UNITED STATES OF AMERICA
BEFORE THE
FEDERAL ENERGY REGULATORY COMMISSION

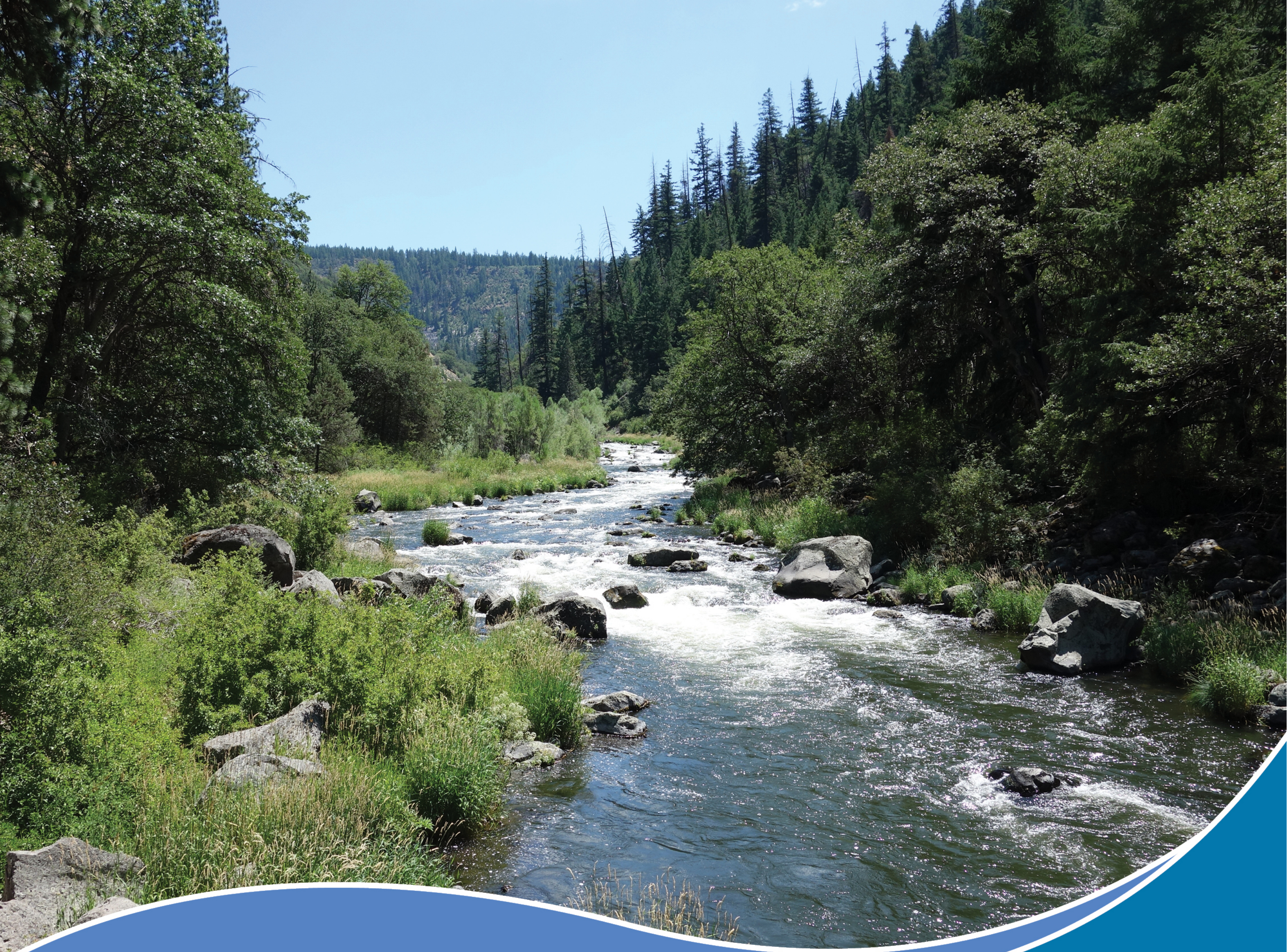
Klamath River Renewal Corporation	**Project Nos. 14803-001;**
PacifiCorp	**2082-063**

AMENDED APPLICATION FOR SURRENDER OF LICENSE FOR MAJOR PROJECT AND REMOVAL OF PROJECT WORKS

Attachment A-8

Lower Klamath Project Biological Assessment

Appendix G
(Species Accounts)



Biological Assessment

Appendix G - Species Accounts

KLAMATH
RIVER RENEWAL
CORPORATION

March 2021

Prepared for:

Klamath River Renewal Corporation

Prepared by:

Renewal Corporation Technical Representatives:

AECOM Technical Services, Inc.
300 Lakeside Drive, Suite 400
Oakland, California 94612

CDM Smith
1755 Creekside Oaks Drive, Suite 200
Sacramento, California 95833

River Design Group
311 SW Jefferson Avenue
Corvallis, Oregon 97333

Table of Contents

List of Tables

List of Figures

Acronyms and Abbreviations

AFA	Aphanizomenon flos-aquae (a nitrogen-fixing cyanobacteria)
BA	Biological Assessment
BLM	Bureau of Land Management
California HSRG	California Hatchery Scientific Review Group
CDFG	California Department of Fish and Game (now CDFW)
CDFW	California Department of Fish and Wildlife
CDWR	California Department of Water Resources
CESA	California Endangered Species Act
CFR	Code of Federal Regulations
CHSU	critical habitat subunit
CNDDB	California Natural Diversity Database
CWA	Clean Water Act
DPS	Distinct Population Segment
EPA	United States Environmental Protection Agency
ESA	federal Endangered Species Act
ESU	Evolutionarily Significant Unit
FERC	Federal Energy Regulatory Commission
FR	Federal Register
HCP	Habitat Conservation Plan
ISAB	Independent Scientific Advisory Board
KPO	Klamath Project Operations
KRRC	Klamath River Renewal Corporation
LRS	Lost River suckers
LWD	large woody debris
mg	milligram
mg/L	milligram per liter
MSNO	Master Site Number
NCRWQCB	North Coast Regional Water Quality Control Board
nDPS	Northern Distinct Population Segment
NMFS	National Marine Fisheries Service
NRC	National Research Council
NSO	northern spotted owl
NWFSC	Northwest Fisheries Science Center
NWR	National Wildlife Refuge
ODFW	Oregon Department of Fish and Wildlife

ODEQ	Oregon Department of Environmental Quality
OSF	Oregon spotted frog
PBF	Physical and Biological Feature
sDPS	Southern Distinct Population Segment
SNS	shortnose suckers
SONCC	Southern Oregon Northern California Coast
TMDL	total maximum daily load
USBR	United States Bureau of Reclamation
USDA	United States Department of Agriculture
USFS	United States Forest Service
USFWS	United States Fish and Wildlife Service
USGS	United States Geological Survey
WDFW	Washington Department of Fish and Wildlife
YOY	young of the year
YTEP	Yurok Tribe Environmental Program

G. SPECIES ACCOUNTS

This appendix provides detailed information for each species covered in the BA. References cited in this appendix are listed in Section 8 of the BA.

G.1 NMFS Species

G.1.1 SONCC Coho Salmon (*Oncorhynchus kisutch*)

G.1.1.1 Species status

The Southern Oregon – Northern California Coastal (SONCC) coho salmon ESU was listed as threatened under the ESA on May 6, 1997 (62 FR 24588). The SONCC coho salmon ESU includes all natural-origin populations of coho salmon in coastal streams between Cape Blanco, Oregon, and Punta Gorda, California. The SONCC coho salmon ESU includes the Klamath River drainage up to Spencer Creek.

Three artificial propagation programs are considered to be part of the ESU: the Cole Rivers Hatchery, Trinity River Hatchery, and Iron Gate Hatchery (NMFS 2001). NMFS has determined that these artificially propagated stocks are no more divergent relative to the local natural-origin populations than what would be expected between closely related natural-origin populations in the ESU (70 FR 37160; June 28, 2005). An updated review of these hatchery programs indicates that all three continue to be operational, and that no substantial changes in their management have been implemented since the last status review that would increase their divergence from natural populations. Based on the updated information, all three programs continue to propagate fish that are considered part of the SONCC coho salmon ESU (NMFS 2016a).

Coho salmon in the Klamath Basin have also been listed by the California Fish and Game Commission as threatened under the California ESA (CESA) (CDFG 2002b).

G.1.1.2 Critical habitat

Critical habitat was designated for SONCC coho salmon in May 1999 (64 CFR § 24049). Critical habitat includes all river reaches accessible to listed coho salmon between Cape Blanco, Oregon and Punta Gorda, California, and includes water, substrate, and adjacent riparian zones of estuarine and riverine reaches, including off-channel habitat. Accessible reaches are defined as those within the historical range of the ESU that can still be occupied by any life stage of coho salmon. Specifically, in the Klamath Basin, all river reaches downstream of Iron Gate Dam on the Klamath River and Lewiston Dam on the Trinity River are designated as critical habitat (64 CFR Section 24049; May 5, 1999). Excluded are: (1) areas upstream of specific dams identified in the FR notice; (2) areas upstream of longstanding natural impassible barriers (i.e., natural waterfalls); and (3) tribal lands. The physical and biological features (PBFs) of habitat considered essential for the conservation of the SONCC ESU include: 1) spawning sites, 2) food resources, 3) water quality and quantity, and 4) riparian vegetation (62 CFR Section 62741, November 1997).

G.1.1.3 Life history

Coho salmon have an anadromous life history in which juveniles are born and rear in freshwater, migrate to the ocean, grow to maturity, and return to freshwater as adults to spawn. Coho salmon adults migrate upstream from September through late December, peaking in October and November. Spawning occurs mainly in November and December, with fry emerging from the gravel in the spring, approximately 3 to 4 months after spawning. Coho salmon tend to spawn in small streams that flow directly into the ocean, or tributaries and headwater creeks of larger rivers (Moyle 2002, Sandercock 1991). Juveniles may spend 1 to 2 years rearing in freshwater (Bell and Duffy 2007) or emigrate to an estuary shortly after emerging from spawning gravels (Tschaplinski 1988). Coho salmon juveniles are also known to redistribute into non-natal rearing streams, lakes, or ponds; often following rainstorms, where they continue to rear (Peterson 1982). Emigration from streams to the estuary and ocean generally takes place from February through June, with the peak period being the end of April through May. The majority of coho salmon in the Klamath River have a 3-year life cycle, with their time being spent about equally between fresh- and saltwater. Some 2-year-old males, known as “jacks,” also return as spawners. Juveniles typically rear in freshwater for 1 full year, then migrate to the sea in the spring after their first winter of life.

G.1.1.4 Geographic distribution

Coho salmon are native to the Klamath Basin. Williams et al. (2006) described nine historical coho salmon populations in the Klamath Basin, including the Upper Klamath River, Shasta River, Scott River, Salmon River, Mid-Klamath River, Lower Klamath River, and three population units in the Trinity River watershed (upper Trinity River, lower Trinity River, and South Fork Trinity River). All nine of these populations occur in the Action Area, including the Upper Klamath River (composed of tributaries and mainstem Klamath River from the mouth of Portuguese Creek upstream to Iron Gate Dam, excluding the Shasta and Scott Rivers); the Middle Klamath River (composed of tributaries and mainstem Klamath River from the Trinity River confluence upstream to the mouth of Portuguese Creek, excluding the Salmon River); the Lower Klamath River (composed of tributaries and mainstem Klamath River downstream of the Trinity River confluence to the Klamath River mouth); the Salmon River; the Scott River; and the Shasta River.

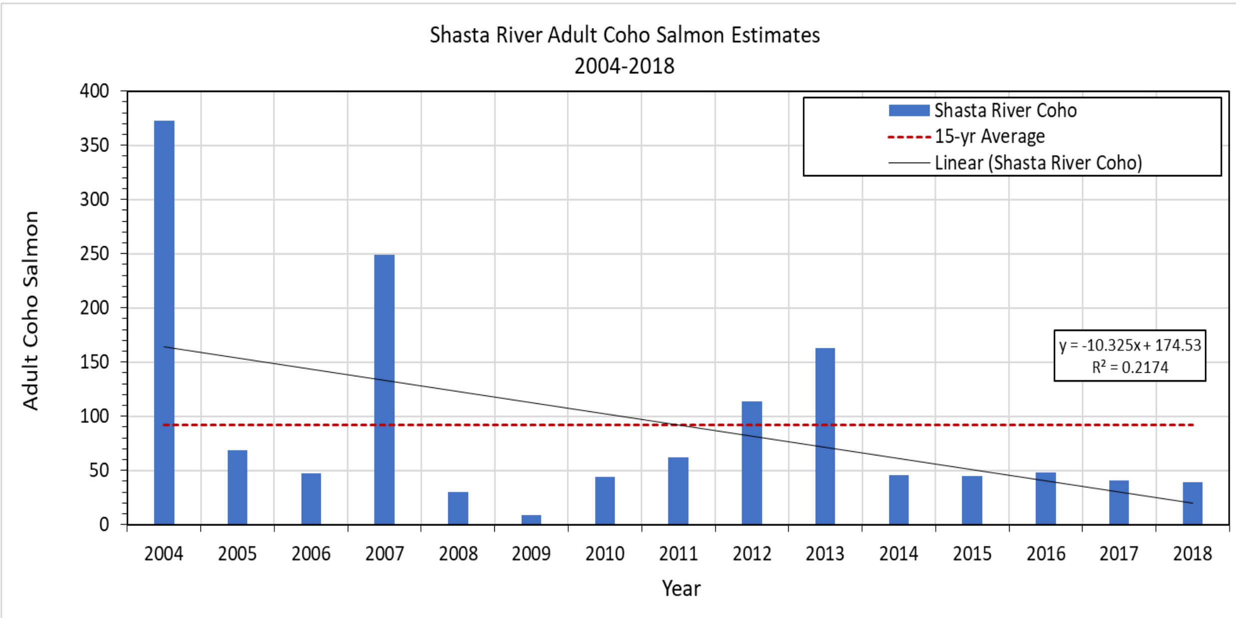
Coho salmon are distributed throughout the Klamath River downstream of Iron Gate Dam, and spawn primarily in tributaries (Trihey and Associates 1996, NRC 2004). During their upstream migration, adult coho salmon from the Upper Klamath River Population Unit may travel upstream as far as Iron Gate Dam. Coho salmon were once numerous and widespread in the Klamath River basin (Snyder 1931) and were formerly known to occupy mainstem and tributary habitat at least as far upstream as Spencer Creek (NRC 2004). The PacifiCorp Hydroelectric Project, of which Iron Gate Dam is the lowest of four mainstem dams, blocks access to approximately 76 miles of spawning, rearing, and migratory habitat for SONCC coho salmon (USBR and CDFW 2012).

Coho salmon use the mainstem Klamath River for some or all their life history stages (spawning, rearing, and migration). However, the majority of returning adult coho salmon spawn in the tributaries to the mainstem (Magneson and Gough 2006, NMFS 2010a). Some fry and age-0+ juveniles enter the mainstem in the spring and summer following emergence (Chesney et al. 2009). Large numbers of age-0 juveniles from tributaries in the mid-Klamath River move into the mainstem in the fall (October through November) (Soto et al. 2008; Hillemeier et al. 2009). Juvenile coho salmon have been observed to move into non-natal rearing streams, off-channel ponds, the Lower Klamath River, and the estuary for overwintering (Soto et al. 2008; Hillemeier et al. 2009). Some proportion of juveniles generally remain in their natal tributaries to rear. Rearing has also been observed in tributary confluence pools in the mainstem Klamath River (NRC 2004).

G.1.1.5 Population trends in the ESU

The following section is largely taken or adopted from the 2016 5-Year Review: Summary and Evaluation of Southern Oregon/ Northern California Coast Coho Salmon (NMFS 2016a), because this document contains a summary of recently gathered information and is at the time of this writing the most up-to-date official status review of the coho populations in the SONCC ESU. The next 5-year status review is scheduled to be released in 2021.

SONCC coho salmon have declined substantially from historical levels. Quantitative population-level estimates of adult spawner abundance spanning more than 9 to 12 years are scarce for independent or dependent populations of coho salmon in the SONCC ESU. Monitoring in California has improved considerably since the 2011 viability assessment because of the implementation of the Coastal Monitoring Plan (CMP) across the California portion of the ESU. Currently in California, seven independent populations are currently monitored at the “population unit” scale. Most of this monitoring produces estimates of adult escapement based on random subsampling in the population area. In contrast, video weir counts from the Shasta and Scott rivers are not based on estimates. In these locations, the actual numbers of adult fish passing a video weir are counted. Currently, only the Shasta River video weir counts meets the minimum duration to assess under the viability criteria. Of great concern is the extremely low number of fish passing the weir in 2014 (46 coho salmon), which is less than the depensation threshold of 144 fish (NMFS 2014), and that only four of those fish were considered to be 3-year-olds (Chesney and Knechtle 2015). The Shasta River count is now 17 years in duration (5+ generations), and from this time series, a decline is apparent, particularly with the low numbers (less than 50) crossing the weir in each of the last 5 years (Figure G-1).



Klamath River Renewal Corporation, Klamath River Renewal Project

Figure G-1: Estimates of Adult Coho Salmon in the Shasta River for 2004 to 2018 (Giudice and Knechtle 2019a)

Video weir counts of adult coho in the Shasta and Scott rivers represent the longest-term population-unit spatial scale monitoring currently underway in the SONCC coho salmon ESU. Although long-term data on coho abundance in the SONCC coho salmon ESU are scarce, all evidence from trends since an earlier 2011 assessment (Williams et al. 2011a) indicates little change. Many independent populations are likely well below low-risk abundance targets based on the limited data available, and several are likely below the high-risk depensation thresholds specified by the NMFS Technical Recovery Team and the Recovery Plan (NMFS 2014). Although population-level estimates of abundance for most independent populations are lacking, it does not appear that any of the seven diversity strata currently supports a single viable population as defined by the Technical Recovery Team's viability criteria, although all diversity strata are occupied. Further, 24 out of 31 independent populations are at high risk of extinction, and six are at moderate risk of extinction (NMFS 2016a).

G.1.1.6 Threats

Stresses are the physical, biological, or chemical conditions and associated ecological processes that may be impeding SONCC coho salmon recovery. General categories of stresses include water quality, competition, disease, access to habitat, instream flows, insufficient quality and quantity of physical habitat, and predation. Threats are activities or impacts that cause or contribute to the stresses that limit recovery of the species, including water diversions, hydropower impacts, land management, invasive species, fish harvest management, and hatchery management. Table G-1 includes a matrix of interrelated threats and stresses that are currently affecting populations of coho salmon in the SONCC ESU. For a comprehensive narrative on these stresses and threats, please see the Final Recovery Plan for SONCC ESU of Coho Salmon (NMFS 2014).

G.1.1.7 Status of Populations in the Action Area

Populations of coho salmon in the SONCC ESU that are expected to be potentially affected by the proposed Action Area include the Upper Klamath River, Middle Klamath River, Lower Klamath River, Shasta River, Scott River, Salmon River, Lower Trinity River, Upper Trinity River, and South Fork Trinity River populations. These nine populations are part of three diversity strata, including the Central Coastal, Interior Klamath, and Interior Trinity. None of the nine populations of coho salmon that could be potentially be affected by the Proposed Action are considered viable (NMFS 2013). Even the most optimistic estimates from Ackerman et al. (2006) indicate each population falls well short of abundance thresholds for the proposed viability criteria that, if met, would suggest that the populations were at low risk of extinction for this specific criterion.

Regarding spatial structure and diversity, Williams et al. (2008) abundance thresholds were based on estimated historical distribution and abundance of spawning coho salmon, and thereby capture the essence of these two viability parameters. By not meeting the low-risk annual abundance threshold, all Klamath River coho salmon populations are likewise failing to meet spatial structure and diversity conditions consistent with viable populations. Several of these populations have also recently failed to meet the high-risk abundance thresholds, underscoring the critical nature of recent low adult returns (NMFS 2013). Six of the nine populations in the Action Area are considered at high risk of extinction, and three are considered at moderate risk of extinction.

Recent abundance estimates are not available for all populations of coho salmon in the Action Area. However, estimates of adult coho salmon in the Action Area that are available are all reduced from historic numbers, and are all estimated to be below the viability threshold each year since 2009 (Table G-2).

Table G-1: Threats and Stresses Affecting Populations of Coho Salmon in the SONCC ESU

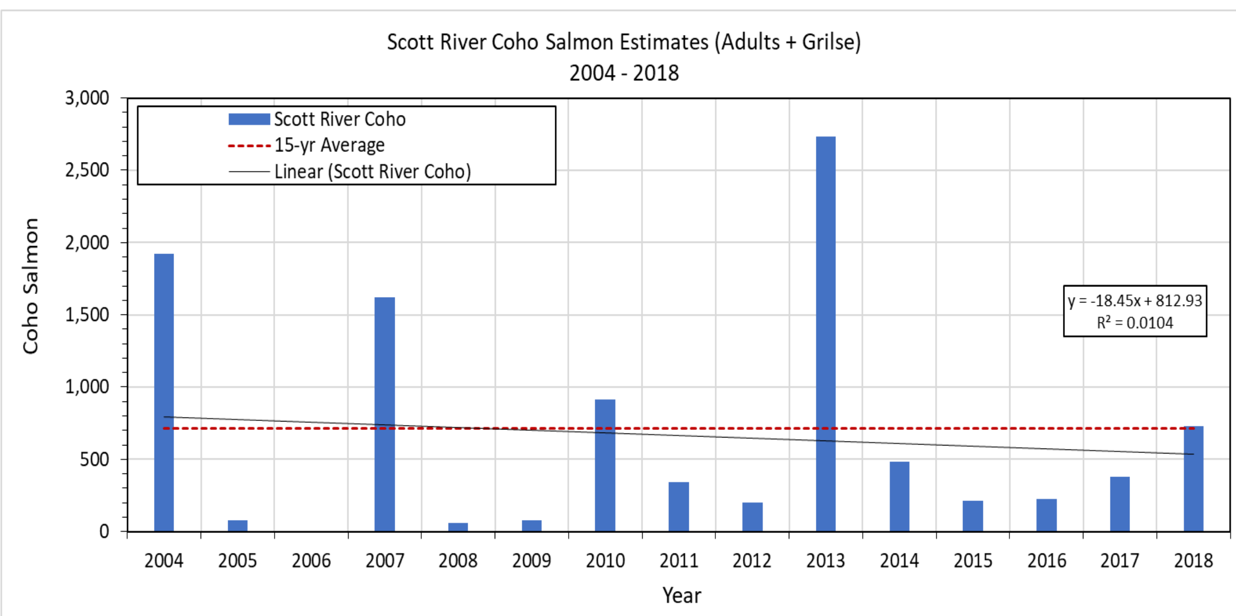
	Stresses									
Threats	Adverse Hatchery-Related Effects	Impaired Water Quality	Degraded Riparian Forest Conditions	Increased Disease/ Predation/ Competition	Altered Sediment Supply	Lack of Floodplain/ Channel Structure	Altered Hydrologic Function	Barriers	Adverse Fishery and Collecting-Related Effects	Impaired Estuary/ Mainstem Function
Climate Change		X	X	X	X	X	X			X
Roads		X	X		X	X	X	X		X
Channelization/ Diking		X	X		X	X	X			X
Agricultural Practices		X	X		X	X	X	X		X
Timber Harvest		X	X		X	X	X	X		X
Urban/Residential/ Industrial Development		X	X		X	X	X	X		X
High-Severity Fire		X	X		X		X			
Mining/Gravel Extraction		X	X		X	X	X	X		X
Dams/Diversions		X	X	X	X	X	X	X		X
Fishing and Collecting									X	
Invasive/ on- Native/ Alien Species				X						X
Hatcheries	X			X						

Table G-2: Estimated naturally spawning and hatchery returning coho salmon abundance estimates for populations where data are available

Stratum	Population or Subset	Year 2009	2010	2011	2012	2013	2014	2015	2016	2017	2018
Interior Klamath	Upper Klamath	<200	<350	<300	<300	<300	<300	<300	<300	<300	<300
	Bogus Creek	7	154	143	185	446	97	14	85	48	47
	Iron Gate Hatchery	70	485	586	644	1,268	384	72	86	122	200
	Shasta River	9	44	62	114	163	46	45	48	41	39
	Scott River	80	918	358	199	2,644	504	290	250	368	727
	Middle Klamath	<1,500	<1,500	<1,500	<1,500	<1,500	<1,500	<1,500	<1,500	<1,500	<1,500
Interior Trinity	Trinity River upstream of Willow Creek Weir	6,396	7947	15,040	18,657	21,906	13,537	4,619	1,325	655	1,486
	Trinity River Hatchery	3,351	4,425	4,810	8,236	6,631	3,908	3,337	527	420	742

Data Sources: Upper Klamath – NMFS 2019a; Iron Gate Hatchery – Giudice and Knechtle 2019a; Shasta River – Giudice and Knechtle 2019b; Scott River – Knechtle and Giudice 2019; Trinity River and TRH – Kier et al. 2019.

In recent years, the highest recorded escapement of adult coho salmon in the Interior Klamath stratum has been to the Scott River sub-basin. The run was relatively high in 2013 (2,644 fish) in comparison to that observed in other years (80 – 918 fish) (Figure G-2).

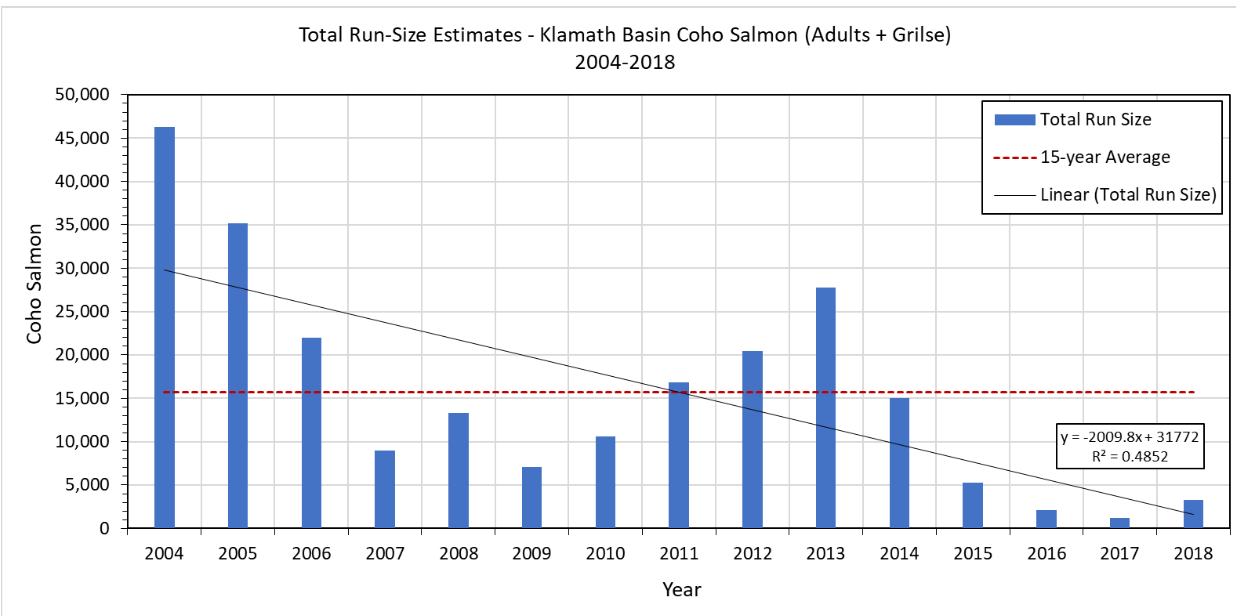


Klamath River Renewal Corporation, Klamath River Renewal Project

Figure G-2: Estimates of Coho Salmon (Adults and Grilse) in the Scott River for 2004 to 2018 (Knechtle and Giudice 2019)

Escapement of coho salmon entering Bogus Creek is monitored by the CDFW annually since about 2004. Over that period, the number of adult coho salmon estimated to have entered Bogus Creek has ranged between 7 fish (2009) and 446 fish (2013) (Table G-2), and the proportion of hatchery coho present in the run has ranged between 0.22 (2017) and 0.88 (2012). Since 2014, the total number of adult coho salmon observed has been less than 100 fish, and the numbers appear to be decreasing over time (Knechtle and Giudice 2018).

Preliminary data available in CDFW's draft coho "megatable" also provides some additional context to recent population trends of SONCC coho in the Klamath Basin. Estimates for the total run size of naturally and hatchery produced coho salmon for the Klamath Basin between 2004 and -2018 have ranged from a maximum of 46,302 (2004) to a minimum of 1,243 (2017) (CDFW 2019a; Figure G-3.

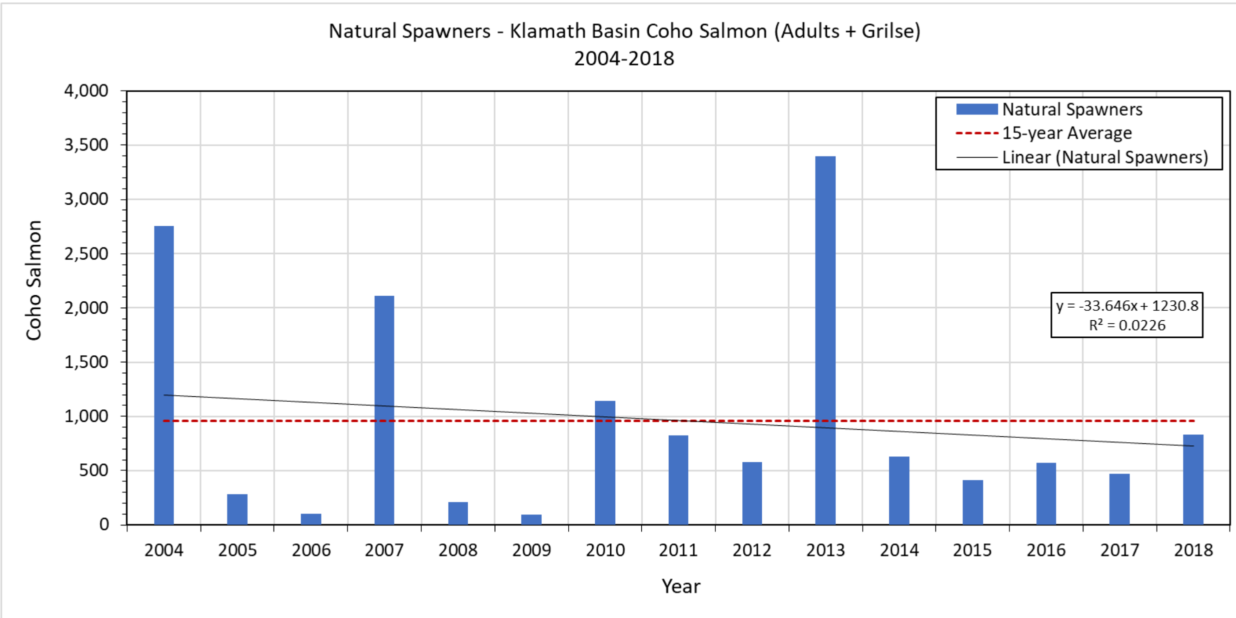


Klamath River Renewal Corporation, Klamath River Renewal Project

Figure G-3: Total Run Size Estimate for Klamath Basin Coho Salmon (Adults and Grilse) from 2004 to 2018 (CDFW 2019a)

Estimates of natural spawners in the Klamath River and select tributaries between 2004 and 2018 show the variability between different year classes, and illustrate how one brood year class (2004, 2007, 2010, 2013) is typically stronger than the other two-year classes (Figure G-4). This is also consistent with counts in the Shasta and Scott rivers (Figure G-1 and Figure G-2). Estimates of naturally spawned coho salmon in the Klamath River are based on the sum of various monitoring surveys that include the mainstem Klamath River, Salmon River basin, Scott River basin, Shasta River basin, Bogus Creek, and miscellaneous Klamath River tributaries downstream of the Yurok Reservation (CDFW 2019a). Estimates of total run size and

Klamath natural spawners are not representative of an actual population estimate for all Klamath River coho, but they are useful in providing historical context and determining trends in abundance.

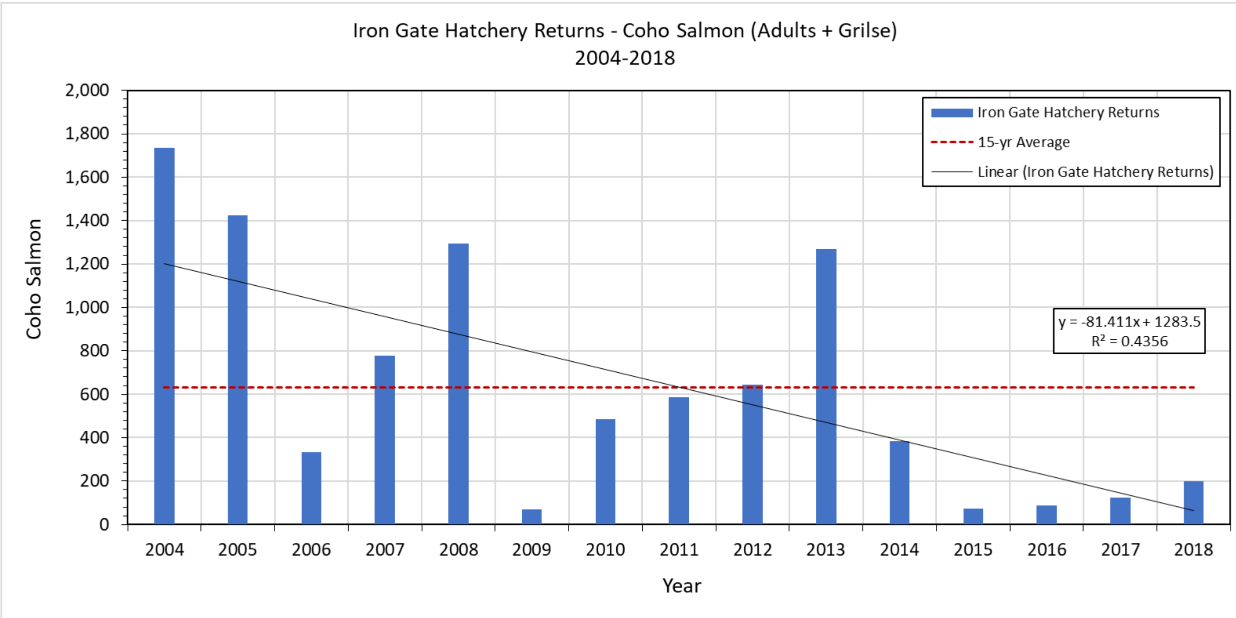


Klamath River Renewal Corporation, Klamath River Renewal Project

Figure G-4: Estimates for Coho Salmon (Adults and Grilse) Natural Spawners in the Mainstem Klamath River and Selected Tributaries from 2004 to 2018 (CDFW 2019a)

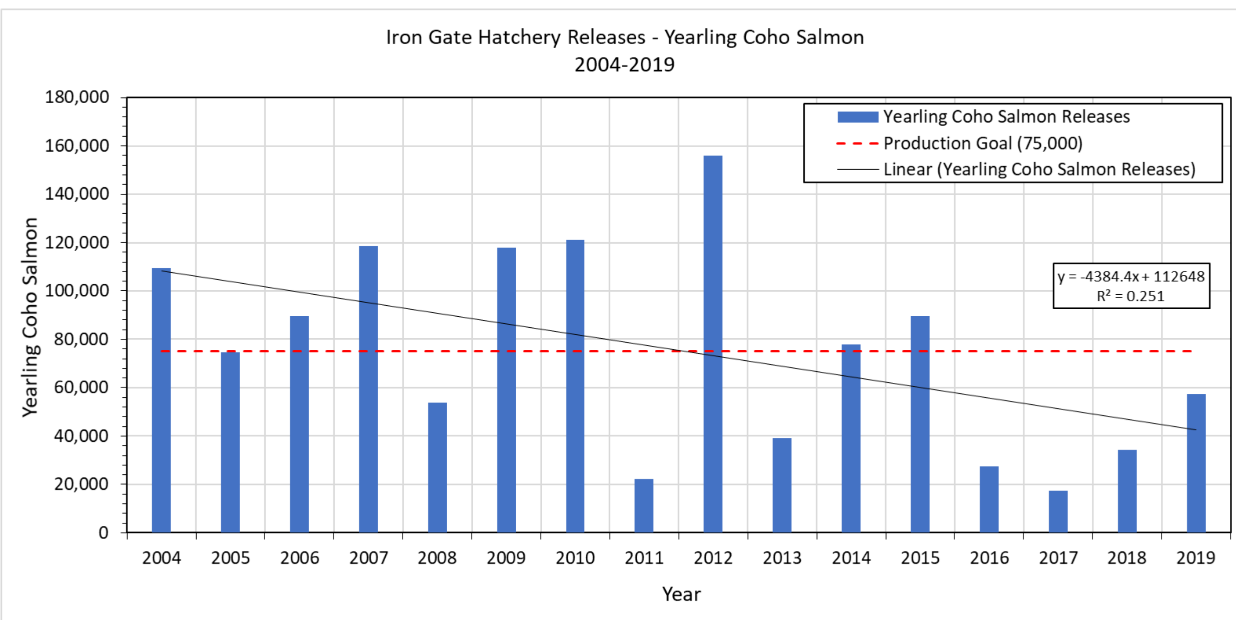
Hatchery coho production at Iron Gate Hatchery provides additional context to the status of populations in the Klamath River. The Iron Gate Hatchery coho program was initiated in the late 1960s to mitigate for impacts resulting from the construction of Iron Gate Dam, and currently operates to produce a program goal of 75,000 yearling coho salmon (California HSRG 2012). The program currently operates under a Hatchery Genetics Management Plan (HGMP) finalized in 2014 to protect and conserve the genetic resources of the Upper Klamath River coho population unit (CDFW and PacifiCorp 2014).

Returns of coho salmon to Iron Gate Hatchery between 2004 and 2018 have ranged from a maximum of 1,734 (2004) to a minimum of 70 in 2009, with a 15-year average of 632 (Figure G-5). The count of hatchery coho includes adult and grilse (reproductively mature after 1 ocean year) salmon. Recent returns have showed a downward trend, with the most recent 5-year average of 173. Similarly, releases of yearling coho salmon from hatchery production at Iron Gate Hatchery between 2004 and 2018 show a downward trend, and the hatchery has only met coho production goals in 1 out of the last 5 years (see Figure G-6).



Klamath River Renewal Corporation, Klamath River Renewal Project

Figure G-5: Returns of Coho Salmon (Adults and Grilse) to the Iron Gate Hatchery from 2004 to 2018 (Giudice and Knechtle 2019b)



Klamath River Renewal Corporation, Klamath River Renewal Project

Figure G-6: Yearling Coho Salmon Releases from the Iron Gate Hatchery from 2004 to 2018 (Giudice and Knechtle 2019b)

G.1.1.8 Abundance and Seasonal Distribution in the Action Area

Much of the following information is directly adopted from NMFS' 2019 BO on the continued operation of the Klamath Project (NMFS 2019a), because this document represents the most updated summary of information on populations of coho salmon in the Action Area.

Upper Klamath River Population

The Upper Klamath River population is currently composed of approximately 64 miles of mainstem habitat and numerous tributaries to the mainstem Klamath River upstream of Portuguese Creek to Iron Gate Dam. Historically, the population extended upstream of Iron Gate Dam to Spencer Creek. The PacifiCorp Hydroelectric Project, of which Iron Gate Dam is the lowest of four mainstem dams, blocks access to approximately 76 miles of spawning, rearing, and migratory habitat for SONCC coho salmon (USBR and CDFW 2012). As a result, coho salmon in the Upper Klamath River population spawn and rear primarily in several of the larger tributaries between Portuguese Creek and Iron Gate Dam; namely, Bogus, Horse, Beaver, and Seiad creeks (NMFS 2016a).

Coho salmon in the Upper Klamath River population spawn and rear primarily in several of the larger tributaries between Portuguese Creek and Iron Gate Dam, including Horse and Seiad creeks. Coho salmon presence was confirmed in six surveyed tributary streams in or near the Project Action Area, including Horse, Seiad, Grider, West Grider, Walker, and O'Neil creeks (Garwood 2012). In surveys from 2014 to 2017, KNF fisheries staff routinely observed hundreds of YOY juvenile coho salmon in lower Horse and Seiad creeks (NMFS 2014).

Due to the low demographics of the Upper Klamath River population, Iron Gate Hatchery coho salmon strays are currently an important component of the adult returns for these populations because of their role in increasing the likelihood that wild/natural coho salmon find a mate and successfully reproduce.

Middle Klamath River Population

Few data on adult coho are available for this stretch of river. Adult spawning surveys and snorkel surveys have been conducted by the US Forest Service and Karuk Tribe, but data from those efforts are insufficient to draw definitive conclusions on run sizes (Ackerman et al. 2006). Ackerman et al. (2006) relied on professional judgment of local biologists to determine what run sizes would be in high, moderate, and low return years to these tributaries; therefore, the run size approximations are judgment-based estimates. NMFS (2014) identifies that the Middle Klamath River population is at moderate risk of extinction.

Most of the juveniles observed in the Middle Klamath have been in the lower parts of the tributaries, which suggests many of these fish are non-natal rearing in these refugial areas. Adults and juveniles appear to be well-distributed throughout the Middle Klamath; however, use of some spawning and rearing areas is restricted by water quality, flow, and sediment issues. Although its spatial distribution appears to be good, many of the Middle Klamath tributaries are used for non-natal rearing, and too little is known to infer its extinction risk based on spatial structure (NMFS 2019a).

Shasta River Population

Adult coho salmon returns to the Shasta River have generally been in decline over the last decade. Since 2007, the number of adult coho observed entering the Shasta River has ranged from a high of 249 fish in 2007 to a low of only 9 fish in 2009 (Giudice and Knechtle 2019b). From 2014 through 2018, the number of adult coho salmon has been less than 50 fish annually. To reduce the risk of demographic extinction, all Iron Gate Hatchery surplus adult coho salmon have been released back to the Klamath River since 2010. Some of these surplus adults have been observed entering the Shasta River, which is about 14 river miles downstream of Iron Gate Hatchery. Since that time, the percentage of hatchery-origin coho salmon observed in the Shasta River spawning population has ranged from about 25 percent to 80 percent. Due to the low demographics of the Shasta River population, Iron Gate Hatchery origin fish play an important role in increasing the likelihood that wild/natural coho salmon find a mate and successfully reproduce. The portion of hatchery-origin adults in the spawning population is unknown for the most recent 4 years (2015 to 2018), because sampling efforts were unable to recover any adult carcasses during this time (Giudice and Knechtle 2019b).

The current distribution of coho salmon spawners is concentrated in the mainstem Shasta River, Big Springs Creek, lower Parks Creek, and in the Shasta River Canyon. Juvenile rearing is also occurring in these same areas (NMFS 2014).

Scott River Population

Abundance estimates on the Scott River are also relatively robust due to the presence of a video fish counting weir (Knechtle and Giudice 2019). Since 2007, a video weir was placed in the Scott River, alleviating concerns about data collection methods. In 2017 and 2018, 368 and 727 adult coho salmon were estimated to have returned to the river, respectively. Spawning activity and redds have been observed in the East Fork Scott River, South Fork Scott River, Sugar, French, Miners, Etna, Kidder, Patterson, Shackleford, Mill, Canyon, Kelsey, Tompkins, and Scott Bar Mill creeks. Fish surveys of the Scott River and its tributaries have been occurring since 2001. These surveys have documented that many of the tributaries do not consistently sustain juvenile coho salmon, indicating that the spatial structure of this population is restricted by available rearing habitat. Many of these tributaries likely have intermittent fish occupation due to low-flow barriers for juvenile and adult migration periods, as described in the previous sections. Juvenile fish have been found rearing in the mainstem Scott River, East Fork Scott River, South Fork Scott River, Shackleford Creek and its tributary Mill Creek, Etna Creek, French Creek and its tributary Miners Creek, Sugar Creek, Patterson Creek, Kidder Creek, Canyon Creek, Kelsey Creek, Tompkins Creek, and Mill Creek (NMFS 2014).

Salmon River Population

Since 2002, the Salmon River Restoration Council, along with CDFW, the Karuk Tribe, USFS, and USFWS, have conducted spawning and juvenile surveys throughout the watershed. Juvenile coho salmon have been found rearing in most of the available tributary habitat with moderate or high intrinsic potential values (NMFS 2014). Juvenile presence/absence and abundance data from a variety of surveys indicate that many of the tributaries throughout the watershed are used, including tributaries to the lower Salmon River, Wooley

Creek, and the North and South Fork Salmon (NMFS 2014). Annual adult coho salmon abundance observed in the Salmon River has varied between 0 and 14 spawning adults since 2002 (Hotaling and Brucker 2010). Between 2002 and 2007, only 18 adults and 12 redds (average of 4 spawners per year) were found in the roughly 15 miles of surveyed habitat. Known coho salmon spawning has been observed in the Nordheimer Creek, Logan Gulch, Brazil Flat, and Forks of Salmon areas along the mainstem Salmon River, in the Knownothing and Methodist Creek reaches of the South Fork Salmon River, and in the lower North Fork Salmon River (Hotaling and Brucker 2010), with the most recent recorded observation being two individuals building a redd in 2017 (Meneks 2018). Without any new information to show coho salmon spawner abundance increased, estimates of the total Salmon River spawner abundance remains at less than 50 individuals. An adult population of 50 or less would represent a population with limited spatial structure (NMFS 2019a).

Lower Klamath River Population

Between 1996 and 2004, coho salmon were found in nearly all surveyed streams, including Salt, High Prairie, Hunter, Hoppaw, Saugep, Waukell, Terwer, McGarvey, Tarup, Omagaar, Blue, Ah Pah, Bear, Surpur, Little Surpur, Pularvasar, One Mile, Tectah, Johnsons, Pecwan, Mettah, Roaches, Cappell, Richardson, and Tully creeks. Coho salmon were generally not well-distributed in tributaries upstream of Blue Creek (NMFS 2014). In general, coho were only observed in the lower reaches of most tributaries; and in some cases, the Yurok Tribe noted that their presence appeared to be non-natal rearing (Voight and Gale 1998, YTEP 2009). Because of the high incidence of non-natal rearing, juvenile survey data cannot be used to determine the distribution of the LKR population. Spawner distribution data provide more accurate information regarding natal population distribution. Spawning data from a few of the major tributaries in the LKR show moderate spawner densities throughout surveyed reaches of these watersheds. Spawning coho salmon have been found in Blue Creek (mainstem), Crescent City Fork of Blue Creek, Hunter, Waukell, McGarvey, Terwer, Ah Pah, Tectah, and Pine (Gale 2009a, 2009b).

For the Lower Klamath River coho salmon population to be at low risk for the population-size threshold, Williams et al. (2008) estimated that a minimum of 29 coho salmon per IP-kilometer of habitat are needed (5,900 spawners total). The current distribution of spawners is well below this threshold. With the exception of McGarvey and Blue creeks (Gale and Randolph 2000), coho salmon are not well-distributed throughout the Lower Klamath River tributaries and continue to occur at modest to very low densities.

G.1.1.9 Status of Critical Habitat within the Action Area

Much of the following information is directly adopted from NMFS' 2019 BO on the continued operation of the Klamath Project (NMFS 2019a), because this document represents the most updated summary of information on habitat conditions for coho salmon in the Action Area downstream of Iron Gate Dam.

Water Quality Conditions

Much of the Klamath Basin is currently listed as water-quality impaired under Section 303(d) of the Clean Water Act (CWA). Water temperature and quality in both mainstem and tributary reaches are often stressful to juvenile and adult coho salmon during late spring, summer, and early fall months. In addition, increased nutrient loading and organic enrichment with associated depletion of dissolved oxygen (DO) are recognized to be stressors for coho salmon in the Action Area (NMFS 2014).

Water Temperature

Unsuitable water temperature is one of the most widespread and significant stresses in the SONCC coho salmon ESU (Williams et al. 2016) and is a recognized stressor seasonally throughout the Action Area. Optimal water, sub-optimal, and lethal temperatures for coho salmon are lifestage-specific (DWR 2004, Carter 2005). Stenhouse et al. (2012) reviewed water temperature thresholds and optima for coho salmon in the Action Area and identified an optimal water temperature range for rearing juvenile coho salmon to be 8°C to 15.6°C. Temperatures above this optimal range are associated with higher disease incidence and increased predation. NMFS (2014) identifies 19°C as the upper limit for coho salmon suitability, and 25°C as the lethal threshold for juvenile coho salmon.

Water temperatures in the Klamath Basin vary seasonally and by location, but water temperatures in the Klamath River regularly exceed temperatures optimal for coho salmon. Daily mean temperature (averaged over 2001 to 2011) exceeded 21°C from early July to late August in the Klamath River downstream of Iron Gate Dam (Asarian and Kann 2013). In 2017, an "extremely wet year," using the Environmental Protection Agency (EPA) guidelines, migrating adult salmon and rearing juvenile salmon temperature criteria were exceeded for between 3 months and 4 summer months at all focal monitoring locations in the Action Area (Romberger and Gwozdz 2018).

Downstream of Iron Gate Dam, water released from the Iron Gate Reservoir, when compared with modeled conditions without the dams, is 1 to 2.5°C cooler in the spring, potentially just below optimal temperatures in some years, and 2 to 10°C warmer in the summer and fall, well above optimal temperatures in most years (PacifiCorp 2004b, Dunsmoor and Huntington 2006, NCRWQCB 2010a, Risley et al. 2012). Farther downstream, water temperatures are more influenced by solar energy, the natural heating and cooling regime of ambient air temperatures, and tributary inputs of surface water.

Dissolved Oxygen

As with temperature, optimal and sub-optimal levels of DO are life-stage-specific for coho salmon (Carter 2005). In addition, there is an interaction effect among DO and other stressors, including water temperature and turbidity. Carter (2005) reviewed effects of various DO concentrations on salmonids and identified a minimum of 6 mg/L DO before production impairment was observed for most life stages, and a minimum 3 mg/L DO for acute mortality.

Generally, DO concentrations in the Klamath River downstream of Iron Gate Dam exceed minimum DO requirements for salmonids and other coldwater species (Asarian and Kann 2013). However, annual minimum DO concentrations from 2001 – 2011 were as low as 3.5 mg/L at Iron Gate Dam, with a general upward trend from 2001 – 2011 (Asarian and Kann 2013). Asarian and Kann (2013) indicated that the lowest DO concentrations (daily minimum DO, averaged over 2001 – 2011) occur from mid-July through late

August, with Klamath River minima (7.3 to 7.0 mg/L when averaged over 2001 to 2011) occurring between Iron Gate Dam and RM 100. Similarly, PacifiCorp (2018b indicated that seasonal minima (approaching 5 mg/L) occurred in August and mid-September within 1 river mile downstream of Iron Gate Dam; DO concentrations at all other monitored Klamath River sites were greater than 8 mg/L during calendar year 2017 (PacifiCorp 2018b).

Nutrients

Primary nutrients, including nitrogen and phosphorus, are affected by the geology of the surrounding watershed of the Klamath River, upland productivity and land uses, and a number of physical processes affecting aquatic productivity in reservoir and riverine reaches. An overabundance of these nutrients in the water can lead to toxic algal blooms and reduced dissolved oxygen levels. Total phosphorus values typically range from 0.1 to 0.25 mg/L in the Klamath River between Iron Gate Dam and Seiad Valley, with the highest values occurring just downstream of the dam. Total nitrogen concentrations in the river downstream of Iron Gate Dam generally range from <0.1 to over 2.0 mg/L and are generally lower than those in upstream reaches due to reservoir retention and dilution by springs in the Klamath Hydroelectric Reach (Asarian et al. 2010). Further decreases in total nitrogen occur in the mainstem Klamath River due to a combination of tributary dilution and natural in-river nutrient removal processes such as uptake by aquatic plants and algae growing on the riverbed (periphyton). However, concentrations of both nitrogen and phosphorous are high enough that other factors (i.e., light, water velocity, or available substrate) are likely more limiting to primary productivity than nutrients, particularly in the vicinity of Iron Gate Dam (FERC 2007, Asarian et al. 2010). Therefore, there is a limit on the extent to which high concentrations of these nutrients can cause periphyton growth in this portion of the Klamath River.

Downstream of the confluence with the Salmon River, nutrient concentrations continue to decrease in the Klamath River due to tributary dilution and nutrient retention. For total nitrogen, Asarian et al. (2010) demonstrated a general upward trend in concentrations from June through October at sites downstream of Iron Gate Dam.

Suspended Sediment Concentrations

High levels of sediment transport can reduce habitat and water quality for salmonids and are also of concern because high densities of *M. speciosa* (freshwater polychaete worms) have been observed in these habitats (Hillemeier et al. 2017, Som and Hetrick 2017). Currently, suspended sediments are more likely to be flushed out of the mainstem portion of the Upper Klamath River reach from annual surface flushing flow events. In addition, tributary rearing habitat currently accessed by Klamath River coho salmon is compromised to some degree, most commonly by high instream sediment concentrations or impaired riparian communities (see NMFS 2014 for review).

G.1.1.10 Juvenile Migratory Habitat Conditions

Juvenile migratory habitat must support both smolt emigration to the ocean and the seasonal redistribution of juvenile fish. This habitat must have adequate water quality, water temperature, water velocity, and passage conditions to support migration. It is important that migratory habitat is available year-round, because juvenile coho salmon spend at least 1 year rearing in freshwater, and have been shown to move upstream, downstream, in the mainstem, and into non-natal tributaries when redistributing to find suitable habitat (Adams 2013, Witmore 2014). Emigrating smolts are usually present in the mainstem Klamath River between February and the beginning of July, with April and May representing the peak migration months. Emigration rate tends to increase as fish move downstream (Stutzer et al. 2006). Juvenile migratory habitat conditions by sub-basin in the Action Area are described below.

Upper Klamath River Reach

In the Upper Klamath River reach, juvenile migration corridors are degraded because of diversion dams, low-flow conditions, poorly functioning road/stream crossings in tributaries, disease effects, and high water temperatures and low water velocities that slow and hinder emigration or upstream and downstream redistribution in both tributaries and the mainstem portion of this reach. The unnatural and steep decline of the hydrograph in the spring, due to anthropogenic factors, including water diversions and timing of water releases, observed in both the mainstem and tributaries, likely slows the emigration of coho salmon smolts, speeds the proliferation of fish diseases in the mainstem, and increases water temperatures more quickly than would occur otherwise. Disease effects, particularly in areas of the mainstem such as the Trees of Heaven campground, have been found to have had a substantial impact on the survival of juvenile coho salmon in this stretch of river (NMFS 2014). Low flows in the mainstem during the spring can slow the emigration of smolt coho salmon, which can in turn lead to longer exposure times for disease, and greater risks due to predation.

Many of the tributaries composing the Upper Klamath River reach population unit are small and may go subsurface near their confluence with the mainstem Klamath River. Yet these intermittent tributaries sometimes remain important rearing habitat for coho salmon, when and where sufficient instream flows, water temperature, and habitat conditions are suitable to sustain them. Coho salmon have adapted life history strategies (spatial and temporal) to use intermittent streams. For example, adult coho salmon will often stage in the mainstem Klamath River at the mouth of natal streams until hydrologic conditions allow them to migrate into tributaries, where they are able to find more suitable spawning conditions, and juveniles can find adequate rearing conditions and cover. In summer, when the downstream sections of these tributaries may go dry, the shaded, forested sections upstream provide cold water and high-quality summer rearing habitat for juvenile coho salmon. By early spring, when emigration of smolt coho salmon typically occurs, tributary flows are elevated, and connectivity to the mainstem Klamath River allows the smolt to emigrate (NMFS 2014).

Middle Klamath River Reach

Similar to the mainstem portion of the Upper Klamath River reach, low flows during the spring can slow the emigration of smolt coho salmon, which can in turn lead to longer exposure times for disease, and greater risks due to predation. In part due to this concern, flow releases to increase the volume of water in the Middle Klamath Reach were incorporated into the NMFS and USFWS (2013) joint opinion. Higher velocities resulting from these flow releases have somewhat addressed the water quality concern by reducing "dead zones" in the channel that can harbor disease pathogens (Hardy et al. 2006), thereby reducing the overall impact of disease infection on coho salmon. Still, summer water diversions downstream of Iron Gate Dam, which further decrease flows, contribute to degraded habitat and/or fish passage issues at tributaries such as Stanshaw, Red Cap, Boise, Camp, Elk Creek, and Fort Goff creeks during low water years.

Shasta River

Smolt emigration in the Shasta River coincides with the drop in flows from irrigation water withdrawal, typically in mid-April. Because there are significant water diversions and impoundments in the Shasta River, the unnatural and steep decline of the hydrograph following the start of the irrigation season in April decreases the quantity of rearing habitat and causes water temperatures to increase more quickly than would occur otherwise. These changes can displace YOY coho salmon, forcing them to redistribute in search of suitable rearing habitat, and thereby increasing their risk of mortality (Gorman 2016). Similarly, the reduction in water quality and quantity likely has a negative impact to emigrating coho salmon smolts, increasing their risk of mortality.

Scott River

A number of physical fish barriers exist in the Scott River watershed. For instance, Big Mill Creek, a tributary to the East Fork Scott River, has a complete fish passage barrier caused by down cutting at a road culvert outfall. Additionally, historical mining has left miles of tailings piles along the mainstem and some tributaries of the Scott River. A 7-mile reach of Scott River goes subsurface every summer due to this channel modification, in combination with low flows, limiting juvenile redistribution. For example, during the summer of 2014, when flows were disconnected in the mainstem Scott River, large numbers of juvenile coho salmon were left stranded, unable to migrate to suitable rearing habitat. A large rescue-relocation effort led to 115,999 coho salmon being moved to cold water habitats; however, monitoring of this effort showed that relocation did not increase the survival of rescued fish (CDFW 2016). For many years, the City of Etna's municipal water diversion dam on Etna Creek effectively blocked fish passage into upper Etna Creek; however, this dam was retrofitted with a volitional fishway in 2010. In addition, valley-wide agricultural surface water withdrawals and diversions, and groundwater extraction have all combined to cause premature surface flow disconnection in the summer and delayed re-connection in the fall along the mainstem Scott River. These conditions can consistently result in restrictions or exclusions to suitable rearing habitat, contribute to elevated water temperatures, and contribute to conditions that cause juvenile fish stranding and mortality.

Salmon River

Juvenile migration corridors exhibit high water temperatures that may hinder juvenile redistribution during the summer. Seasonal low-flow barriers were previously a concern for juvenile migration, but those barriers were largely addressed, and barriers are now a low-level stressor for the Salmon River.

G.1.1.11 Juvenile Rearing Habitat Conditions

Juvenile coho salmon rear in freshwater for a full year and can be found in the mainstem and tributaries. Although their rearing needs and locations may change on a seasonal basis, an interconnected system is critical so that they can access different resources provided in different water bodies. For example, Witmore (2014) and Brewitt and Danner (2014) documented juvenile salmonids rearing in tributaries of the Klamath River, while simultaneously relying on mainstem food sources. These individuals displayed a diurnal movement pattern that highlights the importance of tributary/mainstem connection even during times when the mainstem appears to be inhospitable.

Juvenile rearing habitat conditions by sub-basin in the Action Area are described below.

Upper Klamath River Reach

Juvenile summer rearing areas have been compromised by low-flow conditions, high water temperatures, insufficient dissolved oxygen levels, excessive nutrient loads, habitat loss, disease effects, pH fluctuations, non-recruitment of large woody debris (LWD), and loss of geomorphological processes that create habitat complexity. Water released from Iron Gate Dam during summer months is already at a temperature stressful to juvenile coho salmon, and solar warming can increase temperatures even higher (up to 26ºC) as flows travel downstream (NRC 2004). The period of time when fry and juvenile rearing, as well as smolt migration, is possible along the mainstem has been shortened by these conditions and is therefore a temporal limitation. In the summer, the diversion and impoundment of water continues to lead to poor hydrologic function, disconnection, and diminishment of thermal refugia, and poor water quality in tributaries and the mainstem. Most tributaries with summer rearing potential are highly impacted by agriculture and past timber harvest. Very few remaining areas exist downstream of Iron Gate Dam with the potential and opportunity for summer rearing. Overwinter rearing habitat may be a limiting factor for juvenile coho salmon in the Upper Klamath River each. Human activities such as mining and agriculture have significantly altered the mainstem and tributaries into a more simplified channel with limited access to the floodplain. Additionally, much of the Upper Klamath River reach parallels Highway 96, leaving little room for floodplain complexity. As a result, slow-velocity water, such as side channels, off-channel ponds, and alcoves, have been eliminated, decreasing the ability for juvenile coho salmon to persist during high-velocity flows in the winter (NMFS 2014).

Unlike many of the other tributary streams in the Upper Klamath River reach, Bogus Creek and its largest tributary, Cold Creek, contain several cold-water springs that provide favorable conditions for rearing coho salmon during the summer (Hampton 2010). These springs are upstream of a waterfall (RM 3.48) that prevented anadromous fish access to these locations historically. In 1965, a fish ladder was constructed over this migration barrier, and adult salmon and steelhead have had access to another 6 miles of habitat upstream of the barrier since that time. There are several habitat and water conservation projects that have been completed recently or are currently underway to further improve rearing habitat conditions for juvenile coho salmon in the reach upstream of the ladder. These projects include installation of cattle exclusion fencing, riparian plantings, piping of irrigation ditches, construction of tailwater capture systems, and direct infusion of cold spring water to the channel. The mouth of Bogus Creek is adjacent to Iron Gate Hatchery, and hatchery-origin coho salmon are known to stray, and spawn in Bogus Creek. The CDFW has been monitoring emigration of smolt from Bogus Creek since 2015. Results of this effort indicate that age 1+ coho salmon emigrate from late February through May, and fry coho salmon have been observed from April through mid-June (Knechtle and Giudice 2018).

Over approximately the last 10 years, there has been a large effort to improve over-winter habitat for juvenile coho salmon in the Upper Klamath River reach. In particular, the Mid Klamath Watershed Council and Karuk Tribe have been constructing off-channel pond features in key locations to provide slow-velocity water. Over a dozen ponds have been constructed in locations such as Seiad Creek, Horse Creek, Tom Martin Creek, West Grider Creek, and O'Neil Creek. Monitoring efforts have shown that both natal and non-natal juvenile coho salmon are using these sites in large numbers (Witmore 2014).

Middle Klamath River Reach

There are approximately 79 miles of potentially suitable juvenile rearing habitat spread throughout the mainstem Klamath River and tributaries in the Middle Klamath region (NMFS 2014). However, juvenile summer rearing areas in this stretch of river are degraded relative to the historic state. High water temperatures, exacerbated by water diversions and seasonal low flows, restrict juvenile rearing in the mainstem Klamath River and lessen the quality of tributary rearing habitat (NMFS 2014). Nevertheless, a few tributaries in the Middle Klamath River Population (e.g., Boise, Red Cap, and Indian creeks) support populations of coho salmon and offer critical cool water refugia in their lower reaches when mainstem temperatures and water quality approach uninhabitable levels. Other important tributaries for juvenile rearing include Sandy Bar, Stanshaw, China, Little Horse, Pearch, and Boise creeks (NMFS 2014). However, these cool water tributary reaches can become inaccessible to juveniles when low flows and sediment accretion create passage barriers; therefore, summer rearing habitat can be limited.

Shasta River

Historically, instream river conditions, fostered by unique cold spring complexes, created abundant summer rearing and off-channel overwintering habitat that were favorable for production of coho salmon in the Shasta River basin. However, a reduction in the frequency of large flood flows, along with the elimination of sediment transport processes downstream of Dwinnell Dam have resulted in coarsening of the bed and reduction in habitat diversity immediately downstream of the dam. The loss of woody debris, pools, side channels, springs, and accessible wetlands from land use conversions have also contributed to reduced summer and winter rearing capacity for juvenile coho salmon (NMFS 2014).

Juvenile rearing is currently confined to the mainstem Shasta River, Big Springs Creek, Lower Parks Creek, Shasta River Canyon, Yreka Creek, and the upper Little Shasta River. Stream temperatures for summer rearing are poor throughout much of the mainstem Shasta River from its mouth upstream to near the confluence of Big Springs Creek. The onset of the irrigation season in the Shasta River watershed has a dramatic impact on discharge when large numbers of irrigators begin taking water simultaneously. This results in a rapid decrease in flows downstream of the diversions, stranding coho salmon as channel margin and side-channel habitat disappears; and in some extreme cases, channels can become entirely de-watered.

Low stream flows can decrease rearing habitat availability for juvenile coho salmon. Further alterations to stream channel function from agricultural practices includes a reduction in the number of beaver ponds, which provide important habitat attractive to rearing coho salmon (NMFS 2014).

Historically, the most vital habitat in the Shasta River basin was its cold springs, which created cold water refugia for juvenile coho salmon; decreased overall water temperatures; and allowed for successful summer rearing of individuals in natal and non-natal creeks and mainstem areas. These areas have been significantly adversely affected by water withdrawals, agricultural activities, and riparian vegetation removal. These land use changes have compromised juvenile rearing areas by creating low-flow conditions, high water temperatures, insufficient dissolved oxygen levels, and excessive nutrient loads. However, habitat restoration in the Big Springs complex and on The Nature Conservancy's Nelson Ranch have improved juvenile rearing conditions in those areas.

Streamflow in the Upper Shasta River is primarily controlled through releases from Dwinnell Reservoir, which is owned and operated by the Montague Water Conservation District (MWCD). Dwinnell Reservoir was constructed on the Upper Shasta River in 1928, with the purpose of storing water for irrigation use during the growing season. MWCD holds appropriative water right permits (Permit Numbers 2452 and 2453), which give MWCD the right to divert and store a total of 49,000 acre-feet of water from the upper Shasta River (35,000 acre-feet) and Parks Creek (14,000 acre-feet) annually. There are several ways in which MWCD can release water to the Upper Shasta River downstream of Dwinnell Dam. These include releases of irrigation water to meet prior water right holders downstream; short-term voluntary release of water and participation in water lease agreements to improve instream conditions for salmonids; and release of environmental water as agreed to under their Conservation and Habitat Enhancement and Restoration Program (CHERP), which was developed coincident with a Settlement Agreement with the Klamath River Keeper and Karuk Tribe.

Under the CHERP, once water conservation projects have been completed to their main canal, MWCD will increase instream environmental releases by an average of 4,400 acre-feet downstream of Dwinnell Dam as a conservation measure to improve conditions for coho salmon. The environmental water will be used to support fisheries habitat enhancements through a combination of (a) releases of stored water from Dwinnell Reservoir to the upper Shasta River; (b) bypassing additional flows at its Parks Creek Diversion; (c) augmenting flows in the upper Shasta River through groundwater releases; and (d) potential water exchanges with downstream diverters. MWCD also proposes to implement other infrastructure improvements to support fisheries enhancement and recovery in the upper Shasta River and lower Parks Creek. These improvements include the enlargement of its Cross Canal, which delivers released flow from Dwinnell Reservoir to the Shasta River, and construction of wetland and cold water refugia habitat immediately downstream of Dwinnell Dam. All of these efforts will improve rearing conditions for coho salmon downstream of Dwinnell Dam.

LWD is depleted in the Shasta River due to anthropogenic land use changes, including grazing and agricultural practices. Additionally, water diversions have likely lowered the water table throughout the basin, thereby limiting growth of riparian vegetation and channel-forming wood. The lack of large wood in the

Shasta River creates a deficit of shade and shelter, and decreases habitat complexity and pool volumes, all necessary components for over-summering juvenile survival.

Scott River

Numerous water diversions, dams, and interconnected groundwater extraction for agricultural purposes, and the diking and leveeing of the mainstem Scott River have reduced summer and winter rearing habitat in the Scott River basin, limiting juvenile survival. Although rearing habitat still exists in some tributaries, access to some of these areas is hindered by dams and diversions, the existence of alluvial sills, and the formation of thermal barriers at the confluence of tributaries. Where passage is possible, there are thermal refugial pools and tributaries where the water temperature is several degrees cooler than the surrounding temperature, providing a limited amount of rearing habitat in the basin.

Currently, valley-wide agricultural water withdrawals and diversions, groundwater extraction, and drought have all combined to cause premature surface flow disconnection along the mainstem Scott River. In addition, summer discharge has continued to decrease significantly over time, further exacerbating detrimental effects on coho salmon in the basin. These conditions restrict or exclude available rearing habitat, elevate water temperature, decrease fitness and survival of over-summering juveniles, and sometimes result in juvenile fish strandings and death.

Woody debris is scarce throughout the mainstem Scott River and its tributaries. Mainstem habitat has been straightened, leveed, and armored. Anthropogenic impacts have resulted in a lack of channel complexity from channel straightening and reduced amounts of woody material (Cramer Fish Sciences 2010). The present-day mainstem Scott River bears minor resemblance to its more complex historic form, although meandering channel planforms are still present (Cramer Fish Sciences 2010). Over the last several years, the Scott River Watershed Council has been working collaboratively with the NMFS and CDFW to improve habitat conditions for rearing coho salmon, improve wetland habitat, improve floodplain connectivity, and help maintain surface water and groundwater connectivity through development of beaver dam analogue structures (BDAs) at strategic locations in major tributary streams and in the mainstem Scott River. Fry and juvenile coho salmon have been documented using these restoration sites throughout the year. The Scott River Watershed Council, in collaboration with NMFS, has shown through their long-term monitoring efforts that the fish in these BDA sites have displayed high rates of growth and high rates of over-winter survival (Yokel et al. 2018). Development of more of these types of projects, if combined with improved water conservation and management practices, is anticipated to improve conditions for rearing coho salmon in the future.

Salmon River

The Salmon River watershed has little private landownership and is dominated by public U.S. Forest Service land. Therefore, human-caused stressors are minimal, with few diversions, and little agriculture or channel modification.

According to available juvenile fish survey information beginning in 2002, juvenile coho salmon have been found rearing in most of the available suitable tributary habitat. These streams are tributaries to the South Fork Salmon (Know nothing and Methodist Creek), at least nine tributaries to the North Fork Salmon, and in mainstem Salmon River tributaries, including Nordheimer and Butler Creeks (Hotaling and Brucker 2010). The lower reaches of these tributaries provide substantially cooler summer habitat than mainstem river habitat. During juvenile coho salmon presence/absence surveys conducted from 2015-2017 a total of

89 juvenile coho was observed (0 in 2015, 53 in 2016, 36 in 2017), primarily in the South Fork or its tributaries. In 2018, 54 juvenile coho were observed at the mouth of and within Methodist Creek, a tributary to the South Fork (Amy Fingerle, unpublished data). There is some indication that juvenile coho salmon move up from the mainstem Klamath River into the cooler Salmon River tributaries during summer months when stressed by mainstem water temperatures. Some juveniles found in surveys are thought to reflect non-natal as well as natal rearing (NMFS 2014).

G.1.1.12 Spawning Habitat Conditions

Spawning habitat conditions by sub-basin in the Action Area are described in the following sections.

Upper Klamath River Reach

Coho salmon are typically tributary spawners, low numbers of adult coho salmon annually spawn in the Upper Klamath River mainstem. However, upstream dams block the transport of sediment into this reach of river, and the lack of clean and loose gravel diminishes the quality of salmonid spawning habitat downstream of the dams. This condition is especially critical directly downstream of Iron Gate Dam (FERC 2007). However, water temperatures and water velocities are generally sufficient in this reach for successful adult coho salmon spawning. Gravel augmentation implemented under the PacifiCorp habitat conservation plan will partially restore spawning habitat in the Upper Klamath River reach, particularly between Iron Gate Dam and the confluence with the Shasta River (PacifiCorp 2012). Downstream of Iron Gate Dam, channel conditions reflect the interruption of sediment flux from upstream by reservoir capture and the eventual re-supply of sediment from tributaries entering the mainstem Klamath River (PacifiCorp 2004b). Key Upper Klamath River reach spawning tributaries to which adult coho salmon return annually to spawn include Seiad Creek and Horse Creek in the lower portion of the reach, Beaver Creek in the middle portion of the reach, and Bogus Creek in the upper portion of the reach.

Middle Klamath River Reach

The quality and amount of spawning habitat in the Middle Klamath River reach is naturally limited due to the geomorphology and the prevalence of bedrock in this stretch of river. Coho salmon are typically tributary and headwater stream spawners, so it is unclear if there was historically very much mainstem spawning in this reach. Key Middle Klamath River reach spawning tributaries to which adult coho salmon return annually to spawn include Red Cap and Camp creeks.

Shasta River

The Shasta River, with its cold flows and high productivity, was once especially productive for anadromous fishes. The current distribution of spawners is limited to the mainstem Shasta River, Big Springs Creek, lower Parks Creek, and the Shasta River Canyon. The reduction of LWD recruitment, channel margin degradation, and excessive sediment has limited the development of complex stream habitat necessary to sustain spawning habitat in the Shasta Valley. Persistent low-flow conditions through the end of the irrigation season (October 1) can also constrain the timing and distribution of spawning adult coho salmon. Unlike the majority of the Shasta Valley, the irrigation season in Parks Creek does not end until November 1, and there are also several stock water diversions that continue to divert throughout the fall and winter season. Therefore, persistent low-flow conditions, particularly in dry years, can limit the extent of spawning, and in some years may prevent coho salmon from spawning in Parks Creek.

Coho salmon spawning has been observed in the Shasta River Canyon, lower Yreka Creek, throughout the Big Springs Complex area, and in Lower Parks Creek. In some reaches, particularly in the lower canyon and the reach downstream of the Dwinnell Dam, limited recruitment of coarse gravels is likely contributing to a decline in abundance of spawning gravels (Ricker 1997). The causes of the decline in gravels include gravel trapping by Dwinnell Dam and other diversions, bank-stabilization efforts, and historical gravel mining in the channel. In a 1994 study of Shasta River gravel quality, Jong (1997) found that small sediment particles and fines (<4.75mm) were present in quantities associated with excessive salmon and steelhead egg mortality. Jong (1997) also concluded that gravel quality had deteriorated since 1980, when the DWR performed similar work in the Shasta basin. Greenhorn Dam blocks the movement of gravel down Yreka Creek, and alters the Yreka Creek hydrograph.

Scott River

Gravel transport in the Scott River basin is relatively unimpeded; however, significant water diversions can reduce the volume and power of the mainstem and tributaries so that bedload mobilization is reduced.

Pebble count data and survey data indicate that suitable gravels sizes are found in conjunction with slopes also suitable for spawning (Cramer Fish Sciences 2010). These observations suggest that the amount of coarse sediment and its rate of delivery are not limiting spawning habitat availability in the Scott River Watershed.

Although gravel mobilization is unimpeded, historic land uses create a legacy of effects that are continuing to impact available spawning habitat. Data show that spawning substrate is largely suitable throughout the basin, but the spatial extent of these areas is limited due to mine tailing piles and other legacy mining effects. Current conditions in the Scott River mimic hydraulic conditions similar to bedrock canyons, where sediment used by salmonids has a lower likelihood of persistence due to increased (or more efficient) sediment transport compared to unconfined reaches (Cramer Fish Sciences 2010). The over-extraction of streambed alluvium likely also has stripped the alluvial cover from some river reaches, exposing underlying bedrock, the net result of which is enhanced sediment transport; less persistent alluvium; and an overall loss of physical complexity (Cramer Fish Sciences 2010). Channel confinement by historic mining tailings indirectly affects the diversity of stream habitat that might otherwise be available. Many of these tailing piles are too large for the adjacent watercourse to reshape.

Salmon River

Twelve percent of the 1,414 miles of stream in the Salmon River watershed are able to support anadromous salmonids, due to the mountainous topography and associated hydrology of the landscape (Elder et al. 2002). For this reason, coho salmon in the Salmon River population are naturally restricted in their distribution (NMFS 2014). Coho salmon habitat includes the mainstem Salmon River, Wooley Creek, the North Fork and South Fork Salmon Rivers, and the lower reaches of a few smaller tributaries.

G.1.1.13 Factors Affecting Critical Habitat in the Action Area

Many of the factors affecting the condition of critical habitat in the Action Area are discussed at length in Chapter 4 of the BA. Factors listed in the section below are specific to activities that directly affect listed populations of coho salmon. Much of this section is taken from NMFS 2019a.

Hatcheries

Iron Gate Hatchery was constructed in 1962 to mitigate for lost anadromous salmonid spawning and rearing habitat between Copco No. 2 Dam and Iron Gate Dam. The historic mitigation goals include a release of 6,000,000 Chinook salmon (5,100,000 fingerlings and 900,000 yearlings), 75,000 coho salmon yearlings, and 200,000 steelhead trout yearlings, annually. Returns of adulty hatchery-reared steelhead declined dramatically during the 1990s for unknown reasons, and Iron Gate Hatchery has not produced steelhead since 2012.

Of the 6 million Chinook salmon that are released from the Iron Gate Hatchery, about 5.1 million are released as smolts from mid-May through early June; and about 900,000 are released as yearlings from mid-October through November. The 75,000 coho salmon are released as yearlings after March 15th each spring. Prior to 2001, all of the Chinook salmon smolts were released after June 1 of each year. However, beginning in 2001, CDFW began implementing an early release strategy in response to recommendations provided by the Joint Hatchery Review Committee (CDFG and NMFS 2001). The Joint Hatchery Review Committee stated that the current smolt release times (June 1 to June 15) often coincide with a reduction in the flow of water released by USBR into the Klamath River, and that this reduction in flows also coincides with a deterioration of water quality and reduces the rearing and migration habitat available for both naturally and hatchery-reared fish. In response to these concerns, the CDFW proposed an Early Release Strategy and Cooperative Monitoring Program in April of 2001 (CDFG 2001). The goals of implementing the early release strategy are to:

1. Improve the survival of hatchery-released fall Chinook salmon smolts from Iron Gate Hatchery to the commercial, tribal, and sport fisheries.
2. Reduce the potential for competition between hatchery and natural salmonid populations for habitats in the Klamath River, particularly for limited cold-water refugia habitat downstream of Iron Gate Dam.

Although these management strategies are intended to reduce impacts to wild salmonids, some negative interactions between hatchery and wild populations likely still persist through competition between hatchery and natural fish for food and resources, especially limited space and resources in thermal refugia important during summer months (McMichael et al. 1997, Kostow et al. 2003, Kostow and Zhou 2006).

The SONCC coho salmon ESU, which includes coho salmon produced at Iron Gate Hatchery, is listed as threatened under CESA and the ESA. A Hatchery and Genetics Management Plan (HGMP) and Section 10(a)(1)(A) Enhancement of Survival Permit was issued to the CDFW in 2014 for the Iron Gate Hatchery coho salmon artificial propagation program (Section 10(a)(1)(A) Permit 15755) (CDFW and PacifiCorp 2014).

Under the HGMP, the purpose of the coho salmon program is to aid in the conservation and recovery of the Upper Klamath Population Unit of the SONCC coho salmon ESU by conserving genetic resources and reducing short-term extinction risks prior to future restoration of fish passage upstream of Iron Gate Dam. In addition, the HGMP is also intended to reduce the immediate threat of demographic extinction for both the upper Klamath River and Shasta River populations by encouraging release of adult coho salmon from the hatchery that are not required or suitable for use in the hatchery genetic spawning matrix. Starting in 2010, all returning adult coho salmon to Iron Gate Hatchery that were not used as broodstock were returned back to the Klamath River, where they would have the opportunity to spawn naturally in the upper Klamath River or nearby tributary streams. Under the HGMP, the Iron Gate Hatchery program will operate in support of the basin's coho salmon recovery efforts by conserving a range of the existing genetic, phenotypic, behavioral, life history, and ecological diversity of the run. The program includes conservation measures, genetic analysis, and rearing and release techniques that will improve fitness and reduce adverse impacts that may result from straying of hatchery fish, and limit effects of hatchery releases on wild fish.

The exact effects on juvenile coho salmon the Klamath River from the annual release of 6,000,000 hatchery-reared Chinook salmon smolts from Iron Gate Hatchery are not known precisely. The release of a relatively large number of hatchery-origin juvenile Chinook salmon has the potential to affect wild coho salmon juveniles via competitive interactions, increased predation, and exposure to disease, but habitat partitioning between the two species likely limits these effects. However, although both hatchery and wild origin coho salmon in the system are listed under the ESA, the hatchery releases of yearling coho salmon (75,000 fish) may still compete with wild coho salmon juveniles for rearing habitat, migratory habitat, prey items, and thermal refugia. Hatchery juveniles are often larger and can displace wild juveniles in pools and other high-quality habitats. In addition, when hatchery coho salmon adults return, a small percentage can stray, and spawn with wild adults. Modeling conducted for CDFW's Iron Gate Hatchery HGMP indicates that the release of 75,000 coho salmon juveniles has the potential to reduce wild coho salmon juvenile abundance by up to 6 percent through increased predation, competition, and disease, assuming the wild juvenile coho salmon abundance is 75,000 (CDFW and PacifiCorp 2014).

Harvest

Coho salmon have been harvested in the past in both coho- and Chinook-directed ocean fisheries off the coasts of California and Oregon. However, stringent management measures, which began to be introduced in the late 1980s, reduced coho salmon harvest substantially. The prohibition of coho salmon retention in commercial and sport fisheries in all California waters began in 1994 (NMFS 2014). With the exception of some tribal harvest by the Yurok and Hoopa Valley for subsistence and ceremonial purposes, the retention of coho salmon is prohibited in all California river fisheries.

Tribal fishing for coho salmon in the Yurok Tribe's reservation on the lower Klamath River has been monitored since 1992. The Yurok Tribal Fisheries Program reported that annual harvest of coho salmon from reservation lands on the lower Klamath River has ranged from 25 to 2,452 fish per year and averaged 612 fish between 1992 and 2009 (Williams 2010). Williams (2010) estimated that the Yurok Tribal harvest captured between 0.9 and 16.9 percent (average 3.7 percent) of the Klamath River coho salmon escapement. Similar data reviewed from 2010 to 2018 (CDFW 2019a) showed Yurok Tribal harvest captured between 20 and 416 coho salmon per year and averaged 193 coho salmon. No data on Yurok Tribal harvest was available for 2017 and 2018. The recent Yurok Tribe Fall Harvest Management Plan (Yurok Tribe 2018) includes weekly fishing closures intended to protect coho salmon from harvest.

A review of harvest data from the Hoopa Valley Tribe from 2010 to 2014 showed an average annual harvest of 462 coho salmon per year, with approximately 80 percent of those fish harvested over 5 years being of hatchery origin (CDFW 2019a). No data for Hoopa Tribal harvest since 2014 were available.

With regard to ocean fisheries, in 1995, ocean recreational fishing for coho salmon was closed from Cape Falcon in Oregon to the United States/Mexico border. To comply with the SONCC coho salmon ESU conservation objective, projected incidental mortality rates on Rogue and Klamath River hatchery coho salmon stocks are calculated during the preseason planning process using the coho salmon Fishery Regulation Assessment Model (Kope 2005). Specifically, the Pacific Fishery Management Council applies a SONCC coho salmon ESU consultation standard requirement of no greater than a 13.0 percent marine exploitation rate on Rogue/Klamath hatchery coho salmon, which applies to incidental mortality in the Chinook salmon ocean fisheries from Cape Falcon in Canada to the United States/Mexico border (PFMC 2018a). In summary, although major steps have been taken to limit effects of harvest on SONCC coho salmon, the population is still impacted by incidental mortality associated with various Chinook salmon fisheries, and by subsistence and ceremonial tribal fisheries.

Predation

Predation of adult and juvenile coho salmon occurs from a number of sources, including piscivorous fish, avian predators, pinnipeds, and other mammals. However, the effect of predation on coho salmon in the Action Area is not well understood. Pinniped predation on adult salmon can significantly affect escapement numbers in the Klamath River basin. Hillemeier (1999) assessed pinniped predation rates in the Klamath River estuary during August, September, and October 1997, and estimated that a total of 223 adult coho salmon were consumed by seals and sea-lions during the entire study period. Increased rates of predation of juvenile coho salmon from piscivorous fish (e.g., steelhead) may result from the concentrated hatchery releases from Iron Gate Hatchery (Nickelson 2003). Although the extent of predation is not well understood, given the small number of wild-born juvenile coho salmon, predation at any level may be having an adverse effect on coho salmon in the Action Area (NMFS 2014).

Restoration Activities

There are various restoration and recovery actions underway in the Klamath Basin aimed at removing barriers to salmonid habitat and improving habitat and water quality conditions for anadromous salmonids. Although habitat generally remains degraded across the ESU, restorative actions have effectively improved the conservation value of critical habitat throughout the range of the SONCC coho salmon, including portions of the Interior Klamath Diversity Stratum. In 2002, NMFS began ESA recovery planning for the SONCC and Oregon Coast coho salmon ESU through a scientific technical team created and chaired by the Northwest and Southwest Regional Fishery Science Centers, referred to as the Oregon and Northern California Coast coho salmon technical recovery team. In 2014, NMFS issued a final recovery plan for the SONCC coho salmon ESU (NMFS 2014). Planned and implemented actions intended to help recover SONCC coho salmon, as guided by the recovery plan, include the following:

- PacifiCorp Habitat Conservation Plan: As described in Chapter 4 of the BA, PacifiCorp's Coho Salmon HCP (PacifiCorp 2012) guides a comprehensive program of restoration actions that have benefitted coho salmon habitat in the Action Area.
- USBR has provided $500,000 per year since 2013 (approximately $3 million) for the Klamath Coho Habitat Restoration Program administered by National Fish and Wildlife Foundation (NFWF). The grant program funds restoration activities to improve habitat, water quality, water quantity, and fish passage, as well as research projects for coho salmon recovery. Restoration activities can occur on the mainstem Klamath River and its tributaries, with most restoration being conducted in the Shasta, Scott, and Salmon River Basins. Restoration projects are typically implemented by state, tribal, local, or private non-governmental organizations. USBR has supported three grant cycles (2016, 2017, and 2018) via funding through NFWF for restoration and research/monitoring projects, whereas a total of 21 projects have been selected for full or partial funding. Of these projects, seven have started implementing their projects for the grant years of 2016 and 2017, and three have begun or completed restoration activities.
- Congress authorized $1 million annually from 1986 through 2006 to implement the Klamath River Basin Conservation Area Restoration Program. The Klamath River Basin Fisheries Task Force (Task Force) was established by the Klamath River Basin Fishery Resources Restoration Act of 1986 (Klamath Act) to provide recommendations to the Secretary of the Interior on the formulation, establishment, and implementation of a 20-year program to restore anadromous fish populations in the Klamath River Basin to optimal levels.
- Multiple local watershed groups exist in the Action Area, including: the Shasta River Coordinated Resource Management Planning Group (Shasta sub-basin), Scott River Watershed Council(Scott sub-basin), Siskiyou Resource Conservation District (Scott sub-basin), Scott Valley Water Trust (Scott sub-basin), Salmon River Restoration Council (Salmon sub-basin), Karuk Tribe and Mid-Klamath Watershed Council (mid-Klamath sub-basin), and the Yurok Tribe (lower-Klamath sub-basin). Some key restoration actions that have been implemented in these sub-basins include:
  - Construction of off-channel ponds and side channels to provide winter velocity refugia for juvenile salmonids. These projects typically include connection to groundwater so the habitat can also function as cold water refugia throughout the summer as well.

  + Construction of BDAs to improve floodplain connectivity and instream complexity. The BDAs increase groundwater storage, sort sediment, and provide both winter and summer refugia for juvenile salmonids.
  + Placement of large wood jams in tributaries to improve floodplain connectivity, and provide winter and summer refugia for juvenile salmonids.
  + Remediation of mine tailings and reconstruction of stream reaches to improve sinuosity and floodplain connection.
  + Implementation of off-channel stock watering systems to improve water quality and quantity, as well as riparian vegetation condition.
- NMFS administers several grant programs to further restoration efforts in the Klamath River Basin. Since 2000, NMFS has issued grants to the states of California and Oregon, and Klamath River Basin tribes (Yurok, Karuk, Hoopa Valley and Klamath) through the Pacific Coast Salmon Restoration Fund (PCSRF) for the purposes of restoring coastal salmonid habitat. California integrates the PCSRF funds with their salmon restoration funds and issues grants for habitat restoration, watershed planning, salmon enhancement, research and monitoring, and outreach and education.
- The Klamath National Forest (KNF) continues to implement floodplain and instream habitat restoration projects along the Mid Klamath River corridor to benefit salmonids, including SONCC coho salmon. Most notable of these is a side channel and floodplain restoration project at the confluence of Fish Gulch and mainstem Horse Creek, a tributary to the Klamath River. Completed in fall 2018, this effort has reactivated more than 900 linear feet of salmonid spawning and rearing habitat. The KNF has also undertaken LWD placement projects along this reach of lower Horse Creek, as well as in SONCC coho salmon critical habitat in several other tributaries to the Klamath River.
- Caltrans completed the Fort Goff Creek Fish Passage Restoration Project, which restored fish passage above Highway 96 by replacing a culvert pipe with a concrete/steel-span bridge. Caltrans is currently also planning for two fish passage projects that would occur along Highway 96, very close to the Klamath River at Portuguese Creek and Cabe Creek, with possible construction in 2022 or 2023. Three additional fish passage projects are along Highway 96 at Ti Creek, Coon Creek, and Tom Martin Creek. These projects are on the priority list with potential construction in 2024 or after.
- The Shasta Valley Resource Conservation District conducts various restoration, monitoring, and management activities in the Action Area, including in minor tributaries to the Klamath River from the California State line near Keno to below Happy Camp, and the lower end of the Scott River.
- The Salmon River Restoration Council conducts fish passage projects at stream/river confluences on 50 tributaries to the Klamath from Iron Gate to Bluff Creek in Humboldt County.
- The Mid Klamath Watershed Council conducts various restoration, monitoring, and management activities in the Action Area. Specific projects include the following:
  + Mid Klamath Floodplain Assessment and Mine Tailings Remediation Planning (complete) – Assessment of 71 river miles from Shasta River to Elk Creek (Happy Camp), prioritization of 15 sites with 30 percent design completed.

  + Seiad Creek Coho Habitat Enhancement Project (completed). Restore stream geomorphology and floodplain function in a 0.75-mile, 14.5-acre reach. Remove tailing piles, construct off-channel habitats, instream wood structures.
  + Structural Monitoring of Constructed Off-Channel Habitats (on-going) – Monitoring of physical and biological effectiveness restoration work, including collection of data on water, fish use, and physical parameters for evaluating project effectiveness addressing limiting conditions for salmonids. Project sites are on tributaries to the Klamath River.
  + Horse Creek Wood Loading and Floodplain Design (on-going) – Development of engineered plans for restoring stream geomorphology with instream wood structure, reconnecting floodplain in the lower 1.5 miles, 100 acres of Horse Creek.
  + China Creek Fish Passage and Wood Loading Project (on-going).
  + Klamath River at Horse Trough Springs Floodplain Connection Design Project (on-going).
  + Aikens Creek Instream Habitat Enhancement Project (on-going) – Enhancement of a 0.6-mile reach of Aikens Creek in Humboldt County by installing 24 wood structures for restoring stream geomorphology and reconnecting floodplain habitats for winter and summer rearing.
  + Creation of off-channel ponds and channels at multiple sites (e.g., O'Neil Creek, Seiad Creek, etc.).
- In-river fish monitoring projects include annual adult and juvenile fish abundance/health monitoring for salmonid management purposes. Monitoring is conducted by USFWS, USFS, CDFW, Hoopa Valley Tribe, Karuk Tribe, Yurok Tribe, the Salmon River Restoration Council, and the Mid Klamath Watershed Council.

G.1.2 Southern DPS Green Sturgeon (*Acipenser medirostris*)

G.1.2.1 Species status

NMFS published a final rule listing the sDPS of green sturgeon as threatened in 2006 (71 FR 17757). There are two DPSs defined for green sturgeon: sDPS that spawns in the Sacramento River; and a northern DPS (nDPS) with spawning populations in the Klamath and Rogue rivers (NMFS 2008a). The sDPS includes all spawning populations of green sturgeon south of the Eel River in California, of which only the Sacramento River Basin currently contains a spawning population. The sDPS of green sturgeon has been listed as threatened under the ESA (71 FR 17757), whereas the nDPS is a Species of Concern.

G.1.2.2 Critical habitat

Critical habitat for the sDPS of green sturgeon was designated in 2009 (74 FR 52300). In freshwater, designated critical habitat includes: 1) the Sacramento River from the Sacramento I-Street bridge to Keswick Dam, including the Sutter and Yolo bypasses; 2) the Feather River from its confluence with the Sacramento River upstream to Fish Barrier Dam; 3) the Yuba River from its confluence with the Feather River upstream to Daguerre Point Dam; 4) the American River from its confluence with the Sacramento River upstream to the Highway 160 bridge; and 5) the Sacramento-San Joaquin Delta (as defined by California Water Code Section 12220). In coastal bays and estuaries, designated critical habitat includes: 1) San Francisco, San Pablo, Suisun, and Humboldt bays in California; 2) Coos, Winchester, Yaquina, and Nehalem bays in Oregon; 3) Willapa Bay and Grays Harbor in Washington; and 4) the lower Columbia River estuary from the mouth to RM 46. In coastal marine waters, designated critical habitat includes nearshore waters within the 60-fathom isobath from, and including, Monterey Bay north to the U.S./Canada border (including the Strait of Juan de Fuca).

The specific PBFs essential for the conservation of the sDPS of green sturgeon in freshwater riverine systems include:

- Food resources: abundant prey items for larval, juvenile, sub-adult, and adult life stages.
- Substrate: substrates suitable for egg deposition and development, larval development, and sub-adults and adults. Spawning is believed to occur over substrates ranging from clean sand to bedrock, with preferences for cobble (Emmett et al. 1991, Moyle et al. 1995).
- Water: a flow regime (i.e., the magnitude, frequency, duration, seasonality, and rate-of-change of freshwater discharge over time) necessary for normal behavior, growth, and survival of all life stages.
- Water quality: suitable water quality for normal behavior, growth, and viability of life stages, including temperature, salinity, oxygen content, and other chemical characteristics.

The Klamath River estuary and 1.6 kilometers of the coastal marine areas adjacent to the Yurok Tribal land are excluded from the critical habitat designation. Except for the 1.6 kilometers adjacent to Yurok Tribal land, the coastal marine areas around the Klamath River are designated as critical habitat for the sDPS green sturgeon.

G.1.2.3 Life history

Green sturgeon are believed to spend most of their lives in nearshore oceanic waters, bays, and estuaries. Early life-history stages reside in freshwater, with adults returning to freshwater to spawn when they are more than 15 years of age and more than 4 feet in size. sDPS green sturgeon typically spawn every 3 to 4 years (range 2 to 6 years), and spawning occurs primarily in the Sacramento River (Brown 2007; Poytress et al. 2012). Spawning by sDPS green sturgeon has been recently confirmed in the Feather River (Seesholtz et al. 2015). Adult sDPS green sturgeon enter San Francisco Bay in late winter through early spring, and spawn from April through early July, with peak activity influenced by factors including water flow and temperature (Heublein et al. 2009; Poytress et al. 2011). Spawning primarily occurs in cool sections of the upper mainstem Sacramento River in deep pools containing small- to medium-sized gravel, cobble, or boulder substrate (Poytress et al. 2009-2011; Wyman et al. unpublished). Post-spawn fish may hold for several months in the Sacramento River and outmigrate in the fall or winter or move out of the river quickly during the spring and summer months, although the holding behavior is most commonly observed (Heublein et al. 2009; CDWR 2013; Thomas et al. unpublished). Based on the length of juvenile sturgeon captured in the San Francisco Bay Delta, sturgeon migrate downstream toward the estuary between 6 months and 2 years of age (Radtke et al. 1966). Juvenile green sturgeon spend 1 to 4 years in fresh and estuarine waters before dispersal to saltwater (Beamesderfer and Webb 2002). They disperse widely in the ocean after their out-migration from freshwater (Moyle et al. 1992).

G.1.2.4 Geographic distribution

Green sturgeon is a widely distributed and marine-oriented species found in nearshore waters from Baja California to Canada (NMFS 2008a). Non-spawning adult and subadult nDPS and sDPS green sturgeon spend much of their lives coexisting in marine and estuarine waters from the Bering Sea, Alaska (Colway and Stevenson 2007) to El Socorro, Baja California, Mexico (Rosales-Casian and Almeda-Juaregui 2009).

Telemetry, genetic, and fisheries data suggest that DPS green sturgeon generally occur from Graves Harbor, Alaska to Monterey Bay, California (Moser and Lindley 2007; Lindley et al. 2008, 2011; Schreier et al. 2016), and within this range, frequent coastal waters of Washington, Oregon, Vancouver Island, and San Francisco and Monterey bays (Huff et al. 2012). Adult and subadults DPS green sturgeon occur in relatively large concentrations in summer and autumn in coastal bays and estuaries, including the Columbia River estuary, Willapa Bay, Grays Harbor, and the Umpqua River estuary (Moser and Lindley 2007; Lindley et al. 2008, 2011; Schreier et al. 2016).

Information from fisheries-dependent sampling suggests that green sturgeon only occupy large estuaries during the summer and early fall in the northwestern U.S. Green sturgeon are known to enter Washington estuaries during summer (Moser and Lindley 2007). Commercial catches of green sturgeon peak in October in the Columbia River estuary, and records from other estuarine fisheries (Willapa Bay and Grays Harbor, Washington) support the idea that sturgeon are only present in these estuaries from June until October (Moser and Lindley 2007). Benson et al. (2007a) stated that outmigration of any holding green sturgeon from the Klamath River estuary occurred during the first significant rainfall, usually in November and December. This information suggests that sDPS green sturgeon are likely to use the Klamath River estuary only during the summer and fall months. Because sDPS sturgeon spend the majority of their life in the ocean, and individuals spend some time in a number of estuaries along the West Coast in the summer and fall, only a small proportion of the sDPS green sturgeon would be expected to be present in the Klamath River estuary in any given year.

G.1.2.5 Population trends

Historically, population estimates and trends have been scarce for both sDPS and nDPS green sturgeon. As described in the latest sDPS green sturgeon status review (NMFS 2015b), several recent studies and monitoring efforts are underway to provide better estimates of population trends in the future. Currently, the most useful dataset for examining population trends comes from Dual Frequency Identification Sonar (DIDSON) surveys in the Sacramento River, which began in 2010. These surveys have been used to estimate the abundance of sDPS adults— current estimate 2,106 (95 percent confidence interval [CI] = 1,246-2,966; Mora 2016). Mora (2016) also applied a conceptual demographic structure to that adult population estimate resulting in an sDPS subadult population estimate of 11,055 (95 percent CI = 6,540-15,571). The DIDSON surveys and associated modeling will eventually provide population trend data. Other efforts to track population trends are underway using tagging and fisheries data and larval capture as reviewed in Heublein et al. (2017).

G.1.2.6 Threats

The principal factor in the decline of the sDPS is the reduction of the spawning habitat to a limited section of the Sacramento River (NMFS 2006). The potential for catastrophic events to affect such a limited spawning area increases the risk of the sDPS green sturgeon's extirpation. Insufficient freshwater flow rates in spawning areas, contaminants (e.g., pesticides), bycatch of green sturgeon in fisheries, potential poaching (e.g., for caviar), entrainment and potential stranding of juveniles by water projects, influence of exotic species, small population size, impassable migration barriers, and elevated water temperatures in the spawning and rearing habitat likely also pose threats to this species (NMFS 2006).

An emerging threat is the development and operation of offshore and nearshore kinetic energy projects (NMFS 2015b). Impacts of such projects on North American green sturgeon could occur due to direct mortality impacts or habitat loss, and sensitivity to low levels of electromagnetic fields associated with the operations that could impact migration and habitat use (Nelson et al. 2008).

With respect to threats, the available information indicates that some threats, such as those posed by fisheries and impassable barriers, have been reduced. Recent prohibitions on retention of green sturgeon in recreational and commercial fisheries in all states has eliminated a known threat and is likely having a positive effect on the overall population (NMFS 2015b). The recent decommissioning of the Red Bluff Diversion Dam on the Sacramento River and breaching of the Shanghai Bench on the Feather River has improved spawning conditions, although sDPS green sturgeon still encounter impassible barriers in the Sacramento River Basin that limit their spawning range (NMFS 2015b). The emerging threat posed by nearshore and offshore energy development requires continued attention into the future. Because many of the threats cited in the original listing still exist, the threatened status is still applicable.

G.1.2.7 Status in the Action Area

Both sDPS and nDPS green sturgeon likely use the Klamath River (195 FR 52300). Although sDPS green sturgeon may enter West Coast estuaries to feed in the summer and fall, there has been no evidence of them entering the Klamath River estuary (75 FR 30714). However, if they do enter the Klamath River, they are not anticipated to migrate beyond the estuarine habitat or be in the Action Area during reservoir drawdown.

G.1.3 Southern DPS Eulachon (*Thaleichthys pacificus*)

G.1.3.1 Species status

Eulachon, (commonly called smelt, candlefish, or hooligan) are a small, anadromous fish from the eastern Pacific Ocean. On March 18, 2010, NMFS listed the sDPS of eulachon as threatened under the ESA (75 FR 13012). This DPS encompasses all populations in the states of Washington, Oregon, and California; and extends from the Skeena River in British Columbia (inclusive) south to the Mad River in Northern California (inclusive). The DPS is divided into four sub-areas: Klamath River, Columbia River, Fraser River, and British Columbia coastal rivers south of the Nass River. NMFS' 2016 ESA 5-year review concluded that the DPS' threatened designation remained appropriate.

G.1.3.2 Critical habitat

Critical habitat for the sDPS eulachon in the Klamath River was designated by NMFS on October 20, 2011 (76 FR 65324). NMFS designated approximately 539 miles of riverine and estuarine habitat in California, Oregon, and Washington in the geographical area occupied by the sDPS of eulachon. The designation includes 16 rivers and creeks extending from and including the Mad River, California to the Elwha River, Washington. NMFS did not identify any unoccupied areas as being essential to conservation, and therefore did not designate any unoccupied areas as critical habitat. The designated critical habitat areas contain one or more of the physical or biological features essential to the conservation of the species that may require special management considerations or protection. NMFS excluded from designation all lands of the Lower Elwha Tribe, Quinault Tribe, Yurok Tribe, and Resighini Rancheria, on a determination that the benefits of exclusion outweigh the benefits of designation. In the Klamath River, designated critical habitat extends from the mouth of the Klamath River upstream to Omogar Creek, a distance of 10.7 miles, and excludes tribal lands in the Yurok Reservation and Resighini Rancheria boundaries.

The physical or biological features essential for conservation of this species are:

- Freshwater spawning and incubation sites with water flow, quality, and temperature conditions and substrate supporting spawning and incubation.
- Freshwater and estuarine migration corridors free of obstructions with water flow, quality, and temperature conditions supporting larval and adult mobility, and with abundant prey items supporting larval feeding after the yolk sac is depleted.
- Nearshore and offshore marine foraging habitat with water quality and available prey, supporting juveniles and adult survival.

G.1.3.3 Life history

Eulachon are a short-lived, high-fecundity, high-mortality forage fish, and tend to have extremely large population sizes. Eulachon typically spend 3 to 5 years in saltwater before returning to freshwater to spawn. Spawning grounds in the Klamath River may extend up to Omogar Creek (RM 10.7) (76 FR 65324). Spawning generally occurs at between 0 to 10°C throughout the range of the species (Willson et al. 2006). Adult eulachon have been observed in the Klamath River in January and April (Larson and Belchik 1998). Spawning occurs in January, February, and March in the northern part of the DPS, and later in the spring in the southern parts of the DPS. Males appear to enter rivers prior to females (Spangler 2002 *cited in* Willson et al. 2006). Eulachon may be in river systems only a few days to a few weeks before they spawn (Rogers et al. 1990; Eulachon Research Council 2000; Spangler 2002, *cited in* Willson et al. 2006). Females may be present on spawning grounds for only 1 or 2 days (Eulachon Research Council 2000, *cited in* Willson et al. 2006). Males may be present between 1 and 4 days (Spangler 2002, *cited in* Willson et al. 2006). Most eulachon adults die after spawning. Eggs are fertilized in the water column, sink, and adhere to the river bottom, typically in areas of gravel and coarse sand. Eggs are immediately fertilized by milting males and the eggs adhere to stream substrates where they incubate for 30 to 40 days before the emergence of larvae (0.1 to 0.2 inch in length) (Hart 1973, *cited in* HDR Alaska 2008). Freshets rapidly move eulachon eggs and larvae to estuaries; it is likely that eulachon imprint and home to an estuary into which several rivers drain, rather than to individual spawning rivers (Hay and McCarter 2000). Newly hatched young are transparent, 0.16 to 0.27 inch in length, and drift downstream passively to the ocean (Hay and McCarter 2000).

G.1.3.4 Geographic distribution

Eulachon, an anadromous smelt in the northeastern Pacific Ocean, is composed of numerous populations that spawn in rivers from northern California to southwestern Alaska (NMFS 2017). In the portion of the species' range that lies south of the U.S.–Canada border, most eulachon production originates in the Columbia River Basin, including the Columbia River, the Cowlitz River, the Grays River, the Kalama River, the Lewis River, and the Sandy River (Gustafson et al. 2010). Smith and Saalfeld (1955) stated that eulachon were occasionally reported to spawn up to the Hood River on the Oregon side of the Columbia River prior to the construction of Bonneville Dam in the 1930s. In times of great abundance (e.g., 1945, 1953), eulachon have been known to migrate as far upstream as Bonneville Dam (Smith and Saalfeld 1955; WDFW and ODFW 2001, *cited in* Gustafson et al. 2010) and may extend upstream of Bonneville Dam by passing through the ship locks (Smith and Saalfeld 1955). Eulachon likely reached the Klickitat River on the Washington side of the Columbia River in 1945 via this route (Smith and Saalfeld 1955).

Historically, the only other large river basins in the contiguous United States where large, consistent spawning runs of eulachon have been documented are the Klamath River in northern California and the Umpqua River in Oregon.

G.1.3.5 Population trends

There are few direct estimates of abundance available for eulachon, and there is an absence of monitoring programs for them in the United States. Most population data come from fishery catch records. However, the combination of catch records and anecdotal information indicate that eulachon were present in large annual runs in the past, and that significant declines in abundance have occurred. The Biological Review Team RT concluded that, starting in 1994, the sDPS of eulachon experienced an abrupt decline in abundance throughout its range (Gustafson et al. 2010).

Spawning stock biomass estimations of eulachon in the Columbia River for the years 2000 through 2017 have ranged from a low of 783,400 fish in 2005 to a high of 185,965,200 fish in 2013, with an estimated 18,307,100 fish in 2017. Spawning stock biomass estimations of eulachon in the Fraser River for the years 1995 through 2017 have ranged from a low of 109,129 to 146,606 fish in 2010 to a high of 41,709,035 to 56,033,332 fish in 1996, with an estimated 763,330 to 1,026,251 fish in 2017.

In the Klamath River and the Umpqua River, eulachon were once abundant, but have declined to the point where detecting them has become difficult (NMFS 2010e). The situation in the Klamath River is currently more positive than it was at the time of the 2010 status review, with adult eulachon presence being documented in the Klamath River in the spawning seasons of 2011 to 2014, although it has not been possible to calculate estimates of spawning stock biomass (SSB) in the Klamath River (NMFS 2016a). Since the 2010 status review (Gustafson et al. 2010), there are reports of an estimated 7 (McCovey 2011), 40 (McCovey 2012), 112 (McCovey and Walker 2013), and approximately 1,000 adult eulachon being sampled by Yurok Indian tribal biologists in presence/absence surveys using seines and dip nets in the Klamath River in spring of 2011, 2012, 2013, and 2014, respectively.

There has been no long-term monitoring program targeting eulachon in California, making the assessment of historical abundance and abundance trends difficult (Gustafson et al. 2008). Adult spawning abundance of the sDPS of eulachon has clearly increased since the listing occurred in 2010. A number of data sources indicate that eulachon abundance in some subpopulations in the sDPS were substantially higher from 2011 to 2015, compared to indications of very low abundance from 2005 to 2010. The improvement in estimated abundance in the Columbia River, relative to the time of listing, reflects both changes in biological status and improved monitoring. The documentation of eulachon returning to the Naselle, Chehalis, Elwha, and Klamath rivers over the period from 2011 to 2015 also likely reflects both changes in biological status and improved monitoring (NMFS 2016a).

G.1.3.6 Threats

Habitat loss and degradation threaten eulachon, particularly in the Columbia River basin. Hydroelectric dams block access to historical eulachon spawning grounds and affect the quality of spawning substrates through flow management, altered delivery of coarse sediments, and siltation. The release of fine sediments from behind a United States Army Corps of Engineers sediment retention structure on the Toutle River has been negatively correlated with Cowlitz River eulachon returns 3 to 4 years later and is thus implicated in harming eulachon in this river system, although the exact cause of the effect is undetermined. Dredging activities in the Cowlitz and Columbia rivers during spawning runs may entrain and kill fish or otherwise result in decreased spawning success.

Eulachon have been shown to carry high levels of chemical pollutants; and although it has not been demonstrated that high contaminant loads in eulachon result in increased mortality or reduced reproductive success, such effects have been shown in other fish species. Eulachon harvest has been curtailed significantly in response to population declines. However, existing regulatory mechanisms may be inadequate to recover eulachon stocks.

Global climate change may threaten eulachon, particularly in the southern portion of its range where ocean warming trends may be the most pronounced, and may alter prey availability, as well as spawning and rearing success.

G.1.3.7 Status in the Action Area

Historically, large aggregations of eulachon were reported to have consistently spawned in the Klamath River. Allen et al. (2006) indicated that eulachon usually spawn no further south than the Lower Klamath

River and Humboldt Bay tributaries. The California Academy of Sciences (CAS) ichthyology collection database lists eulachon specimens collected from the Klamath River in February 1916, March 1947, and 1963, and in Redwood Creek in February 1955. During spawning, fish were regularly caught from the mouth of the river upstream to Brooks Riffle, near the confluence with Omogar Creek (Larson and Belchik 1998), indicating that this area contains the spawning and incubation, and migration corridor essential features.

Historically, the Klamath River was described as the southern limit of the range of eulachon (Hubbs 1925; Schultz and DeLacy 1935, both *cited in* NMFS 2010e). Other accounts have described large spawning aggregations of eulachon occurring regularly in the Klamath River (Fry 1979; Moyle et al. 1995; Larson and Belchik 1998; Moyle 2002; Hamilton et al. 2005), and occasionally in the Mad River (Moyle et al. 1995; Moyle 2002) and Redwood Creek (Ridenhour and Hofstra 1994; Moyle et al. 1995). In addition, small numbers of eulachon have been reported from the Smith River (Moyle 2002). The only reported commercial catch of eulachon in northern California occurred in 1963, when a combined total of 25 metric tons (56,000 pounds) was landed from the Klamath River, the Mad River, and Redwood Creek (Odemar 1964). Since 1963, the run size has declined to the point that only a few individual fish have been caught in recent years. Moyle (2002) indicates that eulachon have been scarce in the Klamath River since the 1970s, with the exception of three years: they were plentiful in 1988, and moderately abundant again in 1989 and 1998. After 1998, they were thought to be extinct in the Klamath Basin, until a small run was observed in the estuary in 2004. According to accounts of Yurok Tribal elders, the last noticeable runs of eulachon were observed in the Klamath River in 1988 and 1989 by Tribal fishers (Larson and Belchik 1998). Larson and Belchik (1998) reported that eulachon have not been of commercial importance in the Klamath River in recent years, and that their current run strength is completely unstudied.

However, in January 2007, six eulachon were reportedly caught by tribal fishers on the Klamath River. Since the 2010 status review (Gustafson et al. 2010), there are reports of an estimated 7 (McCovey 2011), 40 (McCovey 2012), 112 (McCovey and Walker 2013), and approximately 1,000 adult eulachon being sampled by Yurok Indian tribal biologists in presence/absence surveys using seines and dip nets in the Klamath River in northern California in spring of 2011, 2012, 2013, and 2014, respectively.

G.1.4 Southern Resident DPS Killer Whale (*Orcinus orca*)

G.1.4.1 Species status

The Southern Resident DPS killer whale (Southern Residents) was listed as an endangered species on November 18, 2005 (70 FR 69903). Prior to the ESA listing, NMFS determined that the Southern Resident stock was below its optimum sustainable population and designated it as depleted under the Marine Mammal Protection Act in May 2003 (68 FR 31980). The Recovery Plan for the Southern Resident Killer Whale was completed in 2008 (NMFS 2008c), and most recent 5-year review was completed in 2016 (NMFS 2016b). The 5-year review concluded that Southern Residents should remain listed as endangered.

Three pods, J, K, and L, make up the Southern Resident population. The minimum historical population size of Southern Residents in the eastern North Pacific was about 140 animals. Following a live-capture fishery in the 1960s for use in marine mammal parks, 71 animals remained in 1974. Although there was some growth in the population in the 1970s and 1980s, with a peak of 98 animals in 1995, the population experienced a decline of almost 20 percent in the late 1990s, leaving 81 whales in 2001, largely driven by lower survival rates in L pod. In 2013 and 2014, there were multiple successful pregnancies. However, the population census at the end of 2016 counted only 78 whales, and several deaths in 2017 brought the total of this struggling population to 76 (NMFS 2018a). As of December 2018, the population has decreased to only 74 whales, a historical low in the last 30 years. This includes 22 whales in J pod, 18 whales in K pod, and 34 whales in L pod (NMFS 2019b).

The NMFS Northwest Fisheries Science Center (NWFSC) continues to evaluate changes in fecundity and mortality rates and update the population viability analyses, which now suggests a downward trend in population growth projected over the next 50 years. This downward trend is in part due to the changing age and sex structure of the population, but also related to the relatively low fecundity rate observed over the period from 2011 to 2016 ((NMFS 2016b).

G.1.4.2 Critical habitat

In November 2006, NMFS designated critical habitat for Southern Resident DPS killer whales. NMFS received a petition requesting an expansion of critical habitat to include areas of the Pacific Ocean between Cape Flattery, Washington, and Point Reyes, California, extending approximately 47 miles (76 kilometers) offshore. NMFS accepted the petition and identified the next steps for modifying the critical habitat designation in a 12-month finding in 2015 (80 FR 9682).

Based on the natural history of the Southern Residents and their habitat needs, the following physical or biological features were identified as essential to conservation: (1) water quality to support growth and development; (2) prey species of sufficient quantity, quality, and availability to support individual growth, reproduction, and development, as well as overall population growth; and (3) passage conditions to allow for migration, resting, and foraging. From observed sightings and other data, three “specific areas” were identified in the geographical area occupied by the species, containing important physical or biological features. The designated areas are: (1) the Summer Core Area in Haro Strait and waters around the San Juan Islands; (2) Puget Sound; and (3) the Strait of Juan de Fuca, which comprise approximately 2,560 square miles of marine habitat in the area occupied by Southern Resident DPS killer whales in Washington. Critical habitat includes all waters relative to a contiguous shoreline delimited by the line at a depth of 20 feet relative to extreme high water. Some of these areas overlap with military sites, which are not designated as critical habitat because they were determined to have national security impacts that outweigh the benefit of designation, and are therefore excluded under ESA Section 4(b)(2).On September 18, 2019, NMFS proposed to expand critical habitat (84 FR 49214) to include ocean waters from Cape Flattery, Washington south to Point Sur, California, between the 6.1-meter and 200-meter depth contours. Revised critical habitat would include Northern California coastal habitat at the mouth of Klamath River and would be in the Action Area.

G.1.4.3 Life history

Most mating in the North Pacific is believed to occur from May to October, when all three Southern Resident killer whale pods frequent inland waters (Nishiwaki 1972, Olesiuk et al. 1990, Matkin et al. 1997). However, small numbers of conceptions apparently happen year-round, as evidenced by births of calves in all months. Gestation periods in captive killer whales average about 17 months (Asper et al. 1988, Walker et al. 1988, Duffield et al. 1995). Mean interval between viable calves is 4 years (Bain 1990). Newborns measure 2.2 to 2.7 meters long and weigh about 200 kilograms (Nishiwaki and Handa 1958, Olesiuk et al. 1990, Clark et al. 2000, Ford 2002). Calves remain close to their mothers during their first year of life, often swimming slightly behind and to the side of the mother's dorsal fin. Weaning age remains unknown but nursing probably ends at 1 to 2 years of age (Haenel 1986, Kastelein et al. 2003). Mothers and offspring maintain highly stable social bonds throughout their lives, and this natal relationship is the basis for the matrilineal social structure (Bigg et al. 1990, Baird 2000, Ford et al. 2000).

Southern Resident females appear to have reduced fecundity compared to Northern Residents (Ward et al. 2013, Velez-Espino et al. 2014). Recent evidence indicates several miscarriages among Southern Residents, particularly in late pregnancy (Wasser et al. 2017). The authors suggest this reduced fecundity is largely due to nutritional limitation.

Southern Resident killer whales feed on a variety of fish species and one species of squid (Ford et al. 1998, Ford et al. 2000, Ford and Ellis 2006, Hanson et al. 2010, Ford et al. 2016). Scale and tissue sampling of Southern Residents from May to September indicate that their diet consists of a high percentage of Chinook salmon (monthly proportions as high as greater than 90 percent) (Hanson et al. 2010; Ford et al. 2016). The diet data also indicate that the whales are consuming mostly larger (i.e., older) Chinook salmon. Recently, Ford et al. (2016) confirmed the importance of Chinook salmon to the Southern Residents in the summer months using DNA sequencing from whale feces. Salmon and steelhead made up to 98 percent of the inferred diet, of which almost 80 percent were Chinook salmon. Coho salmon and steelhead are also found in the diet in spring and fall months when Chinook salmon are less abundant. Specifically, coho salmon contribute to more than 40 percent of the diet in late summer, which is evidence of prey shifting at the end of summer towards coho salmon (Ford et al. 1998; Ford and Ellis 2006; Hanson et al. 2010; Ford et al. 2016, as cited in NMFS 2018b). Less than 3 percent each of chum salmon, sockeye salmon, and steelhead were observed in fecal DNA samples collected in the summer months (May through September).

Observations of whales overlapping with salmon runs (Wiles 2004; Zamon et al. 2007; Krahn et al. 2009) and collection of prey and fecal samples have also occurred in the winter months. Preliminary analysis of prey remains and fecal samples taken during the winter and spring in coastal waters indicated the majority of prey samples were Chinook salmon (80 percent of prey remains and 67 percent of fecal samples were Chinook salmon), with a smaller number of steelhead, chum salmon, and halibut (NWFSC unpubl. data, as cited in NMFS 2018b).

Prey consumption rates of Chinook and chum salmon were calculated by Noren (2011) for the adult Southern Resident DPS killer whale population. Chinook and chum salmon were used because they are the most prevalent salmon species in the diet of Southern Resident DPS killer whales. When only subsisting on

Chinook, the daily consumption rate is from 9 to 12 fish/day. Fish consumption increased significantly to 41 to 49 fish/day when the population consumed only chum. These rates are consistent with Osborne's (1999) estimated 28 to 34 salmon/day based on the average size of all five salmon species. Extrapolation of these estimates indicates that a Southern Resident population of 82 whales would eat 289,131 to 347,000 Chinook/year, or 1,222,003 to 1,466,581 chum/year (Noren 2011). This does not, however, account for any other prey species, and is therefore likely an overestimate of potential salmon consumption.

As discussed further in Appendix J, NMFS and Washington Department of Fish and Wildlife (WDFW) (2018) developed a prioritized list of West Coast Chinook salmon stocks that are important to the recovery of the species, based on a model that analyzes how much the ranges of different stocks overlap with the Southern Residents, and giving extra weight to salmon runs that support the Southern Residents when their access to food is limited. Fall and Spring Klamath River Chinook runs are identified as two of the top ten priority Chinook populations for the recovery of Southern Resident killer whales (NMFS and WDFW 2018).

G.1.4.4 Geographic distribution

All three Southern Resident pods reside for part of the year in the inland waterways of Washington State and British Columbia (Strait of Georgia, Strait of Juan de Fuca, and Puget Sound), principally during the late spring, summer, and fall (Heimlich-Boran 1988, Felleman et al. 1991, Olson 1998, Osborne 1999, Ford et al. 2000, Krahn et al. 2002). Pods visit coastal sites off Washington and Vancouver Island (Ford et al. 2000) and travel as far south as Central California, and as far north as Southeast Alaska (NMFS 2008c, Hanson et al. 2013, and Carretta et al. 2017; as cited in NMFS 2018b).

Offshore movements and distribution, primarily during the winter months, are largely unknown for the Southern Resident DPS killer whale. However, satellite-linked tag deployments have found that K and L pods use the coastal waters along Washington, Oregon, and California during non-summer months. Detection rates of K and L pods on the passive acoustic recorders indicate Southern Residents occur with greater frequency off the Columbia River and Westport and are most common in March (Hanson et al. 2013). J pod has also only been detected on one of seven passive acoustic recorders positioned along the outer coast (Hanson et al. 2013).

G.1.4.5 Threats

The NMFS 2008 Recovery Plan for Southern Resident DPS killer whales cites three primary factors that threaten this species: toxic pollution, vessel activity and sound, and the quantity and quality of prey (NMFS 2008c). Southern Resident DPS killer whale survival and fecundity are correlated with Chinook salmon abundance (Ward et al. 2009, Ford et al. 2009). Many salmon populations are themselves at risk, with 9 ESUs of Chinook salmon listed as threatened or endangered under the ESA. Hanson et al. (2010) found that Southern Resident DPS killer whale stomach contents included several different ESUs of salmon, including Central Valley fall-run Chinook salmon. The population of Southern Resident DPS killer whales experienced a dramatic decline in the mid-1990s; and as a consequence, was listed as Endangered under the ESA in 2005.

In 2014, NMFS compiled a 10-year review of the research and conservation efforts to support recovery of the species (NMFS 2014). The report summarizes major research findings, management activities, and remaining knowledge gaps, and discusses the threats currently faced by Southern Residents, as well as actions to be taken to address them. To address the threat of pollution and contamination, NMFS has worked with the Puget Sound Partnership (PSP), a Washington State agency leading the cleanup of Puget Sound. NMFS is also coordinating with the U.S. Coast Guard, WDFW, and DFO to evaluate the need for regulations or areas with vessel restrictions as described in the Recovery Plan. NMFS published final vessel regulations in 2011 (76 FR 20870). Additional research is aimed at understanding the impacts of vessel activity, including acoustic and physical disturbance, environmental contamination, and prey availability (NMFS 2016b).

NMFS evaluates salmon harvest actions under the ESA to ensure that the harvest management regimes will not jeopardize the continued existence of ESA-listed salmon or killer whales, or adversely modify their designated critical habitat. These analyses have concluded that the harvest actions cause small prey reductions but were not likely to jeopardize the continued existence of ESA-listed Chinook salmon or Southern Residents, or adversely modify their critical habitats (NMFS 2016b). In 2011 and 2012, NMFS and Department of Fisheries and Oceans Canada (DFO) appointed an independent science panel to review the effects of salmon harvest on Southern Resident killer whales. Their report concludes that there is little evidence that a reduction in salmon catch would have long-term benefits for Southern Resident killer whales (Hilborn et al. 2012, as cited in NMFS 2016b). The report noted that efforts to restore important Chinook salmon habitat, unlike a reduction in salmon fishery harvest, would likely have greater long-term benefits for the whales (Hilborn et al. 2012, as cited in NMFS 2016b).

Population modeling by Lacy et al. (2017), which considered sublethal effects and the cumulative impacts of threats (contaminants, acoustic disturbance, and prey abundance) concluded that the effects of prey abundance on fecundity and survival had the largest impact on the population growth rate of Southern Residents. Their model indicated that for the population to reach the recovery target of 2.3 percent growth rate, the acoustic disturbance would need to be reduced in half, and the Chinook salmon abundance would need to be increased by 15 percent (Lacy et al. 2017).

Beginning in 2018, NMFS is participating in the Southern Resident Orca Task Force to identify immediate and long-term actions to benefit Southern Resident killer whales (Southern Resident Orca Task Force 2019). The Task Force recommendations prioritize the improvement of habitat for Chinook salmon, including habitat acquisition and restoration, and a significant increase in the production of hatchery Chinook at facilities in Puget Sound, on the Washington Coast, and in the Columbia River.

G.1.4.6 Status in the Action Area

As previously described, Southern Residents primarily occur in the inland waters of Washington State and southern Vancouver Island, although individuals from this population have been observed off coastal California in Monterey Bay, near the Farallon Islands, and off Point Reyes (NMFS 2008c). Limited data from acoustic monitoring, photo-identification, and contaminant signatures in blubber suggest some individuals spend substantial time in coastal waters off the coasts of Washington, Oregon, and northern California (i.e., Monterey Bay, Farallon Islands, and Point Reyes) in winter (Krahn et al. 2002, 2009, and Riera 2012; as cited in Hilborn et al. 2012).

The Action Area includes the Pacific Ocean, where Southern Residents co-occur with Klamath River Chinook salmon. The exact boundaries of the area of co-occurrence cannot be precisely defined based on current information; however, it includes coastal waters ranging from Northern California through Central Oregon, up to the Columbia River. Satellite-tagged whales spent time off the Northern California Coast from January through April (NWFSC unpubl. data, as cited in NMFS 2019a and 2019b). Tagged whales swam within a relatively narrow north-south corridor off the coast of California compared to when they were off the coasts of Washington or Oregon (Hanson et al. 2017). The median depth of waters used by Southern Residents off the Northern California Coast was 45 meters (147.6 feet), and median distance from shore was 6.3 kilometers (3.9 miles) (NWFSC unpubl. data, as cited in NMFS 2019b).

As described previously, Southern Resident killer whale survival and fecundity are correlated with Chinook salmon abundance (Ward et al. 2009, Ford et al. 2009). Many salmon populations are themselves at risk, with nine ESUs of Chinook salmon listed as threatened or endangered under the ESA. Klamath River Chinook salmon populations have declined from historic numbers over the past few decades, and spawning stocks in the mainstem Trinity and Klamath rivers are increasingly supported by hatcheries (NMFS 2009).

According to NMFS (2009), Klamath fall-run Chinook are likely to have numbered 400,000 to 500,000 in the early 1900s, while runs in the last several decades have ranged from below 50,000 to 225,000 fish. The significant decline in the Upper Klamath-Trinity Rivers spring-run Chinook salmon ESU prompted its consideration for listing (NMFS 2018c). Declines in Klamath River Chinook are attributed to several factors, including habitat loss and degradation from dams, diversions, and mining, and disease, particularly for juvenile salmon infected with *C. shasta*.

G.2 USFWS Species

G.2.1 Lost River (*Deltistes luxatus*) and Shortnose Sucker (*Chasmistes brevirostris*)

G.2.1.1 Species Status

USFWS designated the Lost River sucker (LRS) and shortnose sucker (SNS) as endangered under the ESA on July 18, 1988 (53 FR 27130). The designation was based on threats to the population, including the damming of rivers, instream flow diversions, hybridization, competition and predation by exotic species, dredging and draining of marshes, water quality problems associated with timber harvest, the removal of riparian vegetation, livestock grazing, and agricultural practices (53 FR 27130; July 18, 1988). Loss and alteration of lake and stream habitats in the upper Klamath Basin is considered by USFWS as the most important factor in the decline of both species (USFWS 1993, 2019a, 2019b, 2019c). Both species are also listed as endangered by Oregon and California.

USFWS published an initial recovery plan for LRS and SNS in 1993. Both the LRS and SNS were subsequently petitioned for delisting on June 29, 2009 (74 FR 30996). The USFWS found that the petition did not present substantial scientific or commercial information indicating that either species warranted delisting (74 FR 30996). An updated revised recovery plan was published by USFWS in 2013; and most recently, USFWS published 5-year reviews for LRS (2019a) and SNS (2019b), and the Special Status Assessment for the Endangered Lost River Sucker and Shortnose Sucker (2019c).

G.2.1.2 Critical habitat

Final critical habitat for LRS and SNS was designated by USFWS on December 11, 2012 (77 FR 73740). In total, approximately 146 miles (234 kilometers) of streams and 117,848 acres (47,691 hectares) of lakes and reservoirs for LRS and approximately 136 miles (219 kilometers) of streams and 123,590 acres (50,015 hectares) of lakes and reservoirs for SNS in Klamath and Lake counties in Oregon, and Modoc County in California, fall within the boundaries of the critical habitat designation.

Designated critical habitat was occupied at the time of listing and continues to be occupied in 2019. Critical habitat contains the physical or biological features to support life-history processes essential to the conservation of LRS and SNS. Two units were designated for each species, based on sufficient elements of physical or biological features being present to support LRS and SNS life processes (77 FR 73740). For LRS, the two units, which were occupied at the time of listing and are still occupied, are: (1) Upper Klamath Lake Unit, including Upper Klamath Lake and tributaries as well as the Link River and Keno Reservoir, and (2) Lost River Basin Unit, including Clear Lake Reservoir and tributaries. For SNS, the two units, which were occupied at the time of listing and are still occupied, are: (1) Upper Klamath Lake Unit, including Upper Klamath Lake and tributaries, as well as the Link River and Keno Reservoir, and (2) Lost River Basin Unit, including Clear Lake Reservoir and tributaries, and Gerber Reservoir and tributaries (77 FR 73740). The Hydroelectric Reach reservoirs, which are on the Klamath River downstream of Keno Dam, are not designated critical habitat for either sucker species.

The PBFs identified in the critical habitat proposal are as follows: (1) water of sufficient quantity and suitable quality; (2) sufficient spawning and rearing habitat; and (3) sufficient food resources with an abundant forage base, including a broad array of Chironomidae (non-biting midge family), crustaceans (crayfish), and other aquatic macroinvertebrates (77 FR 73740).

G.2.1.3 Life history

USFWS recently completed a Special Status Assessment (2019c) and USBR completed the Klamath Project Operations (KPO) BA (USBR 2018). The Special Status Assessment is meant to serve as the basis for defining the status and environmental baseline for consultation under Section 7 of the ESA (USFWS 2019c cited in USFWS 2019d). The Special Status Assessment and KPO BA include complementary, up-to-date information pertinent to LRS and SNS inhabiting the Action Area. The following sections include information adopted directly from the KPO BA.

LRS and SNS are long-lived, lake-obligate fishes. Annual survival estimates for adults of both species are typically 90 percent; on average, LRS live 20 years, while SNS live 12 years. However, there is substantial variation in life expectancy; the oldest-aged specimens are 57 years for LRS, and more than 30 years for SNS (Scoppettone 1988, Buettner and Scoppettone 1990, Terwilliger et al. 2010.) Reproductive maturity is reached between 4 and 9 years for LRS, and between 4 and 6 years for SNS (Buettner and Scoppettone 1990, Perkins et al. 2000a). Fecundity of females is related to age and size, and other unidentified factors (Perkins et al. 2000a). LRS produce 44,000 to 236,000 eggs per female, whereas SNS produce 18,000 to 72,000 eggs per female (Perkins et al. 2000a).

In Upper Klamath Lake, there are two main spawning aggregations of LRS; those that spawn in the Williamson and Sprague Rivers (tributary-spawner) and those that spawn at springs emanating from the eastern shoreline of Upper Klamath Lake. Currently, known spawning occurs along the shore of Upper Klamath Lake at Sucker, Silver Building, Ouxy, and Cinder Flats springs (Figure 6-1; Shively et al. 2000, Hayes and Shively 2001, Hayes et al. 2002, 2004, Barry et al. 2007b). Both populations of LRS show a high degree of site fidelity, although a small amount of mixing does occur (Hewitt et al. 2018). SNS spawn only in the Williamson and Sprague rivers (Hewitt et al. 2018). Annual spawning migrations for tributary-spawners in Upper Klamath Lake are triggered by average daily temperatures; 50°F (10°C) for LRS, and 54°F (12°C) for SNS (Hewitt et al. 2018). Suckers begin spawning immediately after migrating up the rivers, and peak egg-drift typically occurs within days of peak adult migration (Hewitt et al. 2011, Ellsworth and Martin 2012). Up to seven males may attempt to spawn with a single female, although two males and one female is most common (Buettner and Scoppettone 1990). Both male and female suckers quiver as females broadcast their eggs and males fertilize the eggs. Spawning typically occurs in water ranging from 0.4 to 2.3 feet (0.12 to 0.70 meter) deep (both tributary and shoreline springs populations) over mixed gravel (20 to 64 mm; 0.80 to 2.5 inches) or course cobble (2.5 to 10 inches; 65 to 256 mm). Spawning has been observed in flows ranging from 0.49 to 2.69 feet/second (15 to 82 cm/second) in the tributaries. Eggs settle in the interstitial space in the substrate, and typically develop in 8 days to 3 weeks. The rate of development is dependent on temperature, but other factors such as light conditions have also been identified as factors that change the rate of development (Ellsworth and Martin 2012, Stone and Jacobs 2015).

Suckers in the Clear Lake (LRS and SNS) and Gerber reservoir (SNS) drainages spawn primarily, if not entirely, in the tributary streams (Koch and Contreras 1973, Buettner and Scoppettone 1991, Perkins and Scoppettone 1996, BLM 2000, Barry et al. 2007a, Leeseberg et al. 2007). Migration of Clear Lake suckers up Willow Creek is initiated when stream temperatures reach or exceed 6°C, and when sufficient flows in Willow Creek are available (Hewitt and Hayes 2013). Spawning has been entirely skipped some years when flows and lake elevations were not sufficient for suckers to access Willow Creek, and opportunistic spawning has been observed during high discharge events (Hewitt and Hayes 2013, Burdick et al. 2018).

Larvae

Approximately 1 week after fertilization, eggs develop into larvae, and larvae emerge from gravels about 10 days after hatch (Coleman and McGie 1988, Buettner and Scoppettone 1990). Emerging larvae are about a third of an inch long (7 to 9 mm) and are mostly transparent with a small yolk sac (Buettner and Scoppettone 1990). Larval suckers need to begin feeding before they exhaust their yolk, or they will starve

(The Klamath Tribes 1996, Cooperman and Markle 2003). Larvae spend relatively little time in the tributaries, and they drift toward the lake shortly after emergence (Buettner and Scoppettone 1990, Perkins and Scoppettone 1996, Cooperman and Markle 2003). The majority of larvae from tributary populations egress from the river toward the lake during dark hours (Buettner and Scoppettone 1990, Cooperman and Markle 2003, Ellsworth and Martin 2012), then exit the river current during daylight hours and move to nearshore shallow habitat (Buettner and Scoppettone 1990, Cooperman and Markle 2003). Diurnal peak egress appears to vary among natal sites (Ellsworth and Martin 2012). Although the majority of larval sucker research has been conducted on tributary populations, it is suspected that larval suckers hatched at shoreline spawning areas also emerge from the gravels in greatest numbers at night.

Seasonal timing of drift varies among natal sites and occurs approximately 4 weeks after the peak in adult spawning (Ellsworth and Martin 2012, Hewitt et al. 2018). Shoreline spawned larvae typically emerge in greatest numbers in April, whereas the majority of larvae from tributaries emerge in May or June (Ellsworth et al. 2008, 2011, Ellsworth and Martin 2012). Larval LRS spawned in tributaries typically egress in one large, rapid pulse; whereas SNSs egress in three smaller pulses, of which, the second is the largest (Wood et al. 2014). Larvae enter Upper Klamath Lake at a slower rate since restoration of the Williamson River Delta began in 2007 (Wood et al. 2014). In 2007 (Tulana) and 2008 (Goose Bay), levees built in the 1940s were breached, effectively changing the mouth of the Williamson River, and attempting to bring the Williamson River wetland back to some semblance of its historic, pre-manipulated condition (Wood et al. 2014).

Larval drift and distribution for all populations of suckers throughout Upper Klamath Lake is a function of larval production timing, wind speed and directionality, discharge from the Williamson River, and lake elevation, although other factors also influence distribution (Wood et al. 2014). Generally, the prevailing water current in Upper Klamath Lake moves clockwise from the Williamson River Delta, south along the eastern shoreline, west across the lake north of Buck Island, then north along the western side through the Trench, the deepest location of Upper Klamath Lake (Wood et al. 2014). A smaller portion of the current is directed south of Buck Island out of Upper Klamath Lake and into the Link River (Wood et al. 2014). Winds typically originate from the west from April to July, and the predominant water current is clockwise, although wind directionality and speed vary diurnally, seasonally, and among years (Burdick and Brown 2010, Wood et al. 2014). When prevailing winds originate from the northwest (which is not typical), the east-shore current is more prominent, and larvae exit Upper Klamath Lake in larger numbers (Wood et al. 2014). Generally, larval retention (for both tributary and springs populations) in Upper Klamath Lake is lower when river discharge is high, and higher when river discharge is low (Wood et al. 2014). Lake elevation does not appear to affect larval distribution or retention in Upper Klamath Lake except when river discharge is low, and winds are counter-prevailing (from the east; Wood et al. 2014). Based on particle transport models that have been verified with extensive lake-wide larval sampling, the effect of lake elevation on larval distribution is unpredictable, and not suspected to be an effective management tool for increasing larval retention (Wood et al. 2014). However, modeled distribution of larvae (based on hydrodynamics models of water currents, wind speed and direction, and lake elevation) failed to predict high densities of larvae captured in the northern part of the lake, suggesting that larval retention may be higher than predicted (Wood et al. 2014). Other factors that may influence larval retention and distribution are changes in lake elevation, the rate lake elevation changes, the initial distribution of larvae, or some other factor (Wood et al. 2014).

Once in Upper Klamath Lake, peak larval sucker catches occur in late May or early June (Cooperman and Markle 2000; Simon et al. 1996, 2000, 2009; Burdick et al. 2009a). Larval suckers are found throughout Upper Klamath Lake; however, the highest concentrations of larvae are generally near the mouth of the Williamson River, and in emergent wetlands (Simon et al. 1995, 1996, 2009; Burdick et al. 2009b; Cooperman and Markle 2003). Diurnal peak egress appears to vary among natal sites (Ellsworth and Martin 2012). Although the majority of larval sucker research has been conducted on tributary populations, it is suspected that larval suckers hatched at shoreline spawning areas also emerge from the gravels in greatest numbers at night.

Larval habitat in Upper Klamath Lake appears to vary between species; SNS are captured more often along the shoreline and are associated with emergent aquatic vegetation, whereas LRS are more common in open-water habitat (Burdick and Brown 2010). Diets of sucker larvae generally consist of pelagic or surface food items, including adult chironomids and indigestible pollen (Markle and Clausen 2006).

Larval sucker ecology and habitat use in the Lost River watershed, particularly Tule Lake, Lost River, and both Clear Lake and Gerber Reservoir, have not been directly studied. Given the lack of direct observations, larval sucker ecology in the Lost River watershed is assumed similar to the observations from Upper Klamath Lake, except for the use of emergent vegetation in some lake environments, because permanent emergent vegetation is generally scarce or absent along the shorelines of Clear Lake and Gerber Reservoir (USBR 2002).

Young-of-Year Juveniles

Larvae typically develop into young-of-the-year (YOY) juveniles by mid-summer. Transition from larvae to juvenile includes changes in physiology, diet, behavior, and ecology. Suckers are considered juveniles at about ¾- to 1-inches total length (20 to 30 mm; Markle and Clausen 2006). Very few studies aimed at identifying prey items for larval and juvenile suckers have been conducted, and those that have been conducted are relatively inconclusive. However, juvenile suckers appear to consume more benthic-oriented prey items than larvae (predominantly pelagic or surface items), and this change in feeding ecology has been characterized as a developmental milestone (Markle and Clausen 2006). Identifiable prey items of juveniles (longer than 40 mm) include chironomid larvae and pupae, chydorids, ostracods, and harpacticoid copepods (Markle and Clausen 2006). Age-0 juveniles longer than 45 mm are habitat generalists and use all available habitat types in Upper Klamath Lake; they are found near-shore, off-shore, and in vegetated and open-water habitats (Buettner and Scoppettone 1990; Simon et al. 2000, 2009; Hendrixson et al. 2007a, 2007b; Terwilliger et al. 2004; Burdick et al. 2009b; Burdick and Martin 2017).

Although adult LRS are about four times more abundant and are more fecund (females produce more eggs) than SNS, juvenile LRS are not proportionally more abundant. For example, in 2016, juvenile LRS only made up 51 percent of all suckers captured in 2016, 25 percent were SNS, 21 percent had genetic information from both species, and 3 percent were not identified to taxa (Burdick et al. 2018). Catches of age-0 suckers in Upper Klamath Lake are typically highest in August, when suckers are greater than 45 mm standard length (SL) (Burdick and Martin 2017). Catches generally decline throughout August, September, and October; and very few age-1 and almost no age-2 juvenile suckers are captured each year (Simon and

Markle 2001, Terwilliger et al. 2004, Terwilliger 2006, Simon et al. 2009, Korson et al. 2011, Korson and Kyger 2012, Burdick and Martin 2017).

Some of the reduced abundance may be associated with advection from Upper Klamath Lake, including both emigration and entrainment (Markle et al. 2009). Directed movement patterns from north to south of age-0 juveniles were detected once in 2004 (Hendrixson et al. 2007b), but this trend was not apparent in other years (2001 to 2003 and 2005 to 2009; Hendrixson et al. 2007a, 2007b; Bottcher and Burdick 2010; Burdick and Martin 2017). Advection of age-0 suckers from Upper Klamath Lake into the Link River is greatest between July and October, generally peaking in August (Gutermuth et al. 1999, 2000a, 2000b; Foster and Bennetts 2006; Tyler 2007; Markle et al. 2009). Advection of suckers from Upper Klamath Lake may be a passive act indicative of compromised health. Generally, juvenile suckers (and other fishes) captured from the pumped fish bypass at the A Canal fish screen and headgates (at the southern end of Upper Klamath Lake where advection occurs), have higher parasite loads, more disease, and more afflictions than suckers captured elsewhere in Upper Klamath Lake (S. Foott; personal communication, August 2018).

The cause(s) of advection of juvenile suckers is not currently understood. Plausible hypotheses include passive movement due to compromised health, natural emigration, avoidance of or impairment from poor water quality events, diminished habitat in the northern end of Upper Klamath Lake (which may concentrate suckers in the southern end of Upper Klamath Lake near the outlet), entrainment, or some other factors (USFWS 2002c, 2008a). Although entrainment may account for some reductions in abundance, poor juvenile sucker survival (high mortality) appears to be the actual cause of reduced abundance of juvenile suckers (Burdick and Martin 2017). Poor juvenile sucker survival has resulted in essentially no substantial recruitment of juveniles into the adult spawning population since a relatively large cohort born in the early 1990s survived (Burdick and Martin 2017, Hewitt et al. 2018). The cause of widespread juvenile mortality is unknown, but it is likely that some combination of poor water quality, disease, parasites, loss of habitat, non-native species (fish and cyanobacteria), and predation interact to reduce annual survival of juveniles to near zero.

In contrast to Upper Klamath Lake, the majority of adult and juvenile suckers in Clear Lake are SNS, or introgressed SNS/KLS (Hewitt and Hayes 2013); for example, 80 percent of juveniles captured in 2016 were SNS or SNS/KLS, 17 percent were LRS, and 2 percent were introgressed LRS/SNS (Burdick et al. 2018). As discussed earlier, the differences between KLS and SNS are not visually apparent at this life stage, and genetic tools to differentiate between SNS and KLS are not available. Little is known about juvenile sucker distribution and habitat use in Clear Lake; but when reservoir elevations are high and both lobes have water (the East Lobe may be dry or extremely shallow some years), juvenile suckers are found almost equally in both lobes. For example, in 2016, 56 percent of juvenile suckers were captured in the West Lobe. Interestingly, the majority (77 percent) of juvenile LRS captured in 2016 were in the shallower East Lobe (Burdick et al. 2018).

The abundance of age-0 suckers in Clear Lake during any given year is associated, at least in part, with the ability of adult suckers to make a spawning run up Willow Creek (Hewitt and Hayes 2013). Adult suckers in Clear Lake have skipped spawning during years when access to spawning tributaries is limited or made smaller runs (fewer individuals) when spring inflows and/or reservoir elevation limited access (Burdick et al. 2018). Recent years that produced larger-year classes had lake elevations of at least 4,524 feet (1,378.9 meters) during the February-to-May spawning run (Burdick et al. 2018). Lake elevations or tributary inflows were too low from 2013 to 2015 for adult suckers to make large spawning runs in Willow Creek; therefore, very few juveniles were present in Clear Lake until 2016 (Burdick et al. 2018). In 2016, juvenile suckers were found in both lobes, although sampling in the East Lobe in September was limited due to low lake elevations (Burdick et al. 2018).

Older Juveniles

Relatively little is known about habitat use, diet, and ecology of age-1 and older juvenile suckers. A few age-1 suckers are captured each year; they are typically captured in water at least equal to or greater than 3.28 feet (1 meter), because this depth is effectively sampled by trap nets. As lake elevations in Upper Klamath Lake decline throughout the summer, some areas (like wetlands near the Williamson River Delta) become inaccessible for sampling, which limits researchers' ability to fully assess changes in abundance relative to habitat type and depth in Upper Klamath Lake (Burdick 2012). Captures of juvenile suckers older than age-1 are extremely rare, and trends are not discernable from sparse data. However, the real limitation in Upper Klamath Lake is poor survival of age-0 and age-1 juveniles. Older juveniles are captured in Clear Lake; however, few extensive studies of juveniles in Clear Lake have been conducted. A consistent juvenile sucker monitoring program began in 2016 but followed several years of limited (2013) or no (2014 and 2015) adult sucker spawning in Willow Creek, an important tributary to Clear Lake for sucker spawning, due to inaccessibility of spawning grounds (Burdick et al. 2018).

Extensive habitat use studies similar to those in Upper Klamath Lake have not been conducted in Clear Lake. Unlike Upper Klamath Lake, the Clear Lake ecosystem is more homogeneous, primarily varying by depth. There are no surrounding wetlands, and there is limited submergent or emergent vegetation. However, juvenile suckers are found throughout Clear Lake.

Adults

Distribution of adult suckers in Upper Klamath Lake varies seasonally. In winter and fall, adult suckers are distributed throughout Upper Klamath Lake. In the spring, adult suckers congregate in the northeastern portion of the lake, staging prior to making their spawning migration (Hewitt et al. 2018). After spawning occurs (described in the previous section), suckers return to Upper Klamath Lake. As summer progresses and water quality conditions decline, suckers congregate in the northern portion of Upper Klamath Lake (Reiser et al. 2001, Banish et al. 2009). When water quality conditions become especially stressful, adult suckers seek refuge in or near Pelican Bay, where springs provide cooler water and higher dissolved oxygen concentrations (Banish et al. 2007, Banish et al. 2009). Many suckers moved to the western side of Upper Klamath Lake into the Eagle Ridge trench in mid-September (Banish et al. 2007, Banish et al. 2009).

After suckers return from spawning locations in Upper Klamath Lake, suckers are found at various depths, but are most often associated with depths greater than 6.56 feet (2 meters). Depths greater than 6.56 feet (2 meters) are thought to provide adequate cover and protection from avian predators, including American white pelicans (*Pelecanus erthrorhynchos*), and provide for adequate food resources (Banish et al. 2007, Banish et al. 2009). In the summer, SNS and LRS prefer depths greater than 6.56 feet (2 meters) and 9.84 feet (3 meters), respectively, but are not found in the deepest waters of Upper Klamath Lake where water depths are greater than 16.4 feet (5 meters) (Banish et al. 2007, Banish et al. 2009). When water quality conditions deteriorate, adult suckers may select depths less than 6.56 feet (2 meters) near springs where conditions are better (Banish et al. 2007, Banish et al. 2009). Many suckers moved into the deepest part of Upper Klamath Lake (up to 49 feet; 15 meters), the Eagle Ridge Trench, in mid-September (Banish et al. 2007, Banish et al. 2009).

In Tule Lake, where much of the lake is shallower than 3.28 feet (1 meter), adult suckers are found primarily in the very limited areas where depths are greater than 3.28 feet (1 meter; Hicks et al. 2000, USBR 2000).

Adult sucker distribution in Clear Lake has not been specifically studied; however, inferences can be made from other fish-sampling efforts there. Adult suckers in Clear Lake are sampled each fall, and are found throughout the West Lobe and in the East Lobe when lake elevations are high enough for safe boat access (B. Hayes, pers. comm., October 19, 2018). Adult suckers appear to exhibit schooling behavior as researchers typically capture many or few suckers in trammel nets (B. Hayes, pers. comm., October 19, 2018). In the West Lobe, the majority of suckers have been captured in either the north or south, but large numbers of suckers have also been captured in central quadrants (B. Hayes, pers. comm., October 19, 2018). Lake level and weather conditions may influence captures and distribution (B. Hayes, pers. comm., October 19, 2018).

Relatively little is known about the diets of suckers; however, the terminal mouth morphology and triangle gill rakers of LRS indicate they may be primarily benthic feeders. The subterminal or terminal mouth orientation and branched gill rakers of SNS may indicate a more pelagic diet that may include filter-feeding zooplankton from the water column (Miller and Smith 1981, Scoppettone and Vinyard 1991).

G.2.1.4 Geographic Distribution within the Action Area

The historical range of LRS and SNS has been severely impacted by the drainage of Lower Klamath and Tule lakes, wetland loss around Upper Klamath Lake, and alteration of river and spring habitats in the Upper Klamath Basin. Both species are endemic to the Upper Klamath Basin, including Upper Klamath Lake and tributaries, and the Lost River and its tributaries. Both species continue to persist in Upper Klamath Lake and tributaries, Clear Lake and tributaries, the Lost River, Tule Lake, and in Klamath River impoundments downstream to J.C. Boyle Reservoir. SNS populations are also present in Gerber Reservoir, Copco No. 1 Reservoir, and Iron Gate Reservoir. Extirpated populations include populations formerly associated with Lower Klamath Lake (including Sheepy Lake), Lake of the Woods, and at spring systems in Upper Klamath Lake, including Barkley Spring and springs along the northwestern shoreline near Pelican Bay (USBR 2019). In general, the quantity of suitable stream/river, lake, and wetland habitats have been reduced by approximately 75 percent (USFWS 2007b; USFWS 2007c) compared to pre-settlement conditions.

Larval, juvenile, and adult suckers are known to emigrate from Upper Klamath Lake into the Link River. The number of emigrating suckers varies annually, likely based on sucker reproduction and other factors such as water quality and lake level. USFWS estimated that up to 2.33 million larvae, 31,627 juveniles, and 111 adults could be entrained annually at Link River Dam and the A Canal. PacifiCorp (2018) discontinued operations at the West Side and East Side Power Canals as a measure to reduce take of listed suckers in Upper Klamath Lake.

Emigrant suckers occupy waterbodies downstream of Upper Klamath Lake, including Lake Ewauna (Kyger and Wilkens 2011), and Copco No. 1 Reservoir, and J.C. Boyle and Iron Gate reservoirs (Beak Consultants 1987, Buettner and Scoppettone 1991, Desjardins and Markle 2000, Renewal Corporation 2020). Although LRS and SNS are known to emigrate from Lake Ewauna back to Upper Klamath Lake via the Link River fish ladder (Kyger and Wilkens 2011), LRS and SNS inhabiting the Hydroelectric Reach reservoirs are considered “sink” populations, because these fish no longer interact with LRS and SNS in Upper Klamath Lake due to steep channel gradients between J.C. Boyle Reservoir and Keno Dam, and poor fish passage conditions in the Keno Dam fish ladder. Additionally, LRS and SNS in the Hydroelectric Reach reservoirs are believed to have low reproductive success due to limited spawning habitat, abundant non-native predatory fish species, and poor water quality. As described in Chapter 4, PacifiCorp finalized an HCP for LRS and SNS in November 2013 (PacifiCorp 2013b). The HCP addressed direct effects to suckers, including entrainment at Project diversions, false attraction at Project tailraces, ramp rates, lake level fluctuations, migration barriers, loss of habitat, and water quality, as well as effects to sucker critical habitat.

As of 2019, the overall distribution of LRS and SNS has not changed substantially at the sub-basin scale since the original species’ listings (USFWS 1988).

G.2.1.5 Threats

The Special Status Assessment includes on-going threats to LRS and SNS persistence (USFWS 2019c). Predominant threats to listed suckers are past and continued loss of spawning and rearing habitats, water diversions, entrainment into irrigation systems, competition and predation by introduced species, disease and parasites, hybridization with other sucker species, isolation of remaining habitat due to barriers, and effects of climate change such as increased frequency and intensity of droughts (USFWS 1988; CDFG 2005; USFWS 2013a; USFWS 2019a).

Water quality impairment related to nutrient-rich basin soils, wetland conversion, timber harvest, dredging and filling activities, removal of riparian vegetation, and livestock grazing may also cause problems for these species (USFWS 1988). Most water bodies currently occupied by LRS and SNS do not meet water quality standards for nutrients, dissolved oxygen, temperature, and pH set by Oregon and California (Boyd et al. 2002; Kirk et al. 2010). These conditions (primarily in summer) have been associated with several incidents of mass adult mortality, which appears to be a consequence of inadequate amounts of dissolved oxygen (Perkins et al. 2000b). The occurrence of mass mortality of fish in Upper Klamath Lake is not new; however, it is believed that the increased dominance of Aphanizomenon flos-aquae (AFA), a blue-green algae, in the system leads to increased regularity of extreme events (NRC 2004). Although conditions are most severe in Upper Klamath Lake and Keno Reservoir, fish throughout the basin are vulnerable to water-quality-related mortality (USFWS 2007b, 2007c). Degraded water quality conditions may also weaken fish, and increase their susceptibility to disease, parasites, and predation (Holt 1997; Perkins et al. 2000b; ISRP 2005).

The primary, short-term threat to the persistence of LRS and SNS in Upper Klamath Lake is the prolonged lack of substantial and sustained recruitment of new individuals into spawning populations (Hewitt et al. 2015). The following information is adopted from USBR (2018).

> *"Adult LRS in [Upper Klamath Lake] have relatively high survivorship; however, there has been little to no recruitment of juveniles into adult populations (Hewitt et al. 2018). Mark-recapture analyses of adult LRS from the lakeshore-spawning subpopulation in [Upper Klamath Lake] indicate annual survival from 2000 to 2015 ranged from 88 to 96 percent for females, and 80 to 98 percent for males (Hewitt et al. 2011, 2012, 2018). LRS from the tributary-spawning subpopulation had annual survival ranging from 88 to 95 percent for females, and 70 to 96 percent for males during this same time period. Despite high survival for most years from 1999 to 2015, the abundance of LRS males in the lakeshore-spawning subpopulation declined approximately 64 percent and the abundance of females declined by approximately 56 percent (Hewitt et al. 2018). Preliminary data from USGS reports that lakeshore-spawning LRS have experienced additional declines of approximately 20 percent from 2016 to the spring of 2018. The abundance of tributary-spawning LRS is likely 32 percent of what it was in 1999 (E. Janney and D. Hewitt, USGS, pers. comm., 16 August 2018). The estimated abundance of lakeshore spawning LRS in [Upper Klamath Lake] is approximately 7,200 individuals (E. Janney and D. Hewitt, USGS, pers. comm., 16 August 2018). Individuals in this population have exceeded the average life expectancy for the species.*
>
> *Changes in abundance for LRS in the tributary spawning sub-population is less clear. Current population assessments suggest that minor recruitment events may have occurred for tributary- spawning LRS, but overall, the decline of both LRS spawning groups from 2000 to 2015 is probably greater than 40 or 50 percent (Hewitt et al. 2012). The declines primarily reflect a lack of recruitment of new individuals into the spawning populations, but reduced survival of LRS occurred some years (Hewitt et al. 2012). Preliminary data from USGS reports that tributary-spawning LRS have experienced additional declines of approximately 50 percent from 2016 to the spring of 2018. The abundance of tributary-spawning LRS is likely 30 percent of what it was in 2001 (E. Janney and C. Hewitt, USGS, pers. comm., 16 August 2018). The estimated abundance of tributary-spawning LRS in [Upper Klamath Lake] is approximately 32,000 individuals (E. Janney and D. Hewitt, USGS, pers. comm., 16 August 2018). Individuals in this population have exceeded the average life expectancy for the species.*
>
> *Annual survival for SNS in [Upper Klamath Lake] has been lower than either population of LRS. Mark- recapture analyses of adult SNS indicate annual survival from 2000 to 2015 ranged from 68 to 95 percent for females, and 74 to 90 percent for males (Hewitt et al. 2011, 2012, 2018). Similar to tributary-spawning LRS, recruitment events of new individuals into the SNS spawning population is less clear. Recruitment events may have occurred in some years though substantial data supporting these events is not comprehensive. The SNS population has declined more than declined by 78 percent and the abundance of females*

declined 77 percent (Hewitt et al. 2018). Preliminary data from USGS reports that SNS have also experienced additional declines of approximately 40 percent from 2016 to the spring of 2018. The abundance of SNS is likely 20 percent of what it was in 2001 (E. Janney and D. Hewitt, USGS, pers. comm., 16 August 2018). The estimated abundance of SNS in [Upper Klamath Lake] is approximately 7,900 individuals (E. Janney and D. Hewitt, USGS, pers. comm., 16 August 2018). Individuals in this population have exceeded average life expectancy and are near the maximum known age for the species (33 years).

Despite relatively high annual survivals from 2000 to 2015 both species have experienced substantial declines in abundance because losses from mortality have not been balanced by recruitment of new individuals (Hewitt et al. 2011, 2012, 2018). All adult sucker populations in [Upper Klamath Lake] appear to be largely comprised of fish that were present in the late 1990s and early 2000s (Hewitt et al. 2011, 2018). Survival analyses show that the two species do not necessarily experience poor survival in the same years and that poor survival on an annual scale is not predictable from fish die-offs observed in the summer and fall (Hewitt et al. 2011). However, little to no recruitment has occurred into these sucker subpopulations in the last 20 years (Hewitt et al. 2011, 2012, 2018)."

G.2.1.6 Status of Populations within the Action Area

This section describes the status of the species in the Upper Klamath Basin and Action Area. The Action Area is less than the Upper Klamath Basin because the Lost River Basin, aside from Tule Lake and a 7.5-mile reach of the Lost River between Anderson-Rose Dam and Tule Lake, is not included in the Action Area. The following section is adopted directly from USBR (2018).

Upper Klamath Lake Population

The following section is adopted from USFWS (2019a).

Upper Klamath Lake likely contains the largest remaining populations of both LRS and SNS, although the SNS population in Clear Lake may be similar in size. Although robust abundance estimates are difficult for this population due to low recapture rates of tagged fish, these recapture rates can be used to obtain rough estimates of abundance. Over the last decade, abundance estimates were roughly 100,000 adult LRS river-spawners, 8,000 adult LRS shoreline-spring-spawners, and 19,000 adult SNS (Hewitt et al. 2014 p. 16). However, in 2018, the estimates of fish participating in spawning aggregations were estimated to be much lower: 32,000, 8,000, and 7,000, respectively (D. Hewitt, USGS, personal communication August 16, 2018). These estimates may not reflect the true population size due to the statistical challenges of estimating abundance from the available data, particularly if some individuals skipped spawning in 2018. Overall, the populations in Upper Klamath Lake are characterized by high annual survival of adults (Hewitt et al. 2018 pp. 12, 17, 21). These adults spawn successfully and produce larvae, but few juveniles survive their first year, and captures of individuals 2 to 6 years old is exceedingly rare (Burdick and Martin 2017 p. 30).

Similarly, there has not been evidence of significant numbers of new individuals joining the adult spawning populations since the late 1990s (Hewitt et al. 2018 p. 24), and the lack of significant recruitment has led to sharp declines in population sizes (Hewitt et al. 2018 pp. 14, 20, 24).

Survival of adult SNS and LRS in Upper Klamath Lake varied little over the past decade. Annual adult survival rates of the SNS in Upper Klamath Lake appear to vary more than the LRS, but adult survival for both species in Upper Klamath Lake appears to have been relatively stable since high-quality estimates became available in the early 2000s (Hewitt et al. 2018 pp. 12, 17, 21). Adult LRS in Upper Klamath Lake average approximately 93 percent survival annually (Hewitt et al. 2017 pp. 15, 21). The approximate average adult SNS annual survival in Upper Klamath Lake is slightly less at 87 percent (Hewitt et al. 2017 p. 28). However, preliminary data indicate that survival from spring 2016 to spring 2017 (i.e., 2016 survival) was low for both species, in some cases lower than has been observed during the period with robust estimates. For SNS, preliminary estimates for 2016 survival are 77 percent for females and 74 percent for males. The preliminary estimates of survival for both sexes are 78 percent for LRS spawning in the Williamson River and 85 percent for LRS spawning at the lakeshore springs (D. Hewitt, USGS, personal communication, August 16, 2018). Additionally, hundreds of dead adult suckers were observed during a die-off in the summer of 2017.

Juvenile mortality and the resulting lack of recruitment of new individuals into the adult populations have led to steep declines in LRS and SNS populations in Upper Klamath Lake. Although there is uncertainty about the rates of decline, the best available estimates indicate that the LRS lakeshore springs spawning population declined by approximately 56 percent for females and 64 percent for males between 2002 and 2015 (Hewitt et al. 2018 p. 10, Figure 6-3). The decline in the Williamson River LRS population is more difficult to assess due to sampling issues specific to that population (Hewitt et al. 2018 pp. 25–26), but it is likely that the population dynamics are similar to those of the shoreline springs population. The SNS population in Upper Klamath Lake has also declined substantially since 2001, losing approximately 77 percent of females and 78 percent of males between 2001 and 2016 (Hewitt et al. 2018 p. 19, Figure 6-3).

Recent LRS and SNS size distribution trends reveal that the adult spawning populations in Upper Klamath Lake are composed of similar-sized, similar-age relatively old individuals. Median lengths of individuals of both species in Upper Klamath Lake generally increased since between the 1990s and 2010; but since about 2010, size distributions have been more or less stable among years (Hewitt et al. 2018, pp. 19, 22–23, 27, 29). This indicates that few new individuals are joining the adult populations. The fish recruited in the 1990s are now approximately 28 years old and are well beyond the average survival: past maturity of 12 years for the SNS, and equal to that of 20 years for the LRS.

The effects of senescence on the survival and reproduction of these two species are unknown at present, but the populations in Upper Klamath Lake are clearly aging (Hewitt et al. 2018 pp. 15, 18, 21). The low recent survival rates could be an early signal that senescence is leading to increased mortality rates and accelerated population declines. Additional years of survival data will help to resolve whether the low survival reveals increased mortality of aging individuals or unique environmental conditions to that year.

Both species spawn successfully in the Sprague River, producing larvae that drift downstream to Upper Klamath Lake. Captures of 1,000s to 10,000s of larvae from the Sprague and Williamson rivers (Cooperman and Markle 2003 pp. 1146–1147; Ellsworth and Martin 2012 p. 32) conservatively suggest that combined larval production of both species is on the order of 1,000,000s: note that these numbers are rough estimates and not a characterization of inter-annual variation, which is also substantial. Successful spawning in the Sprague River suggests that the needs of both species for spawning access and suitable egg incubation habitat are at least minimally met; however, available information does not permit comparisons with historical conditions.

LRS also spawn successfully at groundwater seeps along the Upper Klamath Lake margin. No robust estimates of larval production at these sites exist; but given the number of LRS females and average fecundity, it is likely that millions of larvae hatch annually, even with the expected high mortality of eggs. There is typically access to these areas between February and May; however, lake elevations lower than approximately 4,141.4 to 4,142.0 feet (1,262.3 to 1,262.5 meters) reduce the number of spawning individuals and the amount of time spent on the spawning grounds. Upper Klamath Lake elevations less than 4,142.0 feet (1,262.5 meters) occurred by May 31 in 6 years between 1975 and 2017, which is equivalent to 14 percent of spawning seasons. Therefore, lake elevations have the potential to negatively impact spawning for LRS, but this has rarely occurred over the last 43 years.

Although numerous larvae are produced annually, the number of juveniles captured during sampling efforts is low, and typically decreases to nearly zero in late summer. Very few individuals are captured as age-1 or older (Burdick and Martin 2017 p. 30), suggesting complete cohort failure each year. The declines in captures commonly occur during the periods with the most degraded water quality conditions in Upper Klamath Lake, but a clear empirical link between water quality parameters and mortality rates has not been established. One prominent hypothesis is that water quality is directly responsible for the unnaturally high levels of juvenile mortality. Another is that water quality interacts with other sources of mortality by causing chronic stress that renders the individuals more susceptible to forms of predation or infection (USFWS 2019 pp. 21–41). The specific causes of repeated cohort failure at the juvenile stage are a critical uncertainty challenging recovery because juvenile mortality is the primary factor that contributes to the low resilience of both LRS and SNS populations in Upper Klamath Lake.

Even though viable eggs and larvae are produced each year, there is a lack of recruitment of new adults into Upper Klamath Lake sucker populations, which continue to exist only because of their long life. Although we do not know specifically how this current uniform age distribution compares to historical conditions, healthy adult populations of long-lived species should generally possess multiple reproducing year-classes. Both species are expected to become extirpated from Upper Klamath Lake without significant recruitment, but the current dynamics are particularly untenable for the SNS, and without substantial recruitment in the next decade, the population will be so small that it is unlikely to persist without intervention (Rasmussen and Childress 2018, p. 586).

Tule Lake Population

The following information is adopted from USBR (2018). “Project” in this section refers to USBR’s Klamath Project.

Historically, Tule Lake was a 95,000-acre shallow lake with a small border of fringe wetlands and hosted one of the largest sucker populations. Now located within Tule Lake National Wildlife Refuge (NWR), Tule Lake has been reduced to approximately 10,500 acres of open water and 2,500 acres of shallow wetlands (Hicks et al. 2000). The Lost River and return flows from the Project provide water to Sump 1A and Sump 1B, the deepest, separated remnants of the historic lake (Hicks et al. 2000, USBR 2007). Approximately 17,000 acres of farmland, acres that are part of the Tulelake NWR, surround Tule Lake (Hicks et al. 2000). This refuge was established by an executive order dated 1928. The refuge supports many fish and wildlife species and provides suitable habitat and resources for migratory birds of the Pacific Flyway. Sumps 1A and 1B are refuge facilities that are managed to meet flood control and wildlife needs, including the needs of endangered suckers. USBR, through a contract with the Talent Irrigation District, manages deliveries from the sumps and pumping from Pumping Plant D to aid Tule Lake NWR in maintaining the elevations necessary in the sumps to meet wildlife needs and requirements (USBR 2007).

Both LRS and SNS reside in Sump 1A, the larger sump of Tule Lake. The current number of suckers in Tule Lake sumps are relatively small, probably in the hundreds, possibly the low thousands of individuals, and is dominated by adults (Hodge and Buettner 2007, 2008, 2009). Surface elevations in Sump 1A have been maintained for a minimum elevation of 4,034.0 feet from October 1 through March 31, and a minimum elevation of 4,034.6 feet from April 1 through September 30 each year since the 1992 BO (USFWS 1992), including operations under the 2013 BO (NMFS and USFWS 2013).

Historically, populations of suckers in Tule Lake migrated up the Lost River to spawn at Big Springs near Bonanza, Oregon (RM 45), and probably other shallow riffle areas with appropriate spawning substrate (Coots 1965, ISRP 2005). Access to spawning areas in the Lost River is blocked by upstream diversion dams, including the Lost River Diversion Dam (1912), Anderson-Rose Diversion Dam (1921), and Harpold Dam (1926). Currently, spawning migrations from Tule Lake are limited to a 7-mile portion of the lower Lost River downstream of Anderson-Rose Diversion Dam (Hodge and Buettner 2008).

USBR and the USFWS have monitored endangered spawning runs from Tule Lake into the Lost River infrequently since 1991 (USBR 1998; Hodge and Buettner 2007, 2008, 2009). Spawning is restricted to one riffle area downstream of Anderson-Rose Dam. Spawning runs have occurred in years that Anderson-Rose Dam spills or releases water. Releases were required as provisions of earlier BOs (USFWS 1992, 2001, 2008a). For example, in 2006 and 2007, the Service entered into an agreement with Talent Irrigation District to provide releases during the spawning season (USFWS 2008a). Successful egg incubation and survival of larvae to swim-up has been infrequent in recent years (Hodge and Buettner 2008, USFWS 2008a). Only two juvenile suckers were captured in Tule Lake in 2007, suggesting recruitment continues to be low (Hodge and Buettner 2008). Water levels in Tule Lake Sumps have been managed according to criteria set in previous BOs (USFWS 2002c). From April 1 to September 30, a minimum elevation of 4,034.6 feet was set in part to provide access to spawning areas downstream of Anderson Rose Diversion Dam (USFWS 2008a).

Minimum flows downstream of Anderson-Rose Dam were also previously required by the 2008 BO on Project operations. However, in 2009, the 2008 BO was amended, and those flows were no longer required, as the USFWS stated in their letter dated January 6, 2009 (Reference # 8-10-09-F-070070), "…that habitat conditions in Tule Lake negatively influence recruitment far more than flows at Anderson-Rose Dam, and therefore, we determined that Term and Condition #2 [flows downstream of Anderson-Rose Dam for spawning] is no longer necessary to minimize take of endangered suckers." Today, there are no minimum flows downstream of Anderson-Rose Dam. Stranding of adult and juvenile suckers downstream of Anderson-Rose Dam occurred in the spring of 2016, when flows downstream of the dam receded quickly. USBR coordinated with Talent Irrigation District in the summer of 2016 to install automatic gate controls at the dam that provides Talent Irrigation District with much more control over spill situations at Anderson-Rose Dam; the gate sensors will reduce the likelihood of rapidly fluctuating flows and stranding risk to suckers immediately downstream of the dam. The impact these actions have had on juvenile suckers is poor or no survival. The impact these actions have had on adults is less clear, because adult suckers, while not well studied, appear to be surviving.

Water depths in Tule Lake Sumps 1A and 1B are shallow (less than 5 feet deep). However, lack of deep areas in the sumps and the gradual sedimentation that appears to be occurring (USFWS 2002c) is detrimental to older juvenile and adult suckers that require water depths greater than three feet to avoid predation by piscivorous birds, particularly pelicans (USFWS 2008a). The USFWS has been investigating options to restore deep water habitat including small-scale dredging and flooding existing agricultural lease lands that have subsided (Mauser 2007, pers. comm. cited in USFWS 2008a). Low elevations in Tule Lake Sumps may lead to increased avian predation. PIT tags from adult suckers in Tule Lake have been found at bird nesting colonies and loafing areas (N. Banet, Fish Biologist, USGS Klamath Falls, personal communication, December 13, 2018).

During severe winters with thick ice cover, only small, isolated pockets of water with depths greater than 3 feet exist, increasing the risk of winter die-offs (USFWS 2008a). The April 1 to September 30 minimum elevation of 4,034.6 feet was set in part to provide rearing habitat in Tule Lake (USFWS 2008a) and the October 1 to April 31 minimum elevation of 4,034.0 feet was set to provide suckers with adequate winter water depths for cover and to reduce the likelihood of fish die-offs owing to low DO concentrations beneath ice cover (2008a). The impact harsh winters have on suckers is not well understood, but harsh winters are likely to reduce body condition and fitness, meanwhile increasing stress and mortality associated with increased levels of parasites, disease, and predation.

Hydroelectric Reach Reservoirs Population

The Renewal Corporation completed four sampling efforts to assess the current abundance, demographics, and genetics of LRS and SNS present in the Hydroelectric Reach reservoirs (Renewal Corporation 2020). The Renewal Corporation used standard techniques developed by the USGS and USFWS to sample J.C. Boyle, Copco No. 1, and Iron Gate reservoirs in fall 2018, spring and fall 2019, and spring 2020. The Renewal Corporation captured 222 LRS, SNS, and potential hybridized LRS or SNS across the three reservoirs over the four sampling periods (Table G-3). Recaptured suckers were used to develop population estimates for the three reservoirs, and for the reservoirs combined. Three different methods were used to develop population estimates; the three methods yielded comparable results. The Renewal Corporation's survey-based population estimates suggest that the total number of adult target suckers is highest in Copco No. 1 Reservoir, slightly less in J.C. Boyle Reservoir, and lowest in Iron Gate Reservoir (Table G-4). The 95 percent confidence intervals suggest that there are several thousand adult listed suckers in Copco No. 1 Reservoir and J.C. Boyle Reservoir, and several hundred adult listed suckers in Iron Gate Reservoir. Due to the number of recaptured suckers over the sampling effort, the 95 percent confidence intervals for the population estimates are large compared to the magnitude of the population estimate (i.e., confidence interval widths greater than ±100 percent of the population estimate for Copco No. 1 Reservoir and J.C. Boyle Reservoir. The 95% confidence interval for the estimated total number of listed suckers across the three reservoirs is between 4,500 and 11,500 suckers (Table G-4).

The Renewal Corporation's results are similar to earlier work completed by Beak Consultants (1987) in Copco No. 1 Reservoir, and by Desjardins and Markle (2000) in J.C. Boyle, Copco No. 1, and Iron Gate reservoirs. Sucker catch in these studies were of similar magnitude to the Renewal Corporation's results, and the Renewal Corporation's catch per unit effort (measure of catch efficiency) was similar to Desjardins and Markle (2000), suggesting LRS and SNS populations in the reservoirs have not substantially changed since the late 1990s. SNS median lengths increase in a downstream direction with the smallest SNS, in J.C. Boyle Reservoir, and the largest SNS in Iron Gate Reservoir. However, Iron Gate Reservoir SNS count was slightly more than a quarter of the count of the Copco No. 1 Reservoir catch. A cohort of smaller SNS sampled in Copco No. 1 Reservoir during spring 2020, reduced the median SNS size from 481 mm to 438 mm in Copco No. 1 Reservoir. This cohort was not represented in previous sampling efforts. The Renewal Corporation also found that median SNS lengths were similar to the median length of SNS sampled by USGS in Upper Klamath Lake (Hewitt et al. 2017) and comparable to Desjardins and Markle (2000).

USFWS recently developed the genetic library for the four Klamath Basin sucker species (Smith et al. 2020). USFWS is developing genetic assays that will be used, in part, to assess the genetic integrity of suckers sampled by the Renewal Corporation in the Hydroelectric Reach reservoirs. The Renewal Corporation's sucker tissue samples will be processed by USFWS using the genetic assays by summer 2021. The Renewal Corporation anticipates the assay results will clarify the genetics and the geographic origin of suckers that were sampled in the Hydroelectric Reach reservoirs. This information will inform the approach to the sucker salvage, which will be completed in advance of the Proposed Action. Additionally, USFWS may use the genetic results for future management of the translocated suckers.

Table G-3: Summary Sucker Sampling Results for Trammel Net Sets and Boat Electrofishing for Fall 2018 through Spring 2020

Sampling Metric	Sampling Event	J.C. Boyle	Copco	Iron Gate	Total	Grand Total
Total net-sets	Fall 2018	30	22	24	76	312
	Spring 2019	40	31	25	96	
	Fall 2019	19	30	36	85	
	Spring 2020	7	36	12	55	
Total net hours	Fall 2018	57.9	33.6	37.3	128.8	649.2
	Spring 2019	55.1	42.4	42.6	140.1	

Sampling Metric	Sampling Event	J.C. Boyle	Copco	Iron Gate	Total	Grand Total
	Fall 2019	36.0	50.3	61.0	147.4	
	Spring 2020	49.7	137.5	45.7	233.0	
Total nets with listed suckers	Fall 2018	13 (43%)	13 (59%)	8 (33%)	34 (45%)	126 (40%)
	Spring 2019	19 (48%)	9 (29%)	1 (4%)	29 (30%)	
	Fall 2019	10 (53%)	13 (43%)	8 (22%)	31 (36%)	
	Spring 2020	6 (86%)	24 (67%)	2 (17%)	32 (58%)	
Total Lost River Suckers	Fall 2018	3	0	0	3	27
	Spring 2019	10	0	0	10	
	Fall 2019	4	1	0	5	
	Spring 2020	9	0	0	9	
Total Shortnose Suckers	Fall 2018	21	11	12	44	185
	Spring 2019	19	16	1	36	
	Fall 2019	9	21	10	40	
	Spring 2020	15	48	2	65	
Total listed or potential hybrid suckers	Fall 2018	27	13	17	57	223
	Spring 2019	30	16	1	47	
	Fall 2019	14	21	10	45	
	Spring 2020	24	48	2	71	
Total listed or potential hybrid suckers catch per unit effort (fish/net-hour) – Does not include electrofishing results (3 suckers)	Fall 2018	0.47	0.39	0.46	0.44	0.35
	Spring 2019	0.54	0.38	0.02	0.34	
	Fall 2019	0.39	0.42	0.16	0.31	
	Spring 2020	0.44	0.35	0.02	0.30	
Total recaptures from same sampling event	Fall 2018	0	0	1	1	7
	Spring 2019	2	1	0	3	
	Fall 2019	2	0	1	3	
	Spring 2020	0	0	0	0	
Total recaptures from previous sampling event	Fall 2018	NA	NA	NA	NA	6
	Spring 2019	0	0	1	0	
	Fall 2019	1	1	1	3	
	Spring 2020	1	1	1	3	

Table G-4: Population Estimate Attributes and Preliminary Estimates for Listed and Potential Hybrid Suckers in Each Reservoir and in All Reservoirs Combined

Population Estimate Attributes	J.C. Boyle	Copco	Iron Gate	All Reservoirs Combined
Total suckers PIT-tagged (fall 2018, spring and fall 2019, and spring 2020)	71	83	27	181
Total maiden suckers captured (fall 2018 through spring 2020)	95	98	29	222
Total tagged suckers recaptured (fall 2018 through spring 2020)	3	3	2	8
Recapture efficiency (# recaptured/# Tagged)	4.2%	3.6%	7.4%	4.4%
Chapman Method - Population estimate	1,727	2,078	279	4,509
Bootstrap Method - Mean population estimate	2,766	3,371	399	5,540
Bootstrap Method – 95% confidence interval upper limit	6,496	7,879	943	11,531
Bootstrap Method - 95% confidence interval	±3,730	±4,508	±544	±5,991
Jolly-Seber Model – Mean population estimate	864	1,235	102	2,201
Jolly-Seber Model – 95% Confidence Interval Upper Limit	1,815	2,609	191	4,615
Jolly-Seber Model - 95% confidence interval	±951	±1,374	±89	±2,414
Note: PIT = Passive Integrated Transponder				

Status of Critical Habitat in the Action Area

Upper Klamath Lake

At approximately 64,000 acres (26,000 hectares), Upper Klamath Lake is the largest remaining contiguous habitat for endangered suckers in the Upper Klamath Basin. Upper Klamath Lake is a natural lake that was dammed in 1921 to allow for management of lake elevations both higher and lower to support irrigation deliveries. Approximately 70 percent of the original 50,400 acres (20,400 hectares) of wetlands surrounding the lake, including the Wood River Valley, was diked, drained, or significantly altered between 1889 and 1971 (Gearhart et al. 1995, p. 7). Spawning aggregations at numerous locations in the Upper Klamath Lake system have disappeared, but LRS continue to use two spawning locations in relatively large numbers: the Williamson River and the eastern shoreline springs, and Upper Klamath Lake contain the largest remaining population of LRS by far. SNS are only known to spawn in significant numbers in the Williamson River.

Tule Lake

Tule Lake was extensively diked, and its volume has been greatly reduced through evaporation related to retention of water upstream of dams and irrigation, as well as diversion of water to the Klamath River and to Lower Klamath NWR through the D Pump. The remaining lake habitat, referred to as Sump 1A and Sump 1B, is approximately 9,081 acres and 3,259 acres, respectively. Hundreds of individuals of both species were captured in Tule Lake Sump 1A during a 3-year effort (Hodge and Buettner 2009, pp. 4–6). Spawning aggregations have been observed in the Lost River downstream of Anderson-Rose Dam, but the habitat is not high quality. Locations in the Lost River where historical spawning was documented, such as Olene, are inaccessible from Tule Lake due to multiple dams and inundation behind dams. Therefore, the Tule Lake populations are considered sinks, entirely composed of the offspring of other populations that found their way through the Lost River or the irrigation system into Tule Lake, and without sufficient means to be self-sustaining.

G.2.1.7 Factors Affecting Critical Habitat in the Action Area

The following section includes factors affecting critical habitat in the Action Area. The Proposed Action will have limited effect on critical habitat, because designated critical habitat is upstream of Keno Dam, and the Proposed Action is focused on the Klamath River downstream of Keno Dam. USFWS (2019a) reviewed factors that affect critical habitat throughout the species' range. The following section includes factors that may affect critical habitat in the Action Area; the information is adopted from USBR (2018).

Water Quality

Upper Klamath Lake

Although Upper Klamath Lake was historically eutrophic (Sanville et al. 1974, Johnson et al. 1985), large-scale watershed development from the late-1800s through the 1900s has likely contributed to the current hypereutrophic condition in Upper Klamath Lake (Bortleson and Fretwell 1993). This legacy, combined with current nutrient loading from the watershed and lake sediment, facilitates extensive cyanobacteria blooms (Boyd et al. 2002) that typically result in large diel fluctuations in DO and pH, high concentrations of the hepatotoxin microcystin, and toxic levels of un-ionized ammonia during bloom decomposition (Boyd et al. 2002, Walker et al. 2012). Together, these conditions create a suboptimal environment for native aquatic biota, and likely play a role in the decline of ESA-listed SNS and LRS (Perkins et al. 2000a). Indeed, in recent decades, Upper Klamath Lake has experienced serious water quality issues that have resulted in fish die-offs, as well as re-distribution of fish in response to changes in water quality (Buettner and Scoppettone 1990, Banish et al. 2007, Banish et al. 2009).

Phosphorus is the key driver of water quality issues in Upper Klamath Lake (Boyd et al. 2002, Walker et al. 2012). Phosphorus occurs in relatively high levels in the local geology of the Upper Klamath Basin (Boyd et al. 2002, Walker et al. 2015), but has been, and continues to be, produced through past and current land use activities in the watershed (Walker et al. 2012, Walker et al. 2015). Specifically, average annual external phosphorus load to Upper Klamath Lake is now approximately 40 percent higher than the natural background (Boyd et al. 2002, Walker et al. 2012). Additionally, the intact riparian areas and lake-fringe wetlands that historically filtered and retained phosphorus have been much diminished, further exacerbating the phosphorus loading issue. These factors, combined with internal loading as a result of current and historical external load (Boyd et al. 2002), result in summer water column phosphorus concentrations up to six times higher than the natural background (NRC 2004).

In 1998, the Oregon Department of Environmental Quality (ODEQ) placed Upper Klamath Lake and its tributaries on the 303(d) list of Oregon waters with impaired beneficial uses (ODEQ 1998). Subsequently, the Upper Klamath Lake Drainage total maximum daily load (TMDL) identified phosphorus as the key pollutant, and recommended total phosphorus loading targets as the primary method to improve Upper Klamath Lake water quality (Boyd et al. 2002). Specifically, the TMDL calls for a 40 percent reduction in external total phosphorus loading to limit the underlying causes of adverse water quality conditions (Boyd et al. 2002). Recent work has indicated that a reduction in external phosphorus loading of this magnitude is likely to result in reduced water column phosphorus concentrations, and thereby an improvement in water quality, over a period of years to decades (Wherry and Wood 2018).

The focus on phosphorus loading and concentrations is critical to disrupt the processes directly linked to water quality issues in Upper Klamath Lake; namely, large cyanobacteria blooms during the growing season (Boyd et al. 2002). Of specific concern is the cyanobacteria species AFA, which has only been present in Upper Klamath Lake since the onset of large-scale watershed development in the late 1800s and early 1900s (Eilers et al. 2004, Bradbury et al. 2004). AFA, a nitrogen-fixing cyanobacteria, now dominates the Upper Klamath Lake phytoplankton community during the growing season, with bloom biomass reaching several orders of magnitude greater than that of other phytoplankton species (Nielsen et al. 2017). During bloom development and proliferation, AFA photosynthesis facilitates an increase in pH (Jassby and Kann 2010, Nielsen et al. 2017), often greater than levels thought to be stressful to SNS and LRS (Loftus 2001). At this same time, increasing water temperature and nighttime AFA respiration combine to reduce DO concentrations, which may pose additional challenges to listed suckers. Typically, by late July or early August, and often in tandem with hot and calm conditions, AFA blooms “crash” (Jassby and Kann 2010, Nielsen et al. 2017), resulting in increased organic biomass available for decomposition at the sediment-water interface. Increased decomposition subsequently results in reduced DO, and possibly increased un-ionized ammonia concentrations, both of which may be stressful or lethal to listed suckers (Saiki et al. 1999, Loftus 2001), depending on the extent and duration of the suboptimal concentrations. In addition to changes in these water quality parameters, AFA bloom crashes increase the amount of available nitrogen for uptake by other phytoplankton, primarily the toxin-producing cyanobacteria *Microcystis aeruginosa* (Jassby and Kann 2010); Upper Klamath Lake is often under an Oregon Health Authority recreational use health advisory for the algal toxin microcystin, produced by Microcystis aeruginosa, by early July. Although there is no clear direct evidence that microcystin negatively affects listed suckers, it is another possible chronic stressor (Martin et al. 2015) and has been implicated in fish die-offs in other locations (Zanchett and Oliveira-Filho 2013). Regardless, adverse water quality events associated with AFA bloom dynamics may have lethal impacts to individual suckers (Perkins et al. 2000b) and may reduce the reproductive capacity of the populations by reducing the numbers of larger and more fecund females (Buchanan et al. 2011). Adverse water quality may also affect young suckers (Buchanan et al. 2011, Hereford et al. 2018), but the existing data have been unable to discern a clear relationship.

As mentioned previously, past and current external phosphorus loading and internal loading (as a result of past external loading) are believed to be key drivers behind AFA bloom dynamics and subsequent water quality issues in Upper Klamath Lake (Boyd et al. 2002, Walker et al. 2012). Additionally, there are specific meteorological conditions that further influence bloom dynamics. Both Wood et al. (1996) and Morace (2007) found a relationship between spring air temperature and the timing of the onset of the AFA bloom.

The onset of the AFA bloom was delayed when spring air temperatures were cooler (Wood et al. 1996, Morace 2007). It has also been hypothesized that smoke or cloud cover can reduce the capacity of AFA to recover after a bloom crash (Morace 2007), which can result in depressed DO concentrations for extended periods. Conversely, a decrease in wind speed and an increase in air temperature and solar radiation in July and August can result in thermal stratification of Upper Klamath Lake, which subsequently creates suboptimal conditions for AFA, and typically leads to a bloom crash (Jassby and Kann 2010, Nielsen et al. 2017).

There is some support for the proposition that Upper Klamath Lake surface elevation may also influence AFA bloom dynamics. For instance, Walker (2010) recommended a specific Upper Klamath Lake elevation trajectory that targets higher lake elevations in the spring and early summer, but then "threads the needle" to avoid lake elevations (both high and low) that facilitate lower DO concentrations and higher un-ionized ammonia concentrations in the late summer and early fall. Specifically, Walker (2010) suggests that higher Upper Klamath Lake elevations reduce AFA biomass by reducing light intensity in the water column, and increasing the ratio of sediment to water volume, thereby diluting the effects of internal phosphorus loading. Conversely, increasing Upper Klamath Lake elevations above certain levels in the late summer increases the likelihood of thermal stratification, thereby exacerbating issues related to low DO, and increasing un-ionized ammonia concentrations (Walker 2010). Previous work (Horn and Lieberman 2005) provides some support for the hypothesis that Upper Klamath Lake depth may affect DO concentrations; however, this work relied on prior Upper Klamath Lake bathymetry, assumed a conservative diffusion coefficient (i.e., assumed slight reaeration due to wind and water surface contact with air), and suggested that the changes in probability of DO concentrations stressful or lethal to suckers changed little over the recent range of Upper Klamath Lake elevations (i.e., those observed since implementation of the 2013 BO, a period which included 3 subsequent years of drought and correspondingly low Upper Klamath Lake elevations).

The most recent and best available science regarding water quality for the purposes of ESA Section 7 consultations has not demonstrated a direct, consistent, and discernible relationship between Upper Klamath Lake elevation and water quality (Wood et al. 1996; NRC 2002; Morace 2007; Jassby and Kann 2010; Nielsen et al. 2017; Wherry and Wood 2018; Evan Childress, pers. comm., November 20, 2018). Specifically, NRC (2002) did not find a relationship between Upper Klamath Lake elevation and AFA density (represented by chlorophyll-a concentrations) and determined that the hypothesis that maintaining higher Upper Klamath Lake elevations would effectively dilute internal phosphorus loading and reduce algal density was not supported. NRC (2002) also did not find a quantifiable relationship between Upper Klamath Lake elevation and extremes of DO concentrations or pH. Similarly, Wood et al. (1996) concluded there was little evidence that Upper Klamath Lake elevation affected any of the water quality parameters considered (chlorophyll-a concentrations, DO concentrations, pH, and total phosphorus concentrations) when examining the seasonal distribution of data and a seasonal summary statistic. Further, Wood et al. (1996) found that low DO concentrations, high pH, high phosphorus concentrations, and prolific AFA blooms were observed each year between 1990 and 1994, regardless of Upper Klamath Lake elevation. It is important to note that Wood et al. (1996) did suggest that the very low Upper Klamath Lake elevations in the summer of 1992 may have influenced DO concentrations; however, it was not possible to determine the extent to which Upper Klamath Lake elevation played a role in adverse water quality conditions in 1992. Additionally, Upper Klamath Lake elevations in 1992 were some of the lowest elevations on record (Kann 2010), coinciding with one of the driest years on record in the Klamath Basin; Upper Klamath Lake elevations at or near 1992 levels therefore would only be expected in severe drought conditions, which have occurred relatively infrequently since records began. Wood et al. (1996) also identified a possible relationship between June Upper Klamath Lake elevation and chlorophyll-a concentrations but concluded that the effect was likely due to degree days, and that it was not possible to disentangle the effects of Upper Klamath Lake elevation and air temperature.

Regardless, Morace (2007) replicated the analysis of Wood et al. (1996) with additional years of data and was again unable to identify a discernible relationship between Upper Klamath Lake elevation and water quality. Morace (2007) also did not support previous findings that suggested lower spring Upper Klamath Lake elevations may coincide with an earlier onset of the AFA bloom (Wood et al. 1996).

Conversely, Jassby and Kann (2010) did find preliminary evidence of a relationship between Upper Klamath Lake elevation and May and June chlorophyll-a concentrations (a proxy measure for bloom onset); however, the effect was largely driven by a few influential data points, as stated by the authors of the study. Additionally, Jassby and Kann (2010) did not indicate a clear subsequent effect on water quality during the bloom crash period, when water quality is most concerning for listed suckers. Nielsen et al. (2017) suggest a possible relationship between bloom onset timing and DO concentrations during the bloom crash period; however, the preponderance of data available do not suggest a direct, consistent, and discernable relationship between Upper Klamath Lake elevation and DO concentration during the bloom crash period. In conclusion, the best available science has not demonstrated a discernible and consistent relationship between Upper Klamath Lake elevation and water quality. In other words, currently, the best available science does not indicate that changes in Upper Klamath Lake elevation, within the range typically observed, result in water quality conditions that are harmful to listed suckers. This does not mean that Upper Klamath Lake elevation or water depth does not have an effect on water quality; only that the best available science has not demonstrated a direct, consistent, and discernable relationship, especially within the range of Upper Klamath Lake elevations observed from 1990 to 2016.

Finally, there is some concern that winter water quality conditions under ice cover may also adversely impact suckers (Kann 2010). Ice cover can occur on Upper Klamath Lake from November through March, although the extent and duration are dependent on winter air temperature, precipitation, and other meteorological conditions (USFWS 2008a). The available data, while limited, indicate that winter water quality parameters do not generally fall within levels considered stressful for suckers. It is also unclear how lake elevations through the POR may have contributed to poor under-ice water quality conditions, because there have been no documented winter fish die-offs in Upper Klamath Lake (Buettner 2007, pers. comm., cited in USFWS 2008a).

Tule Lake

Tule Lake is classified as highly eutrophic because of high nutrient concentrations and resultant elevated biological productivity (ODEQ 2017). Tule Lake water quality is affected primarily by the import of Upper Klamath Lake surface water through the Lost River Diversion Channel and A Canal during the irrigation season, and secondarily by local runoff during winter and spring months from lands downstream of Lost

River Diversion Dam on the Lost River. Also, contributing to the eutrophic status of Tule Lake is its shallow bathymetry and internal nutrient cycling from lake sediment. Water quality can vary seasonally and diurnally, especially in summer. Water quality in the sumps is similar to Upper Klamath Lake, with large diurnal fluctuations in DO concentrations and pH (Buettner 2000, Hicks et al. 2000, Beckstrand et al. 2001), largely due to high levels of aquatic macrophyte and green algal biomass during the growing season.

Water quality conditions in Tule Lake during the winter tend to be optimal for suckers, except during prolonged periods of ice cover, when DO concentrations decline (USFWS 2008a). A small adult sucker die-off occurred during the winter of 1992 to 1993 during an extended period of ice cover and low DO concentrations (USBR, unpublished data, cited in USFWS 2008a). A minimum elevation of 4,034.0 feet from October 1 to March 31 was set to provide adequate winter depths for cover, and to reduce the likelihood of fish die-offs owing to low DO concentrations beneath ice cover (USFWS 2008a).

Pesticide and Herbicide Applications

Up to an estimated 60 percent of Project lands (120,000 acres), including private and public, are managed for agricultural production where pesticide use is common. A majority of Project irrigation drainage is received in the area that drains into the Tule Lake sumps in Tule Lake NWR. Therefore, if pesticide residues are present in drain water from these lands, concentrations may be greatest in the Tule Lake sumps.

Surveys regarding pesticide impacts to suckers have largely focused on the Tule Lake sumps as a likely place that agrochemicals may accumulate in the Project. Additionally, the highest concentration of intensively grown crops (e.g., potato, onion, garlic) reside in the Tule Lake area.

Pesticide residues may accumulate in drain waters and discharge into Keno Reservoir from the Project. Additionally, this reach receives drainage from neighboring non-project areas such as Keno Irrigation District and private lands. However, the risk from chemical exposure for suckers in the Lost River and Keno Reservoir is likely to be less than the risk for suckers in the Tule Lake sumps due to fewer intensively grown crops in these areas such as hay, or pastureland for cattle. The risk to the suckers posed by pesticide use is dependent on many factors, including chemical toxicity, mobility, persistence, amount applied, groundwater-surface water interaction, application method, and proximity of application area relative to nearby water bodies.

Once in the sumps, pesticides volatilize, degrade, settle to the bottom with sediment, or remain in the water column where they would be highly diluted (USFWS 2008a). Based on ecological fate analyses for pesticides used on the federal lease lands (USFWS 1995), it is anticipated that pesticide use does not likely pose a threat to LRS and SNS in Tule Lake sumps when label directions are followed, and when appropriate buffers are in place (USFWS 2008a); for example, being consistent with the 1995, 1996, and 2008 BOs on pesticide use.

There is little doubt that at least trace amounts of pesticides reach the Tule Lake sumps. Since the late 1980s, low levels of pesticides were detected in the sumps (Sorenson and Schwarzbach 1991, Dileanis et al. 1996, Cameron 2008). Of the pesticides detected in waters and sediments around Tule Lake, the levels are below those known to be acutely toxic to aquatic life (Dileanis et al. 1996, Eagles-Smith and Johnson 2012), except for detections of bifenthrin and prodiamine during two sample dates in 2011. A nationwide assessment by USGS from 1992 to 2001 found pesticides at low concentrations were nearly ubiquitous in the Nation's streams and rivers, even in undeveloped watersheds (Gilliom et al. 2006).

DaSilva (2016) monitored for 34 active ingredients at Tule Lake Basin sites to include sites near the Tule Lake NWR. Although two herbicides were detected (2,4-D and dicamba) in multiple locations, neither exceeded the Aquatic Life Benchmarks values for fish (DaSilva 2016).

Between 1998 and 2000, several wildlife mortalities and fish die-offs were documented and investigated on Tule Lake NWR, but with the exception of one incident in which off-refuge use of acrolein caused a fish die-off, there was little supporting evidence that implicated pesticides as causative agents in any of the mortality events (Snyder-Conn and Hawkes 2004). However, the results of the study did reveal some evidence of trace wildlife exposure to the herbicides dicamba and 2,4-D, and a few cases of limited acetylcholinesterase inhibition in birds, suggesting potential low-level exposure to organophosphate or carbamate insecticides (Snyder-Conn and Hawkes 2004, Eagles-Smith and Johnson 2012). However, some pesticides and herbicides in use in the Klamath Basin can be toxic at low concentrations (Eagles-Smith and Johnson 2012). Although some products are listed as toxic, the actual risk of these products is a function of exposure or the amount released into the environment.

Based on limited existing data on pesticide impacts and distribution, pesticide use information, benchmark toxicity values, and habitat use of the threatened and endangered species, a 2007 BO (USFWS 2007d) evaluated impacts from direct exposure to the organisms, indirect effects through pesticide-induced reduction in prey populations, and pesticide-induced reductions in water quality. Although the assessment found that some level of pesticide exposure could occur to listed species, the evidence did not support a determination that the pesticide applications were likely to cause harm to the species considered (USFWS 2008a).

Although most of the sampling to date in Tule Lake suggests pesticides may not be present in concentrations that would adversely affect suckers, a lack of detection of toxic pesticides does not necessarily mean they would not have adverse effects on LRS or SNS (USFWS 2008a, Eagles-Smith and Johnson 2012). Highly toxic pesticides, like metam-sodium (Vapam), can harm fish at low concentrations, indicating that some chemicals may be present at low but harmful concentrations, and may escape detection during surveys. Further, many of the newer pesticides are difficult to monitor due to their rapid breakdown (USFWS 2008a). Although USBR indicates bimonthly water samples taken during the Vapam application period resulted in no detections at Tule Lake Sump 1A. USBR (2012a) conducted an ecological risk assessment specific to soil fumigants (e.g., Vapam) used on federal lease lands within Tule Lake NWR, analyzing the toxicity, environmental fate, transport, and exposure pathways. The assessment indicated there is "sufficient information that ecological risks to terrestrial, aquatic, and invertebrate species are negligible" for the majority of exposure scenarios.

In a review of existing pesticide data from the Upper Klamath Basin, Eagles-Smith and Johnson (2012) indicate that monitoring efforts to date have not been sufficient to detect low concentrations, or trace amounts, of pesticides that could have harmful impacts. In addition to possible adverse impacts from chemicals at concentrations below acute-effects low concentrations, or below detectable levels (Eagles-Smith and Johnson 2012), bifenthrin and prodiamine have recently been detected in Tule Lake, and the bifenthrin detection was at a concentration that could adversely impact aquatic life (USBR 2011; Unpublished Data; Syngenta 2008; Australian Government 2010). Although the pesticide compounds bifenthrin and prodiamine were detected, these pesticide compounds currently are not approved for use on federal lease lands. This suggests that the origins of these compounds are coming from pesticide applications on lands not under USBR or USFWS jurisdiction. Current pesticide use on federal lease lands is consistent with and covered under the Lower Klamath, Clear Lake, Tule Lake, Upper Klamath, and Bear Valley NWRs, Final Comprehensive Conservation Plan and Environmental Impact Statement. Tule Lake and Sacramento, California; USFWS, Pacific Southwest Region (USFWS 2017); and pesticide use on Project facilities and rights-of-way is consistent with and covered under previous BOs.

Fish Health

Degraded water quality conditions may compromise fish health and increase their susceptibility to disease and parasites (Holt 1997, Perkins et al. 2000b, ISRP 2005). Several parasites are common in the Upper Klamath Basin, and when combined with other environmental stressors, can have synergistic effects on the health and survival of suckers. The extent that pathogens affect suckers is not fully understood, but some parasites likely contribute to sucker mortality.

Lernaea sp., a parasitic copepod or "anchor worm," which feeds on fish tissues by puncturing the skin of its host (Briggs 1971), is a common parasite on suckers in the Upper Klamath Basin. *Lernaea* infestation was apparently absent prior to 1995. Low-level *Lernaea* infestation was first seen on YOY LRS and SNS in 1995, but prevalence (percent infested) increased substantially in the mid- to late-1990s and peaked for both species in about 2003 and 2004 (Simon et al. 2012).

Lernaea sp. are commonly found on juvenile suckers (both species) in Upper Klamath Lake and Clear Lake during summer months, although infections appear to be more common in LRS, with up to nine attachment sites on some individuals (Burdick et al. 2018. Attachment typically occurs in the dermis, along the dorsal fin or body, but attachment can also occur in the nares (Burdick et al. 2018). Attachment sites can open a pathway for other pathogens or disease, thereby causing secondary infections. Severe inflammation and necrosis (dead tissue) in the skin and muscle occur far and deep beyond the attachment site (Janik et al. 2018). The *Lernaea* that appear to affect suckers in Upper Klamath Lake were identified by Janik et al. to be *Lernaea cyprinacae*. Prevelence of *Lernea* sp. infections appears to vary among years (Burdick et al. 2018)

The trematode metacercariae, *Bolbophorus* sp. (Janik et al. 2018), commonly called "black spot," is a flat worm that infects the skeletal muscle tissue of LRS and SNS in Upper Klamath Lake. Of the two species, prevalence of infection appears to be higher in SNS (Burdick et al. 2018, Janik et al. 2018). Number of metacercariae infections in suckers is typically higher for SNS than LRS; as many as 11 raised cysts have been observed on a single YOY sucker (Burdick et al. 2018, Janik et al. 2018). Host response includes melanization of the skeletal muscle tissue that surrounds the encysted digenean metacercariae; however, the surrounding tissue is typically unaffected (Burdick et al. 2015, 2018).

A number of pathogens have been identified from moribund (dying) suckers, including Gram-negative bacterial infections of apparent *Flavobacterium columnare*, which can damage gills, produce body lesions, which leads to respiratory problems, an imbalance of internal salt concentrations, and provides an entry route for lethal systemic pathogens (ISRP 2005; Foott 1997, 2004; Holt 1997). Apparent columnaris infections were found in some moribund juvenile suckers in mesocosms (Hereford et al. 2016, 2018). Although columnaris infections are suspected to impact suckers in most cases, Morris et al. (2006) found that LRS exposed to *Flavobacterium columnare* and exposed to high concentrations of un-ionized ammonia in laboratory trials, had higher survival than those that were exposed to lower concentrations of un-ionized ammonia, or control fish. Morris et al. (2006) suggested that the columnaris bacterial infection was killed or compromised by the highest un-ionized ammonia concentration, or that suckers exposed to *Flavobacterium columnare* had elevated immune response that allowed them to survive elevated un-ionized ammonia concentrations. These findings suggest that interactions among parasites and water quality conditions may be complex. A total of 304 bacterial genera was detected in skin mucous of YOY juvenile suckers from Upper Klamath Lake, several of which are potentially pathogenic (Burdick et al. 2009b). Further research is necessary to determine which bacteria pose a serious health risk to suckers (Burdick and Hewitt 2012).

One parasite that severely impacts YOY SNS is the nematode larva *Contracaecum* sp. (Janik et al. 2018). This parasite, which is approximately 17 mm in length, has been found in some (19 percent) SNS hearts, and in one (of 75) unidentified sucker heart (Janik et al. 2018). When present, the nematode enlarged and thinned the atrium, and prevented normal heart function (Janik et al. 2018). Although not terribly common, *Contracaecum* sp. is expected to cause cardiovascular failure and inhibit swimming performance (Janik et al. 2018). Affected suckers are not suspected to survive (Janik et al. 2018).

Although its prevalence in wild suckers is not known (Burdick and Martin 2017), *Ichthyobodo* sp. (formerly *Costia* sp.) is a parasite that attaches to the gills or skin (Callahan et al. 2002). This obligate ectoparasite can cause or contribute to mortality of wild juvenile suckers by impairing normal body functions (Hereford et al. 2016, 2018). For example, *Ichthyobodo* sp. infestations in fish can cause anorexia, surface cell-death, reduced oxygen uptake, reduced ion regulation, and impaired circulation (Lom and Dyková, 1992). Interestingly, fish rarely show distress or have changes in behavior prior to mortality (Callahan et al. 2002). This parasite is commonly associated with mortality of juvenile suckers in mesocosms in Upper Klamath Lake (Hereford et al. 2016, 2018). Trichodinid protozoan parasites have been observed on juvenile suckers from both Upper Klamath Lake and Clear Lake (Burdick et al. 2015, Janik et al. 2018).

Parasites were not identified as a threat at the time of listing, but recent information indicates they could be a threat to the suckers (Buchanan et al. 2011). Parasites can lead to direct mortality, provide a route for pathogens to enter fish through wounds, and can make fish more susceptible to predation (Robinson et al. 1998). Although many parasites are common, especially in Upper Klamath Lake, the role Project operations have on their occurrence is unknown.

Typically, there is a direct relationship between prevalence of stress and prevalence of parasites and disease. Many factors may contribute to stress (and therefore prevalence of disease and parasites), including but not limited to fish density, water quality, habitat availability, preferred-food resource availability, predation, seasonality, or some combination of these factors. For juvenile suckers in Upper Klamath Lake, parasites or other signs of stress are relatively common, although not prevalent throughout July, August, and September, and no specific disease or parasite has been found to be widespread (Burdick and Martin 2017). The lack of information regarding disease, parasites, and stress affecting juvenile suckers is likely due to the inherent hardiness of the species, and the difficulty for researchers to capture compromised and affected suckers using passive gear. Several studies (Saiki et al 1999, Meyer and Hansen 2002, Lease et al. 2003, Hereford et al. 2018) have found suckers show little to no sign of distress until immediately before death, despite high parasite loads, compromised water quality conditions, or other factors, which may explain why understanding causes of mortality for juvenile suckers is so difficult. Further, suckers with compromised health may be heavily predated on.

Entrainment Losses

Entrainment of listed suckers can occur from the downstream movement of fish into diversions or spillways by drift, dispersion, and volitional migration (PacifiCorp 2012). Effects to fish associated with entrainment may include harassment, injury, and mortality as fish pass through or over spillways, into canals, or into pumps. Spillway mortality of entrained fish can occur from strikes or impacts with solid objects (e.g., baffles, rocks, or walls in the plunge zone), rapid pressure changes, abrasion with the rough side of the spillway, and the shearing effects of turbulent water (Clay 1995). Entrainment at and lack of passage through Klamath River dams and other irrigation structures were added to the list of threats to the endangered suckers after the original listing (USFWS 1992, NRC 2004). Entrainment into irrigation and power-diversion channels is now recognized as being responsible for losses of “millions of larvae, tens of thousands of juveniles, and hundreds to thousands of adult suckers each year” (NRC 2004). Changes in the physical structure at the southern end of Upper Klamath Lake, such as channel cuts in natural reefs, and changes in lake hydrology likely contribute to entrainment of suckers from Upper Klamath Lake (USFWS 2008a).

Entrainment also occurs at other diversion dams in the Project, including at Clear Lake, Gerber, Miller Creek, Malone, Lost River Diversion and Anderson-Rose dams (USBR 2002). Clear Lake Dam was screened in 2003 to prevent entrainment of juvenile and adult suckers, but not larvae. The effectiveness of the screen in excluding juvenile and adult suckers was verified in 2003, when fish salvage operations conducted downstream of Clear Lake Dam at the end of the irrigation season captured only three suckers (Bennetts et al. 2004) compared to several hundred suckers captured before the screen was installed (Piaskowski 2003). Numerous additional points of diversions or delivery exist in the Project area, including: A Canal (Upper Klamath Lake); J Canal, Q Canal, Pumping Plant D and R Canal (Tule Lake sump); and the Lost River Diversion Channel and its associated lateral canals (USBR 1992, 2001). See USBR (2001) for a more comprehensive list of diversion locations and estimated diversion quantities in the Project. Much of the effort to estimate and understand entrainment of suckers has focused on fish that move downstream of Upper Klamath Lake. Although entrainment has not been measured at all diversions, entrainment of suckers likely occurs at other locations in the Project, particularly at unscreened diversions or diversions nearest to known populations of suckers.

USBR completed construction of a fish screen at the entrance to the A Canal in March 2003 to reduce fish entrainment known to occur at this diversion (USBR 2007). Upper Klamath Lake has been suggested as a better suited environment for suckers than Keno Reservoir due to the food-rich environment in Upper

Klamath Lake, and the frequency and duration of poor water quality events in the Klamath River (Reithel 2006, Markle et al. 2009), and access to spawning (USFWS 2008a). LRS and SNS were particularly vulnerable to entrainment at A Canal before the screen was installed. Entrainment studies at the southern end of Upper Klamath Lake from 1997 to 1999 (Gutermuth et al. 2000a, 2000b) have been used to estimate and understand entrainment from Upper Klamath Lake at the Link River, A Canal, and both the East Side and West Side power developments at the Link River (USFWS 2007c, 2008a; Tyler 2012a, 2012b).

Entrainment of young fish is a potentially important contributor to recruitment failure, given that the entrained larvae that are passed through the A Canal fish screen and YOY juveniles that are entrained at the Lost River Diversion likely originate from known spawning aggregations in the tributaries or shoreline areas, and individuals exiting Upper Klamath Lake to the south may be permanently lost from the population (NRC 2004).

Entrainment estimates from Upper Klamath Lake are typically based on extrapolation of observations from Gutermuth et al. (2000a, 2000b) with A Canal fish screen assumptions and annual updates for inter-annual sucker production and water conveyance (USFWS 2008a; Tyler 2012a, 2012b; NMFS and USFWS 2013). Annual estimates for suckers exiting Upper Klamath Lake via the Link River are variable, and range between 100,000 and 6,000,000 for larvae, between about 10,000 and 140,000 for juveniles, and usually fewer than 230 adult suckers (USFWS 2008a; Korson et al. 2011; Korson and Kyger 2012; Tyler 2012a, 2012b). Not all sucker entrainment at the southern end of Upper Klamath Lake is lethal (PacifiCorp 2013b), because some adults return to Upper Klamath Lake using the Lost River Diversion fish ladder (Kyger and Wilkens 2011, 2012).

Of the number of YOY juvenile suckers entrained each year from Upper Klamath Lake, some individuals may survive in Keno Reservoir (Reithel 2006, Terwilliger et al. 2004, Phillips et al. 2011, Tyler and Kyger 2012). Although this reach does not provide ideal conditions, some of these suckers may survive to older juvenile and adult life history stages, and attempt returns to Upper Klamath Lake via the Lost River Diversion fish ladder. However, the number of individuals that do survive in Keno Reservoir is likely small. Of an estimated 6 million larvae, 100,000 juveniles, and 100 older juvenile/adult suckers that disperse annually into Keno Reservoir from Upper Klamath Lake, an estimated 80 percent of these fish perish (i.e., about 5 million larvae, 80,000 juveniles, and 80 older juvenile/adult suckers annually) due to the impaired water quality conditions downstream of the Link River (USFWS 2007c).

Population impacts due to the loss of larval, juvenile, and adult suckers are uncertain (USFWS 2008a, PacifiCorp 2013b). Numbers of larval suckers that are estimated to be lost through entrainment represent a small proportion of the potential fecundity of the breeding population. Each female shortnose and LRS can produce up to 72,000 and 236,000 eggs per year, respectively (Perkins et al. 2000a). There are thousands of reproductively active female suckers in Upper Klamath Lake each year (Janney et al. 2008, 2009; Hewitt et al. 2011), suggesting a high reproductive potential in any given year.

Although there are no reliable estimates for larval and YOY juvenile suckers (USFWS 2007a, 2007b), there are extrapolations of data from surveys that inform us on the magnitude of early life history stage entrainment from Upper Klamath Lake. Data from The Klamath Tribes (1996) estimated the total annual production for larval suckers at about 73 million. The entrainment of an estimated 6 million larval suckers represents approximately 8.2 percent of the total annual sucker production at that life history stage (USFWS 2007c). More recently, Simon et al. (2012) estimated the number of larval suckers in Upper Klamath Lake between 19 and 29 million based on an extrapolation of early June fish surveys in 2011. Estimated entrainment at the southern end of Upper Klamath Lake was 2.4 million larval suckers in 2011, based on amount of water exiting Upper Klamath Lake and the magnitude of larval sucker production (Tyler 2012b). These numbers suggest that larval entrainment could represent 8 to 13 percent of estimated numbers of larval suckers available in Upper Klamath Lake during a given year. Although using a combination of work by Simon et al. (2012) and Tyler (2012b) represents a higher percent of total annual production than using earlier estimates of larval production, data suggest that sucker larvae in 2011 were mostly retained in Upper Klamath Lake by the central gyre rather than by shoreline retention (Simon et al. 2012). How the number of larval suckers produced and entrained affects recruitment to the adult populations in Upper Klamath Lake is still uncertain (PacifiCorp 2013b).

Entrainment of YOY juvenile suckers is also variable among years and can represent a substantial percent of the annual sucker production. Low-cast net catches of YOY suckers in Lake Ewauna and higher catches in northern and middle Upper Klamath Lake in 2011 suggest that retention of juvenile suckers was relatively high in 2011, with about 850,000 YOY juvenile suckers of both species present in early August of that year (Simon et al. 2012). Estimated entrainment at the southern end of Upper Klamath Lake was about 7,000 YOY juvenile suckers (Tyler 2012b); however, monitoring at the fish bypass at A Canal estimated that about 140,000 YOY juvenile suckers were bypassed back to Upper Klamath Lake (Korson and Kyger 2012). An entrainment estimate of 7,000 juvenile suckers represents less than 1 percent of 2011 YOY juvenile sucker abundance (i.e., 850,000), but using 140,000 bypassed YOY juveniles as an entrainment number represents greater than 16 percent of the 2011 YOY juvenile sucker abundance.

Long-lived LRS and SNS typically exhibit relatively low mortality. However, adult suckers in Upper Klamath Lake are nearing their maximum life expectancy, and mortality appears to be increasing rapidly (Hewitt et al. 2017; E. Janney, USGS, pers. comm., May 11, 2018); likewise, mortality for juvenile suckers continues to be widespread each year as substantial recruitment events into the adult population have not been observed (Hewitt et al. 2017, Burdick et al. 2018). Given the current status of suckers in Upper Klamath Lake, it is likely that entrainment losses through A Canal bypass and Lost River Diversion adversely impact sucker populations through a reduction in the number of suckers available to recruit to the adult populations.

The number of suckers entrained at facilities decreases progressively downstream of the Lost River Diversion (PacifiCorp 2013b). This corresponds to the relative distribution of the suckers in reservoirs downstream of the Lost River Diversion (PacifiCorp 2013b). Each of these reservoirs, including Keno Reservoir, is likely seeded by larval and juvenile suckers emigrating from Upper Klamath Lake (Desjardins and Markle 2000). Based on entrainment studies at Lost River Diversion and fish distribution studies in reservoirs, substantial numbers of larval and juvenile suckers disperse downstream of Upper Klamath Lake to reside in the downstream reservoirs (USFWS 2007c). There is no evidence that self-sustaining populations exist in any of the reservoirs, but it is possible that some larval and juvenile suckers in Keno Reservoir are from spawning in the Link River (Smith and Tinniswood 2007). However, it is more likely that most of the suckers in Keno Reservoir arrived from Upper Klamath Lake (Markle et al. 2009). SNS spawning and larval production occurs in Copco No. 1 Reservoir; however, there is little recruitment into the adult population (USFWS 2007c).

Annual entrainment losses from Keno Reservoir via the spillway at Keno Dam are nearly 570,000 larvae, nearly 15,000 juveniles, and 15 adult suckers (PacifiCorp 2013b). Of these entrainment estimates, approximately 12,000 larvae and nearly 300 juveniles are thought to expire as a result of trauma while passing the spillway at Keno Dam (PacifiCorp 2013b).

Entrainment losses from Keno Reservoir are also likely through the Lost River Diversion Channel and other unscreened diversions (North Canal, Ady Canal, and other diversions). Sampling in the Lost River Diversion Channel between Reeder Road and Tingley Lane captured eight juvenile suckers in 64 trap nets fished on 16 sample dates (Foster and Bennetts 2006). Sampling was conducted weekly from late May through late September and represents 1,200+ hours (Foster and Bennetts 2006). During the same effort, a screw trap was fished on seven dates between mid-July and early September at Station 48 on the Lost River Diversion Channel, capturing two suckers (one juvenile and one dead adult; Foster and Bennetts 2006). Fish entrainment monitoring at Miller Hill Pumping Station, which feeds parts of C Canal from the Lost River Diversion Channel in July and August 2008, did not capture suckers but did capture other fish species (Korson 2010). Fish sampling near Ady and North canals indicated the juvenile suckers are present near both locations during the summer (Phillips et al. 2011). These efforts indicate the presence of suckers in relatively low abundance in the Lost River Diversion Channel and near other diversions that are susceptible to entrainment.

Unquantified sucker entrainment also occurs in the Lost River, Tule Lake Sumps, and at other unscreened diversions throughout the Project (USBR 2001).

Bird Predation

Bird predation on endangered suckers has been studied at Clear Lake and Upper Klamath Lake. American White pelicans and double-crested cormorants (*Phalacrocorax auritus*) are the most abundant avian predators, and both species have nesting colonies at Clear Lake and Upper Klamath Lake (Evans et al. 2016). Pelicans are more common at nesting colonies at Clear Lake, while cormorants are more common at nesting colonies at Upper Klamath Lake. With their larger beak, pelicans are able to consume larger fish (up to 730 mm; Evans et al. 2016) than cormorants (up to 450 mm; Hatch and Weseloh 1999). Individual pelicans are able to forage on suckers up to 4 feet (1.25 meters) deep (Anderson 1991).

However, as cooperative foragers, pelicans often drive fish into shallow water (Anderson 1991). In contrast, cormorants can forage for fish in water up to 33 feet (10 meters) deep but are more limited in the size of fish they can consume. Other avian predators of suckers in the Upper Klamath Basin, including gulls (*Larus* sp.), herons (*Ardea* sp.), and Caspian terns (*Hydroprogne caspia*), nest among pelicans and cormorants, and likely contribute to the sucker mortality (Evans et al. 2016).

Bird predation varies by sucker age-class and species, bird colony location, nesting success, and year (Evans et al. 2016). Relative to their availability, avian predators often select smaller suckers, including juveniles and SNS, although exceptions to this were observed in some years. Deposition rates for avian predators have not been specifically studied for pelicans or cormorants in the Upper Klamath Basin; therefore, specific estimates relative to bird species are not available. Additionally, from the data available, Evans et al. (2016) were able to estimate minimum (not actual) bird predation on both species of suckers at each lake by scanning bird nesting colonies for sucker PIT tags. Again, actual estimates require deposition rates for each avian predator in each lake. Avian predators in Clear Lake had the highest predation rates on suckers in Clear Lake; minimum avian predation rates for Clear Lake nesting birds are estimated to be 4.6 percent for LRS and 4.2 percent for SNS (Evans et al. 2016). Avian predation at Upper Klamath Lake accounts for a minimum of 0.6 percent LRS and 1.8 percent SNS mortality. Recovered PIT tags from Clear Lake included tags that were implanted in suckers that were released at other locations, principally Upper Klamath Lake, demonstrating that piscivorous water birds nesting on islands in Clear Lake traveled to other lakes and streams to consume PIT-tagged suckers (Roby et al. 2011, Evans et al. 2016). Interestingly, pelicans nesting at Clear Lake were more likely to prey on adult suckers spawning at the springs on the eastern side of Upper Klamath Lake, whereas Upper Klamath Lake pelicans were more likely to prey on suckers spawning in tributaries (Evans et al. 2016).

Additional information regarding factors that may influence predation on suckers by fish-eating birds is not currently understood; however, fish age, fish behavior (including that caused by disease or parasites), poor water quality, loss of deep water habitat (due to lake elevation changes or changes in habitat), fish proximity to bird nesting areas, bird colony size and success rate, and the availability of other prey items were suggested as possibly influencing PIT-tag recovery inferences (Roby et al. 2011, Evans et al. 2016). Bird predation may also vary seasonally, although this has not been directly studied.

G.2.1.8 PacifiCorp Habitat Conservation Plan (2013)

PacifiCorp finalized a Habitat Conservation Plan (HCP) for Lost River suckers (LRS) and shortnose suckers (SNS) in 2013 (PacifiCorp 2013b) in accordance with Section 10(a)(1)(B) of the ESA. In response to this plan, the USFWS conducted an intra-service consultation (08EKLA00-2013-F-0043) on the effects to suckers of the authorization of the plan. Actions conducted by PacifiCorp under the HCP have influenced the status of LRS and SNS in the Action Area. A detailed description of these Actions is provided in Chapter 4 of the BA.

G.2.2 Bull Trout (*Salvelinus confluentus*)

G.2.2.1 Species Status

The following information is largely taken from USFWS 2015a. Bull trout populations in the Columbia River and Klamath River basins were defined as distinct population segments (DPS), and federally listed as threatened on June 10, 1998 (63 FR 31647). The Jarbidge River population segment of bull trout were proposed to be listed on June 10, 1998 (63 FR 31693), and bull trout throughout the coterminous United States were listed as threatened on November 1, 1999 (64 FR 58910). The coterminous listing added bull trout of the Coastal-Puget Sound (Olympic Peninsula and Puget Sound regions), Jarbridge River, and Saint Mary-Belly River populations (east of the continental divide in Montana) to the previous listing action. The *Klamath Recovery Unit Implementation Plan* (USFWS 2015a) provides an update on the species status, and the document is complementary to the *Recovery Plan for the Coterminous U.S. Population of Bull Trout* (USFWS 2015b). Bull trout once occupied habitat throughout the Klamath Basin, but forestry practices, agricultural development, and fisheries management practices have greatly reduced bull trout distribution in the watershed (USFWS 2015a). Other factors such as competition and hybridization with non-native brook trout (*Salvelinus fontinalis*) have further affected the three bull trout core areas (Sycan River, Upper Klamath Lake, and Upper Sprague River) in the Klamath Recovery Unit (USFWS 2015a). In the Klamath Recovery Unit, because 9 of 17 known local populations have already been extirpated and the remainder are significantly imperiled and require active management of threats, effective threat management is necessary in 100 percent of core areas, and the geographic range of bull trout in this recovery unit will need to be expanded through reestablishment of extirpated local populations (USFWS 2015b).

G.2.2.2 Critical Habitat

The following information is largely taken from USFWS 2015a. Final critical habitat for the bull trout DPS in the Klamath and Columbia rivers was designated by USFWS on October 6, 2004 (69 FR 59996), and for the species in the coterminous United States on September 26, 2005 (70 FR 56212). A final revision of critical habitat for this species was designated by USFWS on October 18, 2010 (75 FR 63898). The Klamath River Basin Critical Habitat Unit is in south-central Oregon and includes three critical habitat subunits (CHSUs): (1) Upper Klamath Lake CHSU; (2) Sycan River CHSU; and (3) Upper Sprague River CHSU. The Klamath River Basin CSU covers 276.6 miles of river and 9,329.4 acres of reservoirs or lakes designated as critical habitat.

G.2.2.3 Life History

The following information is largely taken from USFWS 2015a. Bull trout exhibit two basic life history strategies: resident, and migratory. Migratory bull trout live in larger river (fluvial) and lake systems (adfluvial) where juvenile fish usually rear from 1 to 4 years before migrating to either a larger river or lake where they spend their adult life, returning to the tributary stream to spawn (Fraley and Shepard 1989). In general, migratory fish are larger than resident fish. Stream-resident bull trout complete their entire life cycle in the tributary streams where they spawn and rear. Research indicates that resident and migratory forms may be found together, and interbred at times, which helped maintain viable populations throughout the range (Rieman and McIntyre 1993).

Bull trout reach sexual maturity in 5 to 7 years, and spawn from the end of August through November (McPhail and Baxter 1996). Spawning may occur annually for some populations, and every other year for the rest. Migration for spawning is initiated by warming water temperatures in downstream reaches. The distances traveled by migratory bull trout to spawn are on average farther than other non-anadromous salmonids (Fraley and Shepard 1989). Bull trout require particularly clean gravel substrates to build their redds. Increased sediment suffocates eggs by reducing dissolved oxygen (Rieman and McIntyre 1996). Bull trout eggs incubate over the winter, and hatch in the late winter or early spring. Emergence usually requires an incubation period of 120 to 200 days.

Juveniles migrate to areas upstream from spawning beds to grow and take advantage of cool headwater temperatures. Bull trout less than 1 year old are generally found in areas along stream margins and in side channels. Most migratory juvenile bull trout remain in headwater tributaries for 1 to 3 years before emigrating downstream to larger stream reaches. Emigration usually takes place from June to August (Rieman and McIntyre 1996).

Migration is important for the persistence of many local subpopulations of bull trout. Migratory corridors that allow bull trout to move from spawning and rearing habitat to foraging and overwintering habitat result in larger, more reproductively successful bull trout (McPhail and Baxter 1996), and also result in increased dispersion, which improves gene flow. Local populations that are extirpated during catastrophic events can be re-established as a result of bull trout movement through migration corridors (Rieman and McIntyre 1996).

Bull trout have more specific habitat requirements than most other salmonids (Rieman and McIntyre 1993). Habitat components that particularly influence their distribution and abundance include water temperature, cover, channel form and stability, spawning and rearing substrate conditions, and migratory corridors (Fraley and Shepard 1989). Bull trout require especially clean and cold water with temperatures below 59°F. They live primarily in cold headwater lakes, and streams and rivers that drain high mountainous areas, especially where snowfields and glaciers are present. Like all salmonids, bull trout require diverse, yet well-connected, habitats with structural components that provide good hiding cover (McPhail and Baxter 1996).

G.2.2.4 Geographic distribution

The following information is largely taken from USFWS 2015a. Bull trout are members of the char subgroup of the family Salmonidae and are native to waters of western North America. Historically, bull trout occurred throughout the Columbia River Basin; east to Montana, south to the Jarbidge River in northern Nevada, the Klamath Basin in Oregon, and the McCloud River in California; and north to Alberta, British Columbia, and possibly southeastern Alaska. The range of the bull trout has decreased compared with the known historical range. Bull trout are now extirpated in northern California (Moyle et al. 2008), and from other watersheds in Oregon and Washington (USFWS 2015a). In areas where bull trout populations occur, many are reduced in size, fragmented, or have been eliminated from the mainstems of large rivers (USFWS 2008d; 2015a).

In the Klamath Basin, the Klamath Recovery Unit is in southern Oregon, and includes three bull trout core areas (Upper Klamath Lake, Sycan River, and Upper Sprague River), all in the upper Klamath River basin (USFWS 2015b). The Upper Klamath Lake core area comprises the northern portion of the lake and its immediate major and minor tributaries. This core area includes two existing local bull trout populations: Threemile Creek, and Sun Creek. Sun Creek originates in Crater Lake National Park, and currently supports the largest local population in the Upper Klamath Lake core area (USFWS 2015b). A mark-resight population estimate was completed by the Oregon Department of Fish and Wildlife (ODFW) in Threemile Creek during 2012. The total population of age 1+ and older bull trout was 577 +/-102 in the reach from approximately 450 feet above the 3519 bridge to the forks. No additional population sampling has been completed since 2012, but the Threemile Creek population is likely increasing due to successful brook trout eradication and stream enhancement efforts (ODFW 2016). In Sun Creek, the bull trout population has increased from 150 individuals in 1989 to over 2,000 individuals by 2017, with the bulk of that population residing in Crater Lake National Park. However, the distribution of bull trout in Sun Creek has increased from approximately 1.2 miles to almost 12 miles during the same timeframe (Buktenica et al. 2018). The Sycan River core area comprises Sycan Marsh, Sycan River, and associated tributaries. This core area is composed of the waters that drain into the Sycan Marsh, including Long, Callahan, and Coyote creeks on the western side, and Sycan River, Chocktoot Creek, Shake Creek, and their tributaries on the eastern side of the marsh. The only local bull trout population in the Sycan River core area occurs in Long Creek. Long Creek is driven by a snowmelt hydrograph, but base flow is largely spring- fed. Bull trout have been found distributed throughout most of the length of Long Creek, and bull trout occupy approximately 2.2 miles of spawning and rearing habitat, and seasonally use 16.1 miles of foraging, migratory, and overwintering habitat (USFWS 2015b).

The Upper Sprague River core area includes five bull trout populations, including Boulder Creek, Dixon Creek, Deming Creek, Leonard Creek, and Brownsworth Creek (USFWS 2015b). Deming Creek is believed to support the largest local population of bull trout in the Upper Sprague River core area. These local populations are at an elevated risk of extinction because the populations are not interconnected.

Management actions have targeted increasing watershed connectivity to improve the potential for population expansion in the Upper Sprague River core area. Fish passage barrier replacement has increased the amount of occupied habitat, and the five populations in the core area are believed to either be stable or expanding their distribution (USFWS 2015b). Non-native brook trout and brown trout present hybridization, predation, and competition concerns for bull trout (USFWS 2015b).

Bull trout in the Klamath Recovery Unit have been isolated from other bull trout populations for the past 10,000 years and are recognized as evolutionarily and genetically distinct (USFWS 2015b). Therefore, there is no opportunity for bull trout in another recovery unit to naturally re-colonize the Klamath Recovery Unit if it were to become extirpated (USFWS 2015b). The Klamath Recovery Unit is also at the southern extent of the species range and is likely susceptible to climate change effects characterized by warming temperatures, decreasing snowpack, and more variable hydrologic conditions.

G.2.2.5 Population trends

In Oregon, bull trout occurrences represent a fraction of the species' historical distribution. A total of 85 bull trout populations in 12 basins are currently identified in Oregon (ODFW 2005). These basins include the Klamath River, Willamette River, Hood River, Deschutes River, John Day River, Umatilla River, Walla Walla River, Grande Ronde River, Imnaha River, Pine Creek, Powder River, and Malheur River. In these basins, bull trout populations are highly fragmented; and in some cases, only exist in a small portion of each basin.

In the Klamath Basin, bull trout abundance and distribution have likely been greatly reduced from historical levels due to habitat degradation and fragmentation, past and present land use practices, agricultural water diversions, and past fisheries management practices (USFWS 2015b). Further discussion of population metrics for bull trout in the Klamath Recovery Unit is provided below.

G.2.2.6 Threats

The factors that have contributed to the decline of bull trout include restriction of migration routes, poor forest management practices, grazing, agricultural practices, road construction, mining, introduction of non-native species (including brook trout), and residential development contributing to habitat modification (USFWS 2015b).

Competition and hybridization with brook trout is considered one of the primary threats to bull trout recovery in all three core areas of the Klamath Recovery Unit (USFWS 2002b). Overall, interspecific interactions, including predation, with non-native species may also exacerbate stresses on bull trout from habitat degradation, fragmentation, isolation, and species interactions (Rieman and McIntyre 1993). Brook trout readily spawn with bull trout, creating a hybrid that is often sterile (Markle 1992).

Warmer temperature regimes associated with global climate change represent another risk factor for bull trout. Increased stream temperature is a recognized effect of a warming climate (ISAB 2007). Species at the southern margin of their range that are associated with colder water temperatures, such as the bull trout, are likely to become restricted to smaller, more disjunct habitat patches, or become extirpated as the climate warms (Rieman et al. 2007).

G.2.2.7 Status in the Action Area

The following information is largely taken from USFWS 2015b. The current spawning distribution of bull trout is highly fragmented and concentrated in a few isolated headwater streams of Upper Klamath Lake, upper Sprague River, and upper Sycan River upstream of Sycan Marsh (USFWS 2015b). The Klamath River Recovery Unit bull trout population is currently composed of eight populations that are in Sun Creek, Threemile Creek, Long Creek, Dixon Creek, Boulder Creek, Deming Creek, Leonard Creek, and Brownsworth Creek. In the Klamath Recovery Unit, at least nine historical local populations of bull trout have become extirpated (USFWS 2015b).

Few data exist to accurately assess abundance of bull trout in the Klamath Basin. Population estimates were initially conducted between 1989 and 1991 (Buchanan et al. 1997; Ziller 1992) and have occurred more recently (USFWS 2015b). Barriers, poor water quality, and lack of a migratory life history in most populations (Long Creek in Sycan River has resident and migratory life histories [N. Banish, USFWS, personal communication]) prevent bull trout in each watershed (i.e., Sprague, Sycan, and Upper Klamath Lake) from mixing.

In the Upper Klamath Lake core area, bull trout formerly occupied Annie Creek, Sevenmile Creek, Cherry Creek, and Fort Creek, but are now extirpated from these locations. Currently, this core area is composed of two local bull trout populations in Sun Creek and Threemile Creek. These local populations likely face an increased risk of extirpation because they are isolated and not interconnected with each other (USFWS 2015b). However, focused efforts on eradicating non-native brook trout from Sun Creek and Threemile Creek have helped stabilize these populations and have led to increases in bull trout abundance and distribution (USFWS 2015b).

The Sycan River core area is composed of one local population, Long Creek. Long Creek likely faces greater risk of extirpation because it is the only remaining local population due to extirpation of all other historical local populations. This core area is considered essential for recovery because bull trout in this core area exhibit both resident and fluvial life histories, which are important for representing diverse life history expression in the Klamath Recovery Unit (USFWS 2015b). No recent statistically rigorous population estimate has been completed for Long Creek; however, the 2002 Draft Bull Trout Recovery Plan reported a population estimate of 842 individuals (USFWS 2002b).

The Upper Sprague River core area is composed of five bull trout local populations, placing the core area at an intermediate risk of extinction. The five local populations include Boulder Creek, Dixon Creek, Deming Creek, Leonard Creek, and Brownsworth Creek. The Upper Sprague River core area population of bull trout has experienced a decline from historical levels, although less is known about historical occupancy in this core area. Bull trout are reported to have historically occupied the South Fork Sprague River, but are now extirpated from this location (Buchanan et al. 1997). Recent efforts have been made to increase the connectivity of existing bull trout populations by addressing fish passage barriers. Therefore, over the past few generations, these populations have likely been stable, and increased in distribution; although a recent documented brook trout invasion in Boulder Creek threatens the stability of this population (N. Banish, USFWS, personal communication).

Population abundance has been estimated recently for Boulder Creek (372 ± 62 percent; Hartill and Jacobs 2007), Dixon Creek (20 ± 60 percent; Hartill and Jacobs 2007), Deming Creek (1,316 ± 342; Moore 2006), and Leonard Creek (363 ± 37 percent; Hartill and Jacobs 2007). No statistically rigorous population estimate has been completed for the Brownsworth Creek local population; however, the 2002 Draft Bull Trout Recovery Plan reported a population estimate of 964 individuals (USFWS 2002b).

Efforts to reduce hybridization and competition with non-native fish, replacement or removal of passage barriers, changes in fishing regulations, and habitat restoration projects have improved several local populations (e.g., Threemile, Sun, and Long creeks; Hamilton et al. 2010). However, the overall status of Klamath River bull trout continues to be depressed. Conservation recommendations in USFWS 2015b also included the reintroduction of anadromous species, such as Chinook salmon and steelhead, that were historically present in the upper Klamath River basin. Reintroduction of anadromous species is expected to support bull trout recovery by increasing prey base and providing marine-derived nutrients (USFWS 2015b).

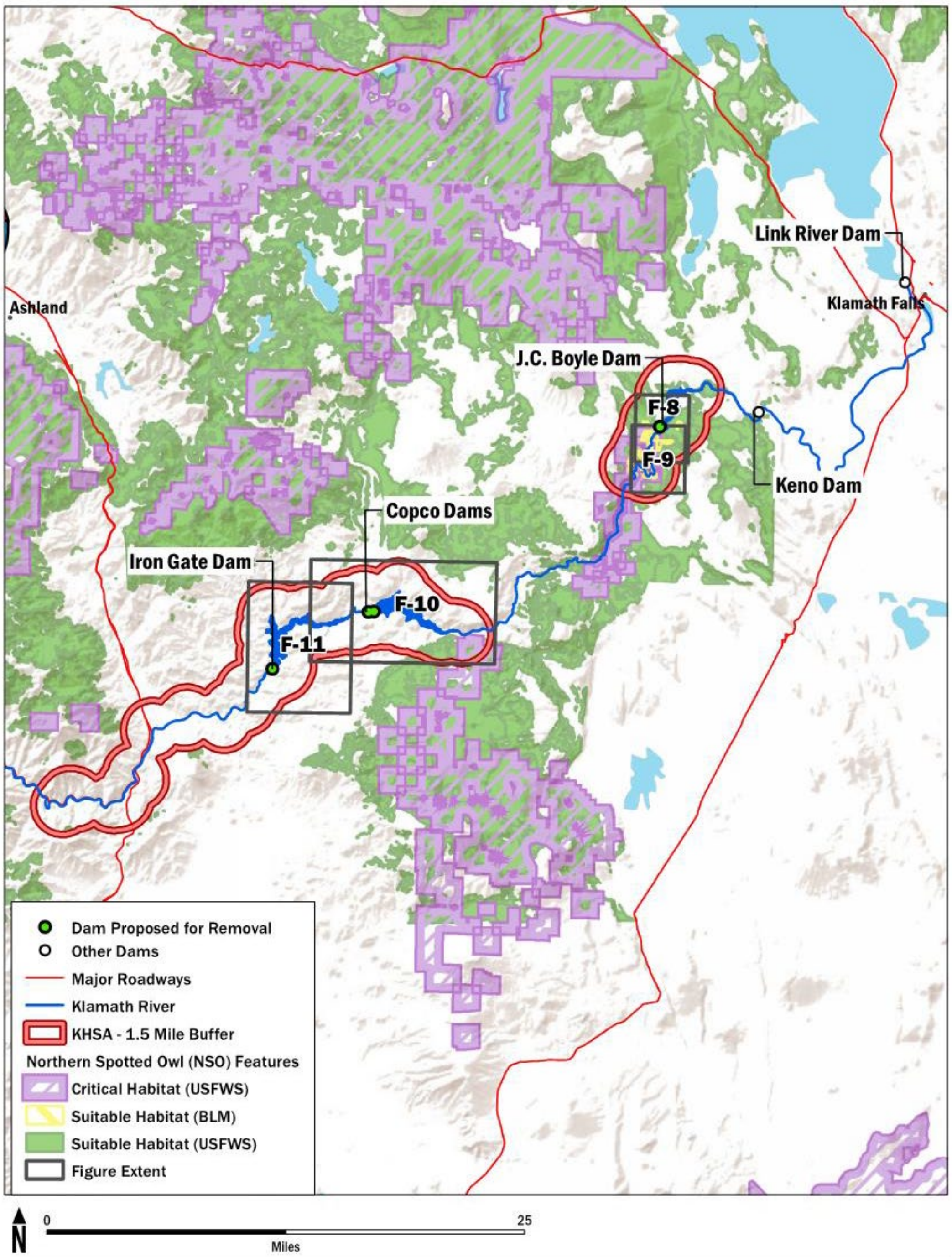
G.2.3 Northern Spotted Owl (*Strix occidentalis caurina*)

G.2.3.1 Species status

The northern spotted owl was federally listed as threatened in 1990 due to widespread loss and adverse impacts on suitable habitat across the owl’s entire range, and the inadequacy of existing regulatory mechanisms to conserve the owl (USFWS 1990).

G.2.3.2 Critical habitat

Northern spotted owl critical habitat is present north of Iron Gate Reservoir, south of the Klamath River east of Copco No. 1 Reservoir, and adjacent to the J.C. Boyle powerhouse, as shown in Figure G-7.



Klamath River Renewal Corporation, Klamath River Renewal Project

Figure G-7: Northern Spotted Owl Habitat Overview

In June 1990, the USFWS issued a final rule listing all northern spotted owl populations as threatened under the authority of the ESA. Critical habitat was originally designated in 1992 (USFWS 1992). Critical habitat was revised in 2012 (USFWS 2012a) based on the Revised Recovery Plan for the Northern Spotted Owl (USFWS 2011a). Critical habitat is designated under the ESA as an area in which biological or physical features essential to the conservation of the species are present in their occupied geographical range and may require special management consideration or protection (USFWS 1992).

In addition, USFWS critical habitat designation (USFWS 2012a) includes the PBFs listed below that are essential to a species' conservation.

1. **Forest types that support the species across its geographic range,** which primarily include early- mid- or late- seral stages of Sitka spruce, western hemlock, mixed conifer and mixed evergreen, grand fir, Pacific silver fir, Douglas-fir, white fir, Shasta red fir, redwood/Douglas-fir, and the moist end of the ponderosa pine coniferous forest zones at elevations up to approximately 3,000 feet (914 meters) near the northern edge of the range, and up to approximately 6,000 feet (1,828 meters) at the southern edge. This feature is essential to the conservation of the species, because it provides biotic communities that are known to be necessary for the spotted owl. This feature must occur with at least one of the additional physical or biological feature described below.
2. **Nesting, roosting, and foraging habitat.** Home ranges require forest types (described in (i) above) that contain one or more habitat types (nesting, roosting, foraging) that provides habitat components essential for survival and successful reproduction of a resident breeding pair. The core area of the home range is used most intensively and usually includes the nesting area. The remainder of the home range is used for foraging and roosting.

   Nesting habitat includes moderate to high (60 to 80 percent) canopy closure, multi-layered and multi- species canopy with greater than 30-inch-diameter-at-breast-height (dbh) overstory trees; high incidence of large trees with various deformities (e.g., large cavities, broken tops, mistletoe platforms); large snags; large accumulation of fallen trees and woody debris on the ground; and sufficient open space below the canopy for flying.

   Roosting habitat provides thermoregulation, shelter, and cover to reduce predation risk while resting or foraging. Habitat characteristics are similar to nesting habitat; however, they exclude features required for nesting (e.g., large cavities, broken tops, mistletoe platforms, snags).

   Foraging habitat provides a food supply for survival and reproduction, and contains some roosting habitat attributes, but can consist of more open and fragmented forests.
3. **Dispersal habitat** includes forest described in (i) above, and could be (a) younger, less-diverse stands than foraging habitat, but include some roosting structures and foraging habitat; or (b) habitat that is generally equivalent to roosting and foraging habitat. Dispersal habitat can occur in between or in larger blocks of nesting, foraging, and roosting habitat. Dispersal habitat is essential to maintaining stable populations by filling territorial vacancies when resident northern spotted owls die or leave their territories, and to provide adequate gene flow across the range of the species.

G.2.3.3 Life history

Spotted owl pairs occupy the same territories each year as long as suitable habitat is present. However, nesting may not occur every year, and survival of offspring varies annually and geographically. Nest trees are often used more than one year; but occasionally, a pair will move to a new nest tree within its home range.

Use of the same nest tree can occur in non-consecutive years, and some sites become re-occupied more than 8 years after the site was last used. Spotted owls begin their annual breeding cycle in late winter (late February to early March) when pairs begin to roost together (Thomas 1990). One to three eggs (usually two) are laid in March or April. Incubation lasts for approximately 30 days, and juvenile owls leave the nest 3 to 5 weeks after hatching. Many leave the nest site well before they are able to fly. Both parents feed the young until August or September. The young become independent in September or October at which time they disperse from the parental nest areas.

Spotted owls are mainly found in old-growth forests characterized by high canopy closure (greater than 70 percent), multi-layered canopy structure, large-diameter trees, downed logs, and snags (Thomas 1990, Buchanan 1991). The multi-layered canopy provides various microclimates, which helps spotted owls regulate their body temperature, and provides foraging, roosting, and nesting habitat. Although nests are found mainly in mature stands, they have also been observed in younger stands where the forest has been managed for uneven-aged stand composition, or in areas managed for rapid tree growth, facilitating habitat development in a relatively short period of time. Nests are found in tree or snag cavities, on platforms (abandoned raptor or raven nests, squirrel nests, mistletoe brooms, debris accumulations), or on top of broken-off snags. In more mature forests, spotted owls tend to use broken-top trees and cavities more frequently than platforms (LaHaye 1988, Buchanan 1991, Gutiérrez et al. 1995). Dispersal habitat typically includes stands that have at least an 11-in-average dbh, and at least 40 percent canopy closure (Thomas 1990); however, spotted owls use a wide variety of forest habitats for dispersal and will traverse very fragmented landscapes (USFWS 2011a).

G.2.3.4 Geographic distribution

The current range of the spotted owl extends from San Francisco Bay in Marin County north through the coast range of California, western Oregon, western Washington, to southwestern British Columbia (USFWS 1990).

G.2.3.5 Threats

Past habitat loss, current habitat loss, and competition by barred owls (*Strix varia*) are the most pressing current threats to the northern spotted owl (USFWS et al. 2008, USFWS 2011a). Davis et al. (2016) evaluated trends in NSO habitat, including late-successional and old-growth (older) forest, on federally administered lands since implementation of the Northwest Forest Plan (beginning in 1994 up to 2013) and found that decreases in the amount of older forests on federal lands managed under the Northwest Forest Plan have been small (a 2.8 to 2.9 percent net decrease) despite gross losses from wildfire (4.2 to 5.4 percent), timber harvest (1.2 to 1.3 percent), and from insects or other causes (0.7 to 0.9 percent).

Provinces that incurred the largest losses of older forest of federal lands were the Oregon Western Cascades, Oregon Klamath, and California Klamath (Davis et al. 2016).

Barred owls, which have expanded their distribution into the western United States, are now found in the Klamath Basin. Barred owls occupy a similar ecological niche to that of spotted owls. They forage in similar habitats, but barred owls have a much broader diet than spotted owls. In addition, barred owls appear to be more tolerant of disturbance and habitat fragmentation (Dark et al. 1998). Barred owls exhibit a behavioral dominance, which can lead to either displacement of spotted owls (Hamer 1988) or hybridization with spotted owls (Hamer et al. 1994). There is also some indication that barred owls may actually prey on spotted owls (Leskiw and Gutiérrez 1998). As part of the Northwest Forest Plan, long-term annual monitoring of northern spotted owls is conducted across the entire NSO range, including the South Cascades Demographic Study Area, about 104 kilometers (65 miles) northwest of the Proposed Action in two BLM Districts in Western Oregon (Medford and Roseburg) (Anthony et al., 2006, Davis et al. 2010). Dugger et al. (2016) estimated that northern spotted owls' populations declined by 3.8 percent per year from 1985 to 2013 in all parts of their range, and that the rate of decline was increasing in many areas, including southern Oregon and northern California. The only exception was in Green Diamond Resources land along the Northern California coast, where lethal removal of barred owls began in 2009. NSO populations started increasing following barred owl removal there. Results of that comprehensive study indicate that competition with barred owls may be the primary cause of northern spotted owl population declines across their range (Dugger et al. 2016).

G.2.3.6 Conservation needs/existing strategies

The USFWS Revised Recovery Plan for the Spotted Owl Recovery Plan (2011a) identifies four steps to conserve the species: (1) habitat modeling application; (2) active forest management and habitat conservation; (3) barred owl management; and (4) research and monitoring.

A spatially explicit demographic modeling application is described in the USFWS 2011a Revised Recovery Plan. The modeling tool evaluated information from over 4,000 spotted owl sites and nesting and roosting geographic data in development of a conservation planning framework. The conservation planning framework integrates a spotted owl habitat model, a habitat conservation planning model, and a population simulation model. Collectively, these modeling tools allow comparison of estimated spotted owl population performance among alternative habitat conservation network scenarios under a variety of potential conditions (USFWS 2011a).

Management strategies to provide suitable habitat and connectivity between populations have been implemented on state and federal lands. In Oregon and California, HCPs and Safe Harbor Agreements cover more than 970,000 acres of non-federal land (USFWS 2010a). Management of federal land under land-use allocations, identified in the Northwest Forest Plan (i.e., Late-Successional Reserves, Managed Late-Successional Areas, and Congressionally Reserved Areas) are intended to directly support northern spotted owl habitat, and connectivity of habitat between populations. Management of other land-use allocations (i.e., Adaptive Management Areas, Administratively Withdrawn Areas, and Riparian Reserves) can provide support for habitat and connectivity between populations; however, that is not the management goal.

G.2.3.7 Status in the Action Area

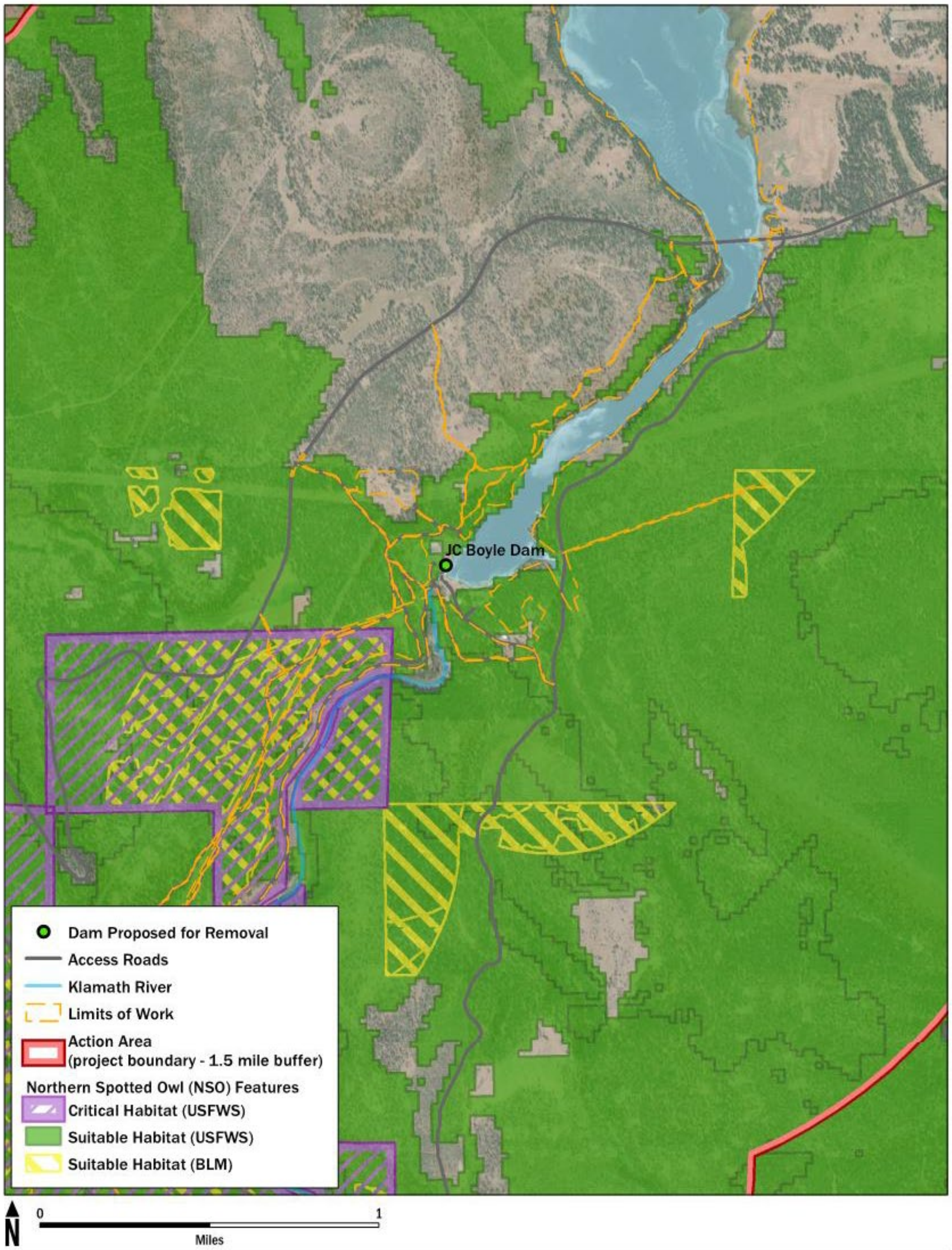
The majority of habitat in the Action Area is considered unsuitable for NSO, especially in the vicinity of Iron Gate Reservoir (see Figures G-8 through G-11). Adjacent to the J.C. Boyle powerhouse, there are small, isolated stands of trees that may provide roosting and foraging opportunities; however, the surrounding area consists of younger forest stands with open canopies that does not support nesting, roosting, or foraging. Southeast of Copco No. 1 Reservoir, there is nesting, roosting, and foraging habitat that supports a known activity center (Figure G-10). The majority of land in this area is owned by private or other entities (which include easements and tribal lands) and BLM (Table G-5).

Table G-5: Land ownership[1] within a 2.4-kilometer (1.5-mile) buffer along the Klamath River from Iron Gate Dam upstream to the eastern side of J.C. Boyle Reservoir

Land ownership[1]	Acres (%)
Private or other[2]	64,281 (75%)
BLM	17,293 (20%)
USDA Forest Service	2,543 (3%)
State agency	1,593 (2%)
Total	85,710

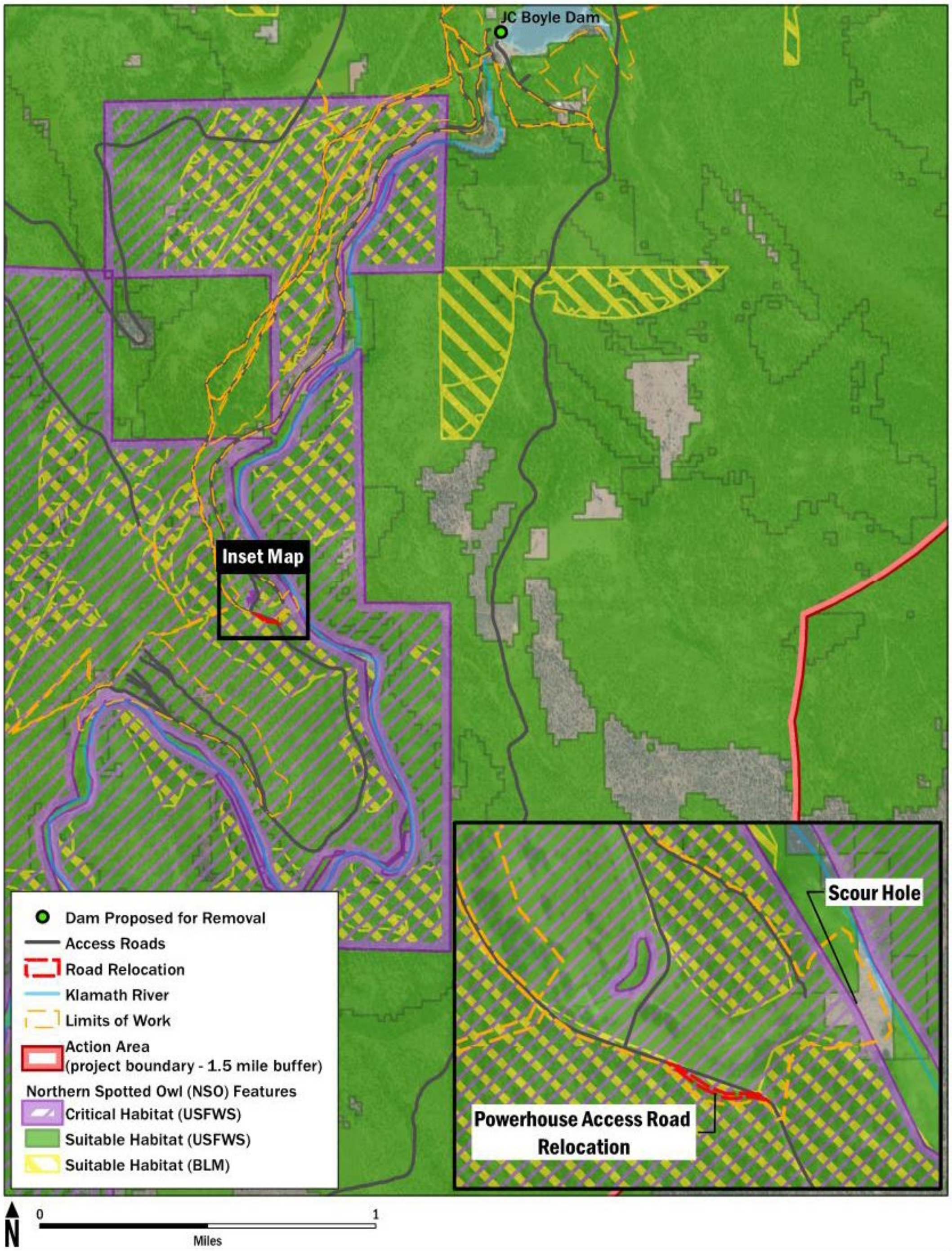
[1]Land ownership layer is BLM surface management data for Oregon and California.
[2]Other lands include those not managed by state or federal agencies. Private lands include easements and tribal lands.

Northern spotted owl activity centers have been documented in the vicinity of Copco No. 1 and J.C. Boyle reservoirs, as described in Table G-6. The activity center in the vicinity of Copco No. 1 Reservoir is approximately 1.3 miles southeast of the eastern end of Copco No. 1 Reservoir. The nearest activity center to the J.C. Boyle Reservoir is approximately 4.6 miles southwest of J.C. Boyle Dam. Nesting, roosting, or foraging habitat within the Action Area is limited to that in the vicinity of the activity center 1.3 miles southeast of the eastern end of Copco No. 1 Reservoir. A portion of this habitat is included in the 1.5-mile buffer surrounding the hydroelectric reach; however, no construction activities would occur in nesting, roosting, or foraging habitat.



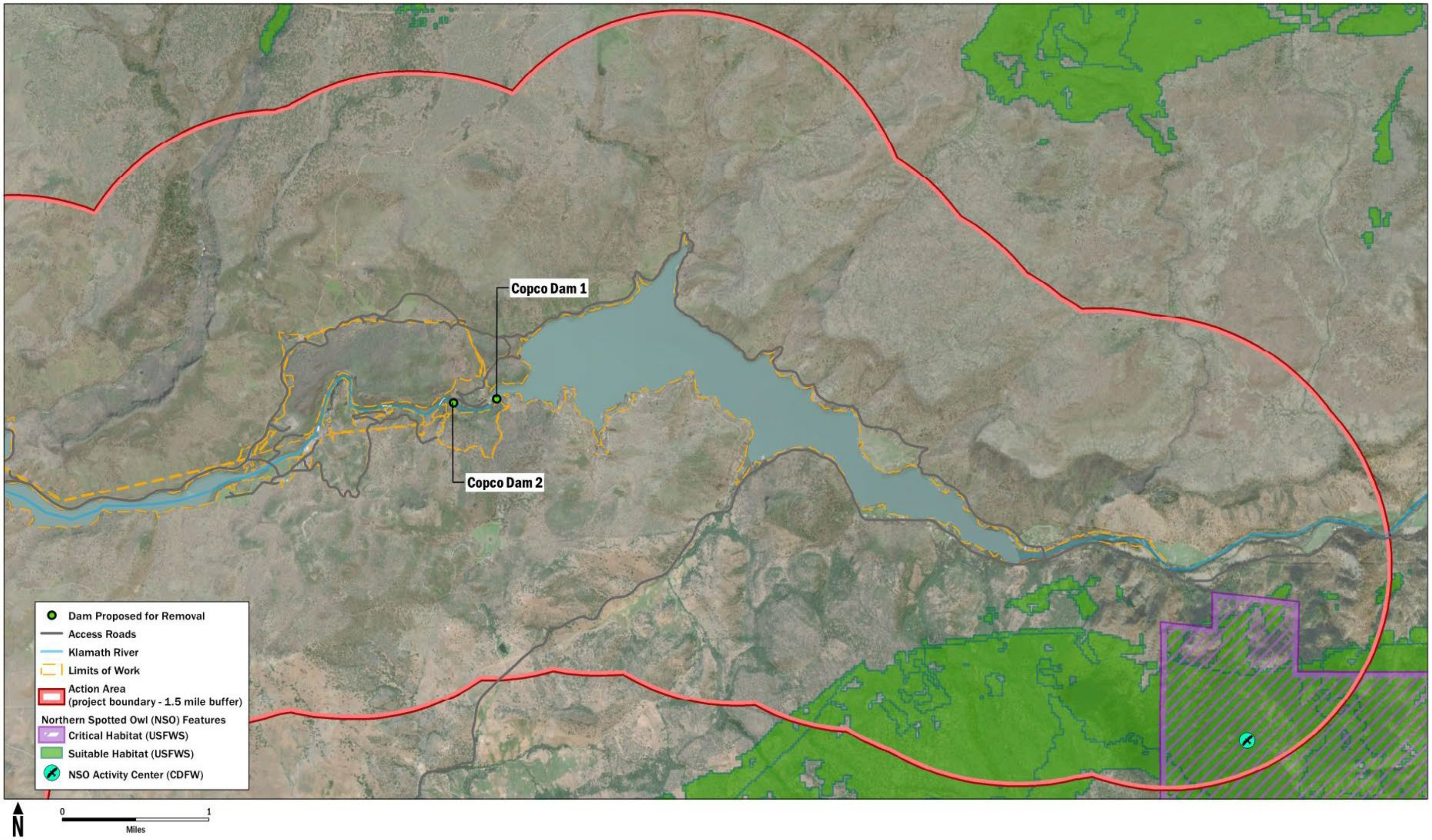
Klamath River Renewal Corporation, Klamath River Renewal Project

Figure G-8: J.C. Boyle (North) Northern Spotted Owl Habitat



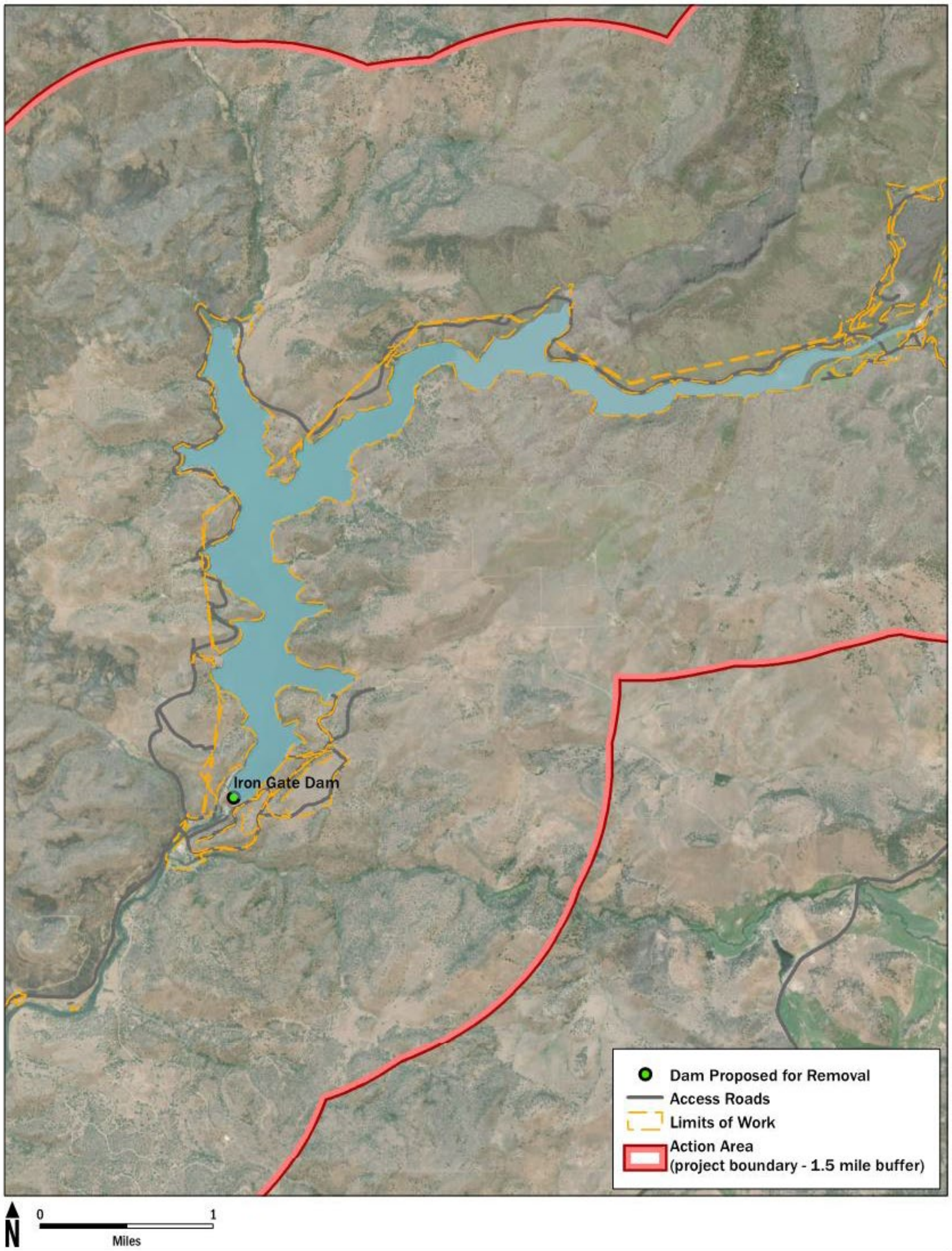
Klamath River Renewal Corporation, Klamath River Renewal Project

Figure G-9: J.C. Boyle (South) Northern Spotted Owl Habitat



Klamath River Renewal Corporation, Klamath River Renewal Project

Figure G-10: Copco Northern Spotted Owl Habitat



Klamath River Renewal Corporation, Klamath River Renewal Project

Figure G-11: Iron Gate Northern Spotted Owl Habitat

Table G-6: Summary of Current Northern Spotted Owl Habitat and Activity Centers Between Iron Gate Dam and J.C. Boyle Reservoir

Construction area	Northern spotted owl habitat and activity centers
Iron Gate Dam and associated construction areas	No suitable nesting, roosting, or foraging habitat is present in the vicinity of Iron Gate Dam (Oakley Consulting 2011; USFWS Relative Habitat Suitability mapping layers provided 2017), and no activity centers are present (D. Freeling, USFS Goosenest Ranger District. Pers comm., June 16, 2017). See Figure G-11.
Copco No. 1 Dam and associated construction areas	Suitable nesting and roosting habitat in the vicinity of Copco No. 1 Dam is about 8 kilometers (5 miles) east of Copco No. 1 Dam (Oakley Consulting 2011; USFWS Relative Habitat Suitability mapping layers provided 2017). Most of this habitat is included in the designated critical habitat (Figure G-10). This suitable habitat consists of mixed conifer in the steep north-facing canyon area that grades into ponderosa pine and oak woodland habitat to the west and the north (Oakley Consulting 2011). The critical habitat between Copco No. 1 Reservoir and mapped suitable nesting and roosting habitat is not identified as suitable nesting and roosting habitat. Suitable habitat is present north of Copco dam sites in Oregon (greater than 3.2 kilometers (2 miles) away, which is primarily on BLM land and is in small 16– to 24-hectare (40– to 60-acre) patches. One activity center is approximately 1.3 miles southeast of the eastern end of Copco No. 1 Reservoir in the suitable habitat described above (Figure G-10). PacifiCorp 2002 and 2003 surveys resulted in four detections in this vicinity. The status of this activity center, CNDDB SIS0301 and BLM Master Site Number (MSNO) 2191, is active. The activity center was confirmed to be occupied by NSO in 2017 (D. Freeling, USFS Goosenest Ranger District. Pers comm., June 16, 2017) and 2018 (CDFW 2019b). A summary of NSO detections documented in the vicinity of this activity center during the preceding 10-year period (i.e., 2008 to 2018) is provided in Table G-7.
Copco No. 2 Dam and associated areas	Suitable habitat is described above for Copco No. 1 Dam. The closest activity center is described above for Copco No. 1 Dam.
J.C. Boyle Dam and associated construction areas	Suitable nesting/roosting and foraging habitat is limited around the J.C. Boyle area Ponderosa pine forests in this area are generally younger and have low to moderate canopy closure. Relative Habitat Suitability mapping layers provided by USFWS (S. Galloway, Biologist USFWS Yreka Office, pers. comm., May 24, 2017) and BLM (S. Hayner, Biologist, Lakeview District, Klamath Falls Resource Area, pers. comm., August 24, 2017) indicate suitable habitat occurs approximately 1 mile away from the J.C. Boyle Reservoir and adjacent to the J.C. Boyle powerhouse (Figures G-8 and G-9, respectively). PacifiCorp surveys resulted in two detections near the J.C. Boyle Powerhouse, and one just north of the Klamath River downstream of the J.C. Boyle powerhouse in 2003; however, specific information on the observations (e.g., behavior status and the status of reproduction) was not able to be verified. The J.C. Boyle powerhouse is in designated critical habitat (USWFS 2012). An activity center (MSNO 1306; known as Buck Mountain), is about 9.5 kilometers (5.9 miles) northwest of J.C. Boyle Dam. The owl pair was last detected reproducing in 2007 but has not been observed in recent surveys. A second activity center (MSNO 2388; known as Topsy) is about 7.5 kilometers (4.6 miles) southwest of J.C. Boyle Dam. Surveys indicated this site was occupied by a single male in 2005 and 2006; was not occupied in 2007, 2008, 2009, 2011, and 2012; and was determined to be abandoned on January 31, 2013 (E. Willy, USFWS Klamath Office, pers. comm. March 26, 2018. BLM (S. Hayner, Lakeview District, Klamath Falls Resource Area, pers. comm., August 24, 2017) confirmed there are no NSO territories within the 1-mile noise disturbance buffer from potential blasting at the J.C. Boyle dam, or within 0.5 mile of the limits of work.

Table G-7: Summary of NSO Detections from 2008 to 2018 for Activity Center CNDDB SIS0301

Year	Single (M/F)	Non-nesting Pair	Nesting Pair	Young Observed	Not Surveyed
2008		X			
2008		X			
2009			X	1	
2010	Unknown			2	
2010	F			1	
2010			X	2	
2011	F				
2012					X
2013					X
2014	M				
2015					X
2016					X
2017	M				
2018	M				
2019		X			
Source: CDFW 2021					

Each NSO home range (or territory) generally includes one or more activity centers (USFWS 2012b). An NSO home range is usually represented by a 1.2-mile radius circle (1.3 miles in California) centered on the activity center. The activity center is a location where NSO have nested, consistently roosted, or where one or more individual NSO is detected multiple times over 1 or more years. There may be multiple activity centers in a given NSO home range. Surrounding most activity centers is a “core area” (defined as an 0.8- kilometer [0.5-mile] radius from the activity center) that receives a high amount of use relative to the remainder of the home range. Immediately surrounding the activity center (radius of 300 meters) there is often a contiguous patch of some of the highest quality habitat available, dominated by large-diameter trees, dense overhead canopy, and a structurally diverse understory. The likelihood of an effect to an owl activity center is determined based on the distance to the activity center, the amount of suitable habitat surrounding each activity center under current conditions, and the amount of habitat modification/removal that is anticipated to occur.

USFWS and BLM provided spatial data on habitat suitability for NSO in the Action Area. The USFWS Relative Habitat Suitability (RHS) model covers the entire Action Area, while the BLM data cover only the J.C. Boyle portion of the Action Area. Based on this habitat suitability information, “highly” suitable habitat for NSO occurs adjacent to the J.C. Boyle powerhouse and within 1 mile of the J.C. Boyle Reservoir. However, field surveys found that habitat for NSO in the J.C. Boyle area was marginal at best. The majority of the forested habitat consisted of younger forest stands with open canopies; however, a small number of isolated patches of habitat that may support roosting and/or foraging were observed. These isolated patches consisted of two or three larger diameter trees in close proximity and with features such as leaning or fallen trees, broken limbs, dense tangles, or other structure. These small, isolated patches would not be expected to support a future nesting pair given the lack of nesting, roosting, and foraging habitat available in the surrounding vicinity.

In the vicinity of the activity center southeast of Copco No. 1 Reservoir, the habitat consists of relatively young deciduous oak woodland in the lower elevations, with relatively open mixed forest at the higher elevations. The nearest NSO detection documented in the CNDDB is more than 1 mile from the bridge that crosses the eastern end of Copco No. 1 Reservoir (CDFW 2019b). The NSO activity center itself is farther to the southeast. In addition, most of the NSO detections documented in the CNDDB are outside of the project viewshed in a draw.

G.2.3.8 Summary of the current viability

Northern spotted owl populations are divided into physiographic provinces, four of which are included in the Action Area: Eastern Oregon Cascades, California Cascades, California Klamath, and California Coast. In general, these provinces include poor distribution and quality of existing habitat and a high level of natural and man-made fragmentation (USFWS 2011a). Davis et al. (2016) estimated the loss of northern spotted owl habitat in these provinces on federal lands, and all (federal and non-federal) lands. Wildfires caused the greatest loss of habitat on federal lands, while harvest contributed to the greatest losses of habitat when non-federal lands were included.

In July 1994, a total of 5,431 occupied spotted owl locations were known; however, because not all areas can or have been surveyed on an annual basis, the current range-wide status is unknown (USFWS 1992, USFWS 1995, and Thomas et al. 1993; all as cited in USFWS 2010a). Because existing survey coverage and effort are insufficient to produce reliable range-wide estimates of population size, researchers use other indices, such as demographic data, to evaluate trends in spotted owl populations. Analysis of demographic data can provide an estimate of the rate and direction of population change [i.e., lambda (λ)]. A λ of 1.0 indicates a stationary population (i.e., neither increasing nor decreasing), a λ less than 1.0 indicates a declining population, and a λ greater than 1.0 indicates a growing population (USFWS 2018c). Dugger et al. (2016) evaluated population trends using range-wide estimates of population size and demographic data for 11 study areas in Oregon, Washington, and California. The weighted mean estimate of λ for all 11 study areas was 0.962 (Standard Error = 0.019, 95 percent Confidence Interval = 0.925–0.999), indicating that between 1986 and 2013, the population declined 3.8 percent per year. Five of the 11 demographic study areas are in northern California and southern Oregon. The populations are declining (Table G-8).

Table G-8: Northern spotted owl parameters from the demographic study areas in northern California and southern Oregon

Demographic study area	Fecundity	Apparent survival[1]	λRJS^2 (SE; 95% CI)	Population change[3]
Klamath	Declining	Declining	0.972 (0.017; 0.940–1.005)	Declining
Southern Cascades	Declining	Declining	0.963 (0.024; 0.916–1.010)	Declining
NW California	Declining	Declining	0.970 (0.009; 0.951–0.989)	Declining
Hoopa	Declining	Declining	0.977 (0.010; 0.958–0.996)	Declining
Green Diamond[4]	Declining	Declining	0.961 (0.018; 0.926–0.996)	Declining

Source: Dugger et al. 2016
[1]Based on modeled average.
[2]Re-parameterized Jolly-Seber method.
[3]Based on estimates of realized population change.
[4]Green Diamond treatment area before barred owls were removed.

G.2.4 Oregon Spotted Frog (*Rana pretiosa*)

G.2.4.1 Species Status

The Oregon spotted frog (OSF) was listed as threatened under the ESA on August 29, 2014 (79 FR 51658).

G.2.4.2 Critical Habitat

Critical habitat for OSF was designated in areas of Washington and Oregon on May 11, 2016 (81 FR 29336). The critical habitat designation for OSF consists of 14 units, delineated by river sub-basins where OSF are extant: (1) Lower Chilliwack River; (2) South Fork Nooksack River; (3) Samish River; (4) Black River; (5) White Salmon River; (6) Middle Klickitat River; (7) Lower Deschutes River; (8) Upper Deschutes River; (9) Little Deschutes River; (10) McKenzie River; (11) Middle Fork Willamette River; (12) Williamson River; (13) Upper Klamath Lake; and (14) Upper Klamath.

In addition, the USFWS critical habitat designation identifies the following PBFs as essential to the species' conservation (81 FR 29354):

1. **Nonbreeding (N), Breeding (B), Rearing (R), and Overwintering Habitat (O)** - Ephemeral or permanent bodies of fresh water, including, but not limited to natural or manmade ponds, springs, lakes, slow-moving streams, or pools within oxbows adjacent to streams, canals, and ditches that have one of more of the following characteristics:
    - Inundated for a minimum of 4 months per year (B, R) – timing varies by elevation but may begin as early as February and last as long as September.
    - Inundated from October through March (O).
    - If ephemeral, areas are hydrologically connected by surface water flow to a permanent water body (e.g., pools, springs, ponds, lakes, streams, canals, or ditches) (B, R).

- Shallow water areas (less than or equal to 30 cm (12 inches), or water of this depth over vegetation in deeper water (B, R).
- Total surface area with less than 50 percent vegetative cover (N).
- Gradual topographic gradient (<3 percent slope) from shallow water toward deeper, permanent water (B, R).
- Herbaceous wetland vegetation (i.e. emergent, submergent, and floating-leaved aquatic plants), or vegetation that can structurally mimic emergent wetland vegetation through manipulation (B, R).
- Shallow water areas with high solar exposure or low (short) canopy cover (B, R).
- An absence or low density of nonnative predators (B, R, N).

2. **Aquatic movement corridors** - Ephemeral or permanent bodies of fresh water that have one or more of the following characteristics:
   - Less than or equal to 5 km (3.1 miles) linear distance from breeding areas;
   - Impediment free (including, but not limited to, hard barriers such as dams, impassable culverts, lack of water, or biological barriers such as abundant predators, or lack of refugia from predators).
3. **Refugia habitat** – Nonbreeding, breeding, rearing, or overwintering habitat or aquatic movement corridors with habitat characteristics (e.g., dense vegetation and/or an abundance of woody debris) that provide refugia from predators (e.g., nonnative fish or bullfrogs).

G.2.4.3 Life History

OSF requires shallow water areas for egg and tadpole survival; perennially deep, moderately vegetated pools for adult and juvenile survival in the dry season; and non-freezing perennial water to protect all age classes during cold weather at higher elevations. Emergent or floating aquatic vegetation is used by OSF for basking and cover. Large concentrations of OSF have been documented in areas with the following characteristics: (1) the presence of high-quality breeding and overwintering sites connected by perennial water; (2) consistent water depth throughout the period between egg-laying and metamorphosis; and (3) the absence of introduced predators, especially bullfrogs and introduced fish such as brook trout and centrarchids (*Micropterus* and *Lepomis* spp.) (Pearl et al. 2009).

Adult OSF generally begin to breed by one to three years of age, depending on sex, elevation, and latitude. Breeding occurs in February or March at lower elevations and between early April and early June at higher elevations (Leonard et al. 1993). Egg masses are typically laid communally in groups of up to several hundred (Licht 1971, Nussbaum et al. 1983, Cook 1984, Hayes 1997, Engler and Friesz 1998). Females deposit their egg masses on sedges and rushes occuring in shallow (i.e., generally less than 14 inches deep) pools of water on gradually receding shorelines, on benches of seasonal lakes and marshes, and in wet meadows.

OSF eggs typically hatch within three weeks of oviposition. Tadpoles are grazers and consume plant tissue and bacteria. Tadpoles metamorphose into froglets during their first summer. Post-metamorphic OSF feed primarily on insects. Predators can strongly affect the abundance of larval and post-metamorphic OSF with the heaviest losses to predation occurring shortly after tadpoles emerge from eggs (Licht 1974). Survival rates appear to increase as tadpoles develop and aquatic vegetation grows to provide improved cover (Licht 1974).

G.2.4.4 Geographic Distribution

Historically, OSFs were documented in 31 sub-basins ranging from British Columbia to the Pit River basin in northeastern California (McAllister et al. 1993, Hayes 1997, McAllister and Leonard 1997, Committee on the Status of Endangered Wildlife in Canada 2011). Currently, OSFs are found in 15 sub-basins ranging from extreme southwestern British Columbia south through the Puget Trough, and the Cascades Range from south-central Washington at least to the Klamath River basin in southern Oregon (79 FR 51662). OSFs have a limited distribution west of the Cascade crest in Oregon, are considered extirpated in the Willamette Valley in Oregon and may be extirpated in the portions of the Klamath and Pit River basins within California (Hayes 1997, Cushman and Pearl 2007).

G.2.4.5 Population Trends

OSF may no longer occupy as much as 90 percent of their historical range (79 FR 51667). In most sub-basins, trend information is limited or unavailable. The best scientific and commercial information available indicates that the trend is undetermined for OSF populations in 13 of the 15 occupied sub-basins and is declining in the Lower Fraser River and Middle Klickitat sub-basins (79 FR 51667).

G.2.4.6 Threats

At the time of listing, the USFWS determined that OSF was being impacted by one or more of the following factors to the extent that the species meets the definition of a threatened species under the ESA (79 FR 51658):

- Habitat necessary to support all life stages of OSF is continuing to be impacted and/or destroyed by human activities that result in the loss of wetlands to land conversions; hydrologic changes resulting from the operation of existing water diversions/manipulation structures, new and existing residential and road development, drought, and removal of beavers; changes in water temperature and vegetation structure resulting from reed canary grass (*Phalaris arundinacea*) invasions, plant succession, and restoration plantings; and increased sedimentation, increased water temperatures, reduced water quality, and vegetation changes resulting from the timing and intensity of livestock grazing (or in some instances, removal of livestock grazing at locations where it maintains early seral stage habitat essential for breeding);
- Predation by nonnative species, including nonnative trout and bullfrogs;
- Inadequate existing regulatory mechanisms that result in significant negative impacts such as habitat loss and modification; and

- Other natural or manmade factors including small and isolated breeding locations, low connectivity, low genetic diversity within occupied sub-basins, and genetic differentiation between sub-basins.

G.2.4.7 Status in the Action Area

The Action Area includes all tributaries to J.C. Boyle Reservoir (i.e., Spencer Creek) and Upper Klamath Lake that will be accessible to salmonids following dam removal up to the limits of anadromy. Thus, the Action Area includes areas inhabited by OSF in the Upper Klamath River sub-basin and overlaps portions of the Upper Klamath and Upper Klamath Lake critical habitat units. According to information available at the time the species was listed in 2014, OSF occupy Buck Lake and may inhabit suitable reaches of Spencer Creek occuring downstream (79 FR 51667). However, recent surveys indicated that the Buck Lake population is in decline (Lerum 2012).

In the Upper Klamath Lake sub-basin area OSF reportedly occupy two watersheds that flow into Upper Klamath Lake: Klamath Lake and Wood River (79 FR 51666–51667). There are four populations in this sub-basin: Crane Creek, Fourmile Creek, Sevenmile Creek, and the Wood River channel in addition to the adjacent but separate BLM Wood River canal. Surveys completed in 2013 identified additional occupied habitat in Sun Creek, Annie Creek, and more locations of Crane Creek and Sevenmile Creek (79 FR 51657). These OSF populations occur in both riverine and wetland habitats. Historically, the Klamath Lake and Wood River watersheds were hydrologically connected. Survey efforts on Fourmile Creek, Sevenmile Creek, and the Wood River channel have been sporadic while Crane Creek and the BLM Wood River canal have been surveyed annually. These data suggest that there is still insufficient information to obtain population trends for all but the BLM Wood River canal population, which is declining. As of 2011, the minimum population estimate for the sub-basin was approximately 374 breeding individuals (male and female) (USGS multiple datasets, BLM multiple datasets). Permission to survey adjacent private lands has not been obtained; however, the private lands surrounding the known populations appear to have suitable habitat and likely contain additional breeding complexes and individuals. Trend data are lacking for three out of four populations in the Upper Klamath Lake.