Appendix C
Water Quality Supporting Technical Information

C.1 Water Temperature

C.1.1 Upper Klamath Basin

C.1.1.1 Wood, Williamson, and Sprague Rivers

The Williamson River, the Sprague River and their major tributaries are listed as impaired under Section 303(d) of the Clean Water Act (CWA) for water temperature based upon the 18ºC (64.4ºF) criteria for salmon and trout rearing (Oregon Department of Environmental Quality [ODEQ] 1998; see Klamath Facilities Removal Environmental Impact Statement/Environmental Impact Report (EIS/EIR) Section 3.2, Table 3.2-8). For waters supporting redband cutthroat trout, the temperature criterion is a 7-day average maximum temperature of 20ºC (68ºF) (see Section 3.2, Table 3.2-8).

Mainstem water temperature in the Williamson and Sprague rivers have maximum values (21ºC [70ºF]) during May–October (ODEQ 2002, Attachment 1). Exposure to solar radiation during summer and early fall months heats surface water rapidly in headwater meadows of the Williamson River (David Evans and Associates [DEA] 2005). Heat energy is dissipated through turbulence occurring downstream of small impoundments and meadows on these rivers, and localized cooling from groundwater springs has been indicated along the mainstem Williamson and Sprague rivers. Spring Creek, Larkin Springs, Wickiup Spring, Williamson River Spring and Kamkaun Spring are examples of large groundwater sources that appear to have a significant cooling effect on surface waters in these tributaries to Upper Klamath Lake (ODEQ 2002, Attachment 1). Seasonal irrigation withdrawals and agricultural return flows increase mainstem stream temperatures during summer time, depending upon relative flow volume and air temperatures. Widespread removal of riparian vegetation and alterations to channel morphology also increase summer water temperatures in the Williamson and Sprague rivers. While the Upper Klamath Lake Total Maximum Daily Load (TMDL) and water quality management plan were completed in 2002, the Williamson and Sprague rivers will retain their water quality limited status until they achieve water quality standards.

The Wood River is listed as attaining cold water fish rearing temperature (18ºC [64.4 F]) from River Mile (RM) 0–17.9 (ODEQ 2002, Attachment 1). Wood River originates from a group of springs near Fort Klamath and stream flow is mostly groundwater-derived.

Campbell et al. (1993) report year-round headwater stream temperatures of 4.7–7.4ºC (40.5–45ºF) and water entering Agency Lake at 2.3–16.8ºC (36–62ºF).

C.1.1.2 Upper Klamath Lake

Water temperatures in Upper Klamath Lake regularly exceed 20ºC (68ºF) during July and August based on data collected since 1990 (Kann 2010). The mean depth of the lake is relatively shallow, at 8 feet when the lake level is at mean summer elevation (1,262.3 m [4,141.3 ft]; Gearhart et al. 1995), increasing the effect of solar radiation on lake temperatures beyond that of deeper lakes. Upper Klamath Lake undergoes periods of intermittent, weak stratification (Kann and Walker 1999), which has implications for dissolved oxygen and pH (Wood et al. 2006). As described above, water inputs from the Williamson River exceed 20ºC (68ºF) during the summer months (U.S. Geologic Survey [USGS] Data Grapher 2010). Groundwater discharges providing cooler water directly to the lake appear to have little effect on overall water temperature, with the exception of springs in or near Pelican Bay on the northwest side of the lake.

C.1.1.3 Link River Dam to Klamath River upstream of J.C. Boyle Reservoir

The Upper Klamath River is listed as impaired under Section 303(d) of the CWA for summertime water temperature (see Section 3.2, Table 3.2-8). Weekly measurements in 2007 in the Link River and upper Keno Impoundment reveal maximum temperatures of 23ºC (73.4ºF) in mid-to-late summer (Appendix B in Sullivan et al. 2008). Keno Impoundment exhibits a weak, intermittent stratification during the summer months, with maximum water temperatures exceeding 25ºC (77ºF) (Deas and Vaughn 2006). Recorded average monthly temperatures (2001–2004) in Keno Impoundment are 22.4ºC (72.3ºF) in July, 20.8ºC (69.4ºF) in August and 18.0ºC (64.4ºF) in September (Federal Energy Regulatory Commission [FERC] 2007). Average monthly temperatures reported by PacifiCorp downstream of Keno Dam are 23.2ºC (73.8ºF), 21.1ºC (70.0ºF), and 16.9ºC (62.4ºF) during July, August, and September (2001–2004), respectively (FERC 2007), exceeding Oregon water quality objectives for core coldwater habitat (16°C [60.8°F]) (see Section 3.2, Table 3.2-3). Similarly, during 2009, summer water temperatures downstream of Keno Dam were generally greater than 16°C (60.8°F) from June–September, with peak temperatures exceeding 26°C (78.8°F) in late-July (Watercourse Engineering, Inc. 2011).

C.1.1.4 Hydroelectric Reach

The Hydroelectric Reach spans the Oregon–California state line; both states include this reach on their Section 303(d) lists of impaired waters for water temperature (see Section 3.2, Table 3.2-8). During summer months, maximum weekly maximum temperatures (MWMTs) in the Hydroelectric Reach regularly exceed the range of chronic effects temperature thresholds (13–20°C [55.4–68°F]) for full salmonid support (North Coast Regional Water Quality Control Board [NCRWQCB] 2010, Kirk et al. 2010).

In general, water temperatures in this reach follow a seasonal pattern, with average monthly water temperatures from March through November ranging from just over 5°C (41ºF) in November to more than 22°C (71.6ºF) during June through August (FERC 2007). Winter water temperatures throughout the reach are largely driven by the temperature of river inflows (Deas and Orlob 1999). In the summer, the relatively shallow J.C. Boyle Reservoir (like the upstream Keno Impoundment) does not exhibit long-term thermal stratification, with a typical vertical temperature difference of less than 2°C (3.6ºF) in the water column (FERC 2007; Raymond 2008, 2009, 2010). Downstream of J.C. Boyle Dam, at approximately RM 224.7, water from cool groundwater springs at a relatively constant 11 to 12ºC (51.8 to 53.6ºF) mixes with river water that can exceed 25ºC (77ºF) in July and August (Kirk et al. 2010). When combined with peaking flows from the upstream J.C. Boyle Powerhouse, the groundwater springs create a unique summertime temperature signal on the Klamath River downstream of J.C. Boyle Dam; non-peaking flows are dominated by the cooler spring water while peaking flows are dominated by warmer water from reservoir discharges (PacifiCorp 2006, Kirk et al. 2010).

Within and downstream of Copco 1 Reservoir, spring, summer and fall temperatures in the Hydroelectric Reach are heavily influenced by the large thermal mass of the two deepest reservoirs, Copco 1 and Iron Gate Reservoirs, and their seasonal stratification patterns. Spring temperatures are generally cooler than would be expected under natural conditions, and summer and fall temperatures are generally warmer (PacifiCorp 2004a, NCRWQCB 2010). Both Iron Gate and Copco 1 Reservoirs thermally stratify beginning in April/May and do not mix again until October/November (Table C-1). The onset of spring/summer stratification and the timing of fall turnover in Iron Gate and Copco 1 Reservoirs are driven by meteorological conditions (Deas and Orlob 1999).

Table C-1. General Reservoir Turnover Dates for Copco 1 and Iron Gate Reservoirs (2007–2009).

Year	Thermally Stable Hypolimnion Establishment Date		Approximate Reservoir Turnover Date		Source
	Copco	Iron Gate	Copco	Iron Gate	
2007	By June 6	By June 6	Before October 23	Before November 28	Raymond 2008
2008	By April 30	By April 30	Before October 22	Before November 19	Raymond 2009
2009	By May 24	By May 24	Before October 13	Before November 17	Raymond 2010

Powerhouse withdrawals for Copco 1 and Iron Gate Dams are primarily from the epilimnion (surface waters). In Copco 1 Reservoir, powerhouse withdrawal is from approximately 6 m (20 ft) below the water surface when the reservoir is full and in Iron Gate Reservoir powerhouse withdrawal is from approximately 12 m (39 ft) below the water surface (National Research Council 2003). Occasionally, withdrawals extend into the hypolimnion; for example, in Iron Gate Reservoir, the withdrawal envelope has been estimated to extend down to approximately 18 m (60 ft) in depth (Deas and Orlob 1999).

Additionally, a small withdrawal (about 50 cfs) for the Iron Gate Hatchery occurs from the hypolimnion at Iron Gate Reservoir. In general, however, temperature in waters discharged from Copco 1 and Iron Gate Reservoirs reflect the warmer temperatures of surface water (National Research Council 2003). Stratification of the reservoirs also prevents mixing of waters within the water column and affects dissolved oxygen, nutrient concentration (and speciation), and pH in the Hydroelectric Reach and the Klamath River just downstream of Iron Gate Dam (Section C.1.2).

C.1.2 Lower Klamath Basin

C.1.2.1 Iron Gate Dam to Salmon River

The middle portion of the Klamath River, including the reach between Iron Gate Dam and the Salmon River, is listed as impaired under Section 303(d) of the CWA for water temperatures (see Section 3.2, Table 3.2-3). Water temperature in the Lower Klamath Basin varies seasonally, with mean monthly temperatures in the river downstream of Iron Gate Dam ranging from 3 to 6°C (37–43°F) in January to 20–22.5°C (68–72.5°F) in July and August (Bartholow 2005, Karuk Tribe of California 2009, Watercourse Engineering, Inc. 2011). Based upon annual water temperature monitoring conducted by the Karuk Tribe since 2000, water temperatures peak during the summer when air temperatures increase and flows decrease in the Klamath Basin (Figure C-1; Karuk Tribe of California 2002, 2003, 2007, 2009, 2010). Daily average summer water temperatures documented immediately downstream of Iron Gate Dam are generally cooler and less variable than those documented farther downstream in the Klamath River; daily average temperatures between June and September are approximately 1–4°C (1.8–7.2°F) higher near Seiad Valley and just downstream of the Salmon River confluence (i.e., at Orleans) than those just downstream of Iron Gate Dam (Figure C-2; Karuk Tribe of California 2009, 2010).

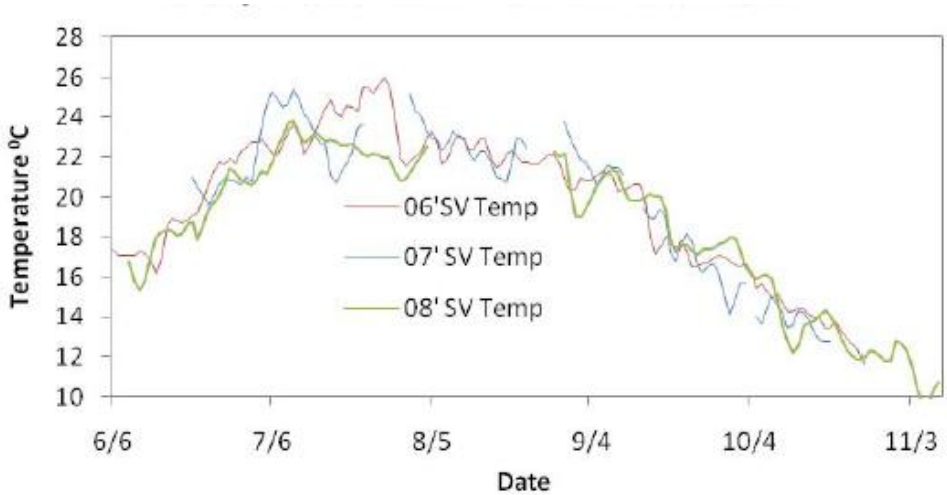

Figure C-1. Daily Average Water Temperature in the Klamath River near Seiad Valley (RM 129.4) June through November 2006, 2007, and 2008. Source: Karuk Tribe of California 2009.

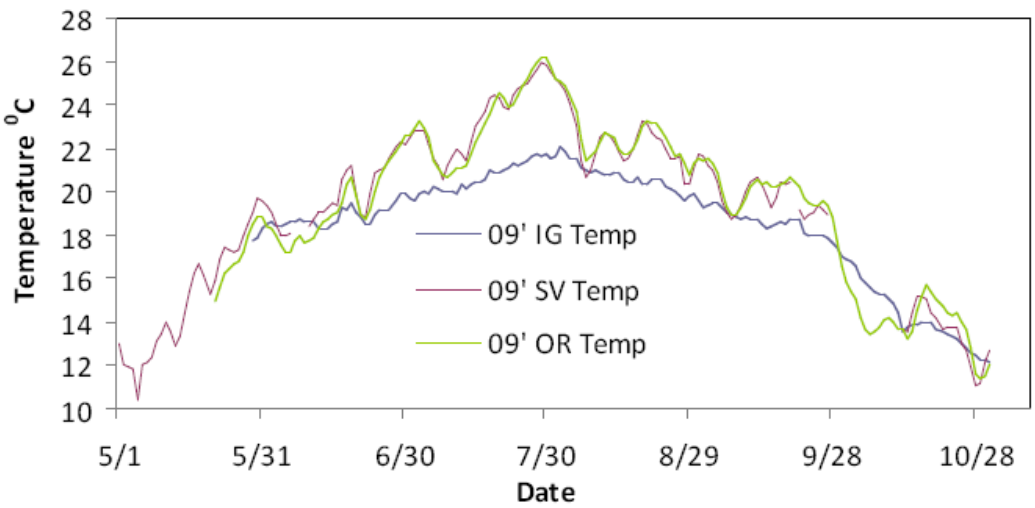

Figure C-2. Daily Average Water Temperature in the Klamath River Downstream of Iron Gate Dam (≈RM 189), near Seiad Valley (RM 129.4) and at Orleans (RM 59) during May through October 2009. Source: Karuk Tribe of California 2010.

With respect to the longer term water temperature record (i.e., prior to 2000), Bartholow (2005) presents evidence that water temperatures in the lower Klamath River have been increasing since before 1950. Bartholow (2005) indicates that the observed multi-decade trend of increasing water temperatures in the lower river is related to the cyclic Pacific Decadal Oscillation and is consistent with a measured average basinwide air temperature increase of 0.33°C/decade (0.59°F/decade). Bartholow (2005) estimates that the season of high temperatures that are potentially stressful to salmonids has lengthened by about 1 month in the Klamath River since the early 1960s, and the average length of the lower river exhibiting a summer water temperatures less than 15°C (59°F) has declined by about 8.2 km/decade (5.1 mi/decade). Potential climate change effects on water temperature are discussed in more detail as part of the effects determination for the No Action/No Project Alternative (see Section 3.2.4.3 – No Action/No Project Alternative – Water Temperature).

C.1.2.2 Salmon River to Estuary

The lower Klamath River between the Salmon River and the Klamath Estuary is listed as impaired under Section 303(d) of the CWA for water temperature (see Section 3.2, Table 3.2-3). Water temperature monitoring by the Karuk Tribe includes data from Orleans (RM 59), which is just downstream of the Salmon River confluence with the mainstem. Daily average water temperature at Orleans was 10.5–26°C (50.9–78.8ºF) from June through November 2006–2008, with the warmest temperatures generally occurring during July (Figure C-3; Karuk Tribe of California 2009). In the mainstem river between the Klamath River's confluence with the Trinity River and the Klamath Estuary, the Yurok Tribe, through the Yurok Tribe Environmental Program (YTEP) has conducted annual water temperature monitoring since 2002 (YTEP 2004). Peak temperatures

generally occur in mid- to late-July, with the highest daily maximum temperatures recorded at the most upstream locations and a small (0.5°C [0.9°F]) cooling effect detected from the contribution of the Trinity River to the mainstem Klamath River (Sinnott 2010). During May through November 2009, water temperatures ranged from approximately 11.1°C (52.0°F) in October to 26.8°C (80.2°F) in July (Sinnott 2010). Daily maximum summer water temperatures were greater than 26°C (78.8°F) just upstream of the confluence with the Trinity River (Weitchpec [RM 43.5]), decreasing to 24.5°C (76.1°F) near Turwar Creek (RM 5.8) (Figure C-4, YTEP 2005, Sinnott 2010). These summer temperatures exceed optimal growth thresholds as well as critical thermal maxima for coho, Chinook salmon, and steelhead (Brett 1952, Armour 1991, Stein et al. 1972, McGeer et al. 1991). Historically, summer water temperature maxima in the lower Klamath River have been greater than in other coastal rivers to the north and south. For example, Blakey (1966, as cited in Bartholow 2005), reports water temperatures in the Klamath River downstream of the Trinity River confluence (RM 42.5) reaching 26.6°C (79.9°F) for up to 10 days per year, in contrast to proximal coastal rivers that never reach this temperature

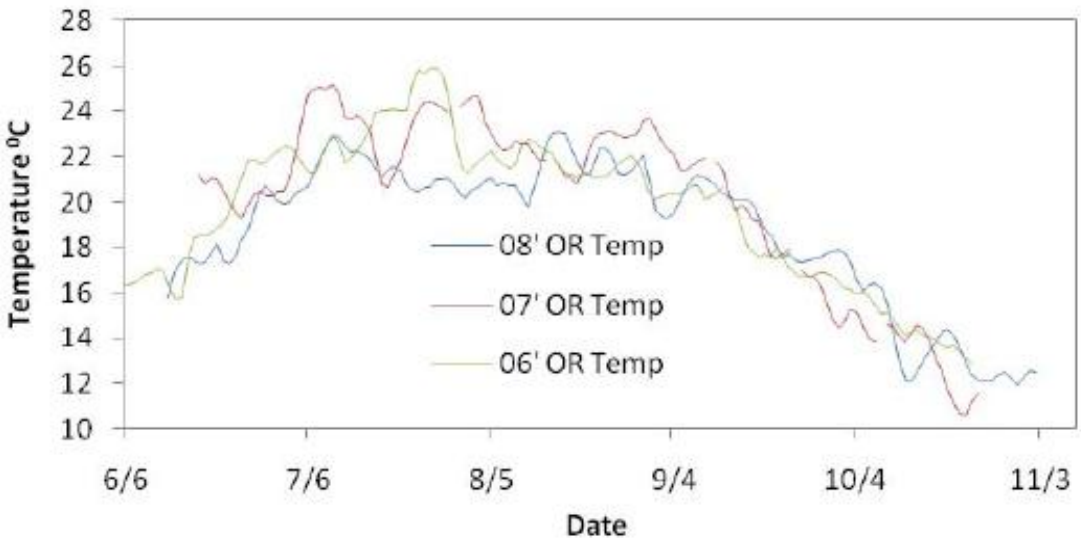

Figure C-3. Daily Average Water Temperature in the Klamath River at Orleans (RM 59) June through November 2006, 2007, and 2008. Source: Karuk Tribe of California 2009.

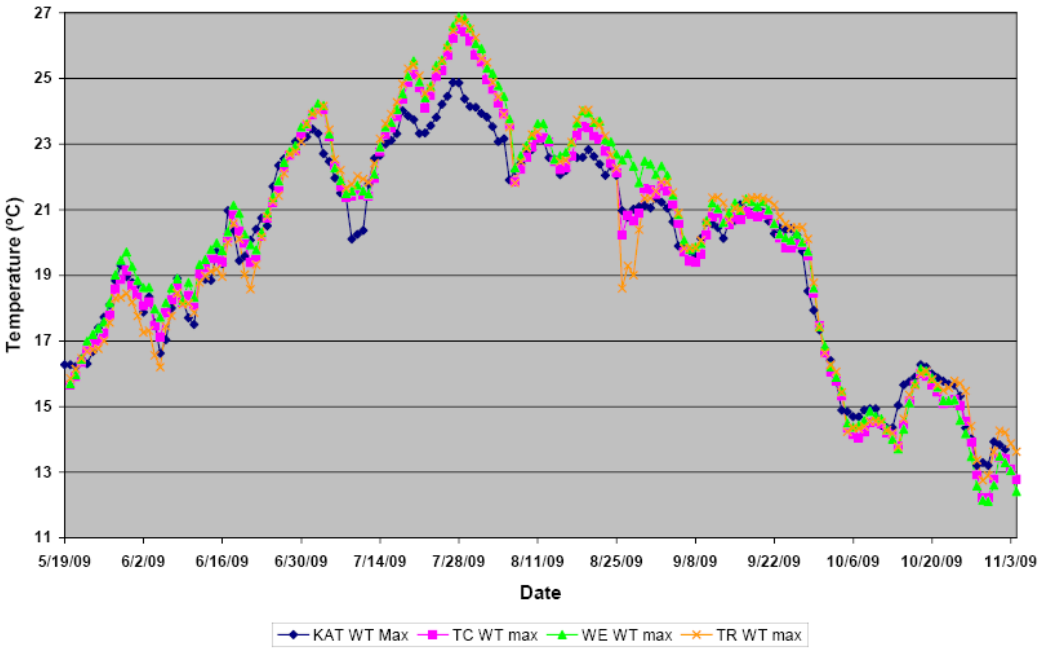

Figure C-4. Daily Maximum Water Temperatures in the Klamath River at Weitchpec (RM 43.5 [WE]), Upstream of Tully Creek (RM 38.5 [TC]), and Upstream of Turwar Boat Ramp (RM 8 [KAT]), as well as in the Trinity River (RM 40) near the Confluence with the Klamath River (RM 0.5 [TR]) May through November 2009. Source: Sinnott 2010.

C.1.2.3 Klamath Estuary

Hydrodynamics and water quality within the estuary are highly variable spatially and temporally and are greatly influenced by season, river flow, vertical water column stratification (thermal and/or chemical), and location of the estuary mouth, the latter changing due to periodic sand bar movement. The lower Klamath River, including the estuary, is listed as impaired under Section 303(d) of the CWA for water temperature (see Section 3.2, Table 3.2-3). Water temperature has been monitored in the Klamath Estuary by California Department of Fish and Game (Wallace 1998) and most recently by the Yurok Tribe Fisheries Program (Hiner 2006) and the YTEP (2005), with support from the NCRWQCB. Water temperatures in the estuary from December through April are roughly 5–12°C (41–54°F) (Hiner 2006). In summer and fall months, warmer air temperatures and lower flows result in increased water temperatures. Under low-flow summertime conditions, water temperatures in the estuary have been observed at 20–24°C (68–75.2°F) (Wallace 1998) or greater than 24°C (75.2°F) (Hiner 2006). During June–September 2009, water temperatures were 18.7–20.7°C (65.7–69.3°F) (Watercourse Engineering, Inc. 2011). These levels exceed optimal growth thresholds for salmonids, as cited in the previous section.

Estuarine water temperature is linked to upstream hydrology and periods of mouth closure because when the estuary mouth is open, denser salt water from the ocean sinks

below the lighter fresh river water, resulting in chemical stratification and a "salt wedge" that moves up and down the estuary with the daily tides (Horne and Goldman 1994, Wallace 1998, Hiner 2006). The salt wedge is also thermally stratified with cooler, high salinity ocean waters remaining near the estuary bottom, and warmer, low salinity river water near the surface. Upstream hydrology can affect the location of the salt water wedge and thus affect thermal structure in the estuary. For example, during pulse flows released from the Lewiston Dam on the Trinity River in August 2004, the upstream extent of the salt wedge moved downstream approximately one mile (YTEP 2005). In the Klamath Estuary, mouth closure has been reported to reduce the size of the salt water wedge, decrease overall salinity, and subsequently increase water temperatures in the estuary (Hiner 2006). Mouth closure, caused by formation of a sand berm across the mouth of the estuary, is a function of off-shore and alongshore wave power and sediment supply, freshwater inflows, the tidal prism, and morphological characteristics of the inlet (Escoffier 1940, Brunn 1966, O'Brien 1971, Barnes 1980). The historical frequency and duration of mouth closure in the Klamath Estuary has not been documented, although it is expected to occur during low-flow periods (June–October).

C.2 Suspended Sediments

For the purposes of the Klamath Facilities Removal EIS/EIR, suspended sediments refer to settleable suspended material in the water column. Bed materials, such as gravels and larger substrates, are discussed in Section 3.3.3.2 Aquatic Resources – Existing Conditions/Affected Environment – Physical Habitat Descriptions. Two types of suspended material are considered for the EIS/EIR analysis: algal-derived (organic) suspended material and mineral (inorganic) suspended material. Sources of each type of suspended material differ, as do spatial and temporal trends for each within the Upper and Lower Klamath Basins.

Often, suspended materials in the water column are quantified by measuring the concentration of total suspended solids (TSS). Turbidity, an optical property referring to the amount of light scattered or absorbed by a fluid, is another common way to quantify suspended materials and is measured in nephelometric turbidity units (NTUs). The exact relationship between turbidity and suspended sediment is dependent on the parent geology and must be determined for each watershed (Montgomery 1985, MacDonald et al. 1991). Turbidity and TSS affect organisms directly (e.g., interfering with vision) or indirectly by changing water temperature and Dissolved Oxygen (DO), and are often associated with the sorption of contaminants from the water column (e.g., polar organics and cationic metal forms). Municipal and domestic water supply beneficial uses can also be adversely affected by changes in suspended sediment concentrations and turbidity in streams.

For the Klamath River, coincident turbidity data is occasionally presented along with TSS data. However, as the dataset is not consistent in space or time, turbidity levels are not used to support significance determinations (see Section 3.2.4.2 – Thresholds of

Significance for Narrative Standards or Water Quality Objectives – Suspended Sediments) and are not analyzed in detail in this EIS/EIR.

C.2.1 Upper Klamath Basin

C.2.1.1 Wood, Williamson, and Sprague Rivers

Recently collected USGS data (2008–2010) for TSS at the Williamson River near Chiloquin indicate a range of 3–63 milligrams per liter (mg/L) with peak values occurring in February and March of 2009 (Figure C-5). Coincident turbidity data collected roughly monthly from March 2008 to July 2009 at this same location range from 1.6 to 35 formazin nephelometric units with similar peaks during late winter/early spring. Causes of fine sediment delivery to the upper Williamson River that may increase suspended sediment and turbidity include stream bank erosion from agricultural lands, lack of interception from riparian vegetation, timber harvest, and road construction and maintenance (DEA 2005, ODEQ 2002). Road density in the upper Williamson River subbasin is approximately 3.45 mi/mi2, with 6.4 percent of total road miles found within 200 feet of a stream channel (DEA 2005). For the Williamson River watershed as a whole, the number of stream crossings per mile of road (road/stream crossing density) is reported as 0.4 (Bureau of Land Management 2005, 2006; as cited in Rabe and Calonje 2009). Most of the roads in the Winema National Forest, in the eastern portion of the Williamson River subbasin, are dirt gravel (unpaved) (Gearhart et al. 1995) and produce fine sediment that can be delivered directly to streams (DEA 2005). Based on a watershed analysis of the Deep, Sand and Aspen Creek tributaries to the Williamson River, estimates of total road sediment yield are 5 to 20 times greater than background rates, with the greatest yield originating in the Sand Creek subbasin (Weyerhaeuser Company 1996, as cited in DEA 2005).

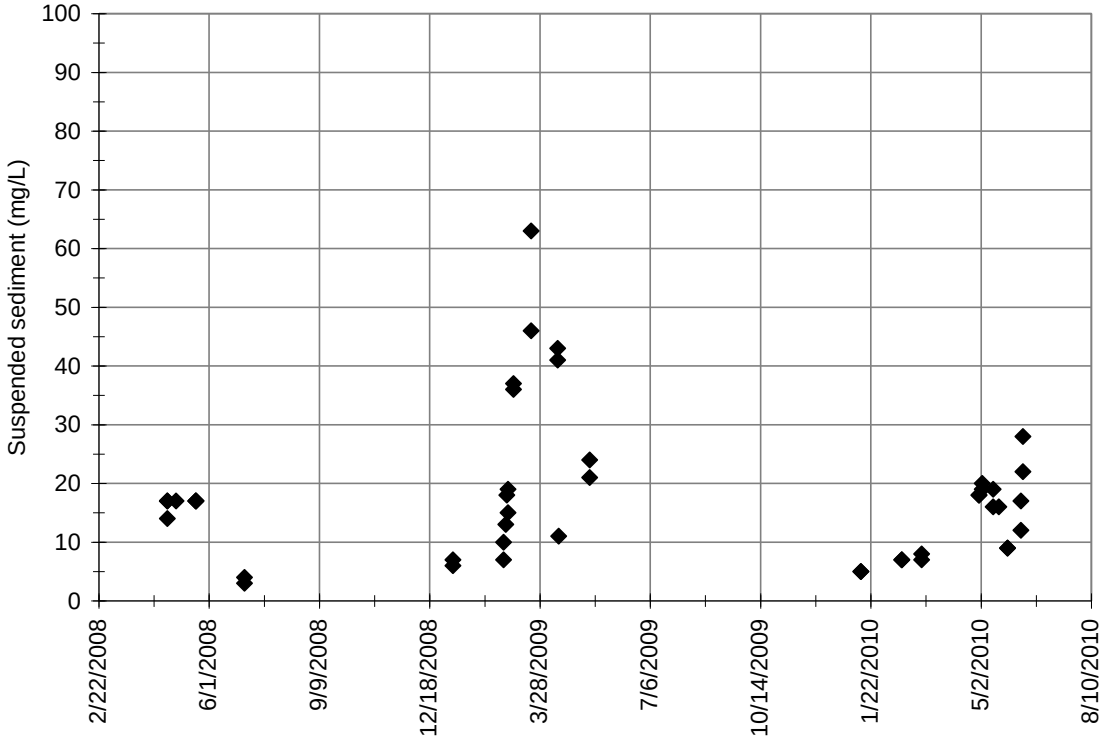

Figure C-5. Suspended Sediment (mg/L) Grab Samples for USGS Williamson River downstream of Sprague River near Chiloquin (USGS Gage No. 11502500) 2008–2010. Source: USGS 2011 (http://waterdata.usgs.gov/nwis)

Recently collected USGS data (2008–2010) for TSS at the Sprague River at Chiloquin range 4–88 mg/L with peak values generally occurring in February and March (Figure C-6). Coincident turbidity data collected roughly monthly from March 2008 to July 2009 at this same location were 2–53 formazin nephelometric units, with similar peaks during late winter/early spring. The Sprague River has been identified as a primary source of sediment to Upper Klamath Lake, based on analyses conducted to determine associated phosphorus loading to the lake (Gearheart et al. 1995). Relatively high and variable rates of runoff and erosion in the Sprague River drainage, as compared to the Williamson River drainage, have been identified as the source of bound phosphorus generated during seasonal runoff events and delivered to Upper Klamath Lake (ODEQ 2002, Connelly and Lyons 2007). The high sediment delivery rates have been identified as a factor affecting water temperatures in the upper basin and have altered relationships between depth and width of stream channels (Gearheart et al. 1995, ODEQ 2002). The sources of the sediment inputs within the Sprague River drainage include agriculture, livestock grazing and forestry activities, and road-related erosion (ODEQ 2002, Connelly and Lyons 2007, Rabe and Calonje 2009).

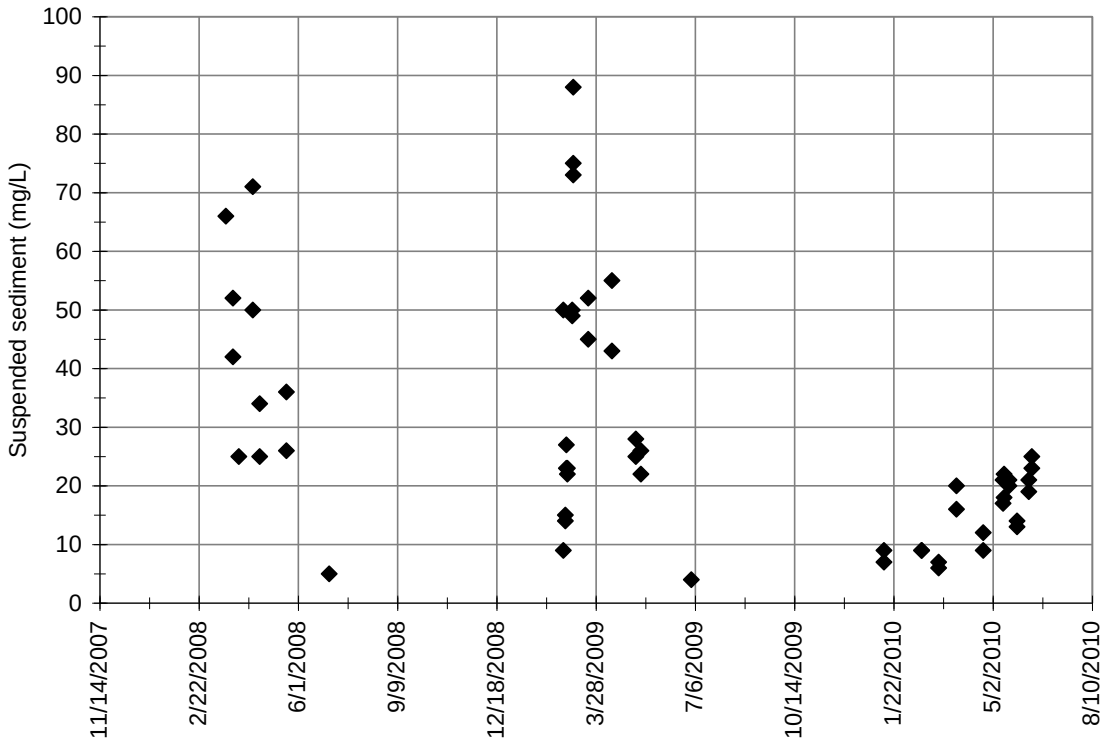

Figure C-6. Suspended Sediment (mg/L) Grab Samples for USGS Sprague River at Chiloquin (USGS Gage No. 11501000) 2008–2010. Source: USGS 2011 (http://waterdata.usgs.gov/nwis)

Campbell et al. (1993) examined TSS levels in the mainstem Wood River and report highly variable concentrations of 0–1.41 mg/L in the headwaters at Dixon Road and 0.6-5.88 mg/L at the mouth near Agency Dike Road. Available data indicate that turbidity in the Wood River averages 0.65 NTUs at the headwaters and 1.0–4.5 NTUs at the mouth (Campbell et al. 1993).

C.2.1.2 Upper Klamath Lake

While not focused on suspended materials per se, a variety of studies indicate that fine sediment delivery to Upper Klamath Lake has been relatively high during the 20th century. Suspended sediment inputs to Upper Klamath Lake have been examined using isotopic dating studies of sediment cores to determine the historical timing and potential sources of sediments to the lake. Using lead isotopic ratios (^{210}Pb:^{206}Pb), Eilers et al. (2004) demonstrate an increase in sediment accumulation rates in Upper Klamath Lake from 3 to 22 grams per square meter per year from the early to late 20th century. High titanium and aluminum concentrations in upper layers of lake sediments further indicate accelerated erosional inputs associated with 20th century watershed disturbances (Eilers et al. 2004). Gearhart et al. (1995) estimate 932 acre-feet per year of reduction in lake volume due to the sediment accumulation between 1920 and 1980.

Bradbury et al. (2004) suggest that a combination of wetland draining and channelization of tributaries to Upper Klamath Lake have increased erosion in the watershed during the 20th century.

Additionally, Eilers et al. (2004) report higher isotopic ratios of nitrogen (^{15}N:^{14}N) in 20th century sediment deposits in Upper Klamath Lake, indicative of nonpoint sediment source inputs of nitrogen-based fertilizers (Fry 1999, as cited in Eilers et al. 2004). A significant increase in ^{15}N after the completion of the Link River Dam at the outlet of Upper Klamath Lake in 1921 suggests that large inputs from nonpoint sediment sources in the upper watershed have occurred (Eilers et al. 2004). Based on sediment core analyses by Eilers et al. (2004), the construction and operation of Link River Dam appears to have altered the timing and quantity of lake flushing flows. The dam may also have contributed to alterations in nutrient retention dynamics in Upper Klamath Lake (as evidenced by the increased ^{15}N:^{14}N ratios) concurrent with the increased nutrient loading due to anthropogenic activities in the basin.

C.2.1.3 Link River Dam to Klamath River Upstream of J.C. Boyle Reservoir

Between Link River at Klamath Falls (RM 253.1) and the upstream end of J.C. Boyle Reservoir (RM 224.7), suspended sediment and turbidity concentrations decrease longitudinally, as algae are exported from Upper Klamath Lake and into the quiescent, relatively long residence time (approximately 4 to 12 days [Sullivan et al. 2011]) waters of Lake Ewauna and the Keno Impoundment, where they largely settle out of the water column (see also chlorophyll-*a* discussion in Section C.6.1.3). Data from June through November during 2000-2005 indicate that the largest relative decrease in mean TSS in the upper Klamath River occurs between Link River Dam and Keno Dam (Figure C-7, Raymond 2008), where mean values dropped from approximately 14 mg/L at Link River at Klamath Falls (RM 253.1) to near 8 mg/L at Keno Dam (RM 233.0). Values in individual years generally conform to this pattern, although year to year variability in the trend is apparent (PacifiCorp 2004a, Raymond 2008, 2009, 2010).

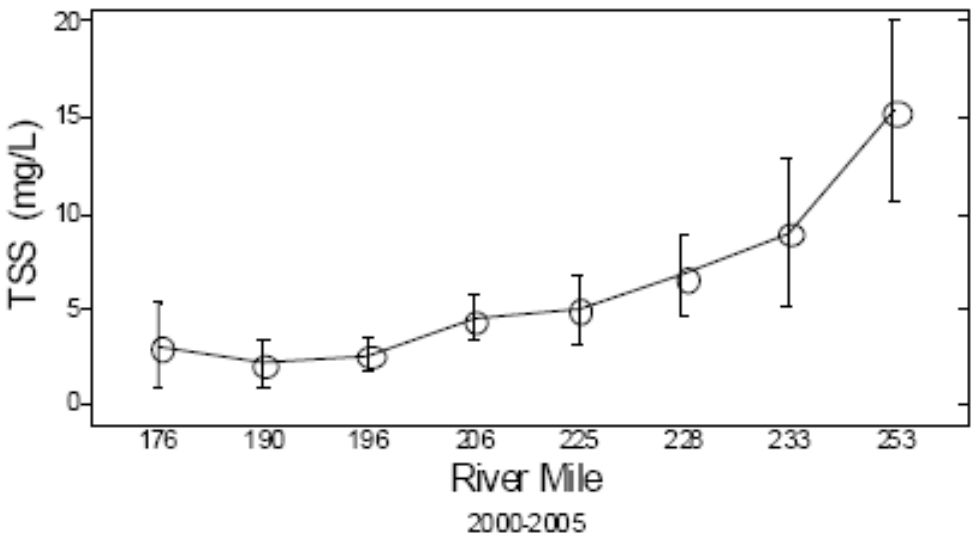

Figure C-7. Mean Total Suspended Solids (TSS) Values for Data Collected from Various Sites in the Klamath River Between 2000 and 2005. Error bars depict 90 percent confidence interval of the mean. Iron Gate Dam (RM 190.1), Copco 1 Dam (RM 198.6), J.C. Boyle Dam (RM 224.7), Keno Dam (RM 233), Link River Dam (RM 253), Source: Raymond 2008.

During summer months, peak values of TSS in this reach are associated with algal blooms from Upper Klamath Lake (PacifiCorp 2004a). Concurrent data from 2003, including chlorophyll-*a*, TSS and turbidity, indicate that elevated organic suspended sediments and turbidity levels are associated with high concentrations of algae (specifically cyanobacteria) downstream of Link River at Lake Ewauna (RM 253.1) in this reach during summer months (PacifiCorp 2004a). Samples collected at Link River mouth (RM 253.1) during July, August, and September 2003 exhibit algal (*Aphanizomenon flos-aquae*) concentrations greater than 20 cubic millimeters per liter (Kann and Asarian 2006), maximum turbidity of 22.5 NTU, and maximum TSS of 46 mg/L (PacifiCorp 2004a). Earlier and later in the summer of 2003, lower suspended sediment and turbidity levels correspond to lower algal levels in the river upstream of J.C. Boyle Dam (RM 224.7). Data from May to October 2005 indicate that suspended sediments in the Keno Impoundment (including Lake Ewauna) (RM 233-253.1) ranges from 2 to 21 mg/L, with concentrations increasing through the spring and reaching a maximum in early summer (Deas and Vaughn 2006). More recent data collected in 2009 for the Keno Impoundment at Miller Island (RM 246) indicate a summer peak in TSS of 17 mg/L (Watercourse Engineering, Inc. 2011).

C.2.1.4 Hydroelectric Reach

Moving downstream, suspended sediments generally continue to decrease through the Hydroelectric Reach. During the winter and spring (November through April), the reservoirs at the Four Facilities intercept and retain mineral (inorganic) sediments delivered from tributaries to the reservoirs (i.e., Shovel Creek, Fall Creek, Jenny Creek), where peak concentrations occur in association with high-flow events. While this may be somewhat beneficial for downstream reaches by decreasing suspended sediment concentrations and turbidity, the interception of mineral (inorganic) sediments by the reservoirs does not appear to be an important mechanism related to sediment delivery in the mainstem Klamath River. This is because a relatively small (3.4 percent) fraction of total sediment supplied to the Klamath River on an annual basis originates from the upper and middle Klamath River (i.e., from Keno Dam to the Shasta River) (Stillwater Sciences 2010) and beneficial uses in the upper Klamath River are currently not impaired due to mineral (inorganic) suspended material (see Section 3.2, Table 3.2-8).

During the growth season (May through October), algal-derived (organic) suspended materials exhibit a general downward longitudinal trend in the Hydroelectric Reach, although the relative decrease through this reach is less than that occurring further upstream in the Keno Impoundment, where algal blooms originating in Upper Klamath Lake largely settle out of the water column (see Figure C-7 and prior discussion in Section C.2.1.3). Further decreases in concentrations of algal-derived (organic) suspended materials can occur in the Hydroelectric Reach, which may be due to the mechanical breakdown of algal remains and sorting of progressively smaller sizes of natural organic matter (NOM) in the turbulent river reaches between Keno Dam and Copco 1 Reservoir, as well as by dilution from the springs downstream of J.C. Boyle Dam.

Despite the mechanisms supporting decreased longitudinal concentrations of algal-derived (organic) suspended materials in the riverine portions of the Hydroelectric Reach, concentrations in this reach can also increase due to large seasonal algal blooms occurring in Copco 1 and Iron Gate Reservoirs. TSS values in Copco 1 Reservoir during the growth season (May through October) typically range <2–20 mg/L and those in Iron Gate Reservoir range <2–14 mg/L, although intense algae blooms can result in TSS levels greater than 20 mg/L (Raymond 2008, 2009, 2010). During 2003 sampling by PacifiCorp, a particularly high TSS measurement of 280 mg/L was recorded in the epilimnion of Copco 1 Reservoir during May. Simultaneous measurements of suspended materials measured in the outflow to the reservoir indicated only 4.8 mg/L TSS (FERC 2007), suggesting that the suspended materials source (algal cells) had largely settled out of the water column within the reservoir. Since powerhouse withdrawals for Copco 1 and Iron Gate Dams are from depths of approximately 6 m (20 ft) to 12 m (39 ft) below the water surface when the reservoirs are full (Section C.1.1.4), only portions of the extensive algal blooms positioned closer to the water surface may be transported to the downstream Klamath River. During 2009 water quality monitoring, total suspended sediments measured in J.C. Boyle Reservoir ranged <2–6.8 mg/L from May through November. Levels in Copco 1 and Iron Gate reservoirs levels were somewhat greater,

with suspended sediments ranging <2–9.6 mg/L in Copco 1 Reservoir (peak in August) and <2–7.2 mg/L in Iron Gate Reservoir (peak in May) (Watercourse Engineering, Inc. 2011).

Estimates of the volume of sediment deposits stored within J.C. Boyle, Copco 1 and 2, and Iron Gate Reservoirs include 13.1 million cubic yards (yd^3) (Greimann et al. 2011), 14.5 million yd^3 (Eilers and Gubala 2003), and 20.4 million yd^3 (Gathard Engineering Consultants 2006). Sediment texture analysis results of the current reservoir deposits indicate that the deposits are composed of predominantly fine material (e.g., silt and clay <0.0625 mm [Gathard Engineering Consultants 2006]; see also Section 3.11 of this Klamath Facilities Removal EIS/EIR) with 3 to 5 percent of the accumulated material as organic carbon, corroborating interpretation of longitudinal suspended sediment patterns and indicating that in-reservoir and upstream algal growth is largely intercepted and retained in reservoir sediments in the Hydroelectric Reach.

C.2.2 Lower Klamath Basin

C.2.2.1 Iron Gate Dam to Salmon River

Immediately downstream of Iron Gate Dam (RM 190.1), mineral (inorganic) suspended materials tend to increase with distance downstream of the dam during winter months. On an annual basis, two of the three tributaries that contribute the largest amount of sediment to the Klamath River are in this reach; the Scott River (607,300 tons per year or 10 percent of the cumulative average annual delivery from the basin), and the Salmon River (320,600 tons per year or 5.5 percent of the cumulative average annual delivery from the basin) (Stillwater Sciences 2010). The Scott River enters the mainstem Klamath River at RM 143 and is listed as impaired under Section 303(d) of the CWA for sedimentation (see Section 3.2, Table 3.2-8).

During the growth season (May–October), suspended materials immediately downstream of Iron Gate Dam are relatively lower than upstream locations, with generally low (<5-8 mg/L) concentrations for 2000–2005 (PacifiCorp 2004a; Raymond 2008, 2009, 2010) (Figure C-7). Between Iron Gate Dam and Seiad Valley (RM 129.4), suspended materials can increase; for example TSS concentrations near the Shasta River confluence (RM 176) for the period 2000-2005 were roughly 1 mg/L greater than those measured further upstream at Iron Gate Dam (Figure C-7), and during 2009 monitoring, TSS ranged 0.87–4.4 mg/L downstream of Iron Gate Dam (RM 190.1), increasing to 2.5-11.5 mg/L downstream of Seiad Valley (RM 129.4) (Watercourse Engineering, Inc. 2011)[1]. This pattern may be related to the transport of some portion of the in-reservoir algal blooms to downstream reaches of Klamath River. River bed scour may also cause resuspension of previously settled materials and increases in summer and fall TSS from 0 to 20 miles downstream of the dam (Figure C-8). Further downstream, near the confluence with the Scott River (RM 143.0) concentrations of suspended materials tend to decrease with distance as suspended materials gradually settle out of the water column

[1] This data set includes measurements in November and December 2009 as well.

farther downstream or are diluted by tributary inputs (Armstrong and Ward 2008). Chlorophyll-*a* data show a similar trend (see Section C.6.2.1).

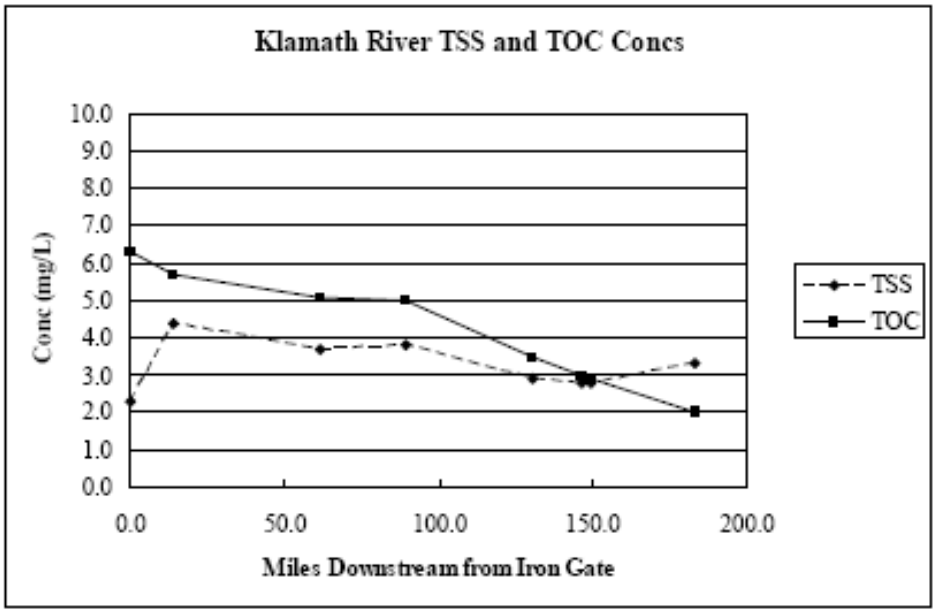

Figure C-8. Average TSS and Total Organic Carbon in the Klamath River Downstream of Iron Gate Dam during June–October 2001–2005. Source: Armstrong and Ward 2008.

C.2.2.2 Salmon River to Estuary

As in other reaches of the Klamath River, seasonal variation in turbidity and suspended materials is evident in the Klamath River from the Salmon River (RM 66.0) to the Estuary (RM 0–2), with peak summer turbidity values associated with organic matter (i.e., algae blooms) and peak spring and winter turbidity values associated with inorganic sediments that are mobilized during high flow events (Stillwater Sciences 2009). The lower Klamath River from the Trinity River (RM 42.5) to the Estuary (RM 0–2) and multiple tributaries downstream of the Trinity River are listed as impaired under Section 303(d) of the CWA for sedimentation (see Section 3.2, Table 3.2-8) (NCRWQCB 2010, State Water Resources Control Board [SWRCB] 2010a).

Historical (1950–1979) suspended sediment data for the Klamath River at Orleans (RM 59) (USGS gage no.11523000) range from less than 5 mg/L during summer (low-flow) periods to greater than 5,000 mg/L during winter (high-flow) periods, although some high (>1,000 mg/L) suspended sediment events have occurred during summer months (e.g., 1974, see Figure C-9). More recent data indicate that suspended material levels in the lower Klamath River from the Salmon River confluence (RM 66.0) to the Estuary (RM 0–2) can be similar to those measured in the upstream reach from Iron Gate Dam to the Salmon River (RM 66). Results from grab samples collected by the Yurok Tribe Environmental Program during the period 2003–2004 indicate that TSS ranged

<1.0-3.2 mg/L upstream of the Trinity River (RM 42.5) and <1.0–14.0 farther downstream at Turwar (RM 5.8), with the peak value (14.0 mg/L) occurring in December 2003 (YTEP 2005). However, the majority of the grab samples were collected from June to September and only two grab samples were collected in December and January. The data exhibit similar values for 2007, with the highest TSS (up to 16.0 mg/L) observed at Turwar in September of that year (Fetcho 2007). During 2009 monitoring, TSS values measured at Orleans were generally 1.1–13.3 mg/L between May and December, with peak values (≈56 mg/L) occurring during October (Watercourse Engineering, Inc. 2011).

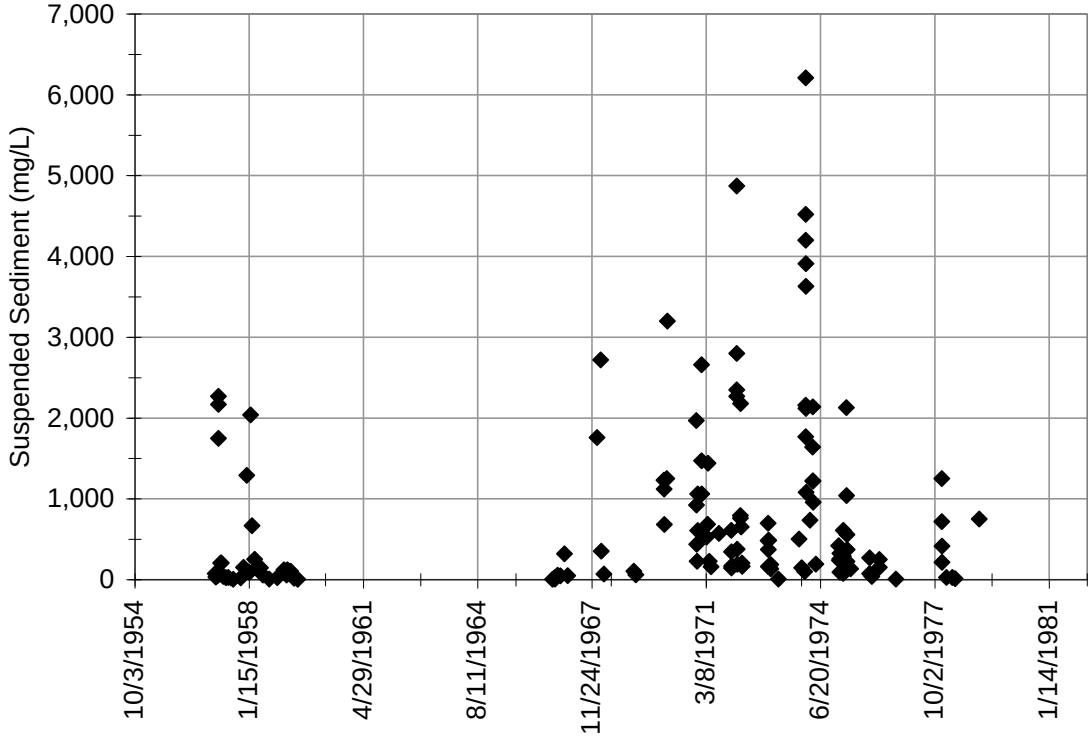

Figure C-9. Suspended Sediment (mg/L) Grab Samples for USGS Klamath River at Orleans (USGS Gage No. 11523000) (RM 59) 1950–1979. Source: USGS 2011 (http://waterdata.usgs.gov/nwis)

The Trinity River contributes 3,317,300 tons per year of sediment to the lower Klamath River or 57 percent of the cumulative average annual delivery from the basin (Stillwater Sciences 2010). Mass wasting, bank erosion, and other natural erosion processes contribute a large portion of the total fine sediment supply to the lower Klamath River, along with management activities such as timber harvest and road construction along tributaries (United States Forest Service 2004, Stillwater Sciences 2010). When combined with the steep terrain, granular soil matrix, and high precipitation, these sources may be a primary contributor to fine sediment deposits found in deep pools near cultural sites in the lower Klamath River (FERC 2007).

C.2.2.3 Klamath Estuary

Available historical (1958–1996) suspended sediment data for the Klamath River at Klamath Glen (RM 7) (USGS gage no. 11530500) indicates values of less than 5 mg/L during summer (low-flow) periods to greater than 500 mg/L during winter (high-flow) periods, although one high (>750 mg/L) suspended sediment event appears to have occurred during the early fall (i.e., October 1977, see Figure C-10).

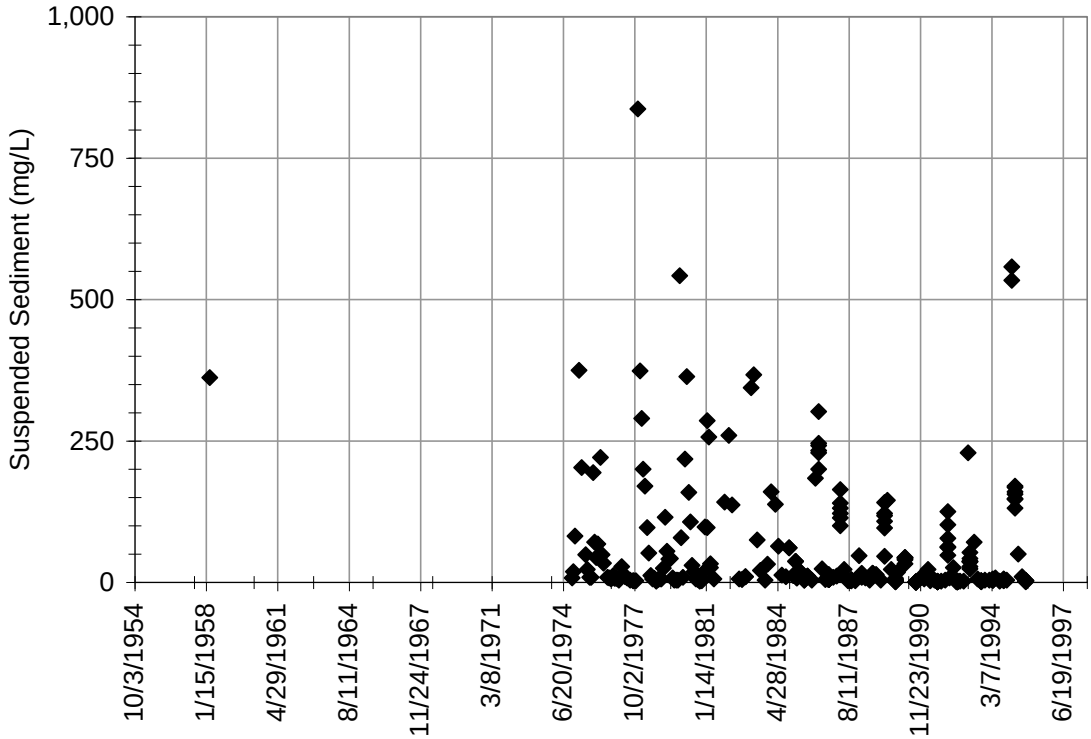

Figure C-10. Suspended Sediment (mg/L) Grab Samples for USGS Klamath River near Klamath (USGS Gage No. 11530500) (RM 7) 1958–1995. Source: USGS 2011 (http://waterdata.usgs.gov/nwis)

An analysis of more recently collected TSS data in the Klamath Estuary indicates that TSS are variable but generally similar to those measured at upstream sites in the lower Klamath River (YTEP 2005, Sinnott 2007). For 2003–2004, TSS levels were <1.0-3.2 mg/L for surface waters in the mid- and lower-estuary, and slightly greater (1.8-10.0 mg/L) at depth (YTEP 2005). During May–December 2009, measured TSS levels were generally 2.1–12.7 mg/L, with the peak value (17.9 mg/L) occurring in May (Watercourse Engineering, Inc. 2011). Turbidity measurements in small tributaries (e.g., McGarvey, Den, Blue, and Turwar Creeks) immediately upstream or within a few river miles upstream of the estuary exhibit peak values during winter high flow periods (i.e., storm events), with measured values exceeding 500 NTU during December through February 2004 (YTEP 2005). During late spring through early fall, when average rates of

precipitation in the Klamath Basin are relatively lower, inorganic (mineral) suspended sediments and turbidity in the Klamath Estuary are generally lower as well.

Algal blooms within and upstream of the estuary have the potential to cause large spikes in turbidity and organic suspended sediments in the estuary. This occurred during the extensive algal bloom detected throughout at least 40 river miles of the lower Klamath River in September 2007 (Kann 2007a–2007d). In the lower estuary, increases in nutrient levels and algae concentrations were correlated with an increase in TSS from 2.2 mg/L on August 21, 2007 to 9.0 mg/L on September 18, 2007, and increases in nutrients, algae levels, and TSS during that period were measured as far upstream as Iron Gate and Copco 1 reservoirs (Asarian et al. 2009). Thus, the observed 2007 increase in estuarine TSS appears to have been influenced by algal growth originating in the two largest reservoirs in the Hydroelectric Reach.

C.3 Nutrients

Nutrients are critical for the support of primary productivity (i.e., plant growth) in both terrestrial and aquatic ecosystems. High levels of nutrients (nitrogen and phosphorus) in lakes and rivers have the potential to impact overall water quality by increasing rates of algal growth and decay, which can lead to increased levels of turbidity, large fluctuations in dissolved oxygen and pH levels, as well as potential increases of toxic substances such as ammonia (NH_4^+/NH_3), hydrogen sulfide (H_2S), and release of heavy metals from low oxidation-reduction potential at the sediment water interface (see Section 3.2.3.1 for additional background information on water quality processes in the Klamath Basin). Dissolved nutrients (e.g., ortho-phosphorus, nitrate, and ammonium) can be used directly by algae, whereas particulate nutrients (e.g., organic phosphorus, organic nitrogen) are not readily bioavailable for most algal species.

Volcanic activity has dominated the geology of Upper Klamath basin for the past 35 million years. Consequently, relatively high levels of phosphorus are present in Upper Klamath Basin's volcanic rocks and soils. Erosion is currently understood to be the major process by which sediment-associated particulate phosphorus is delivered from the upper sub-basins of the Wood, Williamson, and Sprague Rivers to Upper Klamath Lake (ODEQ 2002). During peak flows, particulate phosphorus has been observed to increase to 60 percent of the total phosphorus (TP) load compared to less than 5 percent during summer low flows (Kann and Walker 1999). The observed seasonal increase in particulate phosphorus loading and increase in volume-weighted concentration of TP during high flows may be indicative of degraded watershed conditions (Kann and Walker 1999), where land uses including road building, forestry, grazing and agriculture have altered upland and riparian plant communities and subsequently increased contribution of phosphorus through erosion to Upper Klamath Lake (DEA 2005). Based on available information, local watershed groups have suggested that insufficient data exists to clearly demonstrate the proportion of TP loading due to natural sources and the proportion due to degraded riparian conditions and increased water yields (Connelly and Lyons 2007, Rabe

and Calonje 2009). However, research published in peer reviewed journals demonstrates that although levels of naturally occurring phosphorus are elevated in Upper Klamath Lake, historical land use activities in the Upper Klamath Basin resulted in increased nutrient loading to the lake, subsequent changes in its trophic status, and associated degradation of water quality (Bradbury et al. 2004, Eilers et al. 2004). Nitrogen sources to the lake have been identified as upland erosion, return flows from agricultural lands, and *in situ* nitrogen fixation by cyanobacteria (ODEQ 2002).

C.3.1 Upper Klamath Basin

C.3.1.1 Wood, Williamson, and Sprague Rivers

Based on the mass balance conducted for development of the Upper Klamath Lake TMDLs (ODEQ 2002), the Williamson River contributes an estimated 20.5 percent (10.8 kilograms per square meter per year [kg/km^2/yr]) of the external phosphorous load to Upper Klamath Lake, the Sprague River contributes 26.5 percent (11.5 kg/km^2/yr), and the Wood River contributes 19.1 percent (90 kg/km^2/yr) (Kann and Walker 1999). The Sprague River exhibits a high correlation between river flows and phosphorus loading, particularly during runoff events, suggesting that runoff from peak flow events delivers a significant source of suspended and particulate phosphorus to the Williamson River and subsequently, Upper Klamath Lake (ODEQ 2002). Upland contributions to the TP load are generally bound phosphorus and are associated with peak flows and suspended sediments (Gearhart et al. 1995, McCormick and Campbell 2007).

Agricultural return flows from former wetlands along the Wood River appear to contribute relatively high concentration of phosphorus to Upper Klamath Lake; upstream of former wetlands on the Wood River, the phosphorus load is approximately 64.9 kg/km^2/yr, while downstream it increases to 237 kg/km^2/yr (Kann and Walker 1999). Large increases in TP in the Wood River occur from January to June, corresponding to pumping/drainage of the surrounding inundated lands for grazing and agricultural uses, and peak seasonal runoff (ODEQ 2002).

The estimated TN load of the Williamson River, excluding loads contributed by the Sprague River, is 111 metric tons per year (MT/yr) (Walker 2001). The Sprague River TN load is estimated at 237 MT/yr (Walker 2001). Data collected during 1999–2005 in the upper Sprague River subbasin and 1991–2005 in the lower Sprague-lower Williamson river sub-basins indicate that nitrate is consistently below 0.38 mg/L, the evaluation criteria adopted for watershed assessments in Oregon (ODEQ 2006 as cited in Connelly and Lyons 2007; Klamath Tribes Natural Resource Department 2006 and USGS 2007 as cited in Rabe and Calonje 2009, Watershed Professionals Network 1999)

C.3.1.2 Upper Klamath Lake

Based on the mass balance conducted for development of the Upper Klamath Lake TMDLs, the TP budget for Upper Klamath and Agency Lakes averages 466 MT/yr; external sources supply roughly 40 percent of TP (182 MT/yr) and internal sources

supply roughly 60 percent (285 MT/yr; ODEQ 2002, Kann and Walker 1999). Identified external sources include 1) atmospheric deposition, 2) fluvial sources from tributaries, and 3) diffuse sources such as springs and marshes (ODEQ 2002). Within these external sources, springs that contribute to the base flow of tributaries to Upper Klamath Lake carry a naturally high background of soluble phosphorus ranging from 0.05 to 0.09 mg/L (Gearhart et al. 1995). Agricultural return flows from former wetlands, while contributing only 3 percent of the annual flow into the lake, account for 15 percent of external phosphorus loading (Kann and Walker 1999). The estimated median unit area contribution for agricultural return flows is 220 $kg/km^2/yr$. Former wetlands, drained and diked for agricultural purposes, contain peat soils that decompose under the aerobic conditions of the wet and dry cycles associated with agriculture and release high concentrations of phosphorus (Snyder and Morace 1997).

Seasonal changes in lake TP have been reported by Rykbost and Charlton (2001) and Kann and Walker (1999); spring runoff causes an initial seasonal increase in TP levels in Upper Klamath Lake as phosphorus bound to sediments is transported into the lake. A second increase in phosphorus occurs from June–September due to algal growth and decay cycles; algal blooms incorporate phosphorus into biomass and after a bloom crash occurs, they release soluble reactive phosphorus back into the lake. Dissolved ammonium can also be released following algal bloom crashes. Blooms of the nitrogen-fixing cyanobacteria species, *A. flos-aquae*, in Upper Klamath Lake appear to be phosphorus limited. However, water column samples collected during the annual *A. flos-aquae* bloom in April, May, and August 2006 suggest that iron limitation may play a role in primary productivity in the lake and should be further investigated. Study results suggest that dissolved iron became depleted in the lake water column during the course of 2006 seasonal algal bloom, while dissolved ammonium and soluble reactive phosphorus (SRP) increased (Kuwabara et al. 2009). However, there were no samples collected during the primary bloom period and the study did not account for the low SRP during the initial bloom growth period, suggesting that iron may play a role but it is not likely to be the primary driver for limiting algal growth in Upper Klamath Lake; the more prominent pattern is one of phosphorus limitation during bloom development (e.g., Kann 2010, Lindenberg et al. 2009).

The Total Nitrogen (TN) balance conducted for development of the Upper Klamath Lake TMDLs indicates that the lake is a seasonal source of nitrogen to Link River, with export rate estimates at 234.5 kg/km^2 (Kann and Walker 1999, ODEQ 2002). Internal lake sources of nitrogen appear to exceed external sources given available data, where the main source of increased internal nitrogen loading is nitrogen fixation by the cyanobacteria species, *A. flos-aquae* (Kann and Walker 1999). Regeneration of nitrogen from lake sediments is another identified internal source of nitrogen to Upper Klamath Lake (Kuwabara et al. 2009). Identified external sources of nitrogen to Upper Klamath Lake include tributaries, native soils, precipitation, agricultural pumps, and springs (Kann and Walker 1999, McCormick and Campbell 2007).

While both phosphorus and nitrogen concentrations have increased in Upper Klamath Lake over the past 100 years, increases in these nutrients have not necessarily occurred in

the same proportions (Eilers et al. 2004). A significant decrease in the nitrogen to phosphorus (N:P) ratio in recent sediment layers has been observed and is likely the result of either a decrease in N-fixing algae or an increase in phosphorus loading from external sources. As there is an abundant presence of N-fixing algae in the lake, it is more likely that phosphorus loading has increased over time relative to nitrogen loading (Eilers et al. 2004). The relative increase in phosphorus over nitrogen (i.e., decreasing N:P) favors N-fixing cyanobacteria such as *A. flos-aquae* , which currently contribute to heavy algal blooms in Upper Klamath Lake.

C.3.1.3 Link River Dam to Klamath River upstream of J.C. Boyle Reservoir

Historical (1950–2001) TP data indicate median values of 0.072–2.1 mg/L in the Upper Klamath Basin between Link River Dam and J.C. Boyle Reservoir, with the highest median values occurring at RM 228, which is at the upstream end of J.C. Boyle Reservoir (Figure C-11). Variability over the long-term record in this reach is high, with multiple outlying data points above and below the 95th percentile and indicating TP levels greater than 3 mg/L at multiple sites in the reach. The historical record indicates less overall variability in orthophosphate concentrations in the reach, but still some relatively high concentrations (≈3 mg/L) occurring just downstream of Link River Dam (RM 253) (PacifiCorp 2004b). A review of water quality data near Klamath Straits Drain (RM 240.5) from 1991–1999 also documented elevated concentrations of total phosphorus, with mean concentrations across the monitoring period of 0.41 mg/L at Stateline Highway (Lytle 2000).

Phosphorus data collected in a more recent study from April to November 2007 from just downstream of Link River Dam (RM 253.2) to Keno Dam (RM 234.9) range from 0.05 to 0.50 mg/L for TP and from 0.01 to 0.27 mg/L for filtered orthophosphate (Sullivan et al. 2008); both nutrient concentrations increase from spring to summer and decrease into fall months (Sullivan et al. 2008, Deas and Vaughn 2006). During the 2007 study, orthophosphate concentrations appeared to increase in a downstream direction from the Link River Dam to the downstream end of the Keno Impoundment (Deas and Vaughn 2006). In a recent study of nutrient dynamics in the Klamath River, May through November TP and orthophosphate (reported as SRP) for 2005–2008 follow a decreasing longitudinal pattern, with the highest concentrations (approximately 0.1-0.5 mg/L) measured in the Klamath River downstream of Keno Dam (RM 228–233) (Asarian et al. 2010, Watercourse Engineering, Inc. 2011). Downstream of Keno Dam, orthophosphate concentrations are highly variable, and there appears to be substantial conversion of phosphorus from particulate to dissolved forms in the turbulent section of river between Keno Dam and J.C. Boyle Reservoir (Asarian et al. 2010, Deas 2008). For April to November 2007, peak TP concentrations in this reach tend to occur between July and September (variable by year), declining through the remainder of the fall months (Asarian et al. 2010). A recent study on nutrient cycling the Lower Klamath National Wildlife Refuge indicates that refuge wetland management is simultaneously reducing nutrient loads and increasing the proportion of bioavailable P in wetland outflows, which then enter the Klamath River through the Klamath Straits Drain (RM 240.5) (Mayer 2005).

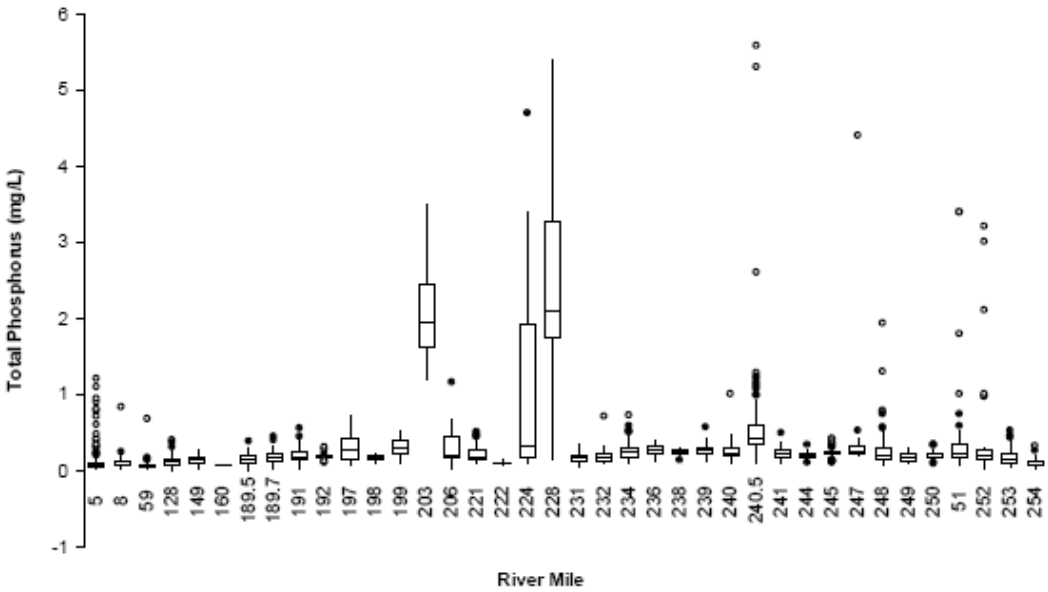

Figure C-11. Box and Whisker Plot of Historical TP Data Collected from Various Sites in the Klamath River from Klamath River at Klamath Glen (RM 5) to Klamath River at Link River Dam (RM 254) Between 1950 and 2001. Source: PacifiCorp 2004b.

Historical (1950–2001) nitrogen data is available as total Kjeldahl nitrogen (TKN; a measure of organic nitrogen plus ammonia), nitrate (NO_3^-) and ammonia (NH_4^+). TKN median values were 1.3–3 mg/L in the Upper Klamath Basin between Link River Dam and J.C. Boyle Reservoir, with the highest median values occurring at RM 240.5, which is just downstream of the Klamath Straits Drain (PacifiCorp 2004b). Variability over the long-term record in this reach is relatively high, with multiple outlying data points above the 95th percentile and indicating TKN levels greater than 5 mg/L at multiple sites in the reach. The historical record indicates similarly high variability in nitrate and ammonia concentrations in the reach, with some relatively high concentrations (>5 mg/L) occurring throughout the reach (PacifiCorp 2004b).

For the period April to November 2007, TN concentrations in the mainstem Klamath River collected from just downstream of Link River Dam (RM 253.2) to near Keno Dam (RM 234.9) range from 0.70 to 5.85 mg/L (Sullivan et al. 2008) with concentrations generally increasing from spring to summer (Sullivan et al. 2008, Deas and Vaughn 2006). Particulate nitrogen (presumably associated with organic particulate matter) concentrations range from 0.08 to 3.93 mg/L, also increasing from spring to summer and generally decreasing in a downstream direction (Sullivan et al. 2008). Average ammonia concentrations measured during April through November increase in a downstream direction (Sullivan et al. 2008, Deas and Vaughn 2006) from 0.089 mg/L at Link River, to 0.413 mg/L at Miller Island (RM 246) and 0.56 mg/L upstream of Keno Dam (RM 234.9, Sullivan et al. 2008). Dissolved nitrite plus nitrate concentrations appear to follow a different seasonal cycle than most nutrients in the Klamath River, where concentrations

are below the reporting level (0.06 mg/L) from late June through mid-August 2007, but are detectable on a site-specific basis in the spring, late summer and fall (Sullivan et al. 2008). In the Asarian et al. (2010) study of nutrient dynamics in the Klamath River, data collected from May 2005 to December 2008 indicate that the highest TN values (approximately 1–4 mg/L) are found downstream of Keno Dam (RM 233) as compared with sites farther downstream (a similar pattern was observed in the 2009 dataset [Watercourse Engineering, Inc. 2011]). For the 2005–2008 study period, peak TN and organic nitrogen concentrations in this reach tend to occur in September, with an additional July peak during 2008, and declining concentrations through the remainder of the fall months (Asarian et al. 2010), corresponding to general temporal trends in chlorophyll-*a* and blooms of cyanobacteria.

Total inorganic nitrogen (nitrate, nitrite, ammonia) concentrations follow a different trend, tending to be relatively low in May–June (<0.2 mg/L), increasing through July, and remaining high (1–1.1 mg/L) August through November (Asarian et al. 2010). Data collected during 2000 to 2002 indicates that 34 percent of ammonia samples in the Klamath Hydroelectric Project (KHP) reach, many of them from the Keno Impoundment (including Lake Ewauna) (i.e., 64 of 178 exceedances), exceeded levels that would be acutely toxic to fish (FERC 2007). Accordingly, the Klamath River in Oregon from RM 231.5 (just upstream of Keno Dam) to the Link River Dam (RM 253) is listed as impaired under Section 303(d) of the CWA for ammonia (see Section 3.2, Table 3.2-8). Ammonia levels just downstream of Keno Dam exceeded 1.0 mg N/L during at least one summer month each year from 2005 to 2008, and exceeded 1.5 mg N/L during August 2005 and 2008 (Asarian et al. 2010).

C.3.1.4 Hydroelectric Reach

The Klamath River from the California-Oregon state line to Iron Gate Dam (including Copco Lake Reservoir [1 and 2] and Iron Gate Reservoir) is listed as impaired under Section 303(d) of the CWA for nutrients (see Section 3.2, Table 3.2-8). Historical and contemporary nutrient data indicate that on an annual basis nutrients in the Hydroelectric Reach tend to be lower than those in the reach from Link River Dam to upstream of J.C. Boyle Reservoir (Figure C-11 through C-13), due to dilution from springs downstream of J.C. Boyle Reservoir and the settling of particulate matter and associated nutrients in the larger KHP reservoirs (PacifiCorp 2004a; FERC 2007, Butcher 2008, Asarian et al. 2009), while on a seasonal basis the TN and TP can increase in this reach due to release of nutrients to the water column during periods of seasonal hypolimnetic anoxia (Kann and Asarian 2006; Asarian and Kann 2006a, 2006b; Butcher 2008; Asarian et al. 2009, et al. 2010). In J.C. Boyle Reservoir (RM 224.7), the furthest upstream reservoir in this reach, concentrations of TN and TP measured between the inflow and outflow are typically similar, likely due to the shallow depth and short residence time characteristic of this impoundment (PacifiCorp 2006), indicating that relatively little nutrient retention occurs in this reservoir. Downstream of J.C. Boyle Dam, TN and TP concentrations generally decrease with distance, with both mean longitudinal concentrations (Raymond 2008, 2009, 2010; Watercourse Engineering, Inc. 2011) and

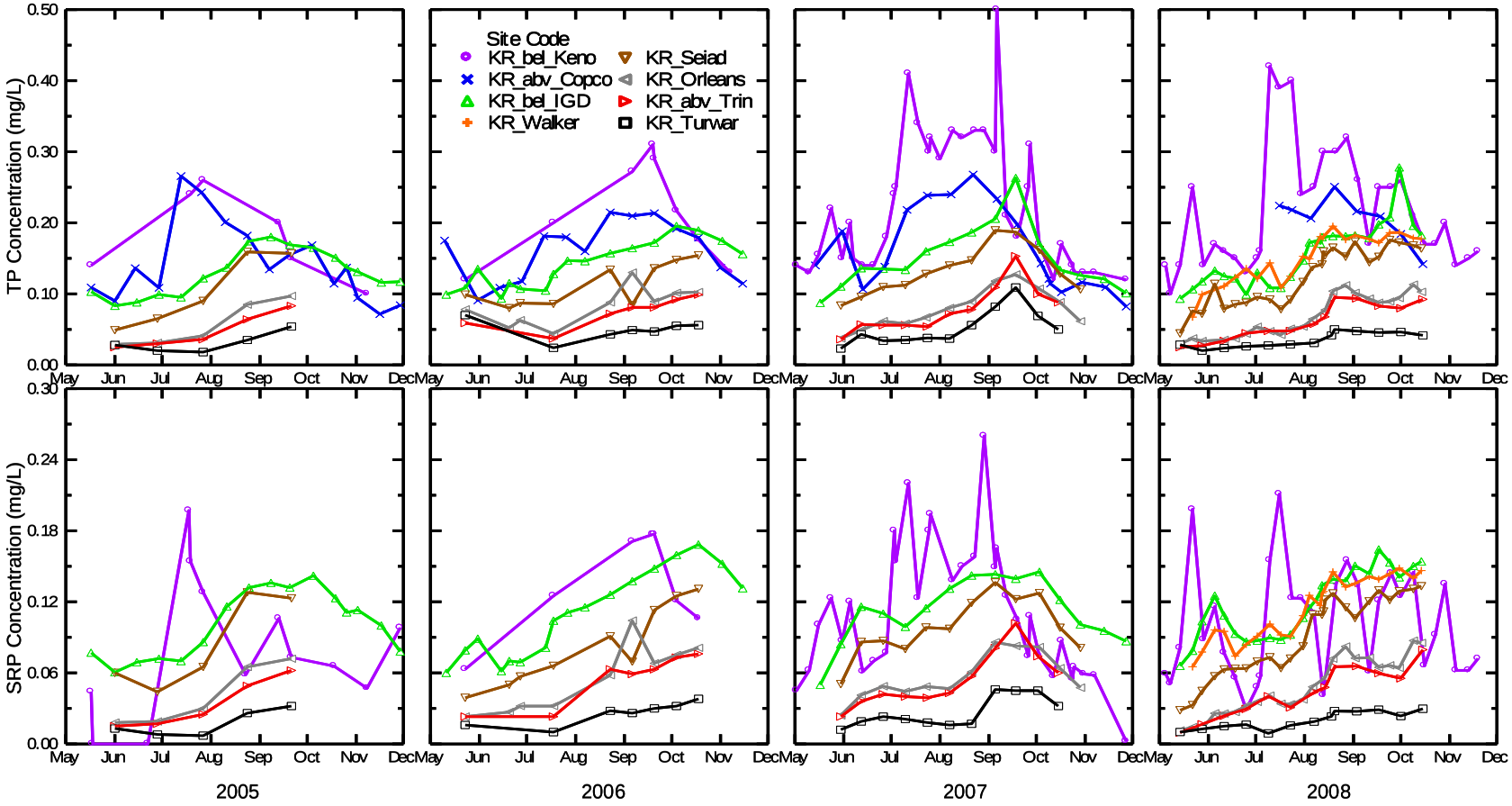

Figure C-12. Time Series of Total Phosphorus (TP) and Soluble Reactive Phosphorus (SRP) Concentrations for Selected Mainstem Klamath Sites from Downstream of Keno Dam (≈RM 233) to Turwar (RM 5.8), May 2005–November 2008. Source: Asarian et al. 2010.

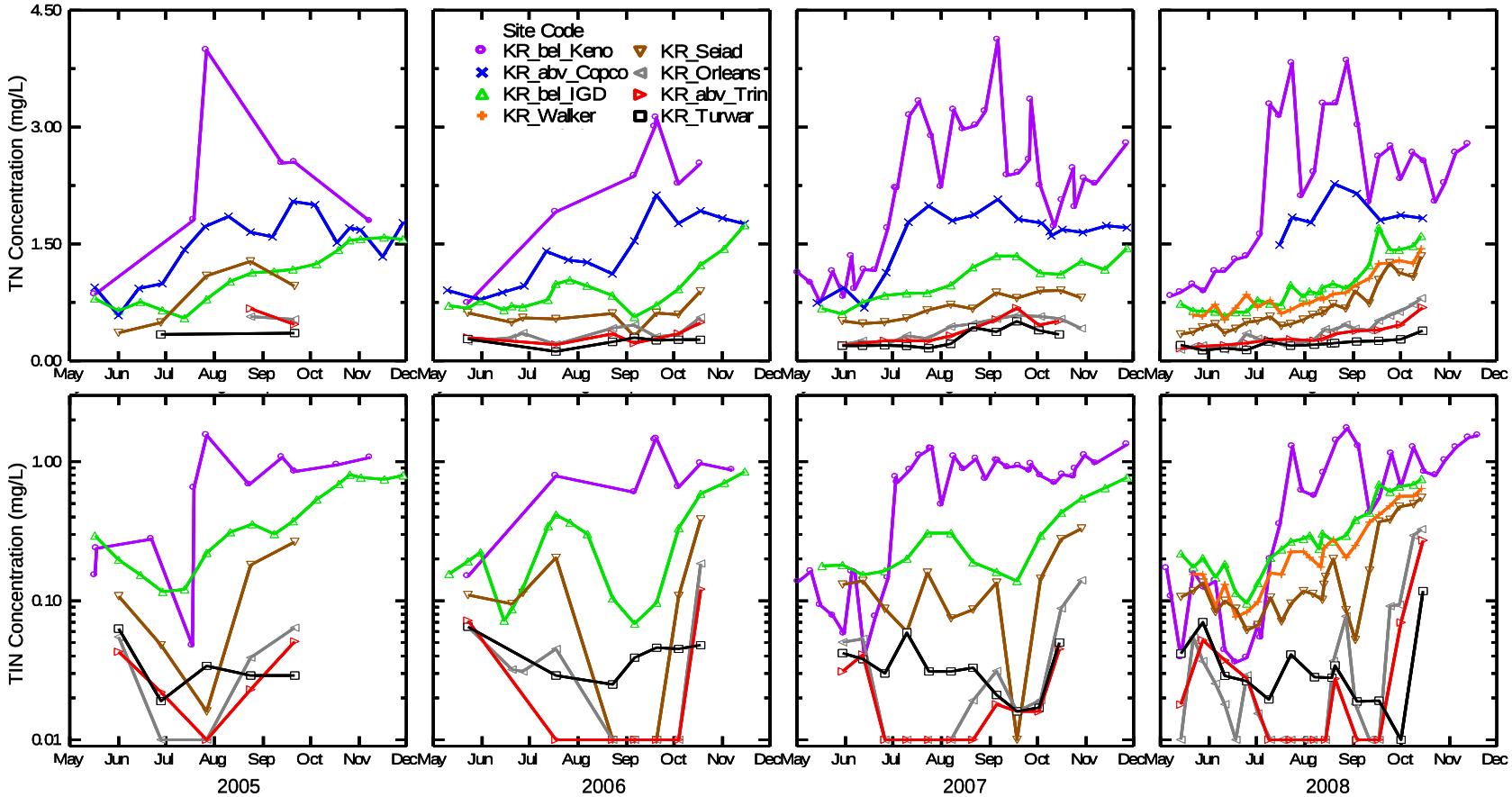

Figure C-13. Time Series of Total Nitrogen (TN) and Total Inorganic Nitrogen (TIN) Concentrations for Selected Mainstem Klamath Sites from downstream of Keno Dam (≈RM 233) to Turwar (RM 5.8), May 2005–November 2008. Source: Asarian et al. 2010.

flow-weighted longitudinal concentrations trending strongly downward through Copco 1 and Iron Gate Reservoirs, particularly for TN (see Figure C-14 and C-15 for flow-weighted concentration data; Asarian et al. 2009, et al. 2010). A frequent and notable exception occurs during August–November, when TP concentrations are often higher at Iron Gate Dam than they are at Keno Dam and upstream of Copco 1 Reservoir; this is likely due to the combination of internally-driven nutrient dynamics related to algal bloom crashes in Copco 1 and Iron Gate reservoirs and a 1–2 month temporal lag due to the longer hydraulic retention time of the reservoirs as compared to free-flowing river reaches (Kann and Asarian 2007, Asarian et al. 2009, et al. 2010, Watercourse Engineering, Inc. 2011).

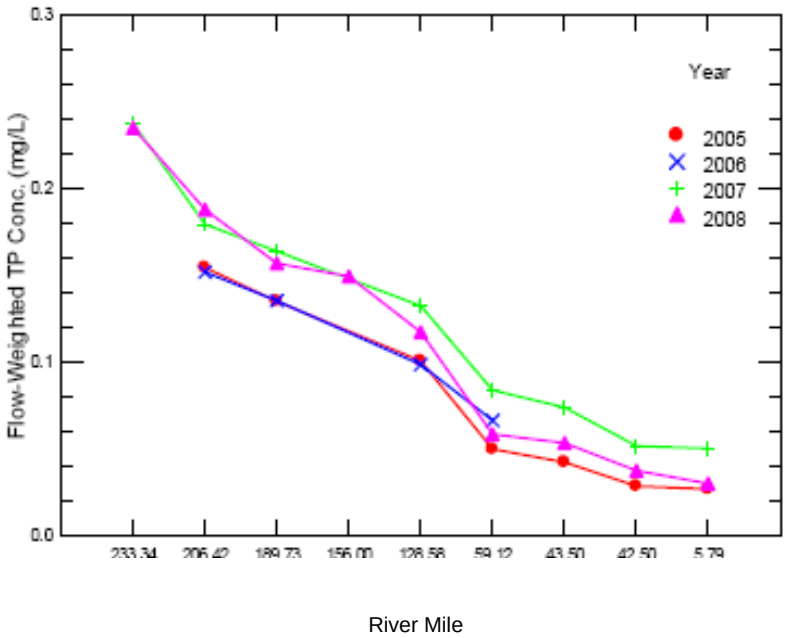

Figure C-14. Summary of Flow-Weighted Mean Concentration (mg/L) for TP at Mainstem Klamath River Sites from downstream of Keno Dam (≈RM 233) to Turwar (RM 5.8) for the Months of June–October (2005–2008). Source: Asarian et al. 2010.

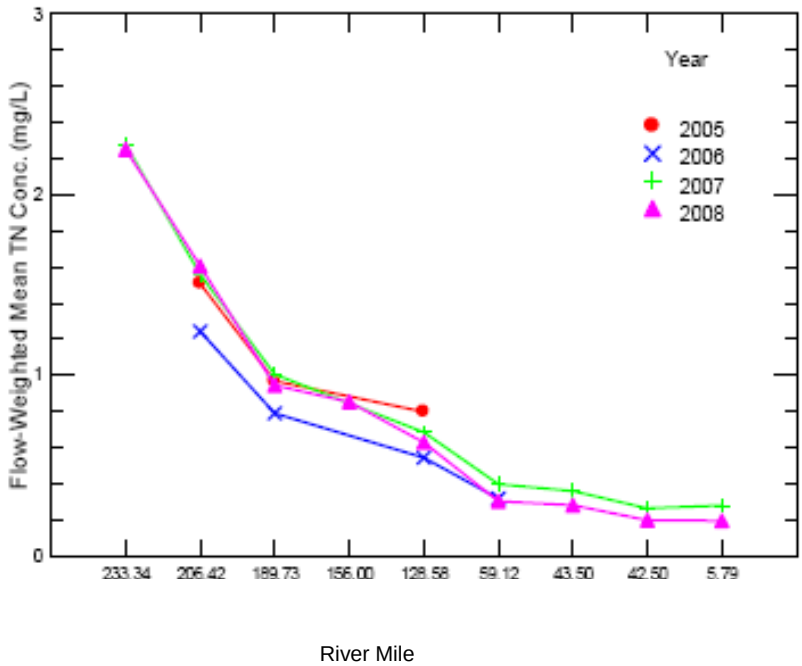

Figure C-15. Summary of Flow-Weighted Mean Concentration (mg/L) for TN at Mainstem Klamath River Sites from Downstream of Keno Dam (≈RM 233) to Turwar (RM 5.8) for the Months of June–October (2005–2008). Source: Asarian et al. 2010.

While annual data from 2000 through 2004 and early modeling studies by PacifiCorp conducted for the FERC relicensing process indicates that Copco 1 and Iron Gate Reservoirs act primarily as TN and TP sinks due to trapping of algal detritus (PacifiCorp 2004a, FERC 2007), subsequent analyses found that while overall annual retention is likely occurring, the KHP reservoirs can also serve as seasonal sources of TN and TP (though far less for TN than for TP) through the release of nutrients from reservoir sediments into the water column during periods of algal decomposition and seasonal hypolimnetic anoxia, and possibly through direct nitrogen fixation from the atmosphere by cyanobacteria (Kann and Asarian 2005, 2006; Asarian and Kann 2006a, 2006b; Butcher 2008; Asarian et al. 2009, et al. 2010). Data presented in Asarian et al. (2009) suggest that much of the TP released from Copco 1 and Iron Gate reservoir sediments during summertime anoxia remains in the hypolimnion until the reservoirs begin to turn over in the fall, rather than being released to downstream river reaches during the summer period of peak periphyton growth. However, in many years TP concentrations during August through October have been observed to be higher downstream of Iron Gate Dam than upstream of Copco 1 Reservoir corresponding to peak in-reservoir algal blooms and indicating that some release of TP can occur at times that downstream periphyton growth downstream may be stimulated.

With respect to nutrient speciation, internally-driven reservoir nutrient dynamics due to stratification patterns and hydraulic residence time in Iron Gate and Copco 1 Reservoirs appear to influence ortho-phosphorus and, to a lesser degree, ammonium concentrations within the Hydroelectric Reach. Soluble reactive phosphorus (surrogate measure for orthophosphate) concentrations in the riverine portions of the Hydroelectric Reach generally follow a decreasing longitudinal trend through this reach for summer and fall months (i.e., May through November; see Figure C-12); however, concentrations in Iron Gate Reservoir can exceed those of upstream sites (i.e., Klamath River downstream of Keno Impoundment, Copco 1 Reservoir) particularly between September and November (Asarian et al. 2009, et al. 2010; Raymond 2009, 2010). Concentrations of orthophosphate are generally constant throughout the water column in winter months when the reservoirs are mixed, while in stratified periods (i.e., May–October/November) vertical concentration gradients develop in Copco 1 and Iron Gate Reservoirs (Raymond 2008, 2009, 2010). For example, concentrations near the bottom of Copco 1 Reservoir reached 1.4 mg/L in September and October of 2008 and 2009, while in Iron Gate Reservoir they remained below 0.5 mg/L throughout the water column.

Data from 2001–2008 indicate that nitrate concentrations often peak in the vicinity of J.C. Boyle Reservoir and decrease through the remainder of the Hydroelectric Reach (Raymond 2009). On a seasonal basis, coupled nutrient and algal data indicate that nitrate levels decrease during phytoplankton blooms in the Hydroelectric Reach; cyanobacteria blooms were recorded in Iron Gate and Copco 1 Reservoirs in summer and fall 2005 coincident with a nitrate decrease of up to 0.8 mg/L between the inflow to Copco 1 and the outflow of Iron Gate Reservoirs (Kann and Asarian 2007). Dilution from the springs downstream of J.C. Boyle Dam also reduces nitrate concentrations in this reach.

Relatively high levels of ammonia have been recorded in the Hydroelectric Reach. While available data collected to date suggests no actual ammonia toxicity events associated with the operation of the Four Facilities (NCRWQCB 2010), elevated ammonia levels in the deeper portions of the hypolimnion of both Copco 1 and Iron Gate Reservoirs in summer of 2005 exceeded 0.6 mg/L (Figures 12 and 14 in Kann and Asarian 2007), indicating that anoxic conditions are likely causing conversion of organic nitrogen in reservoir deposits to ammonia and introducing the potential for episodic in-reservoir toxicity events depending upon reservoir mixing conditions. From 2001 to 2004, June and November mean ammonia concentrations in Iron Gate Reservoir were 0.1–0.2 mg/L to a depth of 45 meters, whereas Copco 1 Reservoir concentrations were consistently higher for the 20–32 meter depth and were 0.9–1.0 mg/L in September and October (FERC 2007). Only minor increases in ammonia (0.05−0.1 mg/L) have been observed to occur in Copco 1 and Iron Gate Reservoirs, most often during October and November (Kann and Asarian 2005, 2007).

C.3.2 Lower Klamath Basin

C.3.2.1 Iron Gate Dam to Salmon River

Historical (1950–2001) TP data indicate median values of 0.11–0.19 mg/L in the Lower Klamath Basin between Iron Gate Dam and Seiad Valley, with the highest values occurring just downstream of the dam (Figure C-11). Variability over the long-term record in this reach is lower than upstream reaches, with concentrations varying from near zero to over 0.3 mg/L for the period of record (Figure C-11). The historical record indicates relatively low variability in orthophosphate concentrations in the reach (as compared with variability in the Upper Klamath Basin), with median values of 0.03–0.1 mg/L (PacifiCorp 2004b).

More recent data indicate that phosphorus dynamics in the Klamath River immediately downstream of Iron Gate Dam are affected by conditions within the Project reservoirs (Section C.3.1.4). During May 2005–November 2008, peak TP concentrations at locations downstream of Iron Gate Dam tended to occur between mid-August and early October, which is roughly 1 to 2 months later than peak timing in upstream reaches and may be due to the hydraulic residence time in Iron Gate and Copco Reservoirs or release of TP from anoxic sediments during summer stratification, or following algal bloom and death (Figure C-12). Highest TP concentrations were approximately 0.25 mg/L downstream of Iron Gate Dam in 2007 and 2008, with a steep late-fall decline in TP in some years and a more gradual decline in others (Figure C-12). Orthophosphate tends to decrease in the mainstem Klamath River with distance downstream of Iron Gate Dam (FERC 2007, Asarian et al. 2010). Seasonal trends in orthophosphate closely follow observed TP concentrations and for the period 2005–2008 this phosphorus species regularly accounts for 60 to 90 percent of TP sampled (Asarian et al. 2010).

For the period 1996–2007, average TN concentrations downstream of Iron Gate Dam (RM 190.1), vary by year, with mean concentrations of 1.2 mg/L (FERC 2007) and a range of measured concentrations from <0.1 to over 2.0 mg/L (NCRWQCB 2010). Additional historical (1951–2001) nitrogen data is available as TKN (a measure of organic nitrogen plus ammonia). TKN median values for this period were 0.6–0.9 mg/L in the Lower Klamath Basin between Iron Gate Dam and Seiad Valley, with the highest median values occurring just downstream of the dam (PacifiCorp 2004b). Variability over the long-term record in this reach is relatively low compared with that of upstream reaches. For 1951–2001, high variability in nitrate concentrations is apparent in the reach between Iron Gate Dam and the Salmon River confluence, with some relatively high concentrations (>5 mg/L) occurring at the Seiad Valley location (RM 129.4) (PacifiCorp 2004b).

More recent data indicate that nitrogen dynamics in the Klamath River immediately downstream of Iron Gate Dam are also affected by conditions within the KHP reservoirs (Section C.3.1.4). Based on data collected May 2005–November 2008, TN concentrations in the river downstream of Iron Gate Dam are generally lower than those in upstream reaches (Figure C-13) due to dilution from the springs downstream of J.C.

Boyle Dam and reservoir retention in the Klamath Hydropower Reach (Asarian et al. 2009). Further decreases in TN occur in the mainstem river downstream of Iron Gate Dam due to a combination of tributary dilution and in-river nitrogen removal processes such as denitrification and/or storage related to biomass uptake (Asarian et al. 2010). On a seasonal basis, TN increases from May through November, with peak concentrations (1−1.5 mg/L) typically observed during September and October (Figure C-13). Previous analysis of the 2001−2004 dataset also indicated that median nutrient concentrations in the Klamath River from Iron Gate Dam to (RM 190.1) to Seiad Valley (RM 129.4) exceeded 0.2 mg/L (Asarian and Kann 2006b). Downstream of Seiad Valley, median TN concentrations were often at or only slightly greater than 0.2 mg/L TN (Asarian and Kann 2006b).

Ratios of TN to TP (TN:TP) measured in the Klamath River suggest that the system is generally N-limited with some periods of co-limitation by N and P; however, concentrations of both nutrients are high enough in the river from Iron Gate Dam (RM 190.1) to approximately Seiad Valley (RM 129.4) (and potentially further downstream) that nutrients are not likely to be limiting primary productivity (i.e., periphyton growth) in this portion of the Klamath River (FERC 2007, Hoopa Valley Tribal Environmental Protection Agency [HVTEPA] 2008, Asarian et al. 2010). In addition, N-fixing species dominate the periphyton communities in the lower reaches of the Klamath River where inorganic nitrogen concentrations are low (Asarian et al. 2010). Since these species can fix their own nitrogen from the atmosphere, nitrogen would not limit their growth.

Based on data collected during 2005–2008, nitrate concentrations also tend to decrease longitudinally in the Klamath River downstream of Iron Gate Dam (Asarian et al. 2010). Although patterns in nutrient concentrations vary between years, nitrate typically increases during August and September, with measured concentrations downstream of Iron Gate Dam greater than 0.25 mg/L each year from 2005 through 2008 (Asarian et al. 2010). In the fall, nitrate concentrations tend to increase again, reaching values of over 0.6 mg/L (Asarian et al. 2010). Mean 2000–2004 nitrate concentrations downstream of Iron Gate Dam were 0.15–0.44 mg/L between March and November, with the highest concentrations observed in early September (FERC 2007). Over the same time period, mean nitrate concentrations farther downstream near the confluence of the Shasta River (RM 176.7) had decreased to 0.02–0.36 mg/L, with peaks observed in early November (FERC 2007). Nitrate generally comprises less than 40 percent of the TN concentration throughout the lower Klamath River (Asarian et al. 2010).

As a result of the seasonal production of ammonia in anoxic hypolimnetic waters of the Project reservoirs (Section C.3.1.4) and the high pH levels (>7.5 pH units) measured seasonally downstream of Iron Gate Dam (YTEP 2005; NCRWQCB 2006), the NCRWQCB evaluated all available sampling data records as part of Klamath River TMDL development. The NCRWQCB analysis showed that for sampling events in which all three parameters (pH, ammonia, and water temperature) were collected simultaneously, no acute or chronic toxicity exceedances of the North Coast Basin Plan criteria for ammonia were indicated (NCRWQCB 2010). For the May–November sampling period in 2005–2008, ammonia concentrations in the Klamath River

downstream of Iron Gate Dam were generally <0.3 mg/L and constituted less than ten percent of the TN concentration (Asarian et al. 2010). Highest concentrations were measured during fall months downstream of Iron Gate Dam (RM 190.1), with late-fall ammonia concentrations generally increasing at this location and values increasing to above 0.2 mg/L during November 2006. For 2000−2004, mean ammonia levels of 0.13 mg/L were reported in Iron Gate Dam outflow (FERC 2007).

Although tributary dilution generally has a proportionally greater effect on nutrient concentration reductions in the Klamath River downstream of Iron Gate Dam, nutrient retention is an important component of overall nutrient dynamics in this reach. In a study of the June–October and July–September periods during 2005–2008, nutrient retention in the reach from Iron Gate Dam to the Klamath River Estuary was calculated after accounting for tributary dilution (Asarian et al. 2010). For the study, positive retention values represented seasonal removal of nutrients from the water column through storage in algae/plant biomass or denitrification, and negative retention represented an internal source of nutrients from sediment release or algal regeneration and nitrogen fixation. Retention rates downstream of Iron Gate Dam were variable but generally positive for TP, although negative retention was observed during some years in the reach between Seiad Valley (RM 129.4) and the Salmon River (RM 66), as well as further downstream to Turwar (RM 5.8) (Asarian et al. 2010). In general, TP and orthophosphate retention increased with distance downstream of Iron Gate Dam while particulate phosphorus retention decreased (or became more negative). Nutrient retention for TN was similarly positive, with instances of negative retention observed during 2005 between Iron Gate Dam (RM 190.1) and Seiad Valley (RM 129.4) (see Section C.3.2.2 for discussion of retention in lower reaches). Additionally, during 2005–2008, total inorganic nitrogen (TIN = nitrite + nitrate + ammonium) retention was consistently positive between Iron Gate Dam and as far downstream as the Trinity River confluence (RM 42.5). The Asarian et al. (2010) analysis indicates that large quantities of nitrogen and phosphorus were retained in the river across the roughly 130 miles from Iron Gate Dam to just downstream of the Salmon River at Orleans (RM 59.0): during July-September of 2007–2008, the incoming nutrient load at Iron Gate Dam was reduced by 24% for TP, 25% for ortho-phosphorus, 21% for particulate phosphorus, 41% for TN, 93% for TIN, and 21% for organic nitrogen.

C.3.2.2 Salmon River to Estuary

Downstream of the confluence with the Salmon River, nutrient concentrations continue to decrease in the Klamath River as compared with those measured farther upstream. Historical (1950–2001) TP data indicate median values of 0.06–0.07 mg/L in river between the Salmon River confluence and near the Klamath Estuary, with generally low variability (Figure C-11). Orthophosphate levels and variability over the long-term record in the reach downstream of the Salmon River are similar to those in the reach downstream of Iron Gate Dam (see previous section). Contemporary data (2005–2008) indicate that TP concentrations in this reach are generally 0.05–0.1 mg/L with peak values occurring in September and October. Downstream of the Trinity River,

orthophosphate often accounts for less than 50 percent of TP, possibly due to dilution from the Trinity River (Asarian et al. 2010).

As with upstream reaches, historical (1951–2001) nitrogen data is available as TKN, nitrate, and ammonia. TKN median values downstream of the Salmon River for this period were 0.25–0.3 mg/L (PacifiCorp 2004b). Variability over the long-term record in this reach is dependent on sampling location, with the greatest variability for the most downstream site at Klamath Glen (RM 5). For 1951–2001, high variability in nitrate concentrations is apparent throughout this reach, with some relatively high concentrations (>3 mg/L) occurring at Orleans (RM 59) and Klamath Glen (RM 5) (PacifiCorp 2004b). Contemporary data indicate that on a seasonal basis, TN increases from May through November, with peak concentrations (<0.5 mg/L) typically observed during September and October (Figure C-13), which are at or above the Hoopa Valley Tribe numeric criterion of 0.2 mg/L TN (see Section 3.2, Table 3.2-6). Downstream of the Trinity River confluence (RM 42.5), TN concentrations are typically less than 0.5 mg/L (YTEP 2005), with general increases from spring to fall months. For the 2005–2008 dataset, TN increases were observed between September and October at Orleans (RM 59), upstream of the confluence with the Trinity River (≈RM 66), and at Turwar (RM 5) (Asarian et al. 2010) (Figure C-13).

Nutrient retention rates in the Klamath River from approximately the Salmon River confluence to the Trinity River are variable for the period 2005–2008, but generally positive for TN and TP. However, from the Trinity River to the Klamath River estuary, nutrient retention rates are generally negative (Asarian et al. 2010). For example, during 2005–2008, total inorganic nitrogen (TIN = nitrite + nitrate + ammonium) retention was consistently negative between the Trinity River confluence and Turwar (RM 5.8) (Asarian et al. 2010). The Asarian et al. (2010) analysis suggests that while nitrogen and phosphorus are largely being removed from the river upstream of the Trinity River confluence (RM 42.5) during the June–October and July–September study periods, downstream of the confluence, nutrients are being added.

C.3.2.3 Klamath Estuary

Nutrient levels in the Klamath River Estuary are highly variable spatially and temporally and are greatly influenced by season, river flow, tidal prism, and location of the estuary mouth. In general, nutrient levels in the Klamath River Estuary are lower than in the Klamath River just upstream of the Trinity River confluence (RM 43.5) and comparable to the nearest river sampling station (RM 5) near Turwar (Sinnott 2007, 2008, 2009, 2010, 2011a). Inter-annual and seasonal variability are apparent in the contemporary data collected by the Yurok Tribe during 2006–2010 (Sinnott 2007, 2008, 2009, 2010, 2011a). For example, measured levels of TP in the Klamath River Estuary are below 0.12 mg/L during the period June–October 2006–2010. Contemporary data (2006–2010) indicate that TP concentrations in the Klamath River Estuary generally range from 0.020–0.100 mg/L with peak values generally occurring in September and October, although 2009 and 2010 data indicated that concentrations of TP can continue to increase into November and December, especially during elevated river flows (Sinnott 2007, 2008, 2009, 2010,

2011a). During peak concentrations, values often exceed the Hoopa Valley Tribe's standard of 0.035 mg/L TP (see Section 3.2, Table 3.2-6). During the same period, orthophosphate is consistently reported at less than 0.05 mg/L. Orthophosphate often accounts for more than 50% of TP from June through October.

Contemporary data (2006–2010) indicate that TN concentrations in the Klamath River Estuary were consistently below 0.7 mg/L, generally ranging from 0.1–0.5 mg/L (Sinnott 2007, 2008, 2009, 2010, 2011a). Concentrations increase from June to October with peak values occurring in September and October, although 2009 and 2010 data indicate that concentrations can continue to increase into November and December, especially during high river flows (Sinnott 2007, 2008, 2009, 2010, 2011a). During peak concentrations, values often exceed the Hoopa Valley Tribe's standard of 0.2 mg/L TN (see Section 3.2, Table 3.2-6). During June–October 2006-2010, measured values of nitrate plus nitrite in the Klamath River Estuary are near or below the reporting limit (0.01 mg/L), with concentrations generally ranging from 0.01 mg/L to 0.05 mg/L. Concentrations of nitrate plus nitrite in the Klamath River Estuary increase from June to October, with peak values during this period occurring in September and October. As with TN, recent data indicates that nitrate plus nitrite concentrations can continue to increase into November and December, especially during elevated river flows (Sinnott 2010, Sinnott 2011a). Measured values of ammonia in the Klamath River Estuary were low, with measurements consistently below 0.1 mg/L during the period June-October 2006-2010, generally ranging from 0.01 mg/L to 0.03 mg/L, with peak values generally occurring in September. Many ammonia samples from the Klamath River Estuary return values near or below the reporting limit of 0.01 mg/L (Sinnott 2007, 2008, 2009, 2010, 2011a). However, the Klamath River Estuary sampling site has more detectable concentrations of ammonia than any other sampling site within the Yurok Reservation. Nutrient retention has not been explicitly measured in the estuary, although measurements have been made just upstream of the estuary in the reach from the Trinity River confluence (RM 43.5) to Turwar (5.8).

C.4 Dissolved Oxygen

C.4.1 Upper Klamath Basin

C.4.1.1 Wood, Williamson, and Sprague Rivers

Limited dissolved oxygen data were collected in the Wood River in 1991, with reported values of 9.8−12.7 mg/L at headwaters and 8.9−10.8 mg/L at the mouth (Kann 1993). Historical dissolved oxygen data for the Williamson and Sprague rivers are not generally available. More recent data collected by ODEQ (2002) for the Sprague River indicates that dissolved oxygen concentrations in this tributary to Upper Klamath Lake undergo large daily cycles due to algal photosynthesis and respiration causing dissolved oxygen supersaturation (>10 mg/L) during the day and depressed (<7 mg/L) levels at night.

Critically low dissolved oxygen conditions occur during summer months in the Sprague River, where slower water column velocities and elevated water temperatures encourage excessive periphyton (i.e., benthic or attached algae) growth.

C.4.1.2 Upper Klamath Lake

Dissolved oxygen concentrations in Upper Klamath Lake range from less than 4 mg/L to greater than 10 mg/L and exhibit high seasonal and spatial variability. High nutrient loading is the primary cause of eutrophication and subsequent low dissolved oxygen levels in Upper Klamath Lake. Water quality data collected by the Klamath Tribes contains periods of weeks during the summer months when dissolved oxygen levels in the lake are continuously below the ODEQ criterion of 5.5 mg/L for support of warm water aquatic life (Kann 2010, Morace 2007). Low (0–4 mg/L) dissolved oxygen concentrations occur most frequently in August, the period of declining algal blooms in the lake and warm water temperatures (Morace 2007, ODEQ 2002, Walker 2001).

Intermittent thermal stratification in Upper Klamath Lake can isolate a near-bottom layer of water, within which high sediment oxygen demand (SOD) and decomposition of algal cells depletes oxygen and creates potentially unsuitable conditions for resident fish (e.g., suckers) (Wood 2001, Wood et al. 2006). In the upper water column, high concentrations of N-fixing algae increase dissolved oxygen concentrations during photosynthesis, often resulting in oxygen super-saturation (>10 mg/L) during the daytime. The resulting water column profiles of oxygen are extreme, with depletion in bottom waters and super-saturation in surface waters. This chemical structure is stressful for fish but is not maintained for long periods of time as thermal stability tends to develop and erode over the course of a day (Wood et al. 2006). Stronger, more extended thermal stratification can occur in the relatively deep trench along Eagle Ridge. This can cause longer-term dissolved oxygen depletion, decreasing dissolved oxygen concentrations in the northern part of Upper Klamath Lake for periods of weeks (Wood et al. 2006, et al. 2008).

C.4.1.3 Link River Dam to Klamath River upstream of J.C. Boyle Reservoir

Historical (1950–2001) dissolved oxygen data collected during daytime at various times during the year indicate median values of 4.2–9.2 mg/L in the Upper Klamath River between Link River Dam and J.C. Boyle Reservoir, with the lowest median values occurring from RM 236 to RM 238, which is downstream of the Klamath Straits Drain (RM 240.5), and from RM 245 to RM 248, which is downstream of Link River Dam (RM 253.7) (Figure C-16). Variability over the long-term record in this reach is high, with multiple outlying data points above and below the 95th percentile, indicating both supersaturated and hypoxic dissolved oxygen conditions (PacifiCorp 2004b); the latter not meeting the ODEQ criterion of 5.5 mg/L for support of warm water aquatic life.

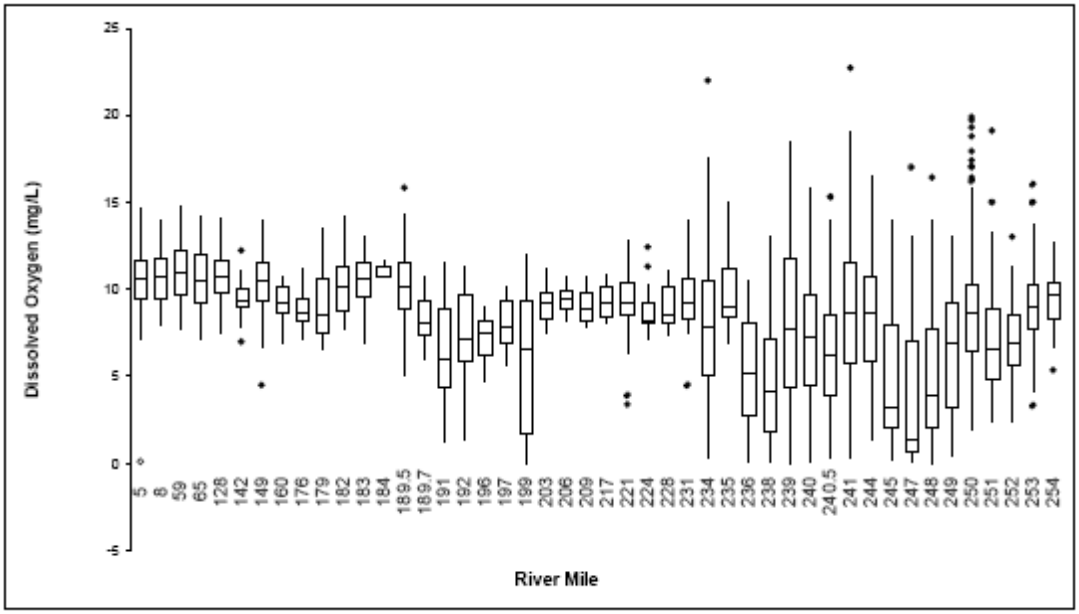

Figure C-16. Box and Whisker Plot of Historical Dissolved Oxygen Data Collected as Daytime Grab Samples from Various Sites in the Klamath River from Klamath River at Klamath Glen (RM 5) to Klamath River at Link River Dam (RM 254) between 1950 and 2001. Source: PacifiCorp 2004b.

More recent continuous *in-situ* data collected in June 2003 show dissolved oxygen concentrations below 4 mg/L and many instances where dissolved oxygen was below 1 mg/L in this reach (Doyle and Lynch 2005). In the downstream Keno Impoundment (including Lake Ewauna), dissolved oxygen reaches very low levels (< 1−2 mg/L) during July−October as algae transported from Upper Klamath Lake settle out of the water and decay (Sullivan et al. 2011) (see also chlorophyll-*a* discussion in Section C.6.1.3). Dissolved oxygen concentrations measured in 2005 from the downstream end of Lake Ewauna (RM 252) to Keno Dam (RM 235) ranged from 7 to 8 mg/L in the early spring, and by late July concentrations were less than 2 mg/L throughout the water column (Deas and Vaughn 2006). During this same period, dissolved oxygen concentrations in Link River inflow were 7–8 mg/L, but apparently had little effect on the dissolved oxygen concentrations in the Keno Impoundment (RM 252–233). Continuous dissolved oxygen data collected by Reclamation at Klamath River upstream of Keno Dam (USGS gage no. 11509370) for the period January 2006–December 2009 exhibit seasonally low dissolved oxygen from July through October (Figure C-17).

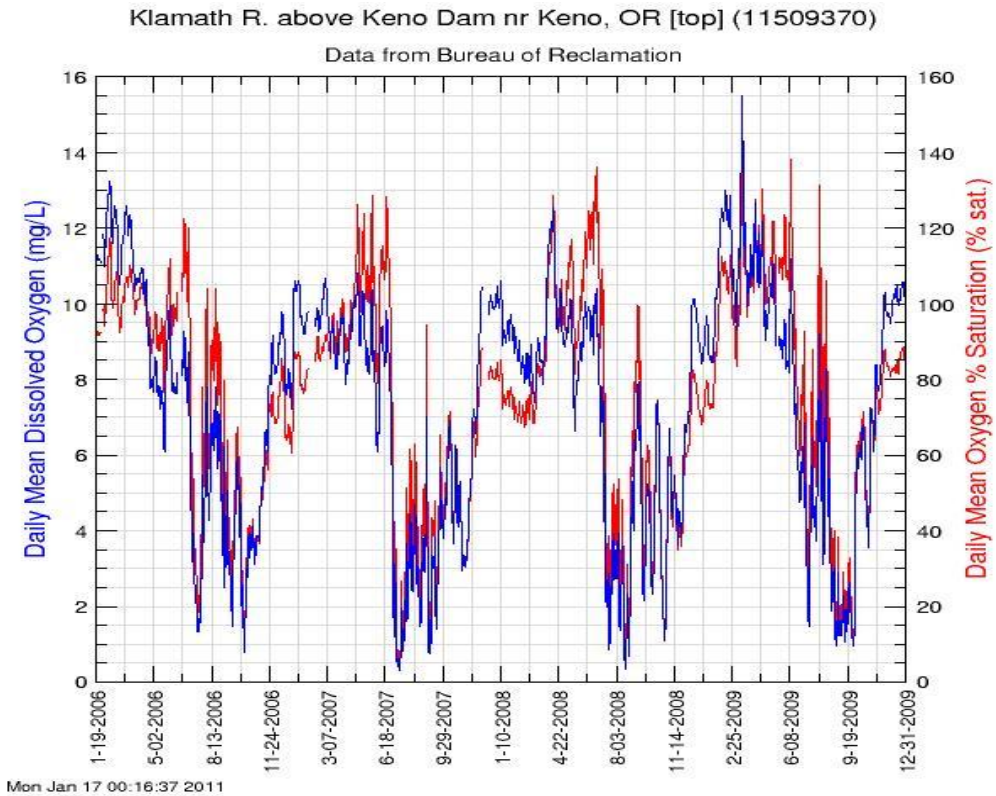

Figure C-17. Daily Mean Dissolved Oxygen Concentration and Percent Saturation for Continuous Data from Klamath River Upstream of Keno Dam, USGS Gage No. 1159370, from January 2006 to December 2009. Source: USGS Data Grapher 2010 (http://or.water.usgs.gov/cgi-bin/grapher).

In addition to water column dissolved oxygen measurements, *in-situ* SOD has been measured at multiple locations in the reach from Link River Dam to the Klamath River upstream of J.C. Boyle Reservoir. The SOD normalized to 20ºC (68ºF) (SOD_{20}) measured in June 2003 in Lake Ewauna (RM 253 to 247) and three sites downstream (Reclamation's monitoring locations at ≈RM 246, at the location of the water intake for the North Canal, and at RM 244 and downstream of Klamath Straits Drain at RM 239) were 0.3–2.9 grams of oxygen per square meter per day (g O_2/m^2/day) (Doyle and Lynch 2005). Eilers and Raymond (2005) report higher summer SOD rates of 2.7–3.6 g O_2/m^2/day in Lake Ewauna, with the equivalent rate of water column oxygen demand due to SOD measured by Doyle and Lynch (2005) at an average of 0.75 mg/L/day (20ºC [68ºF]) and 0.012–1.87 mg/L/day (Sullivan et al. 2010). Following conversion from areal to volumetric units, the author's report that measured water column oxygen demand is equal to or greater than oxygen demand in sediments. Accordingly, a reduction in the upstream load of particulate algal material (i.e., cyanobacteria) to this reach would decrease the high oxygen demand and may limit occurrences of anoxia and hypoxia in the water column (Sullivan et al. 2010, Doyle and Lynch 2005).

Historical water column oxygen demand data is also available for the reach from Link River Dam to the Klamath River upstream of J.C. Boyle Reservoir. The historical record indicates relatively high biological oxygen demand (BOD) (but also high variability) at the downstream end of Lake Ewauna (RM 247) (Figure C-18). Sullivan et al. (2010) measured BOD in the Klamath River between Keno Dam (RM 233) and Link River Dam (RM 253.7), as well as within Klamath Straits Drain (RM 240.5) from April to November 2007. They report the existence of at least two pools of organic matter (i.e., labile and refractory) in these reaches, each possessing a different average rate of decay. The labile pool of organic matter is dominated by particulate organic matter such as *A. flos-aquae* , and decays rapidly, with 80 percent of the associated oxygen demand expressed in 8 days (Sullivan et al. 2010). The refractory pool is also largely composed of particulate matter, but includes some dissolved organic matter and decays at a much slower rate, consuming oxygen for at least 60 days. Since the travel time from Link River to Keno Dam is 6–10 days during summer months, the majority of BOD in this reach during the summer months appears to be attributable to labile, algal-derived organic matter (Sullivan et al. 2010).

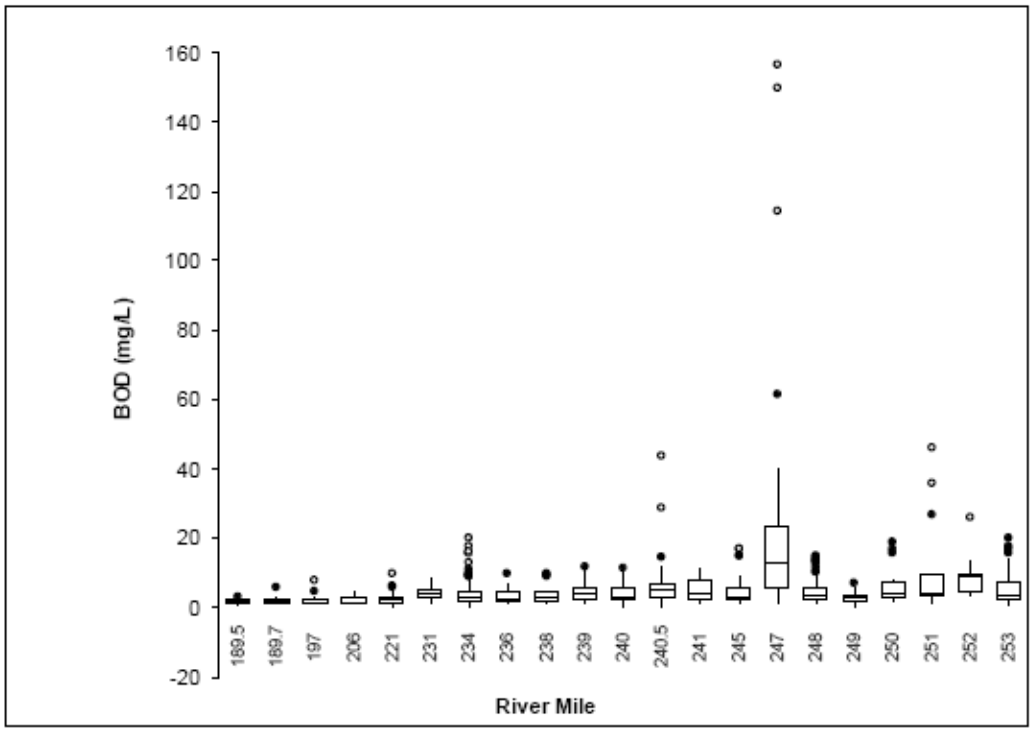

Figure C-18. Box and Whisker Plot of Historical BOD Data Collected from Riverine Sites in the Klamath River Between 1950 and 2001. Link River Dam (RM 253.7), Lake Ewauna and Keno Impoundment (RM 233–253.1), J.C. Boyle Reservoir (RM 224.7–RM 228.3), Copco 1 Reservoir (RM 198.6–203.1), Copco 2 Reservoir (RM 198.3–RM 198.6), Iron Gate Reservoir (RM 1901.1–196.9). Source: PacifiCorp 2004b.

Lastly, four facilities discharge treated wastewater to the Keno Impoundment; however, these facilities contribute a very small amount (<1.5% of the organic material loading) to the overall oxygen demand in the Keno Reach. Overall, decomposition of algae transported from Upper Klamath Lake appears to be the primary driver of low oxygen in the Keno Impoundment (including Lake Ewauna) (Sullivan et al. 2009, Kirk et al. 2010).

The Klamath River in Oregon from the California-Oregon state line to RM 251 is currently listed as impaired for dissolved oxygen under Section 303(d) of the CWA (see Section 3.2, Table 3.2-8). Oregon's Upper Klamath Lake Drainage TMDLs (2002) and Upper Klamath and Lost River draft TMDLs (2010) address dissolved oxygen, among other water quality parameters (see Section 3.2.2.4), and indicate that reductions in BOD loading from Upper Klamath Lake and both point and nonpoint sources in the Upper Klamath River are required in order to achieve water quality standards.

C.4.1.4 Hydroelectric Reach

Dissolved oxygen levels in the Hydroelectric Reach vary on a seasonal and daily basis (e.g., Karuk Tribe of California 2002, 2003; FERC 2007; PacifiCorp 2004b, 2008a; USFWS 2008; FISHPRO 2000, Zedonis and Turner 2010). During summer, the KHP reservoirs exhibit varying degrees of supersaturation in surface waters due to high rates of algal photosynthesis and hypolimnetic anoxia as dissolved oxygen is depleted in bottom waters during seasonal thermal stratification and microbial decomposition of dead algae. J.C. Boyle Reservoir, a relatively long, shallow reservoir, does not stratify. However, J.C. Boyle Reservoir can exhibit large variations in dissolved oxygen due to conditions in the upstream reach from Link River Dam through the Keno Impoundment (including Lake Ewauna), and in Upper Klamath Lake (see previous section). Copco 1 and Iron Gate Reservoirs thermally stratify beginning in April/May and do not mix again until October/November (FERC 2007). Stratification occurs, with dissolved oxygen in Iron Gate and Copco 1 surface waters generally at or, in some cases above[2], saturation and levels in hypolimnetic waters reaching minimum values near 0 mg/L by July (for example, 2008 data shown in Figure C-19).

C.4.2 Lower Klamath Basin

C.4.2.1 Iron Gate Dam to Salmon River

Historical (1950–2001) dissolved oxygen data (reflecting day time grab sampling) are variable. Median values were 8.1–10.8 mg/L in the Klamath River between Iron Gate Dam and the confluence with the Salmon River, with the lowest median values and the greatest general variability in the first mile downstream of the dam (Figure C-16, PacifiCorp 2004b). Based on more recent data, dissolved oxygen concentrations immediately downstream of Iron Gate Dam regularly fall below 8 mg/L, the former North Coast Basin Plan water quality objective for dissolved oxygen during summer months (Karuk Tribe of California 2001, 2002, 2007, 2009; NCRWQCB 2010). Based

[2] At high rates of photosynthesis, oxygen production may exceed the diffusion of oxygen out of the water column and oxygen supersaturation may result.

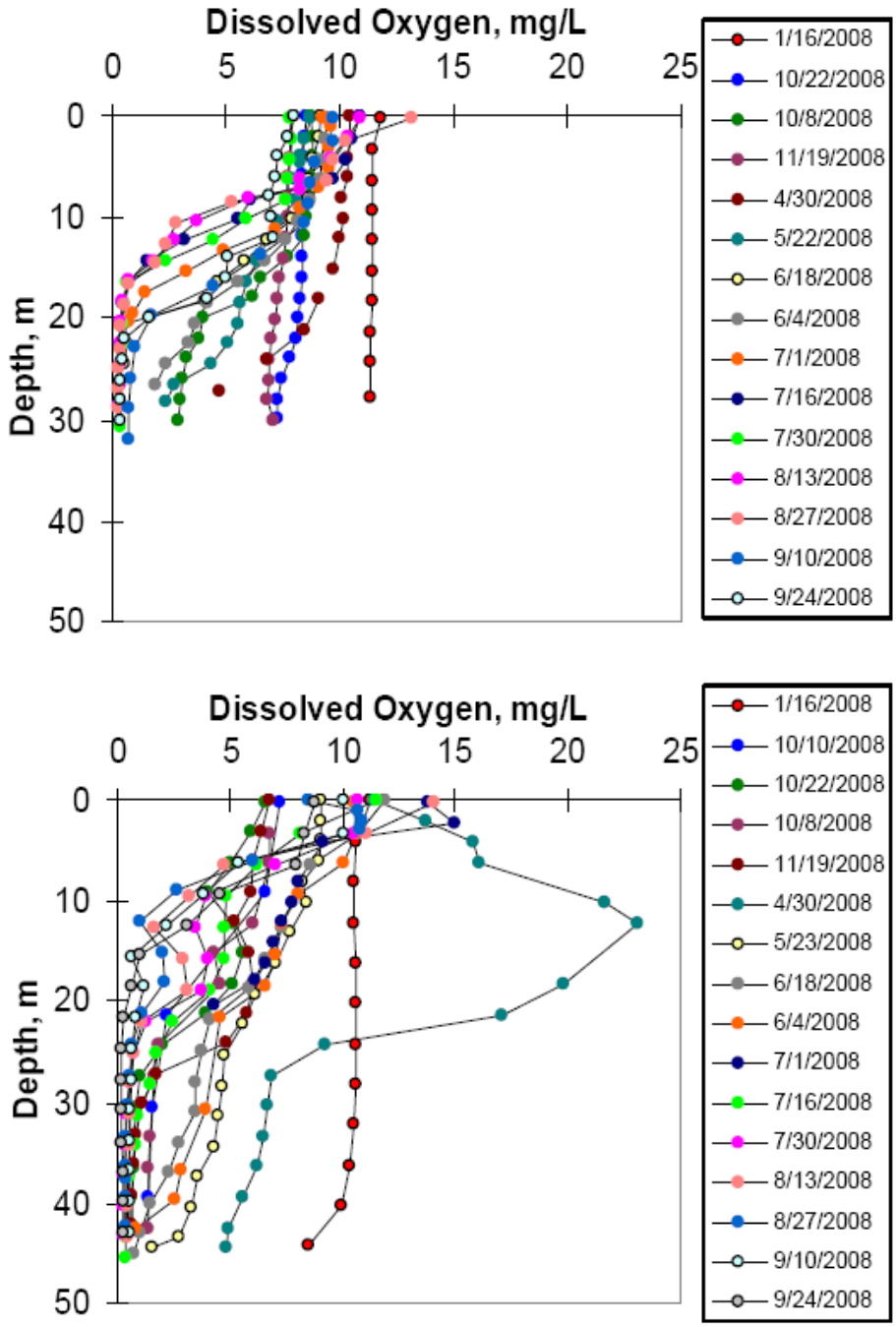

Figure C-19. Vertical Profiles of Dissolved Oxygen Concentration Measured in Copco 1 Reservoir Near the Dam (Top Plot) and Iron Gate Reservoir Near the Dam (Bottom Plot) in 2008. Source: Raymond 2009.

on continuous Sonde data collected at multiple locations in the lower Klamath River during summer 2004–2006, roughly 45 to 65 percent of measurements immediately downstream of Iron Gate Dam did not achieve the (previous) North Coast Basin Plan water quality objective of 8 mg/L (the objective is now based on percent saturation, see Section 3.2, Table 3.2-5). The percent of dissolved oxygen measurements below 8 mg/L decreases with distance downstream, particularly in 2005 and 2006. Table C-2 summarizes dissolved oxygen concentrations in the Lower Klamath River during Summer 2004-2006.

Table C-2. Dissolved Oxygen Concentrations in the Lower Klamath River during Summer 2004–2006.

Location	2004		2005		2006	
	n [(1)]	%	n [(1)]	%	n [(1)]	%
At Iron Gate Dam (RM 190.1)	2,706	64	4,498	45	5,391	61
Upstream of Shasta River (RM 176.7)	5,478	50	5,533	49	-	-
Upstream of Scott River (RM 143)	2,966	58	4,457	47	-	-
Seiad Valley (RM 129.4)	3,381	57	4,713	45	5,526	40
Orleans (RM 59)	57	37	4,533	23	5,349	15
Weitchpec (RM 43.5)	4,142	48	5,400	7	5,332	6
Downstream of Weitchpec (≈ RM 42)	5,500	16	3,529	11	5,293	4
Upstream of Trinity (RM 40)	-	-	5,535	5	5,739	3
Turwar (RM 5.8)	5,066	30	5,543	6	-	-

Source: Ward and Armstrong 2006, NCRWQCB 2010.

[1] Dissolved oxygen measurements were collected at 30-minute increments for a total of forty-eight daily measurements.

Key:

n=number of measurements

%=percent of measurements not achieving the North Coast Basin Plan previous water quality objective of 8 mg/L

Withdrawals occur at depths of approximately 12 meters in Iron Gate Reservoir, and thus downstream dissolved oxygen concentrations tend to reflect oxygen conditions in the lower epilimnion (Section C.4.1.4) when the reservoirs are stratified, with some increases in dissolved oxygen as the water is re-aerated upon discharge. In the fall, before and after reservoir turnover, low dissolved oxygen concentrations from the hypolimnion can be translated downstream. Table C-3 summarizes dissolved oxygen concentrations taken downstream of Iron Gate Dam (RM 190), representing the range of daily average measurements.

Table C-3. Range of Observed Dissolved Oxygen Concentrations Downstream of Iron Gate Reservoir.

Year	General Range of Daily Average Dissolved Oxygen (mg/L) downstream of Iron Gate Dam (RM 190, near USGS Gage No. 11516530)		
	September–October	November	Source
2001	4–6[(1)]	7–8	Karuk Tribe of California 2003
2002	4–9[(2)]	-	Karuk Tribe of California 2003
2004	6–9.5	8–9	Zedonis and Turner 2010
2006	6.5–8	7–8	Karuk Tribe of California 2009
2007	7–9[(2)]	-	Karuk Tribe of California 2009
2008	6.5–8.5	-	Karuk Tribe of California 2009
2009	7–10	-	Karuk Tribe of California 2010
2010	7–10.5	5–7	Preliminary data from PacifiCorp and Karuk Tribe of California, K. Fetcho, Yurok Tribe, pers. comm.

[1] No September data reported
[2] No October data reported

In situ continuous data collected during 2008–2010 by PacifiCorp in the Klamath River downstream of Iron Gate Dam demonstrates the seasonal decreases in dissolved oxygen (measured as percent saturation and concentration) originating from the reservoirs, with the lowest average monthly values occurring in October and November rather than during the months with the warmest water temperatures (i.e., July and August) (Table C-4).

It has been suggested that daily fluctuations of up to 1–2mg/L measured in the Klamath River downstream of Iron Gate Dam (RM 190.1) (Karuk Tribe of California 2002, 2003; YTEP 2005; NCRWQCB 2010) are caused by daytime algal photosynthesis and nighttime bacterial respiration. Low DO can also be often driven by bacterial decomposition of algae in the reservoir (Ward and Armstrong 2010).

Testing of turbine venting at Iron Gate Dam is has been conducted as part of Klamath Hydroelectric Settlement Agreement Interim Measures (Section 1.2.4 Klamath Hydroelectric Settlement Agreement, Interim Measures). Turbine venting is Interim Measure 3 (IM 3), and has a goal of improving dissolved oxygen concentrations downstream of Iron Gate Dam. Test results from 2008 indicate that dissolved oxygen levels immediately downstream of Iron Gate Dam can be increased through the mechanical introduction of oxygen as water passes through the turbines (Carlson and Foster 2008, PacifiCorp 2008a). Monitoring data taken during the tests suggest that an increase of approximately 0.5 to 2 mg/L in dissolved oxygen (approximately 7 to 20 percent saturation) is possible; however, further testing and monitoring are recommended (PacifiCorp 2008a)**.**

Table C-4. Average Monthly Water Temperature, Dissolved Oxygen Percent Saturation, and Dissolved Oxygen Concentration in the Klamath River Downstream of Iron Gate Dam (RM 189.7).

Month	Average Monthly Water Temperature (^oC)	Average Monthly Dissolved Oxygen (% Saturation)	Average Monthly Dissolved Oxygen (mg/L)
2008			
June	18.4	92.2	8.7
July	22.3	90.0	7.8
August	21.8	91.8	8.0
September	18.6	84.5	7.9
October	14.8	66.2	6.7
November	10.3	67.4	7.5
December	7.0	70.0	8.5
2009			
January	3.7	79.4	10.5
February	4.4	83.0	10.8
March	6.7	83.2	10.2
April	8.4	82.2	9.6
May	17.4	94.4	9.0
June	19.3	87.9	8.1
July	21.2	86.8	7.7
August	21.7	99.9	8.8
September	19.4	95.7	8.8
October	14.6	77.7	7.9
November	9.9	71.2	8.1
December	5.0	81.2	10.4
2010 (Preliminary)			
January	3.9	86.6	11.4
February	5.4	92.2	11.1
March	7.2	88.9	10.5
April	9.5	100.2	11.4
May	12.7	96.4	10.2
June	16.8	87.3	8.5
July	21.3	90.9	8.1
August	21.9	88.3	7.7
September	18.4	96.7	9.1
October	15.5	85.1	8.5
November	11.8	57.5	6.2

Raw daily data from http://www.pacificorp.com/es/hydro/hl/kr.html# (PacifiCorp 2008b, 2009, 2010). Data obtained with YSI 6600 V2 or 6900 Multiprobe Datasondes (30-minute intervals).

Farther downstream in the mainstem Klamath River, near Seiad Valley (RM 129.4), dissolved oxygen concentrations tend to increase; however, values are often below 8 mg/L (i.e., 2001, 2002, 2006, and 2009, as reported in Karuk Tribe of California [2001, 2002, 2007, 2009]). Dissolved oxygen concentrations near Seiad Valley continue to exhibit variability, with mean daily values ranging from approximately 6.5 mg/L to

(supersaturated concentrations of) approximately 10.5 mg/L, from June through November, 2001–2002 and 2006–2009.

C.4.2.2 Salmon River to Estuary

Measured concentrations of dissolved oxygen in the mainstem Klamath River downstream of the confluence with the Salmon River (RM 66) continue to increase relative to concentrations at upstream sites. Despite this, values sometimes fall below 8 mg/L in this reach (e.g., at the Orleans gage [RM 59] during 2001, 2002, 2006 as reported in Karuk Tribe of California [2001, 2002, 2007, 2009], Ward and Armstrong 2006, NCRWQCB 2010). Dissolved oxygen concentrations near Orleans also exhibit variability, with mean daily values ranging from approximately 6.5 mg/L to (supersaturated concentrations of) 11.5 mg/L from June through November, 2001–2002 and 2006–2009. Extremely high mean daily dissolved oxygen concentrations (11–15.5 mg/L) (Sonde data) were reported for October 2006 at the Orleans gage (Karuk Tribe of California 2007, 2009).

Dissolved oxygen concentrations in the mainstem Klamath River upstream of the confluence with the Trinity River (RM 42.5) ranged 5.5–10.3 mg/L in September and October 2004, respectively, with minimum dissolved oxygen concentrations below 8 mg/L (the Basin Plan minimum dissolved oxygen criterion prior to 2010) for extended periods of time from mid-August through early September (YTEP 2005). In 2009 at this location, dissolved oxygen concentrations ranged 7.1–11.8 mg/L, with minimum dissolved oxygen concentrations dropping below 8 mg/L (the Basin Plan minimum dissolved oxygen criterion prior to 2010) for an extended period of time from mid-July to early August, and again from late August to early September (Sinnott 2010). In 2010, concentrations ranged 7.9–12.1 mg/L (Sinnott 2011b), with minimum dissolved oxygen concentrations remaining above 2010 amended Basin Plan minimum dissolved oxygen concentration criteria based on percent saturation (e.g., 7.0, 6.9, and 7.8 mg/L for July, August, and September, respectively, see Section 3.2, Table 3.2-5).

Further downstream, at the confluence with the Trinity River (RM 42.5) and at the Turwar gage (RM 5.8), daily minimum dissolved oxygen values at the Trinity River and Turwar sites during May through November are consistently observed to occur late-night or early in the morning, likely due to respiration by aquatic vegetation (YTEP 2005, Sinnott 2010, 2011b). At Turwar (RM 5.8) in 2004, minimum dissolved oxygen concentrations dropped below 8 mg/L (the Basin Plan minimum dissolved oxygen criterion prior to 2010) between late July and late August (YTEP 2005); dissolved oxygen concentrations ranging 5.9–10.1 mg/L were observed in August and September). In 2009, dissolved oxygen concentrations at Turwar ranged 7.3–11.7 mg/L, with minimum dissolved oxygen concentrations dropping below 8 mg/L for an extended period of time from mid-July to early August (Sinnott 2010). In 2010, concentrations ranged 7.8–11.8 mg/L, with minimum values remaining above 2010 amended Basin Plan minimum dissolved oxygen concentration criteria based on percent saturation (e.g., 7.0,

6.9, and 7.8 mg/L for July, August, and September, respectively, see Section 3.2, Table 3.2-5).

C.4.2.3 Klamath Estuary

Dissolved oxygen concentrations within the Klamath River Estuary are highly variable spatially and temporally and are greatly influenced by season, river flow, vertical water column stratification (thermal and/or chemical), and location of the estuary mouth, the latter changing due to periodic sand bar movement. Dissolved oxygen concentrations have been monitored in the Klamath River Estuary by CDFG (Wallace 1998) and most recently by the Yurok Tribe Fisheries (Hiner 2006) and Environmental Programs (YTEP 2005) with support from the North Coast Regional Water Quality Control Board. Concentrations in the deeper, main channel of the estuary are generally greater than 6 to 7 mg/L throughout the year (Hiner 2006, YTEP 2005).

Lower dissolved oxygen concentrations ranging 2.5–5.5 mg/L have been measured near the bottom of deep pools or in heavily vegetated side channels (Wallace 1998). Low dissolved oxygen concentrations (<1 to 5 mg/L) have been observed during summer months in the relatively shallow, heavily vegetated south slough (Hiner 2006, Wallace 1998). The low levels of dissolved oxygen observed in the slough are likely due to high rates of growth and subsequent decomposition of algae and macrophytes, which are not abundant elsewhere in the estuary.

Dissolved oxygen becomes progressively more variable and generally lower in concentration nearer the estuary bottom and the estuary mouth, with concentrations frequently below 6 mg/L during summer months (Hiner 2006, YTEP 2005). Low dissolved oxygen has also been observed during late summer months when a sand berm forms across the river mouth, forcing the river to flow south diagonally between two sand spits. This berm prevents ocean water from entering the estuary, creating ‘lagoon-like’ conditions until higher flows breech the berm (Wallace 1998, Hiner 2006). These conditions were documented in 1994 and 2001; in 2001, a decrease in dissolved oxygen concentrations was measured related to sand berm formation, with especially marked decreases in dissolved oxygen in the south slough (Hiner 2006).

C.5 pH

C.5.1 Upper Klamath Basin

C.5.1.1 Wood, Williamson, and Sprague Rivers

The Sprague River is listed as impaired under Section 303(d) of the CWA for pH in the summer months based upon the 6.5–9.0 range established by ODEQ in the Upper Klamath Lake Drainage TMDLs (see Section 3.2, Table 3.2-8). In the Wood River subbasin, Campbell and Ehinger (1993) report pH levels of 6.9–8.2 in the headwaters and

7.4–8.2 at the mouth. During the summer months, Wood River has little potential to influence the overall pH of downstream Agency Lake, based upon low relative flow volumes. High pH levels (8.5–9.5) have been observed in the Sprague River linked primarily to high rates of periphytic (i.e., benthic or attached algae) algal photosynthesis (ODEQ 2002). The pH criterion (see Section 3.2, Table 3.2-8) has been exceeded on the mainstem Sprague River during the warmest part of the day in August from RM 79.1 to the confluence with the Williamson River.

C.5.1.2 Upper Klamath Lake

Upper Klamath Lake is listed as impaired under Section 303(d) of the CWA for pH based upon the 6.5–9.0 range established in the Upper Klamath Lake Drainage TMDLs (see Section 3.2, Table 3.2-8). During November–April (non-growing season) pH levels in Upper Klamath Lake are near neutral (Aquatic Scientific Resources 2005). However, in summer, instances of pH levels above 10 and extended periods of pH greater than 9 lasting for several weeks have been associated with large summer algal blooms occurring in the lake (Kann 2010; Morace 2007; Kann and Smith 1999). On a daily basis, algal photosynthesis can elevate pH levels during the day, with changes exceeding 2 pH units over a 24-hour period. Elevated pH is linked to nutrient dynamics in the lake, as increases in pH also increase phosphorus flux from internal loading via in-lake sediments to the water column by solubilizing iron-bound phosphorus. A threshold of pH 9.3 has been identified in the lake where the probability of internal phosphorus loading from lakebed sediments sharply increases (Kann and Walker 1999). A positive feedback loop can be created where an initially high phosphorus concentration supports an algal bloom, which then raises pH through photosynthesis and causes the release of phosphorus from lake sediments (Kann and Walker 1999). Estimates of internal loading of phosphorus range from 57% (Miller and Tash 1967) to 61% (Kann and Walker 1999) of the total phosphorus sources, with much of the internal loading source occurring in the summer. Internal loading in an unstratified lake such as Upper Klamath Lake is driven by several mechanisms including 1) high pH resulting in dissolution of iron and aluminum complexed phosphorus, 2) anoxic conditions at sediment-water interface resulting in dissolution of iron-phosphorus complexes, 3) temperature driven microbial metabolism resulting in mineralization of organic phosphorus into dissolved phosphorus, 4) dissolved phosphorus released directly from algal cell in the sediment, wind-driven sediment re-suspension, 5) bioturbation from benthic invertebrates allowing migration phosphorus in deeper sediment to migrate upward, and 6) chemical diffusion from phosphorus concentration gradients in the sediments.

Open water areas in Upper Klamath Lake tend to have the highest measured pH, reaching levels above 10 (Gearhart et al. 1995), while nearshore areas may have relatively lower pH of due to the production of acidic humic substances associated with fringe wetlands and marshes there (Aquatic Scientific Resources 2005). However, an estimated 35,000 acres of marsh and wetlands directly adjacent to Upper Klamath Lake have been converted to pasture or agricultural fields (Gearhart et al. 1995), potentially reducing the buffering effect of the littoral marshes on lake pH level under current conditions.

C.5.1.3 Link River Dam to Klamath River upstream of J.C. Boyle Reservoir

The Klamath River from Link River Dam to upstream of J.C. Boyle Reservoir, inclusive of the Keno Impoundment (and Lake Ewauna), is also listed as impaired under Section 303(d) of the CWA for pH during the summer months (see Section 3.2, Table 3.2-8). Generally, pH in the reach from Link River Dam through the Keno Impoundment increases from spring to early summer and decreases in the fall; however, there are site-dependent variations in the observed trend. Measurements of pH collected by Sullivan et al. (2008) in the Keno Impoundment (≈RM 235 to RM 253) from April to November 2007 indicate pH ranging from 7.2 in April to a one-day peak of 9.9 in November (Sullivan et al. 2008). Downstream at Miller Island, pH was typically 8.5 in the spring, increasing to values at or near 9.5 in the summer, and decreasing to near 7.5 in the fall. In 2009, springtime pH levels at J.C. Boyle Reservoir were also typically 8.5, decreasing during the summer and fall to 7.6–7.9 (Watercourse Engineering, Inc. 2011).

C.5.1.4 Hydroelectric Reach

The Hydroelectric Reach is not listed as impaired under Section 303(d) of the CWA for pH (see Section 3.2, Table 3.2-8). Based upon monitoring conducted by PacifiCorp, pH in the Hydroelectric Reach is seasonally variable, with levels near neutral (7.5–8.0) during the winter and increasing in the spring and summer (7.7–8.1). Peak values (8–9.2) are recorded during May and September (Raymond 2010). Longitudinally, pH ranges from 7.3 to 9.2 in this reach, with the lowest values recorded downstream of J.C. Boyle Reservoir and the highest values in Copco and Iron Gate Reservoirs (Raymond 2008, 2009, 2010). The pH is typically elevated where and when algal photosynthesis rates are high; maximum values (e.g., between 8 and 9) are measured at or near the water surface during periods of thermal stratification and high nutrient concentrations (Raymond 2008).

C.5.2 Lower Klamath Basin

C.5.2.1 Iron Gate Dam to Salmon River

Although not listed as impaired for pH under Section 303(d) of the CWA (see Section 3.2, Table 3.2-8), the California North Coast Basin Plan pH maximum of 8.5 units is regularly exceeded in the lower Klamath River downstream of Iron Gate Dam (USFWS 2008; FERC 2007; FISHPRO 2000; Karuk Tribe of California 2002, 2003; YTEP 2005; NCRWQCB 2006, 2010). During April through October 2000–2004, incidences of pH *below* the minimum North Coast Basin Plan limit of 7.0 were also observed immediately downstream of Iron Gate Dam (RM 190.1) (PacifiCorp 2004b). Because the Klamath River is a weakly buffered system (i.e., has typically low alkalinity <100 mg/L; PacifiCorp [2004a], Karuk Tribe of California [2010]) it is susceptible to photosynthesis-driven daily swings in pH. Observed exceedances of pH water quality objectives usually occur during later afternoon or early evening, following the period of maximum photosynthesis (NCRWQCB 2010). In addition, the highest pH values generally occur during late-summer and early-fall months (August–September). In 2007, daily maximum pH values downstream of Iron Gate Dam (≈RM 189) were 8.2–9.6 with the highest

documented pH occurring in September (Figure C-20); near Seiad (≈RM 128), maximum pH values were slightly lower, at 8.1–9.4 with the highest documented pH occurring in mid-August (and Figure C-21; Karuk Tribe of California 2007).

The most extreme pH exceedances typically occur from Iron Gate Dam (RM 190.1) to approximately Seiad Valley (RM 129.4), with pH values generally decreasing with distance downstream (Figure C-22; FERC 2007; Karuk Tribe of California 2007, 2009, 2010). However, during May–October 2005, the greatest number of pH exceedances in this reach occurred just upstream of the mainstem confluence with the Shasta River (RM 66) (Figure C-23).

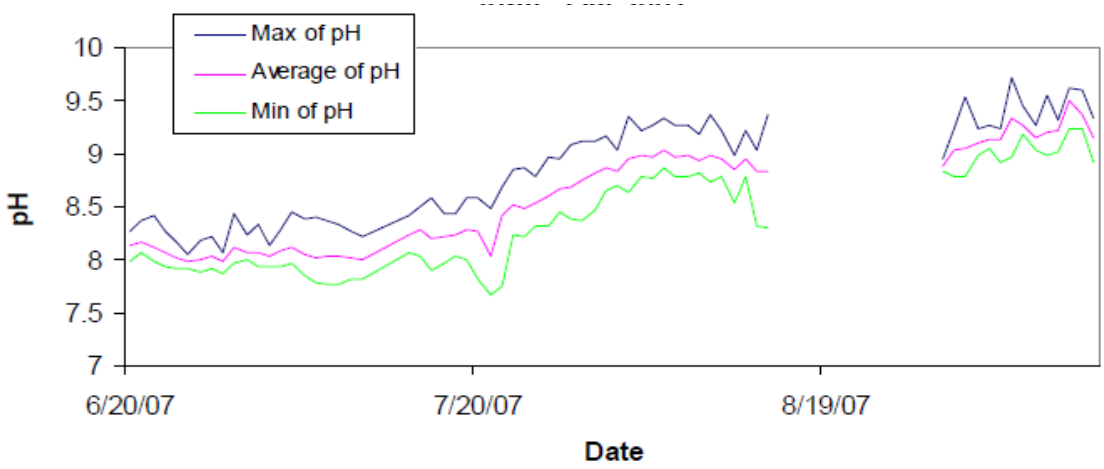

Figure C-20. Daily Maximum, Mean, and Minimum pH Values in the Klamath River Downstream of Iron Gate Dam (RM 190.1) from June to October 2007. Source: Karuk Tribe of California 2007.

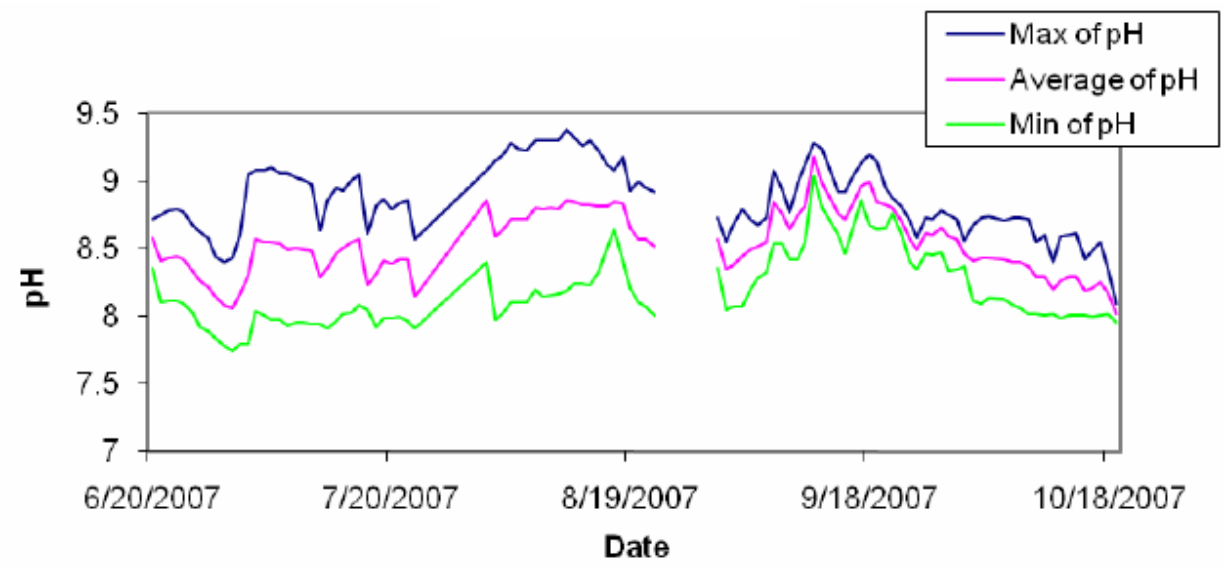

Figure C-21. Daily Maximum, Mean, and Minimum pH Values in the Klamath River near Seiad (≈RM 128) from June to October 2007. Source: Karuk Tribe of California 2007.

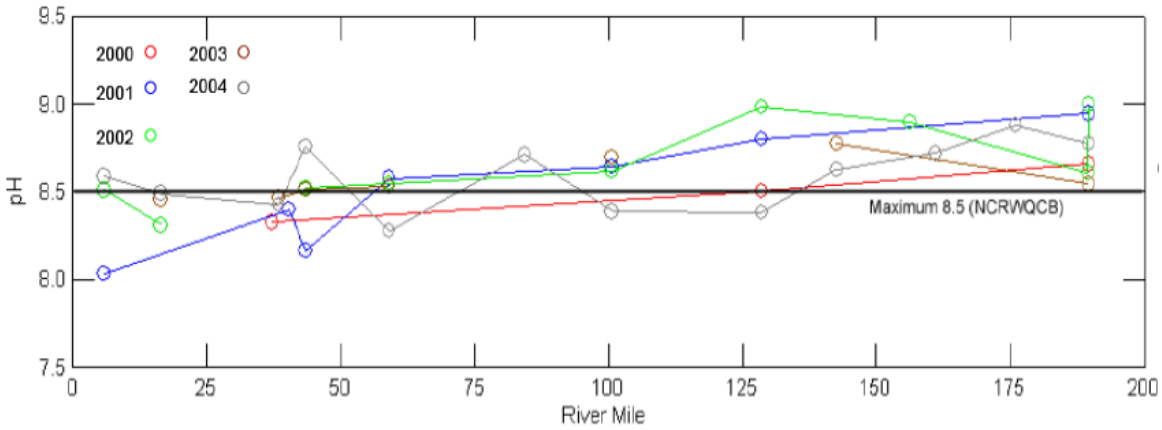

Figure C-22. Average August Daily Maximum pH Values for Locations along the Mainstem Klamath River Downstream of Iron Gate Dam for the Years 2000–2004 using Data Collected by USFWS, USGS, and the Karuk Tribe of California and Yurok Tribe. Source: Kier Associates 2006 as cited in FERC 2007.

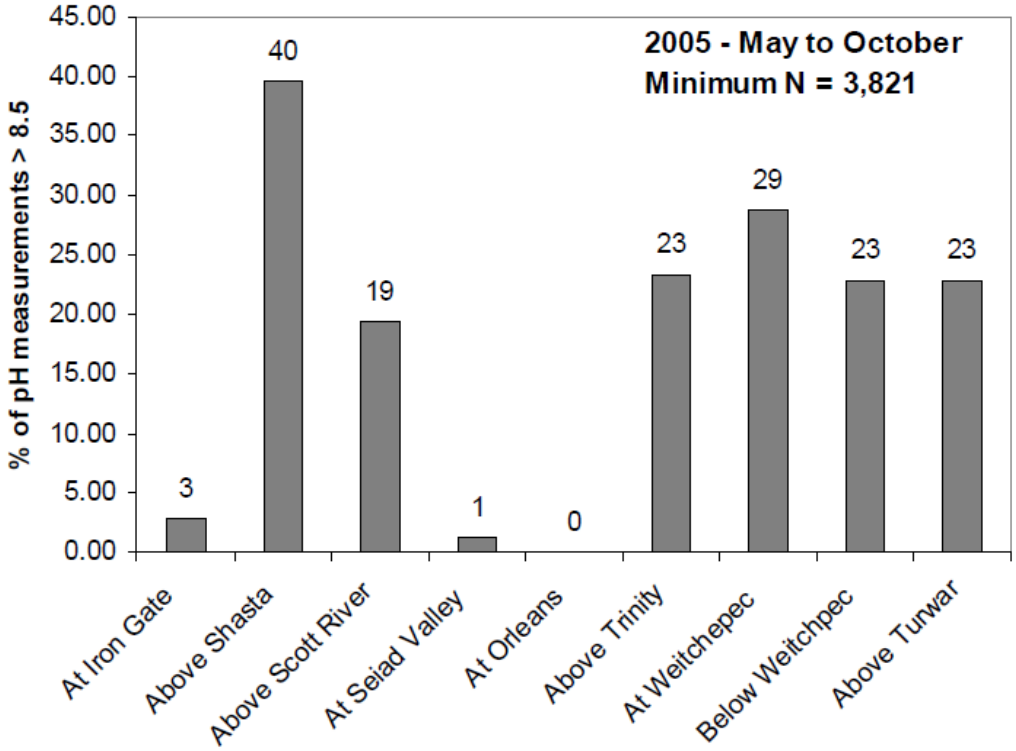

Figure C-23. Percent of pH Measurements in the Lower Klamath River Exceeding the North Coast Basin Plan Water Quality Objective of 8.5 pH Units during 2005. Source: NCRWQCB 2010

C.5.2.2 Salmon River to Estuary

The North Coast Basin Plan pH maximum of 8.5 is also regularly exceeded in the lower Klamath River between the Trinity River and Turwar Creek during summer months (Figure C-19; USFWS 2008; FISHPRO 2000; Karuk Tribe of California 2002, 2003; YTEP 2005; NCRWQCB 2006, 2010). Water quality monitoring by the Karuk Tribe includes pH data from Orleans (RM 59), which is just downstream of the mainstem confluence with the Salmon River (see also Section C.5.2.1). Daily maximum pH values at Orleans were 7.9– 8.9 from June through October 2007, with the highest pH occurring in mid-September (Figure C-24; Karuk Tribe of California 2007). In the mainstem river between the confluence with the Trinity River and the Klamath Estuary, annual water pH monitoring has been conducted by the YTEP since 2002 (YTEP 2004, YTEP 2005, Sinnott 2010). During 2009 monitoring, peak pH values were documented from July through September with the highest daily maximums recorded in early July; the highest pH values were documented at the most upstream location (i.e., just over 9.0 at Klamath River at Weitchpec [RM 43.5]) while both sample locations farther downstream were approximately 8.8 (Klamath River upstream of Tully Creek [RM 38.5] and upstream of Turwar Boat Ramp [RM 8]; Figure C-25; Sinnott 2010).

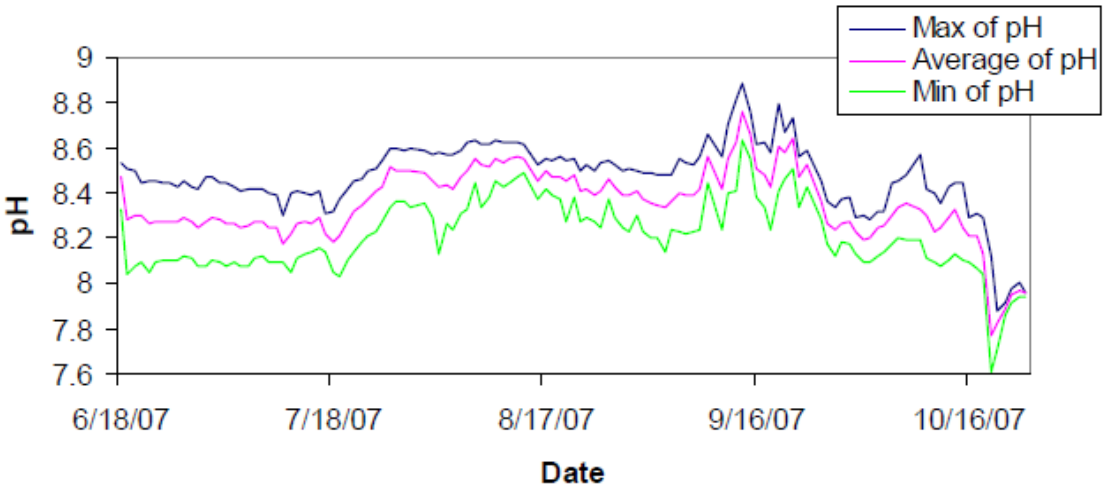

Figure C-24. Daily Maximum, Mean, and Minimum pH Values on the Klamath River near Orleans (RM 59) from June to October 2007. Source: Karuk Tribe of California 2007.

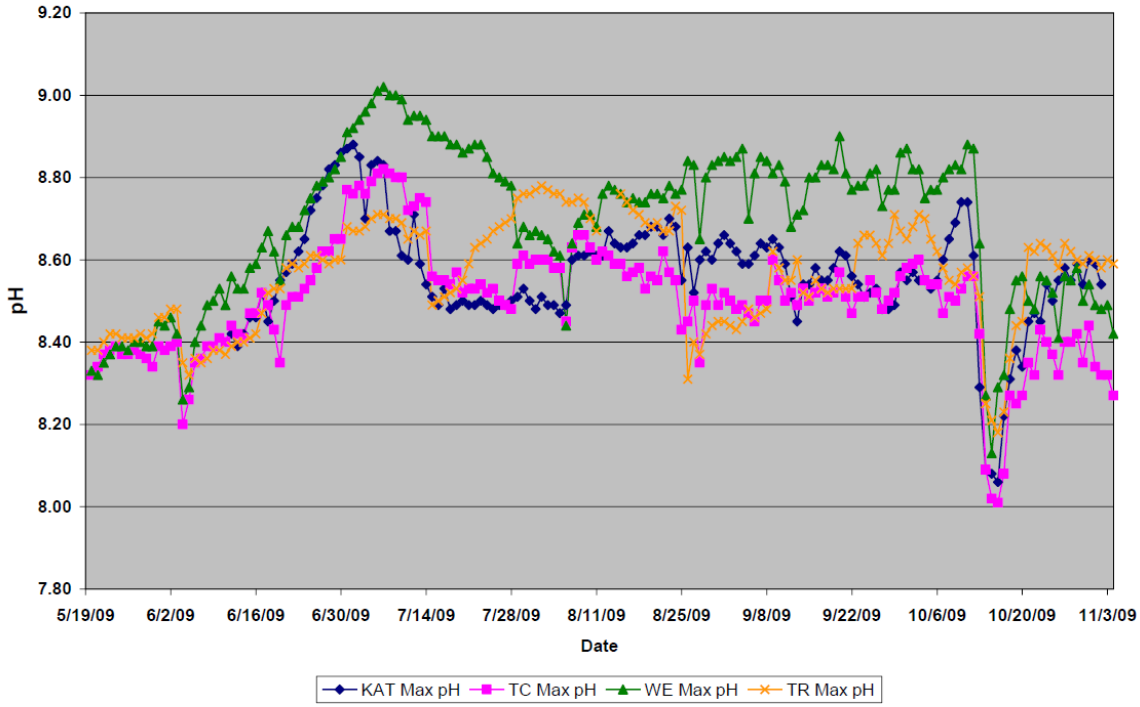

Figure C-25. Daily Maximum pH in the Klamath River at Weitchpec (RM 43.5 [WE]), Upstream of Tully Creek (RM 38.5 [TC]), and Upstream of Turwar Boat Ramp (RM 8 [KAT]), as well as in the Trinity River (RM 40 [TR]) near the Confluence with the Klamath River (RM 0.5 [TR]). Source Sinnott 2010.

C.5.2.3 Klamath Estuary

pH within the Klamath River Estuary is variable spatially and temporally and is influenced by season, river flow, vertical stratification (thermal and/or chemical), and location of the estuary mouth, the latter changing due to periodic sand bar movement. Although not listed as impaired for pH under Section 303(d) of the CWA (see Section 3.2, Table 3.2-8), the North Coast Basin Plan pH maximum of 8.5 is also regularly exceeded in the Klamath Estuary (Figure C-19) (YTEP 2005). Based on Yurok Tribe water quality data, pH in the Klamath Estuary ranges between approximately 7.5 and 9, with peak values generally occurring during the summer months (YTEP 2005, Sinnott 2010, Sinnott 2011b). Daily variations in pH are typically on the order of 0.5 pH units, and fluctuations tend to be somewhat larger in the late summer and early fall. The Environmental Fluid Dynamics Code estuary model component used in the California Klamath River TMDL development as well as observed data show very low algae and chlorophyll-*a* concentrations in the estuary (YTEP 2005), suggesting that local photosynthesis and biological respiration are not significant enough to cause large daily fluctuations of pH, as seen in upstream reaches. When large daily fluctuations are observed, they are likely caused by an upstream daily signal that is subsequently transported into the estuary.

C.6 Algal Toxins and Chlorophyll-a

C.6.1 Upper Klamath Basin

C.6.1.1 Wood, Williamson, and Sprague Rivers

M. aeruginosa occurrence has not been reported in the Wood, Williamson, and Sprague Rivers, and algal toxin data are not available for these rivers. Measured water column chlorophyll-*a* production in the Sprague River does not currently exceed action levels (see Section 3.2, Table 3.2-8) (ODEQ 2002), although data collected prior to 2000 is not readily available. Abundant periphytic algae (i.e., benthic or attached algae) are known to cause water quality impairments for dissolved oxygen and pH in these tributaries to Upper Klamath Lake (Sections C.4.1.2 and C.5.1.2). Chlorophyll-*a* concentrations in Wood River increase from the headwaters to the mouth; flowing water at the headwater springs of the Wood River exhibit a total chlorophyll-*a* concentration of 0–0.3 µg/L, while concentrations at the mouth range from 0.9 to 3.9 µg/L (Campbell et al. 1993).

C.6.1.2 Upper Klamath Lake

In Upper Klamath Lake, large summertime blooms of cyanobacteria are typically dominated by *A. flos-aquae* , with relatively smaller amounts of *M. aeruginosa* present (see Section 3.4, Algae). Despite this, *M. aeruginosa* is believed to be responsible for the production of microcystin in the lake. A preliminary study of the presence, concentration, and dynamics of microcystin in Upper Klamath Lake, particularly as

related to Lost River sucker (*Deltistes luxatus*) and shortnose sucker (*Chasmistes brevirostris*) exposure, USGS collected water samples at multiple lake sites from July to October 2007 and June through September 2008. At most sites and on most sampling dates in 2008, microcystin concentrations were equal to or greater than the World Heath Organization (WHO) limit for drinking water (1 µg/L) and peaked at 17 µg/L, which is above the Oregon Department of Public Health guidelines for issuing public health advisories. Microcystin levels were generally lower in 2007 than 2008, but concentrations at some sites still reached a peak of 6 µg/L. Additional microcystin data collection in Upper Klamath Lake is ongoing (Vanderkooi et al. 2010).

Upper Klamath Lake and Agency Lake are listed as impaired under Section 303(d) of the CWA for chlorophyll-*a* (see Section 3.2, Table 3.2-8). The Klamath Tribes water quality monitoring data from 1990 through 2009 provide chlorophyll-*a* data for Upper Klamath Lake from June through September (Kann 2010). Chlorophyll-*a* concentration varies by location, as related to wind, temperature, pH and nutrients (Morace 2007). Chlorophyll-*a* concentrations measured in 2008 in the relatively sheltered Wocus Bay on the southwest side of the lake, show a series of peaks (1,154 µg/L in mid-July, 862 µg/L in early September) due to concentrated cyanobacteria blooms, whereas concentrations at the mid-lake monitoring station were considerably lower at 116 and 45 µg/L in mid-July and early September, respectively. A correlation between lake mean TP, chlorophyll-*a* and pH was developed by Walker (2001) to develop the TMDL for total phosphorus as the controlling parameter in addressing adverse pH and dissolved oxygen levels in Upper Klamath Lake. Walker (2001) reports a relationship between lake mean TP and chlorophyll-*a* (R^2=0.65) and between coincident measurements of chlorophyll-*a* and lake mean pH (R^2=0.87). Based on this relationship, measured values of pH >9 in Upper Klamath Lake are likely to coincide with chlorophyll-*a* concentrations of >50 µg/L.

C.6.1.3 Link River Dam to Klamath River upstream of J.C. Boyle Reservoir

Multiple years of data characterizing the occurrence of *M. aeruginosa* in the reach from Link River Dam to upstream of J.C. Boyle Reservoir have been collected by PacifiCorp and The Klamath Tribes (Kann 2006). Microcystin data have been collected in this reach only relatively recently (May–December 2009) with concentrations of 0.09–0.66 µg/L (Watercourse Engineering, Inc. 2011). Algal species occurrence is discussed further in Section 3.4, Algae.

The Klamath River from RM 231 to RM 251, including the Keno Impoundment, is listed as impaired under Section 303(d) of the CWA for chlorophyll-*a* (see Section 3.2, Table 3.2-8). Historical (1950–2001) chlorophyll-*a* data indicate median values of 2.8–37 µg/L in the Upper Klamath Basin between Link River Dam and J.C. Boyle Reservoir, with the highest median values occurring at RM 251 in the Keno Impoundment (including Lake Ewauna) (Figure C-26). Variability over the long-term record in this reach is high, with multiple outlying data points above and below the 95th percentile, indicating chlorophyll-*a* levels greater than 100 µg/L at multiple sites in the reach.

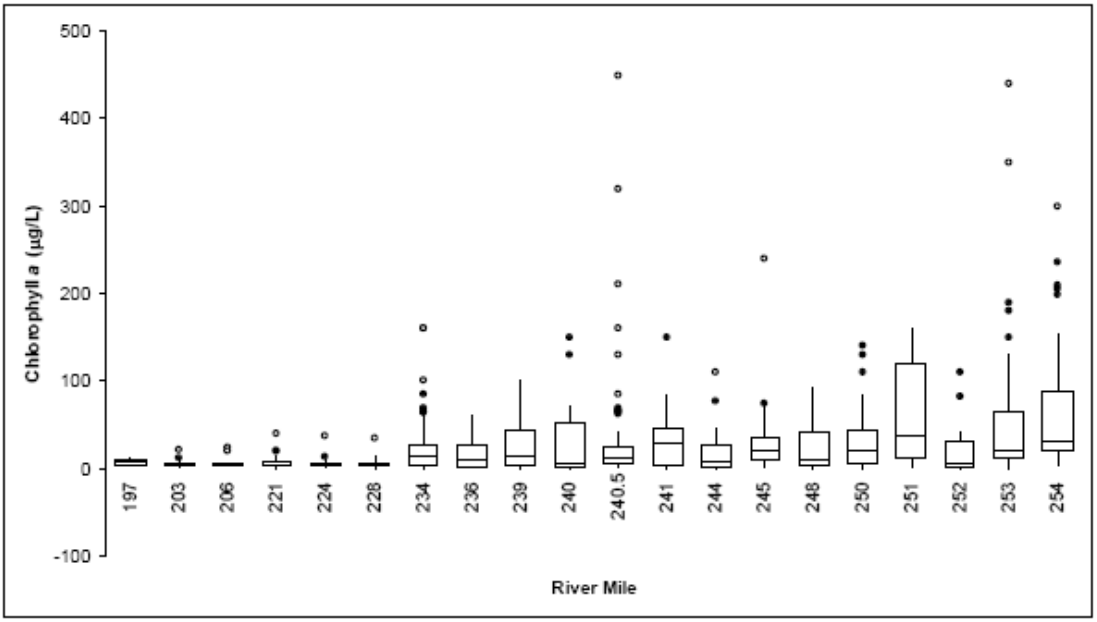

Figure C-26. Box and Whisker Plot of Historical Chlorophyll-*a* Data Collected from Various Sites in the Upper Klamath Basin (Copco 1 Dam Outflow [RM 197] to Link River at Fremont St. Bridge [RM 254]) Between 1950 and 2001. Source: PacifiCorp 2004b.

As with the historical data record, more recent data indicate that high summer chlorophyll-*a* concentrations in the Keno Impoundment (including Lake Ewauna) are due to large populations of algae, predominantly *A. flos-aquae*, entering the Klamath River from Upper Klamath Lake in summer and largely settling out of the water column (FERC 2007, NCRWQCB 2010, Sullivan et al. 2008, et al. 2009, et al. 2010). Chlorophyll-*a* data in the mainstem Klamath River downstream of Upper Klamath Lake follow a seasonal and longitudinal pattern where concentrations tend to be highest (and most variable) at Link River at Klamath Falls (RM 253.1) and decrease toward Keno Dam (RM 235) and the upstream end of J.C. Boyle Reservoir (RM 228.3) (Figures C-27 and C-28). At all locations in this reach, concentrations peak in mid-summer (Sullivan et al. 2008, et al. 2009). For example, for the 2008 growth season, Link River chlorophyll-*a* concentrations range from 9.3 µg/L in May, to 340 µg/L in mid-July, peak in early August at 390 µg/L, and decrease to 8.8 µg/L in November (Figure C-27). Further downstream, in the Keno Impoundment at the Highway 66 bridge, surface chlorophyll-*a* concentrations are considerably lower, peaking at approximately 120 µg/L in mid-July 2008 and 75 µg/L in mid-August, and are generally below 50 µg/L for the rest of the summer. Data reported for 2009 show very high concentrations, with a maximum concentration of 631 µg/L at Link River Dam in early August, 35 µg/L downstream of Keno Dam in late August, and 25 µg/L downstream of J.C. Boyle Dam in late May (Watercourse Engineering, Inc. 2011).

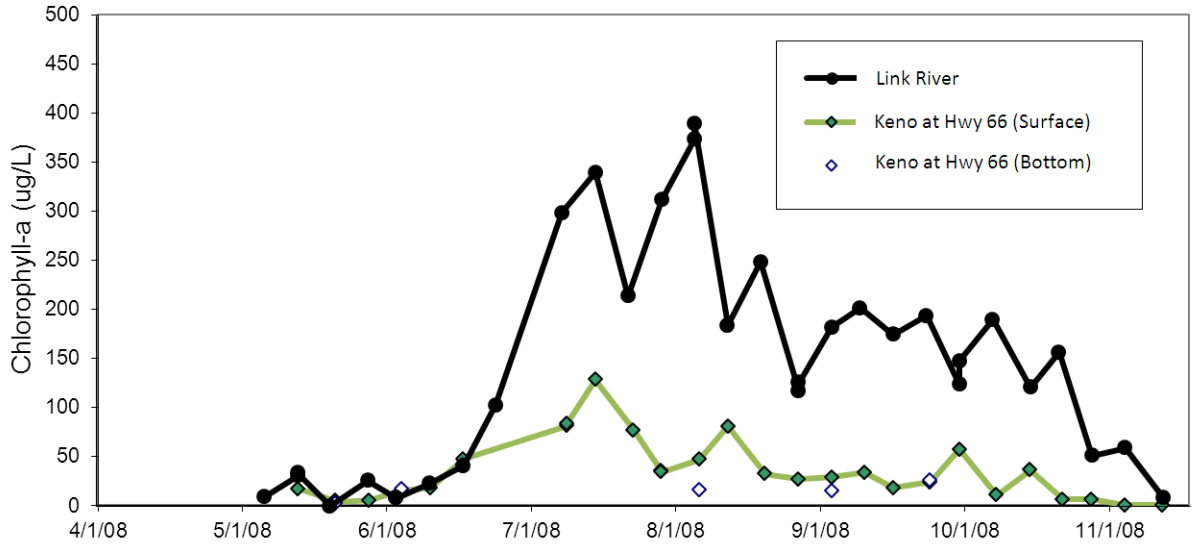

Figure C-27. Decrease in chlorophyll-*a* concentrations from Link River (RM 253.7) to the Keno Impoundment at Highway 66 during May through November, 2008. Graph modified from Appendix B in Sullivan et al. (2009).

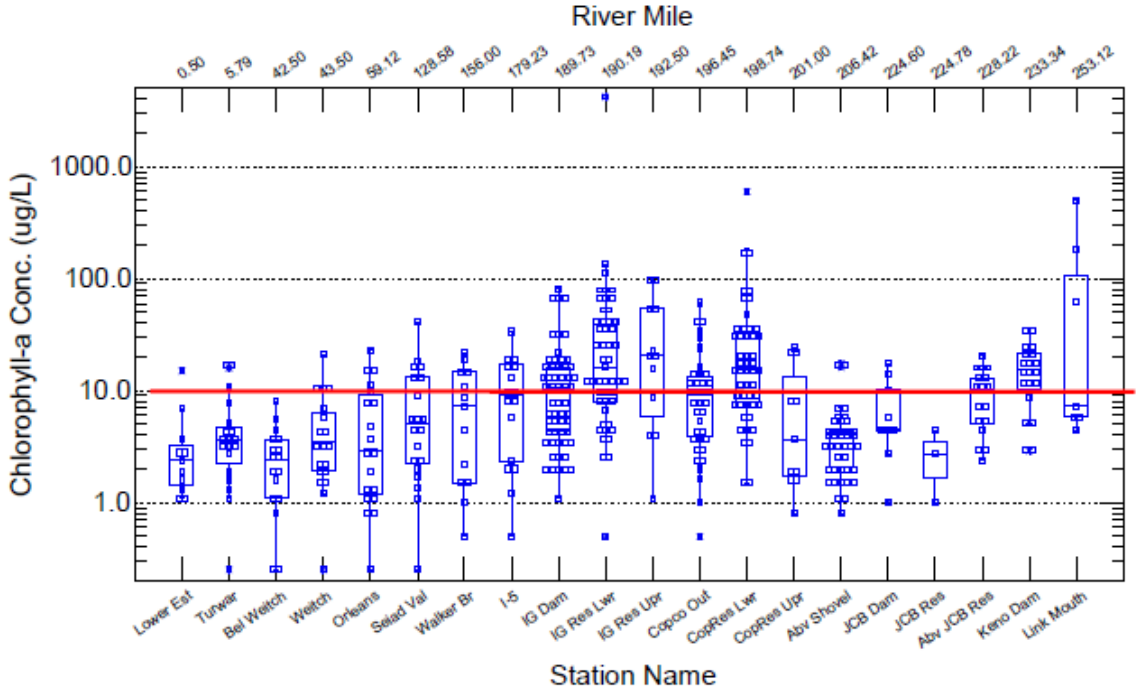

Figure C-28. Longitudinal analysis of summer (May through September) chlorophyll-*a* concentrations from 2005–2007 along the Klamath River. Note the logarithmic scale. Data from the Yurok Tribe, Karuk Tribe of California, North Coast Regional Water Quality Control Board, and PacifiCorp. Source: NCRWQCB 2010.

The seasonal and longitudinal chlorophyll-*a* patterns correspond to patterns measured for algal-derived (organic) suspended material between Link River at Klamath Falls (RM 253.1) and the upstream end of J.C. Boyle Reservoir (RM 228.3) (see Section C.2.1.3), as well as seasonally high SOD and hypoxic dissolved oxygen levels measured in this reach (see Section C.4.1.3).

C.6.1.4 Hydroelectric Reach

Over the past decade, algal toxin and chlorophyll-*a* have become routinely monitored water quality parameters in the Hydroelectric Reach. PacifiCorp chlorophyll-*a* monitoring data for the river upstream of J.C. Boyle Reservoir to just downstream of Iron Gate Dam from 2002 through 2009 (May–October) indicates that annual mean values above 10 µg/L are typical of the dataset and there is generally greater apparent variability upstream of J.C. Boyle Reservoir as compared with just downstream of Iron Gate Reservoir (Raymond 2008, 2009, 2010; PacifiCorp 2004a).

A broader longitudinal analysis of measured chlorophyll-*a* concentrations was conducted using monitoring data compiled during 2005–2007 (May–September) from the Yurok Tribe, Karuk Tribe of California, NCRWQCB, and PacifiCorp (NCRWQCB 2010). Results at numerous locations from the lower Klamath Estuary (RM 0–2) to the Link River Dam (RM 253) demonstrate that median chlorophyll-*a* concentrations within Copco 1 and Iron Gate Reservoirs are 2 to 10 times greater (note the logarithmic scale in Figure C-28) than those documented in free-flowing locations in the mainstem river, with median concentrations greater than 10 µg/L exhibited in the reservoirs and median concentrations less than 10 µg/L exhibited at river locations (NCRWQCB 2010). Upstream, in the Keno Impoundment (including Lake Ewauna), median chlorophyll-*a* concentrations for 2005–2007 are similarly high (i.e., greater than 10 ug/L). Additionally, median chlorophyll-*a* concentrations measured upstream of Copco 1 Reservoir ("Abv Shovel" location in Figure C-28) are greater than those measured downstream of Iron Gate Dam to approximately Seiad Valley ("IG Dam" and "Seiad Val" locations in Figure C-28), suggesting that algal blooms generated in Copco 1 and Iron Gate Reservoirs are exported into the Klamath River downstream of Iron Gate Dam.

Seasonal chlorophyll-*a* patterns in the Hydroelectric Reach indicate that relatively high concentrations can occur during spring diatom blooms (i.e., 30–40 µg/L for Copco 1 and Iron Gate Reservoirs in March 2000–2003), followed by a period of relatively low concentrations, which in previous years (e.g., 2009 and 2010) have included *Anabaena* spp. with sufficient density to require health advisory posting of the reservoirs. A second increase occurs during August and September when dense blooms dominated by both *A. flos-aquae* and *M. aeruginosa* are typical (i.e., 30–60 µg/L for Copco and Iron Gate Reservoir 2000–2003) (FERC 2007).

High chlorophyll-*a* concentrations have also been shown to correlate with the toxigenic cyanobacteria blooms dominated by *Anabaena* spp. and *M. aeruginosa* and sharp increases in microcystin levels above WHO numeric targets (Kann and Corum 2009) and

SWRCB, California Department of Public Health, and Office of Environmental Health and Hazard Assessment (OEHHA) guidelines (Draft Voluntary Statewide Guidance for Blue-Green Algae Blooms [SWRCB 2010b]). Data collected from 2004 through 2009 indicate high levels of microcystin in Copco 1 and Iron Gate Reservoirs, with measured concentrations exceeding the SWRCB/OEHHA public health threshold of 8 µg/L by over 1000 times in Copco Reservoir in 2006, 2007, 2008, and 2009 (Figure C-29) (Kann 2007a–2007d, Kann and Corum 2007 and 2009, Kann et al. 2010a, Jacoby and Kann 2007). Microcystin measured during May–December 2009 at numerous locations in the Klamath River exhibited concentrations less than 1 µg/L, or well below the SWRCB/OEHHA public health threshold of 8 µg/L, at free-flowing river sites from Link River Dam (RM 253) to the Klamath River near Klamath (RM 6) (Watercourse Engineering, Inc. 2011). However, extremely high concentrations (1,000–73,000 µg/L) were measured during algal blooms occurring in July, August, and September in Copco 1 Reservoir in Mallard Cove and Copco Cove, and in Iron Gate Reservoir at Jay Williams (Watercourse Engineering 2011).

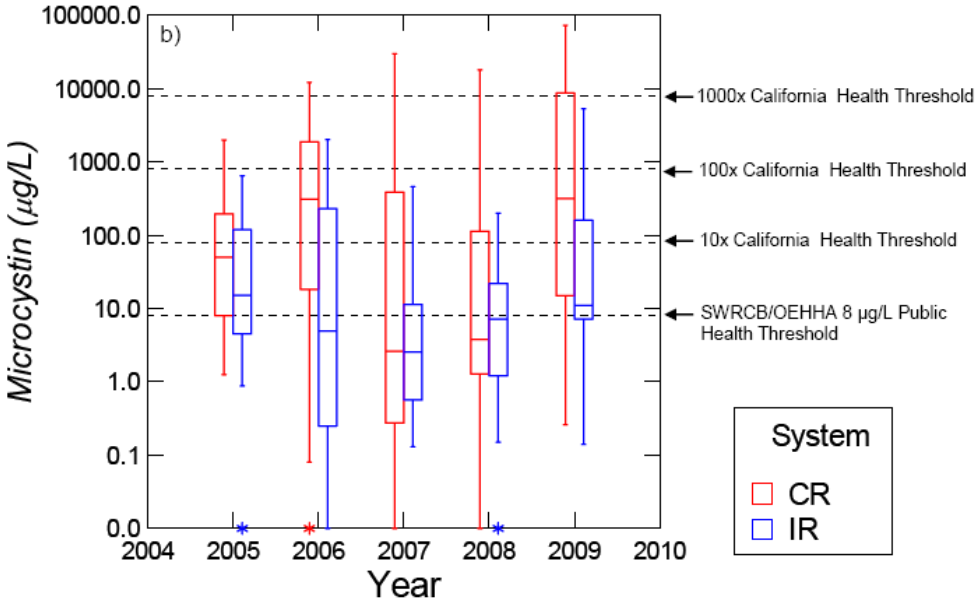

Figure C-29. Inter-annual Comparison of Microcystin Concentration for Copco Reservoir (Red Square) and Iron Gate Reservoir (Blue Square) during July through October 2005–2009. Source: Kann et al. 2010a.

In 2007, a *M. aeruginosa* bloom prompted a Yurok Tribe health advisory along multiple affected reaches in the Klamath River (Kann 2007a–2007d); 85 percent of fish and mussel tissue samples collected during July through September 2007 in the Klamath River, including Iron Gate and Copco 1 Reservoirs, exhibited microcystin bioaccumulation (Kann 2008). Results indicated that all of the WHO total daily intake guideline values were exceeded, including several observations of values exceeding acute total daily intake thresholds (Kann 2008). In a retrospective letter to PacifiCorp (August 6, 2008), the California OEHHA stated that they "would have recommended against consuming mussels from the affected section of the Klamath River, and yellow perch from Iron Gate and Copco Reservoirs, because their average concentrations exceeded 26 nanograms per gram (ng/g)," which is the OEHHA upper bound of advisory tissue levels fish or shellfish consumption (for a single serving per week based on 8 ounces uncooked fish). Data from 2007 also indicate microcystin bioaccumulation in juvenile salmonids reared in Iron Gate hatchery (Kann 2008; see Section 3.3.3.3 Habitat Attributes Expected to be Affected by the Project - Water Quality - Algal Toxins for a discussion of algal toxins as related to fish health). Additional public health advisories were issued in 2009 and 2010 in Copco 1 and Iron Gate Reservoirs, as well as downstream locations in the Klamath River (including locations on the Yurok Reservation), for microcystin levels in ambient and/or freshwater mussel tissue (Kann et al. 2010a, Kann et al. 2010b, Fetcho 2010).

As part of an evaluation of the relationship between *M. aeruginosa* cell density and microcystin concentration, Kann et al. (2010a) compared measured values to the WHO guidelines for a low probability of adverse health effect (20,000 cells/mL *M. aeruginosa*, or 4 µg/L microcystin) and the SWRCB/OEHHA guidelines for protection against a moderate probability of adverse effects (8 µg/L microcystin) for 2009. These results showed that the more conservative guideline of 20,000 cells/mL *M. aeruginosa* decreases the frequency of exceeding the 8 µg/L SWRCB/OEHHA guideline value for microcystin, and is thus more protective of public health. Overall, the 2005–2008 results clearly illustrate that the majority of exceedances to all guidelines and thresholds occurred in the reservoirs in the Hydroelectric Reach (as compared with downstream riverine sites), with the highest overall levels measured in Copco 1 Reservoir (Figure C-30) (Kann and Corum 2009).

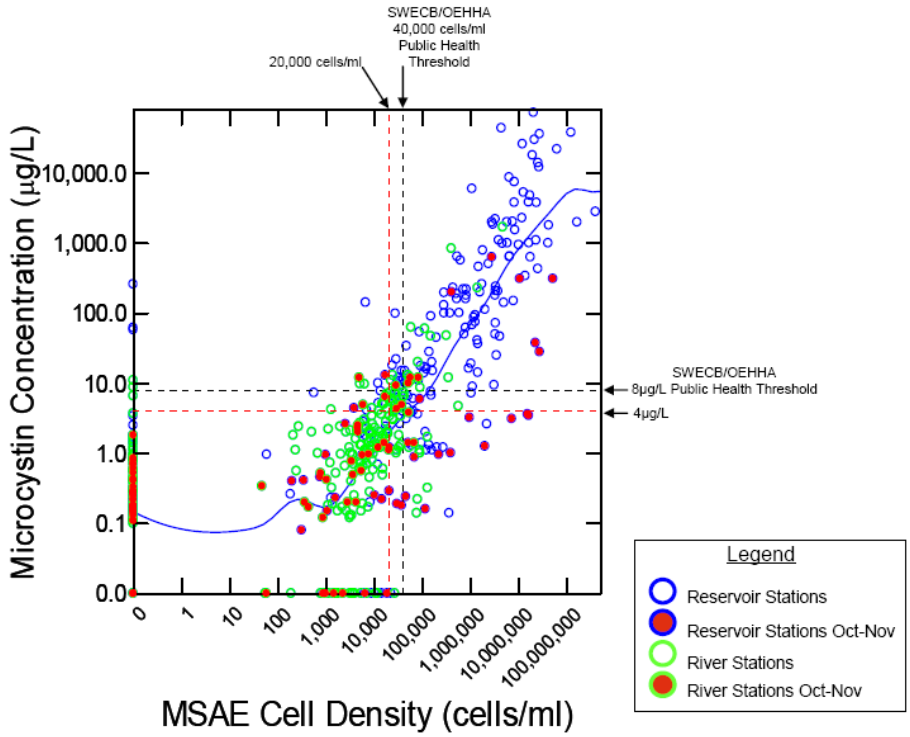

Figure C-30. Relationship between *Microcystis aeruginosa* Cell Density and Microcystin Toxin Concentration for Copco 1 and Iron Gate Reservoirs and Klamath River Stations 2005–2009. Source: Kann et al. 2010a.

C.6.2 Lower Klamath Basin

C.6.2.1 Iron Gate Dam to Salmon River

As discussed above (Section C.6.1.4), 2005–2007 data indicate that during May through September median chlorophyll-*a* concentrations decreased longitudinally with distance downstream of Iron Gate Dam (Figure C-28) and were greater than concentrations measured just upstream of Copco 1 Reservoir. This indicates that algal blooms occurring in the reservoirs were being transported to the downstream river reaches. Ward and Armstrong (2010) report mean annual chlorophyll-*a* concentrations at 5 µg/L for 2001-2005, with concentrations also generally decreasing with distance downstream of Iron Gate Dam (Figure C-31). The highest annual mean value (≈5 µg/L) occurred in 2005 at the confluence with the Shasta River (RM 176.7). In 2009, the Karuk Tribe collected chlorophyll-*a* and pheophytin-*a* (an additional photosynthetic pigment) data from the Klamath River downstream of Iron Gate Dam; chlorophyll-*a* values were approximately 1–35 µg/L and were variable depending on location. Generally speaking, relatively greater values were observed at upstream locations near Iron Gate Dam (RM 190.1) and Walker Bridge sites, but the peak value was observed farther downstream at Orleans (RM 59) (Karuk Tribe of California 2010).

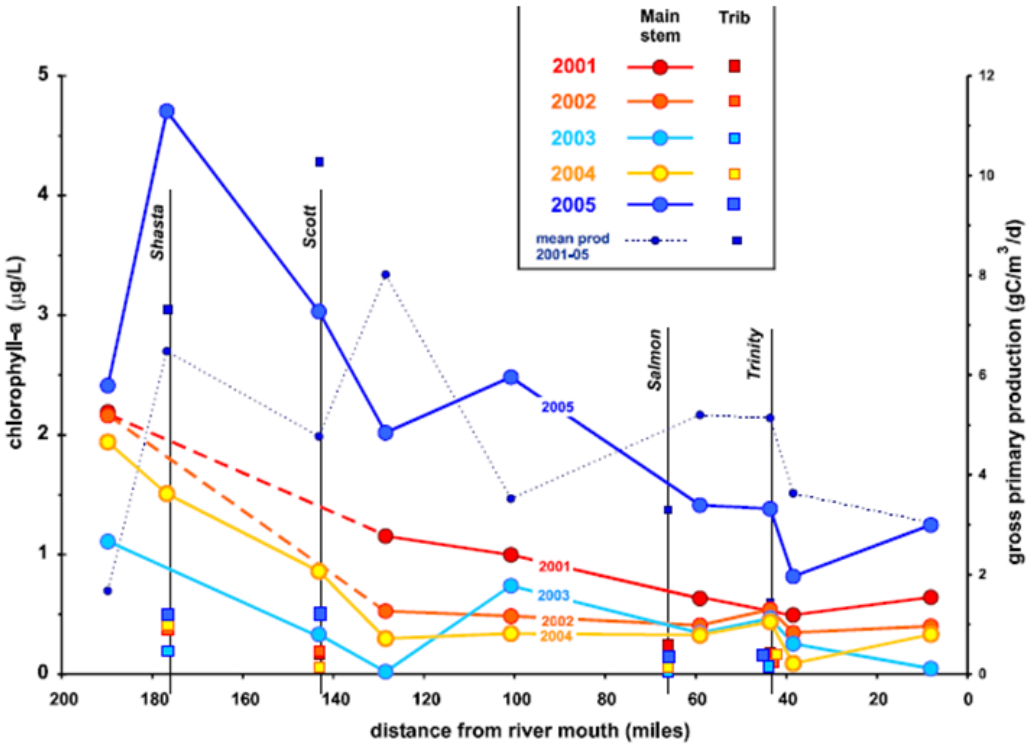

Figure C-31. Annual mean values of chlorophyll-*a* in the Klamath River downstream of Iron Gate Dam during June–September 2001–2005. Source: Ward and Armstrong (2010).

In 2008, the Karuk Tribe collected cyanobacteria concentration (cells/mL) using optical phycocyanin probes to allow more timely assessment of public health threats from toxigenic algal species. Data from downstream of Iron Gate Dam collected during June–October indicated peak values (>25,000 cells/mL) in July and early-to-mid September (Karuk Tribe of California 2009).

Although concentrations of both *M. aeruginosa* and microcystin toxin in the Klamath River downstream of the Hydroelectric Reach are lower relative to the reservoirs (Figure C-32), WHO guidelines for exposure to microcystin (i.e., < 4 µg/L) have been exceeded downstream of Iron Gate Dam on numerous occasions (Kann 2004, Kann and Corum 2009, Kann et al. 2010a, Fetcho 2010), including late-summer/early-fall *M. aeruginosa* blooms in September 2007, 2009, and 2010 from Iron Gate Dam (RM 190.1) to the mouth of the Klamath River (RM 0.0). Health Advisories were posted along this reach of the Klamath River (Iron Gate Dam to Shasta River in 2009 and 2010, due to elevated microcystis cell counts and/or microcystin concentrations in river water. Available data

indicate that algal blooms in Iron Gate and Copco Reservoirs have been responsible for the public health exceedances in the lower river (Kann and Corum 2009).

Additionally, data from 2007 indicate microcystin bioaccumulation in juvenile salmonids reared in Iron Gate hatchery (Kann 2008) and, in 2010, algal toxins were found in salmonid tissues collected near Happy Camp (Kann et al. 2011) (see Section 3.3.3.3 Habitat Attributes Expected to be Affected by the Project - Water Quality - Algal Toxins for a discussion of algal toxins as related to fish health).

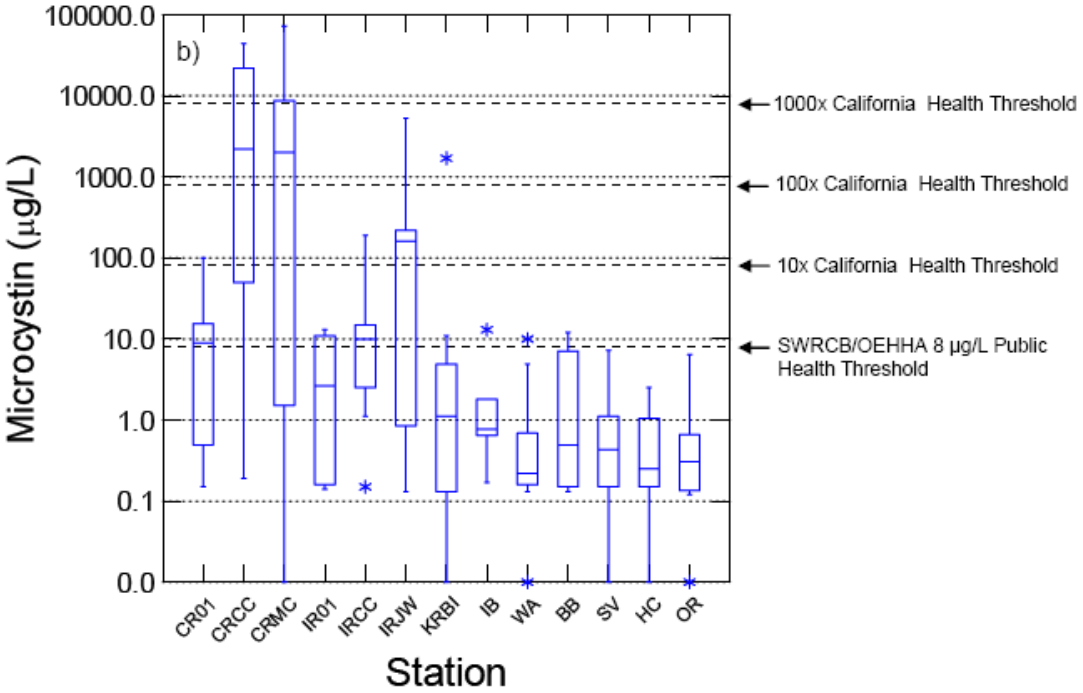

Figure C-32. Microcystin Concentration in Klamath River from Copco 1 (CR01) to Orleans (RM 59) during June–November, 2009. WA=Walker Bridge, SV=Seiad Valley (RM 128), OR=Orleans (RM 59). Source: Kann et al. 2010a.

C.6.2.2 Salmon River to Estuary

Downstream of the confluence with the Salmon River (RM 66.0), algal toxin and chlorophyll-*a* concentrations exhibit variability and are generally lower than those measured farther upstream. During 2009, mean microcystin concentrations from Orleans (RM 57) to Klamath River at Klamath (RM 6.0) were less than 1 µg/L, or well below the SWRCB/OEHHA public health threshold of 8 µg/L (Watercourse Engineering, Inc. 2011). Individual microcystin measurements generally remained less than 1 µg/L as well, with the exception of a sample collected in late-September at Orleans (RM 59.1) for which the concentration was just over 6 µg/L (Watercourse Engineering, Inc. 2011).

For several years, the Yurok Tribe has measured chlorophyll-*a*, cyanobacteria concentrations (cells/mL), and has used optical phycocyanin probes to allow more timely assessment of public health threats from toxigenic algal species on the Yurok Reservation. Chlorophyll-*a* data measured from Weitchpec (RM 43.5) to the estuary during 2003–2004 indicate that, where detectable, concentrations were generally below 5 µg/L. From May–July during 2006–2010 chlorophyll-*a* concentrations were consistently less than 5 µg/L, but increased in during August–October to approximately 20–30 µg/L (Sinnott 2008, 2009, 2010, 2011). Peak values on the Yurok Reservation consistently occurred at Weitchpec (RM 43.5) from late August to mid-October and varied by year (6.7 µg/L in 2008 and 26 µg/L in 2009). In contrast, 2001–2005 chlorophyll-*a* data from Ward and Armstrong (2010) show small relative increases in chlorophyll-*a* with distance downstream, from near the Trinity River confluence (RM 40) to Turwar (RM 5.8), (Figure C-31), suggesting that algal productivity may increase slightly as water moves toward the Klamath Estuary.

As described for the Klamath River from Iron Gate Dam to the Salmon River (Section C.6.2.1), there have been numerous exceedances of public health guidelines in the Klamath River from the Salmon River confluence to the Klamath Estuary, particularly in 2010. Public health advisories were issued in 2009 and 2010 in this reach (including locations on the Yurok Reservation) for elevated microcystin levels in ambient and/or freshwater mussel tissue samples (Kann et al. 2010a, Kann et al. 2010b, Fetcho 2010). In addition, substantial bioaccumulation (exceeding public health guidelines) of microcystin in freshwater mussels has been shown in this reach (Kann 2008, Kann et al. 2010b). See Section 3.3.3.3 Habitat Attributes Expected to be Affected by the Project - Water Quality - Algal Toxins for a discussion of algal toxins as related to fish health.

C.6.2.3 Klamath Estuary

Chlorophyll-a and algal toxin levels in the Klamath Estuary are generally similar to those measured at stations just upstream. From May–July during 2006–2010, chlorophyll-a concentrations in the Klamath Estuary were consistently less than 5 ug/L, increasing slightly during August–October to approximately 1–8ug/L (Sinnott 2008, 2009a, 2009b, 2010, 2011). Peak concentrations of chlorophyll-a during 2006-2010 occurred during late-August to mid-October and varied by year (2.4 ug/L in 2006, 8ug/L in 2009 and 2010).

Algal toxin concentrations in the estuary are generally low, corresponding to relatively low concentrations of *M. aeruginosa*. Exceptions to this include in September 2007 and 2010 when the Yurok Tribe issued advisories because M. aeruginosa concentrations exceeded 40,000 cells/mL, by more than a factor of 2, and in one additional instance in September 2005, concentrations exceeded the WHO guideline for low risk recreational use (20,000 cells/mL). These elevated levels of *M. aeruginosa* corresponded with elevated levels measured farther upstream in Copco 1 and Iron Gate Reservoirs, indicating that *M. aeruginosa* was being transported into the estuary from upstream reservoir blooms (Kann and Corum 2006). See Section 3.3.3.3 Habitat Attributes

Expected to be Affected by the Project - Water Quality - Algal Toxins for a discussion of algal toxins as related to fish health. Lastly, there is emerging evidence that cyanotoxins flushing from coastal rivers into Monterey Bay, California were responsible for numerous sea otter deaths in 2007 (Miller et al. 2010). While it is not known if conditions in Monterey Bay are similar to those in the Klamath River marine nearshore environment, there may be potential for microcystin to adversely impact marine organisms when large blooms are transported through the Klamath Estuary and into the Pacific Ocean.

C.7 Inorganic and Organic Contaminants

C.7.1 Upper Klamath Basin

In general, information regarding contaminants in the Upper Klamath Basin upstream of the Hydroelectric Reach is unavailable. Human activities such as illegal dumping may be a source of inorganic and organic contaminants to the lower Sprague and Williamson river sub-basins (Rabe and Calonje 2009). The exception to this is arsenic; natural geologic sources of arsenic may be causing relatively high levels of this chemical element in the Upper Klamath Basin, as is the case other south central and southeastern Oregon basins (Sturdevant 2010). Recently collected data from a limited number of locations indicate arsenic levels of <1 µg/L to >10 µg/L in surface waters of the Upper Klamath Basin. Limited data are available partly because 5.0 µg/L was the established quantitative limit for the state of Oregon until 2008. It is not known whether these levels represent solely natural geologic sources or are elevated due to anthropogenic activity (Sturdevant 2010).

C.7.1.1 Hydroelectric Reach

Water Column Contaminants

Existing water quality data are available from the California Surface Water Ambient Monitoring Program (SWAMP), which collected water quality data, including inorganic and organic contaminant data, from 2001 through 2005 at eight monitoring sites from the California-Oregon state line (RM 208.5) to Klamath River at Klamath Glen (RM 5.8) (NCRWQCB 2008). Results from the state line site indicated that for the majority of inorganic constituents (i.e., arsenic, cadmium, chromium, copper, lead, mercury, nickel, selenium, silver, and zinc), concentrations were in compliance with water quality objectives at the time of sampling. Aluminum concentrations (50.7–99.2 µg/L) may have exceeded the United States Environmental Protection Agency (USEPA) continuous concentration for freshwater aquatic life protection (87 µg/L) on two of four site visits (50 percent exceedance rate), and exceeded the USEPA secondary Maximum Contaminant Level (MCL) for drinking water (50 µg/L) on all four site visits (100 percent exceedance rate) (NCRWQCB 2008). Grab samples were analyzed for 100 pesticides, pesticide constituents, isomers, or metabolites; 50 polychlorinated biphenyls (PCBs) congeners; and 6 phenolic compounds. Results indicated no PCBs detections, but

one detection of dichlorodiphenyldichloroethylene (1,1-bis-(4-chlorophenyl)-2,2-dichloroethene or DDE) (25 percent of samples) and one detection of trans-nonachlor (25 percent of samples) were found (NCRWQCB 2008).

Sediment Contaminants

To investigate the potential for toxicity of the sediments trapped behind the KHP reservoirs, Shannon & Wilson, Inc. (2006) collected 26 cores from J.C. Boyle, Copco 1, and Iron Gate Reservoirs during 2004–2005 and analyzed them for contaminants including acid volatile sulfides, metals, pesticides, chlorinated acid herbicides, PCBs, volatile organic compounds (VOCs), semi-volatile organic compounds (SVOCs), cyanide, and dioxins. No herbicides or PCBs were found above screening levels and only one sample exceeded Puget Sound Dredge Disposal Analysis screening levels for VOCs ethyl benzenes and total xylenes (Shannon & Wilson, Inc. 2006). While cyanide was detected in multiple sediment cores, it was not found in toxic free cyanide form (HCN or CN^-), and is not likely to be bioavailable or result in adverse effects on fish and other aquatic biota.

Dioxin, a known carcinogen, was measured in three samples from J.C. Boyle, Copco 1, and Iron Gate Reservoirs. Long-term exposure to dioxin in humans is linked to impairment of the immune system, the developing nervous system, the endocrine system and reproductive functions. In the 2004–2005 reservoir samples, measured levels were 2.48–4.83 pg/g (picograms per gram or parts per trillion [ppt] expressed as Toxic Equivalent Concentrations)) and did not exceed Puget Sound Dredged Disposal Analysis screening levels (Shannon & Wilson, Inc. 2006). More comprehensive reviews of dioxin guidelines and sediment studies from watersheds outside of the Klamath basin were conducted by Dillon (2008) and USEPA (2010b), the latter presenting an estimate of background dioxin concentrations (2–5 ppt) for non-source-impacted sediments throughout the U.S. and specifically in the western U.S. (USEPA 2010). Based on the information presented in (USEPA 2010), in addition to being within the range of natural background, Klamath dioxin sediment levels reported by Shannon & Wilson (2006) are one to three orders of magnitude below risk-based USEPA (1,000 pg/g dry weight [DW], toxicity equivalent quotient [TEQ]) preliminary remediation goals in residential soils, and Washington Department of Ecology (11 pg/g DW TEQ) for residential soil clean-up levels (USEPA 2010). They are also generally an order of magnitude below USEPA effects-based ecological receptors thresholds (60–100 pg/g DW TEQ for fish; 2.5–25 pg/g DW TEQ for mammals; 21–210 pg/g DW TEQ for birds). Oregon human health thresholds include risk-based values for subsistence fishers as well as the general consuming public, and hence these values are quite a bit lower (0.0011–1.1 pg/g DW TEQ), with the low end of the range applicable to subsistence fishers (ODEQ 2007). Oregon's dioxin wildlife thresholds include levels at which dioxin would bioaccumulate for the protection of wildlife consumers (0.56 pg/g DW TEQ for fish; 0.052–1.4 pg/g DW TEQ for mammals; 0.7–3.5 pg/g DW TEQ for birds), which are also generally lower than the USEPA effects-based thresholds.

While the existing sediment data (Shannon & Wilson, Inc. 2006) did not indicate a high risk of sediment toxicity, it was not sufficient to evaluate all analytes of interest. Thus, as part of the Secretarial Determination studies, a sediment evaluation was undertaken during 2009–2011 to provide a more comprehensive data set to further guide decision makers in an evaluation of potential impacts from dam removal. The Bureau of Reclamation (Reclamation) and USFWS plan (report in process) expanded the number of sediment cores and the analyte suite examined, including chemicals likely to bioaccumulate, and included biological and elutriate tests (Reclamation 2010). For this evaluation, establishment of toxicity and/or bioaccumulative potential for sediment contaminants relied upon thresholds developed through regional and state efforts such as the Sediment Evaluation Framework (SEF) for the Pacific Northwest Oregon and ODEQ bioaccumulation screening level values (SLVs). Sediment cores were collected at multiple sites and at various sediment depths per site in J.C. Boyle Reservoir (n=26), Copco 1 Reservoir (n=25), Iron Gate Reservoir (n=24), and the Klamath Estuary (n=2), for a total of 77 samples (Department of the Interior 2010). A total of 501 analytes were quantified across the samples, including metals, poly-aromatic hydrocarbons (PAHs), PCBs, pesticides/herbicides, phthalates, VOCs, SVOCs, dioxins, furans, and polybrominated diphenyl ethers (PBDEs) (i.e., flame retardants). Samples were analyzed for sediment chemistry and elutriate (pore water) chemistry, and bioassays and bioaccumulation studies were conducted on the sediment and elutriate using fish and invertebrate national benchmark toxicity species. Using results of these analyses, the following five exposure scenarios were evaluated under Level 2A and 2B of the SEF using multiple lines of evidence: (CDM 2011):

- Scenario 1 – No Action Alternative - Long-term sediment exposure for aquatic biota and humans (via fish consumption) to reservoir sediments.

- Scenario 2 – Proposed Action - Short-term water column exposure for aquatic biota from sediments flushed downstream (suspended sediments, not a bioaccumulation issue).

- Scenario 3 – Proposed Action - Long-term sediment exposure for riparian biota and humans from reservoir terrace deposits and river bank deposits (terrestrial exposures).

- Scenario 4 – Proposed Action - Long-term sediment exposure for aquatic biota and humans from river bed deposits (aquatic exposures).

- Scenario 5 – Proposed Action - Long-term sediment exposure for aquatic biota from estuary and marine near shore deposits.

Based on comparisons of sediment chemistry to screening levels (SLs) and the results of bioassays (see Section C.7.1.1. for more detail), the reservoir sediments do not appear to be highly contaminated. No consistent pattern of elevated chemical composition was observed across discrete sampling locations within a reservoir and no single reservoir was observed to be consistently more or less contaminated. Sediment in all three reservoirs exceeded ecological SLs for nickel, iron, and 2,3,4,7,8-PECDF. Several pesticides and

semi-volatile organic compounds (SVOCs) were not detected; yet, the reporting limits were above the SLs, so other lines of evidence were used to asses these compounds. Similarly, human health SLs were only exceeded for arsenic and nickel, and some legacy pesticides and dioxin-like compounds exceeded the ODEQ Bioaccumulation SLVs. Several pesticides and SVOCs were not detected; yet, the reporting limits were above SLs. Sediment in J.C. Boyle Reservoir does have marginally higher chemical concentrations and more detected chemicals of potential concern (COPCs) as compared to Copco 1 and Iron Gate Reservoir and Klamath Estuary sediments (CDM 2011).

Toxicity tests generally indicated low potential for sediment toxicity to benchmark benthic indicator species; the exception to this occurred in a single sample from J.C. Boyle Reservoir, where survival of the benthic amphipod *Hyalella azteca* indicated a moderate potential for sediment toxicity (CDM 2011). TEQs were calculated for dioxin, furan, and dioxin-like PCBs in reservoir sediment samples to evaluate potential adverse effects from exposure to dioxin, furan, and dioxin-like PCBs. TEQs ranged from approximately 4-9 pg/g for J.C. Boyle Reservoir, 5-10 pg/g for Copco 1 Reservoir, and 2-4 pg/g for Iron Gate Reservoir. In some cases these values are slightly higher than background values reported by USEPA for Region 9 (i.e., 2-5 pg/g), Region 10 (i.e., 4 pg/g), and for non-impacted lakes of the United States (i.e., 5.3 pg/g) (USEPA 2010, CDM 2011). The calculated TEQs may also be within the range of local background values. Since the TEQs are only slightly above regional background concentrations and the nationwide background for non-impacted lakes, they have limited potential for adverse effects for fish exposed to reservoir sediments (CDM 2011).

Lastly, sediment samples were also evaluated for levels of known bioaccumulative compounds; ODEQ bioaccumulation sediment SLVs were not exceeded in J.C. Boyle Reservoir sediments, with the exception of a small number of samples for DDTs (i.e., 4,4’-DDT, 4,4’-DDD, 4,4’-DDE) (CDM 2011).

Overall, using multiple lines of evidence from the 2009-2010 Secretarial Determination study, sediment quality of reservoir sediments does not appear to be highly contaminated. No consistent pattern of elevated chemical composition is observed across discrete sampling locations within a reservoir. No single reservoir is observed to be consistently more or less contaminated based on multiple lines of evidence. Where elevated concentrations of chemicals in sediment are found, the degree of exceedance based on comparisons of measured (i.e., detected) chemical concentrations to SLs is small and in several cases may reflect regional background conditions (CDM 2011).

Contaminants in Aquatic Biota

Separate assessments of contaminants in fish tissue for the Hydroelectric Reach have been undertaken by SWAMP and PacifiCorp. SWAMP data include sport fish tissue samples collected during 2007 and 2008 to evaluate accumulated contaminants in nearly 300 lakes statewide. Sport fish were sampled to provide information on potential human exposure to selected contaminants and to represent the higher aquatic trophic levels (i.e., the top of the aquatic food web).

In the Hydroelectric Reach, fish tissue samples were collected in Copco 1 and Iron Gate Reservoirs and analyzed for total mercury, selenium, and PCBs (Iron Gate Reservoir only) (Davis et al. 2010). SWAMP data for Iron Gate and Copco reservoirs (Table C-5) indicate mercury tissue concentrations above the USEPA criterion of 300 ng/g methylmercury in fish tissue to protect the health of consumers of noncommercial freshwater fish; and greater than the OEHHA public health guideline levels advisory tissue level (Klasing and Brodberg 2008) for consumption for 3 and 2 servings per week (70 and 150 ng/g wet weight, respectively) and the fish contaminant goal (220 ng/g wet weight). Measured selenium concentrations were 3–4 orders of magnitude lower than OEHHA thresholds of concern (2,500–15,000 ng/g wet weight) and PCB concentrations were below the lowest OEHHA threshold (i.e., fish contaminant goal of 3.6 ng/g wet weight) (Davis et al. 2010).

Table C-5. Total Mercury, Selenium, and PCBs in (ng/g wet weight) in Largemouth Bass taken from Iron Gate and Copco 1 Reservoirs During 2007–2008 (Davis et al. 2010).

Contaminant	Species	Iron Gate Reservoir	Copco 1 Reservoir
Methylmercury	Largemouth Bass (LMB)	330	310
Selenium	LMB	80	80
PCBs	LMB	1.31	*Not reported*

In a screening-level study of potential chemical contaminants in fish tissue in Keno, J.C. Boyle, Copco, and Iron Gate Reservoirs, and in Upper Klamath Lake, PacifiCorp analyzed metals (i.e., arsenic, cadmium, chromium, copper, lead, mercury, nickel, selenium, and zinc), organochlorine (pesticide) compounds, and PCBs in largemouth bass (*Micropterus salmoides*) and black bullhead catfish (*Ameiurus melas*) (PacifiCorp 2004c). PacifiCorp reported that, in general, contaminant levels in fish tissue are below both screening level values for protection of human health (USEPA 2000) and recommended guidance values for the protection of wildlife (MacDonald 1994). Exceptions to this include measured fish tissue levels of total mercury in samples from Copco 1 and Iron Gate Reservoirs as compared to the wildlife screening level of 0.00227 ug/g and measured fish tissue levels of arsenic (<0.3 ug/g) that PacifiCorp indicated may equal or exceed the toxicity screening level for subsistence fishers (0.147 ug/g) in samples of largemouth bass from J.C. Boyle, Copco 1, and Iron Gate Reservoirs. Subsequent reanalysis of the PacifiCorp mercury tissue data indicates that all tissue samples exceed the most protective wildlife screening level of 0.00227 ug/g, samples from Keno, J.C. Boyle, Copco 1, and Iron Gate Reservoirs exceed the screening level for subsistence fishers (0.049 ug/g), and samples from Copco 1 and Iron Gate Reservoirs exceed the screening level for recreational fishers (0.4 ug/g) (Table C-6).

Table C-6. Total Mercury Concentrations (ug/g wet weight) in Black Bullhead (BB) and Largemouth Bass (LMB) Composite Tissue Samples taken from Project Reservoirs and Upper Klamath Lake in 2003 (PacifiCorp 2004c).

Sample	Composite	Site	Species	Total Mercury (ug/g wet weight)[1]
L-262-03	1F	Keno Impoundment	BB	0.121
L-262-03	1F Duplicate	Keno Impoundment	BB	0.125
L-262-03	2F	J.C. Boyle Reservoir	LMB	0.153
L-262-03	3F	J.C. Boyle Reservoir	LMB	0.190
L-273-03	1F	Iron Gate Reservoir	LMB	0.564
L-273-03	2F	Iron Gate Reservoir	LMB	0.508
L-273-03	3F	Copco 1 Reservoir	LMB	0.563
L-273-03	4F	Copco 1 Reservoir	LMB	0.389
L-484-03	1F	Upper Klamath Lake	BB	0.031
L-484-04	2F	Upper Klamath Lake	BB	0.035
		Method Detection Limit		0.003[2]
		Method Reporting Limit		0.007[2]
		Screening Levels[3]:		
		Recreational fishers		0.4
		Subsistence fishers		0.049
		Wildlife		0.00227

[1] PacifiCorp (2004c) total mercury data was provided in ng/g dry weight. Data was converted to ug/g wet weight using percent moisture data provided for each sample by Moss Landing Marine Laboratory (A. Bonnema, pers. comm., 17 February 2011).

[2] The Method Detection Limit and Reporting Limit were converted from dry weight to wet weight using an average of the percent moisture data for all samples.

[3] Screening Levels (SLs) are numeric chemical guidelines that are used to assess and characterize the potential toxicity or bioaccumulative nature of environmental samples (i.e., sediments, water, organism tissue).

Additionally, PacifiCorp indicated that some of the fish tissue samples from Upper Klamath Lake, Keno Impoundment, J.C. Boyle Reservoir, and Copco 1 Reservoir exceed the suggested wildlife screening value for total DDTs (Table C-7) (DDE,p,p' was detected; however DDT and DDD were not detected in the study), and total PCB values exceed the screening level for subsistence fishers in black bullhead from Keno Impoundment, and in largemouth bass from J.C. Boyle, Copco 1, and Iron Gate Reservoirs (Table C-8). Dioxins were not tested.

Table C-7. Total DDE Concentration (ng/g) in Black Bullhead (BB) and Large Mouth Bass (LMB) Composite Tissue Samples taken from Project Reservoirs and Upper Klamath Lake in 2003 (PacifiCorp 2004c).

Sample	Composite	Site	Species	DDE,p,p' (ng/g wet weight)
L-262-03	1F	Keno Impoundment	BB	2.41
L-262-03	2F	J.C. Boyle Reservoir	LMB	<2.00
L-262-03	2F Duplicate	J.C. Boyle Reservoir	LMB	<2.00
L-262-03	3F	J.C. Boyle Reservoir	LMB	2.91
L-273-03	1F	Iron Gate Reservoir	LMB	<2.00
L-273-03	2F	Iron Gate Reservoir	LMB	<2.00
L-273-03	3F	Copco Reservoir	LMB	2.16
L-273-03	4F	Copco Reservoir	LMB	<2.00
L-484-03	1F	Upper Klamath Lake	BB	<2.00
L-484-04	2F	Upper Klamath Lake	BB	2.32
		Method Detection Limit		0.56
		Method Reporting Limit		2
		Screening Levels[1] (for Total DDTs):		
		Recreational fishers		117
		Subsistence fishers		14.4
		Wildlife		0.2–1.07

[1] Screening Levels (SLs) are numeric chemical guidelines that are used to assess and characterize the potential toxicity or bioaccumulative nature of environmental samples (i.e., sediments, water, organism tissue).

Table C-8. Total PCB Concentrations (ng/g) in Black Bullhead (BB) and Large Mouth Bass (LMB) Composite Tissue Samples taken from Project Reservoirs and Upper Klamath Lake in 2003 (PacifiCorp 2004c).

Sample	Composite	Site	Species	Total PCB (ng/g wet weight)
L-262-03	1F	Keno Impoundment	BB	2.926
L-262-03	2F	J.C. Boyle Reservoir	LMB	0.885
L-262-03	2F Duplicate	J.C. Boyle Reservoir	LMB	1.397
L-262-03	3F	J.C. Boyle Reservoir	LMB	3.521
L-273-03	1F	Iron Gate Reservoir	LMB	6.574
L-273-03	2F	Iron Gate Reservoir	LMB	4.909
L-273-03	3F	Copco Reservoir	LMB	2.822
L-273-03	4F	Copco Reservoir	LMB	2.158
L-484-03	1F	Upper Klamath Lake	BB	0.846
L-484-04	2F	Upper Klamath Lake	BB	2.015
		Method Detection Limit		Varies
		Method Reporting Limit		0.2
		Screening Levels[1]:		
		Recreational fishers		20
		Subsistence fishers		2.45
		Wildlife		100

[1] Screening Levels (SLs) are numeric chemical guidelines that are used to assess and characterize the potential toxicity or bioaccumulative nature of environmental samples (i.e., sediments, water, organism tissue).

To supplement existing fish tissue data and provide additional lines of evidence in the Secretarial Determination sediment evaluation, the potential for chemicals in sediment and elutriate samples to bioaccumulate in aquatic species at concentrations above screening levels for ecological receptors (i.e., fish, birds, humans/mammals) was investigated (CDM 2011). Bioaccumulation studies were conducted using laboratory invertebrates (Asian clams, *Corbicula fluminea*; and Black worms, *Lumbriculus variegates*) exposed to reservoir-derived sediments and two species of field-caught fish (yellow perch and bullhead) collected during late September 2010 from J.C. Boyle, Copco 1, and Iron Gate Reservoirs (Reclamation/USFWS in prep, see Section C.7.1.1 for more detail). Results indicate that multiple chemicals were found in invertebrate tissue (i.e., acenaphthene, arsenic, benzo(a)pyrene, DDD/DDE, endosulfan I, endosulfan II, endosulfan sulfate, fluoranthene, hexachlorobenzene, lead, mercury, phenanthrene, pyrene, total PBDEs, total PCBs). Of these detected chemicals, only fluoranthene possesses a toxicity reference value (TRV) for the species tested; exceedances of the fluoranthene TRV were only identified above the No Effect TRV, and were below the Low Effect TRV. Tissue-based TRVs are unavailable for the remaining invertebrate chemicals detected, and hexachlorobenzene has no tissue-based TRVs (i.e., for any species). Multiple chemicals were found in fish tissue (i.e., 2,3,7,8-TCDD, arsenic, DDE/DDT, dieldrin, endrin, mercury, mirex, selenium, and total PCBs) as well (CDM 2011). Mercury exceeded tissue-based TRVs for perch in Iron Gate Reservoir and bullhead samples in all three reservoirs (CDM 2011). TRVs are not available the remaining several chemicals detected in yellow perch and bullhead samples.

C.7.2 Lower Klamath Basin

C.7.2.1 Iron Gate Dam to Salmon River

Water Column Contaminants

SWAMP collected water quality data for inorganic and organic contaminants from 2001 through 2005 at eight monitoring sites from the California-Oregon state line (RM 208.5) to Klamath River at Klamath Glen (RM 5.8) (NCRWQCB 2008). As was the case for the SWAMP state line site (Section C.7.1.1), results for the four sites in the reach from Iron Gate Dam to the Salmon River indicated that with the exception of aluminum, all other measured concentrations of inorganic constituents (i.e., arsenic, cadmium, chromium, copper, lead, mercury, nickel, selenium, silver and zinc) were in compliance with all water quality objectives at the time of sampling. Aluminum concentrations (26.30–280.0 µg/L) potentially exceeded the USEPA continuous concentration for freshwater aquatic life protection (87 µg/L) on 23 of 59 site visits (39 percent exceedance rate), exceeded the USEPA secondary MCL for drinking water (50 µg/L) on 37 site visits (63 percent exceedance rate), and exceeded the California Department of Health Services secondary MCL for drinking water (200 µg/L) on five site visits (8 percent exceedances rate) (NCRWQCB 2008).

Sediment Contaminants

Sediment data for inorganic and organic contaminants in the Klamath River from Iron Gate Dam to the Salmon River are not readily available, nor are fish tissue analyses for contaminants in the lower Klamath River.

C.7.2.2 Salmon River to Estuary

Water Column Contaminants

SWAMP collected water quality data for inorganic and organic contaminants from 2001 through 2005 at three monitoring sites in this reach of the Klamath River to Klamath Glen (RM 5.8) (NCRWQCB 2008). With the exception of aluminum, all other measured concentrations of inorganic constituents (i.e., arsenic, cadmium, chromium, copper, lead, mercury, nickel, selenium, silver, and zinc) were in compliance with all water quality objectives at the time of sampling. Aluminum concentrations (8.80 to 565.00 µg/L) potentially exceeded the USEPA continuous concentration for freshwater aquatic life protection (87 µg/L) on 12 of 28 site visits (43 percent exceedance rate), exceeded the USEPA secondary MCL for drinking water (50 µg/L) on 15 site visits (54 percent exceedance rate), and exceeded the California Department of Health Services secondary MCL for drinking water (200 µg/L) on four site visits (14 percent exceedances rate). At one station (Klamath River at Klamath Glen [RM 5.8]), grab samples were analyzed for 100 pesticides, pesticide constituents, or pesticide metabolites; 50 PCB congeners; and 6 phenolic compounds. There were no PCB detections, but the pesticide disulfoton was detected in one sample. Disulfoton is a systemic organophosphate insecticide for which there is no numeric water quality objective.

Sediment Contaminants

Sediment data for inorganic and organic contaminants in the Klamath River from the Salmon River to the estuary are limited. The Yurok Tribe is currently conducting a bioaccumulation study to evaluate levels of bioaccumulatory chemicals in a number of aquatic species; work began in spring of 2010 and is ongoing. Data are not yet available for inclusion in this Klamath Facilities Removal EIS/EIR.

C.7.2.3 Klamath Estuary

Sediment and water column data for inorganic and organic contaminants in the Klamath Estuary are not readily available. However, contaminant conditions in the estuary (RM 0–2) are likely to be similar to those a few miles upstream at the site for which SWAMP data have been recently collected (see previous section).

As part of the Secretarial Determination studies, a sediment evaluation is evaluating potential environmental and human health impacts of the downstream release of sediment deposits currently stored behind the dams under the Proposed Action. Sediment cores were collected during 2009–2010 at multiple sites and at various sediment depths per site, including two locations in the Klamath Estuary (see Section C.7.1.1). Overall, using multiple lines of evidence from the 2009-2010 Secretarial Determination study, sediment quality in the Klamath Estuary does not appear to be highly contaminated. Where

elevated concentrations of chemicals in sediment were found (i.e., arsenic, chromium, iron, nickel, bis[2-ethylhexyl]phthalate), the degree of exceedance based on comparisons of measured (i.e., detected) chemical concentrations to SLs was small and in several cases (i.e., arsenic, nickel) may reflect regional background conditions (CDM 2011). The results of the acute toxicity bioassays for the midge and the amphipod identified no statistically significant difference in survival of either test organism exposed to estuary sediments compared to control sediments. As with the reservoir sediments (Section C.7.1.1), the lone chemical identified in tissue from invertebrates exposed to estuary sediments above TRVs was fluoranthene. Further, it was only identified above the No Effect TRV, and was below the Low Effect TRV. TEQs for dioxin, furan, and dioxin-like PCBs were all below 0.2 pg/g for the Klamath Estuary, thus adverse effects from exposure to TEQs are not expected following exposure to sediment in the estuary (CDM 2011).

C.8 References

Armour CL. 1991. Guidance for evaluating and recommending temperature regimes to protect fish. USFWS Biological Report 90:

Armstong N, and Ward G. 2008. Coherence of nutrient loads and AFWO Klamath River grab sample water quality database. Technical Report. Prepared for USFWS, Arcata Fish and Wildlife Office, Arcata, California.

Asarian E, and Kann J. 2006a. Evaluation of PacifiCorp's Klamath River water quality model predictions for selected water quality parameters. Technical Memorandum. Prepared by Kier Associates, Blue Lake and Arcata, California and Aquatic Ecosystem Sciences, LLC, Ashland, Oregon for the Yurok Tribe Environmental Program, Klamath, California.

Asarian E, and Kann J. 2006b. Klamath River nitrogen loading and retention dynamics, 1996–2004. Final Technical Report Prepared by Kier Associates, Blue Lake and Arcata California and Aquatic Ecosystem Sciences, LLC, Ashland, Oregon for the Yurok Tribe Environmental Program, Klamath, California.

Asarian E, Kann J, and Walker WW. 2009. Multi-year nutrient budget dynamics for Iron Gate and Copco Reservoirs, California. Prepared by Riverbend Sciences and Kier Associates, Eureka, California, Aquatic Ecosystem Sciences, LLC, Ashland, Oregon, and William Walker, Concord, Massachusetts for the Karuk Tribe, Department of Natural Resources, Orleans, California.

Asarian E, Kann J, and Walker WW. 2010. Klamath River nutrient loading and retention dynamics in free-flowing reaches, 2005–2008. Prepared by Kier Associates,

Eureka, California and Aquatic Ecosystem Sciences, LLC, Ashland, Oregon for the Yurok Tribe Environmental Program, Klamath, California.

Aquatic Scientific Resources. 2005. Preliminary research on *Aphanizomenon flos-aquae* at Upper Klamath Lake, Oregon. Investigations to set direction for research of factors with potential for influencing *Aphanizomenon* growth at Upper Klamath Lake. Prepared by Aquatic Scientific Resources, Portland, Oregon for Klamath Basin Ecosystem Restoration Office, Klamath Falls Fish and Wildlife Office, Klamath Falls, Oregon.

Barnes R. 1980. Coastal lagoons, the natural history of a neglected habitat. Cambridge University Press, United Kingdom.

Bartholow J. 2005. Recent water temperature trends in the lower Klamath River, California. North American Journal of Fisheries Management 25: 152–162.

Blakey JF. 1966. Temperature of surface waters in the conterminous United States. Hydrologic Investigations Atlas HA-235. U.S. Geological Survey, Washington, D.C. http://ut.water.usgs.gov/publications/pubsmaps.html.

Bradbury JP, Coleman SM, and Reynolds RL. 2004. The history of recent limnological changes and human impact on Upper Klamath Lake, Oregon. Journal of Paleolimnology 31: 151–161.

Brett JR. 1952. Temperature tolerance in young Pacific salmon, genus *Oncorhynchus.* Journal of the Fisheries Research Board of Canada 9: 265–323.

Bruun P. 1966. Tidal inlets and littoral drift. Universiteforlaget, Oslo, Norway.

Bureau of Land Management. 2005. Oregon/Washington hydrography framework partnership. GIS data file. http://hydro.reo.gov (accessed September 2006). As cited in Rabe and Calonje 2009.

Bureau of Land Management. 2006. Oregon/Washington ground transportation roads publication. ArcGIS data file. ftp://ftp.blm.gov/pub/OR/ (accessed September 2006). As cited in Rabe and Calonje 2009.

Butcher J. 2008. Nutrient dynamics in the Klamath. Memorandum from J. Butcher, Tetra Tech, Inc., Research Triangle Park, North Carolina to the Klamath TMDL Technical Team. 12 February.

Campbell SG, and Ehinger WJ. 1993. Wood River hydrology and water quality study. Pages 7–79 *in* Environmental research in the Klamath Basin, Oregon, 1991 Annual Report, R-93-13. U.S. Department of Interior, Bureau of Reclamation, Denver, Colorado.

Campbell SG, Ehinger WJ, and Kann J. 1993. Chapter 2: Wood River hydrology and water quality study. Pages 9–90 *in* S. G. Campbell, editor. Environmental research in

the Klamath Basin, Oregon, 1992 Annual Report R-93-16. Applied Sciences Branch, Research and Laboratory Services Division, U.S. Department of Interior, Bureau of Reclamation, Denver, Colorado.

Carlson K, and Foster K. 2008. Water quality monitoring during turbine venting tests at the Iron Gate Powerhouse, Klamath Hydroelectric Project. Final Report. Prepared by CH2M HILL and Mason, Bruce, and Girard for PacifiCorp Energy, Portland, Oregon.

CDM. 2011. Klamath settlement process, sediment interpretive report. Internal Working Draft, 2011.

Connelly M, and Lyons L. 2007. Upper Sprague watershed assessment. Prepared by Klamath Basin Ecosystem Foundation, Klamath Falls, Oregon and Oregon State University Klamath Basin Research and Extensions Center with technical assistance from E&S Environmental Chemistry, Inc., Corvallis, Oregon.

David Evans and Associates, Inc (DEA). 2005. Upper Williamson River watershed assessment. Final Report. Prepared by David Evans and Associates, Inc., Portland, Oregon for the Klamath Basin Ecosystem Foundation and Upper Williamson River Catchment Group in cooperation with the Upper Klamath Basin Working Group and the Klamath Watershed Council.

Davis, JA., Melwani AR, Bezalel SN, Hunt JA, Ichikawa G, Bonnema A, Heim WA, Crane D, Swensen S, Lamerdin C, and Stephenson M. 2010. Contaminants in fish from California lakes and reservoirs, 2007–2008. Summary report on a two-year screening survey. Prepared for Surface Water Ambient Monitoring Program, California State Water Resources Control Board, Sacramento, California.

Deas M. 2008. Nutrient and organic matter fate and transport in the Klamath River: June to September 2007. Prepared by Watercourse Engineering, Inc., Davis, California for PacifiCorp, Portland, Oregon.

Deas ML, and Orlob GT. 1999. Klamath River modeling project. Report No. 99-04. Prepared by Center for Environmental and Water Resources Engineering, Department of Civil and Environmental Engineering, Water Resources Modeling Group, University of California, Davis.

Deas M, and Vaughn J. 2006. Characterization of organic matter fate and transport in the Klamath River below Link Dam to assess treatment/reduction potential. Prepared by Watercourse Engineering, Inc., Davis, California for the U.S. Bureau of Reclamation, Klamath Area Office, California.

Department of the Interior (DOI). 2010, Revision 2. Quality assurance project plan: sediment contaminant study, Klamath River sediment sampling program, J.C. Boyle, Copco-1, Copco-2, and Iron Gate reservoirs; Klamath River Estuary. Prepared by U. S.

Bureau of Reclamation, Mid-Pacific Region, Branch of Environmental Monitoring, MP-157.

Dillon J. 2008. Subject: Dioxin in sediments behind the dams on the Klamath River. Technical memorandum to B. Cluer, Team Leader, Scientific Support Team and S. Edmondson, Northern California Habitat Supervisor, from J. Dillon, Water Quality Program Coordinator, National Marine Fisheries Service, Southwest Region, Santa Rosa, California. 8 April 2008.

Doyle MC, and Lynch DD. 2005. Sediment oxygen demand in Lake Ewauna and the Klamath River, Oregon, June 2003. Scientific Investigations Report 2005-5228. Prepared by the U.S. Department of the Interior, U.S. Geological Survey, Reston, Virginia in cooperation with the Bureau of Reclamation.

Eilers JM, and Gubala CP. 2003. Bathymetry and sediment classification of the Klamath Hydropower Project impoundments. Prepared by J. C. Headwaters, Inc. for PacifiCorp, Portland, Oregon.

Eilers JM, Kann J, Cornett J, Moser K, and St. Amand A. 2004. Paleolimnological evidence of change in a shallow, hypereutrophic lake: Upper Klamath Lake, Oregon, USA. Hydrobiologia 520: 7–18.

Eilers JM, and Raymond R. 2005. Appendix E. Sediment oxygen demand in selected sites of the Lost River and Klamath River. Prepared by MaxDepth Aquatics, Inc., Bend, Oregon, and Environmental Science Resources, LLC, Corvallis, Oregon for Tetra Tech, Inc., Fairfax, Virginia.

Escoffier FF. 1940. The stability of tidal inlets. Shore and Beach 8: 114–115.

Federal Energy Regulatory Commission (FERC). 2007. Final Environmental Impact Statement for hydropower license. Klamath Hydroelectric Project (FERC Project No. 2082-027). http://www.ferc.gov/industries/hydropower/enviro/eis/2007/11-16-07.asp.

Fetcho K. 2007. 2006 Klamath River blue-green algae summary report. Prepared by the Yurok Tribe Environmental Program, Klamath, California.

Fetcho K. 2008. 2007 Klamath River blue-green algae summary report. Final Report. Prepared by the Yurok Tribe Environmental Program, Klamath, California.

Fetcho K. 2009. 2008 Klamath River blue-green algae summary report. Final Report. Prepared by the Yurok Tribe Environmental Program, Klamath, California.

Fetcho K. 2010. September 14 and 15 phytoplankton results and recent microcystin results. Memorandum from the Yurok Tribe Environmental Program, Klamath, California.

FISHPRO. 2000. Fish passage conditions on the upper Klamath River. Submitted to Karuk Tribe and PacifiCorp.

Fry B. 1999. Using stable isotopes to monitor watershed influences on aquatic trophodynamics. Canadian Journal of Fisheries and Aquatic Sciences 56: 2,167–2,171.

Gearheart RA, Anderson JK, Forbes MG, Osburn M, and Oros D. 1995. Watershed strategies for improving water quality in Upper Klamath Lake, Oregon. Volumes I–III. Humboldt State University, Arcata, California.

Gathard Engineering Consulting. 2006. Klamath River dam and sediment investigation. Prepared by GEC, Seattle, Washington.

Greimann BP, Varyu D, Godaire J, Russell K, Lai G, and Talbot R. 2011. Hydrology, hydraulics and sediment transport studies for the Secretary's Determination on Klamath River dam removal and basin restoration, Klamath River, Oregon and California, Mid-Pacific Region. Draft Technical Report No. SRH-2011-02. Prepared for USDI Bureau of Reclamation, Mid-Pacific Region, Technical Service Center, Denver, Colorado.

Hiner M. 2006. Seasonal water quality in the Klamath River estuary and surrounding sloughs, 2001–2003.

Hoopa Valley Tribe Environmental Protection Agency (HVTEPA). 2008. Water quality control plan Hoopa Valley Indian Reservation. Approved 11 September 2002, Amendments Approved 14 February 2008. Hoopa Valley Tribal Environmental Protection Agency, Hoopa, California.

Horne A, and Goldman C. 1994. Limnology. McGraw-Hill, New York.

Jacoby JM, and Kann J. 2007. The occurrence and response to toxic cyanobacteria in the Pacific Northwest, North America. Lake and Reservoir Management 23: 123–143.

Kann J. 1993. Chapter 3: limnological trends in Agency Lake, Oregon -1992. Pages 91-98 in SG Campbell, editor. Environmental research in the Klamath Basin, Oregon, 1992 Annual Report R-93-16. Applied Sciences Branch, Research and Laboratory Services Division, U.S. Department of Interior, Bureau of Reclamation, Denver, Colorado.

Kann J. 2004. Memo: Copco Lake analysis. Letter to Kier and Associates from J. Kann, Aquatic Ecologist, Aquatic Ecosystem Sciences, LLC, Ashland, Oregon. 7 December.

Kann J. 2006. *Microcystis aeruginosa* occurrence in the Klamath River system of southern Oregon and northern California. Technical Memorandum.

Kann J. 2007a. Toxic cyanobacteria results for Copco/Iron Gate reservoirs: 31 May and 12–13 June 2007. Technical memorandum from J. Kann, Aquatic Ecologist, Aquatic Ecosystem Sciences, LLC, Ashland, Oregon. 19 June.

Kann J. 2007b. Toxic cyanobacteria results for Copco/Iron Gate reservoirs: 26–27 June. Technical memorandum from J. Kann, Aquatic Ecologist, Aquatic Ecosystem Sciences, LLC, Ashland, Oregon. 29 June.

Kann J. 2007c. Toxic cyanobacteria results for Copco/Iron Gate reservoirs: 10–11 July 2007. Technical memorandum from J. Kann, Aquatic Ecologist, Aquatic Ecosystem Sciences, LLC, Ashland, Oregon. 16 July.

Kann J. 2007d. Toxic cyanobacteria results for Copco/Iron Gate reservoirs: 7–8 August 2007. Technical memorandum from J. Kann, Aquatic Ecologist, Aquatic Ecosystem Sciences, LLC, Ashland, Oregon. 15 August.

Kann J. 2008. Microcystin bioaccumulation in Klamath River fish and freshwater mussel tissue: preliminary 2007 results. Technical Memorandum. Prepared by Aquatic Ecosystem Sciences, LLC, Ashland, Oregon for the Karuk Tribe of California, Orleans, California.

Kann J. 2010. Compilation of Klamath Tribes upper Klamath Lake water quality data, 1990–2009. Prepared by Aquatic Ecosystem Sciences, LLC, Ashland, Oregon for the Klamath Tribefetchs Natural Resources Department, Chiloquin, Oregon.

Kann J, and Asarian E. 2005. 2002 Nutrient and hydrologic loading to Iron Gate and Copco reservoirs, California. Prepared by Aquatic Ecosystem Sciences, LLC, Ashland, Oregon and Kier Associates, Mill Valley and Arcata, California for the Karuk Tribe of California, Department of Natural Resources, Orleans, California.

Kann J, and Asarian E. 2006. Longitudinal analysis of Klamath River phytoplankton data 2001–2004. Technical Memorandum. Prepared by Aquatic Ecosystem Sciences, LLC, Ashland, Oregon and Kier Associates, Blue Lake and Arcata, California for the Yurok Tribe Environmental Program, Klamath, California.

Kann J, and Asarian E. 2007. Nutrient budgets and phytoplankton trends in Iron Gate and Copco reservoirs, California, May 2005–May 2006. Prepared by Aquatic Ecosystem Sciences, LLC, Ashland, Oregon and Kier Associates, Arcata, California and by the Karuk Tribe of California, Department of Natural Resources, Orleans, California for the State Water Resources Control Board.

Kann J, and Corum S. 2006. Summary of 2005 toxic *Microcystis aeruginosa* trends in Copco and Iron Gate reservoirs on the Klamath River, California. Technical Memorandum. Prepared by Aquatic Ecosystem Sciences, LLC, Ashland, Oregon and the Karuk Tribe Department of Natural Resources for the Karuk Tribe Department of Natural Resources, Orleans, California.

Kann J and Corum S. 2007. Summary of 2006 toxic *Microcystis aeruginosa* and microcystin trends in Copco and Iron Gate reservoirs, California. Technical Memorandum. Prepared by Aquatic Ecosystem Sciences, LLC, Ashland, Oregon and the Karuk Tribe Department of Natural Resources for the Karuk Tribe Department of Natural Resources, Orleans, California.

Kann J, and Corum S. 2009. Toxigenic *Microcystis aeruginosa* bloom dynamics and cell density/chlorophyll a relationships with microcystin toxin in the Klamath River, 2005–2008. Technical Memorandum. Prepared by Aquatic Ecosystem Sciences, LLC, Ashland, Oregon and the Karuk Tribe Department of Natural Resources for the Karuk Tribe Department of Natural Resources, Orleans, California.

Kann J, and Smith VH. 1999. Estimating the probability of exceeding elevated pH values critical to fish populations in a hypereutrophic lake. Canadian Journal of Fisheries and Aquatic Sciences 56: 2,262–2,270.

Kann J, and Walker WW. 1999. Nutrient and hydrologic loading to Upper Klamath Lake, Oregon, 1991–1998. Draft Report. Prepared by Aquatic Ecosystem Sciences, LLC, Ashland, Oregon for the Klamath Tribes Natural Resources Department, U.S. Bureau of Reclamation Cooperative Studies.

Kann J, Bowater L, and Corum S. 2010a. Middle Klamath River toxic cyanobacteria trends, 2009. Technical Memorandum. Prepared by Aquatic Ecosystem Sciences, LLC, Ashland, Oregon and the Karuk Tribe Department of Natural Resources for the Karuk Tribe Department of Natural Resources, Orleans, California.

Kann J, Corum S, and Fetcho K. 2010b. Microcystin bioaccumulation in Klamath River freshwater mussel tissue: 2009 results. Prepared by Aquatic Ecosystem Sciences, LLC, for the Karuk Tribe Natural Resources Department, and the Yurok Tribe Environmental Program.

Kann J, Bowater L, Johnson G, and Bowman C. 2011. Preliminary 2010 microcystin bioaccumulation results for Klamath River salmonids. Technical Memorandum. Prepared by Aquatic Ecosystem Sciences LLC for the Karuk Tribe Department of Natural Resources, Orleans California.

Karuk Tribe of California. 2001. Karuk aboriginal territories Indian Creek and Elk Creek water quality monitoring report for the fall 2000 monitoring period. Prepared by the Karuk Tribe of California, Department of Natural Resources, Orleans, California.

Karuk Tribe of California. 2002. Water quality monitoring report, Water Year 2000 and 2001. Karuk Tribe of California, Water Resources, Department of Natural Resources, Orleans, California.

Karuk Tribe of California. 2003. Water quality monitoring report, Water Year 2002. Karuk Tribe of California, Water Resources, Department of Natural Resources, Orleans, California.

Karuk Tribe of California. 2007. 2007 Water quality assessment report for Klamath River, Salmon River, Scott River, Shasta River, Ti-Bar Creek, and Irving Creek. Prepared by Karuk Tribe of California, Water Resources, Department of Natural Resources, Orleans, California.

Karuk Tribe of California. 2009. 2008 Water quality assessment report for Klamath River, Salmon River, Scott River, Shasta River, and Bluff Creek. Prepared by Karuk Tribe of California, Water Quality, Department of Natural Resources, Orleans, California.

Karuk Tribe of California. 2010. Water quality report for the mid-Klamath, Salmon, Scott, and Shasta rivers: May–December 2009. Prepared by Karuk Tribe of California, Water Quality Program, Department of Natural Resources, Orleans, California.

Kier Associates. 2006. Appendix G. Nutrient criteria for the Klamath River on the Hoopa Valley Indian Reservation. Hoopa Valley Indian Reservation water quality control plan. Prepared by Kier Associates, Mill Valley and Arcata, California for the Hoopa Valley Tribal Environmental Protection Agency.

Kirk S, Turner D, and Crown J. 2010. Upper Klamath and Lost River sub-basins total maximum daily load (TMDL) and water quality management plan (WQMP). Oregon Department of Environmental Quality, Bend, Oregon.

Klamath Tribes Natural Resources Department. 2006. Data from various studies provided to Environmental Science &Assessment, LLC on CD-ROM. As cited in Rabe and Calonje 2009.

Klasing S, and Brodberg R. 2008. Development of fish contaminant goals and advisory tissue levels for common contaminants in California sport fish: chlordane, DDTs, dieldrin, methylmercury, PCBs selenium, and toxaphene. Prepared by Pesticide and Environmental Toxicology Branch, Office of Environmental Health Hazard Assessment, California Environmental Protection Agency.

Kuwabara JS, Topping BR, Lynch DD, Carter JL, and Essaid HI. 2009. Benthic nutrient sources to hypereutrophic upper Klamath Lake, Oregon, USA. Environmental Toxicology and Chemistry 28: 516–524.

Lindenberg MK, Hoilman G, and Wood TM. 2009. Water quality conditions in Upper Klamath and Agency Lakes, Oregon, 2006. U.S. Geological Survey Scientific Investigations Report 2008-5201. Prepared by U.S. Department of the Interior, U.S. Geological Survey, Reston, Virginia in cooperation with the Bureau of Reclamation.

Lytle CM. 2000. Subject: Water quality data review and wetland size estimate for the treatment of wastewaters from the Klamath Straits Drain. Draft Technical Memorandum. Prepared for USBLM, Klamath Project Office, Klamath Falls.

MacDonald LH. 1994. Developing a monitoring project. Journal of Soil and Water Conservation May–June: 221–227.

McCormick P, and Campbell SG. 2007. Evaluating the potential for watershed restoration to reduce nutrient loading to upper Klamath Lake, Oregon. U.S. Geological Survey Open-File Report 2007-1168. U.S. Department of the Interior, U.S. Geological Survey, Reston, Virginia.

MacDonald LH, Smart AW, and Wissmar RC. 1991. Monitoring guidelines to evaluate effects of forestry activities on streams in the Pacific Northwest and Alaska. Report No. 910/9-91-001. U. S. Environmental Protection Agency, Region 10, Seattle, Washington and Center for Streamside Studies, University of Washington, Seattle.

Mayer TD. 2005. Water quality impacts of wetland management in the Lower Klamath National Wildlife Refuge, Oregon and California, USA. Wetlands 25: 697–712.

McGeer JC, L Baranyi, and GK Iwama. 1991. Physiological responses to challenge tests in six stocks of coho salmon (*Oncorhynchus kisutch*). Canadian Journal of Fisheries and Aquatic Sciences 48: 1,761–1,771.

Miller WE, and JC Tash. 1967. Upper Klamath Lake studies, Oregon. Interim report. Water Pollution Control Series Paper WP-20-8. Federal Water Pollution Control Administration, Pacific Northwest Laboratory, Corvallis, Oregon.

Miller MA, Kudela RM, Mekebri A, Crane D, Oates SC, et al. 2010. Evidence for a novel marine harmful algal bloom: cyanotoxin (microcystin) transfer from land to sea otters. PLoS ONE 5: e12576. doi:10.1371/journal.pone.0012576.

Montgomery JM. 1985. Water treatment principles and design. J. M. Montgomery Consulting Engineers. John Wiley and Sons, Inc.

Morace JL. 2007. Relation between selected water-quality variables, climatic factors, and lake levels in Upper Klamath and Agency Lakes, Oregon, 1990–2006. U.S. Geological Survey Scientific Investigations Report 2007-5117. http://pubs.water.usgs.gov/sir20075117.

National Research Council. 2003. Endangered and threatened fishes in the Klamath River basin, causes of decline and strategies for recovery. Prepared by the Committee on Endangered and Threatened Fishes in the Klamath River Basin, Board on Environmental Studies and Toxicology, Division on Earth and Life Studies, National Research Council of the National Academies. The National Academies Press, Washington, D.C.

North Coast Regional Water Quality Control Board (NCRWQCB). 2006. Water Quality Control Plan for the North Coast region (Basin Plan). Santa Rosa, California.

North Coast Regional Water Quality Control Board. 2008. Surface water ambient monitoring program (SWAMP) summary report for the North Coast Region (RWQCB-1) for years 2000–2006. Prepared by North Coast Regional Water Quality Control Board, Santa Rosa, California.

North Coast Regional Water Quality Control Board. 2010. Klamath River total maximum daily loads (TMDLs) addressing temperature, dissolved oxygen, nutrient, and microcystin impairments in California, the proposed site specific dissolved oxygen objectives for the Klamath River in California, and the Klamath River and Lost River implementation plans. Final Staff Report. North Coast Regional Water Quality Control Board, Santa Rosa, California.

O'Brien, MP. 1971. Notes on tidal inlets on sandy shores. Hydraulic Engineering Lab Report No. HEL-24-5. University of California.

Oregon Department of Environmental Quality (Oregon DEQ). 1998. Water quality limited streams 303(d) list. Portland, Oregon. http://www.deq.state.or.us/wq/assessment/docs/rpt98.pdf.

Oregon DEQ. 2002. Upper Klamath Lake drainage total maximum daily load (TMDL) and water quality management plan (WQMP). Portland, Oregon.

Oregon DEQ. 2006. Water Quality Program data file. [CD-ROM]. Oregon DEQ, Portland, Oregon. As cited in Connelly and Lyons 2007.

Oregon DEQ. 2007. Guidance for assessing bioaccumulative chemicals of concern in sediment. Final Report, 07-LQ-023A. Prepared by Oregon Department of Environmental Quality, Environmental Cleanup Program, Portland, Oregon.

PacifiCorp. 2004a. Analysis of potential Klamath Hydroelectric Project effects on water quality aesthetics for the Klamath Hydroelectric Project (FERC Project No. 2082). Final Technical Report. Prepared by PacifiCorp, Portland, Oregon.

PacifiCorp. 2004b. Water resources for the Klamath Hydroelectric Project (FERC Project No. 2082). Final Technical Report. Prepared by PacifiCorp, Portland, Oregon.

PacifiCorp. 2004c. Screening level determination of chemical contaminants in fish tissue in selected Project reservoirs for the Klamath Hydroelectric Project (FERC Project No. 2082). Final Technical Report. Prepared by PacifiCorp, Portland, Oregon.

PacifiCorp. 2006. Causes and effects of nutrient conditions in the upper Klamath River for the Klamath Hydroelectric Project (FERC Project No. 2082). PacifiCorp, Portland, Oregon.

PacifiCorp. 2008a. 2008 Iron Gate turbine venting study of dissolved oxygen improvement. PacifiCorp, Portland, Oregon.

PacifiCorp. 2008b. Final datasonde data from below Iron Gate Dam. Water quality preliminary raw data collected with a YSI 6600 Datasonde on the Klamath River downstream of Iron Gate Dam (RM 189.7) from June 20, 2008 through December 23, 2008. Collected by PacifiCorp, Portland, Oregon. http://www.pacificorp.com/es/hydro/hl/kr.html# [Accessed on 12 December 2010].

PacifiCorp. 2009. Final datasonde data from below Iron Gate Dam. Water quality preliminary raw data collected with a YSI 6600 Datasonde on the Klamath River downstream of Iron Gate Dam (RM 189.7) from January 16, 2009 through December 31, 2009. Collected by PacifiCorp, Portland, Oregon. http://www.pacificorp.com/es/hydro/hl/kr.html# [Accessed on 12 December 2010].

PacifiCorp. 2010. Final datasonde data from below Iron Gate Dam. Water quality preliminary raw data collected with a YSI 6600 Datasonde on the Klamath River downstream of Iron Gate Dam (RM 189.7) from January 1 thru November 19, 2010. Collected by PacifiCorp, Portland, Oregon. http://www.pacificorp.com/es/hydro/hl/kr.html# [Accessed on 12 December 2010].

Rabe A, and Calonje C. 2009. Lower Sprague-lower Williamson watershed assessment. Prepared by Rabe Consulting, with maps and figures by E&S Environmental Chemistry, Inc., Corvallis, Oregon.

Raymond R. 2008. Water quality conditions during 2007 in the vicinity of the Klamath Hydroelectric Project. Prepared by E&S Environmental Chemistry, Inc., Corvallis, Oregon for PacifiCorp Energy, Portland, Oregon.

Raymond R. 2009. Water quality conditions during 2008 in the vicinity of the Klamath Hydroelectric Project. Prepared by E&S Environmental Chemistry, Inc., Corvallis, Oregon for CH2MHill, Portland, Oregon and PacifiCorp Energy, Portland, Oregon.

Raymond R. 2010. Water quality conditions during 2009 in the vicinity of the Klamath Hydroelectric Project. Prepared by E&S Environmental Chemistry, Inc., Corvallis, Oregon for PacifiCorp Energy, Portland, Oregon.

Rykbost KA, and Charlton BA. 2001. Nutrient loading of surface waters in the Upper Klamath Basin: agricultural and natural sources. Special Report 1023. Prepared by Klamath Experiment Station, Oregon State University, Klamath Falls, Oregon.

Shannon & Wilson, Inc. 2006. Sediment sampling, geotechnical testing and data review report: segment of Klamath River, Oregon and California. Prepared by Shannon & Wilson, Inc., Seattle, Washington for California Coastal Conservancy, Oakland, California.

Sinnott S. 2007. 2007 Klamath River nutrient summary report. Final Report. Prepared by the Yurok Tribe Environmental Program, Water Division, Klamath, California.

Sinnott S. 2008. 2008 Klamath River nutrient summary report. Final Report. Prepared by Yurok Tribe Environmental Program, Water Division, Klamath, California.

Sinnott S. 2009. 2006 Klamath River nutrient summary report. Final Report. Prepared by Yurok Tribe Environmental Program, Water Division, Klamath, California.

Sinnott S. 2010. 2009 Klamath River datasonde report. Final Report. Prepared by Yurok Tribe Environmental Program, Klamath, California.

Sinnott S. 2011a. 2010 Klamath River nutrient summary report. Final Report. Prepared by Yurok Tribe Environmental Program, Water Division, Klamath, California.

Sinnott S. 2011b. 2010 Klamath River continuous water quality monitoring summary report. Final Report. Prepared by Yurok Tribe Environmental Program, Klamath, California.

Snyder DT, and Morace JL. 1997. Nitrogen and phosphorus loading from drained wetlands adjacent to Upper Klamath and Agency Lakes, Oregon. Water-Resources Investigations Report 97-4059. U.S. Department of the Interior, U.S. Geological Survey, Denver, Colorado in cooperation with the Bureau of Reclamation.

State Water Resources Control Board (SWRCB). 2010a. 2010 California 303(d) list of water quality limited segments, Category 5. Final 2010 Integrated Report (CWA Section 303(d) List/ 305(b) Report). State Water Resources Control Board, Sacramento, California. http://www.waterboards.ca.gov/water_issues/programs/tmdl/2010state_ir_reports/category5_report.shtml.

State Water Resources Control Board, California Department of Public Health and Office of Environmental Health and Hazard Assessment. 2010b. Cyanobacteria in California recreational water bodies: providing voluntary guidance about harmful algal blooms, their monitoring, and public notification. Blue Green Algae Work Group of the State Water Resources Control Board, the California Department of Public Health, and Office of Environmental Health and Hazard Assessment.

Stein RA, Reimers PE, and Hall JD. 1972. Social interaction between juvenile coho (*Oncorhynchus kisutch*) and fall Chinook salmon (*Oncorhynchus tshawytscha*) in Sixes River, Oregon. Journal of the Fisheries Research Board of Canada 29: 1,737–1,748.

Stillwater Sciences. 2009. Dam removal and Klamath River water quality: a synthesis of the current conceptual understanding and an assessment of data gaps. Technical Report. Prepared by Stillwater Sciences, Berkeley, California for State Coastal Conservancy, Oakland, California.

Stillwater Sciences. 2010. Anticipated sediment release from Klamath River dam removal within the context of basin sediment delivery. Final Report. Prepared by Stillwater Sciences, Berkeley, California for State Coastal Conservancy, Oakland, California.

Sullivan AB, Rounds SA, Deas ML, Asbill JR, Wellman RE, Stewart MA, Johnston MW, and Sogutlugil IE. 2011. Modeling hydrodynamics, water temperature, and water quality in the Klamath River upstream of Keno Dam, Oregon, 2006–2009. Scientific Investigations Report 2011-5105. Prepared by Oregon Water Science Center, U.S. Geological Survey, Portland, Oregon in cooperation with the Bureau of Reclamation.

Sullivan AB, Deas ML, Asbill J, Kirshtein JD, Butler K, Stewart MA, Wellman RE, and Vaughn J. 2008. Klamath River water quality and acoustic doppler current profiler data from Link River Dam to Keno Dam, 2007. Open-File Report 2008-1185. Prepared by U.S. Department of Interior, U.S. Geological Survey, Reston, Virginia in cooperation with the Bureau of Reclamation.

Sullivan AB, Deas ML, Asbill J, Kirshtein JD, Butler K, and Vaughn J. 2009. Klamath River water quality data from Link River Dam to Keno Dam, Oregon, 2008. U.S. Geological Survey Open File Report 2009-1105. Prepared by the U.S. Department of the Interior, U.S. Geological Survey, Reston, Virginia.

Sullivan AB, Snyder DM, and Rounds SA. 2010. Controls on biochemical oxygen demand in the upper Klamath River, Oregon. Chemical Geology 269: 12–21.

Sturdevant D. 2010. Water quality standards review and recommendations: arsenic, iron, and manganese. Draft Report. Prepared by Oregon DEQ, Water Quality Standards Program, Portland, Oregon.

U.S. Department of Agriculture, Forest Service. 2004. Cumulative watershed effects analysis for the Klamath National Forest. Quantitative models for surface erosion, mass wasting, and ERA/TOC.

U. S. Department of Interior. 2010. Summary of Klamath Secretarial Determination preliminary dioxin findings. Yreka, California. http://klamathrestoration.gov/keep-me-informed/klamath-river-reservoirs.

U. S. Environmental Protection Agency. 2000. National guidance: guidance for assessing chemical contaminant data for use in fish advisories. Volume 2: risk assessment and fish consumption limits. Third edition. EPA 823-B-00-008. Office of Science and Technology, Office of Water, USEPA, Washington, D.C.

U. S. Environmental Protection Agency. 2010. Subject: compilation and discussion of sediment quality values for dioxin, and their relevance to potential removal of dams on the Klamath River. Memorandum from B. Ross, Region 9 Dredging and Sediment

Management Team and E. Hoffman, Region 10 Environmental Review and Sediment Management Unit, USEPA, San Francisco, California to D. Lynch, USGS, and R. Graham, USBR.

U. S. Fish and Wildlife Service. 2008. Corrected Sonde database. http://www.fws.gov/arcata/fisheries/activities/waterQuality/klamathWQ_reports.html [Accessed on July 2008].

U.S. Geological Survey. 2007. USGS surface-water data for Oregon. http://waterdata.usgs.gov/or/nwis/sw (accessed October 2007). As cited in Rabe and Calonje 2009.

U. S. Geological Survey. 2010. Data grapher, online. Accessed for graphs on the Williamson River Delta West. U.S. Department of the Interior, U.S. Geological Survey, Oregon Water Science Center, Portland, Oregon. http://or.water.usgs.gov/cgi-bin/grapher/graph_setup.pl.

U. S. Geological Survey. 2011. USGS water data for the nation. U.S. Department of the Interior, U.S. Geological Survey. http://waterdata.usgs.gov/nwis [Accessed on January 2011].

VanderKooi SP, Burdick SM, Echols KR, Ottinger CA, Rosen BH, and Wood TM. 2010. Algal toxins in upper Klamath Lake, Oregon: linking water quality to juvenile sucker health. U.S. Geological Survey Fact Sheet 2009-3111. U.S. Geological Survey, Western Fisheries Research Center, Seattle, Washington.

Walker WW. 2001. Development of phosphorus TMDL for Upper Klamath Lake, Oregon. Prepared for Oregon Department of Environmental Quality, Bend, Oregon.

Wallace M. 1998. Seasonal water quality monitoring in the Klamath River estuary, 1991–1994. Administrative Report No. 98-9. California Department of Fish and Game, Inland Fisheries, Arcata, California.

Ward G, and Armstrong N. 2010. Assessment of primary production and associated kinetic parameters in the Klamath River. Draft Report. Prepared for the USFWS, Arcata Fish and Wildlife Office, Arcata, California.

Watercourse Engineering, Inc. 2011. Klamath River baseline water quality sampling, 2009 Annual Report. Prepared for the KHSA Water Quality Monitoring Group.

Watershed Professionals Network. 1999. Oregon watershed assessment manual. Prepared for the Governor's Watershed Enhancement Board, Salem, Oregon

Weyerhaeuser Company. 1996. Deep, Sand, Aspen and Coyote watershed analysis, Parts I and II (with appendices). As cited in David Evans and Associates, Inc. 2005.

Wood TM. 2001. Sediment oxygen demand in upper Klamath and Agency lakes, Oregon, 1999. USGS Water-Resources Investigations Report 01-4080.

Wood TM, Hoilman GR, and Lindenberg MK. 2006. Water quality conditions in Upper Klamath Lake, Oregon, 2002–2004. Scientific Investigations Report 2006–5209. Prepared by U.S. Department of the Interior, U.S. Geological Survey, Reston, Virginia in cooperation with the Bureau of Reclamation and the U.S. Fish and Wildlife Service.

Wood TM, Cheng RT, Gartner JW, Hoilman GR, Lindenberg MK and Wellman RE. 2008. Modeling hydrodynamics and heat transport in upper Klamath Lake, Oregon, and implications for water quality. Scientific Investigations Report 2008-5076. Prepared by U. S. Geological Survey, Reston, Virginia in cooperation with the Bureau of Reclamation.

Yurok Tribe Environmental Program (YTEP). 2004. Water year 2002 (WY02) report, 1 October 2001–30 September 2002. Final Report. Prepared by Yurok Tribe Environmental Program, Klamath, California.

Yurok Tribe Environmental Program. 2005. Water year 2004 (WY04) report, 1 October 2003–30 September 2004. Final Report. Prepared by Yurok Tribe Environmental Program, Klamath, California.

Zedonis P and Turner R. 2010. An investigation of water quality above, within, and below Iron Gate Reservoir during the fall of 2004. 2005 Draft Report. U. S. Fish and Wildlife Service, Arcata Fish and Wildlife Office, Arcata Fisheries Technical Report., Arcata, California.