that the direct loss of fish from the screening and salvage process is in the range of 65%–83.5% for fish from the point they enter the Clifton Court Forebay or encounter the trash racks at the CVP facilities. In addition, mark-recapture experiments indicate that most fish are probably subject to predation prior to reaching the fish salvage facilities (e.g., in the Clifton Court Forebay) (Gingras 1997; Clark et al. 2009; Castillo et al. 2012). Fish entrainment and salvage are of particular concern during Dry Water Years when the distributions of young striped bass, delta smelt, longfin smelt, and other migratory fish species may shift closer to the CVP facilities (Stevens et al. 1985; Sommer et al. 1997).

Salvage estimates reflect the number of fish entrained by CVP and SWP exports, but these numbers alone do not account for other sources of mortality related to the export facilities. These numbers do not include losses that occur in the waterways leading to the diversion facilities, which may in some cases reduce the number of salvageable fish (Gingras 1997; Clark et al. 2009; Castillo et al. 2012). Pre-screen losses are reported to account for most adult and juvenile delta smelt mortality at the SWP export facility (Castillo et al. 2012). In addition, larval fish are not salvaged because they cannot be diverted from the export facilities by existing fish screens. The number of fish salvaged also does not include losses of fish that pass through the louvers intended to guide fish into the fish collection facilities or the losses during collection, handling, transport, and release back into the Delta.

The life stage of the fish at which entrainment occurs may be important for population dynamics (Independent Review Panel 2011). For example, winter entrainment of delta smelt, longfin smelt, and threadfin shad (*Dorosoma petenense*) may correspond to migration and spawning of adult fish, and spring and summer exports may overlap with development of larvae and juveniles. The loss of pre-spawning adults and all their potential progeny may have greater consequences than entrainment of the same number of larvae or juvenile fish because younger life stages would have some level of natural mortality before reaching adulthood.

While swimming through south Delta channels, fish can be subjected to stress from poor water quality (seasonally high temperatures, low DO, high water transparency, and *Microcystis* blooms, etc.) and low water velocities, which create lacustrine-like conditions. Any of these factors can cause elevated mortality rates by weakening or disorienting the fish and increasing their vulnerability to predators (Vogel 2011).

Considerable debate remains regarding the relationship between the export to inflow ratio on the survival of fall-run Chinook salmon and steelhead. The SST (2017) evaluated data from multiple studies and found a positive relationship between April and May ratios of San Joaquin River inflow to exports (I:E) and through-Delta survival of San Joaquin River fall-run Chinook salmon when the Head of Old River Barrier is in place. The SST also found that fall-run Chinook salmon survival in the San Joaquin River from Mossdale to the Turner Cut junction tends to increase for higher I:E values but data for the tidal portion of the Delta are mixed, with Chinook salmon survival being highest for an I:E ratio of approximately 2, and lowest for I:E ratios of approximately 1 or greater than 4. They found no evidence linking survival through the facilities to I:E ratios (Salmonid Scoping Team 2017).

For steelhead, the SST (2017) found survival in the south Delta tended to increase for higher levels the ratio of San Joaquin inflow to exports (I:E ratio), but observations are limited to 2 years of acoustic telemetry data available (2011 and 2012). Survival increased from the Turner Cut junction to Chipps Island, and overall from Mossdale to Chipps Island, as the April to May I:E increased. However, the pattern was weaker than the survival pattern observed for inflow based on SST scatterplots. Survival estimates from Mossdale to the Turner Cut junction were similar regardless of I:E based on SST scatterplots. Survival from the CVP trash rack through the facility to Chipps Island, and from the Clifton Court Forebay radial gates to Chipps Island, increased with I:E for fish released during April and May (Salmonid Scoping Team 2017). They further conclude that the high correlation between inflow and exports limits the ability to evaluate survival over a range of I:E ratios. Although not directly comparable, this contrasts with the results of Zeug and Cavallo (2013), who also found little evidence that large-scale water exports or inflows influenced coded-wire tag recovery rates in the ocean from 1993 to 2003.

Buchanan and Skalski (2020) found that I:E ratio was positively correlated with juvenile Chinook survival in the south Delta but less well supported as a predictor of survival than various other flow and environmental measures. Flow-survival relationships exist for steelhead from the San Joaquin River Basin emigrating through the Delta (Buchanan et al. 2021).

Delaney et al. (2014) reported results of a mark-recapture experiment examining the survival and movement patterns of acoustically tagged juvenile steelhead emigrating through the central Delta and south Delta. Their results indicated that most tagged steelhead remained in the mainstem San Joaquin River (77.6%). However, approximately one quarter (22.4%) of tagged steelhead entered Turner Cut. Route-specific survival probability for tagged steelhead using the Turner Cut route was 27.0%. The survival probability for tagged steelhead using the mainstem route was 56.7% (Delaney et al. 2014). Travel times for tagged steelhead also differed between these two routes, with steelhead using the mainstem route reaching Chipps Island significantly sooner than those that used the Turner Cut route. Travel time was not significantly affected by the limited Old and Middle River flow treatments examined in their study. While not significant, there was some evidence that fish movement toward each export facility could be influenced by the relative volume of water entering the export facility (Delaney et al. 2014).

Cunningham et al. (2015) found a negative influence of the ratio of exports to total Delta inflow (E:I ratio) on the survival of fall-run Chinook salmon populations and a negative influence of increased total Delta exports on the survival of spring-run Chinook salmon populations. An increase in total exports of 1 standard deviation from the 1967 to 2010 average was predicted to result in a 68.1% reduction in the survival of Deer Creek, Mill Creek, and Butte Creek spring-run Chinook salmon. Similarly, an increase in the E:I ratio 1 standard deviation was expected to reduce survival of the four fall-run Chinook salmon populations by 57.8% (Cunningham et al. 2015). Note that the levels of Delta exports were relatively high during this historical period relative to current management under the NMFS (2019a) and USFWS (2019) SWP/CVP Biological Opinions and the CDFW (2020) SWP ITP: the annual mean February–April Delta exports during 1967–2010 was approximately 6,000 cubic feet per second (cfs) with a standard deviation of approximately 2,100 cfs (compared to approximately 3,800 cfs in 2020), the mean annual E:I during 1967–2010 was 0.21 with a standard deviation of 0.14 (compared to approximately 0.20 in 2020). Although a mechanistic explanation for the reduction in survival remains elusive, “direct entrainment mortality seems an unlikely mechanism given performance of the fish rescue programs at the export pumps. Changes to water routing may provide a more reasonable explanation for the estimated survival influence of Delta water exports” (Cunningham et al. 2015).

Research conducted during 2010 and 2011 showed that upriver movements of adult delta smelt are achieved through a form of selective tidal stream transport by using lateral movement to shallow edges of channels on ebb tides to maintain their position, with movement into the river channel on the flood tide (Bennett and Burau 2015). Turbidity gradients could be involved in the lateral positioning of delta smelt within the channels, but large-scale turbidity pulses through the system may not be necessary to trigger upriver migrations of delta smelt if they are already occupying sufficiently turbid water (Independent Review Panel 2011; Gross et al. 2021; Korman et al. 2021).

There are more than 2,200 diversions in the Delta (Herren and Kawasaki 2001). These irrigation diversion pipes are shore-based, typically small (30 to 60 centimeters pipe diameter), and operated via pumps or gravity flow, and most lack fish screens. These diversions increase total fish entrainment and losses and alter local fish movement patterns (Kimmerer and Nobriga 2008). Delta smelt have been found in samples of Delta irrigation diversions (Nobriga et al.

2004), as well as larger wetland management diversions downstream (Pickard et al. 1982). However, Nobriga et al. (2004) found that the low and inconsistent entrainment of delta smelt measured in their study of typical irrigation diversions reflected offshore habitat use by delta smelt and relatively small hydrodynamic influence of the diversions. Concerns were expressed by Kneib (2019) about potential entrainment effects given the relatively limited study of entrainment by Nobriga et al. (2004), such as the need to consider cumulative losses at all diversions (Kneib 2019:13). Nobriga and Herbold (2009:25–26) expanded on the discussion by Nobriga et al. (2004) to conclude that irrigations at small diversions are not a major stressor to delta smelt because (1) as noted above, most diversions have very small hydrodynamic footprints and delta smelt tend to occupy offshore habitat away from the diversions, (2) many of the diversions are not diverting water every day, (3) many diversions are located in the south Delta, where habitat conditions are unsuitable for delta smelt during summer/fall, and (4) agricultural water demand has not increased since the 1930s. Citing some of these reasons, Baxter et al. (2010:41) considered small within-Delta irrigation diversions to be unlikely to have had an effect on POD species, including delta smelt and longfin smelt. The overlap of juvenile salmonid occurrence in the Delta with irrigation diversions is limited and not thought to be of population-level consequence (Vogel 2011:94).

Nonnative Invasive Species

Introduced species may influence the Delta ecosystem by increasing competition and predation on native species, reducing habitat quality (as result of invasive aquatic macrophyte growth), and reducing food supplies by altering the aquatic food web. Many of these introductions become invasive, however CDFW does not consider all introduced species invasive. CDFW defines invasive species as "species that establish and reproduce rapidly outside of their native range and may threaten the diversity or abundance of native species through competition for resources, predation, parasitism, hybridization with native populations, introduction of pathogens, or physical or chemical alteration of the invaded habitat (California Department of Fish and Game 2008)," for example the overbite clam discussed above. Some introduced species have minimal ability to spread or increase in abundance. Others may be invasive but also have commercial or recreational value (e.g., striped bass, largemouth bass).

Many fishes not historically present in California waterways (i.e., nonnative species) have been introduced into the Delta, some of which are invasive. These nonnative fishes have been introduced for sport fishing (game fish such as striped bass, largemouth bass, smallmouth bass [*Micropterus dolomieu*], bluegill [*Lepomis macrochirus*], and other sunfish), as forage for game fish (threadfin shad, golden shiner [*Notemigonus crysoleucas*], and fathead minnow [*Pimephales promelas*]), for vector control (inland silverside [*Menidia beryllina*], western mosquitofish [*Gambusia affinis*]), for human food use (common carp [*Cyprinus carpio*], brown bullhead [*Ameiurus nebulosus*], and white catfish [*Ameiurus catus*]), and from accidental releases (yellowfin goby [*Acanthogobius flavimanus*], Shimofuri goby [*Tridentiger bifasciatus*], and Shokihaze goby [*Tridentiger barbatus*]) (Moyle 2002). Although CDFW does not consider many of these invasive species, they can affect native populations through competition for resources and predation.

Changes in hydrology and water quality, stabilization of natural environmental variability, and other modification to Delta generally favor nonnative, invasive species (Mount et al. 2012;

Moyle et al. 2012) to the detriment of native species. These nonnative species prey on and/or compete with native species for resources to the detriment of native species and are an important factor in the decline of native species populations throughout the region (Matern et al. 2002; Brown and Michniuk 2007; Sommer et al. 2007; Mount et al. 2012).

As described in the discussion of nutrients and foodweb support above, the introductions of two clams from Asia have led to major alterations in the foodweb in the Delta. The *Potamocorbula* and *Corbicula* clams significantly reduce the phytoplankton and zooplankton concentrations in the water column, reducing food availability for native fishes, such as delta smelt and young Chinook salmon (Feyrer et al. 2007; Kimmerer 2002b).

Predation

Predation is an important factor that influences the behavior, distribution, and abundance of prey species in aquatic communities to varying degrees. Predation can have differing effects on a population of fish, depending on the size or age selectivity, mode of capture, mortality rates, and other factors. Predation is a part of every food web, and native Delta fishes were part of the historical Delta food web. The introduction of nonnative predators has likely increased mortality rates for juvenile salmonids in the Delta (Vogel 2011). NMFS (2014) rated predation of juvenile winter-run Chinook salmon and spring-run Chinook salmon during rearing and outmigration as a stressor of “Very High” importance. Predation occurs by fish, birds, and mammals.

A panel of experts was convened to review data on predation in the Delta and draw preliminary conclusions on the effects of predation on salmonids. The panel acknowledged that the system supports large populations of fish predators that consume juvenile salmonids (Grossman et al. 2013). However, the panel concluded that because of extensive flow modification, altered habitat conditions, native and nonnative fish and avian predators, temperature and DO limitations, and the overall reduction in salmon population size, it was unclear what proportion of juvenile salmonid mortality could be attributed to predation. The panel further indicated that predation, while the proximate cause of mortality, may be influenced by a combination of other stressors that make fish more vulnerable to predation.

Striped bass, white catfish, largemouth bass and other centrarchids, and silversides are among the introduced, nonnative species that are notable predators of smaller-bodied fish species and juveniles of larger species in the Delta. Along with largemouth bass, striped bass are believed to be major predators on larger-bodied fish in the Delta. In open-water habitats, striped bass are most likely the primary predator of juvenile and adult delta smelt (California Department of Water Resources et al. 2013) and can be an important open-water predator on juvenile salmonids (Smith et al. 2017:252–266). A recent study using genetic techniques to identify prey species in the guts of striped bass reported that predation on delta smelt and Chinook salmon was substantially higher than previous predation studies in the Delta (Brandl et al. 2021). Native Sacramento pikeminnow (*Ptychocheilus grandis*) also prey on juvenile salmonids and other fishes. While limited sampling of smaller pikeminnows did not find evidence of salmonids in the foregut of Sacramento pikeminnow (Nobriga and Feyrer 2007), this does not mean that Sacramento pikeminnow do not prey on salmonids in the Delta since pikeminnow predation of juvenile salmon has been documented at RBDD (the current location of the RBPP) (Tucker et al. 1998).

Largemouth bass abundance has increased in the Delta over the past few decades (Brown and Michniuk 2007). Although largemouth bass are not pelagic, their presence at the boundary between the littoral and pelagic zones makes it probable that they opportunistically consume pelagic fishes. The increase in salvage of largemouth bass occurred during the period when Brazilian waterweed (*Egeria densa*) was expanding its range in the Delta (Brown and Michniuk 2007). The beds of Brazilian waterweed provide good habitat for largemouth bass and other species of centrarchids. Largemouth bass have a much more limited distribution in the estuary than striped bass, but a higher per capita impact on small fishes (Nobriga and Feyrer 2007). Increases in largemouth bass may have had a particularly important effect on threadfin shad and striped bass, whose earlier life stages occur in littoral habitat (Grimaldo et al. 2004; Nobriga and Feyrer 2007).

Invasive Mississippi silverside (*Menidia audens*) is another potentially important predator of larval and pelagic fishes in the Delta. This introduced species was not believed to be an important predator on delta smelt, but studies using DNA techniques detected the presence of delta smelt in the guts of 12.5% of Mississippi silversides sampled in midchannel trawls across a variety of habitats in the north Delta and identified turbidity as a significant predictor of predation (Baerwald et al. 2012; Schreier et al. 2016). This finding may suggest that predation impacts could be significant, given the increasing numbers of Mississippi silversides in the Delta (Mahardja et al. 2016) and decreasing trends in turbidity (Feyrer et al. 2007).

Predation of fish in the Delta is known to occur in specific areas, for example at channel junctions and areas that constrict flow or confuse migrating fish and provide cover for predatory fish (Vogel 2011). Sabal (2014) found similar results at Woodbridge Dam on the Mokelumne River where the dam was associated with increased striped bass per capita salmon consumption, which decreased outmigrant juvenile salmon survival by 10%–29%. CDFW identified subadult striped bass as the major predatory fish in the Clifton Court Forebay (California Department of Fish and Game 1992). In 1993, for example, striped bass made up 96% of the predators removed (Vogel 2011). Cavallo et al. (2012) studied tagged salmon smolts to test the effects of predator removal on outmigrating juvenile Chinook salmon in the south Delta. Their results suggested that predator abundance and migration rates strongly influenced survival of salmon smolts. Exposure time to predators has been found to be important for influencing survival of outmigrating salmon in other studies in the Delta (Perry et al. 2012).

DWR examined the species distribution and abundance of salvaged fish at the SWP pumping facilities to determine whether alternative release scenarios between salvaged delta smelt and predatory species would increase smelt survival. An initial evaluation of historical records on species distribution of salvaged fish lead to the conclusion that adjusting salvage operations to stop returning predatory fish to the Delta would have little impact on delta smelt survival (California Natural Resources Agency 2017:3).

Aquatic Macrophytes

Aquatic macrophytes are an important component of the biotic community of Delta wetlands and can provide habitat for aquatic species, serve as food, produce detritus, and influence water quality through nutrient cycling and DO fluctuations. Whipple et al. (2012) described likely historical conditions in the Delta, which have been modified extensively, with major impacts on the aquatic macrophyte community composition and distribution. The primary change has been a shift from a high percentage of emergent aquatic macrophyte wetlands to open water and hardened channels.

The introduction of two nonnative invasive aquatic plants, water hyacinth (*Eichhornia crassipes*) and Brazilian waterweed, has reduced habitat quantity and value for many native fishes. Water hyacinth forms floating mats that greatly reduce light penetration into the water column, which can significantly reduce primary productivity and available food for fish in the underlying water column. Brazilian waterweed grows along the margins of channels in dense stands that prohibit access by native juvenile fish to shallow water habitat. In addition, the thick cover of these two invasive plants provides excellent habitat for nonnative ambush predators such as bass, which prey on native fish species. Studies indicate low abundance of native fish, such as delta smelt, Chinook salmon, and Sacramento splittail, in areas of the Delta where submerged aquatic vegetation infestations are thick (Grimaldo et al. 2004, 2012; Nobriga et al. 2005).

Invasive aquatic macrophytes are still equilibrating within the Delta and resulting habitat changes are ongoing, with negative impacts on habitats and foodwebs of native fish species (Toft et al. 2003; Grimaldo et al. 2009). Concerns about invasive aquatic macrophytes are centered on their ability to form large, dense growth that can clog waterways, block fish passage, increase water clarity, provide cover for predatory fish, and cause high biological oxygen demand. DWR, in coordination with the Department of Parks and Recreation and the Division of Boating and Waterways, is actively engaged in a program of aquatic weed control. Building on the state's existing herbicide treatment program, DWR targeted 200 acres of delta smelt habitat at Decker Island in the western Delta and the Cache Slough Complex in the north Delta. Ongoing field studies are investigating the effect of herbicide treatment on delta smelt habitat (California Natural Resources Agency 2017).

Suisun Bay/Marsh

Aquatic Habitat

Suisun Marsh is a brackish-water marsh bordering the northern edge of Suisun Bay. Most of its marsh area consists of diked wetlands managed for waterfowl, and the rest of the acreage consists of tidally influenced sloughs and emergent tidal wetlands (Suisun Ecological Workgroup 2001). The central latitudinal location of Suisun Marsh within the San Francisco Estuary makes it an important rearing area for euryhaline freshwater, estuarine, and marine fishes. Many fish species that migrate or use Delta habitats are also found in the waters of Suisun Bay. Tides reach Suisun Bay and Suisun Marsh through the Carquinez Strait, and most freshwater flows enter at the southeast border of Suisun Marsh at the confluence of the Sacramento and San Joaquin Rivers. The mixing of freshwater outflows from the Central Valley with saline tidal water in Suisun Bay and Suisun Marsh results in brackish water with strong salinity gradients, complex patterns of flow interactions, and generally the highest biomass productivity in the entire estuary (Siegel et al. 2010).

Flow, turbidity, and salinity are important factors influencing the location and abundance of zooplankton and small prey organisms used by Delta species (Kimmerer et al. 1998). The location where net current flowing inland along the bottom reverses direction and sinking particles are trapped in suspension is associated with the higher turbidity known as the estuarine turbidity maximum. Burau et al. (2000) reports that the estuarine turbidity maximum occurs near the Benicia Bridge and in Suisun Bay near Garnet Point on Ryer Island. Zooplanktonic organisms maintain position in this region of historically high productivity in the estuary through vertical movements (Kimmerer et al. 1998).

Salinity in the Suisun Marsh and Bay system is a major water quality characteristic that strongly influences physical and ecological processes. Many fish species native to Suisun Marsh require low salinities during the spawning and rearing periods (Suisun Ecological Workgroup 2001; Kimmerer 2004; Feyrer et al. 2007, 2010; Nobriga et al. 2008). The Suisun Marsh and Bay usually contain both the maximum estuarine salinity gradient and the low- salinity zone. The overall estuarine salinity gradient trends from west (higher) to east (lower) in Suisun Bay and Suisun Marsh. The location of the low- salinity zone gradient is influenced by outflow. Suisun Marsh also exhibits a persistent north-south salinity gradient. Despite low and seasonal flows, the surrounding watersheds have a significant water freshening effect because of the long residence times of freshwater inflows to the marsh, including discharges from the upper sloughs and wastewater effluent. The larger of these surrounding watersheds include Suisun, Green Valley, Ledgewood, Laurel, McCoy, and Union Creeks (Siegel et al. 2010:1-18).

The Suisun Bay and Suisun Marsh system contains a wide variety of habitats such as marsh plains, tidal creeks, sloughs, channels, cuts, mudflats, and bays. These features and the complex hydrodynamics and water quality of the system have historically fostered significant biodiversity within Suisun tidal aquatic habitats, but these habitats, like the Delta, have also been significantly altered and degraded by human activities over the decades.

Categories of tidal aquatic waters include bays, major sloughs, minor sloughs, and the intertidal mudflats in those areas (Engle et al. 2010). These tidal waters total approximately 26,000 acres, with the various embayments totaling about 22,350 acres. Tidal slough habitat is composed of major and minor sloughs. Major sloughs of Suisun Marsh have a combined acreage of about 2,200 acres consisting of both shallow and deep channels. Minor sloughs are made up of shallow channel habitat and have a combined acreage of about 1,100 acres. Habitats in Suisun Marsh bays and sloughs support a diverse assemblage of aquatic species that typically use open-water tidal areas for breeding, foraging, rearing, or migrating.

Fish Entrainment

DWR and Reclamation have constructed several facilities to provide lower-salinity water to managed wetlands in Suisun Marsh, including the Roaring River Distribution System (RRDS), Morrow Island Distribution System, and Goodyear Slough Outfall. Other facilities constructed under the Suisun Marsh Preservation Agreement that could entrain fish include the Lower Joice Island and Cygnus Drain diversions.

The intake to the RRDS is screened to prevent entrainment of fish larger than approximately 1 inch (approximately 25 millimeters [mm]). DWR monitored fish entrainment from September 2004 through June 2006 at the Morrow Island Distribution System to evaluate entrainment losses at the facility. Monitoring took place over several months under various operational configurations and focused on delta smelt and salmonids. More than 20 species were identified during the sampling, but only two fish the size of fall-run Chinook salmon were observed at the

South Intake in 2006, and no delta smelt from entrained water were observed (Enos et al. 2007). The survey only evaluated one period when the probability of occurrence of target fish species was highest (i.e., summer/fall). Therefore, there is some uncertainty regarding the results, which suggest a low probability of entrainment of Chinook salmon and delta smelt into the distribution system. The Goodyear Slough Outfall system is open for free fish movement except near the outfall when flap gates are closed during flood tides (Bureau of Reclamation 2008). Conical fish screens have been installed on the Lower Joice Island diversion on Montezuma Slough.

11.2.4. Sacramento River Flood Bypasses (Butte Basin, Sutter Bypass, and Yolo Bypass)

There are three major floodplain bypasses in the Sacramento River system – Butte Basin, Sutter Bypass, and Yolo Bypass – with a total of 10 overflow structures along the lower Sacramento River (six weirs, three flood relief structures, and an emergency overflow roadway) that provide access to broad, inundated floodplain habitat during wet years.

All six weirs (Moulton, Colusa, Tisdale, Fremont, Sacramento, and Cache Creek) consist of the following: (1) a fixed-level, concrete overflow section; followed by (2) a concrete, energy dissipating stilling basin; with (3) a rock and/or concrete erosion blanket across the channel beyond the stilling basin; and (4) a pair of training levees that define the weir-flow escape channel.

All overflow structures except the Sacramento Weir pass floodwaters by gravity once the river reaches the overflow water surface elevation (WSE). The Sacramento Weir has gates on top of the overflow section that hold back floodwaters until opened manually by the DWR' Division of Flood Management (California Department of Water Resources 2010).

Four other relief structures are concentrated along 18 river miles (RMs) between Big Chico Creek (RM 194) and the upstream end of the left (east) bank levee of the Sacramento River Flood Control Project (near RM 176). These structures function like weirs but are not called weirs because they do not have all four structural characteristics previously described. All of these relief structures convey water into the Butte Basin (a natural trough east of the river) upstream of the levee system designed to guide the flood waters. Three of the structures are designated as flood relief structures (M&T, 3B's, and Goose Lake). If these three fail as designed, a raised 6,000-foot roadway near the south end of Parrott Ranch allows excess floodwaters to escape the Sacramento River to the Butte Basin before being confined by the downstream project levees (California Department of Water Resources 2010).

Unlike other Sacramento River and Delta habitats, floodplains and floodplain bypasses are seasonally dewatered (as high flows recede) during late spring through autumn. This prevents introduced fish species from establishing year-round dominance except in perennial water sources (Sommer et al. 2003). Moreover, many of the native fish are adapted to spawn and rear in winter and early spring (Moyle 2002) during the winter flood pulse. Introduced fish typically spawn during late spring through summer when the majority of the floodplain is not available to them.

11.2.4.1. Butte Basin

The Butte Basin lies east of the Sacramento River and extends from Big Chico Creek near Chico Landing to the north, to the Butte Slough outfall gates near Meridian. Flood flows are diverted out of the Sacramento River into the Butte Basin and Sutter Bypass via two weirs, Moulton and Colusa, and several designated overflow areas (i.e., low points along the east side of the river) that allow high flood flows to exit the Sacramento River channel; M&T, 3B's, and Goose Lake).

Moulton Weir was completed in 1932. It is located along the easterly side (left bank looking downstream) of the Sacramento River approximately eight miles north of the town of Colusa and about 100 miles north of Sacramento. Its primary function is to release overflow waters of the Sacramento River into the Butte Basin at such times when floods exceed the safe carrying capacity of the main channel of the Sacramento River downstream from the weir. The fixed crest reinforced concrete weir is 500 feet long with concrete abutments at each end. The outlet channel is flanked by training levees and is approximately 3,000 feet long. The crest elevation is 76.75 feet and the design capacity of the weir is 25,000 cfs. The Moulton Weir is typically the last of the non-gated weirs to overtop, and spills for the shortest duration (California Department of Water Resources 2010).

Colusa Weir was completed in 1933. It is located along the left bank of the Sacramento River one mile north of the town of Colusa. Its primary function is to release overflow waters of the Sacramento River into the Butte Basin. The fixed crest reinforced concrete weir is 1,650 feet long and is flanked by training levees that connect the river to the basin. The crest elevation is 61.80 feet and the design capacity of the weir is 70,000 cfs. Normally, the Colusa Weir does not overtop until the Tisdale Weir is also spilling, except for flood events that are characterized by rapid rise in Sacramento River stage (California Department of Water Resources 2010).

11.2.4.2. Sutter Bypass

The Sutter Bypass is a floodwater bypass conveying Sacramento River flood flows from the Butte Basin and the Tisdale Weir. The bypass area is an expansive land area in Sutter County used mainly for agriculture.

Tisdale Weir was completed in 1932. It is located along the left bank of the Sacramento River about ten miles southeast of the town of Meridian and about 56 miles north of Sacramento. Its primary purpose is to release overflow waters of the Sacramento River into the Sutter Bypass via the Tisdale Bypass. The fixed crest reinforced concrete weir is 1,150 long. The four-mile leveed bypass channel (Tisdale Bypass) connects the river to the Sutter Bypass. The crest elevation is 45.45 feet and the design capacity of the weir is 38,000 cfs. Typically, the Tisdale Weir is the first of the five weirs in the Sacramento River Flood Control System to overtop and continues to spill for the longest duration (California Department of Water Resources 2010).

The Sutter Bypass, in turn, conveys flows to the lower Sacramento River region at the Fremont Weir near the confluence with the Feather River and into the Sacramento River and the Yolo Bypass.

11.2.4.3. Yolo Bypass

Fremont Weir was completed in 1924. It is the first overflow structure on the river's right bank and its two-mile overall length marks the beginning of the Yolo Bypass. It is located about 15 miles northwest of Sacramento and eight miles northeast of Woodland. South of this latitude the Yolo Bypass conveys 80% of the system's floodwaters through Yolo and Solano Counties until it connects to the Sacramento River a few miles upstream of Rio Vista. The weir's primary purpose is to release overflow waters of the Sacramento River, Sutter Bypass, and the Feather River into the Yolo Bypass. The crest elevation is 33.50 feet and the design capacity of the weir is 343,000 cfs.

The Sacramento Weir was completed in 1916. It is the only weir that is manually operated – all others overflow by gravity on their own. It is located along the right bank of the Sacramento River approximately 4 miles upstream of the Tower Bridge, and about 2 miles upstream from the mouth of the American River. Its primary purpose is to protect the city of Sacramento from excessive flood stages in the Sacramento River channel downstream of the American River. The weir limits flood stages (WSE) in the Sacramento River to design levels through the Sacramento/West Sacramento area. The design capacity of the weir is 112,000 cfs.

Aquatic Habitat

Aquatic habitats in the Yolo Bypass include stream and slough channels for fish migration and when flooded, seasonal spawning habitat and productive rearing habitat (Sommer et al. 2001; CALFED Bay-Delta Program 2000a, 2000b). During years when the Yolo Bypass is flooded, it serves as an important migratory route for juvenile Chinook salmon and other native migratory and anadromous fishes moving downstream. During these times, it provides juvenile anadromous salmonids an alternative migration corridor to the lower Sacramento River (Sommer et al. 2003) and, sometimes, better rearing conditions than the adjacent Sacramento River channel (Sommer et al. 2001, 2005). When the floodplain is activated, juvenile salmon can rear for weeks to months in the Yolo Bypass floodplain before migrating to the estuary (Sommer et al. 2001). Research on the Yolo Bypass has found that juvenile salmon grow substantially faster in the Yolo Bypass floodplain than in the adjacent Sacramento River, primarily because of the greater availability of invertebrate prey in the floodplain (Sommer et al. 2001, 2005). Increased frequency and duration of connectivity between the Sacramento River and the Yolo Bypass may increase off-channel rearing opportunities that expand the life history diversity portfolio for Central Valley Chinook salmon (Takata et al. 2017). When not flooded, the lower Yolo Bypass provides tidal habitat for young fish that enter from the lower Sacramento River via Cache Slough Complex—a network of tidal channels and flooded islands that includes Cache Slough, Lindsey Slough, Liberty Island, the Sacramento Deepwater Ship Channel, and the Yolo Bypass (Mahardja et al. 2019).

Sommer et al. (1997) demonstrated that the Yolo Bypass is one of the single most important habitats for Sacramento splittail. Because the Yolo Bypass is dry during summer and fall, nonnative species (e.g., predatory fishes) generally are not present year-round except in perennial water sources (Sommer et al. 2003). In addition to providing important fish habitat, winter and spring inundation of the Yolo Bypass supplies phytoplankton and detritus that may benefit aquatic organisms downstream in the brackish portion of the San Francisco Estuary (Sommer et al. 2004; Lehman et al. 2008a).

The benefit of seasonal inundation of the Yolo Bypass has been studied by DWR as part of the Delta Smelt Resiliency Strategy, which was developed by DWR and other state and federal resource agencies to boost both immediate- and near-term reproduction, growth rates, and survival of delta smelt (California Natural Resources Agency 2016; Mahardja et al. 2019). The Yolo Bypass has been identified as a significant source of phytoplankton and zooplankton biomass to the Delta in the winter and spring during floodplain inundation. However, little has been known about its contribution to the foodweb during the drier summer and fall months.

One action taken by DWR under the strategy is the implementation of foodweb enhancement projects in the Yolo Bypass. Under this action, DWR worked with farmers as well as irrigation and reclamation districts to direct water through the Yolo Bypass in the form of flow pulses during summer and fall (Frantzich et al. 2018, 2021). The first examination of off-season flow pulses occurred in 2016 when a flow pulse of 12,700 AF was released over 2 weeks in the summer. The second examination occurred during 2018 when a 19,821 AF flow occurred over 4 weeks in the fall. These flow pulses were followed in turn by a significant increase in phytoplankton biomass in the Cache Creek Complex and further downstream in the lower Sacramento River (California Natural Resources Agency 2017; California Department of Water Resources 2019a, 2019b; Frantzich et al. 2021). The increase in phytoplankton biomass was also found to enhance zooplankton growth and production, thereby potentially increasing food supplies for delta smelt and other Delta fish species. During the second year of implementing flow pulses, a managed flow pulse was generated in the fall of 2018. The 2018 Fall North Delta Flow Action generated a flow pulse of 19,821 AF over 4 weeks, which while not coinciding with an increased pulse of phytoplankton moving through the Yolo Bypass, did result in an export of higher densities of zooplankton into downstream habitats of lower Cache Slough and the Sacramento River at Rio Vista (California Department of Water Resources 2019c). The increases in zooplankton in the study were inferred to have potentially provided positive effects on delta smelt, but such effects were not specifically observed as increases in abundance in annual delta smelt monitoring surveys. See further discussion in Impact FISH-8, *Operations Effects on Delta Smelt*.

Studies continued in 2019 on the issue of foodweb enhancement in the Yolo Bypass. Working with the GCID and other partners, DWR tested the benefit of passing water through the Yolo Bypass to enhance delta smelt habitat in the north Delta region (Davis et al. 2019). The action was expected to generate a seasonal positive flow pulse through the Yolo Bypass Toe Drain, which was expected to benefit the foodweb in downstream areas for fishery resources. DWR altered the operation of the Knights Landing Outfall Gates and Wallace Weir to direct agricultural return flows from the CBD through Ridge Cut Slough and Wallace Weir into the Yolo Bypass between late August and late September. Analysis of the results is ongoing (Davis et al. 2019). Water for these studies may be limited in years of drought or limited to wetter than average water type years. Additional discussion of the 2019 studies is provided in Impact FISH-8 for delta smelt.

Studies over the past decade have demonstrated the viability of directing agricultural drainage flows into fields within the Yolo Bypass in order to provide rearing habitat for juvenile Chinook salmon (Sommer et al. 2020), which can provide floodplain-like habitat in years when floodplains are not otherwise inundated (Katz et al. 2017).

Fish Passage

The Fremont Weir is a major impediment to adult Chinook salmon, steelhead, and sturgeon passage and a source of migratory delay and mortality (National Marine Fisheries Service 2009; Sommer et al. 2014). Although fish with strong jumping capabilities (such as salmonids) may be able to pass the weir at higher flows. In 2018, DWR implemented the Fremont Weir Adult Fish Passage Modification Project. The project replaced an old, undersized, inefficient fish ladder in the center of the weir with a wider and deeper gate structure. Monitoring and evaluation of the structure's effectiveness is ongoing.

Adult winter-run, spring-run, and fall-run Chinook salmon and white sturgeon have been documented to migrate into the Yolo Bypass via the Toe Drain and Tule Canal when there is no flow into the floodplain over the Fremont Weir (National Marine Fisheries Service 2009). Fyke trap monitoring by DWR has shown that adult salmon and steelhead migrate up the Toe Drain in autumn and winter regardless of whether the Fremont Weir spills (Harrell and Sommer 2003; Sommer et al. 2014). The Toe Drain does not extend to the Fremont Weir because the channel is fully or partially blocked by roads or other higher ground at several locations and fish are often unable to reach upstream spawning habitat in the Sacramento River and its tributaries (Harrell and Sommer 2003; Sommer et al. 2014). Other structures in the Yolo Bypass, such as the Lisbon Weir, and irrigation dams in the northern end of the Tule Canal may also impede upstream passage of adult anadromous fish (National Marine Fisheries Service 2009). Modifications to some of these structures were made as part of the Fremont Weir Adult Fish Passage Modification Project, and two agricultural road crossings were altered to improve fish passage.

In addition, sturgeon and salmonids attracted by high flows into the Bypass become concentrated behind the Fremont Weir, where they are subject to heavy illegal fishing pressure. Passage blockage of green sturgeon at Fremont Weir could have population-level consequences (Thomas et al. 2013). Adult salmon may access the CBD via this migratory route, which is a migratory dead end with not return to the Sacramento River. The state and federal governments have been working with local landowners to eliminate access to the CBD. For example, the Wallace Weir Fish Rescue Project was completed in 2018. This project updated a flow control structure in the Knights Landing Ridge Cut by adding a fish collection and rescue facility where migratory strays may be captured and returned to the Sacrament River to resume their migration to their spawning grounds.

DWR and Reclamation have been working on the Yolo Bypass Habitat Restoration program, which is developing and implementing six restoration actions in the Yolo Bypass, including removal of several fish passage barriers. Some of these actions are complete, or nearly complete, including the Fremont Weir Adult Fish Passage Modification Project.

Stranding of juvenile salmonids and sturgeon has been reported in the Yolo Bypass in scoured areas behind the weir and in other areas as floodwaters recede (National Marine Fisheries Service 2009; Sommer et al. 2005). However, Sommer et al. (2005) found most juvenile salmon migrated off the floodplain as it drained.

The Yolo Bypass Salmonid Habitat Restoration and Fish Passage Project has been developed to significantly improve fish passage and increase floodplain fisheries rearing habitat in the Yolo

Bypass and the lower Sacramento River basin by constructing a notch with operable gates on Fremont Weir (Bureau of Reclamation 2019). The goal of the project is to increase the number of outmigrating juvenile winter-run Chinook salmon that enter the Yolo Bypass. Downstream outmigration is triggered by the first increased flow events. Once constructed, the gates would open each wet season as early as November 1, based on hydrologic conditions. All gates would be opened when the river stage reaches 15 feet, which is one foot above the lowest gate invert. At this stage, about 130 cfs would enter the gated notch. If the river continues to rise, the gates would stay open until the flow through the gates reached 6,000 cfs (river stage about 28 feet). At this point, the two smaller gates would be programmed to start closing to maintain flows of 6,000 cfs. Once Fremont Weir begins to overtop, the smaller gates would remain in their last position prior to the weir overtopping (generally both would be closed at this point). After the overtopping event is over, the smaller gates would open and close as needed to keep the flow through the gate as close as possible to 6,000 cfs. All gates would close when the river stage falls below 14 feet. Gate operations to increase inundation could continue through March 15 of each year, based on hydrologic conditions. The gates may remain partially open after March 15 to provide adult fish passage.

Construction of the Yolo Bypass Salmonid Habitat Restoration and Fish Passage Project is scheduled to be completed in late 2023. Nevertheless, it is considered part of the environmental setting for purposes of modeling potential Project impacts. The project includes supplemental fish passage, which is on the west side of the Fremont Weir, and this will function as a drain to empty out the water and fish that remain in the energy dissipation basin after an overtopping event.

Adult salmonids also stray into the CBD via the Cache Slough Complex (Gahan et al. 2016). One of the terminus points of the CBD connects to the Knights Landing Ridge Cut Slough which then connects to the eastern Toe Drain of the Yolo Bypass which eventually drains out to the Cache Slough Complex in the northern Delta, near Liberty Island. Tidal influence from the Delta enhances flows from the Yolo Bypass in the Cache Slough Complex, creating attraction flows that draw salmonids into the bypass and subsequently the CBD. This occurs both during flooding and non-flooding of the Yolo Bypass (Gahan et al. 2016).

11.2.5. Upstream of Delta

11.2.5.1. General Description of Rivers and Reservoirs Upstream of Delta

The areas upstream of the Delta in the Central Valley that could potentially be affected by Alternatives 1, 2, and 3 include those areas in the SWP and CVP system that may be affected by alterations in SWP and CVP operations, including the reservoirs, rivers, and other components of the SWP and CVP. These components include the following instream, reservoir, and riparian areas:

- Shasta Lake and the upper and lower Sacramento River
- Lake Oroville, Thermalito Afterbay, and the lower Feather River
- Folsom Lake, Lake Natoma, and the lower American River

The timing, duration, and magnitude of water exports can affect operations of SWP and CVP reservoirs upstream of the Delta, which can affect hydrodynamic conditions for species present in the rivers and tributaries reaches. Flows are important to the movement and migration behaviors, straying potential, habitat availability and suitability, and stranding potential of numerous aquatic species. Operational changes to various flow attributes (timing, duration, magnitude, frequency, and rate of change) can directly affect anadromous species immigration and emigration, spawning and egg incubation, and rearing, as well as resident non-migratory species habitat availability for all life stages.

11.2.5.2. Sacramento River

Description of Shasta Lake, Keswick Reservoir, and Sacramento River

Shasta Lake

Shasta Lake is formed by Shasta Dam, which is located on the Sacramento River just downstream of the confluence of the Sacramento, McCloud, and Pit rivers. Shasta Dam has no fish passage facilities, but it has a fish trapping facility that operates in conjunction with Livingston Stone National Fish Hatchery (NFH) downstream of Shasta Dam to propagate winter-run Chinook salmon.

Shasta Lake fish species include native and introduced warm-water and cold-water species. The reservoir is typically thermally stratified from April through November, during which time the upper layer (epilimnion) can reach a peak water temperature of 80°F (Bureau of Reclamation 2003). The epilimnion supports warm-water game fish, while the lower layers (metalimnion and hypolimnion) support cold-water fishes. Nonnative, warm-water fish species in Shasta Lake include smallmouth bass, largemouth bass, spotted bass (*Micropterus punctulatus*), black crappie (*Pomoxis nigromaculatus*), bluegill, green sunfish (*Lepomis cyanellus*), channel catfish (*Ictalurus punctatus*), white catfish, and brown bullhead (California Department of Water Resources et al. 2013). Cold-water species include rainbow trout (*Oncorhynchus mykiss*), brown trout (*Salmo trutta*), landlocked white sturgeon, landlocked coho salmon (*Oncorhynchus kisutch*) (Bureau of Reclamation et al. 2003), and landlocked Chinook salmon (Bureau of Reclamation 2013). Other fish species in Shasta Lake include golden shiner, threadfin shad, and common carp. Native fish species include hardhead (*Mylopharodon conocephalus*), Sacramento sucker (*Catostomus occidentalis*), and Sacramento pikeminnow (California Department of Water Resources et al. 2013; Bureau of Reclamation 2013).

Water quality in Shasta Lake is generally considered good, largely because of the continual inflow of cool, high-quality water from the major tributaries to the reservoir. The primary water quality concern in the reservoir is turbidity, typically associated with heavy rainfall events that move soils and runoff from abandoned mines in the area into the reservoir.

Warm-water fish habitat in Shasta Lake is influenced primarily by fluctuations in the lake level and the availability of shoreline cover (Bureau of Reclamation 2003). The WSE in Shasta Lake can fluctuate approximately 55 feet annually as a result of operation of Shasta Lake and upper Sacramento River diversions (Bureau of Reclamation 2003). Fluctuations in the reservoir WSE can disturb shallow, nearshore habitats, including spawning and rearing habitat for warm-water fish species. The shoreline of Shasta Lake is generally steep. This terrain, with the large fluctuations in WSE, limits shallow, warm-water fish habitat, and the establishment of vegetation or other shoreline cover (Bureau of Reclamation 2003).

Keswick Reservoir

Keswick Reservoir is a re-regulating reservoir for Shasta Lake. The WSE is relatively constant. Residence time for water in Keswick Reservoir is about a day, compared with a residence time of about a year for water in Shasta Lake. Consequently, water temperatures tend to be controlled by releases from Shasta Dam and average less than 55°F. Despite the cool temperatures, the reservoir supports warm-water and cold-water fishes, including largemouth bass, crappie, catfish, and rainbow trout (Bureau of Reclamation 2003).

Sacramento River

Aquatic resources in the Sacramento River rely on the habitat in and along the river and along the tributaries that connect to the river. Habitat along the river ranges from artificial structures used for water supply and flood management to open spaces that provide more natural types of habitat. The flow regime of the Sacramento River is managed for water supply and flood management.

The upper Sacramento River extends from Keswick Dam for approximately 240 miles to the Delta. The Sacramento River is the largest river in California. The river and its tributaries provide spawning, rearing, and migratory habitat for four races of Chinook salmon, steelhead, and two species of sturgeon. Winter-run and spring-run Chinook salmon are federally and state-listed species and Central Valley steelhead and green sturgeon are federally listed. These species are described further in Appendix 11A.

The level of flow in the upper Sacramento River below Keswick Dam results from controlled releases from Shasta and Keswick reservoirs, as well as transfers from the Trinity River and natural accretions. The releases and transfers are determined by a suite of laws, regulations, contracts, and agreements to address demands of water users, requirements for water quality, and needs of fish populations throughout the river and the Delta. In particular, operations are regulated by the State Water Board D-1641 decision, which requires flow releases to meet Delta standards, and the WRO 90-5 decision, which requires cold-water releases to meet temperature targets at compliance points in the upper Sacramento River. In addition to the requirements set forth as part of the WRO 90-5 decision, the Biological Opinion (BiOp) on the Long-Term Operation of the CVP and the SWP (National Marine Fisheries Service 2019a) prescribes a tiered operational approach that allows for variable water temperature objectives for egg incubation based on water year and cold-water pool conditions.

The upper Sacramento River's water temperatures are controlled by selective withdrawal through the temperature control device (TCD) in Shasta Lake and by balancing releases between Lewiston Reservoir (Trinity River) and Shasta Lake. The water temperature operations have three principal objectives: (1) provide enough cold water to optimize survival of the current year's winter-run Chinook salmon eggs and alevins and those of other salmonids (Chinook salmon and steelhead), (2) stabilize water levels through the fall to avoid dewatering redds and stranding juveniles of winter-run and other salmonids, and (3) conserve and rebuild Shasta storage in the fall and winter to provide the cold-water pool resources needed to optimize survival of the next year's winter-run eggs and alevins and those of other salmonids. Protection of other salmonids, especially spring-run Chinook salmon and steelhead, is an important objective, but winter-run Chinook salmon is the principal target of operations because it alone has no spawning habitat other than that in the upper Sacramento River (with some recent spawning in Battle Creek and Clear Creek, in particular as a result of success in recent years for the Battle Creek reintroduction program (e.g., at least 700 subadults and adults returned in 2020 as a result of juvenile releases undertaken in 2018 and 2019; U.S. Fish and Wildlife Service 2020).

Habitat Conditions and Environmental Stressors in Sacramento River Area

Aquatic Habitat

The mainstem Sacramento River provides habitat for native and introduced (nonnative) fish and other aquatic species. The diversity of aquatic habitats ranges from fast-water riffles and glides in the upper reaches to tidally influenced slow-water pools and glides in the lower reaches (Vogel 2011).

A few miles downstream of Keswick Dam, the river enters the valley and the floodplain broadens. Historically, this area had wide expanses of riparian forests, but the riparian zone now has a great deal of urban encroachment. In the middle Sacramento River between Red Bluff and Chico Landing, the mainstem channel is flanked by broad floodplains (The Nature Conservancy 2007a). In the lower reaches downstream of Verona, much of the Sacramento River is constrained by levees.

Dredging, dams, levee construction, urban encroachment, and other human activities in the Sacramento River have modified aquatic habitat, altered sediment dynamics, simplified stream bank and riparian habitat, reduced floodplain connectivity, and modified hydrology (National Marine Fisheries Service 2009). However, some complex floodplain habitats remain in the system such as reaches with setback levees and the Yolo and Sutter Bypasses.

Beginning in 1995, planning was initiated to restore naturally spawning anadromous fish populations in Battle Creek, and construction began in 2010 on the Battle Creek Salmon and Steelhead Restoration Project (Greater Battle Creek Watershed Working Group 2006). When complete, the restoration project will restore ecological processes along 42 miles of Battle Creek and 6 miles of tributaries, including about 5 miles of habitat suitable for spawning by winter-run Chinook in North Fork Battle Creek. Higher minimum flow requirements will increase instream flows, subsequently cooling water temperatures, increasing stream area, and providing reliable passage conditions for adult salmonids in downstream reaches. The restoration project also will result in improved water quality for the Coleman NFH. In 2018, 220,000 juvenile winter-run Chinook salmon from Livingston Stone NFH were released into Battle Creek; in 2019, 185,000 juveniles were released (Sisco 2019).

Holding Habitat

An abundance of deep, cold-water pools in the mainstem Sacramento River provide habitat for holding adult anadromous salmonids during all months of the year (Vogel 2011). Green sturgeon also use deep pools for holding and spawning, but also hold farther downstream.

Spawning Habitat

Salmonid spawning habitat on the Sacramento River is affected by the amount and quality of coarse sediment and the levels of flow and water temperature, including levels determined by the operations of the CVP and local water diverters.

Water Temperatures

Water temperatures in the upper Sacramento River are influenced by the timing, volume, and temperature of water releases from Shasta and Keswick Dams, and are currently managed according to State Water Board Water Rights Orders 90-05 and 91-01. These orders require Reclamation to operate Keswick and Shasta Dams and the Spring Creek Power Plant to a daily average water temperature of 56°F as far downstream in the Sacramento River as practicable during periods when higher temperatures would be harmful to winter-run Chinook salmon. Under the orders, the water temperature compliance point may be modified to an upstream location when the objective cannot be met at RBPP. A TCD on Shasta Dam allows Reclamation to control the temperature of the water released from the dam. Water temperature directly affects many metabolic functions of fish and affects the availability of DO, which is required for respiration. In recent years, elevated water temperatures have resulted in high mortalities of winter-run eggs and alevin (Martin et al. 2016) and may have adversely affected other salmon runs and other fish species present in the Sacramento River during summer. Drought conditions make efforts to maintain suitable water temperatures in the upper Sacramento River more difficult because reduced storage in Lake Shasta results in a smaller cold-water pool. Low egg to fry survival during 2021 coincided with both high water temperatures and thiamine deficiency in adults returning to spawn. Thiamine deficiency is associated with impacts on fry growth, predator avoidance, prey capture ability, visual development, swimming ability, and immune function. Thiamine deficiency is thought to be passed onto eggs from the mother. As a result, thiamine deficiency, in combination with temperature-dependent mortality, may have contributed to low egg to fry survival in 2021. The 2 years of data on thiamine levels in winter-run currently limit our understanding of the relative contribution of water temperature and thiamine deficiency on winter-run survival.

Sediment Conditions

Shasta and Keswick Dams substantially influence sediment transport in the upper Sacramento River because they block sediment that would normally be transported downstream (The Nature Conservancy 2007a; California Department of Water Resources 1985). The result has been a net loss of coarse sediment, including gravel particle sizes suitable for salmon spawning, in the Sacramento River downstream of Keswick Dam (Bureau of Reclamation 2013). To address the issue of spawning gravel loss downstream of Keswick Dam, Reclamation has placed roughly 5,000 tons of washed spawning gravel into the Sacramento River downstream of Keswick approximately every other year since 1997 (Bureau of Reclamation 2010).

Spawning Habitat Availability

The suitability of physical habitat for salmonid spawning (i.e., not including water quality parameters such as temperature and DO) is largely a function of the availability of clean, coarse gravel for constructing redds, favorable depths, and suitable flow velocities. Instream flow potentially affects all three of these habitat characteristics and often affects the availability of suitable habitat.

Most winter-run Chinook salmon spawn in the uppermost reach of the Sacramento River, from Keswick Dam to the confluence with Clear Creek. Operations of Shasta and Keswick Dams generally determine flow and temperature in this area and largely determine spawning habitat availability.

Like winter-run Chinook salmon, the spawning distributions of spring-run, fall-run, and late fall–run Chinook salmon populations that spawn in the Sacramento River are primarily upstream of RBPP, with fall-run spawning the farthest downstream. About 15% of fall-run redds are constructed downstream of the RBPP. Spring-run spawning locations are generally between those of winter-run and fall-run Chinook salmon, and late fall–run spawning distribution is most similar to that of winter-run Chinook salmon (National Marine Fisheries Service 2017). The spawning substrate, depth, and flow velocity requirements of the four Chinook salmon runs are generally similar, so differences in spawning habitat availability among the runs are most strongly related to differences in their spawning distributions and in the prevailing flow and temperature levels during the time of year that they spawn.

Variations in flow can also have major effects on spawning habitat. Reductions in flow following spawning may result in dewatering of the redds and mortality of incubating eggs and alevins (U.S. Fish and Wildlife Service 2006). Large increases in flow may result in scouring of riverbed sediments, including any redds present, or entombment in deposited sediments.

The spawning distribution of steelhead in the upper Sacramento River is poorly known, but the suitability of spawning habitat is largely determined by the same physical habitat and water quality parameters that affect the Chinook salmon runs, although preferred depths, flow velocities, and spawning substrates are somewhat different than those for Chinook salmon. As for the salmon runs, the availability of suitable spawning habitat for steelhead is strongly affected by instream flow (U.S. Fish and Wildlife Service 2011a).

Rearing Habitat

Rearing habitat suitability for juvenile salmonids is generally related to flow much as described above for spawning habitat suitability, although important habitat features for rearing include cover, and the suitability of specific depths and flow velocities differ from those for spawning (U.S. Fish and Wildlife Service 2005a, 2005b).

Inundated floodplains typically provide large areas of suitable rearing habitat for juvenile salmonids (Sommer et al. 2001). In the Sacramento River between Red Bluff and Chico Landing, the mainstem channel is flanked by broad floodplains. Ongoing sediment deposition in these areas provides evidence of continued inundation of floodplains in this reach (California Department of Water Resources 1994). Between Chico Landing and Colusa, the Sacramento River is bounded by levees that provide flood protection for cities and agricultural areas. However, the levees in this portion of the Sacramento River are, for the most part, set back from the mainstem channel such that flooding can be significant within the river corridor (The Nature Conservancy 2007b). Downstream of Colusa, the Sacramento River channel is tightly constrained by federal levees, dominated by riprap, with no floodplain and only remnant riparian vegetation.

Vogel (2011) suggested that the mainstem Sacramento River may not provide adequate rearing areas for fry-stage anadromous salmonids, as evidenced by rapid displacement of fry from upstream to downstream areas and into non-natal tributaries during increased flow events. Underwater observations of salmon fry in the mainstem Sacramento River suggest that optimal habitats for rearing may be limited at high flows (Vogel 2011).

Fish Passage and Entrainment

Historically, anadromous salmonids had access to a minimum of about 493 miles of habitat in the Sacramento River (Yoshiyama et al. 1996). After completion of Shasta Dam in 1945, access to approximately 207 miles was blocked. Keswick Dam, just downstream of Shasta Dam, is now the upstream extent of available habitat for anadromous fish in the Sacramento River.

On the main stem, the Anderson-Cottonwood Irrigation District (ACID) Dam impedes anadromous fish migration during the spring and summer. ACID completed two state-of-the-art fish ladders and fish screens to improve passage for salmonids at their dam and diversion facility in 2001. These ladders allow access to spawning and rearing habitat up to Keswick Dam, which is an impassable barrier to migration (California Department of Water Resources 2005). However, adult green sturgeon that migrate upstream in April, May, and June continue to be completely blocked by the ACID Dam (National Marine Fisheries Service 2009), rendering approximately 3 miles of spawning habitat upstream of the diversion dam inaccessible. Adult green sturgeon that pass upstream of the intake before April are delayed for 6 months until the flashboards are pulled and they can return downstream to the ocean. Newly emerged green sturgeon larvae that hatch upstream of the ACID Dam must hold for 6 months upstream of the dam or pass over it and be subjected to higher velocities and turbulent flow below the dam (National Marine Fisheries Service 2009).

Downstream of the ACID Dam, the RBDD, which created a partial barrier to upstream migration, was decommissioned in 2013 and no longer impedes upstream fish passage (Tehama-Colusa Canal Authority 2012). The TCCA, in cooperation with Reclamation, completed construction of a state-of-art-fish screen and pumping plant, the RBPP. This facility allowed the RBDD gates to be permanently lifted out of the water, providing unimpeded passage of adult and juvenile Chinook salmon, steelhead, and sturgeon (Tehama-Colusa Canal Authority 2012).

The largest one of the oldest diversions on the Sacramento River is the GCID, which operates a 3,000-cfs pumping plant approximately 4 miles north of Hamilton City. The entrainment of juvenile emigrating salmon into this diversion has been an ongoing concern. At the time winter-run Chinook salmon were listed in 1989, the diversion was protected by a series of poorly functioning rotary drum screens. GCID, Reclamation, and the fishery agencies worked to address this deficiency and in 2001, GCID completed installation of a state-of-the-art flat plate fish screen to exclude migrating fish from the diversion (Glenn-Colusa Irrigation District 2001).

Numerous other diversions are located on the Sacramento River. Herren and Kawasaki (2001) documented up to 431 diversions from the Sacramento River between Shasta Dam and the city of Sacramento. Studies at some unscreened diversions in the Sacramento River found low rates of entrainment of juvenile Chinook salmon (Vogel 2013; Hanson 2001). Mussen et al. (2014) examined the risk to green sturgeon from unscreened water diversions and found that juvenile green sturgeon entrainment susceptibility (in a laboratory setting) was high relative to that estimated for Chinook salmon, suggesting that unscreened diversions could be a contributing mortality source for the threatened southern DPS of green sturgeon. Reclamation is currently coordinating with USFWS via the Anadromous Fish Screen Program to support improvements at other fish screens.

The Coleman Fish Hatchery weir is a barrier to anadromous fish upstream passage in Battle Creek, as are various Pacific Gas and Electric Company dams (e.g., Wildcat) located on the creek (Yoshiyama et al. 1996). Yoshiyama et al. (1996) reported that the Coleman South Fork Diversion Dam is the first impassible barrier on Battle Creek.

Hatcheries

The Livingston Stone NFH, located at the foot of Shasta Dam, is a conservation hatchery that has been producing and releasing juvenile winter-run Chinook salmon since 1998. There is growing concern about the potential genetic effects that may result from the use of a conventional hatchery program to supplement winter-run Chinook salmon populations for extended periods of time. To maintain a low risk of extinction (i.e., a 5% risk of extinction in 100 years), Lindley et al. (2007) recommend that no more than 5% of the naturally spawning population should be composed of hatchery fish. Since 2001, more than 5% of the winter-run Chinook salmon run has been composed of hatchery-origin fish, and in 2005, the contribution of hatchery fish was more than 18% (Lindley et al. 2007).

The Livingston Stone NFH minimizes hatchery effects in the population by preferentially collecting wild adult winter-run Chinook salmon for brood stock (U.S. Fish and Wildlife Service 2011b). Up to 15% of the estimated run size for winter-run Chinook salmon run may be collected for brood stock use (up to a maximum of 120 natural-origin winter-run Chinook salmon per brood year). Although there is no adult production goal, Livingston Stone NFH annually releases up to 250,000 winter-run Chinook salmon in late January or early February. Winter-run Chinook salmon are released at the pre-smolt stage and are intended to rear in the freshwater environment prior to smoltification. The pre-smolts are released into the Sacramento River at Caldwell Park in Redding, about 10 miles downstream of the hatchery. All juvenile winter-run Chinook salmon produced at Livingston Stone NFH are adipose fin-clipped and coded-wire-tagged (California Hatchery Scientific Review Group 2012). As previously noted, 220,000 juvenile winter-run Chinook salmon from Livingston Stone NFH were released into Battle Creek in 2018, and 185,000 were released in 2019 (Sisco 2019).

Coleman NFH, located on Battle Creek, was established in 1942 by Reclamation to partially mitigate habitat and fish losses from historical spawning areas caused by construction of two CVP facilities, Shasta and Keswick Dams. The hatchery is funded by Reclamation and operated by USFWS. The steelhead program at the hatchery was initiated in 1947 to mitigate losses resulting from the CVP (California Hatchery Scientific Review Group 2012).

Disease

Several endemic salmonid-specific pathogens occur in the Sacramento River: *Ceratonova shasta* (salmonid ceratomyxosis), *Parvicapsula minibicornis* (myxosporean parasite), *Flavobacterium columnare* (columnaris), *Novirhabdovirus* (infectious hematopoietic necrosis [IHN] virus), *Renibacterium salmoninarum* (bacterial kidney disease), *Flavobacterium psychrophilum* (cold-water disease), *Ichthyophthirius multifiliis* (white spot disease or Ich), and *Aeromonas salmonicida* (furunculosis) (California Department of Water Resources 2004; Foott 2014). Of the diseases caused by these pathogens, salmonid ceratomyxosis and myxosporean parasite are of most concern for fisheries management in the region (Foott 2014; Foott et al. 2017). The Coleman NFH on Battle Creek employs management practices and protocols to minimize the spread of pathogens to Battle Creek and the Sacramento River (Cramer Fish Sciences 2016).

Salmonid ceratomyxosis is endemic to the Sacramento River Basin. While native fish have developed some resistance to the disease, mortality in all ages of anadromous and resident salmonids still occurs. Steelhead appear to be particularly susceptible to the disease, compared to Chinook (California Department of Water Resources 2004). Fish can become infected at temperatures as low as 39°F; however, mortality predominantly occurs at water temperatures exceeding 50°F (Bartholomew 2012).

The risk of infection in the Sacramento River Basin with salmonid ceratomyxosis is highest when the fish remain for an extended period in an infectious zone. This is an area with a high concentration of infectious pathogen(s) (Foott et al. 2017; Pacific Fishery Management Council 2018). An infectious zone may develop when the following factors coincide: low flow velocity and volume (especially in proximity to spawning areas) and water temperatures above 54°F (Ray and Bartholomew 2013; Foott et al. 2017; Pacific Fishery Management Council 2018).

Predation

On the mainstem Sacramento River, high rates of predation have been known to occur at the diversion facilities and areas where rock revetment has replaced natural river bank vegetation (National Marine Fisheries Service 2009). Chinook salmon fry, juveniles, and smolts may be more susceptible to predation at these locations because Sacramento pikeminnow and striped bass congregate in areas that provide predator refuge (Williams 2006; Tucker et al. 2003). A recent study by Stompe et al. (2020) comparing angling catch per unit effort (CPUE) at engineered (water diversions or beam bridges), riprap, and natural locations found no statistically significant difference for striped bass and a marginally significant difference for Sacramento pikeminnow (greater CPUE) at engineered sites compared to natural sites, with no significant difference in diet composition between sites.

11.2.5.3. Feather River

Description of Lake Oroville, Thermalito Afterbay, and Feather River

The Feather River contributes approximately 25% of the total flow in the Sacramento River (Federal Energy Regulatory Commission 2007). The lower Feather River extends about 67 miles from the Fish Barrier Dam below Oroville Dam to its confluence with the Sacramento River. There are two reaches where Chinook salmon, steelhead and green sturgeon spawn: the low-flow channel (LFC), which extends about 8 miles from the Fish Barrier Dam to the Thermalito Afterbay outlet, and the high-flow channel (HFC), which extends about 15 miles from the Thermalito Afterbay outlet to Honcut Creek (Vogel 2011). Most of the in-river spawning and rearing by salmon and steelhead occurs in the LFC (Federal Energy Regulatory Commission 2007; California Department of Water Resources 2007).

The Oroville Facilities include Oroville Dam, the Thermalito Diversion Dam, the Thermalito Complex, and the Feather River Fish Hatchery. The hatchery, which is managed by CDFW, raises spring and fall-run Chinook salmon and steelhead. A special fish barrier dam leads adults returning to spawn in the Feather River into the Feather River Fish Hatchery (California Department of Water Resources 2007). Central Valley spring-run and Central Valley fall-run Chinook salmon and California Central Valley steelhead are reared at the Feather River Hatchery, using cold water diverted from Lake Oroville. The capacity of this facility is 2.5 million fingerlings a year. Approximately 20% of the salmon and steelhead returning to spawn use the hatchery and 80% spawn in the Feather River (ICF International 2015). Hatchery operations are discussed in more detail below.

Oroville Dam impounds Lake Oroville, which is the principal water storage facility of the SWP. The reservoir has a storage capacity of 3.5 MAF and a surface area of 15,810 acres. The WSE of Lake Oroville is reduced through the summer season as releases from storage are required to meet downstream requirements, including instream flow, environmental requirements, in-basin uses, and urban and agricultural demand. In wetter years, the maximum total release from Lake Oroville typically occurs in February and March, due primarily to the requirement for large releases to meet flood control criteria and maintain sufficient flood storage in the reservoir. In drier years, the highest releases from Lake Oroville typically occur in July (California Department of Water Resources 2007).

Releases from Lake Oroville are made into the Diversion Pool below Oroville Dam, where water can be released through the Thermalito Diversion Dam Power Plant to the LFC of the Feather River or diverted through the Thermalito Power Canal Forebay and into Thermalito Afterbay. Flows can be diverted from Thermalito Afterbay into agricultural canals to meet local Feather River Service Area requirements (including Western Canal and Water District and Richvale Irrigation District) or released through the Thermalito Afterbay outlet back into the Feather River, where they combine with flows passing through the LFC to produce the HFC. Pursuant to a 1982 agreement with CDFW, DWR provides 600 cfs to the LFC to accommodate spawning by anadromous salmonids, and at least 700 cfs the remainder of the year. Water temperatures in the LFC are required to meet stringent standards for protection of all life stages of the anadromous salmonids. In addition, cold water is supplied to the Feather River Fish Hatchery and the return flow from the hatchery helps moderate the temperatures in the LFC.

The Project has operated under an annual license, which extends the terms of the original license. In anticipation of the original license expiration (2007), DWR filed an application for renewal with Federal Energy Regulatory Commission (FERC) in 2005. In 2006 DWR entered into a settlement agreement regarding operation of Oroville Dam with several water agencies, state and federal regulatory agencies, Tribes, local government agencies, and non-governmental organizations. The agreement includes a comprehensive Lower Feather River Habitat Improvement Plan which contains flow and temperature standards for the LFC and HFC, and a number of programs to restore and improve habitat for spawning and rearing of anadromous fish. This agreement will be implemented upon DWR's acceptance of a renewed license from FERC. While issuance of a new license is pending, NMFS has concluded formal consultation with FERC on the proposed new license and accepted the proposed flow and temperature standards as sufficient to avoid jeopardy to list fish under its jurisdiction and the Central Valley Regional

Water Quality Control Board has completed its CWA 401 determination and likewise has acknowledged the suitability of the proposed flow and temperature measures in its determination.

Lake Oroville

Lake Oroville typically thermally stratifies beginning in the spring, begins to de-stratify in the fall, and remains relatively uniform throughout the winter. Because of this stratification regime, Lake Oroville supports both cold-water and warm-water fisheries that are thermally segregated for much of the year. The cold-water fish use the deeper, cooler, well-oxygenated hypolimnion and thermocline, whereas the warm-water fish are found in the warmer, shallower, epilimnetic and littoral zones (California Department of Water Resources 2007).

Lake Oroville's cold-water fishery is primarily composed of coho salmon, although rainbow trout, brown trout, and lake trout (*Salvelinus namaycush*) are periodically caught. The cold-water fishery for coho salmon is sustained by hatchery stocking. The Lake Oroville warm-water fishery is a self-sustaining fishery. Spotted bass are the most abundant bass species in Lake Oroville, followed by largemouth, redeye (*Micropterus coosae*), and smallmouth bass. Catfish species, white crappie (*Pomoxis annularis*), and black crappie are also present. Resident native species include white sturgeon, Sacramento sucker, Sacramento pikeminnow, and hardhead (California Department of Water Resources 2007).

Project operations influence fish habitat in Lake Oroville through manipulation of the amount of cold water released into the Feather River and changes in the WSE of Lake Oroville that result from flood control, power generation, and water releases downstream. Cold water is taken from Lake Oroville's hypolimnion for releases to the downstream fishery in the main channel of the Feather River, thereby potentially limiting the amount of cold water available for salmonids in Lake Oroville. The WSE fluctuations in Lake Oroville occur on a seasonal basis, resulting from seasonal variations in upstream tributary inflows into the reservoir, as well as seasonal variations in Oroville Facilities reservoir releases. Reservoir stage reductions can reduce the amount of littoral fish habitat, bass nest survival, invertebrate food supply, and cold-water pool volume (California Department of Water Resources 2007).

Thermalito Afterbay

Thermalito Afterbay provides habitat for both cold-water and warm-water fishes. This 4,300 surface-acre reservoir has gently sloping banks with large areas of rooted aquatic vegetation along its upper margins. Depths rarely exceed 20 feet. Changes in flow rates, pumpback operations, and WSE resulting from operations affect water temperatures and the quality, quantity, and distribution of fish habitat in the afterbay. Water temperatures can vary widely around the afterbay in the summer, from the low 60s (°F) near the tailrace channel that feeds the afterbay to the mid-80s (°F) in the backwater areas that do not readily circulate (California Department of Water Resources 2007). The operational range of WSE fluctuations is 12 feet, although the normal fluctuation range is between 4 and 8 feet. The WSE can fluctuate rapidly and frequently, resulting in a high degree of variability in water levels from day-to-day and week-to-week, depending on operation (California Department of Water Resources 2007).

Fish species observed in Thermalito Afterbay include largemouth bass, smallmouth bass, rainbow trout, brown trout, sunfish species, black crappie, channel catfish, carp, and large schools of wakasagi (*Hypomesus nipponensis*). The rainbow and brown trout that occur in the afterbay likely are washed in from the smaller and colder Thermalito Forebay. The Thermalito Afterbay likely provides good habitat for largemouth and smallmouth bass, except that nest dewatering from reservoir fluctuations likely limits juvenile recruitment. Large schools of wakasagi provide a plentiful source of prey (California Department of Water Resources 2007).

Lower Feather River

At least 22 fish species have been found in the lower Feather River, although some of these may be non-residents washed down from Lake Oroville. Fish species of management concern present in the lower Feather River include Central Valley spring-run Chinook salmon, Central Valley fall-run Chinook salmon, California Central Valley steelhead, southern DPS green sturgeon, Pacific lamprey (*Entosphenus tridentatus*), river lamprey (*Lampetra ayresii*), hardhead, and Sacramento splittail. Chinook salmon are abundant in the Feather River, with an estimated 30,000 to 170,000 Chinook salmon spawning in the Feather River annually (Federal Energy Regulatory Commission 2007). There is substantial introgression between Feather River spring-run and fall-run Chinook salmon, likely a result of past hatchery practices and loss of geographic separation when Oroville Dam was built (Williams 2006; National Marine Fisheries Service 2009; Vogel 2011).

Oroville Dam releases are managed to benefit cold-water fisheries. Most Feather River spring-run Chinook salmon, fall-run Chinook salmon, and steelhead natural spawning occurs in the LFC, which has flow and water temperature restrictions (Federal Energy Regulatory Commission 2007:122). Flows in the LFC are currently required to be at least 800 cfs from September 9 to March 31 of each water year, to accommodate spawning by anadromous salmonids, and at least 700 cfs the remainder of the year. Water temperatures in the channel LFC range from 47°F in the winter to 65°F in the summer (Vogel 2011). The summer water temperatures can limit salmon production. Gravel recruitment is also an issue for the LFC of the river (Vogel 2011).

The flow regime in the reach of the Feather River extending from the Thermalito Afterbay outlet (RM 59) to the confluence of the Feather and Sacramento Rivers varies depending on runoff and month. Flows in this reach of the Feather River typically vary from the minimum flow requirement up to a flow of 7,500 cfs (Federal Energy Regulatory Commission 2007). Spawning by southern DPS green sturgeon was documented in 2011, a wet year, at the Thermalito Afterbay outlet (Seesholtz et al. 2015).

Habitat Conditions and Environmental Stressors in Feather River Area

Aquatic Habitat

The lower Feather River contained at least 71 miles of suitable spawning habitat for winter-run and spring-run Chinook salmon, steelhead, and green sturgeon prior to the installation of the Oroville Complex on the Feather River (Yoshiyama et al. 2001; Schick et al. 2005). Extensive mining, irrigation, and other dams significantly reduced the amount of suitable habitat for these species (Yoshiyama et al. 2001). Currently, most spawning for these fishes is concentrated in the uppermost 3 miles of accessible habitat downstream of the Feather River Fish Hatchery (Federal

Energy Regulatory Commission 2007). As a result, spawning is concentrated at unnaturally high levels directly downstream of Oroville Dam and the Fish Barrier Dam in the LFC.

The lower Feather River is almost entirely contained within a series of levees. Streamflow is regulated by the Oroville Complex (composed of the Oroville Dam, Thermalito Diversion Dam, and Thermalito Afterbay outlet), and releases from the Oroville Complex are planned weekly to accommodate water deliveries, Sacramento Valley in-basin demands such as Delta requirements, instream flow requirements in the Feather River, and minimum flood management space requirements. The lower Feather River's modified flow regime has reduced the frequency of channel- forming flows, which, along with levees, has reduced the lateral movement of the Feather River and resulted in a more channelized river with reduced sinuosity.

Natural channel processes have also been affected by flow fluctuations, including the interruption of the downstream movement of gravel and wood, lateral river movement forming side channels, and a reduction in the frequency of inundated flood plains. An estimated 97% of the sediment from the upstream watershed has been trapped behind Oroville Dam, only allowing the discharge of fine sediment into the lower Feather River (Federal Energy Regulatory Commission 2007). As a result, optimum habitat features such as gravel and large woody materials from upstream reaches are now limited in the lower Feather River, and the median gravel diameter (D50) in the LFC may generally be too large for successful redd construction by native salmonids (Federal Energy Regulatory Commission 2007). However, the lower Feather River watershed encompasses approximately 803 square miles, with approximately 190 miles of major streams and 695 miles of minor streams, which contribute to the sediment load of the lower Feather River. FERC (2007) noted that the suitability of gravel sizes for spawning Chinook salmon increased with distance downstream of Oroville Dam.

DWR currently operates the Oroville Complex (i.e., the Oroville Dam, Thermalito Diversion Dam, and Fish Barrier Dam) consistent with the applicable NMFS and USFWS BiOps. When the WSE of Lake Oroville is greater than 773 feet, minimum instream flow releases downstream of the Thermalito Afterbay outlet range from 1,000 to 1,700 cfs, depending on water year type and season.

Mean daily flow in the Feather River ranged from approximately 2,000 cfs in Critically Dry Water Years to 18,000 cfs in Wet Water Years prior to the completion of Oroville Dam (Oroville gage 1906–1965), and from approximately 1,000 cfs during Critically Dry Water Years to 12,000 cfs during Wet Water Years after the completion of the Oroville Dam (Gridley gage 1969–2012) (National Marine Fisheries Service 2016).

Modeling and surveys have been conducted to inform management decisions by identifying optimum water flow for native salmonids. According to Oroville Facilities Relicensing (FERC 2100 Relicensing site), optimum Chinook salmon flow suitability for spawning is about 800–825 cfs in the LFC and 1,200 cfs in the HFC. Steelhead appeared to have no optimum flow for spawning in the LFC; however, optimum flow was just under 1,000 cfs in the HFC (California Department of Water Resources 2004).

Fish Passage

The Oroville Complex facilities currently block the upstream migration of anadromous fish from historically available spawning areas in the Feather River and have altered flow regimes as part of ongoing operations, which affects upstream and downstream migration passage in downstream reaches. The loss of spawning habitat and altered passage conditions downstream of the dams have resulted in hybridization of spring-run and fall-run Chinook salmon populations and have decreased population sizes.

In addition to the Oroville Complex, two other potential/partial upstream migration barriers exist in the lower Feather River during low-flow or high-flow conditions (approximately 2,074 cfs or 9,998 cfs, respectively). These are the Sunset Pumps Diversion Dam and Steep Riffle (Federal Energy Regulatory Commission 2007). The Sunset Pumps Diversion Dam is a rock weir structure approximately 27 miles downstream of the Fish Barrier Dam that may impede or delay passage under certain flow conditions. Planning efforts are underway to resolve/correct this feature. Steep Riffle is located approximately 2 miles upstream of the Thermalito Afterbay outlet but is generally considered passable under both low and high flows (Beamesderfer et al. 2004).

Hatcheries

The Feather River Fish Hatchery is part of the SWP Oroville Complex. The hatchery is operated as part of a mitigation measure established to address anadromous fish decline as a result of loss of habitat upstream of Oroville Dam (National Marine Fisheries Service 2009). This facility produces fall-run Chinook salmon, spring-run Chinook salmon, and steelhead and is the sole contributor of hatchery-raised spring-run Chinook salmon found within the Central Valley (California Hatchery Scientific Review Group 2012). The hatchery-raised spring-run Chinook salmon are included in the federal designation of Central Valley spring-run Chinook salmon evolutionarily significant unit (ESU) (70 *Federal Register* [FR] 37160).

The initial Chinook salmon production protocols separated returning spring-run and fall-run Chinook salmon based solely on the run timing periods, which resulted in considerable mixing of spring-run and fall-run Chinook salmon stocks due to the overlap in spawning periods (Cavallo et al. 2009). In 2005, the Feather River Fish Hatchery changed their methodology to prevent any further mixing; only fish entering the hatchery prior to July 1 receive an external tag. These are the only fish used as spring-run Chinook salmon broodstock (California Hatchery Scientific Review Group 2012; Cavallo et al. 2009). Additionally, all hatchery-raised spring-run Chinook salmon are now adipose fin-clipped, coded-wire-tagged (California Hatchery Scientific Review Group 2012), and thermally otolith-marked with race and brood year specific details (Cavallo et al. 2009). The juvenile hatchery production goal is two million smolts, which are released during April or May. Returning hatchery-produced spring-run Chinook salmon are intended to spawn and integrate with the natural population in the lower Feather River, although there are no specific goals for the number of adult spring-run Chinook salmon.

The steelhead program at the Feather River Fish Hatchery traps marked hatchery-origin and unmarked natural-origin steelhead to artificially produce steelhead for later release. However, few unmarked fish are trapped annually, and only fish returning to the Feather River Basin are used for broodstock. There are no specific goals for the quantity of adults produced by this program; however, there is an annual production goal of 450,000 yearlings released in January or

February. All Feather River Hatchery steelhead have their adipose fin clipped prior to release to distinguish them from the naturally spawned population (California Hatchery Scientific Review Group 2012).

The Feather River Fish Hatchery disease management protocols employ an ultraviolet treatment system, periodic testing, prescribed therapeutic treatments, and has modified the stocking practices of Lake Oroville to successfully manage disease (California Department of Water Resources 2004).

Disease

The salmonid-specific diseases that are known to occur in the Feather River Basin are salmonid ceratomyxosis, columnaris, IHN virus, bacterial kidney disease, and cold-water disease (California Department of Water Resources 2004). Each of these diseases has been shown to infect stocked and native salmonids in the Feather River (Federal Energy Regulatory Commission 2007). Salmonid ceratomyxosis and IHN virus are of most concern for fisheries management in the region because of the associated fish mortality rates at the hatchery (California Department of Water Resources 2004).

Salmonid ceratomyxosis is endemic to the Feather River Basin. While native fish have developed some resistance to the disease, mortality in all ages of anadromous and resident salmonids still occurs. Steelhead appear to be particularly susceptible to the disease, compared to Chinook salmon (California Department of Water Resources 2004). Fish can become infected at temperatures as low as 39°F; however, mortality predominantly occurs at water temperatures exceeding 50°F (Bartholomew 2012). While whirling disease has been found in upstream tributaries of the Feather River, it has not been detected downstream of Oroville Dam (California Department of Water Resources 2004).

The Feather River Fish Hatchery experienced severe IHN virus outbreaks in 2000 and 2001. The University of California at Davis and USFWS have indicated that although there were no clinical signs of disease, 28% and 18% of adult salmonids returning to the Yuba and Feather Rivers, respectively, carried IHN virus (Brown et al. 2004). Survivors of IHN virus can become carriers, and the disease can be spread via contaminated water or contact with carriers, making IHN virus particularly difficult to control.

The Feather River Fish Hatchery employs best management practices and protocols to avoid the spread of diseases. By installing an ultraviolet treatment system, modifying the stocking of Lake Oroville, conducting periodic testing, and using prescribed therapeutic treatments, the hatchery has been successful in adaptively managing disease concerns as they arise (California Department of Water Resources 2004).

Predation

Sufficient information is not available to estimate the current rate of predation on juvenile salmonids in the lower Feather River. As reported by FERC (2007), the Fish Barrier Dam concentrates most anadromous salmonid spawning in the LFC. Reported counts of known predators on juvenile anadromous salmonids in the LFC were low. However, significant numbers of predators reportedly do exist throughout the lower Feather River and have been known to congregate at passage impediments, such as the Sunset Pumps Diversion Dam (Seesholtz et al. 2004; Windell et al. 2017). Passage impediments increase the risk of predation by providing habitat for predator fishes that feed on outmigrating juvenile salmonids that become disoriented in the turbulent water below the dam, and by increasing the exposure to predation by delaying passage.

11.2.5.4. American River

Description of Folsom Lake, Lake Natoma, and American River

The American River watershed includes nearly 2,100 square miles (Bureau of Reclamation et al. 2006). Three forks of the American River (north, middle, and south forks) meet upstream of Folsom Dam, and the collective flow is transported through Lake Natoma and the 23-mile lower American River before converging with the Sacramento River just north of the city of Sacramento. Access to the upper reaches of the American River by anadromous fish is blocked at Nimbus Dam.

Folsom Lake has a capacity of approximately 977 TAF. Thermal stratification occurs in Folsom Lake from April until November which results in a thermocline with warmer, less dense water at the surface and colder, denser water underneath (U.S. Army Corps of Engineers et al. 2012). This thermal stratification allows both cold-water and warm-water fish species to coexist in the reservoir. Warm-water fish species include native hardhead, California roach (*Hesperoleucus symmetricus*), Sacramento pikeminnow, and Sacramento sucker, as well as nonnative largemouth bass, smallmouth bass, spotted bass, sunfish, black crappie, and white crappie (Bureau of Reclamation 2007). Cold-water fish species include native rainbow trout and planted Chinook and kokanee salmon (*Oncorhynchus nerka*), as well as nonnative brown trout (Bureau of Reclamation 2007).

Lake Natoma is a regulating afterbay to the Folsom Power plant and provides more uniform releases into the lower American River. Lake Natoma is formed by Nimbus Dam and is relatively shallow (average depth near 16 feet) (Bureau of Reclamation 2005). Lake Natoma also serves as a forebay for diversions to the Folsom South Canal. Surface water elevations in Lake Natoma may fluctuate daily between 4 and 7 feet (U.S. Army Corps of Engineers et al. 2012). Owing to the high fluctuation in surface water elevations and water temperatures, the lake has relatively low productivity as a fishery. The fish species present in Lake Natoma are generally similar to those found in Folsom Lake (Bureau of Reclamation 2007).

Reclamation operates the CVP American River Division for flood control, municipal, industrial, and agricultural water supplies, hydroelectric power generation, fish and wildlife protection, recreation, and Delta water quality. Many other small reservoirs above Folsom Lake with a combined reservoir storage of approximately 820 TAF provide hydroelectric generation and water supply without specific flood control responsibilities. Because Folsom Lake is the closest reservoir among SWP/CVP reservoirs to the Delta, operators can use releases from Folsom to address any Delta regulatory requirements, such as water quality or flow requirements more quickly than upstream reservoirs.

Reclamation operates Folsom Reservoir to meet water rights, contracts, and agreements that are specific to the American River Division and to those that apply to the entire CVP, including the

Delta Division. For lower American River flows (below Nimbus Dam), Reclamation has adopted the minimum release requirement (MRR) and approach proposed by the Sacramento Area Water Forum in 2017 in the 2017 Flow Management Standard Releases (American River Water Agencies 2017) and a "planning minimum." Reclamation works together with the American River water agencies to define the planning minimum, which is an appropriate amount of storage in Folsom Reservoir that represents the lower bound for typical forecasting processes at the end of calendar year that will provide releases of salmonid-suitable temperatures to the lower American River and reliable deliveries (using the existing water supply intakes and conveyance systems) to American River water agencies. Other components of American River operations include seasonal operations for flood control, improvements to the management of the Nimbus Hatchery, and number of spawning and rearing habitat restoration projects on the American River and several tributary creeks (Bureau of Reclamation 2020).

The MRR establishes minimum flows, as measured by the total release at Nimbus Dam, which vary throughout the year in response to the hydrology of the Sacramento and American River Basins. The MRF for January is based on the Sacramento River Index and ranges from 500 cfs to 1750. The MRR for February through December is based on the American River index: from February 1 to March 31 MRR varies between 500 and 1,750 cfs; from April 1 to June 30 it varies from 500 to 1,500 cfs; from July 1 to September 30 it varies from 500 to 1,750 cfs; from October 1 to October 30 it varies from 500 t0 1,500 cfs; and from November 1 to December 31 it varies from 500 to 2,000 cfs. These MRR flows are adjusted by redd dewatering protection adjustments to minimize the likelihood that reservoir operations would result in dewatering of fall-run Chinook salmon redds or steelhead redds.

Water temperatures in the lower American River are influenced primarily by the timing, magnitude, and temperature of water releases from Folsom and Nimbus Dams. Reclamation manages the Folsom/Nimbus Dam complex and the water temperature control shutters at Folsom Dam to maintain a daily average water temperature of 65°F (or other temperature as determined by temperature modeling) or lower at Watt Avenue Bridge from May 15 through October 31, to provide suitable conditions for juvenile steelhead rearing in the lower American River. If the temperature is exceeded for 3 consecutive days, Reclamation will notify NMFS and outline steps being taken to bring the water temperature back into compliance. During the May 15 to October 31 period, if the temperature requirement (as defined in the temperature plan) cannot be met because of limited cold water availability in Folsom Lake, then the target daily average water temperature at Watt Avenue may be increased incrementally (i.e., no more than 1°F every 12 hours) to as high as 68°F. The priority for use of the lowest water temperature control shutters at Folsom Dam shall be to achieve the water temperature requirement for listed species (i.e., steelhead), and thereafter may also be used to provide cold water for fall-run Chinook salmon spawning (Bureau of Reclamation 2019).

Habitat Conditions and Environmental Stressors in American River Area

Aquatic Habitat

The completion of Nimbus Dam in 1955 blocked upstream passage by anadromous fish and restricted available habitat in the lower American River to the approximately 23- river- mile reach below the dam to the confluence with the Sacramento River. Additionally, completion of

Folsom Dam has blocked downstream transport of sediment that contributes to the formation and maintenance of habitat for aquatic species in the American River and further downstream.

Beginning in 2008, in coordination with USFWS and the Sacramento Water Forum, Reclamation implemented a salmonid habitat improvement program in the lower American River. In 2008, approximately 5,000 cubic yards of gravel and cobble were put in the river just upstream of Nimbus Fish Hatchery. The following year, an additional approximately 7,000 cubic yards were placed adjacent to the Nimbus Fish Hatchery. In 2010, approximately 11,688 cubic yards of gravel and cobble were placed at Sailor Bar to enhance spawning habitat for Chinook salmon and steelhead in the lower American River (Merz et al. 2012). Additionally, the 2010 augmentation site contained a constructed cobble island and "scallops" in the substrate designed to add habitat heterogeneity to the main channel and rearing habitat for juvenile Chinook salmon and steelhead. An estimated 5,500 tons of cleaned cobble were also placed downstream of the 2010 augmentation site to divert flow into an adjacent, perched side channel, thereby preventing the dewatering of salmonid redds in a historically important spawning and rearing area if flows dropped.

At higher flows, channel geomorphology in the lower American River consists of bar complexes and side-channel areas, which may disappear at lower flows (National Marine Fisheries Service 2009). Spawning bed materials in the lower American River may begin to mobilize at flows of 30,000 cfs, with more substantial mobilization at flows of 50,000 cfs or greater (Bureau of Reclamation 2008). At 115,000 cfs (the highest flow modeled), particles up to 70 mm median diameter would be moved in the high-density spawning areas around Sailor Bar and Sunrise Avenue. Flood frequency analysis for the American River at the Fair Oaks gage shows that, on average, flood control releases exceed 30,000 cfs about once every 4 years and exceed 50,000 cfs about once every 5 years (Bureau of Reclamation 2008).

In 2008, Reclamation began implementing floodplain and spawning habitat restoration projects in the American River to assist in meeting the requirements of the 1992 CVPIA, Section 3406 (b)(13). Spawning and rearing habitat enhancement projects have occurred each year since 2008 and they are planned to continue.

Fish Passage

Historically, more than 125 miles of riverine habitat was available for anadromous salmonids in the American River watershed (Yoshiyama et al. 1996). Access to the upper reaches of the river has been blocked by a series of impassable dams, which preceded the construction of Folsom Dam, including Old Folsom Dam, first constructed in the American River between 1895 and 1939.

Reclamation operates a fish diversion weir approximately 0.25 mile downstream of Nimbus Dam, which functions to divert adult steelhead and Chinook salmon into Nimbus Fish Hatchery. The weir is installed annually during September prior to the arrival of fall-run Chinook salmon and steelhead and is removed at the conclusion of fall-run Chinook salmon immigration in early January (Bureau of Reclamation and California Department of Fish and Game 2011). Some steelhead may be trapped prior to weir removal, but they are returned to the river. A new fish passageway is being implemented in the Nimbus Dam stilling basin, commonly referred to as

Nimbus Shoals. The passageway will replace the existing fish diversion weir with a new flume and fish ladder that will connect to the existing fish ladder near Nimbus Salmon and Steelhead Hatchery and American River Trout Hatchery (collectively, Nimbus Fish Hatchery).

Hatcheries

The Nimbus Fish Hatchery is located immediately downstream from Nimbus Dam. Facilities for the hatchery include a fish weir, fish ladder, gathering and handling tanks, hatchery-specific buildings, and rearing ponds. Nimbus Fish Hatchery was constructed primarily to mitigate the loss of spawning habitat for Chinook salmon and Central Valley steelhead that were blocked by the construction of Nimbus Dam (Bureau of Reclamation and California Department of Fish and Game 2011). It does not address lost habitat upstream from Folsom Dam (California Hatchery Scientific Review Group 2012). Under contract with Reclamation, CDFW operates the hatchery (Lee and Chilton 2007). Operations include trapping, artificially spawning, rearing, and releasing steelhead and fall- /late fall–run Chinook salmon. The Nimbus Fish Hatchery Winter-Run Steelhead Program is an isolated-harvest program, meaning it does not include natural-origin steelhead in the broodstock, and is designed and implemented to artificially spawn the adipose fin-clipped adult steelhead that seasonally enter the trapping facilities (California Hatchery Scientific Review Group 2012). These fin-clipped fish are not considered part of the Central Valley steelhead DPS by NMFS. The Nimbus Fish Hatchery Winter-Run Steelhead Program propagates fish for recreational fishing opportunities and harvest (California Hatchery Scientific Review Group 2012).

Historically, steelhead were trapped at Nimbus Fish Hatchery as early as the first week of October. Beginning in 2000, the ladder has been opened in early November. Trapping of steelhead has continued to occur as late as the second week of March. Presently, winter-run Steelhead are trapped at Nimbus Fish Hatchery, and artificially spawned adults are marked with an adipose fin clip (California Hatchery Scientific Review Group 2012). Unmarked steelhead adults are not retained at Nimbus Fish Hatchery for use in the annual broodstock and are released back to the river (California Hatchery Scientific Review Group 2012). In addition, marked or unmarked *O. mykiss* that are less than 16 inches long may be resident hatchery-origin trout and are returned to the river (California Hatchery Scientific Review Group 2012).

The hatchery has raised and released an average of approximately 422,000 yearling steelhead per year since brood year 1999 (California Hatchery Scientific Review Group 2012). Since 1998, all steelhead/rainbow trout produced in Nimbus Fish Hatchery have been marked with an adipose fin clip to aid in subsequent identification of hatchery-origin fish.

Juvenile steelhead yearlings are not held past March 30 due to higher hatchery water temperatures and to encourage outmigration during spring. If releases occur during periods of low flows in the Sacramento River and possibly the American River, some released fish migrate back to Nimbus Fish Hatchery and may take up residency rather than migrating downstream (Lee and Chilton 2007). Additionally, juvenile fish are released in February and early March to coincide with State Water Board D-1641 closures of the DCC gates from February 1 through May 20, which reduces straying into the Delta. Reclamation determines the exact timing and duration of the gate closures each year after discussion with USFWS, CDFW, and NMFS.

Reclamation is implementing a genetic screening study of Nimbus Fish Hatchery steelhead. Reclamation, in contract with NMFS, is conducting a parental-based tagging study of American River steelhead and continuing a study to determine a more genetically appropriate stock.

CDFW releases all hatchery-produced steelhead juveniles into the American River at boat ramps or at the confluence of the Sacramento and American Rivers and releases all unclipped steelhead adults returning to Nimbus Fish Hatchery into the lower American River via the river return tube that is just downstream of the fish ladder. In accordance with California law, the current protocol of Nimbus Fish Hatchery is to destroy all surplus eggs to prevent inter-basin transfer of eggs or juveniles to other hatcheries or waters.

The goal of the Nimbus Fish Hatchery Integrated Fall-/Late Fall–-Run Chinook Salmon Program is to release 4 million smolts. Each fall, Nimbus Hatchery staff collect approximately 10,000 adult fall-run Chinook salmon, with an annual goal of harvesting 8,000,000 eggs and releasing the 4,000,000 smolts. All adult fall-run Chinook salmon collected at the hatchery are euthanized, and no trapped salmon are returned to the American River (Bureau of Reclamation 2008).

Disease

CDFW has observed the occurrence of a bacterial-caused inflammation of the anal vent (commonly referred to as “rosy anus”) of steelhead in the lower American River. The inflammation has been linked to relatively warm water temperatures (Water Forum 2005). Disease transmission may be exacerbated by crowding under conditions when water flows are reduced (Water Forum 2005).

Predation

Several predatory fish are variably present in the lower American River, including black basses (largemouth bass, smallmouth bass, and spotted bass) and striped bass, that exert predatory pressure on salmonids and other native fish species. Two factors—reduced cold-water storage in Folsom Lake and using Folsom Lake to meet Delta water quality objectives and demands—influence habitat conditions in the lower American River for warm-water predator species that feed on juvenile salmonids and potentially alter predation pressure (Water Forum 2005). Additionally, isolation of redds in side channels resulting from fluctuations in Folsom Lake releases may increase predation of emergent fry (Water Forum 2005).

11.2.6. San Pablo and San Francisco Bays

11.2.6.1. Description of San Pablo and San Francisco Bays

Hydrologically, the Bay may be divided into two broad subdivisions with differing ecological characteristics: a southern reach consisting of South San Francisco Bay; and a northern reach composed of Central San Francisco, San Pablo, and Suisun Bays (The Bay Institute 1998; CALFED Bay-Delta Program 2000a). The southern reach receives little freshwater discharge, leading to high salinity and poor circulation (high residence time). It also has more extreme tides. The northern reach, which directly receives Delta outflow, is characterized by less extreme tides and a pronounced horizontal salinity gradient, ranging from near full marine conditions in Central San Francisco Bay to near freshwater conditions in Suisun Bay. Central San Francisco Bay and Suisun Bay contain large islands, features not present in San Pablo Bay and South Bay

(The Bay Institute 1998; CALFED Bay-Delta Program 2000a). All of the bays except Central San Francisco Bay include extensive marshlands. Suisun Bay is not treated in this section because it was covered with the Delta in a previous section.

Northern Reach—Central San Francisco and San Pablo Bays

Ecological factors having the greatest influence on fish of Central San Francisco Bay and San Pablo Bay include freshwater inflow from rivers, wetlands, riparian vegetation, and aquatic habitat diversity. Habitats in these bays are tidal perennial aquatic habitat, tidal saline emergent wetland, seasonal wetland, perennial grassland, agricultural land, and riparian habitat. These habitats support a variety of native marine, estuarine, freshwater, and anadromous fish (CALFED Bay-Delta Program 2000a). San Francisco Bay is designated as a coastal estuary Habitat Area of Particular Concern (HAPC) and eelgrass (*Zostera marina*) is designated as seagrass HAPC for Pacific groundfish species. Fish species that currently depend on tidal marshes and adjoining sloughs, mudflats, and embayments include delta smelt, longfin smelt, Chinook salmon, green sturgeon, white sturgeon, Pacific herring (*Clupea pallasii*), starry flounder (*Platichthys stellatus*), Sacramento splittail, American shad, and striped bass (The Bay Institute 1998; CALFED Bay-Delta Program 2000a; Baxter et al. 2008). Other fish commonly found in Central Bay include northern anchovy (*Engraulis mordax*), Pacific halibut (*Hippoglossus stenolepis*), American shad, bay goby (*Lepidogobius lepidus*), white croaker (*Genyonemus lineatus*), Pacific staghorn sculpin (*Leptocottus armatus*), and marine surfperches. English sole (*Parophrys vetulus*), shiner surfperch (*Cymatogaster aggregata*), jacksmelt (*Atherinopsis californiensis*), topsmelt (*Atherinops affinis*), diamond turbot (*Hypsopsetta guttulata*), and speckled sand dab (*Citharichthys stigmaeus*) are common in shallow waters around Central Bay. The leopard shark (*Triakis semifasciata*), sevengill shark (*Notorynchus cepedianus*), and the brown smoothhound (*Mustelus henlei*) are abundant in the intertidal mudflats of the Central Bay. The sand substrate and rock outcrops in the Central Bay support recreational fish such as the halibut, striped bass, rockfish (*Sebastes* spp.), and lingcod (*Ophiodon elongatus*).

11.2.6.2. Habitat Conditions and Environmental Stressors in San Pablo and San Francisco Bay Area

Environmental stressors for fish populations in San Francisco and San Pablo Bays include water and sediment quality, exposure to toxic substances, reduction in Delta outflows, legal and illegal harvest, food availability, reduction in seasonally-inundated wetlands, wave and wake erosion, introduced nonnative plant and animal species, competition for food resources with non-native fish and macroinvertebrates (e.g., filter feeding by the non-native mollusks) (CALFED Bay-Delta Program 2000a; Armor et al. 2005; Baxter et al. 2008).

11.2.7. Local Drainages

Both Stone Corral and Funks Creeks have small watersheds originating in the eastside foothills of the Coast Range at elevations of 700 to 850 feet and flow intermittently, mostly in winter and early spring months. From their origins, both creeks flow through low foothills, across Antelope Valley (the location of the Sites Reservoir footprint), through a series of ridges, and onto the Sacramento Valley floor (Figure 11-2). For much of their course on the valley floor, they are confined to narrow channels between berms along agricultural fields and road prisms[3]. While the stream channels of these creeks are not actively managed, their straight channels and angular turns around agricultural fields and along roads indicate that they were modified from their natural channels at some point in the past. In the upper parts of the watersheds just below the dam locations, these streams are largely devoid of riparian cover due to cattle grazing activity (California Department of Water Resources and Bureau of Reclamation 2008). In the lower reaches where the streams run through and around agricultural fields, riparian habitat is sparse and consists mostly of low shrubs, grasses, and occasional oak and cottonwood trees.

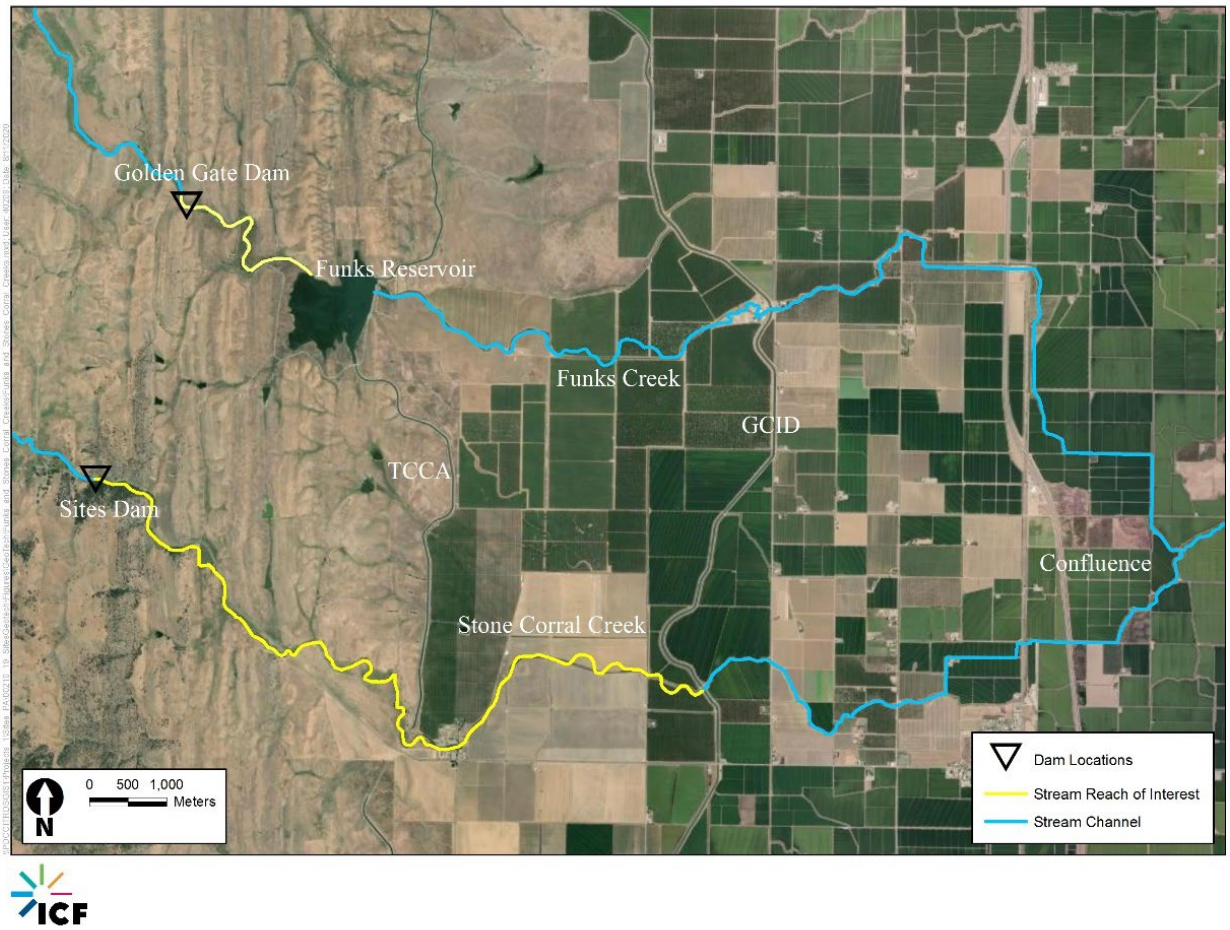


Figure 11-2. Stone Corral Creek and Funks Creek Drainages

11.2.7.1. Stone Corral Creek

Stone Corral Creek has a drainage area of 32.8 square miles. From the location of the Sites Dam, Stone Corral Creek meanders through a shallow canyon onto the valley floor, where it flows through an incised channel across grazing lands. At 4.6 miles from the Sites Dam location, Stone Corral Creek crosses over a siphon in the TC Canal and begins to travel through agricultural lands. About 3 miles below the TC Canal siphon, Stone Corral Creek crosses the GCID Main Canal siphon. Although most of the water in the GCID Main Canal passes under Stone Corral

[3] Characterization of stream channels is based on desktop review of streams using Google Earth.

Creek in the siphon, GCID releases water from the canal to Stone Corral Creek for delivery to agricultural fields downstream. About 5.5 miles below the GCID Main Canal, Stone Corral Creek merges with Funks Creek and then flows an additional 5.7 miles to the CBD.

11.2.7.2. Funks Creek

Funks Creek, a tributary to Stone Corral Creek, has a drainage area of 43 square miles. From the location of Golden Gate Dam, Funks Creek meanders through a series of low ridges and grazing lands for about 1.8 miles to Funks Reservoir. Funks Reservoir is a re-regulating reservoir on the TC Canal and is created by a low dam on Funks Creek. Funks Dam is operated by TCCA mostly for flood control purposes. The Funks Dam gates are opened during large storm events to pass flood waters through the reservoir and downstream to avoid compromising the TC Canal and its operations. California Fish and Game Code Section 5937 requirements maintain sufficient flows in Funks Creek below Funks Reservoir to keep in good condition fish that reside below the dam.

Below Funks Dam, Funks Creek travels 3.9 miles through agricultural fields in a combination of natural and straightened channels to where it crosses the GCID Main Canal. While the GCID Main Canal passes under Funks Creek in a siphon, GCID releases water from the canal to Funks Creek and uses the downstream portions of the creek as part of its conveyance system to deliver water to agricultural fields. Approximately 2 miles northeast of Maxwell and 1 mile east of Interstate 5, Funks Creek flows into Stone Corral Creek.

11.2.7.3. Water Quality

Stone Corral Creek is listed under Section 303(d) as an impaired waterbody for low DO levels (State Water Resources Control Board 2018). The creek was originally listed in 2010 and is scheduled to have a TMDL plan by 2027. This designation is based on samples collected at a sampling site located where Stone Corral Creek crosses 4-mile Road. This location is downstream of the confluence between Funks and Stone Corral Creeks, at the western edge of the Delevan National Wildlife Refuge. The source of the oxygen depletion is listed as unknown (State Water Resources Control Board 2018) but, given the amount of algae visible in Google Earth photos, nutrient loading from the cattle grazing lands and agricultural fields is a likely source in both watersheds. During fish surveys in 1998 and 1999, California Department of Fish and Game (CDFG; now CDFW) noted that water quality was poor and high in dissolved minerals. They reported that the total dissolved solids in the water were so high that it precluded electrofishing as a means of sampling (California Department of Fish and Game 2003).

State Water Board (2018) did not report on water quality in Funks Creek but given similar size, geology, and land use between the two watersheds, the water quality in Funks Creek is likely comparable to Stone Corral Creek.

11.2.7.4. Hydrology

Both streams originate at low elevations below the snow line of the Coast Range and consequently do not receive cold snowmelt water. Rather, they respond rapidly to significant rainfall events and flash flooding and substantial overland flow has been observed.

The U.S. Geological Survey (USGS) collected 25 years of discharge measurements in Stone Corral Creek near the town of Sites from 1958 through 1985. During that time, there were 3 years of no measurable flow: 1972, 1976, and 1977. Yates (1989) estimated the recurrence interval of a winter without flow at 12 to 14 years. The maximum mean daily flow of 2,230 cfs occurred on December 24, 1983. The instantaneous peak flow was 5,700 cfs on January 26, 1983. The 100-year discharge was established in a 1987 Colusa Basin flood flow frequency analysis as 7,870 cfs (California Department of Water Resources and Bureau of Reclamation 2008).

There is no comparable data set for Funks Creek. However, given the comparable size, geology, and topography of the two watersheds and their proximity to each other upstream of their confluence, Stone Corral Creek hydrology is likely representative of Funks Creek hydrology in terms of amount and seasonality of flow. The daily mean hydrology for Stone Corral Creek is included in Table 11-3. It shows the variability of flow over the period of record differs considerably from a static flow of 10 cfs.

Table 11-3. Stone Corral Creek Daily and Monthly Flows Near Sites, USGS 11390672

	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep
Daily Flows (cfs) for Period of Record												
Min	0	0	0	0	0	0	0	0	0	0	0	0
Max	0	74	2,230	1,910	2,150	1,980	619	45	9	1	0	0
Avg	0	1	11	32	39	21	8	1	0	0	0	0
Monthly Flows (AF) for Period of Record												
Min	0	0	0	0	0	0	0	0	0	0	0	0
Max	0	427	11,432	8,825	11,137	15,227	4,451	740	146	19	0	0
Avg	0	37	660	1,946	2,190	1,300	484	83	13	1	0	0

Source: Sites Project Authority and Bureau of Reclamation 2017.
Period of Record 4/1/1958–9/30/1964 and 10/1/1965–9/30/1985
Drainage Area = 38.2 Square Miles

11.3 Methods of Analysis

This section describes the various methods of analyses and results used to evaluate construction, operation, and maintenance activities associated with the Project. In general, qualitative analyses were performed based on characteristics and presence of species and the types of construction or maintenance activities to evaluate construction or maintenance impacts. Quantitative analyses using multiple different models and other lines of evidence were used for evaluating operational impacts.

11.3.1. Construction

The assessment of potential impacts on fish resources consisted of a primarily qualitative evaluation of construction effects of the Project facilities. The impact assessment addressed two primary impact types: (1) temporary and localized impacts associated with construction of the Project facilities; and (2) permanent impacts associated with construction of Project facilities and filling of Sites Reservoir.

The assessment of impacts on fish resources was based on the CEQA significance thresholds described below, and professional judgment that considers current state, federal, and local regulations and policies, currently available peer-reviewed scientific literature, field survey observations, and knowledge of species' distribution, life history, and habitat requirements. Key considerations in the evaluation of impacts included the magnitude of environmental effects (e.g., spatial extent and duration of habitat modification), sensitivity of the species to these effects, and potential exposure or extent to which the population would be affected.

The impact analysis includes the following key conclusions:

- Installation of the two additional TC Canal diversion pumps at the RBPP would not affect special-status fish or their habitat because construction would occur in the existing facility footprint which does not support habitat for special-status fish. No impacts are anticipated and this area is not considered further in this analysis.
- Construction performed at various locations along the GCID Main Canal associated with upgrades to support the operation of Sites Reservoir would not affect special-status fish or their habitat because construction would occur in the existing facility footprint which does not support habitat for special-status fish. No impacts are anticipated and this area is not considered further in this analysis.
- Construction impacts on special-status fish in the Sacramento River, including listed species and associated designated critical habitat, would occur only under Alternative 2 because Alternative 2 has in-water work in the Sacramento River associated with the construction of the Sacramento River discharge. Piles for work platforms to construct the Sacramento River discharge are assumed to be within 200 feet of the Sacramento River channel and could produce underwater noise levels during impact pile driving that exceed the interim criteria for injury to fish.
- The reservoir would replace existing intermittent and ephemeral streams with open water and Alternative 1 or 3 would permanently flood slightly more stream habitat than Alternative 2.

For the purposes of evaluating potential pile driving effects representative project information was used from similar construction in-water construction activities. The evaluation of potential injury to fish from pile driving sounds used data from another coffer dam project on the Sacramento River involving similar pile driving methods (National Marine Fisheries Service 2017:110–116). These data provided estimates of the distances from the pile that sound attenuates (decreases) to the peak or cumulative criteria (i.e., the point at which it is believed that sound no longer causes injury to fish) and identified the anticipated pile driving sound levels. The predicted sound levels determined the potential for injury to fish from exposure to pile driving sounds from the coffer dam installation. Predicted sound levels and distances to the potential for injury to fish are estimated using a spreadsheet model developed by NMFS that is based on measured sound levels for similar piles driven in similar conditions and reported in California Department of Transportation (2015). The NMFS model uses input data such as the size and type of pile to be driven, the number of strikes needed to drive each pile, the number of piles to be driven in a day, and pile location (i.e., in-water or on land). Larger piles, greater number of strikes per pile and piles driven in a day, and piles driven in water generally result in higher sound levels and therefore greater distances from the pile that sound attenuates to the peak or cumulative criteria. Because these pile driving details are not known at this time for

Alternative 2 with respect to the coffer dam, but also other activities that may require pile driving under Alternatives 1 or 3, the current analysis used the results from the other coffer dam project on the Sacramento River to predict the sounds levels that would be likely to occur from coffer dam installation using piles driven in water. This approach is reasonable because similar materials and methods used for the other coffer dam on the Sacramento River are expected to be used to install the coffer dams on the CBD and Sacramento River. The analysis also used these data for evaluating acoustic impacts associated with installing the piles for the work platforms during construction of the discharge and energy dissipation structure for the Sacramento River discharge under Alternative 2, even though piles would be driven only on land and therefore expected to result is lower sound levels than in-water pile driving (all other things being equal). This approach is reasonable because the predicted noise levels associated with impact driving of sheet piles for the coffer dam are expected to be greater than levels predicted for impact driving of piles for the temporary work platforms on the river's shoreline. Therefore, it is assumed that the predicted sound levels for driving the sheet piles for the coffer dam would represent a worst-case scenario for driving piles for the work platforms that would be driven on the river's shoreline for Alternative 2.

The following BMPs, which are described in Appendix 2D, *Best Management Practices, Management Plans, and Technical Studies*, are incorporated into the analysis of potential construction impacts on fish resources.

- BMP-10, Salvage, Stockpiling, and Replacing Topsoil and Preparation of a Topsoil Storage and Handling Plan, requires evaluation of topsoil for salvaging suitability and storage and handling plans when topsoil cannot be used without stockpiling to protect adjacent waterways from sediment, preserve soil properties, and facilitate site rehabilitation.
- BMP-12, Develop and Implement Stormwater Pollution Prevention Plan(s) (SWPPP) and Gain Coverage under Stormwater Construction General Permit (Stormwater and Non-stormwater), requires development and use of erosion control measures, sediment control measures, construction materials management measures, waste management measures, non-stormwater control measures, and post-construction stormwater management measures to prevent the transport and delivery of sediment into adjacent waterways, which would prevent alteration of fish physiology, behavior, habitat conditions, and direct injury or mortality to fish.
- BMP-13, Development and Implementation of Spill Prevention and Hazardous Materials Management/Accidental Spill Prevention, Containment, and Countermeasure Plans (SPCCPs) and Response Measures, requires site-specific plans with measures to minimize effects from spills of hazardous or petroleum substances during construction and operation/maintenance by implementing measures such as physically-distancing equipment from waterways, maintaining spill prevention kits at facilities where hazardous materials may be used, providing the equipment and materials necessary for cleanup of accidental onsite spills, and storing hazardous materials in double containment to avoid and reduce localized water quality degradation and prevent direct injury or mortality, habitat degradation, and reduced prey availability.

- BMP-33, Implementation of a Worker Environmental Awareness Program (WEAP), requires training of all construction crews and contractors on protection and avoidance of biological, cultural, archaeological, paleontological, and other sensitive resources to make personnel aware of these resources to protect fish from direct physical injury or mortality.
- BMP-23, Development and Implementation of an Underwater Construction Noise Control, Abatement, and Monitoring Plan, requires measures to avoid and minimize the effects of underwater construction noise on fish, particularly underwater noise effects associated with impact pile driving activities, by restricting the seasonal and daily timing of pile driving, conducting hydroacoustic monitoring during impact pile driving, and monitoring the in-water work area for signs of distressed or injured fish so that disturbance and injury and/or mortality of special-status fish in Funks and Stone Corral Creeks do not occur.
- BMP-30, Development and Implementation of Hazardous Materials Management Plans, requires hazardous materials management plans (HMMPs) before beginning construction that considers hazardous materials present on site and known historic site contamination, governs the storage, use, or transfer of hazardous materials, outlines emergency spill containment and cleanup procedures, and outlines procedures for handling, hauling, and disposing of hazardous waste generated at work sites to avoid and minimize the potential for hazardous materials and waste to enter aquatic habitat.
- BMP-34, Development and Implementation of Fish Rescue and Salvage Plans for Funks Reservoir, Stone Corral Creek, Funks Creek, and CBD for Alternatives 1, 2, and 3; for Sacramento River for Alternative 2, requires plan to outline fish herding and/or fish rescue operations where dewatering and resulting isolation of fish may occur. The plans will include detailed fish collection, holding, handling, and release procedures to avoid and minimize effects on fish during construction activities that may require dewatering.
- BMP-35, Development and Implementation of Construction Best Management Practices and Monitoring for Fish, Wildlife, and Plant Species Habitats, and Natural Communities, requires a construction monitoring plan for sensitive biological resources and in-water construction activities, use of exclusion fencing around sensitive biological resources, and restricting in-water construction activities on the Sacramento River to September 1 through October 15 and in non-anadromous waters from June 1 through October 15 to protect fish. This BMP would limit direct impacts on sensitive natural communities because they would train construction workers on the importance of preserving sensitive natural communities outside of the construction footprint and require fencing of sensitive natural communities where avoidance is feasible. The BMP would also restrict off-road driving in the construction area, where avoided sensitive natural communities could be damaged or destroyed.

11.3.2. Operations

The assessment of potential impacts to fish resources consisted of a qualitative and quantitative evaluation of Project operations on fish and aquatic resources (refer to Impacts FISH-2 through FISH-19). The assessment of impacts to fish and aquatic resources was based on the CEQA significance thresholds, and professional judgment that considers current state, federal, and local regulations and policies, currently available peer-reviewed scientific literature, field survey observations, and knowledge of species' distribution, life history, and habitat requirements. Key considerations in the evaluation of impacts included the magnitude of environmental effects (e.g., spatial extent and duration of habitat modification), sensitivity of the species to these effects, and potential exposure or extent to which the population would be affected. The quantitative operational impact assessment relies on modeled hydrologic changes in SWP and CVP operations that would occur as a result of Project operations. The monthly flow output of the CALSIM II model was used to assess changes in reservoir WSE, storage, and instream flows associated with implementation of Alternatives 1, 2, and 3. Hydrologic simulation results of end-of-month reservoir storage and elevations provided a quantitative basis to assess the potential impacts of operations on fish species, relative to the bases of comparison, for the period of simulation extending from Water Year 1922 through 2003 (82-year simulation period).

The CALSIM II model uses a monthly time-step. Where feasible and when modelers indicate using them is appropriate, daily model outputs are utilized. However, CALSIM II is a monthly model developed for planning level analyses. The model is run for an 82-year historical hydrologic period, at a projected level of hydrology and demands; and under an assumed framework of regulations. Therefore the 82-year simulation does not provide information about historical conditions, but it does provide information about variability of conditions that would occur at the assumed level of hydrology and demand with the assumed operations, under the same historical hydrologic sequence. Because it is not a physically based model, CALSIM II is not calibrated and cannot be used in a predictive manner. CALSIM II is intended to be used in a comparative manner.

While there are certain components in the model that are downscaled to a daily time-step (simulated or approximated hydrology), such as an air-temperature-based trigger for a fisheries action, the results of those daily conditions are always averaged to a monthly time-step. For example, a certain number of days with and without the action is calculated and the monthly result is calculated using a day-weighted average based on the total number of days in that month. Operational decisions based on those components are again made on a monthly basis. Any reporting or use of sub-monthly results from CALSIM II should include disaggregation methods that are appropriate for the given application, report, or subsequent model.

CALSIM represents a reasonable representation of the range of potential flows affecting fish species, The results of the CALSIM model were used in conjunction with multiple lines of evidence for various impact analyses. The CALSIM II monthly flow output also served as input to many of the other models used to analyze potential impacts to aquatic resources. Refer to Table 11-4 for a summary of tools used for the impact assessment and where they are used in each impact. Information regarding modeling results located throughout Section 11.4, *Impact Analysis and Mitigation Measures*, and detailed results can be found in related technical appendices.

For example, Weighted Usable Area (WUA) was utilized to evaluate potential changes in upstream salmonid habitat conditions for spring-run Chinook salmon. The modeling results and subsequent discussions are located in the Far-Field Effects within Impact FISH-3: Operations Effects on Spring-Run Chinook Salmon. Additional results and discussions related to WUA can also be found in Appendix 11K, *Weighted Usable Area Analysis*. As discussed in Appendix 11K, the availability of rearing habitat was estimated using WUA curves obtained from the literature. Relevant literature is cited and discussed in Appendix 11K. WUA is an index of the surface area of physical habitat available, weighted by the suitability of that habitat. WUA curves are normally developed as part of instream flow incremental methodology studies. Rearing habitat WUA was estimated only for the Sacramento River because no flow versus rearing WUA curves for the Feather or American River were available. The available flow versus rearing WUA information for these rivers is old, limited, and potentially unreliable (Appendix 11K).

Detailed discussion of the specific methodologies and indicators used to evaluate potential impacts due to changes in SWP and CVP operations as a result of Project implementation is provided in Appendix 11B, *Upstream Fisheries Impact Assessment Quantitative Methods*.

Modeling results for all alternatives are presented in tables and graphs throughout this chapter. As discussed in Chapter 3, *Environmental Analysis*, the term *No Project Alternative* is primarily used in this document to represent both the CEQA No Project Alternative and NEPA No Action Alternative. The term *NAA*, which is identical to the No Project Alternative, is used throughout this chapter in the presentation of modeled results and represents no material difference from the No Project Alternative. This chapter also provides modeling results for Alternative 1A and Alternative 1B, which are both considered under Alternative 1, as described in Chapter 2, *Project Description and Alternatives*. This information is provided for the purposes of the operational impact analysis and represents two different operation options under Alternative 1 as a result of the different participation for Reclamation.

In addition, as noted in Chapter 2, all Project components are the same between Alternatives 1 and 3. Therefore, in some instances the impact analyses for Alternatives 1 and 3 are combined under subheadings. If the impact mechanisms and types of impacts are similar across Alternatives 1, 2, and 3, the impact analyses may be aggregated to reduce redundancy and provide ease of comparisons between alternatives. All alternatives have been co-equally analyzed as required by NEPA, even if alternatives are combined under subheadings.

Table 11-4. Methods for Analysis of Potential Effects on Fish and Aquatic Resources

Method	Brief Description	Requires Modeled Output	Impact Key
Anderson-Martin models for egg Mortality (Martin et al. 2017; Anderson 2018)	Estimates water temperature-related mortality of winter-run Chinook salmon eggs to fry.	Yes; HEC5Q	FISH-2
Bypass and Side-Channel Inundated Habitat Area analysis	Examines the surface area of suitable inundated floodplain and side-channel habitat that would be available for salmonids and Sacramento splittail.	Yes; CALSIM	FISH-2 through FISH-5; FISH-11
California Water Resources Simulation Model II (CALSIM II)	A hydrological planning tool for the SWP and CVP system, providing monthly average outputs including channel flows and reservoir storage, for example	No	FISH-2 through FISH-18
Delta Outflow Year-Class Strength Regression Analysis	Evaluates white sturgeon year-class index as a function of April–May or March–July Delta outflow. Also used as a surrogate for green sturgeon.	Yes; CALSIM	FISH-6 and FISH-7
Delta Simulation Model II (DSM2)	1D mathematical model for simulating hydrodynamics and water quality.	Yes; CALSIM	FISH-8
DSM2-QUAL	Water quality in the Delta, primarily concerned with assessment of salinity.	Yes; CALSIM	FISH-13
Eurytemora affinis–X2 analysis	Estimates *E. affinis* (delta and longfin smelt prey) density as a function of spring X2.	Yes; CALSIM	FISH-8 and FISH-9
Flow threshold survival analysis	Estimates outmigration survival based on flow thresholds by Michel et al. 2021 in Sacramento River at Wilkins Slough.	Yes; Sites Reservoir Daily Divertible & Storable Flow Tool	FISH-2 through FISH-4
HEC-RAS model	Hydrologic modeling approach for water flow, sediment transport, and water temperature/water quality in rivers and channels.	No	FISH-2 through FISH-5, FISH-11
IOS (Interactive Object-Oriented Simulation) life cycle model	Stochastic life cycle model for winter-run in the Sacramento River.	Combines data from field studies, long-term monitoring programs and laboratory studies in a simulation framework, based on CALSIM, HEC5Q, and DSM2	FISH-2
Juvenile Stranding Analysis	Estimation of salmonid juvenile stranding based on estimated rearing habitat dewatering between an initial flow and the subsequent minimal flow during the rearing period.	Yes; USRDOM	FISH-2 through FISH-5
Longfin smelt abundance analysis	An analysis reproducing methods by Nobriga and Rosenfield 2016 to include the influence of adult stock and density dependence on Longfin Smelt population dynamics.	Yes; CALSIM	FISH-9
OBAN (Oncorhynchus Bayesian Analysis) life cycle model	Statistical modeling approach to evaluating scenarios effects on winter-run Chinook salmon.	Yes; CALSIM, HEC5Q	FISH-2
PHABSIM	Hydraulic model that estimates, at a given flow, the availability of suitable habitat in a river reach, based on predetermined habitat suitability criteria (HSI) at multiple locations in the river. Results typically used in weighted usable area analyses.	No; results from literature	FISH-2 through FISH-5
Reclamation Temperature Model	Simulates monthly mean vertical temperature profiles and release temperatures from reservoirs and in rivers	Yes; CALSIM	FISH-2
Redd Scour	Estimates frequency of flows high enough to scour or entomb salmonid redds.	Yes; USRDOM	FISH-2 through FISH-5
Redd dewatering analysis	Uses the maximum reduction in flow from the initial flow, or *spawning flow*, that occurs over the duration of an egg cohort plus estimates of redd distributions based on spawning weighted usable area analysis, to estimate percent of redds dewatered.	Yes; USRDOM	FISH-2 through FISH-5
River Temperature Modeling - HEC5Q	Simulates flood control, water quality analysis, etc., that has the capabilities to accept user-specified water quantity and quality needs system-wide. Used to provide water temperatures for the Sacramento and American. Model outputs can be used to evaluate potential (temp-related) changes in upstream habitat conditions.	Yes; CALSIM	FISH-2, FISH-3
SALMOD	Simulates Sacramento River populations of winter-run, spring-run, fall-run, and late fall–run Chinook salmon to assess potential flow- and temperature-related effects on early life stages.	Yes; CALSIM, HEC5Q	FISH-2, FISH-3, FISH-4
Salvage-Density Analysis	Differences in south Delta exports weighted by historical density of fish in salvage.	Yes; CALSIM	FISH-2 through FISH-7, FISH-10 through FISH-16
STARS (Survival, Travel time, And Routing Simulation) model, adapted by Perry	The spreadsheet model was provided by Perry et al. 2018 and reproduces the mean response of the STARS model to estimate through-Delta survival as a function of daily Sacramento River flow at Freeport as well as Delta Cross Channel gate position.	Yes; DSM2-HYDRO	FISH-2, FISH-3, FISH-4

Method	Brief Description	Requires Modeled Output	Impact Key
Threadfin Shad South Delta Entrainment Risk Analysis	Inference regarding potential entrainment risk to threadfin shad, based on particle tracking modeling results from Kimmerer and Nobriga (2008).	Yes; CALSIM	FISH-15
Tidal habitat restoration mitigation calculations for longfin smelt	Combined statistical relationships between export to inflow ratio and proportion of particles entrained from various particle injection locations, weighted by habitat area represented by the injection locations.	Yes; CALSIM	FISH-9
Upper Sacramento River Daily Operations Model (USRDOM)	Models the flows and related operations in the Upper Sacramento on a daily time-scale, downscaled from monthly CALSIM II to a daily scale.	Yes; CALSIM	FISH-2, FISH-3, FISH-4
Upstream temperature mean value and exceedance plot analysis	Evaluates mean water temperatures by month and water year type and exceedance plots that overlap temporally and spatially with each species and life stage.	Yes; HEC5Q, Reclamation Temperature Model	FISH-2, FISH-3
Upstream Water Temperature Index Analysis	Assesses magnitude and frequency of exceeding water temperature index values or being outside a water temperature index range from the scientific literature for each species and life stage within the Sacramento River, Feather River, and American River.	Yes; HEC5Q, Reclamation Temperature Model	FISH-2, FISH-3
Weighted Usable Area (WUA)	Weighted usable area is the sum of water surface area within a study site, weighted by multiplying area by habitat suitability variables. Evaluates potential changes in upstream salmonid habitat conditions.	Yes; CALSIM	FISH-2, FISH-3, FISH-4, FISH-5, FISH-6
Winter-Run Chinook Salmon Life Cycle Model (WRLCM)	Estimates abundance of winter-run Chinook salmon at each geographic region and time-step for all stages of their lifecycle.	Yes, CALSIM	FISH-2
X2-abundance regression	Estimates indices of abundance or survival as a function of X2 for various seasonal periods.	Yes; CALSIM	FISH-12 through FISH-14; FISH-17
X2-Longfin smelt abundance index analysis	Estimates of longfin smelt indices of abundance as a function of January–June X2.	Yes; CALSIM	FISH-9

Upstream fisheries impact assessments were done utilizing quantitative methods related to temperature, spawning habitat WUA, redd dewatering, redd scour/entombment, the Salmonid Population Modeling (SALMOD) model, rearing flows, rearing habitat WUA, juvenile stranding, and upstream migration of salmon and sturgeon adults. Other fisheries impact assessment quantitative methods are discussed in various other appendices. Additional information on upstream methodology is located in Appendix 11B.

Quantitative methods and supplementary results used in the operational impact analyses of delta smelt and longfin smelt include: the *Eurytemora affinis*–X2 analysis for smelt prey, the Delta outflow–longfin smelt abundance analysis (based on Nobriga and Rosenfield 2016), the X2–longfin smelt abundance index analysis, and tidal habitat restoration mitigation calculations for longfin smelt. Additional information is located in Appendix 11F, *Smelt Analysis*.

For potential operational water temperature effects on fish in waterways upstream of the Delta, for each fish species and life stage, the analysis evaluated the frequency (and magnitude for salmonids and green sturgeon) of occurence of daily or monthly water temperature model outputs above a specific water temperature index value or outside a specific water temperature index range during different times of year and in locations that overlap with the fish presence. Additional information and results are located in Appendix 11D, *Fisheries Water Temperature Assessment*.

SALMOD was utilized to simulate potential operational effect to Sacramento River populations of winter-run, spring-run, fall-run, and late fall–run Chinook salmon to assess potential flow- and temperature-related effects on early life stages. Interpretation of SALMOD outputs are presented in Appendix 11H, *Salmonid Population Modeling (SALMOD)*. IOS (Interactive Object-Oriented Simulation) and OBAN (Oncorhynchus Bayesian Analysis) winter-run Chinook salmon life cycle models were also utlized to determine potential operational impacts. Methods and results are discussed in Appendix 11I, *Winter-Run Chinook Salmon Life Cycle Modeling*.

Appendix 11J, *Through-Delta Survival and Delta Rearing Habitat of Juvenile Chinook Salmon*, describes the through-Delta survival analysis of juvenile salmonids based on the model of Perry et al. (2018). The analysis was conducted through a spreadsheet implementation of the model that was provided by Perry (pers. comm.) which reproduces the mean response of the STARS (Survival, Travel time, and Routing Simulation) model estimating through-Delta survival, travel time, and routing of juvenile Chinook salmon as a function of Sacramento River flow at Freeport (Perry et al. 2019).

As described in Appendix 11K, WUA analysis provides estimates of the amount of suitable spawning and rearing habitat of fishes available in rivers and streams at various levels of flow. WUA is computed as the surface area of physical habitat available weighted by its suitability. Habitat suitability is determined from field studies of the distributions of redds or rearing juveniles with respect to flow velocities, depths, and substrate or cover characteristics in the river (Bovee et al. 1998). These data are used in model simulations (PHABSIM) that estimate the availability of suitable habitat in a portion of the river at a given flow. WUA curves showing suitable habitat availability versus flow are generated from the simulations. For this Final EIR/EIS, spawning WUA was estimated for winter-run, spring-run, fall-run, and late fall–run

Chinook salmon and California Central Valley steelhead in the Sacramento, Feather, and American Rivers.

Operational analyses undertaken for sturgeon included the salvage-density analysis for south Delta entrainment risk (green sturgeon and white sturgeon) and the Delta outflow-year class strength regression analysis (white sturgeon). These analyses are discussed in Appendix 11L, *Sturgeon Analyses*. Appendix 11O, *Anderson-Martin Models*, describes two analytical tools used in the Project analysis of potential temperature-related effects on winter-run Chinook salmon egg mortality—the Martin et al. (2017) and Anderson (2018) egg mortality models. Appendix 11P, *Riverine Flow-Survival*, discusses methods applied to assess potential operational effects of Red Bluff and Hamilton City diversions on juvenile Chinook salmon riverine survival in the Sacramento River as a function of flow.

Implementation of Alternatives 1, 2, and 3 could potentially result in alterations to storage volumes and WSE in Shasta Lake, Lake Oroville, Folsom Lake, San Luis Reservoir, New Melones Reservoir, and Millerton Lake, which could potentially affect reservoir fish species. Model output parameters derived from CALSIM II used to determine potential impacts included:

- End-of-month (average annual monthly) reservoir storage volume
- End-of-month (average annual monthly) WSE

During the period when these reservoirs are thermally stratified (generally April through November), cold-water fish within the reservoir reside primarily within the deeper layers of the reservoir where water temperatures remain suitable. Implementation of the cooperative operations agreements with Reclamation and DWR could alter reservoir storage during this period; implementation could also alter the reservoir's cold-water pool volume, thereby changing the quantity of habitat available to cold-water fish species during these months. Reservoir cold-water pool size generally increases as reservoir storage increases, although not always in direct proportion because of the influence of reservoir basin shape. To assess potential storage-related impacts on cold-water fish habitat availability in Shasta Lake, Lake Oroville, and Folsom Lake reservoirs, end-of-month storage was simulated for Alternatives 1, 2, and 3 were compared to end-of-month storage simulated for the NAA for each month of the April through November period (Appendix 11E, *Reservoir Fish Species Analysis*).

Because reservoir warm-water fish species[4] use the warm upper layer of the reservoir and nearshore littoral habitats, seasonal changes in reservoir storage, as it affects reservoir WSE, and the rates at which WSE change during specific periods of the year, can affect warm-water fish nesting and spawning success. To assess the impacts of potential reservoir WSE changes on warm-water fish due to implementation of the Project, the following approach was used. The Authority and Reclamation simulated the magnitude of change, as measured in feet with reference to mean sea level (feet mean sea level), in reservoir WSE occurring each month of the primary spawning period for nest-building fish (March through June) for Alternatives 1, 2, and 3 and compared the modeled results to the NAA. The Authority and Reclamation compared the

[4] Largemouth bass are evaluated as an indicator species in this EIR/EIS analysis to reflect potential impacts on warm-water game fishes.

number of times that reservoir reductions of 6 feet or more per month could occur with implementation of Alternatives 1, 2, and 3 to the number of occurrences of the same modeled for the NAA (Appendix 11E).

A detailed description of the specific methods utilized to evaluate potential impacts on cold-water and warm-water fish species in each of the existing reservoirs potentially affected by implementation of Alternatives 1, 2, and 3 is provided in Appendix 11B.

11.3.3. Maintenance

Maintenance activities may have some limited potential to "harm" juvenile and adult fishes by direct physical contact, including physical injury or mortality (Impact FISH-20). The assessment of impacts from maintenance activities was based on a qualitative evaluation for the facilities included under Alternative 1, 2, or 3 and focuses on maintenance activities that are near waterways and can affect fish, if fish are present. Examples of these activities include debris removal, vegetation control, rodent control, routine inspections. Electrical transmission connections and lines, substations, distribution lines, dam monitoring equipment, and administration and other buildings are not included in the assessment since these facilities are located away from waterways and will not affect fish and aquatic resources.

The following BMPs, which are described in Appendix 2D, are incorporated into the analysis of potential maintenance impacts on fish resources.

- BMP-12, Development and Implementation of SWPPP(s) and Obtainment of Coverage under Stormwater Construction General Permit (Stormwater and Non-stormwater), requires development and use of erosion control measures, sediment control measures, construction materials management measures, waste management measures, non-stormwater control measures, and post-construction stormwater management measures for a minimum of 5 years following completion of construction to prevent the transport and delivery of sediment into adjacent waterways, and prevent the alteration of fish physiology, behavior, habitat conditions, and reduce the potential for direct injury or mortality to fish.
- BMP-13, Development and Implementation of Spill Prevention and Hazardous Materials Management/Accidental Spill Prevention, Containment, and Countermeasure Plans (SPCCPs) and Response Measures , requires site-specific plans with measures to minimize effects from spills of hazardous or petroleum substances during construction and operation/maintenance by implementing measures such as physically-distancing equipment from waterways, maintaining spill prevention kits at facilities where hazardous materials may be used, providing the equipment and materials necessary for cleanup of accidental onsite spills, and storing hazardous materials in double containment. Implementing this BMP will avoid and reduce localized water quality degradation and would prevent or reduce the potential for direct injury or mortality, habitat degradation, and reduced prey availability.
- BMP-30, Development and Implementation of Hazardous Materials Management Plans, requires HMMPs before beginning construction that considers hazardous materials present on site and known historic site contamination, governs the storage, use, or transfer of hazardous materials, outlines emergency spill containment and cleanup

procedures, and outlines procedures for handling, hauling, and disposing of hazardous waste generated at work sites to avoid and minimize the potential for hazardous materials and waste to enter aquatic habitat.

11.3.4. Thresholds of Significance

An impact on aquatic biological resources would be considered significant if the Project would:

- Have a substantial adverse effect, either directly or through habitat modifications, on any fish species identified as a candidate, sensitive, or special-status species in local or regional plans, policies, or regulations, or by the California Department of Fish and Wildlife or U.S. Fish and Wildlife Service.
- Interfere substantially with the movement of any native resident or migratory fish species or with established native resident or migratory wildlife corridors, or impede the use of native wildlife nursery sites.
- Conflict with any local policies or ordinances protecting aquatic biological resources.
- Conflict with the provisions of an adopted habitat conservation plan, Natural Community Conservation Plan, or other approved local, regional, or state habitat conservation plan.

Conflict with local plans or habitat conservation plan, Natural Community Conservation Plan or other approved local, regional or state habitat conservation plan are indirectly assessed through the evaluation of indirect and direct impacts disclosed in the analysis related to construction, operation, and maintenance. If conflicts were to occur as a result of a significant direct or indirect impact to a special-status fish species they are noted in the analysis below.

11.4 Impact Analysis and Mitigation Measures

Impact FISH-1: Construction Effects on Special-Status Fish

No Project

Under the NAA, no Project facilities would be constructed and there would be no associated ground-disturbance that would result in temporary or permanent impacts on special-status fish or aquatic biological resources from construction activities (including ground disturbance) or placement of Project facilities, respectively.

<u>*Significance Determination*</u>

The NAA would not result in construction effects on special-status fish or aquatic biological resources because no new Project facilities would be built. There would be no construction impacts or effects on state and federally listed species or their critical habitats, or other special-status species and species of management concern and their aquatic habitats.

Alternatives 1, 2, and 3

Effects of Alternatives 1, 2, and 3

Acoustic Effects

Under Alternatives 1, 2, and 3, underwater noise would be generated by a variety of construction activities including pile driving (impact and vibratory)[5], dredging in Funks Reservoir, geotechnical investigations, bridge and culvert construction and replacement, and riprap placement for the CBD outlet. Pile driving is proposed for coffer dam construction or other construction activities associated with GCID system upgrades at the head gates, Walker Creek siphon, and Willow Creek siphon; coffer dam and wall construction at the Funks and TRR PGPs; installation of sheet pile vertical shafts for the tunnel crossings for the TRR pipelines at the GCID Main Canal and TC Canal; and coffer dam construction for the Dunnigan Pipeline at the CBD outlet. Underwater noise from construction activities associated with Alternatives 1, 2, and 3 has the potential to disturb or result in the injury and/or mortality of special-status fish in Funks and Stone Corral Creeks and the CBD (Table 11-2).

Under Alternative 2, additional construction activities with potential to generate underwater noise would include the installation of a coffer dam in the Sacramento River and the installation of up to two temporary work platforms, each with 12 to 15 vibrated or hammer-driven pipe piles on the river bank, for the discharge and energy dissipation structure for the Sacramento River discharge. Underwater noise from construction activities for Alternative 2 also has the potential to disturb or result in the injury and/or mortality of state and federally listed species or other special-status fish in the Sacramento River (Table 11-2). Under Alternative 2, the work platforms would be constructed in about 2 weeks, while coffer dam installation is predicted to take 4 to 6 weeks.

Impact pile driving in or near aquatic habitat (i.e., within 200 feet) would be of most concern as it generates sound levels that can injure or kill fish and other aquatic organisms. Alternatives 1, 2, and 3 include physical or structural components that would require vibratory and/or impact driving of temporary and permanent piles during construction. Several of these components involve pile driving activities in or adjacent to waterbodies supporting fish, resulting in their potential exposure to pile driving noise.

Pile Driving Effects on Fish

Research indicates that impact pile driving can result in adverse effects on fish because of the level of underwater sound produced (Popper and Hastings 2009:464–480). The effects of pile driving noise on fish may include behavioral responses, physiological stress, temporary and permanent hearing loss, tissue damage (auditory and non-auditory), and direct mortality. Factors

[5] Impact pile driving uses a hydraulic hammer mounted on a piling rig with a ram mass to dynamically drive piles into the ground, while vibratory pile driving uses a low impact method of creating vertical vibrations that puts soil particles into motion thereby loosening the soil and allowing the pile to penetrate the soil. Impact pile driving results in high intensity impulsive sounds that can potentially cause injury in fish. Vibratory hammers generally produce less sound than impact hammers and are often employed as a mitigation measure to reduce the potential for adverse effects on fish that can result from impact pile driving (California Department of Transportation 2015:2-17). In addition, there are no established injury criteria for vibratory pile driving (California Department of Transportation 2015:2-17); therefore, effects on fish from vibratory pile driving are typically behavioral.

that may influence the magnitude of effects include: (1) species, life stage, and size of fish (smaller fish are more susceptible to injury); (2) type and size of pile and hammer (larger piles and bigger hammers result in more noise); (3) frequency and duration of pile driving (more strikes per day means greater accumulated energy); (4) site characteristics (e.g., water depth, channel bends [sound attenuates faster in shallow water and around bends]); and (5) distance of fish from the source (fish closer to the source of the noise are at greater risk of injury than fish farther away).

Dual interim criteria have been established to provide guidance for assessing the potential for injury of fish resulting from pile driving noise (Fisheries Hydroacoustic Working Group 2008:1) and were used in this evaluation. The dual criteria for impact pile driving are shown in Table 11-5.

Table 11-5. Interim Criteria for Injury to Fish from Impact Pile Driving Activities

Interim Criteria	Agreement in Principle
Peak sound pressure level (SPL)	206 dB re: 1 μPa (for all sizes of fish)
Cumulative sound exposure level (SEL)	187 dB re: 1 μPa^2-sec—for fish size ≥ 2 grams 183 dB re: 1 μPa^2-sec—for fish size < 2 grams

Source: Fisheries Hydroacoustic Working Group 2008
dB = decibel; μPa = microPascal; sec = second

The following criteria relate to impact pile driving only. The peak sound pressure level (SPL) is considered the maximum sound pressure level a fish can receive from a single strike without injury. The cumulative sound exposure level (SEL) is considered the total amount of acoustic energy that a fish can receive from single or multiple strikes without injury. The cumulative SEL threshold is based on the cumulative daily exposure of a fish to noise from sources that are discontinuous (e.g., noise that occurs for only 8 to 12 hours in a day, with 12 to 16 hours between exposure). This quiet period assumes that the fish is able to recover from any pile driving effects during this 8–12-hour period.

Vibratory pile driving methods produce more continuous, lower energy sounds below the thresholds associated with injury. There are currently no established noise thresholds associated with continuous sound waves, and vibratory methods are generally accepted as an effective measure for minimizing or eliminating the potential for injury of fish during in-water pile driving operations, though they could still cause physiological and behavioral changes (McCauley et al. 2003; Popper and Hastings 2009).

The following analysis also considers the potential for pile driving sound to adversely affect fish behavior. Potential mechanisms include startle or avoidance responses that can disrupt or alter normal activities (e.g., migration, holding, feeding) or expose individuals to increased predation risk. Insufficient data are currently available to support the establishment of a noise threshold for behavioral effects (Hastings and Popper 2005:46; Popper and Hastings 2009:464). NMFS generally assumes that a noise level of 150 decibels (dB) root mean square (RMS) is an appropriate threshold for behavioral effects, although neither research data nor related citations have been provided to support this threshold (California Department of Transportation 2015:4-23).

Noise from Impact Pile Driving

For coffer dam sheet piles, the pile driving analysis suggests that the distance from the source pile to sound level thresholds (i.e., upstream and downstream) would extend 30 feet for the 206 dB injury threshold and 2,814 feet for the 187 dB SEL injury threshold, assuming an unimpeded propagation path (National Marine Fisheries Service 2017:110–116). However, the potential for behavioral effects would extend much farther. Based on the threshold of 150 dB RMS, potential behavioral effects are calculated to extend up to 13,058 feet (2.5 miles) from the source pile. While the extent of the 187-dB and 150-dB RMS thresholds overlap and would cover the entire width of the CBD (Alternative 1, 2, or 3) and Sacramento River (Alternative 2), major bends in the CBD (approximately 5,500 feet upstream and downstream from the outlet) and in the Sacramento River (approximately 6,000 feet upstream and 2,500 feet downstream from the discharge site) would limit the propagation of the sound path. These estimates likely reflect maximum distances from the source pile for potential direct injury and behavioral impacts on special-status fish in the CBD and state and federally listed fish and other special-status fish in the Sacramento River (Table 11-2). Maximum distances from the source pile for potential direct injury and behavioral impacts on special-status fish in Funks and Stone Corral Creeks are expected to be considerably less because shallow water, intermittent flow, and the sinuous nature of these creeks would limit the propagation of underwater noise. Furthermore, this analysis assumes use of impact pile driving exclusively (i.e., no vibratory pile driving during initial pile installation). Thus, these effect estimates are conservative.

Noise from Geotechnical Investigations

Geotechnical investigations could result in temporary acoustic effects on fish. Acoustic effects from standard geotechnical penetration tests (i.e., dropping a 140-pound automatic hammer to drive a sampler about 1.5 feet) are limited to minimal, short-duration vibrations (National Marine Fisheries Service 2017:177). Placement of riprap has the potential to result in temporary loud noises, although the available data from analogous situations suggest such effects would be limited. For example, sound data taken during the 2012 installation of rock barriers as part of DWR's Temporary Barriers Project in the South Delta showed that noise levels at 100 meters from construction were below the NMFS threshold for adverse behavioral effects (150 dB). Effects on fish in the CBD would be limited to behavioral effects only while the riprap is being placed.

Pile driving would be performed in accordance with BMP-23. This BMP would reduce the potential for injury to fish from exposure to impact pile driving noise because hydroacoustic monitoring would be conducted during impact pile driving to ascertain compliance with established objectives (e.g., distances to cumulative noise thresholds) and identify corrective actions to be taken should the predicted threshold distances be exceeded. Corrective actions include stopping pile driving for the day when predicted threshold distances are exceeded and limiting the number of pile strikes per day on subsequent days to ensure compliance with the predicted threshold distances. In addition, all pile driving (impact or vibratory) would be restricted to seasonal (September 1 to October 15) and daily (7:00 a.m. to 7:00 p.m.) timing limitations.

The proposed seasonal and daily timing restrictions and localized effect of pile driving and other in-water construction activities would limit adverse noise-generated effects to a small proportion of special-status fish species in Funks Creek, Stone Corral Creek, and the CBD (Table 11-2) under Alternatives 1, 2, and 3. Under Alternative 2, pile driving and other in-water construction activities would be timed (September 1–October 15) for periods when life stages of some fish species are not present (e.g., adult winter-run Chinook salmon and juvenile spring-run Chinook salmon; Table 11-2) or their abundance in the affected reach of the Sacramento River is relatively low (e.g., adult green sturgeon; Table 11-2). Any fish present would be expected to pass through the affected area relatively quickly in response to general construction noise and physical disturbance, thereby limiting their exposure. Restricting pile driving in the Sacramento River to September 1 to October 15 would also limit exposure to only a proportion of the total population of more abundant species (e.g., adult steelhead and fall-run Chinook salmon) migrating through the affected area at this time of year. In combination with seasonal restrictions, limiting pile driving to daylight hours only would give fish a 12-hour period to recover between exposures or migrate through the area unexposed during nighttime hours, further limiting the proportion of any given fish run exposed to underwater noise. Thus, noise generated by pile driving and other in-water construction activities would be expected to affect only a small proportion of these fish populations in the Sacramento River.

Sediment Disturbance

The construction of Alternatives 1, 2, and 3 would involve activities that would potentially cause erosion and the disturbance of sediment and soil, subsequently resulting in sediment transport and delivery to streams. Sediment input to streams could temporarily increase water column turbidity and sedimentation rates above ambient levels and potentially alter fish physiology, behavior, and habitat conditions in waterways. Construction activities that have the potential to result in erosion and sediment transport and delivery to streams include: (1) site clearing and vegetation removal in the inundation area and at the CBD outlet; (2) excavation, grading, and soil placement to construct the main dams, saddle dams, and saddle dikes; (3) topsoil storage and road construction and relocation; (4) culvert and bridge modification and construction; (5) dredging/excavation in Funks Reservoir; (6) bed and bank disturbance at creek crossings; and (7) the placement and removal of coffer dams and stream diversions to construct the main dams. Under Alternative 2, sediment- and turbidity-producing activities would include: (1) constructing the Dunnigan Pipeline to the Sacramento River; (2) site clearing and vegetation removal to construct the Sacramento River discharge; and (3) installing a coffer dam for the Sacramento River discharge. Construction activities that would occur in or immediately adjacent to stream channels (e.g., excavation, grading, vegetation removal, bridge and culvert construction, and the placement and removal of coffer dams and stream diversions) and Funks Reservoir (dredging/excavation) or during the wet season have the greatest potential to disturb stream sediments or cause erosion and contribute sediment to streams.

Elevated levels of suspended sediments have the potential to result in physiological, behavioral, and habitat effects on fish. The severity of these effects depends on the sediment concentration, duration of exposure, proximity of the action to the waterbody, and timing of the disturbance relative to the occurrence of the species and sensitive life stages. Short-term increases in turbidity and suspended sediment may disrupt normal behavior patterns of fish, potentially affecting foraging, rearing, and migration. The level of disturbance may also cause juvenile fish to abandon protective habitat or reduce their ability to detect predators, potentially increasing their vulnerability to predators (e.g., piscivorous birds and fish). Chronic exposure to high turbidity and suspended sediment may affect fish growth and survival by impairing respiratory function, reducing tolerance to disease and contaminants, and causing physiological stress (Waters 1995). Deposition of excessive fine sediment on the stream bottom could eliminate habitat for aquatic insects; reduce density, biomass, number, and diversity of aquatic insects and vegetation; reduce the quality and quantity of spawning habitat; and block the interchange of surface and subsurface waters. Substantial sediment input and deposition could: (1) cause channel braiding; (2) increase width: depth ratios; (3) increase incidence and severity of bank erosion; (4) reduce pool volume and frequency; and (5) increase subsurface flow. See Chapter 7, *Fluvial Geomorphology*, for additional information on potential effects related to elevated levels of suspended sediments.

Based on general observations of similar in-water construction activities, increases in turbidity and suspended sediment generated during construction would be temporary and localized, and unlikely to reach levels causing direct injury or mortality to fish. In some cases, increases in turbidity could produce minor positive effects. For example, turbidity has been known to reduce vulnerability to predation in some species interactions (Gregory and Levings 1998:275). NMFS (2008:95) reviewed observations of turbidity plumes during installation of riprap for bank protection projects on the Sacramento River and concluded that visible plumes are expected to be limited to only a portion of the channel width, extend no more than 1,000 feet downstream, and dissipate within hours of cessation of in-water activities. Based on these observations, NMFS (2008:95) concluded that such activities could result in turbidity levels exceeding 25–75 NTU. This level of effect is considered representative of potential turbidity effects from Alternatives 1, 2, and 3.

Historically and currently, much of the study area is used for agriculture; therefore, soils could be contaminated with pesticides, herbicides and other chemicals used in agriculture, as well as other contaminants (Chapter 27, *Public Health and Environmental Hazards*). Eroded soils have been known to transport pollutants such as nutrients; metals; oils, fuels, and grease; and pesticides, herbicides, and other agricultural chemicals. Eroded soils could result in the potential release and dispersal of these contaminants if contaminated sediments are disturbed during construction and transported and delivered to streams. Fish could be directly exposed to elevated levels of contaminants if they are in immediate proximity to construction activities that disturb contaminated sediments. Channel bed disturbance could also result in indirect effects on fish. Toxins in river channel sediments can enter the foodweb through uptake by benthic organisms. If contaminated sediments are disturbed and become suspended in the water column, they also become available directly to pelagic organisms, including fish species and planktonic food sources of fish species. Thus, construction-related disturbance of contaminated bottom sediments creates another potential pathway to the food chain, and the potential accumulation of these toxins in the tissue (i.e., bioaccumulation) of various fish species. The bioaccumulation of toxins can lead to lethal effects, as well as sublethal effects (e.g., effects on behavior, digestion, and immune system response) (Connon et al. 2011:290). Because toxins in contaminated sediments are adhered to the sediment, increases in suspended sediment generated during construction could release these contaminants to the water column and substrate. However, as described above for turbidity, increases in suspended sediment and associated contaminants would be spatially limited to a portion of channel width and not extend far downstream, dissipating within hours of construction activities ceasing.

Construction of Alternative 1, 2, or 3 would be subject to a construction-related stormwater permit and dewatering requirements of the federal Clean Water Act and National Pollutant Discharge Elimination System program. The Authority would obtain required permits through the Central Valley Regional Water Quality Control Board before any ground-disturbing construction activity occurs and implement BMP-14. As required in BMP-12, the Authority will develop and implement a SWPPP before and throughout the construction period as one of its BMPs to protect fish and aquatic habitat from exposure to elevated levels of contaminants and sediment by preventing water runoff, spills, and sediment from entering waterways in immediate proximity to construction activities by using physical barriers and sediment basins or by locating construction and staging activities not in proximity of waterways to the extent practicable. The SPCCPs and response measures described in BMP-13 would prevent and minimize the introduction of oil during construction activities into surface waters through specific equipment, workforce, procedural, and training requirements for the prevention of, preparedness for, and response to oil discharges (U.S. Environmental Protection Agency 2010). These measures would ensure that stormwater runoff would be controlled with physical and procedural means to reduce or avoid degradation of water quality in watercourses downstream of the construction sites that could have both short- and long-term effects on fish populations and aquatic habitat. All in-water construction activities would be limited to allowable in-water work windows as part of BMP-35 and the Authority or its contractors would manage the salvage, stockpiling, and replacement of topsoil through implementation of BMP-10 for the protection of fish, wildlife, and plant species. Implementation of these BMPs would ensure that ground-disturbance construction activities did not violate water quality standards or waste discharge requirements or otherwise substantially degrade water quality that would adversely affect fish populations and habitat.

Water Quality Effects

Construction of Alternatives 1, 2, and 3 could result in accidental spills of contaminants, including cement, oil, fuel, hydraulic fluids, paint, and other construction-related materials, resulting in localized water quality degradation. This could in turn result in adverse effects on fish through direct injury and mortality (e.g., damage to gill tissue that causes asphyxiation) or delayed effects on growth and survival (e.g., increased stress or reduced feeding), depending on the nature and extent of the spill and the contaminants involved. In addition, under Alternative 2, accidental spills of contaminants could occur while constructing the Dunnigan Pipeline to the Sacramento River, constructing the discharge and energy dissipation structure at the Sacramento River, and installing the coffer dam for the Sacramento River discharge.

The greatest potential for an adverse water quality impact is associated with an accidental spill from construction activities occurring in or near surface waters. Construction of Golden Gate Dam and Sites Dam, installation of coffer dams and stream diversions, road construction and improvements at creek crossings, and pipeline construction at creek crossings in particular involve extensive in-water work. Other construction elements that occur in upland areas or are isolated from fish-bearing waters have little potential for accidental spills that could affect fish because of the distance separating construction activities from receiving waters. Discharge of water from construction sites could also affect water quality for fish. See Chapter 6, *Surface*

Water Quality, for additional information on potential surface water quality effects associated with Project construction.

The BMPs described above for sediment disturbance would also reduce and minimize effects associated with water quality and potential effects on state and federally listed fish and other special-status fish species because they would preventing water runoff, spills, and sediment from entering waterways in immediate proximity to construction activities by using physical barriers and sediment basins or by locating construction and staging activities not in proximity of waterways to the extent practicable.

Direct Physical Injury

In-water construction for Alternatives 1, 2, and 3 may result in direct physical injury or mortality to fish from activities including pile driving, coffer dam installation, and stream and coffer dam dewatering (i.e., fish stranding or entrainment into pumps used during dewatering). In addition, under Alternative 2, direct physical injury or mortality to fish could occur from coffer dam placement in the Sacramento River at the discharge site, and coffer dam dewatering. Installation of piles or placement of riprap (i.e., for the CBD outlet) could involve fish being crushed, although that risk would be expected to be low based on the limited spatial extent of the work, the timing of construction activities, and the high probability of fish avoiding such activities. Displacement of fish away from habitat near construction activities seems the most likely adverse effect. Fish entrapped in construction areas enclosed by coffer dams or diverted stream segments that are subsequently dewatered would die without fish rescue activities, although the number of fish being trapped in such areas would be a low proportion of individuals relative to the overall extent of species' ranges. Fish that are rescued from stream segments proposed for dewatering and from coffer dams prior to or during dewatering could be injured and killed during rescue activities or as a result of handling. BMP-34 would avoid and minimize the potential for direct physical injury and mortality of trapped fish. Implementation of this BMP would ensure the protection of state and federally listed fish and other special-status fish species by moving them away from areas where they could be harmed using standard herding, collecting, handling, and relocation protocols. Dredging/excavation of Funks Reservoir would not result in any direct physical injury to fish because this activity would occur in the dry after the irrigation reservoir is lowered during the non-operational period (i.e., December through February), thus exposing the reservoir bottom.

Reduced Prey Availability

Construction of Alternatives 1, 2, and 3 has the potential to reduce prey availability for fish through disturbance of aquatic habitat. Prey species may be affected by: (1) pile driving (e.g., from noise effects or direct physical contact); (2) dredging (e.g., sediment disturbance); (3) removal of riparian aquatic habitat (i.e., reducing habitat structures for prey in or above water during clearing and grubbing); and (4) riprap placement (e.g., direct physical contact and sediment disturbance) for the CBD outlet. In addition, under Alternative 2, disturbance of aquatic habitat with potential to adversely affect prey availability would occur in the Sacramento River associated with removal of riparian aquatic habitat and installation of the coffer dam for the Sacramento River discharge. Isolation of construction areas with coffer dams would prevent fish access to prey in these areas for the duration that these features are installed at each location. The potential effects would be limited in extent relative to the overall area of habitat available to fish in the affected waterways.

Increased Predation

Construction of Alternatives 1, 2, and 3 has the potential for increased predation risk for juvenile fish associated with the removal of riparian vegetation and the placement of riprap associated with the CBD outlet. The removal of riparian vegetation, installation of a coffer dam, and placement of riprap would reduce habitat complexity and shading. This reduction in habitat complexity and shading could potentially lead to the creation of nonnative predatory habitat for alien fish species (e.g., largemouth bass, sunfish, striped bass), making juvenile fish of special-status species more susceptible to predation. In addition, under Alternative 2, increased predation risk for state and federally listed species, particularly juvenile salmonids, and other special-status fish species would occur in the Sacramento River associated with the removal of riparian vegetation and installation of a coffer dam during construction of the Sacramento River discharge. Overall, however, the potential effects from presence of in-water structure and removal of riparian vegetation would be limited as the overall extent of the removal of riparian vegetation would be low and coffer dams would be temporary. The loss of riparian vegetation at the CBD from construction of Alternatives 1, 2, and 3 would be expected to be limited in terms of additional negative effects.

In addition to riparian vegetation removal and in-water structure effects during construction, the various forms of in-water construction work (pile driving, coffer dam installation, and riprap placement at the CBD) have the potential to increase predation risk for smaller fish species by increasing disturbance and susceptibility to predation (e.g., by masking the sounds of approaching predators, or causing fish to flee disturbed areas), which in turn could increase predation success of larger predatory fish such as black bass and striped bass (Sacramento River only). Such effects would be temporally and spatially limited in extent.

Loss of Riparian Vegetation (Including SRA Cover) and Increased Water Temperature

Shaded riverine aquatic (SRA) cover is a component of riparian vegetation. SRA cover is defined as the unique, nearshore aquatic area occurring at the interface between a river (or stream) and adjacent woody riparian habitat (Fris and DeHaven 1993). Riparian vegetation, including vegetation supporting SRA cover, occurs in three land cover types: forested wetland, scrub-shrub wetland, and upland riparian. The removal of trees in these land cover types where necessary at construction sites (e.g., during clearing and grubbing) under Alternatives 1 and 3 would reduce the extent of riparian vegetation, including vegetation supporting SRA cover habitat (Tables 9-2a and 9-2b in Chapter 9, *Vegetation and Wetland Resources*). Under Alternative 2, tree removal would occur in many of the same locations as under Alternatives 1 and 3, but would also occur along Wilson, Grapevine, and Antelope Creeks during construction of the South Road and along the Sacramento River during construction of the Sacramento River discharge (Tables 9-4a and 9-4b). A comparison of the total impacts on these three land cover types (forested wetland, scrub-shrub wetland, and upland riparian) between Alternative 1 or 3 and Alternative 2 is shown in Table 9-5. Although impact acreages of SRA cover would be much smaller than these riparian impacts, riparian impacts provide a reasonable measure of the relative magnitude of SRA cover impacts under each alternative. Under Alternatives 1 and 3, total impacts on riparian vegetation, and therefore SRA cover, would be the same, while impacts under Alternative 2 would be approximately 40% more than under Alternatives 1 or 3. Sites reservoir inundation and road construction account for the majority of impacts on riparian vegetation and SRA cover under all alternatives.

Riparian vegetation is important in controlling stream bank erosion, contributing to instream structural diversity, and maintaining undercut banks in the absence of rock revetment. In addition, canopy cover (overhanging vegetation [a form of SRA cover]) maintains shade that is necessary to reduce thermal input and provides an energy input to the aquatic habitats in the form of fallen leaves and insects (a food source for fish). SRA cover also provides fish with protection from predators in the form of undercut banks and instream woody material in the form of submerged branches, roots, and logs, and provides habitat for several native, regionally important fish and wildlife species.

Riparian and SRA cover habitats are essential components of salmonid rearing habitat that may limit the production and abundance of salmonids in the Sacramento River. Salmonid populations in particular are highly influenced by the amount of available cover (Raleigh et al. 1984). The amount of existing riparian and SRA cover habitat in the study area and in the region is of variable quality (i.e., may not provide all of the functions described in the previous paragraph and/or provides those functions of SRA habitat to a lesser spatial extent than high-quality SRA habitat) because of past and ongoing impacts, including levee construction and bank protection activities (i.e., placement of rock revetment), livestock grazing, and clearing for agricultural use.

USFWS mitigation policy identifies California's riparian habitats, including SRA cover habitat, as a Resource Category 2 habitat. The designation criteria for habitat in Resource Category 2 is "habitat to be impacted is of high quality for evaluation species and is relatively scarce or becoming scarce on a national basis or in the ecoregion section" (U.S. Fish and Wildlife Service 1993) for which "no net loss of in-kind habitat value" is recommended (46 FR 7644, January 23, 1981). In addition, NMFS typically recommends revegetating onsite at a 3:1 ratio (three units replaced per each unit of affected habitat) with native riparian species to facilitate the development of SRA cover habitat. Implementation of Mitigation Measure VEG-2.1 would reduce the level of impact because all locations of riparian vegetation in and within 300 feet of the Project footprint would be identified and mapped, and implementation of Mitigation Measure VEG-2.2 would compensate for the removal of riparian vegetation and SRA cover habitat.

The removal of SRA cover habitat that contributes to stream shading could potentially increase water temperature and have adverse effects on fish, depending on species-specific temperature preferences. However, such increases would be extremely localized as the linear extent of SRA cover habitat that would be removed at individual construction sites would be relatively small, ranging from a footprint impact as small as several trees at construction sites on tributary streams to a footprint impact as large as approximately 200 to 250 linear feet of continuous riparian habitat along the Sacramento River under Alternative 2. NMFS (2017:220) noted the Sacramento River is a wide, faster-moving waterbody and is less likely to experience warming of water temperatures caused by limited decreases in riparian vegetation, such as would occur with construction of Alternative 2. This is because as river channels become wider, a smaller fraction of the channel is affected by shading from the riparian vegetation found along those riverbanks. As further described by NMFS (2017:220), the volume of water present in the river channel acts as a thermal sink, resisting temperature changes caused by shading along a narrow riparian zone. Temperature changes are more influenced by the greater surface area of exposed open water in the river channel, ambient air temperatures over those exposed areas, solar irradiation, and the influence of water layers mixing within the main river channel. Nonetheless, juvenile salmonids preferentially use river margins during migration and rearing, so any increases in water temperature that occur as a result of SRA cover habitat removal could affect migrating anadromous salmonids (Greenberg et al. 2008; Hellmair et al. 2018). Because any water temperature increases as a result of decreased riparian vegetation under Alternative 2 are anticipated to be small and localized, the effects on fish from changes in water temperature would be expected to be minimal.

Reduced Habitat Extent and Access

Filling of the reservoir would result in permanent modification of aquatic habitat by replacing intermittent and ephemeral stream habitat above the dams with a permanent body of water subject to fluctuations in volume, depth, and surface area. Under Alternatives 1 and 3, filling of the reservoir would result in the permanent loss of a cumulative total of approximately 24.3 miles (61%) of stream habitat in Funks Creek, Stone Corral Creek, Antelope Creek, and Grapevine Creek upstream of the dam sites at maximum reservoir inundation levels. Under Alternative 2, a cumulative total of approximately 24.1 miles (60%) of stream habitat would be permanently lost in these same drainages at maximum reservoir inundation levels. These impacts include the loss of stream habitat in Funks Creek, Stone Corral Creek, Antelope Creek, and Grapevine Creek that supports native Sacramento hitch and Central California roach based on sampling conducted in the inundation area by CDFW (Brown 2000:23–27). Brown (2000:24, 43) characterized these streams as being deeply incised channels with little vegetation on the banks and little cover in the streams, having poor water quality, in part, due to high dissolved minerals, and possessing a highly variable flow regime with periods of high flow during winter, declining flows in spring, and intermittent flow, and perhaps no flow, in late summer. In addition, the high concentration of sediments and aquatic vegetation present in these streams may make them uninhabitable to most fish in summer due to high oxygen demand (Brown 2000:43). Therefore, habitat appears to be marginal for special-status fish species in these creeks. Chapter 7 provides a description of the geologic and topographic setting, and geomorphic characteristics associated with these drainages.

General effects of Sites Reservoir construction can be described based on observations from other California reservoirs and general species habitat requirements. Reservoirs can provide suitable habitat for native and introduced fish species but are generally less productive than natural lakes because of their depth, steep slopes, and fluctuating water levels that typically limit spawning and rearing success (Moyle 2002:36). Some minnows (e.g., hitch) are reported to use reservoirs and, in some cases, have developed large populations where introduced predators or competitors are absent (Moyle 2002:37), while other native minnow species (e.g., roach) typically are absent from reservoirs and exist only in the small tributaries above them (Moyle et al. 2015:8). It is expected that Sites Reservoir would contain predatory species that are the result of either planned fish stocking (Chapter 2) or future potential introductions of nonnative predators or competitors through the transfer of water or accidental or deliberate introductions. Considering the physical and operational characteristics of the reservoir and the expected presence of predatory fish species, the reservoir may not provide suitable habitat for native fish species. Although stream habitat in the inundation area would be lost, native fish populations would be expected to continue to persist in Funks Creek and Stone Corral Creek above the inundation area and in stream reaches downstream of Sites Dam and Golden Gate Dam provided that tailwater releases below the dams provide suitable temperature and habitat conditions for native species, such as hitch (Moyle et al. 2015:3). See Impact FISH-11 for further discussion of operations effects on native minnows downstream of the dams and the steps that would be taken by the Authority to maintain fish in good condition in these stream reaches consistent with California Fish and Game Code Section 5937.

Based on a review of Google Earth imagery, fish movement in Funks and Stone Corral Creeks is currently affected by the intermittent nature of these streams, and the occurrence of road crossings, flow control structures, and other features downstream of the proposed dams. For example, Funks Reservoir and spillway, located on Funks Creek approximately 1.8 miles downstream of proposed Golden Gate Dam, preclude fish downstream of Funks Reservoir from accessing habitats upstream of the proposed Golden Gate Dam. Similarly, at least one low-water road crossing and several flow control structures in lower Stone Corral Creek appear to impede or block fish in the lower valley reaches of the creek and the CBD from ascending Stone Corral Creek and accessing habitat upstream of the proposed Sites Dam. However, the extent to which these native minnow species may move within these creeks is unknown and movement of these species is not considered an essential behavioral component of their life cycle given that they are freshwater resident species, and spawning and rearing habitat for these species would continue to persist upstream and downstream of the dams and reservoir. Construction of Sites Dam and Golden Gate Dam on Stone Corral Creek and Funks Creek, respectively, would preclude fish in those drainages from moving upstream or downstream past the dams, as the dams would not include fish passage facilities. These barriers would prevent juvenile and adult fish in Funks Creek and Stone Corral Creek from making seasonal movements within their respective creeks for spawning or dispersal purposes, or to search for refugia during late spring and summer dry periods, although flow releases from the dams would maintain fish in good conditions in the streams below the dams and the streams upstream of Sites reservoir would not be affected by the dams or reservoir. (It should also be noted that special-status minnows in the Sacramento River would not be affected by construction of Sites Dam and Golden Gate Dam because the Knights Landing Outfall Gates that were recently upgraded to prevent salmon from entering the CBD also prevents these minnow species from ascending the CBD and its tributaries.) Although specific information on fish movements and habitat needs in these streams is lacking, Brown (2000:27) concluded that the general absence of fish larger than 5.9 inches at sampling locations in the inundation area indicated that primarily juvenile fish rear in these areas and that adult fish ascend these seasonal creeks in winter and spawn in these stream reaches in early spring before migrating downstream after spawning, although it is unknown which species of fish to which CDFW was referring. However, it should be noted that some of the roach and hitch collected by CDFW (Brown 2000) in Funks Creek and Stone Corral Creek were likely adults as roach typically become mature after they reach 45–60 mm (1.8–2.4 inches) and hitch can reach sexual maturity when only 49–54 mm (1.9–2.1 inches) (Moyle 2002:138, 142). This suggests that adults are likely present year-round in these streams and that stream reaches unaffected by reservoir inundation would be expected to continue to support roach and hitch. Nevertheless, dams have been implicated as a major factor limiting, or potentially limiting, population viability of

Sacramento hitch and Central California roach due to habitat fragmentation, altered flows and reduced habitat suitability (Moyle et al. 2015:3, 4, 7–11).

Although construction of Sites Reservoir would result in additional habitat fragmentation in Funks Creek and Stone Corral Creek, it is expected that populations of these native minnow species would continue to persist in these streams upstream and downstream of Sites reservoir for the reasons outlined above. The permanent loss of this marginal stream habitat and related negative effects on population size in the Funks and Stone Corral drainages associated with reservoir inundation would represent a small fraction of overall habitat and individuals when put in a broader population-wide context for these species across their respective ranges in California.

CEQA Significance Determination and Mitigation Measures for Alternatives 1, 2, and 3

Construction of Alternative 1, 2, or 3 would result in ground-disturbance activities, the use of heavy equipment and hazardous materials, in-water construction (including pile driving), stream diversion and dewatering, removal of riparian and stream-side vegetation (including vegetation supporting SRA cover), and the filling of Sites Reservoir. Under Alternatives 1 and 3, and all components of Alternative 2 with the exception of construction of the energy dissipation structure for the Sacramento River discharge, these activities would result in temporary impacts on special-status fish during construction activities. These activities would also result in permanent impacts from placement of facilities and the conversion of stream habitat to open-water habitat from the filling of Sites Reservoir. These temporary and permanent impacts would not affect any ESA-listed fish species as construction activities would occur on the upstream streams of the Sacramento River which do not support listed species.

Under Alternative 2, construction of the energy dissipation structure for the Sacramento River discharge would result in ground-disturbance activities, in-water construction (including pile driving and coffer dam installation), dewatering, and the removal of riparian and stream-side vegetation (including vegetation supporting SRA cover). These activities would result in temporary impacts on state and federally listed fish and other special-status fish in the Sacramento River during construction activities, and permanent impacts from the removal of riparian vegetation and SRA cover. Underwater noise generated by pile driving associated with the installation of sheet piles for the coffer dam and pipe piles for the work platforms would be of most concern because of the potential for underwater noise to injure fish.

The Authority will implement BMPs during construction of Alternatives 1, 2, and 3 to avoid and minimize permanent and temporary impacts on state and federally listed fish and other special-status fish species. Implementation of BMP-12, BMP-13, and BMP-14 would control storm water runoff with physical and procedural means to reduce or avoid degradation of water quality in watercourses downstream of the construction sites that could have both short- and long-term effects on fish populations and aquatic habitat. All in-water construction activities would be limited to allowable in-water work windows as part of BMP-35 and the Authority or its contractors would manage the salvage, stockpiling, and replacement of topsoil as part of BMP-10 for the protection of fish, wildlife, and plant species. As a result, the construction would not result in increased or contaminated stormwater runoff or violations of water quality standards that would adversely affect fish populations and habitat.

The Authority will also implement BMP-34 to avoid and minimize the potential for direct physical injury and mortality of trapped fish by removing fish from harm's way prior to initiating in-water activities and dewatering.

Pile driving would be performed in accordance with BMP-23 to reduce the potential for injury to fish from exposure to impact pile driving noise because hydroacoustic monitoring would be conducted during impact pile driving to ascertain compliance with established objectives (e.g., distances to cumulative noise thresholds) and identify corrective actions to be taken should the predicted threshold distances be exceeded. In addition, this BMP would restrict all pile driving (impact or vibratory) to specific seasonal periods and daily (7:00 a.m. to 7:00 p.m.) timing limitations, where appropriate, to minimize and avoid the primary periods when sensitive life stages or species are present and to limit the daily exposure of fish to underwater noise.

In addition, the Authority will implement various mitigation measures that will also benefit special-status fish or compensate for impacts on state and federally listed fish and other special-status fish and their habitat. For example, Mitigation Measures VEG-2.1 and VEG-2.2 will minimize or avoid, and compensate for the permanent loss of riparian habitat, including SRA cover. Mitigation Measure VEG-3.2 will compensate for permanent impacts on wetlands, including forested wetland (riparian) and freshwater marsh. Mitigation Measure VEG-3.3 will compensate for temporary and permanent impacts on state or federally protected non-wetland waters by creating or acquiring and permanently protecting suitable open-water habitat to ensure no net loss of stream or pond habitat functions and values.

Construction of Alternative 1, 2, or 3 would not have a substantial adverse effect, either directly or through habitat modifications, on state or federally listed fish or other special-status fish species or interfere substantially with the movement of any native resident or migratory fish species or with established native resident or migratory wildlife corridors, or impede the use of native wildlife nursery sites. Construction of Alternative 1, 2, or 3 would be less than significant with mitigation.

Mitigation Measure VEG-2.1: Conduct Surveys for Sensitive Natural Communities and Oak Woodlands in the Project Area Prior to Construction Activities

This measure is described in Chapter 9, Section 9.5, *Impact Analysis and Mitigation Measures*.

Mitigation Measure VEG-2.2: Avoid and Compensate for Adverse Effects on Sensitive Natural Communities

This measure is described in Chapter 9, Section 9.5.

Mitigation Measure VEG-3.2: Compensate for Temporary and Permanent Impacts on State- or Federally Protected Wetlands

This measure is described in Chapter 9, Section 9.5.

Mitigation Measure VEG-3.3: Compensate for Temporary and Permanent Impacts on State- or Federally Protected Non-Wetland Waters

This measure is described in Chapter 9, Section 9.5.

NEPA Conclusion for Alternatives 1, 2, and 3

Construction effects on special-status fish species and aquatic biological resources would be the same as described above for CEQA. Construction of Alternative 1, 2, or 3 would involve ground-disturbing activities, the use of heavy equipment and hazardous materials, in-water construction (including pile driving), stream diversion and dewatering, removal of riparian and stream-side vegetation (including vegetation supporting SRA cover), and the filling of Sites Reservoir. None of these activities would occur as part of the NAA. Under Alternatives 1 and 3, and all components of Alternative 2 with the exception of construction of the energy dissipation structure for the Sacramento River discharge, these activities would result in temporary effects on special-status fish during construction activities. These activities would also result in permanent effects from placement of facilities and the conversion of stream habitat to open-water habitat from the filling of Sites Reservoir. These temporary and permanent effects would not affect any ESA-listed fish species as construction activities would occur on the upstream streams of the Sacramento River, which do not support listed species. Under Alternative 2, construction of the energy dissipation structure for the Sacramento River discharge would result in ground-disturbance activities, in-water construction (including pile driving and coffer dam installation), dewatering, and the removal of riparian and stream-side vegetation (including vegetation supporting SRA cover). These activities would result in temporary effects on state and federally listed fish and other special-status fish in the Sacramento River during construction activities, and permanent effects from the removal of riparian vegetation and SRA cover. Underwater noise generated by pile driving associated with the installation of sheet piles for the coffer dam and pipe piles for the work platforms would be of most concern because of the potential for underwater noise to injure fish. The Authority will implement BMP-10, BMP-12, BMP-13, BMP-14, BMP-23, BMP-33, BMP-34, and BMP-35 during construction of Alternatives 1, 2, and 3 to avoid and minimize permanent and temporary effects on state and federally listed fish and other special-status fish species. In addition, the Authority will implement various mitigation measures that would also benefit special-status fish or compensate for effects on state and federally listed fish and other special-status fish and their habitat. These measures include Mitigation Measures VEG-2.1, VEG-2.2, VEG-3.2, and VEG-3.3. Project construction would have no adverse effect on special-status fish and aquatic biological resources.

Impact FISH-2: Operations Effects on Winter-Run Chinook Salmon

No Project

Under the NAA, there would be no associated Sites Reservoir operations under Impacts FISH-2 through FISH-17 and Impact FISH-19. Under the NAA continued implementation of regulatory criteria, including implementation of the 2019 BiOps from the USFWS and NMFS for the Reinitiation of Consultation on the Coordinated Operations of the CVP and SWP (ROC ON LTO) (U.S. Fish and Wildlife Service 2019, National Marine Fisheries Service 2019a) and the Incidental Take Permit for Long-term Operations of the SWP in the Sacramento-San Joaquin Delta (State ITP) (California Department of Fish and Wildlife 2020) would occur. Regulatory criteria would also include a continuation of the State Water Board water rights and water quality criteria related to the CVP and SWP operations. Diversions would continue at the RBPP and GCID Main Canal at Hamilton City as they do under the NAA. Therefore, there would be no measurable change between the NAA and 2020 baseline conditions.

Significance Determination

For Impacts FISH-2 through FISH-17 and Impact FISH-19, the NAA would not result in operational effects on special-status fish or aquatic biological resources because there would be no measurable change from 2020 baseline conditions. There would be no impact/no effect.

Alternatives 1, 2, and 3

Effects of Alternatives 1, 2, and 3

Potential exposure of winter-run Chinook salmon to the effects of Alternatives 1, 2, and 3 depends on the species' spatiotemporal distribution. Several sources of information are available to inform potential exposure. General information on spatiotemporal distribution is presented in the species account in Appendix 11A. Appendix 11A also provides graphs from the SacPAS website (Columbia Basin Research 2020) that give a summary of historical patterns of juvenile winter-run capture at various sampling locations in the Sacramento River and Delta[6]. Generally, run affiliation is assigned based on length-at-date analysis developed by Fisher (1992), which hindcasts a spawning date for a particular fish based on date of capture and assumed growth rate. An 11-year study (1981–1991) of monthly sampling at 13 beach seine sites in the Sacramento River between RM 164 (just upstream of Princeton) and RM 298 (Redding) by Johnson et al. (1992) provides perhaps the best spatial information related to juvenile Chinook salmon occurrence upstream and downstream of the Red Bluff and Hamilton City diversions. A long-term (1981–2019) beach seine database provides temporal occurrence information for juvenile Chinook salmon between RM 71 (Elkhorn) and RM 144 (Colusa State Park). Estimates of adult Chinook salmon escapement by tributary from the GrandTab database (Azat 2021) give perspective on the relative proportion of the ESU that could experience near-field and far-field effects of the diversions for Alternatives 1, 2, and 3.

The main patterns of juvenile winter-run Chinook salmon occurrence at the locations documented in the SacPAS summary in Appendix 11A1, *Juvenile Salmonid Monitoring, Sampling, and Salvage Timing Summary from SacPAS*, include:

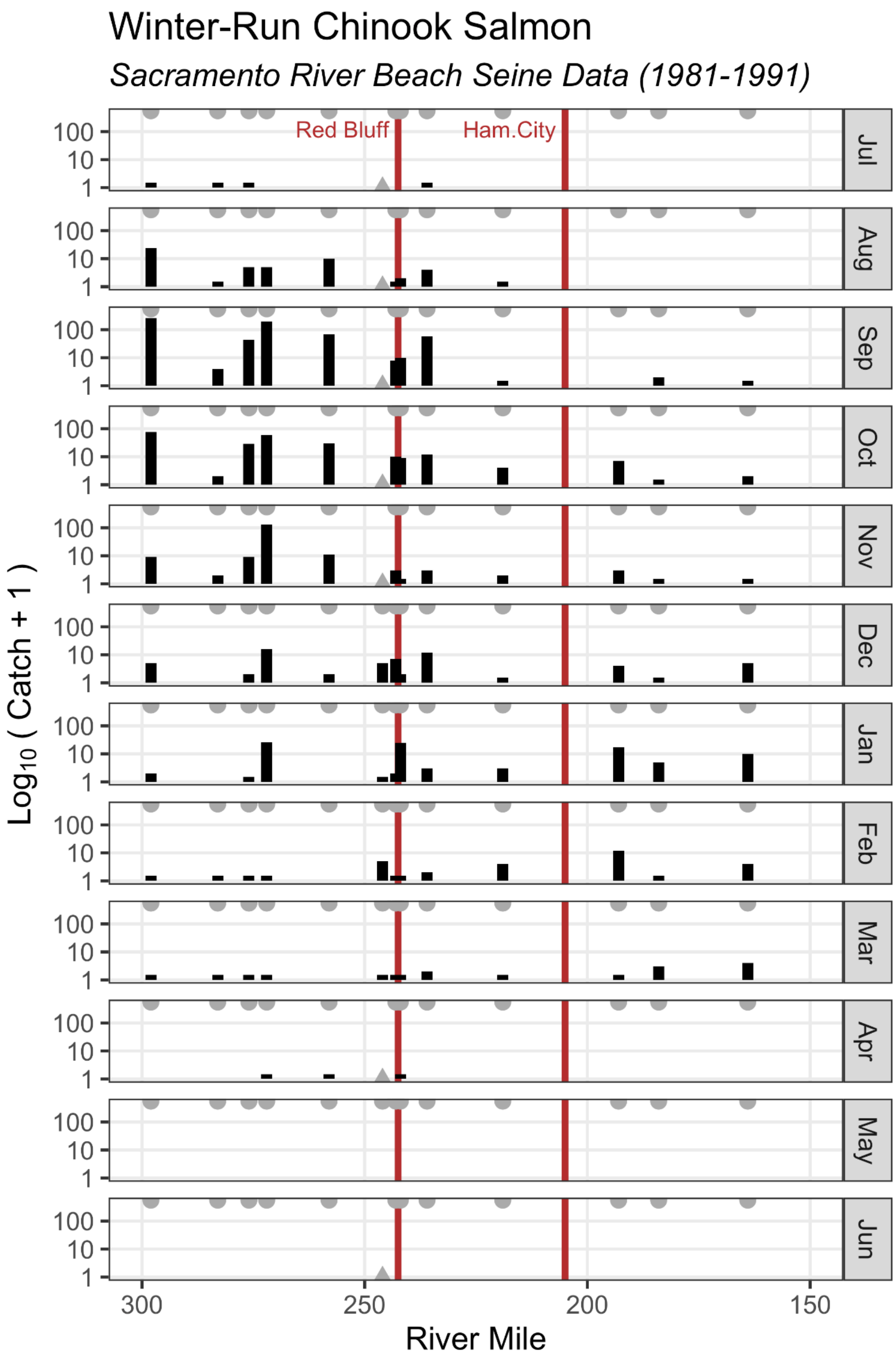
- RBDD rotary screw traps (Appendix 11A1, Figure 11A1-1): Passage begins in early July and ends from early April to late May, and rarely into June. The first half (50%) of the run passes within a 1-month period, late September to late October, whereas the main portion of the run (90%) begins passage through the diversion dam within a few days of September 1 and finishes anywhere from mid-October to as late as the end of December.
- Tisdale Weir rotary screw traps (Appendix 11A1, Figure 11A1-2): Passage begins in early September and lasts until the first of May. The passage of winter-run Chinook at this weir is dynamic: the first half (50%) of the run passes anywhere from the first of

[6] Note that sampling at some locations such as Tisdale Weir and Knights Landing has not always occurred during warmer months when handling of juvenile salmonids may lead to greater chance of mortality.

November to the first of March, whereas the main portion (90%) begins to pass the weir from early October to late November and ends anywhere from the end of December to early March.

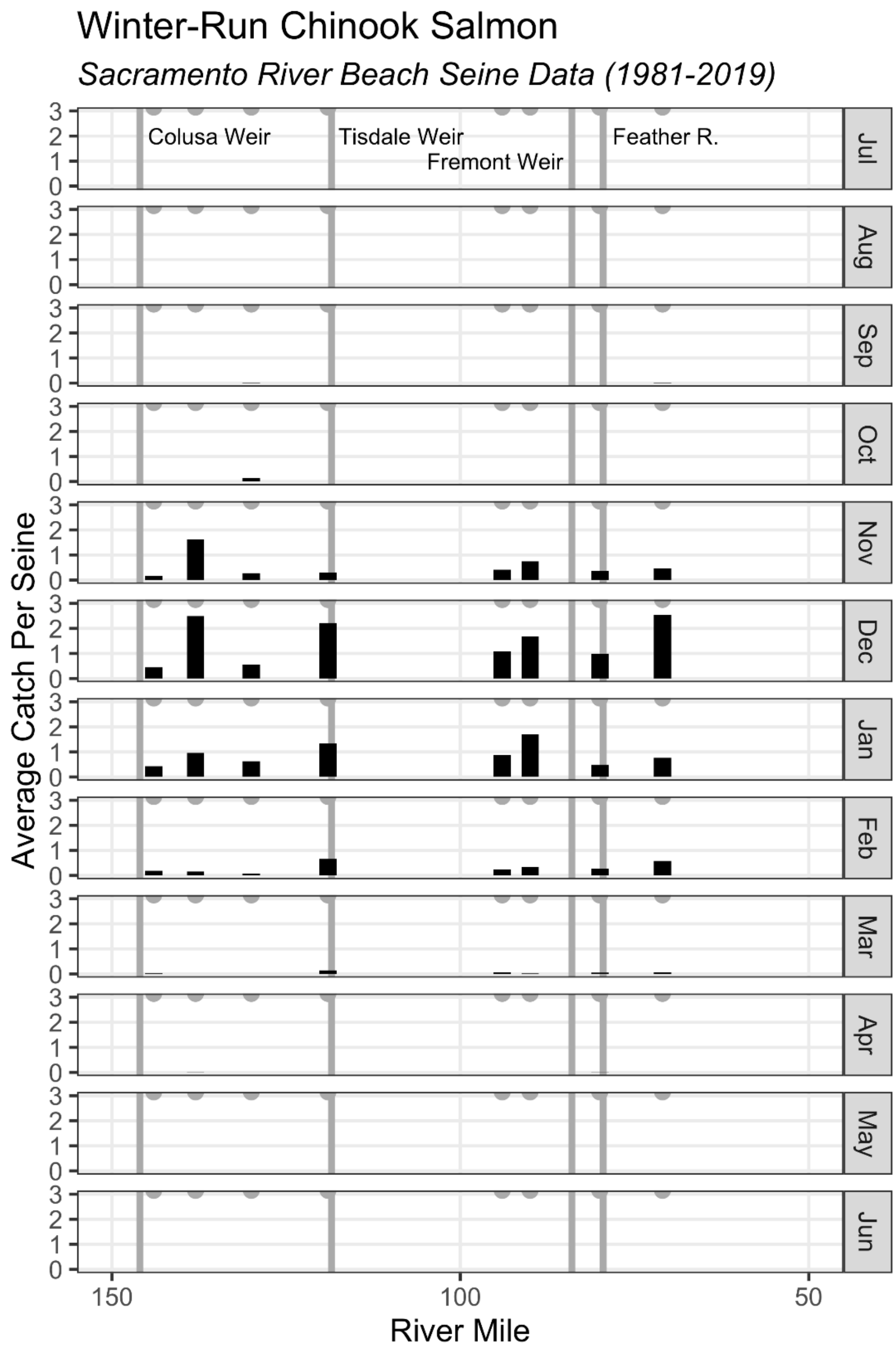
- Knights Landing rotary screw traps (Appendix 11A1, Figure 11A1-3): Passage begins in early to mid-September, though rarely as early as late August. The first half (50%) is variable, happening as early as the first of November or as late as early March. The bulk (90%) of the run begins to pass from early October to mid-early March, and can take only a few weeks to pass, with the bulk of the run finishing in mid-December to mid-March.
- Sacramento beach seines (Appendix 11A1, Figure 11A1-4): Occurrence can begin anywhere from mid-September to mid-December and can end anywhere from mid-January to the first of April. The first half (50%) occurs from early December, rarely mid-November, to the first of March, whereas the main portion (90%) of the run begins to occur in the seines in mid-October to mid-January and ends early January to early March.
- Sacramento trawls (Sherwood Harbor) (Appendix 11A1, Figure 11A1-5): Occurrence begins from early September to early March and ends from the first of March to the end of April. The bulk (50%) occurs anywhere from the end of November to the first of April, whereas the main portion (90%) is highly variable, beginning from late October to mid-March and finishing from Mid- February to early April.
- Chipps Island trawls (Appendix 11A1, Figure 11A1-6): Occurrence begins anywhere from the first of December to early March and finishes consistently from mid-April to the end of May. The first half (50%) occurs from early March to mid-April, whereas the main portion (90%) occurs in late December to late March and finish by late March to late April.
- Salvage (unclipped, length-at-date) (Appendix 11A1, Figure 11A1-7): Salvage begins in early December to early March and finishes from late March to late May. The first half (50%) occurs in early January, though in the bulk of years 50% occurs in March. The main portion of salvage (90%) begins in mid-December to mid-March and finishes mid-January to early April.
- Salvage (clipped, length-at-date) (Appendix 11A1, Figure 11A1-8): Salvage begins in early December to mid-January and finishes from late February to early May. The first half (50%) occurs from late December to early April, whereas the main portion (90%) begins in mid-December to mid-January and finishes from late February to early May. The 2013 cohort was the exception to these trends, having occurred entirely in approximately 2 weeks in mid-March.
- Salvage (clipped, CWT-Race) (Appendix 11A1, Figure 11A1-9): Salvage begins in mid-December to late January and finishes from mid-January to early May. The first half (50%) occurs from early January to mid-February, whereas the main portion (90%) begins in late January to early February and finishes from mid-January to mid-May. The 2013 cohort is an exception to these trends, having occurred entirely in March.

Juvenile winter-run Chinook salmon catches in extensive beach seining during 1981–1991 were more concentrated in the upper reaches of that study area during roughly August–October, spreading downstream in the winter months (Figure 11-3). The catch dropped to essentially zero from April to July in that data set. The pattern of catches spreading downstream in the fall and winter is consistent with the 1981–2019 beach seine data collected further downstream, where the catch rate was low until November, then peaked in December–January, before tapering off in February (Figure 11-4).



Source : Johnson et al. (1992:Table 4). Note: Sampled sites are denoted by grey semicircles at the top of each plot. Sites that were not sampled in a given month are denoted by grey triangles at the bottom of a panel. Values denoted as "<1" by Johnson et al. are shown as 0.5. Red lines indicate locations of Red Bluff and Hamilton City diversions.

Figure 11-3. Mean Monthly Catch Per Beach Seine of Juvenile Winter-Run Chinook Salmon in the Sacramento River Between River Mile 164 and River Mile 298, 1981–1991.



Source: Interagency Ecological Program et al. 2019.

Note: Sampled sites are denoted by grey semicircles at the top of each plot. Grey lines indicate locations of Colusa, Tisdale, and Fremont Weirs, and the confluence with the Feather River.

Figure 11-4. Mean Monthly Catch Per Beach Seine of Juvenile Winter-Run Chinook Salmon in the Sacramento River Between River Mile 71 and River Mile 144, 1981–2019.

All winter-run Chinook salmon spawning occurs upstream of Red Bluff (Azat 2021), so all juvenile winter-run migrating downstream would need to pass the two intake locations at Red Bluff and Hamilton City. Recent analyses based on otolith microchemistry of adults surviving to escape to spawning grounds in the Sacramento River suggest that many juvenile winter-run Chinook salmon rear within non-natal habitat: from 2007 to 2009, between 34% and 51% reared in the natal habitat of the Sacramento River, whereas between 17% and 26% reared in the American River, 7%–34% in tributaries of Mount Lassen (Battle, Mill, or Deer Creeks), and 7%–23% in the Feather River or Delta (Phillis et al. 2018). There is some uncertainty as to when the redistribution of juvenile winter-run Chinook salmon occurs to non-natal rearing habitats, and also to the relative survival within the different non-natal habitats, which could result in the proportion of juveniles surviving to adulthood from each habitat varying considerably from the proportion of juveniles entering each habitat.

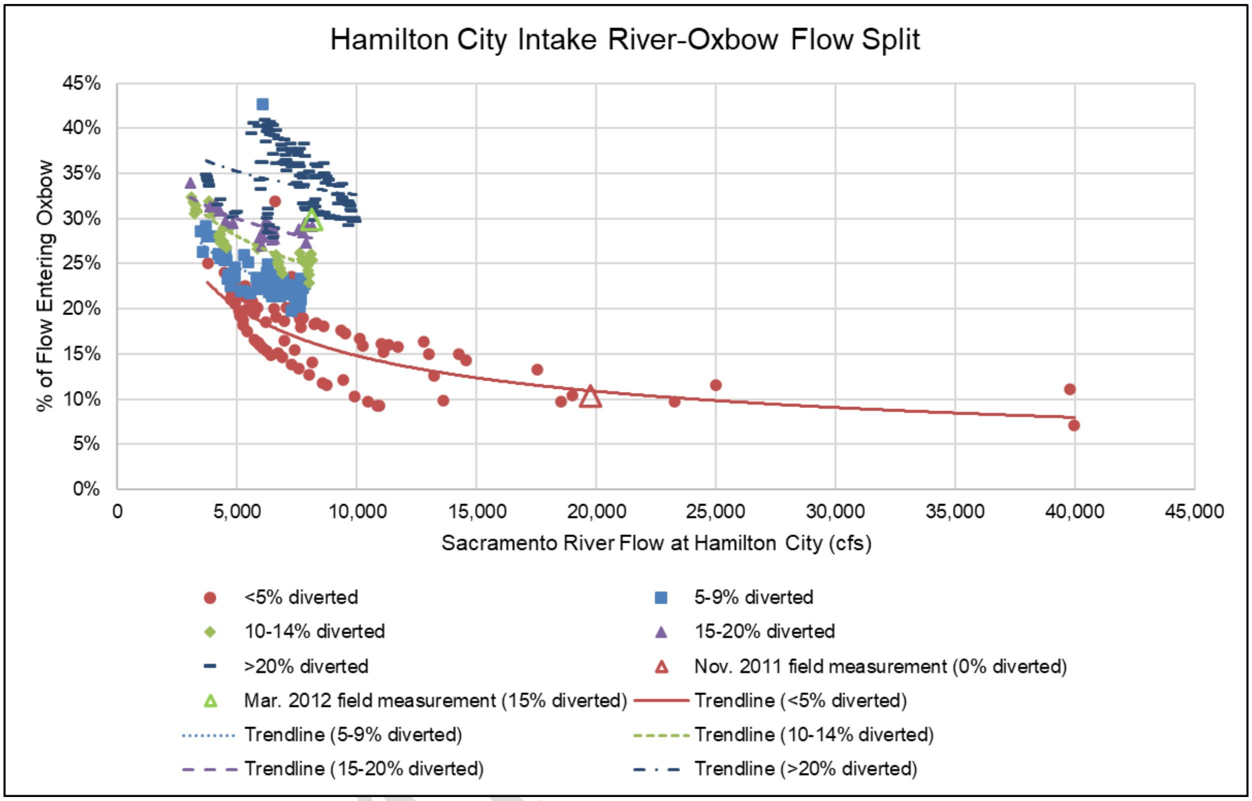
Sacramento River

Near-Field Effects

Distribution in River Channel—Screen Exposure

The spatial distribution of migrating fish within the Sacramento River channel is important for evaluating near-field effects of the Red Bluff and Hamilton City intakes. Juvenile salmonids, including winter-run Chinook salmon, must pass close to the intakes to have the potential for near-field effects from the intakes. Locating the intakes on the outside of river or oxbow channel bends to achieve sweeping velocity requirements may lead to a greater proportion of fish passing close to the intakes than if the fish were occurring evenly distributed across the channel cross section. Several studies on the Sacramento River provide evidence for the distribution of fish toward the outer sides of bends, including at Clarksburg Bend (Burau et al. 2007:Figure C.17), the DCC (Burau et al. 2007:Figure 2.5), and near Fremont Weir (Blake et al. 2017:Figures 2 and 20). The distribution of fish towards the outside of bends is the result of centrifugal and pressure forces in bends which induce a secondary flow that lies in a plane perpendicular to the primary flow direction (Dinehart and Burau 2005) and is reflected in the bathymetry of such areas: the deeper areas, including the thalweg, coincide with the areas subject to the secondary flow (Burau et al. 2007:Figure C.1). These observations agree with the general pattern of downstream-migrating juvenile salmonids in the Pacific northwest often being distributed near the thalweg, or near the shoreline (Smith et al. 2009). It is possible that a relatively large proportion of downstream-migrating juvenile salmonids could pass relatively close to the Red Bluff and Hamilton City intakes, particularly during nighttime periods when most migration occurs (Chapman et al. 2013; Zajanc et al. 2013; Plumb et al. 2016). However, when holding (e.g., during the day), juvenile salmonids could also occur on the inside of river bends, as illustrated at Clarksburg Bend (Burau et al. 2007:Figure C.15).

The Hamilton City intake is on an oxbow of the Sacramento River, with available studies suggesting that the percentage of river flow entering the oxbow varies based on river flow and percentage of river diverted at the intake (Figure 11-5). Note that the data shown in Figure 11-5 reflect a partially obstructed (15%) oxbow channel, which would tend to reduce the percentage of river flow entering the oxbow, as suggested by comparison of the trendlines based on 2018 daily data to two field measurements collected in 2011/2012 (triangles on graph). On the basis of the diversion rates for Alternatives 1, 2, and 3, and assuming that fish are generally moving into the oxbow at a similar or slightly lower percentage as the flow split (Cavallo et al. 2015), it would be expected that approximately 10%–30% of downstream-migrating juvenile salmonids approaching the river-oxbow split would enter the oxbow and have the potential to be exposed to the Hamilton City intake screen. There is some uncertainty in this estimate given that diversion rate data at higher flow (i.e., >10,000 cfs at Hamilton City) generally have been limited to less than 5% of the flow (Figure 11-5).



Source: Developed by ICF based on data provided by Kline (pers. comm.). Notes: Circles indicate daily gage data. During 2018, the oxbow was 15% obstructed, which would tend to somewhat lower the percentage of flow entering it compared to an unobstructed channel. Triangles indicate field measurements from 2011/2012 (McMillen 2013). Trendlines are power functions generated in Microsoft Excel.

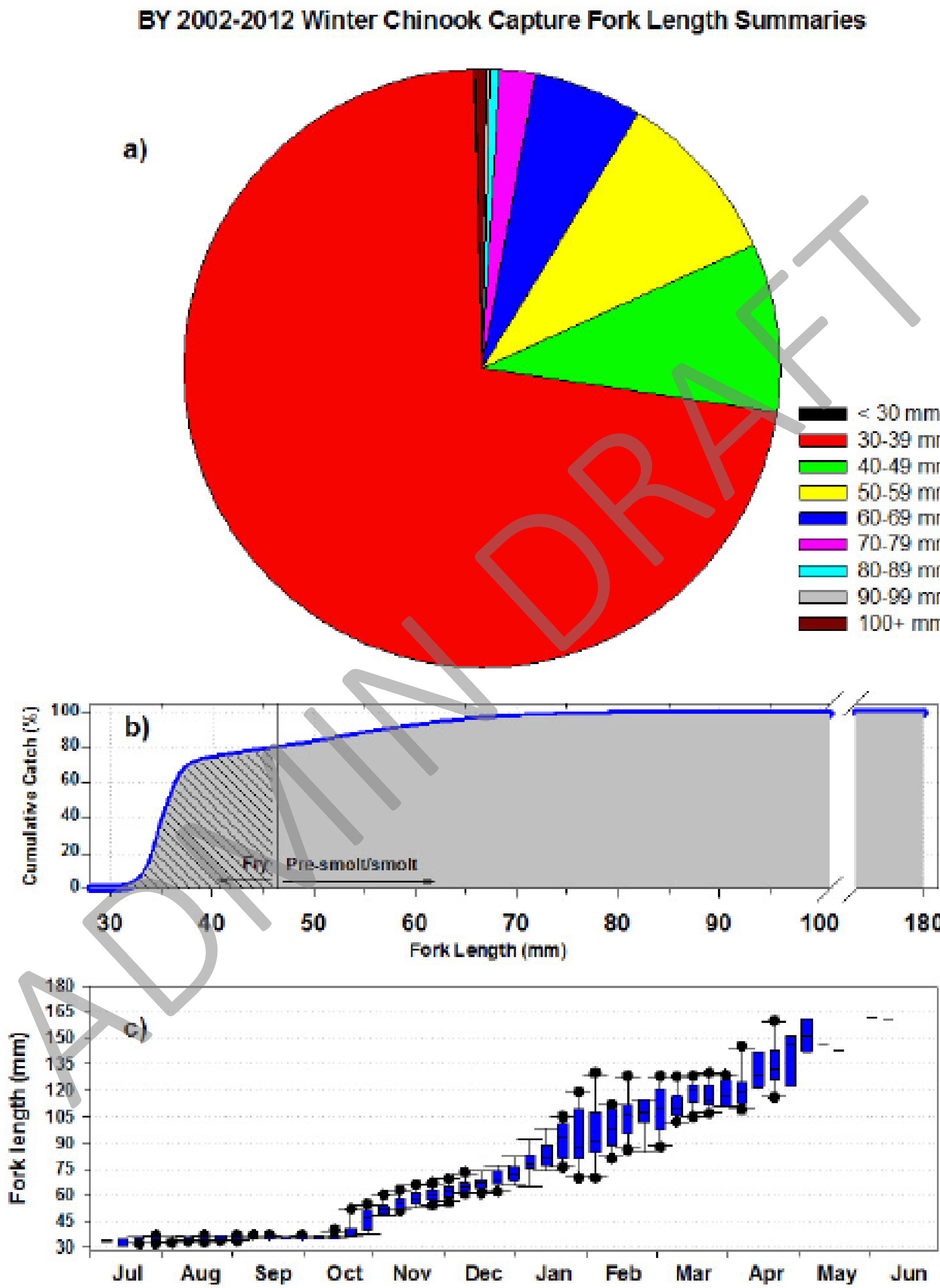
Figure 11-5. Daily Percentage of Sacramento River Flow Entering the Oxbow Containing the Hamilton City Intake, 2018, Divided into Five Groups Based on Percentage of Hamilton City River Flow Diverted by the Intake.

In addition to horizontal distribution, an important consideration for potential screen exposure of juvenile salmonids, including winter-run Chinook salmon, is the vertical distribution of fish in relation to the Project's intakes. In general, migrating juvenile salmonids are located in the upper portion of the water column (Smith et al. 2009). This was illustrated locally in the hydroacoustic study near the DCC, for which fish were particularly abundant between around 4 to 7 meters (13–23 feet) below the surface of the 13-meter (43-foot) deep water column (Blake and Horn 2006:Figure 41) (i.e., greatest abundance was at about 33%–50% of the water column). Preliminary information from annual stage frequency curves at Red Bluff (Kapla pers. comm.) suggests that juvenile Chinook salmon occurring at between 33%–50% of the water column depth could be at a similar depth to the intake screen much of the time. This preliminary stage frequency information includes all months of the year and will be refined to focus only on the main months of the diversions (i.e., November–April) for Alternatives 1, 2, and 3, as well as to include information for the Hamilton City intake (Kapla pers. comm.).

Potential exposure of juvenile salmonids to the Red Bluff and Hamilton City fish screens would be addressed by technical studies focused on diversions at these locations during high winter flow conditions when Project diversions would occur (Appendix 2D).

Entrainment Through Screens

Juvenile salmonids, including winter-run Chinook salmon, migrating downstream past the Red Bluff and Hamilton City intakes would be susceptible to entrainment through the fish screens if sufficiently small. Calculations suggest that a 1.75-mm opening would be effective at excluding juvenile salmonids of 22-mm standard length and greater (ICF International 2016:5-103), which is around 25-mm fork length (FL). The most comprehensive assessment of juvenile salmonid length distribution in the vicinity of the Red Bluff and Hamilton City intakes is from sampling at the RBDD rotary screw traps (Figure 11-6). These data suggest a small proportion of juvenile winter-run Chinook salmon would be of sufficiently small size to be susceptible to entrainment by the intakes' diversions if occurring at the faces of the fish screens, given that few fish were less than 30-mm FL. The small proportion of juvenile winter-run sufficiently small-sized to potentially be susceptible to entrainment occurred in July/August (Figure 11-6), a period during which the diversions for Alternatives 1, 2, and 3 generally would be similar—or in some cases somewhat lower—in magnitude to the NAA (Tables 11-6 and 11-7). Entrainment risk would be expected to be similar between the NAA and Alternatives 1, 2, and 3 for juvenile winter-run Chinook salmon. Entrainment could be monitored at the Red Bluff and Hamilton City intakes during high winter flow conditions when Project diversions would occur (Appendix 2D).



Source: Poytress et al. 2014:84. Note: BY = brood year.

Figure 11-6. Winter-Run Chinook Salmon Fork Length (a) Capture Proportions, (b) Cumulative Capture Size Curve, and (c) Average Weekly Median Boxplots, As Sampled at Red Bluff Diversion Dam Rotary Screw Traps, July 2002–June 2013.

Table 11-6. Red Bluff Diversion as Percentage of Sacramento River Flow, Averaged by Month and Water Year Type, from CALSIM Modeling.

Water Year Type	Month	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	Jan	0%	3%	4%	3%	4%
Wet	Feb	0%	3%	3%	3%	4%
Wet	Mar	0%	3%	4%	3%	4%
Wet	Apr	2%	4%	4%	4%	4%
Wet	May	6%	7%	7%	7%	7%
Wet	Jun	11%	11%	11%	11%	11%
Wet	Jul	10%	10%	10%	10%	10%
Wet	Aug	9%	9%	9%	9%	9%
Wet	Sep	2%	2%	2%	2%	2%
Wet	Oct	2%	2%	2%	2%	2%
Wet	Nov	0%	2%	2%	2%	2%
Wet	Dec	0%	2%	2%	2%	2%
Above Normal	Jan	0%	8%	8%	8%	8%
Above Normal	Feb	0%	8%	8%	8%	8%
Above Normal	Mar	0%	6%	6%	6%	6%
Above Normal	Apr	4%	5%	5%	5%	6%
Above Normal	May	7%	7%	7%	7%	7%
Above Normal	Jun	11%	11%	8%	11%	8%
Above Normal	Jul	9%	9%	7%	9%	4%
Above Normal	Aug	9%	9%	9%	9%	6%
Above Normal	Sep	3%	3%	3%	3%	2%
Above Normal	Oct	2%	2%	2%	2%	2%
Above Normal	Nov	0%	1%	1%	1%	1%
Above Normal	Dec	0%	4%	5%	4%	5%
Below Normal	Jan	0%	4%	4%	4%	4%
Below Normal	Feb	0%	5%	6%	5%	6%
Below Normal	Mar	0%	5%	5%	5%	6%
Below Normal	Apr	5%	6%	5%	6%	5%
Below Normal	May	7%	7%	6%	7%	6%
Below Normal	Jun	9%	9%	7%	9%	6%
Below Normal	Jul	9%	9%	8%	9%	7%
Below Normal	Aug	9%	9%	9%	9%	8%
Below Normal	Sep	3%	3%	3%	3%	3%
Below Normal	Oct	2%	2%	2%	2%	2%
Below Normal	Nov	0%	2%	2%	2%	2%
Below Normal	Dec	0%	5%	5%	5%	5%
Dry	Jan	0%	1%	1%	1%	1%
Dry	Feb	0%	4%	4%	4%	4%

Water Year Type	Month	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Dry	Mar	1%	7%	6%	7%	7%
Dry	Apr	4%	4%	3%	4%	3%
Dry	May	5%	5%	4%	5%	4%
Dry	Jun	6%	6%	5%	6%	4%
Dry	Jul	7%	7%	7%	7%	6%
Dry	Aug	7%	7%	7%	7%	7%
Dry	Sep	3%	3%	3%	3%	4%
Dry	Oct	2%	2%	2%	2%	2%
Dry	Nov	0%	1%	1%	1%	1%
Dry	Dec	0%	3%	3%	3%	3%
Critically Dry	Jan	0%	2%	2%	2%	2%
Critically Dry	Feb	0%	3%	3%	3%	3%
Critically Dry	Mar	0%	3%	3%	3%	3%
Critically Dry	Apr	1%	1%	1%	1%	1%
Critically Dry	May	1%	1%	1%	1%	1%
Critically Dry	Jun	1%	1%	1%	1%	1%
Critically Dry	Jul	1%	1%	1%	1%	1%
Critically Dry	Aug	1%	1%	1%	1%	1%
Critically Dry	Sep	1%	1%	1%	1%	1%
Critically Dry	Oct	1%	1%	1%	1%	1%
Critically Dry	Nov	0%	0%	0%	0%	0%
Critically Dry	Dec	0%	0%	0%	0%	0%

Note: Percentage is calculated based on diversion (CALSIM node D112) divided by (diversion at D112 + Sacramento River below Red Bluff Diversion Dam flow [CALSIM channel C112]).

Table 11-7. Hamilton City Diversion as Percentage of Sacramento River Flow, Averaged by Month and Water Year Type, from CALSIM Modeling.

Water Year Type	Month	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	Jan	0%	2%	2%	2%	2%
Wet	Feb	0%	2%	2%	1%	2%
Wet	Mar	0%	1%	1%	1%	1%
Wet	Apr	3%	5%	5%	5%	6%
Wet	May	17%	18%	17%	18%	17%
Wet	Jun	22%	22%	22%	22%	22%
Wet	Jul	21%	21%	21%	21%	21%
Wet	Aug	21%	21%	21%	21%	21%
Wet	Sep	6%	6%	7%	6%	7%
Wet	Oct	7%	7%	7%	7%	7%
Wet	Nov	8%	8%	8%	8%	8%
Wet	Dec	3%	2%	3%	2%	3%

Water Year Type	Month	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Above Normal	Jan	1%	3%	3%	3%	3%
Above Normal	Feb	0%	2%	2%	2%	2%
Above Normal	Mar	0%	3%	3%	2%	3%
Above Normal	Apr	4%	6%	6%	6%	6%
Above Normal	May	19%	19%	19%	19%	19%
Above Normal	Jun	23%	23%	22%	23%	20%
Above Normal	Jul	19%	19%	18%	19%	12%
Above Normal	Aug	20%	20%	20%	20%	16%
Above Normal	Sep	7%	6%	7%	6%	6%
Above Normal	Oct	8%	8%	8%	8%	8%
Above Normal	Nov	8%	8%	8%	8%	8%
Above Normal	Dec	2%	3%	3%	3%	4%
Below Normal	Jan	1%	2%	2%	2%	2%
Below Normal	Feb	1%	1%	1%	1%	1%
Below Normal	Mar	0%	1%	1%	1%	1%
Below Normal	Apr	6%	7%	7%	7%	7%
Below Normal	May	24%	24%	22%	24%	22%
Below Normal	Jun	24%	24%	24%	24%	21%
Below Normal	Jul	22%	22%	21%	22%	19%
Below Normal	Aug	22%	21%	22%	21%	20%
Below Normal	Sep	10%	11%	11%	11%	11%
Below Normal	Oct	9%	9%	9%	9%	8%
Below Normal	Nov	9%	9%	9%	9%	8%
Below Normal	Dec	2%	3%	3%	3%	3%
Dry	Jan	1%	2%	2%	2%	2%
Dry	Feb	1%	2%	2%	2%	2%
Dry	Mar	0%	1%	1%	1%	1%
Dry	Apr	6%	6%	6%	6%	6%
Dry	May	24%	23%	22%	23%	21%
Dry	Jun	24%	24%	24%	24%	21%
Dry	Jul	22%	18%	18%	18%	17%
Dry	Aug	21%	15%	17%	15%	17%
Dry	Sep	11%	8%	8%	8%	8%
Dry	Oct	10%	8%	9%	8%	9%
Dry	Nov	10%	9%	9%	9%	9%
Dry	Dec	3%	4%	4%	4%	4%
Critically Dry	Jan	1%	2%	2%	2%	2%
Critically Dry	Feb	1%	1%	1%	1%	1%
Critically Dry	Mar	1%	1%	1%	1%	1%
Critically Dry	Apr	8%	9%	9%	9%	8%

Water Year Type	Month	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Critically Dry	May	24%	24%	24%	24%	22%
Critically Dry	Jun	24%	22%	22%	22%	22%
Critically Dry	Jul	23%	19%	19%	19%	19%
Critically Dry	Aug	23%	17%	17%	18%	20%
Critically Dry	Sep	11%	8%	9%	9%	10%
Critically Dry	Oct	8%	8%	7%	8%	7%
Critically Dry	Nov	10%	10%	10%	10%	10%
Critically Dry	Dec	3%	3%	3%	3%	3%

Note: Percentage is calculated based on diversion (CALSIM node D114) divided by (diversion at D114 + Sacramento River at Hamilton City flow [CALSIM channel C114]).

Impingement, Screen Contact, and Screen Passage

Juvenile salmonids, including winter-run Chinook salmon, could be injured or die after coming into contact with the Red Bluff and Hamilton City intake fish screens. The best available information to inform this risk for juvenile Chinook salmon comes from the laboratory-based study of Swanson et al. (2004) undertaken at the fish treadmill facility at the University of California in Davis. These authors found that juvenile Chinook salmon experienced frequent contact with the simulated fish screen but were rarely impinged (defined as prolonged screen contacts >2.5 minutes) and impingement was not related to any of the experimental variables examined. Swanson et al. (2004:274) noted:

> The injury rates of both preexperiment and experimental fish were generally high but most injuries consisted of minor damage to fins and scales. Among the four treatments, significant differences in injury indices were apparently related to the duration of laboratory holding, with larger, older fish exhibiting more damage... Within treatments, the injury index was not significantly affected by either flow regime or screen contact rate (regression and correlation, $P > 0.3$, all tests) and, in general, preexperimental indices were similar to those measured for fish after exposure in the Fish Treadmill.

Of the more than 3,200 fish tested, only five fish from four experiments died during the experiment and one fish, from a fifth experiment, during the 48-hour post-experiment period. Two of the mortalities were from daytime experiments and four were from nighttime experiments. All mortalities were from flow treatments with a sweeping flow component, but the small number precluded the detection of significant flow effects on survival. The death of these fish did not appear to be related to observed impingements.

Swanson et al. (2004) concluded that fish screen designs that minimize screen exposure duration (e.g., via reduced screen length or increased sweeping velocities) should optimally protect juvenile Chinook salmon. To inform potential juvenile Chinook salmon screen passage time past the Red Bluff and Hamilton City intakes, the methods applied in the California WaterFix Biological Assessment (ICF International 2016:Appendix 5.D, 5.D-1) based on Swanson et al. (2004:272) were used. In addition, to estimate potential number of screen contacts for juvenile Chinook salmon, the daytime and nighttime contact rate equations from Swanson et al. (2004:272) were used:

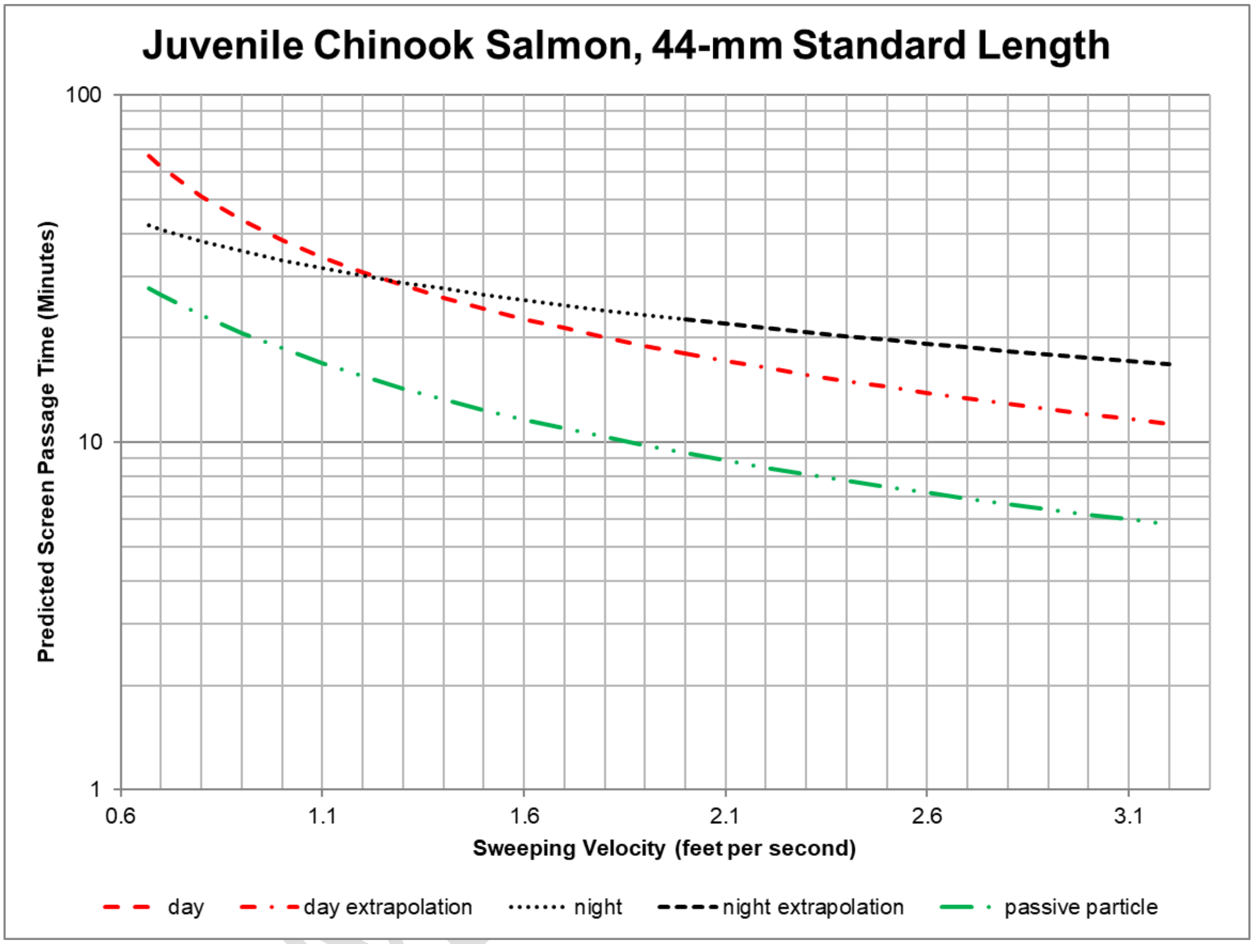
Contact rate (daytime) = 0.158(SL) - 0.008(SWP) - 0.006(D) - 0.001(SL*T)

Contact rate (nighttime) = 0.146(SL)

where SL = standard length (calculated for 4.4-cm [44-mm] and 7.9-cm [79-mm] fish, the minimum and maximum tested), SWP = sweeping velocity (cm/s), D = distance from screen (assumed to be 31 cm, which is halfway across the test channel), T = temperature (°C, assumed to be 12°C). The contact rates (contacts/minute) were multiplied by the screen passage time to give the estimated total number of contacts.

For this assessment, the analyses based on Swanson et al. (2004) were applied to total fish screen lengths of 1,100 feet (i.e., representative of the Red Bluff and Hamilton City intakes; for a range of sweeping velocities from 0.67 to 3.2 feet per second (ft/s); this range covers the lowest sweeping velocity at which diversions could occur (i.e., two times the allowable approach velocity of 0.33 ft/s) up to the estimated surface water velocity during a site visit to the Sacramento River near Maxwell in February 2019 when river flow was approximately 25,000 cfs. The extent to which the relatively benign experimental environment of Swanson et al. (2004) is representative of Sacramento River conditions is uncertain, but the Red Bluff and Hamilton City fish screens have a smooth screen surface and frequent screen cleaning, which would provide additional protection to minimize screen surface impingement of fish, including winter-run Chinook salmon. The smooth surface also would serve to reduce the risk of abrasion and scale loss for any fish that does come into contact with the screens (Swanson et al. 2004). Note also that Swanson et al. (2004) only tested sweeping velocity up to 2 ft/s, so there is added uncertainty in extrapolations above this range. It is important to recognize that the results of these analyses are only relevant to fish passing close to the screens, as the test facility under which the observations were made had a flume width of 1.2 meters (3.9 feet; Swanson et al. 2004).

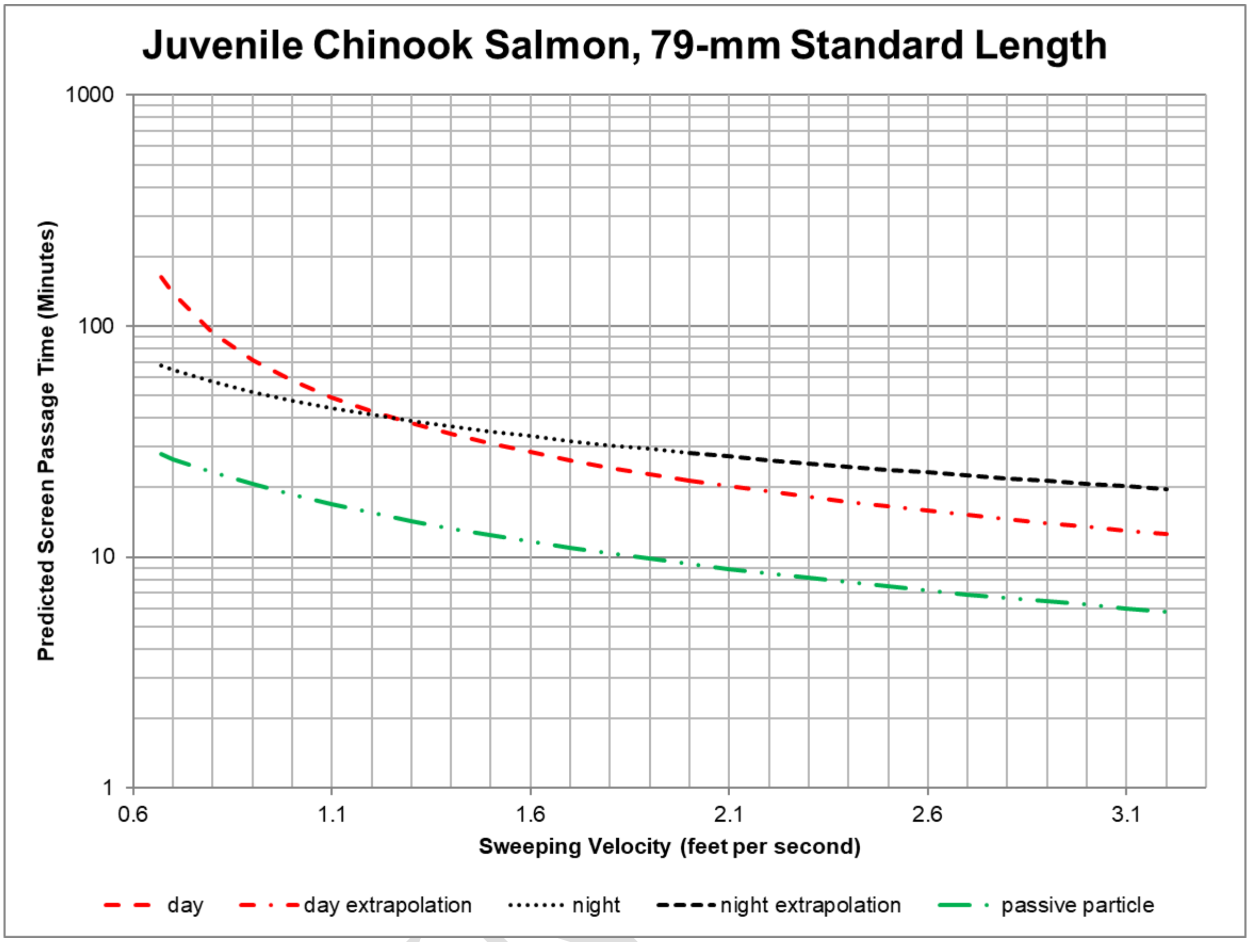
The results of the screen passage time analyses based on equations presented by Swanson et al. (2004) suggest that 44-mm juvenile Chinook salmon screen passage time could range from just over 10 minutes at 3.2-ft/s sweeping velocity by day to around 67 minutes at 0.67-ft/s sweeping velocity by day (Figure 11-7). These estimates are appreciably longer than the rates of passage of passive particles linearly moving past the intakes without being entrained, for which the estimates based on screen length divided by sweeping velocity range from less than 6 minutes to nearly 28 minutes (Figure 11-7). Relative to passive particles, longer passage times for fish calculated based on laboratory experiments reflect the swimming response of the tested juvenile Chinook salmon, which was generally rheotactic (i.e., swimming against the prevailing current). Based on the equations in Swanson et al. (2004), the screen passage time of 79-mm juvenile Chinook salmon was estimated to range from 12.6 minutes at 3.2-ft/s sweeping velocity during the day to over 160 minutes at 0.67-ft/s sweeping velocity during the day (Figure 11-8). Swanson et al. (2004) found that at warmer temperatures (19°C), larger juvenile Chinook salmon had a greater tendency to move downstream with the current (i.e., negative rheotaxis), consistent with a behavioral shift to outmigration; this would result in considerably lower screen passage times. This appears to be consistent with studies of PIT-tagged juvenile Chinook salmon undertaken at the Hamilton City intake[7] in July 1995, which indicated that median passage time was generally similar to sweeping velocity (Vogel and Marine 1995:Figure 21).



Note: The total screen length for the Red Bluff and Hamilton City fish screens is approximately 1,100 feet. Plot only includes mean responses and does not consider model uncertainty. 'Extrapolation' indicates range of predictions beyond range of laboratory-tested sweeping velocities. Passive particle passage time was calculated as screen length divided by sweeping velocity.

Figure 11-7. Predicted Screen Passage Time for Juvenile Chinook Salmon (44-mm Standard Length) Encountering the Red Bluff and Hamilton City Fish Screens at Approach Velocity of 0.33 Feet per Second During the Day and Night.

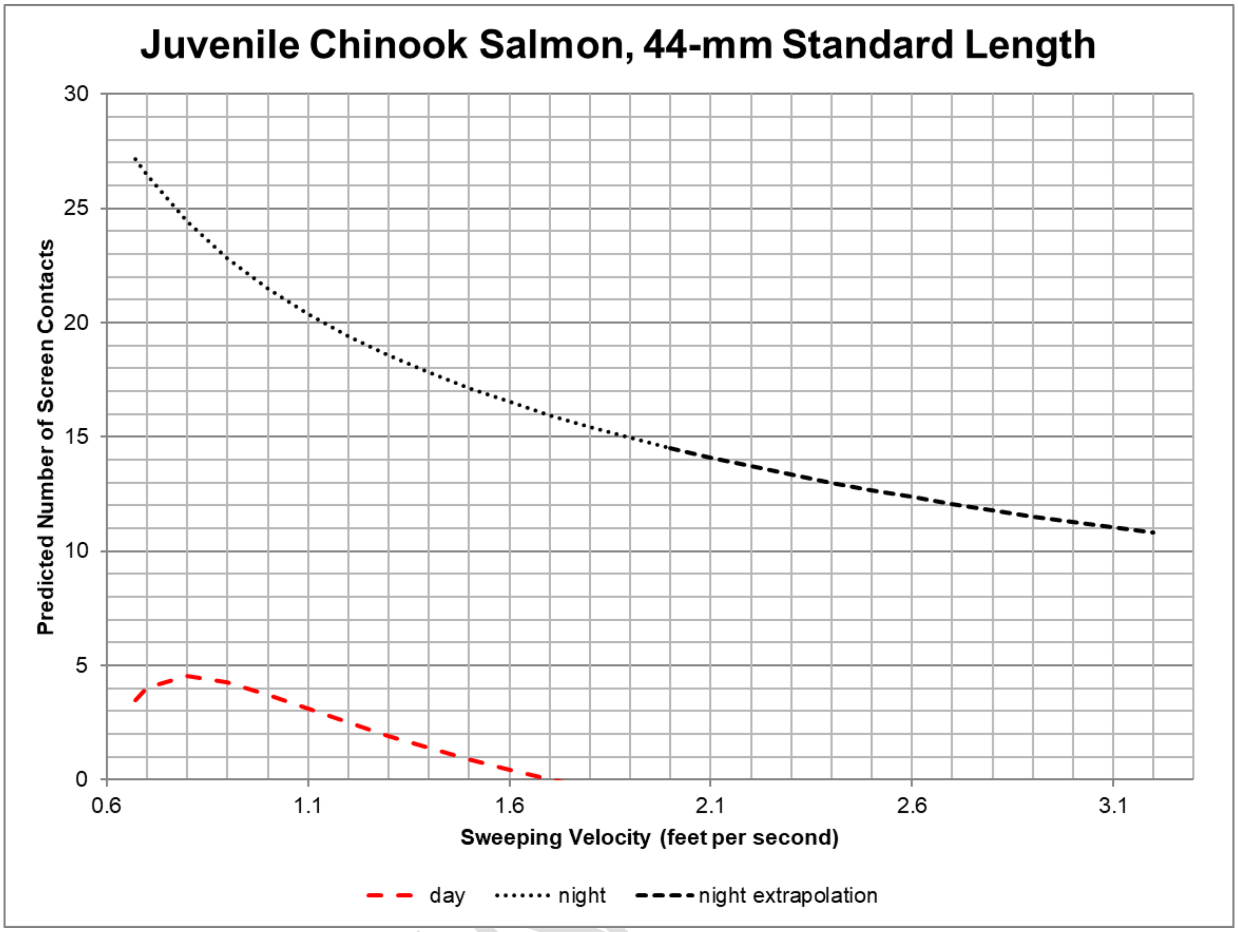
[7] This study occurred when the intake screens were just over 800 feet long, prior to screen lengthening to current 1,100-foot length, but remains useful for assessing screen passage rate.



Note: The total screen length for the Red Bluff and Hamilton City fish screens is approximately 1,100 feet. Plot only includes mean responses and does not consider model uncertainty. 'Extrapolation' indicates range of predictions beyond range of laboratory-tested sweeping velocities. Passive particle passage time was calculated as screen length divided by sweeping velocity.

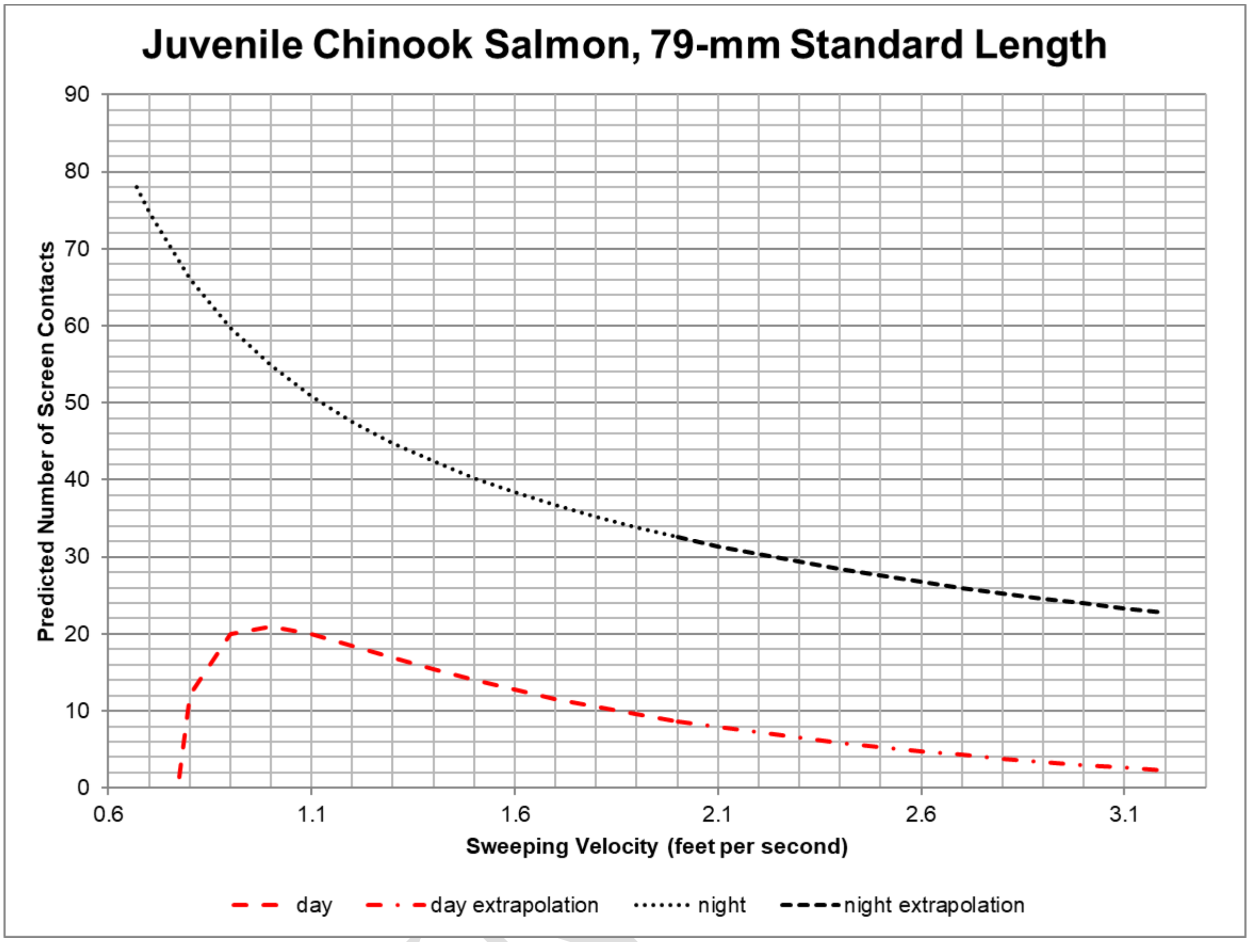
Figure 11-8. Predicted Screen Passage Time for Juvenile Chinook Salmon (79-mm Standard Length) Encountering the Red Bluff and Hamilton City Fish Screens at Approach Velocity of 0.33 Feet per Second During the Day and Night.

The results of the screen contact analysis based on the equations from Swanson et al. (2004) suggested that the total number of screen contacts for juvenile Chinook salmon could range from 0 contacts to nearly 80 contacts, depending on fish size and sweeping velocity (Figures 11-9 and 11-10).



Note: The total screen length for the Red Bluff and Hamilton City fish screens is approximately 1,100 feet. Plot only includes mean responses and does not consider model uncertainty. 'Extrapolation' indicates range of predictions beyond range of laboratory-tested sweeping velocities.

Figure 11-9. Predicted Number of Screen Contacts for Juvenile Chinook Salmon (44-mm Standard Length) Encountering the Red Bluff and Hamilton City Fish Screens at Approach Velocity of 0.33 Feet per Second During the Day and Night.



Note: The total screen length for the Red Bluff and Hamilton City fish screens is approximately 1,100 feet. Plot only includes mean responses and does not consider model uncertainty. 'Extrapolation' indicates range of predictions beyond range of laboratory-tested sweeping velocities.

Figure 11-10. Predicted Number of Screen Contacts for Juvenile Chinook Salmon (79-mm Standard Length) Encountering the Red Bluff and Hamilton City Fish Screens at Approach Velocity of 0.33 Feet per Second During the Day and Night.

Examination of the literature related to impingement does not appear to yield any additional information that would allow refinement of the quantitative results from Swanson et al. (2004), (i.e., that injury could not easily be related to velocity or other variables). Thomas et al. (1969) found that avoidance of impingement on screens situated perpendicular to 0.8-ft/s sweeping velocity flow (no approach velocity was given) by swim-up fall-run Chinook salmon increased to ~90% or more of individuals with 70%–80% absorption of the yolk sac. Avoidance of impingement decreased to ~70% of individuals as 100% of the yolk sac was absorbed, and as the fish got older (approaching 100 days post fertilization or more) avoidance of impingement returned to >90%. Thomas et al. (1969) suggested that the reduced swimming ability at or shortly after complete yolk sac absorption may be an important factor in the timing of downstream migration and survival of juvenile emigrants. Injury and mortality at vertical screens (1.75-mm opening) guiding fish to juvenile fish passage systems were examined for juvenile salmonids at John Day Dam (Brege et al. 2005). Note that these screens consist of a quite different configuration than the Red Bluff or Hamilton City intakes, as they guide fish upward toward the bypass orifice (Brege et al. 2005:Figure 2) and flow and turbulence are relatively high (Brege et al. 2005:19). Results for yearling Chinook salmon in 2002 found low mortality (0.1%) and injury (descaling; 4%). Additional testing for yearling Chinook salmon in 2004 with various vertical screen types to different prototype gate slots found that injury (descaling) ranged from 0.0% to 11.3% and mortality ranged from 0.0% to 10.4%. In 2004, subyearling Chinook salmon descaling ranged from 0.0% to 5.6% and mortality ranged from 0.0% to 11.9% (Brege et al. 2005). Zydlewski and Johnson (2002) found that vertical screens (1.75-mm opening) with approach velocity not exceeding 0.33 ft/s impinged around 12% of bull trout fry (median total length = 25 mm), but that all fry survived 24 h after the experiments were completed. Overall, considering the screen-related mortality data for juvenile Chinook salmon, the mean mortality from the studies of Swanson et al. (2004) and Brege et al. (2005) was 4.1% (median 1.2%, range 0%–45.5%). The mean mortality weighted by the number of fish in each test was 1.4%.

None of the references citing Swanson et al. (2004) provided additional information on impingement-related injury/mortality beyond those listed above or found within Swanson et al. (2004). Note that Swanson et al. (2004:266) stated: "…Hanson and Li (1978) reported that a number of fishes, including juvenile Chinook salmon, contacted a simulated fish screen or become impinged at velocities substantially below their measured swimming abilities." However, examination of Hanson and Li (1978) shows that those authors in fact had made a similar statement ("it has been shown that fish are impinged at velocities substantially lower than those in forced swimming performance trials") without themselves undertaking such studies or providing specific references to which studies were meant.

Limited examples exist for the way existing information has been used to inform effects analyses in the Central Valley and other areas. Cramer et al. (2005) assessed potential fish benefits from screening various Sacramento River diversions, and assumed various screen encounter mortality rates ranging from 2% (an unsedimented screen condition) up to 30% mortality (as a result of higher approach velocity caused by sedimentation and reduction in surface area). Walters et al. (2012) assumed 99% survival per screen encountered for juvenile Chinook salmon emigrating from the heavily irrigated Lemhi River (Idaho) watershed, based on the study of Swanson et al. (2004), but also noted that there could be additional mortality because of other factors such as predation (discussed herein below).

The available information generally suggests that impingement and screen passage/contact-related negative effects of the operation of the Red Bluff and Hamilton City intakes would be limited, particularly given that these effects would only apply to the subset of juvenile winter-run Chinook salmon encountering the intakes. The Red Bluff and Hamilton City fish screens are designed to protective standards for Chinook salmon fry and so near-field effects would be expected to be limited. Impingement could be monitored at the Red Bluff and Hamilton City intakes during high winter flow conditions when Project diversions would occur (Appendix 2D).

Predation

Red Bluff and Hamilton City Intakes

Predation of juvenile salmonids at the Red Bluff and Hamilton City intakes could occur if predatory fish aggregate along the screens. Such aggregation has been previously observed at the Hamilton City intake (Vogel 2008). This represents the only completed study of predation along

long fish screens in the Central Valley that involved calculation of survival along the fish screen based on recapture of marked juvenile Chinook salmon released from several locations. Vogel's (2008) study found that mean survival of tagged juvenile Chinook salmon at the Hamilton City intake in 2007—this being the only year of the study in which flow control blocks at the weir at the downstream end of the fish screen were removed, to reduce predatory fish concentration—was approximately 95% along the fish screen. However, the percentage of tagged juvenile Chinook salmon released at the upstream end of the fish screen that were recaptured at a downstream sampling location was similar or slightly greater than for fish released at the downstream end of the fish screen, when standardized for the distance that the fish had to travel to the recapture site (Table 11-8). These data suggest that survival along the screen was at least similar to survival in the portion of the channel without the screen (i.e., screen survival was similar to baseline survival, if the latter is assumed to be represented by the channel downstream of the screen). However, test fish providing the estimate of survival in the channel downstream of the screen were released prior to the fish that were released at the upstream end of the fish screen, which could have confounded comparisons of relative survival between these groups if predatory fishes became partly satiated prior to the arrival of the fish released at the upstream end of the screen (thus potentially making their survival relatively higher than otherwise would have occurred) (Vogel 2008:12). In addition, Vogel (2008:20) cautioned:

> …the fish mark/recapture survival experiments probably result in higher survival estimates than would be expected to occur for wild juvenile salmon migrating past the site (Vogel 2007). This circumstance is attributable to the fact that wild fish exhibit a more-protracted migration timing and do not migrate en masse like the simultaneous release of hundreds of marked hatchery fish for the short-term survival experiments. Predatory fish in the GCID oxbow channel could more readily consume greater numbers of wild fish "trickling" downstream through the oxbow as compared to an instantaneous release of hundreds of juvenile salmon that move rapidly past the site during a very short period. The acoustic telemetry experiments [undertaken in 2005] provided some empirical evidence of this phenomenon. Seasonally, large numbers of wild fish enter the oxbow inlet channel and become concentrated into a lesser amount of flow by the time the fish pass the flow-control weir because the majority of water is removed from the channel through the fish screens. Notably, this period [referring to spring/summer diversions under baseline conditions] includes the early portion of endangered juvenile winter-run Chinook salmon migration.

Note, however, that Vogel's (2008) experiments in 2007 occurred during May–July, during which time water temperature would be expected to be greater than during the main November–March period of the diversions and predation may be lower because of reduced predator bioenergetic requirements (e.g., Loboschefsky et al. 2012). This, coupled with the fact that the Red Bluff intake is situated on a wider stretch of the Sacramento River than the Hamilton City intake oxbow channel, increases the uncertainty in the applicability of estimates of predation mortality from the Vogel (2008) GCID study to the Red Bluff and Hamilton City intakes.

Table 11-8. Number and Proportion of Juvenile Chinook Salmon Released and Recaptured at the Hamilton City Intake, 2007.

Date	Time	Screen Group Number of Fish Released	Screen Group Number of Fish Recaptured	Screen Group Proportion of Fish Recaptured	Screen Group Proportional Survival/ 100 m[a]	Weir Group Number of Fish Released	Weir Group Number of Fish Recaptured	Weir Group Proportion of Fish Recaptured	Weir Group Proportional Survival/ 100 m[a]
5/22/2007	Day	504	294	0.58	0.95	239	201	0.84	0.97
5/22/2007	Night	474	416	0.88	0.99	254	242	0.95	0.99
5/24/2007	Day	454	304	0.67	0.96	259	145	0.56	0.90
5/24/2007	Night	484	338	0.70	0.96	264	171	0.65	0.93
5/29/2007	Day	504	295	0.59	0.95	260	194	0.75	0.95
5/29/2007	Night	492	417	0.85	0.98	249	169	0.68	0.93
5/31/2007	Day	484	363	0.75	0.97	253	220	0.87	0.98
5/31/2007	Night	503	372	0.74	0.97	258	221	0.86	0.97
6/5/2007	Day	515	428	0.83	0.98	254	219	0.86	0.97
6/5/2007	Night	511	408	0.80	0.98	264	223	0.84	0.97
6/7/2007	Day	504	451	0.89	0.99	261	216	0.83	0.97
6/7/2007	Night	514	383	0.75	0.97	264	212	0.80	0.96
6/12/2007	Day	511	460	0.90	0.99	265	257	0.97	0.99
6/12/2007	Night	514	430	0.84	0.98	264	235	0.89	0.98
6/14/2007	Day	510	495	0.97	1.00	265	197	0.74	0.95
6/14/2007	Night	502	440	0.88	0.99	263	215	0.82	0.97
6/19/2007	Day	510	489	0.96	1.00	264	200	0.76	0.95
6/19/2007	Night	511	434	0.85	0.98	263	214	0.81	0.96
6/21/2007	Day	515	495	0.96	1.00	265	242	0.91	0.98
6/21/2007	Night	512	394	0.77	0.97	264	200	0.76	0.95
6/26/2007	Day	515	495	0.96	1.00	265	256	0.97	0.99
6/26/2007	Night	509	409	0.80	0.98	259	207	0.80	0.96
6/28/2007	Day	507	432	0.85	0.98	265	227	0.86	0.97
6/28/2007	Night	505	396	0.78	0.98	264	189	0.72	0.94
7/3/2007	Day	507	460	0.91	0.99	265	232	0.88	0.98
7/5/2007	Day	515	499	0.97	1.00	265	245	0.92	0.99
7/5/2007	Night	504	344	0.68	0.96	264	195	0.74	0.95

Date	Time	Screen Group Number of Fish Released	Screen Group Number of Fish Recaptured	Screen Group Proportion of Fish Recaptured	Screen Group Proportional Survival/ 100 m[a]	Weir Group Number of Fish Released	Weir Group Number of Fish Recaptured	Weir Group Proportion of Fish Recaptured	Weir Group Proportional Survival/ 100 m[a]
7/10/2007	Day	509	493	0.97	1.00	228	184	0.81	0.96
7/10/2007	Night	513	368	0.72	0.97	263	185	0.70	0.94
7/12/2007	Day	513	471	0.92	0.99	262	226	0.86	0.97
7/24/2007	Day	497	405	0.81	0.98	248	207	0.83	0.97
7/24/2007	Night	508	337	0.66	0.96	257	161	0.63	0.92
7/26/2007	Day	500	380	0.76	0.97	263	223	0.85	0.97
7/31/2007	Day	509	361	0.71	0.97	253	190	0.75	0.95
7/31/2007	Night	503	378	0.75	0.97	264	194	0.73	0.95
8/2/2007	Day	498	420	0.84	0.98	219	166	0.76	0.95
8/7/2007	Night	511	397	0.78	0.97	246	189	0.77	0.96
8/9/2007	Day	504	415	0.82	0.98	265	212	0.80	0.96
8/9/2007	Night	478	307	0.64	0.96	255	169	0.66	0.93
8/21/2007	Night	514	338	0.66	0.96	263	152	0.58	0.91
8/23/2007	Day	429	298	0.69	0.96	247	139	0.56	0.91
8/23/2007	Night	507	323	0.64	0.95	258	142	0.55	0.90
8/27/2007	Day	465	342	0.74	0.97	261	227	0.87	0.98
			Mean	0.80	0.98			0.78	0.96
			Median	0.80	0.98			0.80	0.96
			Min.	0.58	0.95			0.55	0.90
			Max.	0.97	1.00			0.97	0.99

The recent study of acoustically tagged juvenile late fall–run Chinook salmon survival by Henderson et al. (2019) primarily provides information regarding far-field effects of flow (discussed in *Migration Flow-Survival* section below), but also has value in allowing inference regarding near-field effects of diversions. Henderson et al. (2019:Table 1) hypothesized that the density of diversions (number per km) would be negatively related to survival because of higher predator densities near the diversions. In fact, they found the opposite, and speculated that greater survival with higher diversion density may be more a function of habitat conditions where diversions are more abundant, e.g., armored banks resulting in reduced predator density and predation mortality (Henderson et al. 2019:1558). Reach-specific survival estimates by Henderson et al. (2019) provide context for the near-field effects provided by the physical structure of the existing Red Bluff and Hamilton City intakes, albeit during the non-diversion season. During the 2007–2011 study years, survival in the reach including the Red Bluff intake ranged in rank from highest survival (2007, 2011) to second lowest survival of 19 reaches in 2008. Survival in the Hamilton City reach ranged from highest survival (2010, 2011) to 12th highest survival of 19 reaches in 2008. Overall, the studies by Henderson et al. (2019) and Vogel (2008) are not inconsistent in suggesting that near-field survival along large fish screens does not appear to be greatly different from reaches without intakes. This could be because the flat surface of the screens gives relatively high sweeping velocity, which may be greater than the velocity preference of predatory fishes such as Sacramento pikeminnow, for which adult fish generally occur at velocity below 30 cm/s (1 ft/s) and for which critical swimming velocity is around 0.4–0.6 m/s (1.3–2 ft/s; Myrick and Cech 2000) for individuals of the size at which piscivory is common (Nobriga et al. 2006).

The available information suggests that the effects of intakes on predation is limited and so the effects of the diversions for Alternatives 1, 2, and 3 from the Red Bluff and Hamilton City intakes would be limited. Note that in-water structure at the intakes would have the same extent under the NAA and Alternatives 1, 2, and 3, so this aspect of potential predation would not change as a result of Alternatives 1, 2, and 3. Technical studies would be undertaken to assess predator density and distribution at the Red Bluff and Hamilton City intakes. These studies would be focused on diversions at these locations during high winter flow conditions when Project diversions would occur (Appendix 2D).

Dunnigan Pipeline Reservoir Discharge to Sacramento River (Alternative 2)

As described in Chapter 2, the discharge structure to the Sacramento River from the Dunnigan Pipeline would be located at approximately RM 100.8 (Figure 2-40). The Sacramento River discharge would include a transmission pipeline, stilling well, discharge pipes, reinforced concrete headwall, reinforced concrete stilling basin, and a reinforced concrete weir and apron extending to near the edge of the river and tying into the existing bank riprap.

The discharge structure would also include a vertical drop exclusion barrier to prevent the passage of anadromous fish into the pipeline. The weir and apron would meet NMFS guidelines for a combination velocity and vertical drop barrier for the exclusion of fish.

Stranding Behind Screens

At high flows, river flow may overtop the fish screens and associated decks of the intakes. At this point, water spills into the forebay behind the screens. Juvenile salmonids, including winter-run Chinook salmon, entering the forebays in the streamflow overtopping the inundated fish screen structures could be stranded once the stream flows have receded. At the Hamilton City intake, the deck above the fish screens (elevation 154.8 feet North American Vertical Datum of 1988) is overtopped at streamflows greater than approximately 100,000 cfs, as occurred in February 2017 (Figure 11-11). At the Red Bluff intake, the top of the deck is at a similar elevation to the 100-year flood flow (220,000 cfs), so that overtopping would be rare. The portion of the water column that exceeds the elevation of the top of the decks is small relative to the total water column, suggesting that only a limited portion of juvenile winter-run Chinook salmon in the upper water column may be susceptible given their typical vertical distribution (see discussion above in *Distribution in River Channel—Screen Exposure*). Of those juvenile salmon that are susceptible, the turbulence caused by the water passing over the decks (Figure 11-11) may trigger an avoidance response to swim away from the overtopping area. Although overtopping can occur at the Red Bluff and Hamilton City intakes, these are relatively infrequent events (e.g., approximately once per 100 years at Red Bluff) that occur under the NAA and would not be changed by Alternatives 1, 2, and 3.

Source: Glenn-Colusa Irrigation District 2017. Note: Oxbow from which water is diverted is to the left of the picture, with streamflow moving to the right into the forebay behind the screens.

Figure 11-11. Streamflow Overtopping the Fish Screen Structure at Glenn-Colusa Irrigation District Hamilton City Pumping Plant, February 18, 2017.

Attraction to Reservoir Discharge and Pipeline Entry (Alternative 2)

Water discharged from the Dunnigan Pipeline to the Sacramento River under Alternative 2 would primarily originate from the Sacramento River and as such may attract upstream-migrating salmonids, including adult winter-run Chinook salmon, which inhabit the upper Sacramento River and tributaries upstream of the Red Bluff and Hamilton City intakes as natal habitat. As described in Chapter 2, the discharge structure to the Sacramento River from the Dunnigan Pipeline at approximately RM 100.8 would include a vertical drop exclusion barrier to prevent the passage of anadromous fish, including adult winter-run Chinook salmon, into the pipeline. The twenty 36-inch pipes would discharge into a stilling basin, with a 4-foot weir at the basin's downstream end, on the other side of which would be a 16:1-gradient, 60-foot-long apron (Figure 2-40 in Chapter 2). The discharge facility meets draft NMFS (2018a:85–89) criteria for anadromous fish protection. Given that discharges would occur during low-water conditions (May–October/November) and the river would be more than 100 feet away from the pipes discharging water, there would be no risk of entry into the pipeline. The timing of adult winter-run Chinook salmon upstream migration generally has some overlap with the initial period of reservoir discharge in April/May, although peak migration occurs earlier (Vogel and Marine 1991:4; also Appendix 11A). Although discharge flow would be dissipated by the riprap, the rate of flow discharged in April/May (e.g., several hundred cfs in Critically Dry Water Years; see Table 5-19 in Chapter 5, *Surface Water Resources*) could attract some late-migrating adult winter-run Chinook salmon that might attempt to move upstream by leaping out of the river toward the discharge flow, but the design of the apron and weir of the discharge structure would eliminate the risk of stranding for any fish attempting to move up the flow.

Far-Field Effects

The additional water supply provided by Sites Reservoir may provide opportunities for improved management of salmonid habitat, particularly in the Sacramento River above RBPP. By exchanging Sites' water for CVP water, Reclamation has an additional tool to maintain and improve habitat for salmonid spawning, incubation, rearing, and migration. By delivering water to CVP contractors from Sites Reservoir, Reclamation may maintain supply in Shasta Lake for important periods to support these habitat conditions. The possible additional water supply in Shasta Lake can then be allocated during real-time management scenarios for a number of uses (e.g., cold-water pool maintenance, spring pulse or fall pulse flow events, reduced fall flows) that may provide enhanced anadromous fish benefits. These benefits may include protecting and increasing the cold-water pool in Shasta Lake, which is essential for temperature control in the reaches below Keswick Dam that are critical for salmonid egg incubation during Dry and Critically Dry Water Years. Maintenance of water in Shasta Lake may also provide a resource for achieving fall flow schedules to support spawning redds that persist in the wetted margins of the Sacramento River. In years when storm events are weak and natural pulse flows are minimal, this maintenance of supply in Shasta Lake could be used to manufacture a spring pulse flow to assist juvenile salmonids in their migration from the upper Sacramento River through the Delta and to the ocean.

The modeling conducted for this document includes some scenarios to maintain water supply in Shasta Lake. However, these scenarios are difficult to model and rely to a great extent on real-time conditions and real-time management actions. Benefits to anadromous fish can be realized through varying ways depending on specific in-year conditions. Thus, the modeling provided in this document likely underestimates the potential for exchanges between Sites Reservoir and Shasta Lake, and resulting anadromous fish benefits. The Authority and Reclamation intend to work together to better reflect the exchanges in the modeling with the goal of substantiating the Project's benefits to anadromous fish.

The Project also provides an additional capability to adjust to changes in precipitation and runoff patterns expected to result from climate change. As discussed in Chapter 28, climate change is expected to change the frequency and magnitude of intense storms and an increased likelihood of multi-year droughts. While long-term averages in precipitation are not expected to change, a larger percentage of precipitation is expected to fall as rain, resulting in a decreased snowpack and changes to runoff patterns. These changes will likely present challenges for future water management, including management of water for environment. The ability of the Project to capture and store water that cannot otherwise be captured and stored by Reclamation and to exchange water with Shasta Lake may allow for flexibility to provide environmental benefits to anadromous fish in the upper Sacramento River under changing climate scenarios.

Temperature Effects

Operation of Sites Reservoir has the potential to change water temperatures in the Sacramento River that could affect the life stages of winter-run Chinook salmon present. As described in Appendix 11B, the four methods used to analyze temperature-related effects on winter-run Chinook salmon in the Sacramento River were: (1) Physical Model Output Characterization; (2) Water Temperature Index Value/Range Exceedance Analysis; (3) Martin and Anderson Egg Mortality Models; and (4) SALMOD. More details on these methods are provided in Appendix 11B, Appendix 11H, and Appendix 11O.

The Authority and Reclamation evaluated water temperature model outputs during the period of presence and in the locations of each life stage of winter-run Chinook salmon in the Sacramento River upstream of the Delta (see Appendix 11B, Table 11B-2 for timing and locations). Visual observation of exceedance plots and differences in modeled mean monthly temperatures by water year type between Alternatives 1, 2, and 3 and the NAA in the Sacramento River below Keswick, at Balls Ferry, at Bend Bridge, below RBDD,[8] and at Butte City indicates that water temperatures would be predominantly similar among alternatives during the period of presence of each life stage of winter-run Chinook salmon (Appendix 6C, *River Temperature Modeling Results*, Tables 6C-5-1a to 6C-5-4c, Tables 6C-7-1a to 6C-7-4c, Tables 6C-9-1a to 6C-9-4c, Tables 6C-10-1a to 6C-10-4c, Tables 6C-12-1a to 6C-12-4c; Figures 6C-5-1 to 6C-5-18, Figures 6C-7-1 to 6C-7-18, Figures 6C-9-1 to 6C-9-18, Figures 6C-10-1 to 6C-10-18, Figures 6C-12-1 to 6C-12-18). At all locations, mean monthly water temperatures for all months in all water year types under Alternatives 1A and 1B were within 0.6°F of the NAA with the exception of August of critical years at Butte City, in which mean monthly water temperature would be 0.8°F lower under the NAA than under Alternatives 1A and 1B. Water temperature modeling results for Alternatives 2 and 3 were similar to those of Alternative 1 at all locations. Overall, these differences would be biologically inconsequential due to their low frequency and small magnitude.

Results for each alternative of the analysis of exceedance above water temperature index values for winter-run Chinook salmon in the Sacramento River from Appendix 11B, Table 11B-2 are presented in Appendix 11D, Table 11D-1 through Table 11D-19 and summarized in Table 11-9. For each life stage and at all locations evaluated, there were no month and water year type combinations in which both: (1) the percent of days that exceeded the index values was more than 5% greater under Alternative 1 than under the NAA, and (2) the exceedance per day was

[8] The RBDD, which was decommissioned in 2013, and the RBPP are co-located, and the names may be used interchangeably when referring to geographic locations.

more than 0.5°F greater under Alternative 1 than under the NAA. Results of the exceedance analysis for Alternative 2 are similar to Alternative 1 with no month and water year type combinations in which both criteria were met for any life stage at all locations. For Alternative 3, there was one month and water year type combination (July of Above Normal Water Years) at Hamilton City for the juvenile rearing and migration life stages in which there were 10.8% more days than the NAA exceeding the 64°F 7-day average daily maximum (7DADM) index value and the mean daily exceedance on those days was 0.6°F greater than the NAA. Also for Alternative 3, there was also one month and water year type combination (August of Critically Dry Water Years) for both the 53.5°F mean daily and 55.4°F 7DADM spawning and egg incubation index values in which both criteria were met in a favorable way for winter-run Chinook salmon (the percent of days that exceeded the index values was more than 5% lower under Alternative 3 than under the NAA, and the exceedance per day was more than 0.5°F lower under Alternative 3 than under the NAA). Because these biologically meaningful effects occurred in only one month of one water year type, they are not expected to be persistent enough to affect winter-run Chinook salmon at a population level.

Table 11-9. Number of Month and Water Year Type Combinations that Satisfy Both Criteria for being Biologically Meaningful in the Water Temperature Index Value Analysis, Winter-run Chinook Salmon, Sacramento River[1,2,3]

Location	Spawning and Egg Incubation				Juvenile Rearing and Emigration				Adult Immigration				Adult Holding			
	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3
Below Keswick	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Below Clear Creek	0	0	0	2 Favorable (53.5°F Mean Daily and 55.4°F 7DADM: August, Critically Dry Water Years,)	0	0	0	0	NA[4]	NA	NA	NA	NA	NA	NA	NA
Balls Ferry	0	0	0	0	0	0	0	0	NA	NA	NA	NA	0	0	0	1 positive (August, Critically Dry Water Years)
Bend Bridge	0	0	0	0	0	0	0	0	0	0	0	0	NA	NA	NA	NA
Below Red Bluff Diversion Dam	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1 positive (August, Critically Dry Water Years)

Hamilton City	NA	NA	NA	NA	0	0	0	1 Unfavorable (July, Above Normal Water Years)	NA	NA	NA	NA	NA	NA	NA	NA

[1] Biologically Meaningful Criteria include: (1) the difference in frequency of exceedance between the NAA and the alternative was greater than 5%, and (2) the difference in average daily exceedance between the NAA and the alternative was greater than 0.5°F.

[2] Index values for each life stage are located in Appendix 11B, Table 11B-2.

[3] Full results presented in Appendix 11D, Table 11D-1 through Table 11D-219.

[4] NA = Not Analyzed

The additional water temperature analysis to determine the ability to meet the Tiered Management Approach for summer cold-water pool management in the ROC ON LTO proposed action (Bureau of Reclamation 2019:4-28–4-32) is presented in Appendix 11D, Table 11D-20 and Table 11D-21. Results for the 53.5°F target are presented in Table 11D-20 and indicate that water temperature exceedances above the 53.5°F target under Alternatives 1A and 1B in the Sacramento River below Clear Creek would be similar to or lower than exceedance under the NAA in most tiers and periods in which temperatures were assumed to be managed at 53.5°F (see Appendix 11B, Table 11B-8). There were multiple instances in which temperatures would exceed the 53.5°F in >5% fewer days under the Alternatives 1A and 1B than the NAA. However, the difference in exceedance between Alternatives 1A and 1B and the NAA would be <0.5 degrees per day in each instance. Results for Alternatives 2 and 3 would be similar to those of Alternative 1 with one exception. Under Alternative 3, there would be one instance in which temperatures would exceed the 53.5°F in >5% more days than under the NAA. However, the difference in exceedance between Alternative 3 and the NAA would be 0.2 degrees per day.

Results for the 56°F target are presented in Appendix 11D, Table 11D-21 and indicate that water temperature exceedances under Alternatives 1A and 1B in the Sacramento River below Clear Creek would be similar to or lower than exceedance under the NAA in most tiers and periods in which temperatures were assumed to be managed at 56°F (see Appendix 11B, Table 11B-8). There were multiple instances in which temperatures would exceed the 56°F in >5% fewer days under Alternatives 1A and 1B than the NAA and one instance in which temperatures would exceed 56°F in >5% more days under Alternatives 1A and 1B. In all cases, the difference in exceedance between the Project and NAA would be <0.5°F per day. Results for Alternatives 2 and 3 would be similar to those of Alternative 1.

The Martin and Anderson models estimate water temperature-related mortality of winter-run Chinook salmon eggs to fry (Martin et al. 2017, Anderson 2018). Results of these models, which were run by the Authority, are presented in Appendix 11O. Differences between the Martin and Anderson model results are generally small other than in Critically Dry Water Years, in which the Martin model predicts nearly twice the egg mortality that the Anderson model under the NAA and Alternatives 1, 2, and 3 (Appendix 11O, Table 11O-3a, Table 11O-3b, Table 11O-4a, Table 11O-4b, Table 11O-5a, Table 11O-5b, Table 11O-6a, Table 11O-6b). Combining all months during the egg incubation period analyzed (May through September), Alternatives 1A and 1B would have small effects (0.1% difference from the NAA) on egg mortality in most water year types (Appendix 11O, Table 11O-3c, Table 11O-4c). In Critically Dry Water Years, egg mortality under Alternatives 1A and 1B would be 1.4% and 1.6% lower than the NAA, respectively, according to the Martin model and 1.3% and 1.9% lower than the NAA, respectively, according to the Anderson model. Exceedance plots (Appendix 11O, Figures 11O-1 through 11O-12) indicate that the majority of variance between the NAA and Alternatives 1A and 1B would occur between August and October in Critically Dry Water Years. The critical hatching period (in which there is the largest demand for oxygen and the eggs are most sensitive to temperature effects) typically peaks in August and in some drought years (e.g., 2015) the critical hatching period can extend until mid-September. Regardless, the small reductions in egg mortality under Alternative 1 relative to the NAA found in both the Martin and Anderson models are expected to be biologically inconsequential because of the small magnitude of the reduction.

The Martin and Anderson models predict largely similar results for Alternatives 2 and 3 compared to Alternative 1, although the reduction in egg mortality relative to the NAA in Critically Dry Water Years would be greater under Alternative 3 (3% in both Martin and Anderson models) than under Alternatives 1 and 2 (Appendix 11O, Table 11O-3 through Table 11O-6).

Because SALMOD provides temperature- and flow-related outputs, SALMOD results are summarized separately in the section labeled *SALMOD* below.

Overall, these results indicate that effects of Alternatives 1, 2, and 3 on water temperature-related effects on winter-run Chinook salmon in the Sacramento River are expected to be biologically inconsequential due to the low frequency and small magnitude of differences between Alternatives 1, 2, and 3 and the NAA.

Flow-Related Physical Habitat Conditions

Redd Scour Entombment

Loss of redds to scouring and entombment occurs when flows are high enough to mobilize sediments, destroying redds and their incubating eggs and alevins, or entombing the redds when sediments are redeposited. A flow of 40,000 cfs was selected as the scour flow threshold for the Sacramento River based on estimates in the literature (Appendix 11N, *Other Flow-Related Upstream Analyses*, Table 11N-10).

The probability of redd scour and entombment was estimated for winter-run by computing the percentage of days with flows exceeding 40,000 cfs in the USRDOM 82-year daily flow record (29,952 days in total) at four locations between Keswick Dam and the RBPP during the months of winter-run spawning and incubation. The results for the NAA and Alternatives 1, 2, and 3 show that the probability of scour and entombment is consistently low for winter-run (Appendix 11N, Table 11N-22 through Table 11N-25). The results indicate that Alternatives 1, 2, and 3 would have no effect on redd scour and entombment for winter-run Chinook salmon in the Sacramento River.

Redd Dewatering

The percentage of redds in the Sacramento River lost to dewatering was estimated using tables in USFWS (2006) that relate spawning and dewatering flows to percent reductions in species-specific spawning habitat WUA (Appendix 11N). The field studies used for USFWS (2006) were conducted in the Sacramento River between Keswick Dam and Battle Creek. USRDOM flow data, which has a daily time-step, are available for three locations in this river section: Keswick Dam (RM 302), the Sacramento River at Clear Creek (RM 289), and the Sacramento River at Battle Creek (RM 271). A single relationship for flows was developed for the entire river section, but the flows used to estimate redd dewatering in the current analysis were those that best matched the longitudinal distribution of the redds of the different salmon runs in the river as estimated from aerial redd surveys conducted by CDFW from 2003 through 2019. Spawning of winter-run Chinook salmon occurs primarily between Keswick Dam and the confluence with Clear Creek (Table 11-10), so Keswick Dam flows were used to analyze winter-run redd dewatering.

Table 11-10. Distributions of Spawning Redds among WUA River Segments as Percent of Total in the Sacramento River for Chinook Salmon Runs.

Segment	Description	River Miles	Winter-run	Spring-run	Fall-run	Late fall–run
6	Keswick to ACID[1]	302-298.5	45.0%	12.4%	16.3%	67.6%
5	ACID to Cow Creek	298.5-280	54.6%	66.0%	25.9%	12.7%
4	Cow Creek to Battle Creek	280-271	0.4%	12.8%	18.4%	9.2%
3	Battle Creek to RBPP	271-243	0.0%	4.9%	22.8%	4.3%
—	Downstream of RBPP	—	0.0%	4.0%	16.6%	6.2%

[1] ACID = Anderson-Cottonwood Irrigation District

Results are presented using the grand mean percentages of redds dewatered for each month of spawning, April through July, and each water year type and all water year types combined. The expected time for incubation of eggs and alevins is 3 months (Appendix 11N). Because changes in Project-related flow any time during this period can affect redd dewatering, the complete spawning and egg/alevin incubation periods (April–July through July–October) are provided in the results (Table 11N-13). The means of the redd dewatering estimates under the NAA and Alternatives 1, 2, and 3 are compared using absolute differences rather than relative differences (percent change) because many of the values for percentages of redds dewatered are small. Expressing changes of small values as percent changes may result in large values that may be misleading.

The results for winter-run Chinook salmon show few large changes in redd dewatering between the NAA and Alternatives 1, 2, and 3 (Table 11N-13). The largest reduction in redd dewatering, 7% (absolute difference), occurs under Alternative 3 during the incubation period for eggs spawned in July of Above Normal Water Years. Changes for most months and water year types under all Alternatives 1, 2, and 3 are less than 2%. Overall, the effects of Alternatives 1, 2, and 3 on winter-run redd dewatering are minor.

Spawning Habitat Weighted Usable Area

The suitability of physical habitat for salmonid spawning is largely a function of the availability of clean, coarse gravel for constructing redds, favorable depths, and suitable flow velocities. Instream flow potentially affects all these habitat characteristics and often affects the availability of suitable habitat. Habitat suitability for spawning was analyzed using WUA curves developed by USFWS and others from results of field studies and hydraulic modeling (Appendix 11K).

Spawning WUA for winter-run Chinook salmon was determined by USFWS (2003a, 2006) for a range of flows in three segments of the Sacramento River between Keswick Dam and the Battle Creek confluence (Figure 11-12). To estimate changes in winter-run spawning WUA that would result from Alternatives 1, 2, and 3, the winter-run flow versus spawning habitat WUA relationship developed for each of the three segments was used with mean monthly CALSIM II flows for the winter-run spawning period (April through July) under Alternatives 1, 2, and 3 and the NAA in the corresponding segments of the river. Differences in spawning WUA under Alternatives 1, 2, and 3 and the NAA were examined using the grand mean spawning WUA for each month of the spawning period under each water year type and all water year types combined.

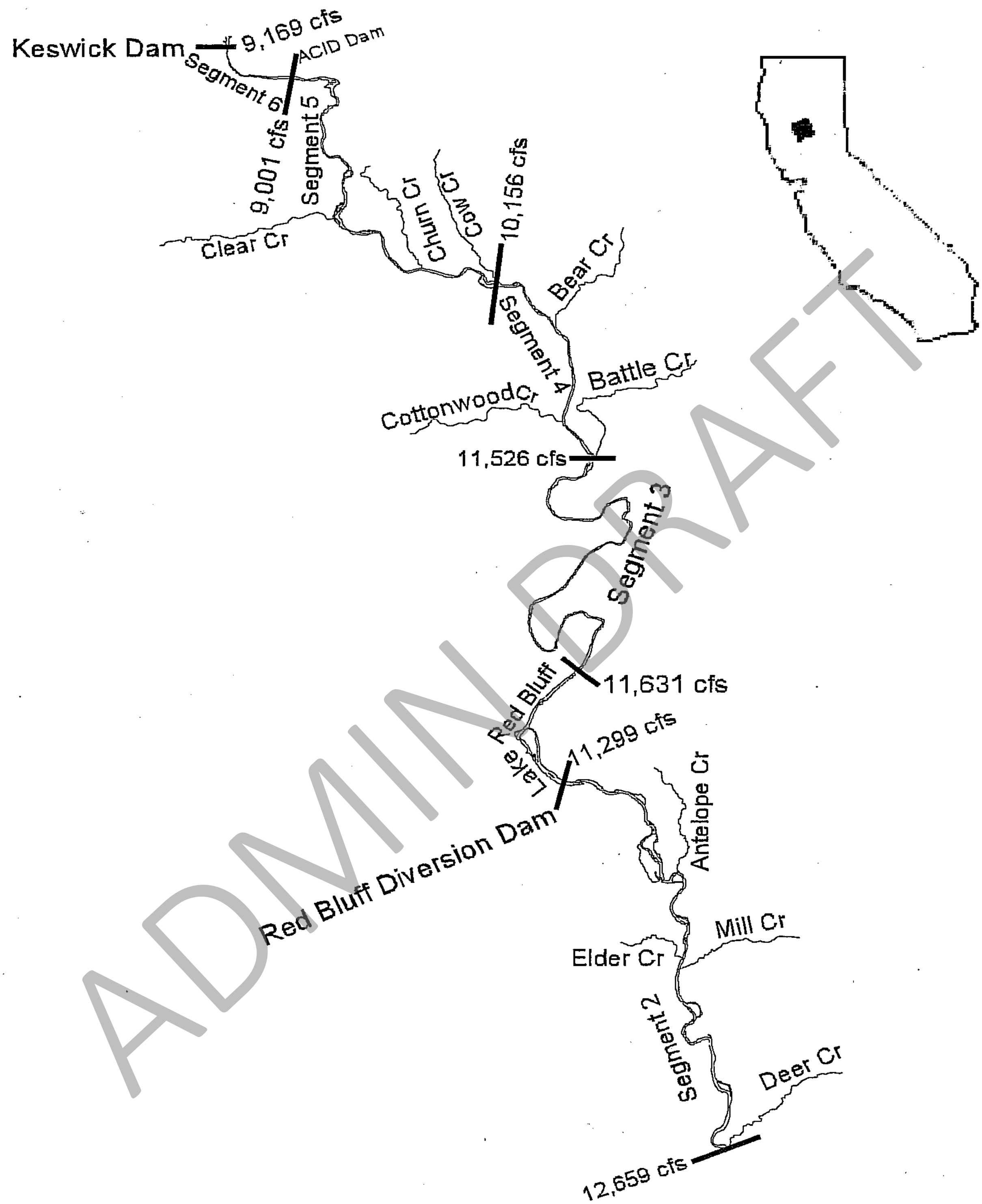


Figure 11-12. Segments 2–6 of the Sacramento River Used in U.S. Fish and Wildlife Service Studies to Determine Spawning and Rearing Weighted Usable Area (WUA) (flows in the figure are the average flows at the upstream boundary of each segment for October 1974 to September 1993). Source: U.S. Fish and Wildlife Service 2003a.

Almost all spawning by winter-run occurs in the upper two segments (Segments 6 and 5) of the Sacramento River, between Keswick Dam and Cow Creek, with spawning density (redds per river mile) especially high in Segment 6 (Table 11-10 and Figure 11-12). Winter-run spawning WUA under Alternatives 1, 2, and 3 and the NAA were estimated using CALSIM II flow estimates for Segments 4–6 for the winter-run spawning period (April through August). Differences in spawning WUA under Alternatives 1, 2, and 3 and the NAA were examined using the grand mean spawning WUA for each month of the spawning period under each water year type and all water year types combined.

Mean winter-run spawning WUA differs by less than 4% for most months and water year types, but mean WUA in Segment 6 under Alternatives 1A and 1B is 8%–9% lower than WUA under the NAA in April of Critically Dry Water Years (Table 11K-2). In Segment 5, WUA consistently differs little between the Project and the NAA, except for a 9% increase in WUA under Alternative 3 in July of Above Normal Water Years (Table 11K-3). In Segment 4, spawning WUA is up to 7% higher under Alternative 3 than under the NAA in June and July of Above Normal and Below Normal Water Years (Table 11K-4).

These results indicate that in April of Critically Dry Water Years, Alternatives 1A and 1B would result in reductions of spawning habitat in Segment 6. In Segments 5 and 4, Alternative 3 would results in increases of spawning habitat during June and July of Above Normal and Below Normal Water Years. Spawning habitat conditions are much more important for winter-run in Segment 6 than in Segment 4 because winter-run use Segment 6 much more than Segment 4 for spawning (Table 11K-1). Most differences in spawning WUA between Alternatives 1–3 and the NAA in all three river segments are less than 3%. The alternatives are not expected to substantially affect winter-run spawning WUA.

Rearing Habitat Weighted Usable Area

The suitability of physical habitat for salmonid rearing is largely a function of water depth, flow velocity, and the availability and type of cover. Instream flow potentially affects all these habitat characteristics and often affects the availability of suitable habitat. Habitat suitability for rearing was analyzed using WUA curves developed by USFWS and others from results of field studies and hydraulic modeling (Appendix 11K).

Rearing WUA for fry and older juvenile life stages of winter-run in the Sacramento River was determined by USFWS (2005b) for a range of flows in three segments of the Sacramento River between Keswick Dam and the Battle Creek confluence (Figure 11-12). For this analysis, fry are defined as fish less than 60 mm and juveniles are young fish (young-of-the-year) greater than 60 mm. To estimate changes in winter-run rearing WUA that would result from the Project, the winter-run flow versus rearing habitat WUA relationships developed for each of the three segments was used with mean monthly CALSIM II flows for the winter-run fry (July through October) and juvenile (September through November) rearing period under Alternatives 1, 2, and 3 and the NAA in the corresponding segments of the river. Differences in winter-run fry and juvenile rearing habitat WUA in each river segment under Alternatives 1, 2, and 3 compared to the NAA were examined using the grand mean rearing WUA for each month of the fry and juvenile rearing periods under each water year type and all water year types combined (Table 11K-23 through Table 11K-28).

All the means for fry rearing WUA differ by less than 3% between Alternatives 1A, 1B, and 2 and the NAA. Under Alternative 3, however, there are a number larger differences and all but one of these (September of Below Normal Water Years in Segment 6) constitutes a reduction in rearing WUA (Table 11K-23 through Table 11K-25). The largest of the reductions under Alternative 3 range up to 6% for July and August of Above Normal Water Years in Segment 5 (Table 11K-24). These results indicate that Alternative 3 would have a negative effect on rearing habitat WUA for winter-run fry in the Sacramento River. Alternatives 1A, 1B, and 2 would have little effect.

All the means for juvenile rearing WUA differ by <3% between Alternatives 1, 2, and 3 and the NAA, except for a 4% increase in Segment 6 for November of Critically Dry Water Years under Alternative 3 and a 3% reduction in Segment 4 for October of Wet Water Years under Alternative 3 (Table 11K-26 and Table 11K-28). These results indicate that Alternatives 1, 2, and 3 would have little effect on rearing habitat WUA for winter-run juveniles in the Sacramento River.

Juvenile Stranding

The juvenile stranding estimation procedure for the Sacramento River computes the surface area of salmonid rearing habitat inundated at an initial flow that is dewatered at a subsequent minimum (stranding) flow, and then converts this area to number of stranded juveniles using estimates of habitat capacity based on field study observations (U.S. Fish and Wildlife Service 2006). The minimum flow over a period of 3 months (90 days) is used for the juvenile stranding analysis in this report because the juveniles are presumed to be most vulnerable to stranding during their first 3 months (i.e., fry stage).

Juvenile stranding is computed using USRDOM daily flow estimates for Alternatives 1, 2, and 3 and the NAA at three locations in the upper Sacramento River: Keswick Dam, Clear Creek, and Battle Creek. Separate tables for converting initial and stranding flows to number of juveniles stranded were developed for periods when the ACID Dam boards are in and when they are out (Table 11N-11 and Table 11N-12). Both tables are used for all the salmonid species and races.

The results are presented using the grand mean number of juveniles stranded for each month of emergence under each water year type and all water year types combined (Table 11N-28 through Table 11N-30). The analysis assumes that under equal flow conditions the fry and early juvenile stage of all runs and species are equally vulnerable to stranding. To determine the results for a given species or run, the estimated months for which the fry typically emerge (Table 11N-27), are consulted in the tables of results. The effects of dewatering flows are tracked by the analysis from the month of emergence through the 3 months following. These periods are given in the tables for each Chinook run and steelhead.

Winter-run fry are present in the upper Sacramento River primarily from about July through October (Table 11N-27). During this period, juvenile stranding under Alternatives 1, 2, and 3 is generally similar to or lower than stranding under the NAA in the Keswick and Clear Creek reaches of the river (Table 11N-28 and Table 11N-29). The largest reductions for both reaches are 35% and occur for both reaches in July of Above Normal Water Years under Alternative 3. In the Battle Creek reach, winter-run stranding in individual months and water year types is mostly similar or slightly higher under Alternatives 1A, 1B, and 2 to stranding under the NAA, but it is mostly lower under Alternative 3 (Table 11N-30). For all reaches, months, and water year types combined, Alternatives 1B and 3 are expected to reduce winter-run stranding of fry juveniles and Alternatives 1A and 2 are expected to increase stranding (Table 11N-31 through Table 11N-33), but none of these effects is substantial.

SALMOD

The Authority used the SALMOD model to ascertain the potential effect of Alternatives 1, 2, and 3 on winter-run Chinook salmon mortality and potential production in the Sacramento River. A full description of the model can be found in the California WaterFix Biological Assessment (California Department of Water Resources 2016), Attachment 5.D.2, *SALMOD Model*. The SALMOD outputs for winter-run Chinook salmon are presented in Appendix 11H, Table 1a-1 through Table 1a-4, Table 2a-1 through Table 2a-4, and Figure B-a-1 through Figure B-a-19. For all water year types combined for all life stages and source of mortality, mean annual winter-run Chinook salmon potential production would be similar under Alternative 1A (0.2% greater) and Alternative 1B (0.4% greater) relative to the NAA (Appendix 11H, Table 2a-1 and Table 2a-2, Figure B-a-1). Further, differences within each water year type in mean annual potential production between Alternatives 1A and 1B and the NAA would be small (<2%). Alternative 2 results would be similar to those of Alternatives 1A and 1B (Appendix 11H, Table 2a-3, Figure B-a-1). Increases in mean annual production under Alternative 3 relative to the NAA would be slightly greater than those under Alternatives 1A and 1B (Appendix 11H, Table 2a-4, Figure B-a-1).

Results by life stage and mortality source are reported in Appendix 11H (Table 1a-1 through Table 1a-4 and Figure B-a-8 through Figure B-a-19). Depending on the life stage and source of mortality (flow-/habitat-based or temperature-based), mean annual mortality under Alternative 1A would be between 100% lower to 38% greater than that under the NAA and mortality under Alternative 1B would be between 100% lower to 1,014% higher than that under the NAA (Appendix 11H, Table 1a-1, Table 1a-2). However, it is important to understand that the model is seeded with 5,913,000 winter-run eggs each year. Although there may be larger percent differences in mean annual mortality between an alternative and the NAA by source and life stage, these differences typically represent a small proportion of overall individuals when put in a broader population-wide context. The largest difference in mean annual mortality (1,014%) would be in habitat-related juvenile mortality between Alternative 1B and the NAA in Above Normal Water Years. In this case, there would be an increase in mean annual mortality of 5 juveniles under Alternative 1B relative to the NAA, accounting for just 0.00008% of the 5,913,000 winter-run Chinook salmon eggs annually seeded into the model. Combining all sources of mortality and life stages together, mean annual mortality under Alternatives 1A and 1B would be 1% and 3% lower, respectively, than that under the NAA for the full simulation period (all water year types combined) and 3% lower to 2% higher and 7% lower to 2% higher, respectively, than that under the NAA depending on water year type. Mortality results for Alternatives 2 and 3 are generally similar to those of Alternatives 1A and 1B (Appendix 11H, Table 1a-3, Table 1a-4). Combining all sources of mortality and life stages, the mean annual mortality under Alternatives 2 and 3 would be 1% and 8% lower, respectively, than under the NAA for the full simulation period (all water year types combined) and 4% lower to 2% higher and 0% to 21% lower, respectively, than under the NAA depending on water year type.

Overall, SALMOD results show a minimal positive effect of each alternative on winter-run Chinook salmon mortality and potential production in the Sacramento River. However, due to the small magnitude of differences between Alternatives 1 and 2 and the NAA, these effects are expected to be biologically inconsequential.

Floodplain Inundation and Access

As described in Chapter 2, the Project's diversion criteria include restrictions to maintain Bend Bridge and Wilkins Slough flows, which have the effect of limiting changes to Yolo Bypass spill frequency and duration under Alternatives 1, 2, and 3 relative to the NAA. In particular, these criteria avoid impacts on Reclamation's ability to implement its obligations in the 2019 NMFS ROC ON LTO BiOp to implement the Yolo Bypass Restoration Salmonid Habitat Restoration and Fish Passage Implementation Plan and provide more than 17,000 acres of inundation in the Yolo Bypass from December to April (National Marine Fisheries Service 2019a). As such, Alternatives 1, 2, and 3 generally would have limited potential for negative effects on Yolo Bypass floodplain inundation and access for winter-run Chinook salmon.

Yolo Bypass and Fremont Weir Spill Flow and Days of Yolo Bypass Inundation

Results for Yolo Bypass Fremont Weir spill events are provided in Appendix 11M2, *Yolo Bypass Spill Events*, Table 1. The results show that Alternatives 1, 2, and 3 would have fewer days of Fremont Weir spill than the NAA. Opportunities for juvenile salmonids to enter the Yolo Bypass for rearing are therefore somewhat reduced under Alternatives 1, 2, and 3 relative to the NAA. A more detailed analysis of juvenile salmonid access to the Yolo Bypass is provided below (see *Juvenile Entry into Yolo Bypass at Fremont Weir*).

Takata et al. (2017) examined various juvenile Chinook salmon biological responses to Yolo Bypass flooding, which they defined as the number of days during January–June with daily mean flows at the downstream end of Yolo Bypass >4,000 cfs; this is the flow at which floodplain inundation occurs. Takata et al. (2017) found that growth and floodplain residence of coded-wire-tagged juvenile Chinook salmon and CPUE of wild juvenile Chinook salmon are significantly positively related to the annual duration of Yolo Bypass flooding (Takata et al. 2017:Figures 3 and 4c). Daily downscaled CALSIM modeling (Appendix 5A7, *Daily Pattern Development for the Estimation of Daily Flows and Weir Spills in CALSIM II*) suggests that operations under Alternatives 1, 2, and 3 may reduce Yolo Bypass inundation in January–June by approximately one day across most water year types (Table 11-11). Given the variability in the observed biological relationships indicated by the spread in the data (Takata et al. 2017:Figures 3 and 4c), and no significant difference in survival to capture in ocean fisheries between coded-wire-tagged hatchery juvenile Chinook salmon released in the Yolo Bypass and those released at the same time in the Sacramento River (Takata et al. 2017), the small differences in Yolo Bypass inundation indicated by the CALSIM modeling suggest that Alternatives 1, 2, and 3 are limited in their potential for negative effects on juvenile Chinook salmon, including winter-run. Note that this analysis was constrained to the months of January–June because those were the months considered by Takata et al. (2017). Expanding the analysis to consider the full period of potential diversions by Alternatives 1, 2, and 3 and the broader potential migratory period of juvenile Chinook salmon (i.e., September–June) gave differences between Alternatives 1, 2, and 3 and the NAA that were similar to those for January–June (Table 11-12).

Table 11-11. Mean Annual Number of Days in January–June With Yolo Bypass Floodplain Inundation by Alternative and Water Year Type.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	72	71 (-1%)	71 (-1%)	71 (-1%)	71 (-1%)
Above Normal	55	54 (-1%)	55 (-1%)	54 (-1%)	54 (-1%)
Below Normal	19	18 (-6%)	18 (-5%)	18 (-7%)	19 (-4%)
Dry	8	7 (-7%)	7 (-7%)	7 (-7%)	7 (-7%)
Critically Dry	3	3 (-5%)	3 (-5%)	3 (-5%)	3 (-5%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. Floodplain inundation is Yolo Bypass flow >4,000 cfs per Takata et al. (2017). The method of daily downscaling is provided in Appendix 11M, Section 11M.2.3.1, *Yolo Bypass*.
Note: Percentage values are rounded; as a result, differences between percentages may not always appear consistent.

Table 11-12. Mean Annual Number of Days in September–June With Yolo Bypass Floodplain Inundation by Alternative and Water Year Type.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	80	79 (-2%)	78 (-2%)	79 (-2%)	78 (-2%)
Above Normal	62	61 (-2%)	61 (-1%)	61 (-2%)	62 (-1%)
Below Normal	29	27 (-7%)	27 (-7%)	27 (-8%)	27 (-6%)
Dry	15	14 (-6%)	15 (-6%)	14 (-6%)	15 (-5%)
Critically Dry	3	3 (-5%)	3 (-5%)	3 (-5%)	3 (-5%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. Floodplain inundation is Yolo Bypass flow >4,000 cfs per Takata et al. (2017). The method of daily downscaling is provided in Appendix 11M, Section 11M.2.3.1, *Yolo Bypass*.
Note: Percentage values are rounded; as a result, differences between percentages may not always appear consistent.

Yolo Bypass Inundated Area

The modeling results of Yolo Bypass inundated suitable habitat show considerable increases in mean inundation acreage under Alternatives 1, 2, and 3 relative to the NAA during August through October, including up to 805 acres for September of Above Normal Water Years under Alternatives 1A and 1B (Table 11-13). These increases are the result of planned agricultural flow releases from Sites Reservoir. The releases reach the Yolo Bypass via the CBD, entirely bypassing the Sacramento River. For this reason and because of the months in which they occur, these summer-fall increases in potential habitat acreage have negligible effects on juvenile Chinook salmon or steelhead, including winter-run.

For November through June, the model results show no change to moderate reductions in Yolo Bypass mean daily habitat acreage under Alternatives 1, 2, and 3 (Table 11-13). Absolute acreage reductions for this period range from minimums of no change during April of Critically Dry Water Years and May and June of all but Wet Water Years to maximums of over 426 to 457 acres during December of Below Normal Water Years under Alternatives 1, 2, and 3 (Table 11-13). Almost all changes during this period consist of reductions in acreage. The majority of the reductions during December through March exceed 100 acres (Table 11-13). The largest percentage reductions in acreage, 12%, occur in November of Below Normal Water Years under

Alternatives 1, 2, and 3. In Table 11-13, the habitat acreages (but not the differences) are reported in thousands of acres to save space in the table.

Table 11-13. Estimated Mean Daily Inundated Habitat (Thousands of Acres <1 Meter Deep) for Juvenile Salmonids in the Yolo Bypass and the Absolute Differences (Acres, in parentheses) for the NAA and Alt 1A, Alt 1B, Alt 2, and Alt 3.

Month	Water Year Type[1]	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
January	Wet	14.52	14.36 (-156)	14.28 (-237)	14.36 (-156)	14.28 (-238)
	Above Normal	10.65	10.36 (-286)	10.36 (-285)	10.36 (-284)	10.38 (-268)
	Below Normal	6.19	6.05 (-146)	6.02 (-174)	6.04 (-157)	6.01 (-182)
	Dry	1.79	1.76 (-34)	1.75 (-41)	1.76 (-34)	1.75 (-39)
	Critically Dry	1.42	1.37 (-45)	1.37 (-45)	1.37 (-45)	1.38 (-44)
	All	7.78	7.66 (-129)	7.62 (-161)	7.65 (-130)	7.62 (-160)
February	Wet	17.26	17.23 (-33)	17.23 (-33)	17.22 (-43)	17.22 (-42)
	Above Normal	17.22	16.95 (-264)	16.97 (-249)	16.94 (-274)	16.89 (-324)
	Below Normal	10.58	10.38 (-197)	10.4 (-177)	10.38 (-196)	10.42 (-153)
	Dry	4.58	4.37 (-217)	4.35 (-236)	4.37 (-217)	4.35 (-238)
	Critically Dry	1.34	1.3 (-31)	1.31 (-30)	1.31 (-30)	1.3 (-31)
	All	10.92	10.78 (-133)	10.79 (-132)	10.78 (-138)	10.78 (-141)
March	Wet	14.62	14.59 (-34)	14.57 (-53)	14.59 (-33)	14.68 (64)
	Above Normal	14.51	14.32 (-196)	14.31 (-205)	14.33 (-183)	14.3 (-216)
	Below Normal	5.33	5.08 (-243)	5.1 (-230)	5.09 (-241)	5.09 (-240)
	Dry	3.78	3.61 (-176)	3.6 (-178)	3.61 (-175)	3.62 (-166)
	Critically Dry	1.37	1.33 (-44)	1.33 (-44)	1.33 (-44)	1.33 (-44)
	All	8.63	8.5 (-125)	8.5 (-130)	8.51 (-122)	8.53 (-94)
April	Wet	11.37	11.24 (-132)	11.24 (-130)	11.24 (-132)	11.16 (-210)
	Above Normal	5.07	5.14 (67)	5.13 (64)	5.14 (66)	5.13 (58)
	Below Normal	1.31	1.3 (-3)	1.3 (-3)	1.3 (-3)	1.3 (-3)
	Dry	1.20	1.2 (-2)	1.2 (-2)	1.2 (-2)	1.2 (0)
	Critically Dry	0.52	0.52 (0)	0.52 (0)	0.52 (0)	0.52 (0)
	All	4.91	4.87 (-34)	4.87 (-34)	4.87 (-35)	4.85 (-60)
May	Wet	2.76	2.64 (-113)	2.64 (-111)	2.64 (-112)	2.65 (-110)
	Above Normal	0.82	0.82 (0)	0.82 (0)	0.82 (0)	0.82 (0)
	Below Normal	0.46	0.46 (0)	0.46 (0)	0.46 (0)	0.46 (0)
	Dry	0.27	0.27 (0)	0.27 (0)	0.27 (0)	0.27 (0)
	Critically Dry	0.17	0.17 (0)	0.17 (0)	0.17 (0)	0.17 (0)
	All	1.16	1.12 (-37)	1.12 (-36)	1.12 (-36)	1.12 (-36)
June	Wet	0.78	0.75 (-29)	0.75 (-28)	0.75 (-28)	0.75 (-28)
	Above Normal	0.16	0.16 (0)	0.16 (0)	0.16 (0)	0.16 (0)
	Below Normal	0.16	0.16 (0)	0.16 (0)	0.16 (0)	0.16 (0)
	Dry	0.16	0.16 (0)	0.16 (0)	0.16 (0)	0.16 (0)

Month	Water Year Type[1]	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
	Critically Dry	0.16	0.16 (0)	0.16 (0)	0.16 (0)	0.16 (0)
	All	0.36	0.35 (-9)	0.35 (-9)	0.35 (-9)	0.35 (-9)
July	Wet	0.12	0.11 (-12)	0.11 (-12)	0.11 (-12)	0.11 (-12)
	Above Normal	0.11	0.1 (-13)	0.1 (-13)	0.1 (-14)	0.1 (-13)
	Below Normal	0.11	0.1 (-6)	0.1 (-6)	0.1 (-7)	0.1 (-5)
	Dry	0.11	0.11 (-5)	0.11 (-5)	0.11 (-5)	0.11 (-5)
	Critically Dry	0.12	0.12 (-1)	0.12 (-1)	0.11 (-6)	0.12 (-1)
	All	0.12	0.11 (-9)	0.11 (-9)	0.11 (-10)	0.11 (-9)
August	Wet	0.32	1.02 (702)	1.02 (703)	1.02 (701)	1.02 (703)
	Above Normal	0.22	0.91 (685)	0.91 (685)	0.99 (762)	0.91 (685)
	Below Normal	0.25	0.62 (364)	0.62 (367)	0.68 (425)	0.56 (304)
	Dry	0.14	0.47 (324)	0.46 (313)	0.44 (302)	0.46 (318)
	Critically Dry	0.13	0.22 (91)	0.22 (93)	0.47 (343)	0.21 (78)
	All	0.23	0.69 (467)	0.69 (465)	0.75 (520)	0.68 (454)
September	Wet	0.21	1 (793)	0.97 (766)	1.03 (827)	0.95 (744)
	Above Normal	0.17	0.98 (805)	0.98 (805)	1.05 (878)	0.94 (773)
	Below Normal	0.28	0.63 (353)	0.55 (269)	0.64 (366)	0.57 (297)
	Dry	0.16	0.28 (116)	0.3 (134)	0.31 (146)	0.29 (129)
	Critically Dry	0.16	0.2 (37)	0.2 (33)	0.31 (147)	0.17 (9)
	All	0.20	0.65 (456)	0.63 (436)	0.7 (502)	0.62 (425)
October	Wet	0.24	1 (756)	0.93 (686)	1.02 (775)	0.83 (592)
	Above Normal	0.09	0.71 (625)	0.7 (608)	0.86 (768)	0.53 (439)
	Below Normal	0.64	0.95 (311)	0.9 (265)	0.96 (321)	0.91 (271)
	Dry	0.11	0.17 (67)	0.17 (67)	0.23 (123)	0.22 (116)
	Critically Dry	0.10	0.18 (75)	0.11 (3)	0.12 (11)	0.1 (0)
	All	0.24	0.65 (407)	0.6 (363)	0.68 (437)	0.56 (322)
November	Wet	1.34	1.36 (25)	1.34 (1)	1.37 (28)	1.34 (-4)
	Above Normal	1.20	1.25 (50)	1.25 (48)	1.27 (64)	1.25 (48)
	Below Normal	1.36	1.19 (-163)	1.19 (-163)	1.19 (-162)	1.2 (-161)
	Dry	0.66	0.6 (-61)	0.6 (-56)	0.6 (-55)	0.62 (-38)
	Critically Dry	0.05	0.06 (8)	0.05 (0)	0.05 (0)	0.05 (0)
	All	0.98	0.96 (-26)	0.95 (-34)	0.96 (-23)	0.95 (-31)
December	Wet	5.02	4.93 (-97)	4.92 (-109)	4.93 (-98)	4.92 (-101)
	Above Normal	5.89	5.59 (-297)	5.63 (-256)	5.6 (-292)	5.7 (-187)
	Below Normal	6.84	6.38 (-457)	6.41 (-426)	6.38 (-457)	6.4 (-433)
	Dry	4.85	4.84 (-7)	4.75 (-96)	4.84 (-7)	4.87 (24)
	Critically Dry	0.92	0.91 (-10)	0.91 (-10)	0.91 (-10)	0.91 (-10)
	All	4.81	4.65 (-153)	4.64 (-166)	4.65 (-153)	4.68 (-128)

[1] Water year type sorting is by hydrologic water year.

A further summary of Yolo Bypass inundated habitat acreages gives the net effect of all the November through May changes between the NAA and Alternatives 1, 2, and 3 in daily habitat acreage (Table 11-14). For this summary, the monthly means were computed for all daily habitat acreages from November through May for all water year types combined. The largest difference is a reduction of 164 acres for December under Alternative 1B, or 3.4% of the NAA acreage, and the largest difference for the entire November through May period is a reduction of 98 acres under Alternative 1B, a 1.8% reduction of the NAA acreage (Table 11-14).

Table 11-14. Estimated Mean Daily November through May Inundated Habitat (Acres <1 Meter Deep) for Juvenile Salmonids in the Yolo Bypass and the Differences (in parentheses) for the NAA and Alt 1A, Alt 1B, Alt 2, and Alt 3.

Month	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
November	970	945 (-25)	937 (-34)	948 (-22)	939 (-31)
December	4,760	4,609 (-151)	4,597 (-164)	4,609 (-151)	4,633 (-127)
January	7,949	7,825 (-125)	7,793 (-157)	7,823 (-126)	7,791 (-158)
February	10,897	10,765 (-132)	10,766 (-131)	10,760 (-137)	10,757 (-140)
March	8,526	8,402 (-124)	8,397 (-129)	8,405 (-122)	8,433 (-93)
April	4,950	4,916 (-34)	4,916 (-33)	4,916 (-34)	4,891 (-59)
May	1,261	1,225 (-36)	1,226 (-35)	1,225 (-36)	1,226 (-35)
All (Nov–May)	5,616	5,527 (-90)	5,519 (-98)	5,527 (-90)	5,524 (-92)

The fish species of management concern most likely to be affected by the changes in Yolo Bypass inundated suitable habitat are Chinook salmon, steelhead, and splittail. Recent studies have shown that, when inundated by high flows in the winter and spring, the Yolo Bypass provides good rearing habitat for juvenile salmonids, as demonstrated by increased growth rates (Sommer et al. 2001, 2005; Katz et al. 2017; Bellido-Leiva et al. 2021). These species use the Yolo Bypass during the winter and spring, the natural period for seasonal floodplain inundation in the Sacramento River Basin. By late summer and fall, when Alternatives 1, 2, and 3 are expected to result in the largest percentage increases in Yolo Bypass inundation (Table 11-13), rearing juvenile salmonids have emigrated from the bypass, except for the relatively few trapped in pools (Sommer et al. 2005).

Significant spilling of the Fremont Weir generally begins in November or December and may occur as late as May. One race or another of juvenile Chinook salmon or steelhead is likely to enter the Yolo Bypass during most of this period. Based on the Knights Landing rotary screw trap data (Appendix 11M, *Yolo and Sutter Bypass Flow and Weir Spill Analysis*), winter-run occurs near the Fremont Weir from October through March. Once on the bypass, the juveniles may remain for a month or more, depending on conditions (Sommer et al. 2005). The March and April reductions in suitable habitat resulting from Alternatives 1, 2, and 3 would potentially affect rearing juveniles of all four Chinook salmon races (including winter-run) and steelhead. The largest differences in mean acreage for March and April are reductions of 230 to 243 acres (about 4.5%) during March of Below Normal Water Years under Alternatives 1, 2, and 3 (Table 11-13). As noted above, almost all changes during December through March consist of reductions and many of the reductions exceed 100 acres, so Alternatives 1, 2, and 3 are expected to have a cumulative negative effect on habitat availability (Table 11-13). However, when the changes in mean monthly acreage between the NAA and Alternatives 1, 2, and 3, irrespective of water year type, are examined, all reductions are less than 200 acres (3.5%) (Table 11-14). The net effect on the Chinook salmon and steelhead rearing habitat is not expected to be substantial.

The results of the frequency analysis of inundation of events for the Yolo Bypass generally show only minor difference between Alternatives 1, 2, and 3 and the NAA (Appendix 11M, Figure 11M-7). However, there are reductions in frequency for Alternatives 1, 2, and 3 compared to the NAA for events of 15,000 to 20,000 acres lasting 8 to 17 days and there are increases for Alternatives 1A and 1B for events >20,000 acres lasting 18 to 24 days. As noted above, inundation lasting 18–24 days has been shown to result in maximum habitat productivity for juvenile salmonids in field studies (Whipple et al. 2019). The differences in frequencies of inundation events of varying duration and acreage show no consistent differences between the NAA and Alternatives 1, 2, and 3.

Sutter Bypass Spill Flow and Duration

The results of the frequency analysis of weir spills shows reductions in the number of spills for the Sutter Bypass, indicating a reduction in bypass entry opportunity for juvenile salmonids (Appendix 11M4, *Sutter Bypass Weir Spill Events*, Table 1 through Table 4). A more detailed analysis of juvenile salmonid access to the Sutter Bypass is provided below (see *Juvenile Entry into Sutter Bypass at Moulton, Colusa, and Tisdale Weir*).

Sutter Bypass Inundated Area

The Sutter Bypass, when inundated, provides important juvenile rearing habitat for Chinook salmon and steelhead, as discussed for the Yolo Bypass (Cordoleani et al. 2020; Bellido-Leiva et al. 2021). For the Sutter Bypass the modeling results indicate that Alternatives 1, 2, and 3 would produce little change in suitable habitat compared to the NAA (Appendix 11M, Table 11M-4). The largest differences are increases of 42 acres for January of Above Normal Water Years under Alternatives 1, 2, and 3 and increases of about 45 acres for February of Above Normal Water Years under Alternatives 1A, 1B, and 2. The largest reduction is 17 acres for December of Dry Water Years under Alternative 3. All differences are less than 1%.

Juvenile Entry into Yolo Bypass at Fremont Weir

The potential for juvenile Chinook salmon entry into Yolo Bypass via Fremont Weir was examined with three different analyses. The first analysis examined the proportion of Sacramento River flow entering the Yolo Bypass via Fremont Weir based on daily downscaled CALSIM data (1922–2003) during the main November–March Yolo Bypass inundation period. This generally showed limited differences between Alternatives 1, 2, and 3 and the NAA during the main periods of Yolo Bypass inundation (i.e., wetter years), with larger relative differences occurring during periods with lower percentages of flow entering the Bypass via Fremont Weir (Table 11-15). This indicates the potential for a somewhat lower proportion of juvenile winter-run Chinook salmon entering the Yolo Bypass via Fremont Weir under Alternatives 1, 2, and 3 relative to the NAA.

Table 11-15. Proportion of Flow Entering Yolo Bypass via Fremont Weir.

Month	Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Nov	Wet	0.009	0.008 (-8%)	0.008 (-12%)	0.008 (-2%)	0.008 (-11%)
Nov	Above Normal	0.011	0.010 (-7%)	0.011 (-5%)	0.012 (4%)	0.011 (0%)
Nov	Below Normal	0.006	0.005 (-18%)	0.005 (-18%)	0.007 (3%)	0.006 (-7%)
Nov	Dry	0.004	0.003 (-11%)	0.004 (-10%)	0.005 (20%)	0.004 (-7%)
Nov	Critically Dry	0.000	0.000	0.000	0.000	0.000
Dec	Wet	0.047	0.046 (-4%)	0.045 (-4%)	0.053 (12%)	0.046 (-4%)
Dec	Above Normal	0.022	0.020 (-9%)	0.021 (-8%)	0.027 (20%)	0.020 (-10%)
Dec	Below Normal	0.040	0.037 (-9%)	0.037 (-8%)	0.048 (19%)	0.037 (-8%)
Dec	Dry	0.041	0.039 (-6%)	0.039 (-5%)	0.051 (23%)	0.040 (-3%)
Dec	Critically Dry	0.002	0.002 (-5%)	0.002 (-5%)	0.003 (26%)	0.002 (-5%)
Jan	Wet	0.187	0.182 (-3%)	0.181 (-3%)	0.193 (3%)	0.181 (-3%)
Jan	Above Normal	0.078	0.071 (-9%)	0.070 (-10%)	0.082 (5%)	0.071 (-9%)
Jan	Below Normal	0.020	0.018 (-9%)	0.018 (-11%)	0.020 (-2%)	0.018 (-11%)
Jan	Dry	0.008	0.007 (-8%)	0.007 (-8%)	0.008 (4%)	0.007 (-8%)
Jan	Critically Dry	0.003	0.002 (-19%)	0.002 (-19%)	0.004 (24%)	0.002 (-19%)
Feb	Wet	0.234	0.229 (-2%)	0.229 (-2%)	0.241 (3%)	0.227 (-3%)
Feb	Above Normal	0.124	0.119 (-4%)	0.120 (-3%)	0.138 (11%)	0.119 (-4%)
Feb	Below Normal	0.056	0.052 (-7%)	0.053 (-6%)	0.057 (2%)	0.054 (-4%)
Feb	Dry	0.024	0.021 (-11%)	0.022 (-11%)	0.025 (2%)	0.021 (-11%)
Feb	Critically Dry	0.008	0.007 (-9%)	0.007 (-8%)	0.008 (3%)	0.007 (-9%)
Mar	Wet	0.140	0.139 (0%)	0.139 (-1%)	0.146 (5%)	0.138 (-1%)
Mar	Above Normal	0.082	0.074 (-9%)	0.075 (-8%)	0.081 (-1%)	0.076 (-7%)
Mar	Below Normal	0.011	0.010 (-12%)	0.010 (-12%)	0.009 (-15%)	0.010 (-14%)
Mar	Dry	0.008	0.007 (-16%)	0.007 (-16%)	0.008 (3%)	0.007 (-16%)
Mar	Critically Dry	0.002	0.002 (-14%)	0.002 (-14%)	0.002 (-6%)	0.002 (-14%)

Notes: Percentage values in parentheses indicate differences of alternatives compared to the NAA.

The second analysis followed the methods of Acierto et al. (2014) by calculating an annual percentage of juvenile Chinook salmon entering the Yolo Bypass via Fremont Weir, based on weighting daily flow data by the historical daily density of juvenile Chinook salmon sampled at the Knights Landing rotary screw trap. Flow data used were for 2009–2017 from the Daily Divertible Flow and Storage Tool Results (see Appendix 11P1, *Sites Reservoir Daily Divertible and Storage Flow Tool*). As with the analysis based on the proportion of flow, this suggested similar or somewhat lower entry into Yolo Bypass under the Project relative to the NAA (Table 11-16).

Table 11-16. Proportion of Juvenile Winter-Run Chinook Salmon Entering Yolo Bypass Via Fremont Weir, Based on Acierto et al. (2014).

Water Year	Water Year Type	NAA	With Project*
2009	Dry	0.049	0.044 (-12%)
2010	Below Normal	0.055	0.051 (-6%)

Water Year	Water Year Type	NAA	With Project*
2011	Wet	0.068	0.062 (-9%)
2012	Below Normal	0.024	0.023 (-3%)
2013	Dry	0.066	0.065 (0%)
2014	Critically Dry	0.014	0.014 (0%)
2015	Critically Dry	0.038	0.038 (0%)
2016	Below Normal	0.011	0.009 (-14%)
2017	Wet	0.083	0.081 (-2%)

Notes: Percentage values in parentheses indicate differences of alternatives compared to the NAA. *Results are the same for Alternatives 1, 2, and 3 because the same operational criteria are applied.

A third analysis used the positive entrainment rating curve into the Yolo Bypass developed by the U.S. Army Engineer Research and Development Center (2017) for the Yolo Bypass Salmonid Habitat Restoration and Fish Passage EIS/EIR. This analysis focused on full operations of the Fremont Weir notch (up to 6,000 cfs) from November 1 through March 15, using flow data for 2009–2018 from the Daily Divertible Flow and Storage Tool Results (see Appendix 11P1). Consistent with the other analyses, this suggested similar or slightly less entrainment into the Yolo Bypass under Alternatives 1, 2, and 3 relative to the NAA (Table 11-17).

Table 11-17. Proportion of Juvenile Chinook Salmon Entering Yolo Bypass Via Fremont Weir, Based on U.S. Army Engineer Research and Development Center (2017).

Water Year	Water Year Type	NAA	With Project *
2009	Dry	0.023	0.023 (0%)
2010	Below Normal	0.044	0.038 (-14%)
2011	Wet	0.073	0.066 (-10%)
2012	Below Normal	0.004	0.004 (0%)
2013	Dry	0.038	0.037 (-3%)
2014	Critically Dry	0.005	0.005 (0%)
2015	Critically Dry	0.030	0.029 (-3%)
2016	Below Normal	0.031	0.030 (-3%)
2017	Wet	0.096	0.094 (-2%)
2018	Below Normal	0.002	0.002 (0%)

Notes: Percentage values in parentheses indicate differences of alternatives compared to the NAA. *Results are the same for Alternatives 1, 2, and 3 because the same operational criteria are applied.

Juvenile Entry into Sutter Bypass at Moulton, Colusa, and Tisdale Weirs

As with the analysis for Yolo Bypass, the potential for juvenile Chinook salmon entry into Sutter Bypass via Moulton, Colusa, and Tisdale weirs was examined with daily downscaled CALSIM data (1922–2003) during the main November–March inundation period. This generally showed limited a limited proportion of flow entering Sutter Bypass via Moulton Weir for any scenario (Table 11-18), whereas there were greater proportions of flow entering Sutter Bypass via Colusa and Tisdale weirs (Tables 11-19 and 11-20). Similar to the pattern observed for proportion of flow entering Yolo Bypass via Fremont Weir as discussed above, differences between Alternatives 1, 2, and 3 and the NAA during the main periods of Sutter Bypass inundation (i.e., wetter years) tended to be more limited, with larger relative differences occurring during periods with lower percentages of flow entering Sutter Bypass (Tables 11-19 and 11-20). This indicates the potential for a somewhat lower proportion of juvenile winter-run Chinook salmon entering the Sutter Bypass from the Sacramento River under Alternatives 1, 2, and 3 relative to the NAA.

Table 11-18. Proportion of Flow Entering Sutter Bypass via Moulton Weir.

Month	Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Nov	Wet	0.000	0.000	0.000	0.000	0.000
Nov	Above Normal	0.000	0.000 (-44%)	0.000 (-35%)	0.000 (-44%)	0.000 (-11%)
Nov	Below Normal	0.000	0.000	0.000	0.000	0.000
Nov	Dry	0.000	0.000	0.000	0.000	0.000
Nov	Critically Dry	0.000	0.000	0.000	0.000	0.000
Dec	Wet	0.001	0.000 (-10%)	0.000 (-8%)	0.000 (-9%)	0.000 (-10%)
Dec	Above Normal	0.000	0.000	0.000	0.000	0.000
Dec	Below Normal	0.001	0.000 (-33%)	0.000 (-33%)	0.000 (-33%)	0.000 (-33%)
Dec	Dry	0.002	0.002 (-10%)	0.002 (-9%)	0.002 (-10%)	0.002 (-4%)
Dec	Critically Dry	0.000	0.000	0.000	0.000	0.000
Jan	Wet	0.010	0.009 (-8%)	0.009 (-7%)	0.009 (-8%)	0.009 (-7%)
Jan	Above Normal	0.000	0.000 (-73%)	0.000 (-73%)	0.000 (-73%)	0.000 (-72%)
Jan	Below Normal	0.000	0.000	0.000	0.000	0.000
Jan	Dry	0.000	0.000	0.000	0.000	0.000
Jan	Critically Dry	0.000	0.000	0.000	0.000	0.000
Feb	Wet	0.014	0.012 (-8%)	0.013 (-7%)	0.013 (-7%)	0.012 (-9%)
Feb	Above Normal	0.003	0.003 (-13%)	0.003 (-13%)	0.003 (-13%)	0.003 (-11%)
Feb	Below Normal	0.000	0.000 (-46%)	0.000 (-45%)	0.000 (-45%)	0.000 (-28%)
Feb	Dry	0.000	0.000	0.000	0.000	0.000
Feb	Critically Dry	0.000	0.000	0.000	0.000	0.000
Mar	Wet	0.006	0.006 (-6%)	0.006 (-6%)	0.006 (-6%)	0.006 (-7%)
Mar	Above Normal	0.001	0.000 (-52%)	0.000 (-52%)	0.000 (-52%)	0.000 (-52%)
Mar	Below Normal	0.000	0.000	0.000	0.000	0.000
Mar	Dry	0.000	0.000	0.000	0.000	0.000
Mar	Critically Dry	0.000	0.000	0.000	0.000	0.000

Notes: Percentage values in parentheses indicate differences of alternatives compared to the NAA.

Table 11-19. Proportion of Flow Entering Sutter Bypass via Colusa Weir.

Month	Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Nov	Wet	0.001	0.000 (-31%)	0.000 (-54%)	0.000 (-30%)	0.000 (-55%)
Nov	Above Normal	0.016	0.015 (-6%)	0.015 (-4%)	0.015 (-6%)	0.016 (-1%)
Nov	Below Normal	0.001	0.000 (-100%)	0.000 (-100%)	0.000 (-100%)	0.000 (-100%)
Nov	Dry	0.003	0.001 (-68%)	0.001 (-67%)	0.001 (-68%)	0.001 (-56%)
Nov	Critically Dry	0.000	0.000	0.000	0.000	0.000
Dec	Wet	0.034	0.031 (-8%)	0.030 (-9%)	0.031 (-8%)	0.031 (-7%)
Dec	Above Normal	0.019	0.016 (-12%)	0.016 (-14%)	0.016 (-12%)	0.014 (-23%)
Dec	Below Normal	0.029	0.023 (-18%)	0.024 (-18%)	0.023 (-18%)	0.023 (-18%)

Month	Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Dec	Dry	0.034	0.031 (-9%)	0.031 (-8%)	0.031 (-9%)	0.035 (2%)
Dec	Critically Dry	0.000	0.000	0.000	0.000	0.000
Jan	Wet	0.144	0.139 (-4%)	0.138 (-4%)	0.139 (-4%)	0.137 (-5%)
Jan	Above Normal	0.059	0.048 (-18%)	0.048 (-18%)	0.048 (-18%)	0.048 (-18%)
Jan	Below Normal	0.006	0.004 (-39%)	0.004 (-39%)	0.004 (-39%)	0.004 (-39%)
Jan	Dry	0.004	0.003 (-28%)	0.003 (-28%)	0.003 (-28%)	0.003 (-28%)
Jan	Critically Dry	0.000	0.000	0.000	0.000	0.000
Feb	Wet	0.182	0.174 (-4%)	0.174 (-4%)	0.176 (-4%)	0.171 (-6%)
Feb	Above Normal	0.099	0.091 (-9%)	0.092 (-7%)	0.090 (-9%)	0.096 (-3%)
Feb	Below Normal	0.024	0.020 (-16%)	0.021 (-14%)	0.020 (-17%)	0.021 (-12%)
Feb	Dry	0.012	0.008 (-35%)	0.008 (-35%)	0.008 (-35%)	0.008 (-35%)
Feb	Critically Dry	0.000	0.000	0.000	0.000	0.000
Mar	Wet	0.105	0.105 (0%)	0.105 (-1%)	0.106 (1%)	0.101 (-4%)
Mar	Above Normal	0.059	0.050 (-15%)	0.050 (-15%)	0.050 (-15%)	0.050 (-14%)
Mar	Below Normal	0.000	0.000	0.000	0.000	0.000
Mar	Dry	0.004	0.002 (-61%)	0.001 (-61%)	0.002 (-61%)	0.002 (-59%)
Mar	Critically Dry	0.000	0.000	0.000	0.000	0.000

Notes: Percentage values in parentheses indicate differences of alternatives compared to the NAA.

Table 11-20. Proportion of Flow Entering Sutter Bypass via Tisdale Weir.

Month	Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Nov	Wet	0.003	0.002 (-23%)	0.002 (-40%)	0.002 (-23%)	0.002 (-39%)
Nov	Above Normal	0.016	0.015 (-3%)	0.015 (-3%)	0.015 (-3%)	0.016 (-1%)
Nov	Below Normal	0.004	0.002 (-56%)	0.002 (-55%)	0.002 (-55%)	0.002 (-42%)
Nov	Dry	0.005	0.004 (-25%)	0.004 (-24%)	0.004 (-25%)	0.004 (-18%)
Nov	Critically Dry	0.000	0.000	0.000	0.000	0.000
Dec	Wet	0.042	0.039 (-7%)	0.039 (-8%)	0.039 (-7%)	0.040 (-7%)
Dec	Above Normal	0.030	0.026 (-12%)	0.026 (-12%)	0.026 (-12%)	0.024 (-18%)
Dec	Below Normal	0.038	0.032 (-16%)	0.032 (-15%)	0.032 (-16%)	0.032 (-16%)
Dec	Dry	0.036	0.032 (-9%)	0.032 (-9%)	0.032 (-9%)	0.035 (-2%)
Dec	Critically Dry	0.001	0.001 (-25%)	0.001 (-25%)	0.001 (-25%)	0.001 (-25%)
Jan	Wet	0.148	0.143 (-3%)	0.141 (-5%)	0.143 (-3%)	0.140 (-5%)
Jan	Above Normal	0.081	0.067 (-18%)	0.066 (-18%)	0.067 (-18%)	0.067 (-17%)
Jan	Below Normal	0.012	0.009 (-23%)	0.008 (-30%)	0.009 (-26%)	0.008 (-31%)
Jan	Dry	0.008	0.006 (-19%)	0.006 (-19%)	0.006 (-19%)	0.006 (-20%)
Jan	Critically Dry	0.000	0.000 (-100%)	0.000 (-100%)	0.000 (-100%)	0.000 (-100%)
Feb	Wet	0.177	0.171 (-4%)	0.170 (-4%)	0.172 (-3%)	0.169 (-5%)
Feb	Above Normal	0.122	0.112 (-8%)	0.114 (-6%)	0.112 (-8%)	0.114 (-6%)
Feb	Below Normal	0.039	0.032 (-19%)	0.033 (-15%)	0.032 (-20%)	0.035 (-10%)
Feb	Dry	0.021	0.017 (-22%)	0.017 (-22%)	0.017 (-22%)	0.017 (-22%)
Feb	Critically Dry	0.002	0.000 (-86%)	0.000 (-86%)	0.000 (-86%)	0.000 (-86%)
Mar	Wet	0.114	0.111 (-2%)	0.110 (-3%)	0.113 (-1%)	0.107 (-6%)
Mar	Above Normal	0.083	0.071 (-14%)	0.071 (-14%)	0.071 (-14%)	0.071 (-14%)

Month	Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Mar	Below Normal	0.002	0.001 (-44%)	0.001 (-44%)	0.001 (-44%)	0.001 (-44%)
Mar	Dry	0.010	0.007 (-34%)	0.007 (-34%)	0.007 (-34%)	0.007 (-32%)
Mar	Critically Dry	0.000	0.000	0.000	0.000	0.000

Notes: Percentage values in parentheses indicate differences of alternatives compared to the NAA.

Adult Upstream Passage at Fremont Weir

Changes in Sacramento River flow and flow entering Yolo Bypass have the potential to change the number of days meeting adult Chinook salmon passage criteria at the modified Fremont Weir. This was assessed using the number of days meeting adult passage criteria[9], based on data for water years 2009–2018 from the Daily Divertible Flow and Storage Tool Results (see Appendix 11P1). The results of this analysis indicated that the number of days meeting fish passage criteria at Fremont Weir generally would be similar with or without the Project (Table 11-21).

Table 11-21. Number of Days Meeting Adult Chinook Salmon and Sturgeon Passage Criteria at Fremont Weir.

Water Year	Water Year Type	NAA	With Project*
2009	Dry	25	25 (0%)
2010	Below Normal	48	58 (21%)
2011	Wet	118	121 (3%)
2012	Below Normal	27	24 (-11%)
2013	Dry	75	75 (0%)
2014	Critically Dry	21	21 (0%)
2015	Critically Dry	35	35 (0%)
2016	Below Normal	74	71 (-4%)
2017	Wet	151	151 (0%)
2018	Below Normal	41	40 (-2%)

Notes: Percentage values in parentheses indicate differences of alternatives compared to the NAA. *Results are the same for Alternatives 1, 2, and 3 because the same operational criteria are applied.

Adult Upstream Passage at Sutter Bypass Weirs

As with the Fremont Weir, changes in Sacramento River flow and flow entering Sutter Bypass as a result of the Project have the potential to change the number of days meeting adult Chinook salmon passage criteria at the three Sutter Bypass weirs (Moulton, Tisdale, and Colusa). This was assessed using the number of days meeting adult passage criteria[10], based on data for water years 2009–2018 from the Daily Divertible Flow and Storage Tool Results (see Appendix 11P1).

[9] The criteria for days to be counted as Yolo Bypass passage days were river stage (1) between 21.14 feet and 29.92 feet during November 1–March 15, (2) between 21.14 feet and 23.35 feet during March 16–April 30, or (3) greater than 35 feet (elevation of the crest of the Fremont Weir). These criteria were based on the YBPASS tool (California Department of Water Resources 2017).

[10] The criteria for days to be counted as passage days were minimum depth of 3 feet and maximum velocity of 4 ft/s, during November–May. These criteria were based on long (>60 feet) criteria from Table 2-1 in the Tisdale Weir Rehabilitation and Fish Passage Project Draft EIR (California Department of Water Resources and Environmental Science Associates 2020).

The results of this analysis indicated that there would be no days meeting adult salmon passage criteria during 2009–2018 at Moulton or Tisdale Weirs. For Colusa Weir, the number of days meeting fish passage criteria generally would be similar with or without the Project, with the main exception being 2016 (nearly 40% fewer days of passage; Table 11-22).

Table 11-22. Number of Days Meeting Adult Chinook Salmon Passage Criteria at Colusa Weir.

Water Year	Water Year Type	NAA	With Project*
2009	Dry	0	0 (0%)
2010	Below Normal	5	4 (-20%)
2011	Wet	23	22 (-4%)
2012	Below Normal	0	0 (0%)
2013	Dry	8	6 (-25%)
2014	Critically Dry	0	0 (0%)
2015	Critically Dry	8	8 (0%)
2016	Below Normal	18	11 (-39%)
2017	Wet	81	76 (-6%)
2018	Below Normal	0	0 (0%)

Notes: Percentage values in parentheses indicate differences of alternatives compared to the NAA. *Results are the same for Alternatives 1, 2, and 3 because the same operational criteria are applied.

Migration Flow-Survival

Diversions from the Sacramento River to Sites Reservoir under Alternatives 1, 2, and 3 have the potential to affect survival of juvenile salmonids, including winter-run Chinook salmon, based on flow-survival relationships. Several recent analyses provided evidence for positive correlations between Sacramento River flow and survival of Chinook salmon (Michel 2018; Henderson et al. 2019; Michel et al. 2021). Henderson et al. (2019) found that riverine flows—along with a number of other covariates related to individual, release group, reach-specific, and time-varying factors—were positively correlated with survival of acoustically tagged late fall–run Chinook salmon juveniles in 19 Sacramento River reaches from Battle Creek to the Delta. Michel (2018) found correlations between mean streamflow at Bend Bridge and survival to adulthood of hatchery-origin juvenile fall-run ($r^2 = 0.35$), late fall–run ($r^2 = 0.45$), and winter-run ($r^2 = 0.57$) Chinook salmon, which were stronger predictors of survival than ocean conditions. Michel et al. (2021) examined survival of a mixture of acoustically tagged hatchery-origin and wild-origin fall-run, spring-run, and winter-run juvenile Chinook salmon in the Sacramento River from the Deer Creek confluence to the Feather River confluence during late March to early June, 2013–2019. Michel (2018) explored various forms of flow-survival relationship based on Sacramento River at Wilkins Slough flow and found the best to be a function based on three flow thresholds, which they defined as minimum (4,259 cfs), historical mean (10,712 cfs), and high (22,872 cfs). Survival varied by flow threshold: 3.0% below minimum, 18.9% between minimum and historical mean, 50.8% between historical mean and high, and 35.3% above high. Although the modeling focused on the spring period, it is hypothesized that similar relationships may occur during other months of juvenile salmon migration (Michel pers. comm.). Hassrick et al. (2022) found winter-run Chinook salmon smolt survival in the Sacramento River from Redding to Sacramento was best predicted by mean annual flow, intra-annual deviations from the mean flow at the reach scale, reach-specific channel characteristics, and travel time. Mean annual flow (at Bend Bridge) had the strongest positive effect on survival. A negative interaction between mean annual flow and intra-annual reach flow indicated that within-year deviations at the reach scale from annual mean flow had larger effects on survival in low-flow years, giving higher survival during years with pulse flows or high flows.

As described in Chapter 2, the Project includes pulse flow protection measures to be applied to precipitation-generated pulse flow events. There is also a 10,700-cfs Wilkins Slough bypass flow threshold from October 1 through June 14, corresponding with the threshold identified by Michel et al. (2021[11]). Adaptive management would be utilized to assess and if necessary refine the diversion criteria based on biological monitoring (see Appendix 2D, Section 2D.6, *Fish Monitoring and Technical Studies Plan and Adaptive Management for Diversions*). The discussion in Appendix 11P, Section 11P.2, *Flow Threshold Survival Analysis to Assess Potential Effects of Sites Reservoir Project Diversion Criteria Based on Historical Juvenile Chinook Salmon Monitoring Data (Michel et al. 2021)*, illustrates that the Sites Reservoir diversion criteria effectively limit diversions during the historical periods of fish movement as reflected in Red Bluff rotary screw trap data, and application of the flow-threshold criteria from Michel et al. (2021) suggests that flow-survival effects on juvenile Chinook salmon (including winter-run Chinook salmon) would be greatly limited by the Project's diversion criteria because there is essentially no difference in predicted survival between Alternatives 1, 2, and 3 and the NAA. See also the discussion of flow-survival effects below in the *Life Cycle Models* discussion.

Sites Reservoir Release Effects

Sites Reservoir releases could temporally overlap with winter-run Chinook salmon presence near the release location in the Sacramento River, specifically migrating adults during November through May and juveniles rearing or migrating during October through March (Appendix 11A, Table 11A-2), although beach seine data suggest juveniles only occur downstream of the discharge location at approximately RM 100 from November onwards (Figure 11-4).

Sites Reservoir releases into the Yolo Bypass via the CBD would occur during August–October. These releases could temporally overlap with juvenile rearing and emigration of winter-run Chinook salmon during October. Adults would not be present in the Yolo Bypass during August through October (Appendix 11A, Table 11A-2).

Temperature Effects

As discussed in Chapter 6, the effects of Alternatives 1A, 1B, 2, and 3 on water temperatures at the Sites Reservoir release site in the Sacramento River would be relatively small with the releases generally tending to cause a slight reduction in water temperature (Tables 6-12a through

[11] In contrast to other studies describing flow-survival relationships that considered only continuous relationships (e.g., Henderson et al. 2019; Hassrick et al. 2022), Michel et al. (2021) examined evidence for both continuous and step function (threshold) relationships, finding the latter to provide the best fit to the data. Their study included three of four runs of Chinook salmon and included more study years than Henderson et al. (2019) and Hassrick et al. (2022). Their examination of both forms of relationship, finding the step function to be statistically better supported, provided the basis for the focus on flow thresholds for the Project.

6-12d). Therefore, temperature-related effects of Alternatives 1A, 1B, 2, and 3 on winter-run Chinook salmon at the Sacramento River release site would be minimal.

Estimated changes in CBD water temperatures due to Sites Reservoir releases during August through October are presented in Table 11-23. These reflect changes in temperatures where water first enters the Yolo Bypass from the CBD but would mix with existing Yolo Bypass water as well as equilibrate as it moves through the Yolo Bypass. For Alternatives 1A, 1B, 2, and 3, water temperatures at this location would either stay the same or be reduced due to Sites Reservoir releases. Specifically for juvenile winter-run Chinook salmon, the small water temperature reductions during October would not increase the risk of mortality. Lower temperatures could reduce their growth somewhat due to reduced metabolism. However, the small reduction in water temperatures is expected to occur during one month and in a limited area of the Yolo Bypass before it equilibrates with atmospheric temperatures. As a result, temperature-related effects of Alternatives 1A, 1B, 2, and 3 in the Yolo Bypass due to Sites Reservoir releases via the CBD would be minimal at a population level.

Table 11-23. Mean and Median of Estimated Change in Colusa Basin Drain Water Temperature (ºF) from Sites Reservoir Releases, August–October

Month and Statistic	Alternative 1A	Alternative 1B	Alternative 2	Alternative 3
August Mean	-1.9	-1.8	-1.6	-1.9
August Median	-1.9	-1.7	-1.5	-1.7
September Mean	-1.1	-1.0	-0.9	-1.0
September Median	-0.9	-0.9	-0.8	-0.8
October Mean	-0.8	-1.2	-0.5	-1.1
October Median	-0.3	-0.5	-0.2	-0.7

As discussed in Chapter 6, Yolo Bypass flows account for only about 5% of the water in the Sacramento River downstream of the Yolo Bypass (e.g., Rio Vista) at this time of year. Even if releases into the Yolo Bypass coincided with large (~5°F) temperature differentials between the Sacramento River and the Yolo Bypass, the maximum change in Sacramento River water temperatures at the mouth of the Yolo Bypass would be 5% of 5°F or 0.25°F, with the effect potentially being somewhat diminished by tidal fluxes. This suggests that there would be minimal temperature-related effects of Alternatives 1A, 1B, 2, and 3 on winter-run Chinook salmon related to Sites Reservoir releases via the CBD and Yolo Bypass in this reach of the Sacramento River.

Water Quality Effects

As described in Appendix 6F, *Mercury and Methylmercury*, and Chapter 6, surface water mercury levels in the Delta could increase during the July–November export period as a result of the Project relative to the NAA. Given the relatively short temporal overlap of winter-run Chinook salmon with the export period, and the low observed levels of mercury in Chinook salmon generally (Appendix 6F, Table 6F-8) in relation to levels of mercury that may result in sublethal effects (Beckvar et al. 2005), it is expected that water quality effects of mercury from releases as a result of the Project would be minimal.

The Project would be expected to have limited effects on the salinity aspect of water quality, given that potential increases in Sacramento River salinity as a result of Salt Lake influence on Sites Reservoir salinity are anticipated to be minimal based on analyses in Chapter 6.

Water released from Sites Reservoir into the Sacramento River primarily would originate from the Sacramento River, with a minor contribution from local tributary inflow, so that far-field olfactory cues for upstream-migrating adult salmonids, including winter-run Chinook salmon, from reservoir releases would not be expected to be greatly different than Sacramento River water. Reservoir releases through the CBD and thence the Yolo Bypass to improve foodweb productivity in the north Delta for delta smelt would increase flow from the Yolo Bypass, but not during the upstream migration period of adult winter-run Chinook salmon or migration/rearing period of juvenile winter-run.

As discussed in Chapter 6 and below for delta smelt, Sites Reservoir releases under Alternatives 1, 2, and 3 would not be expected to increase risk of exposure to pesticides.

Feather River

Temperature Effects

Operation of Sites Reservoir has the potential to change water temperatures in the Feather River that could affect the non-natal rearing of winter-run Chinook salmon juveniles. As described in Appendix 11B, the two methods used to analyze temperature-related effects on winter-run Chinook salmon non-natal rearing in the Feather River were: (1) Physical Model Output Characterization; and (2) Water Temperature Index Value/Range Exceedance Analysis. More details on these methods are provided in Appendix 11B.

The Authority and Reclamation evaluated water temperature model outputs during the July through March winter-run Chinook salmon non-natal juvenile rearing period in the Feather River LFC at Robinson Riffle and in the HFC at Gridley Bridge. Visual observation of exceedance plots and differences in modeled mean monthly temperatures by water year type between alternatives and the NAA at these two locations indicates that water temperatures would be predominantly similar among alternatives during the period of winter-run Chinook salmon non-natal rearing (Appendix 6C, Tables 6C-17-1a to 6C-17-4c, 6C-18-1a to 6C-18-4c; Figures 6C-17-1 to 6C-17-18, 6C-18-1 to 6C-18-18). At both locations, mean monthly water temperatures for all months within all water year types under Alternatives 1A and 1B were within 0.5°F of the NAA. Water temperature modeling results for Alternatives 2 and 3 were similar to those of Alternative 1 at both locations.

Results of the analysis of exceedance above the 64°F 7DADM water temperature index value for winter-run Chinook salmon non-natal rearing in the Feather River (Appendix 11B, Table 11B-3) are presented in Appendix 11D, Table 11D-22 and Table 11D-23. At both locations evaluated, there were no month and water year type combinations in which both: (1) the percent of months that exceeded the index value was more than 5% greater under Alternative 1, 2, or 3 than under the NAA, and (2) the exceedance per month was more than 0.5°F greater under Alternative 1, 2, or 3 than under the NAA. Results of the exceedance analysis for Alternatives 2 and 3 are similar to Alternative 1 with no month and water year type combinations in which both criteria were met at both locations.

Combined, these water temperature results indicate that the Project's implementation under Alternatives 1, 2, and 3 would cause inconsequential temperature-related effects on winter-run non-natal juvenile rearing in the Feather River.

Flow-Related Physical Habitat Conditions

Redd Scour and Entombment, Redd Dewatering

Winter-run do not spawn in the Feather River.

Habitat Weighted Usable Area

Winter-run do not spawn in the Feather River.

Juvenile Stranding

No formal method for analyzing juvenile stranding analysis has been developed for the Feather River.

<u>American River</u>

Temperature Effects

Operation of Sites Reservoir has the potential to change water temperatures in the American River that could affect the non-natal rearing of winter-run Chinook salmon juveniles. As described in Appendix 11B, the two methods used to analyze temperature-related effects on winter-run Chinook salmon in the Feather River were: (1) Physical Model Output Characterization; and (2) Water Temperature Index Value/Range Exceedance Analysis. More details on these methods are provided in Appendix 11B.

The Authority and Reclamation evaluated water temperature model outputs during the July through April winter-run Chinook salmon non-natal rearing period in the American River at Watt Avenue. Visual observation of exceedance plots and differences in modeled mean monthly temperatures by water year type between alternatives and the NAA at Watt Avenue indicates that water temperatures would be predominantly similar among alternatives during the period of winter-run Chinook salmon non-natal rearing (Appendix 6C, Tables 6C-14-1a to 6C-14-4c; Figures 6C-14-1 to 6C-14-18). Mean monthly water temperatures for all months within all water year types under Alternatives 1A and 1B were within 0.3°F of the NAA. Water temperature modeling results for Alternatives 2 and 3 were similar to those of Alternative 1.

Results of the analysis of exceedance above the 64°F 7DADM water temperature index value for winter-run Chinook salmon non-natal rearing in the American River (Appendix 11B, Table 11B-4) are presented in Appendix 11D, Table 11D-22 and Table 11D-23 and summarized in Table 11D-34. There were no month and water year type combinations in which both: (1) the percent of days that exceeded the index value was more than 5% greater under Alternatives 1, 2, and 3 than under the NAA; and (2) the exceedance per day was more than 0.5°F greater under Alternatives 1, 2, and 3 than under the NAA. Results of the exceedance analysis for Alternatives 2 and 3 are similar to Alternative 1 with no month and water year type combinations in which both criteria were met.

Combined, these water temperature results indicate that the Project's implementation under Alternatives 1, 2, and 3 would cause inconsequential temperature-related effects on winter-run non-natal juvenile rearing in the American River.

Flow-Related Physical Habitat Conditions

Redd Scour and Entombment, Redd Dewatering

Winter-run do not spawn in the American River.

Habitat Weighted Usable Area

Winter-run do not spawn in the American River.

Juvenile Stranding

No formal method for analyzing juvenile stranding analysis has been developed for the American River.

<u>Delta</u>

Juvenile Through-Delta Survival

Operations of Alternatives 1, 2, and 3 could affect juvenile winter-run Chinook salmon through changes in Sacramento River flow entering the Delta, which could influence through-Delta survival based on flow-survival relationships. The potential for such effects was assessed using a spreadsheet implementation of the through-Delta survival function formulated by Perry et al. (2018)[12], which estimates through-Delta survival as a function of daily Sacramento River flow at Freeport as well as DCC gate position (Appendix 11J). The results of this analysis showed that during the main period of juvenile winter-run Chinook salmon occurrence in the Delta (i.e., December–April; see Table 11A-3 in Appendix 11A), there were 0%–1% differences in mean through-Delta survival between the NAA and Alternatives 1, 2, and 3 (Table 11-24). The largest differences in through-Delta survival occurred in October (up to 7% greater than the NAA in Below Normal or Dry Water Years), which is a period with lower abundance of juvenile winter-run (Appendix 11A1).

Table 11-24. Probability of Juvenile Chinook Salmon Through-Delta Survival, Averaged by Month and Water Year Type, Based on Perry et al. (2018).

Month	Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Sep	Wet	0.39	0.40 (0%)	0.39 (0%)	0.40 (0%)	0.40 (0%)
Sep	Above Normal	0.40	0.40 (0%)	0.40 (0%)	0.40 (0%)	0.40 (0%)
Sep	Below Normal	0.30	0.30 (1%)	0.30 (1%)	0.30 (1%)	0.30 (1%)
Sep	Dry	0.27	0.28 (4%)	0.28 (4%)	0.28 (4%)	0.28 (3%)
Sep	Critically Dry	0.30	0.31 (3%)	0.31 (3%)	0.31 (3%)	0.30 (2%)
Oct	Wet	0.37	0.37 (0%)	0.37 (0%)	0.37 (0%)	0.37 (0%)
Oct	Above Normal	0.34	0.34 (0%)	0.34 (1%)	0.34 (0%)	0.34 (2%)

[12] The spreadsheet model was provided by Perry (pers. comm.) and reproduces the mean response of the STARS (Survival, Travel time, And Routing Simulation) model (Perry et al. 2019).

Month	Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Oct	Below Normal	0.30	0.31 (1%)	0.31 (1%)	0.31 (1%)	0.32 (7%)
Oct	Dry	0.28	0.30 (7%)	0.30 (7%)	0.30 (5%)	0.29 (3%)
Oct	Critically Dry	0.25	0.25 (1%)	0.25 (1%)	0.25 (1%)	0.25 (1%)
Nov	Wet	0.41	0.41 (0%)	0.41 (-1%)	0.41 (0%)	0.41 (0%)
Nov	Above Normal	0.38	0.38 (0%)	0.38 (0%)	0.38 (0%)	0.39 (2%)
Nov	Below Normal	0.39	0.39 (1%)	0.39 (1%)	0.39 (1%)	0.40 (4%)
Nov	Dry	0.35	0.35 (2%)	0.35 (2%)	0.35 (2%)	0.35 (2%)
Nov	Critically Dry	0.29	0.29 (1%)	0.29 (1%)	0.29 (1%)	0.29 (1%)
Dec	Wet	0.48	0.48 (0%)	0.47 (0%)	0.48 (0%)	0.48 (0%)
Dec	Above Normal	0.47	0.47 (-1%)	0.47 (-1%)	0.47 (-1%)	0.47 (-1%)
Dec	Below Normal	0.47	0.47 (-1%)	0.47 (0%)	0.47 (-1%)	0.47 (0%)
Dec	Dry	0.44	0.44 (-1%)	0.44 (0%)	0.44 (-1%)	0.44 (0%)
Dec	Critically Dry	0.38	0.38 (0%)	0.38 (0%)	0.38 (0%)	0.38 (1%)
Jan	Wet	0.60	0.60 (0%)	0.60 (0%)	0.60 (0%)	0.60 (0%)
Jan	Above Normal	0.55	0.55 (-1%)	0.55 (-1%)	0.55 (-1%)	0.55 (-1%)
Jan	Below Normal	0.49	0.49 (-1%)	0.49 (-1%)	0.49 (-1%)	0.49 (-1%)
Jan	Dry	0.44	0.44 (0%)	0.44 (0%)	0.44 (0%)	0.44 (0%)
Jan	Critically Dry	0.42	0.42 (0%)	0.42 (0%)	0.42 (0%)	0.42 (0%)
Feb	Wet	0.63	0.63 (0%)	0.63 (0%)	0.63 (0%)	0.63 (0%)
Feb	Above Normal	0.59	0.59 (-1%)	0.59 (-1%)	0.59 (-1%)	0.59 (-1%)
Feb	Below Normal	0.55	0.54 (0%)	0.54 (0%)	0.54 (0%)	0.54 (0%)
Feb	Dry	0.50	0.49 (-1%)	0.49 (0%)	0.49 (-1%)	0.49 (0%)
Feb	Critically Dry	0.45	0.45 (0%)	0.45 (0%)	0.45 (0%)	0.45 (0%)
Mar	Wet	0.60	0.60 (0%)	0.60 (0%)	0.60 (0%)	0.60 (0%)
Mar	Above Normal	0.59	0.59 (-1%)	0.59 (-1%)	0.59 (-1%)	0.59 (-1%)
Mar	Below Normal	0.49	0.49 (-1%)	0.49 (-1%)	0.49 (-1%)	0.49 (-1%)
Mar	Dry	0.47	0.47 (-1%)	0.47 (-1%)	0.47 (-1%)	0.47 (-1%)
Mar	Critically Dry	0.42	0.42 (0%)	0.42 (0%)	0.42 (0%)	0.42 (0%)
Apr	Wet	0.57	0.57 (0%)	0.57 (0%)	0.57 (0%)	0.57 (0%)
Apr	Above Normal	0.52	0.52 (0%)	0.52 (0%)	0.52 (0%)	0.52 (0%)
Apr	Below Normal	0.47	0.47 (0%)	0.47 (0%)	0.47 (0%)	0.47 (0%)
Apr	Dry	0.43	0.43 (0%)	0.43 (0%)	0.43 (0%)	0.43 (0%)
Apr	Critically Dry	0.40	0.40 (0%)	0.40 (0%)	0.40 (0%)	0.40 (0%)
May	Wet	0.55	0.55 (0%)	0.55 (0%)	0.55 (0%)	0.55 (0%)
May	Above Normal	0.49	0.49 (0%)	0.49 (0%)	0.49 (0%)	0.50 (0%)
May	Below Normal	0.45	0.45 (0%)	0.45 (0%)	0.45 (0%)	0.45 (0%)
May	Dry	0.41	0.41 (0%)	0.41 (0%)	0.41 (0%)	0.41 (0%)
May	Critically Dry	0.37	0.37 (0%)	0.37 (0%)	0.37 (0%)	0.37 (0%)
Jun	Wet	0.43	0.43 (0%)	0.43 (0%)	0.43 (0%)	0.43 (0%)
Jun	Above Normal	0.37	0.37 (0%)	0.37 (0%)	0.37 (0%)	0.37 (0%)

Month	Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Jun	Below Normal	0.34	0.34 (0%)	0.34 (0%)	0.34 (0%)	0.34 (0%)
Jun	Dry	0.33	0.33 (0%)	0.33 (0%)	0.33 (0%)	0.33 (0%)
Jun	Critically Dry	0.30	0.30 (0%)	0.30 (0%)	0.30 (0%)	0.30 (0%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. Percentage values are rounded; as a result, differences between percentages may not always appear consistent.

Note that the spreadsheet implementation of the Perry et al. (2018) model does not account for the variability in coefficient estimates (Perry et al. 2018:Figure 6), which would likely give appreciable overlap of estimates in through-Delta survival between the NAA and the alternative scenarios, particularly in relation to the relatively small differences (i.e., low single digit percentages) between alternatives. Note also that the CDFW (2020) State ITP requires a Georgiana Slough Migratory Barrier to be installed to reduce juvenile winter- and spring-run Chinook salmon entry into Georgiana Slough. This barrier is part of 2020 baseline conditions and would be present under the NAA and Alternatives 1, 2, and 3. The above analysis with the spreadsheet implementation of the Perry et al. (2018) model did not include a representation of the barrier because the specific operating criteria (e.g., months for installation) are not yet known. However, to illustrate the potential effects of the barrier on relative survival differences between the NAA and Alternatives 1, 2, and 3, a sensitivity analysis was undertaken that assumed the barrier was installed and reduced proportional entry into Georgiana Slough by 50% compared to no barrier (Appendix 11J). Although the sensitivity analysis gave higher absolute estimates of through-Delta survival, as expected, there was little change in the relative pattern of differences between the NAA and Alternatives 1, 2, and 3 (Appendix 11J, Table 11J-1; compared with Table 11-24). In addition to the analysis based on Perry et al. (2018), the IOS life cycle model includes a through-Delta survival analysis (the Delta Passage Model), discussed below in *Life Cycle Models*.

Juvenile Rearing Habitat

Changes in flow entering the Delta as a result of the operations of Alternatives 1, 2, and 3 would be expected to give only limited changes to rearing habitat for juvenile winter-run Chinook salmon in the Delta. Operations of the Project have the potential to affect the extent of riparian and wetland bench rearing habitat inundation (i.e., by lowering WSE in the Delta as a result of Project diversions upstream of the Delta). This could affect the suitability of channel margin rearing habitat for juvenile Chinook salmon. This potential effect was assessed by calculating bench inundation indices for a number of habitat benches in the north Delta; further background and methods are provided in Appendix 11J, Section 11J.3, *Delta Rearing Habitat of Juvenile Chinook Salmon*.

The analysis of bench inundation suggested the potential for changes in inundation under the Alternatives 1, 2, and 3 relative to the NAA, ranging from little difference to over 10% (relative difference) less bench inundation under the Project, depending on season and location (Appendix 11J, Table 11J-4). The largest proportional differences were for riparian benches with generally little difference for wetland benches, which tend to be inundated at lower WSEs that would be available at much lower flows. The Project would result in somewhat less availability of inundated bench habitat for juvenile winter-run Chinook salmon compared to the NAA.

Multiplying the proportional difference in inundation indices between Alternatives 1, 2, and 3 and the NAA (Appendix 11J, Table 11J-4) by the length of bench in each area allows the largest differences as a result of the Project to be estimated; the overall difference relative to the NAA was approximately 2,200–2,400 feet (~5%) less under the Project (Appendix 11J, Table 11J-5).

South Delta Entrainment

Juvenile salmonids, including winter-run Chinook salmon, can be entrained at the south Delta export facilities, principally during the winter/spring (e.g., Williams 2006). There would be little difference in indicators of entrainment risk (south Delta exports and Old and Middle River flows) during winter/spring between the NAA and Alternatives 1, 2, and 3 (Appendix 5B3, *Delta Operations*, Tables 5B3-6-1a through 5B3-6-4c; Figures 5B3-6-1 through 5B3-6-18; and Appendix 5B4, *Regional Deliveries*, Tables 5B4-1-1a through 5B4-1-4c; Figures 5B4-1-1 through 5B4-1-18), and existing restrictive criteria from the NMFS (2019a) ROC ON LTO BiOp and CDFW (2020) State ITP would limit entrainment risk for winter-run under Alternatives 1, 2, and 3. South Delta export of water released from Sites Reservoir would have little to no overlap with winter-run Chinook salmon. This is illustrated by the results of the salvage-density analysis (Appendix 11Q, *Other Delta Species Analyses*), which weights south Delta exports at SWP (Banks) and CVP (Jones) export facilities by historical salvage[13] per unit volume (i.e., salvage density) of juvenile winter-run Chinook salmon. There were minimal differences between the NAA and Alternatives 1, 2, and 3 from the results of the salvage-density analysis (Table 11-25; Table 11-26).

Table 11-25. Entrainment Loss of Juvenile Winter-Run Chinook Salmon At SWP Banks Pumping Plant, Averaged by Water Year Type, Based on the Salvage-Density Method.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	2,291	2,294 (0%)	2,285 (0%)	2,293 (0%)	2,303 (0%)
Above Normal	NA	(2%)	(1%)	(1%)	(0%)
Below Normal	2,006	2,010 (0%)	2,016 (1%)	2,006 (0%)	1,993 (-1%)
Dry	1,095	1,088 (-1%)	1,068 (-2%)	1,086 (-1%)	1,066 (-3%)
Critically Dry	984	1,004 (2%)	1,003 (2%)	1,008 (2%)	998 (1%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. The analysis was based on historical salvage data during 2009–2019 which did not include any Above Normal Water Years, hence that row of the table is noted as 'NA'; the percentage difference in Above Normal Water Years is based on density data for Wet Water Years applied to above normal modeled exports. Results by water year type and month are provided in Appendix 11Q. Percentage values are rounded; as a result, differences between percentages may not always appear consistent.

[13] The salvage-density analysis is sometimes also called the loss-density analysis when applied to juvenile salmonids, because salvage is extrapolated into total entrainment loss by accounting for estimates of predation and other factors during entrainment; however, this extrapolation is consistent between scenarios and the focus is on the relative difference between scenarios (as opposed to absolute differences).

Table 11-26. Entrainment Loss of Juvenile Winter-Run Chinook Salmon At CVP Jones Pumping Plant, Averaged by Water Year Type, Based on the Salvage-Density Method.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	272	270 (-1%)	271 (-1%)	270 (-1%)	267 (-2%)
Above Normal	NA	(-1%)	(-1%)	(-1%)	(0%)
Below Normal	529	528 (0%)	528 (0%)	530 (0%)	534 (1%)
Dry	308	307 (0%)	306 (0%)	308 (0%)	309 (1%)
Critically Dry	72	72 (1%)	75 (5%)	72 (0%)	76 (6%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. The analysis was based on historical salvage data during 2009–2019 which did not include any Above Normal Water Years, hence that row of the table is noted as 'NA'; the percentage difference in Above Normal Water Years is based on density data for Wet Water Years applied to above normal modeled exports. Results by water year type and month are provided in Appendix 11Q. Percentage values are rounded; as a result, differences between percentages may not always appear consistent.

Life Cycle Models

The IOS and OBAN winter-run Chinook salmon life cycle models were run to provide an analysis of the potential integrated effects of Alternatives 1, 2, and 3 on the species relative to the NAA. Appendix 11I includes technical memoranda describing the IOS and OBAN modeling. The IOS model includes flow-survival relationships for migrating juvenile winter-run Chinook salmon in the Delta, whereas the OBAN model does not. The IOS model includes a flow-survival relationship based on Sacramento River at Bend Bridge flows that was adjusted to account for the Red Bluff and Hamilton City diversions. The OBAN model included the Michel et al. (2021) flow-survival relationship to account for Red Bluff and Hamilton City diversions.

IOS

IOS modeling results for female adult winter-run Chinook salmon escapement reflect the integration of all modeled effects of The Project, as discussed in detail in Appendix 11I. Relative to the NAA, the mean female escapement ranged from 3% greater under Alternative 3 in Dry Water Years to 6% less under Alternative 1A in Wet and Below Normal Water Years (Table 11-27). As shown in Appendix 11I, there is considerable overlap in results between Alternatives 1, 2, and 3. The different life stage components of the IOS modeling generally had limited differences between the NAA and Alternatives 1, 2, and 3, with the largest differences being 2%–4% greater mean egg and fry survival under Alternative 3 in Critically Dry Water Years (Table 11-28; Table 11-29; Table 11-30; Table 11-31). Note that because the IOS model includes a Sacramento River flow-survival relationship based on Bend Bridge flow, Red Bluff and Hamilton City diversions (which are downstream of Bend Bridge) were subtracted from the Bend Bridge flows proportional to length of the riverine survival portion of the model (i.e., Red Bluff to Verona) that would be affected by Project diversions, to provide an assessment of potential flow-survival effects of the diversions under Alternatives 1, 2, and 3.

The IOS modeling suggested potential negative effects of Alternatives 1, 2, and 3 relative to the NAA. The differences between the NAA and Alternatives 1, 2, and 3 arise because of flow-survival relationships and less flow because of the Red Bluff and Hamilton City diversions under the Project. As discussed in the *Migration Flow-Survival* section above and Chapter 2, the Project includes pulse flow protection measures to be applied to precipitation-generated pulse flow events. There is also a 10,700-cfs Wilkins Slough bypass flow threshold from October 1 through June 14, corresponding with the threshold identified by Michel et al. (2021). Adaptive management would be utilized to assess and if necessary refine the diversion criteria based on biological monitoring (see Appendix 2D, Section 2D.6).

Table 11-27. Mean Female Adult Winter-Run Chinook Salmon Escapement by Water Year Type Based on IOS.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	4,485	4,234 (-6%)	4,322 (-4%)	4,266 (-3%)	4,295 (-4%)
Above Normal	3,679	3,506 (-5%)	3,579 (-3%)	3,557 (-4%)	3,549 (-4%)
Below Normal	3,967	3,714 (-6%)	3,802 (-4%)	3,735 (-3%)	3,796 (-4%)
Dry	3,625	3,538 (-2%)	3,622 (0%)	3,540 (-1%)	3,722 (3%)
Critically Dry	2,437	2,338 (-4%)	2,388 (-2%)	2,375 (-2%)	2,406 (-1%)
Total	3,763	3,579 (-5%)	3,659 (-3%)	3,606 (-3%)	3,671 (-2%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. Table only includes annual mean responses and does not consider model uncertainty. Percentage values are rounded; as a result, differences between percentages may not always appear consistent.

Table 11-28. Mean Winter-Run Chinook Salmon Proportional Egg Survival by Water Year Type Based on IOS.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	1.00	1.00 (0%)	1.00 (0%)	1.00 (0%)	1.00 (0%)
Above Normal	1.00	1.00 (0%)	1.00 (0%)	1.00 (0%)	1.00 (0%)
Below Normal	1.00	1.00 (0%)	1.00 (0%)	1.00 (0%)	1.00 (0%)
Dry	1.00	1.00 (0%)	1.00 (0%)	1.00 (0%)	1.00 (0%)
Critically Dry	0.95	0.96 (1%)	0.97 (2%)	0.96 (1%)	0.99 (4%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. Table only includes annual mean responses and does not consider model uncertainty. Percentage values are rounded; as a result, differences between percentages may not always appear consistent.

Table 11-29. Winter-Run Chinook Salmon Proportional Fry Survival by Water Year Type Based on IOS.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	0.96	0.96 (0%)	0.96 (0%)	0.96 (0%)	0.96 (0%)
Above Normal	0.96	0.96 (0%)	0.96 (0%)	0.96 (0%)	0.96 (0%)
Below Normal	0.96	0.96 (0%)	0.96 (0%)	0.96 (0%)	0.96 (0%)
Dry	0.96	0.96 (0%)	0.96 (0%)	0.96 (0%)	0.96 (0%)
Critically Dry	0.91	0.92 (0%)	0.92 (1%)	0.92 (0%)	0.93 (2%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. Table only includes annual mean responses and does not consider model uncertainty. Percentage values are rounded; as a result, differences between percentages may not always appear consistent.

Table 11-30. Winter-Run Chinook Salmon Proportional Juvenile River Migration Survival by Water Year Type Based on IOS.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	0.30	0.30 (-1%)	0.30 (-1%)	0.30 (-1%)	0.30 (-1%)
Above Normal	0.27	0.27 (-1%)	0.27 (-1%)	0.27 (-1%)	0.27 (-1%)
Below Normal	0.26	0.26 (-1%)	0.26 (-1%)	0.26 (-1%)	0.26 (-1%)
Dry	0.25	0.25 (0%)	0.25 (0%)	0.25 (0%)	0.25 (0%)
Critically Dry	0.25	0.25 (0%)	0.25 (0%)	0.25 (0%)	0.25 (0%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. Table only includes annual mean responses and does not consider model uncertainty. Percentage values are rounded; as a result, differences between percentages may not always appear consistent.

Table 11-31. Winter-Run Chinook Salmon Proportional Juvenile Through-Delta Migration Survival by Water Year Type Based on IOS.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	0.35	0.35 (0%)	0.35 (0%)	0.35 (0%)	0.35 (0%)
Above Normal	0.32	0.32 (-1%)	0.32 (-1%)	0.32 (-1%)	0.32 (-1%)
Below Normal	0.26	0.25 (-1%)	0.25 (-1%)	0.25 (-1%)	0.25 (-1%)
Dry	0.21	0.20 (-1%)	0.20 (-1%)	0.20 (-1%)	0.20 (-1%)
Critically Dry	0.17	0.17 (0%)	0.17 (0%)	0.17 (0%)	0.17 (0%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. Table only includes annual mean responses and does not consider model uncertainty. Percentage values are rounded; as a result, differences between percentages may not always appear consistent.

OBAN

OBAN modeling results are discussed in detail in Appendix 11I. Alternatives 1A, 1B, 2, and 3 had higher escapement abundance relative to the NAA over the 1922–2002 timeframe, but only Alternative 1B had higher abundance relative to the NAA over the period 1933–2002 (which was examined to address potential issues with production during earlier years because of model initialization). The probability of quasi-extinction (probability of spawner abundance below 100 fish) was higher in all alternatives relative to the NAA except Alternative 1B and Alternative 2, in which the probability was lower than the NAA. Egg to fry survival was also higher in Alternative 1B relative to the NAA, whereas all other alternatives had lower egg to fry survival compared to the NAA. Egg to fry survival is a function of temperature (mean daily water temperature in the Sacramento River at Bend Bridge) and flow (minimum monthly flow in the Sacramento River at Bend Bridge). Both Alternative 1B and Alternative 3 had higher median flow, whereas all alternatives except Alternative 3 had temperatures lower than the NAA on average. Delta survival was similar under the Project and the NAA.

CEQA Significance Determination for Alternatives 1, 2, and 3

The preceding subsections of this impact discussion provide the detailed information used for this CEQA (and NEPA) determination. This section provides a summary of this information.

In-Delta and upstream operational impacts of the Project on winter-run Chinook salmon generally would be limited. Related to temperature effects, observations of exceedance plots and differences in modeled mean monthly temperatures by water year type indicate that Alternatives 1, 2, and 3 and the NAA would be predominantly similar among alternatives during the period of presence of each life stage of winter-run Chinook salmon. However, the Martin and Anderson models predict that Alternative 3, during Critically Dry Water Years, would have slightly lower winter-run egg mortality due to slightly lower temperatures as compared to the NAA. In addition, SALMOD predicts a slight reduction in mortality and increase in annual production, particularly under Alternative 3, relative to the NAA in the Sacramento River.

Juvenile salmonids, including winter-run Chinook salmon, migrating downstream past the Red Bluff and Hamilton City intakes would be susceptible to entrainment through the fish screens if sufficiently small. The small proportion of juvenile winter-run sufficiently small sized to potentially be susceptible to entrainment would occur in July/August, a period during which the diversions for the Project generally would be similar in magnitude—or in some cases somewhat lower—to the NAA. Overall, entrainment risk would be expected to be similar between the NAA and Alternatives 1, 2, and 3 for juvenile winter-run Chinook salmon.

The available information generally suggests that impingement and screen passage/contact-related negative effects of the operation of the Red Bluff and Hamilton City intakes would be limited for all alternatives, particularly given that these effects would only apply to the subset of juvenile winter-run Chinook salmon encountering the intakes. The Red Bluff and Hamilton City fish screens are designed to protective standards for Chinook salmon fry and so near-field effects would be expected to be limited.

Based on available studies of predation effects at water intakes, the predation effects in the vicinity of the Red Bluff and Hamilton City intakes for the Project would be limited. The extent of in-water structure at the intakes would be the same under the NAA and the Project. Although overtopping during high flows can occur at the Red Bluff and Hamilton City intakes, leading to potential stranding of juvenile winter-run Chinook salmon, these are relatively infrequent events (e.g., approximately once per 100 years at Red Bluff) that would occur under the NAA and would not be changed by Alternatives 1, 2, and 3.

Results indicate that Alternatives 1, 2, and 3 would have no effect on redd scour and entombment for winter-run Chinook salmon in the Sacramento River. The results for winter-run Chinook salmon show few large changes in redd dewatering between the NAA and Alternatives 1, 2, and 3. Changes for most months and water year types under Alternatives 1, 2, and 3 are less than 2%. Overall, the effects of Alternatives 1, 2, and 3 on winter-run redd dewatering are minor.

Most differences in spawning WUA between the Project and the NAA in all three river segments are less than 3%. The Project is not expected to substantially affect winter-run spawning WUA. The means for fry rearing WUA differ by less than 3% between Alternatives 1 and 2 and the NAA. Under Alternative 3, however, there are a number of larger differences and all but one of these (September of Below Normal Water Years in Segment 6) constitutes a reduction in rearing WUA. The largest of the reductions under Alternative 3 range up to 6% for July and August of Above Normal Water Years in Segment 5. These results indicate that Alternative 3 would have a negative effect on rearing habitat WUA for winter-run fry in the Sacramento River. Alternatives 1 and 2 would have little effect.

All the means for juvenile rearing WUA differ by <3% between Alternatives 1, 2, and 3 and the NAA, except for a 4% increase in Segment 6 for November of Critically Dry Water Years under Alternative 3 and a 3% reduction in Segment 4 for October of Wet Water Years under Alternative 3. These results indicate that Alternatives 1, 2, and 3 would have little effect on rearing habitat WUA for winter-run juveniles in the Sacramento River.

As described in Chapter 2, the Project's diversion criteria include restrictions to maintain Bend Bridge and Wilkins Slough flows, which have the effect of limiting changes to Yolo Bypass spill frequency and duration under Alternatives 1, 2, and 3 relative to the NAA. Thus, the Project would have limited potential for negative effects on Yolo Bypass floodplain inundation and access for winter-run Chinook salmon. Acreages of Yolo Bypass suitable inundated habitat under Alternatives 1, 2, and 3 would generally be lower than those under the NAA, but overall reductions are less than 2% (100 acres), which is not expected to substantially affect production of winter-run Chinook salmon. The results of the frequency analysis of inundation of events for the Yolo Bypass generally show only minor differences between Alternatives 1, 2, and 3 and the NAA. The Project would operate to avoid effects on the Yolo Bypass Fremont Weir Big Notch Project's (Big Notch) ability to achieve the same level of performance for salmonids in the Sacramento River as it would absent the Project. However, the Adaptive Management Plan for the Project recognized there is uncertainty about the performance of the Big Notch and the effects of the Project on it. Monitoring will be conducted, in cooperation with the State, to determine whether there is an effect and, if so, what the magnitude of that effect would be on entrainment of juvenile salmon into the Yolo Bypass. If there is an adverse effect, a science-based adaptive management approach will be employed to determine how to adjust diversions 158 RMs upstream of the Big Notch to maintain its efficiency for entraining juvenile salmon into the Yolo Bypass.

The Big Notch Project also is intended to reduce migratory delays and loss at the Fremont Weir of adult salmon, steelhead, and sturgeon that are migrating upstream through the Yolo Bypass. As with juvenile passage, the Project would operate to avoid affecting adult passage at Fremont Weir. The number of days meeting adult Chinook salmon passage criteria at Fremont Weir generally would be similar with or without the Project (Table 11-21). This too will be evaluated and adaptively managed as required.

The Sutter Bypass, when inundated, provides important juvenile rearing habitat for Chinook salmon and steelhead, as discussed for the Yolo Bypass. For the Sutter Bypass the modeling results indicate that Alternatives 1, 2, and 3 would produce little change in suitable habitat as compared to the NAA. Changes in Sacramento River flow and flow entering Sutter Bypass as a result of the Project have the potential to change the number of days meeting adult Chinook salmon passage criteria at the three Sutter Bypass weirs (Moulton, Tisdale, and Colusa). The results of this analysis indicated that there would be no days meeting adult salmon passage criteria during 2009–2018 at Moulton or Tisdale Weirs under the NAA or Alternatives 1, 2, and 3. For Colusa Weir, the number of days meeting fish passage criteria generally would be similar with or without the Project.

As described in Chapter 2, the Project includes pulse flow protection measures to be applied to precipitation-generated pulse flow events. There is also a 10,700-cfs Wilkins Slough bypass flow threshold from October 1 through June 14. Adaptive management would be utilized to assess and if necessary refine the diversion criteria based on biological monitoring (see Appendix 2D, Section 2D.6). Sites Reservoir diversion criteria effectively limit diversions during the historical periods of fish movement as reflected in Red Bluff rotary screw trap data, and application of the flow-threshold criteria from Michel et al. (2021) suggests that flow-survival effects on juvenile Chinook salmon (including winter-run Chinook salmon) would be greatly limited by the Project's diversion criteria because there is essentially no difference in predicted survival between the Project and NAA.

Changes in flow entering the Delta as a result of the operations of the Project would be expected to give only limited changes to rearing habitat for juvenile winter-run Chinook salmon in the Delta. Analysis showed that during the main period of juvenile winter-run Chinook salmon occurrence in the Delta (i.e., December–April), there were 0%–1% differences in mean through-Delta survival between the NAA and Alternatives 1, 2, and 3. There were also limited differences in inundation of juvenile Chinook salmon rearing habitat in the north Delta as a result of differences in flow entering the Delta.

The IOS and OBAN life cycle models suggested potential negative effects of the Project relative to the NAA. Such differences between the NAA and the Project arise because of factors including flow-survival relationships and less flow because of the Red Bluff and Hamilton City diversions. As previously discussed, the Project includes pulse flow protection measures to be applied to precipitation-generated pulse flow events. There is also a 10,700-cfs Wilkins Slough bypass flow threshold from October 1 through June 14. Adaptive management would be utilized to assess and if necessary refine the diversion criteria based on biological monitoring (see Appendix 2D, Section 2D.6).

Based on the analyses provided, operation of Alternative 1, 2, or 3 would not have a substantial adverse effect, either directly or through habitat modifications, on winter-run Chinook salmon. In consideration of the analyses within the preceding subsections of this impact discussion and summarized above, operation impacts of Alternative 1, 2, or 3 would be less than significant. No mitigation is necessary.

NEPA Conclusion for Alternatives 1, 2, and 3

Operation effects on Chinook salmon would be the same as described above for CEQA. In-Delta and upstream operational effects of the Project on winter-run Chinook salmon generally would be limited. Related to temperature effects, observations of exceedance plots and differences in modeled mean monthly temperatures by water year type indicate that Alternatives 1, 2, and 3 and the NAA would be predominantly similar among alternatives during the period of presence of each life stage of winter-run Chinook salmon. However, the Martin and Anderson models predict that Alternative 3, during Critically Dry Water Years, would have slightly lower winter-run egg mortality due to temperatures as compared to the NAA. In addition, SALMOD predicts a slight reduction in mortality and increase in annual production, particularly under Alternative 3, relative to the NAA in the Sacramento River.

As described in Chapter 2, the Project's diversion criteria include restrictions to maintain Bend Bridge and Wilkins Slough flows, which have the effect of limiting changes to Yolo Bypass spill frequency and duration under Alternatives 1, 2, and 3 relative to the NAA. Thus, the Project would have limited potential for negative effects on Yolo Bypass floodplain inundation and access for winter-run Chinook salmon. Also described in Chapter 2, the Project includes pulse flow protection measures to be applied to precipitation-generated pulse flow events. There is also a 10,700-cfs Wilkins Slough bypass flow threshold from October 1 through June 14, corresponding with the threshold identified by Michel et al. (2021). Operation of Alternative 1, 2, or 3 would have no adverse effect on winter-run Chinook salmon.

Impact FISH-3: Operations Effects on Spring-Run Chinook Salmon

Alternatives 1, 2, and 3

Effects of Alternatives 1, 2, and 3

Potential exposure of spring-run Chinook salmon to the effects of Alternatives 1, 2, and 3 is dependent on the species' spatiotemporal distribution. As described for winter-run Chinook salmon, several sources of information provide important context.

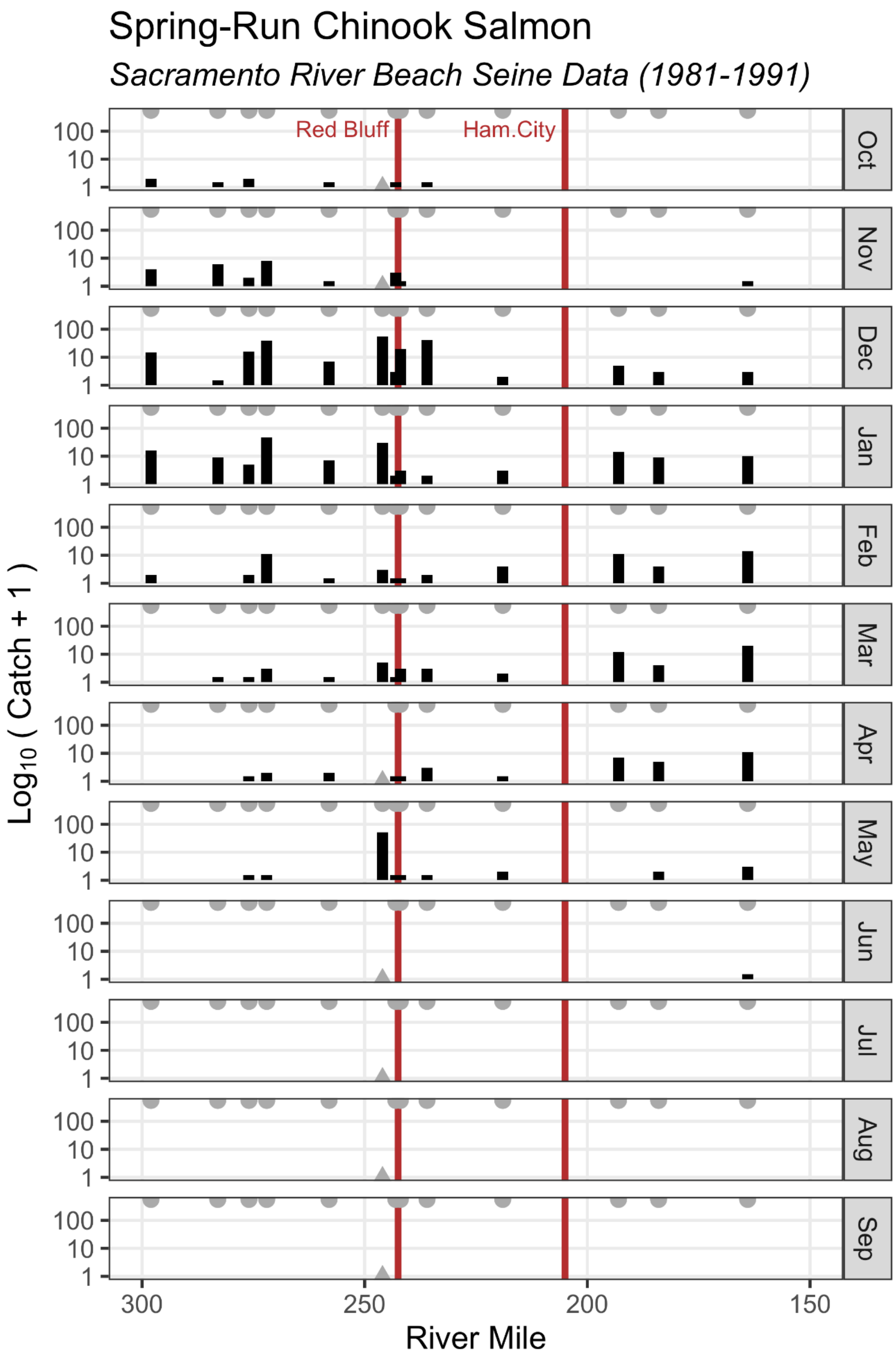
The main patterns of juvenile spring-run Chinook salmon occurrence at the locations documented in the SacPAS summary in Appendix 11A1 include:

- RBDD rotary screw traps (Appendix 11A1, Figure 11A1-10): Passage consistently begins mid-October, apart from the 2017 cohort which did not start until mid-November, ending in early May to the first of August. The first half (50%) passes from mid-November to early April. Prior to 2014, the main portion (90%) began passing from mid-October to late December, though from 2015 to 2017 the main portion began in mid-March. All years, the main portion (90%) ended from mid-April to early May, except for the 2007 cohort which finished in late December.
- Tisdale Weir rotary screw traps (Appendix 11A1, Figure 11A1-11): Passage begins each year from early November to mid-March and is completed in late April to early May. Half of the fish (50%) have passed the weir typically in early April but may be as early as early January. The main portion (90%) of the fish begin to pass in early November to late March and finish from mid-March to late April.
- Knights Landing rotary screw traps (Appendix 11A1, Figure 11A1-12): Passage begins in early October to mid-December and ends from late April to early May. The first half (50%) passes from mid-December to mid-April. The main portion (90%) begins in early December to mid-April and ends in April. The exception to these trends were the 2012 and 2015 cohorts. The 2012 cohort began in late November and finished within approximately 3 weeks (first to last), with the first half (50%) passing in early December. The 2015 cohort completed its entire passage in approximately 2 weeks (first to last) at the end of December.
- Sacramento beach seines (Appendix 11A1, Figure 11A1-13): Occurrence begins the first of November to mid-January and finishes in late March to early May. The first half (50%) of the fish occurs from mid-December to late March. The main portion (90%)

begins to occur in early December to early March and finishes from late January to mid-April.

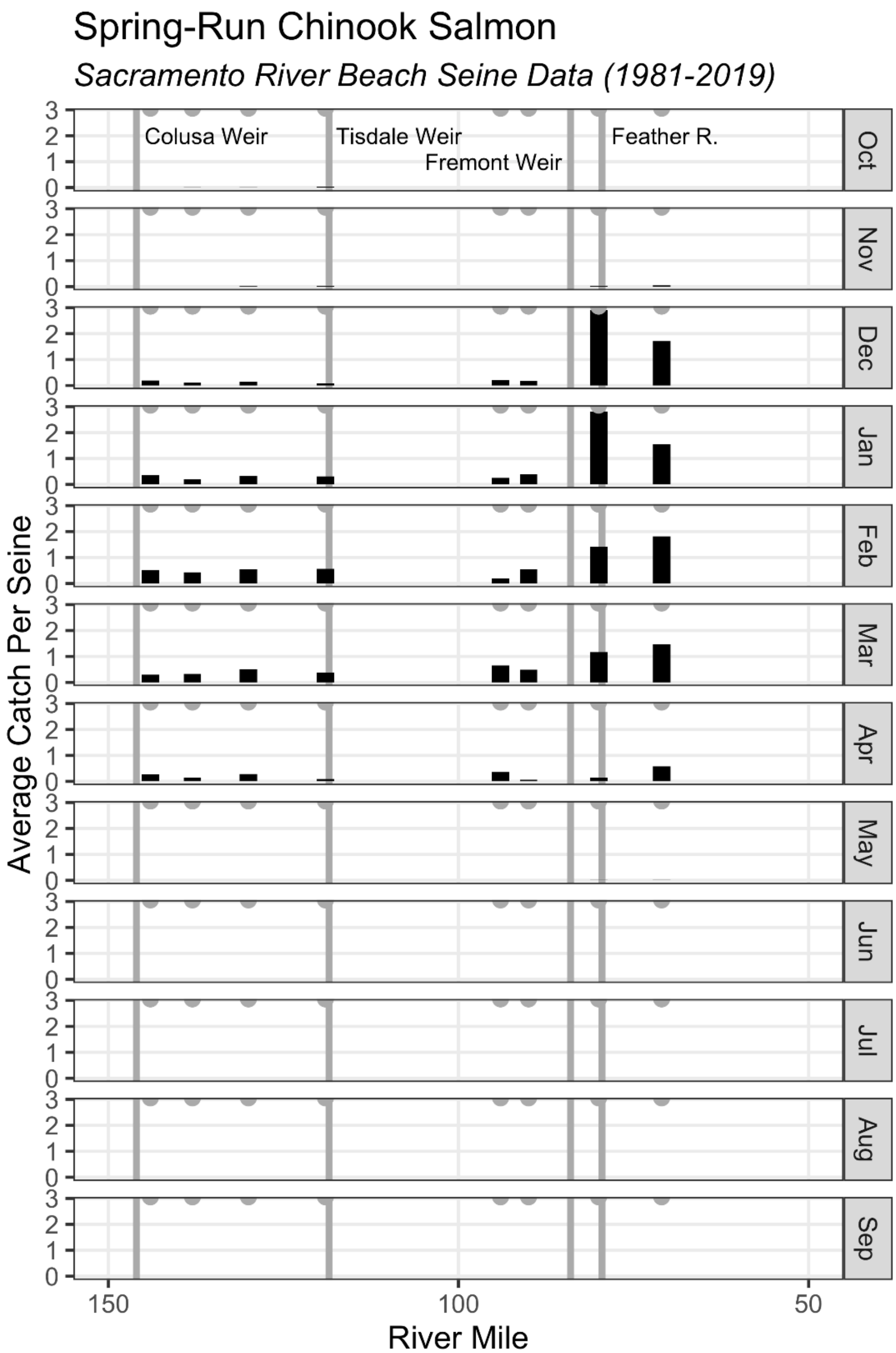
- Sacramento trawls (Sherwood Harbor) (Appendix 11A1, Figure 11A1-14): Occurrence begins in late November to late February and ends from late April to late June. The first half (50%) of the fish occurs by April. The main portion (90%) starts in mid-December to the first of April and ends in about 3 weeks from mid-April to early May.
- Chipps Island trawls (Appendix 11A1, Figure 11A1-15): Occurrence begins from mid-December to early April and ends from mid-May to the end of September. The first half (50%) occurs by April. The main portion (90%) occurs within a small window of approximately 6 weeks or less that begins in late March to early April and ends at the end of April to mid-May.
- Salvage (unclipped, length-at-date) (Appendix 11A1, Figure 11A1-16): Salvaged begins in the first of January to late March, except for the 2000 cohort which started in late September. Salvage ends in mid-May to mid-June. The first half (50%) occurs by early April to mid-May. The main portion (90%) of the fish begin to be salvaged in mid-March to mid-April and finishes from mid-April to the first of June.
- Salvage (clipped, length-at-date) (Appendix 11A1, Figure 11A1-17): Salvage begins in late February to the first of May and ends from early April to mid-June. The first half (50%) occurs by late March to early June. The main portion (90%) begin to occur in early March to early May and ends from late March to early June.
- Salvage (clipped, CWT-Race) (Appendix 11A1, Figure 11A1-18): Salvage begins in mid-March to mid-May and ends from mid-April to early June. The first half (50%) occurs by late March to mid-May. The main portion (90%) begins to occur by mid-March to mid-May and finishes from late March to late May.

Juvenile spring-run Chinook salmon catches in extensive beach seining during 1981–1991 had a similar pattern to winter-run, where catches spread downstream from the upper reaches through time (Figure 11-13). The peaks in spring-run catches, however, lagged behind winter-run by about 2–3 months. In both data sets, there appears to be spatial and temporal overlap between juveniles of winter- and spring-run during the winter. Spring-run juveniles were absent during summer and fall in the lower reaches of the Sacramento River during 1981–2019 sampling (Figure 11-14).



Source: Johnson et al. 1992:Table 4. Note: Sampled sites are denoted by grey semicircles at the top of each plot. Sites that were not sampled in a given month are denoted by grey triangles at the bottom of a panel. Values denoted as "<1" by Johnson et al. are shown as 0.5. Red lines indicate locations of Red Bluff and Hamilton City diversions.

Figure 11-13. Mean Monthly Catch Per Beach Seine of Juvenile Spring-Run Chinook Salmon in the Sacramento River Between River Mile 164 and River Mile 298, 1981–1991.



Source: Interagency Ecological Program et al. 2019. Note: Sampled sites are denoted by grey semicircles at the top of each plot. Grey lines indicate locations of Colusa, Tisdale, and Fremont Weirs, and the confluence with the Feather River.

Figure 11-14. Mean Monthly Catch Per Beach Seine of Juvenile Spring-Run Chinook Salmon in the Sacramento River Between River Mile 71 and River Mile 144, 1981–2019.

Between 2009 and 2020, the mean percentage of in-river spawning adult spring-run Chinook salmon returning to tributaries upstream of Red Bluff was ~7%, and upstream of Hamilton City was ~24%; the mean percentage of total fish (hatchery plus in-river) was just over 3.5% upstream of Red Bluff and 14.5% upstream of Hamilton City (Table 11-32). This indicates that appreciable portions of spring-run Chinook salmon would not be expected to have the potential for near-field effects from the diversions, with most of the ESU occurring downstream in tributaries such as Butte Creek, for example. These downstream fish would have the potential for far-field effects from Alternatives 1, 2, and 3 (e.g., flow-survival effects for juveniles migrating downstream, discussed further below in the *Migration Flow-Survival* section).

Table 11-32. Abundance and Percentage of Spring-Run Chinook Salmon Adult Escapement Upstream and Downstream of the Red Bluff and Hamilton City Intakes, 2009–2020.

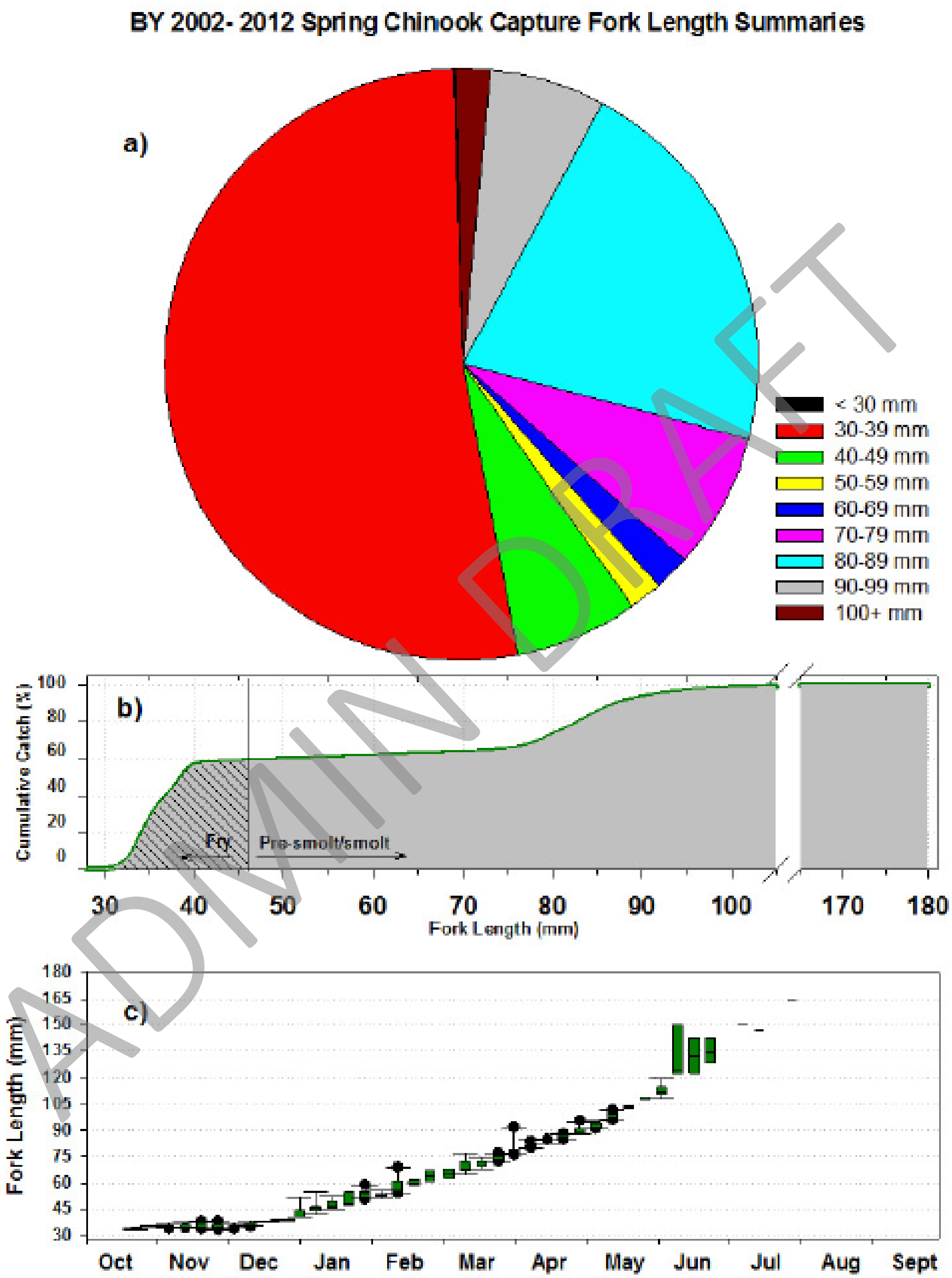
Year	All Fish Abundance Upstream of Red Bluff	% of All Fish Upstream of Red Bluff	In-River Abundance Upstream of Red Bluff	% of In-River Upstream of Red Bluff	All Fish Abundance Upstream of Hamilton City	% of All Fish Upstream of Hamilton City	In-River Abundance Upstream of Hamilton City	% of In-River Upstream of Hamilton City	All Fish Abundance Downstream of Hamilton City	% of All Fish Downstream of Hamilton City	In-River Abundance Downstream of Hamilton City	% of In-River Downstream of Hamilton City
2009	314	4.5	314	9.1	764	17.2	764	22.1	3,682	82.8	2,693	77.9
2010	210	3.5	210	7.1	971	21.0	971	32.8	3,654	79.0	1,993	67.2
2011	169	1.5	169	2.9	812	10.4	812	14.0	6,964	89.6	4,995	86.0
2012	868	2.3	868	4.6	2,371	10.6	2,371	12.7	20,055	89.4	16,317	87.3
2013	1,382	3.5	1,382	7.1	2,734	11.5	2,734	14.0	21,076	88.5	16,782	86.0
2014	526	3.7	526	7.4	2,042	20.6	2,042	28.7	7,859	79.4	5,083	71.3
2015	226	9.5	226	18.9	626	13.7	626	52.4	3,955	86.3	569	47.6
2016	209	1.6	209	3.2	722	8.9	722	11.2	7,390	91.1	5,731	88.8
2017	59	2.8	59	5.6	544	34.2	544	51.4	1,047	65.8	515	48.6
2018	131	2.3	131	4.7	443	9.0	443	15.8	4,472	91.0	2,362	84.2
2019	204	0.6	204	1.3	976	4.9	976	6.0	19,080	95.1	15,213	94.0
2020	235	7.0	235	13.9	407	12.6	407	24.1	2,835	87.4	1,281	75.9
Mean	378	3.6	378	7.1	1,118	14.5	1,118	23.8	8,506	85.5	6,128	76.2
Median	218	3.2	218	6.3	788	12.0	788	18.9	5,718	88.0	3,844	81.1

Source: Adapted from Azat 2021. Note: All fish = hatchery fish + in-river fish. Totals upstream of Red Bluff and Hamilton City are cumulative, so that fish counted as upstream of Red Bluff are also counted as upstream of Hamilton City. Table does not include any estimates of fish introduced as part of the San Joaquin River Restoration Program, or spring-running fish in San Joaquin River Basin tributaries.

Sacramento River

Near-Field Effects

Spring-run Chinook salmon would have the potential for similar types of near-field effects to those previously discussed for winter-run Chinook salmon (i.e., entrainment; impingement and screen contact; predation; stranding behind screens; and attraction to reservoir discharge). The potential for effect would differ relative to winter-run Chinook because of spatiotemporal differences in species occurrence. In particular, as discussed above, less than one quarter of spring-run Chinook salmon would be expected to pass the intakes (Table 11-24). As discussed for winter-run, few spring-run Chinook juveniles would have the potential for entrainment given their size (Figure 11-15). The Red Bluff and Hamilton City fish screens are designed to protective standards for Chinook salmon fry and so near-field effects would be expected to be limited. There would be greater temporal overlap of adult spring-run Chinook with reservoir releases potentially beginning in April/May than for winter-run Chinook (Appendix 11A), although again this would only apply to up to 25% of adult spring-run returning to tributaries upstream of the Sacramento River discharge. (There are no spring-run Chinook-bearing tributaries between Hamilton City and the Sacramento River discharge.) As described for winter-run Chinook salmon, adult spring-run Chinook salmon could be attracted to the flow from the Sacramento River discharge under Alternative 2. The discharge structure would include a vertical drop exclusion barrier to prevent the passage of anadromous fish into the pipeline. The weir and apron would meet NMFS guidelines for a combination velocity and vertical drop barrier for the exclusion of fish. The design of the apron and weir would exclude anadromous fish and eliminate stranding risk.



Source: Poytress et al. 2014:85. Note: BY = brood year.

Figure 11-15. Spring-Run Chinook Salmon Fork Length (a) Capture Proportions, (b) Cumulative Capture Size Curve, and (c) Average Weekly Median Boxplots, As Sampled at Red Bluff Diversion Dam Rotary Screw Traps, October 2002–June 2013.

Far-Field Effects

As described previously in Impact FISH-2, there are likely multiple opportunities that would arise in real-time operations to coordinate exchanges between Sites Reservoir and Shasta Lake that could benefit anadromous fish. These are described in the project description and reflected in the modeling for this document to some extent. However, due to the unique conditions in each year, additional opportunities for exchanges and coordination of real-time operations exist beyond those modeled for this document. The Authority and Reclamation intend to work together to better reflect the exchanges in the modeling with the goal of substantiating the Project's benefits to anadromous fish.

Temperature Effects

Operation of Sites Reservoir has the potential to change water temperatures in the Sacramento River upstream of the Delta that could affect the life stages of spring-run Chinook salmon present. As described in Appendix 11B, the three methods used to analyze temperature-related effects on spring-run Chinook salmon in the Sacramento River were: (1) Physical Model Output Characterization; (2) Water Temperature Index Value/Range Exceedance Analysis; and (3) SALMOD. More details on these methods are provided in Appendix 11B and Appendix 11H.

The Authority and Reclamation evaluated water temperature model outputs during the period of presence and in the locations of each life stage of spring-run Chinook salmon in the Sacramento River upstream of the Delta (see Appendix 11B, Table 11B-2 for timing and locations). Visual observation of exceedance plots and differences in modeled mean monthly temperatures by water year type between Alternatives 1, 2, and 3 and the NAA in the Sacramento River below Keswick, at Balls Ferry, at Bend Bridge, below RBDD[18], and at Butte City indicates that water temperatures would be predominantly similar among Alternatives 1, 2, and 3 during the period of presence of each life stage of spring-run Chinook salmon (Appendix 6C, Tables 6C-5-1a to 6C-5-4c, Tables 6C-7-1a to 6C-7-4c, Tables 6C-9-1a to 6C-9-4c, Tables 6C-10-1a to 6C-10-4c, Tables 6C-12-1a to 6C-12-4c; Figures 6C-5-1 to 6C-5-18, Figures 6C-7-1 to 6C-7-18, Figures 6C-9-1 to 6C-9-18, Figures 6C-10-1 to 6C-10-18, Figures 6C-12-1 to 6C-12-18). At all locations, mean monthly water temperatures for all months within all water year types under Alternatives 1A and 1B were within 0.6°F of the NAA with the exception of August of critical years at Butte City, in which mean monthly water temperature would be 0.8°F lower under the NAA than under Alternatives 1A and 1B. Water temperature modeling results for Alternatives 2 and 3 were similar to those of Alternative 1 at all locations. Overall, these differences would be biologically inconsequential due to their low frequency and small magnitude.

Results for each alternative of the analysis of exceedance above water temperature index values for spring-run Chinook salmon in the Sacramento River from Appendix 11B, Table 11B-2 are presented in Appendix 11D, Table 11D-25 through Table 11D-44 and summarized in Table 11-33. For each life stage and at all locations evaluated, there were no month and water year type combinations in which both: (1) the percent of days that exceeded the index values was more than 5% greater under Alternative 1 than under the NAA; and (2) the exceedance per day was more than 0.5°F greater under Alternative 1 than under the NAA. Results of the exceedance

[18] The RBDD, which was decommissioned in 2013, and the RBPP are co-located, and the names may be used interchangeably when referring to geographic locations

analysis for Alternative 2 are similar to Alternative 1 with no month and water year type combinations in which both criteria were met for any life stage at all locations. For Alternative 3, there was one month and water year type combination (July of Above Normal Water Years) at Hamilton City with an unfavorable result for the juvenile rearing and migration life stages in which there were 10.8% more days than the NAA exceeding the 64°F 7DADM index value and the mean daily exceedance on those days was 0.6°F greater than the NAA. There was also one month and water year type combination (August of Critically Dry Water Years) below Keswick with a favorable result in which both criteria were met in a positive way (the percent of days that exceeded the index values was more than 5% lower under Alternative 3 than under the NAA, and the exceedance per day was more than 0.5°F lower under Alternative 3 than under the NAA) for both the 53.5°F mean daily and 55.4°F 7DADM temperature values analyzed. Because these biologically meaningful effects occurred in only one month of one water year type, they are not expected to be persistent enough to affect spring-run Chinook salmon at a population level.

Table 11-33. Number of Month and Water Year Type Combinations that Satisfy Both Criteria for Being Biologically Meaningful in the Water Temperature Index Value Analysis, Spring-run Chinook Salmon, Sacramento River[1,2,3]

Location	Spawning and Egg Incubation				Juvenile Rearing and Emigration				Adult Immigration				Adult Holding			
	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3
Below Keswick	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Below Clear Creek	0	0	0	2 favorable (53.5°F mean daily and 55.4°F 7DADM, August, Critically Dry Water Years)	0	0	0	0	NA	NA	NA	NA	NA	NA	NA	NA
Balls Ferry	0	0	0	0	0	0	0	0	NA	NA	NA	NA	0	0	0	0
Bend Bridge	0	0	0	0	0	0	0	0	0	0	0	0	NA	NA	NA	NA
Below Red Bluff Diversion Dam	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Hamilton City	NA	NA	NA	NA	0	0	0	1 unfavorable (July, Above Normal Water Years)	NA	NA	NA	NA	NA	NA	NA	NA

[1] Biologically Meaningful Criteria include: (1) the difference in frequency of exceedance between the NAA and the alternative was greater than 5%, and (2) the difference in average daily exceedance between the NAA and the alternative was greater than 0.5°F.

[2] Index values for each life stage are located in Appendix 11B, Table 11B-2.

[3] Full results presented in Appendix 11D, Table 11D-25 through Table 11D-44.

[4] NA = Not Analyzed

Flow-Related Physical Habitat Conditions

Redd Scour Entombment

Loss of redds to scouring or entombment occurs when flows are high enough to mobilize sediments, destroying redds and their incubating eggs and alevins, or entombing the redds when sediments are redeposited. A flow of 40,000 cfs was selected as the scour flow threshold for the Sacramento River based on estimates in the literature (Appendix 11N, Table 11N-10).

The probability of redd scour and entombment was estimated for spring-run by computing the percentage of days with flows exceeding 40,000 cfs in the USRDOM 82-year daily flow record (29,952 days in total) at four locations between Keswick Dam and the RBPP during the months of spring-run spawning and incubation. The results for the NAA and Alternatives 1, 2, and 3 show that the probability of scour and entombment is consistently low for spring-run (Appendix 11N, Table 11N-22 through Table 11N-25). Alternatives 1, 2, and 3 have no adverse effect on the frequency of scouring and entombment flows. The results indicate that Alternatives 1, 2, and 3 would have no adverse effect on redd scour and entombment for spring-run in the Sacramento River.

Redd Dewatering

The percentage of redds in the Sacramento River lost to dewatering was estimated using tables in USFWS (2006) that relate spawning and dewatering flows to percent reductions in species-specific spawning habitat WUA (Appendix 11N). USRDOM flow data, which has a daily time-step, are available for three locations in this river section: Keswick Dam (RM 302), the Sacramento River at Clear Creek (RM 289), and the Sacramento River at Battle Creek (RM 271). A single relationship for flows was developed for the entire river section, but the flows used to estimate redd dewatering in the current analysis were those that best matched the longitudinal distribution of the redds of the different salmon runs in the river as estimated from aerial redd surveys conducted by CDFW from 2003 through 2019. Spawning of spring-run occurs primarily between the ACID Dam and Airport Road (Table 11N-1), so Sacramento River at Clear Creek flows were used to analyze spring-run redd dewatering. As discussed in Appendix 11N, percentage of redd dewatering for spring-run was computed from the Sacramento at Clear Creek flows using the fall-run flows vs. redd dewatering relationship. This substitution was made because field data for spring-run were inadequate for developing the relationship, and fall-run spawning distributions and timing are most similar to those of spring-run (U.S. Fish and Wildlife Service 2006).

Results are presented using the grand mean percentages of redds dewatered for each month of spawning, August through October, and each water year type and all water year types combined. The expected time for incubation of eggs and alevins is 3 months (Appendix 11N). Because changes in Project-related flow any time during this period can affect redd dewatering, the complete spawning and egg/alevin incubation periods (August–November through October–January) are provided in the results (Table 11N-14). The means of the redd dewatering estimates under the NAA and Alternatives 1, 2, and 3 are compared using absolute differences rather than relative differences (percent change) because many of the values for percentages of redds dewatered are small. Expressing changes of small values as percent changes may result in large values that may be misleading.

The results of the redd dewatering analysis for spring-run (Table 11N-14) show 2% increases in redd dewatering for eggs spawned in August and September of Critically Dry Water Years under Alternatives 1, 2, and 3 and reductions of up to 6% under Alternative 3 during August of Above Normal and Below Normal Water Years and September of Below Normal Water Years. Changes for most months and water year types under Alternatives 1, 2, and 3 are less than 2%. In general, Alternatives 1, 2, and 3 are not expected to substantially affect spring-run redd dewatering.

Spawning Habitat Weighted Usable Area

The suitability of physical habitat for salmonid spawning is largely of function of the availability of clean, coarse gravel for constructing redds, favorable depths, and suitable flow velocities. Instream flow potentially affects all these habitat characteristics and often affects the availability of suitable habitat. Habitat suitability for spawning was analyzed using WUA curves developed by USFWS and others from results of field studies and hydraulic modeling (Appendix 11K).

Spawning habitat for spring-run Chinook salmon was not estimated directly by USFWS (2003a, 2006) and no spring-run Chinook salmon WUA curves for the Sacramento River are available, so spring-run spawning habitat was modeled using the WUA curves provided by USFWS (2003b, 2006) for fall-run Chinook salmon. As noted by USFWS (2003a), the validity of using the fall-run WUA curves to characterize spring-run spawning habitat is uncertain.

To evaluate the effects of the NAA and Alternatives 1, 2, and 3 on spring-run spawning habitat, spring-run spawning WUA was estimated for flows during the August through October spawning period under Alternatives 1, 2, and 3 and the NAA in the same three segments of the Sacramento River that were used for winter-run (Figure 11-12). The redd distribution data for spring-run indicate that about 12%, 66% and 13% of spring-run redds occur within Segments 6, 5, and 4, respectively (Table 11-10).

Mean spawning WUA for spring-run under Alternatives 1, 2, and 3 differs from the NAA by more than 3% for only a few months and water year types, with most of these differences occurring under Alternative 3 (Table 11K-5 through Table 11K-7). The largest difference is a 16% increase under Alternative 3 in Segment 5 for August of Above Normal Water Years (Table 11K-6). The largest reduction in spawning WUA for spring-run is a 7% reduction, under Alternative 3 in Segment 4 for October of Wet Water Years. Other relatively large differences in WUA between Alternatives 1, 2, and 3 and the NAA are 4% to 6% reductions occurring in Critically Dry Water Years during August in Segment 5 and during September in Segment 4. These results indicate that Alternatives 1, 2, and 3 would lead to some reductions and increases of spring-run spawning habitat WUA. Spawning habitat conditions for spring-run are most important in Segment 5 (Table 11-10), which has one large increase and several smaller increases and reductions (Table 11K-6). On balance, Alternatives 1, 2, and 3 are not expected to substantially affect spring-run spawning WUA.

Rearing Habitat Weighted Usable Area

The suitability of physical habitat for salmonid rearing is largely a function of water depth, flow velocity, and the availability and type of cover. Instream flow potentially affects all these habitat characteristics and often affects the availability of suitable habitat. Habitat suitability for rearing was analyzed using WUA curves developed by USFWS and others from results of field studies and hydraulic modeling (Appendix 11K).

Rearing habitat WUA for spring-run was not estimated directly but was modeled using the fry and juvenile rearing habitat WUA curves obtained for fall-run Chinook salmon in Segments 4, 5, and 6 (U.S. Fish and Wildlife Service 2005b). The validity of using the fall-run Chinook salmon rearing WUA curves to characterize spring-run Chinook salmon rearing habitat is uncertain (U.S. Fish and Wildlife Service 2005b).

Rearing WUA in the Sacramento River was separately determined for spring-run fry and juveniles for a range of flows in Segments 4, 5, and 6. To estimate changes in rearing WUA that would result from Alternatives 1, 2, and 3 relative to the NAA, the fry and juvenile rearing habitat WUA curves developed for each of these segments were used with mean monthly CALSIM II flow estimates for the corresponding segments of the river under Alternatives 1, 2, and 3 and the NAA during the spring-run fry (November through February) and juvenile (year-round) rearing periods.

The largest reductions and increases in mean fry rearing WUA between Alternatives 1, 2, and 3 and the NAA are 5% reductions in November of Critically Dry Water Years and December of Dry Water Years in Segment 5 under Alternative 3 and 2% increases in January of Above Normal Water Years in Segment 6 under Alternatives 1A and 1B (Table 11K-29 and Table 11K-30). All differences for Alternatives 1A and 1B and most for Alternative 3 are <3%. In Segment 4, all differences are <3% for all three alternatives. These results indicate that Alternatives 1, 2, and 3 would have little effect on rearing habitat availability for spring-run fry in the Sacramento River.

Because some rearing by spring-run juveniles occurs throughout the year, all months are included in the spring-run juvenile rearing WUA analysis (Table 11K-32 through Table 11K-34). In Segment 6, few or none of the means for Alternatives 1A and 1B, but many of the means for Alternative 3 differ from the NAA means by more than 3%. All but one of the >3% differences constitute increases in rearing WUA. The largest differences are 7% increases in June and August of Above Normal Water Years under Alternative 3 and the largest reduction is 3% in September of Dry Water Years under Alternative 3. In Segment 5, relatively few of the differences in rearing WUA between Alternatives 1, 2, and 3 and the NAA are >3%. The largest differences are 5% and 6% increases in June of Above Normal and Below Normal Water Years under Alternative 3 (Table 11K-33). The largest reduction is a 3% reduction in October of Wet Water Years under Alternative 3. Segment 4 has more and larger differences between Alternatives 1, 2, and 3 and the NAA than the other two river segments (Table 11K-34). The largest increases are 17% in June of Above Normal and Below Normal Water Years under Alternative 3, 16% in August of Above Normal Water Years under Alternative 3, 15% in July of Above Normal Water Years under Alternative 3, and 12% increase in June of Above Normal Water Years under Alternative 1B. The largest reductions in Segment 4 are 8% to 9% reductions in August of Critically Dry Years under all three alternatives. In all segments combined, increases in rearing WUA of >3% far outnumber reductions of >3%, especially for Alternative 3.

The results for spring-run juvenile rearing WUA indicate that Alternatives 1, 2, and 3 would generally have little effect on rearing habitat in Segments 6 and 5 (Table 11K-32 and Table 11K-

33), but Alternative 3 would have relatively large effects in Segment 4, including substantial increases during late spring and summer and smaller reductions during late summer and fall (Table 11K-34). Increases in WUA outnumber reductions in the results and more of them are especially large (>10%). Furthermore, the increases occur during spring and summer, when the juveniles are younger and perhaps more vulnerable to reductions in habitat availability. On balance, Alternatives 1, 2, and 3 are expected to have little effect on spring-run fry rearing habitat availability and to increase spring-run juvenile rearing habitat WUA.

Juvenile Stranding

The juvenile stranding estimation procedure, which is identical for all the salmonids, is described in the juvenile stranding discussion for Impact FISH-2.

Spring-run fry are present in the upper Sacramento River primarily from about November through February (Table 11N-27). During this period, juvenile stranding under Alternatives 1, 2, and 3 includes many large increases and reductions in stranding under Alternatives 1, 2, and 3 relative to the NAA in all three reaches of the river (Table 11N-28 and Table 11N-30). The most frequent large changes (>10%) for the period occur in the Keswick and Battle Creek reaches under Alternative 3, with a maximum increase of 18% in January of Wet Water Years in the Keswick reach and a maximum reduction of 17% in January of Dry Water Years in the Keswick reach under Alternative 1B (Table 11N-28). Overall, large (>10%) reductions in stranding are about three times as frequent as large increases (Table 11N-28 and Table 11N-30). The summary tables show reductions in juvenile stranding for spring-run in the Battle Creek reach for Alternatives 1, 2, and 3 (Table 11N-33) and show increases in the Keswick and Clear Creek reaches under Alternative 3 (Table 11N-31 and Table 11N-32). On balance, Alternatives 1, 2, and 3 are not expected to substantially affect spring-run juvenile stranding.

SALMOD

The Authority used the SALMOD model to ascertain the potential effect of Alternatives 1, 2, and 3 on spring-run Chinook salmon mortality and potential production in the Sacramento River. A full description of the model can be found in the California WaterFix Biological Assessment (California Department of Water Resources 2016), Attachment 5.D.2, *SALMOD Model.* The SALMOD model outputs for spring-run Chinook salmon are presented in Appendix 11H, Table 1b-1 through Table 1b-4, Table 2b-1 through Table 2b-4, and Figure B-b-1 through Figure B-b-19. For all water year types combined for all life stages and source of mortality, mean annual spring-run Chinook salmon potential production would be similar under Alternative 1A and 1B (0% difference) relative to the NAA (Appendix 11H, Table 2b-1, Table 2b-2, Figure B-a-1). Further, differences within each water year type in mean annual potential production between Alternatives 1A and 1B and the NAA would be small (<1%). Alternatives 2 and 3 results would be similar to those of Alternatives 1A and 1B (Appendix 11H, Table 2b-3, Table 2b-4, Figure B-b-1) in that differences in production relative to the NAA would be generally <1%.

Results by life stage and mortality source are reported in Appendix 11H, Table 1b-1 through Table 1b-4 and Figure B-b-8 through Figure B-b-19. Depending on the life stage and source of mortality (flow-/habitat-based or temperature-based), mean annual mortality under Alternative 1A would be between 53% lower to 24% greater than that under the NAA and mean annual mortality under Alternative 1B would be between 80% lower to 40% higher than that under the

NAA (Appendix 11H, Table 1b-1, Table 1b-2). However, it is important to understand that the model is seeded with 1,210,000 spring-run eggs each year. Although there may be larger percent differences in mean annual mortality between Alternative 1, 2, or 3 and the NAA by source and life stage, these differences typically represent a small proportion of overall individuals when put in a broader population-wide context. The largest raw difference in mean mortality determined through comparison of primary data would be in pre-spawn mortality between Alternative 1B and the NAA in Above Normal Water Years. In this case, there would be a reduction in mean annual egg mortality of 15,120 eggs under Alternative 1B relative to the NAA, accounting for just 1.2% of the 1,210,000 spring-run Chinook salmon eggs annually seeded into the model. Combining all sources of mortality and life stages together, mean annual mortality would be 1% lower under Alternatives 1A and 1B than the NAA for the full simulation period (all water year types combined). By water year type, mean annual mortality under Alternative 1A would be between 8% lower (in Dry Water Years) and 17% greater (in Below Normal Water Years) than the NAA. Mean annual mortality under Alternative 1B would be between 16% lower (in Dry Water Years) and 14% greater (in Below Normal Water Years). Mean annual mortality results for Alternatives 2 and 3 are generally similar to those of Alternatives 1A and 1B (Appendix 11H, Table 1b-3, Table 1b-4). Combining all sources of mortality and life stages together, mean annual mortality under Alternative 2 would be identical (0% difference) to under the NAA for the full simulation period (all water year types combined) and 11% lower to 24% higher than under the NAA depending on water year type. Mean annual mortality under Alternative 3 would be 9% lower than under the NAA for the full simulation period (all water year types combined) and 12% lower to 9% higher than under the NAA depending on water year type.

Overall, SALMOD results show a minimal positive effect of Alternatives 1, 2, and 3 on spring-run Chinook salmon mortality and potential production in the Sacramento River. However, due to the small magnitude of differences between Alternatives 1 and 2 and the NAA, these effects are expected to be biologically inconsequential.

Floodplain Inundation and Access

As described in Chapter 2 and as discussed for winter-run Chinook salmon, the Project's diversion criteria include restrictions to maintain Bend Bridge and Wilkins Slough flows, which have the effect of limiting changes to Yolo Bypass spill frequency and duration under Alternatives 1, 2, and 3 relative to the NAA. In particular, these criteria avoid impacts on Reclamation's ability to implement its obligations in the 2019 NMFS ROC ON LTO BiOp to implement the Yolo Bypass Restoration Salmonid Habitat Restoration and Fish Passage Implementation Plan and provide more than 17,000 acres of inundation in the Yolo Bypass from December to April (National Marine Fisheries Service 2019a). As such, Alternatives 1, 2, and 3 generally would have limited potential for negative effects on Yolo Bypass floodplain inundation and access for spring-run Chinook salmon. This was confirmed with the modeling summarized in the winter-run Chinook salmon section (Table 11-13) and below.

The results of the run-specific Acierto et al. (2014) method for juvenile entry into Yolo Bypass were consistent with those of winter-run Chinook salmon in suggesting similar or somewhat less entry into Yolo Bypass under the Project compared to the NAA (Table 11-34). As noted for winter-run Chinook salmon, there is the potential for a somewhat lower proportion of juvenile spring-run Chinook salmon entering the Sutter Bypass from the Sacramento River under the

Project relative to the NAA, for the portion of the spring-run Chinook salmon ESU that could enter Sutter Bypass via Moulton, Colusa, and Tisdale weirs (i.e., primarily Mill and Deer Creek populations; the Butte Creek population migrates via Butte Slough into Sutter Bypass).

Table 11-34. Proportion of Juvenile Spring-Run Chinook Salmon Entering Yolo Bypass Via Fremont Weir, Based on Acierto et al. (2014).

Water Year	Water Year Type	NAA	With Project*
2009	Dry	0.019	0.017 (-10%)
2010	Below Normal	0.026	0.025 (-3%)
2011	Wet	0.157	0.148 (-6%)
2012	Below Normal	0.047	0.042 (-10%)
2013	Dry	0.100	0.099 (0%)
2014	Critically Dry	0.012	0.012 (0%)
2015	Critically Dry	0.074	0.074 (0%)
2016	Below Normal	0.050	0.045 (-9%)
2017	Wet	0.231	0.216 (-6%)

Notes: Percentage values in parentheses indicate differences of alternatives compared to the NAA. *Results are the same for Alternatives 1, 2, and 3 because the same operational criteria are applied.

As discussed in Impact FISH-2, and in Appendix 11M, Alternatives 1, 2, and 3 are not expected to substantially affect suitable inundated floodplain habitat on the Yolo and Sutter Bypasses or suitable inundated side-channel habitat on the Sacramento River for rearing juvenile Chinook salmon, including spring-run. This conclusion is based on the results of habitat modeling that showed: (1) little difference in suitable floodplain habitat acreage between the NAA and Alternatives 1, 2, and 3 for the Sutter Bypass (Table 11M-4 and Figure 11M-8); (2) an absence of large differences in acreage of suitable side-channel habitat in the Sacramento River (Table 11M-5 through Table 11M-7 and Figure 11M-9): and (3) a reduction of less than 100 acres (1.8%) in the total suitable habitat acreage on the Yolo Bypass for the November through May period (Table 11-14 and Table 11M-3), despite relatively large reductions of inundated floodplain habitat on the Yolo Bypass or Sacramento River side channels for a few month and water year combinations (Table 11-13 and Table 11M-1).

Adult Upstream Passage at Fremont Weir

Adult Chinook salmon migrate upstream during Fremont Weir overtopping events, as well as during lower flow conditions (National Marine Fisheries Service 2019b:26). Sommer et al. (2014) did not find adult Chinook salmon catch in a fyke trap in the Toe Drain was associated with Yolo Bypass flow events, noting this may have been because most of the Chinook salmon were fall-run, a race known to migrate upstream relatively early before winter and spring flow events. Recent completion of the Wallace Weir fish rescue facility and fish passage facilities at Fremont Weir blocked access to the CDB and improved adult fish passage allowing passage at a variety of flows into the Yolo Bypass, including across the range anticipated under the NAA and Alternatives 1, 2, and 3. As such, the minor differences in flow entering Yolo Bypass at Fremont Weir under Alternatives 1, 2, and 3 relative to the NAA (see discussion above related to inundation and juvenile access) would not result in major differences in adult upstream passage under Alternatives 1, 2, and 3 compared to the NAA. As described for winter-run Chinook salmon in Impact FISH-2, the number of days of adult passage at Fremont Weir was assessed using the number of days meeting adult passage criteria[19], based on data for water years 2009–2018 from the Daily Divertible Flow and Storage Tool Results (see Appendix 11P1). The results of this analysis indicated that the number of days meeting fish passage criteria at Fremont Weir generally would be similar with or without the Project (Table 11-21).

Adult Upstream Passage at Sutter Bypass Weirs

As described for winter-run Chinook salmon, changes in Sacramento River flow and flow entering Sutter Bypass as a result of the Project have the potential to change the number of days meeting adult Chinook salmon passage criteria at the three Sutter Bypass weirs (Moulton, Tisdale, and Colusa). The results of the analysis indicated that there would be no days meeting adult salmon passage criteria during 2009–2018 at Moulton or Tisdale Weirs. For Colusa Weir, the number of days meeting fish passage criteria generally would be similar with or without the Project, with the main exception being 2016 (nearly 40% fewer days of passage; Table 11-22).

Migration Flow-Survival

As described in more detail for winter-run Chinook salmon, Alternatives 1, 2, and 3 include pulse flow protection measures to be applied to precipitation-generated pulse flow events from October through May. There is also a 10,700-cfs Wilkins Slough bypass flow threshold from October 1 through June 14, corresponding with the threshold identified by Michel et al. (2021). Adaptive management would be utilized to assess and if necessary refine the diversion criteria based on biological monitoring (see Appendix 2D, Section 2D.6). The discussion in Section 11P.2 of Appendix 11P illustrates that the Sites Reservoir diversion criteria effectively limit diversions during the historical periods of fish movement as reflected in Red Bluff rotary screw trap data, and application of the flow-threshold criteria from Michel et al. (2021) suggests that flow-survival effects on juvenile Chinook salmon (including spring-run Chinook salmon) would be greatly limited by the Project's diversion criteria because there is essentially no difference in predicted survival between the Project and NAA.

Sites Reservoir Release Effects

Sites Reservoir releases could temporally overlap with spring-run Chinook salmon presence near the release location in the Sacramento River, specifically migrating adults during approximately February through July and juveniles rearing or migrating during November through May (Appendix 11A, Table 11A-4; Figure 11-14).

Sites Reservoir releases into the Yolo Bypass via the CBD would occur during August–October, which will not coincide with juvenile or adult spring-run Chinook salmon presence in the Yolo Bypass (Appendix 11A, Table 11A-4; Figure 11-14).

Temperature Effects

[19] The criteria for days to be counted as Yolo Bypass passage days were river stage (1) between 21.14 feet and 29.92 feet during November 1–March 15, (2) between 21.14 feet and 23.35 feet during March 16–April 30, or (3) greater than 35 feet (elevation of the crest of the Fremont Weir). These criteria were based on the YBPASS tool (California Department of Water Resources 2017).

As discussed in Chapter 6, the effect of Alternatives 1A, 1B, 2, and 3 on water temperatures at the Sites Reservoir release site in the Sacramento River would be relatively small with the releases generally tending to cause a slight reduction in water temperature (Tables 6-12a through 6-12d). Temperature-related effects of Alternatives 1A, 1B, 2, and 3 on spring-run Chinook salmon at the release site in the Sacramento River would be minimal.

There would be no temperature-related effects of Sites Reservoir releases in the Yolo Bypass via the CBD and in the Sacramento River below the Yolo Bypass under Alternatives 1A, 1B, 2, and 3 to spring-run Chinook salmon because no spring-run would be present in these locations during August through October when the Yolo Bypass would receive Sites Reservoir releases.

Feather River

Temperature Effects

Operation of Sites Reservoir has the potential to change water temperatures in the Feather River that could affect the life stages of spring-run Chinook salmon present. As described in Appendix 11B, the two methods used to analyze temperature-related effects on spring-run Chinook salmon in the Feather River were: (1) Physical Model Output Characterization; and (2) Water Temperature Index Value/Range Exceedance Analysis. More details on these methods are provided in Appendix 11B.

The Authority and Reclamation evaluated water temperature model outputs during the period of presence and in the locations of each life stage of spring-run Chinook salmon in the Feather River (see Appendix 11B, Table 11B-3 for timing and locations). Visual observation of exceedance plots and differences in modeled mean monthly temperatures by water year type between alternatives and the NAA in the Feather River LFC below the Fish Barrier Dam and in the HFC at Gridley Bridge indicates that water temperatures would be predominantly similar among alternatives during the period of presence of each life stage of spring-run Chinook salmon (Appendix 6C, Tables 6C-16-1a to 6C-16-4c, Tables 6C-18-1a to 6C-18-4c; Figures 6C-16-1 to 6C-16-18, Tables 6C-18-1 to 6C-18-18). At both locations, mean monthly water temperatures for all months within all water year types under Alternatives 1A and 1B were within 0.5°F of the NAA. Water temperature modeling results for Alternatives 2 and 3 were similar to those of Alternative 1 at both locations. These results suggest that temperature-related effects on spring-run Chinook salmon in the Feather River would be negligible.

Results of the analysis of exceedance above water temperature index values for spring-run Chinook salmon in the Feather River from Appendix 11B, Table 11B-3 are presented in Appendix 11D, Table 11D-45 through Table 11D-52 and summarized in Table 11-35. For each life stage and at all locations evaluated, there were no month and water year type combinations in which both: (1) the percent of months that exceeded the index values was more than 5% greater under Alternatives 1, 2, and 3 than under the NAA, and (2) the exceedance per month was more than 0.5°F greater under Alternatives 1, 2, and 3 than under the NAA. Results of the exceedance analysis for Alternatives 2 and 3 are similar to Alternative 1 with no month and water year type combinations in which both criteria were met for any life stage at all locations. These results indicate that temperature-related effects on spring-run Chinook salmon in the Feather River would be negligible.

Table 11-35. Number of Month and Water Year Type Combinations that Satisfy Both Criteria for Being Biologically Meaningful in the Water Temperature Index Value Analysis, Spring-run Chinook Salmon, Feather River[1,2,3]

Location	Spawning and Egg Incubation				Juvenile Rearing and Migration				Adult Immigration				Adult Holding			
	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3
Low-Flow Channel Below Fish Barrier Dam	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
High-Flow Channel Below Thermalito Afterbay Outlet	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

[1] Biologically Meaningful Criteria include: (1) the difference in frequency of exceedance between the NAA and the alternative was greater than 5%, and (2) the difference in average monthly exceedance between the NAA and the alternative was greater than 0.5°F.
[2] Index values for each life stage are located in Appendix 11B, Table 11B-3.
[3] Full results presented in Appendix 11D, Table 11D-45 through Table 11D-52.

The probability that Alternative 1, 2, or 3 would meet water temperature targets to support anadromous fish in the Feather River included the Settlement Agreement for Licensing of the Oroville Facilities (Appendix 11B, Table 11B-3; California Department of Water Resources 2006:A-18–A-24) was evaluated with monthly water temperature model outputs above Thermalito Afterbay for the LFC and below Thermalito Afterbay for the HFC. Results for each alternative are provided in Appendix 11D, Table 11D-53 and Table 11D-54.

The percent of months above the LFC temperature targets under Alternatives 1A and 1B were predominantly similar to those under the NAA with few exceptions. There would be 5.6% and 8.3% more months above the temperature target in April of Dry Water Years and in June of Critically Dry Water Years, respectively, under Alternative 1A compared to the NAA. Under Alternative 1B, there would be 5.6% more months above the temperature target in April of Dry Water Years relative to the NAA. There would be 8.3% and 5.6% fewer months above the temperature target in April of Critically Dry Water Years and in October of Dry Water Years, respectively, under Alternative 1B compared to the NAA. Despite these changes in frequency of exceedance, the magnitude of exceedance during these month and water year type combinations would be within 0.5°F/month, suggesting that these differences could be resolved through real-time reservoir operations.

The percent of months above the HFC temperature targets under Alternatives 1A and 1B were predominantly similar to those under the NAA with few exceptions. There would be 8.3% more months above the temperature target in October of Critically Dry Water Years, respectively, under both Alternatives 1A and 1B compared to the NAA. Despite these changes in frequency of exceedance, the magnitude of exceedance during these months would be within 0.5°F/month, suggesting that these differences could be resolved through real-time reservoir operations, as explained in the methods description in Appendix 11B.

Results of the evaluation of exceedance of Oroville Settlement Agreement water temperature targets in the LFC and HFC for Alternatives 2 and 3 would be similar to those for Alternative 1.

These results indicate that the Project's implementation under Alternatives 1, 2, and 3 would not substantially change the ability to meet water temperature targets in the Oroville Settlement Agreement relative to the NAA.

Combined, these water temperature results indicate that the Project's implementation under any Alternatives 1, 2, and 3 would cause inconsequential temperature-related effects on spring-run Chinook salmon in the Feather River.

Flow-Related Physical Habitat Conditions

Redd Scour and Entombment

Frequency of scouring flows was not estimated for the Feather River because information on minimum flows required to mobilize sediments is not available for the Feather River (Cain and Monohan 2008) and no existing credible scientific evidence can be used to practically estimate the frequency of scouring flows on the Feather River. However, Feather River flows during the Wet and Above Normal Water Years, when scouring flows would be most likely to occur during the months of spring-run spawning and egg incubation (September through January), are generally similar between the NAA and Alternatives 1, 2, and 3 (Chapter 5). Therefore, no substantial differences on the frequency of scouring flows are expected between the NAA and Alternatives 1, 2, and 3.

Redd Dewatering

As described in Section 11N.2, *Methods*, redd dewatering for Feather River salmon was estimated from the depth distributions of their redds and changes in river stage (depth) at the Gridley gage on the Feather River at different flows. The redd depth distributions were determined from DWR studies of Feather River salmon (Appendix 11N, Figure 11N-2). The river depths were estimated from CALSIM II monthly flow results at the Thermalito Afterbay outlet and DWR's stage-discharge tables for the Gridley gage. Comparisons of the means of the redd dewatering estimates under Alternatives 1, 2, and 3 to means under the NAA use absolute differences rather than relative differences (percent change) because many of the values for percentages of redds dewatered are small. Expressing changes of small values as percent changes may result in very large values that may be misleading. The use of monthly time-step flow estimates likely underestimates redd dewatering rates because they smooth out short-term flow fluctuations. This potential bias is expected to affect all Project scenarios equally.

The results for spring-run show large increases in redd dewatering under Alternatives 1, 2, and 3 for September spawning, especially in Wet Water Years (Table 11N-18). The largest increases occur under Alternatives 1B and 3, including a 7% increase under Alternative 3 for spawning in

September of Below Normal Water Years and 5% increases under Alternative 1B for spawning in September of Wet Water Years and October of Below Normal Water Years. These results indicate that Alternatives 1B and 3 would substantially increase spring-run redd dewatering in the Feather River, especially for spawning during September. However, given that most spawning of Feather River salmonids occurs in the LFC (Figure 11N-1) (Kindopp pers. comm.), the expected increased redd dewatering in the HFC is not expected to greatly affect the Feather River spring-run population.

Spawning Habitat Weighted Usable Area

Spring-run and fall-run are often difficult to distinguish in the Feather River, so a single WUA curve was developed for both runs (Payne and Allen 2004) (Figure 11K-7). The curve was used to compute spawning WUA with flows specific to the months of spawning for the run analyzed. To evaluate the effects of Alternatives 1, 2, and 3 on spring-run spawning habitat in the Feather River, the spawning WUA was estimated under Alternatives 1, 2, and 3 and the NAA for CALSIM II flows below Thermalito Afterbay during September through November, the Feather River spring-run spawning period. Differences in spawning WUA between Alternatives 1, 2, and 3 and the NAA were examined using the grand mean spawning WUA for each month of the spawning period under each water year type and all water year types combined (Table 11K-18).

The largest differences between Alternatives 1, 2, and 3 and the NAA means in spring-run spawning WUA were 5% increases in WUA for October of Dry Water Years under Alternatives 1A, 1B, and 2, and a 5% reduction in September of Below Normal Water Years under Alternative 3. Most differences for both runs are <2%. These results indicate that Alternatives 1, 2, and 3 would have little effect on spring-run spawning WUA in the HFC of the Feather River.

Rearing Habitat Weighted Usable Area

Reliable predictions regarding changes in flow affecting rearing habitat for spring-run Chinook salmon in the Feather River cannot be made. As discussed in Payne and Allen (2005) and quoted below, curves developed for Feather River Chinook rearing habitat WUA are unreliable:

> ….The results for this component of the analysis were more ambiguous and difficult to interpret than those for adult salmon and steelhead. In an effort to reach agreement on the meaning and applicability of the juvenile salmonid PHABSIM findings, an interagency meeting was held on June 3, 2004. At this meeting it was agreed that, given current channel conditions, the results did not support a clear alternative or ideal discharge level. Rearing habitat indexes for fry and juvenile Chinook salmon and steelhead did not respond clearly or significantly to changes in discharge. Furthermore, results differed markedly depending on how areas having no cover were treated in the model. Although the results appear to be valid (i.e., they correctly represent a simplified version of juvenile fish habitat), the amount of suitable habitat seems relatively insensitive to modeled discharge levels. Based on this interpretation, the group agreed that efforts to improve physical habitat for juvenile salmonids (e.g., increasing habitat complexity with side channels, midchannel bars, riparian vegetation and/or instream objects) should be given primary consideration, and that any flow changes should be complimentary to these physical habitat enhancements. However, the group did recommend that juvenile salmonid PHABSIM results be used wherever possible to aide in the design and placement of future habitat enhancements.

WUA curves from other rivers (e.g., Sacramento River) cannot be used to determine the availability of rearing habitat WUA in response to changes in flow in the Feather River because relationships between flow and habitat WUA are generally not transferrable from one river to another (Bovee et al. 1998). As such, quantitative conclusions cannot be made regarding effects of Alternatives 1, 2, and 3 on spring-run rearing WUA.

Juvenile Stranding

No formal method for analyzing juvenile stranding analysis has been developed for the Feather River (Appendix 11K).

American River

Redd Scour and Entombment, Redd Dewatering

Spring-run do not spawn in the American River.

Habitat Weighted Usable Area

Spring-run do not spawn in the American River and no rearing WUA curves are available for the American River (Appendix 11K).

Juvenile Stranding

No formal method for analyzing juvenile stranding analysis has been developed for the American River.

Delta

Juvenile Through-Delta Survival

The analysis of the potential for negative effects on juvenile Chinook salmon through-Delta survival, as assessed by the spreadsheet implementation of the through-Delta survival function formulated by Perry et al. (2018; see further discussion for winter-run Chinook salmon), found that during the main period of juvenile spring-run Chinook salmon occurrence in the Delta (i.e., April–May; see Table 11A-5 in Appendix 11A), there was essentially no difference in mean through-Delta survival between Alternatives 1, 2, and 3 and the NAA (Table 11-24 in winter-run analysis). Overall, during the broader spring-run juvenile migration including the earlier winter/early spring period of lower abundance (i.e., December–March), differences were small (0%–2% less under Alternatives 1, 2, and 3; Table 11-24).

As described for winter-run Chinook salmon, note that the spreadsheet implementation of the Perry et al. (2018) model does not account for the variability in coefficient estimates (Figure 6 of Perry et al. 2018), which would likely give appreciable overlap of estimates in through-Delta survival between the NAA and the alternative scenarios, particularly in relation to the relatively small differences between alternatives.

Juvenile Rearing Habitat

As discussed in more detail for winter-run Chinook salmon, available information from analysis of changes in riparian and wetland bench inundation suggests that Alternatives 1, 2, and 3 would have the potential for ~5% less inundation relative to the NAA for juvenile spring-run Chinook salmon.

South Delta Entrainment

There would be little difference in indicators of entrainment risk (south Delta exports and Old and Middle River flows) during winter/spring between the NAA and Alternatives 1, 2, and 3 (see Appendix 5B3, Tables 5B3-6-1a through 5B3-6-4c; Figures 5B3-6-1 through 5B3-6-18; and Appendix 5B4, Tables 5B4-1-1a through 5B4-1-4c; Figures 5B4-1-1 through 5B4-1-18), and existing restrictive criteria from the NMFS (2019a) ROC ON LTO BiOp and CDFW (2020) State ITP would limit entrainment risk for spring-run under Alternatives 1, 2, and 3. South Delta export of water released from Sites Reservoir would have little to no overlap with spring-run Chinook salmon. This determination is illustrated by the results of the salvage-density analysis (Appendix 11Q) for which there were minimal differences between the NAA and Alternatives 1, 2, and 3 (Table 11-36; Table 11-37).

Table 11-36. Entrainment Loss of Juvenile Spring-Run Chinook Salmon At SWP Banks Pumping Plant, Averaged by Water Year Type, Based on the Salvage-Density Method.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	45,458	45,588 (0%)	45,496 (0%)	45,587 (0%)	45,462 (0%)
Above Normal	NA	(-9%)	(-9%)	(-9%)	(-2%)
Below Normal	4,835	4,834 (0%)	4,833 (0%)	4,842 (0%)	4,810 (-1%)
Dry	2,696	2,698 (0%)	2,687 (0%)	2,698 (0%)	2,707 (0%)
Critically Dry	2,261	2,264 (0%)	2,248 (-1%)	2,265 (0%)	2,247 (-1%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. The analysis was based on historical salvage data during 2009–2019 which did not include any Above Normal Water Years, hence that row of the table is noted as 'NA'; the percentage difference in Above Normal Water Years is based on density data for Wet Water Years applied to above normal modeled exports. Results by water year type and month are provided in Appendix 11Q. High absolute estimates of spring-run Chinook salmon juvenile loss reflect length-at-date misclassification of fall-run Chinook salmon (Harvey et al. 2014); note, however, that focus should be placed on relative differences as opposed to absolute estimates. Percentage values are rounded; as a result, differences between percentages may not always appear consistent.

Table 11-37. Entrainment Loss of Juvenile Spring-Run Chinook Salmon At CVP Jones Pumping Plant, Averaged by Water Year Type, Based on the Salvage-Density Method.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	8,120	8,117 (0%)	8,111 (0%)	8,117 (0%)	8,029 (-1%)
Above Normal	NA	(0%)	(0%)	(0%)	(0%)
Below Normal	3,235	3,241 (0%)	3,274 (1%)	3,241 (0%)	3,294 (2%)
Dry	2,989	2,994 (0%)	3,021 (1%)	2,994 (0%)	3,042 (2%)
Critically Dry	131	130 (0%)	134 (3%)	130 (0%)	135 (4%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. The analysis was based on historical salvage data during 2009–2019 which did not include any Above Normal Water Years, hence that row of the table is noted as 'NA'; the percentage difference in Above Normal Water Years is based on density data for Wet Water Years applied to above normal modeled exports. Results by water year type and month are provided in Appendix 11Q. High absolute estimates of spring-run Chinook salmon juvenile loss reflect length-at-date misclassification of fall-run Chinook salmon (Harvey et al. 2014); note, however, that focus should be placed on relative differences as opposed to absolute estimates. Percentage values are rounded; as a result, differences between percentages may not always appear consistent.

CEQA Significance Determination for Alternatives 1, 2, and 3

The preceding subsections of this impact discussion provide the detailed information used for this CEQA (and NEPA) determination. This section provides a summary of this information.

In-Delta and upstream operational impacts of Alternatives 1, 2, and 3 on spring-run Chinook salmon generally would be limited. Spring-run Chinook salmon would have the potential for similar types of near-field effects to those previously discussed for winter-run Chinook salmon (i.e., entrainment, impingement and screen contact, predation, stranding behind screens, and attraction to reservoir discharge). The potential for these effects would differ relative to winter-run Chinook salmon due to less than one quarter of spring-run Chinook salmon being expected to pass the intakes. As discussed for winter-run, few spring-run Chinook juveniles would have the potential for entrainment given their size. As described for winter-run Chinook salmon, adult spring-run Chinook salmon could be attracted to the flow from the Sacramento River discharge under Alternative 2. The design of the apron and weir would exclude anadromous fish and eliminate stranding risk.

Based on available studies of predation effects at water intakes, the predation effects in the vicinity of the Red Bluff and Hamilton City intakes for Alternatives 1, 2, and 3 would be limited. The extent of in-water structure at the intakes would be the same under the NAA and Alternatives 1, 2, and 3. Although overtopping during high flows can occur at the Red Bluff and Hamilton City intakes, leading to potential stranding of juvenile spring-run Chinook salmon, these are relatively infrequent events (e.g., approximately once per 100 years at Red Bluff) that occur under the NAA and would not be changed by Alternatives 1, 2, and 3.

Results indicate that Alternatives 1, 2, and 3 would have no effect on redd scour and entombment for spring-run Chinook salmon in the Sacramento River. In general, Alternatives 1, 2, and 3 are not expected to substantially affect spring-run Chinook salmon redd dewatering. Feather River flows during the Wet and Above Normal Water Years, when scouring flows would be most likely to occur during the months of spring-run spawning and egg incubation (September through January), are generally similar between the NAA and Alternatives 1, 2, and 3. Therefore, no substantial differences on the frequency of scouring flows in the Feather River are expected between the NAA and Alternatives 1, 2, and 3.

Mean spawning WUA for spring-run Chinook salmon under Alternatives 1, 2, and 3 differs from the NAA by more than 3% for only a few months and water year types, with most of these differences occurring under Alternative 3. The largest difference is a 16% increase under Alternative 3 in Segment 5 for August of Above Normal Water Years. The largest reduction in spawning WUA for spring-run is a 7% reduction, under Alternative 3 in Segment 4 for October of Wet Water Years. Other relatively large differences in WUA between Alternatives 1, 2, and 3 and the NAA are 4% to 6% reductions occurring in Critically Dry Water Years during August in Segment 5 and during September in Segment 4. These results indicate that Alternatives 1, 2, and 3 would lead to some reductions and increases of spring-run spawning habitat WUA. However, on balance, Alternatives 1, 2, and 3 are not expected to substantially affect spring-run spawning WUA.

The results for spring-run Chinook salmon juvenile rearing WUA indicate that Alternatives 1 and 2 would generally have little effect on rearing habitat, but Alternative 3 would have relatively large effects (in Segment 4), including substantial increases during late spring and summer and smaller reductions during late summer and fall. Increases in WUA outnumber reductions in the results and more of them are especially large (>10%). Furthermore, the increases occur during spring and summer, when the juveniles are younger and perhaps more vulnerable to reductions in habitat availability. On balance, Alternatives 1, 2, and 3 are expected to have little effect on spring-run fry rearing habitat availability and to increase spring-run juvenile rearing habitat WUA.

Related to temperature effects, observations of exceedance plots and differences in modeled mean monthly temperatures by water year type indicate that Alternatives 1, 2, and 3 and the NAA would be predominantly similar among alternatives during the period of presence of each life stage of spring-run Chinook salmon. The water temperature indicator value exceedance analysis confirms this similarity among alternatives.

As described in Chapter 2, the Project's diversion criteria include restrictions to maintain Bend Bridge and Wilkins Slough flows, which have the effect of limiting changes to Yolo Bypass spill frequency and duration under Alternatives 1, 2, and 3 relative to the NAA. Thus, Alternatives 1, 2, and 3 generally would have limited potential for negative effects on Yolo Bypass floodplain inundation and access for spring-run Chinook salmon. The results of the frequency analysis of inundation of events for the Yolo Bypass generally show only minor differences between Alternatives 1, 2, and 3 and the NAA.

The Project would operate to avoid effects on the Big Notch's ability to achieve the same level of performance for salmonids in the Sacramento River as it would absent the Project. However, the Adaptive Management Plan for the Project recognized there is uncertainty about the performance of the Big Notch and the effects of the Project on it. Monitoring will be conducted, in cooperation with the State, to determine whether there is an effect and, if so, what the magnitude of that effect would be on entrainment of juvenile salmon into the Yolo Bypass. If there is an adverse effect, a science-based adaptive management approach will be employed to determine how to adjust diversions 158 RMs upstream of the Big Notch to maintain its efficiency for entraining juvenile salmon into the Yolo Bypass.

The Big Notch Project also is intended to reduce migratory delays and loss at the Fremont Weir of adult salmon, steelhead, and sturgeon that are migrating upstream through the Yolo Bypass. As with juvenile passage, the Project would operate to avoid affecting adult passage at Fremont Weir. The number of days meeting adult Chinook salmon passage criteria at Fremont Weir generally would be similar with or without the Project (Table 11-21). This too will be evaluated and adaptively managed as required.

The Sutter Bypass, when inundated, provides important juvenile rearing habitat for Chinook salmon and steelhead, as discussed for the Yolo Bypass. For the Sutter Bypass the modeling results indicate that Alternatives 1, 2, and 3 would produce little change in suitable habitat as compared to the NAA. Changes in Sacramento River flow and flow entering Sutter Bypass as a result of the Project have the potential to change the number of days meeting adult Chinook salmon passage criteria at the three Sutter Bypass weirs (Moulton, Tisdale, and Colusa). The results of this analysis indicated that there would be no days meeting adult salmon passage criteria during 2009–2018 at Moulton or Tisdale Weirs under the NAA or Alternatives 1, 2, and 3. For Colusa Weir, the number of days meeting fish passage criteria generally would be similar with or without the Project.

As described in Chapter 2, Alternatives 1, 2, and 3 include pulse flow protection measures to be applied to precipitation-generated pulse flow events. There is also a 10,700-cfs Wilkins Slough bypass flow threshold from October 1 through June 14. Adaptive management would be utilized to assess and if necessary, refine the diversion criteria based on biological monitoring (see Appendix 2D, Section 2D.6). Sites Reservoir diversion criteria effectively limit diversions during the historical periods of fish movement as reflected in Red Bluff rotary screw trap data, and application of the flow-threshold criteria from Michel et al. (2021) suggests that flow-survival effects on juvenile Chinook salmon (including spring-run Chinook salmon) would be greatly limited by the Project's diversion criteria because there is essentially no difference in predicted survival between the Project and NAA.

Based on the analyses provided, operation of Alternative 1, 2, or 3 would not have a substantial adverse effect, either directly or through habitat modifications, on spring-run Chinook salmon. In consideration of the analyses within the preceding subsections of this impact discussion and summarized above, operation impacts of Alternative 1, 2, or 3 would be less than significant. No mitigation is necessary.

NEPA Conclusion for Alternatives 1, 2, and 3

Operation effects on Chinook salmon would be the same as described above for CEQA. In-Delta and upstream operational effects of Alternatives 1, 2, and 3 on spring-run Chinook salmon generally would be limited. Related to temperature effects, observations of exceedance plots and differences in modeled mean monthly temperatures by water year type indicate that Alternatives 1, 2, and 3 and the NAA would be predominantly similar among alternatives during the period of presence of each life stage of spring-run Chinook salmon. The water temperature indicator value exceedance analysis confirms this similarity among alternatives.

As described in Chapter 2, the Project's diversion criteria include restrictions to maintain Bend Bridge and Wilkins Slough flows, which have the effect of limiting changes to Yolo Bypass spill frequency and duration under Alternatives 1, 2, and 3 relative to the NAA. Thus, Alternatives 1, 2, and 3 generally would have limited potential for negative effects on Yolo Bypass floodplain inundation and access for spring-run Chinook salmon. Also described in Chapter 2, Alternatives 1, 2, and 3 include pulse flow protection measures to be applied to precipitation-generated pulse flow events. There is also a 10,700-cfs Wilkins Slough bypass flow threshold from October 1 through June 14, corresponding with the threshold identified by Michel et al. (2021). Operation of Alternative 1, 2, or 3 would have no adverse effect on spring-run Chinook salmon.

Impact FISH-4: Operations Effects on Fall-Run/Late Fall–Run Chinook Salmon

Alternatives 1, 2, and 3

Potential exposure of fall-run/late fall–run Chinook salmon to the effects of Alternatives 1, 2, and 3 is dependent on the species' spatiotemporal distribution. As described for winter-run Chinook salmon, several sources of information provide important context.

The main patterns of juvenile fall-run/late fall–run Chinook salmon occurrence at the locations documented in the SacPAS summary in Appendix 11A1 include:

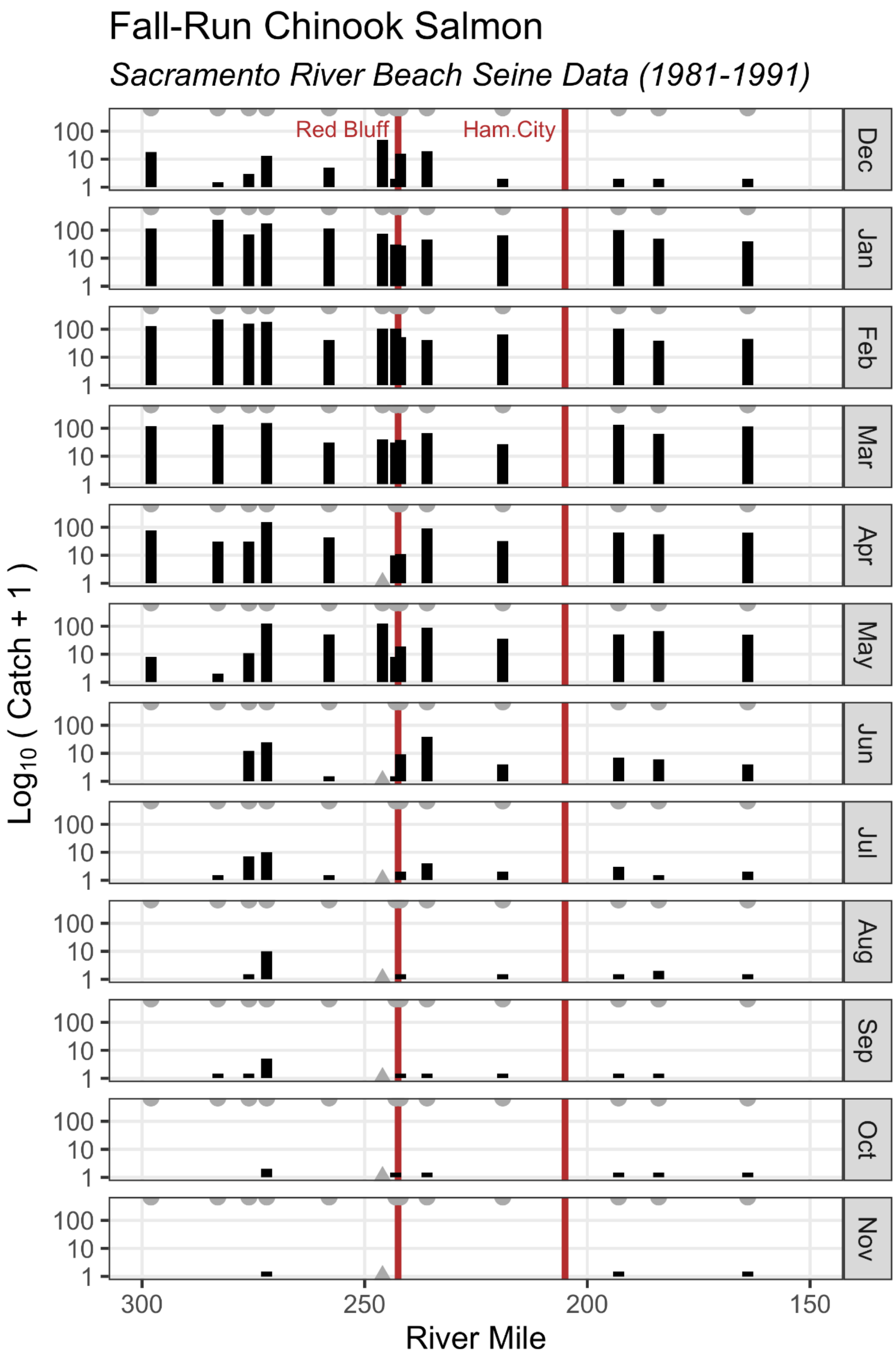
- Fall-run Chinook:
  - RBDD rotary screw traps (Appendix 11A1, Figure 11A1-19): Passage begins around December 1 and finishes in early November (i.e., essentially year-round). The first half (50%) has passed from mid-January to the first of May. The main portion (90%) begins to pass in mid-December to the first of February and finishes passing from mid-March to the end of June.
  - Tisdale Weir rotary screw traps (Appendix 11A1, Figure 11A1-20): Passage begins in early December to mid-January and finishes by early May to early July, except the 2010 cohort, which finishes mid-November. The first half (50%) passes by mid-January to mid-March. The main portion (90%) begins to pass in late December to mid-February and finishes from mid-February to early May.
  - Knights Landing rotary screw traps (Appendix 11A1, Figure 11A1-21): First passage begins from early December to late January and passage finishes from the end of May to late June. The first half (50%) passes by mid-January to mid-April. The main portion (90%) begins to pass in late December to mid-February and finishes from the end of February to the end of April. The exception to these trends are the 2012 and 2015 cohorts. In 2012 the entire cohort passed in approximately 10 days in early December. In 2015 the fish began to pass in mid-December and finished approximately 4 weeks later, with half the fish passing by the end of December.
  - Sacramento beach seines (Appendix 11A1, Figure 11A1-22): Occurrence begins in December and ends from late April to the end of October. The first half (50%) occurs by early February to early March. The main portion (90%) occur beginning in late December to late February and finishes from mid-March to early May.
  - Sacramento trawls (Sherwood Harbor) (Appendix 11A1, Figure 11A1-23): Occurs begins from early December to early February and finishes from late May to early September, except the 1995 cohort which finished at the end of November. The first half (50%) occurs from early February to late April. The main portion (90%) of fish occurs by mid-December to the first of March and ends from early March to early June.
  - Chipps Island trawls (Appendix 11A1, Figure 11A1-24): Occurrence begins in mid-December to late April and finishes from early July to late November. The first half (50%) occurs by a short window from late May to mid-June. The main portion (90%) begins to occur in late April and finishes in mid-May to early June.

- Salvage (unclipped, length-at-date) (Appendix 11A1, Figure 11A1-25): Salvage begins from early August to late March and is complete from early May to late July, except the 2014 cohort, which finished by early March. The first half (50%) of the fish are salvaged by early February to the end of June. The main portion (90%) is salvaged beginning from mid-January to mid-May and finishing from early May to late June, except the 2014 cohort which finished being salvaged in early March.
- Salvage (clipped, length-at-date) (Appendix 11A1, Figure 11A1-26): Salvage is highly variable and can being in late September to mid-May and finish from mid-December to late June. The first half (50%) occurs by early November to early June. The main portion (90%) begins to occur in late September to mid-May and finishes from mid- December to mid-June.
- Salvage (clipped, CWT-Race) (Appendix 11A1, Figure 11A1-27): Salvage begins in early December to mid-May is finished from mid-December to mid-June. The first half (50%) occurs by mid-December to mid-May. The main portion of salvage begins to occur from early December to mid-May and finishes from mid-December to early June.

- Late fall–run Chinook:
  - RBDD rotary screw traps (Appendix 11A1, Figure 11A1-28): Passage begins by the first of April to the end of June and finishes from the first of January to early March. The first half (50%) of passage occurs by mid-May to mid-November. The main portion (90%) begins to pass in early April to late September and finishes from early September to mid-December.
  - Tisdale Weir rotary screw traps (Appendix 11A1, Figure 11A1-29): Passage is highly dynamic, beginning as early as the first of April to the end of January, 9 months later. Passage finishes from late November to the first of February. The first half (50%) of the fish passes from mid-April to late January, 9 months later. The main portion of passage (90%) is similar to the first and last fish passing, i.e., the first of April to the end of January, 9 months later.
  - Knights Landing rotary screw traps (Appendix 11A1, Figure 11A1-30): Passage begins around the first of April to late November and finishes from mid-December to the first of April of the following year. The first half (50%) passes by mid-April to early January, 9 months later. The main portion begins to pass in mid-April to early December and finishes passing from early May to late January, 8 months later.
  - Sacramento beach seines (Appendix 11A1, Figure 11A1-31): Occurrence begins in early April to mid-January, 9 months later, and finishes from the end of November to early February. The first half (50%) passes by early April to mid-January. The main portion (90%) begins to pass in early April to mid-December and finishes passing from early May to early February, 9 months later.
  - Sacramento trawls (Sherwood Harbor) (Appendix 11A1, Figure 11A1-32): Occurrence begins in early May to mid-December and finishes from late October to late March. The first half (50%) occurs by mid-May to mid-January, 8 months later. The main portion (90%) begins to occur from early April to late December and finishes occurring from late July to the end of March.

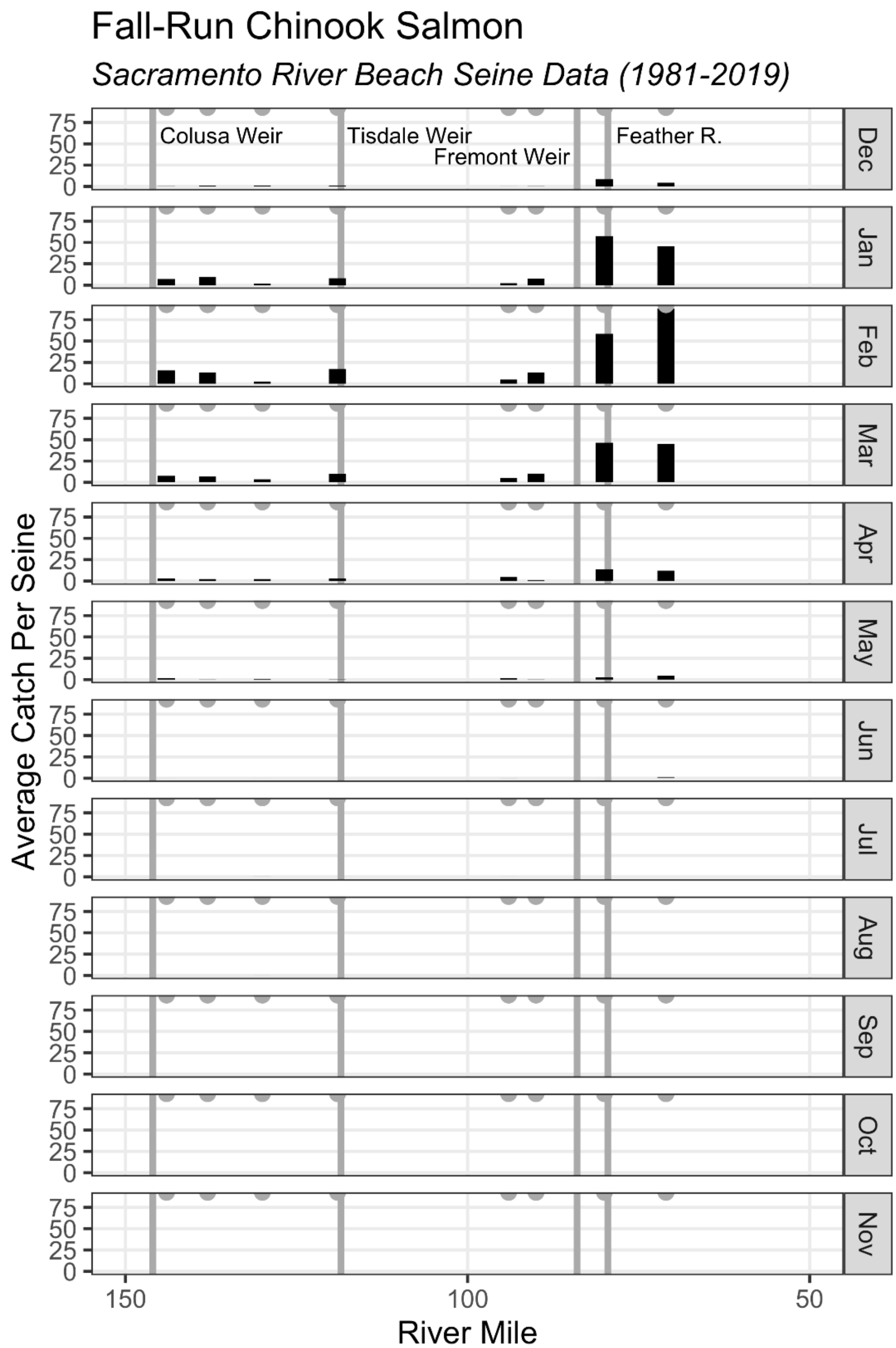
- Chipps Island trawls (Appendix 11A1, Figure 11A1-33): Occurrence begins in early May to mid-January and finishes from early January to early March. The first half (50%) occurs from early December to mid-January, whereas the main portion (90%) begins to occur from mid-September to mid-January and finishes from late December to mid-February.
- Salvage (unclipped) (Appendix 11A1, Figure 11A1-34): Salvage begins in late August to mid-February and finishes from mid-December to mid-June. The first half (50%) is salvaged by mid-November to mid-April. The main portion (90%) begin to be salvaged in early October to mid-February and finishes from mid-December to mid-June.
- Salvage (clipped) (Appendix 11A1, Figure 11A1-35): Salvage begins in mid-November to mid-January and finishes from early January to mid-February. The first half (50%) is salvaged by mid-December to around the first of February. The main portion (90%) begin to be salvaged by the first of December to mid-January and finishes being salvaged from mid-December to mid-February.
- Salvage (clipped, CWT-Race) (Appendix 11A1, Figure 11A1-36): Salvage begins in early December to mid-January and finishes from early January to around the first of February. The first half (50%) of salvage occurs by mid-December to mid-January. The main portion (90%) of salvage is from early December to mid-January and is finished from mid-December to the first of February.

Fall-run beach seine catches were quite uniformly high across the sites included in extensive beach seining during 1981–1991 in December–May, and decreased beginning in June (Figure 11-16). Further downstream at beach seine sites sampled 1981–2019, fall-run catch rates peaked during January–March, and then decreased to zero during the spring and summer (Figure 11-17). Late fall–run beach seine catches were relatively high in the upper reaches of the 1981–1991 study area during April–September, and then started to increase further downstream in October (Figure 11-18). This pattern is consistent with catch rates peaking further downstream at the 1991–2019 sample sites in November–December, although there were also peaks at those locations in April and May (Figure 11-19).



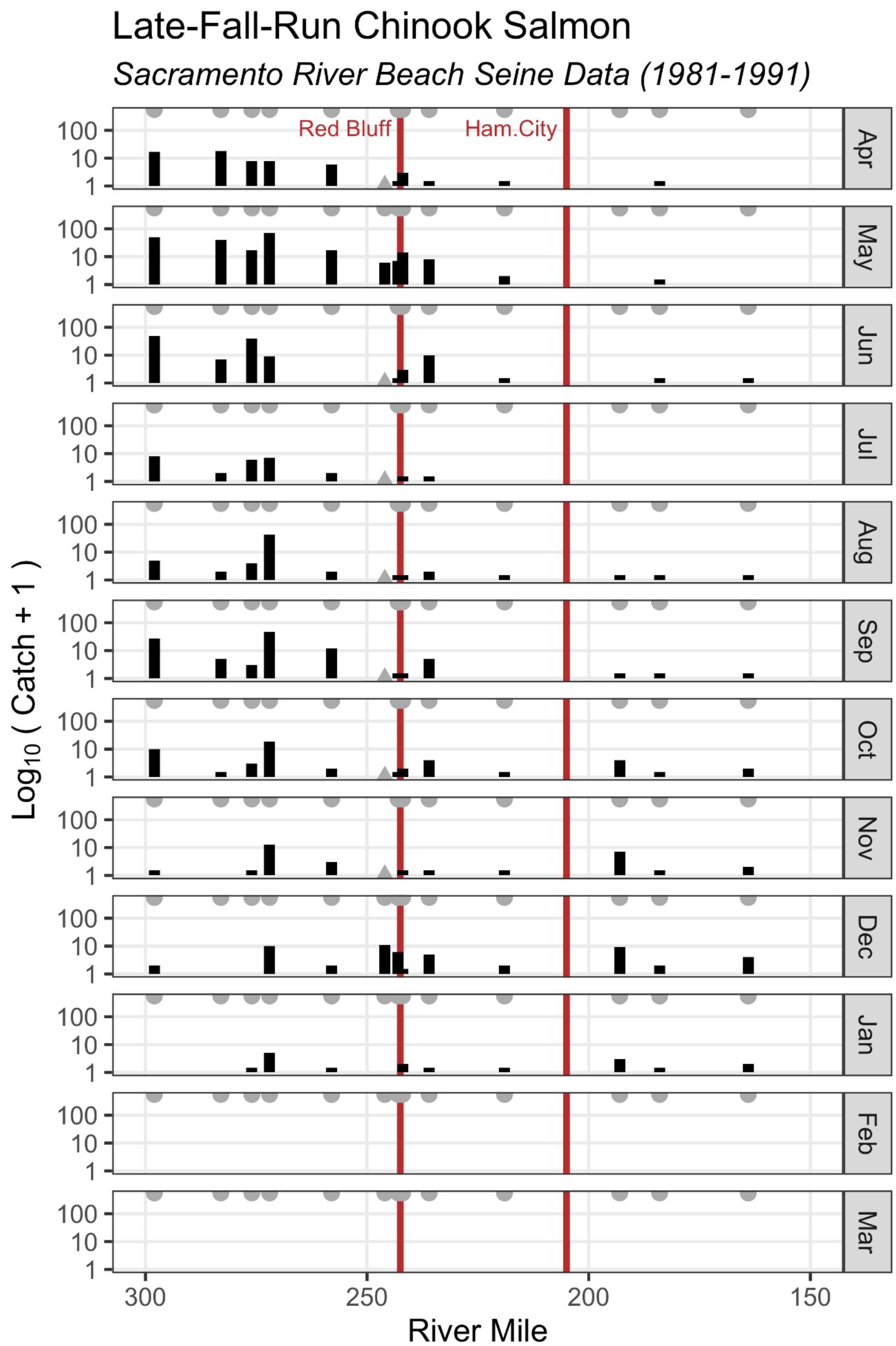
Source: Johnson et al. 1992:Table 4. Note: Sampled sites are denoted by grey semicircles at the top of each plot. Sites that were not sampled in a given month are denoted by grey triangles at the bottom of a panel. Values denoted as "<1" by Johnson et al. are shown as 0.5. Red lines indicate locations of Red Bluff and Hamilton City diversions.

Figure 11-16. Mean Monthly Catch Per Beach Seine of Juvenile Fall-Run Chinook Salmon in the Sacramento River Between River Mile 164 and River Mile 298, 1981–1991.



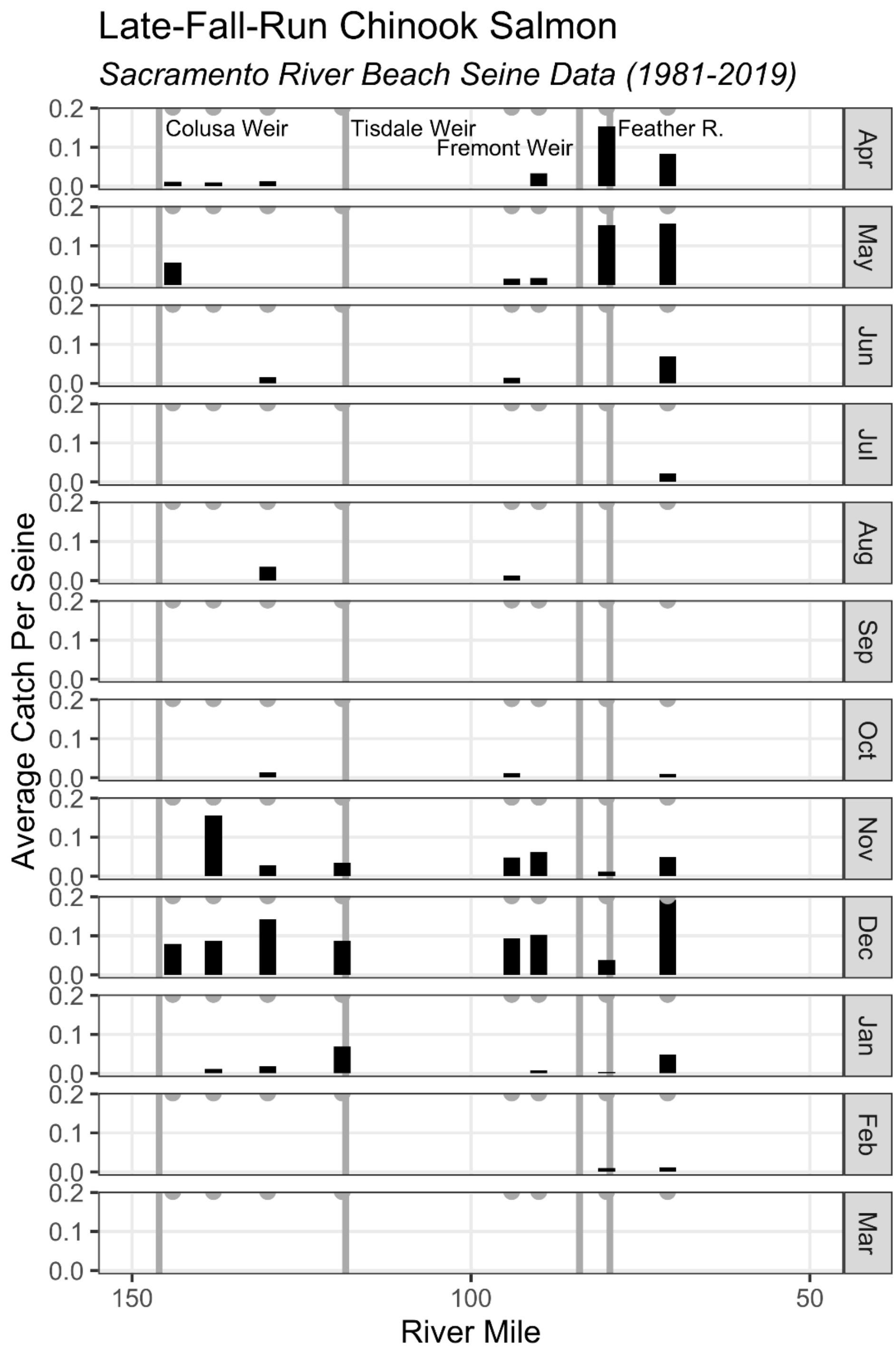
Source: Interagency Ecological Program et al. 2019. Note: Sampled sites are denoted by grey semicircles at the top of each plot. Grey lines indicate locations of Colusa, Tisdale, and Fremont Weirs, and the confluence with the Feather River.

Figure 11-17. Mean Monthly Catch Per Beach Seine of Juvenile Fall-Run Chinook Salmon in the Sacramento River Between River Mile 71 and River Mile 144, 1981–2019.



Source: Johnson et al. 1992:Table 4. Note: Sampled sites are denoted by grey semicircles at the top of each plot. Sites that were not sampled in a given month are denoted by grey triangles at the bottom of a panel. Values denoted as "<1" by Johnson et al. are shown as 0.5. Red lines indicate locations of Red Bluff and Hamilton City diversions.

Figure 11-18. Mean Monthly Catch Per Beach Seine of Juvenile Late Fall–Run Chinook Salmon in the Sacramento River Between River Mile 164 and River Mile 298, 1981–1991.



Source: Interagency Ecological Program et al. 2019. Note: Sampled sites are denoted by grey semicircles at the top of each plot. Grey lines indicate locations of Colusa, Tisdale, and Fremont Weirs, and the confluence with the Feather River.

Figure 11-19. Mean Monthly Catch Per Beach Seine of Juvenile Late Fall–Run Chinook Salmon in the Sacramento River Between River Mile 71 and River Mile 144, 1981–2019.

Between 2009 and 2020, the mean percentage of in-river spawning adult fall-run and late fall–run Chinook salmon returning to tributaries upstream of Red Bluff was 34%, and upstream of Hamilton City was ~37%; the percentage of total fish (hatchery + in-river) was 35%–37% upstream of both intakes (Table 11-38). This indicates that the majority of fall-run and late fall–run Chinook salmon would not be expected to have the potential for near-field effects from the diversions, with a greater proportion of the population occurring downstream in tributaries such as the Feather River and American River, for example. As noted for spring-run Chinook salmon, these downstream fish would have the potential for far-field effects from the Project (e.g., flow-survival effects for juveniles migrating downstream). Note that for hatchery-origin fish, an appreciable portion may be transported by truck for release in the Delta (Huber and Carlson 2015), thereby reducing the potential for effects of the Project.

Table 11-38. Abundance and Percentage of Fall-Run and Late Fall–Run Chinook Salmon Adult Escapement Upstream and Downstream of the Red Bluff and Hamilton City Intakes, 2009–2020.

Year	All Fish Abundance Upstream of Red Bluff	% of All Fish Upstream of Red Bluff	In-River Abundance Upstream of Red Bluff	% of In-River Upstream of Red Bluff	All Fish Abundance Upstream of Hamilton City	% of All Fish Upstream of Hamilton City	In-River Abundance Upstream of Hamilton City	% of In-River Upstream of Hamilton City	All Fish Abundance Downstream of Hamilton City (Sac. R.)	% of All Fish Downstream of Hamilton City (Sac. R.)	In-River Abundance Downstream of Hamilton City (Sac. R.)	% of In-River Downstream of Hamilton City (Sac. R.)	All Fish Abundance Downstream of Hamilton City (Mok. R.)	% of All Fish Downstream of Hamilton City (Mok. R.)	In-River Abundance Downstream of Hamilton City (Mok. R.)	% of In-River Downstream of Hamilton City (Mok. R.)	All Fish Abundance Downstream of Hamilton City (San J. R.)	% of All Fish Downstream of Hamilton City (San R.)	In-River Abundance Downstream of Hamilton City (San J. R.)	% of In-River Downstream of Hamilton City (San J. R.)
2009	35,629	51.1	23,041	56.7	36,368	52.1	23,780	58.5	29,837	42.8	15,085	37.1	2,233	3.2	680	1.7	1,323	1.9	1,077	2.7
2010	61,611	34.5	38,869	32.0	64,908	36.3	42,166	34.7	103,415	57.9	74,347	61.2	7,935	4.4	2,660	2.2	2,423	1.4	2,277	1.9
2011	86,785	36.1	40,056	30.3	89,971	37.4	43,242	32.8	127,555	53.1	82,259	62.3	18,649	7.8	2,727	2.1	4,144	1.7	3,773	2.9
2012	158,976	45.4	71,639	35.2	167,016	47.7	79,679	39.2	161,875	46.3	110,458	54.3	13,162	3.8	6,542	3.2	7,800	2.2	6,800	3.3
2013	165,377	35.9	91,741	26.7	176,478	38.3	102,842	29.9	263,215	57.1	226,517	65.8	12,252	2.7	7,071	2.1	8,695	1.9	7,597	2.2
2014	114,274	41.8	90,128	43.3	123,374	45.1	99,228	47.7	132,465	48.4	100,702	48.4	12,486	4.6	3,670	1.8	5,231	1.9	4,420	2.1
2015	67,267	39.3	44,728	41.2	75,903	44.3	53,364	49.2	73,524	42.9	42,887	39.5	13,083	7.6	4,785	4.4	8,702	5.1	7,496	6.9
2016	25,648	18.2	14,771	16.3	26,950	19.1	16,073	17.8	87,714	62.2	57,931	64.0	10,119	7.2	3,232	3.6	16,214	11.5	13,218	14.6
2017	17,737	16.4	9,676	20.0	18,229	16.8	10,168	21.0	57,609	53.2	21,936	45.4	20,633	19.0	6,314	13.1	11,903	11.0	9,942	20.6
2018	52,438	28.8	34,352	28.3	53,687	29.5	35,601	29.3	104,941	57.6	70,373	58.0	18,263	10.0	11,082	9.1	5,321	2.9	4,346	3.6
2019	87,008	37.4	64,627	39.8	91,218	39.2	68,837	42.4	122,323	52.6	83,924	51.7	13,305	5.7	4,796	3.0	5,634	2.4	4,620	2.8
2020	61,838	37.2	44,509	38.1	62,854	37.8	45,525	39.0	97,831	58.9	69,374	59.4	4,044	2.4	601	0.5	1,440	0.9	1,255	1.1
Mean	77,882	35.2	47,345	34.0	82,246	37.0	51,709	36.8	113,525	52.7	79,649	53.9	12,180	6.5	4,513	3.9	6,569	3.7	5,568	5.4
Median	64,553	36.7	42,283	33.6	70,406	38.1	44,384	36.9	104,178	53.1	72,360	56.1	12,785	5.1	4,228	2.6	5,478	2.1	4,520	2.9

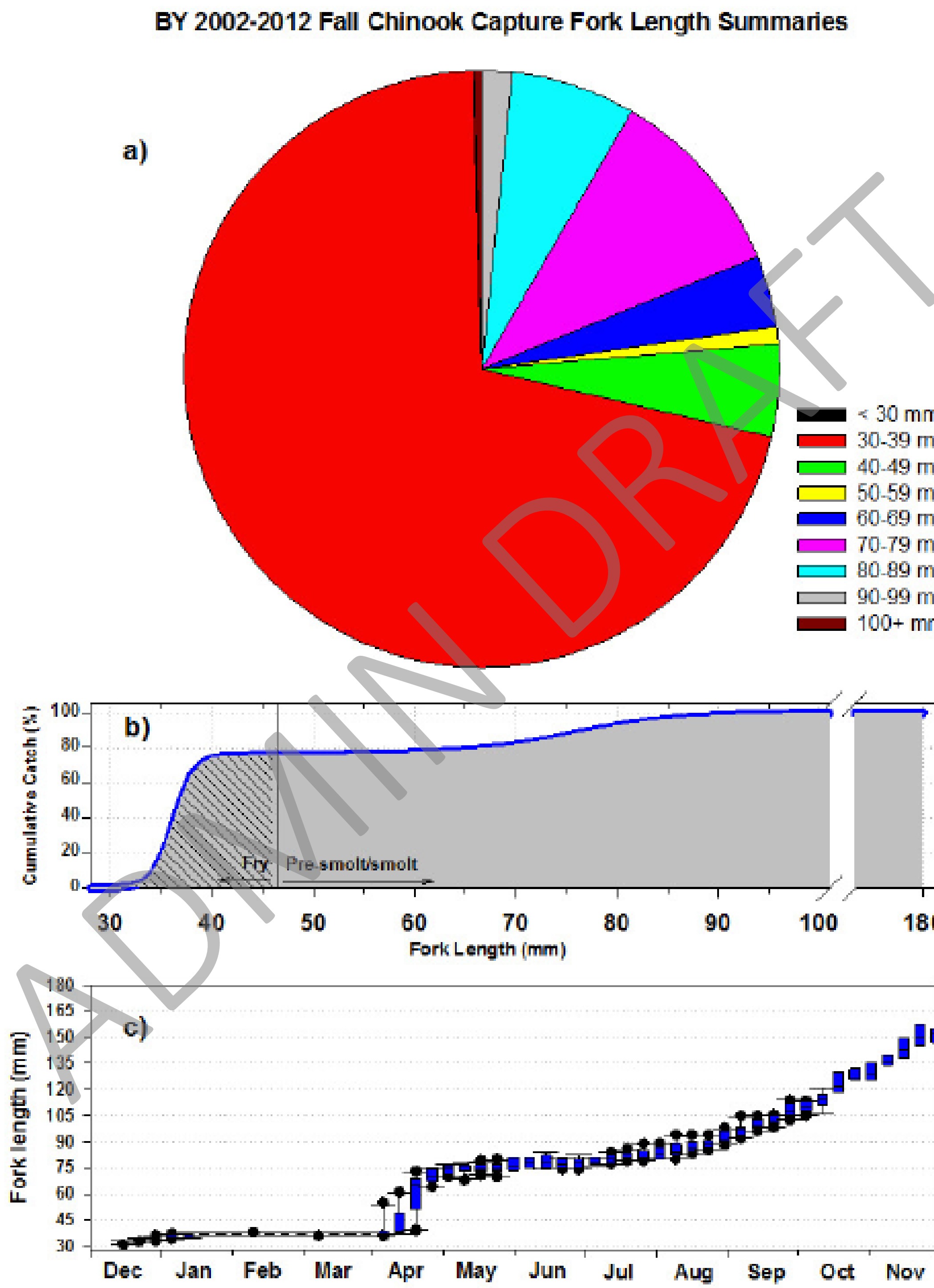
Source: Adapted from Azat (2021). Note: All fish = hatchery fish + in-river fish. Totals upstream of Red Bluff and Hamilton City are cumulative, so that fish counted as upstream of Red Bluff are also counted as upstream of Hamilton City, for example. Table also shows percentages downstream of Hamilton City in the Sacramento River (Sac. R.) basin, downstream of Hamilton City in the Mokelumne (Mok.) River, and downstream of Hamilton City in the San Joaquin (San J.) River Basin.

Effects of Alternatives 1, 2, and 3

Sacramento River

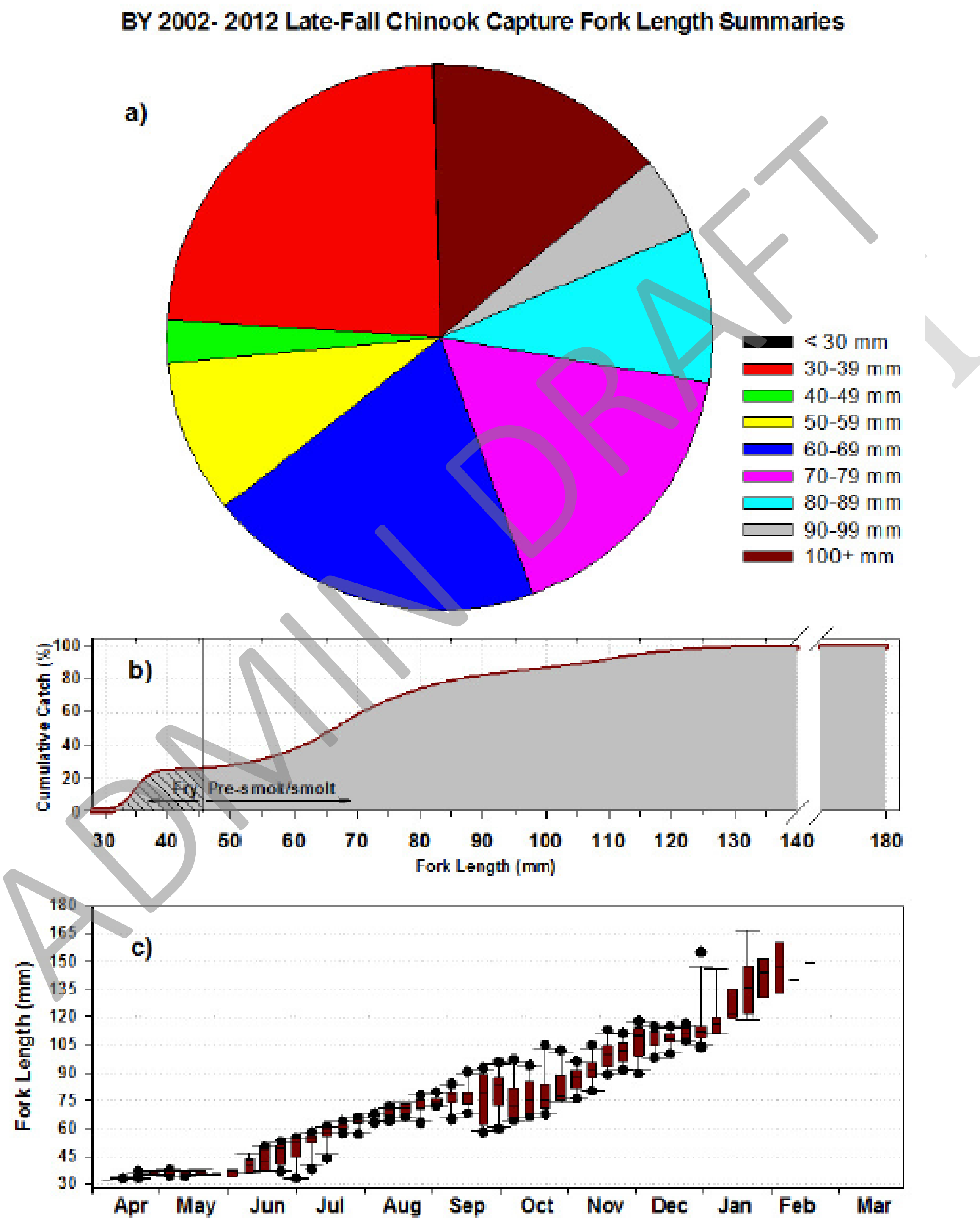
Near-Field Effects

Fall-run/late fall–run Chinook salmon would have the potential for similar types of near-field effects to those previously discussed for winter-run Chinook salmon (i.e., entrainment, impingement and screen contact, predation, stranding behind screens, and attraction to the Sacramento River discharge [Alternative 2 only]). The potential for effect would differ relative to winter-run Chinook because of spatiotemporal differences in species occurrence. As discussed above, around 30%–40% of fall/late fall–run Chinook salmon may pass the intakes (Table 11-38). As discussed for winter-run, few fall-run/late fall–run Chinook juveniles would have the potential for entrainment given their size (Figures 11-20 and 11-21). For late fall–run, the smallest individuals tend to pass the intakes beginning in spring, when there would be less difference in diversions between the NAA and the Project (Tables 11-6 and 11-7). The Red Bluff and Hamilton City fish screens are designed to protective standards for Chinook salmon fry and near-field effects would be expected to be limited. There would be greater temporal overlap of adult fall-run Chinook with reservoir releases than for winter-run Chinook salmon (Appendix 11A). For example, earlier migrant adults (July/August) could encounter reservoir releases to the Sacramento River of several hundred cfs in Critically Dry Water Years, with the main portion of the run (October/November) overlapping with generally lower releases (e.g., Critically Dry Water Year means in October ranging from nearly 170 cfs under Alternative 3 in October to nearly 230 cfs under Alternative 1A; see Chapter 5, Table 5-19). Late fall–run Chinook salmon have peak migration somewhat later than fall-run, and thus have less potential to overlap reservoir releases (Appendix 11A). As described for winter-run Chinook salmon, adult fall-run and late fall–run Chinook salmon could be attracted to the flow from the Sacramento River discharge under Alternative 2 and leap out of the river, but the apron and weir designed to exclude anadromous fish would eliminate stranding risk.



Source: Poytress et al. 2014:82. Note: BY = brood year.

Figure 11-20. Fall-Run Chinook Salmon Fork Length (a) Capture Proportions, (b) Cumulative Capture Size Curve, and (c) Average Weekly Median Boxplots, As Sampled at Red Bluff Diversion Dam Rotary Screw Traps, July 2002–June 2013.



Source: Poytress et al. 2014:83. Note: BY = brood year.

Figure 11-21. Late Fall–Run Chinook Salmon Fork Length (a) Capture Proportions, (b) Cumulative Capture Size Curve, and (c) Average Weekly Median Boxplots, As Sampled at Red Bluff Diversion Dam Rotary Screw Traps, July 2002–June 2013.

Far-Field Effects

As described previously in Impact FISH-2, there are likely multiple opportunities that would arise in real-time operations to coordinate exchanges between Sites Reservoir and Shasta Lake that could benefit anadromous fish. These are described in the Project description and reflected in the modeling for this document to some extent. However, due to the unique conditions in each year, additional opportunities for exchanges and coordination of real-time operations exist beyond those modeled for this document. The Authority and Reclamation intend to work together to better reflect the exchanges in the modeling with the goal of substantiating the Project's benefits to anadromous fish.

Temperature Effects

Operation of Sites Reservoir has the potential to change water temperatures in the Sacramento River that could affect the life stages of fall-/late fall–run Chinook salmon present. As described in Appendix 11B, the three methods used to analyze temperature-related effects on fall-run/late fall–run Chinook salmon in the Sacramento River were: (1) Physical Model Output Characterization; (2) Water Temperature Index Value/Range Exceedance Analysis; and (3) SALMOD. More details on these methods are provided in Appendix 11B and Appendix 11H.

The Authority and Reclamation evaluated water temperature model outputs during the period of presence and in the locations of each life stage of fall-run/late fall–run Chinook salmon in the Sacramento River upstream of the Delta (see Appendix 11B, Table 11B-2 for timing and locations). Visual observation of exceedance plots and differences in modeled mean monthly temperatures by water year type between Alternatives 1, 2, and 3 and the NAA in the Sacramento River below Keswick, at Balls Ferry, at Bend Bridge, below RBDD[20], and at Butte City indicates that water temperatures would be predominantly similar among alternatives during the period of presence of each life stage of fall-run/late fall–run Chinook salmon (Appendix 6C, Tables 6C-5-1a to 6C-5-4c, Tables 6C-7-1a to 6C-7-4c, Tables 6C-9-1a to 6C-9-4c, Tables 6C-10-1a to 6C-10-4c, Tables 6C-12-1a to 6C-12-4c; Figures 6C-5-1 to 6C-5-18, Figures 6C-7-1 to 6C-7-18, Figures 6C-9-1 to 6C-9-18, Figures 6C-10-1 to 6C-10-18, Figures 6C-12-1 to 6C-12-18). At all locations, mean monthly water temperatures for all months in all water year types under Alternatives 1A and 1B were within 0.6°F of the NAA with the exception of August of critical years at Butte City, in which mean monthly water temperature would be 0.8°F lower under the NAA than under Alternatives 1A and 1B. Water temperature modeling results for Alternatives 2 and 3 were similar to those of Alternative 1 at all locations. These results suggest that temperature-related effects on fall-run/late fall–run Chinook salmon in the Sacramento River would be negligible.

Results of the analysis of exceedance above water temperature index values for fall-run Chinook salmon in the Sacramento River from Appendix 11B, Table 11B-2 are presented in Appendix 11D, Table 11D-55 through Table 11D-76 and summarized in Table 11-39. For each life stage and at all locations evaluated, there were no month and water year type combinations in which both: (1) the percent of days that exceeded the index values was more than 5% greater under Alternative 1 than under the NAA; and (2) the exceedance per day was more than 0.5°F greater

[20] The RBDD, which was decommissioned in 2013, and the RBPP are co-located, and the names may be used interchangeably when referring to geographic locations

under Alternative 1 than under the NAA. Results of the exceedance analysis for Alternatives 2 and 3 are similar to Alternative 1 with no month and water year type combinations in which both criteria were met for any life stage at all locations.

Table 11-39. Number of Month and Water Year Type Combinations that Satisfy Both Criteria for Being Biologically Meaningful in the Water Temperature Index Value Analysis, Fall-run Chinook Salmon, Sacramento River[1,2,3]

Location	Spawning and Egg Incubation				Juvenile Rearing and Emigration				Adult Immigration				Adult Holding			
	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3
Below Keswick	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Below Clear Creek	0	0	0	0	0	0	0	0	NA[4]	NA	NA	NA	NA	NA	NA	NA
Balls Ferry	0	0	0	0	0	0	0	0	NA	NA	NA	NA	0	0	0	0
Bend Bridge	0	0	0	0	0	0	0	0	0	0	0	0	NA	NA	NA	NA
Below Red Bluff Diversion Dam	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Hamilton City	NA	NA	NA	NA	0	0	0	0	NA	NA	NA	NA	NA	NA	NA	NA

[1] Biologically Meaningful Criteria include: (1) the difference in frequency of exceedance between the NAA and the alternative was greater than 5%, and (2) the difference in average daily exceedance between the NAA and the alternative was greater than 0.5°F.
[2] Index values for each life stage are located in Appendix 11B, Table 11B-2.
[3] Full results presented in Appendix 11D, Table 11D-55 through Table 11D-76.
[4] NA = Not analyzed

Results of the analysis of exceedance above water temperature index values for late fall–run Chinook salmon in the Sacramento River from Appendix 11B, Table 11B-2 are presented in Appendix 11D, Table 11D-93 through Table 11D-111 and summarized in Table 11-40. For each life stage and at all locations evaluated, there were no month and water year type combinations in which both: (1) the percent of days that exceeded the index values was more than 5% greater under Alternative 1, 2, or 3 than under the NAA; and (2) the exceedance per day was more than 0.5°F greater under the Alternative 1, 2, or 3 than under the NAA. Results of the exceedance analysis for Alternative 2 are similar to Alternative 1 with no month and water year type combinations in which both criteria were met for any life stage at all locations. For Alternative 3, there was one month and water year type combination (July of Above Normal Water Years) at Hamilton City for the juvenile rearing and migration life stages in which there were 10.8% more days than the NAA exceeding the 64°F 7DADM index value and the mean daily exceedance on those days was 0.6°F greater than the NAA. Overall, this difference would be biologically inconsequential due to its low frequency and small magnitude.

Table 11-40. Number of Month and Water Year Type Combinations that Satisfy Both Criteria for Being Biologically Meaningful in the Water Temperature Index Value Analysis, Late Fall–run Chinook Salmon, Sacramento River[1,2,3]

Location	Spawning and Egg Incubation				Juvenile Rearing and Emigration				Adult Immigration			
	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3
Below Keswick	0	0	0	0	0	0	0	0	0	0	0	0
Below Clear Creek	0	0	0	0	0	0	0	0	NA[4]	NA	NA	NA
Balls Ferry	0	0	0	0	0	0	0	0	NA	NA	NA	NA
Bend Bridge	0	0	0	0	0	0	0	0	0	0	0	0
Below Red Bluff Diversion Dam	0	0	0	0	0	0	0	0	0	0	0	0
Hamilton City	NA	NA	NA	NA	0	0	0	1 Unfavorable (July, Above Normal Water Years)	NA	NA	NA	NA

[1] Biologically Meaningful Criteria include: (1) the difference in frequency of exceedance between the NAA and the alternative was greater than 5%, and (2) the difference in average daily exceedance between the NAA and the alternative was greater than 0.5°F.
[2] Index values for each life stage are located in Appendix 11B, Table 11B-2.
[3] Full results presented in Appendix 11D, Table 11D-93 through Table 11D-111.

Because SALMOD provides temperature- and flow-related outputs, SALMOD results are summarized separately in the section labeled *SALMOD* below.

Overall, water temperature–related effects of the Project on fall-/late fall–run Chinook salmon in the Sacramento River are expected to be biologically inconsequential due to the low frequency and small magnitude of differences between Alternatives 1, 2, and 3 and the NAA.

Flow-Related Physical Habitat Conditions

Redd Scour/Entombment

Loss of redds to scouring and entombment occurs when flows are high enough to mobilize sediments, destroying redds and their incubating eggs and alevins, or entombing the redds when sediments are redeposited. A flow of 40,000 cfs was selected as the scour flow threshold for the Sacramento River based on estimates in the literature (Appendix 11N, Table 11N-10).

The probability of redd scour and entombment was estimated for fall-run and late fall–run by computing the percentage of days with flows exceeding 40,000 cfs in the USRDOM 82-year daily flow record (29,952 days in total) at four locations between Keswick Dam and the RBPP during the months of fall-run and late fall–run spawning and incubation. The results for the NAA and Alternatives 1, 2, and 3 show that the probability of scour and entombment is consistently low for both runs, although somewhat higher for late fall–run than for fall-run (Table 11N-22 through Table 11N-25). The probabilities for both runs are higher than for winter-run and spring-run. The results indicate that Alternatives 1, 2, and 3 would have no adverse effect on redd scour and entombment for fall-run and late fall–run in the Sacramento River at any of the four locations.

Redd Dewatering

Fall-run. The percentage of redds in the Sacramento River lost to dewatering was estimated using tables in USFWS (2006) that relate spawning and dewatering flows to percent reductions in species-specific spawning habitat WUA (Appendix 11N). USRDOM flow data, which has a daily time-step, were available for three locations in this river section: Keswick Dam (RM 302), the Sacramento River at Clear Creek (RM 289), and the Sacramento River at Battle Creek (RM 271). USFWS (2006) developed a single relationship between flows and redd dewatering for the entire river section, but the flows used to estimate redd dewatering in the current analysis were those that best matched the longitudinal distribution of the redds of the different salmon runs in the river as estimated from aerial redd surveys conducted by CDFW from 2003 through 2019. Spawning of fall-run occurs primarily between ACID and RBPP (Table 11-10), so Sacramento River at Battle Creek flows were used to analyze fall-run redd dewatering.

Results are presented using the grand mean percentages of redds dewatered for each month of spawning, September through November, and each water year type and all water year types combined. The expected time for incubation of eggs and alevins is 3 months (Appendix 11N). Because changes in Project-related flow any time during this period can affect redd dewatering, the complete spawning and egg/alevin incubation periods (September–December and November–February) are provided in the results (Table 11N-15). The means of the redd dewatering estimates under the NAA and Alternatives 1, 2, and 3 are compared using absolute differences rather than relative differences (percent change) because many of the values for percentages of redds dewatered are small. Expressing changes of small values as percent changes may result in large values that may be misleading.

The results of the redd dewatering analysis for fall-run (Appendix 11N, Table 11N-15) show no large differences in redd dewatering between the NAA and Alternatives 1, 2, and 3. The largest differences are 1.5% increases under Alternatives 1A, 1B, and 2 for spawning in September of Critically Dry Water Years. Most other changes for all months and water year types under all the alternatives are less than 1%. The results indicate that the Project would have little effect on fall-run redd dewatering.

Late Fall–run. Spawning of late fall–run occurs primarily between Keswick Dam and the confluence with Clear Creek (Table 11-10), so Keswick Dam flows were used to analyze late fall–run redd dewatering. Results are presented using the grand mean percentages of redds dewatered for each month of late fall–run spawning, December through March, and each water year type and all water year types combined. The expected time for incubation of eggs and alevins is 3 months (Appendix 11N). Because changes in Project-related flow any time during this period can affect redd dewatering, the complete spawning and egg/alevin incubation periods (December–March through March–June) are provided in the results (Table 11N-16).

Results are presented using the grand mean percentages of redds dewatered for each month of spawning, December through March, and each water year type and all water year types combined. The expected time for incubation of eggs and alevins is 3 months (Appendix 11N). Because changes in Project-related flow any time during this period can affect redd dewatering, the complete spawning and egg/alevin incubation periods (December–March through March–June) are provided in the results (Table 11N-16).

The results for late fall–run redd dewatering show little effect from Alternatives 1A, 1B, and 2 with no differences from the NAA greater than 1.5% (Table 11N-16). Alternative 3 has several larger increases in redd dewatering, including 3% increases for spawning in December of Wet and Dry Water Years and February of Above Normal Water Years. Most other differences under Alternative 3 are small, but almost all represent increases in redd dewatering. The results indicate that Alternatives 1A, 1B, and 2 would have little effect on late fall–run redd dewatering, but Alternative 3 would result in somewhat greater redd dewatering than that under the NAA.

Spawning Habitat Weighted Usable Area

The suitability of physical habitat for salmonid spawning is largely of function of the availability of clean, coarse gravel for constructing redds, favorable depths, and suitable flow velocities. Instream flow potentially affects all these habitat characteristics, and often affects the availability of suitable habitat. Habitat suitability for spawning was analyzed using WUA curves developed by USFWS and others from results of field studies and hydraulic modeling (Appendix 11K).

Fall-run. Spawning habitat WUA for fall-run Chinook salmon in the Sacramento River was determined by USFWS (2003a, 2006) in the same manner that it was determined for winter-run Chinook salmon (Appendix 11K). To evaluate the effects of the Project on fall-run spawning habitat, fall-run spawning WUA was estimated for flows during the September through November spawning period under the NAA and Alternatives 1, 2, and 3 in the same three segments of the Sacramento River that were used for winter-run and spring-run Chinook salmon. However, because fall-run spawning occurs further downstream than spawning of the other runs, fall-run spawning WUA was estimated for an additional downstream segment (Segment 3) (U.S. Fish and Wildlife Service 2005a).

Reductions in mean month and water year type spawning WUA for fall-run are more frequent and typically larger with the Project than increases in mean spawning WUA relative to the NAA, especially in Segment 4 under Alternative 3, for which about half of the means show reductions of >3% (Table 11K-10). Reductions >3% are also frequent in November under Alternative 3 in Segment 6, for which all but Below Normal Water Years show reductions greater than 3% (Table 11K-8). The largest differences are 7% reductions under Alternative 3 for November of Critically Dry Water Years in Segment 6 (Table 11K-8) and October of Wet Water Years in Segment 4 (Table 11K-10). The largest increases are 4% under Alternatives 1A and 2 in September of Above Normal Water Years in Segment 5. The spawning distribution of fall-run is more evenly distributed over the four river segments than that of the other runs (Table 11-10), and therefore any differences resulting from Alternatives 1, 2, and 3 would potentially affect fall-run spawning habitat in any of the segments. The results indicate that Alternative 3 would result in frequent reductions, ranging up to 7%, in fall-run spawning habitat WUA, but these reductions are largely limited to Segments 6 and 4, so the effects are not expected to substantially affect overall fall-run spawning habitat availability.

Late Fall–run. Spawning habitat WUA for late fall–run Chinook salmon was determined by USFWS (2003a, 2006) in the same manner that it was determined for winter-run and fall-run Chinook salmon. To evaluate the effects of the Project on late fall–run spawning habitat, late fall–run spawning WUA was computed for flows during the December through March spawning period under Alternatives 1, 2, and 3 and the NAA in all three segments of the Sacramento River that were used for the other runs, but not Segment 3, which was used for the fall-run effects analysis only. About 90% of late fall–run redds occur in the three upstream segments, and 68% are found in Segment 6 (Table 11-10).

Mean late fall–run spawning WUA with the Project differ little from the NAA (Table 11K-12 and Table 11K-14). The largest differences are 6% reductions under Alternative 3 in Segment 6 for December of Wet and Dry Water Years (Table 11K-12). The largest increases are about 2% in Segment 6 for December of Above Normal Water Years under Alternatives 1, 2, and 3. Most differences in all river segments and all three alternatives are <3%. The results indicate that Alternatives 1A, 1B, and 2 would have little effect on late fall–run spawning WUA and Alternative 3 would have larger adverse effects, but the effects are not expected to substantially affect late fall–run spawning habitat availability.

Rearing Habitat Weighted Usable Area

The suitability of physical habitat for salmonid rearing is largely a function of water depth, flow velocity, and the availability and type of cover. Instream flow potentially affects all these habitat characteristics, and often affects the availability of suitable habitat. Habitat suitability for rearing was analyzed using WUA curves developed by USFWS and others from results of field studies and hydraulic modeling (Appendix 11K).

Fall-run. Rearing habitat WUA for fall-run fry and juveniles was determined by USFWS (2005b) in the same manner that it was determined for winter-run. To estimate changes in rearing WUA that would result from Alternatives 1, 2, and 3, the fall-run WUA curves developed for each of the three river segments (Figure 11-12) were used with mean monthly CALSIM II flow estimates for corresponding river segments under Alternatives 1, 2, and 3 and the NAA during the rearing periods for fry (December through March) and juveniles (February through June).

Most of the means for fry rearing WUA differ by less than 2% between Alternatives 1A, 1B, and 2 and the NAA (Table 11K-35 and Table 11K-36). However, under Alternative 3 in Segment 5, half of the means for water year type by month differ from the NAA by >2% (Table 11K-37). The largest differences for all three river segments and alternatives are 6% reductions under Alternative 3 in Segment 5 during October of Wet Water Years (Table 11K-36) and under Alternatives 1A and 1B in Segment 4 during September of Critically Dry Water Years (Table 11K-37). For juvenile rearing WUA, most differences in Segments 6 and 5 between the NAA and Alternatives 1A, 1B, and 2 are <2%, while about a third of the differences for Alternative 3 are >2% (Table 11K-38 and Table 11K-39). In Segment 4, many more of the means differ by >2%, especially under Alternatives 1B and 3 (Table 11K-40). In all three segments, all differences >4% occur in May and June and all constitute increases in rearing WUA. The largest differences are in Segment 4 and include 17% increases in June of Above Normal and Below Normal Water Years under Alternative 3 and a 12% increase in June of Above Normal Water Years under Alternative 1B. The largest reduction in juvenile rearing WUA, 4%, is in Segment 4 during May of Wet Water Years under Alternative 1B. The results for fall-run rearing WUA show small reductions in fry rearing WUA and substantial increases in juvenile rearing habitat WUA.

Late Fall–run. Rearing habitat WUA for late fall–run Chinook salmon fry and juveniles was determined by USFWS (2005b) in the same manner that it was determined for winter-run and fall-run Chinook salmon. To estimate changes in rearing WUA that would result from Alternatives 1, 2, and 3, the late fall–run Chinook salmon WUA curves developed for each of the three river segments was used with mean monthly CALSIM II flow estimates for corresponding river segments under Alternatives 1, 2, and 3 and the NAA during the rearing periods for late fall–run fry (March through June) and juveniles (May through October).

Most of the differences between Alternatives 1, 2, and 3 and NAA means for late fall–run fry rearing WUA in all river segments are <3% (Table 11K-41 through Table 11K-43). The largest differences are increases of 8% and 9% in May of Critically Dry Water Years and June of Above Normal Water Years, respectively, in Segment 6 under Alternative 3 (Table 11K-41). The largest reduction is 3% and occurs in March of Dry Water Years under Alternative 3 in Segment 5 (Table 11K-42).

Late fall–run mean juvenile rearing WUA differs between Alternatives 1A, 1B, and 2 and the NAA by <2% for most months and water year types in Segments 6 and 5 (Table 11K-44 and Table 11K-45). Many more >2% differences occur for Alternative 3 in these river segments and most of them result from increases in rearing WUA. The largest of these increases is 7% in August of Above Normal Water Years in Segment 6 (Table 11K-44). Segment 4 has many more >2% differences in means between Alternatives 1, 2, and 3 and the NAA than Segments 6 and 5 (Table 11K-46). The largest differences include 15% increases in rearing WUA in June of Above Normal and Below Normal Water Years under Alternative 3 and 8% to 9% reductions in rearing WUA in August of Critically Dry Water years under all three alternatives. The results for juvenile rearing WUA indicate largely minor differences between Alternatives 1, 2, and 3 and the NAA for Segments 6 and 5, but substantial differences for Segment 4. This segment generally has increased juvenile rearing habitat WUA under Alternatives 1B and 3 during late spring and summer and some reduction in habitat during August of Critically Dry Water Years. Alternative 3 has the most frequent and largest differences and, on balance, it is expected to benefit late fall–run rearing habitat availability. Overall, the results indicate that Alternatives 1 and 2 would have little effect on late fall–run fry rearing habitat availability, whereas Alternative 3 would provide a net benefit to late fall–run juvenile rearing habitat availability.

Juvenile Stranding

Fall-run. The juvenile stranding estimation procedure, which is identical for all the salmonids, is described in the *Juvenile Stranding* section of Impact FISH-2, and in Section 11N.2.3, *Juvenile Stranding*, of Appendix 11N.

The results are presented using the grand mean number of juveniles stranded for each month of emergence under each water year type and all water year types combined (Appendix 11N, Table 11N-28 through Table 11N-30). The analysis assumes that under equal flow conditions the fry and early juvenile stage of all runs and species are equally vulnerable to stranding. To determine the results for a given species or run, the estimated months for which the fry typically emerge (Appendix 11N, Table 11N-27), are consulted in the tables of results. The effects of dewatering flows are tracked by the analysis from the month of emergence through the 3 months following. These periods are given in the tables.

Fall-run fry are present in the upper Sacramento River primarily from about December through March (Table 11N-27). During this period, juvenile stranding with the Project results in both increases and reductions in stranding relative to the NAA in all three reaches of the river (Table 11N-28 and Table 11N-30). The most frequent large changes in stranding (>10%) for the period occur in the Keswick reach, with maximum increases including 18% and 19% in March of Critically Dry Water Years under Alternatives 1B and 3 and 18% in January of Above Normal Water Years under Alternative 3 (Table 11N-28). The maximum reductions are 16% and 17% in January of Dry Water Years in the Keswick reach under Alternatives 1A, 1B, and 2 (Table 11N-28). Overall, large (>10%) reductions and increases in stranding are about equally prevalent (Table 11N-28 and Table 11N-30). The juvenile stranding summary results for fall-run fry (Table 11N-31 through Table 11N-33) show no change or increased stranding in the Keswick and Clear Creek reaches under Alternatives 1, 2, and 3, including a 5.5% increase under Alternative 3 in the Keswick reach (Table 11N-31). The summary results for the Battle Creek reach show reductions in standing under Alternatives 1, 2, and 3 (Table 11N-33). On balance, Alternatives 1A, 1B, and 2 are not expected to substantially affect fall-run juvenile stranding, but Alternative 3 is expected to result in increased fall-run juvenile stranding in the Keswick reach.

Late fall–run. The principal period of stranding vulnerability for late fall–run is for fry cohorts emerging in March through June. The late fall–run fry cohorts emerging in April, May, and June are likely to be affected by large spring and summer changes in stranding under Alternatives 1, 2, and 3 relative to the NAA. Although especially large (>20%) increases and reductions in juvenile stranding are predicted for this period in all three river reaches, the large reductions are much more frequent than the increases, so Alternatives 1, 2, and 3 are expected, on balance, to result in reduced stranding of late fall–run fry (Table 11N-28 through Table 11N-30). The summary juvenile stranding results for late fall–run fry (Table 11N-31 through Table 11N-33) shows reductions in stranding in all three river segments under Alternatives 1, 2, and 3 of up to 2.4%, except for a 1.2% increase in the Keswick reach under Alternative 3 (Table 11N-31).

SALMOD

The Authority used the SALMOD model to ascertain the potential effect of each alternative on fall-run/late fall–run Chinook salmon mortality and potential production in the Sacramento River. A full description of the model can be found in the California WaterFix Biological Assessment (California Department of Water Resources 2016), Attachment 5.D.2, *SALMOD*

Model. In this section, SALMOD results for fall-run and late fall–run Chinook salmon are presented separately in their entirety. See Appendix 11H for details on the model.

The SALMOD outputs for fall-run Chinook salmon are presented in Appendix 11H, Table 1c-1 through Table 1c-4, Table 2c-1 through Table 2c-4, and Figure B-c-1 through Figure B-c-19. For all water year types combined for all life stages and source of mortality, mean annual fall-run Chinook salmon potential production would be similar under Alternative 1A and Alternative 1B relative to the NAA (0.4% greater; Appendix 11H, Table 2c-1 and Table 2c-2, Figure B-c-1). Differences within each water year type in mean annual potential production between Alternatives 1A and 1B and the NAA would be small (<1%, except in Above Normal Water Years under Alternative 1B, in which annual production would be 2.1% greater). Alternatives 2 and 3 results would be similar to those of Alternatives 1A and 1B (Appendix 11H, Table 2c-3, Table 2c-4, Figure B-c-1) in that differences in production relative to the NAA would be small (generally <1%, except for 1.3% and 4.2% higher potential production in Above Normal Water Years under Alternatives 2 and 3, respectively, relative to the NAA).

Results by life stage and mortality source are reported in Appendix 11H, Table 1c-1 through Table 1c-4 and Figure B-c-8 through Figure B-c-19. Depending on the life stage and source of mortality (flow-/habitat-based or temperature-based), mean annual mortality under Alternative 1A would be between 44% lower to 68% greater than that under the NAA and mortality under Alternative 1B would be between 80% lower to 126% higher than that under the NAA (Appendix 11H, Table 1c-1, Table 1c-2). However, it is important to understand that the model is seeded with 56,115,000 fall-run eggs each year. Although there may be larger percent differences in mean annual mortality between Alternative 1, 2, or 3 and the NAA by source and life stage, these differences typically represent a small proportion of overall individuals when put in a broader population-wide context. The largest absolute difference in mean annual mortality would be in pre-spawn mortality between Alternative 1B and the NAA in Above Normal Water Years. In this case, there would be a reduction in mean annual egg mortality of 1,870,606 eggs under Alternative 1B relative to the NAA, accounting for just 3.3% of the 56,115,000 fall-run Chinook salmon eggs annually seeded into the model. Combining all sources of mortality and life stages together, mean annual mortality would be 1% lower under Alternative 1A and 2% lower under Alternative 1B compared to the NAA for the full simulation period (all water year types combined). By water year type, mean annual mortality under Alternative 1A would be between 4% lower (in Above Normal Water Years) and 2% greater (in Critically Dry Water Years) than the NAA. Mean annual mortality under Alternative 1B would be between 11% lower (in Above Normal Water Years) and 1% greater (in Critically Dry Water Years). Mean annual mortality results for Alternatives 2 and 3 are generally similar to those of Alternatives 1A and 1B (Appendix 11H, Table 1d-3, Table 1d-4). Combining all sources of mortality and life stages together, mean annual mortality under Alternative 2 would be 2% lower than under the NAA for the full simulation period (all water year types combined) and range from 6% lower than under the NAA to no difference from the NAA depending on water year type. Mean annual mortality under Alternative 3 would be 0% different than under the NAA for the full simulation period (all water year types combined) and range from 21% lower than under the NAA to no difference from the NAA depending on water year type.

Overall, SALMOD results for fall–run Chinook salmon show a negligible effect of Alternatives 1A, 1B, 2, and 3 on mortality and potential production in the Sacramento River.

The SALMOD model outputs for late fall–run Chinook salmon are presented in Appendix 11H, Table 1d-1 through Table 1d-4, Table 2d-1 through Table 2d-4, and Figure B-d-1 through Figure B-d-19. For all water year types combined for all life stages and source of mortality, mean annual late fall–run Chinook salmon potential production would be similar under both Alternatives 1A and 1B relative to the NAA (0.4% higher under Alternative 1A and 0.7% higher under Alternative 1B; Appendix 11H, Table 2d-1 and Table 2d-2, Figure B-d-1). Differences within each water year type in mean annual potential production between Alternatives 1A and 1B and the NAA were <2%. Alternatives 2 and 3 results would be similar to those of Alternatives 1A and 1B (Appendix 11H, Table 2d-3, Table 2d-4, Figure B-d-1) in that differences in production relative to the NAA would be <2% both with all water year types combined and within each water year type.

Results by life stage and mortality source for late fall–run Chinook salmon are reported in Appendix 11H, Table 1d-1 through Table 1d-4 and Figure B-d-8 through Figure B-d-19. Depending on the life stage and source of mortality (flow-/habitat-based or temperature-based), mean mortality under Alternative 1A would be between 100% lower to 116% greater than that under the NAA and mortality under Alternative 1B would be between 100% lower to 248% higher than that under the NAA (Appendix 11H, Table 1d-1, Table 1d-2). However, it is important to understand that the model is seeded with 13,325,000 late fall–run eggs each year. Although there may be larger percent differences in mean mortality between Alternatives 1, 2, or 3 and the NAA by source and life stage, these differences typically represent a small proportion of overall individuals when put in a broader population-wide context. The largest raw difference in mean mortality determined through comparison of primary data would be in egg flow-related mortality between Alternative 1B and the NAA in Wet Water Years. In this case, there would be a reduction in mean annual egg mortality of 149,621 individuals under Alternative 1B relative to the NAA, accounting for just 1.1% of the 13,325,000 late fall–run Chinook salmon eggs annually seeded into the model. Combining all sources of mortality and life stages together, mean annual mortality under Alternatives 1A and 1B would be 2% and 3% lower, respectively, than under the NAA for the full simulation period (all water year types combined). By water year type, mean annual mortality under Alternative 1A would be between 4% lower (in Wet Water Years) and 2% greater (in Above Normal Water Years) than the NAA. Mean annual mortality under Alternative 1B would be between 5% lower than the NAA (in Wet Water Years) and no change from the NAA (in Above Normal and Below Normal Water Years). Mean annual mortality results for Alternatives 2 and 3 are generally similar to those of Alternatives 1A and 1B (Appendix 11H, Table 1d-3, Table 1d-4). Combining all sources of mortality and life stages together, mean annual mortality under Alternative 2 would be 2% lower than under the NAA for the full simulation period (all water year types combined) and 0% to 5% lower than under the NAA depending on water year type. Mean annual mortality under Alternative 3 would be 3% lower than under the NAA for the full simulation period (all water year types combined) and between 1% and 4% lower than under the NAA depending on water year type.

Overall, SALMOD results for Alternatives 1A, 1B, 2, and 3 for late fall–run Chinook salmon show a minimal negative effect on mortality and potential production in the Sacramento River.

Floodplain Inundation and Access

As described in Chapter 2 and as discussed for winter-run Chinook salmon, the Project's diversion criteria include restrictions to maintain Bend Bridge and Wilkins Slough flows, which have the effect of limiting changes to Yolo Bypass spill frequency and duration under Alternatives 1, 2, and 3 relative to the NAA. In particular, these criteria avoid impacts on Reclamation's ability to implement its obligations in the 2019 NMFS ROC ON LTO BiOp to implement the Yolo Bypass Restoration Salmonid Habitat Restoration and Fish Passage Implementation Plan and provide more than 17,000 acres of inundation in the Yolo Bypass from December to April (National Marine Fisheries Service 2019a). As such, Alternatives 1, 2, and 3 generally would have limited potential for negative effects on Yolo Bypass floodplain inundation and access for fall-run/late fall–run Chinook salmon. This was confirmed with the modeling summarized in the winter-run Chinook salmon section (Appendices 11M1, *Acres of Yolo Bypass with Limiting Habitat Suitability Criteria (Depth < 1 meter deep and/or Flow Velocity < 1.5 feet per second) for Rearing Salmonids under Three Different Fremont Weir Spills Levels*, and 11M2). The results of the run-specific Acierto et al. (2014) method for juvenile entry into Yolo Bypass were consistent with those of winter-run Chinook salmon in suggesting similar or somewhat less entry into Yolo Bypass under the Project compared to the NAA (Table 11-41). As noted for winter-run Chinook salmon, there is the potential for a somewhat lower proportion of juvenile fall-run/late fall–run Chinook salmon entering the Sutter Bypass from the Sacramento River under the Project relative to the NAA.

Table 11-41. Proportion of Juvenile Fall-Run Chinook Salmon Entering Yolo Bypass Via Fremont Weir, Based on Acierto et al. (2014).

Water Year	Water Year Type	NAA	With Project*
2009	Dry	0.070	0.061 (-12%)
2010	Below Normal	0.056	0.053 (-6%)
2011	Wet	0.167	0.160 (-4%)
2012	Below Normal	0.032	0.030 (-6%)
2013	Dry	0.097	0.096 (-1%)
2014	Critically Dry	0.014	0.014 (0%)
2015	Critically Dry	0.049	0.049 (0%)
2016	Below Normal	0.037	0.032 (-13%)
2017	Wet	0.398	0.389 (-2%)

Notes: Percentage values in parentheses indicate differences of alternatives compared to the NAA. *Results are the same for Alternatives 1, 2, and 3 because the same operational criteria are applied.

As discussed in Impact FISH-2, and in Appendix 11M, the Project is not expected to substantially affect suitable inundated floodplain habitat on the Yolo and Sutter Bypasses or suitable inundated side-channel habitat on the Sacramento River for rearing juvenile Chinook salmon, including fall-run/late fall–run. This conclusion is based on the results of habitat modeling that showed little difference in suitable floodplain habitat acreage between the NAA and Alternatives 1, 2, and 3 for the Sutter Bypass (Table 11M-4 and Figure 11M-8), an absence of large differences in acreage of suitable side-channel habitat in the Sacramento River (Table 11M-5 through Table 11M-7 and Figure 11M-9), and a reduction of less than 100 acres (1.8%) in the total suitable habitat acreage on the Yolo Bypass (Table 11-14 and Table 11M-3), despite relatively large reductions for a few month and water year combinations (Table 11-13; Table 11M-1).

Adult Upstream Passage at Fremont Weir

Adult Chinook salmon migrate upstream during Fremont Weir overtopping events, as well as during lower flow conditions (National Marine Fisheries Service 2019b:26). Sommer et al. (2014) did not find adult Chinook salmon catch in a fyke trap in the Toe Drain was associated with Yolo Bypass flow events, noting this may have been because most of the Chinook salmon were fall-run, a race known to migrate upstream relatively early before winter and spring flow events. Recent completion of the Wallace Weir fish rescue facility and fish passage facilities at Fremont Weir blocked access to the CBD and improved adult fish passage allowing passage at a variety of flows into the Yolo Bypass, including across the range anticipated under the NAA and Alternatives 1, 2, and 3. As such, the minor differences in flow entering Yolo Bypass at Fremont Weir under Alternatives 1, 2, and 3 relative to the NAA (see discussion above related to inundation and juvenile access) would not result in major differences in adult upstream passage with the Project compared to the NAA.

Adult Upstream Passage at Sutter Bypass Weirs

As described for winter-run Chinook salmon, changes in Sacramento River flow and flow entering Sutter Bypass as a result of the Project have the potential to change the number of days meeting adult Chinook salmon passage criteria at the three Sutter Bypass weirs (Moulton, Tisdale, and Colusa) compared to the NAA. The analysis indicates there would be no days meeting adult salmon passage criteria for Alternative 1, 2, and 3 during 2009–2018 at Moulton or Tisdale Weirs. For Colusa Weir, the number of days meeting fish passage criteria would be little different between the Project and the NAA, with the exception being 2016 (nearly 40% fewer days of passage; Table 11-22).

Migration Flow-Survival

As described in more detail for winter-run Chinook salmon, the Project includes pulse flow protection measures to be applied to precipitation-generated pulse flow events from October through May. There is also a 10,700-cfs Wilkins Slough bypass flow threshold from October 1 through June 14, corresponding with the threshold identified by Michel et al. (2021). Adaptive management would be utilized to assess and if necessary refine the diversion criteria based on biological monitoring (see Appendix 2D, Section 2D.6). The discussion in Section 11P.2 of Appendix 11P illustrates that the Sites Reservoir diversion criteria effectively limit diversions during the historical periods of fish movement as reflected in Red Bluff rotary screw trap data, and application of the flow-threshold criteria from Michel et al. (2021) suggests that flow-survival effects on juvenile Chinook salmon (including fall-run and late fall–run Chinook salmon) would be greatly limited by the Project's diversion criteria because there is essentially no difference in predicted survival between the Project and NAA.

Sites Reservoir Release Effects

Sites Reservoir releases could temporally overlap with fall-run/late fall–run Chinook salmon presence near the release location in the Sacramento River. For fall-run, migrating adults would be present during July through December (Appendix 11A, Table 11A-6) and rearing or emigrating juveniles would be present during December through May (Figure 11-17). For late fall–run, migrating adults would be present during October and November and rearing or emigrating juveniles could be present during all months (Appendix 11A, Table 11A-7; Figure 11-19).

Sites Reservoir releases into the Yolo Bypass via the CBD would not overlap with the juvenile fall-run Chinook salmon rearing and emigration during August through October (Figure 11-17). Releases into the Yolo Bypass would overlap with the adult late fall–run Chinook salmon migration period during October and the juvenile rearing and emigration period during August through October (Appendix 11A, Table 11A-7; Figure 11-19).

In contrast to the other runs of Chinook salmon, the adult fall-run Chinook salmon upstream migration period (Appendix 11A) coincides with increased Yolo Bypass flows in August–October as a result of reservoir releases under Alternatives 1, 2, and 3 (Appendix 5B3, Tables 5B3-3-1a through 5B3-3-4c). Johnston et al. (2020) found that the median probability of acoustically tagged adult fall-run Chinook salmon exiting the Yolo Bypass after entering it was 0.74, indicating that nearly a quarter of fish entering the Toe Drain may not leave. Johnston et al. (2020) suggested that Chinook salmon adults may be drawn into the Yolo Bypass Toe Drain by the higher flows in the Cache Slough Complex relative to those in the Sacramento River; however, the Project would not affect this potential mechanism. Sommer et al. (2014) did not find adult Chinook salmon catch in a fyke trap in the Toe Drain was associated with Yolo Bypass flow events, noting this may have been because most of the Chinook salmon were fall-run, a race known to migrate upstream relatively early before winter and spring flow events. Acoustic telemetry investigations in 2015–2017 found that adult Chinook salmon strongly responded to tidal switches, but these studies did not provide information on effects of the summer/fall managed flow releases, such as whether flows attract more fish into the Yolo Bypass (Singer and Frantzich 2019). Fall-run Chinook salmon entering the Toe Drain may eventually reach the Wallace Weir, where fish rescue and relocation to the Sacramento River by CDFW occurs, either at the recently completed Wallace Weir fish rescue facility or by beach seine in the vicinity of the Wallace Weir. Adult Chinook salmon were captured and relocated to the Sacramento River in 2018 and 2019 during August–October, periods including managed flow actions resulting in greater flow entering the Toe Drain (Table 11-42). In 2019, eight of 340 Chinook salmon rescued from Wallace Weir died, a mortality rate of 2.3% (Davis et al. 2019). This rate of rescue and mortality is low compared the overall ESU size, which numbers in the tens of thousands of fish or greater (Table 11-38). Under the assumption that the increased rate of Chinook salmon occurrence at Wallace Weir is solely a function of the managed flow releases, which is uncertain, then this indicates that Sites Reservoir releases to the CBD and thence the Toe Drain under Alternatives 1, 2, and 3 would result in low ESU-level effects on fall-run Chinook salmon.

Table 11-42. Number of Unmarked and Marked Chinook Salmon Collected at Wallace Weir, in Relation to Fish Trap Capture Effort (Hours).

Year	Month	Hours Fished	Unmarked Salmon	Marked Salmon
2014	8	0	0	0
2014	9	332.5	0	0
2014	10	564.5	10	5

Year	Month	Hours Fished	Unmarked Salmon	Marked Salmon
2015	8	0	0	0
2015	9	0	0	0
2015	10	93.75	0	0
2016	8	0	0	0
2016	9	0	0	0
2016	10	0	0	0
2017	8	0	0	0
2017	9	23	0	0
2017	10	277.75	0	0
2018	8	93.5	0	0
2018	9	288.25	68	5
2018	10	0	3	0
2019	8	85.25	0	0
2019	9	457.5	135	17
2019	10	22.25	176	12
2020	8	0	0	0
2020	9	0	0	0
2020	10	736.5	0	0

Source: Purdy and Kubo 2021. Note: 2019 collection of salmon included individuals collected by beach seining near Wallace Weir in addition to the Wallace Weir trap.

Temperature Effects

As discussed in Chapter 6, the effects of Alternatives 1A, 1B, 2, and 3 on water temperatures at the Sites Reservoir release site in the Sacramento River would be relatively small with the releases generally tending to cause a slight reduction in water temperature (median monthly change <0.8°F; Tables 6-12a through 6-12d). Temperature-related effects of Alternatives 1A, 1B, 2, and 3 on fall-run/late fall–run Chinook salmon at the release site would be minimal.

For Alternatives 1A, 1B, 2, and 3, water temperatures in the CBD would either stay the same or be reduced due to Sites Reservoir releases (Table 11-23). These lower water temperatures are expected to have a negligible effect on adult fall-run and late fall–run Chinook salmon in the Yolo Bypass and would not be expected to have lethal effects on juvenile late fall–run Chinook salmon. Lower temperatures could reduce the growth of juveniles somewhat due to reduced metabolism. However, the small reduction in water temperatures is expected to occur in a limited area of the Yolo Bypass before it equilibrates with atmospheric temperatures. As a result, temperature effects in the Yolo Bypass due to Sites Reservoir releases under Alternatives 1A, 1B, 2, and 3 would be minimal at a population level for fall-run/late fall–run Chinook salmon.

As discussed in Impact FISH-2, temperature changes under Alternatives 1A, 1B, 2, and 3 in the Sacramento River below the Yolo Bypass (from Sites Reservoir releases into the CBD that would flow through the Yolo Bypass), and the effects on fall-run/late fall–run Chinook salmon, would be inconsequential due to the small proportion of Sacramento River water in this reach (Sacramento River below the Yolo Bypass) coming off the Yolo Bypass.

Feather River

Far-Field Effects

Temperature Effects

Operation of Sites Reservoir has the potential to change water temperatures in the Feather River that could affect the life stages of fall-run Chinook salmon present. As described in Appendix 11B, the two methods used to analyze temperature-related effects on winter-run Chinook salmon non-natal rearing in the Feather River were: (1) Physical Model Output Characterization; and (2) Water Temperature Index Value/Range Exceedance Analysis. More details on these methods are provided in Appendix 11B.

The Authority and Reclamation evaluated water temperature model outputs during the period of presence and in the locations of each life stage of fall-run Chinook salmon in the Feather River (see Appendix 11B, Table 11B-3 for timing and locations). Visual observation of exceedance plots and differences in modeled mean monthly temperatures by water year type between Alternatives 1A, 1B, 2 and 3 and the NAA in the Feather River LFC below the Fish Barrier Dam and in the HFC at Gridley Bridge indicates that water temperatures would be predominantly similar among alternatives during the period of presence of each life stage of fall-run Chinook salmon (Appendix 6C, Tables 6C-16-1a to 6C-16-4c, Tables 6C-18-1a to 6C-18-4c; Figures 6C-16-1 to 6C-16-18, Figures 6C-18-1 to 6C-18-18). At both locations, mean monthly water temperatures for all months within all water year types under Alternatives 1A and 1B were within 0.5°F of the NAA. Water temperature modeling results for Alternatives 2 and 3 were similar to those of Alternative 1 at both locations. These results suggest that temperature-related effects on fall-run Chinook salmon in the Feather River would be negligible.

Results of the analysis of exceedance above water temperature index values for fall-run Chinook salmon in the Feather River from Appendix 11B, Table 11B-3 are presented in Appendix 11D, Table 11D-77 through Table 11D-84 and summarized in Table 11-43. For each life stage and at both locations evaluated, there were no month and water year type combinations in which both: (1) the percent of months that exceeded the index values was more than 5% greater under the alternative than under the NAA; and (2) the exceedance per month was more than 0.5°F greater under the alternative than under the NAA. Results of the exceedance analysis for Alternatives 2 and 3 are similar to Alternative 1 with no month and water year type combinations in which both criteria were met for any life stage at both locations. These results indicate that temperature-related effects on fall-run Chinook salmon in the Feather River would be negligible.

Table 11-43. Number of Month and Water Year Type Combinations that Satisfy Both Criteria for Being Biologically Meaningful in the Water Temperature Index Value Analysis, Fall-run Chinook Salmon, Feather River[1,2,3]

Location	Spawning and Egg Incubation				Juvenile Rearing and Migration				Adult Immigration				Adult Holding			
	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3
Low-Flow Channel Below Fish Barrier Dam	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
High-Flow Channel Below Thermalito Afterbay Outlet	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

[1] Biologically Meaningful Criteria include: (1) the difference in frequency of exceedance between the NAA and the alternative was greater than 5%, and (2) the difference in average monthly exceedance between the NAA and the alternative was greater than 0.5°F.
[2] Index values for each life stage are located in Appendix 11B, Table 11B-3.
[3] Full results presented in Appendix 11D, Table 11D-77 through Table 11D-84.

The probability that each alternative would meet water temperature targets to support anadromous fish, including fall-run Chinook salmon, in the Feather River included the Settlement Agreement for Licensing of the Oroville Facilities (California Department of Water Resources 2006:A-18–A-24) was described in Impact FISH-3 above. Those results indicate that the Project's implementation under Alternatives 1, 2, and 3 would not substantially change the ability to meet water temperature targets in the Oroville Settlement Agreement relative to the NAA.

Combined, these water temperature results indicate that the Project's implementation under Alternatives 1, 2, and 3 would cause inconsequential temperature-related effects on fall-run Chinook salmon in the Feather River compared to the NAA.

Flow-Related Physical Habitat Conditions

Redd Scour and Entombment

Frequency of scouring flows was not estimated for the Feather River because information on minimum flows required to mobilize sediments is not available for the Feather River. However, Feather River flows during Wet and Above Normal Water Years, when scouring flows would be most likely to occur during the months of fall-run spawning and egg incubation (October through February), are generally similar between the NAA and Alternatives 1, 2, and 3 (Chapter 5).

Therefore, no substantial differences on the frequency of scouring flows are expected between the NAA and Alternatives 1, 2, and 3.

Redd Dewatering

As described in Section 11N.2, *Methods*, redd dewatering for Feather River Chinook salmon was estimated from the depth distributions of their redds and changes in river stage (depth) at the Gridley gage on the Feather River at different flows. The redd depth distributions were determined from DWR studies of Feather River salmon (Appendix 11N, Figure 11N-2). The river depths were estimated from CALSIM II monthly flow results at the Thermalito Afterbay outlet and DWR's stage-discharge tables for the Gridley gage. Comparisons of the means of the redd dewatering estimates under Alternatives 1, 2, and 3 to means under the NAA use absolute differences rather than relative differences (percent change) because many of the values for percentages of redds dewatered are small. Expressing changes of small values as percent changes may result in very large values that may be misleading. The use of monthly time-step flow estimates likely underestimates redd dewatering rates because they smooth out short-term flow fluctuations. This potential bias is expected to affect all Project scenarios equally. The results for fall-run show few differences in fall-run redd dewatering between Alternatives 1, 2, and 3 and the NAA. Most differences for all the alternatives are < 1%. The one relatively large difference is a 5% increase in redd dewatering for spawning in October of Below Normal Water Years under Alternative 1B. In general, these results indicate that Alternatives 1, 2, and 3 would have little effect on fall-run redd dewatering in the Feather River compared to the NAA.

Spawning Habitat Weighted Usable Area

Spring-run and fall-run are often difficult to distinguish in the Feather River, so a single WUA curve was developed for both runs (Payne and Allen 2004) (Figure 11K-7). The curve was used to compute spawning WUA with flows specific to the months of spawning for the run analyzed. To evaluate the effects of Alternatives 1, 2, and 3 on fall-run spawning habitat in the Feather River, the spawning WUA was estimated under Alternatives 1, 2, and 3 and the NAA for CALSIM II flows below Thermalito Afterbay during October through December, the Feather River fall-run spawning period. Differences in spawning WUA between Alternatives 1, 2, and 3 and the NAA were examined using the grand mean spawning WUA for each month of the spawning period under each water year type and all water year types combined (Table 11K-18 and Table 11K-19).

The largest differences between Alternatives 1, *2*, and 3 and the NAA means were 5% increases in WUA for October of Dry Water Years under Alternatives 1A, 1B, and 2. Most differences for both runs are <2%. These results suggest that Alternatives 1, 2, and 3 have little effect on spring-run or fall-run spawning WUA in the HFC of the Feather River.

Rearing Habitat Weighted Usable Area

As discussed above for Feather River spring-run Chinook salmon, reliable predictions regarding changes in flow affecting rearing habitat for Chinook salmon in the Feather River cannot be made because the only rearing WUA curves developed for Feather River Chinook salmon are unreliable (Payne and Allen 2005). WUA curves from other rivers (e.g., Sacramento River) cannot be used to determine the availability of rearing habitat WUA in response to changes in flow in the Feather River because relationship between flow and habitat WUA are generally not transferrable from one river to another (Bovee et al. 1998). As such, quantitative conclusions regarding effects of Alternatives 1, 2, and 3 on fall-run rearing WUA cannot be drawn.

Juvenile Stranding

No formal method for analyzing juvenile stranding analysis has been developed for the Feather River.

American River

Far-Field Effects

Temperature Effects

Operation of Sites Reservoir has the potential to change water temperatures in the American River that could affect the life stages of fall-run Chinook salmon present. As described in Appendix 11B, the two methods used to analyze temperature-related effects on fall-run Chinook salmon in the American River were: (1) Physical Model Output Characterization; and (2) Water Temperature Index Value/Range Exceedance Analysis. More details on these methods are provided in Appendix 11B.

The Authority and Reclamation evaluated water temperature model outputs during the period of presence and in the locations of each life stage of fall-run Chinook salmon in the American River (see Appendix 11B, Table 11B-4 for timing and locations). Visual observation of exceedance plots and differences in modeled mean monthly temperatures by water year type between Alternatives 1A, 1B, 2, and 3 and the NAA in the American River below Nimbus Dam and Watt Avenue indicates that water temperatures would be predominantly similar among alternatives during the period of presence of each life stage of fall-run Chinook salmon (Appendix 6C, Tables 6C-13-1a to 6C-13-4c, Tables 6C-14-1a to 6C-14-4c; Figures 6C-13-1 to 6C-13-18, Figures 6C-14-1 to 6C-14-18). At both locations, mean monthly water temperatures for all months within all water year types under Alternatives 1A and 1B were within 0.3°F of the NAA. Water temperature modeling results for Alternatives 2 and 3 were similar to those of Alternative 1 at both locations. These results suggest that temperature-related effects on fall-run Chinook salmon in the American River would be negligible.

Results of the analysis of exceedance above water temperature index values for fall-run Chinook salmon in the American River from Appendix 11B, Table 11B-4 are presented in Appendix 11D, Table 11D-85 through Table 11D-92 and summarized in Table 11-44. For each life stage and at both locations evaluated, there were no month and water year type combinations in which both: (1) the percent of days that exceeded the index values was more than 5% greater under the alternative than under the NAA; and (2) the exceedance per day was more than 0.5°F greater under the alternative than under the NAA. Results of the exceedance analysis for Alternatives 2 and 3 are similar to Alternative 1 with no month and water year type combinations in which both criteria were met for any life stage at both locations. These results indicate that temperature-related effects on fall-run Chinook salmon in the American River would be negligible.

Table 11-44. Number of Month and Water Year Type Combinations that Satisfy Both Criteria for Being Biologically Meaningful in the Water Temperature Index Value Analysis, Fall-run Chinook Salmon, American River[1,2,3]

Location	Spawning and Egg Incubation				Juvenile Rearing and Migration				Adult Immigration				Adult Holding			
	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3
Hazel Avenue	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Watt Avenue	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

[1] Biologically Meaningful Criteria include: (1) the difference in frequency of exceedance between the NAA and the alternative was greater than 5%, and (2) the difference in average daily exceedance between the NAA and the alternative was greater than 0.5°F.
[2] Index values for each life stage are located in Appendix 11B, Table 11B-4.
[3] Full results presented in Appendix 11D, Table 11D-85 through Table 11D-92.

Combined, these water temperature results indicate that the Project's implementation under Alternatives 1, 2, and 3 would cause inconsequential temperature-related effects on fall-run Chinook salmon in the American River.

Flow-Related Physical Habitat Conditions

Redd Scour and Entombment

The American River fall-run population is abundant and fairly stable, but over the past decade about 30% of the returning adults have been from the Nimbus Hatchery (California Department of Fish and Wildlife 2022). When hatchery adults spawn in the river, they reduce the genetic fitness of the river's population. This places a premium on natural spawning and availability of high-quality spawning and incubation habitat.

Loss of redds to scouring or entombment occurs when flows are high enough to mobilize sediments, destroying redds and their incubating eggs and alevins, or entombing the redds when sediments are redeposited. A flow of 40,000 cfs was selected as the scour flow threshold for the American River based on estimates in the literature (Appendix 11N, Table 11N-10).

Only CALSIM II monthly flow estimates were available for estimating flows in the American River under Alternatives 1, 2, and 3 and the NAA. Redd scour can occur at a small temporal scale (minutes to hours). This limitation was addressed using historical American River gage data to determine the minimum monthly flow for which at least one daily flow of the month exceeded 40,000 cfs (Appendix 11N). The minimum monthly flow was 19,350 cfs, so this flow was used as the threshold scouring flow with CALSIM II data in American River at Nimbus Dam (Figure 11N-2).

The results indicate that there are few months in the 82-year CALSIM II record for the American River with flow greater than the redd scour or entombment threshold of 19,350 cfs (Appendix 11N, Table 11N-26). There are only 2 months with such flows under the NAA and Alternatives

1, 2, and 3. These results indicate that Alternatives 1, 2, and 3 would have no effect on redd scour or entombment for fall-run in the American River.

Redd Dewatering

The redd dewatering analysis for the lower American River uses relationships between flow, river stage, and redd depth distribution developed by Bratovich et al. (2017). CALSIM II flow estimates at the Nimbus Dam location were used to compute stage at the spawning and dewatering flows, and the redd depth frequency distribution was queried to determine the percentage of the redds that occur between those two stages and would be dewatered. The analyses were conducted for the months of fall-run spawning and incubation. The analysis compared CALSIM II flow estimates below Nimbus Dam for each spawning month with the minimum flow during the 3 months following the spawning month to estimate the percentage of redds dewatered. The use of monthly time-step flow estimates like those obtained from CALSIM II modeling likely underestimates redd dewatering rates because they smoots out short-term flow fluctuations. This potential bias is expected to affect all Project scenarios approximately equally. The means of the redd dewatering estimates under the NAA and Alternatives 1, 2, and 3 are compared using absolute differences rather than relative differences (percent change) because many of the values for percentages of redds dewatered are small. Expressing changes of small values as percent changes may result in large values that may be misleading.

The results of the redd dewatering analysis for American River fall-run (Table 11N-20) show >2% increases in redd dewatering for spawning in November of Above Normal, Dry, and Critically Dry Water Years under Alternative 3. The largest of these increases if 6% for Above Normal Water Years. The other alternatives show few differences >1%. These results indicate that Alternative 3 would increase American River fall-run redd dewatering, but the other alternatives would have little effect. Several factors related to the modeling used for the American River redd dewatering analysis affect the certainty of the results. One of these uncertainty factors is the monthly time-step of the CALSIM II modeling data used in the analysis. As discussed in Appendix 11N, daily flow fluctuations may strongly affect redd dewatering under natural conditions, and the mean monthly flows generated by CALSIM II are likely to underestimate and may otherwise bias estimates of redd dewatering effects. An additional shortcoming of CALSIM II modeling is its limited ability to capture effects of real-time project operations. This includes real-time modifications in operations to minimize potential effects on fish resources, including redd dewatering. The proposed American River Water Agencies Modified Flow Management Standard (Exhibit ARWA-502), which Reclamation has committed to implement (Bureau of Reclamation 2019), includes the following adjustment to their minimum flow requirements (MRR) to protect fall-run redds from dewatering:

> Adjustment for January and February—Protection of Fall-Run Chinook Salmon Redds
> To protect fall-run Chinook salmon redds that have been set during November and December, the January and February MRR shall be adjusted pursuant to this term. During January and February, Permittee shall operate to an MRR that is 70% of the December MRR if that MRR is higher than the MRR that would have indicated under the formulae in Exhibit B based on either: (i) the SRI for January; or (ii) the ARI for February.

The CALSIM II model, because of its monthly time-step, cannot faithfully capture the effects of this protective action. Under real operations, however, Reclamation would operate according to this adjustment and thereby minimize any fall-run redd dewatering. Therefore, while some increase of redd dewatering may result from implementation of Alternatives 1, 2, and 3, real-time Folsom Dam operations will protect against reductions in flow during the fall-run spawning and incubation period that would be great enough to substantially increase levels of redd dewatering. As such, the effects of Alternatives 1, 2, and 3 on redd dewatering are not expected to substantially affect the American River fall-run population.

Habitat Weighted Usable Area

Spawning Habitat Weighted Usable Area

The WUA curve used for fall-run Chinook salmon spawning habitat in the American River (Figure 11K-9) was produced using data obtained from Bratovich et al. (2017), which provides composite spawning WUA tables for fall-run and steelhead downstream of Nimbus Dam.

To evaluate the effects of Alternatives 1, 2, and 3 on fall-run spawning habitat in the American River, fall-run spawning WUA was estimated for CALSIM II flows at Nimbus Dam under the NAA and Alternatives 1, 2, and 3 during the October through December spawning period using the composite fall-run spawning WUA curve (Figure 11K-9).

Differences in fall-run spawning WUA between Alternatives 1, 2, and 3 and the NAA were examined using the grand mean spawning WUA for each month of the spawning period under each water year type and all water year types combined (Appendix 11K, Table 11K-21). The largest difference is a 5% increase in November of Critically Dry Water Years under Alternative 1A. The largest reduction is 3% in November of Below Normal Water Years under Alternative 3. Almost all other differences are <2%. These results indicate that Alternatives 1, 2, and 3 would have only minor effects on fall-run spawning WUA in the American River.

Rearing Habitat Weighted Usable Area

Reliable predictions regarding changes in flow affecting rearing habitat for fall-run Chinook salmon cannot be made because previous curves developed for the American River are old and unreliable and are therefore not applicable (Appendix 11K). In addition, WUA curves from other rivers (e.g., Sacramento River) cannot be used to determine the response of these fish to changes in flow on the American River because relationship between flow and habitat weighted usable area are generally not transferrable from one river to another (Bovee et al. 1998). As such, quantitative conclusions cannot be drawn because no existing credible scientific evidence could have been used to practically estimate the rearing habitat WUA for fall-run Chinook salmon in the American River.

Juvenile Stranding

A formal method for analyzing juvenile stranding analysis has not been developed for the American River. No existing credible scientific evidence could have been used to practically estimate juvenile stranding for fall-run Chinook salmon in the American River.

Delta

Juvenile Through-Delta Survival

The potential for negative effects on juvenile Chinook salmon through-Delta survival, as assessed by the spreadsheet implementation of the through-Delta survival function formulated by Perry et al. (2018; see further discussion for winter-run Chinook salmon), found that during the main period of juvenile fall-run Chinook salmon occurrence in the Delta (i.e., January–May; see Table 11A-6 in Appendix 11A), there was 0%–2% difference in mean through-Delta survival between Alternatives 1, 2, and 3 and the NAA (Table 11-24 in winter-run analysis). Likewise, for juvenile late fall–run Chinook salmon, there was little (Table 11-24 in winter-run analysis) difference between alternatives during the main period of occurrence (i.e., November–May; Table 11A-7 in Appendix 11A).

Juvenile fall-run Chinook salmon from the Mokelumne River watershed emigrate through the Delta via the Mokelumne River, which has a flow-survival relationship and could be affected by Sacramento River flows when the DCC is open. However, as illustrated by implementation of the Perry et al. (2018) flow-survival relationship from the DCC to San Joaquin River via Mokelumne River, during June (the only month with DCC open during the spring juvenile Chinook salmon migration season) there was essentially no difference between Alternatives 1, 2, and 3and the NAA in juvenile Chinook salmon survival in the Mokelumne River (Table 11J-2 in Appendix 11J).

As described for winter-run Chinook salmon, note that the spreadsheet implementation of the Perry et al. (2018) model does not account for the variability in coefficient estimates (Figure 6 of Perry et al. 2018), which would likely give appreciable overlap of estimates in through-Delta survival between the NAA and Alternatives 1, 2, and 3 scenarios, particularly in relation to the relatively small differences between scenarios.

Juvenile Rearing Habitat

As discussed in more detail for winter-run Chinook salmon, available information from analysis of changes in riparian and wetland bench inundation suggests that Alternatives 1, 2, and 3 would have the potential for ~5% less inundation relative to the NAA for juvenile fall-run/late fall–run Chinook salmon.

South Delta Entrainment

There would be little difference in indicators of entrainment risk (south Delta exports and Old and Middle River flows) during winter/spring between the NAA and Alternatives 1, 2, and 3 (Appendix 5B3, Tables 5B3-6-1a through 5B3-6-4c; Figures 5B3-6-1 through 5B3-6-18; and Appendix 5B4, Tables 5B4-1-1a through 5B4-1-4c; Figures 5B4-1-1 through 5B4-1-18), and existing restrictive criteria from the NMFS (2019a) ROC ON LTO BiOp and CDFW (2020) State ITP focused on listed Chinook also would serve to limit entrainment risk for fall-run/late fall–run Chinook under Alternatives 1, 2, and 3. South Delta export of water released from Sites Reservoir would have relatively little overlap with fall-run/late fall–run Chinook salmon. This is illustrated by the results of the salvage-density analysis (Appendix 11Q) for which there generally were minimal differences between the NAA and Alternatives 1, 2, and 3 (Tables 11-45, 11-46, 11-47, and 11-48).

Table 11-45. Entrainment Loss of Juvenile Fall-Run Chinook Salmon At SWP Banks Pumping Plant, Averaged by Water Year Type, Based on the Salvage-Density Method.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	22,110	22,153 (0%)	22,118 (0%)	22,153 (0%)	22,094 (0%)
Above Normal	NA	(-6%)	(-7%)	(-6%)	(-2%)
Below Normal	3,936	3,937 (0%)	3,922 (0%)	3,946 (0%)	3,910 (-1%)
Dry	3,297	3,269 (-1%)	3,198 (-3%)	3,272 (-1%)	3,221 (-2%)
Critically Dry	2,691	2,702 (0%)	2,656 (-1%)	2,698 (0%)	2,647 (-2%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. The analysis was based on historical salvage data during 2009–2019 which did not include any Above Normal Water Years, hence that row of the table is noted as 'NA'; the percentage difference in Above Normal Water Years is based on density data for Wet Water Years applied to above normal modeled exports. Results by water year type and month are provided in Appendix 11Q. Percentage values are rounded; as a result, differences between percentages may not always appear consistent.

Table 11-46. Entrainment Loss of Juvenile Fall-Run Chinook Salmon At CVP Jones Pumping Plant, Averaged by Water Year Type, Based on the Salvage-Density Method.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	9,138	9,142 (0%)	9,148 (0%)	9,141 (0%)	9,128 (0%)
Above Normal	NA	(0%)	(0%)	(0%)	(0%)
Below Normal	2,801	2,807 (0%)	2,862 (2%)	2,807 (0%)	2,882 (3%)
Dry	4,019	4,028 (0%)	4,093 (2%)	4,028 (0%)	4,119 (2%)
Critically Dry	159	159 (0%)	160 (1%)	159 (0%)	160 (1%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. The analysis was based on historical salvage data during 2009–2019 which did not include any Above Normal Water Years, hence that row of the table is noted as 'NA'; the percentage difference in Above Normal Water Years is based on density data for Wet Water Years applied to above normal modeled exports. Results by water year type and month are provided in Appendix 11Q. Percentage values are rounded; as a result, differences between percentages may not always appear consistent.

Table 11-47. Entrainment Loss of Juvenile Late Fall-Run Chinook Salmon At SWP Banks Pumping Plant, Averaged by Water Year Type, Based on the Salvage-Density Method.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	1,677	1,677 (0%)	1,675 (0%)	1,676 (0%)	1,678 (0%)
Above Normal	NA	(0%)	(0%)	(0%)	(-1%)
Below Normal	458	458 (0%)	458 (0%)	458 (0%)	459 (0%)
Dry	1,151	1,089 (-5%)	947 (-18%)	1,094 (-5%)	978 (-15%)
Critically Dry	973	997 (2%)	993 (2%)	988 (2%)	980 (1%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. The analysis was based on historical salvage data during 2009–2019 which did not include any Above Normal Water Years, hence that row of the table is noted as 'NA'; the percentage difference in Above Normal Water Years is based on density data for

Wet Water Years applied to above normal modeled exports. Results by water year type and month are provided in Appendix 11Q. Percentage values are rounded; as a result, differences between percentages may not always appear consistent.

Table 11-48. Entrainment Loss of Juvenile Late Fall-Run Chinook Salmon At CVP Jones Pumping Plant, Averaged by Water Year Type, Based on the Salvage-Density Method.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	297	297 (0%)	297 (0%)	297 (0%)	297 (0%)
Above Normal	NA	(0%)	(0%)	(0%)	(0%)
Below Normal	75	75 (0%)	75 (-1%)	75 (0%)	75 (0%)
Dry	94	95 (1%)	89 (-5%)	95 (1%)	89 (-5%)
Critically Dry	26	27 (1%)	27 (1%)	26 (0%)	28 (5%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. The analysis was based on historical salvage data during 2009–2019 which did not include any Above Normal Water Years, hence that row of the table is noted as 'NA'; the percentage difference in Above Normal Water Years is based on density data for Wet Water Years applied to above normal modeled exports. Results by water year type and month are provided in Appendix 11Q. Percentage values are rounded; as a result, differences between percentages may not always appear consistent.

Adult Straying

Potential effects related to straying of adult Mokelumne River fall-run Chinook salmon to the Sacramento River when the DCC is open during October and November (Setka 2018) were also evaluated. Greater numbers of days with the DCC open have the potential to increase straying risk. Operations criteria would be the same for the NAA and Alternatives 1, 2, and 3. The CALSIM modeling results showed that Alternatives 1, 2, and 3 had a similar mean number of days of DCC open compared to the NAA (Table 11-49). The modeling results do not account for DCC closure in association with Mokelumne River pulse flows, as required under the ROC ON LTO proposed action (Bureau of Reclamation 2019:4-45), and which is part of criteria under which the NAA and the Project are assumed to operate, with implementation as illustrated in October 2021 (Salmon Monitoring Team 2021). Overall, the risk of adult Mokelumne River fall-run Chinook salmon straying would be similar under the NAA and Alternatives 1, 2, and 3.

Table 11-49. Mean Number of Days with Delta Cross Channel Open in October and November.

Month	Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
October	Wet	15	15	15	15	15
October	Above Normal	18	18	19	18	18
October	Below Normal	27	27	27	27	24
October	Dry	22	20	20	21	23
October	Critically Dry	30	30	30	30	30
November	Wet	13	13	13	13	13
November	Above Normal	13	13	13	13	13
November	Below Normal	11	11	12	11	11
November	Dry	13	13	13	13	14

Month	Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
November	Critically Dry	19	19	19	19	19

<u>*CEQA Significance Determination for Alternatives 1, 2, and 3*</u>

The preceding subsections of this impact discussion provide the detailed information used for this CEQA (and NEPA) determination. This section provides a summary of this information.

In-Delta and upstream operational impacts of Alternatives 1, 2, and 3 on fall-run/late fall–run Chinook salmon generally would be limited. Fall-run/late fall–run Chinook salmon would have the potential for similar types of near-field effects to those previously discussed for winter-run and spring-run Chinook salmon (i.e., entrainment, impingement and screen contact, predation, stranding behind screens, and attraction to reservoir discharge). Around 30%–40% of fall-run/late fall–run Chinook salmon may pass the intakes. As discussed for winter-run and spring-run Chinook salmon, few fall-run/late fall–run Chinook juveniles would have the potential for entrainment given their size. As described for winter-run Chinook salmon, adult fall-run/late fall–run Chinook salmon could be attracted to the flow from the Sacramento River discharge under Alternative 2. However, the design of the apron and weir would exclude anadromous fish and eliminate stranding risk.

Related to temperature effects, observations of exceedance plots and differences in modeled mean monthly temperatures by water year type at all locations analyzed in the Sacramento, Feather, and American Rivers indicate that Alternatives 1, 2, and 3 and the NAA would be predominantly similar among alternatives during the period of presence of each life stage of fall-run and late fall–run Chinook salmon.

Results of the analysis of exceedance above water temperature index values for fall-run and late fall–run Chinook salmon indicate that there would be one month and water year type (July of Above Normal Water Years) at one location (Sacramento River at Hamilton City) for one alternative (Alternative 3) in which both: (1) the percent of days that exceeded the index values was more than 5% greater under the alternative than under the NAA; and (2) the exceedance per day was more than 0.5°F greater under the alternative than under the NAA. For all other alternatives at the remainder of locations and for all life stages of fall-run and late fall–run Chinook salmon, there were no month and water year type combinations in which both criteria being met.

Based on available studies of predation effects at water intakes, the predation effects in the vicinity of the Red Bluff and Hamilton City intakes for Alternatives 1, 2, and 3 would be limited. The extent of in-water structure at the intakes would be the same under the NAA and Alternatives 1, 2, and 3. Although overtopping during high flows can occur at the Red Bluff and Hamilton City intakes, leading to potential stranding of juvenile fall-run/late fall–run Chinook salmon, these are relatively infrequent events (e.g., approximately once per 100 years at Red Bluff) that occur under the NAA and would not be changed by Alternatives 1, 2, and 3.

For the Sacramento River, the probability of redd scour and entombment was estimated for fall-run and late fall–run by computing the percentage of days with flows exceeding 40,000 cfs in the USRDOM 82-year daily flow record at four locations between Keswick Dam and the RBPP during the months of fall-run and late fall–run spawning and incubation. The results indicate that Alternatives 1, 2, and 3 would have no adverse effect on redd scour and entombment for fall-run and late fall–run in the Sacramento River at any of the four locations.

Frequency of scouring flows was not estimated for the Feather River because information on minimum flows required to mobilize sediments is not available for the Feather River. However, Feather River flows during the Wet and Above Normal Water Years, when scouring flows would be most likely to occur during the months of fall-run spawning and egg incubation (October through February), are generally similar between the NAA and Alternatives 1, 2, and 3 (Chapter 5). Therefore, no substantial differences on the frequency of scouring flows are expected between the NAA and Alternatives 1, 2, and 3. For redd scour or entombment on the American River, results indicate that Alternatives 1, 2, and 3 would have no effect on redd scour or entombment for fall-run.

Reductions in mean spawning WUA for fall-run are larger and more frequent under Alternatives 1, 2, and 3 than increases, especially for Alternative 3. The largest differences are 7% reductions under Alternative 3 for November of Critically Dry Water Years and October of Wet Water Years. The largest increases are 4% under Alternatives 1A and 2 in September of Above Normal Water Years. The spawning distribution of fall-run is more evenly distributed over the four river segments than that of the other runs, and therefore differences resulting from Alternatives 1, 2, and 3 would potentially affect fall-run spawning habitat in any of the segments. These results indicate that Alternative 3 would result in frequent reductions, ranging up to 7%, in fall-run spawning habitat WUA. However, these reductions are largely limited to two of the four river segments in which most fall-run spawning occurs, so the effects are not expected to substantially affect overall fall-run spawning habitat availability. Alternatives 1A, 1B, and 2 would generally have little effect on fall-run spawning.

Mean late fall–run Chinook salmon spawning WUA under Alternatives 1, 2, and 3 generally differs little from that under the NAA. The largest differences are 6% reductions under Alternative 3 in Segment 6 for December of Wet and Dry Water Years. The largest increases are about 2% in Segment 6 for December of Above Normal Water Years under Alternatives 1, 2, and 3. Most differences in all river segments and all three alternatives are <3%. The results indicate that Alternatives 1A, 1B, and 2 would have little effect on late fall–run spawning WUA and Alternative 3 would have larger adverse effects, but the effects are not expected to substantially affect late fall–run spawning habitat availability.

The results for fall-run Chinook salmon rearing WUA show small reductions in fry rearing WUA and substantial increases in juvenile rearing habitat WUA. Late fall–run Chinook salmon mean juvenile rearing WUA differs between Alternatives 1A, 1B, and 2 and the NAA by <2% for most months and water year types in Segments 6 and 5. Many more >2% differences occur for Alternative 3 in these river segments and most of them result from increases in rearing WUA. The results for juvenile rearing WUA indicate largely minor differences between Alternatives 1, 2, and 3 and the NAA for Segments 6 and 5, but substantial differences for Segment 4. This segment generally has increased juvenile rearing habitat WUA under Alternatives 1B and 3 during late spring and summer and some reduction in habitat during August of Critically Dry Water Years. Alternative 3 has the most frequent and largest differences and, on balance, it is expected to benefit late fall–run rearing habitat availability. Overall, the results indicate that

Alternatives 1, 2, and 3 would have little effect on late fall–run fry rearing habitat availability, whereas Alternative 3 would provide a net benefit to late fall-run juvenile rearing habitat availability.

As described in Chapter 2, the Project's diversion criteria include restrictions to maintain Bend Bridge and Wilkins Slough flows, which should avoid changes to Yolo Bypass spill frequency and duration under Alternatives 1, 2, and 3 relative to the NAA. Thus, Alternatives 1, 2, and 3 generally would have limited potential for negative effects on Yolo Bypass floodplain inundation and access for fall-run/late fall–run Chinook salmon. The results of the frequency analysis of inundation of events for the Yolo Bypass generally show only minor differences between Alternatives 1, 2, and 3 and the NAA.

The Project would operate to avoid effects on the Big Notch's ability to achieve the same level of performance for salmonids in the Sacramento River as it would absent the Project. However, the Adaptive Management Plan for the Project recognized there is uncertainty about the performance of the Big Notch and the effects of the Project on it. Monitoring will be conducted, in cooperation with the State, to determine whether there is an effect and, if so, what the magnitude of that effect would be on entrainment of juvenile salmon into the Yolo Bypass. If there is an adverse effect, a science-based adaptive management approach will be employed to determine how to adjust diversions 158 RMs upstream of the Big Notch to maintain its efficiency for entraining juvenile salmon into the Yolo Bypass.

The Big Notch Project also is intended to reduce migratory delays and loss at the Fremont Weir of adult salmon, steelhead, and sturgeon that are migrating upstream through the Yolo Bypass. As with juvenile passage, the Project would operate to avoid affecting adult passage at Fremont Weir. The number of days meeting adult Chinook salmon passage criteria at Fremont Weir generally would be similar with or without the Project (Table 11-21). This too will be evaluated and adaptively managed as required.

The Sutter Bypass, when inundated, provides important juvenile rearing habitat for Chinook salmon and steelhead, as discussed for the Yolo Bypass. For the Sutter Bypass the modeling results indicate that Alternatives 1, 2, and 3 would produce little change in suitable habitat compared to the NAA. Changes in Sacramento River flow and flow entering Sutter Bypass as a result of the Project have the potential to change the number of days meeting adult Chinook salmon passage criteria at the three Sutter Bypass weirs (Moulton, Tisdale, and Colusa). The results of this analysis indicated that there would be no days meeting adult salmon passage criteria during 2009–2018 at Moulton or Tisdale Weirs under the NAA or Alternatives 1, 2, and 3. For Colusa Weir, the number of days meeting fish passage criteria generally would be similar with or without the Project.

As described in Chapter 2, Alternatives 1, 2, and 3 include pulse flow protection measures to be applied to precipitation-generated pulse flow events. There is also a 10,700-cfs Wilkins Slough bypass flow threshold from October 1 through June 14. Adaptive management would be utilized to assess and if necessary, refine the diversion criteria based on biological monitoring (see Appendix 2D, Section 2D.6). Sites Reservoir diversion criteria effectively limit diversions during the historical periods of fish movement as reflected in Red Bluff rotary screw trap data, and application of the flow-threshold criteria from Michel et al. (2021) suggests that flow-survival effects on juvenile Chinook salmon (including fall-run/late fall–run Chinook salmon) would be greatly limited by the Project's diversion criteria because there is essentially no difference in predicted survival between Alternatives 1, 2, and 3 and NAA.

Based on the analyses provided, operation of Alternative 1, 2, or 3 would not have a substantial adverse effect, either directly or through habitat modifications, on fall-run/late fall–run Chinook salmon. In consideration of the analyses within the preceding subsections of this impact discussion and summarized above, operation impacts of Alternative 1, 2, or 3 would be less than significant. No mitigation is necessary.

NEPA Conclusion for Alternatives 1, 2, and 3

Operation effects on Chinook salmon would be the same as described above for CEQA. In-Delta and upstream operational effects of Alternatives 1, 2, and 3 on fall-run/late fall–run Chinook salmon generally would be limited. Related to temperature effects, observations of exceedance plots and differences in modeled mean monthly temperatures by water year type indicate that Alternatives 1, 2, and 3 and the NAA would be predominantly similar among alternatives during the period of presence of each life stage of fall-run/late fall–run Chinook salmon. The water temperature indicator value exceedance analysis confirms this similarity among alternatives.

As described in Chapter 2, the Project's diversion criteria include restrictions to maintain Bend Bridge and Wilkins Slough flows, which have the effect of limiting changes to Yolo Bypass spill frequency and duration under Alternatives 1, 2, and 3 relative to the NAA. Thus, Alternatives 1, 2, and 3 generally would have limited potential for negative effects on Yolo Bypass floodplain inundation and access for spring-run Chinook salmon. Also as described in Chapter 2, Alternatives 1, 2, and 3 include pulse flow protection measures to be applied to precipitation-generated pulse flow events. There is also a 10,700-cfs Wilkins Slough bypass flow threshold from October 1 through June 14, corresponding with the threshold identified by Michel et al. (2021). Operation of Alternative 1, 2, or 3 would have no adverse effect on fall-run/late fall–run Chinook salmon.

Impact FISH-5: Operations Effects on Central Valley Steelhead

Alternatives 1, 2, and 3

Effects of Alternatives 1, 2, and 3

Potential exposure of steelhead to the effects of Alternatives 1, 2, and 3 is dependent on the species' spatiotemporal distribution. As described for winter-run Chinook salmon, several sources of information provide important context, as documented in the SacPAS summary in Appendix 11A1. The main patterns include:

- RBDD rotary screw traps (Appendix 11A1, Figure 11A1-37): Passage begins in early January to late June and finishes in December. The first half (50%) passes by late May to mid-August. The main portion (90%) begins to pass in late February to around the first of August and finishes passing from mid-August to early October.
- Tisdale Weir rotary screw traps (Appendix 11A1, Figure 11A1-38): Passage begins in early January to late February and finishes from mid-February to the end of December.

The first half (50%) passes by early January to early April. The main portion (90%) begins to pass in early January to mid-March and finishes passing from mid-February to mid-December.

- Knights Landing rotary screw traps (Appendix 11A1, Figure 11A1-39): Passage begins in early January to mid-February and finishes from early April to late December. The first half (50%) passes by mid-February to late April. The main portion (90%) begins to pass from mid-January to mid-March and finishes passing from mid-February to late December.
- Sacramento beach seines (Appendix 11A1, Figure 11A1-40): Occurrence begins from early January to the first of March and finishes from mid-February to late December. The first half (50%) occurs by early January to early June. The main portion (90%) begins to occur in early January to the first of March and finishes from early February to mid-December.
- Sacramento trawls (Sherwood Harbor) (Appendix 11A1, Figure 11A1-41): Occurrence begins around the first of January to early February and finishes from early March to mid-December (first to last). The first half (50%) occurs by late January to mid-April. The main portion (90%) begins to occur in mid-January to early January and finishes from mid-February to the end of December.
- Chipps Island trawls (Appendix 11A1, Figure 11A1-42): Occurrence begins in early January to mid-February and finishes from mid-May to late December. The first half (50%) occurs by early March to mid-April. The main portion (90%) begins to occur in mid-January to late February and finishes from early March to mid-May.
- Salvage (unclipped) (Appendix 11A1, Figure 11A1-43): Salvage begins in late August to the first of February and finishes from early May to around the first of August. The first half (50%) is salvaged by the first of February to mid-May. The main portion (90%) begins to be salvaged by the first of January to mid-March and finishes from early April to mid-June.
- Salvage (clipped) (Appendix 11A1, Figure 11A1-44): Salvage begins in mid-December to mid-February and finishes from mid-February to mid-July. The first half (50%) of salvage occurs by late January to late March. The main portion (90%) begins to be salvaged from early January to early March and finishes by early February to mid-June.

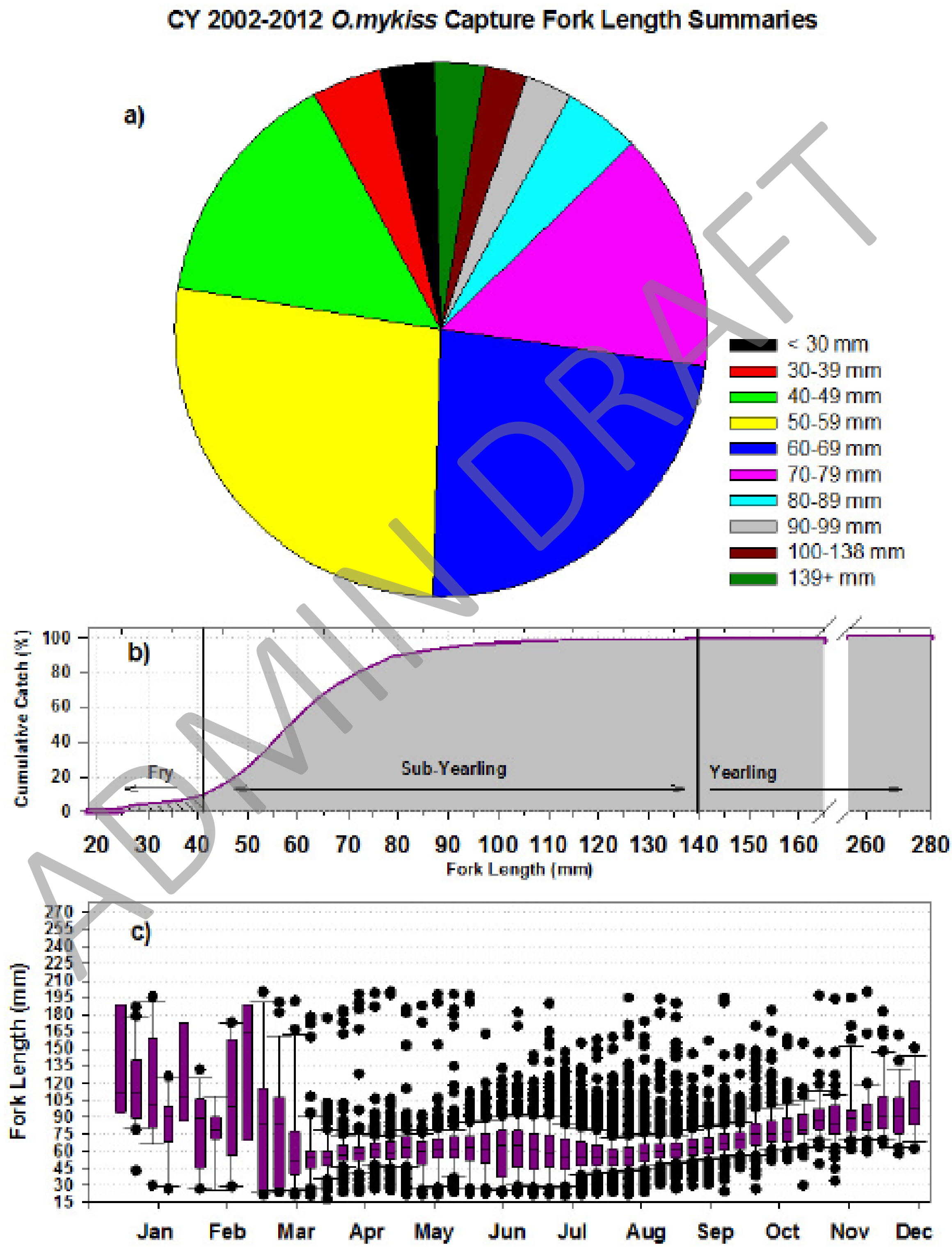
Sacramento River

Near-Field Effects

Steelhead would have the potential for similar types of near-field effects to those previously discussed for winter-run Chinook salmon (i.e., entrainment; impingement and screen contact; predation; stranding behind screens; and attraction to reservoir discharge). Only steelhead originating upstream of the Red Bluff and Hamilton City intakes would have the potential for near-field effects of the intakes as they pass them as juveniles or adults. Little is known of population sizes by tributary, although there are a number of tributaries upstream of the intakes that include steelhead (National Marine Fisheries Service 2014:48). As generally discussed in Appendix 11A and described by NMFS (2016:20), some of the main steelhead populations for which adult abundance has been estimated occur upstream of the Red Bluff and Hamilton City intakes (i.e., Coleman NFH and Clear Creek), whereas other key populations occur downstream of the intakes (American River, Nimbus Hatchery, Feather River Hatchery, and Mokelumne River Hatchery); only the former would have the potential to be exposed to near-field effects.

Rotary screw trap sampling at Red Bluff found around 3.5% of steelhead/rainbow trout sampled were less than 30-mm FL, with the smallest individuals occurring during March–July (Figure 11-22). The earliest occurring individuals would have the potential to be entrained in greater numbers under Alternatives 1, 2, and 3 than under the NAA (Tables 11-7 and 11-8), but entrainment potential would be limited as the species tends to undergo downstream migration as larger juveniles (yearlings or older) (Appendix 11A). This larger size would tend to considerably limit the potential for negative near-field effects on juveniles based on greater swimming ability than juvenile Chinook salmon; as noted for Chinook salmon, the Red Bluff and Hamilton City fish screens are designed to protective standards for Chinook salmon fry.

As with fall-run Chinook salmon in particular, the timing of migrating adult steelhead returning to tributaries upstream of the Sacramento River reservoir release location is such that relatively high numbers of individuals would pass this area during appreciable reservoir release flows to the river, for example during September (Table 5-19 in Chapter 5). As described for winter-run Chinook salmon, adult steelhead could be attracted to the reservoir discharge flow to the Sacramento River under Alternative 2 and leap out of the river, but the apron and weir designed to exclude anadromous fish would eliminate stranding risk.



Source: Poytress et al. 2014:86. Note: CY = calendar year.

Figure 11-22. Steelhead Fork Length (a) Capture Proportions, (b) Cumulative Capture Size Curve, and (c) Average Weekly Median Boxplots, As Sampled at Red Bluff Diversion Dam Rotary Screw Traps, July 2002–June 2013.

Far-Field Effects

As described previously in Impact FISH-2, there are likely multiple opportunities that would arise in real-time operations to coordinate exchanges between Sites Reservoir and Shasta Lake that could benefit anadromous fish. These are described in the Project description and reflected in the modeling for this document to some extent. However, due to the unique conditions in each year, additional opportunities for exchanges and coordination of real-time operations exist beyond those modeled for this document. The Authority and Reclamation intend to work together to better reflect the exchanges in the modeling with the goal of substantiating the Project's benefits to anadromous fish.

Temperature Effects

Operation of Sites Reservoir has the potential to change water temperatures in the Sacramento River that could affect the life stages of steelhead present. As described in Appendix 11B, the two methods used to analyze temperature-related effects on steelhead in the Sacramento River were: (1) Physical Model Output Characterization; and (2) Water Temperature Index Value/Range Exceedance Analysis. More details on these methods are provided in Appendix 11B.

The Authority and Reclamation evaluated water temperature model outputs during the period of presence and in the locations of each life stage of steelhead in the Sacramento River upstream of the Delta (see Appendix 11B, Table 11B-2 for timing and locations). Visual observation of exceedance plots and differences in modeled mean monthly temperatures by water year type between Alternatives 1, 2, and 3 and the NAA in the Sacramento River below Keswick, at Balls Ferry, at Bend Bridge, below RBDD[21], and at Butte City indicates that water temperatures would be predominantly similar among alternatives during the period of presence of each life stage of steelhead (Appendix 6C, Tables 6C-5-1a to 6C-5-4c, Tables 6C-7-1a to 6C-7-4c, Tables 6C-9-1a to 6C-9-4c, Tables 6C-10-1a to 6C-10-4c, Tables 6C-12-1a to 6C-12-4c; Figures 6C-5-1 to 6C-5-18, Figures 6C-7-1 to 6C-7-18, Figures 6C-9-1 to 6C-9-18, Figures 6C-10-1 to 6C-10-18, Figures 6C-12-1 to 6C-12-18). At all locations, mean monthly water temperatures for all months within all water year types under Alternatives 1A and 1B were within 0.6°F of the NAA with the exception of August of critical years at Butte City, in which mean monthly water temperature would be 0.8°F lower under the NAA than under Alternatives 1A and 1B. Water temperature modeling results for Alternatives 2 and 3 were similar to those of Alternative 1 at all locations. These results suggest that temperature-related effects on steelhead in the Sacramento River would be negligible.

Results of the analysis of exceedance above water temperature index values for steelhead in the Sacramento River from Appendix 11B, Table 11B-2 are presented in Appendix 11D, Table 11D-112 through Table 11D-161 and summarized in Table 11-50. For each life stage and at all locations evaluated, there were no month and water year type combinations in which both: (1) the percent of days that exceeded the index values was more than 5% greater under the alternative than under the NAA, and (2) the exceedance per day was more than 0.5°F greater under the alternative than under the NAA. Results of the exceedance analysis for Alternative 2

[21] The RBDD, which was decommissioned in 2013, and the RBPP are co-located, and the names may be used interchangeably when referring to geographic locations

are similar to Alternative 1 with no month and water year type combinations in which both criteria were met for any life stage at all locations. Results of the exceedance analysis for Alternative 3 are similar to Alternative 1 with one exception. There was one month and water year type combination (September of Critically Dry Water Years) in which both criteria met in a favorable way for the 63°F mean daily juvenile rearing index value at Balls Ferry. Because these effects occurred in only one month and water year type combination, it is not expected to be persistent enough to affect steelhead at a population level.

Table 11-50. Number of Month and Water Year Type Combinations that Satisfy Both Criteria for Being Biologically Meaningful in the Water Temperature Index Value Analysis, Steelhead, Sacramento River[1,2,3]

Location	Spawning and Egg Incubation				Kelt Emigration				Juvenile Rearing				Smolt Emigration				Smoltification				Adult Immigration				Adult Holding			
	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3
Below Keswick	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Below Clear Creek	0	0	0	0	NA[4]	NA	NA	NA	0	0	0	0	0	0	0	0	0	0	0	0	NA	NA	NA	NA	NA	NA	NA	NA
Balls Ferry	0	0	0	0	NA	NA	NA	NA	0	0	0	1 favorable (63°F mean daily, September, Critically Dry Water Years)	0	0	0	0	0	0	0	0	NA	NA	NA	NA	0	0	0	0
Bend Bridge	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	NA	NA	NA	NA
Below Red Bluff Diversion Dam	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

[1] Biologically Meaningful Criteria include: (1) the difference in frequency of exceedance between the NAA and the alternative was greater than 5%, and (2) the difference in average daily exceedance between the NAA and the alternative was greater than 0.5°F.

[2] Index values for each life stage are located in Appendix 11B, Table 11B-2.

[3] Full results presented in Appendix 11D, Table 11D-112 through Table 11D-161.

[4] NA = Not analyzed

Overall, effects of Alternatives 1, 2, and 3 on water temperature-related effects on steelhead in the Sacramento River are expected to be biologically inconsequential due to the low frequency and small magnitude of differences between each alternative and the NAA.

Flow-Related Physical Habitat Conditions

Redd Scour Entombment

Loss of redds to scouring and entombment occurs when flows are high enough to mobilize sediments, destroying redds and their incubating eggs and alevins, or entombing the redds when sediments are redeposited. A flow of 40,000 cfs was selected as the scour flow threshold for the Sacramento River based on estimates in the literature (Table 11N-10).

The probability of redd scour and entombment was estimated for steelhead by computing the percentage of days with flows exceeding 40,000 cfs in the USRDOM 82-year daily flow record (29,952 days in total) at four locations between Keswick Dam and the RBPP during the months of steelhead spawning and incubation. Because the steelhead spawning and incubation period includes the wettest months of the year, steelhead redds have the highest probabilities of experiencing scouring flows (Table 11N-22 through Table 11N-25). Alternatives 1, 2, and 3 have little effect on the frequency of redd scouring and entombment.

Redd Dewatering

The percentage of redds in the Sacramento River lost to dewatering was estimated using tables in USFWS (2006) that relate spawning and dewatering flows to percent reductions in species-specific spawning habitat WUA (Appendix 11N). USRDOM flow data, which has a daily time-step, are available for three locations in this river section: Keswick Dam (RM 302), the Sacramento River at Clear Creek (RM 289), and the Sacramento River at Battle Creek (RM 271). A single relationship for flows was developed for the entire river section, but the flows used to estimate redd dewatering in the current analysis were those that best matched the longitudinal distribution of the redds of the different salmon runs in the river as estimated from aerial redd surveys conducted by CDFW from 2003 through 2019. The spawning distribution of steelhead is uncertain, but most spawning is assumed to occur between Keswick Dam and Battle Creek where most salmon spawning occurs and where temperature conditions are most suitable. Flows for the Sacramento River at Clear Creek, which is near the center of this reach, were used to analyze steelhead redd dewatering.

Results are presented using the grand mean percentages of redds dewatered for each month of spawning, November through February, and each water year type and all water year types combined. The expected time for incubation of eggs and alevins is 3 months (Appendix 11N). Because changes in Project-related flow any time during this period can affect redd dewatering, the complete spawning and egg/alevin incubation periods (November–February through February–May) are provided in the results (Table 11N-17). The means of the redd dewatering estimates under the NAA and Alternatives 1, 2, and 3 are compared using absolute differences rather than relative differences (percent change) because many of the values for percentages of redds dewatered are small. Expressing changes of small values as percent changes may result in large values that may be misleading.

The results for steelhead redd dewatering show little effect from Alternatives 1A, 1B, and 2, with few differences from the NAA of more than 1% (Table 11N-17). Alternative 3 has several larger increases in redd dewatering, including 2%–3% increases for spawning in December of Wet and Dry Water Years and January and February of Above Normal Water Years. Most other differences under Alternative 3 are small, but almost all represent increases in redd dewatering. These results indicate that Alternatives 1A, 1B, and 2 would have little effect on steelhead redd dewatering, but Alternative 3 would result in somewhat greater redd dewatering than that under the NAA.

Spawning Habitat Weighted Usable Area

The suitability of physical habitat for salmonid spawning is largely a function of the availability of clean, coarse gravel for constructing redds, favorable depths, and suitable flow velocities. Instream flow potentially affects all these habitat characteristics and often affects the availability of suitable habitat. Habitat suitability for spawning was analyzed using WUA curves developed by USFWS and others from results of field studies and hydraulic modeling (Appendix 11K).

Spawning habitat WUA for steelhead in the Sacramento River was determined USFWS (2003a, 2006) in the same manner that it was determined for winter-run, fall-run and late fall–run, except that HSC previously determined for steelhead in the American River (U.S. Fish and Wildlife Service 2003b) were used in developing the Sacramento River steelhead WUA curves (Appendix 11K). The spawning distribution of steelhead is uncertain, but most spawning is assumed to occur in the upper three segments (Segments 6, 5, and 4), where most salmon spawning occurs and where temperature conditions are most suitable.

To evaluate the effects of Alternatives 1, 2, and 3 on steelhead spawning habitat, steelhead spawning WUA was estimated for CALSIM II flows during the November through February spawning period under Alternatives 1, 2, and 3 and the NAA in the same three segments of the Sacramento River that were used for winter-run, spring-run and late fall–run.

There are a few notable differences in steelhead mean spawning WUA between Alternative 1, 2, and 3 and the NAA in Segments 6 and 5 (Table 11K-15 and Table 11K-17). The largest difference is a 7% reduction under Alternative 3 in Segment 6 for February of Above Normal Water Years (Table 11K-15). Other reductions ranging from 5% to 6% occur in Segment 6 during February of Above Normal Water Years under Alternative 1B, in Segment 6 during December of Wet Water Years under Alternative 3, and in Segment 5 in February of Above Normal Water Years under Alternative 3 (Table 11K-15 and Table 11K-16). Most differences in all river segments and all three alternatives are <3%. These results indicate that Alternatives 1A, 1B, and 2 would have little effect on steelhead spawning WUA. Alternative 3 has more frequent larger negative effects, but these are not expected to substantially affect steelhead spawning habitat availability.

Rearing Habitat Weighted Usable Area

The suitability of physical habitat for salmonid rearing is largely a function of water depth, flow velocity, and the availability and type of cover. Habitat suitability for rearing was analyzed using WUA curves developed by USFWS and others from results of field studies and hydraulic modeling (Appendix 11K).

Rearing habitat WUA for steelhead was not estimated directly by U.S. Fish and Wildlife Service (2005b), but was modeled using the rearing WUA curves obtained for late fall–run Chinook salmon, in the same three Sacramento River segments that were used for the winter-run, fall-run and late fall–run spawning and rearing habitat WUA studies (U.S. Fish and Wildlife Service 2003a, 2005b). The rearing WUA curves for late fall–run Chinook salmon were used because the fry rearing period of late fall–run is similar to that of steelhead in the Sacramento River, and because this substitution follows previous practice (Section 11K.2, *Methods*). The validity of using the late fall–run Chinook salmon WUA curves to characterize Central Valley steelhead rearing habitat is uncertain. For this analysis, fry are defined as fish less than 60 mm, and juveniles are young fish (young-of-year) greater than 60 mm.

To estimate changes in rearing WUA that would result from Alternatives 1, 2, and 3, the late fall–run fry and juvenile WUA curves developed for each of the three river segments was used with mean monthly CALSIM II flow estimates for corresponding river segments under Alternatives 1, 2, and 3 and the NAA during the rearing periods for steelhead fry (February through May) and juveniles (year-round) in the Sacramento River (Table 11A-8 in Appendix 11A).

Few of the differences between Alternatives 1, 2, and 3 and the NAA means for steelhead fry rearing WUA in all river segments are >3% (Table 11K-47 through Table 11K-49). The largest difference is an 8% increase in May of Critically Dry Water Years in Segment 6 under Alternative 3 (Table 11K-47). The largest reduction is 3% and occurs in March of Dry Water Years under Alternative 3 in Segment 5 (Table 11K-48). These results indicate that Alternatives 1, 2, and 3 would have a little effect on steelhead fry rearing habitat availability.

Steelhead mean juvenile rearing WUA differs between Alternatives 1, 2, and 3 and the NAA by <2% for most months and water year types under Alternatives 1A, 1B, and 2 in Segments 6 and 5 (Table 11K-50 and Table 11K-51). Many more >2% differences occur for Alternative 3 in these river segments, including a quarter of the means for Segment 5, and the majority of them result from increases in rearing WUA. The largest of these increases is 7% in August of Above Normal Water Years in Segment 6 (Table 11K-50). Segment 4 has many more >2% differences in means between Alternatives 1, 2, and 3 and the NAA than Segments 6 and 5 (Table 11K-52). The largest differences include 15% increases in rearing WUA in June of Above Normal and Below Normal Water Years under Alternative 3 and 8% to 9% reductions in rearing WUA in August of Critically Dry Water years under all the alternatives. The results for juvenile rearing WUA indicate largely minor differences between Alternatives 1, 2, and 3 and the NAA for Segments 6 and 5, but substantial differences for Segment 4. This segment generally has increased juvenile steelhead rearing habitat WUA under Alternatives 1B and 3 during late spring and summer and some reduction in habitat during August of Critically Dry Water Years. Alternative 3 has the most and largest differences and, on balance, it is expected to benefit steelhead rearing habitat availability. Alternatives 1A, 1B, and 2 are expected to have little effect.

Juvenile Stranding

The juvenile stranding estimation procedure, which is identical for all the salmonids, is described in the *Juvenile Stranding* section of Impact FISH-2.

The principal period of stranding vulnerability for steelhead is for fry cohorts emerging in February through May. The steelhead fry cohorts emerging in April and May are likely to be affected by the large spring and summer changes in stranding under Alternatives 1, 2, and 3. Although especially large (>20%) increases and reductions in juvenile stranding are predicted for this period in all three river reaches, the large reductions are more frequent than the increases (Table 11N-28 through Table 11N-30). However, the largest differences in all three reaches are increases in stranding during May of Below Normal Water Years under Alternative 3. The summary stranding results for steelhead fry (Table 11N-31 through Table 11N-33) show increases (up to 3.3%) in stranding under Alternatives 1B and 3 in the Keswick reach and under Alternative 3 in the Clear Creek reach. The results also show reductions under Alternatives 1, 2, and 3 in the Battle Creek reach (Table 11N-33). On balance, Alternatives 1A and 1B are expected to reduce juvenile stranding of steelhead and Alternative 3 is expected to moderately increase stranding. These effects are not expected to be substantial.

Floodplain Inundation and Access

As described in Chapter 2 and as discussed in the section above for winter-run Chinook salmon, the Project's diversion criteria include restrictions to maintain Bend Bridge and Wilkins Slough flows, which have the effect of limiting changes to Yolo Bypass spill frequency and duration under Alternatives 1, 2, and 3 relative to the NAA. In particular, these criteria avoid impacts on Reclamation's ability to implement its obligations in the 2019 NMFS ROC ON LTO BiOp to implement the Yolo Bypass Restoration Salmonid Habitat Restoration and Fish Passage Implementation Plan and provide more than 17,000 acres of inundation in the Yolo Bypass from December to April (National Marine Fisheries Service 2019a). As such, Alternatives 1, 2, and 3 would have limited potential for negative effects on Yolo Bypass floodplain inundation and access for steelhead. This was confirmed with the modeling summarized in the winter-run Chinook salmon section (Appendices 11M1 and 11M2). As noted for winter-run Chinook salmon, there is the potential for a somewhat lower proportion of juvenile steelhead entering the Sutter Bypass from the Sacramento River under Alternatives 1, 2, and 3 relative to the NAA.

As discussed in Impact FISH-2 and in Appendix 11M, Alternatives 1, 2, and 3 are not expected to substantially affect suitable inundated floodplain habitat on the Yolo and Sutter Bypasses or suitable inundated side-channel habitat on the Sacramento River for rearing juvenile salmonids, including steelhead. This conclusion is based on the results of habitat modeling that showed little difference in suitable floodplain habitat acreage between the NAA and Alternatives 1, 2, and 3 for the Sutter Bypass (Table 11M-4 and Figure 11M-8), an absence of large differences in acreage of suitable side-channel habitat in the Sacramento River (Table 11M-5 through Table 11M-7 and Figure 11M-9), and a reduction of less than 100 acres (1.8%) in the total suitable habitat acreage on the Yolo Bypass for the November through May period (Table 11-14 and Table 11M-3), despite relatively large reductions for a few month and water year combinations (Table 11-13 and Table 11M-1).

Adult Upstream Passage at Fremont Weir

Adult Chinook salmon migrate upstream during Fremont Weir overtopping events (it is reasonable to assume the same for adult steelhead), as well as during lower flow conditions (National Marine Fisheries Service 2019b:26). Sommer et al. (2014) did not find adult Chinook salmon catch in a fyke trap in the Toe Drain was associated with Yolo Bypass flow events, noting this may have been because most of the Chinook salmon were fall-run, a race known to migrate upstream relatively early before winter and spring flow events. Recent completion of the Wallace Weir fish rescue facility and fish passage facilities at Fremont Weir blocked access to the CBD and improved adult fish passage allowing passage at a variety of flows into the Yolo Bypass, including across the range anticipated under the NAA and Alternatives 1, 2, and 3. As such, the minor differences in flow entering Yolo Bypass at Fremont Weir under Alternatives 1, 2, and 3 relative to the NAA (see discussion above related to inundation and juvenile access) would not result in major differences in adult upstream passage under Alternatives 1, 2, and 3 compared to the NAA. As described for winter-run Chinook salmon in Impact FISH-2, the number of days of adult passage at Fremont Weir was assessed using the number of days meeting adult passage criteria[22], based on data for water years 2009–2018 from the Daily Divertible Flow and Storage Tool Results (see Appendix 11P1). The results of this analysis indicated that the number of days meeting fish passage criteria at Fremont Weir generally would be similar with or without the Project (Table 11-21).

Adult Upstream Passage at Sutter Bypass Weirs

As described for winter-run Chinook salmon, changes in Sacramento River flow and flow entering Sutter Bypass as a result of the Project have the potential to change the number of days meeting adult Chinook salmon passage criteria at the three Sutter Bypass weirs (Moulton, Tisdale, and Colusa). Although the analysis was specific to Chinook salmon criteria, this is assumed to be a reasonable representation of potential effects for steelhead as well. The results of the analysis indicated that there would be no days meeting adult salmon passage criteria during 2009–2018 at Moulton or Tisdale Weirs. For Colusa Weir, the number of days meeting fish passage criteria generally would be similar with or without the Project, with the main exception being 2016 (nearly 40% fewer days of passage; Table 11-22).

Migration Flow-Survival

The main juvenile steelhead downstream migration period in the Sacramento River is October–June (Table 11A-8 in Appendix 11A). This period coincides with pulse flow protection measures to be applied to precipitation-generated pulse flow events (Chapter 2) for Alternatives 1, 2, and 3 diversion criteria. River flow-survival relationships analogous to those for juvenile Chinook salmon (e.g., Michel et al. 2021; see also discussion for winter-run Chinook salmon) have not been established for migrating juvenile steelhead. However, assuming that flow may affect survival in a somewhat similar manner to juvenile Chinook salmon, the modeling based on Michel et al. (2021) (Appendix 11P) suggests there would be little difference between Alternatives 1, 2, and 3 and the NAA for juvenile steelhead migration survival as a function of river flow.

Sites Reservoir Release Effects

Sites Reservoir releases could temporally overlap with steelhead presence near the release location in the Sacramento River. Migrating adults would be present during August through

[22] The criteria for days to be counted as Yolo Bypass passage days were river stage (1) between 21.14 feet and 29.92 feet during November 1–March 15, (2) between 21.14 feet and 23.35 feet during March 16–April 30, or (3) greater than 35 feet (elevation of the crest of the Fremont Weir). These criteria were based on the YBPASS tool (California Department of Water Resources 2017).

March, although primarily August through October, and rearing or emigrating juveniles would be present during December through May (Appendix 11A1, Figure 11A1-39).

Sites Reservoir releases into the Yolo Bypass via the CBD would overlap with the adult steelhead upstream migration period during August through October, but not juvenile rearing and emigration (Appendix 11A, Table 11A-8; Figure 11-17).

As discussed in more detail above for fall-run/late fall–run Chinook salmon, Sites Reservoir releases to the CBD and thence the Toe Drain under Alternatives 1, 2, and 3 during August–October could result in increased rates of adult steelhead occurrence at Wallace Weir, although this is uncertain. One marked (hatchery-origin) adult steelhead was collected in the fish trap at Wallace Weir in September 2018, whereas none were collected during trapping in 2014 (September/October), 2015 (October), 2017 (September/October), and 2020 (October) (Table 11-42 in the fall-run/late fall–run Chinook salmon analysis for details of trapping effort and data source). In 2019, a total of 13 adult steelhead, primarily hatchery-origin, were collected at Wallace Weir by trapping at the fish rescue facility and associated beach seining nearby, of which two fish (15%) died (Davis et al. 2019). Assuming this increased occurrence of steelhead at Wallace Weir is the result of the flow action, which is uncertain, then similar flow actions facilitated by reservoir releases under Alternatives 1, 2, and 3 would have the potential to result in a small DPS-level effect on steelhead: overall population numbers as indicated by monitoring of adults returning to hatcheries is in the thousands of fish (National Marine Fisheries Service 2016:12).

Temperature Effects

As discussed in Chapter 6, the effects of Alternatives 1A, 1B, 2, and 3 on water temperatures at the Sites Reservoir release site in the Sacramento River would be relatively small with the releases generally tending to cause a slight reduction in water temperature (Tables 6-12a through 6-12d). Temperature-related effects of Alternatives 1A, 1B, 2, and 3 on steelhead at the release site would be inconsequential.

All changes in water temperature in the Yolo Bypass due to Sites Reservoir releases via the CBD would be zero or negative for Alternatives 1A, 1B, 2, and 3 (Table 11-23). These lower water temperatures are expected to have a negligible effect on adult steelhead in the Yolo Bypass.

As discussed in Impact FISH-2, temperature changes under Alternatives 1A, 1B, 2, and 3 due to Sites Reservoir releases in the Sacramento River below the Yolo Bypass, and the effects on steelhead, would be inconsequential due to the small proportion of Sacramento River water in this reach coming off the Yolo Bypass.

Feather River

Far-Field Effects

Temperature Effects

Operation of Sites Reservoir has the potential to change water temperatures in the Feather River that could affect steelhead. As described in Appendix 11B, the two methods used to analyze temperature-related effects on steelhead in the Feather River were: (1) Physical Model Output

Characterization; and (2) Water Temperature Index Value/Range Exceedance Analysis. More details on these methods are provided in Appendix 11B.

The Authority and Reclamation evaluated water temperature model outputs during the period of presence and in the locations of each life stage of steelhead in the Feather River LFC at Robinson Riffle and in the HFC at Gridley Bridge. Visual observation of exceedance plots and differences in modeled mean monthly temperatures by water year type between Alternatives 1, 2, and 3 and the NAA in these locations indicates that water temperatures would be predominantly similar among alternatives during the period of presence of each life stage of steelhead (Appendix 6C, Tables 6C-16-1a to 6C-16-4c, Tables 6C-18-1a to 6C-18-4c; Figures 6C-16-1 to 6C-16-18, Figures 6C-18-1 to 6C-18-18). At both locations, mean monthly water temperatures for all months within all water year types under Alternatives 1A and 1B were within 0.5°F of the NAA. Water temperature modeling results for Alternatives 2 and 3 were similar to those of Alternative 1 at both locations. Overall, these differences would be biologically inconsequential due to their low frequency and small magnitude.

Results for each alternative of the analysis of exceedance above water temperature index values for steelhead in the Feather River from Appendix 11B, Table 11B-3 are presented in Appendix 11D, Table 11D-162 through Table 11D-181 and summarized in Table 11-51. For each life stage and at both locations evaluated, there were no month and water year type combinations in which both: (1) the percent of months that exceeded the index values was more than 5% greater under the alternative than under the NAA, and (2) the exceedance per month was more than 0.5°F greater under the alternative than under the NAA. Results of the exceedance analysis for Alternatives 2 and 3 are similar to Alternative 1 with no month and water year type combinations in which both criteria were met for any life stage at both locations. These results indicate that water temperature-related effects on steelhead in the Feather River would be negligible.

Table 11-51. Number of Month and Water Year Type Combinations that Satisfy Both Criteria for Being Biologically Meaningful in the Water Temperature Index Value Analysis, Steelhead, Feather River[1,2,3]

Location	Spawning and Egg Incubation				Kelt Emigration				Juvenile Rearing				Smolt Emigration				Smoltification				Adult Immigration				Adult Holding			
	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3
Low-Flow Channel Below Fish Barrier Dam	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
High-Flow Channel Below Thermalito Afterbay Outlet	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

[1] Biologically Meaningful Criteria include: (1) the difference in frequency of exceedance between the NAA and the alternative was greater than 5%, and (2) the difference in average monthly exceedance between the NAA and the alternative was greater than 0.5°F.
[2] Index values for each life stage are located in Appendix 11B, Table 11B-3.
[3] Full results presented in Appendix 11D, Table 11D-162 through Table 11D-181.

The probability that each alternative would meet water temperature targets to support anadromous fish, including steelhead, in the Feather River included the Settlement Agreement for Licensing of the Oroville Facilities (California Department of Water Resources 2006:A-18–A-24) was described in Impact FISH-3 above. Those results indicate that the Project would not substantially change the ability to meet water temperature targets in the Oroville Settlement Agreement relative to the NAA.

Combined, these water temperature results indicate that the Project's implementation under any of the action alternatives would cause inconsequential temperature-related effects on steelhead in the Feather River.

Flow-Related Physical Habitat Conditions

Redd Scour and Entombment

Frequency of scouring flows was not estimated for the Feather River because information on minimum flows required to mobilize sediments is not available for the Feather River. However, Feather River flows during the high-flow months (December through May), when scouring flows would be most likely to occur, are generally similar between the NAA and Alternatives 1, 2, and 3 (Chapter 5). Therefore, no substantial differences on the frequency of scouring flows are expected between the NAA and Alternatives 1, 2, and 3.

Redd Dewatering

As described in Section 11N.2, *Methods*, redd dewatering for Feather River steelhead was estimated from the depth distributions of their redds and changes in river stage (depth) at the Gridley gage on the Feather River at different flows. No sampling of steelhead redds was conducted in the HFC, so the depth distribution of steelhead redds from the American River redd dewatering study (Bratovich et al. 2017) was used for the HFC steelhead redd dewatering analysis. The use of redd depth distributions from a different river adds uncertainty to the analysis, but the proximity of the rivers and their steelhead populations likely reduces differences. The river depths were estimated from CALSIM II monthly flow results at the Thermalito Afterbay outlet and DWR's stage-discharge tables for the Gridley gage. Comparisons of the means of the redd dewatering estimates under Alternatives 1, 2, and 3 to means under the NAA use absolute differences rather than relative differences (percent change) because many of the values for percentages of redds dewatered are small. Expressing changes of small values as percent changes may result in very large values that may be misleading. The use of monthly time-step flow estimates likely underestimates redd dewatering rates because they smooth out short-term flow fluctuations. This potential bias is expected to affect all Project scenarios equally. The results for the Feather River steelhead (Table 11N-19) show little effect of Alternatives 1, 2, and 3 on redd dewatering. Almost all differences from the NAA are less than 1%, except for a 6% reduction in redd dewatering under Alternative 3 for spawning in March of Above Normal Water Years. These results indicate that Alternatives 1, 2, and 3 would not affect steelhead redd dewatering in the Feather River.

Habitat Weighted Usable Area

Spawning Habitat Weighted Usable Area

To evaluate the effects of Alternatives 1, 2, and 3 on steelhead spawning habitat, steelhead spawning WUA was computed from the steelhead spawning WUA curve for the Feather River (Payne and Allen 2004) (Figure 11K-8) for CALSIM II flows below Thermalito Afterbay under Alternatives 1, 2, and 3 and the NAA during the December through March Feather River steelhead spawning period.

There are few sizable differences between Alternatives 1, 2, and 3 and the NAA in steelhead spawning WUA (Table 11K-20). The largest difference is an 8% increase during March of Above Normal Water Years under Alternative 3. The largest reductions are 1.3% in February of Wet Water Years under Alternatives 1A and 2. Only three differences are >2%. These results indicates that Alternatives 1, 2, and 3 would have little effect on steelhead spawning WUA in the Feather River.

Rearing Habitat Weighted Usable Area

As discussed above for Feather River spring-run Chinook salmon, reliable predictions regarding changes in flow affecting rearing habitat for steelhead in the Feather River cannot be made because the only rearing WUA curves developed for Feather River steelhead are unreliable (Payne and Allen 2005). WUA curves from other rivers (e.g., Sacramento River) cannot be used to determine the availability of rearing habitat WUA in response to changes in flow in the Feather River because relationship between flow and habitat WUA are generally not transferrable from one river to another (Bovee et al. 1998). As such, quantitative conclusions regarding effects of Alternatives 1, 2, and 3 on steelhead rearing WUA cannot be drawn.

Juvenile Stranding

No formal method for analyzing juvenile stranding analysis has been developed for the Feather River. No existing credible scientific evidence could have been used to practically estimate juvenile stranding for the Feather River.

American River

Far-Field Effects

Temperature Effects

Operation of Sites Reservoir has the potential to change water temperatures in the American River that could affect the life stages of steelhead present. As described in Appendix 11B, the two methods used to analyze temperature-related effects on steelhead in the American River were: (1) Physical Model Output Characterization; and (2) Water Temperature Index Value/Range Exceedance Analysis. More details on these methods are provided in Appendix 11B.

The Authority and Reclamation evaluated water temperature model outputs during the period of presence and in the locations of each life stage of steelhead in the American River (see Appendix 11B, Table 11B-4 for timing and locations). Visual observation of exceedance plots and differences in modeled mean monthly temperatures by water year type between alternatives and the NAA in the American River below Nimbus Dam and Watt Avenue indicates that water temperatures would be predominantly similar among all alternatives during the period of presence of each life stage of steelhead (Appendix 6C, Tables 6C-13-1a to 6C-13-4c, Tables 6C-

14-1a to 6C-14-4c; Figures 6C-13-1 to 6C-13-18, Figures 6C-14-1 to 6C-14-18). At both locations, mean monthly water temperatures for all months within all water year types under Alternatives 1A and 1B were within 0.3°F of the NAA. Water temperature modeling results for Alternatives 2 and 3 were similar to those of Alternative 1 at both locations. These results suggest that temperature-related effects on steelhead in the American River would be negligible.

Results of the analysis of exceedance above water temperature index values for steelhead in the American River from Appendix 11B, Table 11B-4 are presented in Appendix 11D, Table 11D-182 through Table 11D-201 and summarized in Table 11-52. For each life stage and at both locations evaluated, there were no month and water year type combinations in which both: (1) the percent of days that exceeded the index values was more than 5% greater under the alternative than under the NAA; and (2) the exceedance per day was more than 0.5°F greater under the alternative than under the NAA. Results of the exceedance analysis for Alternatives 2 and 3 are similar to Alternative 1 with no month and water year type combinations in which both criteria would be met for any life stage at both locations. These results indicate that temperature-related effects on fall-run Chinook salmon in the American River would be negligible.

Table 11-52. Number of Month and Water Year Type Combinations that Satisfy Both Criteria for Being Biologically Meaningful in the Water Temperature Index Value Analysis, Steelhead, American River[1,2,3]

Location	Spawning and Egg Incubation				Kelt Emigration				Juvenile Rearing				Smolt Emigration				Smoltification				Adult Immigration				Adult Holding			
	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3
Hazel Avenue	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Watt Avenue	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

[1] Biologically Meaningful Criteria include: (1) the difference in frequency of exceedance between the NAA and the alternative was greater than 5%, and (2) the difference in average daily exceedance between the NAA and the alternative was greater than 0.5°F.
[2] Index values for each life stage are located in Appendix 11B, Table 11B-4.
[3] Full results presented in Appendix 11D, Table 11D-182 through Table 11D-201.

Combined, these water temperature results indicate that the Project would cause inconsequential temperature-related effects on steelhead in the American River.

Flow-Related Physical Habitat Conditions

Redd Scour and Entombment

Loss of redds to scouring or entombment occurs when flows are high enough to mobilize sediments, destroying redds and their incubating eggs and alevins, or entombing the redds when sediments are redeposited. A flow of 40,000 cfs was selected as the scour flow threshold for the American River based on estimates in the literature (Appendix 11N, Table 11N-10).

Only CALSIM II monthly estimates were available for estimating flows in the American River under Alternatives 1, 2, and 3 and the NAA. Redd scour can occur at a small temporal scale (minutes to hours). This limitation was addressed using historical American River gage data to determine the minimum monthly flow for which at least one daily flow of the month exceeded 40,000 cfs (Appendix 11N, Section 11N.2). The minimum monthly flow was 19,530 cfs, so this flow was used as the threshold scouring flow with CALSIM II data in American River at Nimbus Dam (Figure 11N-2).

The results indicate that there are few months in the 82-year CALSIM II record for the American River with flow greater than the redd scour or entombment threshold of 19,350 cfs (Appendix 11N, Table 11N-26). There are only 2 months with such flows under the NAA and Alternatives 1, 2, and 3. These results indicate that Alternatives 1, 2, and 3 would have no adverse effect on redd scour or entombment for steelhead in the American River.

Redd Dewatering

The redd dewatering analysis for the lower American River uses relationships between flow, river stage, and redd depth distribution developed by Bratovich et al. (2017). CALSIM II flow estimates at the Nimbus Dam location were used to compute stage at the spawning and dewatering flows, and the redd depth frequency distribution was queried to determine the percentage of the redds that occur between those two stages and would be dewatered. The analyses were conducted for the months of steelhead spawning and incubation. Based on ranges provided in Bratovich et al. (2017), American River steelhead are estimated to have 2-month incubation periods. The use of monthly time-step flow estimates like those obtained from CALSIM II modeling likely underestimates redd dewatering rates because they smooth out short-term flow fluctuations. This potential bias is expected to affect the Project and NAA scenarios equally. The analysis compared CALSIM II flow estimates below Nimbus Dam for each spawning month with the minimum flow during the 2 months following the spawning month to estimate the percentage of redds dewatered. The use of CALSIM II monthly time-step flow estimates likely underestimates redd dewatering rates. This potential bias is expected to affect all Project scenarios equally. The means of the redd dewatering estimates under the NAA and Alternatives 1, 2, and 3 are compared using absolute differences rather than relative differences (percent change) because many of the values for percentages of redds dewatered are small. Expressing changes of small values as percent changes may result in large values that may be misleading.

The results for steelhead redd dewatering in the American River show little effect from the alternatives (Table 11N-21). Note that the incubation period for steelhead in the American River is 2 months rather than 3 months. The only changes in steelhead redd dewatering of 2% or more are a 2% reduction for February spawning in Above Normal Water Years under Alternative 3 and a 2% increase for February spawning in Critically Dry Water Years under Alternative 2. Overall, Alternatives 1, 2, and 3 are expected to have little effect on American River steelhead redd dewatering.

Habitat Weighted Usable Area

Spawning Habitat Weighted Usable Area

To evaluate the effects of Alternatives 1, 2, and 3 on steelhead spawning habitat in the American River, steelhead spawning WUA was estimated for CALSIM II flows at Nimbus Dam under the NAA and Alternatives 1, 2, and 3 during the December through March spawning period using the steelhead composite spawning WUA curve (Bratovich et al. 2017) (Figure 11K-10).

The largest differences in steelhead mean spawning WUA between Alternatives 1, 2, and 3 and the NAA are 2% reductions and increases, both under Alternative 3, in December of Above Normal Water Years and February of Above Normal Water Years, respectively. Almost all other differences are <1%. These results indicates that Alternatives 1, 2, and 3 would have little effect on steelhead spawning WUA in the American River.

Rearing Habitat Weighted Usable Area

As discussed above for Feather River fall-run Chinook salmon, reliable predictions regarding changes in flow affecting rearing habitat for steelhead in the Feather River cannot be made because the only rearing WUA curves developed for American River steelhead are outdated (U.S. Fish and Wildlife Service 1985). WUA curves from other rivers (e.g., Sacramento River) cannot be used to determine the availability of rearing habitat WUA in response to changes in flow in the American River because relationship between flow and habitat WUA are generally not transferrable from one river to another (Bovee et al. 1998). As such, quantitative conclusions regarding effects of Alternatives 1, 2, and 3 on steelhead rearing WUA cannot be drawn.

Juvenile Stranding

No formal method for analyzing juvenile stranding analysis has been developed for the American River. No existing credible scientific evidence could have been used to practically estimate juvenile stranding for steelhead in the American River.

Delta

Juvenile Through-Delta Survival

As described in Appendix 11A, the main juvenile steelhead migration period in the Delta is February–May. Through-Delta flow-survival relationships analogous to those for juvenile Chinook salmon (e.g., Perry et al. 2018; see also discussion for winter-run Chinook salmon) have not been established for migrating juvenile steelhead. However, assuming that flow may affect survival in a somewhat similar manner to juvenile Chinook salmon, the modeling based on the through-Delta survival function formulated by Perry et al. (2018) suggests there would be little difference between Alternatives 1, 2, and 3 and the NAA (Table 11-24 in the winter-run analysis).

South Delta Entrainment

As discussed for other salmonids, there would be little difference in indicators of entrainment risk (south Delta exports and Old and Middle River flows) during winter/spring between the NAA and Alternatives 1, 2, and 3 (Appendix 5B3, Tables 5B3-6-1a through 5B3-6-4c; Figures 5B3-6-1 through 5B3-6-18; and Appendix 5B4, Tables 5B4-1-1a through 5B4-1-4c; Figures 5B4-1-1 through 5B4-1-18), and existing restrictive criteria from the NMFS (2019a) ROC ON LTO BiOp and CDFW (2020) State ITP would limit entrainment risk for steelhead under Alternatives 1, 2, and 3. South Delta export of water released from Sites Reservoir would have limited overlap with steelhead occurrence. This is illustrated by the results of the salvage-density analysis (Appendix 11Q) for which there were minimal differences between the NAA and Alternatives 1, 2, and 3 (Table 11-53, Table 11-54).

Table 11-53. Entrainment Loss of Juvenile Steelhead At SWP Banks Pumping Plant, Averaged by Water Year Type, Based on the Salvage-Density Method.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	5,187	5,190 (0%)	5,184 (0%)	5,189 (0%)	5,192 (0%)
Above Normal	NA	(1%)	(1%)	(1%)	(-1%)
Below Normal	4,401	4,408 (0%)	4,418 (0%)	4,401 (0%)	4,374 (-1%)
Dry	2,327	2,335 (0%)	2,328 (0%)	2,327 (0%)	2,312 (-1%)
Critically Dry	2,063	2,091 (1%)	2,088 (1%)	2,099 (2%)	2,082 (1%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. The analysis was based on historical salvage data during 2009–2019 which did not include any Above Normal Water Years, hence that row of the table is noted as ‘NA’; the percentage difference in Above Normal Water Years is based on density data for Wet Water Years applied to above normal modeled exports. Results by water year type and month are provided in Appendix 11Q. Percentage values are rounded; as a result, differences between percentages may not always appear consistent.

Table 11-54. Entrainment Loss of Juvenile Steelhead At CVP Jones Pumping Plant, Averaged by Water Year Type, Based on the Salvage-Density Method.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	339	338 (0%)	338 (0%)	338 (0%)	337 (-1%)
Above Normal	NA	(-1%)	(-1%)	(-1%)	(-1%)
Below Normal	939	936 (0%)	938 (0%)	941 (0%)	955 (2%)
Dry	656	655 (0%)	657 (0%)	657 (0%)	664 (1%)
Critically Dry	174	174 (0%)	181 (4%)	174 (0%)	182 (4%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. The analysis was based on historical salvage data during 2009–2019 which did not include any Above Normal Water Years, hence that row of the table is noted as ‘NA’; the percentage difference in Above Normal Water Years is based on density data for Wet Water Years applied to above normal modeled exports. Results by water year type and month are provided in Appendix 11Q. Percentage values are rounded; as a result, differences between percentages may not always appear consistent.

CEQA Significance Determination for Alternatives 1, 2, and 3

The preceding subsections of this impact discussion provide the detailed information used for this CEQA (and NEPA) determination. This section provides a summary of this information.

In-Delta and upstream operational impacts of Alternatives 1, 2, and 3 on steelhead generally would be limited. Steelhead would have the potential for similar types of near-field effects to those previously discussed for winter-run and spring-run Chinook salmon (i.e., entrainment, impingement and screen contact, predation, stranding behind screens, and attraction to reservoir discharge). Entrainment potential would be limited because the species tends to undergo downstream migration as larger juveniles (yearlings or older) (Appendix 11A). This larger size would tend to considerably limit the potential for negative near-field effects on juveniles based on greater swimming ability than juvenile Chinook salmon. As described for winter-run Chinook salmon, steelhead could be attracted to the flow from the Sacramento River discharge under Alternative 2. However, the design of the apron and weir would exclude anadromous fish and eliminate stranding risk.

Related to temperature effects on steelhead, observations of exceedance plots and differences in modeled mean monthly temperatures by water year type at all locations analyzed in the Sacramento, Feather, and American Rivers indicate that Alternatives 1, 2, and 3 and the NAA would be predominantly similar among alternatives. The water temperature indicator value exceedance analysis confirms this similarity among alternatives. Results indicate that Alternatives 1, 2, and 3 would have little effect on frequency of redd scour and entombment for steelhead in the Sacramento River.

The results detailed above related to redd scour or entombment for steelhead in the Sacramento, Feather, and American River, indicate that Alternatives 1, 2, and 3 would have no adverse effect.

There are a few notable differences in steelhead mean spawning WUA between Alternative 1, 2, and 3 and the NAA in Segments 6 and 5. The largest difference is a 7% reduction under Alternative 3 in Segment 6 for February of Above Normal Water Years. Other reductions ranging from 5% to 6% occur in Segment 6 during February of Above Normal Water Years under Alternative 1B, in Segment 6 during December of Wet Water Years under Alternative 3, and in Segment 5 in February of Above Normal Water Years under Alternative 3. Most differences in all river segments and all three alternatives are <3%. These results indicate that Alternatives 1A, 1B, and 2 would have little effect on steelhead spawning WUA. Alternative 3 has more frequent larger negative effects, but these are not expected to substantially affect steelhead spawning habitat availability. Similarly, for steelhead fry rearing WUA, results indicate that Alternatives 1, 2, and 3 would have a little effect on steelhead fry rearing habitat availability. As described in Chapter 2, the Project's diversion criteria include restrictions to maintain Bend Bridge and Wilkins Slough flows, which have the effect of limiting changes to Yolo Bypass spill frequency and duration under Alternatives 1, 2, and 3 relative to the NAA. Thus, Alternatives 1, 2, and 3 generally would have limited potential for negative effects on Yolo Bypass floodplain inundation and access for steelhead. The results of the frequency analysis of inundation of events for the Yolo Bypass generally show only minor differences between Alternatives 1, 2, and 3 and the NAA.

The Project would operate to avoid effects on the Big Notch's ability to achieve the same level of performance for salmonids in the Sacramento River as it would absent the Project. However, the Adaptive Management Plan for the Project recognized there is uncertainty about the performance of the Big Notch and the effects of the Project on it. Monitoring will be conducted, in cooperation with the State, to determine whether there is an effect and, if so, what the magnitude of that effect would be on entrainment of juvenile salmon into the Yolo Bypass. If there is an adverse effect, a science-based adaptive management approach will be employed to determine how to adjust diversions 158 RMs upstream of the Big Notch to maintain its efficiency for entraining juvenile salmon into the Yolo Bypass.

The Big Notch Project also is intended to reduce migratory delays and loss at the Fremont Weir of adult salmon, steelhead, and sturgeon that are migrating upstream through the Yolo Bypass. As with juvenile passage, the Project would operate to avoid affecting adult passage at Fremont Weir. The number of days meeting adult Chinook salmon passage criteria at Fremont Weir generally would be similar with or without the Project (Table 11-21). This too will be evaluated and adaptively managed as required.

The Sutter Bypass, when inundated, provides important juvenile rearing habitat for Chinook salmon and steelhead, as discussed for the Yolo Bypass. For the Sutter Bypass the modeling results indicate that Alternatives 1, 2, and 3 would produce little change in suitable habitat as compared to the NAA. Changes in Sacramento River flow and flow entering Sutter Bypass as a result of the Project have the potential to change the number of days meeting adult Chinook salmon passage (and by proxy steelhead) criteria at the three Sutter Bypass weirs (Moulton, Tisdale, and Colusa). The results of this analysis indicated that there would be no days meeting adult salmon passage criteria during 2009–2018 at Moulton or Tisdale Weirs under the NAA or Alternatives 1, 2, and 3. For Colusa Weir, the number of days meeting fish passage criteria generally would be similar with or without the Project.

As described in Chapter 2, Alternatives 1, 2, and 3 include pulse flow protection measures to be applied to precipitation-generated pulse flow events. There is also a 10,700-cfs Wilkins Slough bypass flow threshold from October 1 through June 14. Adaptive management would be utilized to assess and if necessary, refine the diversion criteria based on biological monitoring (see Appendix 2D, Section 2D.6). Sites Reservoir diversion criteria effectively limit diversions during the historical periods of fish movement as reflected in Red Bluff rotary screw trap data, and application of the flow-threshold criteria from Michel et al. (2021) suggests that flow-survival effects on juvenile Chinook salmon (assumed to be similar for steelhead) would be greatly limited by the Project's diversion criteria because there is essentially no difference in predicted survival between the Project and NAA.

Based on the analyses provided, operation of Alternative 1, 2, or 3 would not have a substantial adverse effect, either directly or through habitat modifications, on steelhead. In consideration of the analyses within the preceding subsections of this impact discussion and summarized above, operation impacts of Alternative 1, 2, or 3 would be less than significant. No mitigation is necessary.

NEPA Conclusion for Alternatives 1, 2, and 3

Operation effects on steelhead would be the same as described above for CEQA. In-Delta and upstream operational effects of Alternatives 1, 2, and 3 on steelhead generally would be limited. Related to temperature effects, observations of exceedance plots and differences in modeled mean monthly temperatures by water year type indicate that Alternatives 1, 2, and 3 and the NAA would be predominantly similar among alternatives during the period of presence of each steelhead life stage. The water temperature indicator value exceedance analysis confirms this similarity among alternatives.

As described in Chapter 2, the Project's diversion criteria include restrictions to maintain Bend Bridge and Wilkins Slough flows, which have the effect of limiting changes to Yolo Bypass spill frequency and duration under Alternatives 1, 2, and 3 relative to the NAA. Thus, Alternatives 1, 2, and 3 generally would have limited potential for negative effects on Yolo Bypass floodplain inundation and access for steelhead. Also described in Chapter 2, Alternatives 1, 2, and 3 include pulse flow protection measures to be applied to precipitation-generated pulse flow events. There is also a 10,700-cfs Wilkins Slough bypass flow threshold from October 1 through June 14, corresponding with the threshold identified by Michel et al. (2021). Operation of Alternative 1, 2, or 3 would have no adverse effect on steelhead.

Impact FISH-6: Operations Effects on Green Sturgeon

Alternatives 1, 2, and 3

Effects of Alternatives 1, 2, and 3

Sacramento River

Near-Field Effects

Relative to juvenile salmonids, there are no field-based investigations informing the risk from near-field effects of Alternative 1, 2, and 3 diversions at Red Bluff and Hamilton City for green sturgeon. There are, however, laboratory investigations from which risk can be inferred. Although screen criteria for green sturgeon have not been developed by NMFS or CDFW, the laboratory studies of Verhille et al. (2014) provided recommendations for intake approach velocity based on flow-tolerance criteria (Figure 11-23). During the primary months in which the Project diversions would operate (i.e., November–March), the flow-tolerance results of Verhille et al. (2014) suggest that approach velocity below 50 cm/s (i.e., 1.6 ft/s) would be protective (Figure 11-23). The approach velocity at Red Bluff and Hamilton City would be no more than 0.33 ft/s in accordance with agency requirements and would be expected to be protective based on the flow-tolerance relationships from Verhille et al. (2014). Verhille et al. (2014) suggested that green sturgeon larvae could be susceptible to diversions regardless of approach velocity during April and May. During April, diversions at Red Bluff under Alternatives 1, 2, and 3 would generally be similar or somewhat (<100 cfs) greater than the NAA, except in ~10% of years, when diversions would be up to ~2,000 cfs, compared to ~500–700 cfs under the NAA (Appendix 5B1, *Project Operations*, Figure 5B1-1-13). In May, diversions under Alternatives 1B and 3 would be up to 200 cfs less than the NAA and Alternatives 1A and 2 at ~40%–85% exceedance, whereas Alternatives 1, 2, and 3 would have considerably greater diversions than the NAA (>2,000 cfs compared to <1,000 cfs) in only a few years (Appendix 5B1, Figure 5B1-1-

14). As shown in Table 11-6, these diversion rates represent water-year-type means of 7% or less of Sacramento River flow. Similar patterns are generally evident at the Hamilton City diversion (Appendix 5B1, Figures 5B1-2-13 and 5B1-2-14), although the water-year-type mean diversion rate is greater (up to 9% in April; 24% or less in May; Table 11-7).

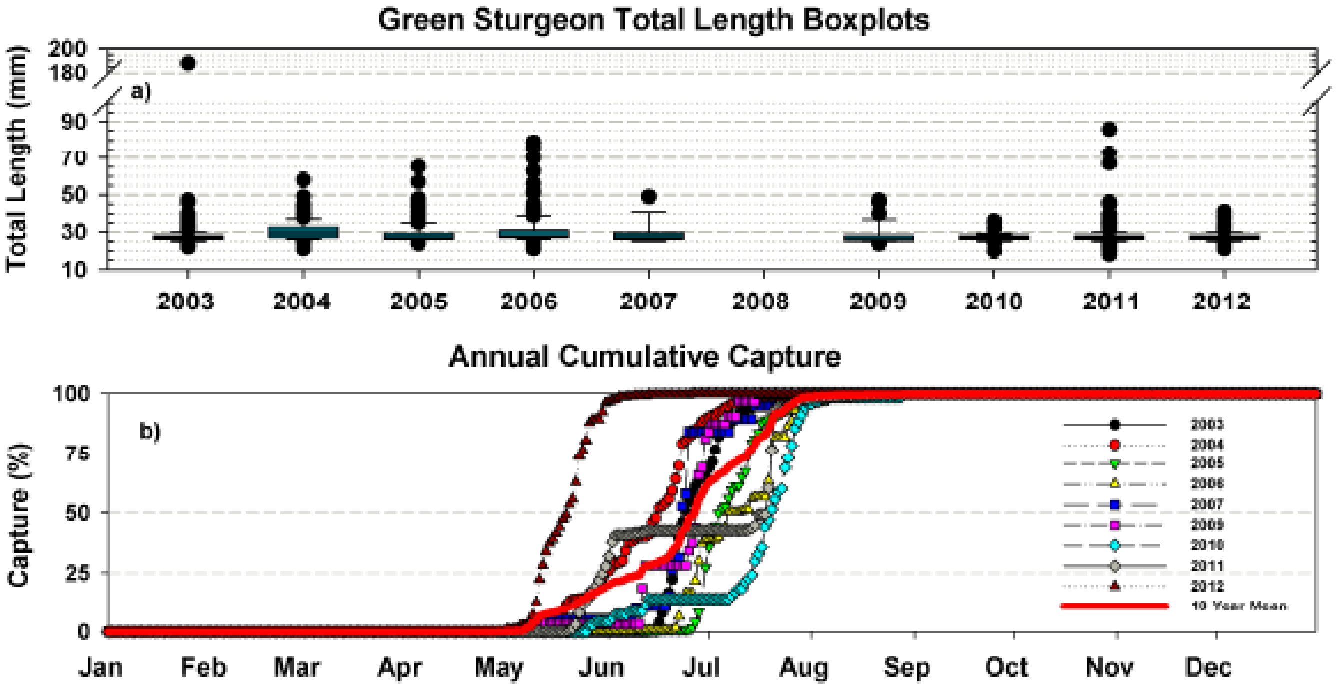
Rotary screw trap data and entrainment sampling would provide refined assessment of potential negative effects on green sturgeon larvae and—if suggested to be necessary from this monitoring—adjustment to operations as part of the adaptive management program, which is not reflected in the modeling of operations of Alternatives 1, 2, and 3. Rotary screw trap monitoring data indicate that green sturgeon larval occurrence at Red Bluff and Hamilton City generally only begins in May (Figure 11-24 and Table 11-55). Green sturgeon post larvae collected at RBDD rotary screw traps during 2002–2012 had minimum annual total lengths ranging from 18 to 25 mm (Figure 11-24). The length distribution of green sturgeon in rotary screw traps does not differ from that of larval sampling nets and indicates that this is the size at initial dispersal from egg incubation and hatching areas (Poytress et al. 2011:15). Based on these sizes, it would be expected that the 1.75-mm-opening intake screens would exclude most larval sturgeon, given that a laboratory study of screen with this size of opening found entrainment of morphologically similar pallid sturgeon at sizes below 20 mm (Mefford and Sutphin 2008). With approach velocity at Red Bluff and Hamilton City of no more than 0.33 ft/s, the two intakes would be expected to be protective of green sturgeon for the remainder of the year based on the flow-tolerance criteria suggested by Verhille et al. (2014) (Figure 11-23). Given that these criteria did not consider behavioral effects (e.g., avoidance of intakes) (Verhille et al. 2014), there is uncertainty in the results based solely on approach velocity. A recent review of green sturgeon biology by Heublein et al. (2017:20) noted: "Larval green sturgeon are present in areas where substantial water volumes are diverted, and, due to small size and relatively poor swimming performance of larvae, it is almost certain that entrainment effects [sic] larval survival…The RBPP [Red Bluff] and GCID [Hamilton City] facilities include modern fish screens to reduce entrainment of juvenile salmonids, but the effectiveness of screens and facility operations in reducing larval green sturgeon entrainment is poorly understood."

Entrainment of green sturgeon eggs by the Alternative 1, 2, and 3 diversions could also occur during the spawning period, which occurs in April–June (Poytress et al. 2015), although such entrainment may be limited by the eggs being demersal and weakly adhesive (Wang 2006:3). In addition, the eggs may be more likely to be subject to impingement than entrainment because their diameter (generally 4.0 mm or greater; Wang 2006:3) is greater than the intake screen openings (1.75 mm). The primarily nocturnal migration of larval green sturgeon (Poytress et al. 2011) is shown on Figure 11-25.

<table>
<tr><th></th><th>Upper river</th><th>Middle river</th><th>Lower river/ delta/bays</th></tr>
<tr><td>January</td><td rowspan="3"><50 cm s^{-1}</td><td></td><td></td></tr>
<tr><td>February</td><td rowspan="4">WS early larvae</td><td></td></tr>
<tr><td>March</td><td></td></tr>
<tr><td>April</td><td rowspan="2">GS early larvae</td><td></td></tr>
<tr><td>May</td><td></td></tr>
<tr><td>June</td><td colspan="3">GS and WS 29 cm s^{-1}</td></tr>
<tr><td>July</td><td></td><td colspan="2">WS 45 cm s^{-1}</td></tr>
<tr><td>August</td><td></td><td colspan="2">GS 50 cm s^{-1}</td></tr>
<tr><td>September</td><td></td><td></td><td></td></tr>
<tr><td>October</td><td rowspan="3"><50 cm s^{-1}</td><td colspan="2" rowspan="2">GS 40 cm s^{-1}</td></tr>
<tr><td>November</td></tr>
<tr><td>December</td><td></td><td></td></tr>
</table>

Source: Verhille et al. 2014. Note: Green sections demarcate tolerable water velocities of ≥50 cm/s; red sections demarcate presence of life stages which are predicted to be intolerant of even very low water velocities; and yellow sections signify recommended water flow velocity limitations to protect present life stages. Behavioral (e.g., avoidance) considerations were not part of this analysis, and they remain an important topic for future research. Based on the text description, Red Bluff and Hamilton City diversions are in the "Upper river".

Figure 11-23. Overview of Flow-Tolerance Limitations of Green (GS) and White (WS) Sturgeon Throughout the Sacramento–San Joaquin Watershed According to Location and Time of Year, Based on Critical Swimming Speed.



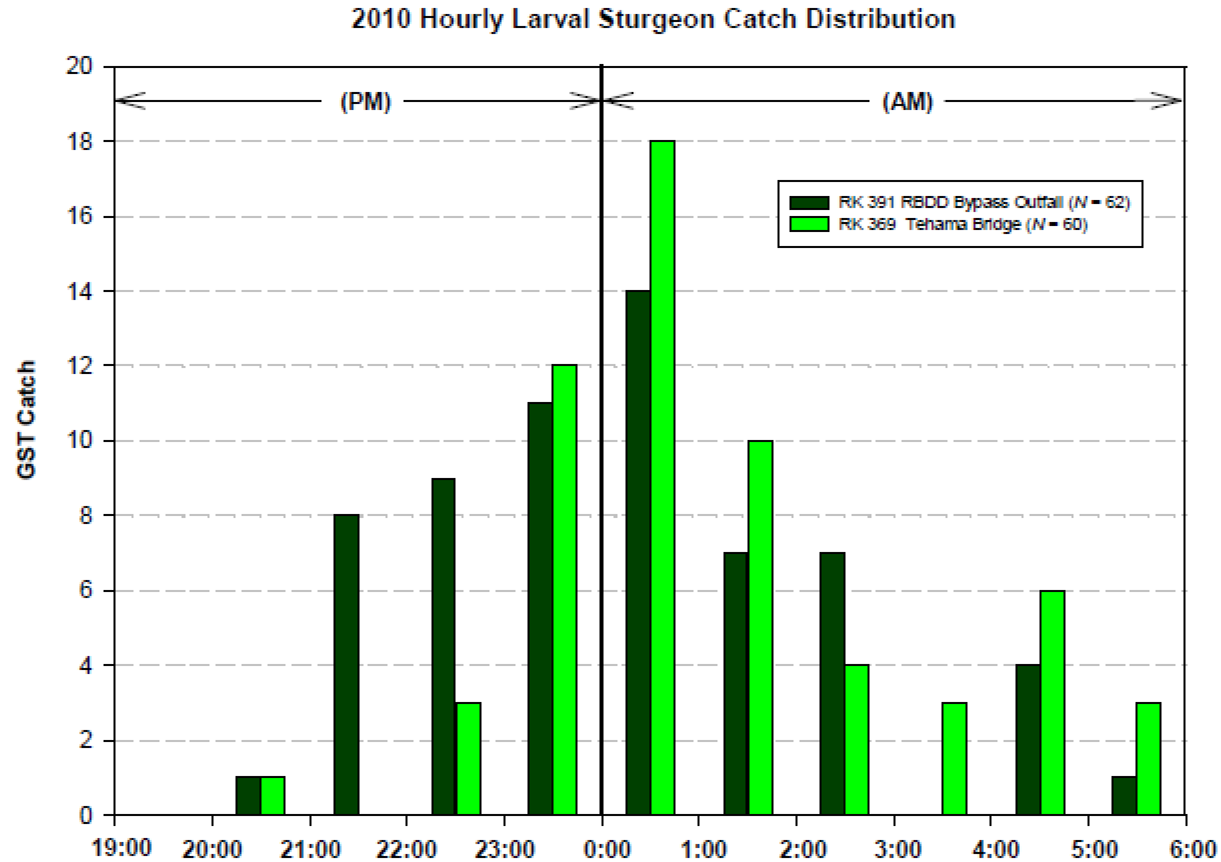
Source: Poytress et al. 2014:99.

Figure 11-24. Green Sturgeon a) Annual Total Length Capture Boxplots and b) Annual Cumulative Capture Trends with 10-Year Mean Trend Line, from Rotary Screw Trap Sampling at Red Bluff Diversion Dam, 2003–2012.

Table 11-55. Rotary Screw Trap Catches of Sturgeon at GCID.

	Sturgeon in CDF&G Screw Trap at GCID														
	1994	**1995**	**1996**	**1997**	**1998**	**1999**	**2000**	**2001**	**2002**	**2003**	**2004**	**2005**	**Average**	**Median**	**Std Dev**
January	0	0	0	0	0	0	0	1	0	0	0	0	0.1	0.0	0.3
February	0	0	0	0	0	0	0	0	0	0	0	0	0.0	0.0	0.0
March	0	0	0	0	0	0	0	0	0	0	0	0	0.0	0.0	0.0
April	0	0	0	0	0	0	0	0	0	0	0	0	0.0	0.0	0.0
May	0	0	113	27	0	0	1	3	8	0	1	0	12.8	0.5	32.5
June	12	20	10	126	0	23	13	13	1	4	3	5	19.2	11.0	34.4
July	6	205	180	52	0	214	18	16	0	3	1	23	59.8	17.0	85.9
August	0	77	109	24	0	52	2	1	0	1	0	4	22.5	1.5	37.0
September	1	4	2	3	0	1	0	0	0	1	0	1	1.1	1.0	1.3
October	0	0	1	4	0	1	1	0	0	0	1	0	0.7	0.0	1.2
November	2	0	0	1	0	0	0	0	0	0	0	0	0.3	0.0	0.6
December	2	1	5	0	0	0	0	0	0	0	0	0	0.7	0.0	1.5
Total	23	307	420	237	0	291	35	34	9	9	6	33	117.0	33.5	151.2

Source: Bureau of Reclamation 2008:11-69



Source: Poytress et al. 2011:41.

Figure 11-25. Nocturnal Distribution Pattern of Capture of Larval Green Sturgeon at Red Bluff Diversion Dam Outfall and Tehama Bridge in 2010.

Far-Field Effects

Temperature Effects

Operation of Sites Reservoir has the potential to change water temperatures in the Sacramento River that could affect the life stages of green sturgeon present. As described in Appendix 11B, the two methods used to analyze temperature-related effects on green sturgeon in the Sacramento River were: (1) Physical Model Output Characterization; and (2) Water Temperature Index Value/Range Exceedance Analysis. More details on these methods are provided in Appendix 11B.

The Authority and Reclamation evaluated water temperature model outputs during the period of presence and in the locations of each life stage of green sturgeon in the Sacramento River upstream of the Delta (see Appendix 11B, Table 11B-2 for timing and locations). Visual observation of exceedance plots and differences in modeled mean monthly temperatures by water year type between Alternatives 1, 2, and 3 and the NAA in the Sacramento River at Bend

Bridge, below RBDD[23], and at Butte City indicates that water temperatures would be predominantly similar among alternatives during the period of presence of each life stage of green sturgeon (Appendix 6C, Tables 6C-9-1a to 6C-9-4c, Tables 6C-10-1a to 6C-10-4c, Tables 6C-12-1a to 6C-12-4c; Figures 6C-9-1 to 6C-9-18, Figures 6C-10-1 to 6C-10-18, Figures 6C-12-1 to 6C-12-18). At all locations, mean monthly water temperatures for all months within all water year types under Alternatives 1A and 1B were within 0.6°F of the NAA with the exception of August of critical years at Butte City, in which mean monthly water temperature would be 0.8°F lower under the NAA than under Alternatives 1A and 1B. Water temperature modeling results for Alternatives 2 and 3 were similar to those of Alternative 1 at all locations. These results suggest that temperature-related effects on green sturgeon in the Sacramento River would be negligible.

Results of the analysis of exceedance above water temperature index values for green sturgeon in the Sacramento River from Appendix 11B, Table 11B-2 are presented in Appendix 11D, Table 11D-202 through Table 11D-213 and summarized in Table 11-56. For each life stage and at all locations evaluated, there were no month and water year type combinations in which both: (1) the percent of days that exceeded the index values was more than 5% greater under Alternative 1A than under the NAA; and (2) the exceedance per day was more than 0.5°F greater under Alternative 1A and 1B than under the NAA. Results of the exceedance analysis for Alternative 2 are similar to Alternative 1A with no month and water year type combinations in which both criteria would be met for any life stage at all locations. Results of the exceedance analysis for Alternative 3 are similar to Alternative 1B in that there was one month and water year type combination, July of Above Normal Water Years, in which both criteria would be met. Because this biologically meaningful effect occurred in only one month and water year type combination, it is not expected to be persistent enough to affect green sturgeon at a population level.

Table 11-56. Number of Month and Water Year Type Combinations that Satisfy Both Criteria for Being Biologically Meaningful in the Water Temperature Index Value Analysis, Green Sturgeon, Sacramento River[1,2,3]

Location	Non-Spawning Adult Presence				Spawning and Egg Incubation				Larval to Juvenile Rearing and Emigration			
	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3
Bend Bridge	0	0	0	0	0	0	0	0	0	0	0	0
Below Red Bluff Diversion Dam	0	0	0	0	0	0	0	0	0	0	0	0
Hamilton City	0	0	0	0	0	0	0	1 Unfavorable (July, Above Normal Water Years)	0	0	0	0

[23] The RBDD, which was decommissioned in 2013, and the RBPP are co-located, and the names may be used interchangeably when referring to geographic locations

[1] Biologically Meaningful Criteria include: (1) the difference in frequency of exceedance between the NAA and the alternative was greater than 5%, and (2) the difference in average daily exceedance between the NAA and the alternative was greater than 0.5°F.
[2] Index values for each life stage are located in Appendix 11B, Table 11B-2.
[3] Full results presented in Appendix 11D, Table 11D-202 through Table 11D-213.

Overall, water temperature-related effects of Alternatives 1, 2, and 3 on green sturgeon in the Sacramento River are expected to be biologically inconsequential due to the low frequency and small magnitude of differences between each alternative and the NAA.

Flow Effects

Spawning and Egg Incubation

The effects of flow on green sturgeon spawning in the Sacramento River were recently studied by Wyman et al. (2018). The authors conducted analyses of habitat attribute suitability and WUA of green sturgeon in spawning pools and determined that the optimal flow-related habitat characteristics include:

- Depths about 25 to 30 feet
- Flow velocity of about 3.3 ft/s
- Substrate of gravel or sand

Based on measurements of depth, flow velocity and substrate in spawning pools at different Sacramento River flows, the authors determined that spawning WUA for the green sturgeon was roughly uniform from about 7,500 cfs, the lowest flow level included the study, to about 12,360 cfs. From 12,360 cfs to the highest flow included in the study, about 23,500 cfs, the WUA fell sharply. Flow velocity was found to be the most important flow-related habitat attribute for green sturgeon habitat selection, and low flows in the Sacramento River would likely result in adverse flow velocity conditions. However, the flow level at which habitat with suitable flow velocities would be unavailable is not known.

For purposes of this effects analysis, Sacramento River flows that exceed 12,360 cfs or fall below 7,500 cfs within the green sturgeon spawning reach and during the spawning period are considered to adversely affect green sturgeon spawning habitat. Note that the lower flow threshold is not based on evidence of less suitable spawning habitat conditions at flows below 7,500 cfs, but rather on the lack of evidence that the lower flows are as suitable. Green sturgeon spawning in the Sacramento River has been observed from upstream of GCID (RM 200) (near Hamilton City) to Inks Creek (RM 265), and possibly to the confluence with Cow Creek (RM 277) (Heublein et al. 2017). The adults spawn primarily from March through July (Heublein et al. 2009, 2017; National Marine Fisheries Service 2018b).

Differences in flow results between Alternatives 1, 2, and 3 and the NAA for the Sacramento River at Bend Bridge (RM 258), which is near the upper limit of the green sturgeon spawning distribution in the Sacramento River, show no flow changes between the NAA and Alternatives 1, 2, and 3 outside the uniform WUA range (Table 11-57). During March through July, CALSIM II estimates of mean monthly flows below RBPP under Alternatives 1, 2, and 3 and the NAA range from about 5,500 cfs in April of Critically Dry Water Years to about 14,000 cfs in April of

Wet Water Years (Table 11-57). Differences in flows between Alternatives 1, 2, and 3 and the NAA are less than 5% in most months, but flows are 6% to 8% lower under Alternatives 1, 2, and 3 in March of Above Normal, Below Normal, and Dry Water Years, and about 7% lower for Alternative 3 in June of Below Normal Water Years (Table 11-58). The reductions in mean flows during March of Above Normal Water Years bring the flows somewhat closer to the upper threshold, 12,360 cfs, of the WUA optimal flow range, but flows for the NAA and Alternatives 1, 2, and 3 remain well above the threshold. For all other months and water year types with appreciable flow reductions, the NAA and Alternatives 1, 2, and 3 all have flows within the range of uniform WUA, 7,500 cfs to 12,360 cfs, so no adverse effect on spawning is expected. At Hamilton City (RM 197), which is near the lower end of the spawning distribution, reductions in mean flow during March are similar to those at RBPP under the NAA (Table 11-59). Hamilton City flows increase under Alternatives 1, 2, and 3 in July of Dry and Critically Dry Water Years, and Above Normal and Below Normal Water Years for Alternative 3. However, all these flow changes occur within the range of uniform spawning WUA.

None of the changes in mean flow under Alternatives 1, 2, and 3 within the green sturgeon spawning reach during the spawning months are expected to substantially affect green sturgeon spawning habitat in the Sacramento River.

Table 11-57. CALSIM II Monthly Average Flow (cfs) by Month and Water Year Type at Bend Bridge for the NAA and Alternatives 1A, 1B, 2, and 3, and Percent Differences between Them (in Parentheses).

Month	Water Year Type[1]	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
January	Wet	27,797	27,768 (-0.1%)	27,821 (0.1%)	27,770 (-0.1%)	27,940 (0.5%)
	Above Normal	17,005	17,159 (0.9%)	17,157 (0.9%)	17,159 (0.9%)	17,325 (1.9%)
	Below Normal	8,918	8,877 (-0.5%)	8,876 (-0.5%)	8,877 (-0.5%)	8,900 (-0.2%)
	Dry	6,762	6,702 (-0.9%)	6,700 (-0.9%)	6,702 (-0.9%)	6,704 (-0.9%)
	Critically Dry	6,027	5,996 (-0.5%)	6,001 (-0.4%)	6,003 (-0.4%)	6,031 (0.1%)
	All	15,224	15,208 (-0.1%)	15,226 (0%)	15,210 (-0.1%)	15,295 (0.5%)
February	Wet	31,525	31,197 (-1%)	31,308 (-0.7%)	31,197 (-1%)	31,368 (-0.5%)
	Above Normal	25,320	25,575 (1%)	25,753 (1.7%)	25,497 (0.7%)	26,310 (3.9%)
	Below Normal	12,889	12,910 (0.2%)	13,077 (1.5%)	12,853 (-0.3%)	13,226 (2.6%)
	Dry	8,889	8,851 (-0.4%)	8,903 (0.2%)	8,851 (-0.4%)	8,882 (-0.1%)
	Critically Dry	6,411	6,504 (1.5%)	6,528 (1.8%)	6,516 (1.6%)	6,504 (1.4%)
	All	18,715	18,648 (-0.4%)	18,751 (0.2%)	18,629 (-0.5%)	18,858 (0.8%)
March	Wet	23,713	24,031 (1.3%)	23,968 (1.1%)	24,031 (1.3%)	23,932 (0.9%)
	Above Normal	18,710	18,628 (-0.4%)	18,680 (-0.2%)	18,650 (-0.3%)	18,786 (0.4%)
	Below Normal	9,347	9,343 (0%)	9,342 (-0.1%)	9,343 (0%)	9,342 (-0.1%)
	Dry	8,648	8,761 (1.3%)	8,660 (0.1%)	8,762 (1.3%)	8,916 (3.1%)
	Critically Dry	6,189	6,169 (-0.3%)	6,367 (2.9%)	6,198 (0.1%)	6,407 (3.5%)
	All	14,604	14,720 (0.8%)	14,711 (0.7%)	14,727 (0.8%)	14,774 (1.2%)
April	Wet	14,117	14,115 (0%)	14,084 (-0.2%)	14,115 (0%)	14,006 (-0.8%)
	Above Normal	10,431	10,483 (0.5%)	10,496 (0.6%)	10,482 (0.5%)	10,540 (1%)

Month	Water Year Type[1]	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
	Below Normal	7,437	7,428 (-0.1%)	7,428 (-0.1%)	7,427 (-0.1%)	7,363 (-1%)
	Dry	6,577	6,611 (0.5%)	6,377 (-3.1%)	6,611 (0.5%)	6,446 (-2%)
	Critically Dry	5,587	5,295 (-5.2%)	5,314 (-4.9%)	5,475 (-2%)	5,488 (-1.8%)
	All	9,542	9,511 (-0.3%)	9,454 (-0.9%)	9,537 (-0.1%)	9,462 (-0.8%)
May	Wet	12,590	12,581 (-0.1%)	12,761 (1.4%)	12,581 (-0.1%)	12,759 (1.3%)
	Above Normal	11,038	11,041 (0%)	11,041 (0%)	11,041 (0%)	11,162 (1.1%)
	Below Normal	9,579	9,565 (-0.2%)	9,341 (-2.5%)	9,564 (-0.2%)	9,299 (-2.9%)
	Dry	9,268	9,328 (0.6%)	9,073 (-2.1%)	9,328 (0.7%)	8,813 (-4.9%)
	Critically Dry	8,813	8,607 (-2.3%)	8,564 (-2.8%)	8,524 (-3.3%)	8,319 (-5.6%)
	All	10,568	10,546 (-0.2%)	10,502 (-0.6%)	10,534 (-0.3%)	10,416 (-1.4%)
June	Wet	10,777	10,794 (0.2%)	10,741 (-0.3%)	10,794 (0.2%)	10,725 (-0.5%)
	Above Normal	11,313	11,299 (-0.1%)	10,656 (-5.8%)	11,299 (-0.1%)	10,470 (-7.4%)
	Below Normal	11,241	11,253 (0.1%)	10,942 (-2.7%)	11,251 (0.1%)	10,170 (-9.5%)
	Dry	11,876	11,886 (0.1%)	11,817 (-0.5%)	11,886 (0.1%)	11,304 (-4.8%)
	Critically Dry	10,137	9,773 (-3.6%)	9,654 (-4.8%)	9,777 (-3.6%)	9,583 (-5.5%)
	All	11,075	11,030 (-0.4%)	10,844 (-2.1%)	11,030 (-0.4%)	10,552 (-4.7%)
July	Wet	13,573	13,574 (0%)	13,548 (-0.2%)	13,574 (0%)	13,564 (-0.1%)
	Above Normal	14,579	14,447 (-0.9%)	14,287 (-2%)	14,382 (-1.3%)	13,285 (-8.9%)
	Below Normal	13,680	13,514 (-1.2%)	13,620 (-0.4%)	13,529 (-1.1%)	13,166 (-3.8%)
	Dry	12,699	12,702 (0%)	12,678 (-0.2%)	12,703 (0%)	12,494 (-1.6%)
	Critically Dry	10,284	10,293 (0.1%)	10,207 (-0.7%)	10,266 (-0.2%)	10,063 (-2.2%)
	All	13,042	12,998 (-0.3%)	12,972 (-0.5%)	12,989 (-0.4%)	12,710 (-2.5%)
August	Wet	11,651	11,648 (0%)	11,599 (-0.4%)	11,648 (0%)	11,596 (-0.5%)
	Above Normal	10,929	10,681 (-2.3%)	10,820 (-1%)	10,658 (-2.5%)	9,986 (-8.6%)
	Below Normal	10,077	9,825 (-2.5%)	9,948 (-1.3%)	9,845 (-2.3%)	9,618 (-4.6%)
	Dry	9,484	9,456 (-0.3%)	9,446 (-0.4%)	9,456 (-0.3%)	9,356 (-1.3%)
	Critically Dry	8,160	8,662 (6.1%)	8,609 (5.5%)	8,645 (5.9%)	8,596 (5.3%)
	All	10,288	10,278 (-0.1%)	10,292 (0%)	10,277 (-0.1%)	10,107 (-1.8%)
September	Wet	10,890	10,891 (0%)	10,800 (-0.8%)	10,891 (0%)	10,814 (-0.7%)
	Above Normal	9,421	9,190 (-2.4%)	9,453 (0.3%)	9,146 (-2.9%)	9,652 (2.5%)
	Below Normal	6,283	6,156 (-2%)	6,247 (-0.6%)	6,154 (-2%)	6,153 (-2.1%)
	Dry	5,315	5,323 (0.1%)	5,344 (0.6%)	5,323 (0.1%)	5,306 (-0.2%)
	Critically Dry	4,860	5,201 (7%)	5,201 (7%)	5,178 (6.5%)	5,116 (5.3%)
	All	7,762	7,762 (0%)	7,786 (0.3%)	7,753 (-0.1%)	7,777 (0.2%)
October	Wet	7,700	7,785 (1.1%)	7,827 (1.7%)	7,788 (1.1%)	8,172 (6.1%)
	Above Normal	6,964	6,946 (-0.3%)	6,988 (0.3%)	7,016 (0.8%)	7,213 (3.6%)
	Below Normal	7,039	6,992 (-0.7%)	7,051 (0.2%)	6,996 (-0.6%)	7,228 (2.7%)
	Dry	6,601	6,616 (0.2%)	6,682 (1.2%)	6,615 (0.2%)	6,830 (3.5%)
	Critically Dry	6,308	6,301 (-0.1%)	6,305 (0%)	6,279 (-0.5%)	6,398 (1.4%)
	All	7,045	7,064 (0.3%)	7,109 (0.9%)	7,071 (0.4%)	7,328 (4%)

Month	Water Year Type[1]	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
November	Wet	10,408	10,377 (-0.3%)	10,428 (0.2%)	10,370 (-0.4%)	10,733 (3.1%)
	Above Normal	7,547	7,746 (2.6%)	7,758 (2.8%)	7,750 (2.7%)	7,900 (4.7%)
	Below Normal	7,216	7,261 (0.6%)	7,262 (0.6%)	7,267 (0.7%)	7,310 (1.3%)
	Dry	7,842	7,877 (0.5%)	8,029 (2.4%)	7,877 (0.4%)	8,036 (2.5%)
	Critically Dry	6,270	6,262 (-0.1%)	6,234 (-0.6%)	6,216 (-0.9%)	6,596 (5.2%)
	All	8,306	8,335 (0.4%)	8,383 (0.9%)	8,328 (0.3%)	8,564 (3.1%)
December	Wet	20,917	21,095 (0.9%)	21,231 (1.5%)	21,143 (1.1%)	21,505 (2.8%)
	Above Normal	10,156	10,006 (-1.5%)	10,059 (-1%)	10,005 (-1.5%)	10,146 (-0.1%)
	Below Normal	8,359	8,379 (0.2%)	8,386 (0.3%)	8,377 (0.2%)	8,354 (-0.1%)
	Dry	7,264	7,365 (1.4%)	7,400 (1.9%)	7,363 (1.4%)	7,692 (5.9%)
	Critically Dry	5,854	5,766 (-1.5%)	5,782 (-1.2%)	5,766 (-1.5%)	5,794 (-1%)
	All	12,106	12,159 (0.4%)	12,222 (1%)	12,174 (0.6%)	12,383 (2.3%)

[1] Water year type sorting is by hydrologic water year.

Table 11-58. CALSIM II Monthly Average Flow (cfs) by Month and Water Year Type Below Red Bluff Diversion Dam for the NAA and Alternatives 1A, 1B, 2, and 3, and Percent Differences between Them (in Parentheses).

Month	Water Year Type[1]	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
January	Wet	28,417	27,478 (-3.3%)	27,358 (-3.7%)	27,488 (-3.3%)	27,461 (-3.4%)
	Above Normal	17,520	16,099 (-8.1%)	16,096 (-8.1%)	16,099 (-8.1%)	16,258 (-7.2%)
	Below Normal	9,135	8,621 (-5.6%)	8,653 (-5.3%)	8,582 (-6.1%)	8,575 (-6.1%)
	Dry	6,902	6,724 (-2.6%)	6,722 (-2.6%)	6,724 (-2.6%)	6,726 (-2.6%)
	Critically Dry	6,114	5,906 (-3.4%)	5,911 (-3.3%)	5,913 (-3.3%)	5,941 (-2.8%)
	All	15,574	14,928 (-4.1%)	14,894 (-4.4%)	14,925 (-4.2%)	14,939 (-4.1%)
February	Wet	32,044	30,779 (-3.9%)	30,914 (-3.5%)	30,891 (-3.6%)	30,675 (-4.3%)
	Above Normal	25,815	24,682 (-4.4%)	24,853 (-3.7%)	24,607 (-4.7%)	25,207 (-2.4%)
	Below Normal	13,179	12,439 (-5.6%)	12,477 (-5.3%)	12,383 (-6%)	12,640 (-4.1%)
	Dry	9,070	8,486 (-6.4%)	8,537 (-5.9%)	8,486 (-6.4%)	8,514 (-6.1%)
	Critically Dry	6,501	6,353 (-2.3%)	6,370 (-2%)	6,359 (-2.2%)	6,352 (-2.3%)
	All	19,053	18,212 (-4.4%)	18,298 (-4%)	18,231 (-4.3%)	18,285 (-4%)
March	Wet	23,997	23,768 (-1%)	23,669 (-1.4%)	23,890 (-0.4%)	23,500 (-2.1%)
	Above Normal	18,968	17,757 (-6.4%)	17,810 (-6.1%)	17,779 (-6.3%)	17,880 (-5.7%)
	Below Normal	9,428	8,839 (-6.2%)	8,839 (-6.3%)	8,840 (-6.2%)	8,701 (-7.7%)
	Dry	8,751	8,119 (-7.2%)	8,084 (-7.6%)	8,119 (-7.2%)	8,239 (-5.8%)
	Critically Dry	6,236	6,010 (-3.6%)	6,207 (-0.5%)	6,039 (-3.2%)	6,247 (0.2%)
	All	14,773	14,270 (-3.4%)	14,265 (-3.4%)	14,317 (-3.1%)	14,233 (-3.7%)
April	Wet	14,113	13,819 (-2.1%)	13,789 (-2.3%)	13,819 (-2.1%)	13,667 (-3.2%)
	Above Normal	10,324	10,186 (-1.3%)	10,199 (-1.2%)	10,186 (-1.3%)	10,243 (-0.8%)

Month	Water Year Type[1]	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
	Below Normal	7,198	7,094 (-1.4%)	7,115 (-1.1%)	7,094 (-1.4%)	7,050 (-2.1%)
	Dry	6,401	6,432 (0.5%)	6,245 (-2.4%)	6,432 (0.5%)	6,311 (-1.4%)
	Critically Dry	5,565	5,272 (-5.3%)	5,290 (-4.9%)	5,452 (-2%)	5,468 (-1.7%)
	All	9,442	9,274 (-1.8%)	9,231 (-2.2%)	9,300 (-1.5%)	9,224 (-2.3%)
May	Wet	11,967	11,840 (-1.1%)	12,019 (0.4%)	11,840 (-1.1%)	12,017 (0.4%)
	Above Normal	10,364	10,367 (0%)	10,367 (0%)	10,367 (0%)	10,488 (1.2%)
	Below Normal	8,943	8,929 (-0.2%)	8,820 (-1.4%)	8,929 (-0.2%)	8,793 (-1.7%)
	Dry	8,771	8,822 (0.6%)	8,736 (-0.4%)	8,822 (0.6%)	8,502 (-3.1%)
	Critically Dry	8,734	8,526 (-2.4%)	8,512 (-2.5%)	8,444 (-3.3%)	8,262 (-5.4%)
	All	10,044	9,980 (-0.6%)	9,998 (-0.4%)	9,968 (-0.8%)	9,920 (-1.2%)
June	Wet	9,684	9,700 (0.2%)	9,670 (-0.1%)	9,700 (0.2%)	9,653 (-0.3%)
	Above Normal	10,164	10,151 (-0.1%)	9,830 (-3.3%)	10,151 (-0.1%)	9,644 (-5.1%)
	Below Normal	10,254	10,267 (0.1%)	10,163 (-0.9%)	10,267 (0.1%)	9,586 (-6.5%)
	Dry	11,123	1,1122 (0%)	11,178 (0.5%)	11,122 (0%)	10,817 (-2.8%)
	Critically Dry	10,013	9,648 (-3.6%)	9,538 (-4.7%)	9,651 (-3.6%)	9,458 (-5.5%)
	All	10,211	10,163 (-0.5%)	10,091 (-1.2%)	10,164 (-0.5%)	9,866 (-3.4%)
July	Wet	12,298	12,301 (0%)	12,281 (-0.1%)	12,301 (0%)	12,321 (0.2%)
	Above Normal	13,258	13,143 (-0.9%)	13,312 (0.4%)	13,078 (-1.4%)	12,704 (-4.2%)
	Below Normal	12,529	12,368 (-1.3%)	12,500 (-0.2%)	12,385 (-1.1%)	12,288 (-1.9%)
	Dry	11,844	11,836 (-0.1%)	11,811 (-0.3%)	11,836 (-0.1%)	11,728 (-1%)
	Critically Dry	10,140	10,147 (0.1%)	10,075 (-0.6%)	10,119 (-0.2%)	9,921 (-2.2%)
	All	12,042	11,999 (-0.4%)	12,021 (-0.2%)	11,990 (-0.4%)	11,881 (-1.3%)
August	Wet	10,641	10,655 (0.1%)	10,605 (-0.3%)	10,655 (0.1%)	10,605 (-0.3%)
	Above Normal	9,899	9,673 (-2.3%)	9,815 (-0.9%)	9,650 (-2.5%)	9,377 (-5.3%)
	Below Normal	9,168	8,922 (-2.7%)	9,043 (-1.4%)	8,943 (-2.4%)	8,883 (-3.1%)
	Dry	8,805	8,769 (-0.4%)	8,758 (-0.5%)	8,768 (-0.4%)	8,690 (-1.3%)
	Critically Dry	8,047	8,547 (6.2%)	8,491 (5.5%)	8,530 (6%)	8,480 (5.4%)
	All	9,498	9,496 (0%)	9,508 (0.1%)	9,494 (0%)	9,409 (-0.9%)
September	Wet	10,641	10,635 (-0.1%)	10,546 (-0.9%)	10,636 (0%)	10,560 (-0.8%)
	Above Normal	9,151	8,920 (-2.5%)	9,185 (0.4%)	8,876 (-3%)	9,472 (3.5%)
	Below Normal	6,105	5,970 (-2.2%)	6,060 (-0.7%)	5,969 (-2.2%)	5,972 (-2.2%)
	Dry	5,139	5,141 (0%)	5,167 (0.5%)	5,141 (0%)	5,118 (-0.4%)
	Critically Dry	4,817	5,148 (6.9%)	5,147 (6.8%)	5,125 (6.4%)	5,057 (5%)
	All	7,569	7,564 (-0.1%)	7,588 (0.3%)	7,555 (-0.2%)	7,588 (0.2%)
October	Wet	7,593	7,656 (0.8%)	7,698 (1.4%)	7,662 (0.9%)	8,038 (5.9%)
	Above Normal	6,859	6,852 (-0.1%)	6,894 (0.5%)	6,922 (0.9%)	7,121 (3.8%)
	Below Normal	6,894	6,871 (-0.3%)	6,929 (0.5%)	6,870 (-0.4%)	7,102 (3%)

Month	Water Year Type[1]	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
	Dry	6,475	6,500 (0.4%)	6,561 (1.3%)	6,498 (0.4%)	6,704 (3.5%)
	Critically Dry	6,216	6,227 (0.2%)	6,233 (0.3%)	6,204 (-0.2%)	6,319 (1.7%)
	All	6,929	6,952 (0.3%)	6,995 (1%)	6,958 (0.4%)	7,211 (4.1%)
November	Wet	10,489	10,120 (-3.5%)	10,148 (-3.2%)	10,114 (-3.6%)	10,453 (-0.3%)
	Above Normal	7,588	7,768 (2.4%)	7,780 (2.5%)	7,772 (2.4%)	7,882 (3.9%)
	Below Normal	7,238	7,201 (-0.5%)	7,202 (-0.5%)	7,207 (-0.4%)	7,246 (0.1%)
	Dry	7,876	7,710 (-2.1%)	7,850 (-0.3%)	7,712 (-2.1%)	7,857 (-0.2%)
	Critically Dry	6,271	6,265 (-0.1%)	6,236 (-0.5%)	6,219 (-0.8%)	6,598 (5.2%)
	All	8,349	8,206 (-1.7%)	8,243 (-1.3%)	8,199 (-1.8%)	8,419 (0.8%)
December	Wet	21,272	20,141 (-5.3%)	20,230 (-4.9%)	20,178 (-5.1%)	20,475 (-3.7%)
	Above Normal	10,315	9,814 (-4.9%)	9,866 (-4.4%)	9,813 (-4.9%)	10,048 (-2.6%)
	Below Normal	8,468	8,297 (-2%)	8,303 (-2%)	8,296 (-2%)	8,271 (-2.3%)
	Dry	7,375	7,258 (-1.6%)	7,323 (-0.7%)	7,257 (-1.6%)	7,539 (2.2%)
	Critically Dry	5,911	5,824 (-1.5%)	5,839 (-1.2%)	5,824 (-1.5%)	5,852 (-1%)
	All	12,295	11,792 (-4.1%)	11,845 (-3.7%)	11,803 (-4%)	11,991 (-2.5%)

[1] Water year type sorting is by hydrologic water year.

Table 11-59. CALSIM II Monthly Average Flow (cfs) by Month and Water Year Type at Hamilton City for the NAA and Alternatives 1A, 1B, 2, and 3, and Percent Differences between Them (in Parentheses).

Month	Water Year Type[1]	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
January	Wet	32,013	30,706 (-4.1%)	30,549 (-4.6%)	30,731 (-4%)	30,585 (-4.5%)
	Above Normal	20,397	18,533 (-9.1%)	18,472 (-9.4%)	18,541 (-9.1%)	18,580 (-8.9%)
	Below Normal	10,428	9,831 (-5.7%)	9,821 (-5.8%)	9,792 (-6.1%)	9,740 (-6.6%)
	Dry	7,704	7,458 (-3.2%)	7,456 (-3.2%)	7,458 (-3.2%)	7,460 (-3.2%)
	Critically Dry	6,689	6,437 (-3.8%)	6,442 (-3.7%)	6,445 (-3.7%)	6,473 (-3.2%)
	All	17,606	16,748 (-4.9%)	16,687 (-5.2%)	16,751 (-4.9%)	16,703 (-5.1%)
February	Wet	35,712	34,006 (-4.8%)	34,109 (-4.5%)	34,216 (-4.2%)	33,715 (-5.6%)
	Above Normal	29,051	27,474 (-5.4%)	27,645 (-4.8%)	27,399 (-5.7%)	27,866 (-4.1%)
	Below Normal	14,949	14,068 (-5.9%)	14,106 (-5.6%)	14,013 (-6.3%)	14,269 (-4.6%)
	Dry	10,465	9,727 (-7.1%)	9,777 (-6.6%)	9,727 (-7.1%)	9,754 (-6.8%)
	Critically Dry	7,297	7,147 (-2.1%)	7,164 (-1.8%)	7,154 (-2%)	7,147 (-2.1%)
	All	21,401	20,302 (-5.1%)	20,377 (-4.8%)	20,353 (-4.9%)	20,297 (-5.2%)
March	Wet	26,997	26,576 (-1.6%)	26,448 (-2%)	26,713 (-1.1%)	26,201 (-2.9%)
	Above Normal	21,852	20,039 (-8.3%)	20,091 (-8.1%)	20,061 (-8.2%)	20,161 (-7.7%)
	Below Normal	10,903	10,219 (-6.3%)	10,171 (-6.7%)	10,232 (-6.2%)	9,927 (-9%)
	Dry	10,338	9,585 (-7.3%)	9,548 (-7.6%)	9,586 (-7.3%)	9,677 (-6.4%)

Month	Water Year Type[1]	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
	Critically Dry	7,125	6,867 (-3.6%)	7,064 (-0.9%)	6,895 (-3.2%)	7,103 (-0.3%)
	All	16,861	16,173 (-4.1%)	16,149 (-4.2%)	16,227 (-3.8%)	16,066 (-4.7%)
April	Wet	16,312	15,660 (-4%)	15,630 (-4.2%)	15,661 (-4%)	15,441 (-5.3%)
	Above Normal	12,134	11,808 (-2.7%)	11,821 (-2.6%)	11,808 (-2.7%)	11,865 (-2.2%)
	Below Normal	8,420	8,188 (-2.8%)	8,209 (-2.5%)	8,188 (-2.8%)	8,144 (-3.3%)
	Dry	7,130	7,161 (0.4%)	7,000 (-1.8%)	7,161 (0.4%)	7,066 (-0.9%)
	Critically Dry	5,670	5,379 (-5.1%)	5,397 (-4.8%)	5,558 (-2%)	5,596 (-1.3%)
	All	10,786	10,453 (-3.1%)	10,416 (-3.4%)	10,480 (-2.8%)	10,391 (-3.7%)
May	Wet	11,888	11,654 (-2%)	11,840 (-0.4%)	11,656 (-2%)	11,840 (-0.4%)
	Above Normal	9,705	9,708 (0%)	9,708 (0%)	9,708 (0%)	9,829 (1.3%)
	Below Normal	7,791	7,778 (-0.2%)	7,865 (1%)	7,778 (-0.2%)	7,875 (1.1%)
	Dry	7,239	7,289 (0.7%)	7,381 (2%)	7,289 (0.7%)	7,274 (0.5%)
	Critically Dry	7,004	6,814 (-2.7%)	6,815 (-2.7%)	6,731 (-3.9%)	6,736 (-3.8%)
	All	9,137	9,041 (-1.1%)	9,139 (0%)	9,030 (-1.2%)	9,120 (-0.2%)
June	Wet	8,371	8,379 (0.1%)	8,352 (-0.2%)	8,379 (0.1%)	8,354 (-0.2%)
	Above Normal	8,467	8,394 (-0.9%)	8,169 (-3.5%)	8,394 (-0.9%)	8,170 (-3.5%)
	Below Normal	8,111	8,130 (0.2%)	8,091 (-0.2%)	8,130 (0.2%)	7,944 (-2.1%)
	Dry	8,668	8,700 (0.4%)	8,755 (1%)	8,698 (0.3%)	8,826 (1.8%)
	Critically Dry	7,743	7,690 (-0.7%)	7,634 (-1.4%)	7,693 (-0.6%)	7,517 (-2.9%)
	All	8,308	8,305 (0%)	8,265 (-0.5%)	8,305 (0%)	8,238 (-0.8%)
July	Wet	9,979	9,992 (0.1%)	9,969 (-0.1%)	9,992 (0.1%)	10,048 (0.7%)
	Above Normal	10,841	10,774 (-0.6%)	10,973 (1.2%)	10,709 (-1.2%)	11,294 (4.2%)
	Below Normal	9,955	9,808 (-1.5%)	9,953 (0%)	9,826 (-1.3%)	10,132 (1.8%)
	Dry	9,351	9,752 (4.3%)	9,713 (3.9%)	9,746 (4.2%)	9,750 (4.3%)
	Critically Dry	7,882	8,341 (5.8%)	8,291 (5.2%)	8,308 (5.4%)	8,186 (3.9%)
	All	9,635	9,759 (1.3%)	9,787 (1.6%)	9,749 (1.2%)	9,877 (2.5%)
August	Wet	8,568	8,580 (0.2%)	8,537 (-0.4%)	8,580 (0.2%)	8,538 (-0.4%)
	Above Normal	7,950	7,789 (-2%)	7,882 (-0.9%)	7,776 (-2.2%)	7,871 (-1%)
	Below Normal	7,147	7,050 (-1.4%)	7,110 (-0.5%)	7,072 (-1.1%)	7,143 (-0.1%)
	Dry	6,945	7,463 (7.5%)	7,338 (5.7%)	7,453 (7.3%)	7,247 (4.3%)
	Critically Dry	6,266	7,164 (14.3%)	7,107 (13.4%)	7,059 (12.6%)	6,855 (9.4%)
	All	7,540	7,751 (2.8%)	7,724 (2.4%)	7,736 (2.6%)	7,672 (1.8%)
September	Wet	10,163	10,125 (-0.4%)	9,979 (-1.8%)	10,125 (-0.4%)	9,993 (-1.7%)
	Above Normal	8,618	8,434 (-2.1%)	8,652 (0.4%)	8,431 (-2.2%)	9,027 (4.8%)
	Below Normal	5,611	5,401 (-3.7%)	5,484 (-2.3%)	5,400 (-3.8%)	5,414 (-3.5%)
	Dry	4,622	4,797 (3.8%)	4,826 (4.4%)	4,794 (3.7%)	4,763 (3%)
	Critically Dry	4,308	4,741 (10.1%)	4,713 (9.4%)	4,700 (9.1%)	4,588 (6.5%)

Month	Water Year Type[1]	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
	All	7,069	7,097 (0.4%)	7,093 (0.3%)	7,090 (0.3%)	7,099 (0.4%)
October	Wet	7,421	7,415 (-0.1%)	7,456 (0.5%)	7,426 (0.1%)	7,757 (4.5%)
	Above Normal	6,563	6,592 (0.5%)	6,634 (1.1%)	6,663 (1.5%)	6,836 (4.2%)
	Below Normal	6,557	6,582 (0.4%)	6,639 (1.3%)	6,575 (0.3%)	6,807 (3.8%)
	Dry	6,130	6,173 (0.7%)	6,228 (1.6%)	6,169 (0.6%)	6,368 (3.9%)
	Critically Dry	5,880	5,900 (0.4%)	5,905 (0.4%)	5,881 (0%)	5,983 (1.8%)
	All	6,649	6,668 (0.3%)	6,710 (0.9%)	6,675 (0.4%)	6,907 (3.9%)
November	Wet	10,937	10,477 (-4.2%)	10,500 (-4%)	10,472 (-4.3%)	10,810 (-1.2%)
	Above Normal	7,518	7,740 (2.9%)	7,742 (3%)	7,743 (3%)	7,833 (4.2%)
	Below Normal	7,010	7,038 (0.4%)	7,012 (0%)	7,024 (0.2%)	7,026 (0.2%)
	Dry	7,724	7,575 (-1.9%)	7,681 (-0.6%)	7,577 (-1.9%)	7,720 (0%)
	Critically Dry	5,850	5,891 (0.7%)	5,852 (0%)	5,845 (-0.1%)	6,188 (5.8%)
	All	8,352	8,206 (-1.7%)	8,227 (-1.5%)	8,196 (-1.9%)	8,400 (0.6%)
December	Wet	23,825	22,439 (-5.8%)	22,488 (-5.6%)	22,489 (-5.6%)	22,612 (-5.1%)
	Above Normal	11,383	10,771 (-5.4%)	10,823 (-4.9%)	10,770 (-5.4%)	11,134 (-2.2%)
	Below Normal	9,096	8,866 (-2.5%)	8,871 (-2.5%)	8,864 (-2.5%)	8,839 (-2.8%)
	Dry	8,018	7,827 (-2.4%)	7,892 (-1.6%)	7,825 (-2.4%)	8,107 (1.1%)
	Critically Dry	6,222	6,141 (-1.3%)	6,157 (-1%)	6,141 (-1.3%)	6,171 (-0.8%)
	All	13,568	12,941 (-4.6%)	12,980 (-4.3%)	12,956 (-4.5%)	13,103 (-3.4%)

[1] Water year type sorting is by hydrologic water year.

Larval and Juvenile Rearing and Emigration

According to field observations, green sturgeon larvae begin to disperse from hatching areas about 18 days post hatch (dph) and dispersion is complete about 35 dph (Heublein et al. 2017). The green sturgeon spawning period is from March through July, so the larval period is considered to be April through September. The downstream distribution of green sturgeon larvae in the Sacramento River is uncertain, but is estimated to extend to the Colusa area, at RM 157 (Heublein et al. 2017). The upstream limit is the Cow Creek confluence.

The green sturgeon juvenile stage begins when metamorphosis of the larva is complete, typically about 45 dph and at about 75 mm in length (Heublein et al 2017). It is likely that juveniles rear near spawning habitat for a few months or more before migrating to the Delta (Heublein et al. 2017). The juveniles rear in the Sacramento River from about May through December (National Marine Fisheries Service 2017:Appendix B). During most of this period, the juveniles are likely to be found anywhere from the upstream spawning habitat near the Cow Creek confluence to the Delta.

The effects of flow on green sturgeon larvae and juveniles are poorly understood. There appears to be a positive relationship between annual outflow and abundance in rotary screw traps at

RBDD[24] of green sturgeon larvae and juveniles (Heublein et al. 2017). Additionally, as noted by NMFS (2018b:12), there are correlations between abundance of juvenile white sturgeon and Delta outflow, which have previously been used to infer potential effects on green sturgeon (ICF International 2016:5-197–5-205). More recently, based on 7 years of netting in the Delta (2015–2021), CDFW found that CPUE of yearling juvenile green sturgeon was about 10 times as high in years following Wet Water Years as in years following Dry and Critically Dry Water Years (Beccio pers. comm.). The causes for these relationships between flow and abundance of young sturgeon are uncertain, but they may result from flows transporting larvae to areas with greater food availability, dispersing larvae over a wider area, and/or enhancing nutrient availability to the Sacramento River and Delta/San Francisco Estuary.

CALSIM II modeling results for the Sacramento River at Bend Bridge, below RBPP and at Hamilton City (Table 11-57 through Table 11-59) indicate that mean monthly flows during the April through September period of larval rearing are generally similar between the NAA and Alternatives 1, 2, and 3, except for moderately lower mean flows during April through July, especially under Alternative 3, and moderately higher flows in August and September of Critically Dry Water Years under Alternatives 1, 2, and 3. At Wilkins Slough (RM 117), the presumed downstream limit of the green sturgeon larval distribution in the Sacramento River, a similar pattern of flow differences between the NAA and Alternatives 1, 2, and 3 is evident, except that there are larger flow increases in July and August, especially in Dry and Critically Dry Water Years (Table 11-60). The July through September increases in flow have the potential to benefit green sturgeon larvae because, as discussed above, there appears to be a positive relationship between annual outflow and abundance of green sturgeon larvae and juveniles, but this conclusion is uncertain and additional studies are needed.

Table 11-60. CALSIM II Monthly Average Flow (cfs) by Month and Water Year Type at Wilkins Slough for the NAA and Alternatives 1A, 1B, 2, and 3, and Percent Differences between Them (in Parentheses).

Month	Water Year Type[1]	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
January	Wet	21,368	21,011 (-1.7%)	20,914 (-2.1%)	21,015 (-1.7%)	20,972 (-1.9%)
	Above Normal	19,059	18,593 (-2.4%)	18,581 (-2.5%)	18,598 (-2.4%)	18,648 (-2.2%)
	Below Normal	11,949	11,558 (-3.3%)	11,571 (-3.2%)	11,528 (-3.5%)	11,497 (-3.8%)
	Dry	8,717	8,611 (-1.2%)	8,611 (-1.2%)	8,611 (-1.2%)	8,609 (-1.2%)
	Critically Dry	7,769	7,525 (-3.1%)	7,527 (-3.1%)	7,531 (-3.1%)	7,566 (-2.6%)
	All	14,596	14,291 (-2.1%)	14,261 (-2.3%)	14,289 (-2.1%)	14,280 (-2.2%)
February	Wet	22,130	21,911 (-1%)	21,988 (-0.6%)	21,933 (-0.9%)	21,902 (-1%)
	Above Normal	22,257	21,715 (-2.4%)	21,743 (-2.3%)	21,684 (-2.6%)	21,562 (-3.1%)
	Below Normal	15,815	15,406 (-2.6%)	15,384 (-2.7%)	15,376 (-2.8%)	15,461 (-2.2%)
	Dry	12,114	11,732 (-3.2%)	11,784 (-2.7%)	11,731 (-3.2%)	11,766 (-2.9%)
	Critically Dry	8,347	8,263 (-1%)	8,281 (-0.8%)	8,268 (-0.9%)	8,265 (-1%)
	All	16,775	16,466 (-1.8%)	16,504 (-1.6%)	16,464 (-1.9%)	16,462 (-1.9%)

[24] The RBDD, which was decommissioned in 2013, and the RBPP are co-located, and the names may be used interchangeably when referring to geographic locations

Month	Water Year Type[1]	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
March	Wet	19,645	19,406 (-1.2%)	19,327 (-1.6%)	19,465 (-0.9%)	19,382 (-1.3%)
	Above Normal	19,753	19,218 (-2.7%)	19,232 (-2.6%)	19,225 (-2.7%)	19,250 (-2.5%)
	Below Normal	12,519	11,921 (-4.8%)	11,874 (-5.2%)	11,934 (-4.7%)	11,628 (-7.1%)
	Dry	11,926	11,390 (-4.5%)	11,354 (-4.8%)	11,390 (-4.5%)	11,476 (-3.8%)
	Critically Dry	8,323	8,066 (-3.1%)	8,263 (-0.7%)	8,094 (-2.8%)	8,305 (-0.2%)
	All	15,003	14,595 (-2.7%)	14,582 (-2.8%)	14,622 (-2.5%)	14,591 (-2.7%)
April	Wet	16,017	15,691 (-2%)	15,664 (-2.2%)	15,690 (-2%)	15,550 (-2.9%)
	Above Normal	13,727	13,631 (-0.7%)	13,645 (-0.6%)	13,630 (-0.7%)	13,686 (-0.3%)
	Below Normal	9,485	9,376 (-1.2%)	9,397 (-0.9%)	9,375 (-1.2%)	9,341 (-1.5%)
	Dry	7,552	7,591 (0.5%)	7,433 (-1.6%)	7,591 (0.5%)	7,499 (-0.7%)
	Critically Dry	5,741	5,434 (-5.3%)	5,443 (-5.2%)	5,616 (-2.2%)	5,644 (-1.7%)
	All	11,181	11,006 (-1.6%)	10,969 (-1.9%)	11,031 (-1.3%)	10,970 (-1.9%)
May	Wet	10,491	10,316 (-1.7%)	10,503 (0.1%)	10,319 (-1.6%)	10,508 (0.2%)
	Above Normal	8,306	8,297 (-0.1%)	8,297 (-0.1%)	8,297 (-0.1%)	8,417 (1.3%)
	Below Normal	5,911	5,892 (-0.3%)	5,978 (1.1%)	5,892 (-0.3%)	5,988 (1.3%)
	Dry	4,685	4,724 (0.8%)	4,824 (3%)	4,724 (0.8%)	4,716 (0.7%)
	Critically Dry	4,460	4,265 (-4.4%)	4,273 (-4.2%)	4,176 (-6.4%)	4,185 (-6.2%)
	All	7,230	7,148 (-1.1%)	7,248 (0.3%)	7,136 (-1.3%)	7,230 (0%)
June	Wet	6,709	6,716 (0.1%)	6,678 (-0.5%)	6,717 (0.1%)	6,678 (-0.5%)
	Above Normal	6,148	6,066 (-1.3%)	5,841 (-5%)	6,066 (-1.3%)	5,838 (-5%)
	Below Normal	5,485	5,498 (0.2%)	5,456 (-0.5%)	5,498 (0.2%)	5,307 (-3.2%)
	Dry	5,654	5,681 (0.5%)	5,727 (1.3%)	5,679 (0.4%)	5,807 (2.7%)
	Critically Dry	4,918	4,869 (-1%)	4,815 (-2.1%)	4,880 (-0.8%)	4,707 (-4.3%)
	All	5,923	5,916 (-0.1%)	5,871 (-0.9%)	5,918 (-0.1%)	5,845 (-1.3%)
July	Wet	7,122	7,130 (0.1%)	7,112 (-0.1%)	7,130 (0.1%)	7,189 (0.9%)
	Above Normal	7,612	7,544 (-0.9%)	7,754 (1.9%)	7,479 (-1.8%)	8,078 (6.1%)
	Below Normal	6,768	6,617 (-2.2%)	6,765 (0%)	6,634 (-2%)	6,950 (2.7%)
	Dry	6,281	6,681 (6.4%)	6,640 (5.7%)	6,675 (6.3%)	6,672 (6.2%)
	Critically Dry	5,011	5,465 (9.1%)	5,420 (8.2%)	5,433 (8.4%)	5,322 (6.2%)
	All	6,624	6,744 (1.8%)	6,775 (2.3%)	6,733 (1.7%)	6,867 (3.7%)
August	Wet	6,140	6,150 (0.2%)	6,109 (-0.5%)	6,150 (0.2%)	6,106 (-0.6%)
	Above Normal	5,766	5,605 (-2.8%)	5,685 (-1.4%)	5,595 (-3%)	5,664 (-1.8%)
	Below Normal	4,725	4,633 (-2%)	4,686 (-0.8%)	4,653 (-1.5%)	4,708 (-0.4%)
	Dry	4,699	5,215 (11%)	5,093 (8.4%)	5,204 (10.7%)	4,997 (6.3%)
	Critically Dry	4,445	5,316 (19.6%)	5,261 (18.4%)	5,213 (17.3%)	5,012 (12.8%)
	All	5,271	5,479 (3.9%)	5,450 (3.4%)	5,464 (3.7%)	5,393 (2.3%)
September	Wet	10,187	10,148 (-0.4%)	10,005 (-1.8%)	10,149 (-0.4%)	10,020 (-1.6%)
	Above Normal	8,533	8,356 (-2.1%)	8,570 (0.4%)	8,352 (-2.1%)	8,952 (4.9%)

Month	Water Year Type[1]	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
	Below Normal	5,467	5,259 (-3.8%)	5,342 (-2.3%)	5,258 (-3.8%)	5,273 (-3.6%)
	Dry	4,435	4,600 (3.7%)	4,635 (4.5%)	4,598 (3.7%)	4,575 (3.1%)
	Critically Dry	4,051	4,451 (9.9%)	4,421 (9.1%)	4,414 (8.9%)	4,305 (6.3%)
	All	6,961	6,984 (0.3%)	6,981 (0.3%)	6,977 (0.2%)	6,990 (0.4%)
October	Wet	7,235	7,237 (0%)	7,276 (0.6%)	7,248 (0.2%)	7,582 (4.8%)
	Above Normal	5,942	5,970 (0.5%)	6,009 (1.1%)	6,040 (1.6%)	6,211 (4.5%)
	Below Normal	6,084	6,112 (0.5%)	6,173 (1.5%)	6,106 (0.4%)	6,327 (4%)
	Dry	5,593	5,644 (0.9%)	5,699 (1.9%)	5,639 (0.8%)	5,837 (4.4%)
	Critically Dry	5,493	5,522 (0.5%)	5,526 (0.6%)	5,502 (0.2%)	5,607 (2.1%)
	All	6,251	6,276 (0.4%)	6,317 (1.1%)	6,283 (0.5%)	6,513 (4.2%)
November	Wet	9,840	9,598 (-2.5%)	9,619 (-2.2%)	9,591 (-2.5%)	9,857 (0.2%)
	Above Normal	6,913	7,141 (3.3%)	7,140 (3.3%)	7,140 (3.3%)	7,217 (4.4%)
	Below Normal	6,653	6,684 (0.5%)	6,655 (0%)	6,671 (0.3%)	6,666 (0.2%)
	Dry	7,243	7,108 (-1.9%)	7,215 (-0.4%)	7,109 (-1.8%)	7,246 (0%)
	Critically Dry	5,042	5,088 (0.9%)	5,049 (0.1%)	5,042 (0%)	5,378 (6.7%)
	All	7,628	7,559 (-0.9%)	7,578 (-0.7%)	7,547 (-1.1%)	7,723 (1.2%)
December	Wet	19,374	18,756 (-3.2%)	18,819 (-2.9%)	18,786 (-3%)	18,926 (-2.3%)
	Above Normal	11,366	11,034 (-2.9%)	11,061 (-2.7%)	11,034 (-2.9%)	10,668 (-6.1%)
	Below Normal	8,734	8,680 (-0.6%)	8,689 (-0.5%)	8,679 (-0.6%)	8,663 (-0.8%)
	Dry	9,067	8,896 (-1.9%)	8,954 (-1.2%)	8,894 (-1.9%)	9,171 (1.1%)
	Critically Dry	6,453	6,374 (-1.2%)	6,392 (-0.9%)	6,376 (-1.2%)	6,391 (-1%)
	All	12,297	11,995 (-2.5%)	12,036 (-2.1%)	12,004 (-2.4%)	12,066 (-1.9%)

[1] Water year type sorting is by hydrologic water year.

During most of the green sturgeon juvenile rearing period, May through November, CALSIM II results are generally similar between the NAA and Alternatives 1, 2, and 3 for the Sacramento River at Bend Bridge, below RBPP, at Hamilton City, and at Wilkins Slough, except for some lower mean flows during May through July, especially under Alternative 3, and some higher flows in July, August, and September of Dry and Critically Dry Water Years under Alternatives 1, 2, and 3, particularly at Hamilton City and Wilkins Slough (Table 11-59 and Table 11-60). However, in December of all water year types, flow is consistently lower under Alternatives 1, 2, and 3 relative to the NAA below RBPP, at Hamilton City and at Wilkins Slough, but not at Bend Bridge.

Many of the April through December flow differences between the NAA and Alternatives 1, 2, and 3 occur in Dry and Critically Dry Water Years (Table 11-57 through Table 11-60). As suggested by results of the recent CDFW sampling discussed above and by the regression analyses of white sturgeon recruitment and Delta outflow detailed below, sturgeon recruitment occurs primarily in the wettest years. Therefore, the reductions and increases in flow, especially during Dry and Critically Dry Water Years, are expected to have little effect on the green sturgeon population. Flow is moderately lower at several of the locations under Alternatives 1, 2, and 3 during April of Wet Water Years, but the flow reductions are small relative to those that drive the relationships between flow and sturgeon recruitment described above. The differences in flow attributable to Alternatives 1, 2, and 3 are not expected to substantially affect rearing and emigration of green sturgeon larvae and juveniles in the Sacramento River.

Adult Migration and Holding

Green sturgeon adults enter the Sacramento River from the Delta as early as February and ultimately make their way upstream to spawn in deep pools from the GCID oxbow (near Hamilton City) to the Cow Creek confluence (Heublein et al. 2017). They spawn most years from March through July (Heublein et al. 2009). After spawning, the adults hold in the river for varying amounts of time, but typically emigrate back to the San Francisco Estuary and the ocean from about October through December (Heublein et al. 2017). Flow pulses during the late winter and spring months may provide important cues for green sturgeon adults to initiate their upstream spawning migrations (Heublein et al. 2009; National Marine Fisheries Service 2018b). Reductions in flow might reduce such migration cues. However, the relationship between flows and green sturgeon spawning migrations is currently too uncertain to make conclusions about such effects.

In addition to potentially affecting environmental cues for upstream spawning migrations, low flows potentially create difficult upstream passage conditions for green sturgeon (National Marine Fisheries Service 2018b). The level of flow in the Sacramento River needed by migrating adult sturgeon for unobstructed upstream passage is uncertain. In a radio tagging study of white sturgeon during spawning migrations in the Sacramento River, Schaffter (1997) noted that when flow at Colusa dropped below a level of 5,300 cfs, adults tended to cease upstream migrations or drifted downstream. Therefore, a low-flow threshold of 5,300 cfs downstream of the Colusa Weir was used in the analyses of potential Project effects on upstream passage of adult sturgeon, as discussed in Appendix 11N, Section 11N.2.4, *Low-Flow Passage Effects on Immigrating Salmon and Sturgeon Adults*. The results of these analyses indicated that Alternatives 1, 2, and 3 would have no effect on green sturgeon upstream passage (Appendix 11N, Table 11N-38).

Adult Upstream Passage at Fremont Weir

As described for winter-run Chinook salmon in Impact FISH-2, the number of days of adult passage at Fremont Weir was assessed using the number of days meeting adult passage criteria[25], based on data for water years 2009–2018 from the Daily Divertible Flow and Storage Tool Results (see Appendix 11P1). The results of this analysis indicated that the number of days meeting fish passage criteria at Fremont Weir generally would be similar with or without the Project (Table 11-21).

Adult Upstream Passage at Sutter Bypass Weirs

As with the Fremont Weir, changes in Sacramento River flow and flow entering Sutter Bypass as a result of the Project have the potential to change the number of days meeting adult sturgeon

[25] The criteria for days to be counted as Yolo Bypass passage days were river stage (1) between 21.14 feet and 29.92 feet during November 1–March 15, (2) between 21.14 feet and 23.35 feet during March 16–April 30, or (3) greater than 35 feet (elevation of the crest of the Fremont Weir). These criteria were based on the YBPASS tool (California Department of Water Resources 2017).

passage criteria at the three Sutter Bypass weirs (Moulton, Tisdale, and Colusa). This was assessed using the number of days meeting adult passage criteria[26], based on data for water years 2009–2018 from the Daily Divertible Flow and Storage Tool Results (see Appendix 11P1). The results of this analysis indicated that there would be no days meeting adult sturgeon passage criteria during 2009–2018 at Moulton or Tisdale Weirs. For Colusa Weir, the number of days meeting fish passage criteria generally would be similar with or without the Project (Table 11-61).

Table 11-61. Number of Days Meeting Adult Sturgeon Passage Criteria at Colusa Weir.

Water Year	Water Year Type	NAA	With Project*
2009	Dry	0	0 (0%)
2010	Below Normal	0	0 (0%)
2011	Wet	18	17 (-6%)
2012	Below Normal	0	0 (0%)
2013	Dry	0	0 (0%)
2014	Critically Dry	0	0 (0%)
2015	Critically Dry	0	0 (0%)
2016	Below Normal	2	0 (-100%)
2017	Wet	49	49 (0%)
2018	Below Normal	0	0 (0%)

Notes: Percentage values in parentheses indicate differences of alternatives compared to the NAA. *Results are the same for Alternatives 1, 2, and 3 because the same operational criteria are applied.

Sites Reservoir Release Effects

Sites Reservoir releases could temporally overlap with green sturgeon presence near the release location in the Sacramento River. Migrating adults would be present primarily during February through April and rearing or emigrating juveniles could be present year-round (Appendix 11A, Table 11A-10).

Sites Reservoir releases into the Yolo Bypass via the CBD would not overlap with the adult green sturgeon migration period, but would coincide with the juvenile rearing and emigration from August through October (Appendix 11A, Table 11A-10). No green sturgeon were collected during rescue efforts at Wallace Weir in 2014–2020 (Purdy and Kubo 2021).

Temperature Effects

As discussed in Chapter 6, the effects of Alternatives 1A, 1B, 2, and 3 on water temperatures at the Sites Reservoir release site in the Sacramento River would be relatively small with the releases generally tending to cause a slight reduction in water temperature (Tables 6-12a through

[26] The criteria for days to be counted as passage days were minimum depth of 5 feet and maximum velocity of 4 feet per second, during January–May. These criteria were based on long (>60 feet) criteria from Table 2-1 in the Tisdale Weir Rehabilitation and Fish Passage Project Draft EIR (California Department of Water Resources and Environmental Science Associates 2020).

6-12d). Therefore, temperature-related effects of Alternatives 1A, 1B, 2, and 3 on green sturgeon at the release site would be minimal.

All changes in water temperature in the Yolo Bypass due to Sites Reservoir releases via the CBD would be zero or negative for Alternatives 1A, 1B, 2, and 3 (Table 11-23). These lower water temperatures are not expected to have a lethal effect on juvenile green sturgeon in the Yolo Bypass. Lower temperatures could reduce the growth of juvenile green sturgeon somewhat due to reduced metabolism. However, the small reduction in water temperatures is expected to occur in a limited area of the Yolo Bypass before it equilibrates with atmospheric temperatures. As a result, temperature effects under Alternatives 1A, 1B, 2, and 3 in the Yolo Bypass due to Sites Reservoir releases would be minimal at a population level for green sturgeon.

As discussed in Impact FISH-2, temperature changes for Alternatives 1A, 1B, 2, and 3 due to Sites Reservoir releases in the Sacramento River below the Yolo Bypass, and effects on green sturgeon, would be minimal due to the small proportion of Sacramento River water in this reach coming off the Yolo Bypass.

Feather River

Far-Field Effects

Temperature Effects

Operation of Sites Reservoir has the potential to change water temperatures in the Feather River that could affect the life stages of green sturgeon present. As described in Appendix 11B, the two methods used to analyze temperature-related effects on green sturgeon in the Feather River were: (1) Physical Model Output Characterization; and (2) Water Temperature Index Value/Range Exceedance Analysis. More details on these methods are provided in Appendix 11B.

The Authority and Reclamation evaluated water temperature model outputs during the period of presence of each life stage of green sturgeon in the Feather River LFC below the Fish Barrier Dam and in the HFC at Gridley Bridge (see Appendix 11B, Table 11B-3 for timing). Visual observation of exceedance plots and differences in modeled mean monthly temperatures by water year type between Alternatives 1, 2, and 3 and the NAA in these locations indicates that water temperatures would be predominantly similar among alternatives during the period of presence of each life stage of green sturgeon (Appendix 6C, Tables 6C-16-1a to 6C-16-4c, Tables 6C-18-1a to 6C-18-4c; Figures 6C-16-1 to 6C-16-18, Figures 6C-18-1 to 6C-18-18). At all locations, mean monthly water temperatures for all months within all water year types under Alternatives 1A and 1B were within 0.5°F of the NAA. Water temperature modeling results for Alternatives 2 and 3 were similar to those of Alternative 1 at all locations. These results suggest that temperature-related effects on green sturgeon in the Feather River would be negligible.

Results of the analysis of exceedance above water temperature index values for green sturgeon in the Feather River from Appendix 11B, Table 11B-3 are presented in Appendix 11D, Table 11D-214 through Table 11D-225 and summarized in Table 11-62. For each life stage and at each location evaluated, there were no month and water year type combinations in which both: (1) the percent of months that exceeded the index values was more than 5% greater under Alternative 1A than under the NAA; and (2) the exceedance per month was more than 0.5°F greater under

Alternative 1A than under the NAA. Results of the exceedance analysis for Alternatives 1B, 2 and 3 are similar to Alternative 1A with no month and water year type combinations in which both criteria were met for any life stage at all locations. These results indicate that temperature-related effects on green sturgeon in the Feather River would be negligible.

Combined, these water temperature results indicate that the Project would cause inconsequential temperature-related effects on green sturgeon in the Feather River.

Table 11-62. Number of Month and Water Year Type Combinations that Satisfy Both Criteria for Being Biologically Meaningful in the Water Temperature Index Value Analysis, Green Sturgeon, Feather River[1,2,3]

Location	Non-Spawning Adult Presence				Spawning and Egg Incubation				Larval to Juvenile Rearing and Emigration			
	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3	Alt 1A	Alt 1B	Alt 2	Alt 3
Low-Flow Channel Below Fish Barrier Dam	0	0	0	0	0	0	0	0	0	0	0	0
High-Flow Channel Below Thermalito Afterbay Outlet	0	0	0	0	0	0	0	0	0	0	0	0
Gridley	0	0	0	0	0	0	0	0	0	0	0	0

[1] Biologically Meaningful Criteria include: (1) the difference in frequency of exceedance between the NAA and the alternative was greater than 5%, and (2) the difference in average monthly exceedance between the NAA and the alternative was greater than 0.5°F.
[2] Index values for each life stage are located in Appendix 11B, Table 11B-3.
[3] Full results presented in Appendix 11D, Table 11D-214 through Table 11D-225.

Flow Effects

Spawning and Egg Incubation

Green sturgeon irregularly spawn in the Feather River. Spawning was documented in 2011 at the Thermalito Afterbay outlet and in 2017 below the Fish Barrier Dam (National Marine Fisheries Service 2018b, Seesholtz et al. 2015). In both 2011 and 2017, water temperature was substantially cooler than average, likely due to the above average flow that occurred in the spring (Heublein et al. 2017). Green sturgeon may spawn in the Feather River only during wet, high-flow years (Heublein et al. 2017, Seesholtz et al. 2015). In most years, water temperatures downstream of the Thermalito Afterbay outlet are too warm for normal egg incubation by late May (Heublein et al. 2017). When green sturgeon spawn in the Feather River, spawning occurs in spring and early summer (Heublein et al. 2017, Seesholtz et al. 2015).

Feather River flow upstream of the Thermalito Afterbay outlet (the LFC) under Alternatives 1, 2, and 3 would be identical to that under the NAA. However, downstream of the Afterbay outlet (the HFC), the flows differ between Alternatives 1, 2, and 3. Feather River mean monthly flow under Alternatives 1, 2, and 3 is similar to or lower than flow under the NAA during April through June, when most green sturgeon spawning and egg incubation likely occurs (Seesholtz et al. 2015), except for 4% to 9% higher flows in April of Above Normal Water Years (Table 11-

62). Flow reductions under Alternatives 1, 2, and 3 are especially large during June of drier water years, including 10% to 16% reductions in Dry and Critically Dry Water Years. The optimal flow range for spawning habitat in the Feather River is not known. Because green sturgeon spawn in deep pools (Wyman et al. 2018), their eggs and embryos are protected from direct effects of flow reduction such as dewatering. Feather River spawning likely occurs only in wetter years, such as the 2 years when spawning was documented, so green sturgeon eggs are likely not present during the drier years in June when flow reductions under Alternatives 1, 2, and 3 are greatest. The flow increases in April occur during Above Normal Water Years and therefore potentially benefit green sturgeon adult migration and spawning (Table 11-63). Differences in the CALSIM II mean monthly flows during April through June of Wet Water Years are small (Table 11-63). Considering all the results, Alternatives 1, 2, and 3 are not expected to have a substantial effect with regard to flow on spawning and egg incubation of green sturgeon in the Feather River.

Table 11-63. CALSIM II Monthly Average Flow (cfs) by Month and Water Year Type in the Feather River at Thermalito Afterbay Outlet for the NAA and Alternatives 1A, 1B, 2, and 3, and Percent Differences between Them (in Parentheses).

Month	Water Year Type[1]	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
January	Wet	9,875	9,892 (0.2%)	9,901 (0.3%)	9,892 (0.2%)	9,916 (0.4%)
	Above Normal	1,630	1,630 (0%)	1,630 (0%)	1,630 (0%)	1,630 (0%)
	Below Normal	1,811	1,815 (0.2%)	1,739 (-3.9%)	1,815 (0.2%)	1,822 (0.6%)
	Dry	1,397	1,396 (0%)	1,397 (0%)	1,396 (0%)	1,397 (0%)
	Critically Dry	1,175	1,175 (0%)	1,175 (0%)	1,175 (0%)	1,175 (0%)
	All	4,260	4,266 (0.1%)	4,256 (-0.1%)	4,266 (0.1%)	4,276 (0.4%)
February	Wet	10,526	10,587 (0.6%)	10,582 (0.5%)	10,577 (0.5%)	10,606 (0.8%)
	Above Normal	3,826	3,797 (-0.7%)	3,761 (-1.7%)	3,814 (-0.3%)	3,829 (0.1%)
	Below Normal	2,911	2,911 (0%)	2,896 (-0.5%)	2,911 (0%)	2,911 (0%)
	Dry	1,534	1,533 (-0.1%)	1,533 (-0.1%)	1,533 (-0.1%)	1,533 (-0.1%)
	Critically Dry	1,370	1,370 (0%)	1,370 (0%)	1,370 (0%)	1,370 (0%)
	All	5,002	5,019 (0.3%)	5,010 (0.1%)	5,017 (0.3%)	5,029 (0.5%)
March	Wet	11,991	11,994 (0%)	11,996 (0%)	11,995 (0%)	12,005 (0.1%)
	Above Normal	5,551	5,577 (0.5%)	5,730 (3.2%)	5,608 (1%)	5,729 (3.2%)
	Below Normal	1,956	1,959 (0.2%)	1,925 (-1.6%)	1,959 (0.2%)	1,816 (-7.2%)
	Dry	1,481	1,484 (0.3%)	1,485 (0.3%)	1,484 (0.3%)	1,485 (0.3%)
	Critically Dry	1,579	1,577 (-0.1%)	1,576 (-0.2%)	1,577 (-0.1%)	1,573 (-0.4%)
	All	5,539	5,544 (0.1%)	5,558 (0.3%)	5,548 (0.2%)	5,540 (0%)
April	Wet	6,281	6,284 (0.1%)	6,284 (0.1%)	6,284 (0.1%)	6,284 (0.1%)
	Above Normal	2,103	2,266 (7.8%)	2,284 (8.6%)	2,239 (6.5%)	2,198 (4.5%)
	Below Normal	983	983 (0%)	983 (0%)	983 (0%)	983 (0%)
	Dry	966	961 (-0.5%)	961 (-0.6%)	961 (-0.5%)	960 (-0.7%)
	Critically Dry	1,042	1,040 (-0.2%)	1,039 (-0.2%)	1,040 (-0.2%)	1,038 (-0.3%)
	All	2,869	2,888 (0.7%)	2,890 (0.8%)	2,885 (0.6%)	2,880 (0.4%)
May	Wet	7,306	7,306 (0%)	7,308 (0%)	7,308 (0%)	7,308 (0%)

Month	Water Year Type[1]	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
	Above Normal	3,725	3,725 (0%)	3,725 (0%)	3,725 (0%)	3,725 (0%)
	Below Normal	1,510	1,513 (0.2%)	1,497 (-0.9%)	1,513 (0.2%)	1,521 (0.7%)
	Dry	1,926	1,949 (1.2%)	1,857 (-3.6%)	1,949 (1.2%)	1,813 (-5.9%)
	Critically Dry	1,846	1,841 (-0.3%)	1,829 (-0.9%)	1,844 (-0.1%)	1,824 (-1.2%)
	All	3,829	3,834 (0.1%)	3,810 (-0.5%)	3,835 (0.2%)	3,804 (-0.7%)
June	Wet	4,963	4,969 (0.1%)	4,963 (0%)	4,969 (0.1%)	4,951 (-0.2%)
	Above Normal	3,245	3,249 (0.1%)	3,240 (-0.1%)	3,249 (0.1%)	3,206 (-1.2%)
	Below Normal	2,714	2,628 (-3.2%)	2,582 (-4.9%)	2,627 (-3.2%)	2,630 (-3.1%)
	Dry	4,529	3,826 (-15.5%)	3,844 (-15.1%)	3,850 (-15%)	3,870 (-14.5%)
	Critically Dry	3,814	3,343 (-12.3%)	3,362 (-11.9%)	3,347 (-12.2%)	3,451 (-9.5%)
	All	4,079	3,842 (-5.8%)	3,837 (-5.9%)	3,848 (-5.7%)	3,857 (-5.4%)
July	Wet	5,823	5,824 (0%)	5,833 (0.2%)	5,826 (0.1%)	5,756 (-1.1%)
	Above Normal	7,853	7,854 (0%)	7,859 (0.1%)	7,854 (0%)	7,712 (-1.8%)
	Below Normal	8,615	8,402 (-2.5%)	8,431 (-2.1%)	8,416 (-2.3%)	8,431 (-2.1%)
	Dry	7,153	6,890 (-3.7%)	6,932 (-3.1%)	6,912 (-3.4%)	7,034 (-1.7%)
	Critically Dry	3,751	3,654 (-2.6%)	3,663 (-2.4%)	3,684 (-1.8%)	3,708 (-1.1%)
	All	6,570	6,460 (-1.7%)	6,479 (-1.4%)	6,472 (-1.5%)	6,465 (-1.6%)
August	Wet	4,491	4,486 (-0.1%)	4,536 (1%)	4,488 (-0.1%)	4,531 (0.9%)
	Above Normal	6,727	6,722 (-0.1%)	6,725 (0%)	6,720 (-0.1%)	6,711 (-0.2%)
	Below Normal	7,521	7,484 (-0.5%)	7,499 (-0.3%)	7,487 (-0.4%)	7,583 (0.8%)
	Dry	1,936	2,106 (8.7%)	2,073 (7.1%)	2,098 (8.4%)	2,085 (7.7%)
	Critically Dry	2,246	2,467 (9.9%)	2,449 (9%)	2,475 (10.2%)	2,513 (11.9%)
	All	4,428	4,489 (1.4%)	4,499 (1.6%)	4,490 (1.4%)	4,523 (2.1%)
September	Wet	6,275	6,351 (1.2%)	6,408 (2.1%)	6,355 (1.3%)	6,394 (1.9%)
	Above Normal	7,660	7,662 (0%)	7,711 (0.7%)	7,662 (0%)	7,668 (0.1%)
	Below Normal	2,706	2,874 (6.2%)	2,945 (8.9%)	2,880 (6.4%)	3,017 (11.5%)
	Dry	1,016	1,100 (8.3%)	1,101 (8.3%)	1,082 (6.5%)	1,115 (9.7%)
	Critically Dry	1,204	1,217 (1.1%)	1,240 (3%)	1,230 (2.2%)	1,233 (2.4%)
	All	3,895	3,971 (2%)	4,012 (3%)	3,971 (2%)	4,018 (3.2%)
October	Wet	2,846	2,946 (3.5%)	2,928 (2.9%)	2,944 (3.4%)	2,927 (2.8%)
	Above Normal	2,628	2,735 (4.1%)	2,743 (4.4%)	2,740 (4.3%)	2,707 (3%)
	Below Normal	2,586	2,728 (5.5%)	2,761 (6.8%)	2,743 (6.1%)	2,657 (2.7%)
	Dry	2,476	2,497 (0.8%)	2,500 (0.9%)	2,489 (0.5%)	2,440 (-1.5%)
	Critically Dry	1,412	1,477 (4.6%)	1,515 (7.3%)	1,456 (3.1%)	1,465 (3.8%)
	All	2,481	2,567 (3.5%)	2,574 (3.8%)	2,565 (3.4%)	2,530 (2%)
November	Wet	2,604	2,637 (1.3%)	2,617 (0.5%)	2,638 (1.3%)	2,625 (0.8%)
	Above Normal	1,650	1,650 (0%)	1,650 (0%)	1,650 (0%)	1,650 (0%)
	Below Normal	1,715	1,813 (5.7%)	1,750 (2.1%)	1,793 (4.6%)	1,764 (2.9%)
	Dry	1,592	1,674 (5.2%)	1,619 (1.7%)	1,674 (5.2%)	1,637 (2.9%)
	Critically Dry	1,322	1,446 (9.4%)	1,492 (12.9%)	1,456 (10.2%)	1,451 (9.8%)

Month	Water Year Type[1]	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
	All	1,915	1,980 (3.4%)	1,957 (2.2%)	1,978 (3.3%)	1,960 (2.3%)
December	Wet	4,521	4,513 (-0.2%)	4,489 (-0.7%)	4,479 (-0.9%)	4,467 (-1.2%)
	Above Normal	2,176	2,146 (-1.4%)	2,177 (0%)	2,146 (-1.4%)	2,178 (0.1%)
	Below Normal	2,434	2,431 (-0.1%)	2,368 (-2.7%)	2,431 (-0.1%)	2,412 (-0.9%)
	Dry	2,312	2,310 (-0.1%)	2,309 (-0.1%)	2,310 (-0.1%)	2,322 (0.5%)
	Critically Dry	1,810	1,794 (-0.9%)	1,803 (-0.4%)	1,793 (-1%)	1,796 (-0.8%)
	All	2,972	2,962 (-0.3%)	2,947 (-0.8%)	2,951 (-0.7%)	2,950 (-0.7%)

[1] Water year type sorting is by hydrologic water year.

Larval and Juvenile Rearing and Emigration

Little information is available on the distribution and timing of green sturgeon larvae or juveniles in the Feather River (National Marine Fisheries Service 2018b, Heublein et al. 2017). Assuming that life stage development times and behaviors for green sturgeon larvae and juveniles in the Feather River are like those in the Sacramento River, the larvae occur in the Feather River from spring to early autumn and are distributed from the Fish Barrier Dam to the confluence with the Sacramento River, while Feather River juveniles are present from May through December and from the Fish Barrier Dam to the Delta.

Mean flows in the HFC of the Feather River are substantially lower under Alternatives 1, 2, and 3 than under the NAA during June of drier years, as described above, and are higher during August through November of drier years (Table 11-63). As noted for green sturgeon in the Sacramento River, higher flows may improve conditions for emigrating green sturgeon larvae and juveniles (Heublein et al. 2017), but this is uncertain.

Green sturgeon larvae are not expected to be present in the Feather River during June of Below Normal, Dry, and Critically Dry Water Years because, as discussed in the previous section, green sturgeon in the Feather River appear to spawn only in high-flow years. As discussed below, however, it is possible that this restriction on their spawning results from their being unable to access spawning habitat upstream of the Sunset Pumps weir, which currently appears to block passage at flows below about 6,000 cfs. Without the Sunset Pump weir barrier, green sturgeon might be able to access spawning habitat with suitable temperatures in the LFC during drier years, in which case larvae might be present in the HFC by June and therefore would potentially be affected by the reductions in flow under Alternatives 1, 2, and 3 (Table 11-63). However, this scenario is too speculative to justify any conclusions regarding effects of Alternatives 1, 2, and 3 on the larvae. Furthermore, the flow increases expected for drier years in August and September of Alternatives 1, 2, and 3 (Table 11-63) would potentially benefit the larvae, resulting in no net effect.

Adult Migration and Holding

Green sturgeon adults have been found throughout the Feather River downstream of the Fish Barrier Dam (National Marine Fisheries Service 2018b, Heublein et al. 2017). However, information on the timing of adult migrations and the duration of the holding period in the Feather River is lacking so it is assumed to be the same as that for the Sacramento River:

immigration from February through June and holding from March through December. CALSIM II results for Feather River mean monthly flows below the Thermalito Afterbay outlet show little difference in flow between Alternatives 1, 2, and 3 and the NAA from February to May, except for flow increases in April of Above Normal Water Years. The results show reductions in flow under Alternatives 1, 2, and 3 during June and July, and large increases in flow during late summer and fall (Table 11-63). All the larger differences in flow occur during Below Normal, Dry, and Critically Dry Water Years, except for the April increases in Above Normal Water Years.

As noted in the previous section, the flow reductions in June have the potential to result in passage barriers to upstream-migrating green sturgeon. Currently, the boulder weir at the Sunset Pumps is the principal passage obstruction in the Feather River (National Marine Fisheries Service 2018b; Seesholtz pers. comm). This weir creates a partial barrier to the only confirmed spawning locations of green sturgeon in the Feather River (Seesholtz et al. 2015; Heublein et al. 2017). USFWS (2016) indicates that the boulder weir is a barrier to upstream passage of green sturgeon when Feather River flow is less than 6,000 cfs. Although some passage at lower flows may occur, the potential effect of Feather River flow reductions under Alternatives 1, 2, and 3 was assessed by enumerating from CALSIM II outputs the frequency of flow lower than 6,000 cfs during the February through June immigration period (Appendix N, Section 11N.2.4, *Low-Flow Passage Effects on Immigrating Salmon and Sturgeon Adults*). The results indicate that a high percentage of the monthly flows under the NAA and Alternatives 1, 2, and 3 are below the 6,000 cfs threshold for passage at the Sunset Pumps, including 78% of flows under the NAA and about 62% of flows under Alternatives 1, 2, and 3 (Appendix 11N, Table 11N-34). These high percentages suggest that a Sunset Pumps passage barrier may contribute to the lack of spawning by green sturgeon in the Feather River during all but the wettest years. As noted in the previous section, without the Sunset Pump weir barrier, green sturgeon might be able to access spawning habitat with suitable temperatures in the LFC during drier years and thereby increase overall production in the Feather River. The results indicate that Alternatives 1, 2, and 3 provide slightly improved flow conditions for upstream passage with regard to low flows. The increased flows under Alternatives 1, 2, and 3 during late summer and fall may improve habitat and passage conditions for adults emigrating from the river after spawning.

Delta

South Delta Entrainment

In contrast to salmonids, for which most south Delta entrainment at the SWP and CVP export facilities occurs in winter/spring, a period of the year when there would be little difference between Alternatives 1, 2, and 3 and the NAA, green sturgeon entrainment can occur in most months of the year, reflecting the year-round presence of juveniles in the Delta. However, salvage of green sturgeon has been low in recent years, and entrainment is regarded as a threat of low importance to the population in the NMFS green sturgeon recovery plan (National Marine Fisheries Service 2018b:26). The salvage-density analysis (Appendix 11Q) was used to assess the potential for differences in south Delta entrainment between alternatives. The method weights south Delta exports at SWP (Banks) and CVP (Jones) export facilities by historical salvage per unit volume (i.e., salvage density) of juvenile green sturgeon. The results of the analysis suggest that there would be minimal difference in south Delta entrainment between the NAA and Alternatives 1, 2, and 3 at either export facility (Table 11-64; Table 11-65).

Table 11-64. Salvage of Juvenile Green Sturgeon At SWP Banks Pumping Plant, Averaged by Water Year Type, Based on the Salvage-Density Method.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	1	1 (0%)	1 (0%)	1 (0%)	1 (1%)
Above Normal	NA	(2%)	(2%)	(2%)	(0%)
Below Normal	1	1 (0%)	1 (0%)	1 (0%)	1 (0%)
Dry	0	0 (0%)	0 (0%)	0 (0%)	0 (0%)
Critically Dry	0	0 (0%)	0 (0%)	0 (0%)	0 (0%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. The analysis was based on historical salvage data during 2009–2019 which did not include any Above Normal Water Years, hence that row of the table is noted as 'NA'; the percentage difference in Above Normal Water Years is based on density data for Wet Water Years applied to above normal modeled exports. Results by water year type and month are provided in Appendix 11Q.

Table 11-65. Salvage of Juvenile Green Sturgeon At CVP Jones Pumping Plant, Averaged by Water Year Type, Based on the Salvage-Density Method.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	7	7 (0%)	7 (0%)	7 (0%)	7 (0%)
Above Normal	NA	(0%)	(0%)	(0%)	(1%)
Below Normal	0	0 (0%)	0 (0%)	0 (0%)	0 (0%)
Dry	0	0 (0%)	0 (0%)	0 (0%)	0 (0%)
Critically Dry	0	0 (0%)	0 (0%)	0 (0%)	0 (0%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. The analysis was based on historical salvage data during 2009–2019 which did not include any Above Normal Water Years, hence that row of the table is noted as 'NA'; the percentage difference in Above Normal Water Years is based on density data for Wet Water Years applied to above normal modeled exports. Results by water year type and month are provided in Appendix 11Q.

Delta Outflow Effects

The NMFS green sturgeon recovery plan suggested that larval abundance and distribution may be influenced by spring and summer outflow and recruitment may be highest in wet years, making water flow an important habitat parameter (National Marine Fisheries Service 2018b:12). As noted by NMFS (2018b:12), there are correlations between white sturgeon and Delta outflow, which have previously been used to infer potential effects on green sturgeon (ICF International 2016:5-197 to 5-205). More recently, as noted above for upstream flow effects in the Sacramento River, CDFW found that CPUE of yearling juvenile green sturgeon sampled in the Delta was about 10 times as high in years following Wet Water Years as in years following Dry and Critically Dry Years (Beccio 2021). As discussed below for white sturgeon, the results of regression analyses for this impact analysis suggest that any potential effects of Alternatives 1, 2, and 3 on Delta outflow and white sturgeon year-class strength[27] would be limited, primarily because the largest recruitment occurs in wetter years (Fish 2010) when there are smaller differences between Alternatives 1, 2, and 3 and the NAA scenarios. This suggests potential Delta outflow effects of Alternatives 1, 2, and 3 on green sturgeon may also be limited.

CEQA Significance Determination for Alternatives 1, 2, and 3

The preceding subsections of this impact discussion provide the detailed information used for this CEQA (and NEPA) determination. This section provides a summary of this information.

Differences in flow results between Alternatives 1, 2, and 3 and the NAA for the Sacramento River at Bend Bridge, near the upper limit of the green sturgeon spawning distribution in the Sacramento River, show no changes between the NAA and Alternatives 1, 2, and 3. In reference to Sacramento River green sturgeon spawning habitat, differences in mean flow between Alternatives 1, 2, and 3 are negligible. Similarly, for green sturgeon larvae rearing habitat in the Sacramento River, differences in mean monthly flows between Alternatives 1, 2, and 3 are minimal and may in certain situations have potential benefits. Related to temperature effects, observations of exceedance plots and differences in modeled mean monthly temperatures by water year type indicate that Alternatives 1, 2, and 3 and the NAA would be predominantly similar among alternatives during the period of presence of each life stage of green sturgeon and the water temperature indicator value exceedance analysis confirms this similarity among alternatives.

Results of analyses based on observations of white sturgeon in Schaffter (1997) indicate that Alternatives 1, 2, and 3 would have no effect on flow conditions for upstream migrations of sturgeon in the Sacramento River relative to the NAA.

In the Feather River, considering all the results, Alternatives 1, 2, and 3 are not expected to have a substantial effect with regard to flow on spawning and egg incubation of green sturgeon. The results also indicate that Alternatives 1, 2, and 3 provide slightly improved flow conditions for upstream passage with regard to low flows. The increased flows under Alternatives 1, 2, and 3 during late summer and fall may improve habitat and passage conditions for adults emigrating from the river after spawning. In-Delta and upstream operations and their impacts associated with Alternatives 1, 2, and 3 on green sturgeon and its spawning habitat would be negligible. Per the above discussion related to green sturgeon, operation of Alternative 1, 2, or 3, would not have a substantial adverse effect, either directly or through habitat modifications, on green sturgeon. Operations impacts of Alternative 1, 2, or 3 would be less than significant.

NEPA Conclusion for Alternatives 1, 2, and 3

Operations effects on green sturgeon would be the same as described above for CEQA. In-Delta and upstream operations and their impacts associated with Alternatives 1, 2, and 3 on green sturgeon and its spawning habitat would be negligible compared to the NAA. Operation of Alternative 1, 2, or 3 would have no adverse effect on green sturgeon.

[27] Year-class strength in this case is an index of year-class abundance based on age-0 and age-1 white sturgeon abundance indices from otter trawling by the San Francisco Bay Study (Fish 2010: 80).

Impact FISH-7: Operations Effects on White Sturgeon

Alternatives 1, 2, and 3

Effects of Alternatives 1, 2, and 3

Sacramento River

Near-Field Effects

White sturgeon spawning, and early life stages potentially vulnerable to entrainment and other near-field effects occurs downstream of the Red Bluff and Hamilton City intakes, as described in Appendix 11A and as reflected in the overview of flow tolerance limitations by location (Figure 11-23 in the analysis for green sturgeon). Near-field effects associated with the Red Bluff and Hamilton City intakes are not expected to occur as a result of operations of Alternatives 1, 2, and 3.

Far-Field Effects

Temperature Effects

Operation of Sites Reservoir has the potential to change water temperatures in the Sacramento River that could affect the life stages of white sturgeon present. As described in Appendix 11B, the two methods used to analyze temperature-related effects on white sturgeon in the Sacramento River were: (1) Physical Model Output Characterization; and (2) Water Temperature Index Value/Range Exceedance Analysis. More details on these methods are provided in Appendix 11B.

The Authority and Reclamation evaluated water temperature model outputs during the period of presence and in the locations of each life stage of white sturgeon in the Sacramento River upstream of the Delta (see Appendix 11B, Table 11B-2 for timing and locations). Visual observation of exceedance plots and differences in modeled mean monthly temperatures by water year type between Alternatives 1, 2, and 3 and the NAA in the Sacramento River at Butte City indicates that water temperatures would be predominantly similar among alternatives during the period of presence of each life stage of white sturgeon (Appendix 6C, Tables 6C-12-1a to 6C-12-4c; Figures 6C-12-1 to 6C-12-18). Mean monthly water temperatures for all months within all water year types under Alternatives 1A and 1B were within 0.6°F of the NAA with the exception of August of critical years at Butte City, in which mean monthly water temperature would be 0.8°F lower under the NAA than under Alternatives 1A and 1B. Water temperature modeling results for Alternatives 2 and 3 were similar to those of Alternative 1. These results suggest that temperature-related effects on white sturgeon in the Sacramento River would be negligible.

Results of the analysis of exceedance above water temperature index values for white sturgeon in the Sacramento River at Hamilton City from Appendix 11B, Table 11B-2 are presented in Appendix 11D, Table 11D-226 through Table 11D-229. For each life stage in each month and water year type, the difference in percent of days exceeding each index value between the NAA and Alternative 1A and between the NAA and Alternative 1B would be <5%. Results of the analysis of exceedance for Alternative 2 would be nearly identical to Alternative 1A.

Results for Alternative 3 would be similar to Alternative 1A with two differences. For juvenile rearing and emigration, the percent of days exceeding the 66°F index value under Alternative 3 in June of Above Normal and Below Normal Water Years would be 9.2% and 6.4% greater, respectively, than the percent of days under the NAA. Due to low frequency and magnitude of differences between each alternative and the NAA in exceedances above water temperature index values in the Sacramento River, they are not expected to be persistent enough to affect white sturgeon at a population level.

Flow Effects

Spawning and Egg Incubation

White sturgeon spawn in deep water in the middle and lower Sacramento River from Verona (RM 80) to just upstream of Colusa (RM 146) from late February to early June, but primarily during March and April (Schaffter 1997; Moyle et al. 2015; Heublein et al. 2017). The adults typically return to the Delta soon after spawning.

During the March and April spawning and egg incubation period of white sturgeon, estimates of mean monthly flows at Wilkins Slough (RM 120) under the NAA and Alternatives 1, 2, and 3 range from about 5,500 cfs for April of Critically Dry Water Years to about 19,000 cfs for March of Wet Water Years (Table 11-60). Differences in flows, which almost all constitute flow reductions under Alternatives 1, 2, and 3, range up to 7% for March of Below Normal Years. Assuming that the relationship between flow and spawning WUA for white sturgeon in the Sacramento River is similar to that described above in the section on green sturgeon, with maximum spawning WUA at flows from 7,500 cfs to 12,360 cfs, the flow reductions in March of Below Normal Water Years would increase spawning WUA of white sturgeon, while the reductions in drier water year types would have negligible effect. It is concluded that Alternatives 1, 2, and 3 would have no flow effect on white sturgeon spawning or egg incubation.

Larval and Juvenile Rearing and Emigration

During the white sturgeon larval and juvenile emigration period, approximately April to July, CALSIM II results at Wilkins Slough indicate that flows would decrease up to 6% in April and May of Critically Dry Water Years under Alternatives 1, 2, and 3 and increase up to 9% in July of Dry and Critically Dry Water Years (Table 11-60). Under Alternative 3, July flow would also increase in Above Normal and Below Normal Water Years (up to 6%). The July increases in flow may benefit the white sturgeon because there appears to be a positive relationship between annual outflow during late winter through July and the abundance of age 0 and yearling white sturgeon (Fish 2010). Alternatives 1, 2, and 3 are expected to have little effect on rearing or emigration of white sturgeon larvae or juveniles.

Adult Upstream Migration and Holding

White sturgeon adults initiate their upstream spawning migrations during late winter and early spring, probably in response to elevated flows (Schaffter 1997, Fish 2010, Jackson et al. 2016). Most upstream migration occurs in March and April (Schaffter 1997). Reduced flows have potentially adverse effects on white sturgeon immigration, because they may reduce flow cues that may initiate migrations (Fish 2010; Jackson et al. 2016) and because low flows can result in passage barriers. The relationship between flows and white sturgeon spawning cues is currently too uncertain to make conclusions about such effects. Schaffter (1997) reported that white sturgeon ceased upstream migrations in the Sacramento River when flow at Colusa (RM 146) fell below about 5,300 cfs. Therefore, a low-flow threshold of 5,300 cfs downstream of the Colusa Weir was used in the analyses of potential Project effects on upstream passage of adult sturgeon, as discussed in Appendix 11N, Section 11N.2.4. The results of these analyses indicated that Alternatives 1, 2, and 3 would have no effect on white sturgeon upstream passage (Appendix 11N, Table 11N-38).

The white sturgeon holding period extends from about January through April. Flow during this period is generally lower under Alternatives 1, 2, and 3 than the NAA, with the majority of the reductions between about 2% and 5% (Table 11-60). However, most of the mean flows under the NAA and Alternatives 1, 2, and 3 throughout the holding period exceed 7,400 cfs, which is likely to be enough for conditions in the deep-water habitat in which the adults hold. The mean flows in April of Critically Dry Water Years are below 5,800 cfs for the NAA and Alternatives 1, 2, and 3. Suitable flow for Sacramento River white sturgeon holding habitat is not known, but the 2% to 5% reductions are not expected to be large enough to substantially affect this habitat.

Adult Upstream Passage at Fremont Weir

As described for winter-run Chinook salmon in Impact FISH-2, the number of days of adult passage at Fremont Weir was assessed using the number of days meeting adult passage criteria[28], based on data for water years 2009–2018 from the Daily Divertible Flow and Storage Tool Results (see Appendix 11P1). The results of this analysis indicated that the number of days meeting fish passage criteria at Fremont Weir generally would be similar with or without the Project (Table 11-21).

Adult Upstream Passage at Sutter Bypass Weirs

As described for green sturgeon, the number of days of adult passage at Sutter Bypass weirs was assessed using the number of days meeting adult passage criteria, based on data for water years 2009–2018 from the Daily Divertible Flow and Storage Tool Results (see Appendix 11P1). The results of this analysis indicated that there would be no days meeting adult sturgeon passage criteria during 2009–2018 at Moulton or Tisdale Weirs. For Colusa Weir, the number of days meeting fish passage criteria generally would be similar with or without the Project (Table 11-61).

Sites Reservoir Release Effects

Sites Reservoir releases could temporally overlap with white sturgeon presence near the release location in the Sacramento River. Migrating adults would be present primarily during December through February and rearing or emigrating juveniles would be present primarily during approximately April through July.

[28] The criteria for days to be counted as Yolo Bypass passage days were river stage (1) between 21.14 feet and 29.92 feet during November 1–March 15, (2) between 21.14 feet and 23.35 feet during March 16–April 30, or (3) greater than 35 feet (elevation of the crest of the Fremont Weir). These criteria were based on the YBPASS tool (California Department of Water Resources 2017).

Sites Reservoir releases in the Yolo Bypass via the CBD would not coincide with the upstream migration period of adult white sturgeon or the juvenile rearing and emigration period. No white sturgeon were collected during rescue efforts at Wallace Weir in 2014–2020 (Purdy and Kubo 2021).

Temperature Effects

As discussed in Chapter 6, the effects of Alternatives 1A, 1B, 2, and 3 on water temperatures at the Sites Reservoir release site in the Sacramento River would be relatively small with the releases generally tending to cause a slight reduction in water temperature (Chapter 6, Tables 6-12a through 6-12d). Temperature-related effects of Alternatives 1A, 1B, 2, and 3 on white sturgeon at the release site would be minimal.

There would be no temperature-related effects of Sites Reservoir releases in the Yolo Bypass via the CBD and in the Sacramento River below the Yolo Bypass under Alternatives 1A, 1B, 2, and 3 because no white sturgeon would be present in these locations during August through October when the Yolo Bypass would receive Sites Reservoir releases.

Feather River

Temperature Effects

Operation of Sites Reservoir has the potential to change water temperatures in the Feather River that could affect the life stages of white sturgeon present. As described in Appendix 11B, the two methods used to analyze temperature-related effects on white sturgeon in the Feather River were: (1) Physical Model Output Characterization; and (2) Water Temperature Index Value/Range Exceedance Analysis. More details on these methods are provided in Appendix 11B.

The Authority and Reclamation evaluated water temperature model outputs during the period of presence and in the locations of each life stage of white sturgeon in the Feather River (see Appendix 11B, Table 11B-3 for timing and locations). Visual observation of exceedance plots and differences in modeled mean monthly temperatures by water year type between Alternatives 1, 2, and 3 and the NAA in the Feather River LFC below the Fish Barrier Dam and in the HFC at Gridley Bridge indicates that water temperatures would be predominantly similar among alternatives during the period of presence of each life stage (Appendix 6C, Tables 6C-16-1a to 6C-16-4c, Tables 6C-18-1a to 6C-18-4c; Figures 6C-16-1 to 6C-16-18, Figures 6C-18-1 to 6C-18-18). At all locations, mean monthly water temperatures for all months within all water year types under Alternatives 1A and 1B were within 0.5°F of the NAA. Water temperature modeling results for Alternatives 2 and 3 were similar to those of Alternative 1 at all locations. These results suggest that temperature-related effects on white sturgeon in the Feather River would be negligible.

Results of the analysis of exceedance above water temperature index values for white sturgeon in the Feather River from Appendix 11B, Table 11B-3 are presented in Appendix 11D, Table 11D-230 through Table 11D-238. In the LFC below the Fish Barrier Dam, there would generally be no differences in exceedance of water temperature indices for any life stage between Alternatives 1A and 1B compared to the NAA in any month or water year type, except for an 8.3% reduction in exceedance above the mean monthly 66°F juvenile rearing and emigration index value in

August of Critically Dry Water Years under both Alternative 1A and 1B relative to the NAA. In the HFC, for each life stage in each month and water year type, the difference in percent of months exceeding each index value between the NAA and Alternative 1A and between the NAA and Alternative 1B would be <5%, with some isolated exceptions. Below Thermalito Afterbay in June of Dry Water Years and Critically Dry Water Years, there would be 11.1% and 16.7% increases in exceedance, respectively, above the juvenile rearing and emigration index value of 66°F under both Alternative 1A and 1B compared to the NAA. At the mouth of the Feather River, there would be no differences in exceedances of water temperature indices for any life stage between Alternatives 1A and 1B compared to the NAA in any month or water year type.

Results of the water temperature exceedance analysis for white sturgeon for Alternative 2 are similar to those for Alternative 1A at all locations.

Results of the water temperature exceedance analysis for white sturgeon for Alternative 3 are predominantly similar to those for Alternative 1A with three exceptions. First, in the LFC at the Fish Dam, there would be no difference in exceedance for any life stage of white sturgeon in any month or water year type. However, below Thermalito Afterbay, there would be a 5.6% reduction in exceedance above the 61°F spawning and egg incubation index value in May of Critically Dry Water Year. There would also be an 8.3% higher exceedance above the 66°F juvenile rearing and emigration index value during September of Critically Dry Water Years.

Due to low frequency and magnitude of differences between each alternative and the NAA in exceedances above water temperature index values in the Feather River, they are not expected to be persistent enough to affect white sturgeon at a population level.

Flow Effects

White sturgeon occasionally use the lower Feather River for spawning, embryo development, and early rearing, although no evidence of spawning has been documented in recent years (Moyle et al. 2015; Heublein et al. 2017). Upstream spawning migrations by white sturgeon adults in the Sacramento River generally occur from late winter to late spring and may coincide with higher flows (Schaffter 1997; Fish 2010; Moyle et al. 2015). Spawning occurs in deep water from late February to May, but primarily during March and April. The larvae and juveniles emigrate from approximately April to July (Moyle et al. 2015; Heublein et al. 2017).

Feather River mean monthly flow under Alternatives 1, 2, and 3 is similar to or lower than flow under the NAA during February through May, when most white sturgeon spawning and egg incubation would likely occur, except for 4% to 9% higher flows in April of Above Normal Water Years (Table 11-63). The largest flow reductions are 6% and 7% reductions in May of Dry Water Years and March of Below Normal Water Years, respectively, under Alternative 3. Because white sturgeon spawn in deep pools (Heublein et al. 2017), their eggs and embryos are protected from direct effects of flow reduction such as dewatering. Flow reductions are greatest (up to 15%) in June of Dry and Critically Dry Water Years (Table 11-62). However, white sturgeon spawning, assuming it occurs at all in the Feather River, would likely occur only in wetter years, such as the years when green sturgeon spawning was documented (Seesholtz et al. 2015). Therefore, white sturgeon eggs and larvae would likely not be present during the years with large June flow reductions under Alternatives 1, 2, and 3. As noted above, flow increases occur in April of Above Normal Water Years. These increases would potentially benefit white sturgeon adult migration and spawning because spawning is more likely to occur in wetter years. Considering all the results, Alternatives 1, 2, and 3 are not expected to have a substantial effect with regard to flow on spawning and egg incubation of white sturgeon in the Feather River.

White sturgeon larvae and juveniles in the Sacramento River emigrate from approximately April to July (Moyle et al. 2015; Heublein et al. 2017). Mean flows in the HFC of the Feather River are substantially lower under Alternatives 1, 2, and 3 than under the NAA during June, as described above, and are higher during August through November of drier years (Table 11-63). As noted for white sturgeon in the Sacramento River, higher flows may improve conditions for emigrating white sturgeon larvae and juveniles (Heublein et al. 2017), but this is uncertain.

Low flows potentially result in passage barriers for upstream-migrating white sturgeon. Flows exceeding the 6,000 cfs threshold for green sturgeon passage over the boulder weir of the Sunset Pumps at Live Oak (Appendix 11N, Section 11N.2.4) occur during the white sturgeon immigration period only in January through April of Wet Water Years, suggesting that in most years these would be the only periods when white sturgeon would be able to move upstream of the Sunset Pumps boulder weir (Table 11-63). During the late winter to late spring upstream migration period, the monthly flows below the Thermalito Afterbay outlet are generally similar between the NAA and Alternatives 1, 2, and 3.

Considering all potential Feather River flow effects, the Project is not expected to substantially affect white sturgeon in the Feather River.

Delta

South Delta Entrainment

As with green sturgeon, the salvage-density analysis (Appendix 11Q) was used to assess the potential for differences in south Delta entrainment of juvenile white sturgeon between alternatives. The results of the analysis suggest that there would be little difference in south Delta entrainment between the NAA and Alternatives 1, 2, and 3 at either export facility (Table 11-66; Table 11-67). Although the mean salvage under Alternative 3 was 9% greater than under the NAA in Below Normal Water Years and 19% greater in Dry Water Years at CVP (Table 11-67), this relative difference should be placed in the context of the low observed historical salvage in recent years, indicating that any increase in entrainment would remain small in population-level terms.

Table 11-66. Salvage of Juvenile White Sturgeon At SWP Banks Pumping Plant, Averaged by Water Year Type, Based on the Salvage-Density Method.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	18	18 (0%)	18 (0%)	18 (0%)	18 (0%)
Above Normal	NA	(-1%)	(-1%)	(-1%)	(-1%)
Below Normal	11	11 (3%)	11 (3%)	11 (3%)	11 (4%)
Dry	5	5 (1%)	5 (1%)	5 (0%)	5 (1%)
Critically Dry	4	4 (0%)	4 (-1%)	4 (1%)	4 (-1%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. The analysis was based on historical salvage data during 2009–2019 which did not include any Above Normal Water Years, hence that row of the table is noted as 'NA'; the percentage difference in Above Normal Water Years is based on density data for Wet Water Years applied to above normal modeled exports. Results by water year type and month are provided in Appendix 11Q. Percentage values are rounded; as a result, differences between percentages may not always appear consistent.

Table 11-67. Salvage of Juvenile White Sturgeon At CVP Jones Pumping Plant, Averaged by Water Year Type, Based on the Salvage-Density Method.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	80	80 (0%)	80 (0%)	80 (0%)	80 (0%)
Above Normal	NA	(0%)	(0%)	(0%)	(2%)
Below Normal	14	15 (2%)	15 (2%)	15 (3%)	16 (9%)
Dry	2	2 (2%)	2 (7%)	2 (3%)	2 (19%)
Critically Dry	0	0 (0%)	0 (0%)	0 (0%)	0 (0%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. The analysis was based on historical salvage data during 2009–2019 which did not include any Above Normal Water Years, hence that row of the table is noted as 'NA'; the percentage difference in Above Normal Water Years is based on density data for Wet Water Years applied to above normal modeled exports. Results by water year type and month are provided in Appendix 11Q. Percentage values are rounded; as a result, differences between percentages may not always appear consistent.

Delta Outflow Effects

Statistically significant positive correlations between white sturgeon year-class strength[29] and Delta outflow have been found for November–February and March–July outflow averaging periods (Fish 2010). Other similar analyses were found that also examined the April–May outflow (ICF International 2016:5-197–5-205). The mechanisms for these correlations are uncertain and could reflect upstream or in-Delta impacts. Appreciable amounts of variation are left unexplained by the relationships (i.e., r^2 of ~70%), with differences possibly reflecting hydrological conditions as opposed to operational differences in outflow. A regression-based approach predicting white sturgeon year-class strength as a function of April–May and March–July averaging periods was undertaken (Appendix 11L). The results of the analysis indicated that there would be little difference in white sturgeon year-class strength between Alternatives 1, 2, and 3 and the NAA based on April–May Delta outflow differences (Table 11-68), whereas the relative difference would be larger in drier (Below Normal and Dry Water Years) years based on March–July Delta outflow differences (Table 11-69). Overall, these results suggest that any potential effects of Alternatives 1, 2, and 3 on Delta outflow and white sturgeon year-class strength would be limited, primarily because the largest recruitment occurs in wetter years (Fish 2010) when there are smaller differences between the Alternatives 1, 2, and 3 and NAA scenarios.

[29] As previously noted in Impact AQUA-6 for green sturgeon, year-class strength in this case is an index of year-class abundance based on age-0 and age-1 white sturgeon abundance indices from otter trawling by the San Francisco Bay Study (Fish 2010: 80).

Table 11-68. Year-Class Strength of White Sturgeon Based on April–May Regression with Delta Outflow.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	148	146 (-1%)	146 (-1%)	146 (-1%)	146 (-1%)
Above Normal	70	69 (0%)	70 (0%)	69 (0%)	70 (0%)
Below Normal	32	32 (-2%)	32 (-2%)	32 (-2%)	32 (-2%)
Dry	7	7 (1%)	7 (-1%)	7 (1%)	7 (0%)
Critically Dry	0	0 (0%)	0 (0%)	0 (0%)	0 (0%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. Table only includes annual mean responses and does not consider model uncertainty.

Table 11-69. Year-Class Strength of White Sturgeon Based on March–July Regression with Delta Outflow.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	120	119 (-1%)	119 (-1%)	119 (-1%)	119 (-1%)
Above Normal	55	54 (-3%)	54 (-3%)	54 (-3%)	54 (-3%)
Below Normal	9	9 (-5%)	9 (-5%)	9 (-5%)	9 (-6%)
Dry	2	2 (-13%)	2 (-15%)	2 (-13%)	2 (-13%)
Critically Dry	0	0 (0%)	0 (0%)	0 (0%)	0 (0%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. Table only includes annual mean responses and does not consider model uncertainty. Percentage values are rounded; as a result, differences between percentages may not always appear consistent.

CEQA Significance Determination for Alternatives 1, 2, and 3

The preceding subsections of this impact discussion provide the detailed information used for this CEQA (and NEPA) determination. This section provides a summary of this information.

If it is assumed that the relationship between flow and spawning WUA for white sturgeon is similar to that of green sturgeon in the Sacramento River, differences in WUA for Alternatives 1, 2, and 3 would have no adverse effect on white sturgeon. Due to low frequency and magnitude of differences between each alternative and the NAA in exceedances above water temperature index values in the Sacramento River, they are not expected to be persistent enough to affect white sturgeon at a population level.

Per the above discussion related to white sturgeon, the operation of Alternative 1, 2, or 3 would not have a substantial adverse effect, either directly or through habitat modifications, on white sturgeon. Operations impacts of Alternatives 1, 2, or 3 on white sturgeon adult immigration, holding, spawning, and egg incubation would be less than significant.

NEPA Conclusion for Alternatives 1, 2, and 3

Operations effects on white sturgeon would be the same as described above for CEQA. Operation of Alternative 1, 2, or 3 would have no adverse effect on white sturgeon.

Impact FISH-8: Operations Effects on Delta Smelt

Alternatives 1, 2, and 3

Effects of Alternatives 1, 2, and 3

As described in Appendix 11A, delta smelt primarily occur within the Delta and Suisun Bay/Marsh, so this impact analysis is focused on these areas.

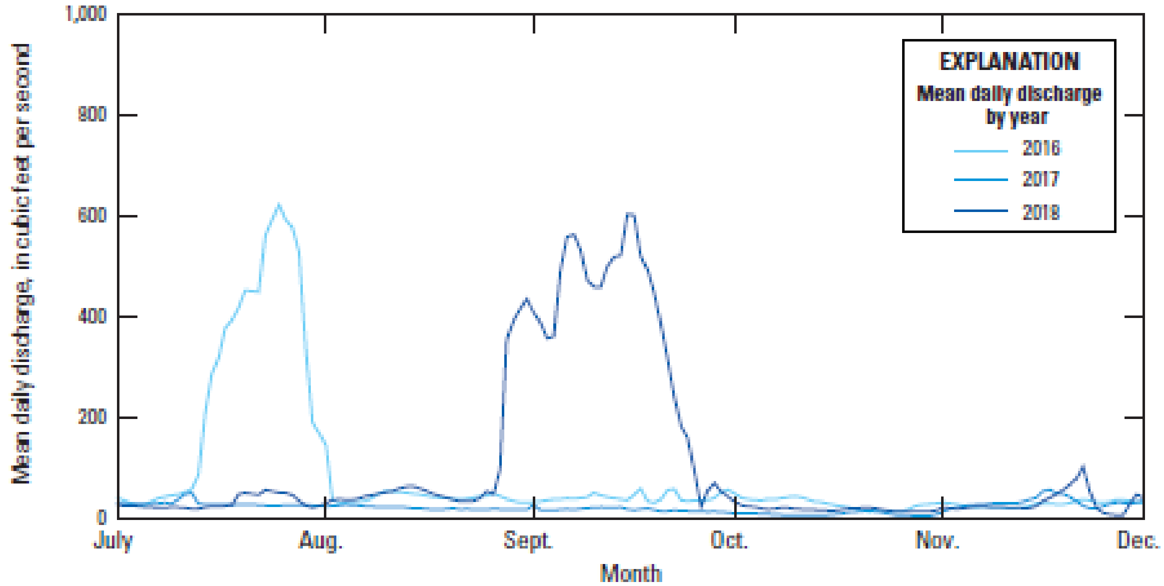
Effects from Reservoir Releases to CBD/Yolo Bypass

Food availability is a key habitat attribute hypothesized to be important for juvenile delta smelt (Interagency Ecological Program Management, Analysis, and Synthesis Team 2015:88). Alternatives 1, 2, and 3 include summer/fall releases of water from Sites Reservoir into the CBD and thence the Yolo Bypass, in order to enhance foodweb productivity in the north Delta for delta smelt. Thus Alternatives 1, 2, and 3 would increase flow through Yolo Bypass in August from ~50 cfs in nearly 90% of years under the NAA to ~400–450 cfs in 60%–70% of years under Alternatives 1, 2, and 3; in September from ~50–60 cfs in 80% of years under the NAA to ~300–350 cfs more under Alternatives 1, 2, and 3 in 40% of years; and in October from <100 cfs in most years under the NAA to ~400 cfs or more in 30%–40% of years under Alternatives 1, 2, and 3 (Appendix 5B3, Tables 5B3-3-1a through 5B3-3-4c).

An average of 23% of delta smelt surviving to adulthood are resident in the Cache Slough Complex/Sacramento Deepwater Ship Channel region throughout their lives, whereas the remainder either migrate to the low-salinity zone or are resident there (Bush 2017; Hobbs et al. 2019)[30]. The portion of the population resident in the north Delta would be most likely to benefit from the summer/fall north Delta food subsidy from CBD, in particular those occurring in the Yolo Bypass Toe Drain (Mahardja et al. 2019). A pilot implementation of this action in 2016, during which flows up to 600 cfs in July were provided by local reclamation districts (Orlando et al. 2020; Figure 11-26), found that primary production in the north Delta increased as a result of the action (Figure 11-27; as had been observed by Frantzich et al. [2018] in previous years with flow pulses). This increased primary production resulted in enhanced zooplankton growth and egg production (California Natural Resources Agency 2017). However, subsequent studies have found that the main diatom occurring in the 2016 phytoplankton blooms (*Aulacoseira*) provided only a minor stimulus to the delta smelt zooplankton prey *Pseudodiaptomus forbesi* during several blooms in 2016 (Jungbluth et al. 2021). Reclamation (2018:2) suggested that a chlorophyll concentration of 10 µg/l, as achieved in 2016 for a number of days during the action (Figure 11-27), could support relatively high zooplankton production (Mueller-Solger et al. 2002) without adversely affecting water quality (e.g., DO concentration). Analyses are underway to determine the potential effectiveness of a 2018 implementation of the action, during which up to 600 cfs of flow was provided by agricultural tailwater released from the CBD (Orlando et al. 2020; Figure 11-26), but preliminary information suggests that chlorophyll concentration above 10 µg/l was limited in duration in the Yolo Bypass (Figure 11-28) and there was no increase at Rio Vista (Figure 11-29).

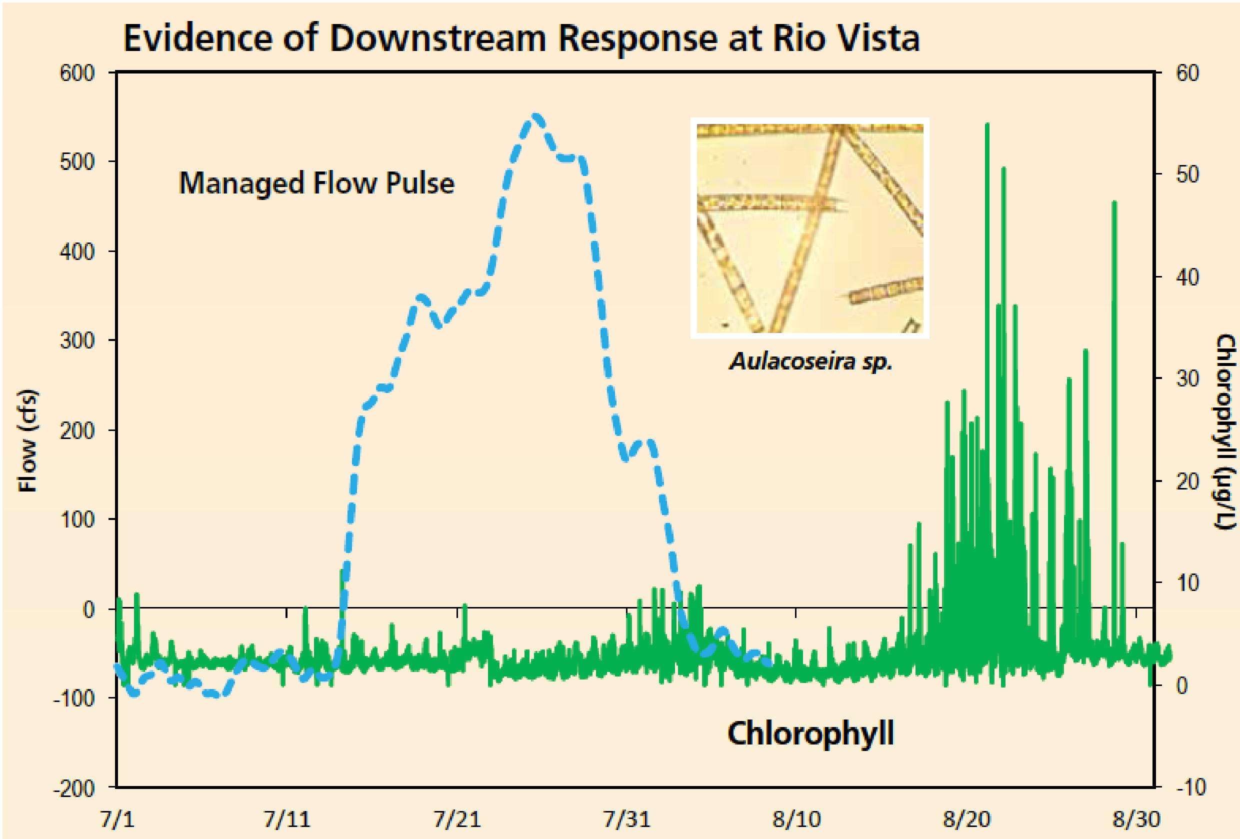
---

[30] During 2005–2014, the proportion remaining resident in the north Delta ranged from 2% in 2006 to 47% in 2010; the proportion remaining resident was negatively correlated with freshwater outflow and mean July temperature (Bush 2017:22–24).



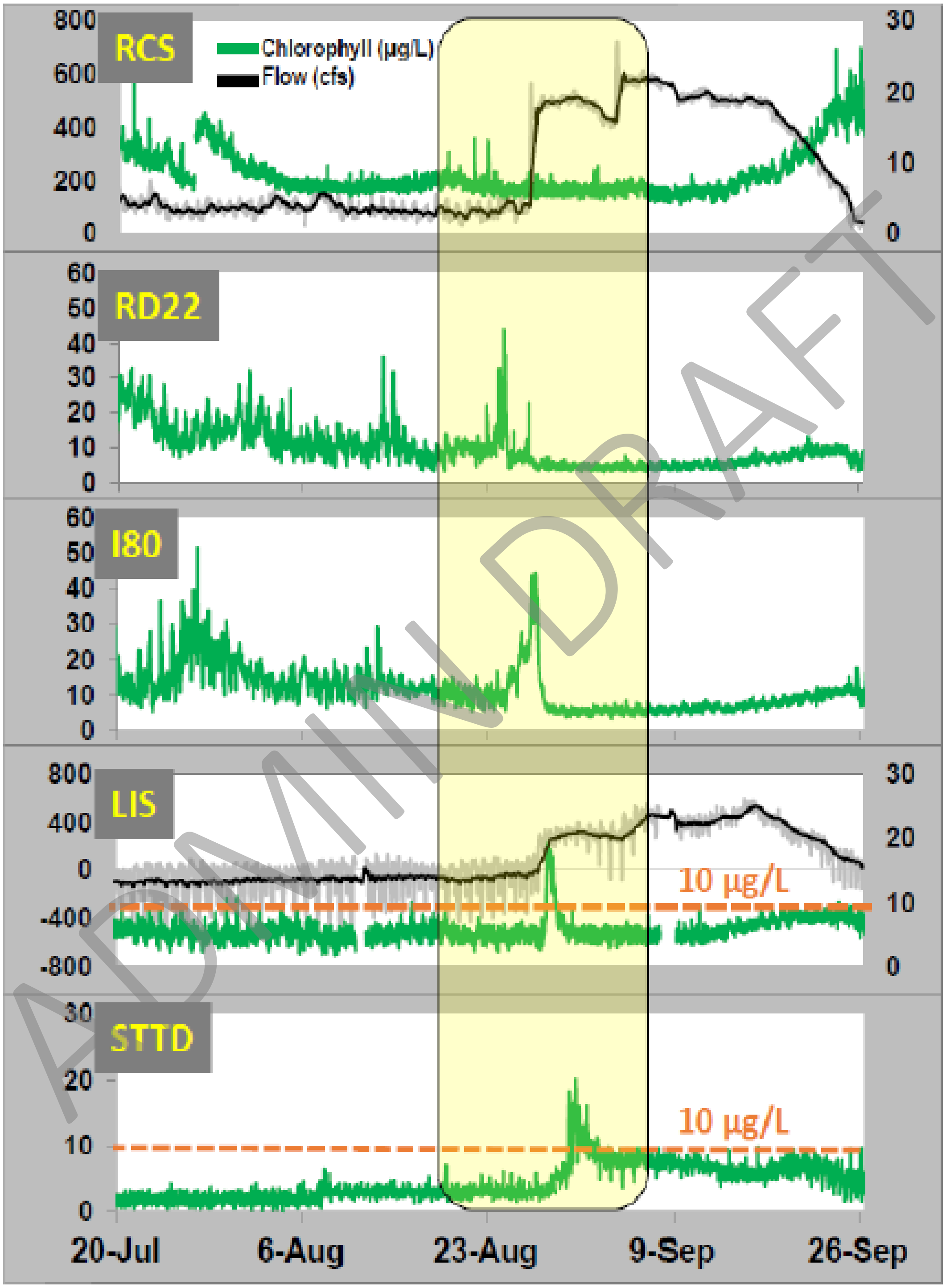
Source: Orlando et al. 2020.

Figure 11-26. Mean Daily Discharge at U.S. Geological Survey site 11453000 Yolo Bypass near Woodland for Summer and Fall 2016, 2017, and 2018.



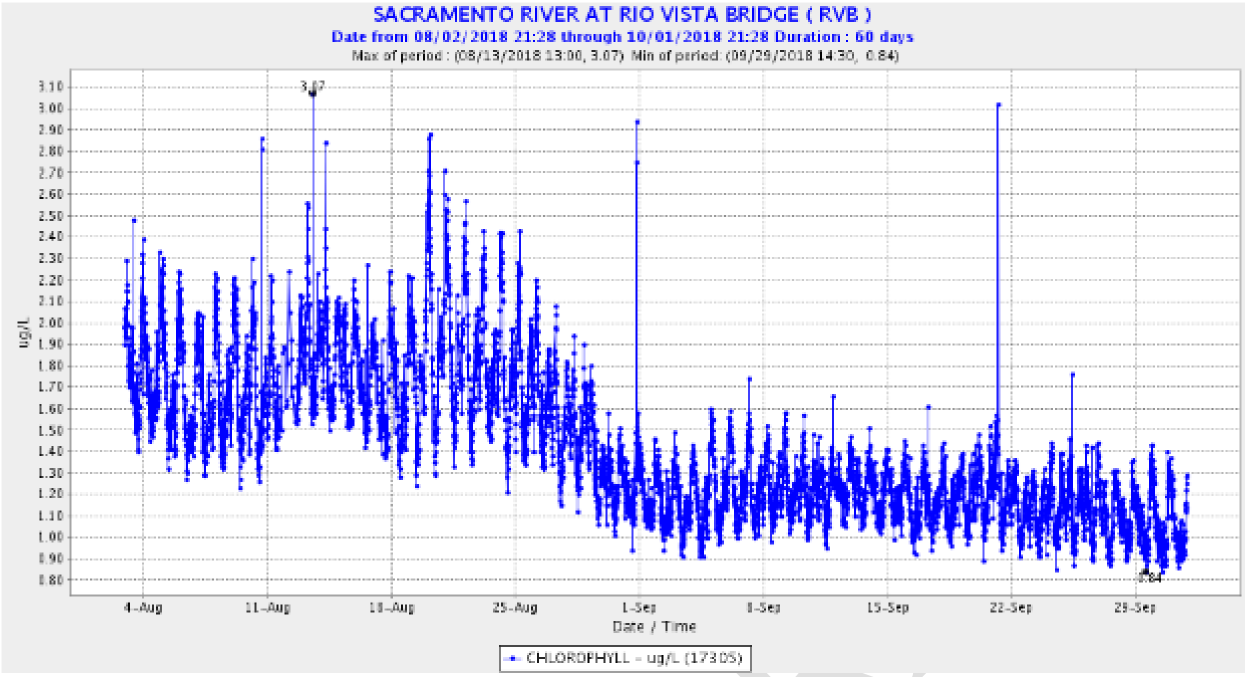
Source: California Natural Resources Agency 2017.

Figure 11-27. Managed Flow Pulse in the Yolo Bypass Toe Drain at Lisbon Weir and Chlorophyll Concentration at Rio Vista During 2016 Pilot North Delta Food Subsidy From Colusa Basin Drain Action.



Source: Northern California Water Association 2018. Note: Yellow box indicates flow pulse into Yolo Bypass from Colusa Basin Drain. Stations are Ridge Cut Slough at Highway 113 (RCS), Toe Drain at Road 22 (RD22), Toe Drain at I80 (I80), Toe Drain below Lisbon Weir (LIS), and Screw Trap at Toe Drain (see Figure 1 of Twardochleb et al. 2021).

Figure 11-28. Managed Flow Pulse in the Yolo Bypass Toe Drain at Lisbon Weir and Chlorophyll Concentration from North (RCS) to South (STTD) in the Yolo Bypass During 2018 Pilot North Delta Food Subsidy From Colusa Basin Drain Action.



Source: California Department of Water Resources 2019d.

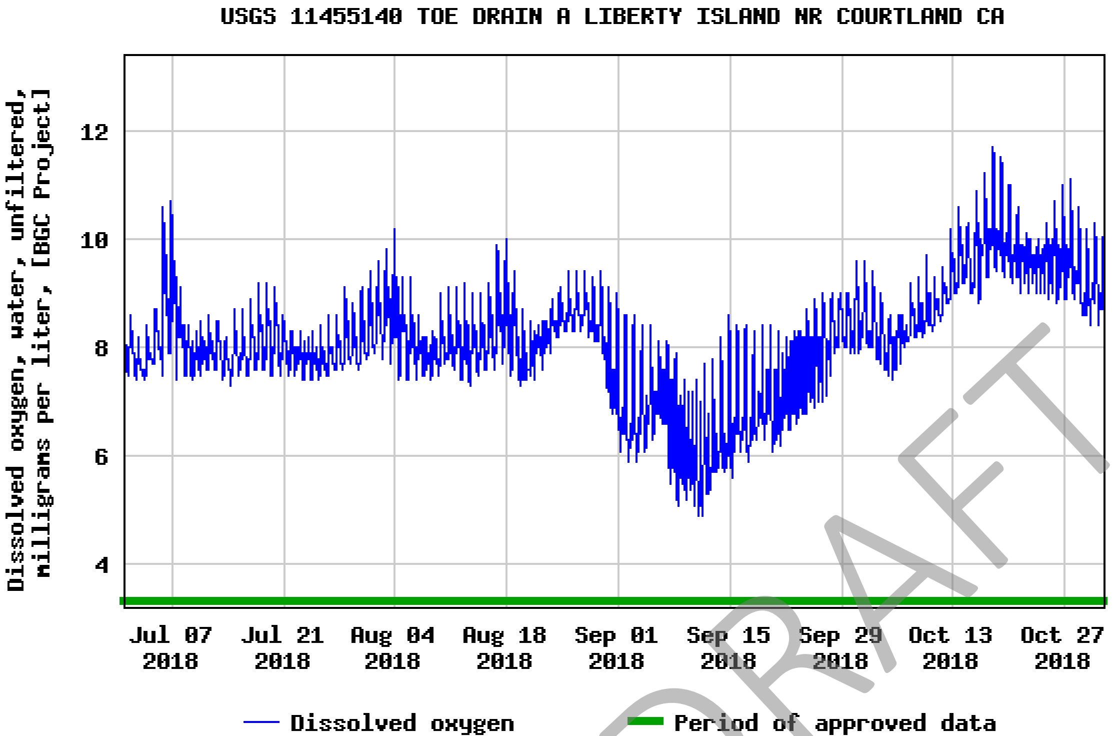
Figure 11-29. Chlorophyll Concentration at Rio Vista Before, During, and After 2018 Pilot North Delta Food Subsidies Action.

Analyses of the effectiveness of a 2019 implementation of an action similar to the 2018 action have been undertaken. The 2018 and 2019 actions were similar but the 2019 action included deployment of caged delta smelt to identify potential effects of the action directly on the fish. Preliminary results showed caged delta smelt in the Yolo Bypass did not survive because of a heat wave in late August, whereas survival during the after-action period was significantly higher, although this was likely due to seasonal effects and not the flow action alone (Davis et al. 2019). Primary productivity, as measured by chlorophyll fluorescence, was moved downstream during the action, with an increase at the downstream-most monitoring station in the Yolo Bypass Toe Drain from <3 micrograms per liter (µg/L) to up to ~7.5 µg/L for a short period, before returning to levels <3 µg/L after the action (Maguire et al. 2020; Twardochleb et al. 2021). Twardochleb et al. (2021) concluded that the 2019 action did not increase food availability downstream by as much as the 2016 action, which used water diverted from the Sacramento River. Twardochleb et al. (2021) further noted that future studies, including repeating the 2016 action using Sacramento River water and an upcoming synthesis comparing the results of managed flow pulses on the north Delta foodweb from 2011–2019, will allow further assessment of the effects of source water (agricultural return flows versus Sacramento River), and other mediating factors such as hydrology, to adaptively manage the flow action to maximize food availability downstream.

In addition to potential positive effects from foodweb materials moving downstream, Alternatives 1, 2, and 3 have the potential to affect water quality, including pesticides, water temperature, and DO. Pesticide concentrations in the Yolo Bypass and Cache Slough Complex were analyzed during augmented pulse flows in July 2016 and September 2018 and also in 2017 during ambient conditions (Orlando et al. 2020). Water samples were taken from the Yolo

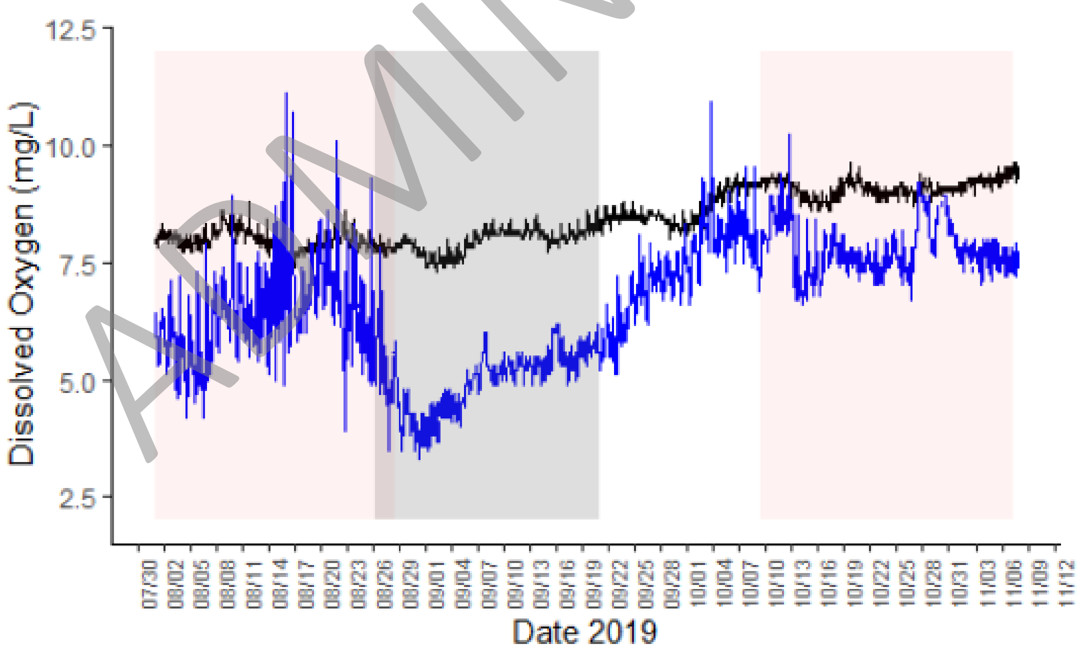
Bypass and Cache Slough Complex before, during, and after the pulse flows to determine if pesticide concentrations increased. Flows in July 2016 were from the Sacramento River into the CBD and the Yolo Bypass Toe Drain. Augmented flows in 2018 were from agricultural tailwater, mainly from rice field discharge water. Results from the three years of study concluded that the 2016 pulse of Sacramento River water reduced pesticide concentration at the upstream end of the Yolo Bypass, but may have moved some pesticide downstream to the lower part of the Yolo Bypass near Lisbon Weir. However, in Cache Slough near Ryer Island, where flow is much higher due to tidal influence of the Sacramento River, there was no apparent increase in pesticides. In 2017, pesticide concentrations remained the same and lessened in the early fall (Orlando et al. 2020:93–99). There was more of a pesticide signature at the downstream end of the Yolo Bypass during 2018, as the flow pulse was composed of agricultural drainage water. However, there was no change in pesticide concentrations evident in Cache Slough at Ryer Island. Similar patterns to 2018 were observed in 2019 (Twardochleb et al. 2021). Pulse flows from Sites Reservoir will generally have a low concentration of pesticides and would not contaminate the Sacramento River and would dilute the relatively high pesticide concentrations in CBD. There is uncertainty in the extent to which delta smelt could be affected by changes in pesticide concentrations under Alternatives 1, 2, and 3 in the lower Yolo Bypass as Sites Reservoir habitat flows would redirect CBD water that is relatively high in pesticides into Yolo Bypass (Chapter 6, Impact WQ-2).

Reductions in DO were observed in the Yolo Bypass Toe Drain from late August through late September during 2018 and 2019 pilot flow actions (Figures 11-30 and 11-31), whereas there was no observable effect further downstream in Cache Slough at Liberty Island (Figures 11-32 and 11-33). Ranges of acceptable DO for delta smelt have not been established because insufficient data exist to do so (Hamilton and Murphy 2020). Jabusch et al. (2008:16, 17, 26) proposed a conceptual model for DO effects on fish in the San Francisco Estuary such that below 7 mg/L reduced growth would occur, whereas below 2.3 mg/L mortality would occur. Assuming that the observed reduction in DO during 2018 and 2019 is representative of what may occur under Alternative 1, 2, or 3 as a result of Sites Reservoir water being released and pushing low DO water from the CBD downstream, these alternatives could result in reduced growth (but not mortality) to delta smelt occurring in the Yolo Bypass Toe Drain. These fish could move downstream toward Cache Slough where DO would be higher. There would not be expected to be reduced growth downstream (e.g., in Cache Slough). Studies of hatchery-origin delta smelt placed in cages in the Yolo Bypass in association with the 2019 flow action did not provide information on survival or growth during the period of low DO because all fish died during a heat wave in late August (Davis et al. 2019).



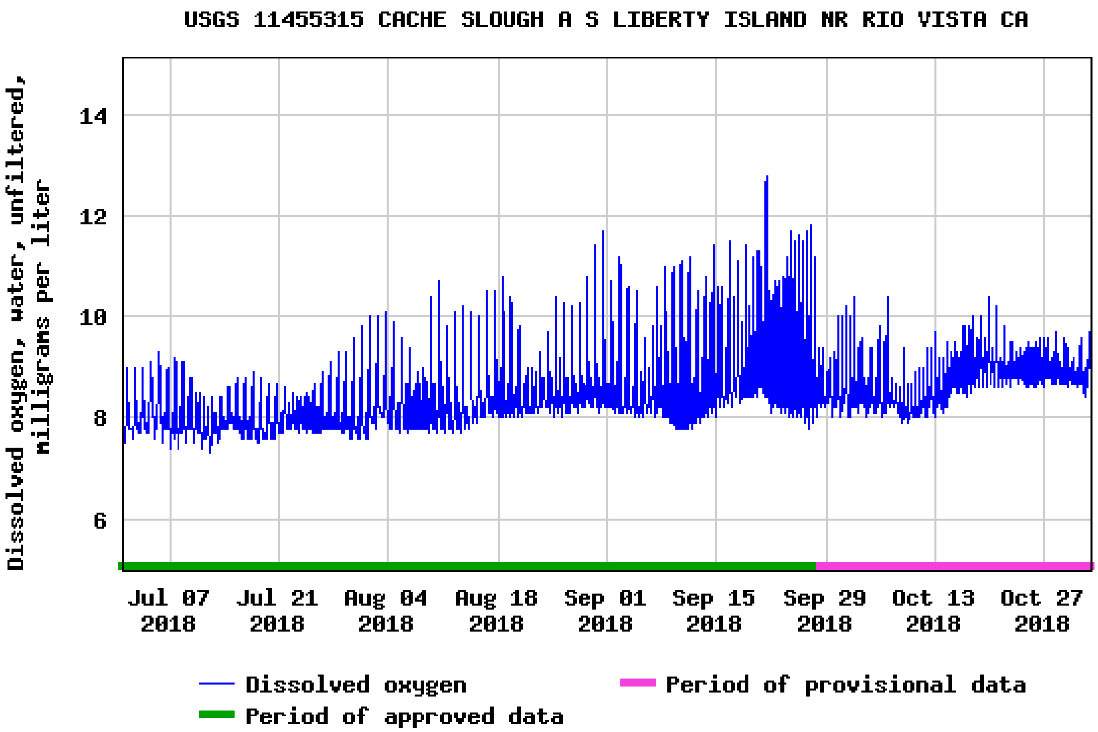
Source: U.S. Geological Survey 2023.

Figure 11-30. Dissolved Oxygen in the Yolo Bypass Toe Drain at Liberty Island During 2018.



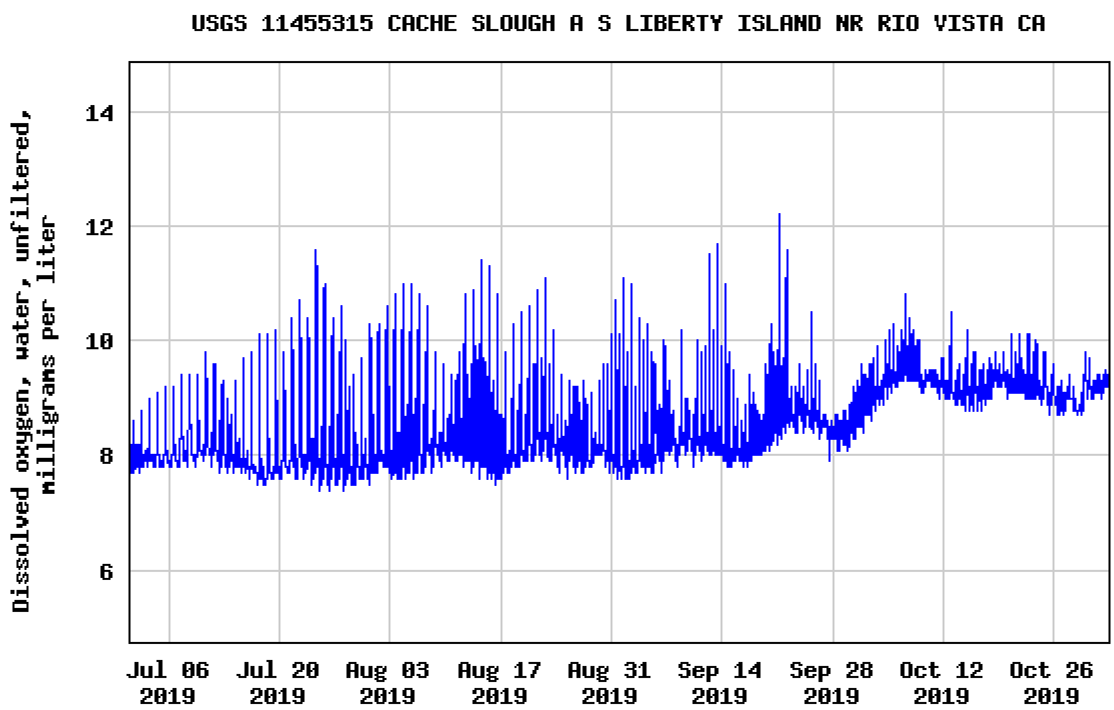
Source: Davis et al. 2019. Note: Shaded regions include the period of the Flow Action (grey), and deployments of caged delta smelt (pink) at Rio Vista and in the Yolo Bypass.

Figure 11-31. Dissolved Oxygen in the Yolo Bypass Toe Drain at Lisbon Weir (Blue Line) and Sacramento River at Rio Vista (Black Line) During 2019.



Source: U.S. Geological Survey 2021.

Figure 11-32. Dissolved Oxygen Cache Slough at Liberty Island During 2018.



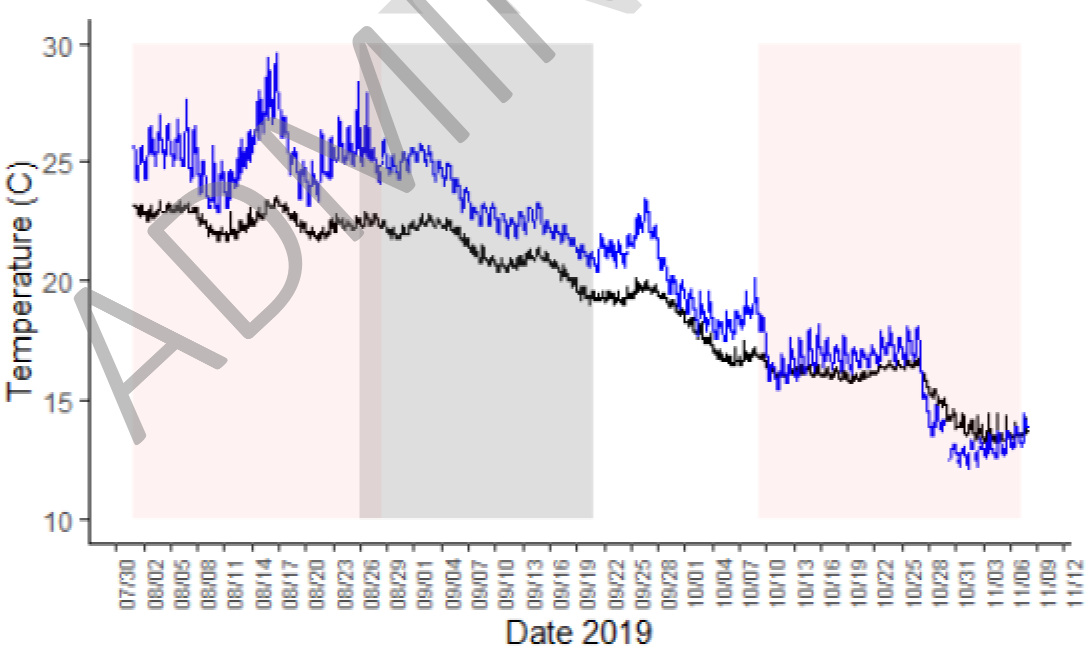
Source: U.S. Geological Survey 2021.

Figure 11-33. Dissolved Oxygen Cache Slough at Liberty Island During 2019.

As described in Chapter 6, temperature would not be expected to be greatly affected by Sites Reservoir releases. This determination was illustrated by the 2018 and 2019 flow actions (Figures 11-34 and 11-35). Because temperature generally is close to or exceeding observed thresholds for delta smelt mortality (Swanson et al. 2000) or occurrence (Nobriga et al. 2008), there is some uncertainty in the potential for effects on delta smelt.

Source: U.S. Geological Survey 2021.

Figure 11-34. Water Temperature in the Yolo Bypass Toe Drain at Liberty Island During 2018.



Source: Davis et al. 2019. Note: Grey shading indicates North Delta Flow Action period.

Figure 11-35. Water Temperature in the Yolo Bypass Toe Drain at Lisbon Weir (Blue Line) and Sacramento River at Rio Vista (Black Line) During 2019.

Entrainment

Adults

Adult delta smelt can be entrained into the south Delta export facilities during dispersal prior to spawning, with this dispersal typically occurring during December–March in response to increases in precipitation, flow, and turbidity (Grimaldo et al. 2009, 2021; Sommer et al. 2011). Alternatives 1, 2, and 3 do not include any changes to the criteria such as Old and Middle River flows from the USFWS (2019) ROC ON LTO BiOp and CDFW (2020) State ITP, which limit export pumping to minimize the risk of adult delta smelt south Delta entrainment. The CALSIM modeling results suggested that during December–March Old and Middle River flows generally would be similar under Alternatives 1, 2, and 3 and the NAA (Appendix 5B3, Figures 5B3-6-9 through 5B3-6-12) which, combined with the same real-time criteria to minimize risk as currently implemented (U.S. Fish and Wildlife Service 2019, California Department of Fish and Wildlife 2020), suggests there would be little difference in entrainment risk for adult delta smelt between the NAA and Alternatives 1, 2, and 3.

Larvae/Early Juveniles

South Delta entrainment of larval/early juvenile delta smelt generally occurs in March–June, with the risk of entrainment limited through criteria related to factors such as Old and Middle River flows (U.S. Fish and Wildlife Service 2019, California Department of Fish and Wildlife 2020). Alternatives 1, 2, and 3 do not include any changes to these criteria. As described in the IEP MAST (2015: 88) conceptual model, larval/early juvenile entrainment risk is a function of exports and spring hydrology, with USFWS (2008:220) demonstrating that entrainment risk increases as Old and Middle River flows decrease and become more negative (reflecting an increasing hydrodynamic influence of the south Delta export facilities) and X2 increases (reflecting less Delta outflow and a greater portion of delta smelt occurring farther upstream in the Delta). CALSIM II modeling suggests little difference in Old and Middle River flows between the NAA and Alternatives 1, 2, and 3 during March–June (Appendix 5B3, Figures 5B3-6-12 through 5B3-6-15). Diversions to Sites Reservoir during March–June would result in some reductions in Delta outflow and small increases in X2 in March (Table 6-16 in Chapter 6; see also discussion below in the discussion of *Flow-Related Effects*). Given the real-time nature of entrainment risk management that currently exists under the USFWS (2019) ROC ON LTO BiOp and CDFW (2020) State ITP, which would not be changed under Alternatives 1, 2, and 3, there would be little difference in south Delta entrainment risk between the NAA and Alternatives 1, 2, and 3. Although there would be somewhat greater pumping at the Barker Slough Pumping Plant under Alternatives 1, 2, and 3, this would be a relatively small increase (10–20 cfs or less) and would occur during July–October (Appendix 5B4, Tables 5B-4-6-1a through 5B4-6-4c; Figures 5B4-6-1 through 5B4-6-18), by which time delta smelt juveniles would be large enough to be excluded from entrainment by the Barker Slough Pumping Plant fish screens (as described by DWR [2020:3-5], the fish screens exclude fish larger than about 25 mm).

Flow-Related Effects

Spring (March–May) X2 correlates with the density of the delta smelt zooplankton prey *Eurytemora affinis* (Kimmerer 2002a; Greenwood 2018), suggesting a potential positive effect of Delta outflow on this prey species, and thereby affecting individual delta smelt growth and survival per the IEP MAST conceptual model (Interagency Ecological Program Management, Analysis, and Synthesis Team 2015:88). The CALSIM II and DSM2 modeling indicates that Alternatives 1, 2, and 3 would have somewhat lower Delta outflow and somewhat greater X2 in March (ranging from a difference of 0.0 km between the mean X2 under the Alternatives 1, 2, and 3 and the NAA in Wet Water Years up to 0.4 km greater mean X2 in Below Normal Years [Alternative 3] and Dry Water Years [Alternative 1B] compared to the NAA), with little difference in April–May (Table 5B3-5-1a–c, Table 5B3-5-2a–c, Table 5B3-5-3a–c, and Table 5B3-5-4a–c in Appendix 5B; see also Table 6-16 in Chapter 6). For this impact analysis, the negative relationship between *E. affinis* density and X2 developed by Greenwood (2018) was applied (Appendix 11F). The results of the analysis suggest that spring food density for delta smelt under Alternatives 1, 2, and 3 would only be minimally negatively affected (Table 11-70). Note that there is appreciable uncertainty in the predictions of *E. affinis* density as a function of X2, with 95% prediction intervals generally spanning two to three orders of magnitude (Tables 11F-1 through 11F-5 in Appendix 11F).

The broad 95% prediction intervals indicate that the 1% estimated mean difference in density of *E. affinis* as a result of operations-related changes in Delta outflow under the Alternatives 1, 2, and 3 compared to the NAA would be unlikely to be statistically detectable given the estimated variability in the underlying relationship. Hennessy and Burris (2017) developed a similar statistical relationship, but limited their analysis to two zooplankton sampling stations within the entrapment zone (a region of high delta smelt abundance), at salinity of 1 ppt and 3 ppt, and used March–June Delta outflow as the predictor of *E. affinis* density. Application of their statistical relationship for this effects analysis gave similarly low differences between Alternatives 1, 2, and 3 and the NAA (Table 11-71) as the X2-*E. affinis* relationship (Table 11-70); as with the X2-*E. affinis* relationship, such small differences would be unlikely to be statistically detectable.

Table 11-70. Density of Adult *Eurytemora affinis* Based on March–May Regression with X2.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	184	183 (0%)	184 (0%)	183 (0%)	183 (0%)
Above Normal	169	169 (0%)	169 (0%)	169 (0%)	169 (0%)
Below Normal	133	133 (0%)	133 (0%)	133 (0%)	132 (-1%)
Dry	108	107 (-1%)	107 (-1%)	107 (-1%)	107 (-1%)
Critically Dry	77	77 (0%)	77 (0%)	77 (0%)	77 (0%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. Table only includes annual mean responses and does not consider model uncertainty.

Table 11-71. Density of Adult + Juvenile *Eurytemora affinis* Based on March–June Regression with Delta Outflow (Hennessy and Burris 2017).

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	1,022	1,017 (-1%)	1,017 (-1%)	1,017 (-1%)	1,015 (-1%)
Above Normal	690	681 (-1%)	681 (-1%)	681 (-1%)	681 (-1%)
Below Normal	405	400 (-1%)	400 (-1%)	400 (-1%)	399 (-1%)
Dry	303	300 (-1%)	299 (-1%)	300 (-1%)	300 (-1%)
Critically Dry	197	196 (-1%)	196 (-1%)	196 (-1%)	196 (-1%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. Table only includes annual mean responses and does not consider model uncertainty. The regression equation is y = 0.00381 + 5.995x, where x = 1/sqrt(*E. affinis)* number per cubic meter, and y = 1/sqrt(mean March–June Delta outflow) [R^2 = 0.58]. Percentage values are rounded; as a result, differences between percentages may not always appear consistent.

The IEP MAST conceptual model (2015) suggests that the probability of egg/larval delta smelt surviving to the juvenile life stage is influenced by predation risk, which may involve different factors such as turbidity, water temperature, and predators (silversides). Operations have limited potential to affect water temperature in the Delta (Wagner et al. 2011), and turbidity during spring would be expected to be similar under the NAA and Alternatives 1, 2, and 3; see also examination of *Upstream Sediment Entrainment* discussed below. The main potential factor being influenced by Alternatives 1, 2, and 3 would be predators (silversides), for which Mahardja et al. (2016) showed that summer (June–September) Delta inflow and spring (March–May) South Delta exports had the strongest correlations with silverside cohort strength (both relationships were negative). Mahardja et al. (2016:12) cautioned that the relationships are not meant to imply causality, given that the mechanisms could not be identified, and that further investigation is merited. Nonetheless, should the relationships be found to be causative, examination of CALSIM-modeled inflow and south Delta exports for Alternatives 1, 2, and 3 allows inference of potential effects on delta smelt predation. The CALSIM modeling results indicate that June–September inflow under Alternatives 1, 2, and 3 would be similar to the NAA in Wet, Above Normal, and Below Normal Water Years, or slightly greater (6%–8%) in Dry and Critically Dry Water Years (Table 11-72). March–May south Delta exports (Table 11-73) generally would have a 0%–1% difference between the NAA and Alternatives 1, 2, and 3 (Table 11-73). Taken together, these results suggest that there would be little difference in silverside predation of delta smelt between alternatives and if there were any difference, predation would be expected to be less under Alternatives 1, 2, and 3 compared to the NAA because of slightly greater June–September inflow under Alternatives 1, 2, and 3.

Table 11-72. Mean June–September Delta Inflow (Cubic Feet per Second) by Alternative and Water Year Type.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	27,016	27,215 (1%)	27,216 (1%)	27,222 (1%)	27,210 (1%)
Above Normal	22,584	22,775 (1%)	22,792 (1%)	22,791 (1%)	22,788 (1%)
Below Normal	18,359	18,536 (1%)	18,551 (1%)	18,554 (1%)	18,516 (1%)
Dry	14,179	15,058 (6%)	15,040 (6%)	15,038 (6%)	15,002 (6%)
Critically Dry	10,330	11,149 (8%)	11,100 (7%)	11,099 (7%)	10,883 (5%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. Delta inflow is calculated as sum of flows: Sacramento River at Freeport (CALSIM channel C169); Yolo Bypass (CALSIM channel C157); Mokelumne River (CALSIM channel C504); and San Joaquin River at Vernalis (CALSIM channel C639). Percentage values are rounded; as a result, differences between percentages may not always appear consistent.

Table 11-73. Mean March–May South Delta Exports (Cubic Feet Per Second) by Alternative and Water Year Type.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	7,246	7,226 (0%)	7,214 (0%)	7,226 (0%)	7,162 (-1%)
Above Normal	5,223	5,183 (-1%)	5,185 (-1%)	5,186 (-1%)	5,192 (-1%)
Below Normal	4,575	4,579 (0%)	4,577 (0%)	4,579 (0%)	4,574 (0%)
Dry	3,415	3,418 (0%)	3,442 (1%)	3,418 (0%)	3,454 (1%)
Critically Dry	2,570	2,584 (1%)	2,657 (3%)	2,589 (1%)	2,667 (4%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. Percentage values are rounded; as a result, differences between percentages may not always appear consistent.

The IEP MAST (2015) conceptual model describes food availability and quality as key components of the June through September transition probability of juvenile delta smelt to subadulthood through growth and survival of individuals. Delta outflow is positively correlated with the subsidy of the delta smelt zooplankton prey *Pseudodiaptomus forbesi* to the low-salinity zone from the freshwater Delta during July–September (Kimmerer et al. 2018). During these months, Delta outflow under Alternatives 1, 2, and 3 would be similar or somewhat greater than under the NAA, indicating that the subsidy of *P. forbesi* would not be expected to be negatively affected by differences in outflow as a result of Alternatives 1, 2, and 3 (Table 11-74). Hennessy and Burris (2017) developed a statistical relationship between *P. forbesi* density at 15 zooplankton sampling stations in Suisun Bay and June–September Delta outflow. Application of their statistical relationship for this effects analysis suggested a 9%–12% greater mean *P. forbesi* density under Alternatives 1, 2, and 3 compared to the NAA in drier years (Table 11-75) as a result of generally higher Delta outflow in these water year types (see Table 11-74 and Appendix 5B3, Tables 5B3-5-1 through 5B3-5-4), with similar density in wetter years (Table 11-75). Although greater than the differences suggested by the X2-*E. affinis* relationship, these differences are also uncertain and would also be likely to be difficult to statistically detect because of the relatively low explanatory power of the statistical relationship ($R^2 = 0.39$; Hennessy and Burris 2017).

Table 11-74. Mean July–September Delta Outflow (Cubic Feet Per Second) by Alternative and Water Year Type.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	10,707	10,959 (2%)	10,953 (2%)	10,967 (2%)	10,944 (2%)
Above Normal	9,190	9,435 (3%)	9,474 (3%)	9,458 (3%)	9,498 (3%)
Below Normal	5,018	5,144 (3%)	5,145 (3%)	5,151 (3%)	5,156 (3%)
Dry	3,972	4,177 (5%)	4,177 (5%)	4,174 (5%)	4,161 (5%)
Critically Dry	3,519	3,671 (4%)	3,666 (4%)	3,704 (5%)	3,615 (3%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. Percentage values are rounded; as a result, differences between percentages may not always appear consistent.

Table 11-75. Density of Adult + Juvenile *Pseudodiaptomus forbesi* Based on June–September Regression with Delta Outflow (Hennessy and Burris 2017).

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	990	1,005 (2%)	1,003 (1%)	1,005 (2%)	1,004 (1%)
Above Normal	740	757 (2%)	759 (3%)	759 (3%)	759 (3%)
Below Normal	312	323 (3%)	322 (3%)	323 (4%)	322 (3%)
Dry	191	208 (9%)	208 (9%)	208 (9%)	207 (8%)
Critically Dry	117	128 (9%)	127 (9%)	130 (12%)	124 (6%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. Table only includes annual mean responses and does not consider model uncertainty. The regression equation is $y = 7.896 - 12550x$, where x = natural log (*P. forbesi* number per cubic meter), and y = 1/ (mean June–September Delta outflow) [$R^2 = 0.39$]. Percentage values are rounded; as a result, differences between percentages may not always appear consistent.

South Delta exports may entrain *P. forbesi* (U.S. Fish and Wildlife Service 2008:228; Kimmerer et al. 2019), and density of *P. forbesi* on the San Joaquin River side of the Delta is relatively high (Kimmerer et al. 2018), so modeled flows in the lower San Joaquin River (QWEST) may provide an indicator of *P. forbesi* downstream subsidy potential to the low-salinity zone (California Department of Water Resources 2020:4-149). There is uncertainty in this mechanism and the magnitude of QWEST flow differences that may be of consequence. Based on the assumption that net positive QWEST provides an indicator of potential downstream subsidy to the low-salinity zone (California Department of Water Resources 2020:4-149), CALSIM modeling showed that Alternatives 1, 2, and 3 had a slightly lower frequency of positive flows in July (16%–17% of years compared to 20% of years under the NAA) and August (6% of years compared to 11% under the NAA), but not September (11% of years under the NAA and Alternatives 1, 2, and 3). Given the few years with positive QWEST under any scenario and uncertainty in the extent to which these modeled differences would be of consequence because of the high rate of grazing in the low-salinity zone (Kimmerer et al. 2019), as well as the distribution of an appreciable portion of delta smelt upstream of the low-salinity zone, i.e., an average of 23% (range 2%–47%) during 2005–2014 (Bush 2017), the differences between scenarios would likely be small. Coupled with the overall increase in June–September Delta outflow under Alternatives 1, 2, and 3 previously discussed, the differences in subadult delta smelt food availability between Alternatives 1, 2, and 3 and the NAA are likely to be limited.

As discussed in the ROC ON LTO Biological Assessment (Bureau of Reclamation 2019:5-385–5-386), various factors have the potential to affect harmful algal blooms (HABs; in particular *Microcystis*) that could affect delta smelt or its prey. Among these factors, RBI (2017) focused on maximum absolute daily velocity as an indicator of turbulent mixing potential, which could disrupt *Microcystis* blooms. The ROC ON LTO Biological Assessment analysis compared a proposed action operations scenario to a without action scenario with no exports and no south Delta temporary barriers, and generally found little difference in *Microcystis* bloom potential between the two because of generally similar maximum absolute daily velocity (Bureau of Reclamation 2019:5-385–5-390). Where differences arose, it was as a result of the south Delta temporary barriers being in place in the proposed action but not being in place in the without action scenario. Given that the south Delta temporary barriers are present in the 2020 baseline conditions, there would be little difference expected in terms of HAB potential between Alternatives 1, 2, and 3 and the NAA.

The IEP MAST (2015:88) delta smelt conceptual model also includes predation risk as an important habitat attribute affecting juvenile survival from June to September (ROC ON LTO BA; Bureau of Reclamation 2019:5-390 to 5-391). The operations-related factors that could influence predation risk would not be expected to differ greatly between Alternatives 1, 2, and 3 and the NAA: turbidity as a function of sediment (discussed below in *Upstream Sediment Entrainment* section), striped bass abundance (based on general similarity in fall X2; see analysis below), and water temperature (which is driven mainly by air temperature and is only slightly affected by freshwater inflow; Wagner et al. 2011).

The IEP MAST (2015:89) delta smelt conceptual model posits that size and location of the low-salinity zone is a habitat attribute that could affect subadult delta smelt survival. The summer-fall delta smelt habitat action from the USFWS (2019) ROC ON LTO BiOp and CDFW (2020) ITP would be undertaken under Alternatives 1, 2, and 3, so differences in the size and location of the low-salinity zone would be limited to differences caused by reservoir releases under Alternatives 1, 2, and 3. DSM2 X2 modeling shows that X2 generally would be similar or somewhat farther downstream under Alternatives 1, 2, and 3 compared to the NAA (Table 11-76), indicating that Alternatives 1, 2, and 3 would not reduce low-salinity zone habitat for delta smelt relative to the NAA based on the negative relationship with X2 (Feyrer et al. 2011). Note that there is debate regarding the importance of low-salinity zone habitat to delta smelt (e.g., Manly et al. 2015; Feyrer et al. 2015; Murphy and Weiland 2019; Greenwood 2018). The IEP MAST (2015:89) conceptual model also posits that food availability affects subadult delta smelt survival, which was discussed above for the early fall period during which there was concluded to be little potential effect of Alternatives 1, 2, and 3 relative to the NAA. The IEP MAST (2015:89) conceptual model also posits turbidity as an important factor affecting subadult delta smelt predation risk, which is discussed below in the next section as a result of upstream sediment entrainment caused by operations for Alternatives 1, 2, and 3.

Table 11-76. Mean X2 (km upstream of Golden Gate Bridge) by Water Year Type, September–November.

Month	Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
September	Wet	77.9	77.3 (-0.5)	77.4 (-0.5)	77.3 (-0.5)	77.4 (-0.4)
September	Above Normal	79.7	79.2 (-0.5)	79.1 (-0.6)	79.2 (-0.5)	79.1 (-0.6)
September	Below Normal	90.1	89.7 (-0.4)	89.7 (-0.4)	89.7 (-0.4)	89.7 (-0.4)
September	Dry	91.8	91.0 (-0.7)	91.0 (-0.8)	91.1 (-0.7)	91.1 (-0.7)
September	Critically Dry	92.9	92.2 (-0.7)	92.2 (-0.7)	92.1 (-0.7)	92.5 (-0.3)
October	Wet	77.7	77.4 (-0.3)	77.4 (-0.3)	77.4 (-0.3)	77.5 (-0.2)
October	Above Normal	80.3	79.8 (-0.4)	79.7 (-0.6)	79.8 (-0.5)	79.6 (-0.6)
October	Below Normal	89.6	89.2 (-0.4)	89.2 (-0.4)	89.2 (-0.4)	88.2 (-1.4)
October	Dry	92.3	91.4 (-0.8)	91.4 (-0.9)	91.4 (-0.8)	91.6 (-0.7)
October	Critically Dry	92.6	92.3 (-0.3)	92.4 (-0.2)	92.3 (-0.3)	92.5 (-0.1)
November	Wet	78.9	79.0 (0.1)	79.1 (0.2)	79.0 (0.1)	79.1 (0.2)

Month	Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
November	Above Normal	82.3	82.1 (-0.1)	82.1 (-0.2)	82.1 (-0.2)	81.7 (-0.6)
November	Below Normal	84.9	85.3 (0.3)	85.2 (0.3)	85.3 (0.3)	83.5 (-1.5)
November	Dry	89.1	88.7 (-0.4)	88.8 (-0.3)	88.7 (-0.4)	88.9 (-0.3)
November	Critically Dry	92.2	92.1 (-0.1)	92.1 (0.0)	92.1 (-0.1)	92.1 (-0.1)

Note: Values in parentheses indicate differences of alternatives compared to the NAA.

In addition to the various habitat indicators discussed above that could be affected by differences in flow, recently published statistical analyses have provided evidence for flow-related variables being directly associated with delta smelt survival and recruitment (Polansky et al. 2021; Smith et al. 2021). Based on the selection of a particular threshold for statistical evidence considered substantial enough to report on[31], Polansky et al. (2021) found September–November X2 and March–May Delta exports to inflow ratio (E:I) to be negatively related to recruitment (i.e., the probability of transitioning from adults to larvae) and June to August Delta outflow to be positively related to post-larval survival. In the context of this effects analysis and the comparison of Alternatives 1, 2, and 3 to the NAA, as discussed above, September–November X2 would be similar or less under Alternatives 1, 2, and 3 (Table 11-76), so there would not be a negative effect on delta smelt recruitment based on this variable. For March–May E:I, the modeling indicated very little difference between Alternatives 1, 2, and 3 and the NAA (Table 11-77). There was also very little difference in June–August Delta outflow between Alternatives 1, 2, and 3 and the NAA, with the small differences indicating greater Delta outflow under Alternatives 1, 2, and 3 than the NAA (Table 11-78). Note that of the three flow-related variables considered to have substantial enough evidence to be reported on by Polansky et al. (2021), the subsequent analysis by Smith et al. (2021) only used the June–August Delta outflow variable because it had the most evidence of an effect meeting a threshold for inclusion in their analysis. Overall, the small differences in the flow-related variables indicate that based on these variables, there would be little difference in delta smelt recruitment or survival between Alternatives 1, 2, and 3 and the NAA, with any small differences in the flow-related variables being positive in favor of Alternatives 1, 2, and 3 with respect to potential effects on delta smelt (i.e., greater June–August outflow and lower September–November X2 than the NAA).

Table 11-77. Mean March–May Delta Exports to Inflow Ratio (E:I) by Water Year Type.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	0.14	0.14 (0.00)	0.14 (0.00)	0.14 (0.00)	0.14 (0.00)
Above Normal	0.15	0.15 (0.00)	0.15 (0.00)	0.15 (0.00)	0.15 (0.00)
Below Normal	0.19	0.19 (0.00)	0.19 (0.00)	0.19 (0.00)	0.19 (0.00)
Dry	0.20	0.20 (0.00)	0.20 (0.00)	0.20 (0.00)	0.20 (0.00)
Critically Dry	0.20	0.20 (0.00)	0.20 (0.01)	0.20 (0.00)	0.21 (0.01)

Note: Values in parentheses indicate differences of alternatives compared to the NAA.

[31] This threshold was evidence of 0.80, i.e., at least 80% of the relationship slope's posterior distribution was above or below zero when the expected effect is positive or negative, respectively.

Table 11-78. Mean June–August Delta Outflow (Thousand Acre-Feet) by Water Year Type.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	2,565	2,586 (1%)	2,583 (1%)	2,586 (1%)	2,584 (1%)
Above Normal	1,699	1,718 (1%)	1,716 (1%)	1,720 (1%)	1,717 (1%)
Below Normal	1,201	1,211 (1%)	1,212 (1%)	1,213 (1%)	1,211 (1%)
Dry	967	992 (3%)	992 (3%)	992 (3%)	990 (2%)
Critically Dry	786	804 (2%)	803 (2%)	809 (3%)	800 (2%)

Note: Values in parentheses indicate differences of alternatives compared to the NAA.

<u>**Upstream Sediment Entrainment**</u>

The IEP MAST (2015) conceptual model identifies predation risk as a habitat attribute affecting delta smelt. Flows interact with erodible sediment supply to affect turbidity (Ferrari et al. 2014). In general, greater turbidity is thought to lower the risk of predation on delta smelt. Large amounts of sediment enter the Delta from winter and spring storm runoff, with resuspension by tidal and wind action. A conceptual model of sedimentation in the Delta includes a submodel for river supply, which notes that dams and reservoirs have contributed to decreased sediment supply to the Delta (Schoellhamer et al. 2012:Figure 4). However, a recent analysis examining future climate scenarios predicted significant increases in large flow events and sediment transport over the next century, which may increase turbidity (Stern et al. 2020). As described in Chapter 7, only construction, maintenance, and operation activities on the Sacramento River—not the effects of Sites Reservoir in capturing sediment from its small upstream tributaries—would have the potential to affect the fluvial geomorphology of the Sacramento River.

Modeling presented in Appendix 11F, Section 11F.3, *Upstream Sediment Entrainment*, estimated that over the 82-year simulation period approximately 2.6%–2.7% of suspended sediment reaching the Red Bluff intake was entrained under the Project, compared to 1.2% under the NAA. For the Hamilton City intake, the estimates were 2.1% for the Alternatives 1, 2, and 3 and 1.8% for the NAA. Because water and sediment would both be diverted, the concentration of the sediment in the water would remain unchanged, so the turbidity of the water would be expected to remain the same at the time the water is being diverted (i.e., principally in the winter/spring).

The slightly reduced sediment load to the Delta under Alternatives 1, 2, and 3 (i.e., suspended sediment entrainment of 2.6%–2.7% of the load at Red Bluff and 2.1% of the load at Hamilton City, compared to 1.2% at Red Bluff and 1.8% at Hamilton City under the NAA) may have only slight effects on turbidity as a result of the reduction in sediment for resuspension at other times of the year[32], but there is some uncertainty in this conclusion. This uncertainty would be addressed through the Sediment Technical Studies Plan and Adaptive Management for Sacramento River (Appendix 2D, Section 2D.5, *Sediment Monitoring Plan and Adaptive Management for Sediment Diverted from the Sacramento River*). This process will include sediment monitoring and modeling to and collaboration with ongoing monitoring and restoration

[32] As described in Chapter 7, *Fluvial Geomorphology*, Impact FLV-1, partial capture of sediment by Sites Reservoir that otherwise could have entered the Sacramento River via Funks and Stone Corral Creeks has the potential to reduce sediment delivered to the Sacramento River; this is inferred to be a small fraction of total sediment yield to the Sacramento River.

efforts to determine whether adaptive management measures such as sediment reintroduction are warranted based on estimated effects on turbidity.

Available estimates of sediment removal by the south Delta export facilities are low, i.e., ~2% of sediment entering the Delta at Freeport in 1999–2002 (Wright and Schoellhamer 2005). This suggests that south Delta exports in association with Alternatives 1, 2, and 3 would remove only a small percentage of sediment entering the Delta. Given the limited expected difference in suspended sediment entering the Delta under Alternatives 1, 2, and 3 relative to the NAA, as well as the small percentage of sediment that would be expected to be removed by the south Delta export facilities and the similarity in south Delta exports between Alternatives 1, 2, and 3 and the NAA during the high-flow period of the year when most sediment is delivered to the Delta (previously discussed in the *South Delta Entrainment* section), the potential negative effect of Alternatives 1, 2, and 3 on turbidity would be expected to be low. As previously noted, uncertainty in the conclusion would be addressed through the Sediment Technical Studies Plan and Adaptive Management for Sacramento River.

<u>CEQA Significance Determination and Mitigation Measures for Alternatives 1, 2, and 3</u>

The preceding subsections of this impact discussion provide the detailed information used for this CEQA (and NEPA) determination. This section provides a summary of this information.

Operations impacts of Alternatives 1, 2, and 3 on delta smelt include small differences assessed for flow-related zooplankton prey and other flow-related habitat attributes during spring, summer, and fall; no increase in south Delta entrainment risk because south Delta exports of Sites Reservoir water do not occur during times of the year when delta smelt are susceptible to entrainment; small reductions in suspended sediment to the Delta, addressed by the Sediment Technical Studies Plan and Adaptive Management for Sacramento River; and potential positive effects from summer/fall Sites Reservoir releases to move foodweb materials into the lower Yolo Bypass and Cache Slough Complex, as well as potential positive effects on prey from greater summer/fall Delta outflow. These impacts would be less than significant.

Impacts on delta smelt would be significant due to uncertainty associated with DO and temperature effects from Sites Reservoir releases (see *Effects from Reservoir Releases to CBD/Yolo Bypass* above) and the population status of delta smelt (Appendix 11A). Mitigation Measure FISH-8.1 will reduce this significant impact by preventing detrimental DO and water temperature effects associated with moving CBD water through the Yolo Bypass. DO and temperature levels suitable to delta smelt would be maintained and would not exceed recognized critical physiological thresholds through implementation of Mitigation Measure FISH-8.1; therefore, impacts would be reduced to less than significant. There is uncertainty in the potential for negative effects from Sites habitat flows redirecting CBD water relatively high in pesticides downstream to the lower Yolo Bypass where delta smelt occur. This potential effect would be addressed by Mitigation Measure WQ-2.2. Operation of Alternative 1, 2, or 3 would not have a substantial adverse effect, either directly or through habitat modifications, on delta smelt. Operational impacts for Alternatives 1, 2, and 3 on delta smelt would be less than significant with mitigation.

Mitigation Measure FISH-8.1: Prevent Detrimental Dissolved Oxygen and Water Temperature Effects on Fish Associated with Moving Colusa Basin Drain Water Through the Yolo Bypass

To evaluate potential water quality effects, when Project releases are made via the Dunnigan Pipeline to the Yolo Bypass DO and water temperature will be measured at 15-minute intervals within 50 feet of the Project discharge location at the Dunnigan Pipeline, at existing California Data Exchange Center stations at the upstream end of the Yolo Bypass at Ridge Cut Slough, and at the downstream end at Lisbon Weir. Measurements of DO and water temperature will occur before and during the period of CBD discharge to the Yolo Bypass, the same as is described for Mitigation Measure WQ-2.2.

Downstream DO and temperature measurements, together with water quality measurements of water released from Sites Reservoir, will be evaluated to determine whether habitat flow releases from Sites Reservoir would lower DO and increase temperatures in the Yolo Bypass Toe Drain and Cache Slough Complex to a level that could be detrimental to delta smelt inhabiting these areas. Dissolved oxygen and temperature criteria for determining effects will be developed in collaboration with the fishery agencies and will maintain existing DO and temperature levels suitable to delta smelt that will not exceed recognized critical physiological thresholds. This evaluation will be part of ongoing monitoring to determine benefits of the Yolo Bypass habitat flows and the Project's funded ecosystem benefits under WSIP. CDFW would have the discretion to modify WSIP water that is released to Yolo Bypass, depending on best available science and fish needs. If measurements indicate DO or temperature criteria are exceeded in the Yolo Bypass Toe Drain and Cache Slough Complex as a result of Project releases and these criteria cannot be maintained for delta smelt, actions to improve DO concentration and temperature will be implemented. Mitigative actions may include, but are not limited to one or more of the following types of measures:

- Use of engineered actions (e.g., installation of aerators) to prevent exceedance of critical physiological thresholds for delta smelt.
- Cessation of releases of flow to the Yolo Bypass until temperature and DO concentration do not exceed critical physiological thresholds for delta smelt.

Mitigation Measure WQ-2.2: Prevent Net Detrimental Metal and Pesticide Effects Associated with Moving Colusa Basin Drain Water Through the Yolo Bypass

This measure is described in Chapter 6, Section 6.4, *Impact Analysis and Mitigation Measures*.

<u>*NEPA Conclusion for Alternatives 1, 2, and 3*</u>

Operations effects on delta smelt would be the same as described above for CEQA. Operations effects from Alternatives 1, 2, and 3 include small differences assessed for flow-related zooplankton prey and other flow-related habitat attributes during spring, summer, and fall; no increase in south Delta entrainment risk because south Delta exports of Sites Reservoir water do not occur during times of the year when delta smelt are susceptible to entrainment; small reductions in suspended sediment to the Delta, addressed by the Sediment Technical Studies Plan and Adaptive Management for Sacramento River; and potential positive effects from summer/fall Sites Reservoir releases to move foodweb materials into the lower Yolo Bypass and Cache Slough Complex, as well as potential positive effects on prey from greater summer/fall Delta outflow.

The effects from operations of Alternative 1, 2, or 3 would be adverse compared to the NAA because of uncertainty associated with DO and temperature effects from Sites Reservoir releases and the population status of delta smelt. Mitigation Measure FISH-8.1 will prevent detrimental DO and water temperature effects on delta smelt associated with the discharge of CBD water to the Yolo Bypass. Existing DO and temperature levels suitable to delta smelt would be maintained and would not exceed recognized critical physiological thresholds with implementation of Mitigation Measure FISH-8.1. There is uncertainty regarding potential adverse effects of Alternative 1, 2, or 3 from Sites habitat flows for redirecting water relatively high in pesticides downstream to the lower Yolo Bypass where delta smelt occur, as compared to the NAA. Implementation of Mitigation Measure WQ-2.2 will address this potential effect. As described above for CEQA, effects would be reduced and would not be adverse with implementation of Mitigation Measures FISH-8.1 and WQ-2.2. Operation of Alternative 1, 2, or 3 would have no adverse effect on delta smelt.

Impact FISH-9: Operations Effects on Longfin Smelt

Alternatives 1, 2, and 3

<u>*Effects of Alternatives 1, 2, and 3*</u>

As described in Appendix 11A, of the geographic areas potentially affected by Alternatives 1, 2, and 3, longfin smelt occur within the Delta and Suisun Bay/Marsh.

<u>Effects from Reservoir Releases to CBD/Yolo Bypass</u>

Longfin smelt would not be affected by summer/fall flow releases from the CBD because the species occurs well downstream of the Delta at this point in its life cycle (Appendix 11A) (Merz et al. 2013).

<u>Entrainment</u>

Key hydrodynamic variables influencing longfin smelt entrainment risk at the south Delta export facilities include Old and Middle River flow (Grimaldo et al. 2009) and flow in the lower San Joaquin River near Jersey Point (QWEST) (California Department of Fish and Game 2009). During the main winter-spring months of potential adult, larval, and juvenile longfin smelt entrainment risk (i.e., December through May), CALSIM modeling indicates little difference in Old and Middle River flows between the NAA and Alternatives 1, 2, and 3 (Appendix 5B3, Figures 5B3-6-9 through 5B3-6-14). There was also little difference in QWEST between Alternatives 1, 2, and 3 and the NAA for the key larval entrainment months of January through March (Tables 11-79, 11-80, and 11-81). CDFG (2009) showed that longfin smelt salvage divided by prior fall midwater trawl index is positively correlated with higher mean December–March X2. CALSIM modeling indicates that there would be little difference in mean December–March X2 between the NAA and Alternatives 1, 2, and 3 (Table 11-82). This and the small differences between Old and Middle River flows and QWEST, together with the fact that south Delta entrainment risk for longfin smelt would continue to be limited based on the CDFW (2020) State ITP under the NAA and Alternatives 1, 2, and 3, suggests that the effect from Alternatives 1, 2, and 3 on longfin smelt south Delta entrainment risk would be similar to the NAA. As discussed for delta smelt, differences in Barker Slough Pumping Plant diversions would occur during July–October and would be relatively small (10–20 cfs or less); the timing of these differences would not result in a change in entrainment risk to longfin smelt because the species is downstream of the Delta at this time of year.

Table 11-79. Mean January QWEST (Cubic Feet Per Second) by Alternative and Water Year Type.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	13,969	13,906	13,890	13,909	13,895
Above Normal	6,252	6,103	6,099	6,103	6,107
Below Normal	1,673	1,602	1,586	1,600	1,571
Dry	-552	-600	-601	-600	-613
Critically Dry	-743	-807	-815	-810	-762

Table 11-80. Mean February QWEST (Cubic Feet Per Second) by Alternative and Water Year Type.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	16,034	15,986	15,986	16,000	15,986
Above Normal	9,237	9,120	9,177	9,114	9,207
Below Normal	5,441	5,359	5,358	5,353	5,370
Dry	1,195	1,117	1,124	1,117	1,082
Critically Dry	-494	-498	-509	-497	-506

Table 11-81. Mean March QWEST (Cubic Feet Per Second) by Alternative and Water Year Type.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	16,611	16,632	16,630	16,649	16,642
Above Normal	9,919	9,804	9,796	9,808	9,778
Below Normal	4,098	4,031	4,028	4,032	3,997
Dry	2,089	1,999	1,994	1,999	2,010
Critically Dry	361	279	86	267	72

Table 11-82. Mean December–March X2 (Kilometers Upstream of Golden Gate Bridge) by Alternative and Water Year Type.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	62.3	62.6 (0.2)	62.6 (0.2)	62.6 (0.2)	62.5 (0.2)

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Above Normal	70.8	71.0 (0.3)	71.0 (0.3)	71.0 (0.3)	70.9 (0.2)
Below Normal	69.4	69.3 (0.0)	69.4 (0.1)	69.3 (0.0)	69.3 (0.0)
Dry	70.8	70.9 (0.2)	70.9 (0.1)	70.9 (0.2)	70.9 (0.1)
Critically Dry	76.2	76.4 (0.2)	76.4 (0.2)	76.4 (0.1)	76.4 (0.2)

Note: Values in parentheses indicate absolute differences (km) of alternatives compared to the NAA.

<u>**Flow-Related Effects**</u>

Winter-spring diversions for Alternatives 1, 2, and 3 would reduce Delta inflow and Delta outflow (Appendix 5B3, Tables 5B3-5-1a to 5B3-5-4c and Figures 5B3-5-1 to 5B3-5-18). The analysis of potential negative effects on the smelt zooplankton prey *Eurytemora affinis* previously discussed for delta smelt showed minimal potential negative effects as a result of Alternatives 1, 2, and 3 (Table 11-70). The density of mysids, another longfin smelt prey item, is positively correlated with spring Delta outflow and negatively correlated with spring X2 (Mac Nally et al. 2010). Hennessy and Burris (2017) developed a statistical relationship between the density of the mysid *Neomysis mercedis* in the entrapment zone and March–May Delta outflow. Application of their statistical relationship for this effects analysis suggested mean density up to 3% lower under Alternatives 1, 2, and 3 relative to the NAA, depending on water year type (Table 11-83). As with the *E. affinis* analyses, the differences for *N. mercedis* are uncertain and would also be likely to be difficult to statistically detect because of the relatively low explanatory power of the statistical relationship ($R^2 = 0.32$; Hennessy and Burris 2017).

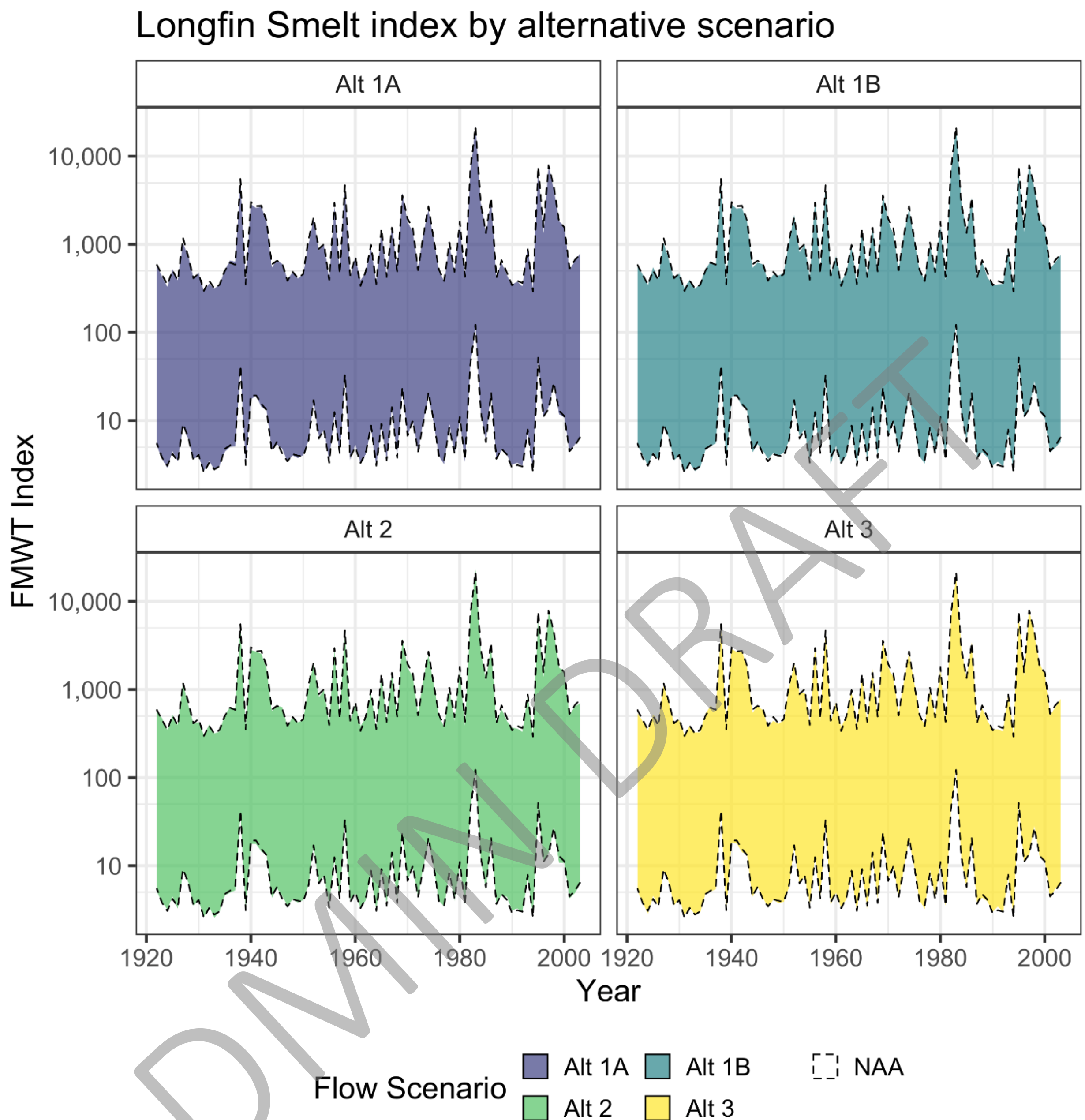
Table 11-83. Density of *Neomysis mercedis* Based on March–May Regression with Delta Outflow (Hennessy and Burris 2017).

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	0.08	0.08 (-1%)	0.08 (-1%)	0.08 (-1%)	0.08 (-1%)
Above Normal	0.05	0.05 (-2%)	0.05 (-2%)	0.05 (-2%)	0.05 (-2%)
Below Normal	0.02	0.02 (-2%)	0.02 (-2%)	0.02 (-2%)	0.02 (-3%)
Dry	0.01	0.01 (-3%)	0.01 (-3%)	0.01 (-3%)	0.01 (-3%)
Critically Dry	0.00	0.00 (-3%)	0.00 (-3%)	0.00 (-3%)	0.00 (-3%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. Table only includes annual mean responses and does not consider model uncertainty. The regression equation is y = -0.602 – 447.59x, where x = natural log(*N. mercedis)* number per cubic meter + 0.001, and y = 1/sqrt(mean March–May Delta outflow) [$R^2 = 0.32$]. Percentage values are rounded; as a result, differences between percentages may not always appear consistent.

As discussed in Appendix 11A, there is a positive correlation between juvenile longfin smelt abundance in the fall and the preceding winter-spring Delta outflow (also often represented as a negative correlation with X2). DWR (2020:4-177–4-178) presents further discussion of this relationship. A model relating the fall midwater trawl abundance index to winter-spring outflow and the fall midwater trawl abundance index 2 years earlier (as an index of parental stock abundance) was used to compare Alternatives 1, 2, and 3 to the NAA, using Delta outflow outputs from CALSIM; additional detail on the method is provided in Appendix 11F, Section 11F.4, *Delta Outflow-Longfin Smelt Abundance Index Analysis*.

Results of the Delta outflow-abundance index model showed considerable overlap between scenarios in longfin smelt fall midwater abundance indices, with values under Alternatives 1, 2, and 3 generally being slightly lower than the NAA (Figure 11-36; Table 11-84). The mean probability of the fall midwater trawl abundance index being less than the NAA ranged from 0.505 (i.e., 50.5%; Alternative 2 in Critically Dry Water Years, Table 11-85) to 0.539 (i.e., 53.9%; Alternative 3 in Above Normal Water Years, Table 11-85). It should be noted that the variability in the modeling predictions of longfin smelt fall midwater trawl abundance index for a given operational scenario is high relative to the differences between scenarios (Figure 11-36). This variability reflects the uncertainty in model parameter estimates, which results in uncertainty in the extent to which operations-related differences in Delta outflow could affect longfin smelt: differences related to operations may be relatively small compared to differences created by hydrological conditions (e.g., wetter vs. drier years) (California Department of Water Resources 2020:4-178). Nevertheless, the results of the analysis suggest the potential for a small negative effect on longfin smelt.



Note: FMWT Index = longfin smelt fall midwater trawl index. See Appendix 11F for method description.

Figure 11-36. 95% Probability Intervals of Longfin Smelt Fall Midwater Trawl Index by Water Year Type from Delta Outflow-Abundance-Index Model.

Table 11-84. Mean Longfin Smelt Fall Midwater Trawl Index by Water Year Type from Delta Outflow-Abundance Index Model.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	408	398 (-2%)	398 (-2%)	399 (-2%)	396 (-3%)
Above Normal	132	127 (-4%)	127 (-4%)	127 (-4%)	127 (-4%)

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Below Normal	58	57 (-1%)	57 (-2%)	57 (-2%)	57 (-2%)
Dry	61	60 (-1%)	60 (-2%)	60 (-2%)	60 (-2%)
Critically Dry	46	46 (0%)	46 (-1%)	46 (-1%)	45 (-1%)

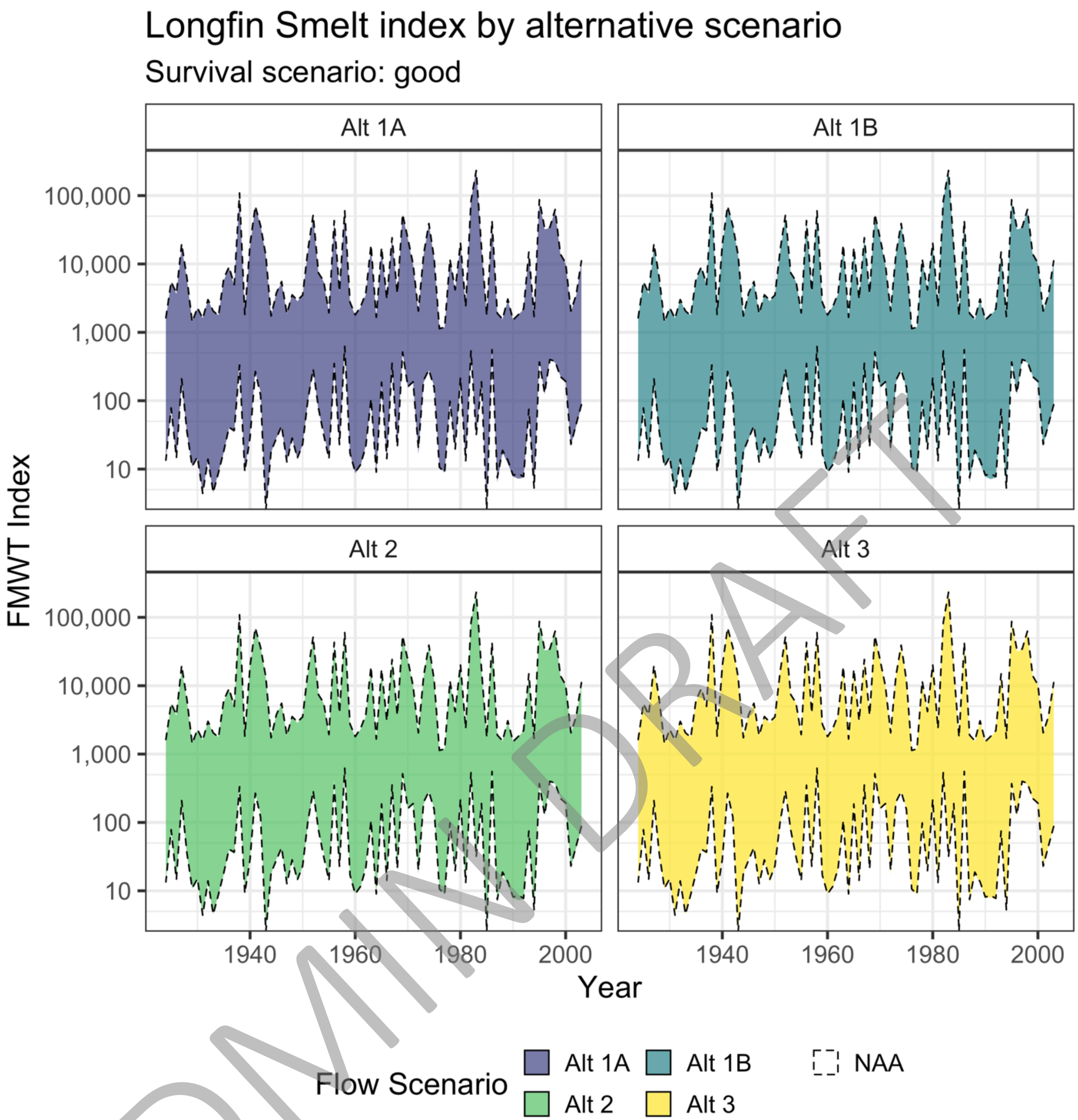
Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. Table only includes annual mean responses and does not consider model uncertainty. Details regarding the method are provided in Appendix 11F. Percentage values are rounded; as a result, differences between percentages may not always appear consistent.

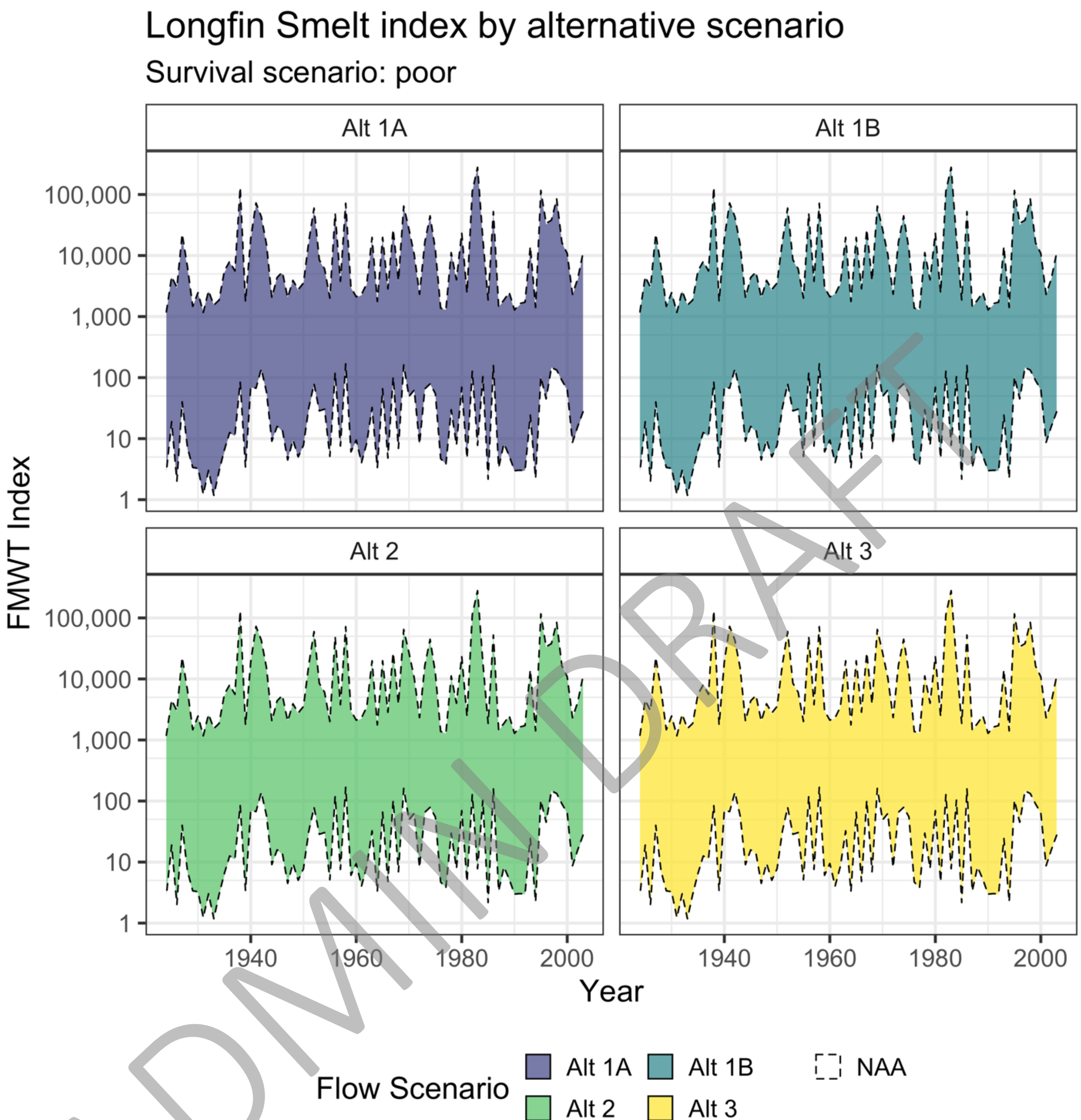
Table 11-85. Mean Longfin Smelt Fall Midwater Trawl Index Probability of Alternatives being less than the NAA by Water Year Type from Delta Outflow-Abundance Index Model.

Water Year Type	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	0.534	0.530	0.523	0.538
Above Normal	0.537	0.537	0.530	0.539
Below Normal	0.520	0.518	0.510	0.522
Dry	0.516	0.516	0.509	0.516
Critically Dry	0.509	0.509	0.505	0.513

Note: Mean probability is based on annual percentage of iterations for which each alternative's fall midwater trawl index was less than the NAA fall midwater trawl index (probability of 0.5 indicates that an equal number of iterations were greater than or less than the NAA fall midwater trawl index value).

Additional analysis was done based on the model of Nobriga and Rosenfield (2016), which is described in Appendix 11F (see Section 11F.5, *Delta Outflow–Longfin Smelt Abundance Analysis (Based on Nobriga and Rosenfield 2016)*). The results of the Nobriga and Rosenfield (2016) method were consistent with the Delta outflow-abundance method in suggesting the potential for small negative effects from Alternatives 1, 2, and 3 relative to the NAA, with high variability around the results relative to the differences (Figure 11-37; Tables 11-86 and 11-87).


Longfin Smelt index by alternative scenario
Survival scenario: good
Alt 1A
Alt 1B
Alt 2
Alt 3
FMWT Index
100,000
10,000
1,000
100
10
1940
1960
1980
2000
Year
Flow Scenario
Alt 1A
Alt 1B
NAA
Alt 2
Alt 3



Note: FMWT Index = longfin smelt fall midwater trawl index. Each chart compares the named alternative (color shades) to the NAA (broken lines). See Appendix 11F for clarification of good vs. poor survival scenarios.

Figure 11-37. 95% Confidence Intervals of Longfin Smelt Fall Midwater Trawl Index by Water Year Type from Nobriga and Rosenfield (2016) Model.

Table 11-86. Mean Longfin Smelt Fall Midwater Trawl Index by Water Year Type from Nobriga and Rosenfield (2016) Model, Based on Good Juvenile Survival Scenario.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	14,179	13,899 (-2%)	13,935 (-2%)	13,906 (-2%)	13,923 (-2%)

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Above Normal	4,482	4,253 (-5%)	4,273 (-5%)	4,253 (-5%)	4,289 (-4%)
Below Normal	1,457	1,420 (-2%)	1,417 (-3%)	1,418 (-3%)	1,424 (-2%)
Dry	848	829 (-2%)	831 (-2%)	829 (-2%)	831 (-2%)
Critically Dry	489	489 (0%)	487 (0%)	489 (0%)	487 (0%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. Table only includes annual mean responses and does not consider model uncertainty. Details regarding the method, including juvenile survival, are provided in Appendix 11F. Percentage values are rounded; as a result, differences between percentages may not always appear consistent.

Table 11-87. Mean Longfin Smelt Fall Midwater Trawl Index by Water Year Type from Nobriga and Rosenfield (2016) Model, Based on Poor Juvenile Survival Scenario.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	10,175	9,871 (-3%)	9,900 (-3%)	9,890 (-3%)	9,879 (-3%)
Above Normal	2,626	2,485 (-5%)	2,494 (-5%)	2,485 (-5%)	2,504 (-5%)
Below Normal	709	690 (-3%)	688 (-3%)	689 (-3%)	692 (-2%)
Dry	440	429 (-2%)	430 (-2%)	429 (-2%)	431 (-2%)
Critically Dry	253	251 (-1%)	251 (-1%)	251 (-1%)	251 (-1%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. Table only includes annual mean responses and does not consider model uncertainty. Details regarding the method, including juvenile survival, are provided in Appendix 11F. Percentage values are rounded; as a result, differences between percentages may not always appear consistent.

A third modeling approach was used to assess potential effects of Alternatives 1, 2, and 3 as a reflection of differences in winter-spring Delta outflow. This modeling approach is considered because it was recently included in the State ITP effects analysis by DWR (2020:Appendix E, Attachment 2: E2-1) at CDFW's behest. The same method was employed herein[33], which produces an estimate of longfin smelt fall midwater trawl index as a function of mean January–June X2. It is acknowledged that the model is limited relative to other models including a representation of parental stock size (i.e., the new Delta outflow-abundance method and the analysis based on Nobriga and Rosenfield [2016], summarized above). The results of the analysis showed that mean fall midwater trawl index under Alternatives 1, 2, and 3 was similar or slightly lower than the NAA (Table 11-88), reflecting somewhat lower Delta outflow and higher X2 during January–June. As with the analyses using the Delta outflow-abundance method and the analysis based on Nobriga and Rosenfield (2016) described above, differences between scenarios were considerably greater than the variability in the estimates for a given scenario as a result of uncertainty in the model's parameters (Tables 11F-10 through 11F-14 in Appendix 11F), but again suggested the potential for a small negative effect.

Table 11-88. Mean Longfin Smelt Fall Midwater Trawl Index Based on January–June X2.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	590	586 (-1%)	586 (-1%)	586 (-1%)	586 (-1%)

[33] The method has been previously referred to as the "Kimmerer regression," reflecting previous similar analyses, e.g., Kimmerer (2002a, 2002b) and Kimmerer et al. (2009); see DWR (2020, Appendix E, Attachment 2:E2-1). The method is described in Appendix 11F.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Above Normal	361	357 (-1%)	357 (-1%)	357 (-1%)	356 (-1%)
Below Normal	159	157 (-1%)	156 (-2%)	157 (-1%)	156 (-2%)
Dry	72	70 (-2%)	70 (-2%)	70 (-2%)	70 (-2%)
Critically Dry	23	23 (-1%)	23 (-2%)	23 (-2%)	23 (-2%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. Table only includes annual mean responses and does not consider model uncertainty. In addition to January–June X2, the predictive model assumed a coefficient representing the Pelagic Organism Decline period (California Department of Water Resources 2020: Appendix E, Attachment 2; method description in Appendix F). Percentage values are rounded; as a result, differences between percentages may not always appear consistent.

CEQA Significance Determination and Mitigation Measures for Alternatives 1, 2, and 3

The analyses of potential impacts of Alternatives 1, 2, and 3 on longfin smelt suggested that entrainment risk under Alternatives 1, 2, and 3 would be similar to entrainment risk under the NAA. The analyses of flow-related effects (differences in Delta outflow/X2) suggested the potential for small negative effects under Alternatives 1, 2, and 3, albeit with uncertainty given the appreciably greater variability of longfin smelt abundance index estimates for a given alternative relative to the difference from the NAA. As identified in Section 11.3, *Methods of Analysis*, operations resulting from Alternatives 1, 2, and 3 would be consistent with all applicable regulations to limit the potential for negative effects on fish and aquatic resources, including the existing spring outflow measures required by the CDFW (2020) State ITP. In order to achieve a less-than-significant impact, mitigation would be required for the small, uncertain negative outflow-related effect of Alternatives 1, 2, and 3 in consideration of longfin smelt's CESA-listed status. Implementation of Mitigation Measure FISH-9.1 would provide tidal habitat restoration mitigation. Tidal habitat restoration would expand the diversity, quantity, and quality of longfin smelt rearing and refuge habitat consistent with recent tidal habitat mitigation required for outflow impacts on the species (California Department of Fish and Wildlife 2020:112). As shown by multiple recent tidal habitat restoration projects in the Delta[34], there are potential feasible opportunities for tidal habitat restoration directly applicable to longfin smelt. Operational impacts for Alternatives 1, 2, and 3 on longfin smelt would be less than significant with mitigation.

Mitigation Measure FISH-9.1: Tidal Habitat Restoration for Longfin Smelt

Tidal habitat restoration mitigation for longfin smelt was calculated based on the same method recently applied by DWR (2019e:5-5). The method is described in more detail in Appendix 11F, Section 11F.7, *Tidal Habitat Restoration Mitigation Calculations for Longfin Smelt*. The mitigation requirement for each alternative varies between 5.1 and 9.7 acres (Table 11-89).

[34] See, for example, the California EcoRestore program's summary of recent projects (California Department of Water Resources 2023).

Table 11-89. Tidal Habitat Restoration Mitigation for Longfin Smelt (Acres).

Alt 1A	Alt 1B	Alt 2	Alt 3
5.1	8.3	5.1	9.7

<u>*NEPA Conclusion for Alternatives 1, 2, and 3*</u>

Operations effects on longfin smelt would be the same as described above for CEQA. The effects of operation of Alternative 1, 2, or 3 on entrainment risk for longfin smelt as compared to the NAA would be similar. Analyses of differences in Delta outflow/X2 for operation of Alternative 1, 2, or 3 suggested the potential for small negative effects, albeit with uncertainty because of the appreciably greater variability of longfin smelt abundance index estimates for a given alternative relative to the difference from the NAA. Implementation of Mitigation Measure FISH-9.1 would be required for the small, uncertain negative outflow-related effect of Alternatives 1, 2, and 3 in consideration of longfin smelt's CESA-listed status. Mitigation Measure FISH-9.1 will provide tidal habitat restoration and expand the diversity, quantity, and quality of longfin smelt rearing and refuge habitat consistent with recent tidal habitat mitigation required for outflow impacts on the species (California Department of Fish and Wildlife 2020:112). Operation of Alternative 1, 2, or 3 would have no adverse effect on longfin smelt.

Impact FISH-10: Operations Effects on Lampreys

Alternatives 1, 2, and 3

<u>*Effects of Alternatives 1, 2, and 3*</u>

<u>**Sacramento River**</u>

Near-Field Effects

Lamprey ammocoetes, smaller than 40–50-mm total length could be entrained by the Red Bluff and Hamilton City intakes if passing close during operational periods. The probability of entrainment will be reduced to almost zero at 60-mm total length (Rose and Mesa 2012). It is not known what proportion of lamprey populations occur upstream of the intakes. Larger migrating juvenile lamprey (macrophthalmia, around 120-mm total length) would not be at risk of entrainment because of their size. Impingement risk for lamprey macrophthalmia would be low given the intakes' fish screens are designed to be protective of Chinook salmon fry and have approach velocity of 0.33 ft/s (Moser et al. 2015). Given the tendency for elevated river flows/precipitation events to coincide with Pacific lamprey macrophthalmia migrating in high numbers (Goodman et al. 2015) or ammocoetes being flushed from burrows (Rose and Mesa 2012), potential near-field effects from diversions would be limited under Alternatives 1, 2, and 3 by inclusion of pulse flow protection measures to be applied to precipitation-generated pulse flow events from October through May (as discussed in Chapter 2). As described in Appendix 11A, river lamprey downstream migration may occur in spring/summer, during which time there would be little difference between the NAA and Alternatives 1, 2, and 3 in terms of diversions. As discussed for winter-run Chinook salmon, there may be some small level of risk to juvenile lamprey from stranding behind the fish screens during high-flow events, but such events would be rare and would not differ between Alternatives 1, 2, and 3 and the NAA in their frequency. There is a lack of information regarding potential near-field effects related to predation of lamprey in the vicinity of the screens, although given that the main period of migration/movement is associated with high-flow events or during periods when diversions would not greatly differ between the NAA and Alternatives 1, 2, and 3, predation risk may be limited and will not differ between the NAA and Alternatives 1, 2, and 3. With respect to attraction of adult lamprey to reservoir releases to the Sacramento River and risk of stranding under Alternative 2, this risk would be minimized by the combination of the 60-foot-long apron and weir, as described further for winter-run Chinook salmon.

Far-Field Effects

Temperature Effects

Visual observation of exceedance plots and differences in modeled mean monthly temperatures by water year type between alternatives and the NAA in the Sacramento River below Keswick Dam and below RBDD[35] indicates that water temperatures would be similar among alternatives during the period of presence of each life stage of Pacific and river lamprey (Appendix 6C, Tables 6C-5-1a to 6C-5-4c, Tables 6C-10-1a to 6C-10-4c; Figures 6C-5-1 to 6C-5-18, Figures 6C-10-1 to 6C-10-18). At both locations, mean monthly water temperatures for all months within all water year types under Alternatives 1A and 1B were within 0.6°F of the NAA. Water temperature modeling results for Alternatives 2 and 3 were similar to those of Alternative 1.

Results of the analysis of exceedance above water temperature index values or occurrence outside index ranges for lamprey in the Sacramento River from Appendix 11B, Table 11B-2 are presented in Appendix 11D, Table 11D-239 through Table 11D-246. At both locations for both lamprey species and both life stages in each month and water year type, the difference in percent of days outside each index range or exceeding each index value between the NAA and Alternative 1A and between the NAA and Alternative 1B would be <5%. Results of the analysis of exceedance for Alternative 2 would be nearly identical to Alternative 1A, with one exception. There would be one month and water year type combination (May of Critically Dry Water Years) in which water temperature would be outside the 50°F to 64°F Pacific lamprey spawning and egg incubation index range under Alternative 2 on 5.4% fewer days than under the NAA.

Results for Alternative 3 would be similar to Alternative 1A with some additional differences in the percent of days outside the 50°F to 64°F index range for both Pacific and river lamprey spawning and egg incubation below Keswick Dam. The percent of days outside the index range under Alternative 3 in multiple water year types during April, June, and July would be 6.3% to 11.4% lower than the percent of days under the NAA. In addition, the percent of days outside the index range under Alternative 3 in May of Critically Dry Water Years would be 5.1% higher than under the NAA.

Flow Effects

Spawning Habitat

Flow-related impacts on spawning habitat of Pacific lamprey and river lamprey were evaluated by estimating effects of flow alterations on redd dewatering risk. Pacific lamprey eggs take about

[35] The RBDD, which was decommissioned in 2013, and the RBPP are co-located, and the names may be used interchangeably when referring to geographic locations.

19 days to incubate (Moyle et al. 2015), during which they must remain covered by water. Incubation times for river lamprey are unknown but assumed to be similar. Rapid reductions in flow can dewater redds leading to mortality. Redds are often found in low gradient stream reaches, in gravel, and at the tailouts of pools and riffles (Goodman and Reid 2012). These areas are vulnerable to dewatering when flows drop suddenly.

A dewatering risk analysis was conducted for redd dewatering in the Sacramento River at Keswick and Red Bluff. Pacific lamprey spawn between January and August and river lamprey spawn between February and June (Moyle 2002; U.S. Fish and Wildlife Service 2012; Goodman and Reid 2022), so flow reductions during those months have the potential to dewater redds, which could result in egg and embryo mortality. Dewatering risk to redds was estimated using CALSIM II outputs, as the number of spawning-period months followed in the next month by a reduction in flow of greater than 50%. The dewatering risk is expressed as number of months with such flow reductions summed over the 82 years of the CALSIM II record. Small-scale spawning location suitability characteristics (e.g., depth, velocity, substrate) of Pacific and river lampreys are not adequately understood to employ a more formal analysis such as a WUA analysis. A month-over-month flow reduction of at least 50% was chosen as a best professional estimate of flow conditions under which redd dewatering would occur, but it is not an empirically derived estimate of redd dewatering events. As such, there is uncertainty that these values represent actual redd dewatering events, and results should be treated as estimates of expected month to month flow fluctuations during egg incubation periods. The redd dewatering results are expressed as the percentage of months with elevated dewatering risk, and percent differences, for Alternatives 1, 2, and 3 and the NAA.

The results for both lamprey species show increased frequency of month-over-month flow reductions >50% for Alternatives 1, 2, and 3 at the Keswick location and reduced frequency at the Red Bluff location (Table 11-90 and Table 11-91). The differences at the two locations likely balance out effects of Alternatives 1, 2, and 3, resulting in no adverse effect on redd dewatering risk of Sacramento River Pacific lamprey or river lamprey.

Table 11-90. Percentage of Pacific Lamprey Spawning Months with >50% Flow Reduction in the Next Month, Used as a Proxy for Redd Dewatering Risk for Locations in the Sacramento River, and Percent Differences (in parentheses) between the NAA and Alt 1A, Alt 1B, Alt 2, and Alt 3.

Location	NAA	Alt 1A vs. NAA	Alt 1B vs. NAA	Alt 2 vs. NAA	Alt 3 vs. NAA
Sacramento River at Keswick	9.5	9.8 (3.2%)	10.1 (6.5%)	9.9 (4.8%)	10.1 (6.5%)
Sacramento River at Red Bluff	9.6	8.7 (-9.5%)	8.5 (-11.1%)	8.7 (-9.5%)	9.1 (-4.8%)

Note: Positive value for percent differences (in parentheses) indicates a higher risk of redd dewatering under Alternative 1–3 vs. the NAA and negative values indicate a lower risk.

Table 11-91. Percentage of River Lamprey Spawning Months with >50% Flow Reduction in the Next Month, Used as a Proxy for Redd Dewatering Risk for Locations in the Sacramento River, and Percent Differences (in parentheses) between the NAA and Alt 1A, Alt 1B, Alt 2, and Alt 3.

Location	NAA	Alt 1A vs. NAA	Alt 1B vs. NAA	Alt 2 vs. NAA	Alt 3 vs. NAA
Sacramento River at Keswick	9.7	10.2 (5%)	10.7 (10%)	10.2 (5%)	10.4 (7.5%)
Sacramento River at Red Bluff	11.7	10.4 (-10.4%)	10.7 (-8.3%)	10.4 (-10.4%)	11.2 (-4.2%)

Note: Positive value for percent differences (in parentheses) indicates a higher risk of redd dewatering under Alternative 1–3 vs. the NAA and negative values indicate a lower risk.

Rearing Habitat

Flow-related impacts on Pacific and river lamprey rearing habitat were evaluated by estimating dewatering of ammocoete rearing habitat resulting from changes in river stage as estimated from changes in flow for Alternatives 1, 2, and 3. An ammocoete is the filter-feeding larval stage of the lamprey. It remains relatively immobile in the sediment at the same location for several years (Moyle et al. 2015), after which it migrates downstream. During the rearing period there is potential for dewatering of ammocoete rearing habitat, also referred to as ammocoete stranding, from rapid reductions in flow, leading to mortality (Goodman and Reid 2012). Suitable habitat for ammocoetes is often at stream margins in areas of low velocity with fine substrate, which are the first areas dewatered when stream flows drop. Ammocoetes do not segregate themselves by age so a single event can affect multiple year classes, significantly impacting a local lamprey population (Goodman and Reid 2012).

Rearing habitat dewatering risks were analyzed for ammocoetes under Alternatives 1, 2, and 3 and the NAA in the Sacramento River at Keswick, Bend Bridge and Hamilton City. Data from WUA studies in the Tualatin River Basin in Oregon indicate that Pacific lamprey ammocoete rearing habitat is primarily located from near surface to about 8 feet deep (David Evans and Associates, Inc. et al. 2007). For this analysis, the ammocoetes are assumed to have a uniform depth distribution over this range. River lamprey ammocoetes are assumed to have a similar depth distribution. Using flow vs. river stage tables for established gages at the three river locations listed above (California Department of Water Resources 2021), changes in river stage were determined from the CALSIM II monthly flow records for Alternatives 1, 2, and 3 and the NAA. A cohort of ammocoetes was assumed to begin every month during the spawning period (January through August for Pacific lamprey and February through June for river lamprey) and spend 7 years rearing upstream. Eggs hatch in about 19 days and ammocoetes quickly disperse to rearing habitat (Moyle et al. 2015), so initiation of an ammocoete cohort was assumed to occur each month of spawning over the 82-year CALSIM II record. At each river location, the stage of the river, estimated from the CALSIM II flow and the stage vs. flow table for that river location, was tracked for each cohort from the month of spawning through seven years of ammocoete rearing. The greatest reduction in stage from the month of spawning during the following 7-year period was used to determine, from the depth distribution of ammocoete habitats, the percentage of the ammocoete cohort stranded. For instance, a stage reduction of 4 feet was estimated to result in the stranding of 50% of the cohort and a stage reduction of 8 feet was estimated to result in a 100% stranding. This procedure assumes that the ammocoetes do not change location during their rearing period, which is not necessarily the case (Goodman and Reid 2012), but any error associated with this assumption is likely to be roughly equal for the NAA and Alternatives 1, 2, and 3.

The results of the stranding analysis are expressed as the mean percentage of ammocoetes stranded in each month and water year type under Alternatives 1, 2, and 3 and the NAA, and the absolute differences in the percentages between Alternatives 1, 2, and 3 and the NAA. Each table gives results for both lamprey species, with specific results for a species corresponding to the spawning months for that species (referred to as "Initial Month of Cohort" in the tables): January through August for Pacific lamprey and February through June for river lamprey.

The results show a range of about 2% to 94% in ammocoete stranding for the Sacramento River at Keswick Dam, 2% to 69% for the river at Bend Bridge, and 1% to 62% for the river at Hamilton City (Table 11-92 through Table 11-94). There was little difference among alternatives in the maximum stranding at any of the three locations. Difference in mean ammocoete stranding between Alternatives 1, 2, and 3 and the NAA are minor. For the Keswick Dam and Bend Bridge locations, almost all of the largest differences (>5) in the percentages occur under Alternative 3 (Table 11-92 and Table 11-93). The largest reductions in stranding (up to 8%) occur in May through August and the largest increases (up to 8%) occur in October and November. At Hamilton City, however, the largest differences in stranding, all of which constitute reductions, occur under all three alternatives in January, February, and March of Above Normal Water Years (Table 11-94). All months with relatively large reductions in stranding are within the cohort initiation periods of both lamprey species, except for January, which is within the Pacific lamprey period only, and none of the months with relatively large increases in stranding are in the cohort initiation period of either species (Table 11-92 and Table 11-93). These results indicate that Alternatives 1, 2, and 3 would have little effect on dewatering risk of the ammocoete rearing habitat of Pacific lamprey or river lamprey in the Sacramento River.

Table 11-92. Percent of Pacific and River Lamprey Ammocoetes Stranded During 7-Year Rearing Period in the Sacramento River at Keswick Dam and Differences in the Percentages for the NAA and Alt 1A, Alt 1B, Alt 2, and Alt 3. January through August Results Pertain to Pacific Lamprey and the February through June Results Pertain to River Lamprey.

Initial Month of Cohort	Water Year Type[1]	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
January	Wet	69.0	68.9 (-0.2)	68.9 (-0.1)	68.9 (-0.2)	69.5 (0.4)
	Above Normal	34.4	34.5 (0.1)	34.4 (0)	34.5 (0.1)	36.4 (2)
	Below Normal	13.2	12.8 (-0.4)	12.9 (-0.3)	12.8 (-0.4)	13 (-0.2)
	Dry	3.9	2.7 (-1.2)	2.7 (-1.2)	2.7 (-1.2)	1.8 (-2)
	Critically Dry	5.1	3.8 (-1.3)	3.9 (-1.2)	3.9 (-1.2)	4.3 (-0.8)
	All	29.9	29.3 (-0.6)	29.3 (-0.5)	29.3 (-0.6)	29.7 (-0.2)
February	Wet	70.1	69.9 (-0.3)	70.8 (0.7)	69.9 (-0.3)	70.7 (0.5)
	Above Normal	64.5	65.9 (1.4)	66.6 (2.1)	65.3 (0.7)	67.2 (2.7)
	Below Normal	35.9	36.1 (0.2)	37.6 (1.7)	35.7 (-0.3)	38.3 (2.4)

Initial Month of Cohort	Water Year Type[1]	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
	Dry	8.2	7.2 (-1)	7 (-1.2)	7.2 (-1)	6.8 (-1.4)
	Critically Dry	6.3	6.6 (0.3)	6.8 (0.5)	6.7 (0.4)	6.5 (0.2)
	All	39.6	39.5 (0)	40.2 (0.6)	39.4 (-0.2)	40.3 (0.7)
March	Wet	67.3	68.2 (0.9)	67.9 (0.6)	68.2 (0.9)	67 (-0.3)
	Above Normal	61.3	61.5 (0.2)	61.5 (0.2)	61.5 (0.2)	60.9 (-0.4)
	Below Normal	15.1	14.8 (-0.3)	14.8 (-0.3)	14.8 (-0.3)	14.4 (-0.7)
	Dry	14.4	15.2 (0.8)	13.9 (-0.4)	15.2 (0.8)	16.3 (1.9)
	Critically Dry	8.8	7.9 (-0.9)	9.9 (1.1)	8.3 (-0.5)	10.2 (1.4)
	All	36.1	36.4 (0.3)	36.4 (0.2)	36.5 (0.4)	36.5 (0.4)
April	Wet	47.7	47.7 (0)	47.7 (0)	47.7 (0)	46.8 (-0.9)
	Above Normal	21.3	21.3 (0)	21.4 (0.1)	21.3 (0)	19.7 (-1.6)
	Below Normal	13.6	13.6 (0)	13.4 (-0.2)	13.6 (0)	12.3 (-1.3)
	Dry	16.1	15.9 (-0.1)	13.9 (-2.2)	15.9 (-0.1)	13.4 (-2.7)
	Critically Dry	15.7	12 (-3.7)	12.5 (-3.2)	14.1 (-1.7)	14.3 (-1.5)
	All	26.0	25.3 (-0.6)	25 (-1)	25.7 (-0.3)	24.4 (-1.5)
May	Wet	63.9	63.9 (-0.1)	65.9 (2)	63.9 (-0.1)	64.6 (0.6)
	Above Normal	56.1	55.9 (-0.2)	56.7 (0.6)	55.9 (-0.2)	58.3 (2.2)
	Below Normal	47.9	47.9 (0)	44.8 (-3)	47.9 (0)	44 (-3.9)
	Dry	50.8	51 (0.2)	49 (-1.8)	51 (0.2)	45.4 (-5.4)
	Critically Dry	51.7	49 (-2.6)	48.7 (-2.9)	48.3 (-3.3)	46.4 (-5.2)
	All	55.1	54.7 (-0.4)	54.4 (-0.7)	54.6 (-0.5)	52.9 (-2.2)
June	Wet	58.5	58.6 (0.1)	58.3 (-0.2)	58.6 (0.1)	57.9 (-0.6)
	Above Normal	70.9	70.8 (-0.1)	64.8 (-6.1)	70.8 (-0.1)	62.9 (-8)
	Below Normal	71.9	72.1 (0.2)	70.5 (-1.5)	72.1 (0.2)	63.6 (-8.4)
	Dry	77.4	77 (-0.4)	76.9 (-0.4)	77 (-0.4)	72.7 (-4.6)
	Critically Dry	66.5	62.6 (-3.9)	61.8 (-4.7)	62.7 (-3.8)	61.2 (-5.3)
	All	68.0	67.3 (-0.7)	66 (-2)	67.3 (-0.7)	63.3 (-4.6)
July	Wet	86.5	86.6 (0.1)	86.4 (0)	86.6 (0.1)	86.2 (-0.3)
	Above Normal	93.8	93.5 (-0.2)	91.5 (-2.2)	93.5 (-0.2)	87.7 (-6.1)
	Below Normal	90.7	90.6 (-0.1)	90.8 (0.1)	90.8 (0.1)	89.6 (-1.2)
	Dry	88.6	88.2 (-0.4)	87.8 (-0.8)	88.2 (-0.4)	85.6 (-3)
	Critically Dry	70.3	69.9 (-0.4)	69.3 (-1)	69.8 (-0.5)	68.3 (-2.1)
	All	86.1	85.9 (-0.2)	85.5 (-0.6)	85.9 (-0.2)	84 (-2.1)
August	Wet	79.7	79.8 (0.1)	79.2 (-0.5)	79.8 (0.1)	78.4 (-1.3)
	Above Normal	74.9	73.7 (-1.3)	74.2 (-0.8)	73.4 (-1.5)	68 (-6.9)
	Below Normal	67.9	66.1 (-1.8)	67.2 (-0.7)	66.3 (-1.6)	63.9 (-4)
	Dry	64.9	64.2 (-0.7)	64.2 (-0.8)	64.2 (-0.7)	62.2 (-2.7)
	Critically Dry	52.9	56.7 (3.7)	56.3 (3.3)	56.7 (3.7)	56.3 (3.4)
	All	69.4	69.4 (0)	69.4 (0)	69.4 (0)	67.3 (-2.1)

Initial Month of Cohort	Water Year Type[1]	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
September	Wet	72.0	72 (0)	71.2 (-0.8)	72.1 (0.1)	70.8 (-1.2)
	Above Normal	59.5	57.2 (-2.2)	59.2 (-0.3)	56.8 (-2.6)	60.1 (0.6)
	Below Normal	30.3	29.4 (-0.9)	30.4 (0.1)	29.3 (-1)	28.7 (-1.5)
	Dry	22.3	21.5 (-0.8)	21.8 (-0.5)	21.5 (-0.8)	21 (-1.3)
	Critically Dry	18.9	21.9 (3)	21.9 (3)	21.5 (2.6)	20.6 (1.8)
	All	43.4	43.3 (-0.1)	43.6 (0.1)	43.2 (-0.3)	42.9 (-0.6)
October	Wet	48.0	47.6 (-0.4)	48 (0)	47.6 (-0.4)	46.8 (-1.2)
	Above Normal	30.5	32.5 (1.9)	33.8 (3.3)	32.6 (2)	37.7 (7.1)
	Below Normal	29.0	28.9 (-0.2)	29.5 (0.4)	29 (-0.1)	35.2 (6.2)
	Dry	19.6	18.1 (-1.5)	18.2 (-1.3)	18.2 (-1.4)	20.1 (0.5)
	Critically Dry	28.4	29.4 (1)	29.8 (1.5)	29.8 (1.4)	30.7 (2.3)
	All	32.9	32.9 (0)	33.4 (0.5)	33 (0.1)	35.2 (2.3)
November	Wet	41.9	41.9 (-0.1)	41.6 (-0.3)	41.9 (-0.1)	40.7 (-1.2)
	Above Normal	37.8	37.9 (0.1)	37.7 (-0.1)	37.7 (-0.1)	39.6 (1.8)
	Below Normal	28.5	30.3 (1.8)	31.5 (3)	30.3 (1.8)	36.9 (8.4)
	Dry	20.9	20.4 (-0.5)	22.3 (1.4)	20.3 (-0.6)	23.4 (2.5)
	Critically Dry	19.9	18.6 (-1.3)	18.6 (-1.3)	18.4 (-1.5)	18.4 (-1.5)
	All	30.8	30.8 (0)	31.3 (0.5)	30.7 (-0.1)	32.5 (1.7)
December	Wet	38.6	38.3 (-0.3)	38.5 (-0.2)	38.3 (-0.3)	38.5 (-0.2)
	Above Normal	35.7	36.9 (1.1)	38.3 (2.6)	38 (2.3)	38 (2.2)
	Below Normal	42.8	43.5 (0.7)	44.9 (2.1)	43.5 (0.7)	46 (3.2)
	Dry	32.6	32.4 (-0.2)	30.8 (-1.8)	32.3 (-0.3)	32.6 (0)
	Critically Dry	4.4	2.9 (-1.5)	3 (-1.4)	2.9 (-1.5)	4.4 (0)
	All	32.2	32.1 (-0.1)	32.3 (0.1)	32.3 (0.1)	33.1 (0.9)

[1] Water year type sorting is by hydrologic water year.

Table 11-93. Percent of Pacific and River Lamprey Ammocoetes Stranded During 7-Year Rearing Period in the Sacramento River at Bend Bridge and Differences in the Percentages for the NAA and Alt 1A, Alt 1B, Alt 2, and Alt 3. January through August Results Pertain to Pacific Lamprey and the February through June Results Pertain to River Lamprey.

Initial Month of Cohort	Water Year Type[1]	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
January	Wet	68.4	68.3 (-0.1)	68.4 (-0.1)	68.3 (-0.1)	69.2 (0.8)
	Above Normal	49.9	49.9 (0)	50.1 (0.2)	49.9 (0)	50.7 (0.8)
	Below Normal	27.5	27.1 (-0.4)	27 (-0.5)	27.1 (-0.4)	27.6 (0.1)
	Dry	7.2	6.8 (-0.4)	7 (-0.2)	6.8 (-0.4)	6.8 (-0.4)
	Critically Dry	7.6	7.7 (0.1)	7.7 (0.1)	7.7 (0.1)	7.7 (0.1)
	All	36.4	36.2 (-0.2)	36.3 (-0.1)	36.2 (-0.2)	36.7 (0.3)
February	Wet	67.2	66.9 (-0.3)	67.3 (0.1)	66.9 (-0.3)	67.7 (0.5)

Initial Month of Cohort	Water Year Type[1]	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
	Above Normal	63.1	63.1 (0)	63.2 (0.1)	63.1 (0)	63.4 (0.3)
	Below Normal	47.7	47.8 (0.1)	48.8 (1.1)	47.5 (-0.3)	50.2 (2.4)
	Dry	22.8	22.5 (-0.3)	22.8 (-0.1)	22.5 (-0.3)	23 (0.1)
	Critically Dry	10.4	10.9 (0.5)	11 (0.6)	11 (0.6)	10.5 (0.1)
	All	45.2	45.1 (-0.1)	45.5 (0.3)	45.1 (-0.1)	45.9 (0.7)
March	Wet	62.8	63 (0.1)	62.8 (0)	63 (0.1)	63.1 (0.2)
	Above Normal	57.7	57.6 (-0.1)	57.6 (-0.1)	57.6 (-0.1)	57.9 (0.2)
	Below Normal	25.0	24.9 (-0.2)	24.9 (-0.2)	24.9 (-0.2)	25.3 (0.3)
	Dry	22.5	23.5 (1)	22.9 (0.4)	23.5 (1)	24.4 (1.9)
	Critically Dry	8.7	8.4 (-0.3)	9.5 (0.8)	8.7 (0)	9.7 (0.9)
	All	38.9	39 (0.2)	39 (0.2)	39.1 (0.2)	39.6 (0.7)
April	Wet	48.6	48.5 (-0.1)	48.6 (0)	48.5 (-0.1)	49.1 (0.5)
	Above Normal	24.2	24 (-0.2)	24.1 (-0.1)	24 (-0.2)	24 (-0.2)
	Below Normal	17.2	16.9 (-0.3)	16.7 (-0.5)	16.9 (-0.3)	17 (-0.2)
	Dry	9.7	10 (0.3)	8.9 (-0.8)	10 (0.3)	9.2 (-0.5)
	Critically Dry	5.1	3.1 (-2.1)	3.5 (-1.7)	4.3 (-0.8)	4.2 (-0.9)
	All	24.8	24.4 (-0.4)	24.3 (-0.5)	24.6 (-0.2)	24.6 (-0.2)
May	Wet	44.2	43.9 (-0.2)	45.4 (1.2)	43.9 (-0.2)	45.6 (1.4)
	Above Normal	29.8	29.3 (-0.4)	30.1 (0.3)	29.3 (-0.4)	31.8 (2.1)
	Below Normal	26.1	26 (-0.1)	23.9 (-2.2)	26 (-0.1)	23.9 (-2.2)
	Dry	23.7	24.2 (0.5)	23.1 (-0.6)	24.2 (0.5)	21 (-2.7)
	Critically Dry	24.6	22.9 (-1.7)	22.4 (-2.2)	22.2 (-2.4)	20.9 (-3.7)
	All	31.6	31.3 (-0.3)	31.2 (-0.4)	31.2 (-0.4)	30.9 (-0.8)
June	Wet	31.2	31.2 (0)	31.2 (0)	31.2 (0)	31.8 (0.5)
	Above Normal	34.3	34.1 (-0.2)	29.8 (-4.6)	34.1 (-0.2)	28.9 (-5.4)
	Below Normal	42.1	42.1 (0)	40.8 (-1.4)	42 (-0.1)	34.9 (-7.2)
	Dry	40.6	40.6 (0)	40.7 (0.1)	40.6 (0)	37.7 (-2.9)
	Critically Dry	34.1	31.2 (-2.9)	30.4 (-3.7)	31.1 (-3)	29.8 (-4.4)
	All	36.0	35.6 (-0.5)	34.6 (-1.4)	35.5 (-0.5)	32.9 (-3.1)
July	Wet	49.3	49.2 (-0.1)	49.1 (-0.2)	49.2 (-0.1)	49.7 (0.4)
	Above Normal	59.0	58.1 (-0.9)	57 (-2)	57.7 (-1.3)	50.7 (-8.4)
	Below Normal	57.3	55.9 (-1.4)	56.7 (-0.5)	56 (-1.2)	54.4 (-2.8)
	Dry	46.5	46.5 (-0.1)	46.6 (0.1)	46.5 (-0.1)	45.3 (-1.2)
	Critically Dry	34.9	34.9 (0)	34.3 (-0.6)	34.7 (-0.2)	33.3 (-1.6)
	All	49.3	48.8 (-0.4)	48.7 (-0.5)	48.8 (-0.5)	47.2 (-2)
August	Wet	39.8	39.6 (-0.1)	39.4 (-0.4)	39.6 (-0.1)	39.9 (0.1)
	Above Normal	36.3	34.8 (-1.6)	35.5 (-0.8)	34.6 (-1.7)	30.8 (-5.5)
	Below Normal	32.8	30.8 (-2)	32 (-0.7)	31.1 (-1.7)	30.1 (-2.7)
	Dry	26.6	26.4 (-0.1)	26.4 (-0.1)	26.4 (-0.1)	25.4 (-1.2)

Initial Month of Cohort	Water Year Type[1]	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
	Critically Dry	19.7	23.3 (3.6)	22.9 (3.2)	23.1 (3.4)	22.6 (2.9)
	All	32.2	32.1 (-0.1)	32.3 (0.1)	32.1 (-0.1)	31.2 (-1)
September	Wet	35.0	34.8 (-0.2)	34.3 (-0.7)	34.9 (-0.1)	35.1 (0.1)
	Above Normal	24.5	22.4 (-2.1)	24 (-0.5)	22.2 (-2.4)	25.3 (0.8)
	Below Normal	7.1	6.8 (-0.3)	7.3 (0.2)	6.8 (-0.3)	7.3 (0.2)
	Dry	2.8	3 (0.2)	3.5 (0.7)	3 (0.2)	3.1 (0.3)
	Critically Dry	1.5	3.2 (1.6)	3.1 (1.5)	3 (1.4)	2.6 (1)
	All	16.6	16.5 (-0.1)	16.7 (0.1)	16.5 (-0.2)	17 (0.4)
October	Wet	20.5	20.2 (-0.3)	20.5 (0)	20.2 (-0.3)	20.9 (0.3)
	Above Normal	9.6	10.5 (0.8)	11.1 (1.5)	10.6 (0.9)	14 (4.3)
	Below Normal	11.9	11.5 (-0.4)	12 (0.1)	11.5 (-0.4)	16.1 (4.2)
	Dry	3.9	3.6 (-0.3)	3.9 (0)	3.6 (-0.3)	4.8 (0.8)
	Critically Dry	7.4	8 (0.6)	8 (0.6)	8.2 (0.8)	8.1 (0.7)
	All	11.9	11.8 (0)	12.2 (0.3)	11.9 (0)	13.6 (1.7)
November	Wet	24.2	24 (-0.1)	24 (-0.2)	24 (-0.2)	24.5 (0.3)
	Above Normal	22.3	22.2 (-0.1)	22.2 (-0.1)	22.1 (-0.2)	23.1 (0.7)
	Below Normal	18.8	20 (1.2)	21.1 (2.2)	20 (1.2)	25.5 (6.7)
	Dry	12.0	12 (0)	12.9 (0.8)	12 (-0.1)	13.6 (1.6)
	Critically Dry	7.9	7 (-0.9)	7.2 (-0.7)	6.8 (-1.1)	6.9 (-1)
	All	18.0	18 (0)	18.3 (0.4)	17.9 (-0.1)	19.5 (1.5)
December	Wet	30.5	30.3 (-0.2)	30.4 (-0.1)	30.3 (-0.2)	30.8 (0.2)
	Above Normal	37.9	38.4 (0.5)	39.1 (1.2)	38.7 (0.8)	38.5 (0.6)
	Below Normal	42.3	42.7 (0.4)	43.6 (1.3)	42.7 (0.4)	45.1 (2.8)
	Dry	29.2	29.5 (0.3)	28.8 (-0.3)	29.4 (0.3)	29.6 (0.4)
	Critically Dry	9.7	8.9 (-0.8)	8.9 (-0.8)	8.9 (-0.8)	9.7 (0)
	All	30.3	30.3 (0)	30.4 (0.2)	30.3 (0)	31 (0.7)

[1] Water year type sorting is by hydrologic water year.

Table 11-94. Percent of Pacific and River Lamprey Ammocoetes Stranded During 7-Year Rearing Period in the Sacramento River at Hamilton City and Differences in the Percentages for the NAA and Alt 1A, Alt 1B, Alt 2, and Alt 3. January through August Results Pertain to Pacific Lamprey and the February through June Results Pertain to River Lamprey.

Initial Month of Cohort	Water Year Type[1]	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
January	Wet	70.7	68.2 (-2.5)	67.2 (-3.5)	68.3 (-2.5)	67.5 (-3.2)
	Above Normal	52.3	45.9 (-6.5)	45.4 (-6.9)	45.9 (-6.5)	46.1 (-6.2)
	Below Normal	27.9	24.9 (-3.1)	24.2 (-3.7)	24.7 (-3.3)	24.2 (-3.7)
	Dry	14.0	13.4 (-0.5)	13.3 (-0.7)	13.4 (-0.5)	13.3 (-0.7)
	Critically Dry	13.5	11.5 (-2)	11.8 (-1.7)	11.5 (-2)	12.1 (-1.3)

Initial Month of Cohort	Water Year Type[1]	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
	All	39.0	36.4 (-2.6)	35.9 (-3.1)	36.4 (-2.7)	36.2 (-2.9)
February	Wet	71.5	68.8 (-2.7)	68.7 (-2.8)	68.9 (-2.5)	68.6 (-2.9)
	Above Normal	70.0	65.2 (-4.9)	65 (-5)	64.9 (-5.1)	65.3 (-4.8)
	Below Normal	41.6	38.2 (-3.5)	37.7 (-3.9)	38 (-3.6)	38.5 (-3.2)
	Dry	27.1	24.2 (-2.9)	24 (-3.1)	24.2 (-2.9)	24 (-3.1)
	Critically Dry	16.4	15.3 (-1)	15.8 (-0.6)	15.3 (-1)	15.6 (-0.7)
	All	47.4	44.5 (-2.9)	44.4 (-3)	44.5 (-2.9)	44.5 (-2.9)
March	Wet	63.4	62.2 (-1.2)	61.4 (-2)	62.7 (-0.7)	61 (-2.4)
	Above Normal	61.6	55 (-6.7)	54.5 (-7.2)	55 (-6.7)	55.1 (-6.5)
	Below Normal	26.4	23.4 (-2.9)	22.9 (-3.5)	23.4 (-2.9)	22.4 (-4)
	Dry	27.3	24.5 (-2.8)	24 (-3.3)	24.5 (-2.8)	24.4 (-2.8)
	Critically Dry	16.1	14.4 (-1.7)	15.5 (-0.5)	14.5 (-1.6)	15.6 (-0.4)
	All	41.0	38.3 (-2.7)	38 (-3)	38.5 (-2.5)	38 (-3)
April	Wet	47.1	44.6 (-2.5)	44.1 (-3)	44.6 (-2.5)	44.3 (-2.8)
	Above Normal	27.8	25.2 (-2.6)	24.9 (-2.9)	25.2 (-2.6)	24.9 (-2.8)
	Below Normal	19.5	17.9 (-1.6)	17.3 (-2.2)	17.9 (-1.6)	17.3 (-2.2)
	Dry	15.5	15.2 (-0.3)	14.4 (-1.2)	15.2 (-0.3)	14.5 (-1)
	Critically Dry	8.7	6.6 (-2.1)	7.2 (-1.5)	7.7 (-1)	8.3 (-0.4)
	All	26.5	24.7 (-1.8)	24.3 (-2.2)	24.9 (-1.6)	24.6 (-1.9)
May	Wet	31.2	29.6 (-1.6)	30.3 (-0.9)	29.6 (-1.6)	30.7 (-0.4)
	Above Normal	22.3	21.1 (-1.2)	21.2 (-1.1)	21.1 (-1.2)	22.7 (0.4)
	Below Normal	14.8	14 (-0.7)	13.6 (-1.2)	14 (-0.7)	13.9 (-0.9)
	Dry	15.1	14.7 (-0.3)	15.2 (0.1)	14.7 (-0.3)	14.5 (-0.5)
	Critically Dry	15.7	14.1 (-1.6)	14.5 (-1.1)	13.8 (-1.9)	14.1 (-1.6)
	All	21.0	19.9 (-1.1)	20.2 (-0.8)	19.9 (-1.1)	20.4 (-0.6)
June	Wet	17.1	16.5 (-0.6)	15.9 (-1.1)	16.5 (-0.6)	16.5 (-0.6)
	Above Normal	19.7	18.3 (-1.4)	17.3 (-2.4)	18.3 (-1.4)	17.7 (-2)
	Below Normal	18.8	18.1 (-0.7)	17.5 (-1.3)	18.1 (-0.7)	17 (-1.9)
	Dry	21.4	21 (-0.4)	21.2 (-0.2)	20.9 (-0.5)	21.1 (-0.3)
	Critically Dry	18.9	17.9 (-1)	18.1 (-0.7)	18 (-0.9)	17.7 (-1.1)
	All	19.0	18.2 (-0.7)	17.9 (-1.1)	18.2 (-0.7)	17.9 (-1)
July	Wet	24.4	23.7 (-0.7)	23.2 (-1.3)	23.7 (-0.7)	23.9 (-0.5)
	Above Normal	29.8	28.4 (-1.4)	28.6 (-1.1)	28.2 (-1.5)	30.4 (0.6)
	Below Normal	25.2	23.9 (-1.3)	24 (-1.2)	23.9 (-1.3)	24.7 (-0.5)
	Dry	24.8	25.9 (1.1)	25.5 (0.7)	25.9 (1)	25.5 (0.6)
	Critically Dry	19.3	20.7 (1.4)	20.9 (1.6)	20.5 (1.2)	20.4 (1.2)
	All	24.5	24.4 (-0.2)	24.2 (-0.4)	24.3 (-0.3)	24.7 (0.1)
August	Wet	20.1	19.4 (-0.7)	18.8 (-1.2)	19.4 (-0.7)	19.5 (-0.6)
	Above Normal	18.5	16.7 (-1.8)	16.8 (-1.7)	16.7 (-1.8)	17.7 (-0.9)

Initial Month of Cohort	Water Year Type[1]	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
	Below Normal	14.3	13.1 (-1.2)	13 (-1.3)	13.3 (-1)	13.1 (-1.2)
	Dry	14.9	16.4 (1.5)	15.7 (0.8)	16.4 (1.5)	15 (0.1)
	Critically Dry	11.8	15.7 (3.9)	15.6 (3.8)	15.2 (3.4)	14.5 (2.7)
	All	16.4	16.7 (0.3)	16.3 (-0.1)	16.6 (0.2)	16.3 (-0.1)
September	Wet	26.2	25.4 (-0.8)	24.3 (-1.8)	25.4 (-0.8)	25.2 (-1)
	Above Normal	20.4	18.4 (-2)	18.9 (-1.5)	18.4 (-2)	20.8 (0.3)
	Below Normal	5.2	3.6 (-1.6)	3.4 (-1.8)	3.6 (-1.6)	3.5 (-1.7)
	Dry	2.0	2.4 (0.4)	2.6 (0.6)	2.4 (0.4)	2.2 (0.2)
	Critically Dry	1.2	3.4 (2.1)	3.5 (2.2)	3.4 (2.1)	2.8 (1.5)
	All	12.3	11.9 (-0.4)	11.7 (-0.6)	11.9 (-0.4)	12.1 (-0.3)
October	Wet	18.1	17.2 (-0.9)	16.9 (-1.2)	17.2 (-0.9)	17.4 (-0.7)
	Above Normal	11.8	11.2 (-0.6)	11.5 (-0.3)	11.3 (-0.5)	13.1 (1.3)
	Below Normal	10.8	8.7 (-2)	8.7 (-2.1)	8.7 (-2)	12 (1.3)
	Dry	3.8	4.1 (0.2)	3.8 (-0.1)	4 (0.2)	4.8 (0.9)
	Critically Dry	8.3	8.4 (0.1)	8.6 (0.3)	8.4 (0.1)	9.6 (1.3)
	All	11.3	10.6 (-0.7)	10.5 (-0.8)	10.6 (-0.7)	11.9 (0.6)
November	Wet	23.2	21.6 (-1.6)	20.9 (-2.3)	21.6 (-1.6)	21.5 (-1.7)
	Above Normal	24.2	23.3 (-0.9)	22.8 (-1.4)	23.3 (-0.9)	23.9 (-0.2)
	Below Normal	16.5	15.9 (-0.6)	16.2 (-0.3)	15.9 (-0.6)	18.4 (1.9)
	Dry	13.0	13 (0)	13.3 (0.3)	12.7 (-0.3)	14 (1.1)
	Critically Dry	9.5	8.6 (-0.9)	8.8 (-0.7)	8.6 (-0.9)	8.8 (-0.7)
	All	17.7	16.9 (-0.8)	16.7 (-1)	16.8 (-0.9)	17.6 (-0.1)
December	Wet	33.7	31.9 (-1.9)	31.2 (-2.5)	31.9 (-1.9)	32 (-1.7)
	Above Normal	38.2	34.7 (-3.5)	34.5 (-3.7)	35 (-3.3)	33.7 (-4.5)
	Below Normal	37.9	34.4 (-3.5)	34.4 (-3.5)	34.3 (-3.5)	35.2 (-2.6)
	Dry	33.4	32.7 (-0.7)	32.3 (-1)	32.7 (-0.7)	32.4 (-0.9)
	Critically Dry	14.6	14.1 (-0.5)	14.3 (-0.3)	14.1 (-0.5)	15.1 (0.5)
	All	32.0	30.1 (-1.9)	29.8 (-2.2)	30.1 (-1.9)	30.3 (-1.8)

[1] Water year type sorting is by hydrologic water year.

Feather River

Temperature Effects

Visual observation of exceedance plots and differences in modeled mean monthly temperatures by water year type between alternatives and the NAA in the Feather River LFC below the Fish Barrier Dam and in the HFC at Gridley Bridge and the mouth indicates that water temperatures would be predominantly similar among alternatives during the period of presence of both life stages of Pacific and river lamprey (Appendix 6C, Tables 6C-16-1a to 6C-16-4c, Tables 6C-18-1a to 6C-18-4c, Tables 6C-19-1a to 6C-19-4c; Figures 6C-16-1 to 6C-16-18, Figures 6C-18-1 to 6C-18-18, Figures 6C-19-1 to 6C-19-18). At all locations, mean monthly water temperatures for all months within all water year types under Alternatives 1A and 1B were within 0.5°F of the NAA. Water temperature modeling results for Alternatives 2 and 3 were similar to those of Alternative 1 at all locations.

Results of the analysis of exceedance above water temperature index values or occurrence outside index ranges for Pacific and river lampreys in the Feather River from Appendix 11B, Table 11B-3 are presented in Appendix 11D, Table 11D-247 through Table 11D-258. In the LFC below the fish dam, there would generally be no differences in exceedance of water temperature indices for any life stage of both lampreys between Alternatives 1A and 1B compared to the NAA in any month or water year type with one exception. The percent of months outside the 50°F to 64°F river lamprey spawning and egg incubation temperature index range under Alternative 1A would be 9.1% greater than under the NAA. In the HFC, for each life stage in each month and water year type for both lampreys, the difference in percent of months exceeding each index value between the NAA and Alternatives 1A and 1B would be <5%, with some isolated exceptions. Below Thermalito Afterbay in August of Dry Water Years, there would be a 5.6% increase in water temperatures outside the spawning and egg incubation index range of 50°F to 64°F for both lampreys under Alternatives 1B compared to the NAA. At the Feather River mouth in July of Below Normal Water Years, there would be a 7.1% increase in exceedance above the 72°F Pacific and river ammocoete rearing and emigration index value for both lampreys under both Alternatives 1A and 1B compared to the NAA. Also, at the Feather River mouth in August of Critically Dry Water Years, there would be 8.3% reductions in exceedance above the juvenile rearing and emigration index value of 72°F under both Alternative 1A and 1B compared to the NAA. Results of the water temperature exceedance analysis for Pacific and river lamprey for Alternative 2 are nearly identical to those for Alternative 1A and 1B at all locations analyzed in the Feather River.

Results of the water temperature exceedance analysis for Pacific and river lamprey for Alternative 3 are similar to those for Alternative 1A with some exceptions. In the LFC below the fish dam there would be no differences in exceedance of water temperature indices for any life stage of both lampreys between Alternative 3 and the NAA in any month or water year type. Similar to Alternative 1B, below Thermalito Afterbay in August of Dry Water Years, there would be a 5.6% increase in water temperatures outside the spawning and egg incubation index range of 50°F to 64°F for both lampreys under Alternative 3 compared to the NAA. At the Feather River mouth during July of Above Normal Water Years, there would be 9.1% more months in which water temperature exceeds the 72°F ammocoete rearing and emigration index value for both Pacific and river lamprey. Also, during August of Dry and Critically Dry Water Years, there would be 5.6% and 8.3% fewer months, respectively, under Alternative 3 relative to the NAA in which water temperature exceeds the 72°F ammocoete rearing and emigration index value for both Pacific and river lamprey.

Flow Effects

Spawning Habitat

Spawning habitat for Pacific and river lamprey was evaluated for the Feather River by analyzing redd dewatering risk as described above for the Sacramento River. CALSIM II flow outputs for the Feather River at the Thermalito Afterbay outlet were used to compare the redd dewatering risks between Alternatives 1, 2, and 3 and the NAA. The results for both lamprey species showed only small differences between Alternatives 1, 2, and 3 and the NAA and it was concluded that Alternatives 1, 2, and 3 would not substantially affect redd dewatering risk in the Feather River for either lamprey species (Table 11-95).

Table 11-95. Percentage of Pacific and River Lamprey Spawning Months with >50% Flow Reduction in the Next Month, Used as a Proxy for Redd Dewatering Risk in the Feather River at Thermalito Afterbay, and Percent Differences (in parentheses) between the NAA and Alt 1A, Alt 1B, Alt 2, and Alt 3.

Location	NAA	Alt 1A vs. NAA	Alt 1B vs. NAA	Alt 2 vs. NAA	Alt 3 vs. NAA
Pacific Lamprey	16.8	17.1 (1.8%)	17.2 (2.7%)	17.1 (1.8%)	17.1 (1.8%)
River Lamprey	14.1	14.3 (1.7%)	14.3 (1.7%)	14.3 (1.7%)	14.6 (3.4%)

Note: Positive value for percent differences (in parentheses) indicates a higher risk of redd dewatering under Alternatives 1, 2, and 3 vs. the NAA and negative values indicate a lower risk.

Rearing Habitat

Rearing habitat for Pacific and river lamprey was evaluated for the Feather River by analyzing ammocoete habitat dewatering risk as described above for the Sacramento River. CALSIM II flow outputs for the Feather River at the Thermalito Afterbay outlet were used to compare the dewatering risks between Alternatives 1, 2, and 3 and the NAA. No river stage versus flow table was found for the Feather River at the Thermalito Afterbay outlet, with the closest location being the Gridley gage about 7 miles downstream. However, there are no major diversions or inflows on the Feather River between the Thermalito Afterbay outlet and the Gridley gage locations, so the CALSIM II flow estimates for the Thermalito Afterbay outlet are expected to provide a reasonable approximation of flows at the Gridley gage. The CALSIM II flows at Thermalito were used with the river stage versus flow table for Gridley gage to estimate changes in river stage attributable to Alternatives 1, 2, and 3.

The results show a range of about 1% to 60% in ammocoete stranding for the Feather River, with little difference among alternatives in the maximum stranding (Table 11-96). There are only minor difference in mean ammocoete stranding between Alternatives 1, 2, and 3 and the NAA. The largest differences are 3% to 5% (absolute scale) increases in the stranding index for October of Dry Water Years under Alternatives 1, 2, and 3. However, October is outside of the cohort initiation periods of both lamprey species and therefore does not represent actual ammocoete stranding. Alternatives 1, 2, and 3 would have no adverse effect on dewatering risk of ammocoete rearing habitat for Pacific and river lamprey in the Feather River.

Table 11-96. Percent of Pacific and River Lamprey Ammocoetes Stranded During 7-Year Rearing Period in the Feather River at Gridley Gage and Differences in the Percentages for the NAA and Alt 1A, Alt 1B, Alt 2, and Alt 3. January through August Results Pertain to Pacific Lamprey and the February through June Results Pertain to River Lamprey.

Initial Month of Cohort	Water Year Type[1]	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
January	Wet	41.0	41.1 (0.1)	41.1 (0.1)	41.1 (0.1)	41.2 (0.2)

Initial Month of Cohort	Water Year Type[1]	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
	Above Normal	12.7	12.7 (0)	12.7 (0)	12.7 (0)	12.8 (0.1)
	Below Normal	10.3	10.4 (0.1)	9.6 (-0.7)	10.4 (0.1)	10.5 (0.2)
	Dry	6.3	6.3 (0)	6.3 (0)	6.3 (0)	6.3 (0)
	Critically Dry	3.9	3.9 (0)	3.9 (0)	3.9 (0)	4.1 (0.2)
	All	18.6	18.6 (0)	18.5 (-0.1)	18.6 (0)	18.7 (0.1)
February	Wet	53.7	53.9 (0.2)	53.8 (0.1)	53.9 (0.2)	54.1 (0.4)
	Above Normal	26.1	26.1 (0)	26 (-0.2)	26.1 (0)	26.2 (0.1)
	Below Normal	16.4	16.4 (0)	16.3 (-0.1)	16.4 (0)	16.5 (0.1)
	Dry	7.8	7.8 (0)	7.8 (0)	7.8 (0)	7.8 (0)
	Critically Dry	6.1	6.1 (0)	6.1 (0)	6.1 (0)	6.3 (0.2)
	All	26.2	26.3 (0.1)	26.2 (0)	26.3 (0)	26.4 (0.2)
March	Wet	60.0	60 (0)	60 (0)	60 (0)	60.2 (0.2)
	Above Normal	30.4	30.5 (0.1)	30.5 (0.1)	30.5 (0.1)	29.1 (-1.2)
	Below Normal	10.3	10.3 (0)	10.4 (0.2)	10.3 (0)	10.4 (0.1)
	Dry	7.9	8 (0.1)	8 (0.1)	8 (0.1)	8 (0.1)
	Critically Dry	8.5	8.5 (0)	8.5 (0)	8.5 (0)	8.9 (0.4)
	All	28.2	28.2 (0)	28.3 (0.1)	28.2 (0)	28.2 (0)
April	Wet	41.0	41 (0)	41 (0)	41 (0)	41.2 (0.2)
	Above Normal	9.7	11 (1.3)	11.2 (1.5)	10.8 (1.1)	10.8 (1.1)
	Below Normal	1.3	1.3 (0)	1.3 (0)	1.3 (0)	1.6 (0.3)
	Dry	1.9	1.8 (-0.1)	1.8 (-0.1)	1.8 (-0.1)	1.8 (-0.1)
	Critically Dry	2.9	2.9 (0)	2.9 (0)	2.9 (0)	3.2 (0.3)
	All	15.5	15.7 (0.2)	15.7 (0.2)	15.6 (0.2)	15.8 (0.3)
May	Wet	45.4	45.4 (0)	45.4 (0)	45.4 (0)	45.6 (0.1)
	Above Normal	16.7	16.7 (0)	16.7 (0)	16.7 (0)	16.8 (0.1)
	Below Normal	7.9	7.7 (-0.2)	7.7 (-0.3)	7.7 (-0.2)	7.9 (0)
	Dry	11.7	11.9 (0.2)	11.3 (-0.4)	11.9 (0.2)	10.9 (-0.8)
	Critically Dry	11.9	11.8 (-0.1)	11.7 (-0.2)	11.9 (0)	12 (0)
	All	22.5	22.5 (0)	22.4 (-0.2)	22.5 (0)	22.4 (-0.1)
June	Wet	32.2	32.2 (0)	32.2 (0)	32.2 (0)	32.4 (0.2)
	Above Normal	19.5	19.5 (0)	19.1 (-0.4)	19.5 (0)	18.7 (-0.8)
	Below Normal	19.6	18.7 (-0.9)	18.4 (-1.1)	18.7 (-0.9)	19 (-0.5)
	Dry	32.3	28.1 (-4.2)	28 (-4.3)	28.3 (-4)	28.2 (-4.2)
	Critically Dry	30.7	27 (-3.7)	27.2 (-3.5)	27.1 (-3.6)	27.8 (-2.9)
	All	28.0	26.4 (-1.6)	26.3 (-1.7)	26.4 (-1.6)	26.5 (-1.5)
July	Wet	36.0	36 (0)	36.1 (0.1)	36 (0)	36.3 (0.3)
	Above Normal	50.2	50.3 (0.1)	50.3 (0.1)	50.3 (0.1)	49.1 (-1.1)
	Below Normal	60.2	58.9 (-1.3)	59.1 (-1)	59 (-1.1)	59.2 (-1)
	Dry	48.7	47.2 (-1.6)	47.5 (-1.2)	47.3 (-1.4)	48 (-0.7)

Initial Month of Cohort	Water Year Type[1]	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
	Critically Dry	30.1	29.3 (-0.7)	29.4 (-0.6)	29.6 (-0.5)	29.9 (-0.2)
	All	44.0	43.4 (-0.7)	43.6 (-0.5)	43.5 (-0.6)	43.7 (-0.4)
August	Wet	28.8	28.8 (0)	29.4 (0.6)	28.8 (0)	29.6 (0.8)
	Above Normal	44.2	44.2 (0)	44.2 (0)	44.2 (0)	44.7 (0.4)
	Below Normal	54.3	54.2 (-0.2)	54.3 (-0.1)	54.2 (-0.2)	54.6 (0.3)
	Dry	13.5	15.1 (1.5)	14.7 (1.2)	15.1 (1.5)	15 (1.5)
	Critically Dry	16.9	18.8 (1.9)	18.6 (1.7)	18.9 (2)	19.6 (2.7)
	All	30.2	30.7 (0.6)	30.8 (0.7)	30.8 (0.6)	31.2 (1.1)
September	Wet	40.8	41.2 (0.5)	41.5 (0.8)	41.4 (0.6)	41.6 (0.8)
	Above Normal	50.3	50.3 (0)	50.6 (0.2)	50.3 (0)	50.4 (0.1)
	Below Normal	13.4	15.7 (2.3)	16.5 (3)	15.7 (2.3)	17.4 (4)
	Dry	2.7	3.8 (1.1)	3.6 (0.9)	3.7 (1)	3.6 (0.9)
	Critically Dry	4.8	5.2 (0.4)	5.4 (0.6)	5.3 (0.5)	5.5 (0.7)
	All	23.6	24.4 (0.9)	24.6 (1.1)	24.4 (0.9)	24.8 (1.3)
October	Wet	24.4	24.3 (0)	24.4 (0.1)	24.3 (0)	24.8 (0.4)
	Above Normal	25.2	25.2 (0)	25.3 (0.1)	25.2 (0)	25.4 (0.2)
	Below Normal	13.4	13.7 (0.3)	13.2 (-0.3)	13.7 (0.3)	13.1 (-0.4)
	Dry	3.9	8.1 (4.2)	8.9 (5)	8.1 (4.2)	7.2 (3.3)
	Critically Dry	4.1	5.4 (1.3)	4.9 (0.8)	5.1 (1)	4.1 (0)
	All	15.2	16.3 (1.2)	16.4 (1.2)	16.3 (1.1)	16 (0.8)
November	Wet	16.4	16.4 (0)	16.4 (0)	16.4 (0)	16.6 (0.2)
	Above Normal	13.5	13.5 (0)	13.3 (-0.2)	13.5 (0)	13 (-0.5)
	Below Normal	9.1	9.8 (0.7)	9.9 (0.8)	9.8 (0.7)	10.3 (1.2)
	Dry	4.3	6.6 (2.3)	5.5 (1.2)	6.6 (2.3)	5.7 (1.4)
	Critically Dry	0.7	1.8 (1)	1.8 (1)	1.8 (1)	1.9 (1.1)
	All	9.8	10.6 (0.8)	10.3 (0.5)	10.5 (0.8)	10.4 (0.7)
December	Wet	25.2	24.9 (-0.3)	24.6 (-0.6)	24.7 (-0.5)	24.9 (-0.3)
	Above Normal	19.4	19.3 (-0.1)	19.3 (-0.1)	19.3 (-0.1)	19.2 (-0.2)
	Below Normal	11.7	11.8 (0.1)	11.7 (0)	11.7 (0)	11.7 (0)
	Dry	10.6	10.8 (0.2)	10.9 (0.3)	10.7 (0.1)	10.9 (0.3)
	Critically Dry	8.6	8.6 (0)	8.6 (0)	8.6 (0)	8.7 (0.1)
	All	16.4	16.3 (-0.1)	16.3 (-0.2)	16.3 (-0.1)	16.4 (0)

[1] Water year type sorting is by hydrologic water year.

American River

Temperature Effects

Visual observation of exceedance plots and differences in modeled mean monthly temperatures by water year type between alternatives and the NAA in the American River below Nimbus Dam, at Watt Avenue, and at the mouth indicates that water temperatures would be predominantly similar among alternatives during the period of presence of each life stage of the two lamprey species (Appendix 6C, Tables 6C-13-1a to 6C-13-4c, Tables 6C-14-1a to 6C-14-4c, Tables 6C-15-1a to 6C-15-4c; Figures 6C-13-1 to 6C-13-18, Figures 6C-14-1 to 6C-14-18, Figures 6C-15-1 to 6C-15-18). At all locations, mean monthly water temperatures for all months within all water year types under Alternatives 1A and 1B were within 0.3°F of the NAA. Water temperature modeling results for Alternatives 2 and 3 were similar to those of Alternative 1 at all locations.

Results of the analysis of exceedance above water temperature index values or occurrence outside index ranges for Pacific and river lampreys in the American River from Appendix 11B, Table 11B-4 are presented in Appendix 11D, Table 11D-259 through Table 11D-270. At all three locations analyzed (below Nimbus Dam, Watt Avenue, and the confluence with the Sacramento River), there would be no differences in exceedance of water temperature indices for any life stage of both lampreys between Alternatives 1A and 1B compared to the NAA in any month or water year type. Results of the water temperature exceedance analysis for Pacific and river lamprey for Alternative 2 are similar to those for Alternative 1A.

Results of the water temperature exceedance analysis for Pacific and river lamprey for Alternative 3 are similar to those for Alternative 1A with one exception. At Watt Avenue in July of Critically Dry Water Years, there would be 5.1% fewer days on which water temperatures exceed the 71.6°F index value for ammocoete rearing and emigration of both lampreys under Alternative 3 relative to the NAA.

Flow Effects

Spawning Habitat

Spawning habitat effects for Pacific and river lamprey were evaluated for the American River by analyzing redd dewatering risk as described above for the Sacramento River. CALSIM II flow outputs for the American River at Nimbus Dam and the H Street Bridge were used to compare the American River redd dewatering risks between Alternatives 1, 2, and 3 and the NAA.

The results for Pacific lamprey of the potential redd dewatering risks primarily show small reductions in frequency of month-over-month >50% reductions in flow during spawning months under Alternatives 1, 2, and 3 at both river locations (Table 11-97). The results for river lamprey primarily show increases in frequency of month-over-month >50% reductions in flow under Alternatives 1, 2, and 3 at both river locations, including larger increases under Alternatives 1A and 1B (Table 11-98). The increased frequency of >50% flow reductions during the river lamprey spawning period has the potential to increase redd dewatering risk for river lamprey, negatively affecting their spawning habitat. However, the effect is not large; the largest increase is 1% on an absolute scale (6.3% relative difference). Overall, Alternatives 1, 2, and 3 are not expected to substantially affect redd dewatering risk of Pacific or river lamprey in the American River.

Table 11-97. Percentage of Pacific Lamprey Spawning Months with >50% Flow Reduction in the Next Month, Used as a Proxy for Redd Dewatering Risk for Locations in the American River, and Percent Differences (in parentheses) between the NAA and Alt 1A, Alt 1B, Alt 2, and Alt 3.

Location	NAA	Alt 1A vs. NAA	Alt 1B vs. NAA	Alt 2 vs. NAA	Alt 3 vs. NAA
American River at Nimbus Dam	13.9	13.9 (0%)	13.9 (0%)	13.7 (-1.1%)	13.3 (-4.4%)
American River at H Street	15.1	15.2 (1%)	15.2 (1%)	15.1 (0%)	14.6 (-3%)

Note: Positive value for percent differences (in parentheses) indicates a higher risk of redd dewatering under Alternative 1–3 vs. the NAA and negative values indicate a lower risk.

Table 11-98. Percentage of River Lamprey Spawning Months with >50% Flow Reduction in the Next Month, Used as a Proxy for Redd Dewatering Risk for Locations in the American River, and Percent Differences (in parentheses) between the NAA and Alt 1A, Alt 1B, Alt 2, and Alt 3.

Location	NAA	Alt 1A vs. NAA	Alt 1B vs. NAA	Alt 2 vs. NAA	Alt 3 vs. NAA
American River at Nimbus Dam	12.9	13.6 (5.7%)	13.8 (7.5%)	13.1 (1.9%)	13.1 (1.9%)
American River at H Street	15.3	16 (4.8%)	16.3 (6.3%)	15.8 (3.2%)	15.8 (3.2%)

Note: Positive value for percent differences (in parentheses) indicates a higher risk of redd dewatering under Alternative 1–3 vs. the NAA and negative values indicate a lower risk.

Rearing Habitat

Rearing habitat effects for Pacific and river lamprey were evaluated for the American River by analyzing ammocoete habitat dewatering risk as described above for the Sacramento River. CALSIM II flow outputs for the American River at Nimbus Dam were used to compare the ammocoete stranding risks between Alternatives 1, 2, and 3 and the NAA. The river stage vs. flow relationships used for the American River analysis were obtained from the salmonid redd dewatering study (Bratovich et al. 2017) that was used to develop the procedures for estimating effects of Alternatives 1, 2, and 3 on redd dewatering of fall-run Chinook salmon and steelhead in the American River (Appendix 11N).

The results show a range of about 0% to 52% in ammocoete stranding for the American River, with little difference among alternatives in the maximum stranding (Table 11-99). There are only minor difference in mean stranding between Alternatives 1, 2, and 3 and the NAA. The largest differences are 3% (absolute scale) reductions and increases under Alternative 3 for Above Normal Water Years in September and November, respectively. Alternatives 1, 2, and 3 would have little effect on dewatering risk of ammocoete rearing habitat for Pacific and river lamprey in the American River.

Table 11-99. Percent of Pacific and River Lamprey Ammocoetes Stranded During 7-Year Rearing Period in the American River at Nimbus Dam and Differences in the Percentages for the NAA and Alt 1A, Alt 1B, Alt 2, and Alt 3. January through August Results Pertain to Pacific Lamprey and the February through June Results Pertain to River Lamprey.

Initial Month of Cohort	Water Year Type[1]	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
January	Wet	48.5	48.7 (0.2)	48.5 (0)	48.4 (-0.2)	48.4 (-0.2)
	Above Normal	32.8	33 (0.1)	32.7 (-0.1)	32.7 (-0.2)	32.5 (-0.3)
	Below Normal	17.4	17.9 (0.5)	17.4 (0)	17.8 (0.4)	17.2 (-0.2)
	Dry	9.6	9.8 (0.2)	9.3 (-0.2)	9.5 (-0.1)	9.4 (-0.2)
	Critically Dry	9.4	9.4 (0)	9.4 (0)	8.9 (-0.5)	9.2 (-0.2)
	All	26.6	26.8 (0.2)	26.6 (-0.1)	26.5 (-0.1)	26.4 (-0.2)
February	Wet	51.3	51.5 (0.2)	51.3 (0)	51.2 (-0.2)	51.3 (0)
	Above Normal	39.8	39.9 (0.1)	39.5 (-0.3)	39.6 (-0.1)	39.1 (-0.7)
	Below Normal	33.2	33.3 (0.1)	33 (-0.2)	33 (-0.2)	32.8 (-0.4)
	Dry	16.2	16.3 (0.1)	16.2 (0)	15.9 (-0.3)	16.2 (0)
	Critically Dry	11.4	12.6 (1.3)	12.6 (1.3)	12.2 (0.9)	12.2 (0.8)
	All	33.0	33.3 (0.3)	33.1 (0.1)	33 (0)	33 (0)
March	Wet	37.4	37.7 (0.3)	37.4 (0)	37.3 (-0.2)	37.4 (0)
	Above Normal	33.0	33.2 (0.1)	33 (0)	32.9 (-0.1)	33 (-0.1)
	Below Normal	15.8	16 (0.1)	15.8 (0)	15.9 (0)	15.4 (-0.4)
	Dry	14.6	14.6 (0.1)	14.6 (0)	14.3 (-0.3)	14.6 (0)
	Critically Dry	8.6	8.3 (-0.4)	8.6 (-0.1)	7.5 (-1.1)	8.3 (-0.3)
	All	23.9	24 (0.1)	23.9 (0)	23.6 (-0.3)	23.7 (-0.1)
April	Wet	37.8	38 (0.3)	37.8 (0)	37.6 (-0.2)	37.8 (0)
	Above Normal	26.0	26.2 (0.1)	26 (0)	25.9 (-0.1)	25.9 (-0.1)
	Below Normal	19.8	20.2 (0.4)	19.8 (0)	20 (0.2)	19.7 (-0.1)
	Dry	9.8	9.9 (0.1)	9.7 (-0.1)	9.6 (-0.3)	10.1 (0.3)
	Critically Dry	10.9	10.8 (-0.1)	11.3 (0.4)	9.3 (-1.6)	9.3 (-1.6)
	All	22.9	23.1 (0.2)	23 (0)	22.6 (-0.3)	22.7 (-0.2)
May	Wet	46.7	46.9 (0.2)	46.7 (0)	46.5 (-0.2)	46.7 (0)
	Above Normal	33.4	33.6 (0.1)	33.4 (0)	33.3 (-0.2)	33.4 (-0.1)
	Below Normal	29.6	29.8 (0.1)	29.6 (0)	29.7 (0)	29.3 (-0.3)
	Dry	16.5	16.6 (0.1)	15.9 (-0.5)	16.2 (-0.2)	15.9 (-0.6)
	Critically Dry	9.0	8.1 (-0.9)	8.1 (-1)	9 (-0.1)	9 (-0.1)
	All	29.7	29.7 (0)	29.4 (-0.3)	29.5 (-0.2)	29.5 (-0.2)
June	Wet	35.7	35.9 (0.2)	35.7 (0)	35.5 (-0.2)	35.7 (0)
	Above Normal	24.5	24.7 (0.1)	23.7 (-0.8)	24.4 (-0.1)	24 (-0.6)
	Below Normal	19.9	20 (0.1)	19.6 (-0.3)	19.9 (0)	19.6 (-0.3)
	Dry	17.4	17.3 (0)	17.3 (-0.1)	17 (-0.4)	16.9 (-0.5)
	Critically Dry	8.2	8.1 (0)	8.1 (0)	7.9 (-0.2)	8.5 (0.3)

Initial Month of Cohort	Water Year Type[1]	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
	All	23.3	23.4 (0.1)	23.1 (-0.2)	23.1 (-0.2)	23.1 (-0.2)
July	Wet	26.8	27 (0.2)	26.8 (0)	26.6 (-0.2)	26.8 (0)
	Above Normal	30.2	30.4 (0.1)	29.6 (-0.6)	30.1 (-0.2)	28.4 (-1.9)
	Below Normal	33.5	33.8 (0.3)	33.7 (0.2)	34.2 (0.7)	32.3 (-1.3)
	Dry	19.4	19.3 (-0.1)	19.2 (-0.2)	18.9 (-0.5)	19.2 (-0.2)
	Critically Dry	10.8	11.4 (0.5)	11 (0.2)	10.5 (-0.3)	10.8 (0)
	All	24.4	24.6 (0.2)	24.4 (-0.1)	24.3 (-0.1)	23.9 (-0.5)
August	Wet	25.8	26 (0.1)	25.6 (-0.2)	25.5 (-0.3)	25.8 (0)
	Above Normal	22.7	22.9 (0.2)	22.6 (-0.1)	22.6 (-0.2)	21.1 (-1.6)
	Below Normal	17.3	17.2 (-0.1)	17.1 (-0.3)	17.4 (0.1)	16.4 (-1)
	Dry	15.3	15.8 (0.5)	15.7 (0.4)	15.4 (0.1)	16.1 (0.8)
	Critically Dry	11.3	9.6 (-1.6)	9.9 (-1.4)	9.5 (-1.7)	9.3 (-2)
	All	19.4	19.4 (-0.1)	19.2 (-0.2)	19.1 (-0.3)	18.9 (-0.5)
September	Wet	17.6	17.8 (0.1)	18 (0.4)	17.4 (-0.3)	17.7 (0.1)
	Above Normal	20.2	20.3 (0.1)	19.9 (-0.3)	19.7 (-0.5)	17.6 (-2.6)
	Below Normal	18.5	18.4 (-0.1)	18.2 (-0.2)	18.3 (-0.2)	17.9 (-0.6)
	Dry	12.3	12.4 (0.1)	12.3 (0)	12 (-0.3)	12.3 (0.1)
	Critically Dry	5.5	5.5 (0)	5.5 (0)	5.3 (-0.2)	5.5 (0)
	All	15.1	15.2 (0.1)	15.2 (0)	14.9 (-0.3)	14.7 (-0.4)
October	Wet	14.3	14.4 (0.1)	14.3 (0)	14 (-0.3)	14.2 (0)
	Above Normal	13.9	14 (0.1)	14.1 (0.2)	13.7 (-0.2)	14.2 (0.3)
	Below Normal	11.5	11.6 (0)	11.6 (0.1)	11.4 (-0.2)	12.2 (0.7)
	Dry	6.0	5.8 (-0.2)	5.8 (-0.1)	5.4 (-0.5)	6 (0)
	Critically Dry	4.1	4.1 (0)	4.1 (0)	3.8 (-0.3)	4.2 (0)
	All	10.5	10.5 (0)	10.5 (0)	10.2 (-0.3)	10.6 (0.2)
November	Wet	30.5	30.6 (0.2)	29.9 (-0.5)	30.1 (-0.4)	30.4 (-0.1)
	Above Normal	21.9	21.7 (-0.2)	22.4 (0.4)	21.8 (-0.2)	24.8 (2.8)
	Below Normal	20.3	20.3 (0)	20 (-0.3)	20.1 (-0.2)	21.2 (0.9)
	Dry	9.8	9.5 (-0.3)	10 (0.3)	9.1 (-0.7)	11 (1.2)
	Critically Dry	2.6	4 (1.4)	3.5 (0.9)	3.3 (0.8)	4 (1.4)
	All	18.9	19 (0.2)	18.9 (0)	18.6 (-0.2)	19.9 (1)
December	Wet	26.7	26.8 (0.2)	26.8 (0.1)	26.4 (-0.2)	26.8 (0.1)
	Above Normal	26.3	26.2 (0)	26.1 (-0.2)	26.1 (-0.1)	26.3 (0)
	Below Normal	28.4	28.7 (0.3)	28.7 (0.3)	28.5 (0.1)	29.1 (0.7)
	Dry	23.1	23.3 (0.2)	23.3 (0.2)	23.1 (-0.1)	23.9 (0.8)
	Critically Dry	0.3	0.7 (0.3)	0.3 (0)	0 (-0.3)	0.3 (0)
	All	22.3	22.4 (0.2)	22.4 (0.1)	22.1 (-0.1)	22.6 (0.3)

[1] Water year type sorting is by hydrologic water year.

Delta

Entrainment

Both Pacific and river lamprey are entrained at the south Delta export facilities. Concerns exist over the relatively low salvage efficiency of these facilities for the lamprey species (Goodman et al. 2017). Potential differences in entrainment risk between Alternatives 1, 2, and 3 and the NAA were assessed with the salvage-density method (Appendix 11Q) for each species as well as for lamprey not identified to species (indicated as 'unknown' in the historical salvage database). The differences between Alternatives 1, 2, and 3 and the NAA were minor (Tables 11-100, 11-101, 11-102, 11-103, 11-104, and 11-105), indicating entrainment risk under Alternatives 1, 2, and 3 would be similar to the NAA.

Table 11-100. Salvage of Pacific Lamprey At SWP Banks Pumping Plant, Averaged by Water Year Type, Based on the Salvage-Density Method.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	0	0 (0%)	0 (0%)	0 (0%)	0 (0%)
Above Normal	NA	NA	NA	NA	NA
Below Normal	0	0 (0%)	0 (0%)	0 (0%)	0 (0%)
Dry	0	0 (0%)	0 (0%)	0 (0%)	0 (0%)
Critically Dry	0	0 (0%)	0 (0%)	0 (0%)	0 (0%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. The analysis was based on historical salvage data during 2009–2019 which did not include any Above Normal Water Years, hence that row of the table is noted as 'NA'; the percentage difference in Above Normal Water Years is based on density data for Wet Water Years applied to above normal modeled exports. Results by water year type and month are provided in Appendix 11Q.

Table 11-101. Salvage of Pacific Lamprey At CVP Jones Pumping Plant, Averaged by Water Year Type, Based on the Salvage-Density Method.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	79	79 (0%)	79 (0%)	79 (0%)	78 (-1%)
Above Normal	NA	(0%)	(0%)	(0%)	(1%)
Below Normal	773	773 (0%)	774 (0%)	775 (0%)	779 (1%)
Dry	21	21 (1%)	22 (1%)	21 (1%)	22 (4%)
Critically Dry	372	372 (0%)	380 (2%)	372 (0%)	383 (3%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. The analysis was based on historical salvage data during 2009–2019 which did not include any Above Normal Water Years, hence that row of the table is noted as 'NA'; the percentage difference in Above Normal Water Years is based on density data for Wet Water Years applied to above normal modeled exports. Results by water year type and month are provided in Appendix 11Q. Percentage values are rounded; as a result, differences between percentages may not always appear consistent.

Table 11-102. Salvage of River Lamprey At SWP Banks Pumping Plant, Averaged by Water Year Type, Based on the Salvage-Density Method.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	0	0 (0%)	0 (0%)	0 (0%)	0 (0%)
Above Normal	NA	NA	NA	NA	NA

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Below Normal	0	0 (0%)	0 (0%)	0 (0%)	0 (0%)
Dry	0	0 (0%)	0 (0%)	0 (0%)	0 (0%)
Critically Dry	0	0 (0%)	0 (0%)	0 (0%)	0 (0%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. The analysis was based on historical salvage data during 2009–2019 which did not include any Above Normal Water Years, hence that row of the table is noted as 'NA'; the percentage difference in Above Normal Water Years is based on density data for Wet Water Years applied to above normal modeled exports. Results by water year type and month are provided in Appendix 11Q.

Table 11-103. Salvage of River Lamprey At CVP Jones Pumping Plant, Averaged by Water Year Type, Based on the Salvage-Density Method.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	1	1 (0%)	1 (0%)	1 (0%)	1 (0%)
Above Normal	NA	(0%)	(0%)	(0%)	(0%)
Below Normal	5	5 (0%)	5 (0%)	5 (0%)	5 (0%)
Dry	0	0 (0%)	0 (0%)	0 (0%)	0 (0%)
Critically Dry	4	4 (0%)	5 (11%)	4 (0%)	5 (12%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. The analysis was based on historical salvage data during 2009–2019 which did not include any Above Normal Water Years, hence that row of the table is noted as 'NA'; the percentage difference in Above Normal Water Years is based on density data for Wet Water Years applied to above normal modeled exports. Results by water year type and month are provided in Appendix 11Q. Percentage values are rounded; as a result, differences between percentages may not always appear consistent.

Table 11-104. Salvage of Unknown Species of Lamprey At SWP Banks Pumping Plant, Averaged by Water Year Type, Based on the Salvage-Density Method.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	869	870 (0%)	870 (0%)	870 (0%)	869 (0%)
Above Normal	NA	(0%)	(0%)	(0%)	(-1%)
Below Normal	159	159 (0%)	159 (0%)	159 (0%)	161 (1%)
Dry	59	58 (-2%)	56 (-6%)	58 (-1%)	56 (-5%)
Critically Dry	49	50 (1%)	50 (1%)	50 (1%)	50 (0%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. The analysis was based on historical salvage data during 2009–2019 which did not include any Above Normal Water Years, hence that row of the table is noted as 'NA'; the percentage difference in Above Normal Water Years is based on density data for Wet Water Years applied to above normal modeled exports. Results by water year type and month are provided in Appendix 11Q. Percentage values are rounded; as a result, differences between percentages may not always appear consistent.

Table 11-105. Salvage of Unknown Species of Lamprey At CVP Jones Pumping Plant, Averaged by Water Year Type, Based on the Salvage-Density Method.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	7,625	7,613 (0%)	7,617 (0%)	7,610 (0%)	7,600 (0%)
Above Normal	NA	(0%)	(-1%)	(0%)	(0%)

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Below Normal	1,679	1,675 (0%)	1,678 (0%)	1,683 (0%)	1,687 (0%)
Dry	1,556	1,570 (1%)	1,484 (-5%)	1,564 (0%)	1,489 (-4%)
Critically Dry	38	38 (1%)	39 (2%)	38 (0%)	38 (2%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. The analysis was based on historical salvage data during 2009–2019 which did not include any Above Normal Water Years, hence that row of the table is noted as 'NA'; the percentage difference in Above Normal Water Years is based on density data for Wet Water Years applied to above normal modeled exports. Results by water year type and month are provided in Appendix 11Q. Percentage values are rounded; as a result, differences between percentages may not always appear consistent.

CEQA Significance Determination for Alternatives 1, 2, and 3

Similar to winter-run Chinook salmon, there may be a low level of risk to juvenile lamprey from stranding behind the fish screens during high-flow events, but such events would be rare and would not differ between Alternatives 1, 2, and 3 and the NAA in their frequency. With respect to attraction of adult lamprey to reservoir releases to the Sacramento River and risk of stranding under Alternative 2, this risk would be minimized by the combination of the 60-foot-long apron and weir. Both lamprey species at the two Sacramento River locations analyzed indicate that Alternatives 1, 2, and 3 would have no adverse effect on the frequency of months predicted to experience a month-over-month 50% reduction in flow. Thus, Alternatives 1, 2, and 3 are not expected to affect redd dewatering risk of Pacific lamprey or river lamprey in the Sacramento River. Results indicate that Alternatives 1, 2, and 3 would have no adverse effect on dewatering risk of the ammocoete rearing habitat of Pacific lamprey or river lamprey in the Sacramento River.

The results of the analysis of exceedance above water temperature index values or occurrence outside index ranges for Pacific and river lampreys in the Feather River are presented in Appendix 11B. Any differences between Alternatives 1, 2, or 3 and the NAA are not large enough or frequent enough to be considered a population-level impact.

Spawning habitat for Pacific and river lamprey was evaluated for the Feather River by analyzing redd dewatering risk as described above for the Sacramento River. The results for both lamprey species showed no differences between Alternatives 1, 2, and 3 and the NAA or minor reductions under Alternatives 1, 2, and 3 in the percentage of months with elevated dewatering risk (Table 11-95).

There are only minor difference in mean ammocoete stranding between Alternatives 1, 2, and 3 and the NAA. Alternatives 1, 2, and 3 would have no adverse effect on dewatering risk of ammocoete rearing habitat for Pacific and river lamprey in the Feather River.

Additional effects analyses details are found within the preceding subsections, including applicable discussions related to Sacramento River, Feather River, and American River analyses. Overall, the results indicate for Pacific lamprey and river lamprey show that Alternatives 1, 2, and 3 are not expected to affect redd dewatering risk in the Sacramento, Feather, or American River. Results also indicate that Alternatives 1, 2, and 3 would have no adverse effect on dewatering risk of the ammocoete rearing habitat of Pacific lamprey or river lamprey in the

Sacramento, Feather, or American River. For these reasons, operation of Alternative 1, 2, or 3 would not have a substantial adverse effect, either directly or through habitat modifications, on Pacific lamprey or river lamprey. Operations impacts of Alternative 1, 2, or 3 would be less than significant.

<u>*NEPA Conclusion for Alternatives 1, 2, and 3*</u>

Operations effects on Pacific lamprey and river lamprey would be the same as described above for CEQA.

Operation of Alternative 1, 2, or 3 would have no adverse effect on Pacific lamprey and river lamprey.

Impact FISH-11: Operations Effects on Native Minnows (Sacramento Splittail, Sacramento Hitch, Hardhead, and Central California Roach)

Alternatives 1, 2, and 3

<u>*Effects of Alternatives 1, 2, and 3*</u>

<u>**Sacramento River**</u>

Near-Field Effects

Species accounts for the four native minnow species are provided in Appendix 11A (Sections 11A.1.12, *Sacramento Hitch*, 11A.1.13, *Sacramento Splittail*, 11A.1.14, *Hardhead*, and 11A.1.15, *Central California Roach*). Early life stages of native minnows occurring at the Red Bluff and Hamilton City intakes would be susceptible to risk of near-field effects such as entrainment at greater levels than the NAA if present during the main winter-early spring period of Alternative 1, 2, and 3 diversions to Sites Reservoir. As discussed in Appendix 11A, Sacramento splittail spawning occurs primarily in areas downstream of the two intakes, so the potential for negative near-field effects would be limited. Sacramento hitch spawning may occur during March–June, with juveniles occurring at stream margins before exiting shallow water after 2 months at about 50-mm FL (Moyle et al. 2015; see also Appendix 11A). As such, the risk of near-field effects under Alternatives 1, 2, and 3 generally would be similar to the NAA because there would be little difference in diversions during the May–August period when juveniles would be most likely to exit shallow water rearing habitat (Tables 11-6 and 11-7 in the winter-run Chinook salmon analysis). Hardhead spawning is mainly in April and May and as with other native minnows the young generally occur at stream margins before moving into deeper habitats at larger size (Appendix 11A). Similar to Sacramento hitch, this timing means that potential exposure of juveniles to the Red Bluff and Hamilton City intakes would occur when there would be little difference in diversions between Alternatives 1, 2, and 3 and the NAA. Central California roach tend to occur in small streams or in backwaters with dense riparian cover along the mainstem rivers (Appendix 11A). This distribution would tend to limit potential for negative near-field effects, although Moyle et al. (2015) noted that larval drift may be significant form of dispersal during some years, with the late spring timing and apparent short drift period possibly being adaptations to reduce risk of drifting downstream to unsuitable habitats such as the Central Valley floor. This lack of systematic, broad-scale migration past the Red Bluff and Hamilton intakes and the late spring timing would limit the potential for greater near-field effects on Central California roach under Alternatives 1, 2, and 3 given the general similarity of diversions to the NAA during this period (Tables 11-6 and 11-7 in the winter-run Chinook salmon analysis).

<u>Funks and Stone Corral Creeks</u>

Funks and Stone Corral Creeks would be modified downstream of the proposed dams. On Stone Corral Creek, the reach of interest is from the downstream face of the Sites Dam to just above the GCID Main Canal (7.7 miles); on Funks Creek, it is from the downstream face of Golden Gate Dam to the upper end of Funks Reservoir (1.8 miles). While these reaches have been modified by agricultural activities and minor diversions, they still have available fish habitat and both native and nonnative fish have been observed in each drainage. Additional information on Funks and Stone Corral Creeks can be found in Chapter 7.

Stone Corral Creek would receive bypass flows from the reservoir from an outlet on the Sites Dam, and Funks Creek would receive augmented flow from the Funks pipelines to its reaches upstream of Funks Reservoir. Using information from field studies, along with currently available information, the Authority will prepare a Funks and Stone Corral Creeks flow schedule that could be incorporated into the Reservoir Operations Plan that would identify the approach for releases, including release schedules and volumes, a monitoring plan, and an adaptive management plan to maintain fish in good condition consistent with California Fish and Game Code Section 5937 in Funks and Stone Corral Creeks (Chapter 2, Section 2.5.2.1, *Water Operations*, Funks Creek and Stone Corral Creek Releases). The field studies would be initiated once access is obtained and before final designs for Sites and Golden Gate Dams are completed. Additional information regarding the Stone Corral Creek and Funks Creek Aquatic Study Plan and Adaptive Management is provided in Section 2D.4 of Appendix 2D).

Far-Field Effects

Temperature Effects

Operation of Sites Reservoir has the potential to change water temperatures in the Sacramento River that could affect the life stages of native minnows present. As described in Appendix 11B, the two methods used to analyze temperature-related effects on native minnows in the Sacramento River were: (1) Physical Model Output Characterization; and (2) Water Temperature Index Value/Range Exceedance Analysis. More details on these methods are provided in Appendix 11B.

The Authority and Reclamation evaluated water temperature model outputs during the period of presence and in the locations of each life stage of native minnows (Sacramento splittail, Sacramento hitch, and hardhead) in the Sacramento River (see Appendix 11B, Table 11B-2 for timing and locations). Visual observation of exceedance plots and differences in modeled mean monthly temperatures by water year type between Alternatives 1, 2, and 3 and the NAA in the Sacramento River below Keswick Dam, below RBDD[36], Hamilton City, and Butte City indicates that water temperatures would be predominantly similar among alternatives during the period of

[36] The RBDD, which was decommissioned in 2013, and the RBPP are co-located, and the names may be used interchangeably when referring to geographic locations.

presence of each life stage (Appendix 6C, Tables 6C-5-1a to 6C-5-4c, Tables 6C-10-1a to 6C-10-4c, Tables 6C-11-1a to 6C-11-4c, Tables 6C-12-1a to 6C-12-4c; Figures 6C-5-1 to 6C-5-18, Figures 6C-10-1 to 6C-10-18, Figures 6C-11-1 to 6C-11-18, Figures 6C-12-1 to 6C-12-18). At all sites, mean monthly water temperatures for all months within all water year types under Alternatives 1A and 1B were within 0.6°F of the NAA with the exception of August of critical years at Butte City, in which mean monthly water temperature would be 0.8°F lower under the NAA than under Alternatives 1A and 1B. Water temperature modeling results for Alternatives 2 and 3 were similar to those of Alternative 1. These results suggest that temperature-related effects on native minnows in the Sacramento River would be negligible.

Results of the analysis of occurrence outside the 45°F to 75°F spawning index range for Sacramento splittail in the Sacramento River at Hamilton City from Appendix 11B, Table 11B-2 are presented in Appendix 11D, Table 11D-271. The percent of months outside the index range would be similar between the NAA and Alternatives 1, 2, and 3.

Results of the analysis of occurrence of water temperatures outside index range of Sacramento hitch spawning in the Sacramento River below RBDD[37] and at Butte City from Appendix 11B, Table 11B-2 are presented in Appendix 11D, Table 11D-272 and Table 11D-273. Below RBDD, the percent of days outside the index range would be similar between Alternatives 1A and 1B compared to the NAA with one exception. In June of Above Normal Water Years below RBDD, the occurrence outside the index range under Alternative 1B would be 8.9% lower than under the NAA. In addition, the occurrence of water temperatures outside the index range under Alternative 1B in July of Dry Water Years would be 5.9% greater than under the NAA. At Butte City, the percent of months outside the index range would be similar between Alternatives 1A and 1B compared to the NAA for all months of occurrence and water year types. Results for Alternative 2 would be similar to Alternative 1A in that the percent of days (below RBDD) or months (at Butte City) outside the index range would be similar to that of the NAA. Alternative 3 results would be similar to Alternative 1A except for some larger reductions in the occurrence of water temperatures outside the index range during June and July (up to 13.3%) relative to the NAA.

Results of the analysis of occurrence outside index ranges for hardhead in the Sacramento River below Keswick Dam and below RBDD from Appendix 11B, Table 11B-2 are presented in Appendix 11D, Table 11D-274 through Table 11D-277. At Keswick, the percent of days outside the spawning index range under Alternative 1A and 1B would be similar to the NAA. Results for Alternatives 2 and 3 would be similar to Alternative 1. Below RBDD, the percent of days outside the index range would be similar between Alternatives 1A and 1B compared to the NAA with one exception. For the 59°F to 64°F spawning range, there would be an 11.13% reduction in occurrence outside the index range under Alternative 1B compared to the NAA in June of Above Normal Water Years. Results for Alternative 2 would be similar to Alternative 1A. Results for Alternative 3 would be similar to Alternative 1B except that there would be 12.8% and 5.5% reductions in occurrence outside the index range under Alternative 3 compared to the NAA in June of Above Normal Water Years and Below Normal Water Years.

[37] The RBDD, which was decommissioned in 2013, and the RBPP are co-located, and the names may be used interchangeably when referring to geographic locations.

Due to low frequency and magnitude of differences between each alternative and the NAA in occurrences of daily or monthly water temperatures outside the index ranges in the Sacramento River, they are not expected to be persistent enough to affect native minnows at a population level.

Floodplain Inundation

Among the native minnows, floodplain inundation is especially important as spawning and early rearing habitat in the life history of Sacramento splittail (Appendix 11A). As described in Chapter 2 and as discussed for winter-run Chinook salmon, the Project's diversion criteria include restrictions to maintain Bend Bridge and Wilkins Slough flows, which have the effect of limiting changes to Yolo Bypass spill frequency and duration under Alternatives 1, 2, and 3 relative to the NAA. In particular, these criteria avoid impacts on Reclamation's ability to implement its obligations in the 2019 NMFS ROC ON LTO BiOp to implement the Yolo Bypass Restoration Salmonid Habitat Restoration and Fish Passage Implementation Plan and provide more than 17,000 acres of inundation in the Yolo Bypass from December to April (National Marine Fisheries Service 2019a). The Project would operate to avoid effects on the Big Notch's ability to achieve the same level of performance for salmonids in the Sacramento River as it would absent the Project. However, the Adaptive Management Plan for the Project recognized there is uncertainty about the performance of the Big Notch and the effects of the Project on it. Monitoring will be conducted, in cooperation with the State, to determine whether there is an effect and, if so, what the magnitude of that effect would be on entrainment of juvenile salmon into the Yolo Bypass. If there is an adverse effect, a science-based adaptive management approach will be employed to determine how to adjust diversions 158 RMs upstream of the Big Notch to maintain its efficiency for entraining juvenile salmon into the Yolo Bypass. As such, Alternatives 1, 2, and 3 would have limited potential for negative effects on Yolo Bypass floodplain inundation and access for Sacramento splittail. This was confirmed with the modeling summarized in the winter-run Chinook salmon section and included in Appendices 11M1 and 11M2.

Yolo Bypass Inundated Area

As described in Appendix 11M, inundated suitable habitat for salmonids and Sacramento splittail is defined as habitat with flow velocity <1.5 ft/s and depth <1 meter. These criteria are based on studies of habitat use by rearing juvenile salmonids and Sacramento splittail (U.S. Fish and Wildlife Service 2005b; Sommer et al. 2008; Merced Irrigation District 2013; Whipple et al. 2019). However, Sacramento splittail adults spawn at depths between about 0.5 and 2 meters (Moyle 2002; Merced Irrigation District 2013), so the suitable habitat acreages in Table 11-89 do not fully represent suitable habitat for Sacramento splittail spawning.

The modeling results of Yolo Bypass inundation show considerable increases in mean inundation acreage under Alternatives 1, 2, and 3 relative to the NAA during August through October (Table 11-106). These increases are the result of planned agricultural flow releases from Sites Reservoir. The releases reach the Yolo Bypass via the CBD, entirely bypassing the Sacramento River. For this reason and because of the months in which they occur, these summer-fall increases in potential habitat acreage have only minor effects on Sacramento splittail.

Significant spilling of the Fremont Weir generally begins in November or December and may occur as late as May. For November through June, the model results range from no change to moderate reductions in Yolo Bypass mean daily habitat acreage under Alternatives 1, 2, and 3 (Table 11-106). Absolute acreage reductions for this period range from minimums of no change during April of Critically Dry Water Years and May and June of all but Wet Water Years to maximums of over 426 to 457 acres during December of Below Normal Water Years under Alternatives 1, 2, and 3 (Table 11-106). Almost all changes during this period consist of reductions in acreage. The majority of the reductions during December through March exceed 100 acres (Table 11-106). The largest percentage reductions in acreage, 12%, occur in November of Below Normal Water Years under Alternatives 1, 2, and 3.

Table 11-106. Estimated Mean Daily Inundated Habitat (Thousands of Acres <1 Meter Deep) for Sacramento Splittail in the Yolo Bypass and the Absolute Differences (Acres, in parentheses) for the NAA and Alt 1A, Alt 1B, Alt 2, and Alt 3.

Month	Water Year Type[1]	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
January	Wet	14.52	14.36 (-156)	14.28 (-237)	14.36 (-156)	14.28 (-238)
	Above Normal	10.65	10.36 (-286)	10.36 (-285)	10.36 (-284)	10.38 (-268)
	Below Normal	6.19	6.05 (-146)	6.02 (-174)	6.04 (-157)	6.01 (-182)
	Dry	1.79	1.76 (-34)	1.75 (-41)	1.76 (-34)	1.75 (-39)
	Critically Dry	1.42	1.37 (-45)	1.37 (-45)	1.37 (-45)	1.38 (-44)
	All	7.78	7.66 (-129)	7.62 (-161)	7.65 (-130)	7.62 (-160)
February	Wet	17.26	17.23 (-33)	17.23 (-33)	17.22 (-43)	17.22 (-42)
	Above Normal	17.22	16.95 (-264)	16.97 (-249)	16.94 (-274)	16.89 (-324)
	Below Normal	10.58	10.38 (-197)	10.4 (-177)	10.38 (-196)	10.42 (-153)
	Dry	4.58	4.37 (-217)	4.35 (-236)	4.37 (-217)	4.35 (-238)
	Critically Dry	1.34	1.3 (-31)	1.31 (-30)	1.31 (-30)	1.3 (-31)
	All	10.92	10.78 (-133)	10.79 (-132)	10.78 (-138)	10.78 (-141)
March	Wet	14.62	14.59 (-34)	14.57 (-53)	14.59 (-33)	14.68 (64)
	Above Normal	14.51	14.32 (-196)	14.31 (-205)	14.33 (-183)	14.3 (-216)
	Below Normal	5.33	5.08 (-243)	5.1 (-230)	5.09 (-241)	5.09 (-240)
	Dry	3.78	3.61 (-176)	3.6 (-178)	3.61 (-175)	3.62 (-166)
	Critically Dry	1.37	1.33 (-44)	1.33 (-44)	1.33 (-44)	1.33 (-44)
	All	8.63	8.5 (-125)	8.5 (-130)	8.51 (-122)	8.53 (-94)
April	Wet	11.37	11.24 (-132)	11.24 (-130)	11.24 (-132)	11.16 (-210)
	Above Normal	5.07	5.14 (67)	5.13 (64)	5.14 (66)	5.13 (58)
	Below Normal	1.31	1.3 (-3)	1.3 (-3)	1.3 (-3)	1.3 (-3)
	Dry	1.20	1.2 (-2)	1.2 (-2)	1.2 (-2)	1.2 (0)
	Critically Dry	0.52	0.52 (0)	0.52 (0)	0.52 (0)	0.52 (0)
	All	4.91	4.87 (-34)	4.87 (-34)	4.87 (-35)	4.85 (-60)
May	Wet	2.76	2.64 (-113)	2.64 (-111)	2.64 (-112)	2.65 (-110)

Month	Water Year Type[1]	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
	Above Normal	0.82	0.82 (0)	0.82 (0)	0.82 (0)	0.82 (0)
	Below Normal	0.46	0.46 (0)	0.46 (0)	0.46 (0)	0.46 (0)
	Dry	0.27	0.27 (0)	0.27 (0)	0.27 (0)	0.27 (0)
	Critically Dry	0.17	0.17 (0)	0.17 (0)	0.17 (0)	0.17 (0)
	All	1.16	1.12 (-37)	1.12 (-36)	1.12 (-36)	1.12 (-36)
June	Wet	0.78	0.75 (-29)	0.75 (-28)	0.75 (-28)	0.75 (-28)
	Above Normal	0.16	0.16 (0)	0.16 (0)	0.16 (0)	0.16 (0)
	Below Normal	0.16	0.16 (0)	0.16 (0)	0.16 (0)	0.16 (0)
	Dry	0.16	0.16 (0)	0.16 (0)	0.16 (0)	0.16 (0)
	Critically Dry	0.16	0.16 (0)	0.16 (0)	0.16 (0)	0.16 (0)
	All	0.36	0.35 (-9)	0.35 (-9)	0.35 (-9)	0.35 (-9)
July	Wet	0.12	0.11 (-12)	0.11 (-12)	0.11 (-12)	0.11 (-12)
	Above Normal	0.11	0.1 (-13)	0.1 (-13)	0.1 (-14)	0.1 (-13)
	Below Normal	0.11	0.1 (-6)	0.1 (-6)	0.1 (-7)	0.1 (-5)
	Dry	0.11	0.11 (-5)	0.11 (-5)	0.11 (-5)	0.11 (-5)
	Critically Dry	0.12	0.12 (-1)	0.12 (-1)	0.11 (-6)	0.12 (-1)
	All	0.12	0.11 (-9)	0.11 (-9)	0.11 (-10)	0.11 (-9)
August	Wet	0.32	1.02 (702)	1.02 (703)	1.02 (701)	1.02 (703)
	Above Normal	0.22	0.91 (685)	0.91 (685)	0.99 (762)	0.91 (685)
	Below Normal	0.25	0.62 (364)	0.62 (367)	0.68 (425)	0.56 (304)
	Dry	0.14	0.47 (324)	0.46 (313)	0.44 (302)	0.46 (318)
	Critically Dry	0.13	0.22 (91)	0.22 (93)	0.47 (343)	0.21 (78)
	All	0.23	0.69 (467)	0.69 (465)	0.75 (520)	0.68 (454)
September	Wet	0.21	1 (793)	0.97 (766)	1.03 (827)	0.95 (744)
	Above Normal	0.17	0.98 (805)	0.98 (805)	1.05 (878)	0.94 (773)
	Below Normal	0.28	0.63 (353)	0.55 (269)	0.64 (366)	0.57 (297)
	Dry	0.16	0.28 (116)	0.3 (134)	0.31 (146)	0.29 (129)
	Critically Dry	0.16	0.2 (37)	0.2 (33)	0.31 (147)	0.17 (9)
	All	0.20	0.65 (456)	0.63 (436)	0.7 (502)	0.62 (425)
October	Wet	0.24	1 (756)	0.93 (686)	1.02 (775)	0.83 (592)
	Above Normal	0.09	0.71 (625)	0.7 (608)	0.86 (768)	0.53 (439)
	Below Normal	0.64	0.95 (311)	0.9 (265)	0.96 (321)	0.91 (271)
	Dry	0.11	0.17 (67)	0.17 (67)	0.23 (123)	0.22 (116)
	Critically Dry	0.10	0.18 (75)	0.11 (3)	0.12 (11)	0.1 (0)
	All	0.24	0.65 (407)	0.6 (363)	0.68 (437)	0.56 (322)
November	Wet	1.34	1.36 (25)	1.34 (1)	1.37 (28)	1.34 (-4)
	Above Normal	1.20	1.25 (50)	1.25 (48)	1.27 (64)	1.25 (48)

Month	Water Year Type[1]	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
	Below Normal	1.36	1.19 (-163)	1.19 (-163)	1.19 (-162)	1.2 (-161)
	Dry	0.66	0.6 (-61)	0.6 (-56)	0.6 (-55)	0.62 (-38)
	Critically Dry	0.05	0.06 (8)	0.05 (0)	0.05 (0)	0.05 (0)
	All	0.98	0.96 (-26)	0.95 (-34)	0.96 (-23)	0.95 (-31)
December	Wet	5.02	4.93 (-97)	4.92 (-109)	4.93 (-98)	4.92 (-101)
	Above Normal	5.89	5.59 (-297)	5.63 (-256)	5.6 (-292)	5.7 (-187)
	Below Normal	6.84	6.38 (-457)	6.41 (-426)	6.38 (-457)	6.4 (-433)
	Dry	4.85	4.84 (-7)	4.75 (-96)	4.84 (-7)	4.87 (24)
	Critically Dry	0.92	0.91 (-10)	0.91 (-10)	0.91 (-10)	0.91 (-10)
	All	4.81	4.65 (-153)	4.64 (-166)	4.65 (-153)	4.68 (-128)

[1] Water year type sorting is by hydrologic water year.

A further summary of Yolo Bypass inundated habitat acreages gives the net effect of all the November through May changes between the NAA and Alternatives 1, 2, and 3 in habitat acreage (Table 11-107). For this summary, the monthly means were computed for all daily habitat acreages from November through May for all water year types combined. The largest difference is a reduction of 164 acres for December under Alternative 1B, or 3.4% of the NAA acreage.

Table 11-107. Mean Daily November through May Inundated Habitat (Acres <1 Meter Deep) for Juvenile Salmonids in the Yolo Bypass and the Absolute Differences (in parentheses) for the NAA and Alt 1A, Alt 1B, Alt 2, and Alt 3.

Month	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
November	970	945 (-25)	937 (-34)	948 (-22)	939 (-31)
December	4,760	4,609 (-151)	4,597 (-164)	4,609 (-151)	4,633 (-127)
January	7,949	7,825 (-125)	7,793 (-157)	7,823 (-126)	7,791 (-158)
February	10,897	10,765 (-132)	10,766 (-131)	10,760 (-137)	10,757 (-140)
March	8,526	8,402 (-124)	8,397 (-129)	8,405 (-122)	8,433 (-93)
April	4,950	4,916 (-34)	4,916 (-33)	4,916 (-34)	4,891 (-59)
May	1,261	1,225 (-36)	1,226 (-35)	1,225 (-36)	1,226 (-35)
All (Nov–May)	5,616	5,527 (-90)	5,519 (-98)	5,527 (-90)	5,524 (-92)

The Yolo Bypass is the most important spawning, nursery, and juvenile rearing habitat for Sacramento splittail (Sommer et al. 2001, 2002, 2008; Moyle et al. 2004; Feyrer et al. 2006a, 2006b). Splittail use the bypass during the winter and spring, the natural period for seasonal floodplain inundation in the Sacramento River Basin. By late summer and fall, when Alternatives 1, 2, and 3 are expected to result in the largest percentage increases in Yolo Bypass inundation (Table 11-106), rearing Sacramento splittail have emigrated from the bypass, except for the relatively few trapped in pools (Sommer et al. 2005).

Adult Sacramento splittail begin their upstream spawning migrations from the Delta during winter and spring and spawn on the Yolo Bypass from late winter to late spring in years when the bypass is inundated. Timing of spawning depends on the timing of inundation, but most often peaks during March (Feyrer et al 2006a). Egg incubation and larval development require a few weeks to a month, depending on water temperature (Moyle et al. 2004). The juveniles rear in the bypass for as long as conditions are suitable, and typically return to the Delta by April through July (Feyrer et al. 2005).

Splittail benefit from Yolo Bypass inundation primarily during the spawning and rearing periods, which typically run from February through April or May. This period partially overlaps the period with the greatest and most consistent habitat reductions associated with Alternatives 1, 2, and 3 (Table 11-106). However, as noted above, the net effect of all daily differences between the NAA and Alternatives 1, 2, and 3 are relatively small reductions in habitat acreage (Table 11-107). The largest reduction for all water year types combined during the February through May period is a reduction of 140 acres (1.3%) during February under Alternative 3 (Table 11-107). Therefore, the habitat reductions are not expected to substantially affect the splittail population.

The results of the frequency analysis of inundation of events for the Yolo Bypass generally show only minor difference between Alternatives 1, 2, and 3 and the NAA (Appendix 11M, Figure 11M-7). However, there are reductions in frequency for Alternatives 1, 2, and 3 compared to the NAA for events of 15,000 to 20,000 acres lasting 8 to 17 days, with frequencies ranging from once per 3.8 years for Alternative 1A to once per 2.9 years for the NAA. There are increases for Alternatives 1A and 1B for events of >20,000 acres lasting 18 to 24 days, with frequencies ranging from once per 4.7 years for the NAA to once per 4.3 years for Alternative 1B. The differences in frequencies of inundation events of varying duration and acreage show no consistent differences between the NAA and Alternatives 1, 2, and 3.

Sutter Bypass Inundated Area

The Sutter Bypass when inundated, much as discussed for the Yolo Bypass, provides important spawning and rearing habitat for splittail (Moyle et al. 2004; Feyrer et al. 2006b; Cordoleani et al. 2020; Bellido-Leiva et al. 2021). For the Sutter Bypass the modeling results indicate that Alternatives 1, 2, and 3 would produce limited change in mean daily suitable habitat as compared to the NAA (Appendix 11M, Table 11M-4). The largest increases during the February through May period are 45 acres for February of Above Normal Water Years under Alternatives 1A, 1B, and 2. The only reductions during this period are 2- and 3-acre reductions in May of Below Normal Water Years under Alternatives 1, 2, and 3. Habitat changes on the Sutter Bypass resulting from Alternatives 1, 2, and 3 are not expected to affect the Sacramento splittail population.

Sacramento River Inundated Side-Channel Habitat Area

Like the floodplain habitat of the Yolo and Sutter Bypasses, inundated side-channel in the Sacramento River is believed to provide important habitat for several fish species, including Sacramento splittail. Splittail use inundated side-channel habitat for spawning and rearing (Moyle et al. 2004, 2015; Feyrer et al. 2005). Adult splittail have been found as far upstream as RBPP, but juveniles are generally not found upstream of about Colusa, so the upstream limit of splittail spawning is uncertain (Moyle et al. 2004; Feyrer et al. 2005). Some juvenile splittail have been found in upstream portions of the Sacramento River year-round, but most juveniles have emigrated to the Delta by July (Feyrer et al. 2005).

The modeling results for acreage of suitable side-channel habitat in the three reaches of the Sacramento River analyzed (Reach 1 = Bend Bridge to Hamilton City, Reach 2 = Hamilton City to Colusa, and Reach 3 = Colusa to Knights Landing) indicate that Alternatives 1, 2, and 3 would produce minor changes in mean daily suitable habitat compared to the NAA in all three reaches. Most of the changes constitute reductions in habitat (Appendix 11M, Table 11M-5 through Table 11M-7). The largest differences in inundated acreages during February through July, when most splittail spawning and rearing in off-channel habitats likely occurs, are reductions of 23 to 87 acres in February and March of all water year types except Critically Dry Water Years under Alternatives 1, 2, and 3 in Reach 2 (Table 11-6). The largest of these reductions is 4.7% of the NAA acreage.

The results of the frequency analysis of inundation of events for all three reaches combined also show some differences between the NAA and Alternatives 1, 2, and 3 (Appendix 11M, Figure 11M-9). The largest differences are 6% to 10% increases in frequency of events under Alternatives 1, 2, and 3 for 6,500- to 8,000-acre events lasting 18–24 days, and 6% reductions for the same range of acreages lasting over 24 days. Other results show little change.

Alternatives 1, 2, and 3 would result in both reductions and increases in acreage and frequency of suitable inundated side-channel habitat in the Sacramento River. On balance, however, the effects would not be large enough to substantially affect the Sacramento splittail population.

Feather River

Operation of Sites Reservoir has the potential to change water temperatures in the Feather River that could affect the life stages of native minnows present, Sacramento splittail, Sacramento hitch, and hardhead. As described in Appendix 11B, the two methods used to analyze temperature-related effects on native minnows in the Feather River were: (1) Physical Model Output Characterization; and (2) Water Temperature Index Value/Range Exceedance Analysis. More details on these methods are provided in Appendix 11B.

The Authority and Reclamation evaluated water temperature model outputs during the period of presence and in the locations of each life stage of native minnows in the Feather River (see Appendix 11B, Table 11B-3 for timing and locations). Visual observation of exceedance plots and differences in modeled mean monthly temperatures by water year type between Alternatives 1, 2, and 3 and the NAA in the Feather River LFC below the Fish Barrier Dam and in the HFC at Gridley Bridge and at the mouth indicates that water temperatures would be predominantly similar among Alternatives 1, 2, and 3 during the periods of presence of native minnows (Appendix 6C, Tables 6C-16-1a to 6C-16-4c, Tables 6C-18-1a to 6C-18-4c, Tables 6C-19-1a to 6C-19-4c; Figures 6C-16-1 to 6C-16-18, Figures 6C-18-1 to 6C-18-18, Figures 6C-19-1 to 6C-19-18). At all locations, mean monthly water temperatures for all months in all water year types under Alternatives 1A and 1B were within 0.5°F of the NAA. Water temperature modeling results for Alternatives 2 and 3 were similar to those of Alternative 1 at all locations. These results suggest that temperature-related effects on native minnows in the Feather River would be negligible.

Results of the analysis of occurrence outside the 45°F to 75°F spawning index range for Sacramento splittail at the mouth of the Feather River from Appendix 11B, Table 11B-3 are presented in Appendix 11D, Table 11D-278 The percent of months outside the index range would be similar between the NAA and Alternatives 1, 2, and 3.

Results of the analysis of occurrence of water temperatures outside index range of Sacramento hitch spawning in the Feather River LFC below the fish dam and in the HFC below Thermalito Afterbay outlet from Appendix 11B, Table 11B-3 are presented in Appendix 11D, Table 11D-279 and Table 11D-280. In the LFC below the fish dam, the percent of months with water temperatures outside the index range would be similar between Alternatives 1A and 1B compared to the NAA with two exceptions. In June of Dry and Critically Dry Water Years, there would be 5.6% to 11.1% fewer months under Alternatives 1A and 1B relative to the NAA in which water temperatures would be outside the index range. Results for Alternative 2 are similar to Alternative 1A except for an 8.3% increase in frequency outside the index range under Alternative 2 relative to the NAA in July of Critically Dry Water Years. Results for Alternative 3 are similar to Alternative 1B except for a 7.1% increase in frequency outside the index range under Alternative 3 relative to the NAA in June of Below Normal Water Years. Below Thermalito Afterbay outlet, the percent of months with water temperatures outside the index range would be similar between Alternatives 1A and 1B compared to the NAA except for a 5.6% reduction outside the index range in April of Dry Water Years. Results for Alternative 2 are identical to Alternative 1A. Results for Alternative 3 are similar to Alternative 1 with one exception. In April of Above Normal Water Years, there would be 9.1% fewer months under Alternative 3 relative to the NAA in which water temperatures are outside the index range.

Results of the analysis of occurrence outside index ranges for hardhead in the Feather River LFC below the fish dam and in the HFC below Thermalito Afterbay outlet and at the mouth from Appendix 11B, Table 11B-3 are presented in Appendix 11D, Table 11D-281 through Table 11D-286. In the LFC below the fish dam, the percent of months with water temperatures outside the hardhead spawning and juvenile and adult presence index ranges would be similar between Alternatives 1A and 1B compared to the NAA, except for an 8.3% increase in frequency of occurrence outside the juvenile and adult presence index range in August of Critically Dry Water Years in both Alternative 1A and 1B. Results for Alternatives 2 and 3 would be similar to Alternative 1, except there would be no difference in frequency of occurrence outside the index ranges in any month or water year type.

Below Thermalito Afterbay, the percent of months with water temperatures outside the hardhead index ranges would be similar between Alternatives 1A and 1B compared to the NAA with three exceptions. In May of Dry Water Years, there would be 5.6% fewer months under Alternatives 1A and 1B relative to the NAA in which water temperatures are outside the index range. In June of Dry Water Years, there would be 5.6% fewer months under Alternatives 1A and 1B relative to the NAA in which water temperatures are outside the spawning index range. In June of Critically Dry Water Years, there would be 8.3% more months under Alternatives 1A and 1B relative to the NAA in which water temperatures are outside the spawning index range. In September of Dry Water Years, there would be 11.1% fewer months under Alternatives 1A and 1B relative to the NAA in which water temperatures are outside the juvenile and adult presence index range.

Results for Alternatives 2 and 3 would be similar to Alternative 1, except for some minor differences.

At the Feather River mouth the percent of months with water temperatures outside the hardhead index ranges would be identical between Alternatives 1, 2, and 3 compared to the NAA.

Due to low frequency and magnitude of differences between each alternative and the NAA in exceedances above water temperature index values in the Feather River, they are not expected to be persistent enough to affect native minnows at a population level.

American River

Operation of Sites Reservoir has the potential to change water temperatures in the American River that could affect the life stages of native minnows present, Sacramento splittail, Sacramento hitch, and hardhead. As described in Appendix 11B, the two methods used to analyze temperature-related effects on native minnows in the American River were: (1) Physical Model Output Characterization; and (2) Water Temperature Index Value/Range Exceedance Analysis. More details on these methods are provided in Appendix 11B.

The Authority and Reclamation evaluated water temperature model outputs during the period of presence and in the locations of each life stage of native minnows in the American River (see Appendix 11B, Table 11B-4 for timing and locations). Visual observation of exceedance plots and differences in modeled mean monthly temperatures by water year type between Alternatives 1, 2, and 3 and the NAA in the American River below Nimbus Dam, at Watt Avenue, and at the mouth indicates that water temperatures would be predominantly similar among alternatives during the period of presence of native minnows: Sacramento splittail, Sacramento hitch, and hardhead (Appendix 6C, Tables 6C-13-1a to 6C-13-4c, Tables 6C-14-1a to 6C-14-4c, Tables 6C-15-1a to 6C-15-4c; Figures 6C-13-1 to 6C-13-18, Figures 6C-14-1 to 6C-14-18, Figures 6C-15-1 to 6C-15-18). At all locations, mean monthly water temperatures for all months in all water year types under Alternatives 1A and 1B were within 0.3°F of the NAA. Water temperature modeling results for Alternatives 2 and 3 were similar to those of Alternative 1 at all locations. These results suggest that temperature-related effects on native minnows in the American River would be negligible.

Results of the analysis of occurrence outside the 45°F to 75°F spawning index range for Sacramento splittail at the mouth of the American River from Appendix 11B, Table 11B-4 are presented in Appendix 11D, Table 11D-287. The percent of days outside the index range would be similar between the NAA and Alternatives 1, 2, and 3.

Results of the analysis of occurrence of water temperatures outside the 57°F to 79°F spawning index range of Sacramento hitch spawning in the American River below Nimbus Dam and at Watt Avenue from Appendix 11B, Table 11B-4 are presented in Appendix 11D, Table 11D-288 and Table 11D-289. At both locations, the percent of days outside the index range would be similar between the NAA and Alternatives 1, 2, and 3.

Results of the analysis of occurrence outside the 59°F to 64°F spawning index range and 65°F to 82.4°F juvenile and adult index range for hardhead in the American River at Watt Avenue from

Appendix 11B, Table 11B-4 are presented in Appendix 11D, Table 11D-290 and Table 11D-291. The percent of days outside both index ranges would be similar between the NAA and Alternatives 1, 2, and 3.

Due to low frequency and magnitude of differences between each alternative and the NAA in exceedances above water temperature index values in the American River, they are not expected to be persistent enough to affect native minnows at a population level.

Delta

Within the Delta, Alternatives 1, 2, and 3 have the potential to affect native minnows by increasing south Delta exports during summer/fall relative to the NAA. Few hitch, Central California roach, or hardhead have been salvaged historically and so increases in exports during summer/fall under Alternatives 1, 2, and 3 would not result in appreciable additional salvage; as described in Appendix 11A, these species occur mostly upstream of the Delta. For Sacramento splittail, the salvage-density method (Appendix 11Q) suggested entrainment risk under Alternatives 1, 2, and 3 generally would be similar to the NAA (Tables 11-108 and 11-109). Any differences would not give population-level consequences because the main driver of splittail population dynamics is floodplain habitat availability and entrainment is not an important driver (Sommer et al. 1997).

Table 11-108. Salvage of Sacramento Splittail At SWP Banks Pumping Plant, Averaged by Water Year Type, Based on the Salvage-Density Method.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	662,010	664,222 (0%)	663,036 (0%)	664,224 (0%)	661,902 (0%)
Above Normal	NA	(-11%)	(-11%)	(-11%)	(-2%)
Below Normal	5,936	6,093 (3%)	6,055 (2%)	6,085 (3%)	6,158 (4%)
Dry	597	605 (1%)	601 (1%)	604 (1%)	599 (0%)
Critically Dry	459	473 (3%)	470 (2%)	473 (3%)	468 (2%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. The analysis was based on historical salvage data during 2009–2019 which did not include any Above Normal Water Years, hence that row of the table is noted as 'NA'; the percentage difference in Above Normal Water Years is based on density data for Wet Water Years applied to above normal modeled exports. Results by water year type and month are provided in Appendix 11Q.

Table 11-109. Salvage of Sacramento Splittail At CVP Jones Pumping Plant, Averaged by Water Year Type, Based on the Salvage-Density Method.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	7,270,745	7,283,075 (0%)	7,285,624 (0%)	7,282,994 (0%)	7,269,881 (0%)
Above Normal	NA	(0%)	(0%)	(0%)	(0%)
Below Normal	73,412	73,630 (0%)	75,567 (3%)	73,632 (0%)	76,137 (4%)
Dry	1,388	1,395 (1%)	1,407 (1%)	1,395 (1%)	1,441 (4%)
Critically Dry	13	13 (1%)	13 (2%)	13 (1%)	13 (-1%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. The analysis was based on historical salvage data during 2009–2019 which did not include any Above Normal Water Years, hence that row of the table is noted as 'NA'; the percentage difference in Above Normal Water Years is based on density data for Wet Water Years applied to above normal modeled exports. Results by water year type and month are provided in Appendix 11Q.

CEQA Significance Determination for Alternatives 1, 2, and 3

Operations of Alternative 1, 2, or 3 would provide bypass flows to Stone Corral and Funks Creeks. These flows would be refined through studies required as part of the Project (Section 2.5.2.1, *Water Operations*) and described in Section 2D.4. These flows would support processes in these channels that support the fish (including native minnows) assemblage below the dams. Per the discussion above, native minnow spawning is not anticipated to be adversely affected due to operations of the Red Bluff and Hamilton City intakes for Alternatives 1, 2, and 3. Mean monthly temperatures by water year types between Alternatives 1, 2, and 3 and the NAA in the Sacramento, Feather, and American Rivers would be similar during the presence of each life stage of native minnows. Alternatives 1, 2, and 3 would have limited potential for negative effects on Yolo Bypass floodplain inundation and access for Sacramento splittail. Within the Delta, few hitch, Central California roach, or hardhead have been salvaged historically and so increases in exports during summer/fall under Alternatives 1, 2, and 3 would not result in appreciable additional salvage. For the reasons analyzed and discussed throughout Impact FISH-11, Alternative 1, 2, or 3 would not have a substantial adverse effect, either directly or through habitat modifications, on native minnows. Operations impacts of Alternative 1, 2, or 3 would be less than significant.

NEPA Conclusion for Alternatives 1, 2, and 3

Operations effects on native minnows would be the same as described above for CEQA. Operation of Alternative 1, 2, or 3 would have no adverse effect on native minnows.

Impact FISH-12: Operations Effects on Starry Flounder and Northern Anchovy

Alternatives 1, 2, and 3

Effects of Alternatives 1, 2, and 3

Within the Delta, Alternatives 1, 2, and 3 have the potential to affect starry flounder by increasing south Delta exports during summer/fall relative to the NAA. Starry flounder are entrained in relatively small numbers at the south Delta export facilities. The salvage-density method (Appendix 11Q) suggested entrainment risk under Alternatives 1, 2, and 3 could increase in drier years (Table 11-110 and 11-111) as the species temporally overlaps the period with greater exports under Alternatives 1, 2, and 3. Given the small numbers of starry flounder salvaged under the NAA, any increase in salvage would remain a small number of fish, particularly relative to the overall range of the species along the Pacific coast (Appendix 11A).

Table 11-110. Salvage of Starry Flounder At SWP Banks Pumping Plant, Averaged by Water Year Type, Based on the Salvage-Density Method.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	65	65 (0%)	65 (0%)	65 (0%)	65 (0%)
Above Normal	NA	(-1%)	(-1%)	(-1%)	(-2%)

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Below Normal	125	126 (1%)	125 (0%)	126 (1%)	128 (2%)
Dry	20	21 (8%)	21 (8%)	21 (8%)	21 (8%)
Critically Dry	13	15 (12%)	15 (10%)	15 (11%)	14 (9%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. The analysis was based on historical salvage data during 2009–2019 which did not include any Above Normal Water Years, hence that row of the table is noted as 'NA'; the percentage difference in Above Normal Water Years is based on density data for Wet Water Years applied to above normal modeled exports. Results by water year type and month are provided in Appendix 11Q. Percentage values are rounded; as a result, differences between percentages may not always appear consistent.

Table 11-111. Salvage of Starry Flounder At CVP Jones Pumping Plant, Averaged by Water Year Type, Based on the Salvage-Density Method.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	14	14 (0%)	14 (0%)	14 (0%)	14 (0%)
Above Normal	NA	(0%)	(0%)	(0%)	(2%)
Below Normal	26	26 (1%)	26 (1%)	26 (1%)	26 (3%)
Dry	14	14 (1%)	15 (5%)	14 (1%)	15 (8%)
Critically Dry	7	8 (1%)	8 (2%)	8 (0%)	7 (0%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. The analysis was based on historical salvage data during 2009–2019 which did not include any Above Normal Water Years, hence that row of the table is noted as 'NA'; the percentage difference in Above Normal Water Years is based on density data for Wet Water Years applied to above normal modeled exports. Results by water year type and month are provided in Appendix 11Q. Percentage values are rounded; as a result, differences between percentages may not always appear consistent.

Kimmerer et al. (2009) found a statistically significant negative relationship between annual mean March–June X2 (an index of Delta outflow) and annual mean starry flounder bay otter trawl abundance indices[38], which they suggested could be related to an increase in residual circulation in the San Francisco Estuary with increasing Delta outflow; if such an increase translates to more rapid or more complete entrainment of starry flounder early life stages into the estuary, or more rapid transport to their rearing grounds, then presumably, survival from hatching to settlement would be higher under high-flow conditions (Kimmerer et al. 2009:385). Note that this relationship only pertains to starry flounder within the Delta and does not consider the broad range of the species along the Pacific coast (Appendix 11A). A comparison of modeled bay otter trawl indices for Alternatives 1, 2, and 3 compared to the NAA as a function of modeled X2 was undertaken using the regression coefficients presented by Kimmerer et al. (2009:Table 2; also Appendix 11Q). This indicated that there would be limited difference in abundance indices expected between Alternatives 1, 2, and 3 and the NAA as a function of X2 (Table 11-112).

[38] An otter trawl is a type of bottom trawl particularly suited for sampling species occurring on or near a waterbody's substrate, such as starry flounder and California bay shrimp.

Table 11-112. Starry Flounder Bay Otter Trawl Index, Averaged by Water Year Type, as a Function of Mean March–June X2.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	201	200 (0%)	201 (0%)	200 (0%)	201 (0%)
Above Normal	155	155 (0%)	155 (0%)	155 (0%)	155 (0%)
Below Normal	104	103 (-1%)	103 (-1%)	103 (-1%)	103 (-1%)
Dry	71	71 (-1%)	71 (-1%)	71 (-1%)	71 (-1%)
Critically Dry	42	41 (0%)	41 (0%)	41 (0%)	41 (-1%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. Table only includes annual mean responses and does not consider model uncertainty. Percentage values are rounded; as a result, differences between percentages may not always appear consistent.

Northern anchovy generally occur well downstream of the Delta. Any potential changes in salinity as a result of Alternatives 1, 2, and 3 would be small relative to the salinity tolerance of northern anchovy (Baxter et al. 1999). Kimmerer et al. (2009) showed for northern anchovy that neither indices of abundance nor indices of habitat extent were related to X2, which is an index of Delta outflow and its effects. This observation, coupled with the small differences in salinity between Alternatives 1, 2, and 3 and the NAA, indicates that Alternatives 1, 2, and 3 would have minimal effects on northern anchovy.

CEQA Significance Determination for Alternatives 1, 2, and 3

Operations impacts of Alternatives 1, 2, and 3 on starry flounder and northern anchovy would be less than significant, as indicated by the small differences compared to the NAA in bay otter trawl abundance for starry flounder as a function of mean March–June X2, limited entrainment risk, and small effects on salinity relative to northern anchovy salinity tolerance. Operation of Alternative 1, 2, or 3 would not have a substantial adverse effect, either directly or through habitat modifications, on starry flounder and northern anchovy. Operations impacts for Alternative 1, 2, or 3 would be less than significant.

NEPA Conclusion for Alternatives 1, 2, and 3

Operations effects on starry flounder and northern anchovy would be the same as described above for CEQA. Operation of Alternative 1, 2, or 3 would have no adverse effect on starry flounder and northern anchovy.

Impact FISH-13: Operations Effects on Striped Bass

Alternatives 1, 2, and 3

Effects of Alternatives 1, 2, and 3

Sacramento River

Near-Field Effects

As described in Appendix 11A, striped bass spawning occurs in the Sacramento River from Colusa to Sacramento, which is downstream of the Red Bluff and Hamilton City intakes. Thus smaller life stages potentially vulnerable to entrainment or other near-field effects would not occur at the intakes.

Far-Field Effects

Temperature Effects

Operation of Sites Reservoir has the potential to change water temperatures in the Sacramento River that could affect the life stages of striped bass present. As described in Appendix 11B, the two methods used to analyze temperature-related effects on striped bass in the Sacramento River were: (1) Physical Model Output Characterization; and (2) Water Temperature Index Value/Range Exceedance Analysis. More details on these methods are provided in Appendix 11B.

The Authority and Reclamation evaluated water temperature model outputs during the period of presence and in the locations of each life stage of striped bass in the Sacramento River (see Appendix 11B, Table 11B-2 for timing and locations). Visual observation of exceedance plots and differences in modeled mean monthly temperatures by water year type between Alternatives 1, 2, and 3 and the NAA in the Sacramento River at Butte City indicates that water temperatures would be predominantly similar among alternatives during the period of presence of each life stage (Appendix 6C, Tables 6C-12-1a to 6C-12-4c; Figures 6C-12-1 to 6C-12-18). Mean monthly water temperatures for all months in all water year types under Alternatives 1A and 1B were within 0.8°F of the NAA. Water temperature modeling results for Alternatives 2 and 3 were similar to those of Alternative 1. These results suggest that temperature-related effects on striped bass in the Sacramento River would be negligible.

Results of the analysis of occurrence outside index ranges for striped bass in the Sacramento River at Butte City from Appendix 11B, Table 11B-2 are presented in Appendix 11D, Table 11D-292 and Table 11D-293. The percent of months with water temperatures outside each index range under Alternatives 1A and 1B would be similar to those under the NAA with some exceptions. For spawning, embryo incubation and initial rearing in April of Dry Water Years, there would be 11.1% fewer months with water temperatures outside the index range under Alternative 1B compared to the NAA. In June of Critically Dry Water Years, there would be 8.3% fewer months with water temperatures outside the index range under both Alternatives 1A and 1B than under the NAA. For larvae, fry, and juvenile rearing and emigration, in April of Dry Water Years, there would be 5.6% more months with water temperatures outside the index range under Alternative 1A relative to the NAA. There would be 8.3% more months with water temperatures outside the index range under Alternative 1B relative to the NAA. There would also be 9.1% fewer months with water temperatures outside the index range under both Alternatives 1A and 1B relative to the NAA in September of Above Normal Water Years. Results of the analysis of exceedance for Alternative 2 are similar to Alternatives 1A and 1B with some exceptions. There would be no difference in the occurrence of water temperatures outside the spawning, embryo incubation, and initial rearing index range under Alternative 2 relative to the NAA. There would be a 5.6% and 8.3% increase in occurrence outside the index range under Alternative 2 relative to the NAA in April of Dry Water Years and July of Critically Dry Water Years, respectively.

Results for Alternative 3 are similar to Alternative 1A and 1B with some differences. For spawning, embryo incubation and initial rearing, there would be 9.1% and 11.1% fewer months with water temperatures outside the index range in April of Above Normal Water Years and Dry

Water Years, respectively, under Alternative 3 compared to the NAA. There would also be 9.1% and 7.1% more months with water temperatures outside the index range in June of Above Normal Water Years and Below Normal Water Years, respectively, under Alternative 3 compared to the NAA. For the larvae, fry, and juvenile rearing and emigration index, there would be 7.7% and 5.6% fewer months with water temperatures outside the index range in April of Wet Water Years and Dry Water Years, respectively, under Alternative 3 compared to the NAA. There would also be 8.3% and 9.1% more months with water temperatures outside the index range in April of Critically Dry Water Years and September of Above Normal Water Years, respectively, under Alternative 3 compared to the NAA.

Due to low frequency of differences between each alternative and the NAA in occurrence outside the water temperature index ranges in the Sacramento River, they are not expected to be persistent enough to affect striped bass at a population level.

Flow Effects

Striped bass spawn in the Sacramento River primarily between about Verona (RM 78) and Wilkins Slough (RM 121) during April through June (Moyle 2002). No spawning occurs until water temperature reaches 57°F (Moyle 2002). The eggs are free-floating and negatively buoyant and hatch in about two days after spawning (at 66°F) as they drift downstream. Low flows can result in eggs settling on the bottom, which they cannot survive for long. The larvae may inhabit shallow, open water of the lower river from April to mid-June and then are carried by flows to the Delta and Suisun Bay (Stevens et al. 1987; Moyle 2002). Juvenile striped bass generally do not occur in the Sacramento River upstream of the Delta. Adult striped bass are found in the upper Sacramento River at RBPP and upstream, primarily from late spring through early fall, where they forage heavily on juvenile salmon and other fish (Tucker et al. 1998). Striped bass also spawn in the Feather and American Rivers, but little is known about their spawning behavior in these rivers.

High flows in April through June benefit striped bass eggs because they help prevent them from settling to the river bottom. High flows likely also accelerate transport of striped bass larvae to their nursery habitats in the Delta and Suisun Bay (Moyle 2002). CALSIM II flow results for Wilkins Slough indicate that mean monthly flow during April through June under Alternatives 1, 2, and 3 would be generally similar to or lower than flow under the NAA, including up to 4% to 6% lower flows during May of Critically Dry Water Years (Table 11-60). These reductions result in mean flows of about 4,200 cfs under Alternatives 1, 2, and 3. The reductions would potentially result in less favorable conditions for transport of striped bass eggs downstream, but this is uncertain because the actual level of flow in the river that negatively affects egg transport is not known. Substantial increases in flow occur in July through September of drier (Dry and Critically Dry) water years under Alternatives 1, 2, and 3, and also under Alternative 3 in October of all water year types. Adult striped bass may reside throughout the Sacramento River in summer and early fall (Tucker et al. 1998).

The CALSIM II flow results indicate that Alternatives 1, 2, and 3 would result in reductions in Sacramento River flow near Wilkins Slough during May of Critically Dry Water Years that could adversely affect survival of striped bass eggs drifting downstream from spawning locations. However, it is considered unlikely that this potential impact would affect the striped bass population for the following two reasons:

1. The CALSIM flow for Alternatives 1, 2, and 3 in May of Critically Dry Water Years was 10% or more below the NAA for a total of 3 months for Alternatives 1, 2, and 3, which constitutes 1% of all months (April–June) of the spawning period in the CALSIM record. Note that any reductions in the other months and water year types would likely have less effect because May of Critically Dry Water Years has the lowest flows (Table 11-60).
2. Analyses of the effects of X2 on juvenile striped bass indices of abundance in the Delta indicate that the striped bass indices under Alternatives 1, 2, and 3 differed little from those under the NAA (discussed in the *Delta* section below). This result indicates that the effects of X2, and indirectly of Delta outflow, on striped bass abundance did not differ substantially between Alternatives 1, 2, and 3 and the NAA. Delta outflow combines flow from many sources that affect the striped bass population and provides a good indication of overall flow effects of Alternatives 1, 2, and 3 on the striped bass population as a whole.

<u>Feather River</u>

Operation of Sites Reservoir has the potential to change water temperatures in the Feather River that could affect the life stages of striped bass present. As described in Appendix 11B, the two methods used to analyze temperature-related effects on striped bass in the Feather River were: (1) Physical Model Output Characterization; and (2) Water Temperature Index Value/Range Exceedance Analysis. More details on these methods are provided in Appendix 11B.

The Authority and Reclamation evaluated water temperature model outputs during the period of presence and in the locations of each life stage of striped bass present in the Feather River (see Appendix 11B, Table 11B-3 for timing and locations). Visual observation of exceedance plots and differences in modeled mean monthly temperatures by water year type between Alternatives 1, 2, and 3 and the NAA in the Feather River HFC at Gridley Bridge and at the mouth indicates that water temperatures would be predominantly similar among Alternatives 1, 2, and 3 during the period of presence of each life stage of striped bass (Appendix 6C, Tables 6C-18-1a to 6C-18-4c, Tables 6C-19-1a to 6C-19-4c; Figures 6C-18-1 to 6C-18-18, Figures 6C-19-1 to 6C-19-18). At both locations, mean monthly water temperatures for all months in all water year types under Alternatives 1A and 1B were within 0.5°F of the NAA. Water temperature modeling results for Alternatives 2 and 3 were similar to those for Alternative 1 at all locations. These results suggest that temperature-related effects on striped bass in the Feather River would be negligible.

Results of the analysis of occurrence outside the 59°F to 68°F spawning, egg incubation, and initial rearing water temperature index range for striped bass below Thermalito Afterbay outlet and at the mouth of the Feather River from Appendix 11B, Table 11B-3 are presented in Appendix 11D, Table 11D-294 and Table 11D-295. Below Thermalito, the percent of months outside the index range would be similar between the NAA and Alternatives 1A and 1B, except during May of Dry Water Years (8.3% higher under Alternatives 1A and 1B) and in June of Below Normal, Dry, and Critically Dry Water Years, in which occurrences of water temperatures outside the index range would be 7.1%, 11.1%, and 8.3% higher, respectively, under both

Alternative 1A and Alternative 1B than for the NAA. At the mouth of the Feather River, the percent of months outside the index range would be similar between the NAA and Alternatives 1A and 1B, except under Alternative 1B in May of Below Normal Water Years, in which the occurrences of water temperatures outside the index range would be 7.1% under Alternative 1B relative to the NAA. Results for Alternatives 2 and 3 below Thermalito would be similar to those of Alternative 1A and 1B with one exception. Results for Alternatives 2 and 3 at the mouth of the Feather River would be similar to those of Alternative 1B.

Results of the analysis of occurrence outside the 61°F to 71°F larvae, fry, and juvenile rearing and emigration index range for striped bass below Thermalito Afterbay outlet and at the mouth of the Feather River from Appendix 11B, Table 11B-3 are presented in Appendix 11D, Table 11D-296 and Table 11D-297. The percent of months outside the index range would be largely similar between the NAA and Alternatives 1A and 1B, with some exceptions. At the Feather River mouth, there would be 8.3% and 5.6% more months with water temperatures outside the index range in June of Critically Dry Water Years and October of Dry Water Years, respectively, under both Alternative 1A and 1B compared to the NAA. Also, there would be 9.1% more months water temperatures outside the index range in September of Above Normal Water Years and October of Dry Water Years, respectively, under both Alternative 1A and 1B compared to the NAA. Alternative 2 and 3 results would be largely similar to those of Alternative 1B with few exceptions.

Due to low frequency and magnitude of differences between each alternative and the NAA in occurrence outside the water temperature index ranges in the Feather River, they are not expected to be persistent enough to affect striped bass at a population level.

American River

Operation of Sites Reservoir has the potential to change water temperatures in the American River that could affect the life stages of striped bass present. As described in Appendix 11B, the two methods used to analyze temperature-related effects on striped bass in the American River were: (1) Physical Model Output Characterization; and (2) Water Temperature Index Value/Range Exceedance Analysis. More details on these methods are provided in Appendix 11B.

The Authority and Reclamation evaluated water temperature model outputs during the period of presence and in the locations of each life stage of striped bass in the American River (see Appendix 11B, Table 11B-4 for timing and locations). Visual observation of exceedance plots and differences in modeled mean monthly temperatures by water year type between alternatives and the NAA in the American River below Nimbus Dam, Watt Avenue, and at the mouth indicates that water temperatures would be predominantly similar among alternatives during the period of presence of each life stage of striped bass (Appendix 6C, Tables 6C-13-1a to 6C-13-4c, Tables 6C-14-1a to 6C-14-4c, Tables 6C-15-1a to 6C-15-4c; Figures 6C-13-1 to 6C-13-18, Figures 6C-14-1 to 6C-14-18, Figures 6C-15-1 to 6C-15-18). At all locations, mean monthly water temperatures for all months within all water year types under Alternatives 1A and 1B were within 0.3°F of the NAA. Water temperature modeling results for Alternatives 2 and 3 were similar to those of Alternative 1 at all locations. These results suggest that temperature-related effects on striped bass in the American River would be negligible.

Results of the analysis of occurrence outside the water temperature index ranges for striped bass at Watt Avenue and at the mouth of the American River from Appendix 11B, Table 11B-4 are presented in Appendix 11D, Table 11D-298 and Table 11D-301. At both locations, the percent of months outside the 59°F to 68°F spawning, egg incubation, and initial rearing index range would be similar between the NAA and Alternatives 1, 2, and 3. At both locations, the percent of months outside the 61°F to 71°F larvae, fry, and juvenile rearing and emigration index range would be largely similar between the NAA and Alternatives 1A and 1B, except at Watt Avenue, where there would be5.1% and 5.4% reductions in occurrence of water temperatures outside the index range under Alternative 1A relative to the NAA in Critically Dry Water Years during July and December, respectively. Results for Alternatives 2 and 3 at both locations would be similar to those for Alternative 1B, except for a 5.3% reduction in occurrence of water temperatures outside the index range in April of Critically Dry Water Years under Alternative 3 relative to the NAA.

Due to low frequency and magnitude of differences between each alternative and the NAA in occurrence outside the water temperature index values in the American River, they are not expected to be persistent enough to affect striped bass at a population level.

Delta

Entrainment

As described in Appendix 11A, striped bass are vulnerable to entrainment at the south Delta export facilities. The spawning and egg/larval downstream movement of striped bass occurs in spring, during which time south Delta exports that could result in entrainment risk would be similar between Alternatives 1, 2, and 3 and the NAA (Appendix 5B4, Tables 5B4-1-1a through 5B4-1-4c; Figures 5B4-1-1 through 5B4-1-18). Juvenile striped bass entrainment and salvage occurs throughout the year, including the summer/fall period when south Delta exports would differ most between Alternatives 1, 2, and 3 and the NAA. The salvage-density method (Appendix 11Q) gave somewhat greater estimated salvage under Alternatives 1, 2, and 3, particularly in Critically Dry Water Years at SWP (Tables 11-113 and 11-114). As noted in Appendix 11A, available studies suggest that even considerable levels of historical estimated population-level entrainment (33%–99% of the population) did not give discernible population-level effects. This indicates that the differences in entrainment risk suggested by the salvage-density method would not be expected to give differing population-level effects between Alternatives 1, 2, and 3 and the NAA.

Table 11-113. Salvage of Striped Bass At SWP Banks Pumping Plant, Averaged by Water Year Type, Based on the Salvage-Density Method.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	311,948	312,470 (0%)	313,121 (0%)	312,455 (0%)	313,029 (0%)
Above Normal	NA	(0%)	(0%)	(0%)	(-2%)
Below Normal	345,368	355,241 (3%)	354,091 (3%)	354,956 (3%)	355,025 (3%)
Dry	124,975	128,787 (3%)	124,176 (-1%)	128,692 (3%)	123,386 (-1%)
Critically Dry	82,549	88,347 (7%)	87,976 (7%)	87,986 (7%)	87,326 (6%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. The analysis was based on historical salvage data during 2009–2019 which did not include any Above Normal Water Years, hence that row of the table is noted as 'NA'; the percentage difference in Above Normal Water Years is based on density data for Wet Water Years applied to above normal modeled exports. Results by water year type and month are provided in Appendix 11Q.

Table 11-114. Salvage of Striped Bass At CVP Jones Pumping Plant, Averaged by Water Year Type, Based on the Salvage-Density Method.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	62,978	62,937 (0%)	62,932 (0%)	62,934 (0%)	62,930 (0%)
Above Normal	NA	(0%)	(0%)	(0%)	(3%)
Below Normal	98,828	99,331 (1%)	99,922 (1%)	99,415 (1%)	102,293 (4%)
Dry	172,783	173,679 (1%)	175,085 (1%)	173,706 (1%)	179,207 (4%)
Critically Dry	60,034	60,095 (0%)	59,887 (0%)	60,143 (0%)	59,317 (-1%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. The analysis was based on historical salvage data during 2009–2019 which did not include any Above Normal Water Years, hence that row of the table is noted as 'NA'; the percentage difference in Above Normal Water Years is based on density data for Wet Water Years applied to above normal modeled exports. Results by water year type and month are provided in Appendix 11Q.

Flow-Related Effects

Kimmerer et al. (2009) found several statistically significant negative relationships between annual mean April–June X2 and various indices of striped bass juvenile abundance or survival. Application of these relationships to Alternatives 1, 2, and 3 and the NAA indicated minimal differences between Alternatives 1, 2, and 3 and the NAA (Tables 11-115, 11-116, 11-117, 11-118). This reflects limited differences in mean X2 between Alternatives 1, 2, and 3 and the NAA during this period.

Juvenile striped bass abundance indices are also negatively related to fall (September–December) X2 (Mac Nally et al. 2010). Under Alternatives 1, 2, and 3, mean fall X2 would be similar or somewhat lower than the NAA (Table 11-119), so there would not be negative effects on striped bass from Alternatives 1, 2, and 3 relative to the NAA conditions as a result of differences in fall X2.

Table 11-115. Striped Bass Summer Townet Abundance Index, Averaged by Water Year Type, as a Function of Mean April–June X2.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	1.5	1.5 (0%)	1.5 (0%)	1.5 (0%)	1.5 (0%)
Above Normal	1.2	1.2 (0%)	1.2 (0%)	1.2 (0%)	1.2 (0%)
Below Normal	1.0	1.0 (0%)	1.0 (0%)	1.0 (0%)	1.0 (0%)
Dry	0.8	0.8 (0%)	0.7 (0%)	0.8 (0%)	0.8 (0%)
Critically Dry	0.5	0.5 (0%)	0.5 (0%)	0.5 (0%)	0.5 (0%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. Table only includes annual mean responses and does not consider model uncertainty.

Table 11-116. Striped Bass Fall Midwater Trawl Abundance Index, Averaged by Water Year Type, as a Function of Mean April–June X2.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	346	346 (0%)	346 (0%)	346 (0%)	346 (0%)
Above Normal	308	308 (0%)	308 (0%)	308 (0%)	308 (0%)
Below Normal	268	268 (0%)	268 (0%)	268 (0%)	268 (0%)
Dry	230	230 (0%)	230 (0%)	230 (0%)	230 (0%)
Critically Dry	192	192 (0%)	192 (0%)	192 (0%)	192 (0%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. Table only includes annual mean responses and does not consider model uncertainty.

Table 11-117. Striped Bass Bay Midwater Trawl Abundance Index, Averaged by Water Year Type, as a Function of Mean April–June X2.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	1,825	1,824 (0%)	1,827 (0%)	1,824 (0%)	1,825 (0%)
Above Normal	1,360	1,363 (0%)	1,364 (0%)	1,363 (0%)	1,362 (0%)
Below Normal	975	972 (0%)	972 (0%)	972 (0%)	971 (0%)
Dry	662	662 (0%)	661 (0%)	662 (0%)	661 (0%)
Critically Dry	423	423 (0%)	422 (0%)	423 (0%)	422 (0%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. Table only includes annual mean responses and does not consider model uncertainty. Percentage values are rounded; as a result, differences between percentages may not always appear consistent.

Table 11-118. Striped Bass Bay Otter Trawl Abundance Index, Averaged by Water Year Type, as a Function of Mean April–June X2.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	3,243	3,241 (0%)	3,245 (0%)	3,242 (0%)	3,243 (0%)
Above Normal	2,735	2,739 (0%)	2,740 (0%)	2,739 (0%)	2,738 (0%)
Below Normal	2,240	2,237 (0%)	2,237 (0%)	2,237 (0%)	2,235 (0%)
Dry	1,791	1,791 (0%)	1,788 (0%)	1,791 (0%)	1,789 (0%)
Critically Dry	1,375	1,376 (0%)	1,375 (0%)	1,376 (0%)	1,374 (0%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. Table only includes annual mean responses and does not consider model uncertainty.

Table 11-119. Mean Fall (September–December) X2, Averaged by Water Year Type.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	77.4	77.2	77.3	77.2	77.3
Above Normal	79.8	79.6	79.5	79.6	79.3
Below Normal	85.1	85.1	85.1	85.1	84.3
Dry	88.0	87.6	87.5	87.6	87.6
Critically Dry	91.3	91.1	91.1	91.1	91.2

The State Water Board Bay-Delta water quality control plan includes an April–May electrical conductivity objective for the San Joaquin River between Jersey Point and Prisoners Point for striped bass spawning water quality. DSM2-QUAL modeling for Alternatives 1, 2, and 3 indicates that compliance with this objective would be the same as under the NAA.

CEQA Significance Determination for Alternatives 1, 2, and 3

Striped bass spawning occurs in the Sacramento River from Colusa to Sacramento, which is downstream of the Red Bluff and Hamilton City intakes. Smaller life stages potentially vulnerable to entrainment or other near-field effects would not occur at the intakes.

Temperatures by water year type between Alternatives 1, 2, and 3 and the NAA in the Sacramento, Feather, and American Rivers indicates that water temperatures would be predominantly similar among Alternatives 1, 2, and 3 during the period of presence of each life stage of striped bass.

While CALSIM II flow results indicate that Alternatives 1, 2, and 3 would result in reductions in Sacramento River flow near Wilkins Slough during May of Critically Dry Water Years, it is considered unlikely that this potential effect would affect the striped bass population. The analysis indicates limited differences in mean X2 between Alternatives 1, 2, and 3. Thus, there would not be negative effects on striped bass from Alternatives 1, 2, and 3 relative to the NAA as a result of differences in fall X2.

Additionally, any differences in entrainment risk suggested by the salvage-density method would not be expected to give differing population-level effects between Alternatives 1, 2, and 3 and the NAA.

Based on the above analysis in Impact FISH-13, operation of Alternative 1, 2, or 3 would not have a substantial adverse effect, either directly or through habitat modifications, on striped bass. Operations impacts of Alternative 1, 2, or 3 would be less than significant.

NEPA Conclusion for Alternatives 1, 2, and 3

Operations effects on striped bass would be the same as described above for CEQA. The operation effects of Alternative 1, 2, or 3 on striped bass as compared to the NAA are anticipated to be minimal and would not have an adverse effect, either directly or through habitat modifications. When compared to the NAA, there would be limited differences in mean fall X2, water temperature would be similar during the period of presence of each life stage, and there would be no expected differences in entrainment risk resulting in population-level effects. Operation of Alternative 1, 2, or 3 would have no adverse effect on striped bass.

Impact FISH-14: Operations Effects on American Shad

Alternatives 1, 2, and 3

Effects of Alternatives 1, 2, and 3

Sacramento River

Near-Field Effects

As described in Appendix 11A, the main spawning areas for American shad are the Sacramento, Feather, American, and Yuba Rivers. Based on the nursery areas of juveniles (Stevens et al. 1987), spawning in the Sacramento River seems to be primarily from Colusa downstream and downstream of the Red Bluff and Hamilton City intakes. As such, the smaller American shad life stages potentially vulnerable to entrainment or other near-field effects would not be expected to occur at the intakes in substantial numbers, and given that the main spawning period is May–July, there would be little difference in diversion between Alternatives 1, 2, and 3 and the NAA (Tables 11-6 and 11-7 in the winter-run Chinook salmon analysis).

Far-Field Effects

Temperature Effects

Operation of Sites Reservoir has the potential to change water temperatures in the Sacramento River that could affect the life stages of American shad present. As described in Appendix 11B, the two methods used to analyze temperature-related effects on American shad in the Sacramento River were: (1) Physical Model Output Characterization; and (2) Water Temperature Index Value/Range Exceedance Analysis. More details on these methods are provided in Appendix 11B.

The Authority and Reclamation evaluated water temperature model outputs during the period of presence and in the locations of each life stage of American shad in the Sacramento River (see Appendix 11B, Table 11B-2 for timing and locations). Visual observation of exceedance plots and differences in modeled mean monthly temperatures by water year type between Alternatives 1, 2, and 3 and the NAA in the Sacramento River below RBPP and Butte City indicates that water temperatures would be predominantly similar among Alternatives 1, 2, and 3 during the period of presence of each life stage of American shad (Appendix 6C, Tables 6C-10-1a to 6C-10-4c, Tables 6C-12-1a to 6C-12-4c; Figures 6C-10-1 to 6C-10-18, Figures 6C-12-1 to 6C-12-18). At both sites, mean monthly water temperatures for all months within all water year types under Alternatives 1A and 1B were within 0.6°F of the NAA with the exception of August of critical years at Butte City, in which mean monthly water temperature would be 0.8°F lower under the NAA than under Alternatives 1A and 1B. Water temperature modeling results for Alternatives 2 and 3 were similar to those of Alternative 1. These results suggest that temperature-related effects on American shad in the Sacramento River would be negligible.

Results of the analysis of occurrence outside index ranges for American shad in the Sacramento River from Appendix 11B, Table 11B-2 are presented in Appendix 11D, Table 11D-302 through Table 11D-305. For the 60°F to 70°F spawning, embryo incubation, and initial rearing life stages, the percent of days below RBPP and months at Butte City with water temperatures outside each index range under Alternatives 1A and 1B would be similar to those under the NAA with some exceptions of up to 16.7% reductions in the occurrence of water temperatures outside the index range under Alternatives 1A and 1B. For the 63°F to 77°F larvae, fry, and juvenile rearing and emigration life stages, there would be minimal differences between Alternatives 1A and 1B compared to the NAA below RBPP. At Butte City, occurrence of water temperatures outside the index range would be predominantly similar between Alternatives 1A and 1B compared to the NAA, with some exceptions ranging from 5.6% more and 14.3% fewer occurrences under Alternatives 1A and 1B compared to the NAA. Alternatives 2 and 3 results would be similar to those under Alternative 1, although Alternative 3 had more reductions in occurrence outside the index range at Butte City during the larvae, fry, and juvenile rearing and emigration period.

Due to low frequency and magnitude of differences between each alternative and the NAA in exceedances above water temperature index values in the Sacramento River, they are not expected to be persistent enough to affect American shad at a population level.

Flow Effects

American shad migrate upstream in the Sacramento River starting in March, and typically spawn from April to June. They also spawn in the lower Feather River and, to a lesser degree, in the American River (Appendix 11A). Shad eggs settle to the river bottom or drift downstream from spawning areas and hatch in about 2.5 days at 77°F to about 8.5 days at 59°F (Marschall et al. 2020). Larval shad are planktonic for about 4 weeks, after which they metamorphose to actively swimming juveniles. Juveniles spend the next several months in fresh water. In the Sacramento River, summer rearing habitat occurs in the main river from Colusa to the north Delta (Stevens et al. 1987). As the season progresses, juvenile shad move downstream and enter salt water primarily during September through November (Moyle 2002). In general, variations in river discharge and temperature during early larval development are considered important regulators of year-class strength and recruitment of American shad (Hinrichsen et al. 2013; Marschall et al. 2020). Adequate flow is needed to disperse eggs and larvae, exposing them to a range of habitat conditions (Marschall et al. 2020). Although the importance of various potential mechanisms is unknown, the abundance of juvenile American shad in the Delta has been shown to be positively correlated with freshwater inflow during the April through June spawning and nursery periods (Stevens et al. 1987, Kimmerer 2002b, Kimmerer et al. 2009).

During the spawning and larval rearing period (April through June), CALSIM II modeling results at Wilkins Slough indicate that mean monthly flow during April through June under Alternatives 1, 2, and 3 would be generally similar to or lower than flow under the NAA, with the largest flow reductions between 5% and 6% during April and May of Critically Dry Water Years and June of Above Normal Water Years (Table 11-60). The reductions during May of Critically Dry Water Years result in mean flows of about 4,200 cfs under Alternatives 1, 2, and 3. The reductions would potentially result in less favorable conditions for transport of American shad eggs and larvae downstream, but this effect is uncertain because the actual level of flow in the Sacramento River that negatively affects egg and larval transport is not known. Larger reductions (>10%) in flow occur in the Feather River at the Thermalito Afterbay in June of Dry and Critically Dry Water Years (Table 11-63). Substantial increases in flow occur in July through September of drier water years (Dry and Critically Dry Water Years) at Wilkins Slough under Alternatives 1, 2, and 3 (Table 11-60). The importance of river flows for juvenile shad in July through October is unknown.

1. The CALSIM II flow results indicate that Alternatives 1, 2, and 3 would result in reductions in Sacramento River flow near Wilkins Slough during May of Critically Dry Water Years that could adversely affect survival of American shad eggs and larvae drifting downstream from spawning locations. However, it is considered unlikely that this potential impact would affect the American shad population for the following two reasons: The CALSIM flow for Alternatives 1, 2, and 3 in May of Critically Dry Water Years was 10% or more below the NAA a total of 3 months for each alternative, which constitutes 1% of all months (April–June) of the spawning period in the CALSIM record. Note that any reductions in the other months and water year types would likely have less effect because May of Critically Dry Water Years has the lowest flows (Table 11-60).

2. Analyses of the effects of X2 on juvenile American shad indices of abundance in the Delta indicate that the American shad indices under Alternatives 1, 2, and 3 differed little from those under the NAA (discussed in the *Delta* section below). The position of X2 is strongly related to Delta outflow. This result indicates that effects of X2, and indirectly of Delta outflow, on American shad abundance differed little between Alternatives 1, 2, and 3 and the NAA. Delta outflow combines flow from a number of sources that affect the American shad population, and therefore provide a good indication of overall flow effects of Alternatives 1, 2, and 3 on the American shad population as a whole.

Feather River

Operation of Sites Reservoir has the potential to change water temperatures in the Feather River that could affect the life stages of American shad present. As described in Appendix 11B, the two methods used to analyze temperature-related effects on American shad in the Feather River were: (1) Physical Model Output Characterization; and (2) Water Temperature Index Value/Range Exceedance Analysis. More details on these methods are provided in Appendix 11B.

The Authority and Reclamation evaluated water temperature model outputs during the period of presence and in the locations of each life stage of American shad in the Feather River (see Appendix 11B, Table 11B-3 for timing and locations). Visual observation of exceedance plots and differences in modeled mean monthly temperatures by water year type between Alternatives 1, 2, and 3 and the NAA in the Feather River HFC at Gridley Bridge and at the mouth indicates that water temperatures would be predominantly similar among Alternatives 1, 2, and 3 during the period of presence of each life stage of American shad (Appendix 6C, Tables 6C-18-1a to 6C-18-4c, Tables 6C-19-1a to 6C-19-4c; Figures 6C-18-1 to 6C-18-18, Figures 6C-19-1 to 6C-19-18). At both locations, mean monthly water temperatures for all months in all water year types under Alternatives 1A and 1B were within 0.5°F of the NAA. Water temperature modeling results for Alternatives 2 and 3 were similar to those of Alternative 1 at all locations. These results suggest that temperature-related effects on American shad in the Feather River would be negligible.

Results of the analysis of occurrence outside the water temperature index ranges for American shad below Thermalito Afterbay outlet and at the mouth of the Feather River from Appendix 11B, Table 11B-3 are presented in Appendix 11D, Table 11D-306 through Table 11D-309. At both locations, the percent of months outside the 60°F to 70°F spawning, egg incubation, and initial rearing index range would be similar between the NAA and Alternatives 1A and 1B, except below Thermalito Afterbay outlet during June of Dry Water Years when the occurrence of water temperatures outside the range would be 5.6% lower under Alternatives 1A and 1B relative to the NAA and during June of Above Normal Water Years when the occurrence of water temperatures outside the range would be 9.1% lower under Alternatives 1A and 1B relative to the NAA. At both locations, the percent of months outside the 63°F to 77°F larvae, fry, and juvenile rearing and emigration index range would be similar between the NAA and Alternatives 1A and 1B with some exceptions. Below Thermalito Afterbay outlet during September of Below Normal and Dry Water Years, there would be 7.1% and 5.6% fewer months, respectively, in which water temperatures are outside the index range. At the mouth of the Feather River in July of Below Normal Water Years and in October of Below Normal and Dry Water Years, there would be between 5.6% and 7.1% more months in which water temperatures are outside the index range. Results for Alternatives 2 and 3 would be predominantly similar to those of Alternative 1.

Due to low frequency and magnitude of differences between each alternative and the NAA in exceedances above water temperature index values in the Feather River, they are not expected to be persistent enough to affect American shad at a population level.

American River

Operation of Sites Reservoir has the potential to change water temperatures in the American River that could affect the life stages of American shad. As described in Appendix 11B, the two methods used to analyze temperature-related effects on American shad in the American River were: (1) Physical Model Output Characterization; and (2) Water Temperature Index Value/Range Exceedance Analysis. More details on these methods are provided in Appendix 11B.

The Authority and Reclamation evaluated water temperature model outputs during the period of presence and in the locations of each life stage of American shad in the American River (see Appendix 11B, Table 11B-4 for timing and locations). Visual observation of exceedance plots and differences in modeled mean monthly temperatures by water year type between Alternatives 1, 2, and 3 and the NAA in the American River below Nimbus Dam, at Watt Avenue, and at the mouth indicates that water temperatures would be predominantly similar among Alternatives 1, 2, and 3 during the period of presence of each life stage of American shad (Appendix 6C, Tables 6C-13-1a to 6C-13-4c, Tables 6C-14-1a to 6C-14-4c, Tables 6C-15-1a to 6C-15-4c; Figures 6C-13-1 to 6C-13-18, Figures 6C-14-1 to 6C-14-18, Figures 6C-15-1 to 6C-15-18). At all locations, mean monthly water temperatures for all months within all water year types under Alternatives 1A and 1B were within 0.3°F of the NAA. Water temperature modeling results for Alternatives 2 and 3 were similar to those of Alternative 1 at all locations. These results suggest that temperature-related effects on American shad in the American River would be negligible.

Results of the analysis of occurrence outside the water temperature index ranges for American shad at Watt Avenue and at the mouth of the American River from Appendix 11B, Table 11B-4 are presented in Appendix 11D, Table 11D-310 through Table 11D-313. At both locations, the percent of months outside the two index ranges would be similar between the NAA and Alternatives 1A and 1B. Results for Alternative 2 would be similar to those of Alternative 1. Results for Alternative 3 would be predominantly similar to those of Alternative 1, except that water temperatures in April of Critically Dry Water Years would be outside the 63°F to 77°F larvae, fry, and juvenile rearing and emigration index range 5.6% less often under Alternative 3 relative to the NAA.

Due to low frequency and magnitude of differences between each alternative and the NAA in exceedances above water temperature index values in the American River, they are not expected to be persistent enough to affect American shad at a population level.

Delta

Entrainment

Some of the highest densities of American shad observed historically in south Delta export facility salvage samples have occurred during July and August, so application of the salvage-density method (Appendix 11Q) suggested the potential for greater salvage under Alternatives 1, 2, and 3 than the NAA at the SWP facility (Table 11-120) as a result of greater exports under Alternatives 1, 2, and 3 during this period; there was little difference in salvage at the CVP facility (Table 11-121). Although it has been suggested that declines observed in American shad populations since the 1970s were related to diversions in the Delta, as well as upstream (Appendix 11A), there has not been a population-level examination of the influence of south Delta exports on the population, and statistical analyses have focused instead on the influence of Delta outflow/X2 (discussed in *Flow-Related Effects* below). Despite the salvage-density method suggesting the potential for greater entrainment as a result of greater summer exports under Alternatives 1, 2, and 3, the overall density of American shad in the south Delta is low relative to other areas occupied by the species, including the north Delta, Sacramento River from Colusa to Sacramento, and Feather River below the Yuba River (Stevens et al. 1987). This suggests that increased entrainment potential under Alternatives 1, 2, and 3 relative to the NAA would not have population-level consequences for American shad.

Table 11-120. Salvage of American Shad At SWP Banks Pumping Plant, Averaged by Water Year Type, Based on the Salvage-Density Method.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	318,771	319,175 (0%)	320,043 (0%)	319,168 (0%)	319,788 (0%)
Above Normal	NA	(0%)	(0%)	(0%)	(-2%)
Below Normal	270,845	279,207 (3%)	279,080 (3%)	279,069 (3%)	275,794 (2%)
Dry	129,338	144,989 (12%)	138,612 (7%)	144,482 (12%)	137,247 (6%)
Critically Dry	82,598	98,717 (20%)	97,960 (19%)	97,038 (17%)	94,773 (15%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. The analysis was based on historical salvage data during 2009–2019 which did not include any Above Normal Water Years, hence that row of the table is noted as ‘NA’; the percentage difference in Above Normal Water Years is based on density data for Wet Water Years applied to above normal modeled exports. Results by water year type and month are provided in

Appendix 11Q.

Table 11-121. Salvage of American Shad At CVP Jones Pumping Plant, Averaged by Water Year Type, Based on the Salvage-Density Method.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	243,321	243,207 (0%)	243,164 (0%)	243,203 (0%)	243,399 (0%)
Above Normal	NA	(0%)	(0%)	(0%)	(2%)
Below Normal	60,139	61,001 (1%)	60,627 (1%)	61,091 (2%)	63,011 (5%)
Dry	72,828	73,622 (1%)	72,340 (-1%)	73,649 (1%)	74,146 (2%)
Critically Dry	3,433	3,498 (2%)	3,492 (2%)	3,478 (1%)	3,498 (2%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. The analysis was based on historical salvage data during 2009–2019 which did not include any Above Normal Water Years, hence that row of the table is noted as 'NA'; the percentage difference in Above Normal Water Years is based on density data for Wet Water Years applied to above normal modeled exports. Results by water year type and month are provided in Appendix 11Q.

Flow-Related Effects

Kimmerer et al. (2009) found statistically significant negative relationships between annual mean February–May X2 and indices of American shad juvenile abundance from fall midwater trawl and bay midwater trawl sampling. Application of these relationships to Alternatives 1, 2, and 3 and the NAA indicated minimal differences between Alternatives 1, 2, and 3 and the NAA (Tables 11-122, 11-123; see methods description in Appendix 11Q). This reflects limited differences in mean X2 between Alternatives 1, 2, and 3 and the NAA during this period.

Table 11-122. American Shad Fall Midwater Trawl Abundance Index, Averaged by Water Year Type, as a Function of Mean February–May X2.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	2,991	2,989 (0%)	2,990 (0%)	2,989 (0%)	2,989 (0%)
Above Normal	2,792	2,785 (0%)	2,785 (0%)	2,785 (0%)	2,784 (0%)
Below Normal	2,341	2,333 (0%)	2,332 (0%)	2,333 (0%)	2,328 (-1%)
Dry	1,942	1,932 (-1%)	1,931 (-1%)	1,932 (-1%)	1,932 (0%)
Critically Dry	1,517	1,513 (0%)	1,512 (0%)	1,512 (0%)	1,513 (0%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. Table only includes annual mean responses and does not consider model uncertainty.

Table 11-123. American Shad Bay Midwater Trawl Abundance Index, Averaged by Water Year Type, as a Function of Mean February–May X2.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	7,662	7,653 (0%)	7,659 (0%)	7,653 (0%)	7,654 (0%)
Above Normal	6,960	6,939 (0%)	6,940 (0%)	6,939 (0%)	6,935 (0%)
Below Normal	5,460	5,435 (0%)	5,430 (-1%)	5,435 (0%)	5,420 (-1%)
Dry	4,226	4,194 (-1%)	4,193 (-1%)	4,194 (-1%)	4,197 (-1%)

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Critically Dry	2,996	2,985 (0%)	2,984 (0%)	2,984 (0%)	2,985 (0%)

CEQA Significance Determination for Alternatives 1, 2, and 3

American shad spawning in the Sacramento River is downstream of the Red Bluff and Hamilton City intakes. Exceedance plots and differences in modeled mean monthly temperatures by water year type between Alternatives 1, 2, and 3 and the NAA in the Sacramento, Feather, and American Rivers indicates that water temperatures would be predominantly similar among Alternatives 1, 2, and 3 during the period of presence of each life stage of American shad. It is unlikely that flow associated with Alternatives 1, 2, and 3 compared to the NAA would have any potential effect on American shad. Any increased entrainment potential under Alternatives 1, 2, and 3 relative to the NAA would not have population-level consequences. Related to flow, the data indicate minimal differences between the Alternatives 1, 2, and 3 and the NAA. Operation of Alternative 1, 2, or 3, would not have a substantial adverse effect, either directly or through habitat modifications, on American shad. The operational impacts of Alternatives 1, 2, and 3 on American shad would be less than significant.

NEPA Conclusion for Alternatives 1, 2, and 3

Operations effects on American shad would be the same as described above for CEQA. The operation effects of Alternative 1, 2, or 3 on American shad as compared to the NAA are anticipated to be minimal and would not have an adverse effect, either directly or through habitat modifications. When compared to the NAA, there would be similarities in water temperature during the period of presence of each life stage and the population-level effects from changes in entrainment risk would be small. Operation of Alternative 1, 2, or 3 would have no adverse effect on American shad.

Impact FISH-15: Operations Effects on Threadfin Shad

Alternatives 1, 2, and 3

Effects of Alternatives 1, 2, and 3

Increases in south Delta exports during the summer under Alternatives 1, 2, and 3 have the potential to increase threadfin shad entrainment at the south Delta export facilities. This is illustrated by the results of the salvage-density method (Appendix 11Q), which showed appreciable increases in SWP exports weighted by historical salvage density during Dry and Critically Dry Water Years (Table 11-124), in contrast to little difference in CVP exports (Table 11-125). To examine this potential increase in entrainment in more detail, consideration was given to the differences in entrainment risk in the main region of the Delta occupied by threadfin shad (i.e., the San Joaquin River in the southeast Delta) (Feyrer et al. 2009). As described further in Appendix 11Q, relationships between proportional entrainment of particles and E:I ratio developed from particle tracking modeling by Kimmerer and Nobriga (2008) were used to assess differences in entrainment risk during the months of June–November for particles released in the San Joaquin River at Medford Island, Potato Slough, and Stockton. This modeling found that there was little difference in particle proportional entrainment between Alternatives 1, 2, and 3 and the NAA except in July of Critically Dry Water Years, for which the mean proportion of particles entrained under Alternatives 1, 2, and 3 was 13%–18% greater than under the NAA. The extent to which this difference represents a potential increase in actual entrainment risk is uncertain because passive particles may not be representative of threadfin shad. The population-level importance of changes in exports has only been examined for spring and fall, with some evidence supporting exports as having a negative effect on population trends (Mac Nally et al. 2010; Thomson et al. 2010). Thomson et al. (2010:1444) suggested that threadfin shad may be especially vulnerable to exports throughout the year because they occupy freshwater throughout the year (Appendix 11A) but noted that proportional loss estimates have not been made for the species.

Table 11-124. Salvage of Threadfin Shad At SWP Banks Pumping Plant, Averaged by Water Year Type, Based on the Salvage-Density Method.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	455,895	456,651 (0%)	458,370 (1%)	456,627 (0%)	457,872 (0%)
Above Normal	NA	(0%)	(0%)	(0%)	(-2%)
Below Normal	1,493,498	1,514,692 (1%)	1,512,551 (1%)	1,514,027 (1%)	1,503,483 (1%)
Dry	1,116,651	1,350,309 (21%)	1,351,449 (21%)	1,347,652 (21%)	1,338,766 (20%)
Critically Dry	337,034	576,274 (71%)	565,370 (68%)	557,968 (66%)	542,742 (61%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. The analysis was based on historical salvage data during 2009–2019 which did not include any Above Normal Water Years, hence that row of the table is noted as 'NA'; the percentage difference in Above Normal Water Years is based on density data for Wet Water Years applied to above normal modeled exports. Results by water year type and month are provided in Appendix 11Q.

Table 11-125. Salvage of Threadfin Shad At CVP Jones Pumping Plant, Averaged by Water Year Type, Based on the Salvage-Density Method.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	740,390	740,983 (0%)	740,847 (0%)	740,982 (0%)	741,600 (0%)
Above Normal	NA	(0%)	(0%)	(0%)	(2%)
Below Normal	892,697	906,765 (2%)	898,899 (1%)	907,235 (2%)	922,140 (3%)
Dry	2,475,778	2,533,588 (2%)	2,537,256 (2%)	2,542,495 (3%)	2,692,596 (9%)
Critically Dry	219,138	222,281 (1%)	220,997 (1%)	222,461 (2%)	219,795 (0%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. The analysis was based on historical salvage data during 2009–2019 which did not include any Above Normal Water Years, hence that row of the table is noted as 'NA'; the percentage difference in Above Normal Water Years is based on density data for Wet Water Years applied to above normal modeled exports. Results by water year type and month are provided in Appendix 11Q.

CEQA Significance Determination for Alternatives 1, 2, and 3

The entrainment analyses indicate the potential for appreciably greater south Delta entrainment of threadfin shad in July of Critically Dry Water Years under Alternatives 1, 2, and 3 compared to the NAA. However, greater summer entrainment appears unlikely to have population-level consequences because abundance in the fall (when most of the commercial harvest in the Delta occurs; Feyrer et al. 2009) is poorly related to abundance in summer, potentially as a result of factors such as toxicity of *Microcystis* blooms (Acuña et al. 2012a, 2020) being more important

(Feyrer et al. 2009; Baxter et al. 2010). The lack of relationship between summer and fall abundance of threadfin shad indicates that the operation of Alternative 1, 2, or 3, would not have a substantial adverse effect, either directly or through habitat modifications, on threadfin shad. Operations impacts of Alternative 1, 2, or 3 would be less than significant.

NEPA Conclusion for Alternatives 1, 2, and 3

Operations effects on threadfin shad would be the same as described above for CEQA. Entrainment analyses indicate the potential for appreciably greater south Delta entrainment of threadfin shad in July of Critically Dry Water Years under Alternatives 1, 2, and 3 compared to the NAA. However, greater summer entrainment appears unlikely to have population-level consequences because of the lack of correlation between the species' summer and fall abundance. Operation of Alternative 1, 2, or 3 would have no adverse effect on threadfin shad.

Impact FISH-16: Operations Effects on Black Bass (Largemouth Bass, Smallmouth Bass, and Spotted Bass)

Alternatives 1, 2, and 3

Effects of Alternatives 1, 2, and 3

Sacramento River

Near-Field Effects

Although black bass would be susceptible to near-field effects such as entrainment or impingement at the Red Bluff and Hamilton City intakes, population-level effects would be expected to be minimal because the smallest life stages would tend to occur during spring/early summer (Appendix 11A) when there would be little difference in diversions between Alternatives 1, 2, and 3 and the NAA (Tables 11-6 and 11-7 in the winter-run Chinook salmon analysis). In addition, the species are widespread in the Central Valley (and particularly in the Delta) without specific migratory patterns (e.g., those of anadromous fish) that would cause them to systematically move past the intakes.

Far-Field Effects

Temperature Effects

Operation of Sites Reservoir has the potential to change water temperatures in the Sacramento River that could affect the life stages of black basses present. As described in Appendix 11B, the two methods used to analyze temperature-related effects on black basses in the Sacramento River, using largemouth bass to represent all black basses due to similar life cycles and biological requirements, were: (1) Physical Model Output Characterization; and (2) Water Temperature Index Value/Range Exceedance Analysis. More details on these methods are provided in Appendix 11B.

The Authority and Reclamation evaluated water temperature model outputs during the period of presence and in the locations of each life stage of largemouth bass in the Sacramento River (see Appendix 11B, Table 11B-2 for timing and locations). Visual observation of exceedance plots and differences in modeled mean monthly temperatures by water year type between Alternatives

1, 2, and 3 and the NAA in the Sacramento River below Keswick and below RBPP indicates that water temperatures would be predominantly similar among Alternatives 1, 2, and 3 during the period of presence of each life stage of largemouth bass (Appendix 6C, Tables 6C-5-1a to 6C-5-4c, Tables 6C-10-1a to 6C-10-4c; Figures 6C-5-1 to 6C-5-18, Figures 6C-10-1 to 6C-10-18). At both sites, mean monthly water temperatures for all months in all water year types under Alternatives 1A and 1B were within 0.6°F of the NAA. Water temperature modeling results for Alternatives 2 and 3 were similar to those of Alternative 1. These results suggest that temperature-related effects on black bass in the Sacramento River would be negligible.

Results of the analysis of occurrence outside the 54°F to 75°F spawning index range for largemouth bass in the Sacramento River (Appendix 11B, Table 11B-2) are presented in Appendix 11D, Table 11D-314 and Table 11D-315. At both locations, the percent of days with water temperatures outside each index range under Alternatives 1A and 1B would be similar to those under the NAA. The results for Alternatives 2 and 3 would be similar to those for Alternative 1. These results indicate that thermal effects of each alternative on black basses in the Sacramento River would be negligible.

Flow Effects

The main channel of the Sacramento, Feather, and American Rivers upstream of the Delta generally provide poor habitat conditions for largemouth bass because of large seasonal flow fluctuations, relatively cold water, and lack of suitable nesting and rearing habitat. Largemouth bass populations upstream of the Delta are largely dependent on off-stream habitats, including reservoirs, side-channel and backwater ponds and sloughs, and irrigation canals that provide suitable conditions for spawning and rearing during the late spring and summer months. Smallmouth bass are better adapted to the more rapid flows and cooler water temperatures of the rivers upstream of the Delta and are abundant in the Sacramento River.

All three black bass species spawn and rear in the spring and summer (Moyle 2002). The males construct and guard nest depressions. In streams, nesting and reproduction can be disrupted by flow reductions that lead to nest dewatering or elevated flows that wash embryos and fry out of nests or lower water temperatures excessively (Graham and Orth 1986; Lukas and Orth 1995). Prior to damming of the Central Valley rivers for flood and agricultural storage, the annual hydrologic cycle included much more variability, with higher winter, spring, and early summer flow and lower late summer and fall flow (Bureau of Reclamation 2019). These conditions are believed to have favored native fish. The reduction of this natural variability by dams and their associated water project operations are believed to have contributed to the decline of the native fish because they contributed to the invasion of exotic species such as the black basses, which outcompete the native species under more stable flow conditions (Brown and Moyle 2005). The Project, by diverting Sacramento River water during winter and spring (to storage in Shasta Lake and Sites Reservoir) and releasing more water in the summer and fall (Table 11-57 through Table 11-60), potentially furthers the reduction in seasonal variability of flow that has favored exotic species such as the black basses. Similar effects on seasonal flow are evident in the results for the Feather River (Table 11-63). However, the reductions in seasonal flow variability expected under Alternatives 1, 2, and 3 are not expected to be large enough to affect these species.

Feather River

Operation of Sites Reservoir has the potential to change water temperatures in the Feather River that could affect the life stages of black basses present. As described in Appendix 11B, the two methods used to analyze temperature-related effects on black basses, using largemouth bass to represent all black basses due to similar life cycles and biological requirements, were: (1) Physical Model Output Characterization; and (2) Water Temperature Index Value/Range Exceedance Analysis. More details on these methods are provided in Appendix 11B.

The Authority and Reclamation evaluated water temperature model outputs during the period of presence and in the locations of each life stage of largemouth bass in the Feather River (see Appendix 11B, Table 11B-3 for timing and locations). Visual observation of exceedance plots and differences in modeled mean monthly temperatures by water year type between Alternatives 1, 2, and 3 and the NAA in the Feather River HFC at Gridley Bridge and at the mouth indicates that water temperatures would be predominantly similar among Alternatives 1, 2, and 3 during the spawning period of presence of each life stage of largemouth bass (Appendix 6C, Tables 6C-18-1a to 6C-18-4c, Tables 6C-19-1a to 6C-19-4c; Figures 6C-18-1 to 6C-18-18, Figures 6C-19-1 to 6C-19-18). At both locations, mean monthly water temperatures for all months in all water year types under Alternatives 1A and 1B were within 0.5°F of the NAA. Water temperature modeling results for Alternatives 2 and 3 were similar to those of Alternative 1 at all locations. These results suggest that temperature-related effects on black bass in the Feather River would be negligible.

Results of the analysis of occurrence outside the 54°F to 75°F spawning index range for largemouth bass below Thermalito Afterbay outlet and at the mouth of the Feather River (Appendix 11B, Table 11B-3) are presented in Appendix 11D, Table 11D-316 and Table 11D-317. At both locations, the percent of months outside the spawning index range would be similar between the NAA and Alternatives 1A and 1B with some exceptions. Below Thermalito Afterbay outlet, the occurrence of water temperatures outside the index range would be 5.6% to 9.1% lower under Alternatives 1A and 1B relative to the NAA in March of Above Normal Water Years and Below Normal Water Years and in April of Dry Water Years. In addition, water temperatures would be outside the index range in 5.6% more months under Alternatives 1A and 1B than the NAA in June of Dry Water Years. Results for Alternatives 2 and 3 would be similar to those of Alternative 1 at both locations.

Due to low frequency and magnitude of differences between each alternative and the NAA in occurrence outside the water temperature index ranges in the Feather River, they are not expected to be persistent enough to affect black basses at a population level.

American River

Operation of Sites Reservoir has the potential to change water temperatures in the American River that could affect the life stages of black basses present. As described in Appendix 11B, the two methods used to analyze temperature-related effects on black basses, using largemouth bass to represent all black basses due to similar life cycles and biological requirements, were: (1) Physical Model Output Characterization; and (2) Water Temperature Index Value/Range Exceedance Analysis. More details on these methods are provided in Appendix 11B.

The Authority and Reclamation evaluated water temperature model outputs during the period of presence and in the locations of each life stage of largemouth bass in the American River (see Appendix 11B, Table 11B-4 for timing and locations). Visual observation of exceedance plots and differences in modeled mean monthly temperatures by water year type between Alternatives 1, 2, and 3 and the NAA in the American River at Watt Avenue indicates that water temperatures would be predominantly similar among Alternatives 1, 2, and 3 during the period of presence of largemouth bass (Appendix 6C, Tables 6C-14-1a to 6C-14-4c; Figures 6C-14-1 to 6C-14-18). At all locations, mean monthly water temperatures for all months within all water year types under Alternatives 1A and 1B were within 0.3°F of the NAA. Water temperature modeling results for Alternatives 2 and 3 were similar to those of Alternative 1 at all locations. These results suggest that temperature-related effects on black basses in the American River would be negligible.

Results of the analysis of occurrence outside the 54°F to 75°F spawning index range for largemouth bass in the American River at Watt Avenue (Appendix 11B, Table 11B-4) are presented in Appendix 11D, Table 11D-318. The percent of months outside the spawning index range would be similar between the NAA and Alternatives 1A and 1B. Results for Alternatives 2 and 3 would be similar to those of Alternative 1. These results suggest that temperature-related effects on black basses in the American River would be negligible.

<u>**Delta**</u>

Differences in south Delta exports between Alternatives 1, 2, and 3 and the NAA could change south Delta entrainment risk for black bass. Historical salvage data show few smallmouth or spotted bass are entrained, whereas largemouth bass are entrained in relatively high numbers. The seasonality of largemouth bass salvage at the south Delta export facilities results in highest salvage occurring during May–July, which overlaps the period during which exports would be greater under Alternatives 1, 2, and 3, in particular at the SWP facility in drier years as illustrated by the salvage-density method (Table 11-126; Table 11-127; see Appendix 11Q for description of salvage-density method). However, the salvage-density method is solely a calculation of differences in south Delta exports between Alternatives 1, 2, and 3 weighted by historical density of observed fish in salvage and is not a prediction of actual salvage expected. Analyses by Grimaldo et al. (2009) did not find a significant relationship between largemouth bass salvage and Old and Middle River flows, an important indicator of entrainment risk for other species such as delta smelt and longfin smelt. Grimaldo et al. (2009) suggested that the littoral (nearshore) habitat occupied by the species probably provides a buffer from entrainment. As such, the differences in entrainment risk suggested by the salvage-density method are likely to be small. This observation, combined with the widespread occurrence of largemouth bass (e.g., Conrad et al. 2016; Mahardja et al. 2017), indicates population-level effects from changes in entrainment risk as a result of Alternatives 1, 2, and 3 would be small.

Table 11-126. Salvage of Largemouth Bass At SWP Banks Pumping Plant, Averaged by Water Year Type, Based on the Salvage-Density Method.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	18,955	18,990 (0%)	19,037 (0%)	18,990 (0%)	19,015 (0%)
Above Normal	NA	(0%)	(0%)	(0%)	(-3%)
Below Normal	16,159	16,279 (1%)	16,242 (1%)	16,282 (1%)	16,273 (1%)

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Dry	7,612	8,769 (15%)	8,779 (15%)	8,750 (15%)	8,646 (14%)
Critically Dry	3,024	4,091 (35%)	4,023 (33%)	4,023 (33%)	3,964 (31%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. The analysis was based on historical salvage data during 2009–2019 which did not include any Above Normal Water Years, hence that row of the table is noted as 'NA'; the percentage difference in Above Normal Water Years is based on density data for Wet Water Years applied to above normal modeled exports. Results by water year type and month are provided in Appendix 11Q.

Table 11-127. Salvage of Largemouth Bass At CVP Jones Pumping Plant, Averaged by Water Year Type, Based on the Salvage-Density Method.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	50,330	50,329 (0%)	50,344 (0%)	50,327 (0%)	50,374 (0%)
Above Normal	NA	(0%)	(0%)	(0%)	(3%)
Below Normal	65,628	65,976 (1%)	66,492 (1%)	66,000 (1%)	68,172 (4%)
Dry	69,940	70,396 (1%)	70,927 (1%)	70,445 (1%)	72,708 (4%)
Critically Dry	42,103	42,243 (0%)	42,009 (0%)	42,267 (0%)	41,427 (-2%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. The analysis was based on historical salvage data during 2009–2019 which did not include any Above Normal Water Years, hence that row of the table is noted as 'NA'; the percentage difference in Above Normal Water Years is based on density data for Wet Water Years applied to above normal modeled exports. Results by water year type and month are provided in Appendix 11Q.

CEQA Significance Determination for Alternatives 1, 2, and 3

Black bass are anticipated to have minimal population-level effects related to entrainments at the Red Bluff and Hamilton City intakes. Exceedance plots and differences in modeled mean monthly temperatures by water year type between Alternatives 1, 2, and 3 and the NAA in the Sacramento, Feather, and American Rivers indicates that water temperatures would be predominantly similar among Alternatives 1, 2, and 3 during the period of presence of each life stage of black bass. All three black bass species are adaptable, so it is highly unlikely that the relatively small differences in flow between Alternatives 1, 2, and 3 and the NAA would have more than minimal effects on the black bass populations of the rivers upstream of the Delta. Population-level effects from changes in entrainment risk as a result of Alternatives 1, 2, and 3 would be small. Operation of Alternative 1, 2, or 3, would not have a substantial adverse effect, either directly or through habitat modifications, on black bass. Operations impacts of Alternative 1, 2, or 3 would be less than significant.

NEPA Conclusion for Alternatives 1, 2, and 3

Operations effects on black bass would be the same as described above for CEQA. The operation effects of Alternative 1, 2, or 3 on black bass populations as compared to the NAA are anticipated to be minimal and would not have an adverse effect, either directly or through habitat modifications. When compared to the NAA, there would be similarities in water temperature during the period of presence of each life stage and the population-level effects from changes in entrainment risk would be small. In addition, the species are adaptable to different conditions. Operation of Alternative 1, 2, or 3 would have no adverse effect on black bass.

Impact FISH-17: Operations Effects on California Bay Shrimp

Alternatives 1, 2, and 3

Effects of Alternatives 1, 2, and 3

Kimmerer et al. (2009) found a statistically significant negative relationship between annual mean April–June X2 and the California bay shrimp bay otter trawl abundance index. Application of this relationship to Alternatives 1, 2, and 3 and the NAA indicated minimal differences between Alternatives 1, 2, and 3 and the NAA (Table 11-128; see methods description in Appendix 11Q). This reflects limited differences in mean X2 between Alternatives 1, 2, and 3 and the NAA during this period.

Table 11-128. California Bay Shrimp Bay Otter Trawl Abundance Index, Averaged by Water Year Type, as a Function of Mean April–June X2.

Water Year Type	NAA	Alt 1A	Alt 1B	Alt 2	Alt 3
Wet	365	364 (0%)	365 (0%)	364 (0%)	364 (0%)
Above Normal	328	328 (0%)	328 (0%)	328 (0%)	327 (0%)
Below Normal	246	245 (0%)	245 (0%)	245 (0%)	244 (-1%)
Dry	190	189 (-1%)	189 (-1%)	189 (-1%)	189 (-1%)
Critically Dry	128	127 (0%)	127 (-1%)	127 (-1%)	127 (-1%)

Note: Percentage values in parentheses indicate differences of alternatives compared to the NAA. Table only includes annual mean responses and does not consider model uncertainty. Percentage values are rounded; as a result, differences between percentages may not always appear consistent.

CEQA Significance Determination for Alternatives 1, 2, and 3

Operation of Alternatives 1, 2, and 3, would not have a substantial adverse effect, either directly or through habitat modifications, on California bay shrimp as indicated by the small differences in bay otter trawl abundance as a function of mean April–June X2. Operations impacts of Alternative 1, 2, or 3 would be less than significant.

NEPA Conclusion for Alternatives 1, 2, and 3

Operations effects on California bay shrimp would be the same as described above for CEQA. There were small differences in the bay otter trawl abundance index for California bay shrimp as a function of mean April–June X2 for operation of Alternative 1, 2, or 3 as compared to the NAA. Operation of Alternative 1, 2, or 3 would have no adverse effect on California bay shrimp.

Impact FISH-18: Operations Effects on Reservoir Fish Species

Reservoir fish species are not State or federal special-status species but are evaluated for their recreational importance. Populations of some of these species are artificially augmented or sustained through periodic fish stocking programs.

A detailed description of the methods and results of this analysis, including a summary of changes in aquatic habitat conditions for cold-water and warm-water reservoir fish species in Lake Oroville, Folsom Lake, San Luis Reservoir, and Shasta Lake resulting from implementation of Alternatives 1, 2, and 3 relative to the NAA, is presented in Appendix 11E.

This analysis does not include reservoir fish species in Trinity Reservoir because implementation of Alternatives 1, 2, and 3 would result in negligible differences in storage volume and WSE in this reservoir. Further, aside from the aforementioned reservoirs, no other CVP or SWP reservoirs (e.g., Millerton and New Melones Reservoirs) would be affected by the alternatives.

It is important to note that storage volume and elevations of the reservoirs listed above have experienced high variation within the year under historical operations. Therefore, this analysis focuses on the comparison of model outputs for each alternative relative to the NAA.

No Project

Under the NAA, the operations of Shasta Lake, Lake Oroville, Folsom Lake, San Luis Reservoir, New Melones Reservoir, and Millerton Lake would continue and there would be no change compared to 2020 baseline conditions for cold-water and warm-water fish species at these reservoirs. There would be no changes in the reservoir storage volumes and WSE of those reservoirs because the Sites Reservoir would not be built and operated.

Significance Determination

The NAA would not result in operations effects on reservoir fish species because there would be no measurable change from 2020 baseline conditions. There would be no impact/no effect.

Alternative 1, 2, and 3

Effects of Alternative 1, 2, and 3

As described in Appendix 11E, Alternatives 1, 2, and 3 would provide a benefit to cold-water and warm-water reservoir fish species relative to the NAA because Sites Reservoir, and the new habitat it would create, would not exist under the NAA (Table 11E-1).

Storage volume in Shasta Lake, Folsom Lake, and Lake Oroville under Alternatives 1, 2, and 3 would generally be similar to or greater than storage under the NAA (Appendix 11E, Table 11E-2 through Table 11E-4). Shasta Lake storage would be consistently >5% higher under Alternative 3 in critical years between June and September, representing a beneficial effect on cold-water reservoir species.

Reservoir warm-water reservoir fish species habitat conditions in Shasta Lake, Lake Oroville, Folsom Lake, and San Luis Reservoir generally would be similar or slightly more suitable under Alternatives 1, 2, and 3 relative to the NAA. This is based on modeling results indicating minor differences in the frequency of monthly WSE reductions of 6 feet or more during the evaluation period (Appendix 11E, Table 11E-5 through Table 11E-13).

<u>*CEQA Significance Determination for Alternatives 1, 2, and 3*</u>

The analyses of potential impacts of Alternatives 1, 2, and 3 on cold-water reservoir fish species suggested that construction of the Sites Reservoir would be beneficial through the provision of new habitat. The creation of Sites Reservoir under Alternatives 1, 2, and 3 would provide new habitat for reservoir warm-water reservoir fish species.

The analyses of potential operational impacts of Alternatives 1, 2, and 3 on reservoir fish species in Shasta Lake, Lake Oroville, Folsom Lake, and San Luis Reservoir suggested that changes in the Sites Reservoir storage and WSE would be similar to changes in these reservoirs.

For these reasons, operation of Alternative 1, 2, or 3, would not have a substantial adverse effect, either directly or through habitat modifications, on warm-water and cold-water reservoir fish species. Operations impacts of Alternative 1, 2, or 3 would be less than significant.

<u>*NEPA Conclusion and Mitigation Measures for Alternatives 1, 2, and 3*</u>

Operations effects on reservoir fish species would be the same as described above for CEQA. The analyses of potential operation effects from changes in storage and WSE on reservoir fish species in Shasta Lake, Lake Oroville, Folsom Lake, and San Luis Reservoir suggested that the corresponding effects in Sites Reservoir would be similar as compared to the NAA. In addition, the creation of Sites Reservoir under Alternatives 1, 2, and 3 would provide new habitat for reservoir warm-water reservoir fish species as compared to the NAA. The operation of Alternative 1, 2, or 3 would have a beneficial effect on reservoir fish species.

Impact FISH-19: Operations Effects on Southern Resident Killer Whale

Alternatives 1, 2, and 3

<u>*Effects of Alternatives 1, 2, and 3*</u>

Southern Resident killer whale is found in coastal waters off British Columbia, Washington, and Oregon in summer and fall (National Marine Fisheries Service 2008). During winter, killer whales are sometimes found off the Central California coast, but are more frequently reported off the Washington coast (Hilborn et al. 2012). The 2005 NMFS endangered listing (70 FR 17386) for the Southern Resident killer whale DPS lists several factors that may be limiting the recovery of killer whales (including the quantity and quality of prey).

Project operations would not directly affect ocean conditions; however, operations have the potential to affect killer whales indirectly by influencing the number of Chinook salmon that enter the Pacific Ocean and become available as a food supply. This potential impact was evaluated qualitatively based on the potential impacts to Chinook salmon, particularly any changes in production. Alternatives 1, 2, and 3 have the potential to affect Southern Resident killer whale by altering the number of Chinook salmon from the Central Valley that enter the Pacific Ocean. Chinook salmon is an important component of the killer whale diet, and the Independent Science Panel reported that Southern Resident killer whales depend on Chinook salmon as a critical food resource (Hilborn et al. 2012). Hanson et al. (2010) analyzed tissues from predation events and feces to confirm that Chinook salmon were the most frequent prey item for killer whales in two regions of the whale’s summer range off the coast of British

Columbia and Washington state, representing more than 90% of the diet in July and August. Samples indicated that when Southern Resident killer whales are in inland waters from May to September, they consume Chinook salmon stocks that originate from regions that include the Fraser River, Puget Sound, the Central British Columbia Coast, West and East Vancouver Island, and the Central Valley of California (Hanson et al. 2010). Available fish harvest data and killer whale diet and contaminants analyses suggest that Central Valley Chinook salmon make up a significant portion of the total abundance of Chinook salmon available to killer whale throughout its range in most, if not all, years (National Marine Fisheries Service 2019a).

Significant changes in food availability for killer whales have occurred over the past 150 years, largely because of human impacts on prey species. Salmon abundance has been reduced over the entire range of the Southern Resident killer whale, from British Columbia to California. The Recovery Plan for Southern Resident killer whale (National Marine Fisheries Service 2008) indicates that wild salmon have declined primarily because of degraded aquatic ecosystems, overharvesting, and production of fish in hatcheries. The recovery plan supports restoration efforts to rebuild depleted salmon populations and other prey to ensure an adequate food base for Southern Resident killer whales. Central Valley streams produce Chinook salmon that contribute to the diet of Southern Resident killer whales. The number of Central Valley salmon that annually enter the ocean and survive to a size susceptible to predation by killer whales is not known. However, estimates of total Chinook salmon production produced by the Comprehensive Assessment and Monitoring Program, administered by USFWS and Reclamation, provide an approximation of the size of the ocean population of Central Valley Chinook salmon potentially available to killer whales.

Data on the abundance and composition of Central Valley Chinook salmon indicates that approximately 75% of all Central Valley-origin Chinook salmon available for consumption by Southern Resident killer whales are produced by Central Valley fall-run Chinook salmon hatcheries (Palmer-Zwhalen and Kormos 2013). Most Central Valley hatchery fall-run Chinook salmon are released directly into San Francisco Bay, and thus bypass potential impacts from project operations. Even where there might be a nexus with CVP and SWP operations. The purpose of Central Valley fall-run Chinook salmon hatchery programs is to produce large numbers of fish independent of freshwater conditions.

Since fall-run Chinook salmon hatcheries began operating more than 40 years ago, the only period of exceptionally low returns was principally attributed to unusual ocean conditions (Lindley et al. 2007). Ocean commercial and recreational fisheries annually harvest hundreds of thousands of Chinook salmon. The Northwest Region of NMFS used a model that estimates prey reduction associated with the salmon fishery and which considers the metabolic requirements of killer whales and the remaining levels of prey availability (National Marine Fisheries Service 2009). Their analysis concluded that the salmon fishery was not likely to result in jeopardy for Southern Resident killer whale.

CEQA Significance Determination for Alternatives 1, 2, and 3

Given conclusions and discussions from NMFS (2009, 2019a), and that at least 75% of fall-run Chinook salmon available for Southern Resident killer whale are produced by Central Valley hatcheries, it is likely that Central Valley fall-run Chinook salmon as a prey base for killer whales would not be appreciably affected by the operations of Alternatives 1, 2, or 3. Operation of Alternative 1, 2, or 3, would not have a substantial adverse effect, either directly or through habitat modifications, on Southern Resident killer whale. Operations impacts of Alternative 1, 2, or 3 would be less than significant.

NEPA Conclusion for Alternatives 1, 2, and 3

Operations effects on Southern Resident killer whale would be the same as described above for CEQA. Based on information from NMFS (2009, 2019a) and the proportion of the existing prey base for Southern Resident killer whale that is produced by Central Valley hatcheries, the operation of Alternative 1, 2, or 3 would not appreciably affect Southern Resident killer whale as compared to the NAA. Operation of Alternative 1, 2, or 3 would have no adverse effect on Southern Resident killer whale.

Impact FISH-20: Maintenance Effects on Fish and Aquatic Biological Resources

The assessment of impacts from maintenance activities is based on a qualitative evaluation for the facilities included under Alternatives 1, 2, and 3 and focuses on maintenance activities that are near waterways and could affect fish and other aquatic biological resources. Electrical transmission connections and lines, substations, distribution lines, dam monitoring equipment, and buildings (i.e., administration and operations, maintenance and storage) are not included in the assessment because these facilities would be located away from waterways and would not affect fish and aquatic resources.

No Project

Under the NAA, no new facilities and infrastructure that would require maintenance would be constructed and operated. Maintenance activities would continue at existing facilities such as the GCID Main Canal, RBPP, and Hamilton City Pump Station. For example, GCID typically dewaters its Main Canal for up to 6 weeks each year between early January and late February for maintenance. GCID and TCCA have established operations and maintenance plans that would be followed and have been issued regulatory permits/approvals.

Significance Determination

Under the NAA, no new facilities would be constructed and there would be no new maintenance activities with the potential to affect fish and other aquatic resources. There would be no impact/no effect.

Alternatives 1, 2, and 3

Maintenance activities for new facilities, including recreation areas, that would be constructed and operated under Alternatives 1, 2, and 3 would include debris removal, vegetation control, rodent control, erosion control and protection, routine inspections (e.g., of dams, tunnels, pipelines, PGPs, I/O Works, fencing, signs, and gates), painting, cleaning, repairs, and other routine tasks to maintain the facilities in accordance with design standards after construction and commissioning. Routine visual inspection of the facilities would be conducted to monitor performance and prevent mechanical and structural failures.

The Authority will implement BMP-12, BMP-13, and BMP-30 to avoid and minimize potential water quality impacts potentially associated with facility operations and maintenance. These BMPs would avoid and minimize potential water quality effects by preventing spills and reducing runoff that may cause sediment or contaminants to flow into waterbodies. The limited extent of possible water quality effects associated with facility maintenance combined with the implementation of these BMPs would prevent facility operation and maintenance activities from causing substantial degradation of water quality.

Effects of Alternatives 1, 2, and 3

Sediment Disturbance

Maintenance activities associated with Alternatives 1, 2, and 3 have the potential to cause erosion, sediment, and soil disturbance. These activities result in sediment transport and delivery to streams. Sediment entering streams could temporarily increase water column turbidity and sedimentation rates above ambient levels and potentially alter fish physiology, behavior, and habitat conditions.

Maintenance activities that have the potential to result in erosion and sediment transport and delivery to streams include: (1) debris removal at fish screens; (2) vegetation maintenance activities for land around facilities that involve grading, tilling, disking, or controlled burns would occur on an as-needed basis and could affect wetlands or non-wetland waters if they are present in the vegetation maintenance areas; (3) erosion control in and around new pipeline inlets/outlets and dams; and (4) road and bridge maintenance. Under Alternative 2, sediment- and turbidity-producing activities would include maintenance of the Sacramento River discharge, which may entail vegetation removal or other maintenance activities. Maintenance activities that occur in or immediately adjacent to stream channels (e.g., vegetation removal, road and bridge work) or during the wet season have the greatest potential to disturb stream sediments or cause erosion and contribute sediment to waterways. Maintenance activities near the Sacramento River can affect special-status fish species such as Chinook salmon, steelhead, green sturgeon, Pacific lamprey, and others as listed in Table 11-2. Maintenance activities in and around the reservoirs could affect black bass species (Table 11-2).

As discussed under Impact FISH-1, elevated levels of suspended sediments have the potential to result in physiological, behavioral, and habitat effects on fish. The severity of these effects depends on the sediment concentration, duration of exposure, proximity of the action to the waterbody, and timing of the disturbance relative to the occurrence of the species and sensitive life stages. Short-term increases in turbidity and suspended sediment may disrupt normal behavior patterns of fish, potentially affecting foraging, rearing, and migration. The level of disturbance may also cause juvenile fish to abandon protective habitat or reduce their ability to detect predators, potentially increasing their vulnerability to predators (e.g., piscivorous birds and fish). Chronic exposure to high turbidity and suspended sediment may affect fish growth and survival by impairing respiratory function, reducing tolerance to disease and contaminants, and causing physiological stress (Waters 1995). Deposition of excessive fine sediment on the stream bottom could eliminate habitat for aquatic insects; reduce density, biomass, number, and diversity of aquatic insects and vegetation; reduce the quality and quantity of spawning habitat; and block the interchange of surface and subsurface waters.

The maintenance footprint for Alternatives 1, 2, and 3 includes areas with known and potentially contaminated sediments (e.g., metals, hydrocarbons such as oil and grease, organochlorine pesticides, and PCBs), indicating the potential for release and dispersal of these contaminants if these sediments are disturbed during maintenance activities. Fish and aquatic species could be directly exposed to elevated levels of contaminants if they are in immediate proximity to maintenance activities that disturb contaminated sediments. Bed disturbance could also result in indirect effects on fish and aquatic species. Toxins in river channel sediments can enter the food chain via benthic organisms. If contaminated sediments are disturbed and become suspended in the water column, they also become available directly to pelagic organisms, including fish species and planktonic food sources of fish species. Thus, maintenance-related disturbance of contaminated bottom sediments creates another potential pathway to the food chain, and the potential bioaccumulation of these toxins in various fish species. The bioaccumulation of toxins can lead to lethal effects, as well as sublethal effects (e.g., effects on behavior, digestion, and immune system) (Connon et al. 2011:290). The toxins in contaminated sediments are adhered to the sediment and as described above for turbidity, elevated suspended sediment because of maintenance activity for Alternatives 1, 2, and 3 would be spatially limited to a small portion of channel width and not extend far downstream, dissipating within hours of maintenance activities ceasing.

The Project is subject to a maintenance-related stormwater permit and dewatering requirements of the federal Clean Water Act and National Pollutant Discharge Elimination System program. The Project operators would obtain required permits through the Central Valley Regional Water Quality Control Board before any ground-disturbing construction activity occurs. The SWPPP would include a long-term maintenance plan that will require erosion and sedimentation control measures as part of maintenance activities to prevent erosion and sedimentation off-site from entering local waterbodies that could occur from ground disturbance. These effects would be of limited duration and intensity. The Project would also limit maintenance activities to potential in-water work windows established by CDFW, NMFS, and USFWS. As a result, there would be minimal changes to surface water quality from maintenance activities that would result in increased or contaminated stormwater runoff or violations of water quality standards that would negatively affect fish populations and habitat.

<u>**Water Quality Effects**</u>

Maintenance activities for Alternatives 1, 2, and 3 could result in accidental spills of contaminants, including cement, oil, fuel, hydraulic fluids, paint, and other maintenance-related materials, resulting in localized water quality degradation. This could in turn result in adverse effects on fish and aquatic species, through direct injury and mortality (e.g., damage to gill tissue, causing asphyxiation) or delayed effects on growth and survival (e.g., increased stress or reduced feeding), depending on nature and extent of the spill and the contaminants involved.

The greatest potential for an adverse water quality impact is associated with an accidental spill from maintenance activities occurring in or near surface waters. Maintenance of pumps include cleaning and lubricating with oil products and heavy equipment use near waterways which could release fuel or hydraulic fluid. Special-status fish species located in waterways near pumps include Chinook salmon, steelhead, and green sturgeon (Table 11-2).

The Authority will implement BMP-13 and BMP-30 for maintenance activities for Alternatives 1 and 3 to avoid and minimize permanent and temporary impacts on aquatic species and habitat. These BMPs would limit direct impacts on aquatic species and habitat because they would provide storage, use, or transfer of hazardous materials guidance to be consistent with regulatory agencies requirements. The Land Management Plan (Appendix 2D, Section 2D.7, *Land Management Plan*) will require a qualified biologist to provide annual training to maintenance personnel on general measures and practices to be employed during maintenance activities to protect aquatic species and habitat.

Reduced Prey Availability

Maintenance activities for Alternatives 1, 2, and 3 have the potential to reduce prey availability for fish and aquatic species through disturbance of aquatic habitat. Prey species may be affected by debris or other vegetation removal (i.e., reducing habitat structures for prey in or above water during clearing). The potential effects would be limited in extent relative to the overall area of habitat available to fish and aquatic species in the affected waterways.

CEQA Significance Determination for Alternatives 1, 2, and 3

BMP-12, BMP-13, and BMP-30 that would be implemented for maintenance include measures to ensure sedimentation and contaminant releases are controlled by minimizing soil disturbance to prevent the alteration of fish physiology, behavior, habitat conditions, and reduce the potential for direct injury or mortality to fish by developing plans to and procedures to avoid and minimize sedimentation and contaminants from entering aquatic habitat. Maintenance activities for Alternative 1, 2, or 3 would not have a substantial adverse effect, either directly or through habitat modifications, on special-status fish species or interfere substantially with the movement of any native resident or migratory fish species or with established native resident or migratory wildlife corridors or impede the use of native wildlife nursery sites. Maintenance impacts for Alternative 1, 2, or 3 would be less than significant.

NEPA Conclusion for Alternatives 1, 2, and 3

Maintenance effects on fish and aquatic biological resources would be the same as described above for CEQA. Maintenance activities for Alternative 1, 2, or 3 that are near waterways would involve BMPs to avoid and minimize effects on fish and aquatic biological resources. These BMPs would include preventing spills and reducing runoff that may cause sediment or contaminants to flow into waterbodies and affect water quality as compared to the NAA. The limited extent of possible water quality effects associated with facility maintenance combined with the implementation of these BMPs would prevent maintenance activities from substantially affecting fish and aquatic resources. Maintenance activities for Alternative 1, 2, or 3 would have no adverse effect on fish and other aquatic biological resources.

11.5 References

11.5.1. Printed References

Acierto, K. R., J. Israel, J. Ferreira, and J. Roberts. 2014. Estimating juvenile winter-run and spring-run Chinook salmon entrainment onto the Yolo Bypass over a notched Fremont Weir. California Fish and Game 100(4):630–639.

Acuña, S., D. Baxa, and S. Teh. 2012a. Sublethal Dietary Effects of Microcystin Producing Microcystis on Threadfin Shad, *Dorosoma petenense*. Toxicon 60(6):1191–1202.

Acuña, S., D. Deng, P. Lehman, S. Teh. 2012b. Sublethal Dietary Effects of Microcystis on Sacramento Splittail, *Pogonichthys macrolepidotus*. Aquatic Toxicology 110-111: 1–8. doi: 10.1016/j.aquatox.2011.12.004.

Acuña, S., D. Baxa, P. Lehman, F. C. Teh, D. F. Deng, and S. Teh. 2020. Determining the exposure pathway and impacts of *Microcystis* on Threadfin shad, *Dorosoma petenense*, in San Francisco Estuary. Environmental Toxicology and Chemistry 39(4):787–798.

Alpers, C., C. Eagles-Smith, C. Foe, S. Klasing, M. Marvin-DiPasquale, D. Slotton, and L. Winham-Myers. 2008. Mercury Conceptual Model. Sacramento (CA): Delta Regional Ecosystem Restoration Implementation Plan.

American River Water Agencies. 2017. *Modified Flow Management Standard Proposed Water-Right Terms and Conditions*. Exhibit ARWA-502. November.

Anderson, J. 2018. Using River Temperature to Optimize Fish Incubation Metabolism and Survival: A Case for Mechanistic Models. Researchgate Preprint. 10.1101/257154.

Armor, C., R. Baxter, B. Bennett, R. Bruer, M. Chotkowski, P. Coulston, D. Denton, B. Herbold, W. Kimmerer, K. Larsen, M. Nobriga, K. Rose, T. Sommer, and M. Stacey. 2005. Interagency Ecological Program Synthesis of 2005 Work to Evaluate the Pelagic Organism Decline (POD) in the Upper San Francisco Estuary. Interagency Ecological Program, Sacramento, CA.

Arthur, J. F., M. D. Ball, and S. Y. Baughman. 1996. Summary of Federal and State Water Project Environmental Impacts in the San Francisco Bay-Delta Estuary, California. In The San Francisco Bay: The Ecosystem, edited by J. T. Hollibaugh, 445–495. Seventy-fifth annual meeting of the Pacific Division, American Association for the Advancement of Science. Held at San Francisco State University, June 19–24, 1994. San Francisco, CA.

Azat, J. 2021. GrandTab 2021.06.307. California Central Valley Chinook Population Database Report. California Department of Fish and Wildlife.

Baerwald, M. R., B. M. Schreier, G. Schumer, and B. May. 2012. Detection of Threatened Delta Smelt in the Gut Contents of the Invasive Mississippi Silverside in the San Francisco

Estuary Using TaqMan Assays. Transactions of the American Fisheries Society 141: 1600–1607.

Bartholomew, J. L. 2012. 3.2.4 Salmonid ceratomyxosis. AFS-FHS (American Fisheries Society-Fish Health Section). FHS Blue Book: Suggested Procedures for the Detection and Identification of Certain Finfish and Shellfish Pathogens, 2014 Edition.

Baxter, R., K. Hieb, S. DeLeón, K. Fleming, and J. Orsi. 1999. Report on the 1980-1995 fish, shrimp, and crab sampling in the San Francisco Estuary, California. Interagency Ecological Program for the Sacramento-San Joaquin Estuary, Technical Report 63, California Department of Fish and Game, Stockton, CA, 503 p.

Baxter, R., R. Breuer, L. Brown, M. Chotkowski, F. Feyrer, M. Gingras, B. Herbold, A. Mueller-Solger, M. Nobriga, T. Sommer, and K. Souza. 2008. Pelagic Organism Decline Progress Report: 2007 Synthesis of Results. Technical Report 227. Interagency Ecological Program for the San Francisco Estuary.

Baxter, R., R. Breuer, L. Brown, L. Conroy, F. Feyrer, S. Fong, K. Gehrts, L. Grimaldo, B. Herbold, P. Hrodey, A. Mueller-Solger, T. Sommer, and K. Souza. 2010. Pelagic Organism Decline Work Plan and Synthesis of Results. Interagency Ecological Program for the San Francisco Estuary.

Beamesderfer, R., M. Simpson, G. Kopp, J. Inman, A. Fuller, and D. Demko. 2004. Historical and Current Information on Green Sturgeon Occurrence in the Sacramento and San Joaquin Rivers and Tributaries. Prepared by for State Water Contractors, Sacramento, CA.

Beckon, W.N., and T.C. Maurer. 2008. Potential Effects of Selenium Contamination on Federally Listed Species Resulting from Delivery of Federal Water to the San Luis Unit. U.S. Fish and Wildlife Service. Sacramento Fish and Wildlife Office. Environmental Contaminants Division.

Beckvar, N., T. M. Dillon, and L. B. Read. 2005. Approaches for linking whole-body fish tissue residues of mercury or DDT to biological effects thresholds. Environmental Toxicology and Chemistry: An International Journal 24(8):2094–2105.

Bellido-Leiva, F., R. A. Lusardi, and J. R. Lund. 2021. Modeling the Effect of Habitat Availability and Quality on Endangered Winter-Run Chinook Salmon (*Oncorhynchus tshawytscha*) Production in the Sacramento Valley. *Ecological Modelling* 447.

Bennett, W. A. 2005. Critical Assessment of the Delta Smelt Population in the San Francisco Estuary, California. San Francisco Estuary and Watershed Science 3: Article 1.

Bennett, W. A., and J. R. Burau. 2015. Riders on the Storm: Selective Tidal Movements Facilitate the Spawning Migration of Threatened Delta Smelt in the San Francisco Estuary. Estuaries and Coasts 38(3):826–835.

Bennett, W. A., J. A. Hobbs, and S. J. Teh. 2008. Interplay of Environmental Forcing and Growth-Selective Mortality in the Poor Year Class Success of Delta Smelt in 2005. Final Report. Fish Otolith and Condition Study 2005. Prepared for the Pelagic Organism Decline Management Team.

Bever, A. and M. MacWilliamson. 2018. Evaluation of Sedimentation in the Low Salinity Zone. Presented to the California Water and Environmental Modeling Forum 2018 Annual Meeting, April 2018.

Blake, A., and M. J. Horn. 2006. Acoustic Tracking of Juvenile Chinook Salmon Movement in the Vicinity of the Delta Cross Channel, Sacramento River, California – 2001 Study Results. Draft report. United States Geological Survey.

Blake, A., P. Stumpner, and J. Burau. 2017. A Simulation Method for Combining Hydrodynamic Data and Acoustic Tag Tracks to Predict the Entrainment of Juvenile Salmonids onto the Yolo Bypass under Future Engineering Scenarios. July 21. West Sacramento, CA: U.S. Geological Survey.

Bovee, K.D., Lamb, B.L., Bartholow, J.M., Stalnaker, C.B. and Taylor, J. 1998. *Stream habitat analysis using the instream flow incremental methodology*. Geological Survey Reston VA Biological Resources Div. p.11-65.

Brandl, S., Schreier, B., Conrad, J. L., May, B., & Baerwald, M. 2021. Enumerating Predation on Chinook Salmon, Delta Smelt, and Other San Francisco Estuary Fishes Using Genetics. North American Journal of Fisheries Management 41(4):1053–1065.

Bratovich, P., J. Weaver, C. Addley, and C. Hammersmark. 2017. *Lower American River. Biological Rationale, Development and Performance of the Modified Flow Management Standard*. Exhibit ARWA-702. Prepared for Water Forum. Sacramento, CA.

Brege, D. A., L. G. Gilbreath, R. A. Absolon, B. J. Sandford, and G. M. Matthews. 2005. Studies to Evaluate the Effectiveness of Extended-Length Screens at John Day Dam, 2004. October. Seattle, WA: National Marine Fisheries Service, Northwest Fisheries Science Center, Fish Ecology Division.

Brooks, M. L., E. Fleishman, L. R. Brown, P. W. Lehman, I. Werner, N. Scholz, C. Mitchelmore, J. R. Lovvorn, M. L. Johnson, D. Schlenk, and S. van Drunick. 2012. Life Histories, Salinity Zones, and Sublethal Contributions of Contaminants to Pelagic Fish Declines Illustrated with a Case Study of San Francisco Estuary, California, USA. Estuaries and Coasts 35(2):603–621.

Brown, C. J. 2000. *North of the Delta Offstream Storage Investigation Progress Report*. Appendix D: Fish Survey Summary. Assisted by W. Yip, G. Gorden, G. Low, and A. Scholzen. CALFED Bay-Delta Program.

Brown, L., and P. Moyle. 2005. Native Fishes of the Sacramento–San Joaquin Drainage, California: A History of Decline. *American Fisheries Society Symposium* 45:75–98.

Brown, L. R., and D. Michniuk. 2007. Littoral Fish Assemblages of the Alien dominated Sacramento–San Joaquin Delta, California 1980–1983 and 2001–2003. Estuaries and Coasts 30: 186–200.

Brown, R., B. Cavallo, and K. Jones. 2004. The Effects of the Feather River Hatchery on Naturally Spawning Salmonids. Oroville Facilities Relicensing (FERC Project Number 2100). Draft Report SP-F9. Prepared by California Department of Water Resources, Sacramento, California, and Pacific States Marine Fisheries Commission, Portland, OR.

Buchanan, R. A., and J. R. Skalski. 2020. Relating Survival of Fall-Run Chinook Salmon through the San Joaquin Delta to River Flow. Environmental Biology of Fishes 103(5):389–410.

Buchanan, R. A., E. Buttermore, and J. Israel. 2021. Outmigration Survival of a Threatened Steelhead Population through a Tidal Estuary. Canadian Journal of Fisheries and Aquatic Sciences 78:1869–1886.

Burau, J. R., S. G. Monismith, M. T. Stacey, R. N. Oltmann, J. R. Lacy, and D. H. Schoellhamer. 2000. Recent research on the hydrodynamics of the Sacramento-San Joaquin River Delta and North San Francisco Bay. IEP Newsletter 13(2): 45-55. Interagency Ecological Program for the Sacramento-San Joaquin Estuary, Sacramento, California. http://www.water.ca.gov/iep/newsletters/2000/IEPNewsletter_Spring2000.pdf. Accessed: May 19, 2021

Burau, J., A. Blake, and R. Perry. 2007. Sacramento/San Joaquin River Delta Regional Salmon Outmigration Study Plan: Developing Understanding for Management and Restoration. December 10.

Bureau of Reclamation. 2003. Ecosystem Restoration Opportunities in the Upper Sacramento River Region.

Bureau of Reclamation. 2005. Lake Natoma Temperature Curtain and Channel Modification Study, 2001–2002. Hydraulic Laboratory Report HL-2005-02.

Bureau of Reclamation. 2007. Folsom General Plan/Resource Management Plan: Folsom Lake Recreation Area and Folsom Powerhouse State Historic Park.

Bureau of Reclamation. 2008. Biological Assessment on the Continued Long-Term Operations of the Central Valley Project and the State Water Project. Bureau of Reclamation Mid-Pacific Region, Sacramento CA. August.

Bureau of Reclamation. 2010. CVPIA Sacramento River Spawning Gravel Addition Project at Keswick Dam, Categorical Exclusion Checklist. October 5.

Bureau of Reclamation. 2013. Shasta Lake Water Resources Investigation, California. Draft Environmental Impact Statement. June.

Bureau of Reclamation. 2019. *Reinitiation of Consultation on the Coordinated Long-Term Operation of the Central Valley Project and State Water Project*. Final Biological Assessment. January. Mid-Pacific Region, Bureau of Reclamation.

Bureau of Reclamation. 2020. *Annual Report on the Long-Term Operation of the Central Valley Project and State Water Project for Water Year 2020*. Central Valley Project, California-Great Basin Region. December.

Bureau of Reclamation and California Department of Fish and Game. 2011. Final Environmental Impact Statement/Environmental Impact Report for the Nimbus Hatchery Fish Passage Project.

Bureau of Reclamation, California Department of Water Resources, U.S. Fish and Wildlife Service, National Marine Fisheries Service, and California Department of Fish and Game. 2003. Environmental Water Account Draft Environmental Impact Statement/Environmental Impact Report.

Bureau of Reclamation, U.S. Fish and Wildlife Service, National Marine Fisheries Service, California Department of Fish and Game, and Water Forum. 2006. Lower American River Flow Management Standard. Draft Report.

Bush, E. E. 2017. "*Migratory Life Histories and Early Growth of the Endangered Estuarine Delta Smelt* (Hypomesus transpacificus)." MS Thesis. University of California, Davis. Davis, CA.

Cain, J. and C. Monohan, 2008. *Estimating Ecologically Based Flow Targets for the Sacramento and Feather Rivers.* April. The Natural Heritage Institute.

CALFED Bay-Delta Program. 2000a. Ecosystem Restoration Program Plan. Volume I: Ecological Attributes of the San Francisco Bay-Delta Watershed.

CALFED Bay-Delta Program. 2000b. Ecosystem Restoration Program Plan - Volume II: Ecological Management Zone Visions. Final Programmatic EIS/EIR Technical Appendix. Sacramento, CA.

California Department of Fish and Game. 1992. A Re-examination of Factors Affecting Striped Bass Abundance in the Sacramento-San Joaquin Estuary. WRINT-DFG-Exhibit 2. Entered by the California Department of Fish and Game for the State Water Resources Control Board 1992 Water Rights Phase of the Bay-Delta Estuary Proceedings.

California Department of Fish and Game. 2003. *Fisheries Studies at Stony Creek, Thomes Creek, Sites Newville Projects, and Colusa Basin Drain Report*. California Department of Fish and Game, Central Valley Bay-Delta Branch.

California Department of Fish and Game. 2008. *California Aquatic Invasive Species Management Plan*. January. Available: https://wildlife.ca.gov/Conservation/Invasives/Plan. Accessed: May 14, 2012.

California Department of Fish and Game. 2009. California Endangered Species Act Incidental Take Permit No. 2081-2009-001-03. Department of Water Resources California State Water Project Delta Facilities and Operations. Yountville, CA: California Department of Fish and Game, Bay Delta Region.

California Department of Fish and Wildlife. 2018. Franks Tract Future? Exploring options for Multi-Benefit Restoration and Increased Resilience in the Central Delta Corridor. June 2018.

California Department of Fish and Wildlife. 2020. California Endangered Species Act Incidental Take Permit No. 2081-2019-066-00. Long-Term Operation of the State Water Project in the Sacramento San Joaquin Delta. Sacramento, CA: California Department of Fish and Game, Ecosystem Conservation Division.

California Department of Fish and Wildlife. 2022. *California Central Valley Chinook Escapement Database Report*. GrandTab 2022.07.20. Available: https://nrm.dfg.ca.gov/FileHandler.ashx?DocumentID=84381&inline. Accessed: January 13, 2023.

California Department of Transportation. 2015. Technical Guidance for Assessment and Mitigation of the Hydroacoustic Effects of Pile Driving on Fish. California Department of Transportation, Sacramento, CA.

California Department of Water Resources. 1985. Sacramento River Spawning Gravel Studies, Executive Summary. June.

California Department of Water Resources. 1994. Sacramento River Bank Erosion Investigation. Memorandum Progress Report. September 23.

California Department of Water Resources. 2004. Evaluation of Project Effects on Instream Flows and Fish Habitat. SP F-16 Phase 2 Report. Oroville Facilities Relicensing, FERC Project No. 2100.

California Department of Water Resources. 2005. Bulletin 250. Fish Passage Improvement: An Element of CALFED's Ecosystem Restoration Program. 263 pp. Available from https://cawaterlibrary.net/wp-content/uploads/2017/05/CalFed-Fish-Passage-Report.pdf.

California Department of Water Resources. 2006. Settlement Agreement for Licensing of the Oroville Facilities. FERC Project No. 2100. Available: https://www.waterboards.ca.gov/waterrights/water_issues/programs/water_quality_cert/docs/oroville_ferc2100/oroville2006settlementagreement.pdf. Accessed April 1, 2021.

California Department of Water Resources. 2007. Oroville Facilities Relicensing FERC Project No. 2100 Draft Environmental Impact Report. May.

California Department of Water Resources. 2010. *State Plan of Flood Control Descriptive Document*.

California Department of Water Resources. 2016. *Biological Assessment for the California WaterFix*. Available: https://www.waterboards.ca.gov/waterrights/water_issues/programs/bay_delta/california_waterfix/exhibits/exhibit104/docs/1_Cover_TitlePage_TOC_RevisedDraftBA.pdf. Accessed: July 27, 2021.

California Department of Water Resources. 2017. *Evaluating adult salmonid and sturgeon passage potential for multiple modified Fremont Weir configurations: application of the Yolo Bypass Passage for Adult Salmonid and Sturgeon (YBPASS) Tool*. Technical memorandum for the Yolo Bypass Salmonid Habitat Restoration and Fish Passage Project. Sacramento, California.

California Department of Water Resources. 2019a. California EcoRestore Projects. https://water.ca.gov/Programs/All-Programs/EcoRestore#. Accessed June 4, 2020.

California Department of Water Resources. 2019b. Work Plan for Monitoring and Assessment of Proposed Suisun Marsh Salinity Control Gates Action, 2019. West Sacramento, CA: California Department of Water Resources, Division of Environmental Sciences. June 3.

California Department of Water Resources. 2019c. North Delta Flow Action Study – 2019 Operation Plan, May 2019.

California Department of Water Resources. 2019d. California Data Exchange Center. Available: http://cdec.water.ca.gov/jspplot/jspPlotServlet.jsp?sensor_no=17305&end=10%2F01%2F2018+21%3A28&geom=huge&interval=60&cookies=cdec01. Accessed: January 2, 2019.

California Department of Water Resources. 2019e. Incidental Take Permit Application for Long-Term Operation of the California State Water Project. December 13. Available: https://water.ca.gov/-/media/DWR-Website/Web-Pages/Programs/State-Water-Project/Files/1_DWR_LTO_ITP_Application_2019-12-13_a_y19.pdf Accessed: February 24, 2020.

California Department of Water Resources. 2020. Final Environmental Impact Report for Long-term Operation of the California State Water Project. State Clearinghouse No. 2019049121. March.

California Department of Water Resources. 2021. California Data Exchange Center. Available: https://cdec.water.ca.gov/rtables. Accessed: May 21, 2021.

California Department of Water Resources. 2023. California EcoRestore. Available: https://water.ca.gov/Programs/All-Programs/EcoRestore. Accessed: January 31, 2023.

California Department of Water Resources and Bureau of Reclamation. 2008. *North-of-the-Delta Offstream Storage Investigation Plan Formulation Report*. September.

California Department of Water Resources and Environmental Science Associates. 2020. *Tisdale Weir Rehabilitation and Fish Passage Project Draft Environmental Impact Report*. Sacramento, California.

California Department of Water Resources, Bureau of Reclamation, U.S. Fish and Wildlife Service, and National Marine Fisheries Service. 2013. *Environmental Impact Report/ Environmental Impact Statement for the Bay Delta Conservation Plan*. Draft. December.

California Hatchery Scientific Review Group. 2012. California Hatchery Review Report. Prepared for the U.S. Fish and Wildlife Service and Pacific States Marine Fisheries Commission. June.

California Natural Resources Agency. 2016. Delta Smelt Resiliency Strategy – July 2016.

California Natural Resources Agency. 2017. Delta Smelt Resiliency Strategy – Progress Report, June 2017.

Castillo, G., J. Morinaka, J. Lindberg, R. Fujimura, B. Baskerville-Bridges, J. Hobbs, G. Tigan, L. Ellison. 2012. Pre-screen Loss and Fish Facility Efficiency for Delta Smelt at the South Delta's State Water Project, California. *San Francisco Estuary and Watershed Science* 10(4).

Cavallo, B., R. Brown, and D. Lee. 2009. Hatchery and Genetic Management Plan for Feather River Hatchery Spring-run Chinook Salmon Program. June. Auburn, California: Cramer Fish Sciences.

Cavallo, B., J. Merz, and J. Setka. 2012. Effects of Predator and Flow Manipulation on Chinook Salmon (*Oncorhynchus tshawytscha*) Survival in an Imperiled Estuary. Environmental Biology of Fishes 96(2–3): 393–403. doi. 10.1007/s10641-012-9993-5

Cavallo, B., P. Gaskill, J. Melgo, and S.C. Zeug. 2015. Predicting juvenile Chinook Salmon routing in riverine and tidal channels of a freshwater estuary. Environmental Biology of Fishes, 98(6):1571–1582.

Chapman, E. D., A. R. Hearn, C. J. Michel, A. J. Ammann, S. T. Lindley, M. J. Thomas, P. T. Sandstrom, G. P. Singer, M. L. Peterson, R. B. MacFarlane, and A. P. Klimley. 2013. Diel Movements of Out-migrating Chinook Salmon (*Oncorhynchus tshawytscha*) and Steelhead Trout (*Oncorhynchus mykiss*) Smolts in the Sacramento/San Joaquin Watershed. Environmental Biology of Fishes 96(2-3):273–286.

Clark, K. W., M. D. Bowen, R. B. Mayfield, K. P. Zehfuss, J. D. Taplin, and C. H. Hanson. 2009. Quantification of Pre-Screen Loss of Juvenile Steelhead in Clifton Court Forebay. March.

Columbia Basin Research. 2020. SacPAS CDFW GrandTab Adult Escapement. University of Washington. Available:

http://www.cbr.washington.edu/sacramento/data/query_adult_grandtab.html Accessed: May 13, 2020.

Conrad, J. L., A. J. Bibian, K. L. Weinersmith, D. De Carion, M. J. Young, P. Crain, E. L. Hestir, M. J. Santos, and A. Sih. 2016. Novel Species Interactions in a Highly Modified Estuary: Association of Largemouth Bass with Brazilian Waterweed *Egeria densa*. Transactions of the American Fisheries Society 145(2):249–263.

Connon, R. E., S. Beggel, L. S. D'Abronzo, J. P. Geist, J. Pfeiff, A. V. Loguinov, C. D. Vulpe, and I. Werner. 2011. Linking molecular biomarkers with higher level condition indicators to identify effects of copper exposures on the endangered delta smelt (*Hypomesus transpacificus*). Environmental Toxicology and Chemistry 30(2):290–300.

Cordoleani, F., E. Holmes, and C. Jeffres. 2020. *Evaluating the Role(s) of the Butte Sink and Sutter Bypass for Butte Creek Spring- Run Chinook Salmon and Other Central Valley Juvenile Salmonid Populations - 2019 Study Year*.

Cramer, S. P., D. B. Demko, C. G. Chapman, M. L. Simpson, and G. J. R. Kopp. 2005. Potential Fish Benefits from Screens or Screening Modifications at Sutter Mutual Water Company, Reclamation District 108, and M&T Ranch/Llano Seco. Technical Memorandum. S.P. Cramer & Associates. June 29.

Cramer Fish Sciences. 2016. Coleman National Fish Hatchery Adaptive Management Plan. Public Review Draft, March 1. Prepared for Bureau of Reclamation, Mid-Pacific Region. Auburn, CA.

Cunningham, C., N. Hendrix, E. Dusek-Jennings, R. Lessard, and R. Hilborn. 2015. Delta Chinook Final Report to the Delta Science Panel. Available: deltacouncil.ca.gov/sites/default/files/2039%20Final%20Report.pdf.

David Evans and Associates, Inc., MWH Americas, Inc., and R2 Resource Consultants. 2007. Clean Water Services Tualatin Basin Water Supply Project. Fish Habitat, Draft Technical Report. Portland, OR.

Davis, B., M. Bedwell, H. Wright, and J. Martinez. 2019. North Delta flow action 2019: Delta smelt and salmonids report. Department of Water Resources.

Delaney, D., P. Bergman, B. Cavallo, and J. Melgo. 2014. Stipulation Study: Steelhead Movement and Survival in the South Delta with Adaptive Management of Old and Middle River Flows. Prepared under the direction of Kevin Clark, Bay-Delta Office, Biotelemetry and Special Investigations Unit. California Department of Water Resources.

Dinehart, R. L., and J. R. Burau. 2005. Averaged Indicators of Secondary Flow in Repeated Acoustic Doppler Current Profiler Crossings of Bends. Water Resources Research 41(9):W09405.

Dugdale, R. C., F. P. Wilkerson, V. E. Hogue, and A. Marchi. 2007. The Role of Ammonium and Nitrate in Spring Bloom Development in San Francisco Bay. Estuarine, Coastal and Shelf Science 73:17–29.

East Bay Municipal Utility District. 2012. Mokelumne River hatchery fish information received by Jose Setka, Fisheries and Wildlife Division.

Edmunds, J. L., K. M. Kuivila, B. E. Cole, and J. E. Cloern. 1999. Do Herbicides Impair Phytoplankton Primary Production in the Sacramento-San Joaquin River Delta? In Proceedings of the Technical Meeting: Toxic Substances Hydrology Program. Volume 2: Contamination of Hydrologic Systems and Related Ecosystems. U.S. Geological Survey Water Resources Investigation Report 99.4018B.

Engle, J., C. Enos, K. McGourty, T. Porter, B. Reed, J. Scammell-Tinling, K. Schaeffer, S. Siegel, and E. Crumb. 2010. Suisun Marsh Tidal Marsh and Aquatic Habitats Conceptual Model. Chapter 2: Aquatic Environment. Final Review Draft. Suisun Marsh Habitat Management, Restoration, and Preservation Plan.

Enos, C., J. Sutherland, and M. L. Nobriga. 2007. Results of a Two Year Fish Entrainment Study at Morrow Island Distribution System in Suisun Marsh. IEP Newsletter 20(1): 10-19.

Federal Energy Regulatory Commission. 2007. Final Environmental Impact Statement for Hydropower License, Oroville Facilities, FERC Project No. 2100-052, California.

Ferrari, M.C., Ranåker, L., Weinersmith, K.L., Young, M.J., Sih, A. and Conrad, J.L., 2014. Effects of turbidity and an invasive waterweed on predation by introduced largemouth bass. Environmental Biology of Fishes 97(1):79–90.

Feyrer, F., B. Herbold, S. A. Matern, and P. B. Moyle. 2003. Dietary Shifts in a Stressed Fish Assemblage: Consequences of a Bivalve Invasion in the San Francisco Estuary. Environmental Biology of Fishes 67: 277–288.

Feyrer, R., T. R. Sommer, and R. D. Baxter. 2005. Spatial-Temporal Distribution and Habitat Associations of Age-0 Splittail in the Lower San Francisco Watershed. Copeia 2005(1):159–168.

Feyrer, F., Sommer, T., and W. Harrell. 2006a. Managing floodplain inundation for native fish: production dynamics of age-0 splittail (*Pogonichthys macrolepidotus*) in California's Yolo Bypass. Hydrobiologia, 573(1):213–226.

Feyrer, F., T. Sommer, and W. Harrell. 2006b. Importance of flood dynamics versus intrinsic physical habitat in structuring fish communities: evidence from two adjacent engineered floodplains on the Sacramento River, California. North American Journal of Fisheries Management 26:408–417.

Feyrer, F., M. L. Nobriga, and T. R. Sommer. 2007. Multi-decadal Trends for Three Declining Fish Species: Habitat Patterns and Mechanisms in the San Francisco Estuary, California, U.S.A. *Canadian Journal of Fisheries and Aquatic Sciences* 64:723–734.

Feyrer, F., T. Sommer, and S. B. Slater. 2009. Old School vs. New School: Status of Threadfin Shad (*Dorosoma petenense*) Five Decades After Its Introduction to the Sacramento–San Joaquin Delta. San Francisco Estuary and Watershed Science 7(1).

Feyrer F., J. Hobbs, and T. Sommer. 2010. Salinity Inhabited by Age-0 Splittail (*Pogonichthys macrolepidotus*) as Determined by Direct Field Observation and Retrospective Analyses with Otolith Chemistry [Online]. *San Francisco Estuary and Watershed Science* 8(2). Available: http://escholarship.org/uc/item/14j5160h.

Feyrer, F, K. Newman, M. Nobriga, and T. Sommer. 2011. Modeling the Effects of Future Freshwater Flow on the Abiotic Habitat of an Imperiled Estuarine Fish. *Estuaries and Coasts* 34:120–128.

Feyrer, F., K. Newman, M. Nobriga, and T. Sommer. 2015. Delta Smelt Habitat in the San Francisco Estuary: A Reply to Manly, Fullerton, Hendrix, and Burnham's "Comments on Feyrer et al. Modeling the Effects of Future Outflow on the Abiotic Habitat of an Imperiled Estuarine Fish." Estuaries and Coasts 39(1):287–289.

Fish, M. A. 2010. A White Sturgeon Year-Class Index for the San Francisco Estuary and Its Relation to Delta Outflow. IEP Newsletter 23(2). Spring.

Fisher, F.W. 1992. Chinook Salmon, *Oncorhynchus tshawytscha*, growth and occurrence in the Sacramento–San Joaquin River System. Sacramento (CA): California Department of Fish and Game, Inland Fisheries Division.

Fisheries Hydroacoustic Working Group. 2008. Agreement in Principal for Interim Criteria for Injury to Fish from Pile Driving Activities. National Marine Fisheries Service Northwest and Southwest Regions, U.S. Fish and Wildlife Service Regions 1 and 8, California/Washington/Oregon Departments of Transportation, California Department of Fish and Game, and U.S. Federal Highway Administration. Memorandum to Applicable Agency Staff. June 12.

Fong, S., S. Louie, I. Werner, J. Davis, and R. E. Connon. 2016. Contaminant Effects on California Bay–Delta Species and Human Health. San Francisco Estuary and Watershed Science 14(4). Available: http://www.escholarship.org/uc/item/52m780xj.

Foott, J. S. 2014. Sacramento and Feather River Juvenile Chinook Pathogen Survey Spring 2014. U.S. Fish and Wildlife Service California – Nevada Fish Health Center, Anderson, CA. Available: https://www.fws.gov/canvfhc/Reports/NWFHS%20Misc/Foott%20JS.%202014%20FR-LS%20NWFS%20Narrative_Sacramento%20and%20Feather%20River%20Juvenile%20Chinook%20Pathogen%20Survey.pdf. Accessed: June 1, 2020.

Foott, J. S., R. Stone, S. Voss, and K. Nichols. 2017. *Ceratonova shasta* and *Parvicapsula minibicornis* (Phylum Cnidaria: Myxosporea) infectivity for juvenile Chinook salmon (*Oncorhynchus tshawytscha*) in the Upper Sacramento River: July–November 2016. U.S. Fish and Wildlife Service California – Nevada Fish Health Center, Anderson, CA. Available: https://www.fws.gov/canvfhc/Reports/Sacaramento%20and%20San%20Joaquin%20River/Foot,%20J.Scott,%20R.%20Stone,%20S.%20Voss%20and%20K.%20Nichols,%202017.%20Ceratonova%20shasta%20and%20Parvicapsula%20minibicornis%20(Phylum%20Cnidaria%20Myxosporea)%20infectivity.pdf. Accessed: June 1, 2020.

Frantzich, J., T. Sommer, and B. Schreier. 2018. Physical and Biological Responses to Flow in a Tidal Freshwater Slough Complex. San Francisco Estuary and Watershed Science 16(1).

Frantzich, J., Davis, B. E., MacWilliams, M., Bever, A., & Sommer, T. 2021. Use of a Managed Flow Pulse as Food Web Support for Estuarine Habitat. San Francisco Estuary and Watershed Science, 19(3).

Fris, M. B. and R. W. DeHaven. 1993. A Community-Based Habitat Suitability Index Model for Shaded Riverine Aquatic Cover, Selected Reaches of the Sacramento River System. February. U.S. Fish and Wildlife Service, Ecological Services. Sacramento Field Office. Sacramento, CA.

Gahan, G., M. Healey, C. McKibbin, H. Kubo, and C. Purdy. 2016. Colusa Basin Drain and Wallace Weir Fish Trapping and Relocation Efforts. November 2013 – June 2014. California Department of Fish and Wildlife, North Central Region. August.

Gingras, M. 1997. Mark/Recapture Experiments at Clifton Court Forebay to Estimate Pre-screening Loss to Juvenile Fishes, 1976-1993. Technical Report 55. Interagency Ecological Program for the San Francisco Bay/Delta Estuary. September.

Glenn-Colusa Irrigation District. 2001. Fish Screen Project: long-term protection for fisheries. Brochure published by GCID. 4 pp. https://912afe62-5b11-482e-8c47-c2358db4f96b.filesusr.com/ugd/c88b6b_36f9a9e00f8942b8960ff2b404ebd0af.pdf

Glenn-Colusa Irrigation District. 2017. https://www.youtube.com/watch?v=u_DOLE7QXa8 Accessed: February 11, 2021.

Glibert, P. M., D. Fullerton, J. M. Burkholder, J. C. Cornwell, and T. M. Kana. 2011. Ecological Stoichiometry, Biogeochemical Cycling, Invasive Species, and Aquatic Food Webs: San Francisco Estuary and Comparative Systems. Reviews in Fisheries Science 19(4):358–417.

Glibert, P. M., F. P. Wilkerson, R. C. Dugdale, A. E. Parker, J. Alexander, S. Blaser, and S. Murasko. 2014. Phytoplankton communities from San Francisco Bay Delta respond differently to oxidized and reduced nitrogen substrates—even under conditions that would otherwise suggest nitrogen sufficiency. Frontiers in Marine Science 1(17):1–16.

Goodman, D. H. and S. B. Reid. 2012. Pacific Lamprey (*Entosphenus tridentatus*) Assessment and Template for Conservation Measures in California. U.S. Fish and Wildlife Service, Arcata, California.

Goodman, D. H. and S. B. Reid. 2022. Rapid development of larval Pacific lamprey *Entosphenus tridentatus* in southern populations provides adaptive benefits for uncertain flow regimes. Environmental Biology of Fish. https://doi.org/10.1007/s10641-022-01236-6

Goodman, D. H., S. B. Reid, N. A. Som, and W. R. Poytress. 2015. The punctuated seaward migration of Pacific lamprey (*Entosphenus tridentatus*): environmental cues and implications for streamflow management. Canadian Journal of Fisheries and Aquatic Sciences 72(12):1817–1828.

Goodman, D. H., Reid, S. B., Reyes, R. C., Wu, B. J., and B.B. Bridges. 2017. Screen efficiency and implications for losses of lamprey macrophthalmia at California's largest water diversions. North American Journal of Fisheries Management, 37(1):30–40.

Gould, A., and W. J. Kimmerer. 2010. Development, growth, and reproduction of the cyclopoid copepod *Limnoithona tetraspina* in the upper San Francisco Estuary. Marine Ecology Progress Series 412:163–177. 10.3354/meps08650.

Graham, R. J. and D. J. Orth. 1986. Effects of Temperature and Streamflow on Time and Duration of Spawning by Smallmouth Bass. *Transactions of the American Fisheries Society* 115:693–702.

Greater Battle Creek Watershed Working Group. 2006. Battle Creek: An Extraordinary Restoration Opportunity. Available: https://www.battle-creek.net/docs/gbcwwg/gbcwwg_brochure_03_13_06.pdf.

Greenberg, J. A., Hestir, E. L., & Ustin, S. L. 2008. The impacts of riparian tree removal on water temperatures in the Sacramento-San Joaquin River Delta using Lidar data analysis. In AGU Fall Meeting Abstracts (Vol. 2008, pp. B41A-0348).

Greene, V. E., L. J. Sullivan, J. K. Thompson, W. J. Kimmerer. 2011. Grazing Impact of the Invasive Clam *Corbula amurensis* on the Microplankton Assemblage of the Northern San Francisco Estuary. Marine Ecology Progress Series 431:183–193.

Greenfield, B. K., S. J. Teh, J. R. M. Ross, J. Hunt, G. H. Zhang, J. A. Davis. G. Ichikawa, D. Crane, S. O. Hung, D. F. Deng, F. C. Teh, P. G. Green. 2008. Contaminant Concentrations and Histopathological Effects in Sacramento Splittail (*Pogonichthys macrolepidotus*). Environmental Contamination & Toxicology 55(2): 270–281. August.

Greenwood, M. 2018. *Potential Effects on Zooplankton From California WaterFix Operations*. Technical Memorandum to California Department of Water Resources. July 2. Available: https://www.waterboards.ca.gov/waterrights/water_issues/programs/bay_delta/california_ waterfix/exhibits/docs/petitioners_exhibit/dwr/part2_rebuttal/dwr_1349.pdf. Accessed: November 30, 2018.

Gregory, R. S., and C. D. Levings. 1998. Turbidity Reduces Predation on Migrating Juvenile Pacific Salmon. Transactions of the American Fisheries Society 127(2):275–285.

Grimaldo, L. F., R. E. Miller, C. M. Peregrin, and Z. P. Hymanson. 2004. Spatial and Temporal Distribution of Native and Alien Ichthyoplankton in Three Habitat Types of the Sacramento–San Joaquin Delta. American Fisheries Society Symposium 39: 81–96.

Grimaldo, L. F., T. Sommer, N. Van Ark, G. Jones, E. Holland, P. B. Moyle, B. Herbold, and P. Smith. 2009. Factors Affecting Fish Entrainment into Massive Water Diversions in a Freshwater Tidal Estuary: Can Fish Losses be Managed? *North American Journal of Fisheries Management* 29:1253–1270.

Grimaldo, L., R. E. Miller, C. M. Peregrin, and Z. Hymanson. 2012. Fish Assemblages in Reference and Restored Tidal Freshwater Marshes of the San Francisco Estuary. San Francisco Estuary and Watershed Science 10(1).

Grimaldo, L. F., Smith, W. E., & Nobriga, M. L. 2021. Re-Examining Factors That Affect Delta Smelt (Hypomesus transpacificus) Entrainment at the State Water Project and Central Valley Project in the Sacramento–San Joaquin Delta. *San Francisco Estuary and Watershed Science* 19(1).

Gross, E. S., Korman, J., Grimaldo, L. F., MacWilliams, M. L., Bever, A. J., & Smith, P. E. 2021. Modeling Delta Smelt Distribution for Hypothesized Swimming Behaviors. *San Francisco Estuary and Watershed Science* 19(1).

Grossman, G. D., T. Essington, B. Johnson, J. Miller, N. E. Monsen, and T. N. Pearsons. 2013. Effects of Fish Predation on Salmonids in the Sacramento River-San Joaquin Delta and Associated Ecosystems. September 25.

Hallock, R. J., R. F. Elwell, and D. H. Fry, Jr. 1970. Migrations of Adult King Salmon, *Oncorhynchus tshawytscha*, in the San Joaquin Delta. California Department of Fish and Game Bulletin 151:1–92.

Hamilton, S. A., and D. D. Murphy. 2020. Use of affinity analysis to guide habitat restoration and enhancement for the imperiled delta smelt. Endangered Species Research 43:103-120.

Hanson, C. H., and H. W. Li. 1978. A Research Program to Examine Fish Behavior in Response to Hydraulic Flow Fields – Development of Biological Design Criteria for Proposed Water Diversions. Completion report for Project C-7679, Grant No. 14-34-001-7254, Office of Water Research and Technology, United States Department of the Interior. February. Davis, CA: University of California.

Hanson, C. H. 2001. Are Juvenile Chinook Salmon Entrained at Unscreened Diversions in Direct Proportion to the Volume of Water Diverted? Contributions to the Biology of Central Valley Salmonids. California Department of Fish and Game Fish Bulletin 179: 331–342.

Hanson, M. B., Baird, R. W., Ford, J. K. B., Hempelmann, J. A., Van Doornik, D. M., Candy, J. R., Emmons, C. K., Schorr, G. S., Gisborne, B., Ayres, K. L., Wasser, S. K., Balcomb, K. C., Balcomb-Bartok, K., Sneva, J. G., and Ford, M. J. 2010. Species and stock identification of prey consumed by endangered Southern Resident Killer Whales in their summer range. Endangered Species Research 11:69–82. doi: 10.3354/esr00263.

Harrell, W. C., and T. R. Sommer. 2003. Patterns of Adult Fish Use on California's Yolo Bypass Floodplain. California Riparian Systems: Processes and Floodplain Management, Ecology, and Restoration. Pages 88–93 in P. M. Faber, editor, California Riparian Systems: Processes and Floodplain Management, Ecology, and Restoration. 2001 Riparian Habitat and Floodplains Conference Proceedings. Riparian Habitat Joint Venture.

Harvey, B. N., D. P. Jacobson, and M. A. Banks. 2014. Quantifying the Uncertainty of a Juvenile Chinook Salmon Race Identification Method for a Mixed-Race Stock. North American Journal of Fisheries Management 34(6):1177–1186.

Hassrick, J. L., A. J. Ammann, R. W. Perry, S. N. John, and M. E. Daniels. 2022. Factors Affecting Spatiotemporal Variation in Survival of Endangered Winter-Run Chinook Salmon Out-migrating from the Sacramento River. North American Journal of Fisheries Management 42(2):375–395.

Hastings, M. C., and A. N. Popper. 2005. Effects of sound on fish. Report prepared for California Department of Transportation. Jones & Stokes, Sacramento, CA.

Hellmair, M., Peterson, M., Mulvey, B., Young, K., Montgomery, J., & Fuller, A. 2018. Physical characteristics influencing nearshore habitat use by juvenile Chinook salmon in the Sacramento River, California. North American Journal of Fisheries Management, 38(4):959–970.

Henderson, M. J., I. S. Iglesias, C. J. Michel, A. J. Ammann, and D. D. Huff. 2019. Estimating spatial-temporal differences in Chinook salmon outmigration survival with habitat and predation related covariates. Canadian Journal of Fisheries and Aquatic Sciences 76(9):1549–1561.Hennessy, A., and Z. Burris. 2017. Preliminary analysis of current relationships between zooplankton abundance and freshwater outflow in the upper San Francisco Estuary. Memorandum to S. Louie, Senior Environmental Scientist, California Department of Fish and Wildlife, Water Branch. February 21. Stockton, CA: California Department of Fish and Wildlife, Bay-Delta Region.

Hennessy, A., and T. Enderlein. 2013. Zooplankton Monitoring 2011. IEP Newsletter 26(1): 23–30.

Herren, J. R., and S. S. Kawasaki. 2001. Inventory of Water Diversions in Four Geographic Areas in California's Central Valley. Contributions to the Biology of Central Valley Salmonids. Edited by R. L. Brown. *California Department of Fish and Game Fish Bulletin* 179(2):343–355.

Hestir, E. L., D. H. Schoellhamer, J. Greenberg, T. Morgan-King, and S. L. Ustin. 2016. The Effect of Submerged Aquatic Vegetation Expansion on a Declining Turbidity Trend in the Sacramento-San Joaquin River Delta. Estuaries and Coasts 39(4):1100-1112. Available: http://dx.doi.org/10.1007/s12237-015-0055-z.

Heublein, J. C., J. T. Kelly, C. E. Crocker, A. P. Klimley, and S. T. Lindley. 2009. Migration of Green Sturgeon (*Acipenser medirostris)* in the Sacramento River. *Environmental Biology of Fishes* 84: 245–258.

Heublein, J., R. Belmer, R. D. Chase, P. Doukakis, M. Gingras, D. Hampton, J. A. Israel, Z. J. Jackson, R. C. Johnson, O. P. Langness, S. Luis, E. Mora, M. L. Moser, L. Rohrbach, A. M. Seesholtz, T. Sommer, and J. S. Stuart. 2017. Life History and Current Monitoring Inventory of San Francisco Estuary Sturgeon. National Marine Fisheries Service, NOAA Technical Memorandum NOAA-TM-NMFS-SWFSC-589.

Hilborn, R., S. P. Cox, F. M. D. Gulland, D. G. Hankin, N. T. Hobbs, D. E. Schindler, and A. W. Trites. 2012. The Effects of Salmon Fisheries on Southern Resident Killer Whales: Final Report of the Independent Science Panel. Prepared for National Marine Fisheries Service and Fisheries and Oceans Canadas.

Hinrichsen, R. A., D, J. Hasselman, C. C. Ebbesmeyer, B. A. Shields. 2013. The Role of Impoundments, Temperature, and Discharge on Colonization of the Columbia River Basin, USA. *Transactions of the American Fisheries Society* 142(4): 887–900.

Hobbs, J. A., Lewis, L. S., Willmes, M., Denney, C., & Bush, E. 2019. Complex life histories discovered in a critically endangered fish. Scientific Reports 9(1):1-12.

Houlton, B., and J. Lund. 2018. Sacramento Summary Report. California's Fourth Climate Change Assessment. Publication number: SUM-CCCA4-2018-002.

Huber, E. R., and S. M. Carlson. 2015. Temporal Trends in Hatchery Releases of Fall-Run Chinook Salmon in California's Central Valley. San Francisco Estuary and Watershed Science 13(2).

Hutton, P. H., J. S. Rath, L. Chen, M. J. Ungs, and S. B. Roy. 2015. Nine Decades of Salinity Observations in the San Francisco Bay and Delta: Modeling and Trend Evaluations. *Journal of Water Resources Planning and Management* 142(3):04015069.

ICF International. 2015. Butte Regional Conservation Plan Public Draft Environmental Impact Statement/Environmental Impact Report. May. (ICF 00736.10.) Sacramento, CA. Prepared for Butte County Association of Governments, Chico, CA.

ICF International. 2016. Biological Assessment for the California WaterFix. July. (ICF 00237.15.) Sacramento, CA. Prepared for U.S. Department of the Interior, Bureau of Reclamation, Sacramento, CA.

Independent Review Panel. 2011. Report of the 2011 Independent Review Panel (IRP) on the Implementation of Reasonable and Prudent Alternative (RPA) Actions Affecting the Operations Criteria And Plan (OCAP) for State/Federal Water Operations. Prepared by J. J. Anderson, J. A. Gore, R. T. Kneib, M.S. Lorang, and J. Van Sickle. Delta Stewardship Council/Delta Science Program.

Interagency Ecological Program, B. Mahardja, J. Speegle, A. Nanninga, and D. Barnard. 2019. Interagency Ecological Program: Over four decades of juvenile fish monitoring data from the San Francisco Estuary, collected by the Delta Juvenile Fish Monitoring Program, 1976-2018 ver 3. Environmental Data Initiative. https://doi.org/10.6073/pasta/87dda12bed2271ce3d91abdb7864c50c. Accessed January 9, 2023.

Interagency Ecological Program Management, Analysis, and Synthesis Team. 2015. An updated conceptual model of Delta Smelt biology: our evolving understanding of an estuarine fish. Technical Report 90. January. Interagency Ecological Program for the San Francisco Bay/Delta Estuary, Sacramento, CA.

Jabusch, T., A. Melwani, K. Ridolfi, and M. Connor. 2008. Effects of Short-term Water Quality Impacts Due to Dredging and Disposal on Sensitive Fish Species in San Francisco Bay. Prepared for U.S. Army Corps of Engineers, San Francisco District. September. Oakland, CA: San Francisco Estuary Institute.

Jackson, Z.J., Gruber, J.J. and Van Eenennaam, J.P. 2016. White sturgeon spawning in the San Joaquin River, California, and effects of water management. Journal of Fish and Wildlife Management, 7(1), pp.171-180.

Jassby, A. D., W. J. Kimmerer, S. G. Monismith, C. Armor, J. E. Cloern, T. M. Powell, J. R. Schubel, and T. J. Vendlinski. 1995. Isohaline Position as a Habitat Indicator for Estuarine Populations. *Ecological Applications* 5:272–289.

Jassby, A. D., J. E. Cloern, and B. E. Cole. 2002. Annual Primary Production: Patterns and Mechanisms of Change in a Nutrient-rich Tidal Ecosystem. *Limnology and Oceanography* 47:698–712.

Jeffries, K. M., R. E. Connon, B. E. Davis, L. M. Komoroske, M. T. Britton, T. Sommer, A. E. Todgham, and N. A. Fangue. 2016. Effects of high temperatures on threatened estuarine fishes during periods of extreme drought. Journal of Experimental Biology 219(11):1705–1716.

Johnson, R. R., D. C. Weigand, and F. W. Fisher. 1992. Use of Growth Data to Determine the Spatial and Temporal Distribution of Four Runs of Juvenile Chinook Salmon in the

Sacramento River, California. North Central Valley Fishery Resource Office, Red Bluff, CA.

Johnson, M. L, I. Werner, S. J. Teh, and F. Loge. 2010. Evaluation of Chemical, Toxicological, and Histopathologic Data to Determine Their Role in the Pelagic Organism Decline. Available: Accessed 2015 October 24. Final report to the California State Water Resources Control Board and Central Valley Regional Water Quality Control Board. University of California, Davis.

Johnston, M., J. Frantzich, M. B. Espe, P. Goertler, G. Singer, T. Sommer, and A. P. Klimley. 2020. Contrasting the migratory behavior and stranding risk of White Sturgeon and Chinook Salmon in a modified floodplain of California. Environmental Biology of Fishes 103:481–493.

Jungbluth, M., C. Lee, C. Patel, T. Ignoffo, B. Bergamaschi, and W. Kimmerer. 2021. Production of the Copepod *Pseudodiaptomus forbesi* Is Not Enhanced by Ingestion of the Diatom *Aulacoseira granulata* During a Bloom. Estuaries and Coasts 44:1083–1099.

Katz, J. V., C. Jeffres, J. L. Conrad, T. R. Sommer, J. Martinez, S. Brumbaugh, N. Corline, and P. B. Moyle. 2017. Floodplain Farm Fields Provide Novel Rearing Habitat for Chinook Salmon. *PLOS ONE*. Available: http://journals.plos.org/plosone/article?id=10.1371/journal.pone.0177409. Accessed: June 7, 2021.

Kimmerer, W. J. 2002a. Physical, biological, and management responses to variable freshwater flow into the San Francisco Estuary. *Estuaries* 25:1275–1290.

Kimmerer, W. J. 2002b. Effects of freshwater flow on abundance of estuarine organisms: physical effects or trophic linkages. Marine Ecology Progress Series 243:39–55.

Kimmerer, W. J. 2004. Open Water Processes of the San Francisco Estuary: from Physical Forcing to Biological Responses. *San Francisco Estuary and Watershed Science* 2(1).

Kimmerer, W. J. 2005. Long-Term Changes in Apparent Uptake of Silica in the San Francisco Estuary. Limnology and Oceanography 50:793–798.

Kimmerer, W. J. 2006. Response of anchovies dampens effects of the invasive bivalve *Corbula amurensis* on the San Francisco Estuary foodweb. Marine Ecology Progress Series. 324:207–218.

Kimmerer, W. J., and M. Nobriga. 2008. Investigating Particle Transport and Fate in the Sacramento San Joaquin Delta Using a Particle Tracking Model. *San Francisco Estuary and Watershed Science* 6(1).

Kimmerer, W. J. and J. K. Thompson. 2014. Phytoplankton Growth Balanced by Clam and Zooplankton Grazing and Net Transport into the Low-Salinity Zone of the San Francisco Estuary. Estuaries and Coasts, Pre-print published online: January 7.

Kimmerer, W. J., E. Gartside, and J. J. Orsi. 1994. Predation by an Introduced Clam as the Likely Cause of Substantial Declines in Zooplankton of San Francisco Bay. Marine Ecology Progress Series 113:81–93.

Kimmerer, W. J., J. R. Burau, W. A. Bennett. 1998. Tidally Oriented Vertical Migration and Position Maintenance of Zooplankton in a Temperate Estuary. Limnol. Oceanogr. 43(7): 1697–1709.

Kimmerer, W. J., J. H. Cowan, Jr., L. W. Miller, and K. A. Rose. 2000. Analysis of an Estuarine Striped Bass (*Morone saxatilis*) Population: Influence of Density-dependent Mortality Between Metamorphosis and Recruitment. Canadian Journal of Fisheries and Aquatic Sciences 57:478–486.

Kimmerer, W. J., N. Ferm, M. H. Nicolini, C. Penalva. 2005. Chronic Food Limitation of Egg Production in Populations of Copepods of the Genus *Acartia* in the San Francisco Estuary. Estuaries 28(4): 541–550. August.

Kimmerer, W., L. Brown, S. Culberson, P. Moyle, M. Nobriga, and J. Thompson. 2008. Aquatic Ecosystems. Pages 73–101 in M. Healey, editor, The State of Bay-Delta Science. CALFED Science Program.

Kimmerer, W. J., E. S. Gross, and M. L. MacWilliams. 2009. Is the Response of Estuarine Nekton to Freshwater Flow in the San Francisco Estuary Explained by Variation in Habitat Volume? *Estuaries and Coasts* 32:375–389. Doi 10.1007/s12237-008-9124-x.

Kimmerer, W. J., T. R. Ignoffo, K. R. Kayfetz, and A. M. Slaughter. 2018. Effects of Freshwater Flow and Phytoplankton Biomass on Growth, Reproduction, and Spatial Subsidies of the Estuarine Copepod *Pseudodiaptomus forbesi*. Hydrobiologia 807:113–130.

Kimmerer, W. J., E. S. Gross, A. M. Slaughter, and J. R. Durand. 2019. Spatial Subsidies and Mortality of an Estuarine Copepod Revealed Using a Box Model. Estuaries and Coasts 42(1):218–236.

Kneib, R. T. 2019. Independent Review of the Coordinated Long-Term Operation of the Central Valley Project and State Water Project. Prepared for National Marine Fisheries Service and U.S. Fish and Wildlife Service.

Korman, J., E. S. Gross, and L. F. Grimaldo. 2021. Statistical Evaluation of Behavior and Population Dynamics Models Predicting Movement and Proportional Entrainment Loss of Adult Delta Smelt in the Sacramento–San Joaquin River Delta. *San Francisco Estuary and Watershed Science* 19(1).

Kuivila, K. M., and C. G. Foe. 1995. Concentrations, Transport and Biological Effects of Dormant Spray Pesticides in the San Francisco Estuary, California. Environmental Toxicology and Chemistry 14: 1141–1150.

Lee, D. P., and J. Chilton. 2007. Hatchery and Genetic Management Plan for Nimbus Fish Hatchery Winter-Run Steelhead Program. Draft Report. Prepared by CDFG under Contract 03CS200006 Modification 0004 with Bureau of Reclamation, Folsom, CA, and Nimbus Fish Hatchery, Rancho Cordova, CA.

Lee, G. F., and A. Jones-Lee. 2003. Summary of Findings on the Causes and Factors Influencing Low DO in the San Joaquin River Deep Water Ship Channel near Stockton, CA. Report of G. Fred Lee & Associates, El Macero, CA. March. Available: http://www.gfredlee.com/psjriv2.htm.

Lee, J. W., De Riu, N., Lee, S., Bai, S. C., Moniello, G., and S.S. Hung. 2011. Effects of dietary methylmercury on growth performance and tissue burden in juvenile green (*Acipenser medirostris*) and white sturgeon (*A. transmontanus*). Aquatic Toxicology, 105(3-4):227–234.

Lee, P., and J. Manning. 2020. Mercury Imports and Exports of Four Tidal Wetlands in the Sacramento-San Joaquin Delta, Yolo Bypass, and Suisun Marsh for Delta Mercury Control Program Compliance. California Department of Water Resources.

Lehman, P. W., G. Boyer, C. Hall, and K. Gehrts. 2005. Distribution and Toxicity of a New Colonial *Microcystis aeruginosa* Bloom in the San Francisco Bay Estuary, California. Hydrobiologia (2005) 541:87–99. DOI 10.1007/s10750-004-4670-0.

Lehman, P. W., T. Sommer, and L. Rivard. 2008a. The Influence of Floodplain Habitat on the Quantity of Riverine Phytoplankton Carbon Produced During the Flood Season in San Francisco Estuary. Aquatic Ecology 42:363–378.

Lehman, P. W., G. Boyer, M. Satchwell, and S. Waller. 2008b. The Influence of Environmental Conditions on the Seasonal Variation of Microcystis Cell Density and Microcystins Concentration in San Francisco Estuary. Hydrobiologia 600(1): 187–204.

Lehman, P. W., S. J. Teh, G. L. Boyer, M. L. Nobriga, E. Bass, C. Hogle. 2010. Initial Impacts of *Microcystis aeruginosa* Blooms on the Aquatic Food Web in the San Francisco Estuary. Hydrobiologia (2010) 637:229–248. DOI 10.1007/s10750-009-9999-y.

Lehman, P.W., Kurobe, T., Huynh, K., Lesmeister, S. and Teh, S.J., 2021. Covariance of Phytoplankton, Bacteria, and Zooplankton Communities Within Microcystis Blooms in San Francisco Estuary. Frontiers in microbiology, p.1184.

Lindley, S. T., R. Schick, E. Mora, P. B. Adams, J. J. Anderson, S. Greene, C. Hanson, B. P. May, D. R. McEwan, R. B. MacFarlane, C. Swanson, and J. G. Williams. 2007. Framework for Assessing Viability of Threatened and Endangered Chinook Salmon and steelhead in the Sacramento-San Joaquin Basin. San Francisco Estuary and Watershed Science 5: 26.

Linville, R.G., Luoma, S.N., Cutter, L. and Cutter, G.A., 2002. Increased selenium threat as a result of invasion of the exotic bivalve *Potamocorbula amurensis* into the San Francisco Bay-Delta. Aquatic Toxicology, 57(1-2):51–64.

Loboschefsky, E., G. Benigno, T. Sommer, K. Rose, T. Ginn, A. Massoudieh, and F. Loge. 2012. Individual-level and Population-level Historical Prey Demand of San Francisco Estuary Striped Bass Using a Bioenergetics Model. *San Francisco Estuary and Watershed Science* 10(1).

Lukas, J. A. and D. J. Orth. 1995. Factors Affecting Nesting Success of Smallmouth Bass in a Regulated Virginia Stream. *Transactions of the American Fisheries Society* 124: 726–735.

Mac Nally, R., Thomson, J. R., Kimmerer, W. J., Feyrer, F., Newman, K. B., Sih, A. et al. 2010. Analysis of Pelagic Species Decline in the Upper San Francisco Estuary Using Multivariate Autoregressive Modeling (MAR). *Ecological Applications* 20(5):1417–1430.

Maguire, A., C. Stuart, M. Bedwell, B. Davis, and J. Frantzich. 2020. North Delta Flow Action 2019: Continuous Water Quality Monitoring in the Yolo Bypass. Poster presentation at 2020 Interagency Ecological Program Workshop. Available: https://nrm.dfg.ca.gov/FileHandler.ashx?DocumentID=182044 Accessed: 27 August 2020.

Mahardja, B., J. L. Conrad, L. Lusher. And B. Schreier. 2016. Abundance Trends, Distribution, and Habitat Associations of the Invasive Mississippi Silverside (*Menidia audens*) in the Sacramento–San Joaquin Delta, California, USA. San Francisco Estuary and Watershed Science 14(1).

Mahardja, B., M. J. Farruggia, B. Schreier, and T. Sommer. 2017. Evidence of a Shift in the Littoral Fish Community of the Sacramento-San Joaquin Delta. PLoS ONE 12(1):e0170683.

Mahardja, B., J. A. Hobbs, N. Ikemiyagi, A. Benjamin, and A. J. Finger. 2019. Role of Freshwater Floodplain-Tidal Slough Complex in the Persistence of the Endangered Delta Smelt. PLOS ONE 14(1):e208084.

Manly, B. F. J., D. Fullerton, A. N. Hendrix, and K. P. Burnham. 2015. Comments on Feyrer et al. "Modeling the Effects of Future Outflow on the Abiotic Habitat of an Imperiled Estuarine Fish." Estuaries and Coasts 38(5):1815–1820.

Marschall, E. A., D. C. Glover, M. E. Mather, D. L. Parrish. 2020. Modeling larval American shad recruitment in a large river. Special Section: 4th Mississippi–Yangtze River Basins Symposium. *North American Journal of Fisheries Management*.

Marston, D., C. Mesick, A. Hubbard, D. Stanton, S. Fortmann-Roe, S. Tsao, and T. Heyne. 2012. Delta Flow Factors Influencing Stray Rate of Escaping Adult San Joaquin River fall-run

Chinook Salmon (*Oncorhynchus tshawytscha*). *San Francisco Estuary and Watershed Science* 10(4).

Martin, B. T., A. Pike, S. N. John, N. Hamda, J. Roberts, S. T. Lindley, and E. M. Danner. 2016. Phenomenological vs. biophysical models of thermal stress in aquatic eggs. Ecology Letters 2016.

Martin, B. T., A. Pike, S. N. John, N. Hamda, J. Roberts, S. T. Lindley, and E. M. Danner. 2017. Phenomenological vs. biophysical models of thermal stress in aquatic eggs. Ecology Letters 20: 50–59. doi: 10.1111/ele.12705.

Matern, S. A., P. B. Moyle, and L. C. Pierce. 2002. Native and Alien Fishes in a California Estuarine Marsh: Twenty-one Years of Changing Assemblages. Transactions of the American Fisheries Society 131(5): 797–816, DOI: 10.1577/1548-8659(2002)131<0797:NAAFIA>2.0.CO;2.

McCauley, R., J. Fewtrell, and A. Popper. 2003. High Intensity Anthropogenic Sound Damages Fish Ears. *The Journal of the Acoustical Society of America* 113:638–642. 10.1121/1.1527962.

McMillen, LLC. 2013. *Glenn-Colusa Irrigation District (GCID) Gradient Facility, Sacramento River, CA*. Water Data Technical Memorandum. June 21.

Mefford, B., and Z. Sutphin. 2008. *Intake Diversion Dam Fish Screens*. Evaluation of Fish Screens for Protecting Early Life Stages of Pallid Sturgeon. Hydraulic Laboratory Report HL-2007-010. Draft. September. Denver, CO: Bureau of Reclamation, Technical Service Center, Water Resources Research Laboratory.

Merced Irrigation District. 2013. Technical Memorandum 3-5. Instream Flow (PHABSIM) Downstream of Crocker-Huffman Dam. Merced Hydroelectric Project, FERC Project No. 2179.

Merz, J., B. Rook, C. Watry, and S. Zeug. 2012. Evaluation of the 2008-2010 Sailor Bar Gravel Placements on the Lower American River, California. 2010-2011 Data Report. Prepared for City of Sacramento Water Forum, and Bureau of Reclamation and U.S. Fish and Wildlife Service, CVPIA Gravel Program. Contract 2010-1049.

Merz, J., P. Bergman, J. Melgo, and S. Hamilton. 2013. Longfin Smelt: Spatial Dynamics and Ontogeny in the San Francisco Estuary, California. *California Fish and Game* 99:122–148.

Michel, C. J. 2010. River and Estuarine Survival and Migration of Yearling Sacramento River Chinook Salmon (*Oncorhynchus tshawytscha*) Smolts and the Influence of Environment. Master’s Thesis. University of California, Santa Cruz.

Michel, C. J. 2018. Decoupling outmigration from marine survival indicates outsized influence of streamflow on cohort success for California's Chinook salmon populations. Canadian

Journal of Fisheries and Aquatic Sciences. Available: https://doi.org/10.1139/cjfas-2018-0140.

Michel, C. J., A. J. Ammann, S. T. Lindley, P. T. Sandstrom, E. D. Chapman, M. J. Thomas, G. P. Singer, A. P. Klimley, and R. B. MacFarlane. 2015. Chinook Salmon Out-Migration Survival in Wet and Dry Years in California's Sacramento River. Canadian Journal of Fisheries and Aquatic Sciences 72: 1749–1759.

Michel, C., J. Notch, F. Cordoleani, A. Ammann, and E. Danner. 2021. Nonlinear survival of imperiled fish informs managed flows in a highly modified river. Ecosphere. DOI: 10.1002/ecs2.3498

Mitchell, C. P. J, T. E. Jordan, A. Heyes, and C. C. Gilmour. 2012. Tidal Exchange of Total Mercury and Methylmercury Between a Salt Marsh and a Chesapeake Bay Sub-Estuary. Biogeochemistry 111 (1-3):583–600. Available: https://doi.org/10.1007/s10533-011-9691-y.

Moser, M. L., A. D. Jackson, M. C. Lucas, and R. P. Mueller. 2015. Behavior and potential threats to survival of migrating lamprey ammocoetes and macrophthalmia. Reviews in Fish Biology and Fisheries 25(1):103–116.

Mount, J., W. Bennett, J. Durand, W. Fleenor, E. Hanak, J. Lund, and P. B. Moyle. 2012. Aquatic Ecosystem Stressors in the Sacramento–San Joaquin Delta. Public Policy Institute of California, San Francisco.

Moyle, P. B. 2002. Inland Fishes of California. Second edition. University of California Press, Berkeley.

Moyle, P. B., R. D. Baxter, T. Sommer, T. C. Foin, and S. A. Matern. 2004. Biology and Population Dynamics of Sacramento Splittail (*Pogonichthys macrolepidotus*) in the San Francisco Estuary: a Review. *San Francisco Estuary and Watershed Science* 2:3.

Moyle, P. B., and W. A. Bennett. 2008. The Future of the Delta Ecosystem and Its Fish. Technical Appendix D, Comparing Futures for the Sacramento–San Joaquin Delta. Public Policy Institute of California, San Francisco.

Moyle, P., W. Bennett, J. Durand, W. Fleenor, B. Gray, E. Hanak, J. Lund, and J. Mount. 2012. Where the Wild Things Aren’t: Making the Delta a Better Place for Native Species. Public Policy Institute of California, San Francisco, CA. Available: http://www.ppic.org/content/pubs/report/R_612PMR.pdf.

Moyle, P. B., R. M. Quiñones, J. V. Katz and J. Weaver. 2015. Fish Species of Special Concern in California. Sacramento: California Department of Fish and Wildlife. Available: https://nrm.dfg.ca.gov/FileHandler.ashx?DocumentID=104282&inline.

Mueller-Solger, A. B., A. D. Jassby, and D. C. Muller-Navarra. 2002. Nutritional Quality of Food Resources for Zooplankton (Daphnia) in a Tidal Freshwater System (Sacramento–San Joaquin River Delta). Limnol. Oceanogr. 47(5):1468–1476.

Murphy, D.D., and P.S. Weiland. 2019. The Low-Salinity Zone in the San Francisco Estuary as a Proxy for Delta Smelt Habitat: A Case Study in the Misuse of Surrogates in Conservation Planning. *Ecological Indicators* 105:29–35. Available: https://doi.org/10.1016/j.ecolind.2019.05.053.

Mussen, T. D., D. Cocherell, J. B. Poletto, J. S. Reardon, Z. Hockett, A. Ercan, H. Bandeh, M. L. Kavvas, J. J. Cech, and N. A. Fangue. 2014. Unscreened Water-Diversion Pipes Pose an Entrainment Risk to the Threatened Green Sturgeon, Acipenser medirostris. PLoS ONE 9: e86321.

Myrick, C. A., and J. J. Cech. 2000. Swimming Performances of Four California Stream Fishes: Temperature Effects. Environmental Biology of Fishes 58(3):289–295.

National Marine Fisheries Service. 2008. Biological Opinion on 24,000 Linear Feet of Sacramento River Bank Protection Project, Phase II. 2007/07158. July 2, 2008. Long Beach, CA.

National Marine Fisheries Service. 2009. Biological Opinion and Conference Opinion on the Long-Term Operations of the Central Valley Project and State Water Project. National Marine Fisheries Service Southwest Region. June 4.

National Marine Fisheries Service. 2014. Recovery Plan for the Evolutionarily Significant Units of Sacramento River Winter-Run Chinook Salmon and Central Valley Spring-Run Chinook Salmon and the Distinct Population Segments of California Central Valley Steelhead. July. National Marine Fisheries Service West Coast Region.

National Marine Fisheries Service. 2016. Central Valley Recovery Domain 5-Year Review: Summary and Evaluation of California Central Valley Steelhead Distinct Population Segment. National Marine Fisheries Service, West Coast Region.

National Marine Fisheries Service. 2017. Endangered Species Act Section 7(a)(2) Biological Opinion, Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Response, and Fish and Wildlife Coordination Act Recommendations for the California WaterFix Project in Central Valley, California. NMFS Consultation Number: WCR-2016-5506. June 16. Portland, OR: United States Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, West Coast Region.

National Marine Fisheries Service. 2018a. NOAA Fisheries West Coast Region Anadromous Salmonid Passage Design Guidelines. Review Draft. August 16. National Marine Fisheries Service, West Coast Region, Portland, Oregon.

National Marine Fisheries Service. 2018b. Recovery Plan for the Southern Distinct Population Segment of North American Green Sturgeon (*Acipenser medirostris*). California Central Valley Area Office, National Marine Fisheries Service, Sacramento, CA.

National Marine Fisheries Service. 2019a. Biological Opinion on Long-term Operation of the Central Valley Project and the State Water Project. Consultation Tracking Number: WCRO-2016-00069. West Coast Region, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, U.S. Department of Commerce. October 21.

National Marine Fisheries Service. 2019b. Endangered Species Act Section 7(a)(2) Biological Opinion, and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Response for the Yolo Bypass Salmonid Habitat Restoration and Fish Passage Project. May 10. West Coast Region, National Marine Fisheries Service, Sacramento, CA.

National Research Council. 2012. Sustainable Water and Environmental Management in the California Bay-Delta. Prepared by the Committee on Sustainable Water and Environmental Management in the California Bay Delta. The National Academies Press, Washington, D.C.

Nixon, S. W. 1988. Physical Energy Inputs and the Comparative Ecology of Lake and Marine Ecosystems. Limnology and Oceanography, Part II 33: 1005–1025.

Nobriga, M. L. 2009. Bioenergetic Modeling Evidence for a Context-dependent Role of Food Limitation in California's Sacramento–San Joaquin Delta. California Fish and Game 95(3): 111–121.

Nobriga, M. L., and F. Feyrer. 2007. Shallow-Water Piscivore-Prey Dynamics in California's Sacramento–San Joaquin Delta. San Francisco Estuary and Watershed Science 5(2): Article 4.

Nobriga, M. and B. Herbold. 2009. *The Little Fish in California's Water Supply: A Literature Review and Life-History Conceptual Model for Delta Smelt (*Hypomesus transpacificus*) for the Delta Regional Ecosystem Restoration and Implementation Plan (DRERIP).* Delta Regional Ecosystem Restoration Implementation Plan (DRERIP). Sacramento, CA.

Nobriga, M. L., and J. A. Rosenfield. 2016. Population Dynamics of an Estuarine Forage Fish: Disaggregating Forces Driving Long-Term Decline of Longfin Smelt in California's San Francisco Estuary. Transactions of the American Fisheries Society 145(1):44–58.

Nobriga, M. L., Z. Matica, and Z. P. Hymanson. 2004. Evaluating Entrainment Vulnerability to Agricultural Irrigation Diversions: A Comparison among Open-Water Fishes. In F. Feyrer, L. R. Brown, R. L. Brown, and J. J. Orsi, editors, Early Life History of Fishes in the San Francisco Estuary and Watershed. American Fisheries Society Symposium 39: 281–295.

Nobriga, M. L., F. Feyrer, R. D. Baxter, and M. Chotkowski. 2005. Fish Community Ecology in an Altered River Delta: Spatial Patterns in Species Composition, Life History Strategies and Biomass. Estuaries 28(5): 776–785.

Nobriga, M. L., F. Feyrer, and R. D. Baxter. 2006. Aspects of Sacramento Pikeminnow Biology in Nearshore Habitats of the Sacramento-San Joaquin Delta, California. Western North American Naturalist 66(1):106–114

Nobriga, M. L., T. R. Sommer, F. Feyrer, and K. Fleming. 2008. Long-term Trends in Summertime Habitat Suitability for Delta Smelt, *Hypomesus transpacificus*. San Francisco Estuary and Watershed Science 6: Article 1.

Northern California Water Association. 2018. *Food Production Program Continues to Improve Delta Smelt Conditions*. October 8. Available: https://mavensnotebook.com/wp-content/uploads/2018/10/FoodWeb_PressRelease_FactSheet.pdf. Accessed: May 23, 2019.

Orlando, J. L., M. De Parsia, C. Sanders, M. Hladik, and J. Frantzich. 2020. Pesticide concentrations associated with augmented flow pulses in the Yolo Bypass and Cache Slough Complex, California. Open-File Report 2020-1076. Reston, VA: U.S. Geological Survey.

Orsi, J. and W. L. Mecum. 1986. Zooplankton Distribution and Abundance in the Sacramento–San Joaquin Delta in Relation to Certain Environmental Factors. Estuaries 9(4B):326–339.

Pacific Fishery Management Council. 2018. Salmon Rebuilding Plan for Sacramento River Fall Chinook. Pacific Fishery Management Council, Portland, OR.

Palmer-Zwahlen, M., and B. Kormos. 2013. Recovery of Coded-Wire Tags from Chinook salmon in California's Central Valley Escapement and Ocean Harvest in 2011. Fisheries Branch Administrative Report 2013-02.

Payne, T. R. and M. Allen. 2004. *Evaluation of Project Effects on Instream Flows and Fish Habitat.* Phase 2 Interim Report. SP F-16. Sacramento, CA. Prepared for T. J. Mills, T. Sommer, and B. Cavallo, Oroville Facilities Relicensing Environmental Work Group, DWR. Sacramento, CA.

Payne, T. R. and M. Allen. 2005. *Addendum to Phase II Report Evaluation of Project Efforts on Instream Flows and Fish Habitat.* SP-F16 Sacramento, CA. Prepared for Oroville Facilities Relicensing Environmental Work Group, DWR. Sacramento, CA.

Perry, R. W. 2010. Survival and migration dynamics of juvenile Chinook Salmon (*Oncorhynchus tshawytscha*) in the Sacramento–San Joaquin River Delta. Doctoral dissertation. University of Washington, Seattle.

Perry, R. W., J. G. Romine, S. J. Brewer, P. E. LaCivita, W. N. Brostoff, and E. D. Chapman. 2012. Survival and Migration Route Probabilities of Juvenile Chinook Salmon in the Sacramento-San Joaquin River Delta During the Winter of 2009–10. U.S. Geological Survey Open-File Report 2012-1200, 30 p.

Perry, R. W., P. L. Brandes, J. R. Burau, P. T. Sandstrom, and J. R. Skalski. 2015. Effect of Tides, River Flow, and Gate Operations on Entrainment of Juvenile Salmon into the Interior Sacramento–San Joaquin River Delta. *Transactions of the American Fisheries Society* 144:445–455.

Perry, R. W., A. C. Pope, J. G. Romine, P. L. Brandes, J. R. Burau, A. R. Blake, A. J. Ammann, C. J. Michel. 2018. Flow-Mediated Effects on Travel Time, Routing, and Survival of Juvenile Chinook Salmon in a Spatially Complex, Tidally Forced River Delta. *Canadian Journal of Fisheries and Aquatic Sciences* 75(11):1886–1901.

Perry, R. W., A. C. Hansen, S. D. Evans, and T. J. Kock. 2019. Using the STARS Model to Evaluate the Effects of Two Proposed Projects for the Long-Term Operation of the State Water Project Incidental Take Permit Application and CEQA Compliance. Open-File Report 2019-1127. Version 2.0. February. U. S. Geological Survey, Reston, VA.

Phillis, C. C., A. M. Sturrock, R. C. Johnson, and P. K. Weber. 2018. Endangered winter-run Chinook salmon rely on diverse rearing habitats in a highly altered landscape. Biological Conservation 217:358–362.

Pickard, A., A. Baracco, and R. Kano. 1982. Occurrence, abundance, and size of fish at the Roaring River Intake, Suisun Marsh, California during the 1980-81 and the 1981-82 diversion seasons. IEP Technical Report 3. Interagency Ecological Program for the San Francisco Bay/Delta Estuary, Sacramento, CA.

Plumb, J. M., N. S. Adams, R. W. Perry, C. M. Holbrook, J. G. Romine, A. R. Blake, and J. R. Burau. 2016. Diel Activity Patterns of Juvenile Late Fall-run Chinook Salmon with Implications for Operation of a Gated Water Diversion in the Sacramento–San Joaquin River Delta. River Research and Applications 32(4):711–720.

Polansky, L., K. B. Newman, and L. Mitchell. 2021. Improving inference for nonlinear state-space models of animal population dynamics given biased sequential life stage data. Biometrics 77(1):352–361.

Popper, A., and M. Hastings. 2009. The effects of anthropogenic sources of sound on fishes. Journal of Fish Biology 75(3):455–489.

Poytress, W. R., J. J. Gruber, and J. P. Van Eenennaam. 2011. *2011 Upper Sacramento River Green Sturgeon Spawning Habitat and Larval Migration Surveys*. Annual Report of U.S. Fish and Wildlife Service to Bureau of Reclamation, Red Bluff, CA.

Poytress, W. R., J. J. Gruber, F. D. Carrillo, and S. D. Voss. 2014. *Compendium Report of Red Bluff Diversion Dam Rotary Screw Trap Juvenile Anadromous Fish Production Indices* *for Years 2002–2012*. Prepared for California Department of Fish and Wildlife Ecosystem Restoration Program and the Bureau of Reclamation. July. U.S. Fish and Wildlife Service, Red Bluff, CA.

Poytress, W. R., J. J. Gruber, J. P. Van Eenennaam, and M. Gard. 2015. Spatial and Temporal Distribution of Spawning Events and Habitat Characteristics of Sacramento River Green Sturgeon. *Transactions of the American Fisheries Society* 144(6):1129–1142.

Purdy, C., and S. Kubo. 2021. Wallace Weir and Colusa Basin Drain - Fish Salvage Operations. Available: https://www.calfish.org/ProgramsData/ConservationandManagement/CentralValleyMonitoring/SacramentoValleyTributaryMonitoring/YoloandSutterBypasses-Monitoring/WallaceWeirandColusaBasinDrain.aspx Accessed: March 2021.

Pyper, B., J. B. Lando, and C. Justice. 2006. Analysis of Weir Counts and Spawning Surveys of Adult Chinook Salmon in the Stanislaus River. September. Cramer Fish Sciences, Gresham, OR.

Raleigh, R. F., T. Hickman, R. C. Solomon, and P. C. Nelson. 1984. *Habitat Suitability Information: Rainbow Trout*. USFWS-BSP, Fort Collins, CO.

Ray, R. A., and J. L. Bartholomew. 2013. Estimation of transmission dynamics of the *Ceratomyxa shasta* actinospore to the salmonid host. Parasitology, 140(7):907–916.

RBI. 2017. *Report on the Effects of the California WaterFix on Harmful Algal Blooms in the Delta*. Prepared for California Department of Water Resources. March. Elk Grove, CA: Robertson-Bryan, Inc. Available: https://www.waterboards.ca.gov/waterrights/water_issues/programs/bay_delta/california_waterfix/exhibits/docs/petitioners_exhibit/dwr/DWR-653.pdf. Accessed: December 10, 2018.

Rose, B. P., and M. G. Mesa. 2012. Effectiveness of Common Fish Screen Materials to Protect Lamprey Ammocoetes. North American Journal of Fisheries Management 32(3):597–603.

Rosenfield, J. A., and R. D. Baxter. 2007. Population Dynamics and Distribution Patterns of Longfin Smelt in the San Francisco Estuary. *Transactions of the American Fisheries Society* 136:1577–1592.

Sabal, M. 2014. Interactive Effects of Non-Native Predators and Anthropogenic Habitat Alterations on Native Juvenile Salmon. Master's Thesis. University of California, Santa Cruz, Santa Cruz, CA.

Saiki, M. K., M. R. Jennings, and R. H. Wiedmeyer. 1992. Toxicity of Agricultural Subsurface Drainwater from the San Joaquin Valley, California, to Juvenile Chinook Salmon and Striped Bass. *Transactions of the American Fisheries Society* 121:73–93.

Saleh, D., and J. Domagalski. 2021. Concentrations, Loads, and Associated Trends of Nutrients Entering the Sacramento–San Joaquin Delta, California. *San Francisco Estuary and Watershed Science* 19(4).

Salmon Monitoring Team. 2021. Weekly Assessment of CVP and SWP Delta Operations on ESA-listed Species. Proposed Action Assessment 10/19/2021. Available: https://www.usbr.gov/mp/bdo/docs/samt-weekly-assessment-2021-10-19.pdf. Accessed: December 10, 2021.

Salmonid Scoping Team. 2017. Effects of Water Project Operations on Juvenile Salmonid Migration and Survival in the South Delta. Available: https://www.fisheries.noaa.gov/resource/document/effects-water-project-operations-juvenile-salmonid-migration-and-survival-south.

San Joaquin Tributaries Authority. 2012. Review of Scientific Information Pertaining to SWRCB's February 2012 Technical Report on the Scientific Basis for Alternative San Joaquin River Flow Objectives. Prepared for State Water Resources Control Board Phase II Comprehensive Review Workshops, Workshop 2, "Bay-Delta Fisheries" to be held October 1–2, 2012. Prepared by Doug Demko, Michael Hellmair, Matt Peterson, Shaara Ainsley, Michele Palmer, and Andrea Fuller.

Sandahl, J.F., Baldwin, D.H., Jenkins, J.J., Scholz, N.L., 2007. A sensory system at the interface between urban stormwater runoff and salmon survival. Environmental Science and Technology 41:2998–3004.

Schaffter, R. G. 1997. White Sturgeon Spawning Migrations and Location of Spawning Habitat in the Sacramento River, California. *California Fish and Game* 83(1):1-20.

Schick, R. S., A. L. Edsall, and S. T. Lindley. 2005. Historical and Current Distribution of Pacific Salmonids in the Central Valley, CA. Technical Memorandum 369. National Marine Fisheries Service, Santa Cruz, CA.

Schoellhamer, D. H., S. A. Wright, and J. Drexler. 2012. A Conceptual Model of Sedimentation in the Sacramento–San Joaquin Delta. *San Francisco Estuary and Watershed Science* 10(3).

Schreier, B. M., M. R. Baerwald, J. L. Conrad, G. Schumer, and B. May. 2016. Examination of Predation on Early Life Stage Delta Smelt in the San Francisco Estuary Using DNA Diet Analysis. Transactions of the American Fisheries Society 145(4): 723–733.

Seesholtz, A., B. J. Cavallo, J. Kindopp, and R. Kurth. 2004. Juvenile Fishes of the Lower Feather River: Distribution, Emigration Patterns, and Associations with Environmental Variables. In F. Feyrer, L. R. Brown, R. L. Brown, and J. J. Orsi, editors, Early Life History of Fishes in the San Francisco Estuary and Watershed. American Fisheries Society, Symposium 39: 141–166.

Seesholtz, A. M., M. J. Manuel, and J. P. Van Eenennaam. 2015. First documented spawning and associated habitat conditions for Green Sturgeon in the Feather River, California. Environmental Biology of Fishes 98: 905–912.

Setka, J. 2018. Written Testimony of Jose D. Setka, Expert Witness for East Bay Municipal Utility District. Exhibit EBMUD-155. Part 2 of Hearing in the matter of California Department of Water Resources and United States Bureau of Reclamation Request for a Change in Point of Diversion for California WaterFix Before the California State Water Resources Control Board. Available: https://www.waterboards.ca.gov/waterrights/water_issues/programs/bay_delta/california_waterfix/exhibits/docs/EBMUD/part2/ebmud_155.pdf. Accessed: April 28, 2022.

Siegel, S., C. Enright, C. Toms, C. Enos, and J. Sutherland. 2010. Suisun Marsh Tidal Marsh and Aquatic Habitats Conceptual Model. Chapter 1: Physical Processes. Suisun Marsh Habitat Management, Restoration and Preservation Plan. Final Review Draft. Prepared by WWR and DWR.

Singer, G., and J. Frantzich. 2019. Task 8: (Salmon Telemetry) Adult fall-run Chinook Salmon acoustic telemetry to investigate residence time and movement in the Yolo Bypass. Pages 149–174 in Investigating Yolo Bypass as a Fall Food Web Subsidy for the Delta. CDFW Contract Agreement D1683001 00 Report. Completed and Submitted: 12/5/2019. Sacramento, CA: California Department of Water Resources.

Sisco, J. 2019. Reclaiming the lost population. US Fish and Wildlife Service, Pacific Southwest Region. 18 April. Available: https://www.fws.gov/cno/newsroom/highlights/2019/winter-run-2019/ Accessed: 11 May 11, 2020.

Sites Project Authority and Bureau of Reclamation. 2017. Draft Environmental Impact Report/ Environmental Impact Statement. Draft. Prepared by Sites Project Authority and the U.S. Department of the Interior, Bureau of Reclamation, Mid-Pacific Region. Volume 1.

Smith, D. L., J. M. Nestler, G. E. Johnson, and R. A. Goodwin. 2009. Species-Specific Spatial and Temporal Distribution Patterns of Emigrating Juvenile Salmonids in the Pacific Northwest. Reviews in Fisheries Science 18(1):40–64.

Smith, J. M., C. J. Michel, S. A. Hayes, D. A. Demer, G. R. Cutter Jr., S. C. Manugian, I. S. Iglesias, S. T. Lindley, T. P. Quinn, and D. D. Huff. 2017. The Influence of Environmental Covariates on Chinook Salmon Predation Events by Non-native Fish Predators in the San Joaquin River, CA. Chapter 7 in *Testing the Effects of Manipulated Predator Densities and Environmental Variables on Juvenile Salmonid Survival in the lower San Joaquin River*. June. Agreement Number 4600010100. Prepared for California Department of Water Resources, Bay-Delta Office. Sacramento, CA.

Smith, W. E., L. Polansky, and M. L. Nobriga. 2021. Disentangling risks to an endangered fish: using a state-space life cycle model to separate natural mortality from anthropogenic losses. Canadian Journal of Fisheries and Aquatic Sciences 78(8):1008–1029.

Sommer, T., and F. Mejia. 2013. A Place to Call Home: A Synthesis of Delta Smelt Habitat in the Upper San Francisco Estuary. San Francisco Estuary and Watershed Science 11(2).

Sommer, T. R., R. Baxter, and B. Herbold. 1997. Resilience of Splittail in the Sacramento–San Joaquin Estuary. *Transactions of the American Fisheries Society* 126:961–976.

Sommer, T., B. Harrell, M. Nobriga, R. Brown, P. Moyle, W. Kimmerer, and L. Schemel. 2001. California's Yolo Bypass: Evidence that flood control can be compatible with fisheries, wetlands, wildlife, and agriculture. *Fisheries* 26(8):6–16.

Sommer, T., L. Conrad, G. O'Leary, F. Feyrer, and W. Harrell. 2002. Spawning and Rearing of Splittail in a Model Floodplain Wetland. *Transactions of the American Fisheries Society* 131:966–974.

Sommer, T. R., W. C. Harrell, M. L. Nobriga, and R. Kurth. 2003. Floodplain as Habitat for Native Fish: Lessons from California's Yolo Bypass. Pages 81–87 in P. M. Faber, editor, California Riparian Systems: Processes and Floodplain Management, Ecology, and Restoration. 2001 Riparian Habitat and Floodplains Conference Proceedings. Riparian Habitat Joint Venture, Sacramento, CA.

Sommer, T. R., W. C. Harrell, A. Mueller-Solger, B. Tom, and W. Kimmerer. 2004. Effects of Flow Variation on Channel and Floodplain Biota and Habitats of the Sacramento River, California, USA. Aquatic Conservation: Marine and Freshwater Ecosystems 14: 247–261.

Sommer, T., W. Harrell, and M. Nobriga. 2005. Habitat Use and Stranding Risk of Juvenile Chinook Salmon on a Seasonal Floodplain. North American Journal of Fisheries Management 25: 1493–1504

Sommer, T. R., C. Armor, R. Baxter, R. Breuer, L. Brown, M. Chotkowski, S. Culberson, F. Feyrer, M. Gingras, B. Herbold, W. Kimmerer, A. Mueller-Solger, M. Nobriga, and K. Souza. 2007. The Collapse of Pelagic Fishes in the Upper San Francisco Estuary. Fisheries 32: 270–277.

Sommer, T. R., W. C. Harrell, Z. Matica, and F. Feyrer. 2008. Habitat Associations and Behavior of Adult and Juvenile Splittail (*Cyprinidae: Pogonichthys macrolepidotus*) in a Managed Seasonal Floodplain Wetland. *San Francisco Estuary and Watershed Science* 5(2):Article 3. Available: http://www.escholarship.org/uc/item/85r15611.

Sommer, T., F. H. Mejia, M. L. Nobriga, F. Feyrer, and L. Grimaldo. 2011. The Spawning Migration of Delta Smelt in the Upper San Francisco Estuary. *San Francisco Estuary and Watershed Science* 9(2).

Sommer, T.R., W.C. Harrell, and F. Feyrer. 2014. Large-Bodied Fish Migration and Residency in a Flood Basin of the Sacramento River, California, USA. Ecology of Freshwater Fish 23: 414–423.

Sommer, T., B. Schreier, J. L. Conrad, L. Takata, B. Serup, R. Titus, C. Jeffres, E. Holmes, and J. Katz. 2020. Farm to Fish: Lessons from a Multi-Year Study on Agricultural Floodplain Habitat. San Francisco Estuary and Watershed Science 18(3).

State Water Resources Control Board. 1999. Water Right Decision 1641. December 29. Available: https://cawaterlibrary.net/document/water-rights-decision-1641-d-1641/. Accessed: January 31, 2023.

State Water Resources Control Board. 2018. *The 303(d) List of Impaired Water Bodies*. California Regional Water Quality Control Board San Francisco Bay Region.

Stern, M. A., L. E. Flint, A. L. Flint, N. Knowles, and S. A. Wright. 2020. The Future of Sediment Transport and Streamflow Under a Changing Climate and the Implications for Long-Term Resilience of the San Francisco Bay-Delta. Water Resources Research 56(9):e2019WR026245.

Stevens, D. E., D. W. Kohlhorst, L. W. Miller, and D. W. Kelley. 1985. The Decline of Striped Bass in the Sacramento-San Joaquin Estuary, California. *Transactions of the American Fisheries Society* 114:12–30.

Stevens, D. E., H. K. Chadwick, and R. E. Painter. 1987. American shad and striped bass in California's Sacramento-San Joaquin River system. American Fisheries Society Symposium 1:66–78.

Stewart, A. R., S. N. Luoma, C. E. Schlekat, M. A. Doblin, and K. A. Hieb. 2004. Food web pathway determines how selenium affects aquatic ecosystems: a San Francisco Bay case study. Environmental Science & Technology 38:4519–4526.

Stompe, D. K., J. D. Roberts, C. A. Estrada, D. M. Keller, N. M. Balfour, and A. I. Benet. 2020. Sacramento River Predator Diet Analysis: A Comparative Study. San Francisco Estuary and Watershed Science 18(1).

Strange, J. S. 2010. Upper Thermal Limits to Migration in Adult Chinook Salmon: Evidence from the Klamath River Basin. Transactions of the American Fisheries Society 139: 1091–1108.

Strong, A. L., M. M. Mills, I. B. Huang, G. L. van Dijken, S. E. Driscoll, G. Berg, R. M. Kudela, S. G. Monismith, C. A. Francis, and K. R. Arrigo. 2021. Response of Lower Sacramento River phytoplankton to high-ammonium wastewater effluent. Elementa: Science of the Anthropocene 9(1). DOI: 10.1525/elementa.2021.040.

Swanson, C., T. Reid, P. S. Young, and J. J. Cech Jr. 2000. Comparative environmental tolerances of threatened delta smelt (*Hypomesus transpacificus*) and introduced wakasagi (*H. nipponensis*) in an altered California estuary. Oecologia 123(3):384–390.

Swanson, C., P. S. Young, and J. J. Cech. 2004. Swimming in Two-Vector Flows: Performance and Behavior of Juvenile Chinook Salmon near a Simulated Screened Water Diversion. Transactions of the American Fisheries Society 133(2):265–278.

Suisun Ecological Workgroup. 2001. Suisun Ecological Workgroup Final Report to the State Water Resources Control Board.

Takata, L., T. R. Sommer, J. L. Conrad, and B. M. Schreier. 2017. Rearing and migration of juvenile Chinook salmon (*Oncorhynchus tshawytscha*) in a large river floodplain. Environmental Biology of Fishes 100(9): 1105–1120.

Tamburello, N., B. M. Connors, D. Fullerton, and C. C. Phillis. 2019. Durability of environment–recruitment relationships in aquatic ecosystems: insights from long-term monitoring in a highly modified estuary and implications for management. Limnology and Oceanography, 64(S1):S223–S239.

Tehama-Colusa Canal Authority. 2012. Fish Passage Improvement Project at the Red Bluff Diversion Dam. Brochure published by TCCA. 4pp. https://www.tccanal.com/RBDD-Bro-Sept2012-NoCrop.pdf

The Bay Institute. 1998. From the Sierra to the Sea: The Ecological History of the San Francisco Bay-Delta 7 Watershed. San Francisco, CA: The Bay Institute of San Francisco.

The Nature Conservancy. 2007a. Sacramento River Ecological Flows Study. Gravel Study Final Report.

The Nature Conservancy. 2007b. Linking Biological Responses to River Processes: Implications for Conservation and Management of the Sacramento River—A Focal Species Approach. Final Report.

Thomas, A. E., J. L. Banks, and D. C. Greenland. 1969. Effect of Yolk Sac Absorption on the Swimming Ability of Fall Chinook Salmon. Transactions of the American Fisheries Society 98(3):406–410.

Thomas, M. J., M. L. Peterson, N. Friedenberg, J. P. Van Eenennaam, J. R. Johnson, J. J. Hoover, and A. P. Klimley. 2013. Stranding of Spawning Run Green Sturgeon in the Sacramento River: Post-Rescue Movements and Potential Population-Level Effects. North American Journal of Fisheries Management 33(2): 287–297.

Thomson, J. R., W. J. Kimmerer, L. R. Brown, K. B. Newman, R. Mac Nally, W. A. Bennett, F. Feyrer, and E. Fleishman. 2010. Bayesian Change Point Analysis of Abundance Trends for Pelagic Fishes in the Upper San Francisco Estuary. *Ecological Applications* 20(5):1431–1448.

Tierney, K. B., D. H. Baldwin, T. J. Hara, P. S. Ross, N. L. Scholz, and C. J. Kennedy. 2010. Olfactory Toxicity in Fishes. *Aquatic Toxicology* 96:2–26.

Toft, J. D., C. A. Simenstad, J. R. Cordell, and L. F. Grimaldo. 2003. The Effects of Introduced Water Hyacinth on Habitat Structure, Invertebrate Assemblages, and Fish Diets. Estuaries 26(3): 746–758.

Tucker, M. E., C. M. Williams, and R. R. Johnson. 1998. Abundance, Food Habits, and Life History Aspects of Sacramento Squawfish and Striped Bass at the Red Bluff Diversion Complex, including the Research Pumping Plant, Sacramento River, California: 1994-1996. Red Bluff Fish Passage Program. U.S. Fish and Wildlife Service, Red Bluff, CA.

Tucker, M. E., C. D. Martin, and P. D. Gaines. 2003. Spatial and Temporal Distribution of Sacramento Pikeminnow and Striped Bass at the Red Bluff Diversion Complex, including the Research Pumping Plant, Sacramento River, California: January, 1997 to August, 1998. Red Bluff Research Pumping Plant Report Series. U.S. Fish and Wildlife Service, Red Bluff, CA.

Twardochleb, L., A. Maguire, L. Dixit, M. Bedwell, J. Orlando, M. MacWilliams, A. Bever, and B. Davis. 2021. North Delta Food Subsidies Study: Monitoring Food Web Responses to the North Delta Flow Action, 2019 Report. California Department of Water Resources, Division of Environmental Services.

U.S. Army Corps of Engineers, Bureau of Reclamation, Sacramento Area Flood Control Agency, and California Central Valley Flood Protection Board. 2012. Folsom Dam Modification Project Approach Channel, Draft Supplemental Environmental Impact Statement/ Environmental Impact Report. July.

U.S. Army Engineer Research and Development Center. 2017. *Scenario Analysis of Fremont Weir Notch - Integration of Engineering Designs, Telemetry, and Flow Fields*. Technical memorandum for the Yolo Bypass Salmonid Habitat Restoration and Fish Passage Project. Sacramento, California.

U.S. Environmental Protection Agency. 2010. The SPCC Rule and Recent Amendments Presentation. November. Available: https://www.epa.gov/oil-spills-prevention-and-preparedness-regulations/spill-prevention-control-and-countermeasure-16. Accessed: June 29, 2021.

U.S. Fish and Wildlife Service. 1985. Flow Needs of Chinook Salmon in the Lower American River. Final Report on the 1981 Lower American River Flow Study. Sacramento, CA. Prepared for U.S. Bureau of Reclamation, Sacramento, CA.

U.S. Fish and Wildlife Service. 1993. *Mitigation Policy: 501 FW 2*. Series Interagencies Activities. February.

U.S. Fish and Wildlife Service. 2003a. *Flow-Habitat Relationships for steelhead and fall, late-fall, and winter-run Chinook salmon spawning in the Sacramento River between Keswick Dam and Battle Creek*. February 4, 2003. Sacramento, CA.

U.S. Fish and Wildlife Service. 2003b. *Comparison of PHABSIM and 2-D Modeling of Habitat for Steelhead and Fall-run Chinook Salmon Spawning in the Lower American River*. February 4, 2003. Sacramento, CA.

U.S. Fish and Wildlife Service. 2005a. Flow Habitat Relationships for Fall-run Chinook Salmon Spawning in the Sacramento River between Battle Creek and Deer Creek. Prepared by Staff of Energy Planning and Instream Flow Branch. Sacramento, CA.

U.S. Fish and Wildlife Service. 2005b. Flow-Habitat Relationships for Chinook Salmon Rearing in the Sacramento River between Keswick Dam and Battle Creek. Prepared by Staff of Energy Planning and Instream Flow Branch. Sacramento, CA.

U.S. Fish and Wildlife Service. 2006. Relationships between Flow Fluctuations and Redd Dewatering and Juvenile Stranding for Chinook Salmon and steelhead in the Sacramento River between Keswick Dam and Battle Creek. Prepared by Staff of Energy Planning and Instream Flow Branch. Sacramento, CA.

U.S. Fish and Wildlife Service. 2008. Biological Opinion on the Coordinated Operations of the Central Valley Project and State Water Project in California.

U.S. Fish and Wildlife Service. 2011a. Flow-Habitat Relationships for Fall-Run Chinook Salmon and Steelhead/Rainbow Trout Spawning in Clear Creek between Clear Creek Road and the Sacramento River.

U.S. Fish and Wildlife Service. 2011b. Biological Assessment of Artificial Propagation at Coleman National Fish Hatchery and Livingston Stone National Fish Hatchery: Program Description and Incidental Take of Chinook Salmon and Steelhead. July.

U.S. Fish and Wildlife Service. 2012. Pacific Lamprey (Entospenus tridentatus) Assessment and Template for Conservation Measures in California. U.S. Fish and Wildlife Service. Sacramento, CA.

U.S. Fish and Wildlife Service. 2016. *Central Valley Improvement Act Fisheries Investigations*. Annual Progress Report. Prepared by Staff of Anadromous Fish Restoration Program. Lodi, CA.

U.S. Fish and Wildlife Service. 2019. Biological Opinion for the Reinitiation of Consultation on the Long Term Operation of the Central Valley Project and State Water Project. USFWS Pacific Southwest Region. Sacramento, CA.

U.S. Fish and Wildlife Service. 2020. 700 Winter-Run Chinook Salmon Return to Battle Creek. October 22. Downloaded from https://www.fws.gov/news/ShowNews.cfm?ref=700-winter-run-chinook-salmon-return-to-battle-creek&_ID=36797 on January 28, 2021 (no longer available at that URL).

U.S. Geological Survey. 2023. Dissolved oxygen data for station USGS 11455140 Toe Drain A Liberty Island Nr Courtland CA. Available:

https://nwis.waterdata.usgs.gov/usa/nwis/uv/?cb_00010=on&cb_00060=on&cb_00065=on&cb_00095=on&cb_00300=on&cb_00301=on&cb_00400=on&cb_00480=on&cb_32295=on&cb_32316=on&cb_63160=on&cb_63680=on&cb_63680=on&cb_70969=on&cb_72137=on&cb_72255=on&cb_99133=on&cb_99409=on&format=gif_default&site_no=11455140&legacy=1&period=&begin_date=2018-07-01&end_date=2018-10-31 Accessed: January 10, 2023.

Van Nieuwenhuyse, E. E. 2007. Response of Summer Chlorophyll Concentration to Reduced Total Phosphorus Concentration in the Rhine River (Netherlands) and the Sacramento–San Joaquin Delta (California, USA). *Canadian Journal of Fisheries and Aquatic Sciences* 64:1529–1542.

Verhille, C. E., J. B. Poletto, D. E. Cocherell, B. DeCourten, S. Baird, J. J. Cech, and N. A. Fangue. 2014. Larval Green and White Sturgeon Swimming Performance in Relation to Water-Diversion Flows. *Conservation Physiology* 2(1):cou031.

Vogel, D.A. 2007. 2006 Biological evaluation of the fish screens at the Glenn-Colusa Irrigation District's Sacramento River pump station. Report prepared for the multi-agency Technical Oversight Committee. Natural Resource Scientists, Inc. June 2007. 24 p

Vogel, D. A. 2008. Biological Evaluations of the Fish Screens at the Glenn–Colusa Irrigation District's Sacramento River Pump Station: 2002–2007. Natural Resource Scientists, Inc., Red Bluff, CA.

Vogel, D. A. 2011. Insights into the Problems, Progress, and Potential Solutions for Sacramento River Basin Native Anadromous Fish Restoration. Prepared for Northern California Water Association and Sacramento Valley Water Users.

Vogel, D. 2013. Evaluation of Fish Entrainment in 12 Unscreened Sacramento River Diversions. Final Report. Prepared for CVPIA Anadromous Fish Screen Program (U.S. Fish and Wildlife Service and Bureau of Reclamation) and Ecosystem Restoration Program (California Department of Fish and Wildlife, U.S. Fish and Wildlife Service, and National Marine Fisheries Service).

Vogel, D. A., and K. R. Marine. 1991. Guide to Upper Sacramento River Chinook Salmon Life History. CH2M HILL, Redding, CA.

Vogel, D. A., and K. R. Marine. 1995. 1995 Evaluation of Juvenile Chinook Salmon Transport Timing in the Vicinity of the New Fish Screens at the Glenn-Colusa Irrigation District's Sacramento River Pump Station. November. Natural Resource Scientists, Inc., Red Bluff, CA.

Vroom, J., M. Van der Wegen, R. C. Martyr-Koller, and L. V. Lucas. 2017. What determines water temperature dynamics in the San Francisco Bay-Delta system? *Water Resources Research* 53(11):9901–9921.

Wagner, R. W., M. Stacey, L. R. Brown, and M. Dettinger. 2011. Statistical Models of Temperature in the Sacramento–San Joaquin Delta under Climate-Change Scenarios and Ecological Implications. *Estuaries and Coasts* 34:544–556. DOI 10.1007/s12237-010-9369-z.

Walters, A. W., D. M. Holzer, J. R. Faulkner, C. D. Warren, P. D. Murphy, and M. M. McClure. 2012. Quantifying Cumulative Entrainment Effects for Chinook Salmon in a Heavily Irrigated Watershed. Transactions of the American Fisheries Society 141(5):1180–1190.

Wang, J. C. S. 2006. Early Life History Comparison of the Green Sturgeon, *Acipenser medirostris*, and White Sturgeon, *Acipenser transmontanus*, of the Sacramento-San Joaquin River Delta, California. Tracy Technical Bulletin 2006-1. U.S. Department of the Interior, Bureau of Reclamation, Mid-Pacific Region, Denver, CO.

Water Forum. 2005. Addendum to the Report Titled "Impacts on Lower American River Salmonids and Recommendations Associated with Folsom Reservoir Operations to Meet Delta Water Quality Objectives and Demands." Prepared for City-County Office of Metropolitan Water Planning, Sacramento, CA. Prepared by Surface Water Resources, Inc., Sacramento, CA. September.

Waters, T. 1995. *Sediment in Streams: Sources, Biological Effects, and Control*.

Weston, D. P., J. You, and M. J. Lydy. 2004. Distribution and Toxicity of Sediment-Associated Pesticides in Agriculture-Dominated Water Bodies of California's Central Valley. Environmental Science and Technology 38: 2752–2759.

Weston, D. P., A. M. Asbell, S. A. Lesmeister, S. J. Teh, and M. J. Lydy. 2014. Urban and Agricultural Pesticide Inputs to a Critical Habitat for the Threatened Delta Smelt (*Hypomesus transpacificus*). *Environmental Toxicology and Chemistry* 33:920–929.

Weston, D. P., C. Moschet, T. M. Young, N. Johanif, H. C. Poynton, K. M. Major, R. E. Connon, and S. Hasenbein. 2019. Chemical and Toxicological Effects on Cache Slough after Storm-Driven Contaminant Inputs. *San Francisco Estuary and Watershed Science* 17(3).

Whipple, A. A., R. M. Grossinger, D. Rankin, B. Stanford, and R. A. Askevold. 2012. Sacramento–San Joaquin Delta Historical Ecology Investigation: Exploring Pattern and Process. Prepared for the California Department of Fish and Game and Ecosystem Restoration Program. Historical Ecology Program Publication 672, San Francisco Estuary Institute-Aquatic Science Center, Richmond, California.

Whipple A., T. Grantham, G. Desanker, L. Hunt, A. Merrill, B. Hackenjos, R. Askevold. 2019. Chinook Salmon Habitat Quantification Tool: User Guide (Version 1.0). Prepared for American Rivers. Funded by the Natural Resources Conservation Service Conservation Innovation Grant (#69-3A75-17-40), Water Foundation and Environmental Defense Fund. A report of SFEI-ASC's Resilient Landscapes Program, Publication #953. San Francisco Estuary Institute, Richmond, CA.

Wilkerson, F. P., R. C. Dugdale, V. E. Hogue, and A. Marchi. 2006. Phytoplankton Blooms and Nitrogen Productivity in San Francisco Bay. *Estuaries and Coasts* 29:401–416.

Williams, J. G. 2006. Central Valley Salmon: A Perspective on Chinook and Steelhead in the Central Valley of California. San Francisco Estuary and Watershed Science 4.

Windell, S., P. L. Brandes, J. L. Conrad, J. W. Ferguson, P. A. L. Goertler, B. N. Harvey, J. Heublein, J. A. Israel, D. W. Kratville, J. E. Kirsch, R. W. Perry, J. Pisciotto, W. R. Poytress, K. Reece, B. G. Swart, and R. C. Johnson. 2017. Scientific Framework for Assessing Factors Influencing Endangered Sacramento River Winter-Run Chinook Salmon (*Oncorhynchus tshawytscha*) across the Life Cycle. NOAA Technical Memorandum NMFS-SWFSC-586. NOAA National Marine Fisheries Service, Southwest Fisheries Science Center Fisheries Ecology Division Santa Cruz, CA.

Wood, M. L., P. W. Morris, J. Cooke, and S. J Louie. 2010. Amendments to the Water Quality Control Plan for the Sacramento River and San Joaquin River Basins for the Control of Methylmercury and Total Mercury in the Sacramento-San Joaquin Delta Estuary. Staff Report. California Central Valley Regional Water Quality Control Board, Rancho Cordova, California.

Wright, S. A., and D. H. Schoellhamer. 2004. Trends in the Sediment Yield of the Sacramento River, California, 1957–2001. San Francisco Estuary and Watershed Science 2(2).

Wright, S. A., and D. H. Schoellhamer. 2005. Estimating sediment budgets at the interface between rivers and estuaries with application to the Sacramento-San Joaquin River Delta. Water Resources Research 41(9):W09428.

Wyman, M.T., M.J. Thomas, R.R. McDonald, A.R. Hearn, R.D. Battleson, E.D. Chapman, P. Kinzel, J.T. Minear, E.A. Mora, J.M. Nelson, M.D. Pagel, and A.P. Klimley. 2018. Fine-scale habitat selection of green sturgeon (*Acipenser medirostris*) within three spawning location in the Sacramento River, California. Canadian Journal of Fisheries and Aquatic Sciences 75:779–791.

Yates, E. B. 1989. *Water Quality and Supply on Cortina Rancheria, Colusa County, California*. Water-Resources Investigations Report 89-4004. U.S. Geological Survey. Sacramento, CA.

Yoshiyama, R. M., E. R. Gerstung, F. W. Fisher, and P. B. Moyle. 1996. Historical and Present Distribution of Chinook Salmon in the Central Valley Drainage of California in Sierra Nevada Ecosystem Project. Final Report to Congress. Volume III: Assessments, Commissioned Reports, and Background Information. University of California, Davis, Centers for Water and Wildland Resources.

Yoshiyama, R. M., E. R. Gerstung, F. W. Fisher, and P. B. Moyle. 2001. Historical and Present Distribution of Chinook Salmon in the Central Valley Drainage of California. Contributions to the Biology of Central Valley Salmonids. California Department of Fish and Game Fish Bulletin 179(1): 71–176.

Zajanc, D., S. H. Kramer, N. Nur, and P. A. Nelson. 2013. Holding Behavior of Chinook Salmon (*Oncorhynchus tshawytscha*) and Steelhead (*O. mykiss*) Smolts, as Influenced by Habitat Features of Levee Banks, in the Highly Modified Lower Sacramento River, California. Environmental Biology of Fishes 96(2-3):245–256.

Zeug, S. C. and B. J. Cavallo. 2013. Influence of Estuary Conditions on the Recovery Rate of Coded-Wire-Tagged Chinook Salmon (*Oncorhynchus tshawytscha*) in an Ocean Fishery. Ecology of Freshwater Fish 22(1):157–168.

Zydlewski, G. B., and J. R. Johnson. 2002. Response of Bull Trout Fry to Four Types of Water Diversion Screens. North American Journal of Fisheries Management 22(4):1276–1282.

11.5.2. Personal Communications

Beccio, Marc. Environmental Scientist. California Department of Fish and Wildlife, Rancho Cordova, CA. August 12, 2021. Email with unpublished relative abundance of juvenile green sturgeon in the San Francisco Bay Delta Estuary, provided to Monique Briard and Sophie Unger, ICF, Sacramento CA.

Kapla, James. Principal Project Manager, Jacobs, Bellevue, WA. May 21, 2019—Email describing potential Delevan discharge velocity and infrastructure requirements provided to Marin Greenwood, Aquatic Ecologist, ICF, Sacramento, CA.

Kindopp, Jason. Senior DWR Environmental Scientist. California Department of Water Resources. April 23, 2021—Email to Sophie Unger, Senior Fish Biologist, ICF, Sacramento, CA.

Kline, Phil. IT Support. Glenn-Colusa Irrigation District, Willows, CA. March 14, 2019—Glenn-Colusa Irrigation District data provided to John Spranza, Senior Ecologist/Regulatory Specialist, HDR, Sacramento, CA, via Google documents link.

Michel, Cyril. Assistant Project Scientist. University of California, Santa Cruz; affiliated with Southwest Fisheries Science Center – Fisheries Ecology Division, National Marine Fisheries Service, Santa Cruz, CA. October 26, 2020—Comment during Sites Joint Aquatic Workshop #1.

Perry, Russell. Research Fisheries Biologist, Quantitative Fisheries Ecology Section, USGS Western Fisheries Research Center, Columbia River Research Laboratory, Cook, WA. June 18, 2019—Email containing Excel file <North Delta Routing Management Tool v2.1.xlsx> sent to Marin Greenwood, Aquatic Ecologist, ICF, Sacramento, CA.

Seesholtz, Alicia. Staff Environmental Scientist. California Department of Water Resources, Sacramento, CA. February 2021—Phone conversations with Rick Wilder, Senior Environmental Scientist, ICF, Sacramento, CA.