Appendix 1

Nutrient Numeric Endpoint Analysis for the Klamath River, CA

Prepared for:

U.S. EPA Region IX

and

North Coast Regional Water Quality Control Board

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1 Introduction: The California NNE Approach

The Klamath River in California is listed as impaired for temperature, nutrients, and low DO/organic enrichment. The North Coast Regional Board is developing TMDLs in collaboration with Oregon and USEPA to address these impairments. For TMDL development, Tetra Tech is applying a set of linked simulation models consisting of CE-QUAL-W2 (for reservoirs) and RMA (for free-flowing reaches). The TMDL runs have primarily addressed numeric criteria for DO and temperature.

Tetra Tech, under contract to EPA Region IX and the California State Water Resources Control Board also developed an approach for calculating nutrient numeric endpoints (NNE) for use in California Water Quality Programs (Tetra Tech, 2006). The "Technical Approach to Develop Nutrient Numeric Endpoints for California," referred to as the California NNE approach, is a risk-based approach in which targets are developed for response variables (or secondary indicators) such as algal density. These response targets can then be converted to site-specific nutrient targets through use of modeling tools.

The California NNE approach recognizes that there is no clear scientific consensus on precise levels of nutrient concentrations or response variables that result in impairment of a designated use. To address this problem, waterbodies are classified in three categories, termed Beneficial Use Risk Categories (BURCs). BURC I waterbodies are not expected to exhibit impairment due to nutrients, while BURC III waterbodies have a high probability of impairment due to nutrients. BURC II waterbodies are in an intermediate range, where additional information and analysis may be needed to determine if a use is supported, threatened, or impaired. Tetra Tech (2006) lists consensus targets for response indicators defining the boundaries between BURC I/II and BURC II/III.

Tetra Tech (2006) also documents a set of relatively simple but effective spreadsheet tools for application in lake/reservoir or riverine systems to assist in evaluating the translation between response indicators and nutrient concentrations or loads.

One important use of the NNE is for setting initial nutrient endpoints for waterbodies requiring nutrient TMDLs. Tetra Tech (2007), under contract with USEPA, conducted a case study of potential NNE endpoints on the Klamath River. That study, "Nutrient Numeric Endpoints for TMDL Development: Klamath River Case Study", addressed only periphyton in the riverine portion of the watershed and used water quality data for 2000-2003, coupled with periphyton observations from 2004. Since that time, significantly more data have become available, and corrections have been made to earlier data. At the request of the North Coast Regional Water Quality Control Board, USEPA has funded this follow-on study. The two major purposes are (1) to extend the NNE analysis to the two reservoirs (Iron Gate and Copco) on the California portion of the Klamath system, and (2) to update the stream periphyton analysis to reflect more recent and corrected data.

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2 Iron Gate and Copco Reservoirs

2.1 USES AND IMPAIRMENTS

Beneficial uses of Copco and Iron Gate reservoirs are defined in the Water Quality Control Plan (NCRWQCB, 2007) and are summarized in Table 1. Both existing and potential uses are protected. Uses related to the protection of endangered salmonid fish species (COLD, RARE, MIGR, SPWN) are of particular interest to many stakeholders in the Klamath River system.

Table 1. Beneficial Uses of Copco and Iron Gate Reservoirs

Code	Use	Copco	Iron Gate
MUN	Municipal and Domestic Supply	E	P
AGR	Agricultural Supply	E	P
IND	Industrial Service Supply	E	P
PRO	Industrial Process Supply	P	P
FRSH	Freshwater Replenishment	E	E
NAV	Navigation	E	E
POW	Hydropower Generation	E	E
REC1	Water Contact Recreation	E	E
REC2	Non-Contact Water Recreation	E	E
COMM	Commercial and Sport Fishing	E	E
WARM	Warm Freshwater Habitat	E	E
COLD	Cold Freshwater Habitat	E	E
WILD	Wildlife Habitat	E	E
RARE	Rare, Threatened, or Endangered Species	E	E
MIGR	Migration of Aquatic Organisms	E	E
SPWN	Spawning, Reproduction, and/or Early Development	E	E
SHELL	Shellfish Harvesting	NA	E
AQUA	Aquaculture	E	E

Notes: E – Existing Use; P – Potential Use; NA – Use not applicable.

California's 2006 Section 303(d) list identified the Klamath River hydrologic unit from the Oregon border to Iron Gate (including both Iron Gate and Copco reservoirs) as impaired due to nutrients, organic enrichment/low dissolved oxygen, and temperature.

By letter of 13 March 2008, Alexis Strauss, Director, Water Division, USEPA Region IX determined that, in addition to this listing, "one Klamath River segment is impaired due to the presence of elevated concentrations of microcystin toxins, specifically the Oregon to Iron Gate segment which includes the Copco and Iron Gate reservoirs." EPA's decision came in response to a suit filed by the Klamath

Riverkeeper on 30 July 2007 (*Klamath Riverkeeper v. USEPA,* Docket No. C 07-3908 (SBA) (N.D. Cal.)). Microcystins are a class of toxic chemicals produced by some strains of the cyanobacteria *Microcystis aeruginosa* that are released into waters when cyanobacterial cells die or cell membranes degrade. These chemicals are a human health risk, capable of inducing skin rashes, sore throat, oral blistering, nausea, gastroenteritis, fever, and liver toxicity (USEPA Region IX, 2008). Microcystin toxins have also been shown to produce effects on animals including acute livestock poisoning and tumor production in fish guts and liver. Microcystin can thus potentially impair a number of beneficial uses of a waterbody. While California has not established numeric water quality objectives for microcystin toxins, EPA based its decision on observations that exceed the World Health Organization guidelines for moderate probability of adverse health effects of microcystin concentrations above 20 µg/L in recreational waters (WHO, 2003), resulting in impairment of the REC-1 beneficial use and the narrative toxicity objective for Iron Gate and Copco reservoirs.

2.2 POTENTIAL NNE TARGETS

Nutrient concentrations in Iron Gate and Copco reservoirs, along with associated physical conditions, are associated with the formation of summer algal blooms, including the formation of extensive blooms of the cyanobacteria *Microcystis aeruginosa*. Algal blooms in the Klamath reservoirs potentially impact designated beneficial uses in a number of ways, including the following linkages between algal growth and beneficial use impairment:

1. The presence of visible algal blooms can directly impact contact and non-contact recreational uses (REC1, REC2) by creating unaesthetic conditions and unpleasant conditions for contact recreation. This is foremost a function of the total algal biomass present during blooms, but a given biomass of cyanobacteria that form visible scums or mats may present a greater problem than a comparable biomass of planktonic algae.
2. Microcystin toxins, produced by blooms of *Microcystis aeruginosa*, have been determined by EPA to cause impairment in the reservoirs. The beneficial uses threatened by elevated microcystin levels include MUN, AGR, REC1, REC2, COMM, WARM, COLD, WILD, RARE, MIGR, SPWN, AQUA and SHELL.
3. Excess algal growth disrupts the dissolved oxygen (DO) balance, leading to super-saturation during daylight periods of high productivity, and depletion of DO during nighttime respiration and as a result of the decay of dead biomass in the water column. Excess productivity typically results in an increase in organic matter loading to the bottom (hypolimnetic) waters of a reservoir, resulting in rapid DO depletion during stratified conditions. In addition, there can be a self-reinforcing feedback loop, as oxygen depletion at the sediment-water interface can promote the release of phosphorus and ammonium from the sediment, which in turn can support additional algal growth. High algal densities can also disrupt pH, as CO_2 is consumed during the day (at depths with sufficient light for photosynthesis) and released during nighttime respiration. Algal-induced changes to the DO balance can thus impair REC1, REC2, COMM, WARM, COLD, WILD, RARE, SPWN, and AQUA beneficial uses.
4. Excess algal growth results in an increase in the export of organic matter from the reservoirs, which in turn can exert an oxygen demand and potentially impair the DO balance and associated beneficial uses in the stretches of the Klamath River downstream from the reservoirs. On the other hand, algal uptake and settling may reduce the transport of inorganic nutrients downstream during the growing season, potentially mitigating impacts in the reaches below the reservoirs.
5. Conditions that lead to dominance by cyanobacteria in the plankton community can have adverse effects on the fishery (other than direct toxicity), as cyanobacteria generally support a much less rich population of planktonic invertebrates, which in turn support forage and juvenile game fish

populations. This potentially affects REC2, COMM, WARM, COLD, WILD, RARE, and SPWN uses.

Of these five impact linkages, the current TMDL effort, driven by the Consent Decree schedule, focuses on numbers 3 and 4, specifically addressing the need to meet DO (as well as temperature) numeric criteria. For these impacts, the target is already established in the numeric water quality criteria.

The required reductions in nutrient and organic matter loads to meet DO criteria will also reduce impacts associated with the other three impact linkages, but are not developed to specifically address these issues. These three risk hypotheses involve narrative, rather than numeric criteria. The Basin Plan contains the following statements of objectives relevant to nutrients in the Klamath:

> ***Biostimulatory Substances***
>
> *Waters shall not contain biostimulatory substances in concentrations that promote aquatic growths to the extent that such growths cause nuisance or adversely affect beneficial uses.*
>
> ***Toxicity***
>
> *All waters shall be maintained free of toxic substances in concentrations that are toxic to, or that produce detrimental physiological responses in human, plant, animal, or aquatic life. Compliance with this objective will be determined by use of indicator organisms, analyses of species diversity, population density, growth anomalies, bioassays of appropriate duration, or other appropriate methods as specified by the Regional Water Board.*

EPA, in establishing the 303(d) listing for microcystin toxins, cites WHO guidance on microcystin targets. However, the scientific understanding does not seem to be sufficiently advanced to translate microcystin levels into quantitative target levels of *Microcystis* biomass or biovolume. As stated in the 2008 EPA staff report

> "WHO used a number of studies to estimate an approximate microcystin concentration that would be expected from a given cell density of *Microcystis aeruginosa*. However, WHO acknowledges that the cyanobacterial cell density may not be a reliable proxy for microcystin toxin concentrations, because different cyanobacterial strains may be present and their genetic capacity may not produce toxins. In fact, some blooms of *Microcystis aeruginosa* may produce little to no microcystin toxins… For Section 303(d) purposes, EPA considered the cyanobacterial cell density results as part of our assessment but we did not rely on this ancillary information as definitive evidence of corresponding ambient concentrations of microcystin toxins."

Further, quantitative prediction of *Microcystis* cell density as a function of nutrient loading is exceedingly difficult, as it involves a combination of the total potential algal growth supported by nutrient loads, the factors that may promote cyanobacterial dominance within the planktonic algal community, and the factors that may enable *Microcystis* to out-compete other cyanobacteria. To achieve narrative standards and protect beneficial uses, linkage (1) requires an appropriate limit on total algal biomass, linkage (5) requires control of cyanobacterial dominance within blooms, and linkage (2) requires control of toxin-producing strains of *Microcystis* within cyanobacterial blooms. Notably, the risks associated with impact linkages (2) and (5) would also be controlled if the general risk of algal blooms was reduced.

Proposed nutrient numeric endpoints developed for the draft CA NNE framework are expressed as two numbers: the boundary between BURC I/II, indicating a concentration below which impacts are unlikely, and the boundary between BURC II/III, indicating a concentration above which impacts are likely. Table 3-2 in Tetra Tech (2006) recommended algal density targets for summer average chlorophyll *a*. These proposed targets were selected by Regional and State Board staff, based on input from Tetra Tech, at the State Water Board Nutrient Numeric Training Workshop held on May 18-19, 2005 in Sacramento, CA, as shown in Table 2.

Table 2. Proposed CA NNE Planktonic Algal Biomass Targets in Lakes and Reservoirs (as µg/L chlorophyll *a* expressed as a summer mean)

Risk Category Boundary	Beneficial Use				
	COLD	WARM	REC1	REC2	MUN
I/II	5	10	10	10	5
II/III	10	25	20	25	10

The most restrictive recommendations are for the COLD and MUN beneficial uses, both of which apply to Copco and Iron Gate reservoirs. Therefore, the BURC II/III boundary of 10 µg/L summer average chlorophyll *a* provides one potential target for managing these reservoirs. It should be noted, however, that the CA NNE targets are still in draft form, and have not been adopted by the State Board or incorporated into the North Coast Water Quality Control Plan at this time.

The CA NNE document (Tetra Tech, 2006) also considered cyanobacterial density as a potential target, but did not propose specific BURC boundary values. One potential target for cyanobacteria would be to reduce the frequency of cyanobacterial dominance. For example, British Columbia states that waters classified for primary recreation and aquatic life uses should have planktonic populations consisting of less than 50 percent of cyanobacterial cells by volume (MELP, 1992). Volumetric predictions are difficult with simple models, and Downing et al. (2001) instead recommend a target of less than 50 percent of total algal biomass for cyanobacteria. Their work demonstrated that there is typically a rapid phase change between low cyanobacteria densities (less than 20 percent of biomass) to cyanobacterial dominance (> 80 percent of biomass) as nutrient concentrations and total phytoplankton biomass increase. Cyanobacterial dominance is also conveniently expressed using the BG index (BGI), where BGI = ln(%BG/(100 - %BG)), in which %BG is the cyanobacterial biomass expressed as a percentage of the total algal biomass (Trimbee and Prepas, 1987). The 50 percent breakpoint is equivalent to BGI = 0, while values greater than zero indicate increasing cyanobacterial dominance. Downing et al. also found that the risk of greater than 50% cyanobacteria in individual lakes increased proportionately with the BGI.

Downing et al. also undertook regression analysis for prediction of BGI, using data from 99 lakes around the world. Contrary to expectation, they found that TN/TP ratio was not a good predictor of BGI (R^2=26%). The best predictors were phytoplankton biomass, total chlorophyll *a*, and total nitrogen, with R^2 value of 42-43 percent. Total phosphorus was also a better predictor of BGI than TN/TP (R^2 = 34%). The authors argue that "the most potentially useful of these relationships is that with total P, because total P predicts phytoplankton biomass…and discriminates incisively the lakes dominated by Cyanobacteria…," although the correlation coefficient is decreased by a few outliers and a nonlinear asymptote. The equations for predicting BGI from TN and TP are given as follows:

$$BGI = -10.0 + 3.03 \log_{10} TN$$

$$BGI = -4.16 + 1.88 \log_{10} TP$$

In sum, management of Iron Gate and Copco reservoirs to achieve designated beneficial uses appears to require some or all of the following: controls on total algal biomass, the percent of cyanobacteria within total algal biomass, and the dominance of *Microcystis* within the cyanobacterial population.

2.3 APPLICATION OF NNE SCOPING TOOLS

The NNE BATHTUB scoping tool was applied to Copco and Iron Gate reservoirs for the two years of 2002 and 2005, selected because these are the years for which extensive monitoring data are available.

After documenting a reasonable agreement with observations, the tool was then applied to the 2000 TMDL model year.

2.3.1 BATHTUB Tool

In support of the CA NNE approach, Tetra Tech developed a spreadsheet application of the U.S. Army Corps of Engineers BATHTUB model (Walker, 1996) to establish screening level nutrient loading targets for lakes and reservoirs by estimating algal response to nutrient loading. BATHTUB is a steady-state model that calculates nutrient concentrations, chlorophyll *a* concentrations (or algal densities), turbidity, and hypolimnetic oxygen depletion based on nutrient loadings, hydrology, lake morphometry, and internal nutrient cycling processes. It explicitly addresses conditions in run-of-river, and short residence time reservoirs. BATHTUB uses a steady-state mass balance model approach that estimates the distribution of external and internal nutrient loads between the water column, outflows, and sediments. External loads can be specified from various sources including stream inflows, nonpoint source runoff, atmospheric deposition, groundwater inflows, and point sources. Internal nutrient loads from cycling processes may include sediment release and macrophyte decomposition. Since BATHTUB is a steady-state model, it focuses on long-term average conditions rather than day-to-day or seasonal variations in water quality. Algal concentrations are predicted for the summer growing season when water quality problems are most severe. Annual differences in water quality, or differences resulting from different loading or hydrologic conditions (e.g., wet vs. dry years), can be evaluated by running the model separately for each scenario.

BATHTUB first calculates steady-state phosphorus and nitrogen balances based on nutrient loads, nutrient sedimentation, and transport processes (lake flushing, transport between segments). Several options are provided to allow first-order, second-order, and other loss rate formulations for nutrient sedimentation that have been proposed from various nutrient loading models in the literature. The resulting nutrient levels are then used in a series of empirical relationships to calculate chlorophyll *a*, oxygen depletion, and turbidity. Phytoplankton concentrations are estimated from mechanistically based steady-state relationships that include processes such as photosynthesis, settling, respiration, grazing mortality, and flushing. Both nitrogen and phosphorus can be considered as limiting nutrients, at the option of the user. Several options are also provided to account for variations in nutrient availability for phytoplankton growth based on the nutrient speciation in the inflows. The empirical relationships used in BATHTUB were derived from field data from many different lakes, including those in EPA's National Eutrophication Survey and lakes operated by the Army Corps of Engineers. Default values are provided for most of the model parameters based on extensive statistical analyses of these data.

It is important to emphasize that the model is a simple screening tool for prediction of average conditions, and that more informative results can be obtained from more detailed, calibrated models. However, BATHTUB's ease of use makes it ideal for rapid evaluation of potential nutrient-algal interactions.

2.3.2 Data Assembly

BATHTUB application to Copco and Iron Gate reservoirs addressed conditions observed near the dams, representing each reservoir as a single longitudinal segment with a stratified water column. Relatively intensive monitoring data for the two reservoirs exists for 2002 and 2005. These data, along with an analysis of mass balances, are presented in Kann and Asarian (2005), and Kann and Asarian (2007). Due to very short residence times in the winter high-flow season, summer algal concentrations in these reservoirs are most strongly affected by loading in and shortly prior to the growing season, consistent with the recommendations of Walker (1996). Flows and loads were therefore calculated for April to September in 2002 and May to September in 2005 (data are not available for April 2005), based on the results calculated by Kann and Asarian, as shown in Table 3.

Table 3. Flow and Nutrient Data for BATHTUB Application

	Copco		Iron Gate	
	2002 (Apr-Sep)	2005 (May-Sep)	2002 (Apr-Sep)	2005 (May-Sep)
Inflow (hm^3)	434	379	532	402
TP Load (kg)	119,380	59,000	122,300	53,700
TN Load (kg)	480,710	545,100	511,500	421,400
TIP Load (kg)	71,489	35,331	79,925	35,094
TIN Load (kg)	182,031	206,414	130,586	107,583
Summer TP (mg/L)	0.24	0.15	0.19	0.13
Summer TN (mg/L)	1.14	1.23	1.19	1.01
Summer Chlorophyll *a* (µg/L)	8.3	12.2	19.5	19.2

2.3.3 BATHTUB Application

Both Copco and Iron Gate reservoirs are known to have low net trap efficiency for nutrients, due to a combination of short residence times and apparent nutrient regeneration from the sediments under stratified conditions (Butcher, 2008). Kann and Asarian (2007) estimated that Copco Reservoir (for 2004-2005 conditions) retained about 9 percent of influent TN and TP, while Iron Gate retained about 3 percent of influent TP and 10 percent of influent TN. The TMDL model estimated (for 2000 conditions) that Copco retained about 1 percent of TP and 4 percent of TN, while Iron Gate retained about 6 percent of TP and 18 percent of TN. The low net retention rates suggest that net sedimentation rates should be lower than the defaults specified for the BATHTUB scoping tool. Accordingly, the TN and TP sedimentation calibration factors were set to 0.1.

With the revised sedimentation factors, the BATHTUB scoping tool provides a good representation of summer average TN and TP observed in the epilimnion near the dam in both reservoirs (Figure 1 and Figure 2). The model also captures the spatial gradient from Copco to Iron Gate and the relative temporal change between 2002 and 2005 conditions for TP.

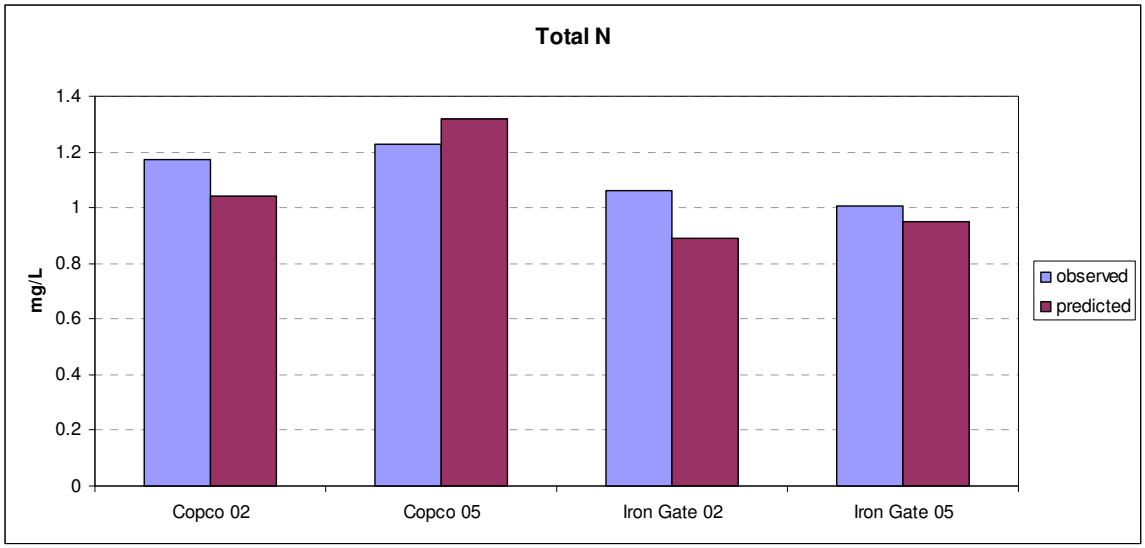


Figure 1. Observed and Predicted Total Nitrogen Concentrations in Copco and Iron Gate Reservoirs

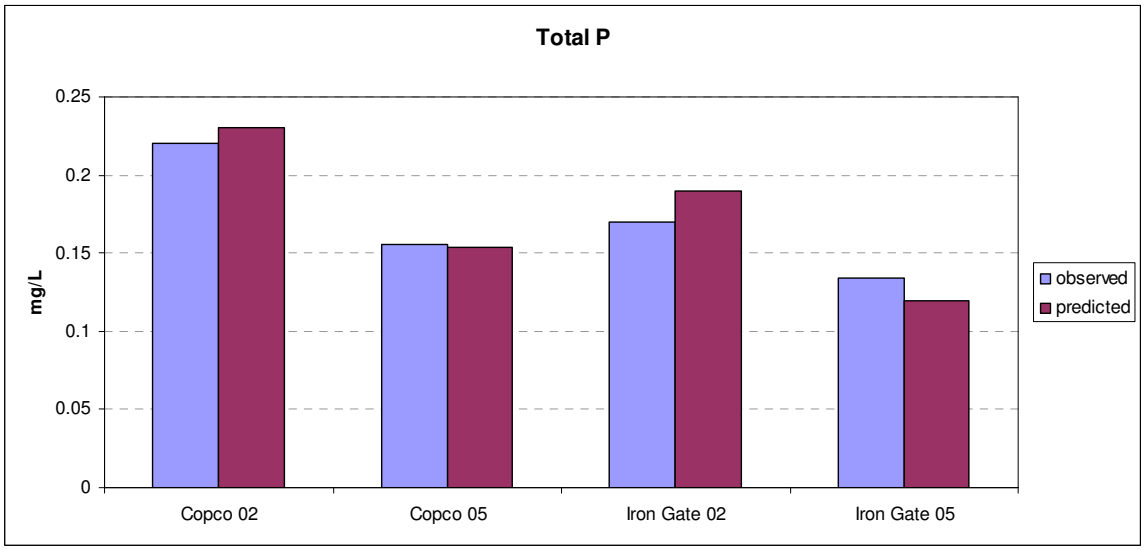


Figure 2. Observed and Predicted Total Phosphorus Concentrations in Copco and Iron Gate Reservoirs

Chlorophyll *a* results were generated without any changes to the default calibration factor of 1.0, and provide a reasonable match to observations (Figure 3). Given that chlorophyll *a* concentrations are highly variable in space and time, as well as the fact that chlorophyll *a* measurements may provide an imprecise measure of cyanobacterial density, these results are considered reasonable. In particular, small samples from right-skewed distributions, such as is typically observed for chlorophyll *a*, are prone to under-estimate the true mean concentration. Predictions for Copco could be brought closer in line with observations by decreasing the chlorophyll *a* calibration factor; however, the quantity and precision of available data do not appear to be sufficient to warrant such fine-scale adjustments.

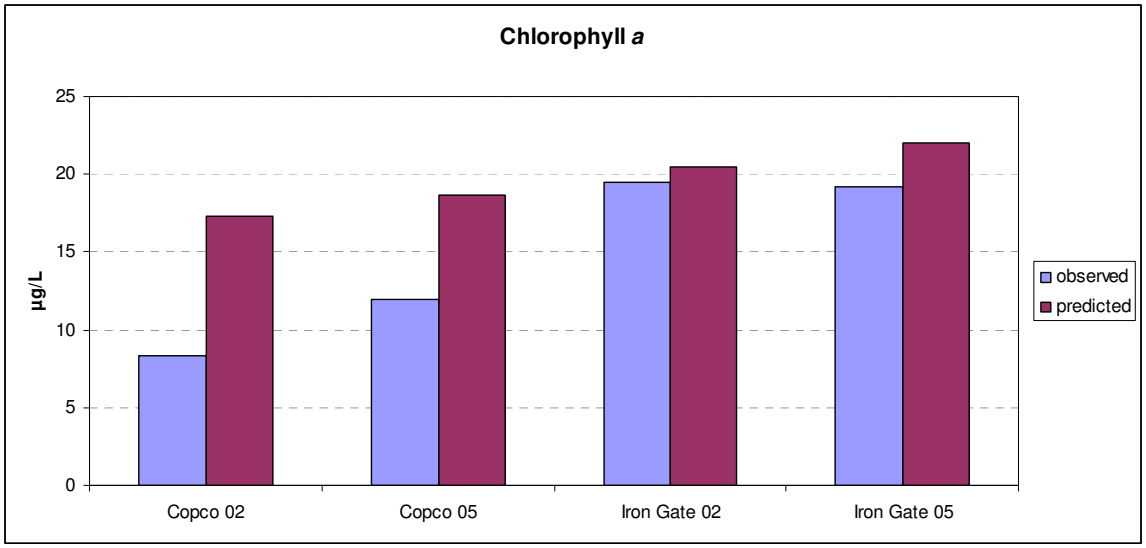


Figure 3. Observed and Predicted Chlorophyll *a* Concentrations in Copco and Iron Gate Reservoirs

The scoping model also predicts the exceedance probability for different concentration levels, based on the coefficient of variation (CV, standard deviation normalized to the mean) of concentrations. Results using the BATHTUB default CV (in natural log space) of 0.42 are shown in Figure 4, suggesting that occasional blooms in excess of 100 µg/L are consistent with the predicted summer average concentrations in Iron Gate, as well as in Copco Reservoir.

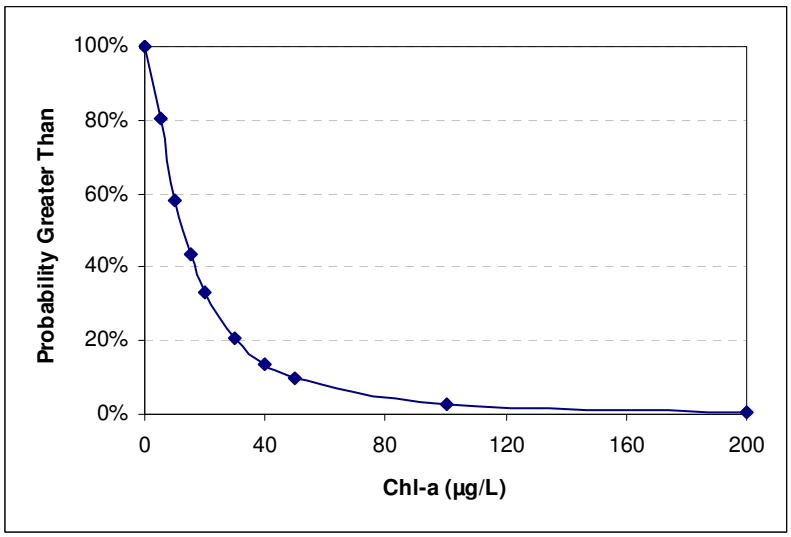


Figure 4. Predicted Distribution Curve for Chlorophyll *a* in Iron Gate Reservoir, 2005

Downing et al.'s (2001) regression equations for BGI as a function of TN and TP concentrations were applied to the predicted nutrient concentrations, and suggest that the algal community is likely to include a significant fraction of cyanobacteria on average (Table 4). The percentage of cyanobacteria predicted from TN concentrations is consistently lower than that predicted from TP concentrations, but both relationships indicate a potential for episodic cyanobacterial blooms, increasing the risk for microcystin toxin production.

Table 4. Cyanobacterial Dominance Predicted for Copco and Iron Gate Reservoirs

	Copco 2002	Copco 2005	Iron Gate 2002	Iron Gate 2005
BGI-P	0.27	-0.15	0.14	-0.27
Cyanobacteria % from BGI-P	56.7%	46.3%	53.5%	43.2%
BGI-N	-0.85	-0.54	-1.06	-0.98
Cyanobacteria % from BGI-N	29.8%	36.7%	25.7%	27.3%

Note: The "Blue Green Index" (BGI) is calculated using the regression relationships presented by Downing et al. (2001).

Application of the spreadsheet tool for year 2000 based on flows and nutrient loads predicted by the Klamath TMDL model yield similar results, with growing season average chlorophyll *a* estimated at 19.7 µg/L for Copco and 23.2 µg/L for Iron Gate. The cyanobacterial fractions of algal biomass are estimated at 64.0 and 59.9 percent using BGI-P, and 39.2 and 30.2 percent using BGI-N.

2.3.4 Potential Nutrient Numeric Endpoints

The BATHTUB scoping tool solves for combinations of TN and TP loading that are consistent with achieving a target growing season average concentration of chlorophyll *a*. Results consistent with achieving the CA NNE recommended BURC II/III boundary of 10 µg/L chlorophyll *a* as a growing season average concentration are shown in Figure 5 for the three years of model application.

The scoping tool predicts that the desired chlorophyll *a* target can be met by reducing phosphorus loading *or* nitrogen loading. The percentage reductions needed to achieve the 10 µg/L target are shown in Table 5. In terms of total algal biomass, it is not necessary to reduce loads of both nutrients to meet the target, as the growth will be controlled by the availability of the most limiting nutrient. These suggest that beneficial uses can be attained by reducing the TP load by approximately 90 percent or by reducing the TN load by approximately 65 percent. However, control by reducing only one nutrient would alter the N:P ratio, and changing the N:P ratio may well have other consequences for algal dynamics in the reservoir, as is discussed further in Section 2.4.

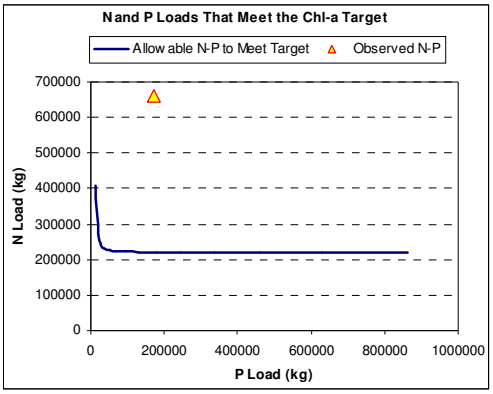
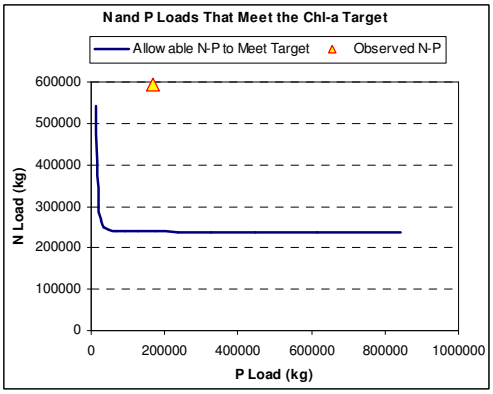
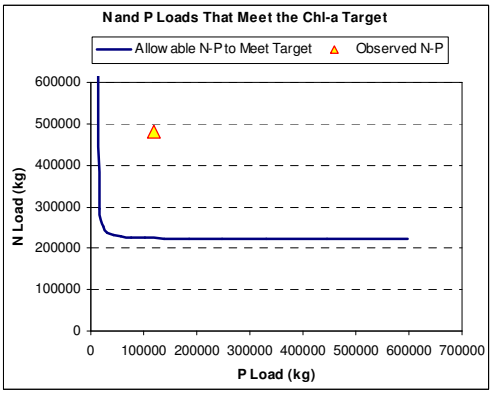
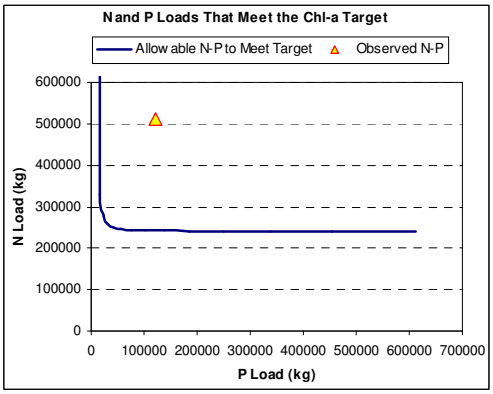
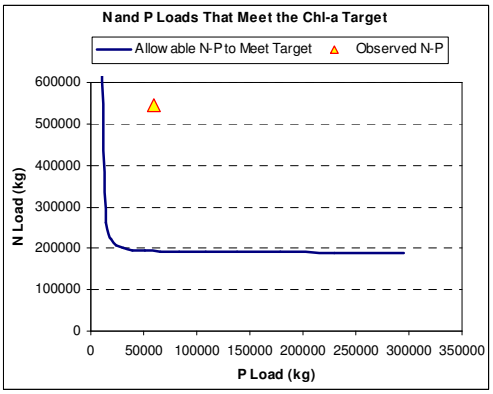
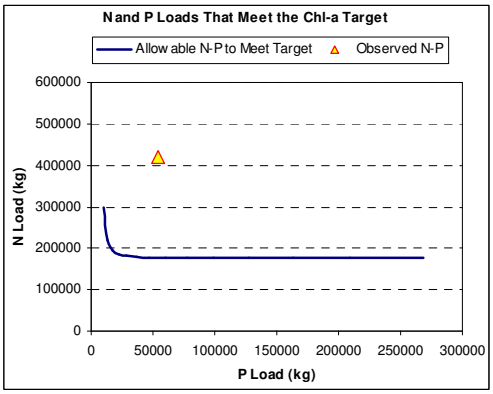
Year	Copco	Iron Gate
2000		
2002		
2005		

Figure 5. Allowable Load Curves to Achieve a 10 µg/L Summer Average Chlorophyll *a* Target

Table 5. Single Component Nutrient Reductions to Achieve a 10 µg/L Summer Average Chlorophyll *a* Target (April-September Loads)

Year	Total Phosphorus		Total Nitrogen	
	Copco	Iron Gate	Copco	Iron Gate
2000	89%	92%	67%	60%
2002	85%	89%	54%	53%
2005	81%	80%	65%	58%

The TMDL model is already calling for significant nutrient reductions to meet DO criteria. Under the dams-in water quality compliance scenario (T4BS1), the April-September 2000 phosphorus loads to Copco are reduced by 89 percent while the nitrogen loads are reduced 73 percent; the reductions in loads to Iron Gate are 88 percent and 74 percent, respectively. Notably, the proposed phosphorus reductions are very similar to those suggested in Table 5, while the proposed total nitrogen reductions in the compliance scenario are greater. Therefore, the T4BS1 scenario developed for dissolved oxygen management would also be expected to meet the algal density target, as developed in this document, to support the COLD and other beneficial uses in Iron Gate and Copco reservoirs.

In addition to reducing the total nitrogen and phosphorus loads, the T4BS1 scenario results in a change in the inorganic fraction of incoming nutrients, with a smaller inorganic fraction, which should also help damp algal response. Application of the BATHTUB tool for 2000 conditions with the T4BS1 nitrogen and phosphorus loads results in a predicted growing season average concentration of 6.6 µg/L in Copco and 4.1 µg/L in Iron Gate. Using Walker's default coefficient of variation for the natural log of chlorophyll *a* of 0.42 suggests that concentrations would be greater than 10 µg/L on 17.4 percent of growing season days in Copco and 2.8 percent of growing season days in Iron Gate.

The T4BS1 scenario also predicts reductions in cyanobacterial populations. With the reduced nutrient and algal settling rates used for BATHTUB application to existing conditions, calculations of the BGI from TN are low (10.8 and 7.2 percent of biomass as cyanobacteria in Copco and Iron Gate, respectively), while the BGI based on TP is reduced to near 25 percent (24.9 and 22.9 percent, respectively).

These results should be considered conservative (that is, including an implicit margin of safety) because the low net sedimentation rates of nutrients assumed for the application to existing conditions have not been altered. In fact, the T4BS1 scenario should result in greater dissolved oxygen concentrations at the sediment-water interface, resulting in lower rates of recycling of nutrients from the sediments, in turn causing higher net sedimentation rates for nutrients. If it is assumed that the effective net sedimentation rates increase to the default values given by Walker, the predicted summer average chlorophyll *a* concentrations in Copco and Iron Gate would decline to 5.0 and 3.0 µg/L, while the predicted cyanobacterial fractions of algal biomass would be 21 and 20 percent, respectively.

Predicted summer average nutrient concentrations in Iron Gate from the BATHTUB scoping tool – relevant to the analysis of downstream effects – are summarized in Table 6 for year 2000 conditions.

Table 6. Summer Average Nutrient Concentrations Predicted for Iron Gate Reservoir (Year 2000 Conditions)

	Existing Loads	T4BS1 Loads with Existing Sedimentation	T4BS1 Loads with Default Sedimentation	Change
TN (mg/L)	1.057	0.288	0.255	-76%
TP (mg/L)	0.267	0.037	0.030	-89%

These BATHTUB results are in good agreement with the CE-QUAL-W2 simulation of concentrations in Iron Gate outflow for the June-September 2000 period. The T4BS1 simulation (without benthic nutrient flux) shows a change relative to existing conditions of -73 percent for TN concentrations and -88.5 percent for TP concentrations.

2.4 MANAGEMENT TO REDUCE *MICROCYSTIS* BLOOMS IN RESERVOIRS

Conditions in Iron Gate and Copco reservoirs, including the risk of microcystin toxins, can clearly be mitigated by a general decrease in eutrophication potential, which would in turn reduce the frequency of cyanobacterial blooms, including *Microcystis* blooms. Other potential strategies to address microcystin levels include control of cyanobacterial dominance within blooms, and control of toxin-producing strains of *Microcystis* within cyanobacterial blooms. As demonstrated by Downing et al. (2001), the risk of cyanobacterial dominance increases with increasing levels of TN, TP, and algal biomass, and is also best addressed through a general reduction in eutrophication potential.

Many Cyanobacteria are able to control buoyancy, enabling them to alternate between light-rich (but nutrient poor) surface waters and nutrient rich (but light poor) waters lower in the water column, yielding a competitive advantage against passively floating algal species (Hyenstrand et al., 1998). Many bloom-forming Cyanobacteria are also able to tolerate higher temperatures than true algae. Lake management strategies that increase vertical mixing (counteracting the cyanobacterial buoyancy advantage) and decrease surface water temperatures may thus be useful pieces of an overall control strategy.

Earlier authors (e.g., Smith, 1983) had theorized that a key factor in promoting cyanobacterial dominance was a low N:P ratio, as many bloom-forming Cyanobacteria can fix atmospheric N_2 (although not *Microcystis aeruginosa*). Downing et al. demonstrate that this ratio is not a good predictor of cyanobacterial dominance.

While the N:P ratio is not a good predictor of general cyanobacterial dominance, it may play an important role in competition between different species of Cyanobacteria. Significantly, *Microcystis aeruginosa* does not fix atmospheric nitrogen, but the competing cyanobacterium *Aphanizomenon* does – suggesting that manipulation of nutrient ratios could cause a shift within cyanobacterial blooms from the toxin-producing *Microcystis* to non-toxin producing *Aphanizomenon*. Moisander et al. (2008) recently reported results of ongoing nitrogen and phosphorus fertilization experiments in Iron Gate and Copco using in-lake incubation chambers. Addition of inorganic nitrogen resulted in an increase in total phytoplankton biomass, *Microcystis* abundance, and microcystin concentrations under both high and low light conditions. Phosphorus additions increased *Microcystis* abundance only under low light conditions, whereas the addition of nitrogen or phosphorus decreased the relative abundance of *Aphanizomenon* by promoting growth of *Microcystis*. Based on this research, Moisander concluded that inputs of dissolved inorganic nitrogen to the reservoirs during the summer season are maintaining and increasing toxic blooms of *Microcystis*, and that reduction of nitrogen inputs to the reservoirs would reduce blooms of *Microcystis*. This suggests that management by reduction of nitrogen loads would yield dual benefits by both reducing the total algal biomass and shifting the cyanobacterial population away from *Microcystis* toward *Aphanizomenon*. The work is ongoing, and may yield valuable insights into optimal management of the reservoirs.

In sum, the proposed nutrient reductions appear to have good potential to address all five of the linkages between algal growth and beneficial use impairment discussed in Section 2.2.

1. Frequency of visible algal blooms will be reduced as average algal biomass decreases.
2. Production of microcystin toxins should decline as total algal biomass decreases and cyanobacterial dominance within the algal population is reduced.
3. Algal effects on the DO balance will be mitigated, as demonstrated in the existing TMDL model.

4. Export of organic matter downstream will be reduced as algal growth is reduced.
5. Reduction in cyanobacterial dominance will potentially result in a healthier aquatic ecosystem that supports an improved fishery.

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3 Klamath River below Iron Gate

The Klamath River watershed encompasses 15,722 square miles in the states of Oregon and California, flowing from the Cascades in Oregon westerly and southerly to the Pacific Ocean in Del Norte Co., CA (see Figure 6). The analysis in this section addresses the major part of the flowing, freshwater portions of the mainstem Klamath River in California, running from the outlet of Iron Gate Reservoir near the Oregon border in Siskiyou County, CA to the confluence with the Trinity River in Humboldt County, CA and represents a major update to the analysis presented in Tetra Tech (2007).

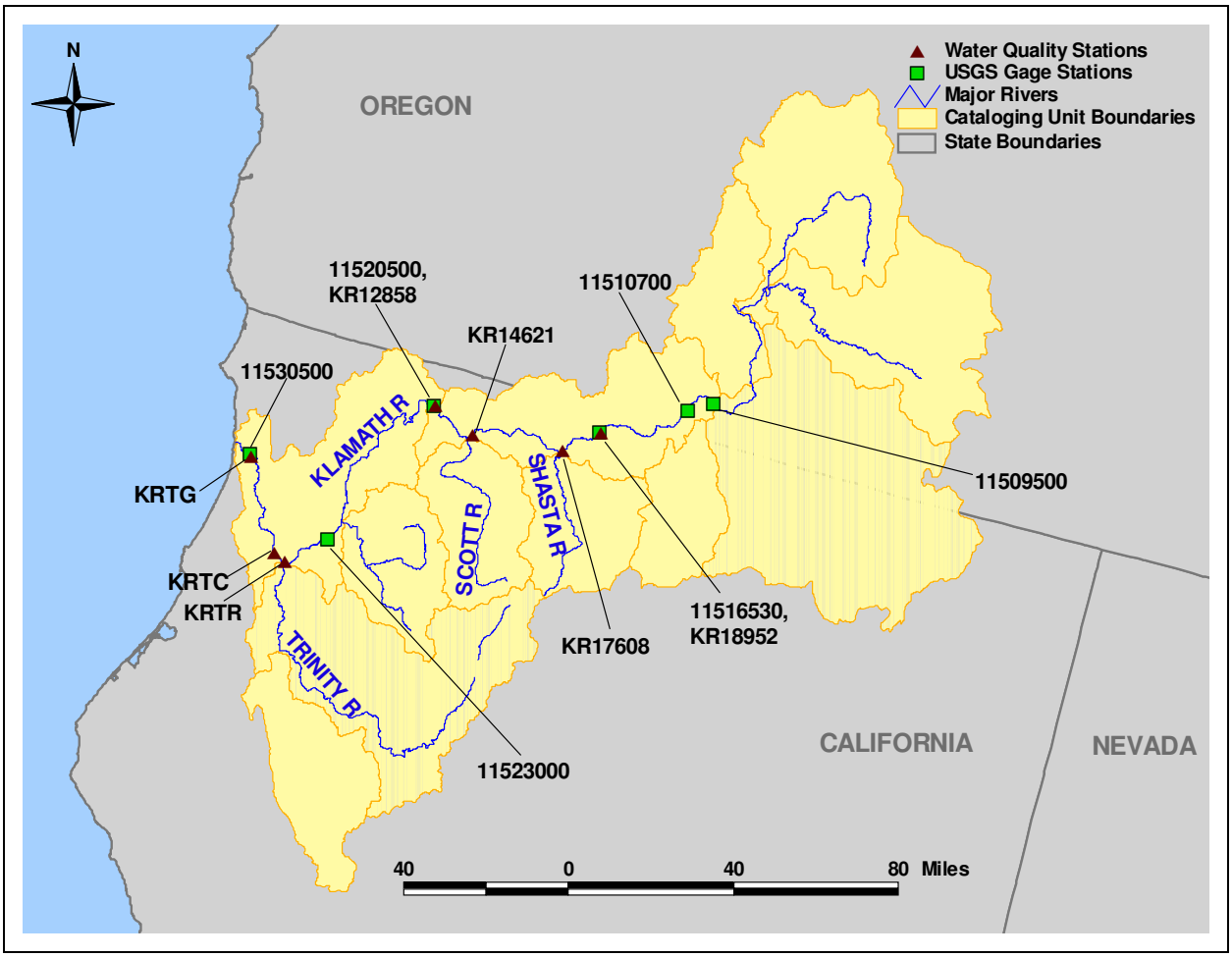

Figure 6. The Klamath River, Showing Selected Water Quality Sampling Stations and Flow Gages on the Lower Klamath River

3.1 Uses and Impairments

The Water Quality Control Plan (NCRWQCB, 2007) establishes multiple beneficial uses for the Klamath River below Iron Gate Reservoir (Table 7). A small portion of the river just upstream of the confluence with Trinity River is under the jurisdiction of the Hoopa Valley Tribe, while much of the Klamath River downstream of the Trinity River is under jurisdiction of the Yurok Tribe.

Table 7. Beneficial Uses of Klamath River below Iron Gate Reservoir

Code	Use	Status
MUN	Municipal and Domestic Supply	E
AGR	Agricultural Supply	E
IND	Industrial Service Supply	E
PRO	Industrial Process Supply	E
FRSH	Freshwater Replenishment	E
NAV	Navigation	E
POW	Hydropower Generation	E
REC1	Water Contact Recreation	E
REC2	Non-Contact Water Recreation	E
COMM	Commercial and Sport Fishing	E
WARM	Warm Freshwater Habitat	E
COLD	Cold Freshwater Habitat	E
WILD	Wildlife Habitat	E
RARE	Rare, Threatened, or Endangered Species	E
MIGR	Migration of Aquatic Organisms	E
SPWN	Spawning, Reproduction, and/or Early Development	E
AQUA	Aquaculture	P
CUL	Native American Culture	E

Notes: E – Existing Use; P – Potential Use; NA – Use not applicable.

California's North Coast Regional Water Quality Control Board has included the free-flowing portion of Klamath River down to the Trinity River on its Clean Water Act Section 303(d) list of impaired waters. Identified impairments include excursions of criteria for nutrients, temperature, and organic enrichment/low DO for segments of the river in California, which are classified for COLD and SPWN beneficial uses.

3.2 POTENTIAL NNE TARGETS

Nutrient loading in the Klamath River produces high levels of periphytic algae. The Hoopa Valley Tribal Environmental Protection Agency has adopted periphyton criteria for the reach of the Klamath River within the Hoopa Valley Indian Reservation. To date, the North Coast Regional Board has not established targets for this endpoint.

While periphyton is included in the Klamath River TMDL models, limited periphyton data were available for model calibration during the years of interest. Calibration focused largely on DO concentrations and diurnal variability in DO, which implicitly include the effects of periphyton and other aquatic vegetation, rather than calibrating directly to periphyton density.

It is important to evaluate periphyton as a response endpoint for several reasons. First, periphyton affects the balance of DO and pH in the river. Second, excess periphyton growth can directly impair COLD, SPWN, and REC designated uses. Finally, in the Klamath River excess periphyton growth (particularly development of *Cladophora* beds) may present an additional important source of risk for maintenance of a healthy salmonid population. This risk hypothesis is summarized in Kier Associates (2005) as follows:

> …*Ceratomyxa shasta* is a myxozoan parasite that causes major problems for the health of juvenile salmonids in the Klamath River. Infection rates are extremely high and in many years results in the death of significant portion of the juvenile salmonids in the Klamath River. Nichols and Foott (2005) estimated that in 2004, 45% of juvenile fall-run Chinook salmon were infected with *C. Shasta* and that the majority of those fish would not survive, and that impact of a loss of that many fish could rival the 2002 adult fish-kill where over 33,000 adult salmon died.
>
> High nutrient levels may be stimulating luxuriant growth of *Cladophora*, a filamentous green algal species. *Cladophora* beds are a favored habitat for polychaete worms that are a host for *C. Shasta* (Stocking and Bartholomew, 2004). The high incidence of *C. Shasta* in the Klamath River may be due to an increase in polychaete populations caused by an increase in polychaete habitat (Stocking and Bartholomew, 2004)… To reduce the incidence of *C. Shasta* infection in the Klamath River, it may be insufficient to improve pH and D.O. alone to reduce fish stress. It also may require reduction in parasite loads by reducing nutrients to reduce the prevalence of *Cladophora* and hence *C. Shasta*'s polychaete host.

Water quality objectives for DO and pH are defined in basin plans, and the relationship between these endpoints, planktonic algal growth, and nutrients is well addressed in the existing calibrated TMDL model. Where a site-specific calibrated nutrient response model exists, this provides the best means of developing appropriate site-specific nutrient numeric endpoints. The North Coast Regional Water Quality Control Board, however, has not yet proposed criteria for periphyton in this river (although the Hoopa have), and this aspect of nutrient response was not the primary focus of the existing TMDL modeling effort.

The Hoopa Valley Tribal Environmental Protection Agency (Kier Associates, 2005; Hoopa Valley TEPA, 2008) recently adopted periphyton standards for the short section of the lower Klamath River on the Hoopa Valley Reservation at Saints Bar just upstream of Trinity River. In addition to DO and pH, they selected periphyton density as an endpoint for criteria development, and initially recommended a maximum annual periphyton biomass of 100 mg/m^2 of periphyton chlorophyll *a*. The criterion was subsequently revised to read as follows (Hoopa Valley TEPA, 2008):

> *Periphyton -For the Klamath River only (Trinity River standards yet to be developed), the maximum annual periphyton biomass shall not exceed 150 mg chlorophyll* a*/m^2 of streambed area.*

The California NNE Approach (Tetra Tech, 2006) recommends setting response targets for benthic algal biomass in streams based on maximum density as mg/m^2 chlorophyll *a*. For the COLD and SPWN beneficial uses, the recommended BURC I/II boundary is 100 mg/m^2, while the BURC II/III boundary is 150 mg/m^2. Existing conditions in the Klamath are clearly often above the BURC II/III boundary, indicating impairment of these uses.

Of particular interest for the Klamath, the risk of *Cladophora* (a filamentous green algae) prevalence (and corresponding large polychaete populations) increases with increasing maximum benthic chlorophyll *a*. Welch et al. (1988) found that 20 percent or more cover by filamentous green algae was correlated with maximum benthic chlorophyll *a* greater than 100 mg/m^2, while Horner et al. (1983) concluded that biomass levels greater than 150 mg/m^2 often occurred with enrichment and when filamentous forms were more prevalent. These findings support the use of the BURC boundaries in establishing targets for the Klamath River. The Klamath River was historically mesotrophic (Kier Associates, 2005), and water

quality conditions in the lower river are exacerbated by large blooms of nitrogen-fixing cyanobacteria (blue-green algae) in Upper Klamath Lake and in the Klamath reservoirs. This suggests that the BURC II/III boundary of 150 mg/m^2 maximum benthic chlorophyll *a* may be most appropriate for the Klamath. The CA NNE approach, however, also recognizes that nutrients occur naturally, and vary in relationship to soils, geology, and land cover, in some cases potentially resulting in benthic chlorophyll *a* concentrations in excess of 150 mg/m^2 under natural conditions. Where this is the case, the natural condition would supersede the proposed target.

3.3 APPLICATION OF NNE SCOPING TOOLS

The CA NNE approach proposes a numeric target for benthic chlorophyll *a*, which is a secondary or response indicator relative to nutrients. To achieve the target, an analysis is required to link nutrient concentrations or load to benthic algal response. Under a previous Work Assignment, Tetra Tech (2007) developed an analysis of potential nutrient numeric endpoints for the lower Klamath downstream of Iron Gate. That analysis relied on a compilation of nutrient monitoring data through 2004. Since that time, data have become available for 2005-2007, and a detailed review of the monitoring data has resulted in modifications of the data through 2004. The sections that follow thus represent an update, revision, and extension of the previous analysis for the free-flowing reaches of the Klamath River below Iron Gate Dam.

3.3.1 Benthic Biomass Tool

The CA NNE *Technical Approach to Develop Nutrient Numeric Endpoints for California* (Tetra Tech, 2006) includes (Appendix 3) the development of a simplified scoping tool of maximum periphyton density in streams. This NNE Benthic Biomass spreadsheet tool is distributed as an Excel spreadsheet. The tool calculates both algal density under average conditions and benthic chlorophyll *a*. Both are estimated using a variety of methods:

- Dodds (1997) method (both mean and maximum)
- Dodds (2002) method (both mean and maximum, using corrected parameters from 2006 erratum[1])
- Standard QUAL2K Model method (maximum)
- Revised QUAL2K Model method (maximum)
- Revised QUAL2K, with adjustment for days of biomass accrual (maximum)

The maximum algal contribution to dissolved oxygen deficit is also calculated, using the Revised QUAL2K Model method. Lastly, the tool allows the user to supply a target (either algal density or benthic chlorophyll *a*), select a calculation method, and the tool will display a graph of allowable TN and TP to meet the target.

The QUAL2K approach is based on the steady-state limit approximation of the benthic algae simulation contained in version 1 of the QUAL2K model (Chapra and Pelletier, 2003). This simulates benthic algal response to nutrient concentrations and light availability. An estimate of the maximum (spatially averaged) response to a given set of forcing functions is obtained as the steady-state asymptote of the model. Because detailed validation data were not available for California, parameters of the model were

[1] The original equations appeared in Dodds, W.K., V.H. Smith, and K. Lohman, 2002, Nitrogen and phosphorus relationships to benthic algal biomass in temperate streams (*Can. J. Fish. Aquat. Sci.*, 59:865-874). The equations were corrected in a 2006 erratum (*Can. J. Fish. Aquat. Sci.*, 63: 1190-1191). The Algal Biomass Spreadsheet beginning with v. 13 (2/28/07) incorporates the corrected coefficients provided in the erratum.

adjusted to obtain approximate agreement with the Dodds (2002) empirical model when applied to California EMAP and Regional Board 6 periphyton data (see Tetra Tech, 2006). It should be noted that this approach introduces considerable uncertainty into predictions for individual streams, and development of a calibrated, site-specific model would be preferable when sufficient data are available. Version 2 of QUAL2K (Chapra et al., 2006) contains significant modifications to the simulation of benthic algae, including an evaluation of nutrient limitation based on the Droop model of changes in intracellular nutrient quotas. Our analysis shows, however, that the changes to Version 2 result in only minor changes to the shape of the steady-state solution, and do not improve the ability of the model to match the Dodds predictions.

Tetra Tech (2006) also developed a "revised QUAL2K" method for predicting maximum periphyton biomass – also tuned to the Dodds (2002) results for the California data set. This approach uses the QUAL2K v.1 solution, but assumes that the "available" fraction of total nutrient used in the model varies as a function of concentration:

$$Availability\ Fraction\ (AF) = 1 - \frac{\gamma}{1 + \exp(\alpha - \beta \log_{10} C)}$$

in which α, β, and γ are parameters from a logistic regression model fit to data, as described in Tetra Tech (2006), and C is the total nutrient concentration. Availability here represents more than just the inorganic fraction of nutrients, as it may also reflect factors such as mat thickness, vertical gradients in the water column, and temporal variability in the inorganic fraction.

Interestingly, the total effect on the Monod growth limitation can be equivalently expressed as an effect on nutrient availability or as an inverse effect on the half-saturation constant:

$$\phi = \frac{AF \cdot C}{k_s + AF \cdot C} = \frac{C}{k_s / AF + C},$$

in which AF, the available fraction, is a function of total nutrient concentration, C, and k_s is the constant Monod half-saturation constant used in the standard QUAL2K model.

The NNE Benthic Biomass spreadsheet tool provides a simple, but robust method for relating nutrient concentrations to benthic algal density. Specifically, the maximum spatially averaged periphyton density is predicted as a function of summer nutrient concentrations and other hydrologic and physical characteristics. A variety of established prediction methods are included. These yield results that are generally similar but differ from one another, reflecting the uncertainty that is present in such predictions.

It is important to provide some clarification on the "maximum" density that is predicted by the tool. What the model predicts is the spatially averaged maximal supported response to a given set of forcing conditions, without reductions by grazing or intermittent die off. In other words, it is the average concentration expected under optimal growth conditions for a given set of nutrient concentrations. It is <u>not</u> the maximum point density that can be observed on a single rock, which can be considerably higher. In addition, it should not be considered as the maximum response to average nutrient conditions: if nutrient concentrations fluctuate above average conditions for a sufficient length of time, additional algal growth will likely occur. Finally, it should be noted that the maximum is difficult to observe. Even if accurate spatially averaged densities are measured, they will often be less than the model-predicted maximum. When performing correctly, the tool should provide an approximate upper-bound envelope on spatially averaged observations.

Because the NNE tools provide only a scoping-level analysis of nutrient targets, they may be superseded by a site-specific calibrated nutrient response model where available. The existing Klamath River TMDL models include, but are not calibrated to periphyton. Instead, calibration focused on DO because of

concerns regarding the representativeness of the periphyton data that are available from the Klamath, due to small sample size and lack of replication. As noted above, accurate prediction of DO implicitly requires a reasonable representation of periphyton and other aquatic vegetation. Continued and improved periphyton sampling would further strengthen the TMDL model application and allow its extension to quantitative analysis of impacts other than DO.

3.3.2 Data

Data have been collected at many sites on the Klamath River, but few stations have consistent long runs of data. For the purpose of this analysis, seven sites on the mainstem Lower Klamath River in California were selected that had reasonable amounts of water quality and periphyton data. These sites are (see also Figure 6 above):

Table 8. Selected Water Quality Monitoring Stations on the Lower Klamath River

Station Number	Station Name	River Mile
KR18952	Klamath River below Iron Gate Dam	189.52
KR17608	Klamath River above Shasta River	176.08
KR14261	Klamath River above Scott River	142.61
KR12858	Klamath River at Seiad Valley	128.59
KRWE	Klamath River above Trinity River (Weitchpec)	43
KRTC	Klamath River below Trinity River above Tulley Creek	35.5-39.2
KRTG	Klamath River at Turwar	5.79

3.3.2.1 Algal Response Data

USEPA and cooperators undertook four rounds of periphyton sampling in the river in 2004 (Eilers, 2005). The published report describes the results of only one of these sampling rounds; results for the remainder were provided by the North Coast Regional Water Quality Control Board. All four sampling rounds followed the same sampling and analytical methodology.

Results of the periphyton sampling include benthic chlorophyll *a*, percent coverage, wet weight, and ash-free dry weight (AFDW). Unfortunately, the information on periphyton density (benthic chlorophyll *a* and AFDW) was obtained from relatively small and separate samples. Specifically, as described in Eilers (2005), determinations of benthic chlorophyll *a* and AFDW were each made by scraping an area of 25 mm x 75 mm from a single rock. The two measurements were made on separate samples, from separate rocks. Because there is not information from multiple points on multiple transects, the measurements may reflect a considerable amount of local variability, and may not be assumed to be representative of average densities in the reach sampled. Further, as the chlorophyll *a* and AFDW estimates come from separate rocks they are not necessarily paired samples, and inferences regarding the ratio of chlorophyll *a* and AFDW are suspect.

Results of the 2004 sampling are summarized for selected stations in Table 9.

Table 9. Summer 2004 Periphyton Sampling in the Klamath River

Station	Average Periphyton Chlorophyll *a* (mg/m^2)	Maximum Periphyton Chlorophyll *a* (mg/m^2)	Average Ash-Free Dry Weight (g/m^2)	Maximum Ash-Free Dry Weight (g/m^2)	Autotrophic Index (Average)
KR18952 – Klamath River below Iron Gate Dam	304.1	462.0	20.9	33.9	606.3
KR17608 – Klamath River above Shasta River	706.1	186.0	44.8	150.9	528.0
KR14261 – Klamath River above Scott River	120.4	353.0	68.7	141.3	684.6
KR12858 – Klamath River at Seiad Valley	65.5	122.0	25.6	54.4	1,982.2
KRWE – Klamath River above Trinity River	126.4	312.5	84.7	202.0	2,420.9
KRTC – Klamath River below Trinity	8.0	10.6	47.6	106.1	6,283.0
KRTG – Klamath River at Turwar	15.1	15.1	71.4	122.5	1,596.5

Notes: Samples at KR14261 combined with nearby samples from Walker Bridge Rd. Samples at KR17608 combined with nearby samples at Colliers Rest and Cottonwood Creek.

As noted above, the chlorophyll *a* and ash-free dry weight (AFDW) results are obtained from separate samples. Nonetheless, the autotrophic index (AI; ratio of AFDW to chlorophyll *a*) values are generally high, and appear to increase downstream. Collins and Weber (1978) suggest that an AI value greater than 400 is generally representative of "polluted" conditions in which the periphyton contains a high percentage of heterotrophs. In the lower Klamath, the AI values may reflect high levels of input of organic matter from eutrophic reservoirs upstream. The 2004 samples at KRTC and KRTG have very low chlorophyll *a* densities, but moderately high AFDW, suggesting largely heterotrophic communities.

Unfortunately, this sampling effort does not appear to provide a firm basis for calculating the ratio of chlorophyll *a* to AFDW (as mg/g), which is a key parameter for application of the QUAL2K-based prediction methods. The ratios from individual sample events reported by Eilers range from 0.1 to 96 mg/g, well outside of the range expected from algal stoichiometry, with a median of 1.1 and average of 7.1 (Figure 7) – probably due to the fact that the analyses are not from the same samples.

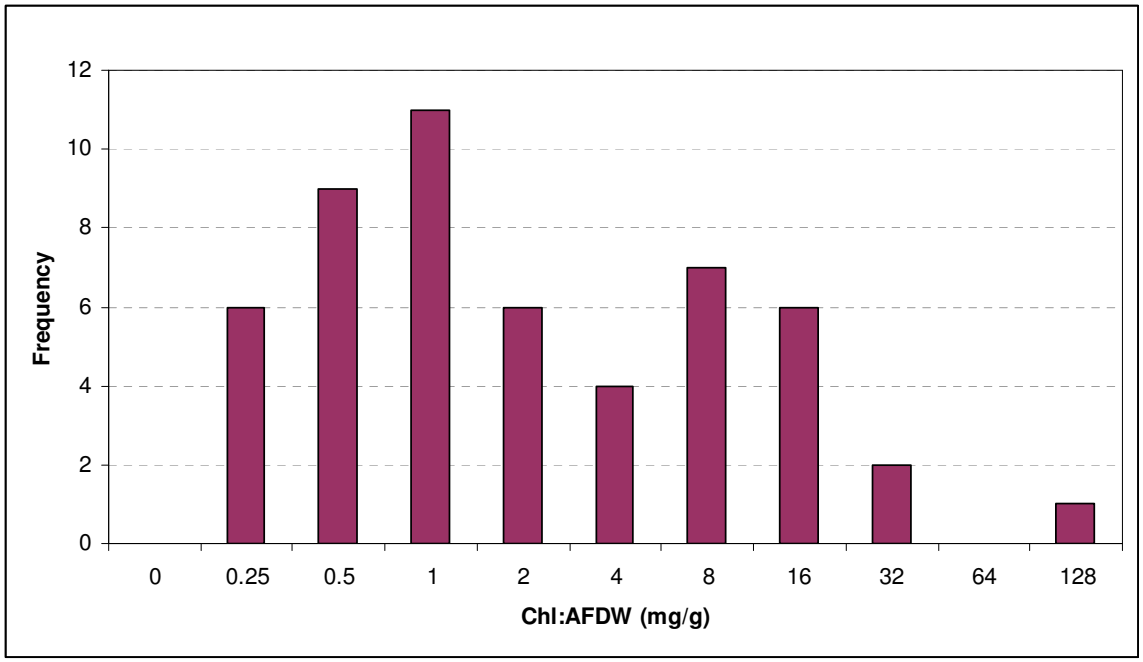

Figure 7. Histogram of Apparent Chlorophyll *a* to AFDW Ratios in 2004 Periphyton Data

Additional periphyton samples were collected by the Yurok Tribe in 2004 and 2006-2007 at KRWE (Weitchpec) and KRTG (Turwar). The 2004 results contain species composition data and AFDW, but not chlorophyll *a*. At KRWE, the average AFDW was 87.2, the maximum 122.5. At KRTG, the average AFDW was 108.2, the maximum 134.5. The 2006-2007 chlorophyll *a* results are shown in Table 10, reflecting revisions to the 2007 laboratory results reported to Tetra Tech by the Regional Board on April 7, 2008. AFDW was not reported for these data. The 2006 chlorophyll *a* results appear anomalously high, for unknown reasons. Communities at these stations were usually dominated by diatoms.

Table 10. Yurok Periphyton Sampling Results for 2006-2007

Station	Year	Average Periphyton Chlorophyll *a* (mg/m^2)	Maximum Periphyton Chlorophyll *a* (mg/m^2)
KRWE (Weitchpec)	2006	609.3	1086.2
	2007	123.6	326.0
KRTG (Turwar)	2006	325.8	651.7
	2007	73.4	163.0

It should be emphasized that it is very difficult to obtain reach average chlorophyll *a* densities in the Klamath, due to its size, depth, and velocity. It appears that all samples taken to date do not qualify as spatially averaged values, but are more representative of point concentrations. As a result, some of the observed maximum values are likely to be greater than the model predictions, which represent spatially averaged algal response under optimal growth conditions, not the maximum point density.

3.3.2.2 Chemical Water Quality

In contrast to periphyton, an extensive database of chemical water quality exists collected by multiple agencies. Earlier data were compiled into an Access database in 2004. Some of the earlier data have

since been corrected and substantial amounts of additional data have been collected since 2004. Accordingly, Tetra Tech worked with the Regional Board to develop a comprehensive tabulation of nutrient monitoring data in the Klamath.

The river data were separated into three time periods, 1996-2001, 2002-2004, and 2005-2007, which correspond approximately to the periods for which the reservoir BATHTUB scoping tools have been developed (2000, 2002, and 2005), and the periods during which periphyton samples are available (2004 and 2006-2007).

Statistics were calculated for the summer season (June – September). As periphyton is expected to have a moderately long response time to ambient nutrient concentrations, extreme values may not be particularly relevant. Therefore, the central tendency and range of the ambient data were described by the mean, median, 25th percentile, and 75th percentile (Table 11). To account for the influence of fluctuations in nutrient concentration on maximum algal response, predictions are made at the 75th percentile value. The ratio of total N to total P at these stations is typically less than the Redfield ratio of 7.2 (representing the typical cellular composition of algae), suggesting that nitrogen may frequently be the nutrient that is most limiting on algal growth.

3.3.2.3 Physical Data

Flow gaging data, and associated measurements, are available from five USGS gages between Iron Gate Dam and the Klamath estuary. Additional information on stream geometry, velocity, and stage is available from the calibrated hydrodynamic model of the Lower Klamath (PacifiCorp, 2005).

Table 11. Summer Nutrient Water Quality at Klamath River Stations below Iron Gate

		1996-2001				2002-2004				2005-2007			
	Station	Count (days)	Mean	25%^{le}	75%^{le}	Count (days)	Mean	25%^{le}	75%^{le}	Count (days)	Mean	25%^{le}	75%^{le}
PO_4-P (mg/L)	KR18952	42	0.152	0.110	0.173	32	0.120	0.100	0.143	32	0.105	0.088	0.130
	KR17608	6	0.198	0.150	0.240	16	0.131	0.113	0.160	21	0.105	0.080	0.133
	KR14261	6	0.204	0.140	0.250	14	0.117	0.091	0.140	19	0.103	0.088	0.120
	KR12858	41	0.124	0.083	0.150	24	0.084	0.060	0.110	8	0.067	0.049	0.075
	KRWE	0	ND	ND	ND	24	0.039	0.027	0.053	5	0.041	0.021	0.062
	KRTC	11	0.041	0.031	0.051	19	0.027	0.015	0.036	4	0.035	0.033	0.044
	KRTG	9	0.025	0.020	0.032	29	0.022	0.014	0.031	4	0.024	0.022	0.029
Org-P (mg/L)	KR18952	42	0.046	0.009	0.053	32	0.069	0.040	0.080	32	0.039	0.010	0.053
	KR17608	6	0.026	0.000	0.040	14	0.076	0.030	0.110	19	0.032	0.000	0.035
	KR14261	6	0.022	0.000	0.028	14	0.085	0.051	0.086	19	0.055	0.013	0.056
	KR12858	41	0.106	0.009	0.040	22	0.050	0.028	0.071	8	0.024	0.016	0.030
	KRWE	0	ND	ND	ND	24	0.051	0.029	0.060	5	0.015	0.013	0.017
	KRTC	11	0.036	0.013	0.059	19	0.054	0.032	0.080	4	0.015	0.014	0.016
	KRTG	9	0.068	0.035	0.081	29	0.050	0.020	0.071	4	0.017	0.015	0.019

	Station	1996-2001				2002-2004				2005-2007			
		Count (days)	Mean	25%le	75%le	Count (days)	Mean	25%le	75%le	Count (days)	Mean	25%le	75%le
NO_2+NO_3-N (mg/L)	KR18952	50	0.296	0.110	0.421	31	0.161	0.110	0.205	43	0.169	0.097	0.238
	KR17608	6	0.166	0.064	0.260	25	0.122	0.070	0.160	20	0.187	0.118	0.263
	KR14261	6	0.117	0.050	0.167	13	0.094	0.050	0.130	17	0.129	0.096	0.120
	KR12858	37	0.172	0.050	0.260	16	0.079	0.040	0.110	16	0.069	0.005	0.107
	KRWE	0	ND	ND	ND	26	0.042	0.033	0.040	4	0.005	0.005	0.005
	KRTC	8	0.084	0.040	0.100	26	0.071	0.020	0.040	4	0.006	0.005	0.006
	KRTG	8	0.076	0.040	0.100	22	0.039	0.040	0.040	4	0.026	0.023	0.028
NH_3-N (mg/L)	KR18952	50	0.091	0.043	0.085	29	0.059	0.050	0.050	43	0.024	0.005	0.039
	KR17608	6	0.043	0.024	0.047	25	0.067	0.020	0.060	20	0.020	0.005	0.034
	KR14261	6	0.041	0.028	0.044	12	0.031	0.000	0.050	19	0.011	0.005	0.005
	KR12858	37	0.032	0.000	0.040	17	0.065	0.050	0.050	16	0.008	0.005	0.011
	KRWE	0	ND	ND	ND	21	0.042	0.050	0.050	4	0.004	0.005	0.005
	KRTC	8	0.058	0.050	0.050	25	0.087	0.010	0.050	4	0.005	0.005	0.005
	KRTG	8	0.061	0.050	0.050	15	0.075	0.050	0.050	4	0.005	0.005	0.005
Org-N (mg/L)	KR18952	42	0.816	0.488	0.727	23	0.761	0.488	1.027	32	0.898	0.675	1.072
	KR17608	6	0.641	0.560	0.680	14	0.756	0.505	0.964	19	0.944	0.760	1.034
	KR14261	6	0.670	0.661	0.724	6	0.834	0.558	1.036	19	0.796	0.575	0.936
	KR12858	37	0.577	0.380	0.650	10	0.434	0.355	0.469	8	0.492	0.384	0.600
	KRWE	0	ND	ND	ND	12	0.432	0.225	0.502	5	0.257	0.213	0.291
	KRTC	8	0.289	0.150	0.388	23	0.306	0.120	0.335	4	0.200	0.175	0.221
	KRTG	8	0.356	0.146	0.375	10	0.212	0.138	0.238	4	0.205	0.191	0.244
Total N (mg/L)	KR18952	42	1.210	0.758	1.150	27	0.942	0.630	1.118	41	1.083	0.866	1.260
	KR17608	6	0.849	0.720	0.971	18	0.878	0.615	1.108	18	1.051	0.889	1.185
	KR14261	6	0.828	0.758	0.872	6	0.949	0.673	1.176	17	0.937	0.693	1.125
	KR12858	37	0.781	0.500	1.000	16	0.566	0.540	0.600	17	0.559	0.479	0.648
	KRWE	0	ND	ND	ND	14	0.480	0.265	0.530	5	0.235	0.180	0.272
	KRTC	8	0.431	0.240	0.538	23	0.386	0.190	0.440	4	0.211	0.189	0.231
	KRTG	8	0.493	0.296	0.538	14	0.305	0.240	0.328	4	0.231	0.212	0.272

Notes: Total Nitrogen calculated as some of Total Kjeldahl Nitrogen (TKN) plus NO_3-N plus NO_2-N where available. Non-detects treated as one-half the detection limit. Organic N calculated as TKN minus NH_3-N. Organic P calculated as Total P minus PO_4-P.

3.3.3 NNE Tool Application

The California NNE benthic biomass scoping tool was applied to the Klamath River in California to provide a scoping-level estimate of nutrient targets. Details on the development and use of this tool are available in Tetra Tech (2006).

Physical parameters for the scoping tool are summarized in Table 12 and explained further below.

Table 12. Parameters Specified for the NNE Tool Application

Station	Typical Summer Velocity (m/s)	Summer Depth for Analysis (m)	Unshaded Summer Solar Radiation (cal/cm^2/d)	Light Extinction Coefficient (m^{-1})	Days of Accrual	Chlorophyll *a* to AFDW Ratio
KR18952	0.65	0.45	528	0.725	185.7	5
KR17608	0.65	0.45	584	0.725	185.7	4
KR14261	0.69	0.45	527	0.725	122.8	4
KR12858	0.61	0.45	527	0.725	122.8	4
KRWE	0.69	0.45	524	0.760	81.9	4
KRTC	0.69	0.45	524	0.760	81.9	4
KRTG	0.69	0.45	526	0.760	69.1	4

Velocity

Stream velocity at each site was input as the "typical" summer value shown in the output of the RMA model of the Klamath River.

Depth

The RMA model output provides information on stage (or maximum depth) at each station, and average depth can be inferred from flow and cross-sectional area. However, the Klamath is a relatively wide river, and much of the potential benthic algal problem is believed to be associated with shallower water. It is therefore appropriate to evaluate impact at shallower depths, where light extinction in the water column is less of a factor. The 2004 periphyton samples were all collected in shallow water at a depth of approximately 0.45 m. Therefore, this depth was used in the scoping model applications.

Solar Radiation

Unshaded solar radiation for the summer period (June-August) was estimated based on latitude using the routine incorporated in the Benthic Biomass spreadsheet. The spreadsheet incorporates an approximation for shading effects on light availability as well. No data on local canopy and topographic shading were available; however, the majority of the Lower Klamath channel appears to be relatively open, so no shading was assumed, except at Seiad Valley. In that reach, the river flows in a N-S direction, whereas other sampled reaches have an approximately E-W orientation. Therefore, there is likely to be more topographic and canopy shading at Seiad Valley, and a value of 40 percent shading was selected.

Light Extinction Coefficient

Light extinction was estimated from turbidity. In general, light extinction is a function of water itself, dissolved colored organic material, phytoplankton, and inanimate particulate matter (Effler et al., 2005), and occurs through a combination of adsorption and scattering. In flowing streams, scattering by inorganic particulates is usually the dominant factor in light extinction, while scattering in the water

column is directly measured by a nephelometric turbidity meter as NTU (Gallegos, 1994). Therefore, an approximately linear relationship of light extinction to turbidity is expected in streams. Rather than implementing a complete optics model, we therefore rely on the simple empirical relationship of Walmsley et al. (1980), who established a regression relationship $K_e(PAR) = 0.1\ T + 0.44$, where $K_e(PAR)$ is the extinction rate of photosynthetically active radiation (PAR, per meter) and T is nephelometric turbidity (NTU). The relationship will vary according to the nature of suspensoids (Kirk, 1985), but is similar to results of other authors who suggest slopes of K_e relative to turbidity in the range of 0.06 to 0.12. Because turbidity has only a small effect on available light at the depths analyzed, the Walmsley relationship appears acceptable. The extinction coefficient was then estimated based on median summer turbidity, which ranged from 2.5 to 3.2 NTU.

Accrual

The scoping model provides an option to evaluate effects on expected maximum algal density based on days of accrual, using the relationship of Biggs (2000), where accrual time is defined as the number of days between events three-times the median flow. Accrual time was analyzed at each of the USGS gages. Because the Klamath is a large river with a multi-day response time, the number of events per year was estimated based on the count of times the hydrograph crossed the three-times-median threshold, rather than the number of individual days above the threshold. Resulting estimates (Table 13) were extrapolated to the nearest water quality monitoring station. The system shows a pattern of decreasing time between scouring events with distance downstream as additional major tributaries join.

Table 13. Estimated Days of Accrual (1985-2005 Data)

USGS Gage	Average Days of Accrual
11516530: Klamath River below Iron Gate	185.7
11520500: Klamath River near Seiad Valley	122.8
11523000: Klamath River at Orleans	81.9
11530500: Klamath River at Klamath	69.1

Half-Saturation Constants

Lacking site-specific data, half-saturation constants for nutrients are set at the levels described in Tetra Tech (2006). For the standard QUAL2K model, the optimized half-saturation constants were 0.206 mg/L for inorganic N and 0.00853 mg/L for inorganic P (Table 4 in Appendix 3 of Tetra Tech, 2006). For the revised QUAL2K model, the half-saturation constants are defined in relation to total nutrient concentrations, and vary from 0.0260 to 2.83 mg/L for total N, and from 0.0205 to 0.0470 mg/L for total P following the logistic regression model (Table 6 in Appendix 3 of Tetra Tech, 2006).

Chlorophyll *a* to AFDW Ratio

One of the most problematic parameters is the chlorophyll *a* to AFDW ratio (mg/g), where AFDW represents the fixed carbon biomass. The need for this parameter arises when the model formulation predicts biomass (as is done in the QUAL2K-based approach), while the target is specified as chlorophyll *a*. The ratio translates between the two, but can be highly variable. As noted above, only the 2004 sampling examined both AFDW and benthic chlorophyll *a*, but analyses were from small samples and the chlorophyll *a* and AFDW measures were obtained from scrapings from different rocks. These data do not provide a reliable basis for estimating the ratio in the Klamath.

Selection of an appropriate ratio is complicated by the fact that periphytic communities contain a mix of photosynthesizing autotrophs and heterotrophs, including bacteria and fungi, whose growth is based on

allochthonous carbon sources. The models are supposed to predict only photosynthetic biomass, but heterotrophs can also take up nutrients from the water column, so the predicted response of biomass as a function of nutrient concentrations likely includes both heterotrophic and autotrophic biomass. Further complications arise because (1) some algae exhibit mixotrophy, in which they are able to assimilate energy from fixed carbon compounds as well as by photosynthesis, and (2) exudates of benthic phototrophic algae may support bacterial and fungal heterotrophic populations, thus tying the heterotroph density to photosynthetic production.

In the development of the QUAL2K method (Tetra Tech, 2006), parameters of QUAL2K were "tuned" to provide a match to the predictions of Dodds' (2002) empirical model of maximum algal density when a chlorophyll *a* to AFDW ratio of 2.5 was assumed. Selection of this value was an appropriate compromise for a cross-sectional dataset, as the ratio of 2.5 corresponds to an autotrophic index of 400, generally presented as the upper limit of clean water conditions.

The CA NNE document (Tetra Tech, 2006) also noted that "alternate, site-specific ratios may be appropriate in specific waterbodies where appropriate information is available." For the Klamath, the Dodds method appears to underpredict maximum observed chlorophyll *a*, which also introduces a tendency for the QUAL2K-based methods, which are tuned to the Dodds method, to underpredict the maxima. Therefore, the chlorophyll *a* to AFDW ratio was increased from 2.5 to 4.0 at all stations except the station below Iron Gate, where a value of 5.0 was used.

3.3.4 Exploratory Data Analysis

Before applying the spreadsheet tool, an exploratory analysis was undertaken to examine the correlation between benthic chlorophyll *a* and nutrient concentrations. Both the average and the maximum benthic chlorophyll *a* from the 2004 sampling are plotted against the 75th percentile of summer average TN and TP concentrations from 2002-2004 water quality monitoring data in Figure 8. This suggests that observed periphyton density is indeed correlated to nutrient concentrations, with the strongest correlation (shown by higher R^2 value) between the observed maximum chlorophyll *a* and TN concentrations.

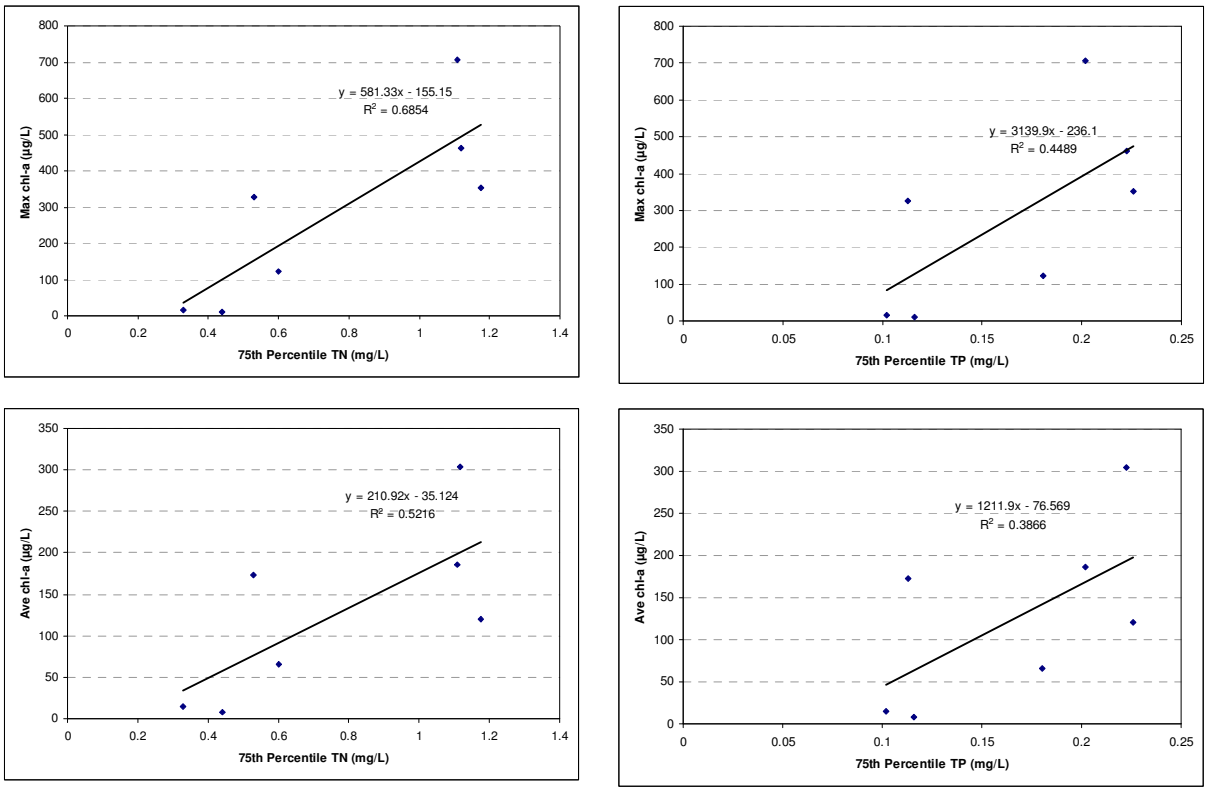

Figure 8. Relationship of Observed Periphyton Chlorophyll *a* to Nutrient Concentrations, 2004 Klamath Sampling

3.3.5 NNE Results

The NNE Benthic Biomass Predictor tool provides a variety of empirical and simplified parametric model approaches to predicting benthic algal response to ambient physical and chemical conditions. For this application, the tool was first used to predict maximum benthic chlorophyll *a* at each of the sites. As discussed in Tetra Tech (2006), benthic algal density is highly variable in time and space, and simplified models generally seem to do a better job of predicting the upper-bound estimate that describes maximum benthic algal density. The tool provides access to multiple predictions, but only three are presented here, all calculated at the 75th percentile summer nutrient concentration, as described above in Section 3.3.2. Of the empirical approaches, results are shown for the latest version of the Dodds model (Dodds, 2002), while for parametric approaches the results for both the standard QUAL2K and revised QUAL2K models (which are tuned to correspond to the Dodds' results on small streams) are shown, the latter both with and without an accrual adjustment (Table 14). The accrual adjustment has little effect on the upstream stations (where the estimated days of accrual are large), but does have a noticeable effect from station KRWE downstream. Of the other available methods, the 1997 version of the Dodds model has been superseded by the more detailed analysis of Dodds (2002). The Dodds method is of particular interest for comparison because results do not depend on the chlorophyll *a* to AFDW ratio.

Table 14. Predicted and Observed Maximum Benthic Chlorophyll *a* (mg/m^2)

Station	Period	Standard QUAL2K	Revised QUAL2K	Revised QUAL2K with Accrual Adjustment	Dodds 2002	Observed Maximum	Observed Average
KR18952 (below Iron Gate)	1996-2001	547	478	477	245		
	2002-2004	426	489	488	248	462	304.1
	2005-2007	441	504	503	241		
KR17608 (above Shasta)	1996-2001	399	363	362	236		
	2002-2004	344	404	403	251	706	186
	2005-2007	398	433	432	236		
KR14261 (above Scott)	1996-2001	333	347	314	231		
	2002-2004	307	406	368	244	353	120.4
	2005-2007	249	375	339	214		
KR12858 (Seiad Valley)	1996-2001	294	264	238	214		
	2002-2004	217	204	185	181	122	65.5
	2005-2007	181	222	201	169		
KRWE (above Trinity)	1996-2001	ND	ND	ND	ND		
	2002-2004	200	261	188	160	312.5	126.4
	2005-2007	30	172	124	115	1086 (2006) 326 (2007)	609 (2006) 124 (2007)
KRTC (below Trinity)	1996-2001	281	250	180	153		
	2002-2004	200	212	153	142	10.6	8
	2005-2007	34	147	105	98		
KRTG (Turwar)	1996-2001	281	246	155	153		
	2002-2004	200	181	114	125	15.1	15.1
	2005-2007	91	163	103	99	652 (2006) 163 (2007)	326 (2006) 73 (2007)

Model predictions of maximum benthic chlorophyll *a* concentrations for 2002-2004 are plotted against the 2004 observations of maximum and average concentrations in Figure 9.

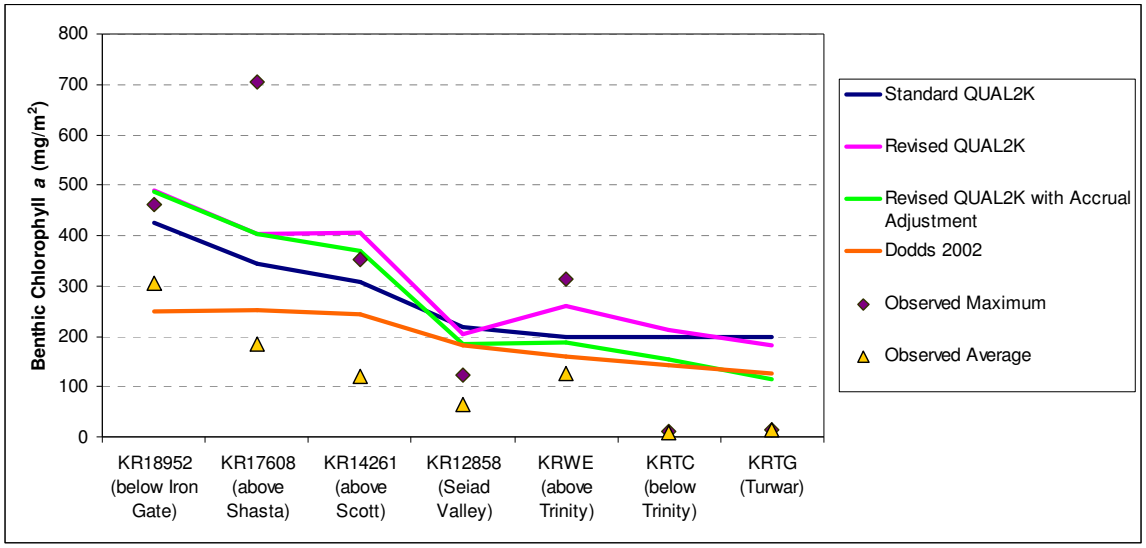

Figure 9. Model Predictions of Maximum Benthic Chlorophyll *a* for 2002-2004 and Observed Densities for 2004

None of the methods provide a perfect match to observations. Indeed, only general qualitative comparisons can be made, as the model predicts spatially averaged responses, whereas the observations reflect point data. In general, the revised QUAL2K approaches appear to do a reasonable job of replicating the spatial trend in observed maxima, while the Dodds results tend to be low. At three of seven stations, the predicted maximum using the QUAL2K approach is greater than the observed – which may only mean that the maximum was not sampled. At two other stations, the QUAL2K predictions are well less than the observed maximum. This may reflect the fact that the observed data are obtained from very small samples, without replication, that may not be representative of spatially averaged conditions in the reach.

Additional comments are warranted regarding several of the stations. For the station above Shasta, the plotted maximum of 706.1 mg/m^2 is for a sample taken at the mouth of Cottonwood Creek, a few miles upstream of station KR17608. Two samples taken at KR17608 had a maximum of only 81.5 mg/m^2. Reported maxima at the downstream stations of KRTC and KRTG were very low in 2004 (less than 20 mg/m^2); however, the Yurok samples from 2006 had a maximum of 652 mg/m^2 at KRTG. The 2004 results at these stations may be biased low relative to the seasonal maximum because they do not include samples from late summer, when periphyton densities are typically at their peak.

Both the data and the model representation of the data are subject to considerable uncertainty. Conditional on the suitability of the model, the tool can then be used to predict nutrient concentration targets needed to achieve a specified maximum algal density. As noted above, for the COLD and SPWN uses present in the Klamath, Tetra Tech (2006) recommends that the target should generally be between 100 mg/m^2 (BURC I/II boundary below which conditions may be deemed acceptable) and 150 mg/m^2 (BURC II/III boundary above which conditions are deemed unacceptable) for these designated uses.

For the Klamath, the models generally suggest that smaller reductions in total nitrogen than in total phosphorus are needed to reach the target range, and further that total phosphorus concentrations would need to be reduced to very low levels to achieve control of benthic algal growth by phosphorus alone. (Achieving the 100 mg/m^2 target by limiting phosphorus alone would require a total P goal of 2 µg/L.) This is consistent with the low observed total N to total P ratios, which suggest nitrogen limitation on algal growth. Therefore, nutrient limitations to achieve the maximum chlorophyll *a* targets are best

expressed in terms of total nitrogen goals (from which corresponding total phosphorus goals may be inferred through use of the Redfield ratio of 7.2, as in Dodds et al., 1997). The resulting total nitrogen goals for a maximum benthic chlorophyll *a* concentration target of 150 mg/m^2 are shown in Table 15, while Table 16 shows the corresponding estimates for a target of 100 mg/m^2 maximum benthic chlorophyll *a*.

Table 15. Total Nitrogen and Phosphorus Goals (mg/L) for Target of 150 mg/m^2 Maximum Benthic Chlorophyll *a* (TP values based on using Redfield ratio of 7.2)

Station	Revised QUAL2K	Revised QUAL2K with Accrual Adjustment	Dodds 2002
KR18952	0.18/0.025	0.18/0.025	0.34/0.047
KR17608	0.23/0.032	0.23/0.032	0.30/0.042
KR14261	0.23/0.032	0.28/0.039	0.33/0.046
KR12858	0.38/0.053	0.44/0.061	0.38/0.053
KRWE	0.24/0.033	0.41/0.057	0.50/0.069
KRTC	0.24/0.033	0.41/0.057	0.49/0.068
KRTG	0.24/0.033	0.51/0.071	0.53/0.074

Table 16. Total Nitrogen and Phosphorus Goals (mg/L) for Target of 100 mg/m^2 Maximum Benthic Chlorophyll *a* (TP values based on using Redfield ratio of 7.2)

Station	Revised QUAL2K	Revised QUAL2K with Accrual Adjustment	Dodds 2002
KR18952	0.08/0.011	0.08/0.011	0.11/0.015
KR17608	0.11/0.015	0.11/0.015	0.10/0.014
KR14261	0.11/0.015	0.14/0.019	0.11/0.015
KR12858	0.19/0.026	0.23/0.032	0.13/0.018
KRWE	0.11/0.015	0.21/0.029	0.17/0.024
KRTC	0.11/0.015	0.21/0.029	0.17/0.024
KRTG	0.11/0.015	0.26/0.036	0.18/0.025

Results for the 150 mg/m^2 target are re-expressed as reductions in TN concentration relative to observed summer average concentrations for the 2005-2007 period based on the revised QUAL2K with accrual adjustment analysis in Table 17. Concentrations observed in the 1996-2001 period are somewhat different, but suggest a similar spatial pattern of needed reductions (Table 18).

Table 17. Reductions in TN Concentrations Relative to 2005-2007 Observations to Achieve the 150 mg/m^2 Target

Station	Percent Reduction in Summer TN Concentration	TN/TP 2005-2007 Summer Average Concentration (mg/L)	Revised QUAL2K with Accrual Adjustment TN/TP Goal (mg/L)
KR18952	83%	1.08/0.14	0.18/0.025
KR17608	78%	1.05/0.14	0.23/0.032
KR14261	70%	0.94/0.16	0.28/0.039
KR12858	21%	0.56/0.091	0.44/0.061
KRWE	0%	0.24/0.056	0.41/0.057
KRTC	0%	0.21/0.050	0.41/0.057
KRTG	0%	0.23/0.041	0.51/0.071

Table 18. Reductions in TN Concentrations Relative to 1996-2001 Observations to Achieve the 150 mg/m^2 Target

Station	Percent Reduction in Summer TN Concentration	TN/TP 2005-2007 Summer Average Concentration (mg/L)	TN/TP Goal (mg/L)
KR18952	85%	1.21/0.20	0.18/0.025
KR17608	73%	0.85/0.22	0.23/0.032
KR14261	66%	0.83/0.23	0.28/0.039
KR12858	44%	0.78/0.23	0.44/0.061
KRWE	No data	No data	0.41/0.057
KRTC	5%	0.43/0.077	0.41/0.057
KRTG	0%	0.49/0.093	0.51/0.071

Table 17 and Table 18 suggest that to achieve the desired reductions in benthic algal density at all stations would require reductions in summer TN concentrations of up to 85 percent. (Achieving targets by controlling TP directly would require reductions of approximately 98 percent at stations through Seiad Valley.) The results for the T4BS1 allocation scenario in Iron Gate Reservoir (see Section 3.3) indicate that this scenario, which is predicted to achieve lake targets, would result in reductions of about 73 percent in summer TN concentrations and about 89 percent in summer TP concentrations for the 2000 simulation. Thus, load reductions in excess of those needed to meet DO criteria and achieve lake planktonic chlorophyll *a* targets may be needed to meet maximum periphyton chlorophyll *a* targets in the Klamath below Iron Gate.

Application of the benthic biomass tool using the 75th percentile summer concentrations in the outflow from Iron Gate predicted by the T4BS1 scenario with benthic flux off results in a prediction of maximum benthic algal chlorophyll *a* at Station KR18952 below Iron Gate of 164 mg/m^2 – slightly in excess of the target – using the revised QUAL2K-based methods. For the same conditions, the Dodds (2002) approach yields a prediction of 84 mg/m^2, well below the target.

3.3.6 Natural Conditions Analysis

The draft CA NNE document (Tetra Tech, 2006) recommended an upper limit of 150 mg/m^2 chlorophyll *a* to support uses in waters of the State of California, yet also recommends that the target should not be set lower than the value expected under natural conditions. This target of 150 mg/m^2 has not been adopted by the State Board and remains open for further evaluation. [The Hoopa Valley Tribe has a regulatory target of 150 mg/m^2 that has been adopted and approved, and applies to the small section of the Klamath River that passes through the Hoopa Valley Tribal lands.]

To examine potential natural conditions in the Klamath River, concentration results from the TMDL Model T1BS natural conditions run were summarized for summer (June-September) conditions. This model run has point sources eliminated and dams out. Current flow leaving Upper Klamath Lake and Klamath Straits Drain in Oregon is continued, but with concentrations reduced to be compliant with the Upper Klamath Lake TMDL. The T1BS run uses 2000 meteorological conditions.

In the dams out simulation, nutrient retention and processing by the Klamath reservoirs is eliminated. This results in changes in the magnitude, timing, and speciation of nutrient loads reaching the lower Klamath River.

Output from the T1BS natural conditions was used to provide input to the NNE benthic biomass tool. Evaluation was made at four locations: below Iron Gate (KR18952), Seiad Valley (KR12858), above Trinity (KRWE), and at Turwar (KRTG). The range of summer average water quality for the natural conditions run is summarized at the 75th percentile level (as was done with the NNE tool to predict maximum benthic chlorophyll *a* in previous sections) and compared to recent observed water quality in Table 19.

Table 19. 75th Percentile of Natural Condition Water Quality (Model Run T1BS) Compared to Observed Water Quality in the Klamath River

	Station	T1BS	1996-2001 Observed	2002-2004 Observed	2005-2007 Observed
PO_4-P (mg/L)	KR18952 – Klamath River below Iron Gate	0.0190	0.173	0.143	0.130
	KR12858 – Klamath River at Seiad Valley	0.0515	0.150	0.110	0.075
	KRWE – Klamath River above Trinity River	0.0601	No Data	0.053	0.062
	KRTG – Klamath River at Turwar	0.0768	0.032	0.031	0.029
Org-P (mg/L)	KR18952	0.0216	0.053	0.080	0.053
	KR12858	0.0220	0.040	0.071	0.030
	KRWE	0.0184	No Data	0.060	0.017
	KRTG	0.0161	0.081	0.071	0.019
Total P (mg/L)	KR18952	0.0406	0.226	0.223	0.183
	KR12858	0.0735	0.190	0.181	0.105
	KRWE	0.0785	No Data	0.113	0.079
	KRTG	0.0929	0.113	0.102	0.108
NO_2+NO_3-N (mg/L)	KR18952	0.0777	0.421	0.205	0.238
	KR12858	0.0957	0.260	0.110	0.107
	KRWE	0.1093	No Data	0.040	0.005
	KRTG	0.1298	0.100	0.040	0.006
NH_3-N (mg/L)	KR18952	0.0831	0.085	0.050	0.039
	KR12858	0.1077	0.040	0.050	0.011
	KRWE	0.1256	No Data	0.050	0.011
	KRTG	0.1467	0.050	0.050	0.005
Org-N (mg/L)	KR18952	0.2671	0.727	1.027	1.072
	KR12858	0.2838	0.650	0.469	0.600
	KRWE	0.2502	No Data	0.502	0.291
	KRTG	0.2598	0.538	0.328	0.244
Total N (mg/L)	KR18952	0.4279	1.150	1.118	1.260
	KR12858	0.4872	1.000	0.600	0.648
	KRWE	0.4851	No Data	0.530	0.272
	KRTG	0.5364	0.538	0.328	0.272

Unlike monitoring results for existing (dams-in) conditions (see above, Table 11), the 75th percentile total nutrient concentrations during the summer tend to increase downstream under the TMDL Model T1BS run. This seems to occur because concentrations in most of the downstream tributaries were kept at existing levels for the T1BS scenario, while upstream concentrations leaving Iron Gate Dam decreased significantly. In addition, the model output reflects continuous subhourly simulation, while the observations are discrete day time grab samples, which may confound direct comparison.

Table 19 also shows that the 75th percentile summer total nitrogen concentrations under natural conditions appear to be greater than the concentrations estimated as needed to meet the 150 mg/m^2 maximum benthic chlorophyll *a* target in the analysis of existing conditions provided above in Table 15. This suggests that natural conditions may result in a tendency for elevated benthic algal densities in the Klamath River.

The dams-out condition will also result in more frequent scouring flows and less days of accrual (time between potential scouring events), which may tend to reduce maximum benthic algal growth. However, data were not available for a long-term analysis of the frequency of scouring flows for the T1BS model conditions. To approximate this effect, the days of accrual for the Revised QUAL2K application with accrual adjustment was set at 69.1 days – the value currently used for the Turwar gage, which is furthest downstream and least affected by the dams on the upper Klamath.

Results of applying the benthic biomass spreadsheet tool to the TMDL Model T1BS conditions are summarized in Table 20. Consistent with the predicted nutrient concentrations, there is no longer a strong spatial gradient in predicted maximum benthic chlorophyll *a* concentrations under the T1BS natural conditions scenario. The standard QUAL2K predictions are much higher than the other approaches due to the increased fraction of inorganic nutrients, which enter directly into the solution for this model, but not the other approaches. The Revised QUAL2K model with accrual adjustment suggests maxima right around the 150 mg/m^2 target, while the QUAL2K approaches without accrual adjustment predict higher densities. The Dodds (2002) approach also predicts maximum densities less than 150 mg/m^2, but results from this model were generally much lower than that obtained for other approaches in the analysis of existing conditions.

The predicted ability to meet the 150 mg/m^2 target using the Revised QUAL2K approach (with accrual adjustment) only occurs due to the assumption of reduced days of accrual. For example, if days of accrual at KRWE are assumed to be 81.9, as in the existing conditions (dams-in) application, the resulting predicted maximum benthic chlorophyll *a* density would be 166, rather than 145 mg/m^2. It is thus not clear from the benthic biomass spreadsheet analysis that the 150 mg/m^2 target could be met under natural conditions A more detailed analysis of the frequency of scouring flows expected under the dams-out natural conditions may be advisable to ascertain the extent to which this phenomenon is likely to limit excess benthic algal density.

Table 20. Predicted Maximum Benthic Chlorophyll *a* (mg/m2) Under TMDL Model Run T1BS Natural Conditions (Dams Out) for Year 2000

Station	Standard QUAL2K	Revised QUAL2K	Revised QUAL2K with Accrual Adjustment	Dodds 2002
KR 18952 (below Iron Gate)	338	250	157	113
KR12858 (Seiad Valley)	246	174	109	135
KRWE (above Trinity)	350	231	145	137
KRTG (Turwar)	377	246	154	147

3.4 DISCUSSION OF KLAMATH RIVER NNE RESULTS

Prediction of periphyton biomass is inherently difficult. This problem is compounded by several factors, including the weak relationship between periphyton biomass and benthic chlorophyll *a* and the sparse and uncertain data available for the Klamath River. The biomass to chlorophyll *a* relationship is expressed through the chlorophyll *a* to AFDW ratio, which is clearly a major source of uncertainty in the QUAL2K-based applications. The observed data are limited, and have been obtained from small samples that may not accurately reflect the reach-averaged conditions predicted by the tool.

Due to the uncertainties in predicting benthic chlorophyll *a*, it may be preferable to define periphyton targets for the Klamath River in terms of AFDW, although more data are needed to establish such a target.

As a result of these caveats, the main value of the benthic biomass tool is in predicting relative changes in benthic chlorophyll *a*, rather than precise estimates. It is clear that significant reductions in summer nutrient concentrations would be needed to meet a target of 150 mg/m^2 maximum benthic chlorophyll *a*; however, the predicted magnitude of the needed reductions is highly uncertain. The reductions that would occur as a result of the T4BS allocation scenario to achieve DO criteria will certainly result in improvements in periphyton density in the Klamath River, but may or may not be sufficient to achieve the BURC II/III target of 150 mg/m^2 maximum benthic chlorophyll *a*. Due to the considerable uncertainty in the NNE analysis. additional data should continue to be collected to build a better understanding of the relationship between nutrient concentrations and periphyton density and support the development of more sophisticated, site-specific models. The RMA model application for Klamath River TMDL development already provides a potential framework for evaluating the benthic algal target in the river; however, the model predictions need to be refined with more data to better assess impacts on beneficial uses in addition to effects on DO, including the formation of periphyton mats that may impair recreational uses, alter the benthic community, and potentially increase the detrimental effects of parasites on salmonid populations.

Finally, although the draft CA NNE document (Tetra Tech, 2006) recommends an upper limit of 150 mg/m^2 chlorophyll *a* to support uses, this target has not been adopted by the State or Regional Board and remains open for further evaluation. As mentioned previously, this target does apply to the small section of the Klamath River that passes through the Hoopa Valley Tribal lands, where a criterion of 150 mg/m^2 has been adopted and approved.

Targets should not be set lower than a value expected under natural conditions. As discussed in Section 3.3.6, the natural condition maximum benthic chlorophyll *a* concentration on Hoopa Valley lands is likely to be near 150 mg/m^2, but may be somewhat higher, depending on the assumptions regarding frequency of scour.

Perhaps more importantly, it would be consistent with the CA NNE approach to develop a site-specific target based on a risk analysis to support beneficial uses in the system. A key here may be establishing the periphyton conditions (and relevant indicator metrics) that are consistent with managing the parasite *Ceratomyxa shasta* at levels that are consistent with maintaining a healthy salmonid population; however, research has not yet advanced to the point where a quantitative target can be set on this basis.

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APPENDIX 2

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MEMORANDUM

To:	Klamath TMDL Technical Team	**Date:**	February 12, 2008
From:	Jonathan Butcher	**Project:**	Klamath
Subject:	Nutrient Dynamics in the Klamath	**Tt Pjn:**	20729-02

Oregon DEQ (DEQ) and California's North Coast Regional Water Quality Control Board (Board) are developing Total Maximum Daily Loads (TMDLs) for the Klamath River to address impairments associated with dissolved oxygen, temperature, nutrients, pH, and chlorophyll *a* – all of which are ultimately affected by the dynamics of nutrients, algal growth, and organic matter transport in the system. The river begins at Upper Klamath Lake in Oregon and encompasses 15,722 square miles in the states of Oregon and California, flowing from the Cascades in Oregon westerly and southerly to the Pacific Ocean in Del Norte, CA. The system has several unusual characteristics. First, the gradient increases downstream. Second, the source of the river is an aging eutrophic lake, which leads to nutrient concentrations that are highest in the headwaters, while downstream tributaries generally have lower nutrient concentrations than the mainstem. Finally, there are a series of dams in the upper third of the watershed, forming a segmented system.

The Klamath is one of the major salmon rivers of the western United States, so interest in the protection, management, and restoration of the system is high. Nutrients in the system are dominated by the loading leaving Upper Klamath Lake, which cannot easily be controlled. The temporal and spatial pattern of water quality downstream is largely controlled by processes that retain, transform, or release nutrients within the impounded and free-flowing reaches, including growth of planktonic algae (primarily in impoundments), settling of nutrients to and regeneration of nutrients from the sediment (also primarily in impoundments), and uptake/release of nutrients by periphytic algae (primarily in free-flowing reaches).

PacifiCorp, which operates federally-licensed hydroelectric projects on the river, developed a simulation model of the river, linking CE-QUAL-W2 models of the impoundments with an RMA-11 model of the free-flowing reaches (PacifiCorp, 2005). This model was subsequently updated and recalibrated by Tetra Tech for USEPA, DEQ, and the Board, and forms a basis for developing the TMDL and potential management scenarios to meet the TMDL. While Tetra Tech has developed a draft model calibration report, a final public version of the report has not yet been released. Stakeholder review of the modeling effort has, however, provoked some questions regarding the processes controlling nutrient dynamics in the impounded and free-flowing reaches (particularly their relative importance), and whether the model accurately represents these processes (Asarian, 2007; PacifiCorp, 2007). In addition, several reports have been published on nutrient dynamics in the system.

To help resolve these issues, the Board and USEPA requested a review of the reports that analyze nutrient dynamics in the Klamath River system, with particular emphasis on evaluation of the effects of reservoirs in California on water quality relative to free-flowing reaches. The review was conducted by Dr. Jonathan Butcher, who is familiar with the Klamath River system, but was not directly involved in the model development effort.

1 Key Issues

The disputes over nutrient dynamics in the Klamath River system are intimately tied to the policy debate over the potential removal of hydropower dams in the Upper Klamath. PacifiCorp (2005) created the original linked water quality simulation models of the system (RMA-11 for the free-flowing reaches and CE-QUAL-W2 for the impoundments), and applied them to the 2000-2004 period, with calibration based on more intensive data collected in 2000. In a series of reports, Asarian and Kann (working variously on behalf of the Yurok and Karuk tribes) raised concerns that the model overestimated nutrient retention rates in the impoundments, while underestimating nutrient retention rates in the free-flowing reaches, resulting in an unrealistic estimate of benefits of the impoundments in controlling downstream nutrient concentrations (see Asarian and Kann, 2006a, 2006b).

Tetra Tech, for USEPA, subsequently recalibrated the PacifiCorp models for year 2000 (with validation application to the reaches above Iron Gate Dam for 2002). The recalibration effort did result in lower nutrient retention rates within the reservoirs and higher retention rates within the free-flowing segments – suggesting that some of the original criticisms of the model were correct. Asarian (2007) has continued to express concerns that the model under-represents nutrient retention in the free-flowing reaches of the river downstream of Iron Gate Dam – and thus does not provide a fair evaluation of the conditions that would result from conversion of impoundments back to free-flowing reaches. On the other side, PacifiCorp (2007) has contended that the analyses of Asarian and Kann are flawed in a variety of ways, while defending the model results.

The truth of the matter is that water quality data are fairly sparse in this system – which lends considerable uncertainty to both direct evaluations of observations and model setup and calibration. The station immediately below Iron Gate Dam (RM 189.73) has samples from 1996-2004, on a biweekly basis for 1998-2002 and monthly in other years. The key downstream station for evaluating nutrient retention at Seiad Valley (RM 130.85) has a similar density of samples only for 1998-2001. Further, samples have generally not been collected in January through April, meaning that a large portion of the total annual flow has not been sampled (see Figure 1). As a result, the mass balance and retention calculations of Asarian and Kann (2006a) are based on June through October, not complete years, preventing full closure of the mass balance.

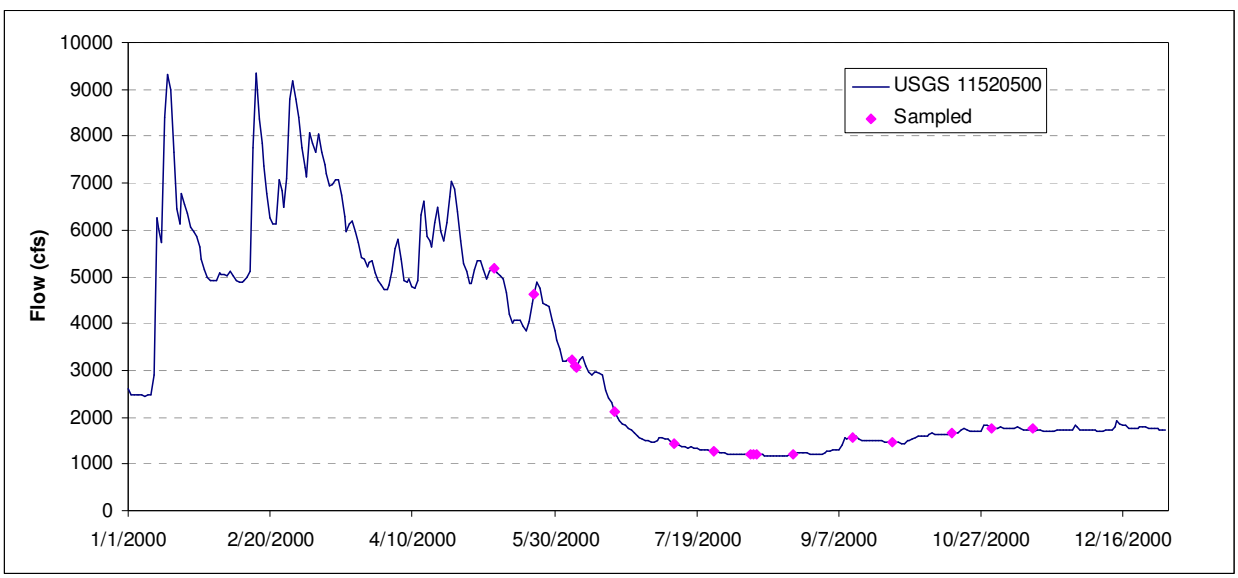

Figure 1. Daily Flows and Water Quality Samples, Klamath River at Seiad Valley, 2000

For the modeling, focus has been on 2000, as this is the year with the best information on tributary loads. Tetra Tech has run the complete, recalibrated model *only* for 2000. Indeed, due to the short residence time and elevated loads from the headwaters at Upper Klamath Lake, the model predictions are strongly determined by the boundary conditions (upstream load and relative dilution provided by the downstream tributaries), and results for years in which the boundary conditions are not well-defined are largely speculative.

It is also important to recognize that the patterns of nutrient retention are likely to vary significantly from year to year. The data analyzed by Asarian and Kann (2006a) show consistent seasonal retention of total Nitrogen between Iron Gate and Seiad Valley in some years (e.g., 2001 and 2002), but not in others. Notably, for 2000 the retention calculated for most sample dates appears to be close to the expected range of uncertainty for nitrogen concentration measurements[1] except for one date, which results in an estimation of net retention over the June-October period. The authors note (p. 42): "most of the positive retention in the Iron Gate (KR18973) to Seiad Valley (KR12858) reach for the 2000 season was due to a single high sample on 7/11/2000 at Iron Gate, when TKN was 4.5 mg/L… There are 313 TKN samples in the database below Iron Gate taken from 1971 to 2004 and this was the highest measurement, with no other samples above 2.0 mg/L…"

Both the empirical analyses of nutrient trend and the model predictions are uncertain as a direct result of the sparse data. Biweekly or monthly samples are likely to provide an inaccurate estimate of the nutrient mass entering or leaving a reach, introducing uncertainty into direct estimates of load, while similarly creating uncertainty in model forcing functions. Further, it is not appropriate to compare long term trends from the data to model results from a single year, as year-to-year variability is likely significant.

In sum, the empirical work of Asarian and Kann is most appropriate as a long-term, statistical estimate of typical removal rates. The model output, while also uncertain, provides an estimate for a specific set of flow and boundary conditions. The two should be qualitatively similar, but not identical.

2 River Retention of Nutrients

Physical and biological processes in river reaches can result in net removal or temporary retention of nutrients. One of the major processes for temporary retention is uptake of nutrients by periphyton. Periphytic algae, as well as heterotrophic organisms, require nutrients for growth and remove inorganic nutrients from the water column, converting them to organic biomass. Heterotrophs also remove organic matter as foodstock. This storage, however, is temporary. In addition to normal dieoff and predation, periphyton is subject to scour and transport downstream during high flow events.

Tanner and Anderson (1996) demonstrated that periphyton (dominantly *Cladophora*) were very effective in reducing dissolved inorganic nitrogen loads downstream of wastewater treatment plants in the South

[1] Total nitrogen is estimated as the sum of total Kjeldahl nitrogen (TKN) and nitrite-plus-nitrate nitrogen. For USGS analyses, relative standard deviations (RSDs) for these measurements in natural waters are typically in the range of 8 to 10 percent (e.g., Lambing and Cleasby, 2006), although Campbell (2001) reported RSD's less than 5 percent for Klamath River nutrient samples. Other laboratories may achieve differing levels of precision. More importantly, TKN is reported to the nearest 0.1 mg/L, while nitrite and nitrate nitrogen are reported to the nearest 0.01 mg/L, reflecting the underlying precision in the analytical methods. In the Klamath below Iron Gate, TKN often constitutes about 90 percent of total nitrogen. At a typical total nitrogen concentration of about 0.8 mg/L, there is thus a built in reporting uncertainty (reflecting the underlying analytical uncertainty) of around ±14 percent. Additional uncertainty is introduced by sampling procedures, as samples may not be fully representative of the complete flow-weighted average concentration passing a given point, and there may also be systematic changes in nutrient concentrations over the diurnal cycle due to algal influences. Finally, calculation of nutrient retention requires use of flows, which are also subject to measurement uncertainty, further decreasing the precision of mass transport estimates obtained by multiplying flow times concentration.

Umpqua River, OR. Similarly, locations in the Bow River in Alberta supported dense *Cladophora* and macrophyte growths that were sensitive to nitrogen load and effective in removing inorganic N from the water column (Sosiak, 2002). Such biological uptake is, however, largely temporary in nature, as biomass follows seasonal cycles with release of nutrients as biomass declines in the fall. Decaying periphyton mats may also promote anoxic conditions that lead to denitrification and loss of nitrogen from the system. Dodds (2003) summarized the role of periphyton in removing phosphorus from aquatic systems. Some of this storage is also temporary; however, Dodds also points out that localized increases in pH during photosynthesis can lead to increased precipitation of calcium phosphate, concurrent deposition of carbonate-phosphate complexes, and long term burial losses of phosphorus.

Temporary retention in river reaches also occurs as a result of settling and storage of particulate matter, including organic detritus. Inorganic orthophosphate and, to a lesser extent, ammonium can also sorb to sediment particles and settle out. These processes also largely constitute temporary retention, as the stored particulate matter can be remobilized by scouring flows.

Permanent removal of nutrient mass can also occur in several ways. For nitrogen, denitrification and conversion to nitrogen gas results in a loss of nitrogen from the water to the air. This may be balanced by fixation of atmospheric nitrogen by certain types of cyanophytes, but these are usually not dominant in flowing waters. Water lost to deep groundwater, agricultural diversions, or riparian wells can remove nutrients, and is more important for nitrogen, which is more soluble than phosphorus. Effective removal of phosphorus may also occur due to burial in deposits that are not readily remobilized (due, for instance, to stream meander and cutoffs), export to the floodplain, or conversion to tightly bound, insoluble mineral forms. These latter processes tend to be of less importance in higher gradient systems, so net rates of removal for TP are expected to be less than net rates of removal for TN in a system like the Klamath.

In general, temporary retention is most important during lower flow periods, which tend to coincide with the growing season. Temporary retention does not, in the end, change the nutrient load that is delivered downstream; however, it can significantly affect both the timing and bioavailability of load delivery. While extensive periphyton communities remain intact below a source area, they may substantially reduce the nutrients available to support algal growth downstream – as appears to be observed in the Klamath. High flows in the Klamath typically occur in the winter and spring. The net effect of temporary retention should thus be to shift much of the nutrient load from the summer and fall to the winter and early spring – a period for which there are very few water quality observations.

The likelihood of significant scour of periphyton can be evaluated in terms of days of accrual, using the relationship of Biggs (2000), where accrual time is defined as the number of days between events three-times the median flow. Accrual time was analyzed at each of the USGS gages. Because the Klamath is a large river with a multi-day response time, the number of events per year was estimated based on the count of times the hydrograph crossed the three-times-median threshold, rather than the number of individual days above the threshold. Resulting estimates (Table 1) show a pattern of decreasing time between scouring events with distance downstream as additional major tributaries join.

Table 1. Estimated Days of Accrual (1985-2005 Data)

USGS Gage	Average Days of Accrual
11516530: Klamath River below Iron Gate	185.7
11520500: Klamath River near Seiad Valley	122.8
11523000: Klamath River above Shasta River	81.9
11530500: Klamath River at Klamath	69.1

Nutrient concentrations measured in the Klamath tend to decrease with distance downstream from Iron Gate Dam, as has been noted by various authors. It is important to note, however, that nutrient retention needs to be evaluated as a mass balance, based on loads, not concentration trends. In the Klamath below Iron Gate, the concentrations of nutrients in many of the tributaries are generally lower than those in the mainstem; thus a reduction in mainstem concentration is expected solely as a result of dilution, regardless of whether there is any true retention.

2.1 REVIEW OF NUTRIENT MASS BALANCE ESTIMATES FOR THE LOWER KLAMATH

Asarian and Kann (2006b) evaluated the PacifiCorp modeling and contended that it overestimated retention in the reservoirs, while underestimating retention in flowing reaches. This analysis, however, is based primarily on comparison of longitudinal concentration distributions in the Klamath mainstem. This is potentially misleading for two reasons. First, the rate of change in concentration along the length of the river is in large part a function of nutrient concentrations in incremental tributary flow, which is not well characterized. Second, the sparseness of the monitoring data introduces considerable uncertainty into the analysis.

Asarian and Kann (2006a) do provide a mass balance evaluation in terms of loads, again concluding that retention in flowing reaches is underestimated by the model. The conclusions of this effort depend in part on how loads are estimated, which can be problematic for sparse data. Asarian and Kann did this as follows:

- Total N is calculated as the sum of Total Kjeldahl Nitrogen (TKN) and NO_3-N+NO_2-N.
- The biweekly (or monthly) concentration data were used to create a continuous daily estimate using linear interpolation between points.
- Resulting daily TN concentration was multiplied by flow to obtain load.
- Retention rates between stations were calculated for June-October, the period for which most monitoring is available.

As noted above, because only a seasonal estimate can be made, the approach measures net retention, not removal, and a significant portion of the retained load may be mobilized and moved downstream by winter flows. Methodologically, the other major concern is the use of linear interpolation.

Estimating constituent mass loads from point-in-time measurements of water-column concentrations presents many difficulties. Load is determined from concentration multiplied by flow, and while measurements of flow are continuous, only intermittent measurements of concentration are available. Calculating total load therefore requires “filling in” concentration estimates for days without samples. The process is further complicated by the fact that concentration and flow are often highly correlated with one another, and many different types of correlation may apply. For instance, if a load occurs primarily as a result of nonpoint soil erosion, flow and concentration will tend to be positively correlated; that is, concentrations will increase during high flows, which correspond to precipitation-washoff events. On the other hand, if load is attributable to a relatively constant point discharge, concentration will decrease as additional flow dilutes the constant load. In most cases, a combination of processes is found.

Preston et al. (1989) undertook a detailed study of advantages and disadvantages of various methods for calculating annual loads from tributary concentration and flow data. Their study demonstrates that simply calculating loads for days when both flow and concentration have been measured and using results as a basis for averaging is seldom a good choice. A method dependent on interpolation between measured concentrations is likely to have similar problems. Depending on the nature of the relationship between flow and concentration, more reliable results may be obtained by one of three approaches:

- Averaging Methods: An average (e.g., yearly, seasonal, or monthly) concentration value is combined with the complete time series of daily average flows
- Regression Methods: A linear, log-linear, or exponential relationship is assumed to hold between concentration and flow, thus yielding a rating-curve approach
- Ratio Methods: Adapted from sampling theory, load estimates by this method are based on the flow-weighted average concentration times the mean flow over the averaging period and performs best when flow and concentration are only weakly related.

No single method provided superior results in all cases examined by Preston et al.; the best method for extrapolating from limited sample data depends on the nature of the relationship between flow and concentration, which is typically not known in detail.

Thus, the accuracy of the interpolation approach will depend in large part on whether there is correlation between flow and concentration. If the two variables are truly independent, then no error will be introduced by this approach. If they are correlated, the approach is sub-optimal.

Reducing the potential impact of these issues is the fact that the Upper Klamath is a highly controlled system, with multiple reservoirs. These reservoirs should serve to damp out correlations between flow and concentration, particularly for the reach between Iron Gate and Seiad. As the river accumulates more uncontrolled tributary flow downstream, correlation may reemerge, rendering the load estimates suspect. (Indeed, this might be why Kann and Assarian detected no nitrogen retention below Orleans).

Campbell (2001) analyzed monitoring data from 1996 through 1998 from Keno Dam to Seiad Valley, and concluded that there was a negative correlation between flow and concentration for all nutrient species, and that this negative correlation became stronger downstream (as far as Seiad Valley). This type of situation can arise when nutrient loads are dominated by lake sources, and tributary stormflow contributions – even if elevated relative to baseflow – serve to dilute the mainstem concentration. Reexamination of the detailed 2000 USGS monitoring at Seiad Valley confirmed a negative correlation between TKN and flow, but showed essentially no correlation between TP and flow.

Further downstream, USGS nutrient water quality monitoring from 1973 to 2003 is available at Klamath, CA (USGS gage 11530500), and these data were examined as a check. As shown in Figure 2, concentration is only weakly related to flow at this station, and, while the slope is positive, it is not significant. The relationships remain weak if results are stratified into summer and winter seasons.

To the extent that flow and concentration are truly independent of one another, the simple interpolation method used by Asarian and Kann will not introduce error. However, where and when a negative correlation between flow and concentration exists, the interpolation approach will tend to overerestimate total load – because concentrations that are too high will tend to be applied to the high flow events that constitute the bulk of the total movement of mass. If, as expected, the negative correlation is strongest immediately downstream of Iron Gate and decreases downstream, this could introduce a bias that results in overestimation of the nutrient retention between Iron Gate and Seiad Valley.

Despite these caveats, the approach taken by Asarian and Kann seems likely to provide reasonable estimates of seasonal nutrient retention over the long term (although not removal, as full-year mass balances are not available). Asarian and Kann, however, attempt to take the analysis further, evaluating retention by reach first on a yearly, and then on a monthly basis. As each month is represented by only one or two sampling events (and these events likely did not sample the same parcel of water), this is asking too much of the sparse data. Due to the uncertainty present in the data and the method, the best that can be hoped for is a statistical convergence to a reasonable estimate. Interpretations of the magnitude of retention for individual months are at best suspect.

Due to changes in sample locations, more than two years of data are available only for the reach Iron Gate to Seiad Valley. For the whole analysis period evaluated by Asarian and Kann (1998-2002), the June-

October net retention of TN is 18.6 percent, or 0.31 percent per mile. It should be remembered that this seasonal estimate is likely an upper bound on the annual retention, as it includes temporary storage that will be flushed downstream during winter high flows.

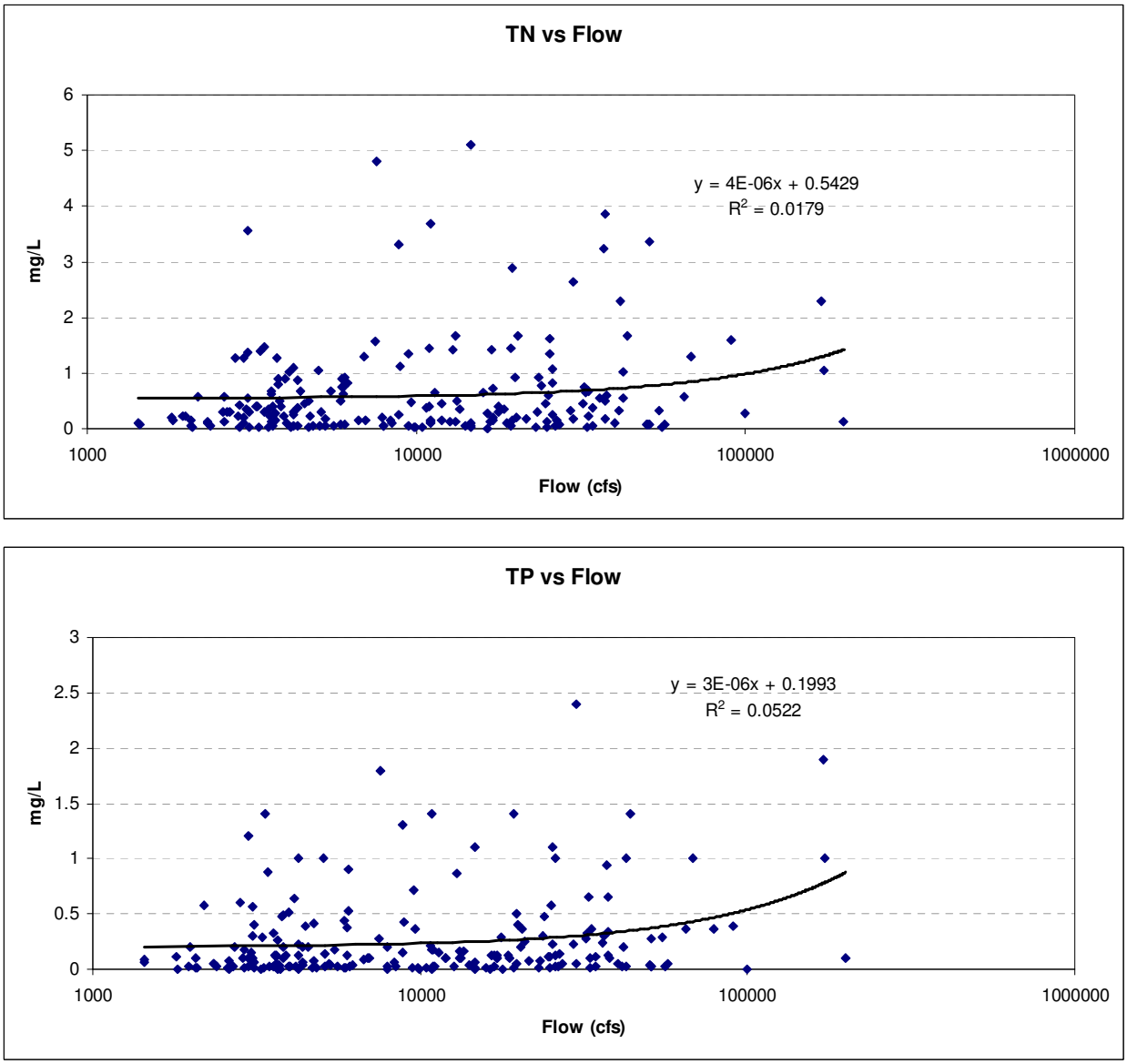

Figure 2. Relation of Nutrient Concentration to Flow, Klamath River at Klamath, CA

2.2 EXPECTED RANGE OF RIVER RETENTION ESTIMATES

USGS, as part of its Spatially Referenced Regressions on Watershed Attributes (SPARROW) project had developed generalized reach removal coefficients for TN and TP based on analysis of monitoring records of 381 USGS National Stream Quality Accounting Network (NASQAN) monitoring sites throughout the US (Smith et al., 1997). Removal is represented as an exponential decay process, such that the retention in a reach is given by

$$\text{Retention} = 1 - \exp(-\delta t),$$

where δ is a loss coefficient (day^{-1}) and t is travel time in days. The national coefficients originally developed by Smith et al. (1997) were revised by Smith et al. (2003a), as reported in Smith et al. (2003b). Median flows in the Klamath downstream of Iron Gate are greater than 1000 cfs, but less than 10,000 cfs, so the relevant national decay coefficients are 0.118 day^{-1} for TN and 0.098 day^{-1} for TP. The mean travel time from Iron Gate to Seiad is on the order of 2 days, so we would expect, as a long-term average, loss rates of about 21 percent for TN and 17 percent for TP. The growing season TN retention estimates for this reach given by Asarian and Kann range from 20 to 61 percent – but retention over the growing season is likely to be higher than annual net retention due to temporary storage in periphyton.

The SPARROW estimates of loss depend entirely on the decay coefficients, which are subject to considerable uncertainty and vary as a result of site-specific conditions. Other evidence is available in recently completed work of Armstrong and Ward (2008), who developed a simplified model of Klamath summer monitoring data for 2001-2005. Their application is a spreadsheet plug-flow model in which the decay rates for TN and TP are taken as calibration parameters. The methodology has some potential problems: First, it is sensitive to assumptions about tributary loads, which are poorly characterized for many of these years. Second, the approach fits individual decay rate estimates for each month in the dataset, which likely leads to over-fitting of the data. Finally, a single decay rate is applied for each month to the entire distance between Iron Gate and Turwar. However, the results are useful in providing another independent estimate of potential retention / loss rates in the Klamath River. Finally, the analyses cover only the June through October period, so one can examine only seasonal retention, not ultimate loss.

The decay rates fit by Armstrong and Ward range between 0.005 and 0.15 day^{-1} for both TN and TP, and are generally higher in the summer, suggesting that periphyton uptake may be a significant component of the retention. The authors do not provide an integrated summary of their decay rate estimates; however, examination of their Figure 33 suggests that the median decay rates were 0.005 day^{-1} for TN and 0.075 day^{-1} for TP. For two day travel between Iron Gate and Seiad, this would imply nitrogen retention of only 1 percent and phosphorus retention of 14 percent. (For individual months in which the estimated decay rate reached 0.15 day^{-1} the retention would be 26 percent.) These results suggest that the seasonal retention rate for TN in the Klamath could well be much less than predicted by SPARROW; however, they do not preclude the possibility that retention rates are much higher between Iron Gate and Seiad Valley than between Seiad Valley and Turwar, as suggested by Asarian and Kann (2006a). The results of Armstrong and Ward are also subject to considerable uncertainty. They report relative percent error on model-predicted concentration by month, ranging from 2 to 102 percent for total nitrogen (with median of 19 percent) and 3 to 22 percent for total phosphorus (with median of 11 percent).

2.3 RMA-11 IMPLEMENTATION

RMA-11 (King, 1998) simulates four state variables for nitrogen (nitrate, nitrite, ammonia, and organic N) and two state variables for phosphorus (dissolved P and organic P). Algal biomass also acts as a store of nutrients. Sorption of inorganic P to suspended sediment is represented as a sink, not a state variable, and thus forms a loss pathway from the system. Settling rates can be specified for organic N and organic P, again representing losses. While both deposition and scour of sediment are simulated by the model, the sediment mass balance is not directly linked to the nutrient mass balance (except through sorption of inorganic P). Algae take up inorganic nutrients during photosynthesis and convert them to organic nutrients, which are released during respiration (as inorganic nutrients) or decay (as organic nutrients). Algae can settle to the sediment, creating a sink for nutrients. (While the model documentation also discusses algal losses to grazing, this is apparently not implemented in the version of the model used for the Klamath.)

Thus, RMA-11 can potentially simulate internal nutrient losses (other than advection out of the system) in four ways: settling or grazing of algae, settling of organic N, settling of organic P, and sorption of inorganic P. RMA-11 can also simulate releases from the sediment of inorganic P and ammonium. Within the model, the phytoplankton settling rate was initially set to zero (PacifiCorp, 2005), but subsequently revised by Tetra Tech to match organic matter settling rates (0.05 m/day). PacifiCorp does not document the values for the other relevant rate constants; however, inspection of the model input shows that sorption losses of inorganic P are not simulated because TSS is not simulated. Net settling of organic N and organic P occurs in accordance with an organic matter settling velocity of 0.05 m/day (a very low value based on assumptions that net settling will be minimal in a fast-flowing river). No sediment releases of inorganic P or ammonium are specified.

Within the Klamath, periphytic algae can play an important role in nutrient cycling although quantitative data are limited and have been cited as a significant data gap (Flint et al., 2005). RMA-11 (King, 1998) has a rather simplistic representation of periphytic algae and algal-related nutrient cycling. This was somewhat remedied by project-motivated modifications to the RMA-11 (v41) code that separated algal respiration and death and added organic matter as a state component. Benthic algae are simulated as if they were planktonic algae, except not subject to advection or settling.

Despite the modifications to the algal code for the Klamath project, RMA-11 omits various processes that could potentially reduce its ability to accurately represent nutrient cycling in the free-flowing reaches of the Klamath. The importance of these processes is generally not known for the Klamath, so they can only be discussed in a speculative context. The majority of these processes are omitted from most other river water quality models as well. Among these are the following:

- RMA-11 does not simulate **denitrification**, the rationale being that this is not a significant pathway in free-flowing, relatively shallow rivers since denitrification bacteria require hypoxic conditions, as well as ample fixed carbon and nitrate supplies. In some rivers, however, there is evidence of significant denitrification occurring within the bed. The general thinking is that denitrification is of little significance in the well-oxygenated sediments of gravel-bed streams (Allan, 1995), and gravel-bed streams are more likely to convert ammonium to nitrate (e.g., research in the Williamette reported by Fernald et al., 2006). Denitrification in the Klamath mainstem is likely limited by the availability of organic matter in the bed. Anoxia under decaying periphyton mats could, however, form a locus for denitrification (Triska and Oremland, 1981). Omission of this pathway may cause the RMA-11 model to underestimate nitrogen loss rates in the river; however, the magnitude of this error is not known. Similarly, RMA-11 (like most other river water quality models) does not simulate chemical reactions that may precipitate inorganic phosphorus in inorganic forms.
- RMA-11 does not simulate **luxury uptake** of nutrients by periphytic algae. In many cases, algae may be able to uptake and store excess nutrients against future growth requirements (Droop, 1983). This phenomenon is more likely to be a significant factor when there is strong temporal variability in nutrient availability. The presence of upstream impoundments tends to smooth out temporal variability in the lower Klamath, and may thus reduce the impact of luxury uptake on retention.
- The Klamath version of RMA-11 simulates respiration as the inverse of photosynthesis. That is, while photosynthesis involves the uptake of inorganic nutrients and inorganic carbon to form biomass, respiration is assumed to release equal amounts of inorganic nutrients and carbon (PacifiCorp, 2005, Appendix A). In reality, the releases of nutrients (if they occur) are likely to be at least in part in organic form. Related to the previous bullet, there is no provision for intracellular storage of nutrients as fixed-carbon energy sources are oxidized. The result is that the model simulation of diurnal ammonium concentrations shows greater variability than

observed data during the growing season, with concentrations depressed during periods of photosynthesis, then enhanced during night-time respiration.

- The RMA-11 model, like most other river models, does not consider the interaction of stream nutrients with riparian perennial vegetation, which may provide for long term sequestration of nutrients. Riparian vegetation is most often thought to intercept nutrients derived from upland sources. However, in a gravel bed river nitrogen in the hyporheic zone derived from the river can be taken up directly by the roots of woody vegetation (Peterjohn and Correll, 1994; Naiman et al., 2000). Species such as alder may act as net nitrogen sources, however, due to their ability to fix atmospheric nitrogen. The net balance of these processes is unknown for the Klamath.

Together, these simplifying assumptions in RMA-11 would tend to result in an underestimate of retention and an over-estimation of the downstream transport of inorganic nutrients during the algal growing season. Denitrification seems likely to be the most significant omitted factor in the Klamath, but its impact is not known for this system.

As implemented for the Klamath, there are very few ultimate sinks for nutrients in RMA-11. Settling of organic nutrients and algae will constitute a loss, but this loss will be small, given the low settling velocity and short travel times. Nutrients taken up by periphyton will be temporarily retained, but will be re-released as periphyton dies off or is scoured. The simulation of periphyton in the model indeed begins with low periphyton biomass, increases to higher biomass in summer, then declines back toward zero (Figure 3).

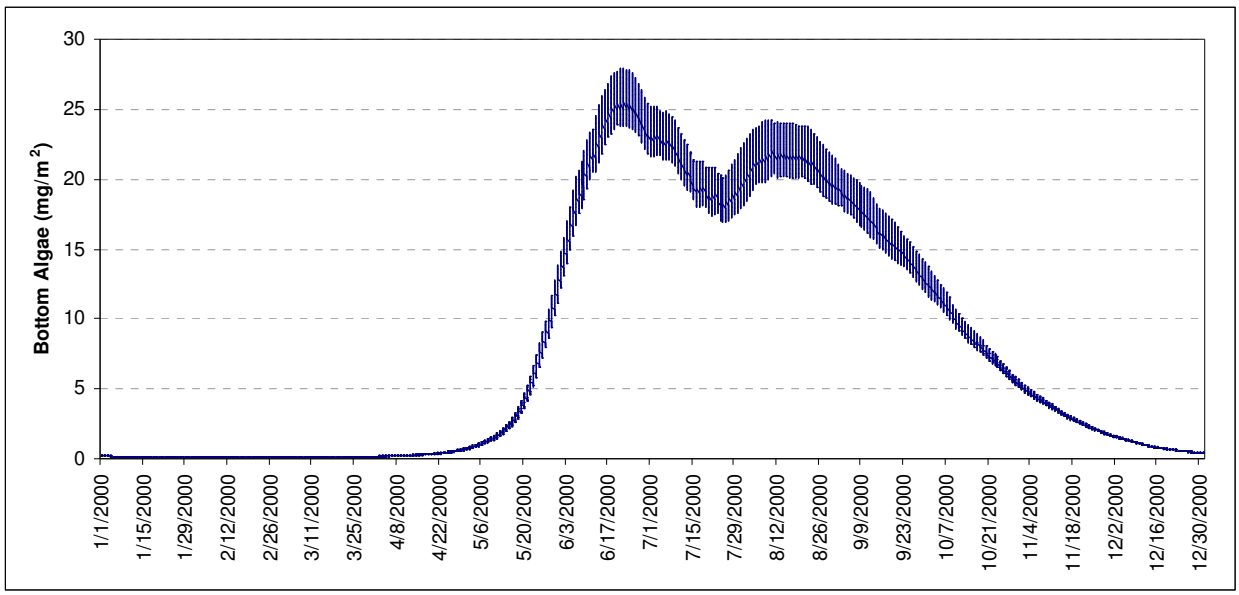

Figure 3. Periphyton Biomass Concentration Simulated for Klamath Upstream of Scott River

From Figure 3 we can conclude that periphyton, as simulated in the model, do not represent a significant net sink of nutrients, because the biomass declines back toward zero by the end of the year. It is also of interest to note that there is only one brief period (late May to early June) in which rapid accrual of periphyton biomass is simulated. Only during this period would we expect to see significant nutrient retention predicted for periphyton. During other periods, the periphyton are simulated as remaining at approximately steady biomass (in which case inorganic nutrient uptake should be balanced by output of inorganic and organic nutrients), or declining (in which case they will be creating a net increase in nutrient loads.)

Given the limitations of the RMA-11 code and the way it is parameterized for the Klamath, it appears clear that the model would not be expected to predict net removal of more than a few percent of total nutrient loads in free-flowing reaches of the river over the course of a year.

To test model behavior, a mass balance was constructed from the river and boundary flow and concentration data in the model. Loads were estimated on a monthly basis for TN, TP, NO_2+NO_3, PO_4, organic N, and organic P. The organic N and P components of non-living organic matter were calculated from organic matter concentrations according to the stoichiometry described in the model calibration report (organic N = 0.07 x organic matter; organic P = 0.0055 x organic matter). Within the Klamath, a noticeable fraction of the nutrient load may be transported in the form of living planktonic algae. Therefore, the planktonic algal biomass was also converted to organic N and organic P, using the same stoichiometry.

Model mass balance results were calculated for each river reach from Iron Gate to Seiad, but the net results over this whole distance are most informative (Table 2 and Table 3). For 2000, the model predicts TN loss/retention of 0.37 percent and TP loss/retention of 0.52 percent. For the period of June – September analyzed by Asarian and Kann (2006a), the TN retention predicted by the model is 1.05 percent. These results are less than estimated from the SPARROW methods or by the analyses of Asarian and Kann. They are, however, generally consistent with the analyses of the 2001-2005 data by Armstrong and Ward (2008), which suggest a TN retention rate of approximately 1 percent between Iron Gate and Seiad. As noted above, all these estimates of retention are subject to considerable uncertainty. This is an unavoidable result of the sparse data available. It is possible that RMA-11 would tend to underestimate seasonal nutrient retention rates due to the omission of various processes that can enhance nutrient retention and loss. However, the data are not sufficient to determine whether such an underestimation exists or is statistically significant. A comparison of nutrient losses on a full-year basis cannot be made, because both Asarian and Kann (2006a) and Ward and Armstrong (2008) worked with only seasonal nutrient data.

Table 2. Total P Mass Balance (kg) for RMA-11 Application for Year 2000, Klamath River from Iron Gate Dam to Seiad, CA

Month	IN	OUT	Change	Retention
1	37,751	36,027	-1,724	4.57%
2	68,409	68,106	-303	0.44%
3	72,501	72,325	-176	0.24%
4	55,820	55,551	-270	0.48%
5	56,498	56,220	-277	0.49%
6	38,889	39,155	266	-0.68%
7	23,875	23,416	-460	1.93%
8	27,364	26,949	-416	1.52%
9	25,776	26,123	348	-1.35%
10	23,951	24,095	143	-0.60%
11	19,410	19,746	337	-1.73%
12	15,159	15,288	129	-0.85%
Whole Year	465,403	462,999	-2,404	0.52%

Table 3. Total N Mass Balance (kg) for RMA-11 Application for Year 2000, Klamath River from Iron Gate Dam to Seiad, CA

Month	IN	OUT	Change	Retention
1	260,614	260,409	-205	0.08%
2	264,131	261,344	-2,787	1.06%
3	281,346	278,604	-2,742	0.97%
4	244,653	243,079	-1,573	0.64%
5	154,400	156,288	1,888	-1.22%
6	105,152	105,184	32	-0.03%
7	62,008	59,356	-2,652	4.28%
8	73,882	71,031	-2,852	3.86%
9	78,354	80,464	2,110	-2.69%
10	86,365	87,557	1,192	-1.38%
11	83,931	84,289	357	-0.43%
12	88,934	89,489	555	-0.62%
Whole Year	1,783,771	1,777,092	-6,679	0.37%

The model does predict a seasonal pattern of temporary retention in the spring and summer, followed by releases in the fall as periphyton densities decline. Plots of the cumulative changes in nitrogen and phosphorus component loads over the simulation year 2000 (Figure 4 and Figure 5) suggest that the spring to early summer period is dominated by the decay of organic matter to inorganic nutrients, followed by uptake of ammonium and inorganic P as periphyton growth accelerates in May and release of organic nutrients as the periphyton reaches senescence.

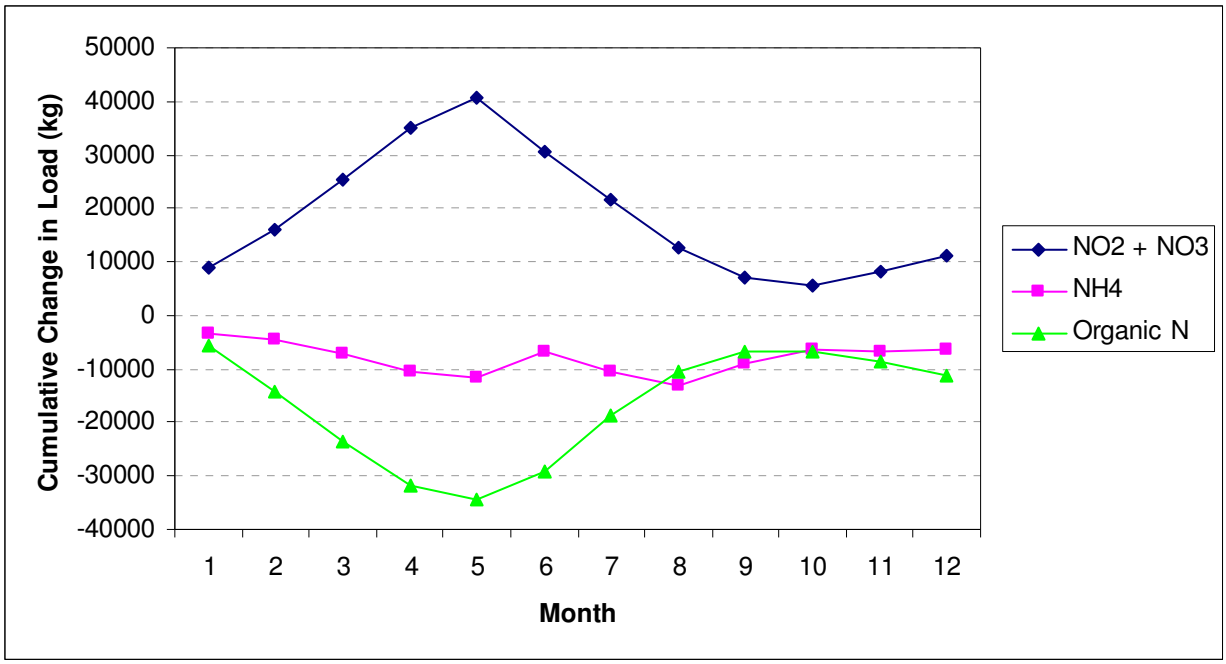

Figure 4. Cumulative Change in Nitrogen Load Predicted by Klamath River Model, Iron Gate to Seiad, for Year 2000

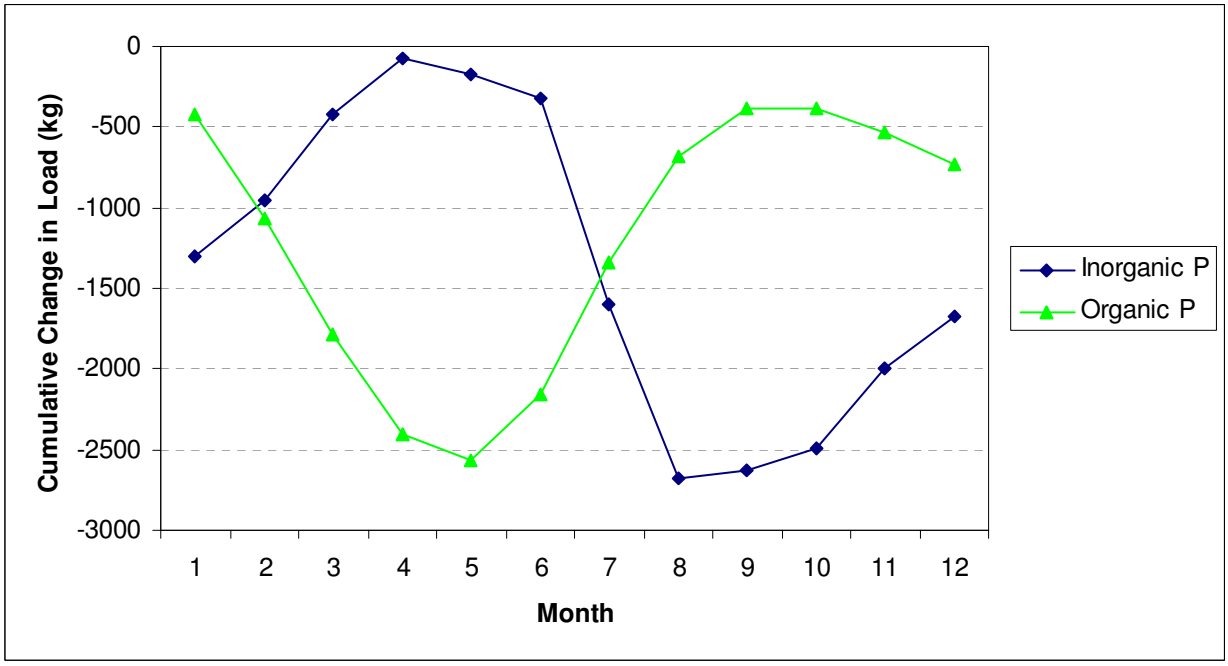

Figure 5. Cumulative Change in Phosphorus Load Predicted by Klamath River Model, Iron Gate to Seiad, for Year 2000

While the model predicts little change in total loads over the course of a year during transit from Iron Gate to Seiad Valley, the annual flow-weighted concentration does decline, by about 17 percent for TN and 19.5 percent for TP (Figure 6). This decline is almost entirely due to accumulation of flow from cleaner tributaries, which dilutes the load originating above Iron Gate dam – and points out the pitfalls inherent in trying to assess retention based only on concentration.

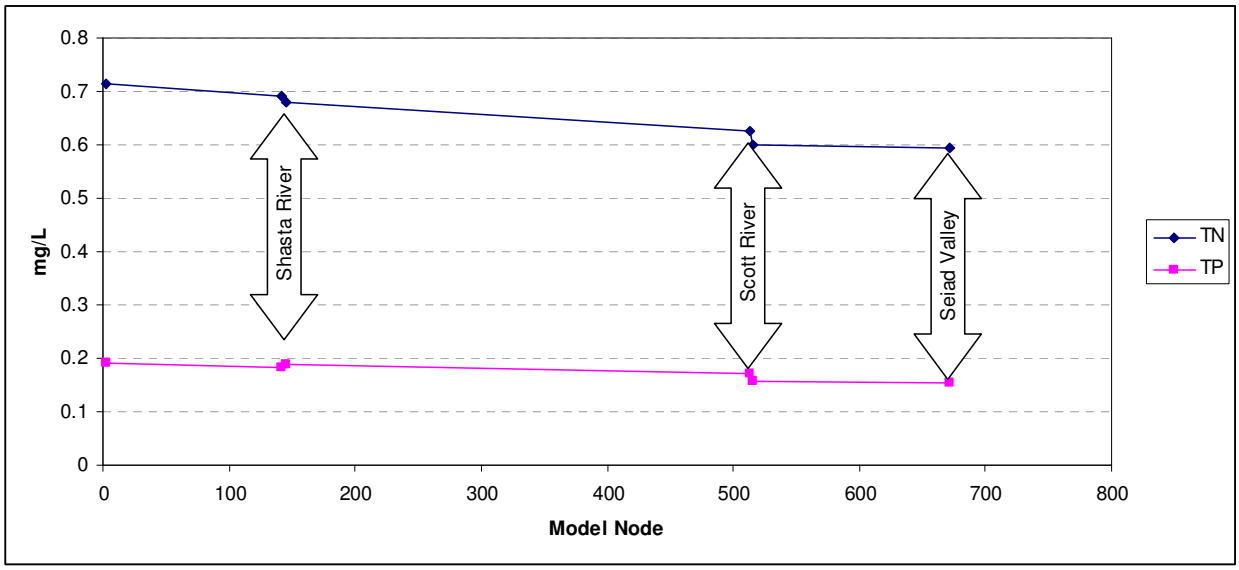

Figure 6. Annual Flow-Weighted Concentrations for 2000 Model Run

In sum, the RMA-11 model application predicts little nutrient loss in the free-flowing reaches of the Klamath, although there is seasonal retention. Even adjusted for partial year estimates, this is at odds with the estimates of Asarian and Kann (2006a), as well as estimates from the USGS SPARROW model. The result is not unexpected, given that RMA-11 provides few permanent sinks for nutrients. The importance of potential sinks (such as denitrification) is unclear, and the estimates from Asarian and Kann (based on limited data) and SPARROW (an approximate, national-scale method) are subject to considerable uncertainty. However, it does appear that the RMA-11 model may have some tendency to underestimate nutrient losses in the free-flowing reaches of the Klamath. The data are not sufficient to resolve whether these factors are significant relative to the overall mass balance, and the magnitude of any such effect cannot be fully resolved without more intensive sampling, coupled with model application to intensively sampled years.

3 Reservoir Retention of Nutrients

Reservoirs can be effective traps of influent nutrients. By design, reservoirs represent areas of a river system in which velocity is decreased and residence time increased. This encourages the settling of particulate material, including both nutrient-bearing organic detritus and nutrients sorbed to inorganic sediment. Reservoirs also encourage the growth of planktonic algae, and settling of dead algal detritus can increase loss rates.

In general, these factors would lead to increased nutrient loss in reservoirs as opposed to free-flowing reaches. The difference, however, depends on reservoir residence time (which is short in the Klamath reservoirs). In addition, there are several compensating factors. First, algal growth in deeper reservoirs is primarily planktonic, and plankton are readily advected downstream, unlike periphytic algae, so reservoirs with short retention times may provide less retention of nutrients than free-flowing reaches.

Second, under anoxic conditions there is typically significant evolution of phosphorus and ammonium from lake sediments (internal loading).

3.1 MASS BALANCE ANALYSES FOR RESERVOIR RETENTION IN THE KLAMATH

The first attempt at nutrient budgets for Iron Gate Reservoir was made by USEPA (1978), based on 1975 monthly sampling. They concluded that the mass of nitrogen leaving Iron Gate was 21 percent higher than inflow, while the reservoir retained 7 percent of the phosphorus mass. As noted by Kann and Asarian (2005), these estimates are based on limited data and are not corrected for changes in reservoir storage.

Kann and Asarian (2005, 2007) have produced two reports on the nutrient budgets of Iron Gate and Copco reservoirs. The first evaluated monthly sampling collected for relicensing purposes in 2002. Unfortunately, data were collected only for March through November, and do not include the full turnover period, so a full year mass balance cannot be created. As with their work on mass balances in the river, Kann and Asarian interpolated concentrations between sampling dates and attempted to evaluate the mass balance on a monthly basis. As with the river, the limited sampling basis and the associated uncertainty in mass calculations renders monthly calculations suspect. Over the period from April 1 to November 13, they estimated net retention in Copco of 36.29 metric tons of TP (26.3 percent of influent loads), and 48.20 metric tons of TN (8.1 percent of influent loads). Over the same period, Iron Gate was estimated to retain 32.4 metric tons of TP (27.3 percent of influent loads) and 65.8 metric tons of TN (12.4 percent of influent loads). Because the calculations do not include the full fall turnover or winter flushing flows, these can be taken as upper bounds on the annual retention. Both reservoirs showed a noticeable shift from inorganic N in the influent to organic N in the effluent, apparently due to algal uptake.

Kann and Asarian (2007) analyzed a complete year of biweekly data collected from May 2005 to May 2006 at several locations in Iron Gate and Copco and their tributaries. Over the entire year, Copco was estimated to retain 9.4 percent of influent phosphorus and 9.1 percent of influent nitrogen, while Iron Gate was estimated to retain 3.1 percent of influent phosphorus and 10.0 percent of influent nitrogen. The annual balances are, however, subject to considerable uncertainty due to large uncertainties in flow measurements (as well as the uncertainties in water quality sampling) over the winter high flow period. For the growing season, defined as May 18 2005 to October 5 2005, Copco was estimated to retain 3.3 percent of influent phosphorus and 18.2 percent of influent nitrogen, while Iron Gate was estimated to retain 0.5 percent of influent phosphorus and 15.3 percent of influent nitrogen. The phosphorus retention rates estimated for the 2005 growing season are dramatically lower than those obtained for 2002 data. Kann and Asarian attribute this difference to higher levels of dissolved P in inflows, coupled with internal P loading. Nitrogen retention estimates for the growing season are twice those obtained earlier for Copco and similar to the 2002 results for Iron Gate.

The differences in the retention rates estimated in the two studies may reflect actual differences in reservoir behavior from year to year. However, it is likely that much of the difference in estimates reflects uncertainties in estimation from limited data. For example, in the 2005-2006 data for Copco inflows, the coefficient of variation (standard deviation divided by the mean) is 0.29 for TP and 0.25 for TN, while the standard error on the mean is 6.2 percent of the influent mean value for TP and 5.3 percent of the influent mean value for TN. Over the summer growing period, the standard errors on the mean for both TP and TN are approximately 9 percent of the influent mean value. The retention estimates are thus generally within the range of two standard errors on the influent mean. This uncertainty or variation in the data supports the need for a modeling approach to interpolate through the limited observations.

As in the earlier report, Kann and Asarian (2007) attempted analyses of mass balances on a monthly basis, and detected periods of negative retention. As each month has only two samples, these estimates are highly uncertain, and results for any given month may be only an artifact of the data. As noted by PacifiCorp (2006), there is a lag time between nutrients entering Copco and being discharged from Iron Gate. As a result of this lag, "it is expected that at times the nutrient concentration in release waters from Iron Gate Reservoir…may be greater than in the inflowing waters to Copco reservoir on the same day, even though the reservoirs act to retain and reduce the loads from these nutrient 'events' as they move through the reservoirs."

3.2 EXPECTED RANGE OF RETENTION

As with the SPARROW estimates for stream reaches, there are simplified empirical methods for estimating nutrient retention in reservoirs that can be used to evaluate whether estimates based on limited observations are reasonable.

Under steady-state conditions,

$$\text{Retention} = 1 - \frac{C}{C_i},$$

where C is the mixed concentration at the dam, and C_i is the influent concentration. In a simple, first-order representation of sedimentation loss, this yields

$$\text{Retention} = 1 - \frac{1}{1 + BT},$$

where B is a first-order sedimentation loss coefficient and T is residence time. For TN, Bachman (1980) gives an estimate of B in terms of flushing rate, as

$$B_{TN} = 0.693\, T^{-0.55}.$$

Similarly, for TP, Vollenweider (1976) estimated

$$B_{TP} = T^{-0.5}.$$

An alternate analysis of TP retention based on an analysis of oxic lakes, exclusive of internal sources, is given by Nürnberg (1984) as

$$\text{Retention} = \frac{15}{18 + z/T},$$

where z is the average depth.

For the two downstream reservoirs in the Klamath system, Copco and Iron Gate, the relevant parameters are given in Table 4. Determination of a residence time is problematic for run-of-river reservoirs that are dominated by winter flow-through. Not only does residence time vary throughout the year, but in addition the reservoirs are not well-mixed in summer, and retention time in the hypolimnion may be much longer than in the epilimnion. For the period of May 2005 through May 2006 reported by Kann and Asarian (2007), the overall residence time in both reservoirs was on the order of 6 days, but the summer residence time of surface waters was around 20-25 days for Copco and 25-35 days for Iron Gate (but can reach as high as 50 days in Iron Gate). For this simple comparison, compromise values of 14 and 16 days were used, combined with summer mean depth.

Table 4. Hydraulic Parameters for Klamath Reservoirs (May 2004 – May 2005)

Impoundment	Residence Time (*T*, yrs)	Mean Depth (*z*, m)
Copco	0.0384	11.7
Iron Gate	0.0484	16.6

Note: Approximate values for summer growing period based on analysis of 2002 and 2004-2005 data in Kann and Asarian (2005, 2007)

Reservoir nutrient retention estimates obtained from the several empirical methods are shown in Table 5, along with the full-year estimates provided by Kann and Asarian (2007). The latter also provide a range of literature-based estimates (Kann and Asarian, 2007, Section 3.4.5.2). For short residence time lakes, the Vollenweider method gives significantly higher retention than that of Nürnberg, with the latter likely being more appropriate. The estimates of Kann and Asarian (2007) on an annual basis are in general agreement with the empirical estimates. Estimated retention is perhaps a little higher than predicted in Copco and lower in Iron Gate – which may reflect the fact that these two reservoirs are in series, with more easily removable material being retained upstream in Copco.

Table 5. Estimated Nutrient Retention for Copco and Iron Gate Reservoirs, 2004-2005

Parameter	Method	Copco	Iron Gate
TP	Vollenweider (1976)	16.4%	17.3%
TP	Nürnberg (1984)	4.6%	3.8%
TP	Range of 5 methods cited by Kann and Asarian (2007)	1.4% - 29%	-1.9% - 29%
TP – 2004-2005 data	Kann and Asarian (2007)	9.4%	3.1%
TN	Bachman (1980)	13.8%	14.5%
TN	Range of 2 methods cited by Kann and Asarian (2007)	8.7% - 10.3%	9.4% - 10.0%
TN – 2004-2005 data	Kann and Asarian (2007)	9.1%	10.0%

3.3 CE-QUAL-W2 IMPLEMENTATION

The CE-QUAL-W2 model (Cole and Wells, 2005) is a two-dimensional, longitudinal/vertical (laterally averaged) hydrodynamic and water quality model that is frequently applied to reservoirs. The model simulates inorganic nutrients (orthophosphate, ammonium, nitrite/nitrate) along with organic matter (labile and refractory, dissolved and particulate) and algae. Decay of organic matter and respiration/death of algae releases inorganic nutrients, while algal growth converts inorganic nutrients to organic matter. In addition to inflow and outflow, the model represents the following internal sources and sinks of nutrients:

Sources	Sinks
Release of PO_4 from sediment under anaerobic conditions	Settling of PO_4
Release of NH_4 from sediment under anaerobic conditions	Settling of organic matter
	Settling of algae
	Denitrification (loss to atmosphere)

Unlike RMA-11 applications to rivers, the nutrient sinks in CE-QUAL-W2 can be significant. The current version of the model also has the ability to simulate macrophytes with direct uptake of nutrients from the sediment; however, this pathway is not considered important and is not implemented in the Klamath models.

Several other potential source/sink pathways are not included in the model, including:

- Release of inorganic nutrients from the sediment under aerobic conditions (usually not a significant process except in shallow lakes where wind-induced scour can redistribute sediment-sorbed nutrients into the water column).
- Release of organic matter from the sediment.
- Settling of NH_4 (like phosphorus, ammonium can sorb to particulate matter and settle out of the water column).
- Nitrogen fixation (some cyanophytes can fix gaseous nitrogen, resulting in a net input of nitrogen to the system).

It is expected that none of these pathways will be significant in the nutrient mass balance for the Klamath reservoirs. Nitrogen fixation is important in some systems where nitrogen supply is limited. The Klamath reservoirs, however, usually have adequate inorganic nitrogen supplies to support algal growth. Under these circumstances, N-fixing algae tend to uptake dissolved nitrogen directly from the water column as opposed to the air, as nitrogen fixation is a highly energy-demanding process (Welch and Jacoby, 2004).

As with the river simulation, the Klamath River model run output for year 2000 was used to examine the mass balance of nutrients in Iron Gate and Copco Reservoirs. The modelers provided the initial and ending storage volumes, which were combined with concentrations to estimate the change in nutrient storage over the course of the simulation. TN and TP mass balances for the two reservoirs for 2000 are summarized in Table 6 through Table 9.

For TP, the annual retention rate estimated for the model is 6.11 percent for Iron Gate and 1.22 percent for Copco. These are in the range predicted by the Nürnberg (1984) model, although the retention rate for Copco appears a bit low, and also in the range of literature estimates reported by Kann and Asarian. For TN, the annual retention rate estimated by the model is 17.63 percent for Iron Gate and 3.61 percent for Copco. The estimate for Iron Gate is similar to that from the Bachman (1980) estimator and higher than that estimated by Kann and Asarian (2007) for 2004-2005. The TN retention rate for Copco appears low relative to both the Bachman estimate and the analysis of Kann and Asarian based on 2004-2005 data. This could simply reflect differences between years. For instance, dam operations vary significantly over time and can have a major impact on nutrient retention. Another potential explanation for lower retention rates in Copco is the low concentrations of particulate organic matter in inflow to this reservoir, while the buildup of algal biomass in both Copco and Iron Gate may contribute to higher retention rates in the downstream impoundment.

Table 6. Total P Mass Balance (kg) for CE-QUAL-W2 Application for Year 2000, Iron Gate Reservoir

Month	IN	OUT	Change	Retention
1	33,813	22,900	-10,913	32.27%
2	54,050	52,012	-2,039	3.77%
3	57,213	52,394	-4,818	8.42%
4	40,121	36,947	-3,174	7.91%
5	40,596	36,641	-3,955	9.74%
6	27,439	25,963	-1,475	5.38%
7	23,515	19,633	-3,883	16.51%
8	28,110	26,485	-1,625	5.78%
9	20,355	24,417	4,062	-19.96%
10	20,773	21,126	352	-1.70%
11	11,517	16,129	4,612	-40.04%
12	11,343	11,670	327	-2.89%
Whole Year	368,845	346,318	-22,527	6.11%
Whole Year Retention (corrected for change in storage)			-22,521	6.11%

Table 7. Total N Mass Balance (kg) for CE-QUAL-W2 Application for Year 2000, Iron Gate Reservoir

Month	IN	OUT	Change	Retention
1	185,087	191,369	6,282	-3.39%
2	233,737	188,997	-44,740	19.14%
3	264,713	196,327	-68,387	25.83%
4	183,398	155,264	-28,135	15.34%
5	103,453	96,293	-7,160	6.92%
6	79,847	55,506	-24,342	30.49%
7	87,133	49,336	-37,797	43.38%
8	109,349	70,070	-39,279	35.92%
9	88,370	72,398	-15,972	18.07%
10	99,909	73,439	-26,470	26.49%
11	75,331	69,115	-6,217	8.25%
12	80,020	72,802	-7,217	9.02%
Whole Year	1,590,349	1,290,916	-299,433	18.83%
Whole Year Retention (corrected for change in storage)			-280,376	17.63%

Table 8. Total P Mass Balance (kg) for CE-QUAL-W2 Application for Year 2000, Copco Reservoir

Month	IN	OUT	Change	Retention
1	35,621	29,907	-5,714	16.04%
2	48,378	49,192	814	-1.68%
3	53,116	52,587	-529	1.00%
4	36,611	35,340	-1,270	3.47%
5	43,803	39,143	-4,659	10.64%
6	25,702	26,740	1,038	-4.04%
7	30,243	22,211	-8,033	26.56%
8	21,346	26,837	5,491	-25.72%
9	15,653	19,395	3,742	-23.90%
10	17,280	19,858	2,578	-14.92%
11	9,569	10,912	1,343	-14.04%
12	8,905	9,720	816	-9.16%
Whole Year	346,226	341,843	-4,383	1.27%
Whole Year Retention (corrected for change in storage)			-4,240	1.22%

Table 9. Total N Mass Balance (kg) for CE-QUAL-W2 Application for Year 2000, Copco Reservoir

Month	IN	OUT	Change	Retention
1	143,374	157,413	14,040	-9.79%
2	196,915	202,659	5,744	-2.92%
3	242,555	232,490	-10,065	4.15%
4	152,922	152,036	-886	0.58%
5	94,216	91,737	-2,480	2.63%
6	95,602	74,430	-21,172	22.15%
7	128,797	78,769	-50,028	38.84%
8	108,494	101,022	-7,473	6.89%
9	82,163	82,274	111	-0.14%
10	96,552	94,102	-2,450	2.54%
11	75,761	71,285	-4,476	5.91%
12	60,175	69,181	9,007	-14.97%
Whole Year	1,477,525	1,407,398	-70,127	4.75%
Whole Year Retention (corrected for change in storage)			-53,278	3.61%

Model-predicted reservoir cumulative retentions for nitrogen and phosphorus species are summarized for the two reservoirs in Figure 7 through Figure 10 (uncorrected for changes in storage). For Iron Gate, there is a steady loss of organic nutrients, accompanied by seasonal patterns in inorganic nutrient uptake and release that result in little net change over the course of the year. For Copco, the predicted cumulative loss of organic nutrients is smaller, with increases in inorganic nutrient mass in the fall.

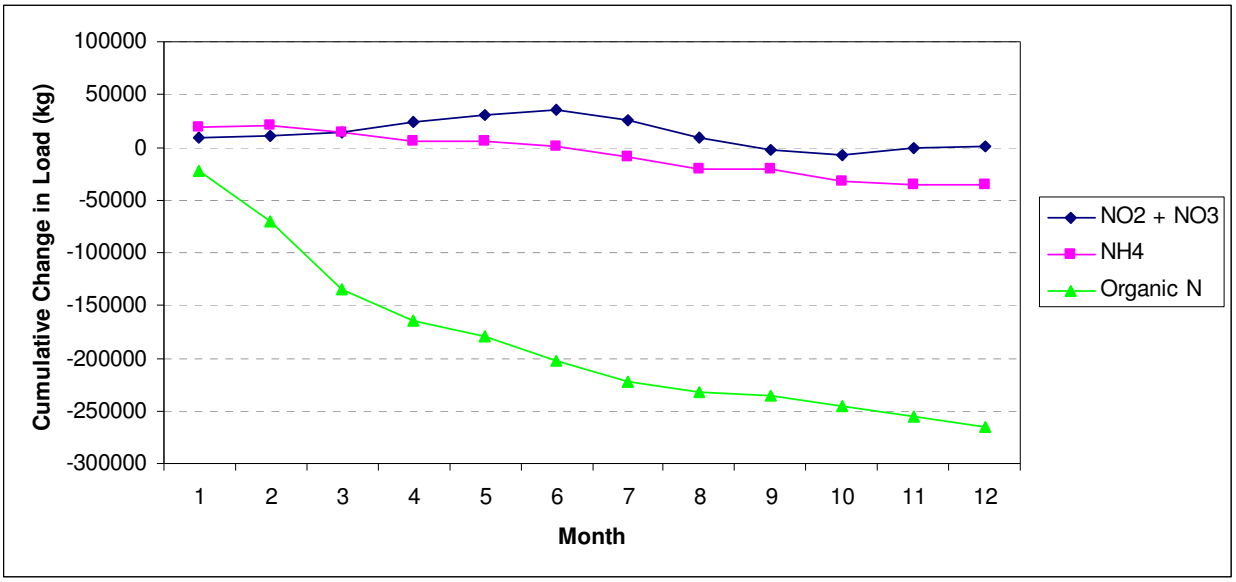

Figure 7. Cumulative Change in Nitrogen Load Predicted by Klamath River Model, Iron Gate Reservoir, for Year 2000

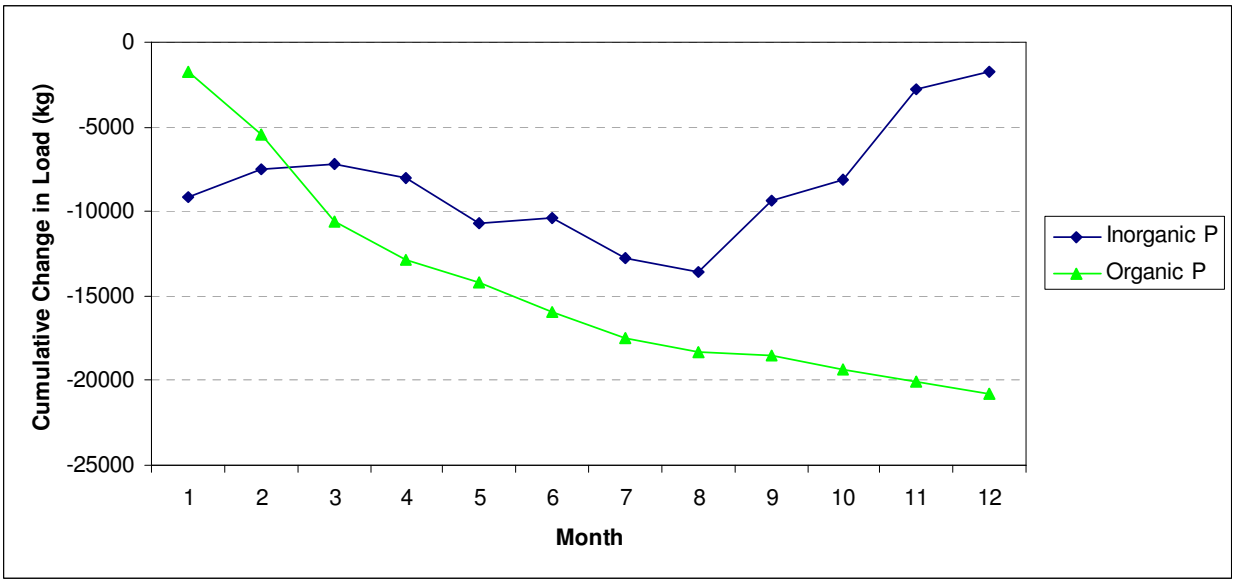

Figure 8. Cumulative Change in Phosphorus Load Predicted by Klamath River Model, Iron Gate Reservoir, for Year 2000

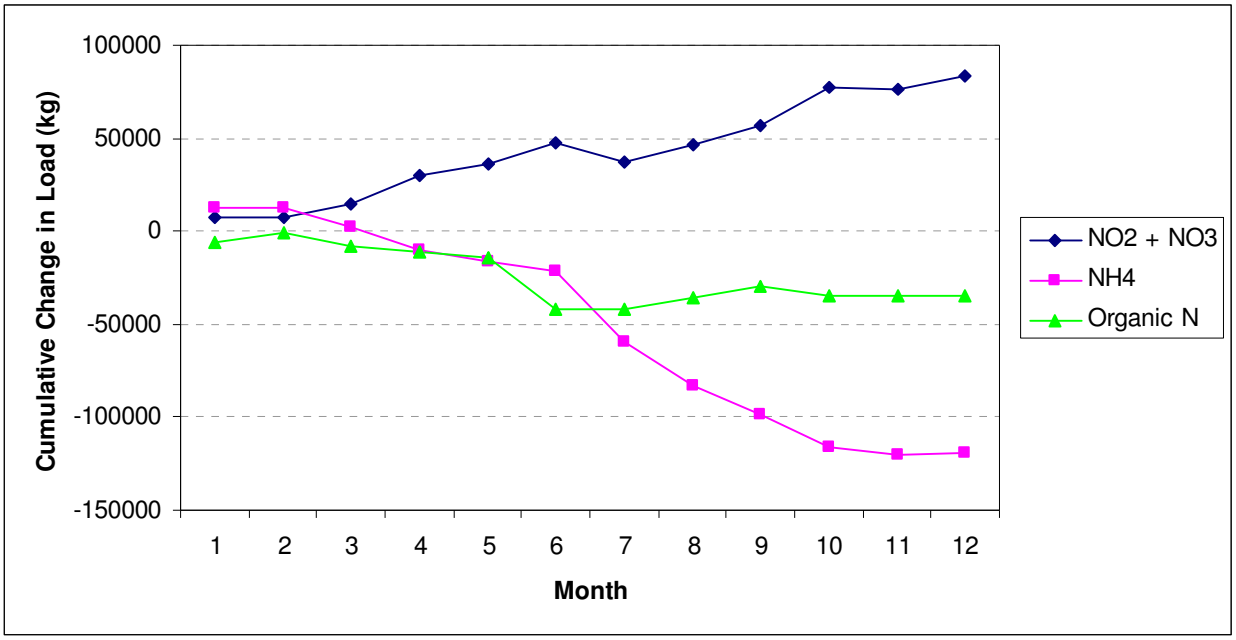

Figure 9. Cumulative Change in Nitrogen Load Predicted by Klamath River Model, Copco Reservoir, for Year 2000

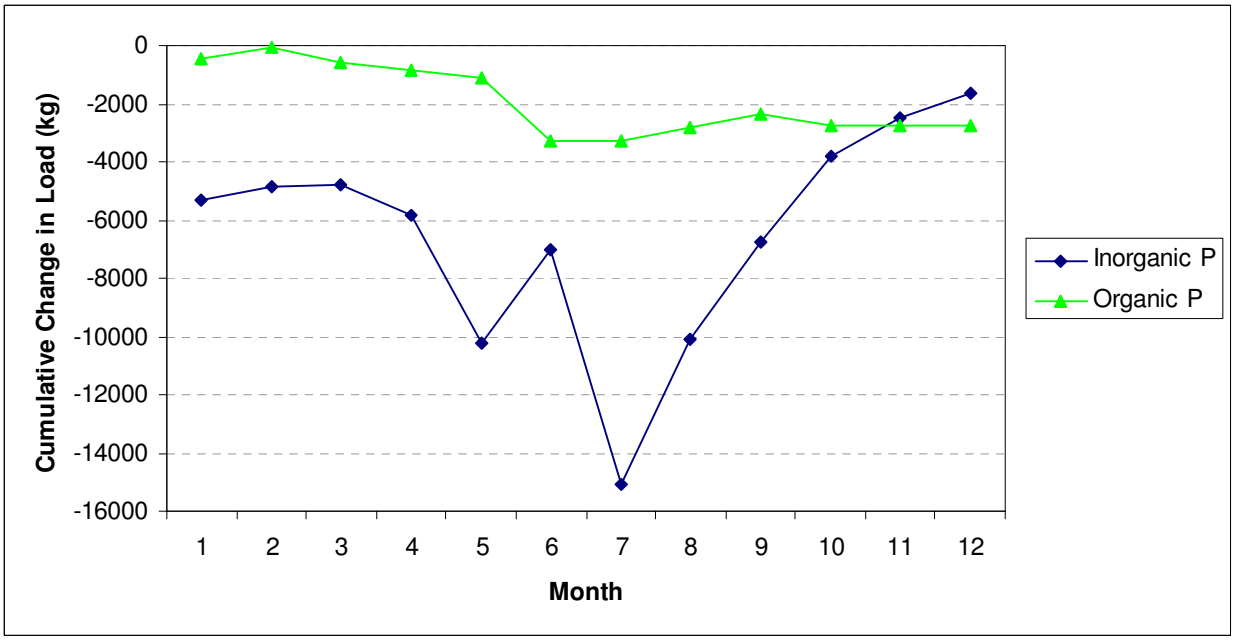

Figure 10. Cumulative Change in Phosphorus Load Predicted by Klamath River Model, Copco Reservoir, for Year 2000

4 Conclusions

The available monitoring data in the Klamath River system is not sufficient to provide a tight closure for nutrient mass balances, and estimates of nutrient retention and loss rates are thus uncertain. It is important to keep in mind the distinction between retention – which delays the transport of nutrients, but does not ultimately remove them – and loss – which results in the long-term removal of nutrients through transfer to stable sediments or the atmosphere.

Both flowing and impounded reaches of the Klamath provide opportunities for retention and loss of nutrients. The dominant process for retention in the system appears to be uptake by algae, which both delays transport downstream and converts inorganic to organic nutrient forms. Given the short residence time in the Klamath reservoirs, retention is likely more significant in flowing reaches, dominated by attached algae, than in the reservoirs, where planktonic algae may be washed downstream. Temporary retention benefits downstream reaches by reducing nutrient loads during the growing season; however, nutrient mass retained in algae is ultimately transported downstream to the estuary, largely after the end of the growing season.

Important loss pathways include denitrification with loss to the atmosphere of nitrogen and deposition of relatively insoluble forms of phosphorus to the sediment. These processes occur in both flowing and impounded reaches. Denitrification permanently removes nitrogen from the aquatic system, and is likely to be significant in the impoundments when the hypolimnion is anoxic. The argument is less clear for the flowing reaches, where oxic conditions are maintained, although some losses likely do occur in conjunction with decaying periphyton mats. For phosphorus, complexes that are insoluble under oxic conditions can often be remobilized under anoxic reducing conditions. This likely limits the annual removal of phosphorus in reservoirs where rapid deep burial is not occurring. (For the Klamath, the presence of reservoirs in series likely limits deep burial rates in the more downstream reservoirs.) For flowing reaches that maintain oxygenation, precipitated phosphorus may remain insoluble – but is prone to transport downstream sorbed to sediment.

Both the CE-QUAL-W2 model and the data analyses of Kann and Asarian (2005, 2007) predict limited amounts of TN and TP removal in Copco and Iron Gate reservoirs. Given the limitations of the available data and associated uncertainty, the model and data-based analyses appear to be in reasonable agreement with one another and with retention estimates based on empirical methods in the literature. The uncertainty in the data supports the need for a modeling approach to interpolate through the limited observations.

For the free-flowing reaches of the Klamath below Iron Gate dam, the RMA-11 model predicts some seasonal retention, but little ultimate loss of nutrients (less than 1% of annual TN and TP loads). In contrast, Asarian and Kann (2006a) contend that there is significant retention of TN between Iron Gate and Seiad. Their estimates are based on seasonal data only, so the annual rate of loss is unknown; however, the estimated seasonal retention rates are much greater than predicted by RMA-11, while appearing to be in approximate agreement with the USGS SPARROW model (Smith et al., 1997). The independent analyses of Armstrong and Ward (2008) do suggest that nutrient loss rates in the Klamath may indeed be quite low, which is consistent with the representation in the calibrated RMA-11 model. Overall, the available data are insufficient to precisely determine the true rates of annual nutrient loss in the free-flowing reaches of the Klamath. Although the RMA-11 model omits several processes such as denitrification that potentially affect nutrient loss rates, most of these processes are also omitted from most other river quality models and it is unclear if they are significant in the Klamath River.

In sum, the linked CE-QUAL-W2 and RMA-11 models of the Klamath appear to provide reasonable estimates of nutrient dynamics in the impoundments, while it is inconclusive whether or not nutrient retention and loss rates in the free-flowing reaches of the Klamath are significantly underestimated.

Despite this unresolved issue, the RMA-11 model appears to be a reasonable tool for assembling the TMDL, as long as the influence of model uncertainty on management decisions is acknowledged.

For the purposes of TMDL-based load allocations, the potential for underestimation of nutrient retention could be treated as a margin of safety (MOS). That is, the actual deleterious impacts of nutrient loads are likely to extend a lesser distance, and create less total algal biomass, than is predicted by the model. Therefore, a TMDL based on the model would include an implicit MOS insofar as the efficacy of nutrient reductions in controlling periphytic algal biomass in the river and its effect on the diurnal DO cycle is likely to be underestimated to some degree. Scenario analyses that depend on the relative retention rates of nutrients in flowing and impounded reaches (such as dam removal scenarios) must be approached with particular care, given the current level of uncertainty in the simulation of retention and loss rates. Interpretation of such scenarios will need to include an evaluation of the decision implications of model and data uncertainty. Additional, focused studies on potential nutrient loss mechanisms are recommended to further reduce uncertainty.

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PEER REVIEW DRAFT

Appendix 3

Effects of Temperature, Dissolved Oxygen/Total Dissolved Gas, Ammonia, and pH on Salmonids

Implications for California's North Coast TMDLs

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Environmental Scientist
North Coast Regional Water Quality Control Board

July 2008

Draft-Do Not Cite-Subject to Revision

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PEER REVIEW DRAFT

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CHAPTER 1. TEMPERATURE

1.1 Introduction

Temperature is one of the most important environmental influences on salmonid biology. Most aquatic organisms, including salmon and steelhead, are poikilotherms, meaning their temperature and metabolism is determined by the ambient temperature of water. Temperature therefore influences growth and feeding rates, metabolism, development of embryos and alevins, timing of life history events such as upstream migration, spawning, freshwater rearing, and seaward migration, and the availability of food. Temperature changes can also cause stress and lethality (Ligon et al. 1999). Temperatures at sub-lethal levels can effectively block migration, lead to reduced growth, stress fish, affect reproduction, inhibit smoltification, create disease problems, and alter competitive dominance (Elliott 1981, USEPA 1999a). Further, the stressful impacts of water temperatures on salmonids are cumulative and positively correlated to the duration and severity of exposure. The longer the salmonid is exposed to thermal stress, the less chance it has for long-term survival (Ligon et al. 1999).

A literature review was performed to evaluate temperature needs for the various life stages of steelhead trout (*Oncorhynchus mykiss*), coho salmon (*Oncorhynchus kisutch*), and Chinook salmon (*Oncorhynchus tschawytscha)*. The purpose of this review was to identify temperature thresholds that are protective of salmonids by life stage, as a basis for evaluating stream temperatures in California temperature TMDLs within the North Coast region.

This review included USEPA temperature guidance, Oregons' and Washingtons' temperature standards reviews, reports that compiled and summarized existing scientific information, and laboratory and field studies. When possible, species-specific needs were summarized by the following life stages: migrating adults, spawning and incubation/emergence, and freshwater rearing and growth. Additionally, the effects of temperature on disease and lethality are also discussed. Some of the references reviewed covered salmonids as a general class of fish, while others were species specific. Information for fall run coho salmon, spring/summer, fall, and winter steelhead, and spring and fall run Chinook salmon are compiled by life stage in Table 1 through Table 12.

1.2 Temperature Metrics

In considering the effect of temperature on salmonids, it is useful to have a measure of chronic and acute (i.e. sub-lethal and lethal) temperature exposures. A common measure of chronic exposure is the maximum weekly average temperature (MWAT). The MWAT is the maximum seasonal or yearly value of the mathematical mean of multiple, equally spaced, daily temperatures over a running seven-day consecutive period (Brungs and Jones 1977, p.10). In other words, it is the highest single value of the seven-day moving average temperature. A common measure of acute effects is the instantaneous maximum. A third metric, the maximum weekly maximum temperature (MWMT), can be used as a

measure of both chronic and acute effects. The MWMT is also known as the seven-day average of the daily maximum temperatures (7-DADM), and is the maximum seasonal or yearly-value of the daily maximum temperatures over a running seven-day consecutive period. The MWMT is useful because it describes the maximum temperatures in a stream, but is not overly influenced by the maximum temperature of a single day.

Much of the information reported in the literature characterizes temperature needs with terms such as "preferred" or "optimum". Preferred stream temperatures are those that fish most frequently inhabit when allowed to freely select temperatures in a thermal gradient (USEPA 1999a). An optimum range provides suitable temperatures for feeding activity, normal physiological response, and normal behavior (without symptoms of thermal stress) (USEPA 1999a). Optimal temperatures have also been described as those temperatures at which growth rates, expressed as weight gain per unit of time, are maximal for the life stage (Armour 1991).

Salmonid stocks do not tend to vary much in their life history thermal needs, regardless of their geographic location. In the 2001 USEPA document, *Summary of Technical Literature Examining the Physiological Effects of Temperature on Salmonids,* the case is made that there is not enough significant genetic variation among stocks or among species of salmonids to warrant geographically specific water temperature standards.

> Climate conditions vary substantially among regions of the State and the entire Pacific Northwest. …Such [varying climatic] conditions could potentially have led to evolutionary adaptations, resulting in development of subspecies differences in thermal tolerance. …[However,] the literature on genetic variation in thermal effects indicates occasionally significant but very small differences among stocks and increasing differences among subspecies, species, and families of fishes. Many differences that had been attributed in the literature to stock differences are now considered to be statistical problems in analysis, fish behavioral responses under test conditions, or allowing insufficient time for fish to shift from field conditions to test conditions (Mathur & Silver 1980, Konecki et al. 1993, both as cited in USEPA 2001).

Additionally:

> There are many possible explanations why salmonids have not made a significant adaptation to high temperature in streams of the Pacific Northwest. Temperature tolerance is probably controlled by multiple genes, and consequently would be a core characteristic of the species not easily modified through evolutionary change without a radical shift in associated physiological systems. Also, the majority of the life cycle of salmon and steelhead is spent in the ocean rearing phase, where the smolt, subadults, and adults seek waters with temperatures less than 59°F (15°C) (Welch et al, 1995, as cited in USEPA 2001).

As a result, literature on the temperature needs of coho and Chinook salmon and steelhead trout stemming from data collected in streams outside Northern California are cited in this document and are considered relevant to characterizing the thermal needs of salmonids, which use Northern California rivers and streams.

1.3 Adult Migration and Holding

All of the adult migration and holding temperature needs referenced in this section can be found in Table 1 through Table 3. Salmon and trout respond to temperatures during their upstream migration (Bjornn and Reiser 1991). Delays in migration have been observed in response to temperatures that were either too cold or too warm. Most salmonids have evolved with the temperature regime they historically used for migration and spawning, and deviations from the normal pattern can affect survival (Spence et al. 1996).

In a 2003 USEPA document entitled *EPA Region 10 Guidance for Pacific Northwest State and Tribal Water Quality Standards*, it is recommended that the 7-DADM should not exceed 18°C in waters where both adult salmonid migration and "non-core" juvenile rearing occur during the period of summer maximum temperatures. The document does not define what constitutes the "summer" period. Non-core juvenile rearing is defined as moderate to low density salmon and trout rearing usually occurring in the mid or lower part of the basin, as opposed to areas of high density rearing which are termed "core" rearing areas. This criterion is derived from analysis and synthesis of past laboratory and field research. The USEPA believes that this temperature recommendation will protect against lethal conditions, prevent migration blockage, provide optimal or near optimal juvenile growth conditions, and prevent high disease risk by minimizing the exposure time to temperatures which can lead to elevated disease rates.

A 7-DADM temperature of 20°C is recommended by the USEPA (2003) for waterbodies that are used almost exclusively for migration during the period of summer maximum temperatures.

> EPA believes that a 20°C criterion would protect migrating juveniles and adults from lethal temperatures and would prevent migration blockage conditions. However, EPA is concerned that rivers with significant hydrologic alterations (e.g., rivers with dams and reservoirs, water withdrawals, and /or significant river channelization) may experience a loss of temperature diversity in the river, such that maximum temperatures occur for an extended period of time and there is little cold water refugia available for fish to escape maximum temperatures. In this case, even if the river meets a 20°C criterion for maximum temperatures, the duration of exposure to 20°C temperatures may cause adverse effects in the form of increased disease and decreased swimming performance in adults, and increased disease, impaired smoltification, reduced growth, and increased predation for late emigrating juveniles…(USEPA 2003).

Therefore, the USEPA recommends a narrative provision to protect and, if possible, restore the natural thermal regime accompany the 7-DADM 20°C criterion for rivers with significant hydrologic alterations.

In an exhaustive study of both laboratory and field studies of temperature effects on salmonids and related species, USEPA (1999a, 2001) concluded that temperatures of approximately 22-24°C limit salmonid distribution, i.e., they totally eliminate salmonids from a location. USEPA (1999a) also notes that changes in competitive interactions between fish species can lead to a transition in dominance from salmonids to other species at temperatures 2-4°C lower than the range of total elimination.

1.3.1 Steelhead Trout Migration

In a 2002 review of numerous studies, Washington State Department of Ecology (WDOE) concluded that daily average temperatures of 21-24°C are associated with avoidance behavior and migration blockage in steelhead trout. WDOE suggests that the MWMT should not exceed 17-18°C, and daily maximum temperatures should not exceed 21-22°C to be fully protective of adult steelhead migration.

Table 1: Effects of Temperature in Considering Adult Steelhead and Migration

<table>
<tr><th>C</th><th colspan="3">MIGRATION</th></tr>
<tr><td>24</td><td rowspan="3">21-24 Average daily temperature associated with avoidance and migration blockage (2)</td><td colspan="2" rowspan="2">22-24 Temperature range which eliminates salmonids from an area (3,4)</td></tr>
<tr><td>23</td></tr>
<tr><td>22</td><td rowspan="2">21-22 Daily maximum temperature should not exceed this to be fully protective (2)</td><td rowspan="5">18-22 Temperature range at which transition in dominance from salmonids to other species occurs (4)</td></tr>
<tr><td>21</td></tr>
<tr><td>20</td><td colspan="2">20 MWMT should not exceed this in waterbodies used almost exclusively for migration. Should be used in conjunction with a narrative provision about protecting/restoring the natural thermal regime for rivers with significant hydrologic alterations (1)</td></tr>
<tr><td>19</td><td colspan="2"></td></tr>
<tr><td>18</td><td rowspan="2">17-18 MWMT should not exceed this to be fully protective (2)</td><td>18 MWMT should not exceed this where migration and non-core rearing occur (1)</td></tr>
<tr><td>17</td><td></td></tr>
</table>

Sources: (1) USEPA 2003, (2) WDOE 2002, (3) USEPA 2001, (4) USEPA 1999a

1.3.2 Chinook Salmon Migration and Holding

USEPA (2001) cited various literature sources that identified thermal blockages to Chinook salmon migration at temperatures ranging from 19-23.9°C, with the majority of references citing migration barriers at temperatures around 21°C.

A radio tracking study on spring Chinook revealed that when maximum temperatures of 21.1°C were reached, a thermal barrier to migration was established (Bumgarner et al. 1997, as cited by USEPA 1999a). Bell (1986) reviewed various studies and notes spring Chinook migrate at water temperatures ranging from 3.3-13.3°C, while fall Chinook migrate at temperatures of 10.6-19.6°C. Preferred temperatures for Chinook range from 7.2-14.5°C (Bell 1986). Based on a technical literature review, WDOE (2002) concluded that daily maximum temperatures should not exceed 21-22°C during Chinook migration.

Table 2: Effects of Temperature in Considering Adult Chinook and Migration and Holding

°C	MIGRATION				
24			22-24 Temperature range which eliminates salmonids from an area (3,5)	19-23.9 Range of temperatures causing thermal blockage to migration (3)	
23	23 Klamath Basin fall Chinook begin migration upstream at temperatures as high as 23C if temperatures are rapidly falling (6)				
22	22 Klamath Basin fall Chinook will not migrate upstream when mean daily temperatures are 22C or greater (6)				18-22 Temperature range at which transition in dominance from salmonids to other species occurs (5)
21	21-22 Daily maximum temperature should not exceed this range to be protective of migration (2)		21 Most references cite as thermal block to migration (3) 21 Klamath Basin fall Chinook will not migrate upstream if temperatures are 21C or above and rising (6)		
20	20 MWMT should not exceed this in waterbodies used almost exclusively for migration. Should be used in conjunction with a narrative provision about protecting/restoring the natural thermal regime for rivers with significant hydrologic alterations (1)				
19			10.6-19.6 Temperature range where adult fall Chinook migrate (4)		
18				18 MWMT should not exceed this where migration and non-core rearing occur (1)	
17	16-17 MWMT should be below this where Chinook are holding (2)				
16					
15					
14	7.2-14.5 Preferred temperatures for Chinook (4)			13-14 Average daily temperature should be below this where spring Chinook are holding (2)	
13					
12				3.3-13.3 Temperature range where adult spring Chinook migrate (4)	
11					
10					
9					
8					
7					
6					
5					
4					
3					

Sources: (1) USEPA 2003, (2) WDOE 2002, (3) USEPA 2001, (4) Bell 1986, (5) USEPA 1999a, (6) Strange 2006

Utilizing radio telemetry to track the movements and monitor the internal body temperatures of adult fall Chinook salmon during their upriver spawning migration in the Klamath basin, Strange (2006) found that fall Chinook will not migrate upstream when mean daily temperatures are $\geq$22°C. Strange also noted that adult fall Chinook in the

Klamath basin will not migrate upstream if temperatures are 21°C or above and rising, but will migrate at temperatures as high as 23°C if temperatures are rapidly falling.

Spring Chinook begin entering freshwater streams during a relatively cool-water season but must hold throughout the warm summer period, awaiting cooler spawning temperatures (ODEQ 1995a). The cumulative effects of management practices such as elevated water temperatures, reduced cover from large woody debris, and reduced resting pool area due to pool filling increase the susceptibility of holding adult fish to mortality from thermal effects (The Oregon Department of Environmental Quality [ODEQ] 1995a). WDOE states that where spring Chinook are holding over for the summer prior to spawning the average daily water temperature should be below 13-14°C and the MWMT should be below 16-17°C (WDOE 2002).

1.3.3 Coho Salmon Migration

Migration for coho is delayed when water temperatures reach 21.1°C, and the preferred water temperatures for coho range from 11.7-14.5°C (Bell 1986). In California coho salmon typically migrate upstream when water temperatures range from 4-14°C (Briggs, 1953 and Shapovalov and Taft, 1954, as cited by Hassler, 1987). WDOE reviewed various studies and concluded that to be protective of adult coho migration, MWMTs should not exceed 16.5°C (WDOE 2002).

Table 3: Effects of Temperature in Considering Adult Coho and Migration

<table>
<tr><th>°C</th><th colspan="2">MIGRATION</th></tr>
<tr><td>24</td><td rowspan="3">22-24 Temperature range which eliminates salmonids from an area (3,6)</td><td></td></tr>
<tr><td>23</td><td></td></tr>
<tr><td>22</td><td rowspan="5">18-22 Temperature range at which transition in dominance from salmonids to other species occurs (6)</td></tr>
<tr><td>21</td><td>21.1 Migration is delayed when temperatures reach this value (4)</td></tr>
<tr><td>20</td><td>20 MWMT should not exceed this in waterbodies used almost exclusively for migration. Should be used in conjunction with a narrative provision about protecting/restoring the natural thermal regime for rivers with significant hydrologic alterations (1)</td></tr>
<tr><td>19</td><td></td></tr>
<tr><td>18</td><td>18 MWMT should not exceed this where migration and non-core rearing occur (1)</td></tr>
<tr><td>17</td><td colspan="2"></td></tr>
<tr><td>16</td><td colspan="2">16.5 MWMT should not exceed this value to be fully protective (2)</td></tr>
<tr><td>15</td><td colspan="2"></td></tr>
<tr><td>14</td><td rowspan="4">11.7-14.5 Preferred temperature range (4)</td><td rowspan="5">4-14 Temperature range at which migration typically occurs (5)</td></tr>
<tr><td>13</td></tr>
<tr><td>12</td></tr>
<tr><td rowspan="2">11</td></tr>
<tr><td>11.4 Preferred temperature (7)</td></tr>
</table>

Sources: (1) USEPA 2003, (2) WDOE 2002, (3) USEPA 2001, (4) Bell 1986, (5) Briggs 1953, Shapovalov and Taft 1954, as cited by Hassler 1987, (6) USEPA 1999a, (7) Reutter and Herdendorf 1974

1.4 Spawning, Incubation, and Emergence

All of the spawning, incubation, and emergence temperature needs referenced in this section can be found in Table 4 through Table 7. Many sources have stated that temperature affects the time of migration i adults and thus the time of spawning, which influences the incubation temperature regime, which in turn influences survival rates, development rates, and growth of embryos and alevins (Murray and McPhail 1988). USEPA Region 10 recommends that the 7-DADM temperatures should not exceed 13°C for salmonid spawning, egg incubation, and fry emergence (USEPA 2003). Optimum temperatures for salmonid egg survival ranges from 6-10°C (USEPA 2001).

1.4.1 Steelhead Spawning, Incubation, and Emergence

In a discussion paper and literature summary evaluating temperature criteria for fish species including salmonids and trout, WDOE (2002) cites studies showing that steelhead were observed spawning in temperatures ranging from 3.9-21.1°C, and that the preferred temperatures for steelhead spawning range from 4.4-12.8°C. In a review of various studies, Bell (1986) concludes that steelhead spawning occurs at water temperatures ranging from 3.9-9.4°C.

Steelhead and rainbow trout eggs had the highest survival rates between 5-10°C according to Myrick and Cech (2001), and while they can tolerate temperatures as low as 2°C or as high as 15°C, mortality is increased at these temperatures. WDOE (2002) reviewed literature on the survival of steelhead and rainbow trout embryos and alevins at various temperatures and concluded that the average water temperature should not exceed 7-10°C throughout development, and the maximum daily average temperature should be below 11-12°C at the time of hatching (WDOE 2002).

Table 4: Effects of Temperature in Considering Steelhead Incubation and Emergence

<table>
<tr><th>°C</th><th colspan="3">INCUBATION AND EMERGENCE</th></tr>
<tr><td>15</td><td colspan="3">15 Steelhead and rainbow trout eggs can survive at temperatures as high as this but mortality is high compared to lower temperatures (3)</td></tr>
<tr><td>14</td><td colspan="3"></td></tr>
<tr><td>13</td><td colspan="3">13 MWMT should not exceed this value to be protective of spawning, egg incubation, and fry emergence (1)</td></tr>
<tr><td>12</td><td colspan="3" rowspan="2">11-12 Maximum daily average temperature should be below this range at the time of hatching (2)</td></tr>
<tr><td>11</td></tr>
<tr><td>10</td><td rowspan="6">5-10 Steelhead and rainbow trout eggs had the highest survival within this range (3)</td><td rowspan="5">6-10 Optimum temperature for salmonid eggs survival to hatching (4)</td><td rowspan="4">7-10 Average daily temperature should not exceed this range throughout embryo development (2)</td></tr>
<tr><td>9</td></tr>
<tr><td>8</td></tr>
<tr><td>7</td></tr>
<tr><td>6</td><td></td></tr>
<tr><td>5</td><td></td></tr>
<tr><td>4</td><td colspan="3"></td></tr>
<tr><td>3</td><td colspan="3"></td></tr>
<tr><td>2</td><td colspan="3">2 Steelhead and rainbow trout eggs can survive at temperatures as low as this but mortality is high compared to higher temperatures (3)</td></tr>
</table>

Sources: (1) USEPA 2003, (2) WDOE 2002, (3) Myrick and Cech 2001, (4) USEPA 2001

Table 5: Effects of Temperature in Considering Steelhead, Chinook, and Coho Spawning

°C	Steelhead			Chinook			Coho		All Salmonids
21	3.9-21.2 Steelhead observed spawning in this temp. range (2)								
20									
19									
18									
17						5.6-17.7 Range of temps. associated with spawning from references reviewed (2)			
16									
15				13-15.5 Temp. range at which pre-spawning mortality becomes pronounced in ripe spring Chinook (4)					
14					14.5 Majority of refs. cite daily max temps. associated with spawning below this level (2)				
13					5.6-13.9 Recommended temperature range for spawning (5)		13 Daily maximum temp. not to exceed this value to be protective (6)	4.4-13.3 Typical temps. during which spawning occurs (2)	13 MWMT not exceed this value during spawning, egg incubation, and fry emergence (1)
12			4.4-12.8 Preferred temp. range for spawning (2)	5.6-12.8 Recom-mended temperature range for spawning (4)					
11									
10							10 MWAT not exceed this value to be protective (6)		
9		3.9-9.4 Temp. range where spawning occurs (3)					4.5-9.4 Preferred spawning temperature range (3)		
8									
7									
6									
5									
4									
3									

Sources: (1) USEPA 2003, (2) WDOE 2002, (3) Bell 1986, (4) ODEQ 1995a, (5) Reiser and Bjornn 1979 as cited by Armour et al. 1991, (6) Brungs and Jones 1977

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1.4.2 Chinook Spawning, Incubation, and Emergence

ODEQ (1995a) reviewed numerous studies and recommended a temperature range of 5.6-12.8°C for spawning Chinook. A discussion paper and literature summary by WDOE in 2002 found that the literature reviewed noted a wide range of temperatures associated with Chinook spawning (5.6-17.7°C), although the majority of these temperature observations cite daily maximum temperatures below 14.5°C. A spawning temperature range of 5.6-13.9°C is recommended for spring, summer, and fall Chinook salmon populations in the Pacific Northwest (Reiser and Bjornn 1979, as cited by Armour et al. 1991). When ripe adult spring Chinook females experience temperatures above 13-15.5°C, pre-spawning adult mortality becomes pronounced (ODEQ 1995a). Additionally, there is decreased survival of eggs to the eyed stage and alevin development is inhibited due to the exposure of the ripe female to warm temperatures, even if the stream temperatures during the egg and alevin development are appropriate (ODEQ 1995a).

Table 6: Effects of Temperature in Considering Chinook Incubation and Emergence

<table>
<tr><th>°C</th><th colspan="5">INCUBATION AND EMERGENCE</th></tr>
<tr><td>20</td><td colspan="5" rowspan="4">17.5-20 The highest single day maximum temperature should not exceed this range to protect eggs and embryos from acute lethal conditions (2)</td></tr>
<tr><td>19</td></tr>
<tr><td>18</td></tr>
<tr><td>17</td></tr>
<tr><td>16</td><td colspan="4"></td><td rowspan="15">1.7-16.7 Eggs can survive these temps. but mortality is greatly increased at the extremes (3)</td></tr>
<tr><td>15</td><td colspan="4"></td></tr>
<tr><td>14</td><td rowspan="12">5-14.4 Recom-mended temp. range for incubation (4)</td><td rowspan="2">13.5-14.5 Daily maximum temperatures should not exceed this from fertilization through initial fry development (5)</td><td colspan="2">14 Moderate embryo survival (6)</td><td rowspan="13">2-14 Range of temps. for normal embryo develop-ment (6)</td></tr>
<tr><td>13</td><td colspan="2">13 MWMT should not exceed this value to be protective of spawning, egg incubation, and fry emergence (1)</td></tr>
<tr><td>12</td><td></td><td rowspan="4">4-12 Lowest levels of egg mortality at these temps. (3)</td><td rowspan="2">11-12.8 Average daily temperatures should be below this range at beginning of incubation (2)</td></tr>
<tr><td>11</td><td>11 High embryo survival (6)</td></tr>
<tr><td>10</td><td rowspan="2">9-10 Optimal temp. should be below this range (5)</td><td rowspan="4">6-10 Optimum temperature for salmonid eggs survival to hatching (5)</td></tr>
<tr><td>9</td></tr>
<tr><td>8</td><td>8-9 Seasonal ave. temps. should not exceed this range from fertilization through initial fry development (2)</td></tr>
<tr><td></td><td>8 High embryo survival (6)</td></tr>
<tr><td>7</td><td></td></tr>
<tr><td>6</td><td></td></tr>
<tr><td>5</td><td>5 High embryo survival (6)</td></tr>
<tr><td>4</td><td></td><td></td></tr>
<tr><td>3</td><td colspan="3"></td></tr>
<tr><td>2</td><td colspan="3">2 Poor embryo survival (6)</td></tr>
<tr><td>1</td><td colspan="4"></td></tr>
</table>

Sources: (1) USEPA 2003, (2) WDOE 2002, (3) Myrick and Cech 2001, (4) Reiser and Bjornn 1979, as cited by Armour et al. 1991, (5) USEPA 2001, (6) Murray and McPhail 1988

WDOE (2002) reviewed numerous references on the effects of various temperatures on Chinook incubation and development and used these studies to derive the temperatures that are protective of Chinook salmon from fertilization through fry development. These reviewed references include laboratory studies assessing Chinook embryo survival at various constant temperatures, studies attempting to mimic naturally fluctuating temperatures experienced by incubating eggs, studies which have made stepwise reductions in the incubation temperatures as incubation progressed to evaluate survival of eggs, and studies on the effects of transferring eggs to optimal constant incubation temperatures after they had been exposed to higher temperatures for various periods. As a result of this review, WDOE (2002) recommends that average daily temperatures remain below 11-12.8°C at the initiation of incubation, and that the seasonal average should not exceed 8-9°C in order to provide full protection from fertilization through initial fry development. The highest single day maximum temperature should not exceed 17.5-20°C to protect eggs and embryos from acute lethal conditions (WDOE 2002).

USEPA (2001) reviewed multiple literature sources and concluded that optimal protection from fertilization through initial fry development requires that temperatures be maintained below 9-10°C, and that daily maximum temperatures should not exceed 13.5-14.5°C. Reiser and Bjornn (1979, as cited by Armour et al. 1991) recommended temperatures of 5.0-14.4°C for spring, summer and fall Chinook salmon incubation in the Pacific Northwest. Myrick and Cech (2001) reviewed studies on the Sacramento-San Joaquin River and concluded that the lowest levels of Chinook egg mortality occurred at temperatures between 4-12°C, and while eggs can survive at temperatures from 1.7-16.7°C, mortality is greatly increased at the temperature extremes.

Embryo survival was studied in a laboratory experiment conducted by Murray and McPhail (1988). They incubated five species of Pacific salmon, including Chinook, at five incubation temperatures (2, 5, 8, 11, 14°C). Chinook embryo survival was high at 5, 8, and 11°C, but survival was moderate at 14°C and poor at 2°C. As a result of their study, Murray and McPhail concluded that the range of temperatures for normal embryo development is > 2°C and <14°C (Murray and McPhail 1988).

1.4.3 Coho Spawning, Incubation, and Emergence

In 2002, WDOE found that several studies and literature reviews state that spawning activity in coho may typically occur in the range of 4.4-13.3ºC. According to a review by Bell (1986), preferred spawning temperatures range from 4.5-9.4ºC. Brungs and Jones (1977) used existing data on the optimum and range of temperatures for coho spawning and embryo survival to create criteria using protocols from the National Academy of Sciences and National Academy of Engineering. The resultant criteria were that the MWAT should not exceed 10ºC and the daily maximum temperature should not exceed 13ºC to be protective of coho (Brungs and Jones 1977, p.16).

In a discussion paper and literature summary WDOE (2002) reviewed studies that assessed the survival of embryos and alevin at various temperatures. Based on the findings of these studies WDOE (2002) has determined that the average daily temperature during the incubation period should be at or below 8-10ºC to fully support

this coho salmon life stage. According to a review of various literature sources by Bell (1986), the preferred emergence temperatures for coho range from 4.5-13.3ºC. USEPA (2001) concluded that to fully support pre-emergent stages of coho development MWMTs should not exceed 9-12°C.

Table 7: Effects of Temperature in Considering Coho Incubation and Emergence

<table>
<tr><th>°C</th><th colspan="5">INCUBATION AND EMERGENCE</th></tr>
<tr><td>14</td><td colspan="5">14 Upper limit for normal embryo development (5)</td></tr>
<tr><td>13</td><td colspan="2">13 MWMT should not exceed this value to be protective of spawning, egg incubation, and fry emergence (1)</td><td colspan="2">13 Daily maximum temperature should not exceed this value to be protective (6)</td><td rowspan="10">4.5-13.3 Preferred emergence temperature range (3)</td></tr>
<tr><td>12</td><td colspan="2"></td><td colspan="2" rowspan="2">9-12 MWMT should not exceed this range to be fully protective (4)</td><td></td></tr>
<tr><td>11</td><td colspan="2"></td><td></td></tr>
<tr><td>10</td><td rowspan="6">6-10 Optimum temperature for salmonid eggs survival to hatching (4)</td><td rowspan="4">8-10 Ave. daily temp. during incubation should be at or below this to be supportive (2)</td><td rowspan="2">10 MWAT should not exceed this to be protective (6)</td></tr>
<tr><td>9</td><td></td></tr>
<tr><td>8</td><td colspan="2"></td><td></td></tr>
<tr><td>7</td><td colspan="3"></td></tr>
<tr><td>6</td><td colspan="3"></td></tr>
<tr><td>5</td><td colspan="3"></td></tr>
<tr><td>4</td><td colspan="3"></td></tr>
</table>

Sources: (1) USEPA 2003, (2) WDOE 2002, (3) Bell 1986, (4) USEPA 2001, (5) Murray and McPhail 1988, (6) Brungs and Jones 1977

Murray and McPhail (1988) incubated five species of Pacific salmon, including coho, at five temperatures (2, 5, 8, 11, 14°C) to determine embryo survival at various temperatures. Coho embryos suffered increased mortality above 11°C although survival was still high. They concluded that the upper limit for normal coho embryo development is 14°C (Murray and McPhail 1988).

1.5 Freshwater Rearing and Growth

All of the freshwater rearing and growth temperature needs referenced in this section can be found in Table 8 through Table 10. Temperature affects metabolism, behavior, and survival of both juvenile fish as well as other aquatic organisms that may be food sources. In streams of the Northern California Coast, including the Klamath River, young Chinook, coho and steelhead may rear in freshwater from one to four years before migrating to the ocean.

In an exhaustive study of both laboratory and field studies of temperature effects on salmonids and related species, USEPA (1999a) concluded that temperatures of approximately 22-24°C limit salmonid distribution, i.e., they totally eliminate salmonids from a location. USEPA (1999a) also notes that changes in competitive interactions between fish species can lead to a transition in dominance from salmonids to other species at temperatures 2-4°C lower than the range of total elimination.

To protect salmon and trout during summer juvenile rearing the USEPA (2003) for Region 10 provided a single guidance metric designating 16°C as the 7-DADM

temperature that should not be exceeded in areas designated as "core" rearing locations. Core rearing areas are defined as areas with moderate to high densities of summertime salmonid juvenile rearing generally found in the mid- to upper portions of river basins. This criterion will protect juvenile salmonids from lethal temperatures, provide optimal to upper optimal conditions for juvenile growth depending on the time of year, avoid temperatures where salmonids are at a competitive disadvantage with other fish species, protect against increased disease rates caused by elevated temperatures, and provide temperatures which salmonids prefer according to scientific studies.

1.5.1 Steelhead Freshwater Rearing and Growth

Nielsen et al. (1994) studied thermally stratified pools and their use by juvenile steelhead in three California North Coast rivers including the Middle Fork Eel River, Redwood Creek at Redwood National Park, and Rancheria Creek, located in the Navarro River watershed. In detailed observations of juvenile steelhead behavior in and near thermally stratified pools in Rancheria Creek, Nielsen et al. (1994) noted behavioral changes including decreased foraging and increased aggressive behavior as pool temperature reached approximately 22°C. As pool temperature increased above 22°C, juveniles left the observation pools and moved into stratified pools where temperatures were lower.

Wurtsbaugh and Davis (1977, as cited by USEPA 2001) found that steelhead trout growth could be enhanced by temperature increases up to 16.5°C. Using a risk assessment approach which took into account "realistic food estimates", Sullivan et al. (2000) report temperatures of 13-17.0°C (MWAT), 14.5-21°C (MWMT), and 15.5-21°C (annual maximum) will ensure no more than a 10% reduction from maximum growth for steelhead. Reduction from maximum growth will be ≤20% for temperatures ranging from 10-19.0°C (MWAT), 10-24°C (MWMT), and 10.5-26°C (annual maximum).

A literature review was conducted by WDOE (2002) in which studies to determine the water temperature that would allow for maximum growth of steelhead trout were analyzed. These included laboratory studies conducted at constant and fluctuating temperatures. One of the studies was conducted using feeding rates comparable to those observed in natural creeks, although most of the laboratory studies were conducted under satiated feeding conditions. As a result of this review of laboratory studies conducted at constant temperatures, WDOE (2002) concludes that under satiated rations growth may be maximized at temperatures as high as 17.2-19°C. Results from laboratory studies using variable temperatures show maximum growth occurs at average daily temperatures between 15.5-18°C, and that under feeding rates similar to natural conditions at various times of the year maximum growth rates occurred at mean temperatures of 13.3°C (spring season), 15.2°C (fall season) and 16.2°C (summer season).

Table 8: Effects of Temperature in Considering Juvenile Steelhead Rearing and Growth

°C	REARING AND GROWTH
26	21-26 Annual maximum temp. which will ensure no more than 20% reduction from max. growth (4)
25	
24	22-24 Temperature range which totally eliminates salmonids from area, limiting their distribution (6); 21-24 MWMT which will ensure no more than 20% reduction from max growth (4)
23	>22 Juveniles left observation pools and moved to pools with lower temperatures (2)
22	22 Decreased foraging, increased aggressive behavior (2); 18-22 Temperature range at which transition in dominance from salmonids to other species occurs (6)
21	14.5-21 MWMT which will ensure no more than 10% reduction from maximum growth (4); 15.5-21 Annual maximum temperature which will ensure no more than 10% reduction from maximum growth (4)
20	
19	17-19 MWAT will ensure no more than 20% reduction from max. growth (4); 17.2-19 Growth may be maximized at temperatures as high as this under satiated feeding conditions, lab studies at constant temperature (5)
18	
17	15.5-18 Average daily temperatures at which maximum growth occurs under satiated feeding, lab studies at varying temps (5); 13-17 MWAT range which will ensure no more than 10% reduction from maximum growth (4)
16	16.5 Growth enhanced by temp. increases up to this temp. (3); 16.2 Mean temp. at which max. growth occurred during the summer, lab studies using natural feeding conditions and varying temps. (5); 16 MWMT should not exceed this value to be protective of core rearing locations (1)
15	15.2 Mean temp. at which max. growth occurred during the fall, lab studies using natural feeding conditions and varying temps. (5); 10.5-15.5 Annual maximum temperature which will ensure no more than 20% reduction from maximum growth (4)
14	
13	13.3 Mean temp. at which max. growth occurred during the spring, lab studies using natural feeding conditions and varying temps. (5); 10-13 MWAT will ensure no more than 20% reduction from maximum growth (4); 10-14.5 MWMT which will ensure no more than 20% reduction from maximum growth (4)
12	
11	
10	

Sources: (1) USEPA 2003, (2) Nielsen et al. 1994, (3) Wurtsbaugh and Davis 1977, as cited by USEPA 2001, (4) Sullivan et al. 2000, (5) WDOE 2002, (6) USEPA 1999a

1.5.2 Chinook Freshwater Rearing and Growth

In a laboratory study, Brett (1952) demonstrated that juvenile Chinook salmon, acclimated to a temperature of 20°C, selectively aggregated in areas where the temperature was in the region of 12-13°C.

ODEQ (1995a), reviewed numerous studies and concluded for juvenile spring Chinook salmon rearing, positive growth takes place at temperatures between 4.5-19°C, and that optimum rearing production is between 10.0-15.6°C. However, as the extremes of this temperature range are reached growth reaches zero. Above and below these thresholds growth becomes negative as feeding ceases and respiration rates increase and/or decrease rapidly.

After synthesizing data from several sources USEPA (2001), came up with the same recommended optimum temperature zone for all Chinook salmon as ODEQ (1995a) of 10.0-15.6°C. While there is research suggesting that some Chinook stocks exhibit adequate rearing capabilities above 15.6°C, USEPA (2001) conclude that anything over this threshold significantly increases the risk of mortality from warm-water diseases.

In a laboratory study Marine and Cech (2004) studied the incremental effects of chronic exposure to three temperature regimes (13-16 °C, 17-20 °C, and 21-24 °C) on Chinook juveniles during rearing and smoltification. Their findings reflected that Chinook juveniles reared at the 17-20 ºC and 21-24 ºC temperature ranges experienced significantly decreased growth rates, impaired smoltification indices, and increased predation vulnerability compared with juveniles reared at 13-16 ºC.

In a field study Chinook grew faster in a stream where temperatures peaked at 16°C compared to a stream where temperatures peaked at 20°C (ODFW 1992, as cited by WDOE 2002). WDOE (2002) reviewed literature on Chinook growth including laboratory studies conducted at a constant temperature, laboratory studies conducted at fluctuating temperatures, and field studies to evaluate the water temperature that would be protective of Chinook and allow for maximum growth. Most of the laboratory studies were conducted under satiated feeding conditions, although one of the studies was conducted using feeding rates more comparable to those observed in natural creeks. As a result of this review of laboratory studies conducted at constant temperatures, WDOE (2002) concludes that maximum growth is expected to occur with exposure to constant temperatures from 15.6-19°C. However, increased growth at temperatures above 15.6°C was inconsistently greater, and under natural rations the temperatures at which maximum growth occurs may decline by as much as 4.2°C. Recommendations based on the review of two laboratory studies conducted at fluctuating temperatures are that "…average temperatures below 19°C are necessary to support maximum growth rates in Chinook salmon, and that the average temperature that produces maximum growth rates likely lies between 15-18°C (median 16.5°C)".

Table 9: Effects of Temperature in Considering Juvenile Chinook Rearing and Growth

°C	REARING AND GROWTH				
24	22-24 Temperature range which totally eliminates salmonids from area, limiting their distribution (7)			21-24 Decreased growth, impaired smoltification, increased predation compared to juveniles reared at 13-16 (6)	
23					
22					
21			18-22 Temperature range at which transition in dominance from salmonids to other species occurs (7)		
20				17-20 Decreased growth, impaired smoltification, increased predation compared to juveniles reared at 13-16 (6)	
19	19 Temperatures above this do not support maximum growth, lab studies at varying temperatures (3)	15.6-19 Maximum growth expected according to lab studies conducted at constant temperature and satiated rations. Under natural feeding conditions maximum growth may occur at temperatures as much as 4.2C lower (3)			4.5-19 Temperature range at which positive growth takes place (5)
18	15-18 Average temperature where maximum growth occurs, lab studies conducted at varying temperatures (3)				
17					
16			16 Chinook grew faster in a stream where temperatures peaked at 16 than when they peaked at 19C (3)	13-16 Increased growth, unimpaired smoltification, lower predation compared to juveniles reared at 21-24, or 17-20 (6)	
			16 MWMT should not exceed this value to be protective of core rearing locations (2)		
15			10-15.6 Optimal temperature range for rearing (5)		
14	10-15.6 Temperature range for optimal growth. Anything over this threshold increases the risk of mortality from warm water disease (1)				
13				12-13 Juvenile Chinook acclimated to 20 selectively aggregate to these water temperatures (4)	
12					
11					
10					
9					
8					
7					
6					
5					
4					

Sources: (1) USEPA 2001, (2) USEPA 2003, (3) WDOE 2002, (4) Brett 1952, (5) ODEQ 1995a, (6) Marine and Cech 2004, (7) USEPA 1999a

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1.5.3 Coho Freshwater Rearing and Growth

In a study of juvenile coho presence and absence in the Mattole watershed, Welsh et al. (2001) used logistic regression to determine that an MWAT greater than 16.8°C or a MWMT greater than 18.1°C may preclude the presence of juvenile coho salmon in the stream. The criterion correctly determined the presence or absence of juvenile coho in 18 of 21 streams. Welsh et al. (2001) also reported that juvenile coho were found in all streams with an MWAT less than 14.5°C, or a MWMT less than 16.3°C.

Sullivan et al. (2000) reviewed sub-lethal and acute temperature thresholds from a wide range of studies, incorporating information from laboratory-based research, field observations, and risk assessment approaches. Using a risk assessment approach based on "realistic food estimates" Sullivan et al (2000) suggest that MWATs ranging from 12.5-14.5°C for coho will result in no more than a 10% reduction from maximum growth, and that a range for the MWAT of 9-18.5°C will reduce growth no more than 20% from maximum. Sullivan et al. (2000) also calculated temperature ranges for MWMT (13-16.5°C) and the annual maximum temperature (13-17.5°C) that will result in no more than a 10% reduction in maximum growth. They further calculated ranges for MWMT (9-22.5°C) and the annual maximum temperature (9.5-23°C) that will result in no more than a 20% growth loss.

In an attempt to determine the water temperature that will allow for maximum growth of coho salmon, WDOE (2002) reviewed literature on laboratory studies conducted at a constant temperature and fluctuating temperatures, and field studies. The two laboratory studies reviewed were conducted under satiated feeding conditions. Shelbourn (1980, as cited by WDOE 2002) found that maximum growth occurred at a constant temperature of 17°C, while Everson (1973, as cited by WDOE 2002) tested fish at different temperatures and determined that coho had the greatest growth at the temperature test regime from 12.1-20.8°C (median 16.5°C). While the various field studies reviewed did not provide an estimate of the temperature best for maximum growth they did allow for WDOE (2002) to conclude that weekly average temperatures of 14-15°C were more beneficial to growth than lower temperature regimes, and daily maximum temperatures of 21-26°C were detrimental to growth.

Brett (1952) acclimated five different species of salmon to various temperatures ranging from 5-24°C and found that coho salmon showed the greatest preference for temperatures between 12-14°C. It was also determined that coho showed a general avoidance of temperatures above 15°C even in fish who were acclimated to temperatures as high as 24°C.

Konecki et al. (1995a) raised two groups of juvenile coho salmon under identical regimes to test the hypothesis that the group from a stream with lower and less variable temperature would have a lower and less variable preferred temperature than the group from a stream with warmer and more variable temperatures. Results reflected that the two groups tended to differ in their preferred temperature range as predicted above, but the differences were slight. Konecki et al. (1995a) concluded that the temperature preference of juvenile coho salmon in their study was 10-12°C.

Table 10: Effects of Temperature in Considering Juvenile Coho Rearing and Growth

°C	REARING AND GROWTH				
26	21-26 Daily maximum temperatures in this range are detrimental to growth, according to field studies (3)				
25					
24			22-24 Temperature range which totally eliminates salmonids from an area, limiting their distribution (9)		
23					17.5-23 Annual maximum temperature will ensure no more than 20% reduction from maximum growth (2)
22					
21			16.5-22.5 MWMT will ensure no more than 20% reduction from maximum growth (2)	18-22 Temperature range at which transition in dominance from salmonids to other species occurs (9)	
20		12.1-20.8 Greatest growth occurs in this temperature range under satiated conditions, lab study (7)			
19					
18	18.1 MWMT above this may preclude the presence of juvenile coho in steams (5)				
17	14.5-18.5 MWAT will ensure no more than 20% reduction from maximum growth (2)			17 Maximum growth at this constant temperature, at satiated rations in a lab study (6)	13.5 17.5 Annual maximum temperature will ensure no more than 10% reduction from maximum growth (2)
16				16.8 MWAT above this may preclude the presence of juvenile coho in streams (5)	
				16.3 Juveniles found in all streams with MWMT less than this value (5)	
			13-16.5 MWMT will ensure no more than 10% reduction from maximum growth (2)	16 MWMT not exceed this value to be protective of core rearing locations (1)	
15				>15 Juveniles show avoidance, even those acclimated to 24C (4)	
14				14-15 Weekly average temperatures in this range are more beneficial than lower temperatures (3)	
	14.5 Juvenile coho found in all streams with MWAT less than this value (5)			12-14 Preferred temperature range (4)	
13	12.5-14.5 MWAT will ensure no more than 10% reduction from maximum growth (2)				9.5-13.5 Annual maximum temperature will ensure no more than 20% reduction from max. growth (2)
12			9-13 MWMT will ensure no more than 20% reduction from maximum growth (2)	10-12 Preferred temperature range (8)	
11	9-12.5 MWAT will ensure no more than 20% reduction from maximum growth (2)				
10					
9					

Sources: (1) USEPA 2003, (2) Sullivan et al. 2000, (3) WDOE 2002, (4) Brett 1952, (5) Welsh et al. 2001, (6) Shelbourn 1980, as cited by WDOE 2002, (7) Everson 1973, as cited by WDOE 2002, (8) Konecki et al. 1995a, (9) USEPA 1999a

1.6 Lethality

All of the lethal temperatures referenced in this section can be found in Table 11. WDOE (2002) reviewed literature on three types of studies (constant exposure temperature studies, fluctuating temperature lethality studies, and field studies) and used this information to calculate the MWMT that, if exceeded, may result in adult and juvenile salmonid mortality. The resultant MWMTs for these various types of studies are as follows: constant exposure studies 22.64°C, fluctuating lethality studies 23.05°C , and field studies 22.18°C.

1.6.1 Steelhead Lethality

Coutant (1970, as cited by USEPA 1999a) found that Columbia River steelhead, which were acclimated to a river temperature of 19°C, had a lethal threshold of 21°C. Bell (1986) reviewed various studies and states that the lethal threshold for steelhead is 23.9°C. According to the California Department of Fish and Game (2001, p.419), temperatures of 21.1°C have been reported as being lethal to adults.

1.6.2 Chinook Lethality

In a laboratory study, Brett (1952) acclimated five different species of juvenile salmon to various temperatures ranging from 5-24°C. At temperatures of 24°C and below there was 100% survival of fish during the one-week duration of the experiment. Brett (1952) concluded that the lethal temperature (temperature where survival becomes less than 100%) was between 24.0 and 24.5°C, and the ultimate upper lethal temperature was 25.1°C (temperature at which 50% of the population is dead after infinite exposure). A review of numerous studies led Bell (1986) to conclude that the upper lethal temperature for Chinook is 25°C. Myrick and Cech (2001) reviewed literature on studies from the Central Valley and found data to suggest that the chronic (exposure >7 days) upper lethal limit for juvenile Chinook is approximately 25°C.

1.6.3 Coho Lethality

In a review of various literature sources, Bell (1986) found that the upper lethal temperature for coho is 25.6°C. Brett (1952) concluded that the ultimate upper lethal temperature of juvenile coho salmon was 25.0°C (temperature at which 50% of the population is dead after infinite exposure). Thomas et al. (1986) conducted a study to determine the mortality of coho subjected to fluctuating temperatures. It was determined that the LT50 (the temperature at which 50% of the population will die) for fish acclimated to a 10-13°C cycle was 26°C for presmolts (age-2 fish), and 28°C for age-0 fish.

Table 11: Effects of Temperature in Considering Lethality and Salmonids

°C	Steelhead	Chinook	Coho	All Salmonids
28			28 LT50[1] for age 0-fish acclimated to a 10-13C cycle (6)	
27				
26			26 LT50[1] for presmolts (age 2-fish) acclimated to a 10-13C cycle (6)	
25		25.1 Upper lethal temp. at which 50% of the population would die after infinite exposure, juvenile Chinook acclimated to temperatures from 5-24C (4)	25.6 Upper lethal threshold (3)	
		25 Upper lethal threshold (3)	25 Upper lethal temp. at which 50% of the population would die after infinite exposure, juvenile coho acclimated to temps. from 5-24C (4)	
		25 Chronic (exposure >7 days) upper lethal limit for juvenile Chinook (5).		
24		24-24.5 Survival becomes less than 100% for juvenile Chinook acclimated to temperatures from 5-24C (4)		
23	23.9 Upper lethal threshold for steelhead (3)			23.05 do not exceed this value to prevent adult and juvenile mortality, data from fluctuating temp. studies (1)
22				22.64 do not exceed this value to prevent adult and juvenile mortality, data from constant exposure studies (1)
				22.18 do not exceed this value to prevent adult and juvenile mortality, data from field studies (1)
21	21.1 Temperature lethal to adults (7)			
	21 Lethal threshold for steelhead acclimated to 19C (2)			

[1] Maximum temperature in the cycle at which 50% mortality occurred
Sources: (1) WDOE 2002, (2) Coutant 1970, as cited by USEPA 1999a, (3) Bell 1986, (4) Brett 1952, (5) Myrick and Cech 2001, (6) Thomas et al. 1986, (7) California Department of Fish and Game (CDFG) 2001

1.7 Disease

All of the effects of temperatures on disease risk in salmonids referenced in this section can be found in Table 12. WDOE (2002) reviewed studies of disease outbreak in salmonids and estimates that an MWMT of less than or equal to 14.38°C (midpoint of 12.58-16.18 range) will virtually prevent warm water disease effects. To avoid serious

Table 12: Effects of Temperature in Considering Disease and Salmonids

°C	Ich		Ceratomyxosis		Columnaris	Disease (general)	
26		21-26.7 Optimum temp. range for Ich, compilation of temps. from three references (3,4,5)					
25							
24	>24 Lifecycle takes less than 4 days (5)						
23	21-23.9 Life cycle takes as few as 3-4 days (5)		23.3 Juvenile coho salmon and rainbow trout time from exposure to death is 12.5 and 14 days respectively (9)	6.7-23.3 Juvenile rainbow trout have little or no ability to overcome infection, and mortality varied from 75-86% (9)	23.3 Juvenile spring Chinook mortality was 92%, and time from exposure to death was 2.3 days (13)		
22					22.2 Mortality is 100% in juvenile sockeye exposed to *C. columnaris* (10)		
21	18.3-21.2 Serious outbreaks of Ich occur (4)				>21.1 Temperatures at this level are associated with a 28-74% infection rate in Chinook (11)		
20		20 Lifecycle takes 1 week (6)	20.5 Mortality is 84% in juvenile coho exposed to *C. shasta* (9).		20.5 Mortality in juvenile steelhead and coho from Columnaris was 100%, and 70% in juvenile spring Chinook (13)	>20.88 MWMTs over this value can result in severe infections and catastrophic outbreaks (1)	18-20 Temperature range which is associated with a high risk of disease in rearing juveniles and migrating adults (2)
					20.5 In juvenile steelhead and coho time from exposure to death was 1.6-1.7 days (13)		
					20 Average water temperature at which low virulence strains show signs of outbreak (3, 12)		
19							
18							
17					17.8 Mortality rates were 52, 92, and 99% for juvenile spring Chinook, steelhead and coho respectively (13)	17.38 MWMT should not be exceeded to avoid serious rates of infection and mortality (1)	14-17 Temperature range which is associated with an elevated risk of disease in rearing juveniles and migrating adults (2)
16					16.1 Mortality is 30% in juvenile sockeye exposed to *C. columnaris* (10)		
15	>15.6 Associated with outbreaks in salmonid fingerlings, especially Chinook (3)		15 Mortality is 22% in juvenile coho exposed to *C. shasta* (9).		15.6 Average water temperature at which low virulence strains show signs of outbreak (3)		
	15.5 Lifecycle of Ich takes 2 weeks (5)				15 Mortality was 31, 56, and 51% for juvenile spring Chinook, steelhead, and coho respectively (13)		
14						14.38 MWMT will virtually prevent all warm water disease (1)	

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Table 12 (continued):Effects of Temperature in Considering Disease and Salmonids

°C	Ich	Ceratomyxosis		Columnaris	Disease (general)
13			6.7-23.3 Juvenile rainbow trout have little or no ability to overcome infection, and mortality varied from 75-86% (9)		12-13 Temperature range which minimizes the risk of disease in rearing juveniles and migrating adults (2)
12				12.8 After 7 days of infection mortality is 60-100% (majority of tests 100%) (12)	
				12.2 Mortality was 4-20% in juvenile spring Chinook, steelhead, and coho respectively. Time from exposure to death ranged from 7.6-12.2 days (13).	
11		10-11 *C. shasta* appears to be come infective (4)			
10	10 Lifecycle takes more than 5 weeks (5)	<10 Steelhead show evidence of *C. shasta* in ~38 days (8)			
9		9.4 Juvenile coho time from exposure to death is 146 days, mortality is 2% (9)		3.9-9.4 No mortality in spring Chinook, steelhead, or coho from Columnaris (13)	
8					
7	7 Lifecycle takes 20 days (6)				
	<7 Lifecycle takes more than 5 weeks (7)				
6			6.7 Juvenile rainbow trout time from exposure to death is 155 days (9)		
			3.9-6.7 No mortality in Juvenile coho exposed to *C. shasta* (9)		
5					
4					
3					

Sources: (1) WDOE 2002, (2) USEPA 2003, (3) Bell 1986, (4) CDWR 1988, (5) Piper et al 1982, (6) Nigrelli et al. 1976, as cited by Dickerson et al. 1995, (7) Durborow et al. 1998, (8) Leitritz and Lewis, 1976, (9) Udey et al. 1975, (10) Ordal and Rucker 1944, as cited by Pacha et al. 1970, (11) USEPA 1999a, (12) Pacha et al. 1970, (13) Holt et al. 1975

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rates of infection and mortality the MWMT should not exceed 17.38°C (midpoint of 15.58-19.18 range), and that severe infections and catastrophic outbreaks become a serious concern when the MWMTs exceed 20.88°C (midpoint of 18.58-23.18 range).

In a summary of temperature considerations, USEPA (2003) states that disease risks for juvenile rearing and adult migration are minimized at temperatures from 12-13°C, elevated from 14-17°C, and high at temperatures from 18-20°C.

Acknowledging that there are many diseases that affect salmonids, the following discussion will focus on three which are common in the Klamath Basin: Ichthyophthiriasis (Ich), Ceratomyxosis, and Columnaris. *Ichthyophthirius multifiliis* is a protozoan parasite that causes the disease known as Ichthyophthiriasis (Ich). The disease ceratomyxosis is caused by a parasite, *Ceratomyxa shasta* (*C. shasta*). Columnaris disease is a bacterial infection caused by *Flavobacterium columnare* (synomyms*: Bacillus columnaris, Chondrococcus columnaris, Cytophaga columnaris, Flexibacter columnaris*).

1.7.1 Ichthyophthiriasis (Ich)

Nigrelli et al. (1976, as cited by Dickerson et al. 1995) proposed that there are physiological races of Ich, which are related to the temperature tolerance of the host fishes. Thus, there are races of Ich that infect cold-water (7.2-10.6°C) fishes such as salmon, and others that infect warm-water (12.8-16.1°C) tropical fishes. Bell (1986) discusses Ich and states that at water temperatures above 15.6°C, this disease often breaks out in salmon fingerlings, especially Chinook. CDWR (1988) states that serious outbreaks of Ich occur at temperatures from 18.3-21.2°C.

Numerous studies and reviews have been conducted on the optimal temperature for Ich. Piper et al. (1982, p.316.) wrote that optimal temperatures range from 21-23.9°C. CDWR (1988) stated the optimum temperature for Ich is in the range of 25 to 26.7°C, while Bell (1986) states optimum temperatures are noted from 21.2-26.7°C.

Temperature is an important factor in the persistence of Ich infections in salmonids. The growth period varies from 1 week at 20 °C to 20 days at 7 °C (Nigrelli et al. 1976, as cited by Dickerson et al. 1995). Piper et al. (1982, p.316) state that at optimal temperatures of 21-23.9°C, the life cycle may take as few as 3-4 days. The cycle requires 2 weeks at 15.5°C, and more than 5 weeks at 10°C (Piper et al. 1982, p.316). Durborow et al. (1998) note that to complete its lifecycle, Ich requires from less than 4 days at temperatures higher than 24°C, to more than 5 weeks at temperatures lower than 7°C. Although studies report varying lengths of time for Ich to complete its lifecycle at similar temperatures, it is clear that the speed at which Ich develops increases as temperatures increase.

1.7.2 Ceratomyxosis

In reviewing the literature on Ceratomyxosis (a disease caused by the parasite, C. shasta), it is clear that as water temperatures increase, the intensity of the disease increases, and

the incubation period decreases (CDWR 1988, Letritz and Lewis, Udey et al. 1975). At water temperatures greater than 10°C, steelhead will show evidence of Ceratomyxosis in approximately 38 days (Leitritz and Lewis 1976, p.154). In a study of juvenile coho salmon by Udey et al. (1975), time from exposure to death was more than 90% temperature dependent, and increased from 12.5 days at 23.3°C, to 146 days at 9.4°C. These results show the accelerating effect of higher temperatures on the progress of the disease. The time from exposure to death of juvenile rainbow trout was nearly 97% temperature dependent, increasing from 14 days at 23.3°C to 155 days at 6.7°C (Udey et al. 1975).

C. shasta appears to become infective at temperatures around 10-11°C (CDWR 1988). According to Leitritz and Lewis (1976, p.154), steelhead from the Klamath River are quite susceptible to *C. shasta* infections and suffer severe losses when exposed.

Udey et al. (1975) conducted a study to determine the relation of water temperature to Ceratomyxosis in juvenile rainbow trout and coho salmon. Rainbow trout from the Roaring River Hatchery, and coho from Fall Creek Salmon Hatchery (both in Oregon) were used in this experiment. Groups of 25 fish exposed to *C. shasta* were transferred to 12.2°C water, and then were tempered to one of eight experimental temperatures from 3.9 to 23.3°C (2.8°C increments).

In the juvenile coho salmon experiment, Udey et al. (1975) found that percent mortality increased progressively from 2% at 9.4°C to 22% at 15.0°C and 84% at 20.5°C. No deaths occurred in coho salmon maintained at 3.9 and 6.7°C, indicating that ceratomyxosis in coho can be suppressed by water temperatures of 6.7°C or below (Udey et al. 1975).

Tests conducted by Udey et al. (1975) on rainbow trout juveniles indicate that once infection is initiated, juvenile rainbow trout have little or no ability to overcome *C. shasta* infections at water temperatures between 6.7 and 23.3°C. Fatal infections varied from 75-86% at temperatures ranging from 6.7 to 15.0°C (Udey et al. 1975). Mortality in trout held at 20.5 and 23.3°C were lower (72% and 52% respectively) due to losses from *Flexibacter columnaris,* which occurred well before the onset of deaths caused by *C. shasta,* in spite of efforts to control it with terramycin (Udey et al. 1975). The results from Udey et al. (1975) also reflected no deaths occurred in juvenile trout held at 3.9°C.

1.7.3 Columnaris

The importance of temperature on infections of Columnaris has been demonstrated in numerous laboratory studies. Ordal and Rucker (1944, as cited by Pacha et al. 1970) exposed juvenile sockeye salmon to *C. columnaris* and studied the effect of temperature on the disease. In these studies, the overall mortality ranged from 30% in fish held at 16.1°C to 100% in those held at 22.2°C (Ordal and Rucker 1944, as cited by Pacha et al. 1970). USEPA (1999a) cites studies that conducted surveys of Columnaris infection frequency on Chinook in the Snake River in July and early August of 1955-1957, which revealed 28-75% of fish infected when water temperature was >21.1°C.

Low virulence strains of Columnaris show signs of outbreak when average water temperatures are over 20°C (Bell 1986, Pacha et al. 1970). Bell (1986) states that outbreaks of high virulence strains occur when average water temperatures reach 15.6°C, and Pacha et al. (1970) found mortalities of 60-100% (majority of tests 100%) occur at temperatures of 12.8°C after 7 days of infection. With regard to strains of higher virulence, while these strains are capable of beginning infection and producing disease at water temperatures as low as 12.8°C, the disease process becomes progressively slower as the water temperature is lowered (Pacha et al. 1970).

Holt et al. (1975) performed a study on the relation of water temperature to Columnaris in juvenile steelhead trout and juvenile coho and spring Chinook salmon. Tests were performed on groups of 25-35 fish at eight temperatures ranging from 3.9°C to 23.3°C (2.8°C increments). At 20.5°C mortality was 100% in juvenile steelhead trout and coho salmon, 70% in juvenile spring Chinook salmon, and at temperatures 23.3°C juvenile spring Chinook mortality was 92% (Holt et al. 1975). Mortality rates were 52, 92, and 99% at 17.8°C for juvenile spring Chinook, steelhead trout, and coho salmon respectively, and mortality dropped to 31, 56, and 51% at 15.0°C (Holt et al. 1975). At 12.2°C mortality varied from 4 to 20% among juveniles of the three species, and at temperatures of 9.4°C and below, no deaths due to the experimental infection with *F. columnaris* occurred (Holt et al. 1975). Holt et al. (1975) state that these results indicate that under the conditions of these experiments Columnaris disease was completely suppressed by water temperatures of 9.4°C or below.

In general, data from laboratory studies indicates that as water temperatures increase, the time to death decreases (Pacha et al. 1970). With juvenile steelhead trout and juvenile coho and spring Chinook salmon as the temperature increased above 12.2°C, the disease process was progressively accelerated, resulting in a minimum time to death at 20.5 or 23.3°C and a maximum at 12.2°C (Holt et al. 1975). In these juvenile salmonids Holt et al. (1975) found the mean time to death decreased from 7.6-12.2 days at 12.2°C to 1.6-1.7 days at 20.5°C for juvenile coho and steelhead, and 2.3 days at 23.3°C for juvenile spring Chinook (Holt et al. 1975).

1.8 TMDL Temperature Thresholds

Currently there are no numeric temperature standards in the *Water Quality Control Plan for the North Coast Region* (Basin Plan). Thus, information from this literature review will be utilized by Regional Water Board staff to selected chronic and acute temperature thresholds for evaluation of stream temperatures in TMDLs. Chronic temperature thresholds (MWMTs) were selected from the USEPA document *EPA Region 10 Guidance For Pacific Northwest State and Tribal Temperature Water Quality Standards* (2003), and are presented in Table 13. The Region 10 guidance is the product of a three-year interagency effort, and has been reviewed by both independent science review panels and the public. Acute lethal temperature thresholds were selected based upon best professional judgment of the literature, and are presented in Table 14.

Table 13: Life Stage Temperature Thresholds

Life Stage	MWMT (°C)
Adult Migration	20
Adult Migration plus Non-Core[1] Juvenile Rearing	18
Core[2] Juvenile Rearing	16
Spawning, Egg Incubation, and Fry Emergence	13

[1] Non-Core is defined as moderate to low density salmon and trout rearing usually occurring in the mid or lower part of the basin (moderate and low not defined).
[2] Core is defines as areas of high density rearing (high is not specifically defined).
Source: USEPA 2003

Table 14: Lethal Temperature Thresholds

Lethal Threshold[1] (°C)			
Life Stage	**Steelhead**	**Chinook**	**Coho**
Adult Migration and Holding	24	25	25
Juvenile Growth and Rearing	24	25	25
Spawning, Egg Incubation, and Fry Emergence	20	20	20

[1] The lethal thresholds selected in this table are generally for chronic exposure (greater than seven days). Although salmonids may survive brief periods at these temperatures, they are good benchmarks from the literature for lethal conditions.

In some cases it may be necessary to calculate MWATs for a given waterbody, and compare these to MWAT thresholds. USEPA (2003) states that for many rivers in the Pacific Northwest the MWMT is about 3°C higher than the MWAT (USEPA 2003, as cited by Dunham et al. 2001and Chapman 2002). Rather than list MWAT thresholds in this document using the 3°C difference suggested above, the Regional Water Board will consider stream temperatures within each individual TMDL waterbody. Thus the Regional Water Board will calculate both MWMTs and MWATs for the waterbody, and characterize the actual difference between these temperature metrics for the watershed using an approach similar to that used in Sullivan et al. (2000). Once this relationship is understood, MWAT thresholds for each life stage can be identified and compared to the watershed MWATs.

The freshwater temperature thresholds presented in this section are applicable during the season or time of year when the life stage of each species is present. Periodicity information is not discussed in this document and will be presented in each individual TMDL staff report. Where life history, timing, and/or species needs overlap, the lowest of each temperature metric applies.

CHAPTER 2. DISSOLVED OXYGEN and TOTAL DISSOLVED GAS

2.1 Introduction

Adequate concentrations of dissolved oxygen in fresh water streams are critical for the survival of salmonids. Fish have evolved very efficient physiological mechanisms for obtaining and using oxygen in the water to oxygenate the blood and meet their metabolic demands (WDOE 2002). Reduced levels of dissolved oxygen can impact growth and development of different life stages of salmon, including eggs, alevins, and fry, as well as the swimming, feeding and reproductive ability of juveniles and adults. Such impacts can affect fitness and survival by altering embryo incubation periods, decreasing the size of fry, increasing the likelihood of predation, and decreasing feeding activity. Under extreme conditions, low dissolved oxygen concentrations can be lethal to salmonids. High levels of total dissolved gas concentrations (TDG), including dissolved oxygen, can result in gas bubble disease and death for salmonids.

Literature reviewed for this analysis included EPA guidance, other states' standards, reports that compiled and summarized existing scientific information, and numerous laboratory studies. When possible, species-specific requirements were summarized for the following life stages: migrating adults, incubation and emergence, and freshwater rearing and growth. The following information applies to salmonids in general, with specific references to coho, Chinook, steelhead, and other species of salmonids as appropriate.

2.2 Effects of Low Dissolved Oxygen Concentrations on Salmonids

2.2.1 Adult Migration

Reduced concentrations of dissolved oxygen can negatively affect the swimming performance of migrating salmonids (Bjornn and Reiser 1991). The upstream migration by adult salmonids is typically a stressful endeavor. Sustained swimming over long distances requires high expenditures of energy and therefore requires adequate levels of dissolved oxygen. Migrating adult Chinook salmon in the San Joaquin River exhibited an avoidance response when dissolved oxygen was below 4.2 mg/L, and most Chinook waited to migrate until dissolved oxygen levels were at 5 mg/L or higher (Hallock et al. 1970).

2.2.2 Incubation/Emergence

Low levels of dissolved oxygen can be directly lethal to salmonids, and can also have sublethal effects such as changing the rate of embryological development, the time to hatching, and size of emerging fry (Spence et al. 1996). The embryonic and larval stages of salmonid development are especially susceptible to low dissolved oxygen levels as their ability to extract oxygen is not fully developed and their relative immobility inhibits their ability to migrate to more favorable conditions. The dissolved oxygen requirements

for successful incubation of embryos and emergence of fry is tied to intragravel dissolved oxygen levels. Intragravel dissolved oxygen is typically a function of many chemical, physical, and hydrological variables, including: the dissolved oxygen concentration of the overlying stream water, water temperature, substrate size and porosity, biochemical oxygen demand of the intragravel water, sediment oxygen demand, the gradient and velocity of the stream, channel configuration, and depth of water. As a result the dissolved oxygen concentration within the gravels can be depleted causing problems for salmonid embryos and larvae, even when overlying surface water oxygen levels are suitable (USEPA 1986a).

Studies note that water column dissolved oxygen concentrations are typically estimated to be reduced by 1-3 mg/L as water is transmitted to redds containing developing eggs and larvae (WDOE 2002). USEPA (1986a) concluded that dissolved oxygen levels within the gravels should be considered to be at least 3 mg/L lower than concentrations in the overlying water. ODEQ (1995b) expect the loss of an average of 3 mg/L dissolved oxygen from surface water to the gravels.

2.2.3 Incubation Mortality

Phillips and Campbell (1961, as cited by Bjornn and Reiser 1991) concluded that intragravel dissolved oxygen must average 8 mg/L for embryos and alevins to survive well. After reviewing numerous studies Davis (1975) states that a dissolved oxygen concentration of 9.75 mg/L is fully protective of larvae and mature eggs, while at 8 mg/L the average member of the incubating population will exhibit symptoms of oxygen distress, and at 6.5 mg/L a large portion of the incubating eggs may be affected. Bjornn and Reiser (1991) reviewed numerous references and recommend that dissolved oxygen should drop no lower than 5 mg/L, and should be at or near saturation for successful incubation.

In a review of several laboratory studies, ODEQ (1995b) concluded that at near optimum (10°C) constant temperatures acute mortality to salmonid embryos occurs at relatively low concentrations of dissolved oxygen, near or below 3 mg/L. Field studies reviewed by ODEQ (1995b) demonstrate that embryo survival is low when the dissolved oxygen content in the gravels drops near or below 5 mg/L, and survival is greater at 8 mg/L.

Silver et al. (1963) performed a study with Chinook salmon and steelhead trout, rearing eggs at various constant dissolved oxygen concentrations and water velocities. They found that steelhead embryos held at 9.5°C and Chinook salmon embryos held at 11°C experienced complete mortality at dissolved oxygen concentrations of 1.6 mg/L. Survival of a large percentage of embryos reared at oxygen levels as low as 2.5 mg/L appeared to be possible by reduction of respiration rates and consequent reduction of growth and development rates.

In a field study Cobel (1961) found that the survival of steelhead embryos was correlated to intragravel dissolved oxygen in the redds, with higher survival at higher levels of

dissolved oxygen. At 9.25 mg/L survival was 62%, but survival was only 16% at 2.6 mg/L. A laboratory study by Eddy (1971) found that Chinook salmon survival at 10.4 mg/L (13.5 °C) was approximately 67%, however at dissolved oxygen levels of 7.3 mg/L (13.5 °C) survival dropped to 49-57.6%. At temperatures more suitable for Chinook incubation (10.5 °C) Eddy (1971) found the percent survival remained high (over 90%) at dissolved oxygen levels from 11 mg/L to 3.5 mg/L; however, as dissolved oxygen levels decreased, the number of days to hatching increased and the mean dry weight of the fry decreased substantially. WDOE (2002) also points out that the studies above did not consider the act of emerging through the redds, and the metabolic requirements to emerge would be expected to be substantial. Therefore, it is likely that higher oxygen levels may be needed to fully protect hatching and emergence, than to just support hatching alone.

2.2.4 Incubation Growth

Embryos can survive when dissolved oxygen is below saturation (and above a critical level), but development typically deviates from normal (Bjornn and Reiser 1991). Embryos were found to be smaller than normal, and hatching either delayed or premature, when dissolved oxygen was below saturation throughout development (Doudoroff and Warren 1965, as cited by Bjornn and Reiser 1991).

Garside (1966) found the number of days it took for rainbow trout to go from fertilization to hatching increased as dissolved oxygen concentrations and water temperature decreased. In this study, rainbow trout were incubated at temperatures between 2.5 - 17.5°C and dissolved oxygen levels from 2.5 - 11.3 mg/L. At 10°C and 7.5°C the total time for incubation was delayed 6 and 9 days respectively at dissolved oxygen levels of 2.5 mg/L versus embryos incubated at approximately 10.5 mg/L.

Silver et al. (1963) found that hatching of steelhead trout held at 9.5°C was delayed 5 to 8 days at dissolved oxygen concentrations averaging 2.6 mg/L versus embryos reared at 11.2 mg/L. A smaller delay of hatching was observed at oxygen levels of 4.2 and 5.7 mg/L, although none was apparent at 7.9 mg/L. For Chinook salmon held at 11°C, Silver et al. observed that embryos reared at oxygen levels lower than 11 mg/L experienced a delay in hatching, with the most significant delay in those reared at dissolved oxygen levels of 2.5 mg/L (6 to 9 days). The size of both Chinook and steelhead embryos increased with increases in dissolved oxygen up to 11.2 mg/L. External examination of embryos revealed abnormal structural development in Chinook salmon tested at dissolved oxygen concentrations of 1.6 mg/L, and abnormalities in steelhead trout at concentrations of 1.6 and 2.6 mg/L. The survival of Chinook salmon after hatching was only depressed at the 2.5 mg/L level, the lowest level at which hatching occurred, with lower mortalities occurring at higher velocities. Post hatching survival of steelhead trout could not be determined due to numerous confounding factors.

Shumway et al. (1964) conducted a laboratory study to determine the influence of oxygen concentration and water movement on the growth of steelhead trout and coho salmon

embryos. The experiments were conducted at a temperature of 10°C and oxygen levels generally ranging from 2.5 - 11.5 mg/L and flows from 3 to 750 cm/hour. It was concluded that the median time to hatching decreased and size of fry increased as dissolved oxygen levels increased. For example, steelhead trout embryos reared at 2.9 mg/L hatched in approximately 41 days and had a wet weight of 17 mg, while embryos reared at 11.9 mg/L hatched in 36 days and weighed 32.3 mg. The authors found that a reduction of either the oxygen concentration or the water velocity will reduce the size of fry and increase the incubation period, although the affect of various water velocities tested was less than the effect of the different dissolved oxygen concentrations tested.

WDOE (2002) reviewed various references and found that at favorable incubation temperatures a mean oxygen concentration of 10.5 mg/L will result in a 2% reduction in growth. At other oxygen concentrations, growth is reduced as follows: 8% reduction at oxygen levels of 9 mg/L, 10% reduction at 7 mg/L, and a 25% reduction at 6 mg/L.

2.2.5 Incubation Avoidance/Preference

Alevin showed a strong preference for oxygen concentrations of 8 - 10 mg/L and moved through the gravel medium to these concentrations, avoiding concentrations from 4 - 6 mg/L (WDOE 2002).

2.2.6 Emergence Mortality

"The hatching time, size, and growth rate of developing embryos is proportional to the dissolved oxygen concentrations up to 8 mg/L or greater. The ability of fry to survive their natural environment may be related to the size of fry at hatch (ODEQ 1995b)." McMahon (1983) recommends dissolved oxygen levels be ≥ 8 mg/L for high survival and emergence of fry. In a review of controlled field and lab studies on emergence, WDOE (2002) states that average intragravel oxygen concentrations of 6 - 6.5 mg/L and lower can cause stress and mortality in developing embryos and alevin. It is also noted that field studies on emergence consistently cite intragravel oxygen concentrations of 8 mg/L or greater as being associated with or necessary for superior health and survival, oxygen concentrations below 6 - 7 mg/L result in a 50% reduction in survival through emergence, and oxygen concentrations below 5 mg/L result in negligible survival. According to various laboratory studies, the threshold for complete mortality of emerging salmonids is noted to occur between 2 - 2.5 mg/L (WDOE 2002).

After reviewing numerous literature sources, the USEPA (1986a) concluded that the embryonic and larval stages of salmonid development will experience no impairment when water column dissolved oxygen concentrations are 11 mg/L. This translates into an intragravel dissolved oxygen concentration of 8 mg/L (USEPA assumes a 3 mg/L loss between the surface water and gravels). Table 15 from the USEPA (1986a) lists the water column and intragravel dissolved oxygen concentrations associated with various health effects. These health affects range from no production impairment to acute mortality.

Table 15: Dissolved oxygen concentrations and their effects salmonid embryo and larval stages

Level of Effect	Water Column DO (mg/L)	Intragravel DO (mg/L)
No Production Impairment	11	8*
Slight Production Impairment	9	6*
Moderate Production Impairment	8	5*
Severe Production Impairment	7	4*
Limit to Avoid Acute Mortality	6	3*

* A 3 mg/L loss is assumed between the water column dissolved oxygen levels and those intragravel.
Source: USEPA 1986a

2.2.7 Freshwater Rearing and Growth

2.2.7.1 Swimming and Activity

Salmonids are strong active swimmers requiring highly oxygenated waters (Spence 1996), and this is true during the rearing period when the fish are feeding, growing, and avoiding predation. Salmonids may be able to survive when dissolved oxygen concentrations are low (<5 mg/L), but growth, food conversion efficiency, and swimming performance will be adversely affected (Bjornn and Reiser 1991). Davis (1975) reviewed numerous studies and reported no impairment to rearing salmonids if dissolved oxygen concentrations averaged 9 mg/L, while at oxygen levels of 6.5 mg/L "the average member of the community will exhibit symptoms of oxygen distress", and at 4 mg/L a large portion of salmonids may be affected. Dahlberg et al. (1968) state that at temperatures near 20°C any considerable decrease in the oxygen concentration below 9 mg/L (the air saturation level) resulted in some reduction of the final swimming speed. They found that between dissolved oxygen concentrations of 7 to 2 mg/L the swimming speed of coho declined markedly with the decrease in dissolved oxygen concentration.

In a laboratory study, Davis et al. (1963) reported that the maximum sustainable swimming speeds of wild juvenile coho salmon were reduced when dissolved oxygen dropped below saturation at water temperatures of 10, 15, and 20°C. Air-saturation values for these dissolved oxygen concentrations were cited as 11.3, 10.2, and 9.2 mg/L respectively. They found that the maximum sustained swimming speeds (based on first and second swimming failures at all temperatures) were reduced by 3.2 - 6.4%, 5.9 - 10.1%, 9.9 - 13.9%, 16.7 - 21.2%, and 26.6 - 33.8% at dissolved oxygen concentrations of 7, 6, 5, 4, and 3 mg/L respectively. The authors also conducted tests on juvenile Chinook salmon and found that the percent reductions from maximum swimming speed at temperatures ranging from 11 to 15°C were greater than those for juvenile coho. At the dissolved oxygen concentrations listed above swimming speeds were decreased by 10%, 14%, 20%, 27%, and 38% respectively.

WDOE (2002) reviewed various data and concluded that swimming fitness of salmonids is maximized when the daily minimum dissolved oxygen levels are above 8 - 9 mg/L. Jones et al. (1971, as cited by USEPA 1986a) found the swimming speed of rainbow trout was decreased 30% from maximum at dissolved oxygen concentrations of 5.1 mg/L

and 14°C. At oxygen levels of 3.8 mg/L and a temperature of 22°C, they found a 43% reduction in the maximum swimming speed.

2.2.7.2 Growth

In a review of constant oxygen exposure studies WDOE (2002) concluded salmonid growth rates decreased less than 10% at dissolved oxygen concentrations of 8 mg/L or more, less than 20% at 7 mg/L, and generally less than 22% at 5 - 6 mg/L. Herrmann (1958) found that the mean percentage of weight gain in juvenile coho held at constant dissolved oxygen concentrations was 7.2% around 2 mg/L, 33.6% at 3 mg/L, 55.8% near 4 mg/L, and 67.9% at or near 5 mg/L. In a laboratory study Fischer (1963) found that the growth rates of juvenile coho exposed to constant oxygen concentrations ranging from 2.5 to 35.5 mg/L (fed to satiation, temperature at approximately 18 °C) dramatically decreased with decreases in the oxygen concentration below 9.5 mg/L (air saturation level). WDOE (2002) concludes that a monthly or weekly average concentration of 9 mg/L, and a monthly average of the daily minimum concentrations should be at or above 8 - 8.5 mg/L to have a negligible effect (5% or less) on growth and support healthy growth rates.

Food conversion efficiency is related to dissolved oxygen levels and the process becomes less efficient when oxygen concentrations are below 4 - 4.5 mg/L (ODEQ 1995b). Bjornn and Reiser (1991) state that growth, food conversion efficiency, and swimming performance are adversely affected when dissolved oxygen concentrations are <5 mg/L. The USEPA (1986a) reviewed growth data from a study conducted by Warren et al. (1973) where tests were conducted at various temperatures to determine the growth of coho and Chinook. USEPA cites that, with the exception of tests conducted at 22 °C, the results supported the idea that the effects of low dissolved oxygen become more severe at higher temperatures.

Brett and Blackburn (1981) performed a laboratory study to determine the growth rate and food conversion efficiency of young coho and sockeye salmon fed full rations. Tests were performed at dissolved oxygen concentrations ranging from 2 to 15 mg/L at a constant temperature of 15°C, the approximate optimum temperature for growth of Pacific Salmon. Both species showed a strong dependence of growth on the environmental oxygen concentrations when levels were below 5 mg/L. For coho, zero growth was observed at dissolved oxygen concentrations of 2.3 mg/L. The mean value for maximum coho growth occurred at 4 mg/L, and at dissolved oxygen concentrations above this level growth did not appear to be dependant on the dissolved oxygen. Sockeye displayed zero growth at oxygen levels of 2.6 mg/L, and reached the zone of independence (growth not dependant on dissolved oxygen levels) at 4.2 mg/L. Brett and Blackburn (1981) conclude that the critical inflection from oxygen dependence to independence occurs at 4 - 4.2 mg/L for coho and sockeye.

Herrmann et al. (1962) studied the influence of various oxygen concentrations on the growth of age 0 coho salmon held at 20 °C. Coho were held in containers at a constant mean dissolved oxygen level ranging from 2.1 - 9.9 mg/L and were fed full rations. The

authors concluded that oxygen concentrations below 5 mg/L resulted in a sharp decrease in growth and food consumption. A reduction in the mean oxygen levels from 8.3 mg/L to 6 and 5 mg/L resulted in slight decreases in food consumption and growth. Weight gain in grams per gram of food consumed was slightly depressed at dissolved oxygen concentrations near 4 mg/L, and were markedly reduced at lower concentrations. At oxygen levels of 2.1 and 2.3 mg/L, many fish died and the surviving fish lost weight and consumed very little food.

USEPA (1986a) calculated the median percent reduction in growth rate of Chinook and coho salmon fed full rations at various dissolved oxygen concentrations. They calculated no reduction in growth at dissolved oxygen concentrations of 8 and 9 mg/L, and a 1% reduction in growth at 7 mg/L for both species. At 6 mg/L Chinook and coho growth were reduced by 7% and 4% respectively. Dissolved oxygen levels of 4 mg/L result in a 29% reduction in growth for Chinook salmon and 21% reduction in growth for coho. At 3 mg/L there was a 47% decrease in Chinook growth and a 37% reduction in coho growth. USEPA (1986a) states that due to the variability inherent in growth studies the reductions in growth rates seen above 6 mg/L are not usually statistically significant, while reductions in growth at dissolved oxygen levels below 4 mg/L are considered severe.

2.2.7.3 Avoidance and Preference

Salmonids have been reported to actively avoid areas with low dissolved oxygen concentrations, which is likely a useful protective mechanism that enhances survival (Davis 1975). Field and laboratory studies have found that avoidance reactions in juvenile salmonids consistently occur at concentrations of 5 mg/L and lower, and there is some indication that avoidance is triggered at concentrations as high as 6 mg/L. Therefore these dissolved oxygen levels should be considered a potential barrier to the movement and habitat selection of salmonids (WDOE 2002).

Spoor (1990) performed a laboratory study on the distribution of fingerling brook trout in dissolved oxygen concentration gradients. Sixteen gradients between 1 and 8.9 mg/L were used for the study to determine what level of dissolved oxygen is preferred by the brook trout. It was found that in the absence of a gradient with dissolved oxygen concentrations at 6 mg/L or more throughout the system, the fish moved freely without showing preference or avoidance. Movement from low to higher oxygen concentrations were noted throughout the study. Fish moved away from water with dissolved oxygen concentrations from 1 - 1.9 mg/L within one hour, moved away from water with dissolved oxygen concentrations of 2 - 2.9 mg/L within 1 - 2 hours, and moved away more slowly from concentrations of 3 - 3.9 mg/L. From his study, Spoor (1990) concluded that brook trout will avoid oxygen concentrations below 4 mg/L, and preferred oxygen levels of 5 mg/L or higher.

Whitmore et al. (1960) performed studies with juvenile coho and Chinook salmon to determine their avoidance reaction to dissolved oxygen concentration of 1.5, 3 , 4.5, and 6 mg/L at variable river water temperatures. Juvenile Chinook salmon showed marked avoidance of oxygen concentrations near 1.5, 3, and 4.5 mg/L in the summer at mean

temperatures ranging from 20.7 - 22.8°C, but no avoidance to levels near 6 mg/L at a mean temperature of 18.4°C. Chinook did not show as strong an avoidance to these oxygen levels in the fall when water temperatures were lower, ranging from 11.8 - 13.2°C. Chinook showed little avoidance of dissolved oxygen concentrations near 4.5 mg/L during the fall, and no avoidance to concentrations near 6 mg/L. In all cases avoidance became progressively larger with reductions in the oxygen concentration below 6 mg/L. Seasonal differences of avoidance are most likely due to differences in water temperature. At temperatures ranging from 18.4 - 19°C juvenile coho salmon showed some avoidance to all of the above oxygen concentrations, including 6 mg/L. Their behavior was more erratic than that of Chinook, and their avoidance of concentrations near 4.5 mg/L and lower was not as pronounced at corresponding temperatures. The juvenile coho often started upon entering water with low dissolved oxygen and then darted around until they found their way out of the experimental channel.

USEPA (1986a) performed a literature review and cites the effects of various dissolved oxygen concentrations on salmonid life stages other than embryonic and larval (Table 16). These effects range from no impairment at 8 mg/L to acute mortality at dissolved oxygen levels below 3 mg/L.

Table 16: Dissolved oxygen concentrations and their effects on salmonid life stages other than embryonic and larval

Level of Effect	Water Column DO (mg/L)
No Production Impairment	8
Slight Production Impairment	6
Moderate Production Impairment	5
Severe Production Impairment	4
Limit to Avoid Acute Mortality	3

Source: USEPA 1986a

2.2.8 Lethality

Salmonid mortality begins to occur when dissolved oxygen concentrations are below 3 mg/L for periods longer than 3.5 days (USEPA 1986a). A summary of various field study results by WDOE (2002) reports that significant mortality occurs in natural waters when dissolved oxygen concentrations fluctuate the range of 2.5 - 3 mg/L. Long-term (20 - 30 days) constant exposure to mean dissolved oxygen concentrations below 3 - 3.3 mg/L is likely to result in 50% mortality of juvenile salmonids (WDOE 2002). According to a short-term (1 - 4 hours) exposure study by Burdick et al. (1954, as cited by WDOE, 2002), in warm water (20 - 21°C) salmonids may require daily minimum oxygen levels to remain above 2.6 mg/L to avoid significant (50%) mortality. From these and other types of studies, WDOE (2002) concluded that juvenile salmonid mortality can be avoided if daily minimum dissolved oxygen concentration remain above 3.9 mg/L, and the monthly or weekly average of minimum concentrations remains above 4.6 mg/L.

2.3 Effects of High Total Dissolved Gas Concentrations on Salmonids

High levels of total dissolved gas (TDG), including dissolved oxygen, can be harmful to

salmonids and other fish and result in "gas bubble disease". This occurs when dissolved gases in their circulatory system come out of solution and form bubbles which block the flow of blood through the capillary vessels (USEPA 1986b). There are several ways TDG supersaturation can occur, including excessive algal photosynthesis which can create supersaturated dissolved oxygen conditions (USEPA 1986b). Thus, to protect salmonids and other freshwater fish the USEPA has set criteria for TDG stating that levels should not exceed 110% of the saturation value.

Numerous studies have been conducted to determine the mortality rate of salmonids exposed to various levels of TDG. Mesa et al. (2000) conducted laboratory experiments on juvenile Chinook and steelhead, exposing them to different levels of TDG and found no fish died when held at 110% TDG for up to 22 days. When fish were exposed to 120% TDG, 20% of juvenile Chinook died within 40 to 120 hours while 20% of juvenile steelhead died within 20 to 35 hours. At TDG levels of 130% Chinook mortality reached 20% after 3 to 6 hours and steelhead mortality was 20% after 5 to 7 hours. Gale et al. (2001) held adult female spring Chinook at mean TDG levels ranging from 114.1% to 125.5% and found the time to first mortality ranged from 10 to 68 hours.

USEPA (1986b) discusses various studies on the effects of TDG on salmonids. The following studies are all cited from the USEPA 1986 water quality criteria document. Bouck et al. (1975) found TDG levels of 115% and above to be acutely lethal to most species of salmonids, and levels of 120% TDG are rapidly lethal to all salmonids. Conclusions drawn from Ebel et al. (1975) and Rulfison and Abel (1971) include the following:

- Adult and juvenile salmonids confined to shallow water (1 m) with TDG levels above 115% experience substantial levels of mortality.
- Juvenile salmonids exposed sublethal levels TDG supersaturation are able to recover when returned to normally saturated water, while adults do not recover and generally die.

DRAFT

CHAPTER 3. AMMONIA

3.1 Introduction

According to the USEPA (1986b, p.17), acute concentrations of ammonia can cause loss of equilibrium, hyperexcitability, increased breathing, cardiac output and oxygen uptake, and, in extreme cases, convulsions, coma, and death of fish. Lower concentrations of ammonia can result in reduced hatching success, reduced growth and morphological development, and pathologic changes in tissues of gills, livers, and kidneys.

The information in the following sections was extracted from the USEPA document titled: *1999 Update of Ambient Water Quality Criteria for Ammonia*. The information presented applies to salmonids in general.

3.2 Ammonia Speciation

Ammonia in water exists primarily in two forms, un-ionized ammonia (NH_3) and ammonium ion (NH_4^+) (USEPA 1999b, p.2). The fraction of each of these two forms, or ammonia speciation, varies markedly with temperature and pH (USEPA 1999b, p.2). The pH-dependence of the relative amounts of un-ionized ammonia and ammonium ion at 25°C are presented in Figure 1 below (USEPA 1999b, p.2). Ammonia speciation also depends on ionic strength, although in freshwater this effect is much smaller than the effects of temperature and pH (USEPA 1999b, p.3)

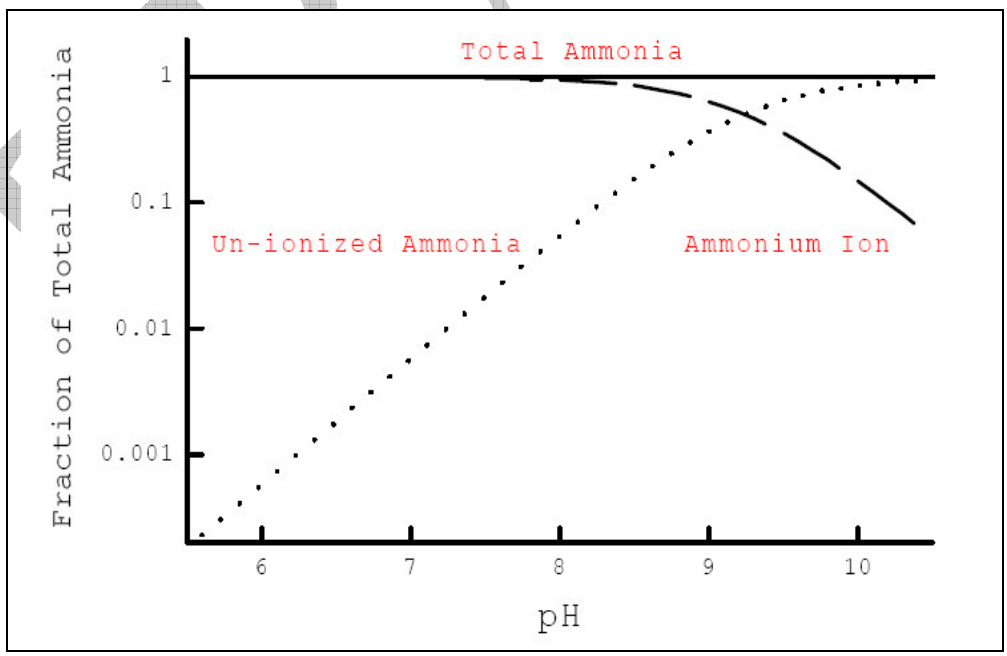

Figure 1: Chemical Speciation of Ammonia
Source: USEPA 1999b, p.3

3.3 Ammonia Toxicity

These speciation relationships are important to ammonia toxicity because un-ionized ammonia is much more toxic than ammonium ion. The importance of un-ionized ammonia was first recognized when it was observed that increased pH caused total ammonia to appear to be much more toxic (Chipman 1934; Wuhrmann and Woker 1948). It is not surprising that un-ionized ammonia is the more toxic form, because it is a neutral molecule and thus is able to diffuse across the epithelial membranes of aquatic organisms much more readily than the charged ammonium ion. Ammonia is unique among regulated pollutants because it is an endogenously produced toxicant that organisms have developed various strategies to excrete, which is in large part by passive diffusion of un-ionized ammonia from the gills. High external un-ionized ammonia concentrations reduce or reverse diffusive gradients and cause the buildup of ammonia in gill tissue and blood (USEPA 1999b, p.3).

Because of the importance of un-ionized ammonia, it became a convention in the scientific literature to express ammonia toxicity in terms of un-ionized ammonia, and water quality criteria and standards followed this convention. However, there are reasons to believe that ammonium ion can contribute significantly to ammonia toxicity under some conditions. Observations that ammonia toxicity is relatively constant when expressed in terms of un-ionized ammonia come mainly from toxicity tests conducted at pH>7.5. At lower pH, toxicity varies considerably when expressed in terms of unionized ammonia and under some conditions is relatively constant in terms of ammonium ion (Erickson 1985). Also, studies have established that mechanisms exist for the transport of ammonium ion across gill epithelia (Wood 1993), so this ion might contribute significantly to ammonia exchange at gills and affect the buildup of ammonia in tissues if its external concentration is sufficiently high. Thus, the very same arguments employed for the importance of un-ionized ammonia can also be applied in some degree to ammonium ion. This is not to say that ammonium ion is as toxic as unionized ammonia, but rather that, regardless of its lower toxicity, it can still be important because it is generally present in much greater concentrations than un-ionized ammonia (USEPA 1999b, p.3,4).

3.4 Ammonia Criteria

The USEPA has utilized the above information to create pH-dependant acute and pH- and temperature-dependent chronic criterion for total ammonia (NH_3 and NH_4^+) as nitrogen in freshwater (Tables 17, 18, 19).

Table 17: pH-Dependent Values of the Criterion Maximum Concentration (CMC) of Total Ammonia as Nitrogen (mg N/L) in Freshwater when Salmonids are Present

Acute Criterion[1]			
pH	**CMC Total NH3 mgN/L**	**pH**	**CMC Total NH3 mgN/L**
6.5	32.6	7.8	8.11
6.6	31.3	7.9	6.77
6.7	29.8	8.0	5.62
6.8	28.1	8.1	4.64
6.9	26.2	8.2	3.83
7.0	24.1	8.3	3.15
7.1	22.0	8.4	2.59
7.2	19.7	8.5	2.14
7.3	17.5	8.6	1.77
7.4	15.4	8.7	1.47
7.5	13.3	8.8	1.23
7.6	11.4	8.9	1.04
7.7	9.65	9.0	0.885

[1] The one-hour average concentration of total ammonia nitrogen (NH_3 and NH_4^+) should not exceed this value more than once every 3 years.
Source: USEPA 1999b, p.86

Table 18: Temperature and pH-Dependent Values of the Criterion Continuous Continuation (CCC) for Total Ammonia as Nitrogen (mg N/L) in Freshwater when Fish Early Life Stages are Present

Chronic Criterion[1]										
CCC for Fish Early Life Stages Present, mg N/L										
pH	Temperature, C									
	0	14	16	18	20	22	24	26	28	30
6.5	6.67	6.67	6.06	5.33	4.68	4.12	3.62	3.18	2.80	2.46
6.6	6.57	6.57	5.97	5.25	4.61	4.05	3.56	3.13	2.75	2.42
6.7	6.44	6.44	5.86	5.15	4.52	3.98	3.50	3.07	2.70	2.37
6.8	6.29	6.29	5.72	5.03	4.42	3.89	3.42	3.00	2.64	2.32
6.9	6.12	6.12	5.56	4.89	4.30	3.78	3.32	2.92	2.57	2.25
7.0	5.91	5.91	5.37	4.72	4.15	3.65	3.21	2.82	2.48	2.18
7.1	5.67	5.67	5.15	4.53	3.98	3.50	3.08	2.70	2.38	2.09
7.2	5.39	5.39	4.90	4.31	3.78	3.33	2.92	2.57	2.26	1.99
7.3	5.08	5.08	4.61	4.06	3.57	3.13	2.76	2.42	2.13	1.87
7.4	4.73	4.73	4.30	3.78	3.32	2.92	2.57	2.26	1.98	1.74
7.5	4.36	4.36	3.97	3.49	3.06	2.69	2.37	2.08	1.83	1.61
7.6	3.98	3.98	3.61	3.18	2.79	2.45	2.16	1.90	1.67	1.47
7.7	3.58	3.58	3.25	2.86	2.51	2.21	1.94	1.71	1.50	1.32
7.8	3.18	3.18	2.89	2.54	2.23	1.96	1.73	1.52	1.33	1.17
7.9	2.80	2.80	2.54	2.24	1.96	1.73	1.52	1.33	1.17	1.03
8.0	2.43	2.43	2.21	1.94	1.71	1.50	1.32	1.16	1.02	0.897
8.1	2.10	2.10	1.91	1.68	1.47	1.29	1.14	1.00	0.879	0.773
8.2	1.79	1.79	1.63	1.43	1.26	1.11	0.973	0.855	0.752	0.661
8.3	1.52	1.52	1.39	1.22	1.07	0.941	0.827	0.727	0.639	0.562
8.4	1.29	1.29	1.17	1.03	0.906	0.796	0.700	0.615	0.541	0.475
8.5	1.09	1.09	0.990	0.870	0.765	0.672	0.591	0.520	0.457	0.401
8.6	0.920	0.920	0.836	0.735	0.646	0.568	0.499	0.439	0.386	0.339
8.7	0.778	0.778	0.707	0.622	0.547	0.480	0.422	0.371	0.326	0.287
8.8	0.661	0.661	0.601	0.528	0.464	0.408	0.359	0.315	0.277	0.244
8.9	0.565	0.565	0.513	0.451	0.397	0.349	0.306	0.269	0.237	0.208
9.0	0.486	0.486	0.442	0.389	0.342	0.300	0.264	0.232	0.204	0.179

[1] The thirty-day average concentration of total ammonia (NH_3 and NH_4^+) should not exceed this value more than once every three years. Additionally, the highest four day average within the thirty-day period should not exceed 2.5 times the CCC (USEPA 1999b, p.87).

Table 19: Temperature and pH-Dependent Values of the Criterion Continuous Continuation (CCC) for Total Ammonia as Nitrogen (mg N/L) in Freshwater when Fish Early Life Stages are Absent

Chronic Criterion[1]

CCC for Fish Early Life Stages Absent, mg N/L										
pH	Temperature									
	0-7	8	9	10	11	12	13	14	15*	16*
6.5	10.8	10.1	9.51	8.92	8.36	7.84	7.35	6.89	6.46	6.06
6.6	10.7	9.99	9.37	8.79	8.24	7.72	7.24	6.79	6.36	5.97
6.7	10.5	9.81	9.20	8.62	8.08	7.58	7.11	6.66	6.25	5.86
6.8	10.2	9.58	8.98	8.42	7.90	7.40	6.94	6.51	6.10	5.72
6.9	9.93	9.31	8.73	8.19	7.68	7.20	6.75	6.33	5.93	5.56
7.0	9.60	9.00	8.43	7.91	7.41	6.95	6.52	6.11	5.73	5.37
7.1	9.20	8.63	8.09	7.58	7.11	6.67	6.25	5.86	5.49	5.15
7.2	8.75	8.20	7.69	7.21	6.76	6.34	5.94	5.57	5.22	4.90
7.3	8.24	7.73	7.25	6.79	6.37	5.97	5.60	5.25	4.92	4.61
7.4	7.69	7.21	6.76	6.33	5.94	5.57	5.22	4.89	4.59	4.30
7.5	7.09	6.64	6.23	5.84	5.48	5.13	4.81	4.51	4.23	3.97
7.6	6.46	6.05	5.67	5.32	4.99	4.68	4.38	4.11	3.85	3.61
7.7	5.81	5.45	5.11	4.79	4.49	4.21	3.95	3.70	3.47	3.25
7.8	5.17	4.84	4.54	4.26	3.99	3.74	3.51	3.29	3.09	2.89
7.9	4.54	4.26	3.99	3.74	3.51	3.29	3.09	2.89	2.71	2.54
8.0	3.95	3.70	3.47	3.26	3.05	2.86	2.68	2.52	2.36	2.21
8.1	3.41	3.19	2.99	2.81	2.63	2.47	2.31	2.17	2.03	1.91
8.2	2.91	2.73	2.56	2.40	2.25	2.11	1.98	1.85	1.74	1.63
8.3	2.47	2.32	2.18	2.04	1.91	1.79	1.68	1.58	1.48	1.39
8.4	2.09	1.96	1.84	1.73	1.62	1.52	1.42	1.33	1.25	1.17
8.5	1.77	1.66	1.55	1.46	1.37	1.28	1.20	1.13	1.06	0.990
8.6	1.49	1.40	1.31	1.23	1.15	1.08	1.01	0.951	0.892	0.836
8.7	1.26	1.18	1.11	1.04	0.976	0.915	0.858	0.805	0.754	0.707
8.8	1.07	1.01	0.944	0.885	0.829	0.778	0.729	0.684	0.641	0.601
8.9	0.917	0.860	0.806	0.756	0.709	0.664	0.623	0.584	0.548	0.513
9.0	0.790	0.740	0.694	0.651	0.610	0.572	0.536	0.503	0.471	0.442

[1] The thirty-day average concentration of total ammonia (NH_3 and NH_4^+) should not exceed this value more than once every three years. Additionally, the highest four day average within the thirty-day period should not exceed 2.5 times the CCC.
Source: USEPA 1999b, p.88

CHAPTER 4. pH

4.1 Introduction

The pH of freshwater streams is important for adult and juvenile salmonid development. Chronic effects from low pH can occur at levels that are not toxic to adult fish but that impair reproduction including altered spawning behavior, reduced egg viability, decreased hatchability, and reduced survival of the early life stages (Jordahl and Benson 1987). The early life stages of salmonid development are most vulnerable to low pH (Jordahl and Benson 1987). Chronic high pH levels in freshwater streams can decrease activity levels of salmonids, create stress responses, decrease or cease feeding, and lead to a loss of equilibrium (Murray and Ziebell 1984; Wagner et al. 1997). Additionally, high temperatures can exacerbate the effects of high pH levels on salmonids (Wagner et al. 1997). If pH reaches extremely low or high levels, death can occur (Wagner et al. 1997).

Literature reviewed for this analysis included numerous laboratory and field studies. The following information applies to salmonids in general.

4.2 Effects of High pH

Wagner et al. (1997) conducted laboratory and field studies and found that pH values of greater than 9.4 will result in the death of rainbow trout, especially at temperatures ranging from 19-22 C. Fresh water pH values of 9.0 or greater resulted in significant stress responses in rainbow trout.

Wilkie and Wood (1996) found that Lahontan cutthroat trout exposed to high pH waters (9.4) permanently lowered their rate of nitrogenous waste production to avoid the potentially toxic build-up of internal ammonia. However, rainbow trout, kokanee, and brown trout were unable to adapt to the high pH and died.

Murray and Ziebell (1984) found that rainbow trout are not able to acclimate to pH levels of 10.0 or higher and that their ability to tolerate pH above 9.0 depends on the rate of acclimation. Gradual acclimation (0.2 to 0.4 of a pH unit/day) allowed rainbow trout to acclimate to a pH of 9.8 and continue feeding, although they showed signs of distress and their activity was greatly reduced by the end of 4 days when the pH reached 9.9 (Table 20). The maximum pH tolerated before fish began dying was 10.2.

Rapid acclimations tests conducted by Murray and Ziebell (1984) yeilded the following results:

> Rainbow trout mortalities were 40% or greater in preliminary acclimation tests in which pH was increased to 9.6 and 9.7 in 3 and 5 hours. These results were comparable to previous shock tests (unpublished data). Consequently, in later experiments, acclimation time was increased to 6 hours and pH values were lowered to 9.3 and 9.5.

Trout in the pH 9.5 experiments exhibited adverse reactions and mortalities were similar to those seen in preliminary tests at pH 9.6 and 9.7. All fish began to show marked signs of stress within 12 hours, and within 24 hours the mortalities in replicated experiments were 30, 40, and 50% respectively. At 49 hours the last deaths occurred that brought mortalities to 50% in each replicate. All remaining living fish were distressed and did not feed. After 72 hours had elapsed, the survivors resumed feeding and their condition improved until the experiments were terminated at 120 hours.

In the pH 9.3, 6-hour acclimation experiments trout exhibited only minor adverse reactions. The primary behavioral changes were a decrease in swimming activity and a temporary loss of appetite. After 48 hours all fish resumed normal feeding and became progressively more active. No mortalities occurred in any of the replicated experiments, and all fish behaved normally when the experiments were terminated at 120 hours Murray and Ziebell (1984).

Table 20: Reactions of 10 rainbow trout to various pH levels during gradual acclimation experiments (0.2 to 0.4 of a pH unit/day)

Day	pH Range	Reactions and Condition of Trout
1	8.6-8.9	Normal
2	8.9-9.2	Activity decreased but feeding normal
3	9.2-9.7	Activity further decreased but feeding continued
4	9.7-9.9	Minor distress shown but feeding continued
5	9.9-10.3	Some fish lost equilibrium at 10.0, and feeding ceased. Loss of equilibrium increased at 10.1 and eyes of some fish developed corneal opacities; 50% of fish lost equilibrium at pH 10.2 and mortality was 60% at pH 10.3

Source: Murray and Ziebell (1984)

4.3 Effects of Low pH

"Chronic effects of low pH on fish populations may occur at pH levels that are not toxic to adult fish but that impair reproduction, and ultimately lead to population extinction (Jordahl and Benson 1987)." A study was conducted by Weiner et al. (1986) to determine the effects of low pH on the reproductive success of rainbow trout. It was determined that exposure of adult salmonids to pH values below 5.5 negatively effected reproduction. Adult rainbow trout were exposed to pH 4.5, 5.0, 5.5, and 6.5-7.1 during the final 6 weeks of reproductive maturation. Weiner et al. found that pH values of 5.5 and below impaired the creation of eggs in females and sperm in males.

Jordahl and Benson (1987) report that reproductive failure occurred in adult brook trout due to low pH in a freshwater stream with pH levels ranging from 5.0-5.8, while trout in a reference stream with pH ranging from 6.1-7.2 did not experience reproductive failure. Additionally, brook trout were absent from a highly acidic freshwater stream with pH

ranging from 4.7-5.4 leading Jordahl and Benson to conclude that breeding females may avoid acidic tributaries.

In addition to effecting adult salmonids, highly acidic freshwater (low pH) can have a detrimental effect on eggs and juvenile salmonids. Weiner et al. (1986) determined that juvenile rainbow trout mortality was greatly increased at pH levels of 5.5 and below, and that no eggs survived when exposed to pH levels below 4.5. Hulsman and Powels (1983) found the mortality of rainbow trout yolk-sac larvae approached 100% within 5 days of exposure to pH 4.6 and 5.4, whereas exposure to pH 6.0 resulted in less than 3% mortality.

Jordahl and Benson (1987) conducted a study to determine the effect of low pH on juvenile brook trout survival and found that survival rates were highest in a freshwater stream with pH values ranging from 6.1-7.2 and lower in acidified streams with pH levels of 4.7-5.8. At pH values of 5.0 and lower, growth was retarded and the development of yolk-sac larvae was considerably prolonged. Additionally, larval activity was depressed, pigmentation was reduced, and incomplete hatching was observed in streams with low pH values of 4.7-5.8, but not in the stream with pH ranging from 6.1-7.2. Jordahl and Benson concluded that mean pH values of 5.0-5.4 can cause acid stress on developing juveniles, while pH levels from 6.1-7.2 are above ranges that negatively effect juvenile brook trout development and survival.

DRAFT

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Appendix 4

Fish and Fishery Resources of the Klamath River Basin

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July 2008

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1.1 Introduction

The Klamath River basin contains 83 species of fish, 45 of which are native to the Klamath drainage and 38 that have been introduced and are non-native. Fourteen of the native fish species in the basin have been granted special federal and/or state status (Table 1).

Table 1: Native Fish Species in the Klamath River Basin with Special Federal and/or State Status

SPECIES	STATUS
Shortnose sucker, *Chasmistes brevirostris*	Endangered-OR, CA, and Federal
Lost River sucker, *Deltistes luxatus*	Endangered-OR, CA, and Federal
Coho salmon, *Oncorhynchus kisutch*	Critical -OR; Threatened-CA and Federal;
Bull trout, *Salvelinus confluentus*	Critical-OR; Endangered-CA; Threatened-Federal
Delta Smelt, *Hypomesus transpacificus*	Threatened-CA and Federal
Coastal cutthroat trout, *Oncorhynchus clarki clarki*	Special Concern-CA
Eulachon, *Thaleichthys pacificus*	Special Concern-CA
Longfin smelt, *Spirinchus thaleichthys*	Special Concern-CA
Redband/Rainbow trout, *Oncorhynchus mykiss gairdneri*	Vulnerable-OR
Chum Salmon, *Oncorhynchus keta*	Special Concern-CA
Klamath largescale sucker, *Catostomus snyderi*	Special Concern-Federal
Slender sculpin, *Cottus tenuis*	Special Concern-Federal
Pacific Lamprey, *Lampetra tridentata*	Vulnerable-OR; Special Concern-Federal
Green sturgeon, *Acipenser medirostris*	Special Concern- CA and Federal

Sources: California Department of Fish and Game (CDFG) 2006b, p.4-6; National Research Council (NRC) 2004, p.181, 251, & 252; Oregon Natural Heritage Information Center (ONHIC) 2004, p.8-11.

The following discussion of fish species and resources in the basin is divided into three parts: fish species found above Iron Gate Dam in California and Oregon, fish species found from Iron Gate Dam to the Ocean in California, and Chinook, steelhead, and coho salmonids from Iron Gate Dam to the Ocean in California.

1.2 Fish above Iron Gate Dam in the Klamath River Basin, California and Oregon

The Klamath River basin above Iron Gate Dam hosts 18 native and 19 non-native fish species (Table 2). Native fish persisting in this area of the basin include lamprey, trout, and sucker species including the endangered shortnose and Lost River suckers. Introduced fish include various sunfish, catfish, and perch species.

Table 2: Fish Found Above Iron Gate Dam in the Klamath River Basin

NATIVE	
Klamath River lamprey, *Lampetra similis*	Klamath largescale sucker, *Catostomus snyderi*
Miller Lake Lamprey, *Lampetra milleri*	Klamath smallscale sucker, *Catostomus rimiculus*
Pit-Klamath brook lamprey, *Lampetra lethophaga*	Redband/Rainbow trout, *Oncorhynchus mykiss gairdneri*
Klamath tui chub, *Siphatales bicolor bicolor*	Bull trout, *Salvelinus confluentus*
Blue chub, *Gila coerulea*	Klamath Lake sculpin, *Cottus princeps*
Klamath speckled dace, *Rhinichthys osculus klamathensis*	Slender sculpin, *Cottus tenuis*
Shortnose sucker, *Chasmistes brevirostris*	Upper Klamath marbled sculpin, *Cottus klamathensis klamathensis*
Lost River sucker, *Deltistes luxatus*	

Table 2 (continued): Fish Found Above Iron Gate Dam in the Klamath River Basin

NON-NATIVE	
Goldfish, *Carassius auratus*	Brown trout, *Salmo trutta*
Golden shiner, *Notemigonus chrysoleucas*	Sacramento perch, *Archoplites interruptus*
Fathead minnow, *Pimephales promelas*	White crappie, *Pomoxis annularis*
Yellow bullhead, *Ameiurus natalis*	Black crappie, *Pomoxis nigromaculatus*
Brown bullhead, *Ameiurus nebulosus*	Green sunfish, *Lepomis cyanellus*
Black bullhead, *Ameiurus melas*	Bluegill, *Lepomis macrochirus*
Channel catfish, *Ictalurus punctatus*	Pumpkinseed, *Lepomis gibbosus*
Kokanee salmon, *Oncorhynchus nerka*	Largemouth bass, *Micropterus salmoides*
Brook trout, *Salvelinus fontinalis*	Yellow perch, *Perca flavescens*

Source: NRC 2004, p.181, 189; PacifiCorp 2004, p.4-5 to 4-7.

1.2.1 Distribution and Status of Native Fish

The following information on fish distribution and status is mainly derived from NRC 2004, p.181-193, with additional information taken from Behnke 1992, p.19, 20 and PacifiCorp 2004, p. 4-12, 4-13, and 4-33.

Four species of suckers inhabit the Klamath River basin above Iron Gate Dam. The shortnose and Lost River suckers are large, long-lived, late-maturing and live in lakes but spawn primarily in streams. Shortnose and Lost River suckers have been found in the reservoirs between Keno and Iron Gate Dam. The Klamath Tribes refer to the shortnose and Lost River suckers as qapdo and c'waam, respectively. These fish were a primary food source for the Klamath and Modoc Indians from historic times until the 1980s when severe declines in the fish populations caused the Klamath Tribes to close their fishery. Historically, Lost River and shortnose suckers were present in the Lost River and Klamath River above Iron Gate Dam and their tributaries (Moyle 2002, USFWS 2002 Appendix D as cited by NRC 2004, p.190, 191). Their current distribution and numbers have decreased from a combination of extirpations and redistribution through water management (NRC p.191). The Klamath Tribes historically harvested tens of thousands of pounds of c'waam and qapdo. Now they are restricted to a single fish each year for ceremonial purposes. Both species are currently on the federal, Oregon, and California endangered species list. Klamath smallscale suckers are considered to be rare in the Klamath River basin. The status of Klamath largescale suckers is poorly understood although it is surmised that lake populations are probably declining in abundance while river populations are probably abundant. Periodicity information for all four sucker species within the Klamath River basin in Oregon is presented in Figure 1.

Ancestors of the redband trout (resident rainbow trout) entered the Klamath River basin when it was connected to the Columbia basin via the Snake River, and coastal rainbow trout (steelhead) later entered the basin from the ocean. Redband/rainbow trout have persisted in the basin above Iron Gate Dam because of their ability to thrive in lake and stream conditions that would be lethal to most salmonids. Currently, redband/rainbow trout numbers are high in both lakes and rivers of the basin above Iron Gate Dam, and these trout support a strong summer fishery. Redband/rainbow trout are currently present in both Copco and Iron Gate Reservoir (Pacificorp 2004, p.4-53 – 4-55, 4-58).

The redband/rainbow trout population in the J.C. Boyle peaking reach (J.C. Boyle Dam to Copco 1 reservoir) supports a high quality recreational fishery and has been described by the National Park Service as highly productive and self-sustaining. In 1984 the adult population in the upper 6 miles of the reach was estimated as 890 fish per mile, and in the 5 miles below this area (near the Oregon-California border) the population was estimated to be 1,911 fish per mile. Populations in Shovel Creek are healthy according to CDFG surveys in the early 1990's. Periodicity information for redband/rainbow trout within the Klamath River basin in Oregon is presented in Figure 1.

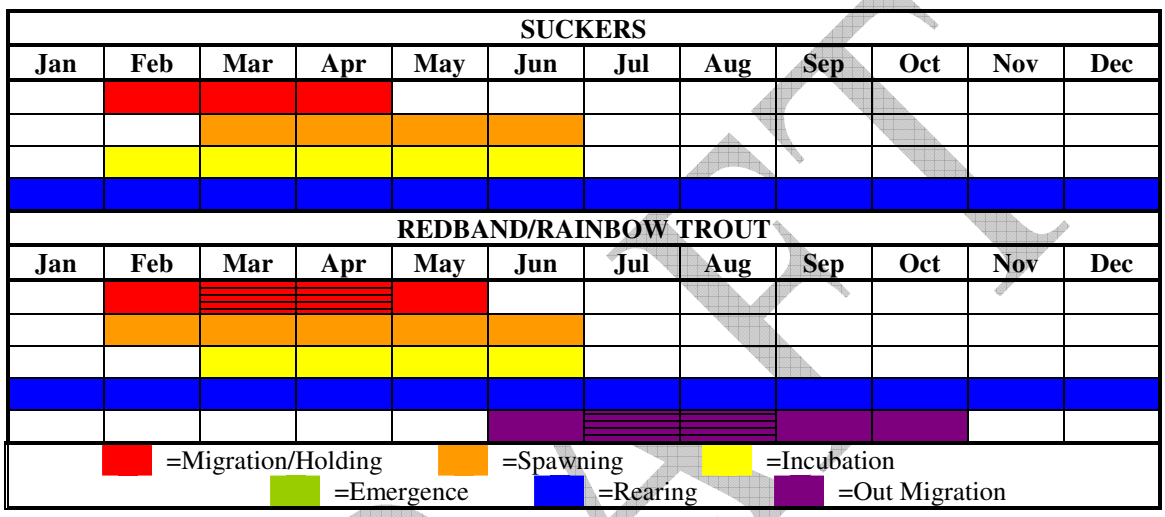

Figure 1: Sucker and Redband/Rainbow Trout Periodicity for the Klamath River in Oregon

= Lesser Use =Lesser Use

Source: FISHPRO 2000

Bull trout have been extirpated, or are at risk of extirpation, from most of the areas they once existed in the Klamath River basin. Bull trout are known to be or have been present in 10 creeks in the basin above Iron Gate Dam: four tributaries to the Sprague River, four tributaries to the Sycan River, and two tributaries to Upper Klamath Lake. The current distribution of bull trout is limited to the headwaters upstream of Upper Klamath Lake. Populations are listed as threatened by the federal government, critical by Oregon, and endangered by California.

The abundance of Klamath Lake sculpin in the basin above Iron Gate Dam is estimated to be in the millions. The presence of these sculpin has not changed from historical distributions and they are found in springs and creeks flowing into the west side of Upper Klamath Lake, Upper Klamath Lake, and Agency Lake. Upper Klamath marbled sculpin are the most widely distributed sculpin present in the basin. They are found in most streams and rivers in the Klamath River basin above Iron Gate Dam, and common in Upper Klamath Lake although they are largely absent from the reservoirs in California. Although slender sculpin were historically common in Upper Klamath Lake and the Williamson, Sprague, Sycan, and Lost Rivers, a survey conducted during the mid 1990's

found them to be present only in Upper Klamath Lake and the Lower Williamson River. Overall, the slender sculpin have disappeared from much of their native range, are uncommon in areas where they are present today, and are listed as a species of concern by the federal government.

Klamath River and Pit-Klamath brook lamprey are abundant and widespread in small streams of the basin above Iron Gate Dam. Klamath tui chub have decreased in abundance in the Lost River over the last 30 years, but are typically among the most abundant species found during fish kills in Upper Klamath Lake. Blue chub populations throughout the basin are in decline, however they are probably the most abundant native fish in Upper Klamath Lake. The status of Klamath speckled dace is currently unknown although it appears to be common in the basin with the exception of the Lost River.

1.2.2 Distribution and Status of Non-Native Fish

The following information on fish distribution and status is mainly derived from NRC 2004, p.188-189, with additional information taken from PacifiCorp 2004, p.4-30, 4-31.

Fifteen of the non-native species in the Klamath River basin above Iron Gate Dam were introduced for sport fishing or for bait. Most of these species are not common in the basin, although some are abundant and widespread. The effect of these fish on native fishes is poorly understood.

Yellow perch, brown bullhead, and pumpkinseed are abundant in the reservoirs, sloughs and ponds of the basin above Iron Gate Dam. Brook trout, brown trout, and non-native strains of rainbow trout are common in streams above Iron Gate Dam and have replaced native redband/rainbow trout and bull trout in many areas. Bullhead and perch are the most abundant non-native species found in Copco Reservoir, while Iron Gate Reservoir hosts large populations of perch, bass, and crappie. Non-native trout are also found in Iron Gate and Copco reservoirs. Fathead minnows are often the most abundant species encountered during fish sampling, and are common in Upper Klamath Lake and the Lost River system. Declines in tui and blue chub numbers have been associated with the increased presence of fathead minnows. Sacramento perch is also present in the Klamath River in the area from below Upper Klamath Lake to Iron Gate Reservoir and throughout the Lost River, although its numbers are not particularly high where present.

1.3 Fish below Iron Gate Dam in the Klamath River Basin, California

A total of 46 fish species (27 native and 19 non-native) are found in the Klamath River basin below Iron Gate Dam (Table 3). Native fish currently present in this region of the Klamath River include lamprey, sturgeon, sculpin, and salmonids including the state and federally listed coho salmon. Introduced fish include bass, bullheads, and several species of sunfish.

Table 3: Fish Found Below Iron Gate Dam in the Klamath River Basin

NATIVE	
Pacific Lamprey, *Lampetra tridentata**	Eulachon, *Thaleichthys pacificus**
River lamprey, *Lampetra ayersi**	Surf smelt, *Hypomesus pretiosus*
Klamath River lamprey, *Lampetra similes*	Longfin smelt, *Spirinchus thaleichthys**
Green sturgeon, *Acipenser medirostris**	Prickly sculpin, *Cottus asper*
White sturgeon, *Acipenser transmontanus**	Sharpnose sculpin, *Clinocottus acuticeps*
Klamath speckled dace, *Rhinichthys klamathensis osculus*	Coastrange sculpin, *Cottus aleuticus*
Klamath smallscale sucker, *Catostomus rimiculus*	Pacific staghorn sculpin, *Leptocottus armatus*
Shiner perch, *Cymatogaster aggregata*	Lower Klamath marbled sculpin, *Cottus klamathensis polyporus*
Starry flounder, *Platichthys stellatus*	Threespine stickleback, *Gasterosteus aculeatus***
Pacific herring, *Clupea pallasi*	Arrow goby, *Clevelandia ios*
Topsmelt, *Atherinops affinis*	Pink salmon, *Oncorhynchus gorbuscha**
Coho salmon, *Oncorhynchus kisutch**	Steelhead (rainbow trout), *Oncorhynchus mykiss**
Chinook salmon, *Oncorhynchus tshawytscha**	Coastal cutthroat trout, *Oncorhynchus clarki clarki**
Chum salmon, *Oncorhynchus keta**	
NON-NATIVE	
American shad, *Alosa sapidissima**	Sockeye salmon, *Oncorhynchus nerka*
Goldfish, *Carassius auratus*	Bluegill, *Lepomis macrochirus*
Fathead minnow, *Pimephales promelas*	Pumpkinseed, *Lepomis gibbosus*
Golden shiner, *Notemigonus chrysoleucas*	Largemouth bass, *Micropterus salmoides*
Brown bullhead, *Ameiurus nebulosus*	Spotted bass, *Micropterus punctulatus*
Yellow bullhead, *Ameiurus natalis*	Smallmouth bass, *Micropterus dolomieui*
Wakasagi, *Hypomesus nipponensis*	Yellow perch, *Perca flavescens*
Green sunfish, *Lepomis cyanellus*	Delta smelt, *Hypomesus transpacificus*
Brook trout, *Salvelinus fontinalis*	Brown trout, *Salmo trutta***

*Anadromous.
**Some anadromous, some non-migratory
Source: NRC 2004, p.251-253; PacifiCorp 2004, p.4-5 to 4-7.

1.3.1 Distribution and Status of Native Fish

Unless otherwise noted, the following information on fish distribution and status is derived from NRC 2004, p.252-277. Chinook and coho salmon, and steelhead trout habitat and distribution, populations, and periodicity are discussed in great detail in section 1.4.

Anadromous species present in the Klamath River basin below Iron Gate Dam include Chinook, coho, pink, and chum salmon, steelhead and coastal cutthroat trout, eulachon, white and green sturgeon, and Pacific lamprey.

Chum salmon are periodically observed in the basin, and maintain a small population in the Klamath River. Historically chum were more abundant, although their numbers were never very large. Coastal cutthroat trout mainly occur in smaller tributaries in the lower 22 miles of the Klamath River, and have been observed in tributaries to the Trinity River. Pink salmon probably once existed in the Klamath River, although they appear to be extirpated from all areas in California and only occasionally stray into streams along the California coast. In 2003, 2 pink salmon fry were found in the Klamath River between Iron Gate Dam and the Interstate 5 bridge (Corum 2006).

Spring/summer steelhead were once widely distributed in the Klamath River and Trinity River basins and were present in the headwaters of most larger tributaries. Their

numbers have declined from historic levels, and NMFS considers stocks depressed and in danger of extinction. Fall and winter steelhead are currently widely distributed in the basin below Iron Gate Dam. Their numbers are believed to be declining from historic levels although past and present estimates of abundance are not readily available. NMFS considers winter steelhead to be in low abundance and at some risk of extinction (Busby et al. 1994 as cited by NRC 2004, p.233), but has not listed them under the ESA (NRC 2004, p.274).

Spring and fall run Chinook populations and distribution have decreased dramatically since the early 1900's. Historically, spring Chinook were found in tributaries throughout the basin, although they are now only present in the Salmon and Trinity Rivers. Large numbers of fall Chinook used to spawn in the basin above Iron Gate Dam, but no longer have access to these areas. In the early 1900's as many as 100,000 spring Chinook were found in the basin, but current populations range from 100 to 1000 fish. Fall Chinook populations have also declined as is evidenced by the Shasta River run which were around 80,000 fish in the 1930's and in the last 10 years have generally been well below 10,000 fish.

Coho were once abundant and widely distributed in the Klamath River and its tributaries at least as far up as Spencer Creek in Oregon (Hamilton et. al. 2005, p.16). Trinity River wild coho stocks have experienced a 96% decline in numbers from historic levels. Coho in the Klamath River basin are currently on the state and federal endangered species lists due to the long-term decline in numbers and distribution.

Eulachon were historically present in large numbers in the lower 8 miles of the river, however since the 1970's their numbers have been too low to support the once flourishing tribal fishery. It is estimated that 70-80% of all green sturgeon are produced in the lower Klamath and Trinity Rivers where several hundred are taken every year by the tribal fishery. There is some evidence that green sturgeon numbers in the basin below Iron Gate Dam have decreased in recent years, although a proposal to list them as threatened was declined by the NMFS in 2003. At the present time they are listed as a species of special concern by the federal government. The historic distribution of Pacific lamprey is unknown, however it is certain that they entered the area above Klamath Falls, Oregon in the basin above Iron Gate Dam at least occasionally. Today Pacific lamprey populations are declining in all coastal rivers, and they are listed as a species of concern by the federal government.

Non-anadromous species common in the Klamath River below Iron Gate Dam and its low gradient tributaries include speckled dace, Klamath smallscale suckers, lower Klamath marbled sculpin, threespine stickleback, and Klamath River lamprey. Dace, stickleback, sculpin, and suckers probably utilize nutrients brought into the streams by anadromous species, and may suffer heavy predation by juvenile salmonids.

1.3.2 Distribution and Status of Non-Native Fish

The following information on non-native fish distribution and status is derived from NRC 2004, p.236-237.

The Klamath River basin below Iron Gate Dam is dominated by native fish, although non-native species have a stronger presence in highly altered areas such as reservoirs and ponds. Large populations of brown bullhead and other non-natives are present in the Shasta River due to introductions and the warmth of these waters. Non-native fish continually enter the Klamath River below Iron Gate Dam from the basin above the Dam where they are extremely abundant (NRC 2004, p.277).

1.4 Chinook salmon, steelhead trout, and coho salmon below Iron Gate Dam in the Klamath River Basin, California

Anadromous salmonids in the Klamath River basin are limited to the area of the basin within California below Iron Gate Dam, which is a barrier to anadromy. Anadromous salmonid runs currently utilizing this portion Klamath River basin include spring and fall Chinook, coho salmon, and spring/summer, fall, and winter steelhead trout. Some authors recognize three runs of steelhead in the basin based on the timing of their entrance to the estuary and tributaries (Hopelain 1998; Shaw et al. 1998; Trihey and Associates, Inc. 1996; USFWS 1979), while others recognize two runs based on sexual maturity at the time of entrance to the river (Hardy 1999; Hardy and Addley 2006; KRBFTF 1991; Moyle 2002). This appendix discusses steelhead based on three runs: spring/summer, fall, and winter. All six salmonid runs in the Klamath River basin have experienced declines in populations and distribution since the early 1900's. The decline of anadromous species in the basin can be attributed to a variety of factors including over harvest, land-use practices, mining, stream habitat alterations, agriculture, and changes in water quality and temperature (Hardy and Addley 2006, p.7). Significant effects are also attributed to water allocation practices and dam construction, which has altered flow regimes (Hardy and Addley 2006, p.7). The following discussion reviews the habitat and distribution, status, and periodicity of these six salmonid runs.

1.4.1 Habitat and Distribution

The information in this section was synthesized from the following sources: CDFG 1965, p.369; Hamilton et al. 2005; Hardy 1999, p.19, 20; Hardy and Addley 2006, p.3, 5, 10-20; and NRC 2004, p.289, 290, 295, & 296.

The continued survival and persistence of sustainable populations of salmonids in the Klamath River basin depends on the amount and suitability of the habitat. Historically, anadromous species within the basin extended above Upper Klamath Lake in Oregon, and into the Sprague and Williamson River systems and other tributaries. Chinook salmon historically migrated into tributaries of Upper Klamath Lake, and steelhead trout were found in the Klamath River basin above Iron Gate Dam as well. Coho salmon distribution extended at least to the vicinity of Spencer Creek.

In 1918, the completion of Copco No.1 Dam on the Klamath River became the first migration barrier for anadromous species and eliminated over 100 miles of potential anadromous fish habitat in the basin. However, reduced access to tributaries in the upper areas of the Klamath River basin likely occurred as early as 1912-1914 when the Lost River diversion canal and Chiloquin Dam were constructed (Hardy and Addley 2006, p.5). The final barrier to upstream migration in the mainstem Klamath River occurred in 1962 with the completion of Iron Gate Dam. The construction of Lewiston and Trinity dams on the Trinity River in 1963 blocked access to over 109 miles of salmonid spawning habitat. Dwinnell dam was constructed on the Shasta River in 1926 and created a barrier to migration, blocking access to 22% of the historical salmonid spawning habitat.

A habitat survey published by the CDFG in 1965 found that there were 805 miles of habitat in the Klamath River basin below Iron Gate Dam suitable for Chinook, 813 miles of habitat suitable for coho, and 1,616 miles of habitat suitable for steelhead. More current information from Hardy and Addley estimate that there are about 701 miles of Chinook, 786 miles of coho, and 1121 miles of steelhead habitat in the basin below Iron Gate Dam.

The following figures show the current distribution of Chinook (Figure 2), steelhead (Figure 3), and coho (Figure 4) runs in the Klamath River basin as well as the areas where these species have been extirpated in the basin. These figures are based on readily available data and thus do not necessarily reflect all locations of presence or areas of extirpation. Rather, the figures show the general decrease in the distribution of salmonids in the basin from historic levels. Locations at which fish presence is not indicated on the map do not necessarily indicate the absence of fish in these areas, as surveys to determine presence/absence may not have been conducted at all locations within the basin.

"Spring Chinook have been known to occupy the lower reaches of many mid-Klamath tributaries during their adult migration (Cyr 2006)." In addition to those areas shown in Figure 2, spring Chinook are occasionally found in very small numbers in the following locations: Beaver Creek, Lower Scott River, and Bogus Creek (Brucker 2006; USFS 2006). The occasional presence of spring Chinook in tributaries of the Klamath River above the Salmon River in very low, dwindling numbers, reflects the fact that they are at high risk of extirpation from these areas.

It is believed that fall and spring Chinook and coho, in addition to steelhead, were once present above Dwinnell Dam on the Shasta River. Hardy and Addley (2006, p.12) report that Dwinnell Dam blocked access to habitat that was historically utilized by steelhead in the headwaters of the Shasta River, and thus Figure 3 reflects steelhead extirpation above Dwinnell Dam. The NRC (2004, p.289) state that the construction of Dwinnell Dam blocked access to 22% of the historical salmonid habitat above the dam. However, to date no reference was found which specifically stated the historic presence of Chinook or coho above Dwinnell Dam, though the habitat was suitable for their presence in many tributaries above the dam.

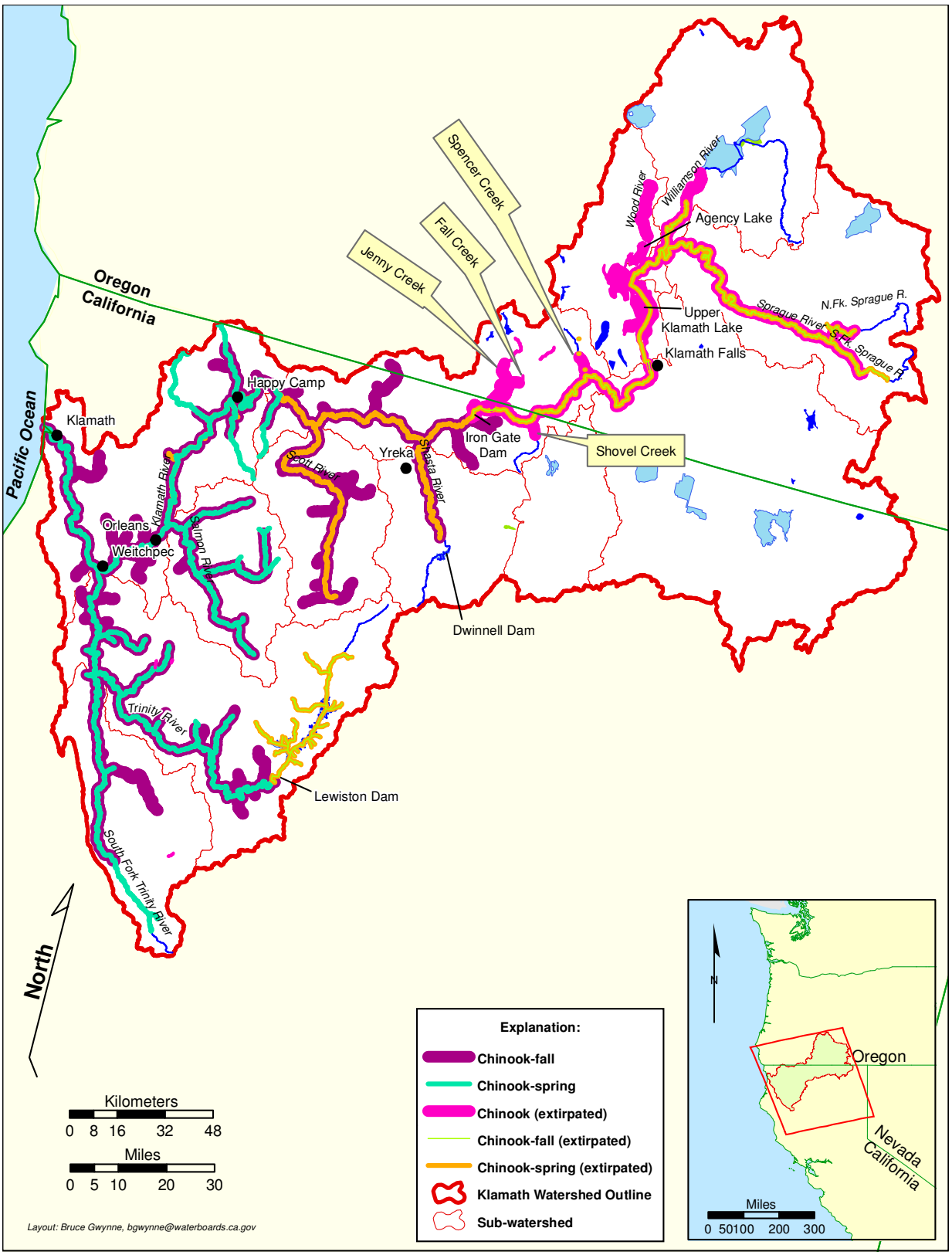

Figure 2: Current Distribution and Areas of Extirpation of Chinook Salmon Runs in the Klamath River Basin
Note: The data for "Chinook (extirpated)" did not differentiate between fall and spring Chinook runs.
Sources: Hamilton et al. 2005, p.12; Moffett and Smith 1950, p.23 & 27; Moyle 2002, p.259; USFS 1996; USFS 2006.

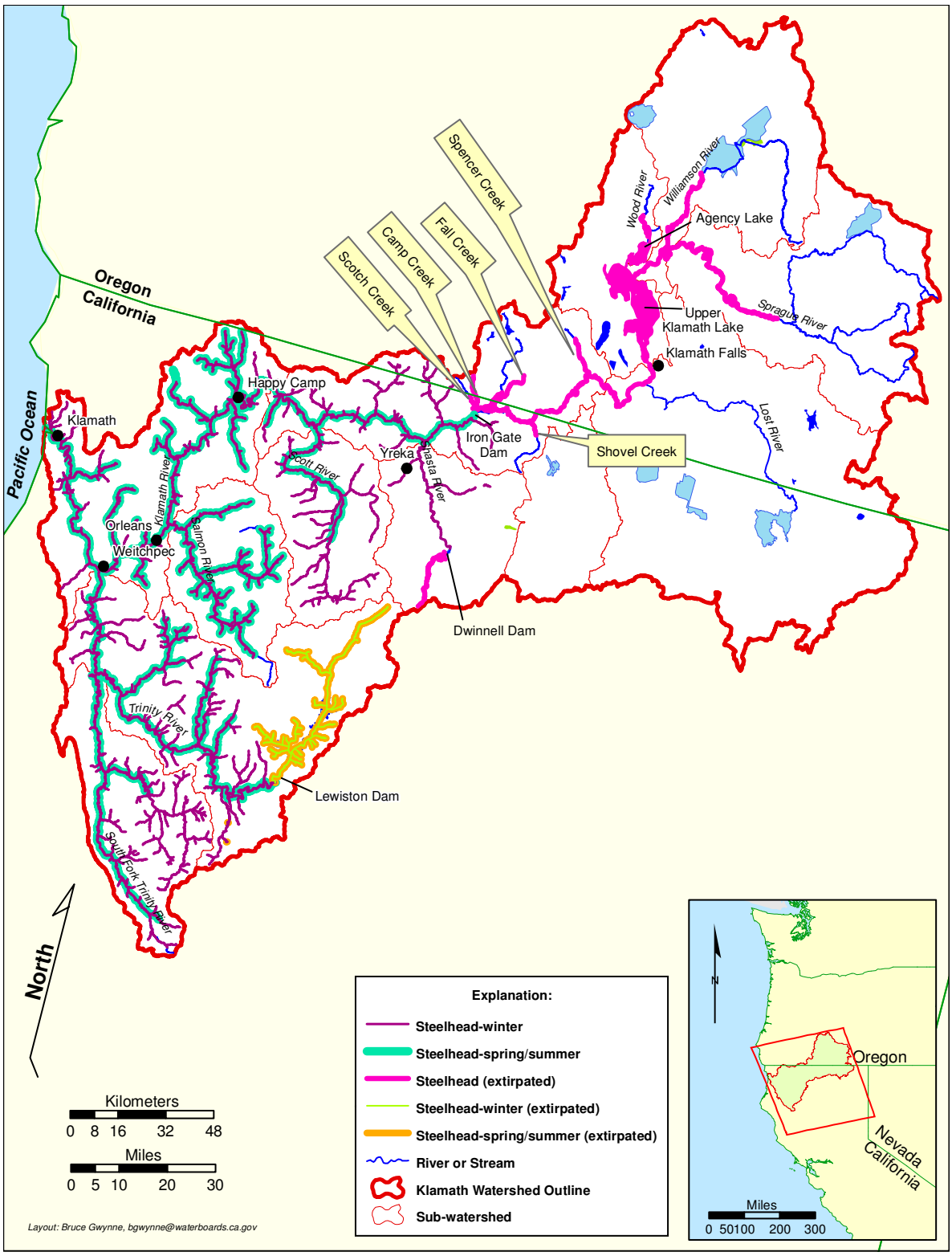

Figure 3: Current Distribution and Areas of Extirpation of Steelhead Trout Runs in the Klamath River Basin
Note: The data for "Steelhead (extirpated)" did not differentiate between spring/summer, fall, and winter runs. The USFS recognizes only winter steelhead (as opposed to fall and winter) and thus information for "Steelhead-winter" and "Steelhead-winter (extirpated)" included data on both fall and winter steelhead runs.
Sources: Hamilton et al. 2005, p.12; Hardy and Addley 2006, p.12; Rushton 2005, p.16 & 17; USFS 1996.

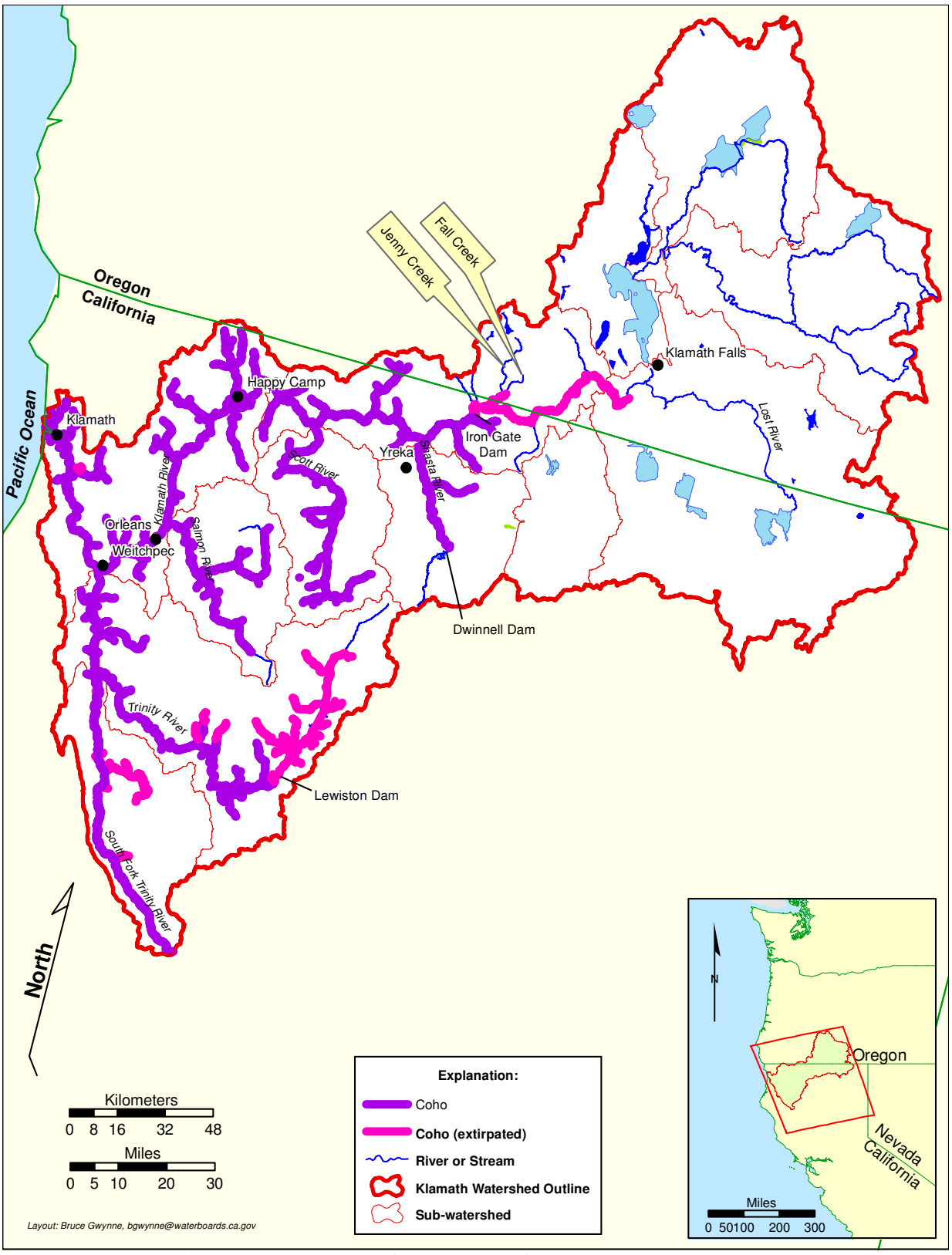

Figure 4: Current Distribution and Areas of Extirpation of Coho Salmon Runs in the Klamath River Basin
Sources: Brown and Moyle 1991, p.14; Brown et al. 1994, p.243; CDFG 2002, p.42; Cyr 2006; Hamilton et al. 2005, p.12; USFS 1996.

1.4.2 Populations

Salmonid populations in the Klamath River basin have declined since the early 1900s. During the period from 1876 to 1933 the salmon runs entering the Klamath River supported a large commercial fishery and several canneries near the mouth of the river (Moyle 2002, p.258). In 1931, Snyder wrote that the fishery of the Klamath River basin is very important because with proper management it can be maintained, although he also states that depletion of the Klamath salmon is apparent and occurring at an "alarming rate" which artificial propagation alone may not remedy (Snyder 1931, p.9, 121). Utilizing information from Snyder (1931), the NRC (2004, p.267, 268) estimated that the annual total catch in the Klamath River basin during the period from 1916-1927 was probably 120,000 to 250,000 fish, and thus the number of potential spawners was considerably higher. Although historically there were large runs of salmonids in the basin, data indicate that current populations are much lower than historic levels.

1.4.2.1 Chinook Salmon

Historic and current records reflect that Chinook salmon were, and continue to be, the most abundant anadromous species in the Klamath River basin. An approximation of total annual catch plus escapement for the period from 1915-1928 estimated there were 300,000 to 400,000 Chinook in the basin (Rankle 1982 as cited by Hardy and Addley 2006, p.7). An estimate of spawner abundance from CDFG in 1965 estimated that on average there were 168,000 Chinook per year in the Klamath River basin (CDFG 1965, p.369). In 1972, Coots estimated that 148,000 Chinook entered the basin (Coots 1973 as cited by Hardy and Addley 2006, p.7).

1.4.2.2 Fall Chinook Salmon

Overall, fall Chinook numbers in the Klamath River basin have dramatically declined during the past century (Hardy and Addley 2006, p.7). The fall Chinook run once totaled as many as 500,000 fish annually (Moyle 2002, p.258). Fall Chinook numbers in the Shasta River watershed alone, historically numbered 20,000-80,000 fish per year (Regional Water Board 2006, p.1-25). Fall Chinook population estimates in the Klamath River basin for the period from 1978-2007 have ranged from a high of 239,559 fish in 1987 to fewer than 35,000 fish in 1991 (Figure 5). In 2002 it was estimated that the fall Chinook population in the basin was 170,014 fish, of which approximately 32,533 were killed (97.1% of the total fish killed) in mid to late September due to a combination of factors including disease, high water temperatures, and low river flow (CDFG 2004, p.III; USFWS 2003a, p.ii; USFWS 2003b, p.ii). This conservative estimate of the number of fall Chinook killed in 2002 is figured from the number of dead fish observed in the area of the fish kill and does not included dead fish that were washed out of the estuary or settled too deep in the water to be visible during surveys (USFWS 2003b, p.1-7). Thus, the estimate of 32,533 dead fall Chinook (19% of the estimated population in 2002) is very conservative and it is likely the actual number of fall Chinook killed was much higher (CDFG 2004, p.III, 158; USFWS 2003b, p.13). Information for 2007 reflect a total estimated run size of 132,167 fall Chinook in the Klamath River basin (CDFG 2008). NRC (2004, p.268) states that in some respects, "…it is remarkable that fall-run Chinook salmon in the Klamath River are doing as well as they seem to be. Both adults migrating

upstream and juveniles moving downstream face water temperatures that are bioenergetically unsuitable or even lethal."

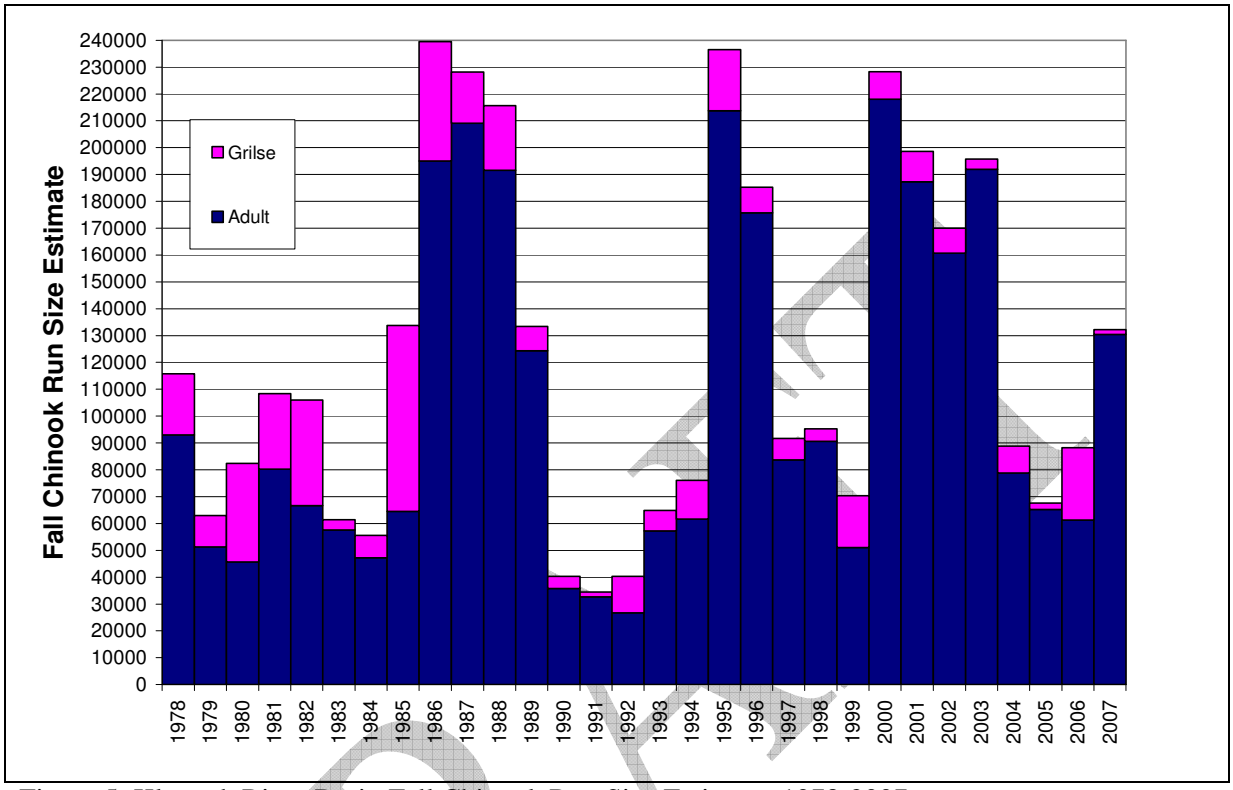

Figure 5: Klamath River Basin Fall Chinook Run Size Estimate, 1978-2007
Note: Run size estimate includes hatchery spawners, natural spawners, and in-river harvest totals. Data from 2007 is preliminary. Grilse are Chinook that return to freshwater to spawn after spending only one year in the ocean.
Source: CDFG 2008

Hatchery returns to Iron Gate and Trinity River hatcheries comprised 6-44% of the fall Chinook populations during the period from 1978-2007 (CDFG 2008). Natural spawners in the Klamath River and its tributaries comprised and estimated 13-39% (Figure 6) of the population during 1991-2007 (CDFG 2008). During 2004 and 2005 the estimated number of fall Chinook natural spawners in the basin has fallen below the Pacific Fishery Management Council goal of a minimum of 35,000. Natural spawning numbers in the basin during these years was estimated to be 28,516 and 27,857 respectively.

In 2006, the National Marine Fishery Service expected the number of natural spawners to be below 35,000 and established an emergency management measure, which closed a majority of the commercial fisheries and greatly reduced the recreational fishery from Cape Falcon, OR, to Point Sur, CA during the period from May 1 through August 31, 2006 (Federal Register 2006, p.26254, 26257). The actual number of natural spawners returning to the Klamath River basin in 2006 (as estimated by CDFG) was 44,546.

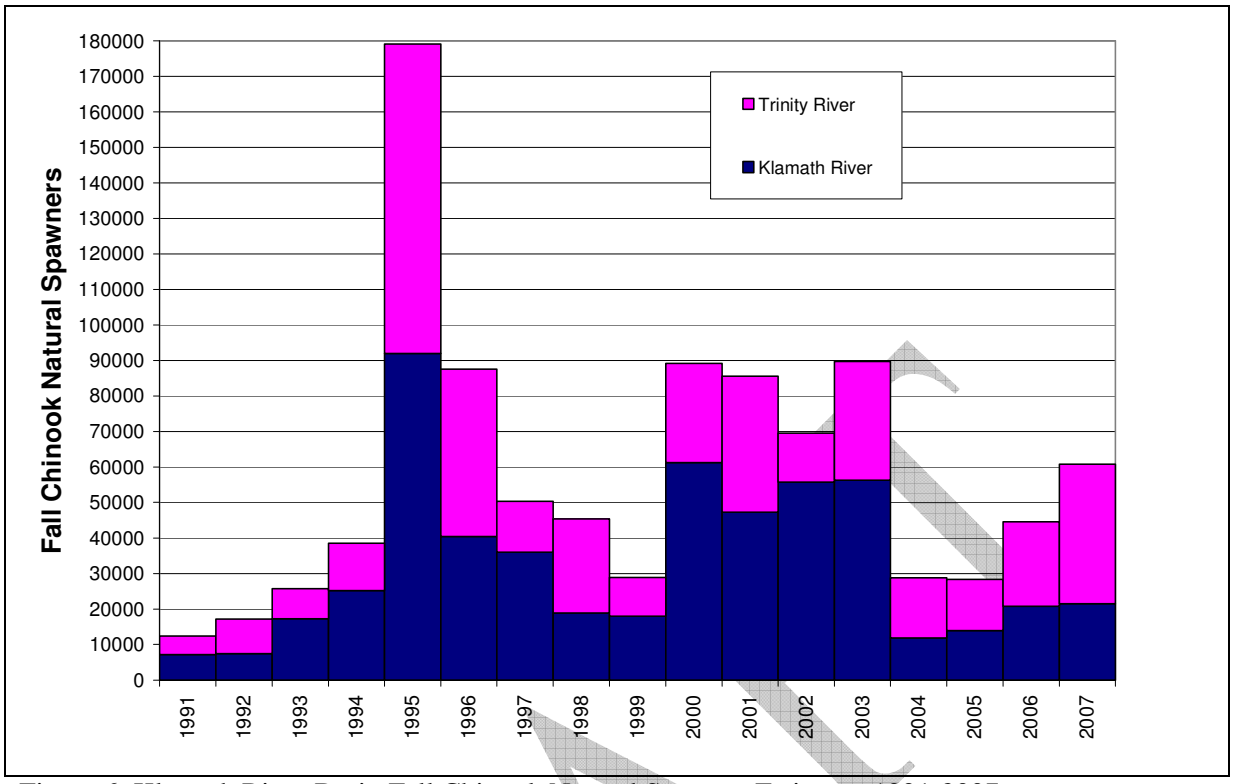

Figure 6: Klamath River Basin Fall Chinook Natural Spawner Estimate, 1991-2007
Note: Data from 2007 is preliminary.
Source: CDFG 2008

The following text on hatchery fall Chinook is from NRC (2004, p.267):

> Hatcheries for Chinook salmon have been operating continuously since 1917. Both the Iron Gate Hatchery and the Trinity River Hatchery produce large numbers of spring-run (13%) and fall-run (87%) juvenile Chinook of native stock (Myers et al. 1998). The hatcheries release 7-12 million juveniles into the river each year (about 70% from the Iron Gate Hatchery, all fall run). The fish generally have been released over a 2-3 days in late May or early June and take 1-2 mo (mean, 31 days) to reach the estuary (M. Wallace, CDFG, unpublished data, 2002), although some fish probably remain in pools for most of summer. Smaller fish take longer than larger fish to reach the estuary, but because they are feeding and growing on the way downstream, all juveniles are about the same size when they reach it. About 40% of the juvenile fish in the estuary in 2000 were of hatchery origin (CDFG, unpublished data, 2000); this is presumably a fairly typical figure. Adult Chinook returning to the hatcheries are roughly one third of the total run—30% in 1999, 44% in 2000, and 28% 2001 (CDFG, unpublished data, 2001). There has been an increase in the percentage of hatchery fish in the run in recent years—up from 18% in 1978-1982, and 26% in 1991-1995 (Meyers et al. 1998). Their contribution to natural spawning is not known, but estimates for the

> Trinity River suggest that it is roughly the same as the percentage of hatchery returns (Myers et al. 1998).

1.4.2.3 Spring Chinook Salmon

The Klamath River basin was known historically for its large run of spring Chinook salmon, which is currently a vestige of its former self (West 1991, p.3). In 1931, Snyder wrote that the spring Chinook migration in the basin, while once very pronounced, "has now come to be limited as to the number of individuals, and is of relatively little economic importance" (Snyder 1931, p.19). A population of more than 100,000 spring-run Chinook was once present in the basin, although this estimate is probably low because spring-run fish were the main run of Chinook in the Klamath River in the 1800's (Moyle 2002, p.259). Access to prime coldwater habitat in the headwaters of the Shasta, Klamath, and Trinity Rivers has been blocked by the construction of dams thus contributing to the decline of spring Chinook. Spring Chinook runs above Trinity Dam historically included an estimated 5,000 fish in the mainstem Trinity and 1,000-5,000 fish in each of four tributaries above Lewiston Dam (Moyle et al. 1995, p.40). Historic run size estimates in each of the Sprague River, Williamson River, Shasta River, and Scott River alone were at least 5,000 fish (CDFG 1990 as cited by Moyle 2002, p.259).

> Runs in the Sprague and Williamson Rivers were probably extirpated before 1900 as the result of dams constructed in Oregon; if any fish remained, they were eliminated with the construction of Copco Dam across the main river in California in 1917. The run in the Shasta River, probably the largest tributary run in the Klamath drainage, disappeared in the early 1930s as a result of habitat degradation and increased summer water temperatures caused by Dwinnell Dam. The smaller Scott River run was extirpated in the early 1970s by a variety of causes (Moyle 2002, p.259).

By the 1980's, habitat alterations had reduced or eliminated much of the cold water habitat and deep pools that spring-run Chinook require resulting in their elimination from much of their former habitat (NRC 2004, p.269). It is estimated that only 3% of the historic habitat available to spring Chinook is currently used by this run (Spring Salmon Summit 2005, p.10). Extant spring run Chinook populations in the Klamath River basin only remain in the Trinity and the Salmon Rivers. Population estimates for spring Chinook during the period from 1980-2006 have ranged from a high of 69,004 fish to fewer than 1,945 (Figure 7). Trinity River Hatchery returns have made up 14-68% of these populations during 1980-2006, and on average comprise 28% of the population.

An average of 10,320 natural spring Chinook spawners have returned to the Klamath River basin annually during the period from 1990-2006, and estimates have ranged from 1,618-35,719 fish (Figure 8). The only substantial wild populations still persisting in the basin are found in the Salmon River (Campbell and Moyle 1991 as cited by Moyle et al. 1995, p.40). Monitoring records of spring Chinook adults in the Salmon River during this period reflect an average of 601 fish returning to the stream annually, with a range from 90 (2005) to 1,485 (1995).

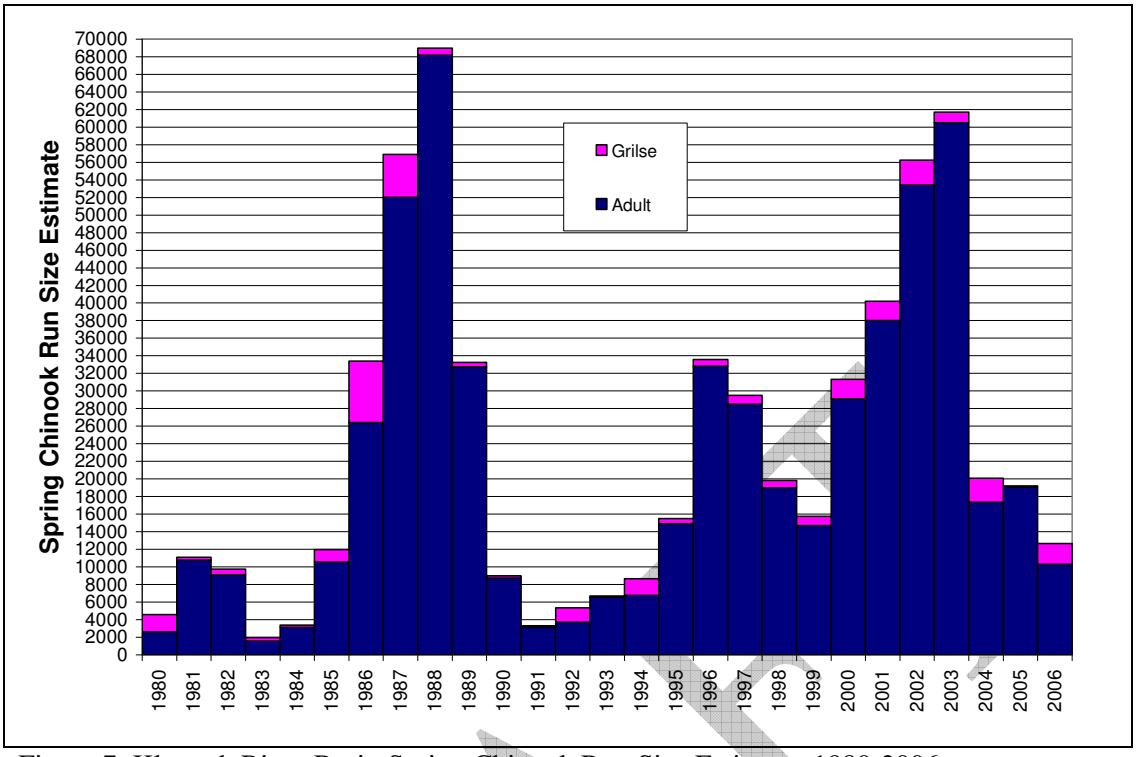

Figure 7: Klamath River Basin Spring Chinook Run Size Estimate, 1980-2006
Note: Run size estimate includes hatchery spawners, natural spawners, and in-river harvest totals. Grilse are Chinook that return to freshwater to spawn after spending only one year in the ocean.
Source: CDFG 2006a

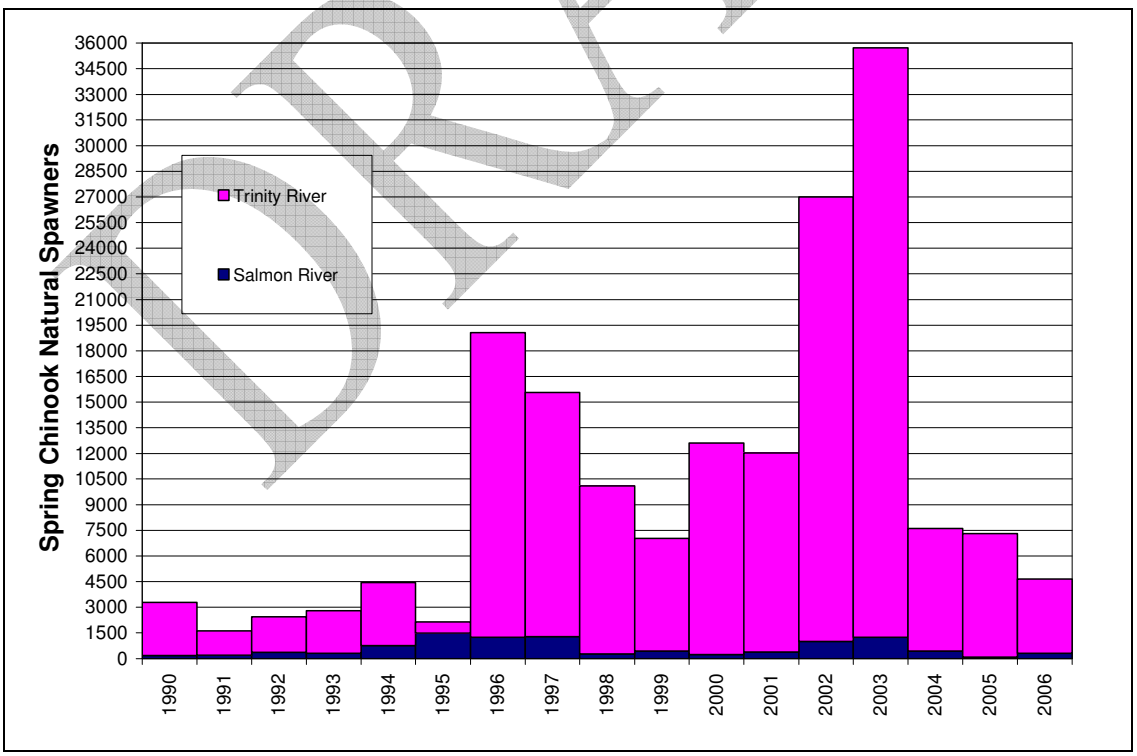

Figure 8: Klamath River Basin Spring Chinook Natural Spawner Estimates, 1990-2006
Source: CDFG 2006a

1.4.2.4 Steelhead Trout

Steelhead are common in the Klamath River basin below Iron Gate Dam where three runs are known to occur: spring/summer, fall, and winter. All three of these runs have a life-history stage called the half-pounder, which is an immature fish that migrates to the sea in the spring but returns to freshwater in the late summer. Fall and winter run steelhead are fairly common in the basin, although they are less abundant than historical levels, while spring/summer steelhead are in danger of extinction (Moyle 2002, p.280). It is likely that steelhead runs exceeded several million fish prior to the 1900s (Hardy and Addley 2006, p.6). An estimate of steelhead spawner abundance by CDFG (1965, p.369) estimated an average of 221,000 steelhead in the Klamath River basin annually.

Hardy and Addley (2006, p.6) state:

> The best quantitative historical run sizes in the Klamath and Trinity river systems were estimated at 400,000 fish in 1960 (USFWS 1960, cited in Leidy and Leidy 1984), 250,000 in 1967 (Coots 1967), 241,000 in 1972 (Coots 1972) and 135,000 in 1977 (Boydston 1977). Busby et al., (1994) reported that the hatchery influenced summer/fall-run in the Klamath Basin (including the Trinity River stocks) during the 1980's numbered approximately 10,000 while the winter-run component of the run was estimated to be approximately 20,000. Monitoring of adult steelhead returns to the Iron Gate Hatchery have shown wide variations since monitoring began in 1963. However, estimates during the 1991 through 1995 period have been extremely low and averaged only 166 fish per year compared to an average of 1935 fish per year for 1963 through 1990 period (Hiser 1994). In 1996, only 11 steelhead returned to Iron Gate Hatchery. The National Marine Fisheries Service (NMFS) considers that based on available information, Klamath Mountain Province steelhead populations are not self-sustaining and if present trends continue, there is a significant probability of endangerment (NMFS 1998); however, steelhead were not listed under the Endangered Species Act of 1973 (ESA).

Annual counts of spring/summer steelhead in holding areas throughout the Klamath River basin have ranged from 500 to 3,000 fish (Roeloffs 1983, as cited by Hopelain 1998, p.1). In the 1990's it was estimated that there were 1000-1500 spring/summer steelhead adults divided among eight populations in the basin (Barnhart 1994, Moyle et al. 1995, Moyle 2002 as cited by NRC 2004, p.274). NMFS considers spring/summer steelhead stocks depressed and in danger of extinction (Busby et al. 1994 as cited by NRC 2004, p.274).

Fall steelhead represent the largest of the three steelhead runs and were estimated to include 55,000-75,000 spawning adults and 150,000-225,000 half-pounders during the period from 1980-1982 (D.P. Lee, CDFG, pers. comm. as cited by Hopelain 1998, p.1).

Run size estimates for winter steelhead were 170,000 in the 1960s, 129,000 in the 1970s, and 100,000 in the 1980s (Busby et al. 1994 as cited by NRC 2004, p.273). Current population estimates for winter steelhead have not been conducted, although Hopelain (1998, p.1) estimated a run-size of about 5,000 to 25,000 during 1980-1982. It is presumed that winter steelhead abundance is still declining although estimates, both past and present, are not very reliable (NRC 2004, p.273).

The following text on hatchery winter steelhead is from NRC (2004, p.272, 273):

> The Iron Gate Hatchery produces about 200,000 and the Trinity River Hatchery about 800,000 winter steelhead smolts per year (Busby et al. 1994). The fish are released into the rivers in the last 2 wk of March, and most reach the estuary about a month later (M. Wallace, CDFG, personal communication), coincident with the emigration of wild smolts. Diets of outmigrating smolts are similar to those of wild smolts, although the consumption of a greater variety of taxa and fewer organisms by the hatchery fish than by wild fish suggests that they have lower feeding efficiency than wild fish (Boles 1990). Otherwise, the interactions between hatchery and wild fish in the Klamath are not known, although hatchery steelhead released into a stream will dominate the wild steelhead (McMichael et al. 1999), potentially increasing the mortality in wild fish from predation, injury, or reduced feeding. Hatchery steelhead also can have adverse effects on juveniles of other salmonids, especially Chinook and coho salmon, through aggressive behavior and predation (Kelsey et al. 2002).
>
> In the 1970s and early 1980s, adults of hatchery origin made up about 8% of the run of Klamath River steelhead and 20-34% of the run in the Trinity River (Busby et al. 1994). As numbers of wild steelhead decline, the percentage of hatchery fish in the population presumably will increase. There is some indication that the runs most heavily influenced by hatchery steelhead in the Trinity River have a lower frequency of half-pounders in the population than do wild populations (Hopelain 1998).

Although steelhead population estimates for the Klamath River basin have not been conducted on a regular basis, adult steelhead return numbers to the Iron Gate Hatchery, Trinity River Hatchery, and Shasta River Fish Counting Facility are available and are presented for the period from 1970-2005 in Figure 9. Adult steelhead returns to these three locations in the basin have ranged from a high of 10,837 fish in 2004 to a low of 529 fish in 1999, with an average return of 3,328 fish over the last 36 years.

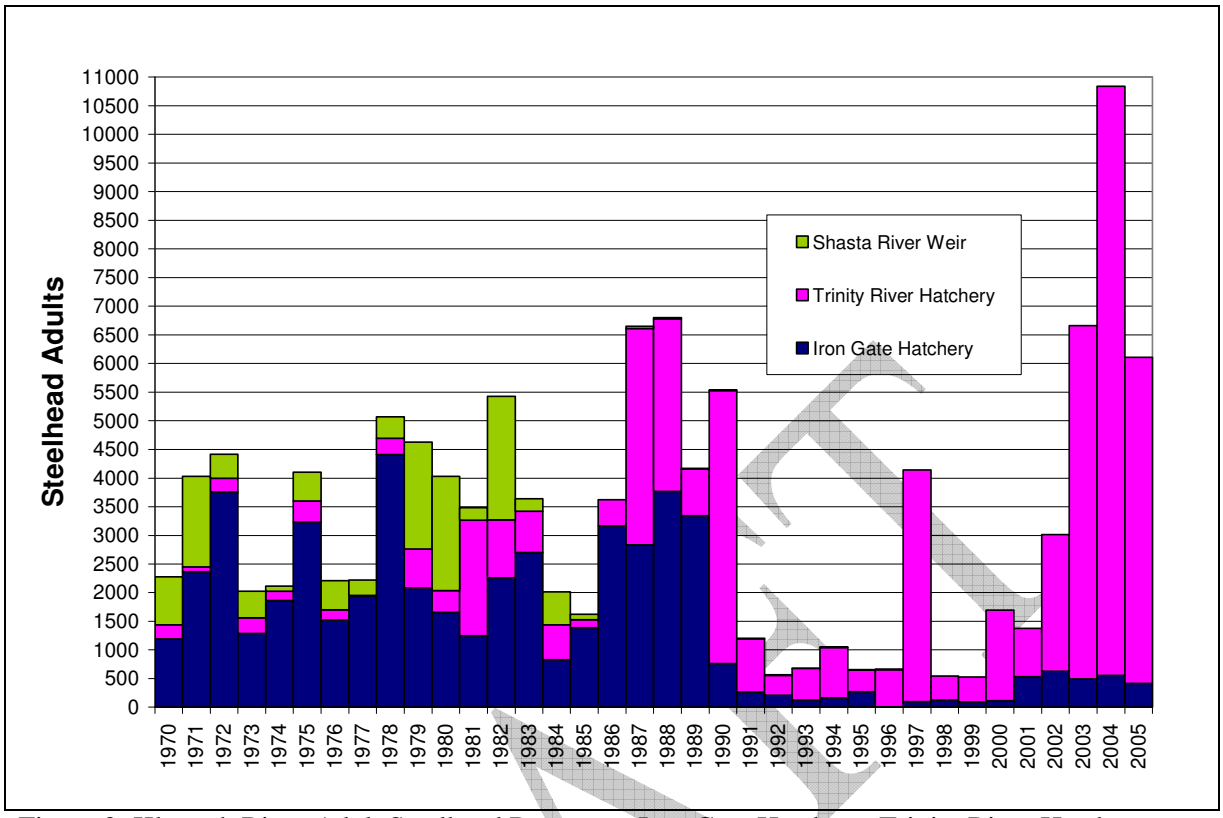

Figure 9: Klamath River Adult Steelhead Returns to Iron Gate Hatchery, Trinity River Hatchery, and the Shasta River Fish Counting Facility, 1970-2005
Note: Steelhead data were not available from the Shasta River Fish Counting Facility for the period of 1997-2005 and thus information is only presented for Iron Gate and Trinity River Hatchery Returns.
Sources: Klamath River Information System (KRIS) 2006; Marshall 2005; and Rushton 2005.

1.4.2.5 Coho Salmon

It is clear from the information available that coho salmon populations statewide have undergone a dramatic decline from historic levels (Brown and Moyle 1991, p.8; Brown et al. 1994; CDFG 2002, p.1). The Southern Oregon/Northern California Coast Evolutionary Significant Unit (SONCC ESU), which encompasses Klamath River basin stocks, has been listed as threatened by the State of California and the federal government. Coho salmon occupy only 61% of the SONCC ESU streams that were previously identified as historical coho salmon streams (CDFG 2002, p.2).

Maximum estimates for coho spawners in California during the 1940's range from 200,000-500,000 fish (Sagar and Glova 1988 as cited by Moyle 2002, p.250). Brown et al. (1994) state that California coho populations are probably less than 6% of what they were in the 1940s, and there has been at least a 70% decline since the 1960s. In 1994, Brown et al. estimated the coho salmon population in California to be 30,000 fish, with natural spawners comprising 43% of the total population or 13,240 fish. This figure is said to be "optimistic because we assumed coho salmon still occur in streams for which there are no current data; it is likely, therefore, that we have underestimated the magnitude of decline (Brown et al. 1994)."

Historical spawning escapement estimates for the Klamath River basin approximate 15,400-20,000 coho, with 8,000 of these fish originating in the Trinity River (USFWS 1979, App. as cited by Brown et al. 1994). In 1965, CDFG estimated 15,400 coho spawners per year in the basin (CDFG 1965, p.369). In 1994, Brown et al. estimated a total abundance of 18,125 coho in the basin, including 1,860 native and naturalized fish. Brown et al. (1994) published the results of presence and absence counts for coho salmon in the basin. Of the 41 tributaries monitored (113 tributaries where they were known to have existed historically), coho were detected in only 21 and absent in the other 20 (Brown et al. 1994). Current population estimates for coho in the Klamath River basin have not been conducted, although adult coho return numbers to the Iron Gate Hatchery, Trinity River Hatchery, and Shasta River Fish Counting Facility are available and are presented for the period from 1964-2005 in Figure 10. Adult coho returns from these combined locations during the last 42 years averaged 5949 fish.

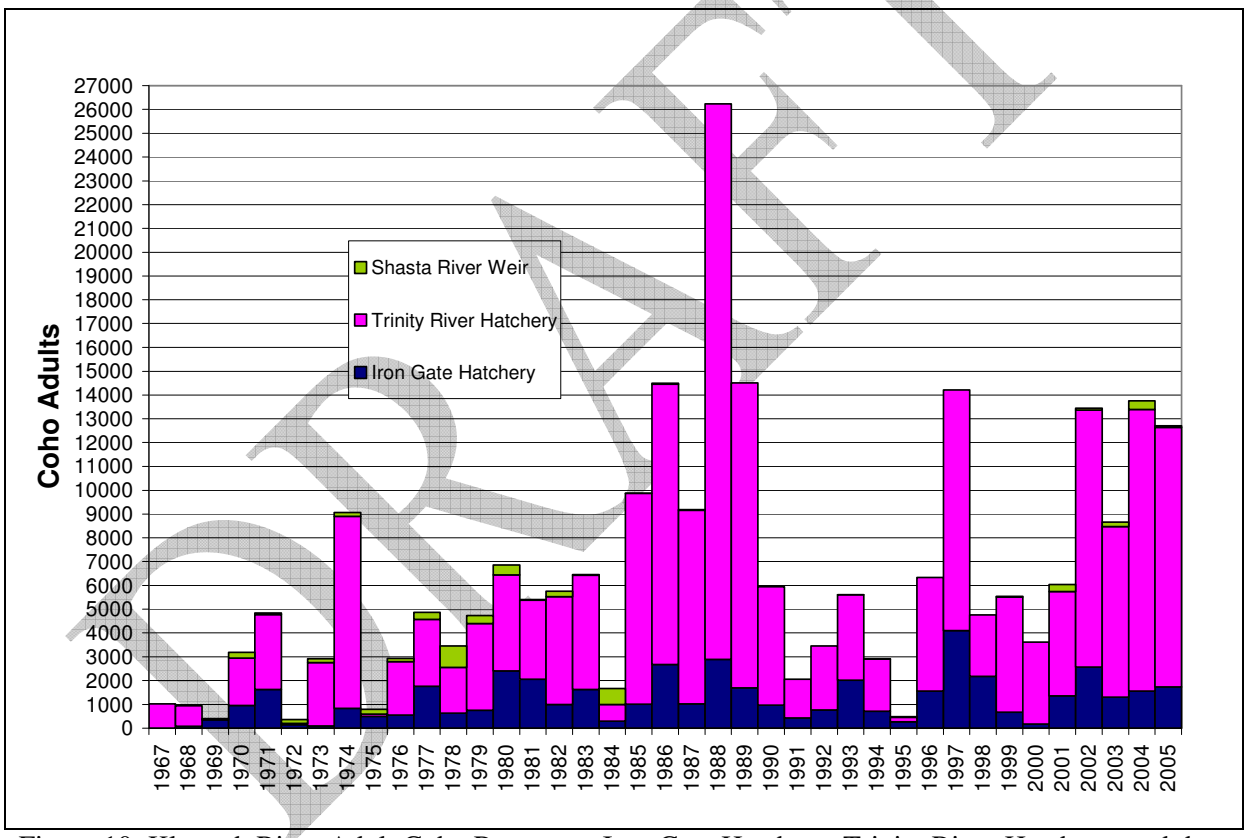

Figure 10: Klamath River Adult Coho Returns to Iron Gate Hatchery, Trinity River Hatchery, and the Shasta River Fish Counting Facility, 1964-2005
Sources: Hampton 2004, p.1; Hampton 2005a, p.1; Hampton 2005b; KRIS 2006; Marshall 2005; and Rushton 2005.

Natural production of coho salmon in the Klamath River basin is considered minimal, with Iron Gate and Trinity River Hatcheries the major sources of most coho salmon in the basin (KRFMC 1991, App. as cited by Brown et al. 1994). The following text from NRC (2004, p.262, 263) discusses hatchery coho in the Klamath River basin:

Coho salmon have been an important part of the Klamath basin fish fauna since prehistoric times (CDFG 2002), and many attempts have been made to augment their populations in the Klamath basin. The first attempt occurred in 1895, when 460,000 fish from Redwood Creek—part of the same evolutionarily significant unit (ESU) as Klamath River coho—were stocked in the Trinity River. It is not known whether these fish, which were taken from a small stream, survived and contributed to later populations. Hatchery production of coho salmon in the Klamath basin began in the 1910-1911 season and continued for another 5 yr. From 1919 to 1942, six additional plants of hatchery-reared fish, all apparently of local origin, were conducted (CDFG 2002). The principal hatcheries today are the Iron Gate Hatchery (operating since 1966) on the Klamath and the Trinity River Hatchery (operating since 1963) on the Trinity River. Faced with a declining egg supply, operators of the two hatcheries at various times brought in fertilized eggs from the Eel and Noyo rivers in California and the Cascade and Alsea rivers in Oregon (CDFG 2002). Thus, present hatchery stocks probably are of mixed origin. Although a few hatchery fish have been planted in tributaries, hatchery fish are for the most part released as smolts into the main stem on the assumption that they will head directly to the sea.

Genetic studies of the contribution of hatchery coho to wild populations in the Klamath basin are not available. Brown et al. (1994) inferred that most wild coho stocks in the basin were partially mixed with hatchery stocks because the two hatcheries are at the far upstream end of coho distribution and produce large numbers of fish. In recent years, the Trinity River Hatchery has released an average of 525,000 coho per year and the Iron Gate Hatchery about 71,000 per year (CDFG 2002), although historically the Iron Gate Hatchery has released about 500,000 coho per year (CDFG, unpublished data, 2002). The coho typically are reared to the smolt stage and marked with a maxillary clip before release, which occurs between March 15 and May 1. They reach the estuary in concert with wild smolts, which peak in late May and early June, but typically are longer than the wild fish—about 170-185 mm vs 135-145 mm (M. Wallace, CDFG, unpublished data, 2002). Although the effect of large numbers of hatchery coho on wild coho is not known for the Klamath, hatchery fish may dominate wild fish when the two are together (Rhodes and Quinn 1998). In any event, hatchery fish are apparently more numerous than their wild counterparts. In 2000 and 2001, 61% and 73%, respectively, of the smolts captured in the estuary were of hatchery origin (M. Wallace, CDFG, unpublished data, 2002).

The percentage of hatchery fish in the spawning population has not been estimated directly, but Brown et al. (1994) estimated that 90% of the adult coho in the system returned directly to the hatcheries or spawned in the rivers in their immediate vicinity. Other hatchery coho no doubt stray into

other streams, but the percentage is not known (CDFG 2002). In a survey of spawning coho in the Shasta River in 2001, individuals from the Iron Gate and Trinity River hatcheries were identified; seven of 23 carcasses examined were hatchery fish (CDFG, unpublished data, 2001). Regardless of origin, natural-spawning coho in the basin's tributaries have managed to maintain timing of runs and other life-history features that fit the basin's hydrologic cycle well.

1.4.3 Periodicity

Adult and juvenile Chinook and steelhead are present year round in the Klamath River basin below Iron Gate Dam (Figures 11 and 13), and mainstem Klamath River below the Dam (Figures 12 and 14). Adult coho are present in the basin from August to February (Figure 15) and in the mainstem Klamath River from September through January (Figure 16), while juvenile coho are present year round. The following sections discuss the presence of salmonids in the Klamath River basin at various life stages throughout the year, known as "periodicity." Data on individual Chinook and steelhead runs in Figures 11, 12, 13, 14 below, are based on readily available information and do not necessarily reflect the entire use period for that run/species. The "All" information rows for Chinook and steelhead represent periods where one or more of the runs are utilizing the basin, and thus is a summary of all information on the individual runs and run timing information for the species in general.

Unless otherwise noted, the text in the following section is primarily from Hardy 1999 (p.5-7), Hardy and Addley 2006 (p.14), NRC 2004 (p.254-258, 264-266, 270-274), and USFWS 1979 (p.16, 27, 29, 30), with additional information from Moyle (2002, p.254), and Salmon River Restoration Council (SRRC) and Klamath Tribe Department of Natural Resources (KTDNR) 2006.

1.4.3.1 Chinook Salmon

Chinook periodicity information for the Klamath River basin below Iron Gate Dam is presented in Figure 11, and information for Chinook periodicity in the mainstem Klamath River is presented in Figure 12.

Fall Chinook are generally slower in their upstream migration than spring Chinook. It takes approximately 2-4 weeks for fall Chinook to reach upstream spawning grounds (USGS 1998 as cited by NRC 2004, p.265). Fall Chinook spawn in the lower reaches of tributaries and in the mainstem Klamath River, although less than 33% spawn in the mainstem. Half of the spawning that occurs in the mainstem takes place in the 13 miles of the river below Iron Gate Dam, although significant spawning occurs as far down as Happy Camp and limited spawning occurs as far downstream as Orleans. Eggs generally incubate for 50-60 days when water temperatures range from 5-14.4 C. Fry move downstream after emergence and often take up residence in shallow water on the edges of the stream in flooded vegetation, where they remain for various lengths of time, although some outmigrate directly to the estuary. The NRC (2004, p.265) reports that juveniles rearing in the Klamath River or larger tributaries reside there for 3-9 months but move

downstream continuously. Shaw et. al. (1998, p.29) state that juvenile rearing in the mainstem between Iron Gate Dam and Seiad Creek is likely to occur year round. Belchik (1997) reports that rearing juvenile salmonids reside in pockets of cool water, thermal refugia, in the Klamath River when mainstem water temperatures become unsuitable during the summer. Type I, II, and III Chinook exist in the Klamath River basin and thus outmigration occurs year round. Juvenile fall Chinook move into the estuary at a smaller size and reside there longer when conditions in the river are unfavorable, such as times of warm water temperatures. It appears that many juveniles leave the estuary after only a few weeks, and outmigrate to the ocean (Wallace 2000 as cited by NRC 2004, p.265).

Spring Chinook are thought to migrate more deliberately and further upstream than fall Chinook. Spring Chinook migrate as far upstream as they can go in larger tributaries, which allows them to access habitat often inaccessible to fall Chinook due to low flows and high temperatures during the summer and fall. Spring Chinook are persistent in their upstream migration and don't rest until they have reached holding areas where they remain until they spawn. Migrating spring Chinook hold in deep pools for 2-

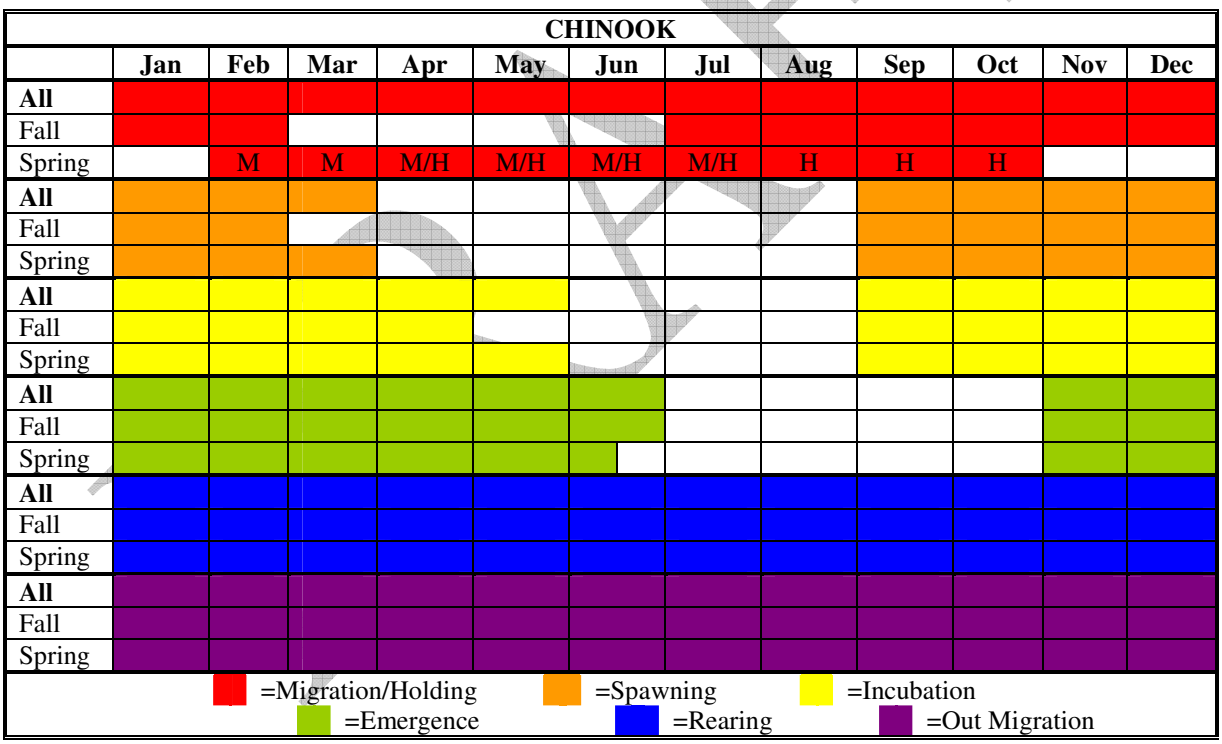

Figure 11: Chinook Periodicity in the Klamath River Basin
M=Migration, M/H=Migration and Holding, H=Holding
Sources: Hardy 1999, p.6, 7, & 34; Hardy and Addley 2006, p.96; KRBFTF 1991, p.4-8, 4-9, 4-12; Leidy and Leidy 1984; NRC 2004, p.269; Olson per comm., as cited by West 1991, p.9; PacifiCorp 2004, p.4-25; Shaw et al. 1998; Snyder 1931, p.19; SRRC and KTDNR 2006; Scott River Watershed Council (SRWC) 2004, p.6-17; Trihey and Associates, Inc. 1996, p.12, 17; USFWS 1979, p.16; USFWS 1999, p.19, 38; USFWS 2001, p. 13, 22; West 1991, p.9.

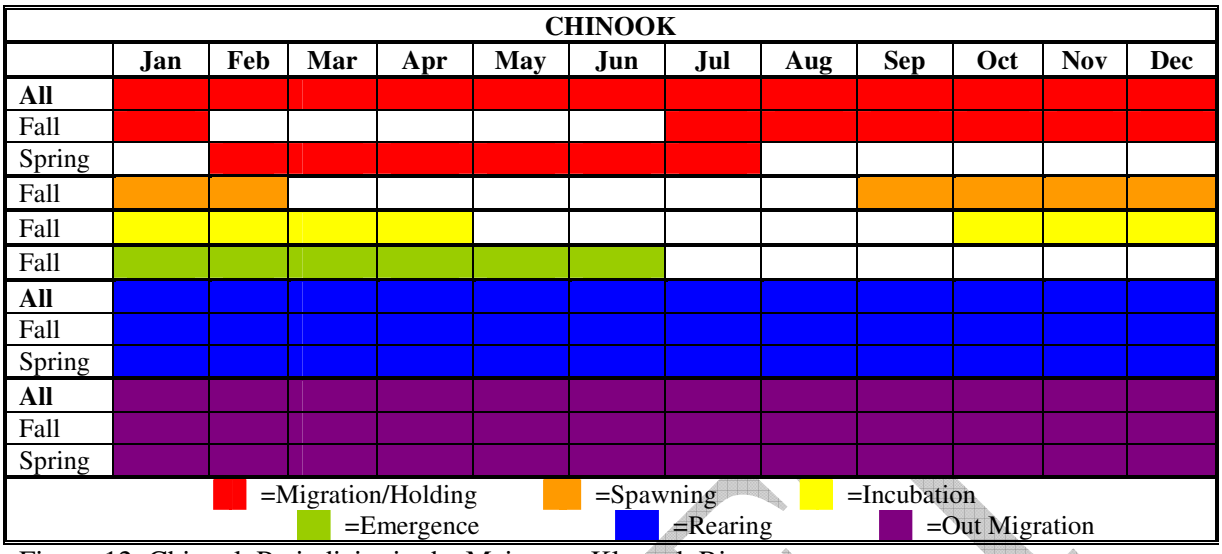

Figure 12: Chinook Periodicity in the Mainstem Klamath River
Sources: Hardy 1999, p.34; Hardy and Addley 2006, p.96; NRC 2004, p.264-267; Rushton 2005; Shaw et al. 1998; USFWS 1979, p.16; USFWS 1999, p.16; USFWS 2001, p. 59, 62, 65, & 68

4 months (throughout the summer) as their gonads fully develop, and then spawn the following fall and winter. Spring Chinook are susceptible to high water temperatures that can result in decreased fecundity of females (decreases egg viability) as they hold throughout the summer (McCullough 1999 as cited by NRC 2004, p.269). Incubation takes approximately 40-60 days, and alevin and fry remain in the gravel for 2-4 weeks before emergence. Spring Chinook will typically rear in freshwater for a year after emergence before heading to the ocean (Healey 1991 as cited by NRC 2004, p.268).

1.4.3.2 Steelhead Trout

Information on steelhead periodicity in the Klamath River basin below Iron Gate Dam is presented in Figure 13, and steelhead periodicity in the mainstem Klamath River is presented in Figure 14.

With the exception of half-pounders, steelhead remain in the ocean for 1-3 years before initiating their spawning run and may spawn 3-4 times during their life. Incidence of repeat spawning reported by Hopelain (1998, p.21) were 17.6-47.9% for fall steelhead, 40-63.3% for spring/summer steelhead, and 33.1% for winter steelhead. Although steelhead generally use the mainstem Klamath River as a migration corridor, some spawning does occur in the mainstem. The mainstem is also very important to the juvenile rearing life stage of steelhead. Fall run steelhead typically enter the Klamath River basin during the summer and hold for several months before moving to spawning areas in smaller tributaries. The early part of the fall steelhead run consists primarily of half-pounders. Franklin (2006) notes that half-pounders have entered the Klamath River as early as July, although most references cite migration beginning in August. Spring/summer steelhead enter the Klamath River in early spring and hold in deep pools until they spawn the following winter. High water temperatures can decrease the

viability of eggs in female spring/summer steelhead holding throughout the summer and fall. Steelhead eggs typically incubate for 4 –7 weeks, although the length of time for incubation is a function of water temperature, taking longer in cooler temperatures.

STEELHEAD	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
All												
Spring/summer	H	H	H	M/H	M/H	M/H	M/H	H	H	H	H	H
Fall	H						M	M/H	M/H	M/H	M/H	H
Winter												
Half-pounder												
All												
Spring/summer												
Fall												
Winter												
All												
winter												
All												
Winter												
All												
Winter												
Half-pounder												
All												
Spring/summer												
Fall/winter												

=Migration/Holding =Spawning =Incubation
=Emergence =Rearing =Out Migration

Figure 13: Steelhead Periodicity in the Klamath River Basin
M=Migration, M/H=Migration and Holding, H=Holding
Sources: Barnhart 1994 as cited by NRC 2004, p.271; Hardy 1999, p.5, 6 & 34; Hardy and Addley 2006, p.96 ; Hopelain 1998, p.1; Leidy and Leidy 1984; NRC 2004, p.271, 272; Shaw et al. 1998; Trihey and Associates, Inc. 1996, p.13; USFWS 1979, p.29, 30; USFWS 1999, p.28, 49; USFWS 2001, p.36, 44.

STEELHEAD	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
All												
Spring/summer												
Fall												
Winter												
Half-pounder												
All												
All												
All												
All												
All												

=Migration/Holding =Spawning =Incubation
=Emergence =Rearing =Out Migration

Figure 14: Steelhead Periodicity in the Mainstem Klamath River
Sources: Hardy 1999, p.34; Hardy and Addley 2006, p.96; Hopelain 1998, p.12; NRC 2004, p.271-273; Rushton 2005; Shaw et al. 1998; USFWS 1979, p.29 & 30; USFWS 1999, p.27; USFWS 2001, p. 61, 64, 67, & 70.

After emergence, fall and winter steelhead juveniles distribute themselves widely throughout the basin, and many move out of the tributaries and into the mainstem to rear (NRC 2004, p.271). Juvenile spring/summer steelhead typically occupy the same upper stream reaches where they were spawned. Shaw et al. (1998, p.31) report that young of the year steelhead emigrate to the mainstem and most likely rear there for a year before emigrating as two year olds. Cool water areas, thermal refugia, of the mainstem Klamath River are utilize by rearing juvenile salmonids during the summer once mainstem temperatures become unsuitably warm (Belchik 1997). Juvenile steelhead normally spend 2 years in freshwater before they enter the ocean, although some emigrate after 1 or 3 years.

1.4.3.3 Coho Salmon

Coho periodicity information for the Klamath River basin below Iron Gate Dam is presented in Figure 15, and information on coho periodicity in the mainstem Klamath River is presented in Figure 16.

In the Klamath River basin, coho salmon have a 3-year lifecycle during which they spend 1-1½ years in freshwater before moving to the ocean, and then return to the river to spawn at age 3. Occasionally males, called "jacks", will return to the river to spawn as 2-year-olds. Coho upstream migrations are typically linked to pulse flows associated with rain events in the basin. They generally spawn in tributaries, however they have been observed spawning at tributary confluences, in side channels, and along the shoreline of the mainstem Klamath River. Eggs incubate for approximately 7 weeks and alevins remain in the gravel for 2-3 weeks before emerging.

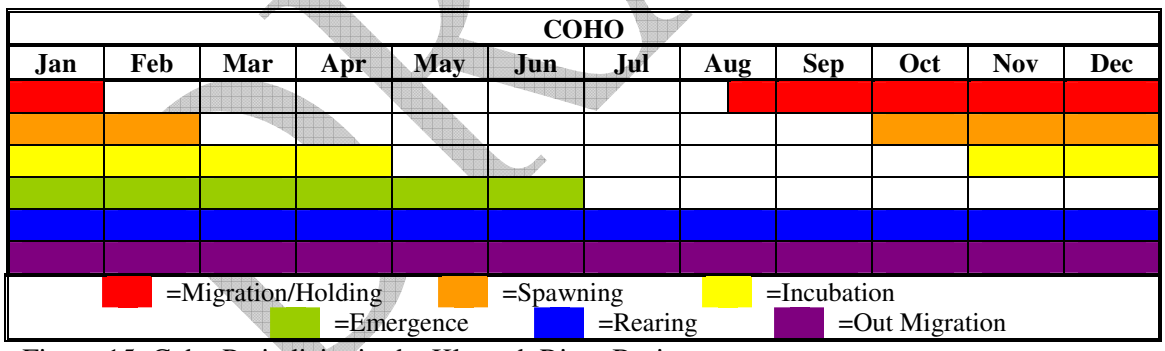

Figure 15: Coho Periodicity in the Klamath River Basin

Sources: Hardy 1999, p.6 & 34; Hardy and Addley 2006, p.96; Leidy and Leidy 1984; NRC 2004, p.7, 8; Shaw et al. 1998; Snyder 1931, p.16, 23; SRRC and KTDNR 2006; SRWC 2004, p.6-17; Trihey and Associates, Inc. 1996, p.13, 17; USFWS 1979, p.27; USFWS 1999, p.26, 43; USFWS 2001p.32, 40.

Upon emergence from the gravels coho juveniles seek areas of low velocity with an abundance of food, such as the stream margins. The NRC reports that coho juveniles live in the mainstem Klamath River despite temperatures that regularly exceed 24C (M. Rhode, CDFG, personal communication, USFWS, unpublished data, 2002 as cited by NRC 2004, p.257). These juveniles are mainly found in pools at the mouths of tributaries where temperatures are 2-6C lower than in the mainstem. Belchik (1997) reports that

cool water areas, thermal refugia, of the mainstem Klamath River are utilize by rearing juvenile salmonids, including coho, during the summer once mainstem temperatures become unsuitably warm. The Karuk tribe have collected data which shows that coho use the mainstem Klamath River even during the hottest periods of the year if suitable thermal refugia is available (Corum 2006). Shaw et. al. (1998, p.30) states that coho likely rear in the mainstem Klamath River between Iron Gate Dam and Seiad Creek year round, although they do not necessarily inhabit the mainstem on a continuous basis due to the high bioenergetic demands. Coho juveniles typically rear in freshwater for 1 year before outmigration occurs.

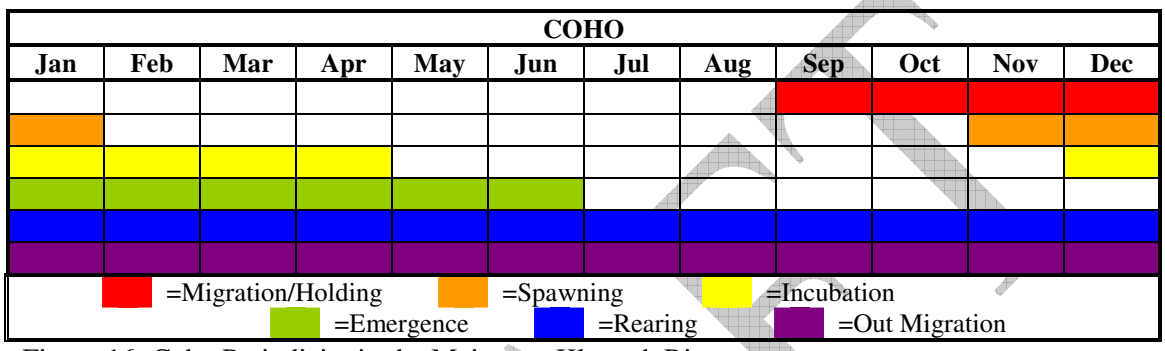

Figure 16: Coho Periodicity in the Mainstem Klamath River
Sources: Hardy 1999, p.34; NRC 2004, p.254, 255, 258, & 259; Rushton 2005; Shaw et al. 1998; USFWS 1979, p.27, USFWS 1999, p.23; USFWS 2001, p. 60, 63, 66, & 69.

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Klamath River Model for TMDL Development

September 2008

Prepared for:

U.S. Environmental Protection Agency Region 9
U.S. Environmental Protection Agency Region 10
Oregon Department of Environmental Quality
North Coast Regional Water Quality Control Board

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Model Configuration and Results

Klamath River Model for TMDL Development

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Prepared for:
U.S. Environmental Protection Agency Region 9
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Prepared by:
Tetra Tech, Inc.

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1.0 INTRODUCTION

The Klamath River watershed traverses the states of Oregon and California, encompassing an area of approximately 15,722 square miles. The headwaters of the Klamath River originate in the Cascade Mountains and the river flows to the southwest from Oregon into northern California toward its confluence with the Pacific Ocean (Figure 1-1). Major tributaries to the Klamath River include the Lost River Diversion Channel and the Shasta, Scott, Salmon, and Trinity Rivers.

The watershed includes portions of Jackson, Josephine, Klamath, and Lake Counties in Oregon, and Del Norte, Humboldt, Modoc, Siskiyou, and Trinity Counties in California. Nearly 63 percent of the watershed (approximately 9,933 square miles) lies in California, while 37 percent (5,727 square miles) is in Oregon. The Klamath River watershed includes twelve U.S. Geological Survey (USGS) 8-digit hydrologic cataloging units, numbers 18010201 through 18010212.

The Oregon Department of Environmental Quality (ODEQ) and California's North Coast Regional Water Quality Control Board (NCRWQCB) have both included the Klamath River on their corresponding Clean Water Act section 303(d) lists as a result of observed water quality criteria exceedances. Impairments include dissolved oxygen (DO), chlorophyll *a*, temperature, pH, and ammonia for various portions of the Klamath River and its tributaries in Oregon and nutrients, temperature, and organic enrichment/low DO for segments of the river and its tributaries in California.

The states are required to develop total maximum daily loads (TMDLs) for applicable water quality parameters. The TMDL process identifies the maximum load of a pollutant a waterbody is able to assimilate and still fully support its designated uses. The TMDL process also allocates portions of the allowable load to all sources, identifies the necessary controls that may be implemented voluntarily or through regulatory means, and describes a monitoring plan and associated corrective feedback loop to ensure that uses are fully supported. Watershed and water quality modeling is often used during the development of TMDLs to help with one or more of these tasks. Modeling is a quantitative approach to better understand complex environmental processes and relationships. Models can be used to help fill in gaps in observed water quality data, estimate existing pollutant sources throughout a watershed, calculate allowable loads, and assess the potential effectiveness of various control options.

The first steps in the TMDL development process were previously conducted, including compiling available data; evaluating monitoring data to identify the extent, location, and timing of water quality impairments; and developing a technical approach to analyze the relationship between source pollutant loading contributions and in-stream response. These steps were detailed in *Data Review and Modeling Approach—Klamath and Lost Rivers TMDL Development* (Tetra Tech, Inc. 2004). Subsequent steps include model configuration, model testing (calibration), and scenario analysis. This document discusses the configuration of the Klamath River model and presents modeling results for the Klamath River from Upper Klamath Lake (UKL) to the river's outlet at the Pacific Ocean for the river and reservoir segments (2000 and 2002) and estuarine segment (2004).

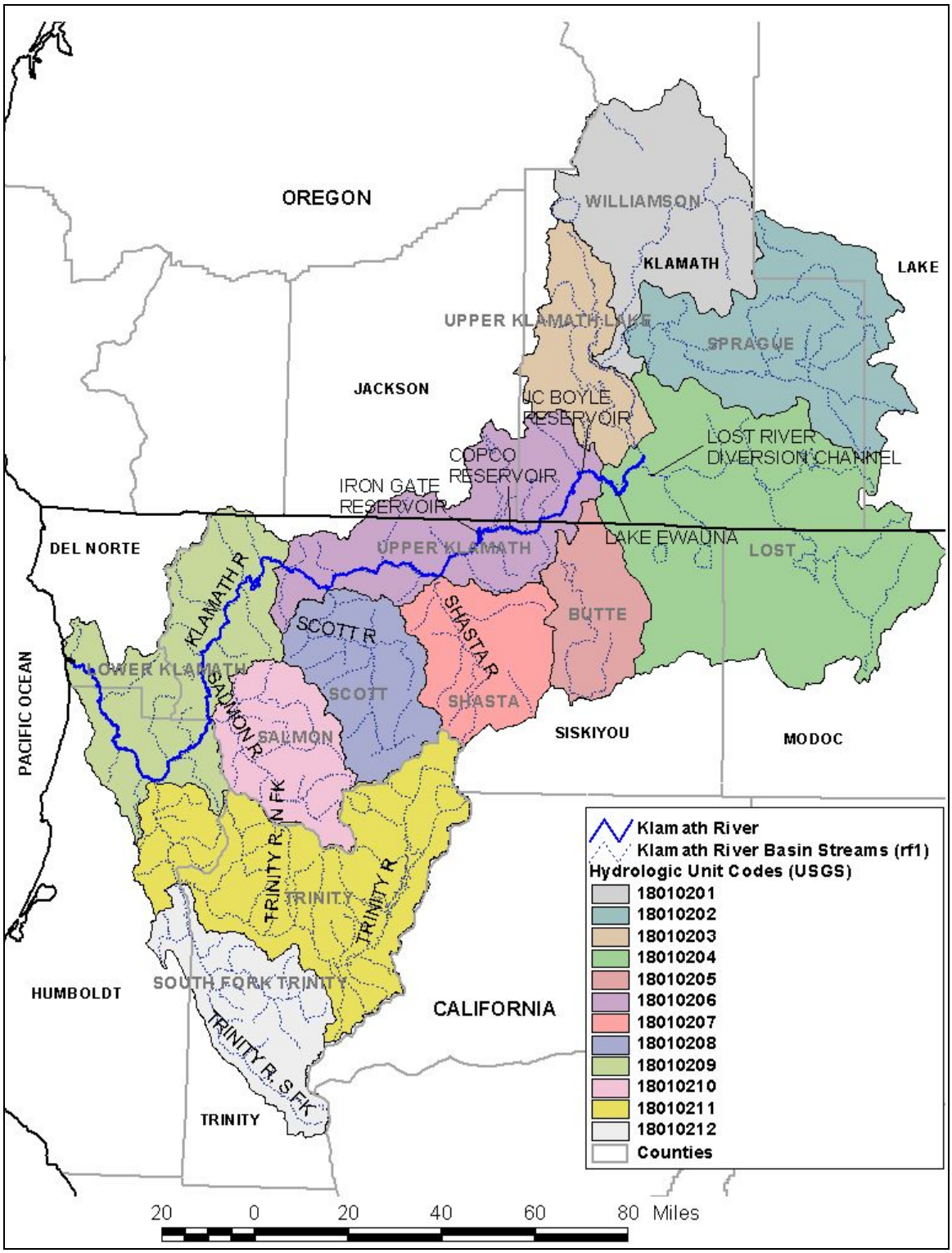


Figure 1-1. Extent of the Klamath River watershed

2.0 MODELING APPROACH

2.1 Model Selection

To support TMDL development for the Klamath River system, the need for an integrated receiving water hydrodynamic and water quality modeling system was identified. The following model capabilities were identified as being necessary to support TMDL development. The model must be

- Capable of predicting hydrodynamics, nutrient cycles, DO, temperature, pH, and other parameters and processes pertinent to the TMDL development effort;
- Capable of simulating the multiple flow control structures along the length of the Klamath River;
- Dynamic (time-variable) and thus capable of representing the highly variable flow and water quality conditions within years and between years;
- Capable of considering the steep channel slope of the Klamath River; and
- Capable of representing conditions in the Klamath Estuary.

A model for the Klamath River had already been developed by PacifiCorp to support studies for the Federal Energy Regulatory Commission Hydropower relicensing process (Watercourse Engineering, Inc. 2004) when this project began. The version of the model available in 2004 is hereafter referred to as the *PacifiCorp Model*. The PacifiCorp Model and other models, including the operational models MODSIM and CALSIM, were evaluated for applicability to Klamath River TMDL development (Tetra Tech, Inc. 2004). NCRWQCB, ODEQ, and the U.S. Environmental Protection Agency (EPA) determined that the existing PacifiCorp Model would provide the optimal basis, after making some enhancements, for TMDL model development. Section 2.2 provides a description of the enhancements made to the PacifiCorp Model. It should be noted that PacifiCorp has since updated the model after receiving comments from reviewers (PacifiCorp 2005).

The original PacifiCorp Model consisted of Resource Management Associates (RMA) RMA-2 and RMA-11 models and the CE-QUAL-W2 model. Specifically, the RMA-2 and RMA-11 models were applied for Link River (which is the stretch of the Klamath River from UKL to Keno Dam), Keno Dam to J.C. Boyle Reservoir, Bypass/Full Flow Reach, and Iron Gate Dam to Turwar. RMA-2 simulates hydrodynamics, while RMA-11 represents water quality processes. The CE-QUAL-W2 model was applied for Lake Ewauna-Keno Dam, J.C. Boyle Reservoir, Copco Reservoir, and Iron Gate Reservoir. In addition to addressing the model needs identified above, the PacifiCorp Model

- Uses hydrodynamic and water quality models with a proven track record in the environmental arena, including historical application to the Klamath River;
- Has already been reviewed by most stakeholders in the watershed;
- Can be directly compared to ODEQ, NCRWQCB and tribal water quality criteria;
- Has been preliminarily calibrated for the Klamath River and its applicability demonstrated; and
- Uses the public domain model CE-QUAL-W2 and a version of RMA that can be distributed to the public for purposes of TMDL application.

Because the estuarine portion of the Klamath River (Turwar to the Pacific Ocean) was not included in the original PacifiCorp Model, it was necessary to identify a model appropriate for modeling that portion of the river. After reviewing a 2004 bathymetric survey and grab, multiprobe, and cross-sectional profile data, it was determined that a laterally averaged 2-D model, such as CE-QUAL-W2, was not the ideal choice for modeling the estuarine portion of the Klamath River.

Hydrodynamics and water quality within the estuary are highly variable spatially and throughout the year and are greatly influenced by time of year, river flow, tidal cycle, and location of the estuary mouth (which changes because of sand bar movement). Additionally, transect temperature and salinity data in the lower estuary showed significant lateral variability, as did DO to a lesser extent. The Environmental Fluid Dynamics Code (EFDC), which is a full 3-D hydrodynamic and water quality model, was selected to model the complex estuarine environment instead.

EFDC is an EPA-endorsed and widely applied 3-D model (particularly for TMDL development). EFDC allows for representing the complex geometry of the Klamath Estuary with a boundary-fitted, curvilinear grid. The model is capable of simulating important physical processes and features, such as the circulation pattern near the funnel-shaped mouth and islands. The mouth of the estuary is very wide; however, it does not open to the Pacific Ocean completely because of the presence of a sand bar. As a result, the estuary can communicate with the ocean only through a very narrow opening in the sand bar. Configuring a CE-QUAL-W2 grid for this portion of the system would likely result in a very wide segment at the downstream end of the river. The wide segment would be linked to a very narrow segment representing the opening in the sand bar. This configuration runs the risk of resulting in computational error because of the sudden change in segment width at the most dynamic portion of the system (Cole and Wells, 2003). Although it is impossible to simulate the evolution of the existing sand bar at the estuary mouth with available technology, EFDC can potentially be used to efficiently evaluate the implications of mouth locations on hydrodynamics and water quality.

Hydrodynamics and water quality in the estuary are also highly variable throughout the year and are greatly influenced by time of year, river flow, tidal cycle, and location of the mouth. Hydrodynamic data also show significant lateral variability, as does DO to a lesser extent. It is desirable to apply a model that has the potential to represent this variability and EFDC is capable.

Additional factors leading to the choice of EFDC for modeling the estuary include the following:

- EFDC is capable of predicting hydrodynamics, nutrient cycles, DO, temperature, and other parameters and processes pertinent to the TMDL development effort for the estuarine section.
- The EFDC model is dynamic (time-variable) and thus capable of representing the highly variable flow and water quality conditions within years and between years.
- EFDC has a proven track record in the environmental arena—particularly with regard to TMDLs.
- Model results can be directly compared to ODEQ, NCRWQCB and tribal water quality criteria.
- EFDC is EPA-endorsed and supported and is included in the EPA TMDL Modeling Toolbox. It is a public domain model, fully transparent (i.e., model code), and is available free of charge. EPA also provides training and support on the application free of charge.

- EFDC has a function for blocking flows between computational cells, and this allows for efficiently evaluating the effect of the location and size of the sand bar opening, with a single grid configuration.
- The EFDC water quality module possesses a fully numerical sediment diagenesis module to predict sediment oxygen demand (SOD) and benthic nutrient flux based on organic loading to the waterbody. This improves the reliability of the model for DO and nutrient TMDLs. Although this component was not used in this study because of time and data limitations, it can be implemented in the future using the existing framework when sufficient data become available.

The combination of the enhanced PacifiCorp Model (RMA and CE-QUAL-W2) and EFDC resulted in the Klamath River model used for TMDL development. Table 2-1 identifies the modeling elements applied to each river segment.

Table 2-1. Model components applied to each Klamath River segment

Modeling segment	Segment type	Model(s)	Dimensions
Link River	River	RMA-2/RMA-11	1-D
Lake Ewauna-Keno Dam	Reservoir	CE-QUAL-W2	2-D
Keno Dam to J.C. Boyle Reservoir	River	RMA-2/RMA-11	1-D
J.C. Boyle Reservoir	Reservoir	CE-QUAL-W2	2-D
Bypass/Full Flow Reach	River	RMA-2/RMA-11	1-D
Copco Reservoir	Reservoir	CE-QUAL-W2	2-D
Iron Gate Reservoir	Reservoir	CE-QUAL-W2	2-D
Iron Gate Dam to Turwar	River	RMA-2/RMA-11	1-D
Turwar to Pacific Ocean	Estuary	EFDC	3-D

The linkages between the riverine and reservoir/estuary models identified in Table 2-1 were made by transferring time-variable flow and water quality from one model to the next (e.g., output from the Link River model became input for the Lake Ewauna-Keno Dam model). This modeling framework is consistent with available models appropriate for application to riverine/reservoir systems and is based on the PacifiCorp Model's existing modeling approach for the river system.

Each model type included (RMA, CE-QUAL-W2, and EFDC) is discussed in more detail below.

RMA

The hydrodynamic component of the RMA modeling suite, RMA-2, is a model specifically designed to assess flow response in complex river systems (Deas 2000). RMA-2 solves the full-flow equations, known as the St. Venant Equations. These equations use all terms of the conservation of momentum formulation and provide a complete description of dynamic flow conditions. The model has been widely applied (it is one of the most used full hydrodynamic models in the United States) to a variety of river and estuary systems in the United States as well as internationally.

The water quality component, RMA-11, is a general-purpose water quality model, compatible in geometry with the configuration of the RMA-2 hydrodynamic model. The model simulates advective heat transport and air-water heat exchange processes, as well as fate and transport of water quality parameters (e.g., nutrients), to produce dynamic descriptions of temperature and constituent concentration along the river reach. Input requirements include temperatures and quality of boundary flows, and meteorological data defining atmospheric conditions governing heat exchange at the air-water interface. Model output is in the form of longitudinal profiles of temperature and water quality parameters along river reaches, or time series at fixed locations.

CE-QUAL-W2

The U.S. Army Corps of Engineers' CE-QUAL-W2 is a 2-D, longitudinal/vertical (laterally averaged), hydrodynamic and water quality model (Cole and Wells, 2003). The model allows for application to streams, reservoirs, and estuaries with variable grid spacing, time-variable boundary conditions, and multiple inflows and outflows from point/nonpoint sources and precipitation.

The two major components of the model include hydrodynamics and water quality kinetics, which simulate changes in constituent concentrations. Both of these components are coupled (i.e., the hydrodynamic output is used to drive the water quality at every timestep). The hydrodynamic portion of the model predicts water surface elevations, velocities, and temperature. The ULTIMATE-QUICKEST numerical scheme used in the CE-QUAL-W2 model is designed to reduce the numerical diffusion in the vertical direction to a minimum and in areas of high gradients, reduce the undershoots and overshoots that could produce small negative concentrations. The water quality kinetics portion can simulate 21 water quality parameters including DO, nutrients, phytoplankton interactions, and pH.

EFDC

EFDC is a general purpose modeling package for simulating 1-D, 2-D, and 3-D flow, transport, and biogeochemical processes in surface water systems including rivers, lakes, estuaries, reservoirs, wetlands, and coastal regions. The EFDC model was originally developed at the Virginia Institute of Marine Science for estuarine and coastal applications. This model is now being supported by EPA and has been used extensively to support TMDL development throughout the country. In addition to hydrodynamic, salinity, and temperature transport simulation capabilities, EFDC is capable of simulating cohesive and non-cohesive sediment transport, near-field and far-field discharge dilution from multiple sources, eutrophication processes, the transport and fate of toxic contaminants in the water and sediment phases, and the transport and fate of various life stages of finfish and shellfish. Cohesive sediment refers to silt and clay particles while non-cohesive refers to anything larger than silt (e.g., sand, gravel). The EFDC model has been extensively tested, documented, and applied to environmental studies worldwide by universities, governmental agencies, and environmental consulting firms.

The structure of the EFDC model includes four major modules: (1) a hydrodynamic model, (2) a water quality model, (3) a sediment transport model, and (4) a toxics model. The water quality portion of the model simulates the spatial and temporal distributions of 22 water quality parameters including DO, suspended algae (3 groups), attached algae, various components of carbon, nitrogen, phosphorus and silica cycles, and fecal coliform bacteria. Salinity, water temperature, and total suspended solids are needed for computation of the 22 water quality parameters, and they are simulated by the hydrodynamic model.

EFDC's water quality model also includes a sediment process model, which uses a slightly modified version of the Chesapeake Bay 3-D model (Park et al. 1995). Upon receiving the particulate organic matter (OM) deposited from the overlying water column, it simulates its diagenesis and the resulting fluxes of inorganic substances (ammonium, nitrate, phosphate and silica) and SOD back to the water column. The coupling of the sediment process model with the water quality model not only enhances the model's predictive capability of water quality parameters, but also enables it to simulate the long-term changes in water quality conditions in response to changes in nutrient loads.

2.2 Model Enhancements

Although the original PacifiCorp Model (Watercourse Engineering, Inc. 2004) is capable of addressing the identified water quality issues, a number of enhancements to the model were necessary to expedite and strengthen the model for the rigors of TMDL development for the Klamath River.

Selected algorithms in the PacifiCorp Model were considered for augmentation of the modeling framework to address specific processes and support TMDL development. Enhancements were made in the following areas:

- BOD/OM unification
- Algae representation in Lake Ewauna
- Monod-type continuous SOD and OM decay
- pH simulation in RMA
- OM-dependent light extinction simulation in RMA
- Reaeration formulations
- Dynamic OM partitioning

2.2.1 BOD/Organic Matter Unification

It was determined that biochemical oxygen demand (BOD) should not be modeled in addition to OM because BOD itself is a surrogate for OM. The BOD compartment in the modeling system was eliminated for both the riverine (RMA model component) and reservoir (CE-QUAL-W2 model component) sections. To maintain consistency between the new version of the model and the original version in terms of organic loading, the BOD concentration in the original version was converted to an OM component by using a stoichiometric ratio of BOD:OM = 1.4. This stoichiometric coefficient was derived on the basis of the original RMA-11 model and is also consistent with representation in the CE-QUAL-W2 model. This converted OM was combined with the OM in the original version to form the total OM in the new version. This conversion allowed the model to represent consistent amounts of OM in the original and new versions.

Additionally, in the original CE-QUAL-W2 models for the reservoirs, particulate OM was not included in the tributary boundary condition files. This is expected to result in an underestimation of particulate OM into the system. Therefore, for the major tributaries that are highly productive, such as the Lost River Diversion Channel, particulate OM loading was represented on the basis of data and appropriate assumptions. Concentration boundary condition files were modified using a labile particulate OM (LPOM) to labile dissolved OM (LDOM) ratio of 4.0 (LPOM:LDOM = 0.8:0.2), which is same as for the Link River boundary condition. Labile OM refers to the portion that is decomposed relatively quickly.

2.2.2 Algae Dynamics in Lake Ewauna

Very low levels of phytoplankton were observed in lower Lake Ewauna, as shown in the year 2000 monitoring data for the Highway 66 station. Phytoplankton biomass in Lake Ewauna also shows significant variability from upstream (Miller Island water quality station) to downstream (Highway 66 water quality station). In addition, observed data show that there was a sharp drop in algae concentration at the Miller Island station during the summer. These phenomena were not

predicted by the existing PacifiCorp Model, although it is configured to simulate all algae-DO-nutrient interactions represented in the W2 model code.

Simulated algae biomass in the existing PacifiCorp Model is similar at both locations, and algae concentrations remained high throughout the summer period. This is due to the dominant upstream inflow (from UKL) that causes water to flow quickly from upstream to downstream. The inability to accurately predict the temporal and spatial distribution of phytoplankton has significant implications on the water quality dynamics. Therefore, model refinement was needed to better represent the algae dynamics in this lake.

It appears that with the existing kinetic structure in the model, it is not possible to reproduce this type of spatial distribution of algae biomass. Sometimes during the summer period, the entire Lake Ewauna water column becomes hypoxic (exhibiting very low DO concentrations), and even anoxic (exhibiting DO concentrations near zero). Figure 2-1 shows the observed DO data, along with chlorophyll *a*, at the Miller Island and Highway 66 locations for 2000 (at approximately 1 meter from the surface). It is believed that the summer hypoxia/anoxia might directly or indirectly influence the spatial variability of algae biomass in Lake Ewauna. Algal mortality has been causally linked with low DO in UKL (National Research Council 2004) and found to be related to anoxic conditions in other systems (Baric et al. 2003). Available data show no other explanation for the observed phenomenon.

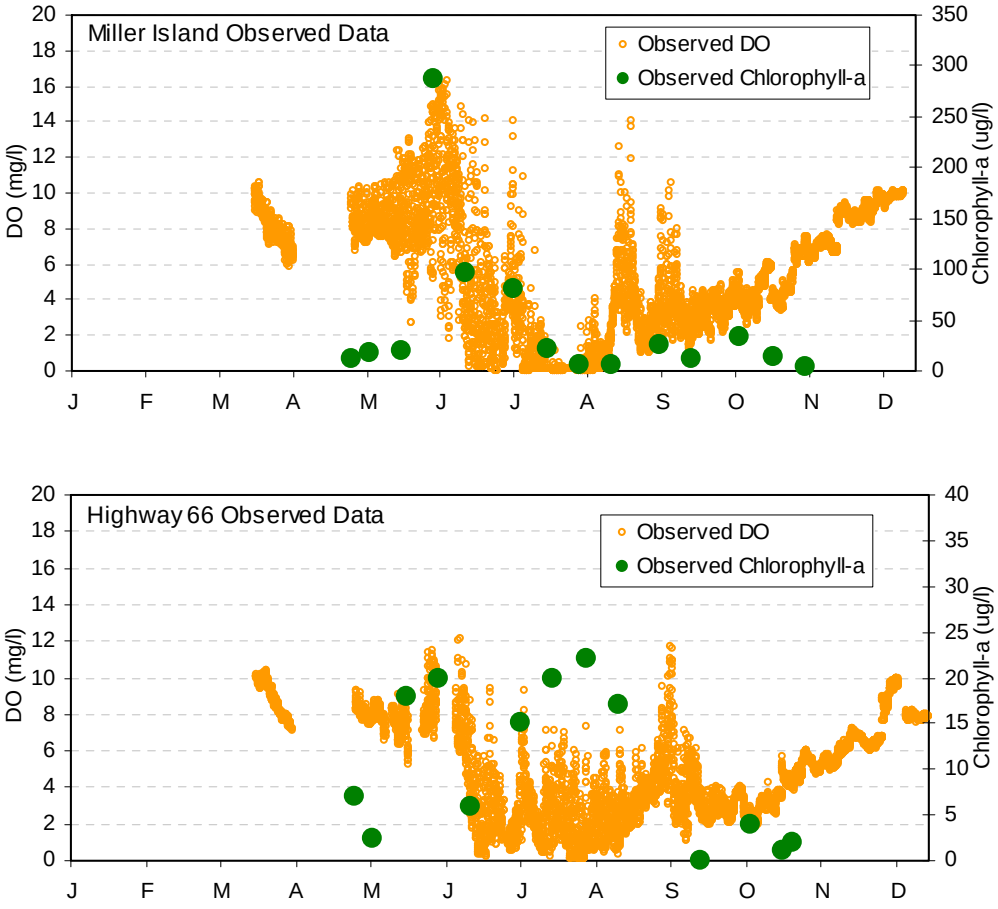


Figure 2-1. Observed DO and Algae at sites in Lake Ewauna (2000)

Algae need oxygen to respire. Thus, when oxygen levels become low or depleted, algae metabolism is expected to be affected. Growth is likely to be slowed down, and death/excretion is likely to increase. In addition to directly affecting algae metabolism, the hypoxia/anoxic condition from top to bottom in the lake can result in excessive concentration of undesirable chemicals such as H2S in the water column, which is highly toxic to phytoplankton.

The major assumptions associated with the DO-dependent algae kinetics are the following:

- Low DO concentrations can restrict algae growth and enhance algae death due to either the directly effect on metabolism or indirect effect through chemical toxicity.
- The longer algae are exposed to hypoxia/anoxia or toxic water environment associated with these conditions, the more the algae mortality is enhanced.

Using these assumptions, a model capable of simulating the impact of hypoxic/anoxic conditions on algae dynamics not only needs to represent the dependence of algae on DO concentrations, but it needs to track the duration of exposure of algae to these conditions. For example, a cluster of algae in Link River (before entering hypoxic Lake Ewauna) is free of the effects of low DO concentrations. However, after this cluster enters Lake Ewauna, it is subjected to local hypoxic conditions (while being transported downstream). The effect of the hypoxic/anoxic conditions on the algae becomes more severe with distance downstream because the exposure time to these conditions increases. This exposure time is hereby referred to as *ET*, and it is not the same as the time that the DO is hypoxic/anoxic at a specific location, which is referred to as *THA*.

CE-QUAL-W2, like most numerical hydrodynamic and water quality models, is based on the Eulerian system that does not track the history of travel of particles. Therefore, it is very difficult to directly track the ET of a specific cluster of algae. As an alternative, a two-state algae transformation approach was applied to the model as a surrogate for representing the effect of the ET. The approach involves defining two states of algae, where one state represents the *healthy* group which is free of the hypoxic/anoxic impact and the other state represents the *unhealthy* group (for which the physiological condition is severely disturbed because of hypoxia/anoxia). The healthy group is represented using typical algae growth and respiration rates; however, the unhealthy algae growth and respiration rates are very low or 0.0 (because of the hypoxic/anoxic condition's effect on the algae's physiology).

The ET is indirectly represented through conversion of healthy algae to unhealthy algae using a first-order transformation mechanism when the algae are in the presence of low-DO conditions. The longer the algae are exposed to hypoxia/anoxia, the higher the fraction of algae transformed into the unhealthy state becomes. Thus, the overall growth and respiration rate of the entire algae cluster is reduced because of the larger amount of unhealthy algae. In the following equations, *A* denotes the healthy group and *B* denotes the unhealthy group:

$$\frac{dC_A}{dt} = L_A - K_1 C_A + K_2 C_B \qquad (1)$$

$$\frac{dC_B}{dt} = L_B + K_1 C_A - K_2 C_B \qquad (2)$$

where

C_A = concentration of algae group A

C_B = concentration of algae group B
L_A = total sources and sinks including kinetics for algae group A as represented in the original model
L_B = total sources and sinks including kinetics for algae group B as represented in the original model
K_1= transformation rate from algae group A to B
K_2 = recovering rate from algae group B to A

To account for the effect of low DO on the kinetic parameters K_1 and K_2, a Monod-type function is used:

$$K_1 = K_{1(0)} \frac{HDO_1}{HDO_1 + DO} \quad (3)$$

$$K_2 = K_{2(0)} \frac{DO}{HDO_2 + DO} \quad (4)$$

where

$K_{1(0)}$ = the base transformation rate from algae group A to B
$K_{2(0)}$ = the base recovering rate from algae group B to A
DO = the DO concentration in the water column
HDO_1 = the half saturation coefficient in mg oxygen /L for K_1
HDO_2 = the half saturation coefficient in mg oxygen/L for K_2

The mechanisms described in Equations (1) through (4) were incorporated into the CE-QUAL-CE-QUAL-W2 source code. The model was then run for Lake Ewauna, and the results indicated that the model was capable of explaining the observed algae patterns reasonably well. It should be noted that the assumptions, mathematical formulations, and code development for algae kinetics are based on general scientific knowledge of algae metabolism. It is recommended that further research be conducted to investigate the impact and response relationships among DO, water chemistry changes, and algae metabolism to better understand this complex phenomenon.

2.2.3 Monod-Type SOD and OM Decay

The PacifiCorp Model used the CE-QUAL-W2 code, which represents SOD and OM decay as a delta function. With the delta function, the SOD and OM decay are activated when DO is greater than a prespecified value (generally close to zero), and deactivated when DO is lower than the value. This leads to abruptly turning SOD and OM decay on and off when DO is low or fluctuates around the prespecified value. A Monod-type continuous SOD and OM decay formulation was thus incorporated into the CE-QUAL-W2 code to represent a smoother transition of SOD and OM decay effects when DO is low.

$$SOD = \frac{DO}{DO + HDO} SOD_s$$

$$Kd = \frac{DO}{DO + HDM} Kd_s$$

where

SOD = effective SOD (g O_2/m^2/day)
DO = DO concentration in the water column for Kd, or in the bottom water for SOD (milligrams per liter [mg/L])
HDO = half saturation DO concentration (mg/L)
SOD_s = SOD before being adjusted by the water column DO (g O_2/m^2/day)
Kd = effective OM decay rate (1/day)
HDM = half saturation DO concentration for OM decay rate adjustment (mg/L)
Kd_s = OM decay rate before DO adjustment (1/day)

2.2.4. pH Simulation Module in RMA

The standard RMA modeling framework does not have the capability of simulating interactions among nutrients, phytoplankton/benthic algae, and pH. Because pH is a key water quality target for Klamath TMDL development, the modeling framework was enhanced to dynamically simulate pH dynamics.

A pH simulation module was developed and incorporated into the RMA framework to simulate the pH in the river, considering the impact of boundary conditions, phytoplankton, periphyton, benthic sources, and atmospheric-water exchange. The state variables for the pH module include Total Inorganic Carbon (TIC) and Alkalinity (Alk). Their transport is simulated using the same algorithms used for transporting other dissolved constituents in RMA. While Alk is assumed to be conservative in the water column, TIC changes due to several physical (water-air interface exchange), chemical (OM decay and benthic sources), and biological (phytoplankton and benthic algae metabolism) factors. The mathematical equations for the pH module are based on those described in Chapter 39 of Chapra (1997), and are detailed in Appendix A.

2.2.5. OM-dependent Light Extinction Formulation in RMA

The existing RMA model does not have the capability of representing the effect of OM on light conditions. Thus, it can inaccurately predict periphyton or phytoplankton growth in the presence of OM. OM can reduce sunlight and limit aquatic vegetation growth. An OM-dependent light extinction formulation was developed using the same formulation in the CE-QUAL-W2 model and incorporated into the RMA code to provide a more realistic representation of the system:

$$Ke = Ke' + OM \times KEOM$$

where

Ke = effective light extinction coefficient
Ke' = light extinction coefficient before OM adjustment
OM = OM concentration
KEOM = light extinction coefficient adjustment factor related to OM concentration

2.2.6. Reaeration Formulation Modification

In the existing RMA-11 model, the flow velocity used for reaeration calculation was forced to be greater than or equal to 0.5 meters per second (m/s). This resulted in excessive reaeration when the flow velocity was actually slower (e.g., during low-flow conditions). A modification was made to this formulation to set the lower bound of the velocity to 0.03 m/s based on Chapra (1997). This modification results in more reasonable DO predictions under low flow, critical conditions.

2.2.7 Dynamic OM Partitioning

Key updates were made to the original PacifiCorp Model to transfer model results between segments represented using CE-QUAL-W2 and those represented using RMA. Originally, the upstream boundary conditions for a CE-QUAL-W2 reservoir model were based on model results from the upstream riverine RMA model. In RMA all the OM is represented as a lumped parameter, while in W2 they are partitioned into four different components: LPOM, RPOM, LDOM, and RDOM. Therefore, when transferring the RMA OM results to W2, the OM output must be partitioned into the four components for W2. In the existing PacifiCorp Model, a static partitioning ratio of 0.8:0.2 was used to partition the OM into LPOM and LDOM, respectively. This static conversion factor does not account for the change in OM composition that occurs throughout the system. Therefore, a dynamic OM partitioning scheme was implemented that calculates and tracks the time-variable partitioning ratio in the reservoir models and applies the ratios to downstream segments. Using this approach, different ratios are implemented for J.C. Boyle and Copco reservoirs. For J.C. Boyle Reservoir, the OM in the upstream river flow is partitioned in such a way that, on average, LDOM accounts for 62 percent, LPOM 36 percent, RDOM 1 percent, and RPOM 1 percent. For Copco Reservoir, the corresponding values are: 67 percent, 30 percent, 2 percent, and 1 percent. These values demonstrate that the fraction of dissolved OM increases with downstream distance, while the fraction of particulate OM decreases (because of the effect of settling).

2.3 Model Configuration

Model configuration involved setting up the model computational grid (bathymetry) using available geometric data, designating the model's state variables, setting boundary conditions, and setting initial conditions. This section describes briefly the configuration process and key components of the model in greater detail.

2.3.1 Segmentation/Computational Grid Setup

The computational grid setup defines the process of segmenting the entire Klamath River into smaller computational segments for application of the model. In general, bathymetry is the most critical component in developing the grid for the system.

The Klamath River model includes the entire Klamath River from Link Dam (at the outlet of the UKL) to the Pacific Ocean. The river is impounded by five dams along its length: Keno, J.C. Boyle, Copco, Copco 2, and Iron Gate Dam. For this modeling study, the Klamath River was divided into nine waterbodies (or Model Segments). Figures 2-2 and 2-3 show each of the waterbodies from upstream to downstream. Note that distances for each waterbody are approximate. Appendix B presents an excerpt (verbatim) from *Klamath River Water Quality Model Implementation, Calibration, and Validation* (PacifiCorp 2005) that summarizes geometric

information for all waterbodies. Each of these waterbodies was represented using unique geometric and hydrological characteristics in the model and is detailed in subsequent sections.

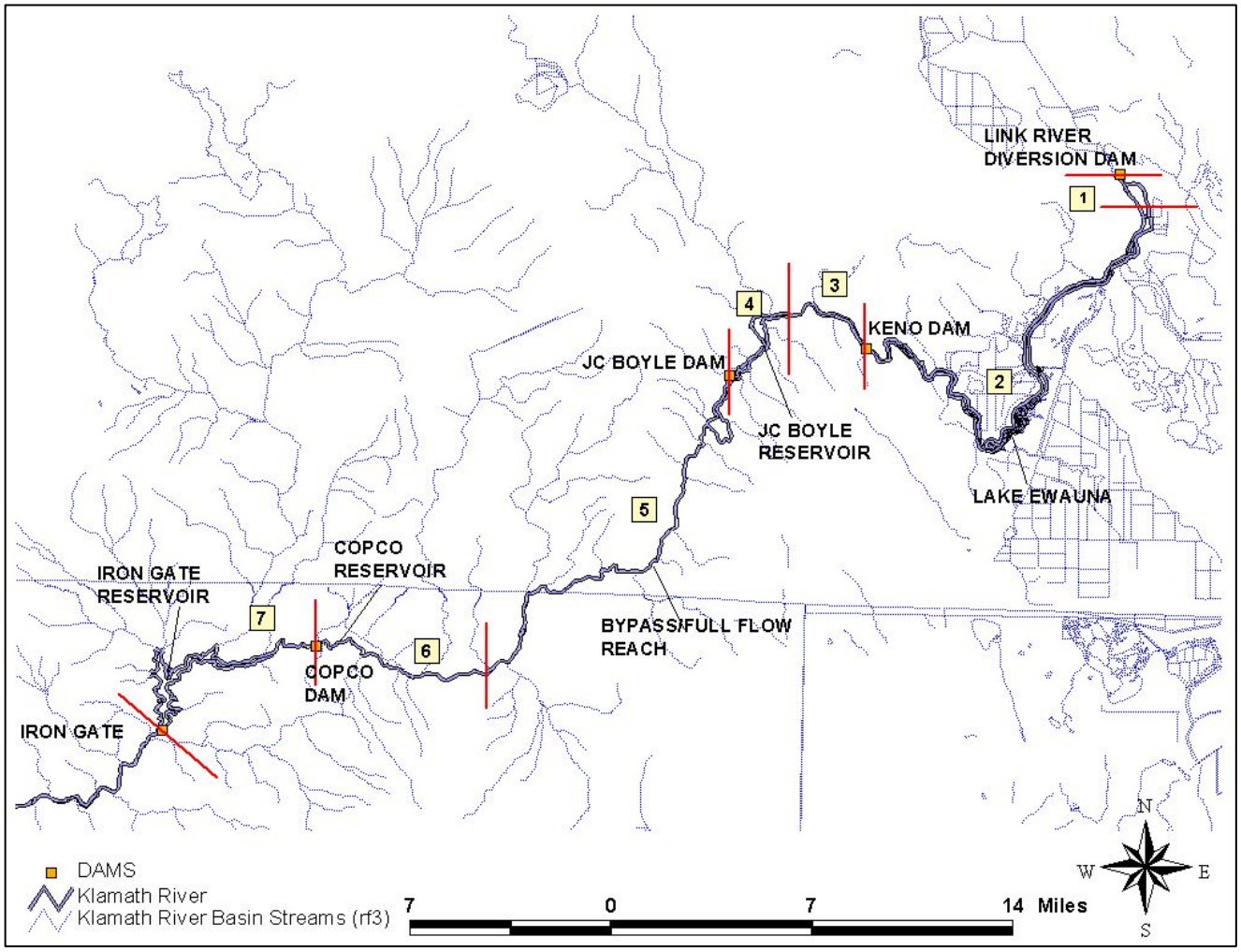


Figure 2-2. Location of waterbodies 1 through 7 along the Klamath River

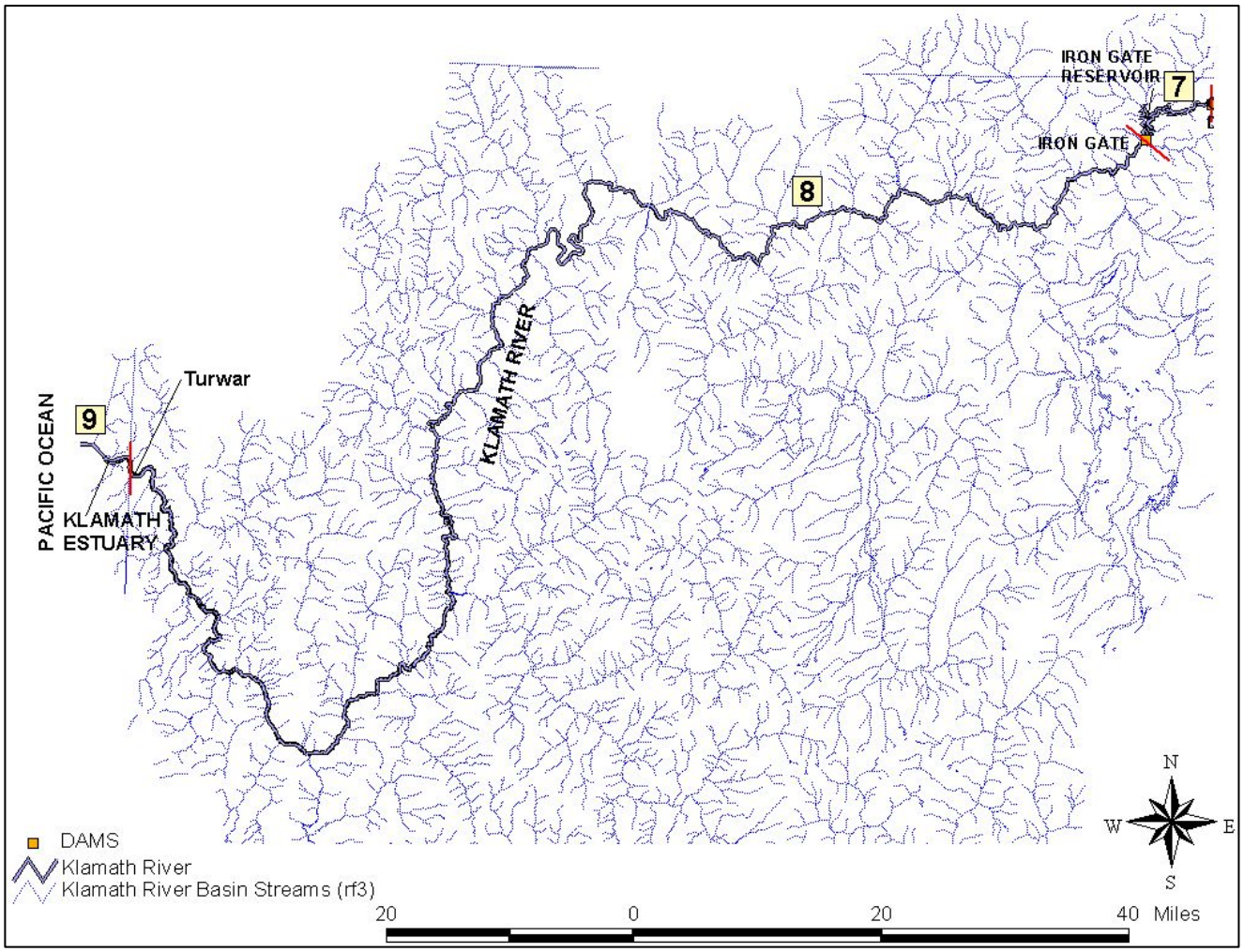


Figure 2-3. Location of waterbodies 7 through 9 along the Klamath River

2.3.1.1 Segmentation of River and Reservoir Segments

Within each of these separate Model Segments (excluding the Klamath Estuary) the primary waterbody (either a Klamath River section or a reservoir) was subdivided into higher resolution elements for greater detail in modeling. The TMDL modeling framework components were segmented similarly to the existing PacifiCorp Model. For the reservoir/lake models in the existing PacifiCorp Model (Lake Ewauna, J.C. Boyle Reservoir, Copco Reservoir, and Iron Gate Reservoir), the corresponding CE-QUAL-W2 models have layer thicknesses (depths) ranging from 0.61 to 1.0 meters and segment lengths ranging from 37 to 714 meters. For the riverine reaches (Link River, Keno reach, Bypass/Full Flow Reach downstream of J.C. Boyle Reservoir, and Klamath River from Iron Gate Dam to Turwar), the corresponding finite element model RMA has node distances ranging from 75 to approximately 300 meters (and are assumed to be homogeneous in the vertical direction). For greater detail on model segmentation in the river and reservoir segments of the Klamath River model, see Section 2.3 of the *PacifiCorp Report.*

Only the mainstem Klamath River and its reservoirs were simulated with the Klamath River model. All tributaries to the river were represented as boundary conditions (i.e., they were not

explicitly modeled). More detailed information regarding the specific tributaries to be included or inflows to the model are included in Section 2.3.3 of this report.

2.3.1.2 Segmentation of the Klamath Estuary Segment

The tidal portion of the Klamath River from Turwar to the Pacific Ocean, which was not included in the existing PacifiCorp Model, was modeled using EFDC. The first step to configure EFDC was to discretize the waterbody into a computational grid. A boundary-fit curvilinear grid was developed to accurately represent the shape of the river. Significant hydraulic features (channels, shorelines, and major bathymetric variability) and their locations were considered in preparation of the grid. The grid consists of 138 curvilinear grid cells, with widths ranging from 99 to 209 meters and lengths from 192 to 1590 meters. Within the modeling domain, each cell is represented by four vertical layer(s), therefore a 3-D spatial representation represents the estuary portion. The open boundary at the downstream end of the estuary was extended into the Pacific Ocean to reduce the effect of boundary reflection. This would likely occur if the open boundary was set directly at the opening of the sand bar. Figure 2-4 presents the computational grid for the Klamath Estuary EFDC model. It should be noted that this grid was developed and refined through an iterative process wherein model resolution, accuracy, and simulation time were optimized. The bold red line in Figure 2-4 represents the impermeable barriers added to the EFDC grid. The barriers allow for the water to flow through the outlet, as seen in the 2004 bathymetry data. These barriers may be reconfigured in the future, if needed, to simulate the sandbar opening at a different location.

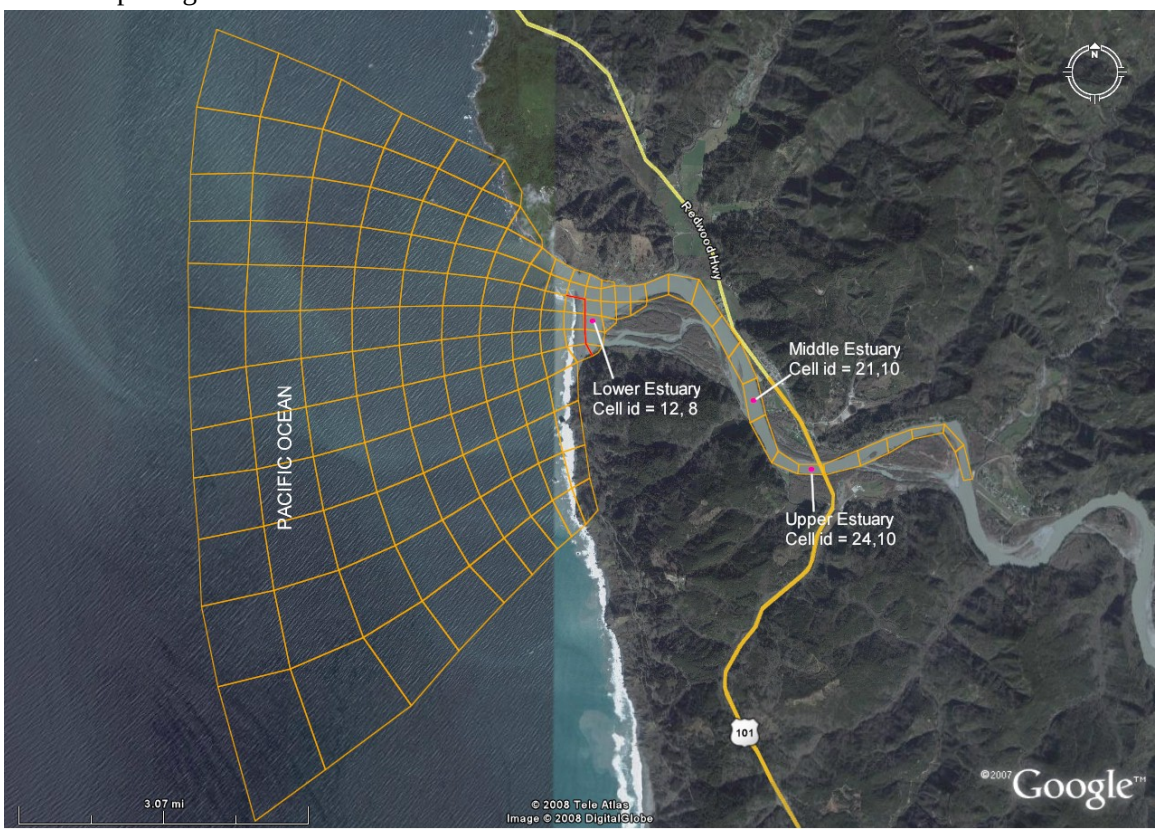


Figure 2-4. The Klamath Estuary grid for EFDC (source of image: Google)

Bathymetry data for the Klamath Estuary were obtained from the NCRWQCB and represent bathymetric conditions in the year 2004. These data contain xyz format coordinate elevation data

relative to North American Datum of 1988 (NAVD88) and were directly incorporated into the grid generation process. Bathymetry data were available from the outlet of the Klamath at the Pacific Ocean upstream to the Rt. 101 Bridge (Hwy 101 @ Klamath). No bathymetry data were available from the Rt. 101 Bridge upstream to the USGS Klamath River near Klamath station (USGS 11530500) (a distance of approximately 3,300 meters). To address this data gap, a constant bed slope similar to the upstream portion of the bathymetry data (where the bathymetry was measured) was assumed. This slope was refined until tidal impacts were properly represented (i.e., no tidal effect was observed at Turwar).

River bank boundaries for the grid were defined using digital orthophotography obtained from the California Spatial Information Library (CaSIL) (http://gis.ca.gov/). Orthophotography was used instead of USGS quadrangle images due to the age of the USGS quadrangles. For example, the Requa, CA USGS quadrangle was last revised in 1966, while Requa orthophotography represents conditions in 1998. The images were georeferenced to a Universal Transverse Mercator (UTM) Zone 10 projection, using a NAD 1927 horizontal datum. This coordinate system was then used to develop the horizontal dimensions of the grid and calculate the dimensions of the computational cells.

2.3.2 State Variables

Selection of appropriate model state variables to represent water quality processes of concern is a critical factor in model configuration. For this study, state variables were selected to most accurately predict TMDL impairments and related physical, chemical, and biological processes. State variables varied for each model type in the Klamath River model (RMA, CE-QUAL-W2, and EFDC). Note that pH is not a state variable. It is computed from alkalinity and TIC. Alkalinity and TIC are transported by the model and are thus state variables. The following state variables were configured for the riverine segments of the Klamath River model (for the RMA portions of the model):

1. Arbitrary Constituent (configured as a tracer to evaluate the mass balance)
2. DO
3. Organic matter (OM)
4. Orthophosphorus (PO_4)
5. Ammonium (NH_4)
6. Nitrite (NO_2)
7. Nitrate (NO_3)
8. Phytoplankton
9. Temperature
10. Periphyton
11. Total inorganic carbon (TIC)
12. Alkalinity (Alk)

The reservoir segments of the Klamath River, where the CE-QUAL-W2 model was applied, were configured using the following active state variables:

1. Labile dissolved organic matter (LDOM)
2. Refractory dissolved organic matter (RDOM)
3. Labile particulate organic matter (LPOM)
4. Refractory particulate organic matter (RPOM)
5. Inorganic Suspended Solids (ISS)

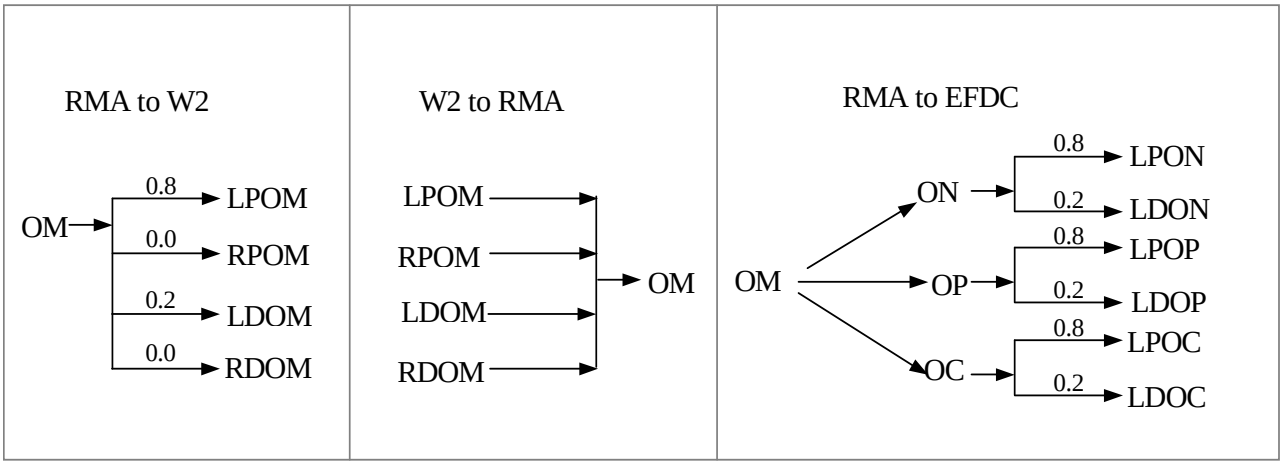
6. PO_4
7. NH_4
8. NO_2/NO_3
9. DO
10. Phytoplankton
11. Alk
12. TIC
13. Temperature
14. Tracer
15. TDS
16. Age (to track detention time at different locations)
17. Coliform bacteria

The estuarine portion of the Klamath River, which was modeled using EFDC, was configured with the following constituents as state variables:

1. Phytoplankton
2. Labile particulate organic carbon (LPOC)
3. Labile dissolved organic carbon (LDOC)
4. Labile particulate organic phosphorous (LPOP)
5. Labile dissolved organic phosphorous (LDOP)
6. PO_4
7. Labile particulate organic nitrogen (LPON)
8. Labile dissolved organic nitrogen (LDON)
9. NH_4
10. NO_2/NO_3
11. DO
12. Temperature
13. Salinity
14. Periphyton

The schematic below (Figure 2-5) shows the flow of OM to and from each of the models. The 0.8:0.2 ratio for partitioning OM was used in the existing PacifiCorp Model and was based on the CE-QUAL-W2 algae partition coefficient (APOM = 0.8). The RMA to W2 conversion shown in Figure 2-5 does not apply to the upstream boundary conditions for J.C. Boyle and Copco reservoirs, because dynamic OM partitioning was implemented (as previously discussed). For Link River, where no upstream dynamic reservoir model was available, the static partitioning shown in Figure 2-5 was implemented.

For model calibration, the EFDC upstream boundary condition was derived using 2004 monitoring data. Therefore, no OM conversion from the upstream RMA model was necessary. When modeling scenarios were run (which are not addressed in this report), the upstream RMA model representing Irongate to Turwar was linked to the estuary EFDC model. For this case, OM from the RMA model was converted into organic nitrogen (ON), organic phosphorous (OP), and organic carbon (OC) components for the EFDC model. This conversion is based on the stoichiometric ratio used in the upstream W2 and RMA models, where OM multipliers are the following: ON = 0.07, OP = 0.0055, and OC = 0.45 (Cole and Wells 2003). To further partition the ON, OP, and OC into labile/refractory and particulate/dissolved components, the ratios presented in Figure 2-5 were used.



Note: The RMA to W2 conversion shown above does not apply to the upstream boundary conditions for J.C. Boyle and Copco reservoirs, where dynamic OM partitioning was applied. The RMA to EFDC representation does not reflect calibration conditions, rather it summarizes transfer used during scenario analysis.

Figure 2-5. Schematic showing the transfer of OM between models

2.3.3 Boundary Conditions

To run the Klamath River model, external forcing factors known as boundary conditions must be specified for the system. These forcing factors are a critical component in the modeling process and have direct implications on the quality of the model's predictions. External forcing factors include a wide range of dynamic information:

- Upstream Inflow Boundary Conditions: Upstream external inflows, temperature, and constituent boundary conditions
- Tributary (or Lateral) Inflow Boundary Conditions: Tributary inflows, temperature, and constituent boundary conditions
- Withdrawal Boundary Conditions
- Surface Boundary Conditions: Atmospheric conditions (including wind, air temperature, solar radiation)

Upstream external inflows essentially represent the inflow at the model's *starting* point. Tributary inflows represent the major tributaries that feed into the Klamath River. All water removed from the system is combined within the withdrawals category.

The surface boundary conditions are determined by the meteorological or atmospheric conditions and include air temperature, dew point temperature, wind speed, wind direction, and cloud cover. The meteorological file from the original PacifiCorp Model was maintained since it was based on real data and was intensively reviewed. Data obtained from station KFLO near Klamath Falls were used to represent the conditions from Link Dam to J.C. Boyle Dam. Data from Brazie Ranch represent the boundary conditions from J.C. Boyle Dam to Seiad Valley. For the reach from Seiad Valley to Turwar, the weather data from Hoopa and Somes Bar were used to represent the meteorological boundary conditions. Data from the Arcata Eureka Airport were applied to the estuarine portion of the Klamath River (Turwar to the Pacific Ocean), as described later in Section 2.3.3.9.

The following subsections provide a detailed description of the boundary conditions used to represent each modeled segment. The descriptions begin upstream at the Link River segment and continue downstream to the Klamath Estuary segment. In the existing PacifiCorp Model, boundary conditions were set as time series at each location on the basis of observed data or other assumptions where data were not available. For periods when no data were available, the model internally estimates the boundary on the basis of linearly interpolating the time series provided in the boundary condition files. In some situations, boundary conditions were updated using more recently acquired monitoring data. Both types of modification are further described in this section. Plots of time series boundary conditions are included as an electronic Appendix with this report (Appendix C).

The upper and middle segments of the model (Model Segments 1 through 8) were tested (calibrated) using data from the year 2000. In addition, the calibration of the upper segments (Model Segments 1 through 5) was further corroborated (validated) with observed data for 2002. The estuarine portion (Model Segment 9), which was modeled with EFDC, was calibrated using data from the year 2004. As described in Section 3.0, these periods were selected because of data availability. In subsequent discussions, boundary condition descriptions are first described for the year 2000. Any deviations from the year 2000 representation for the year 2002 are then noted.

2.3.3.1 Model Segment 1: Link River

The Link River segment begins at the outlet of UKL (Link Dam) and ends at Lake Ewauna. Four types of boundary conditions were included in this model segment: upstream inflow boundary conditions, tributary boundary conditions, downstream stage-discharge boundary conditions, and surface boundary conditions (discussed above).

<u>Upstream Inflow Boundary Conditions:</u> The inflow to Link River is from UKL through releases from Link Dam. Since there were no observed data available at the head of Link River for 2000, observed water quality data at Pelican Marina (in UKL) were used as the basis for upstream boundary conditions. This representation is different than that in the existing PacifiCorp Model which used multiple year composite data for Link River at Fremont Bridge as the basis of boundary condition. Considering the significant inter-year variability in water quality in UKL, it is preferable to use data collected during the modeling year rather than other years to represent the external forces at boundaries. Monitoring data for NH_4, NO_2/NO_3, phytoplankton, DO, and temperature were directly applied to the boundary conditions using a linear interpolation method to obtain daily values for dates without data. OM boundary conditions were derived using observed total phosphorus (TP), dissolved PO_4, and chlorophyll *a* data and following these steps:

Step 1: derive algal P as: $OP_{alg} = Chla \times CCHA / AGP$
Step 2: derive non-algal P as: $OP_{non\text{-alg}} = TP - \text{dissolved } PO_4 - OP_{alg}$
Step 3: derive OM as: $OM = OP_{non\text{-alg}} \times OMP$

where

Chla = observed chlorophyll *a* concentration (µg/L)
CCHA = 0.067 (mg-algae per µg-chlorophyll *a*); derivation: Algae = Chla × 67 × (1 mg/1000 ug), where 67 represents the Algae:Chla ratio defined as 30/0.45 (on the basis of the WASP model default ratio of 30 for Algae-C:Chla and the CE-QUAL-W2 model default ratio of 0.45 for Algae-C:Algae)
AGP = algal P content coefficient (mg-algae / mg-P)
OMP = organic matter P content coefficient (mg-OM / mg-P)

OMP was determined to be 180.0 based on 2002 data for Link River at Fremont Bridge (where the average organic carbon:organic phosphorus ratio is 81, and thus the OM:OP ratio is 81 / 0.45 = 180.0). AGP was assumed to be the same as OMP because phytoplankton is the major source of OM in UKL. BOD was not configured for the model because all OM are represented using a single state variable (as previously noted).

Initially, the total PO_4 boundary condition was represented using the dissolved PO_4 monitoring data at Pelican Marina. It was found, however, that setting the PO_4 boundary concentration at Link River to the observed dissolved PO_4 value resulted in a significant underprediction of PO_4 concentration at Miller Island, in Lake Ewauna. Because Link River flow is dominant in upper Lake Ewauna, the PO_4 concentration at Miller Island should be similar in magnitude and pattern to that at the head of Link River. Several model sensitivity analyses confirmed this. The difference between the dissolved PO_4 data at Pelican Marina and the PO_4 data at Miller Island suggests that the dissolved PO_4 at Pelican Marina is likely not a good representation of conditions at the head of Link River. Therefore, the observed PO_4 data at Miller Island were used as the basis for configuring the PO_4 boundary condition at Link Dam.

The alkalinity boundary condition was configured on the basis of alkalinity monitoring data at Link River and Miller Island. In 2000 there was a limited amount of alkalinity data at Link River (on seven discrete dates). These data were insufficient to accurately predict alkalinity concentrations at Miller Island. Therefore, Miller Island data were used to supplement the Link River data in constructing the boundary condition. The first step in doing this was to compare the flow from Link River and the Lost River Diversion Channel to determine the period during which Link River flow was dominant. For this period the alkalinity at Miller Island would be similar in magnitude to that at Link River. Therefore, data at Miller Island were incorporated into the Link River data to form an expanded data set. The upstream boundary condition for alkalinity was then configured using this expanded data set. There were no data available for TIC; therefore, the TIC boundary condition was obtained through the pH calibration process for Miller Island. Initially, TIC at the Link River boundary was derived on the basis of pH at Miller Island and alkalinity at Link River. These estimates were refined to achieve a better calibration of pH at Miller Island in Lake Ewauna.

The upstream boundary condition for the 2002 model was derived using a method similar to that used for 2000. Available data at both the head of Link River (Fremont Bridge) and at Pelican Marina were combined to form a composite data set for 2002 boundary condition derivation. Monitoring data for PO4, NH_4, NO_2/NO_3, phytoplankton, DO, and temperature were directly applied to the boundary conditions using a linear interpolation method to obtain daily values for dates without data. The OM boundary condition was derived using the same approach used for the 2000 model. The alkalinity boundary condition was derived on the basis of monitoring data at Fremont Bridge. And the TIC boundary condition was derived on the basis of alkalinity data and pH data at the same location.

<u>Tributary Boundary Conditions:</u> There are two diversions from UKL at Link Dam. These diversions are two powerhouses that discharge water from UKL into the Link River segment (East Side and West Side). USGS gage 11507500 (Link River at Klamath Falls, Oregon) is between the powerhouse discharges.

The constituent concentrations for the tributary boundary conditions were set to be the same as the upstream boundary conditions because the powerhouses have the same water source as the upstream boundary (UKL).

Downstream Boundary Conditions: Downstream boundary conditions were configured using a stage-discharge relationship. Although this type of boundary condition does not allow for representation of the backflow condition that occasionally occur at the mouth of Link River, it is a better predictive tool than using the Lake Ewauna elevation as the downstream boundary condition. The backflow condition does not have a significant impact on the loading rate from Link River to Lake Ewauna, thus, it does not significantly impact the water quality in the lake.

2.3.3.2 Model Segment 2: Lake Ewauna to Keno Dam

This segment extends from the point where Link River enters Lake Ewauna to the outlet at Keno Dam. Five types of boundary conditions were included in the Klamath River model for the Lake Ewauna segment. They are upstream inflow boundary conditions, tributary boundary conditions, withdrawal boundary conditions, downstream outflow boundary conditions, and surface boundary conditions.

Upstream Inflow Boundary Conditions: The upstream boundary condition was defined as the water flowing into Lake Ewauna from Link River (Model Segment 1). Link River's flow was determined by using the observed flows at USGS flow gage 11507500 minus the flow from the PacifiCorp West Turbine (powerhouse) gage, which is downstream of the USGS gage.

The upstream boundary conditions for water quality constituent concentrations were based on the model results in the downstream region of the Link River Model Segment. PO_4, NH_4, NO_2/NO_3, DO, phytoplankton, Alk, TIC, and temperature were directly transferred from the RMA-11 model (from Link River) to the CE-QUAL-W2 input data file for Lake Ewauna. Output for OM from Link River was applied to the Lake Ewauna segment and partitioned into four components: LDOM, RDOM, LPOM, and RPOM with partition ratios as 0.2, 0.0, 0.8, and 0.0, respectively. These ratios were based on the CE-QUAL-W2 ALPOM value and the assumption that all OM are labile. These assumptions were justified because the majority of the organic matter from UKL are likely generated by phytoplankton blooms and metabolism. Therefore, the CE-QUAL-W2 ALPOM value can be used to represent partitioning. In reality significant spatial and temporal variability associated with organic matter composition may exist, however insufficient data are available to more accurately represent the organic matter boundary conditions. In addition, CE-QUAL-W2 isn't capable of representing seasonal variability of OM composition for the boundary conditions.

Tributary Boundary Conditions: There are 18 tributary discharges in the Lake Ewauna to Keno Dam river segment. These discharges include 11 stormwater locations, Columbia Plywood discharge, Klamath Falls Wastewater Treatment Plant, South Suburban Sanitation District, two discharges at Collins Forest Products, Lost River Diversion Channel, and Klamath Straits Drain (KSD).

The inflow from the stormwater locations was calculated as an average percentage of total stormwater runoff. The flow from Columbia Plywood was calculated from the discharger's monthly monitoring reports as an average of 0.01 cubic feet per second (cfs). Variable daily flows were used for the Klamath Falls Wastewater Treatment Plant and ranged from approximately 4 to 12 cfs. Variable daily flows were also used for South Suburban Sanitation District and generally ranged from 1 to 4 cfs. The two discharges at Collins Forest Products had average daily flows of approximately 1.4 cfs and 0.1 cfs. Daily flows from Lost River Diversion

Channel and KSD into Lake Ewauna were obtained from U.S. Bureau of Reclamation's (USBR's) flow gages at these locations.

The water quality constituent concentrations of the tributary boundary conditions were set to be the same as in the previous PacifiCorp Model except for the major tributaries and point sources, including KSD, Klamath Falls Wastewater Treatment Plant, and South Suburban Sanitation District. For these major tributaries and point sources, available data for 2000 and 2002 were used to update the boundary conditions. The details of updating these boundary conditions are summarized as follows:

a) KSD

The concentration boundary condition at KSD was represented using data at station Pump F in the KSD. The formulas used to convert observed data to model boundary conditions are listed below. For each constituent notation, the one on the left-hand side corresponds to the model boundary condition, while the one on the right-hand side corresponds to observed data. For parameters not listed, observed values were used directly.

> Algae [mg/L] = Chlorophyll *a* [µg/L] × 0.067, where 0.067 was derived similarly to CCHA for the UKL boundary condition (previously described)
> LDOM [mg/L] = (TP [mg/L] – PO4 [mg/L]) × 180.0 × 0.7, where 180.0 was derived similarly to OMP for the UKL boundary condition (previously described), and 0.7 was derived on the basis of 2002 data at KSD (Dissolved TP / TP)
> RDOM [mg/L] = 0.0
> LPOM [mg/L] = (TP – PO4) × 180.0 × 0.3, where the ratio 0.3 was derived from (1.0–0.7, where 0.7 represents LDOM).
> RPOM [mg/L] = 0.0
> ISS [mg/L] = TSS [mg/L]
> TIC [mg/L] = *f*(Alk, temperature, pH), where *f* represents the functional form relating TIC to Alkalinity [mg/L], temperature [^{o}C], and pH. Detailed equations can be found in Chapra 1997.

b) Klamath Falls Wastewater Treatment Plant and South Suburban Sanitation District

The water quality constituent concentrations for both the Klamath Falls Wastewater Treatment Plant and South Suburban Sanitation District were set to be the same as in the previous PacifiCorp Model, except for where 2000 facility discharge monitoring report data were available. The formulas used to derive the boundary conditions based on data are as follows:

LDOM = BODu / 1.4 × 0.2, where the ratio 1.4 is based on the W2 stoichiometric ratio, and 0.2 is the same as that used for the UKL boundary condition.
LPOM = BODu / 1.4 × 0.8, where the ratio 1.4 is based on the W2 stoichiometric ratio, and 0.8 is the same as that used for the UKL boundary condition.
ISS = TSS
PO4 = TP – BODu / 1.4 / 180.0

<u>Withdrawal Boundary Conditions:</u> There are seven withdrawals in this segment. The Lost River, North Canal, and ADY Canal are the three withdrawals in Lake Ewauna for the Klamath Reclamation Project. Daily flows at all three of these withdrawals are gaged by USBR.

There is a lack of available daily withdrawal rates for non-USBR irrigation diversions, therefore, withdrawal rates from a previous study were applied to all four of the irrigation withdrawals (Watercourse Engineering, Inc. 2004).

The hourly flow rate at Keno Dam was available from USGS gage 11509500 (Klamath River near Keno, Oregon). The flows ranged from less than 500 cfs to more than 4,000 cfs. All these boundary conditions were kept the same as in the previous PacifiCorp Model.

Downstream Outflow Boundary Conditions: For Lake Ewauna, the downstream boundary condition was set as the outflow at the point before entering Keno Reach (Keno Dam to J.C. Boyle Reservoir). The downstream boundary condition was set to be outflow; therefore, no water quality concentration boundary condition was needed.

2.3.3.3 Model Segment 3: Keno Dam to J.C. Boyle Reservoir (Keno Reach)

There were three types of boundary conditions included in this section of the model: upstream inflow boundary conditions, downstream outflow boundary conditions, and surface boundary conditions.

Upstream Inflow Boundary Conditions: The upstream inflow to this reach for the 2000 and 2002 models is based on the outflow from Lake Ewauna for the corresponding year. This segment was dominated by upstream water quality, therefore the simulated loading time series for phytoplankton, temperature, PO_4, NH_4, NO_2/NO_3, DO, TIC, and Alk from the Lake Ewauna to Keno Dam segment were applied. The four OM constituents predicted by the CE-QUAL-W2 model were combined into one OM constituent and applied to the boundary conditions (as previously discussed).

Downstream Outflow Boundary Conditions: Hydrodynamic downstream boundary condition was set as a stage-discharge relationship, which represents the downstream flow as only outflow; therefore, no concentration boundary condition was needed.

2.3.3.4 Model Segment 4: J.C. Boyle Reservoir

The J.C. Boyle Reservoir extends from the J.C. Boyle headwaters (Keno Reach to J.C Boyle Reservoir) to the J.C. Boyle Dam. There were four types of boundary conditions included in this portion of the model. They are upstream inflow boundary conditions, tributary boundary conditions, downstream outflow boundary conditions, and surface boundary conditions.

Upstream Inflow Boundary Conditions: Klamath River inflow for the 2000 and 2002 models to J.C. Boyle dam is represented by discharge from the Keno Reach during the corresponding year. The upstream boundary conditions for water quality constituents were based on the model results at the downstream node of the Keno Reach portion of the model for the corresponding year. PO_4, NH_4, NO_2/NO_3, DO, phytoplankton, temperature, TIC, and Alk were directly transferred from the RMA-11 model results for Keno Reach to the CE-QUAL-W2 input data file for J.C. Boyle Reservoir. Output for OM from the Keno Reach model was applied to the J.C. Boyle Reservoir (see Section 2.2.1) and was partitioned into four components: LDOM, RDOM, LPOM, and RPOM. The aforementioned dynamic partitioning scheme was applied. This scheme uses the LDOM, RDOM, LPOM, and RPOM fractions derived from model results from the last segment of Keno Reservoir to partition the OM into the four components.

Tributary Boundary Conditions: There is one tributary to J.C. Boyle Reservoir, Spencer Creek. Spencer Creek has very limited inflow information. Therefore, it is not configured as a separate tributary in this model. The minor contribution of flow from Spencer Creek is lumped into the upstream headwater in the original PacifiCorp model, and directly adopted in the TMDL model. The net reservoir accretion/depletion was calculated through a flow balance process aiming to reproduce the observed surface water elevation in the reservoir. This accretion/depletion was configured as a distributed tributary boundary condition in the model. The concentration of the tributary inflow was set to be the same as in the upstream boundary condition.

Downstream Outflow Boundary Conditions: The outflow from the reservoir was calculated as the sum of all recorded releases to the four outlets in the reservoir (powerhouse canal, dam spillway, bypass releases, and fish ladder releases).

The downstream boundary condition was set to be outflow; therefore, no concentration boundary condition was needed.

2.3.3.5 Model Segment 5: Bypass/Full Flow Reach

The Bypass/Full Flow Reach extends from the J.C. Boyle Dam to the headwaters of Copco Reservoir. There were four types of boundary conditions included in this portion of the model: upstream inflow boundary conditions, tributary boundary conditions, downstream outflow boundary conditions, and surface boundary conditions.

Upstream Inflow Boundary Conditions: There are two inflows to the Bypass/Full Flow Reach. They are releases from J.C. Boyle Dam directly to the Klamath River and the J.C. Boyle Powerhouse tailrace. Measured releases from the dam for 2000 and 2002 were obtained from PacifiCorp and used to represent both inflows for the model for the corresponding years.

For the upstream water quality constituent concentration boundary conditions, the simulated loading time series for phytoplankton, temperature, PO_4, NH_4, NO_2/NO_3, and DO from the J.C. Boyle Reservoir were applied. The four OM constituents predicted by the CE-QUAL-W2 model were combined into one OM constituent as applied in the Keno Reach boundary condition.

Tributary Boundary Conditions: There are no major tributaries, but there are three springs represented by a constant flow of 75 cfs each. The flow rate, temperature, DO, and phytoplankton boundary conditions for the springs were the same as in the original PacifiCorp Model, while the concentrations for the major nutrients (i.e., PO_4, NH_4, and NO_2/NO_3) were derived through model calibration. After several iterations, the concentrations for NH_4, NO_2/NO_3, and PO_4 were determined to be 0.029 mg/L, 0.25 mg/L, and 0.066 mg/L, respectively. OM concentrations were assumed to be a small value of 0.5 mg/L considering the springs are mainly groundwater. These concentrations were applied to both the 2000 and 2002 models for this reach.

Downstream Outflow Boundary Conditions: The downstream boundary condition for the Bypass/Full Flow Reach was configured as a stage-discharge relationship. No concentration boundary condition was needed for the downstream boundary conditions because only outflow exists there.

2.3.3.6 Model Segment 6: Copco Reservoir

The Copco Reservoir model segment extends from Copco Reservoir's headwaters to Copco Dam. Four types of boundary conditions were included in the portion of the model for Copco Reservoir. They are upstream inflow boundary conditions, tributary boundary conditions, downstream outflow boundary conditions, and surface boundary conditions.

Upstream Inflow Boundary Conditions: The inflow for Copco Reservoir was represented as the sum of the inflow into the reservoir and the estimated accretion/depletion for the reservoir. The flows from Bypass/Full Flow Reach were used as inflow to the Copco Reservoir because there are no flow data available at the headwaters of Copco Reservoir. The daily accretion/depletion was the sum of the daily change in storage in the Copco reservoir and the daily average outflow from the reservoir (minus the daily average inflows from Bypass/Full Flow Reach).

The upstream water quality constituent concentration boundary conditions were based on the model results at the downstream node of the Bypass/Full Flow Reach portion of the model. PO_4, NH_4, NO_2/NO_3, DO, phytoplankton, and temperature were directly transferred from the RMA-11 model results for Bypass/Full Flow Reach to the CE-QUAL-W2 input data file for Copco Reservoir. Output for OM from the Bypass/Full Flow Reach portion of the model was applied to Copco Reservoir (see Section 2.2.1) and is partitioned into four components: LDOM, RDOM, LPOM, and RPOM using the dynamic-partitioning approach as described above.

Tributary Boundary Conditions: The concentrations of the distributed tributary boundary conditions were set to be the same as the upstream concentration boundary condition in the same manner as in the original model except for TIC and Alk, which were set to be the same as those for Jenny Creek. This was arrived at through the calibration process, which indicated that when TIC and Alk were set to be the same as the upstream boundary condition, the model significantly overpredicted pH in the reservoir.

Downstream Outflow Boundary Conditions: The two main outlets for the Copco Dam are a spillway and two waterway intakes at the Copco powerhouse (treated as a single outlet). Hourly outflow data for the powerhouse and the spillway were available from PacifiCorp and used as reservoir outflow flows.

The downstream boundary condition was set to be outflow; therefore, no concentration boundary condition was needed.

2.3.3.7 Model Segment 7: Iron Gate Reservoir

The Iron Gate Reservoir model segment extends from the headwaters of the Iron Gate Reservoir to Iron Gate Dam. Five types of boundary conditions were included in the portion of the model for Iron Gate Reservoir. They are upstream inflow boundary conditions, tributary boundary conditions, withdrawal boundary conditions, downstream outflow boundary conditions, and surface boundary conditions.

Upstream Inflow Boundary Conditions: There is no gage to measure inflow to Iron Gate Reservoir; therefore, the flows from the Copco Reservoir were used to represent inflow.

Simulated water quality outflow values from Copco Reservoir were applied as the Iron Gate Reservoir inflow water quality constituent concentration boundary conditions, and they were the same configuration as for Lake Ewauna.

Tributary Boundary Conditions: There are three tributaries to the Iron Gate Reservoir. They are Camp Creek, Jenny Creek, and Fall Creek. Limited flow information was available for these creeks. The hourly accretion/depletion for the reservoir was calculated as the sum of the daily inflow, outflow, and change in storage in Iron Gate Reservoir. Jenny Creek was represented by this accretion/depletion. Neither Camp Creek nor Fall Creek were explicitly configured with contributions in the model. Tributary boundary conditions were not changed from the original PacifiCorp Model.

There were no water quality data available for the three tributaries, so monthly estimated constituent concentrations from Bogus Creek (in the Iron Gate to Turwar Reach) were applied to Jenny Creek. The only exception was for Alk, which was set to be the same as in the original PacifiCorp Model. The TIC concentration was estimated through an iterative calibration process aiming to reproduce observed pH in the reservoir.

Withdrawal Boundary Conditions: The dam's spillway was modeled as a withdrawal because it draws water to the side of the dam, not over or through the dam. Representing the spillway as a withdrawal more accurately represents the system. If the spillway were represented as a spillway in W2, water would flow to the end of the reservoir instead of the side, and this can affect the hydrodynamic simulation.

Downstream Outflow Boundary Conditions: The Iron Gate dam has four primary outlets: a spillway, penstock, and two fish hatchery intakes. Outflow from the reservoir was based on the outflow in the original PacifiCorp Model. Outflow was determined from PacifiCorp daily flow records for the Powerhouse release and spill and estimates of fish hatchery releases (50 cfs for lower hatchery release and 0 cfs for upper hatchery release).

The downstream boundary condition was set to be outflow; therefore, no concentration boundary condition was needed.

2.3.3.8 Model Segment 8: Iron Gate Dam to Turwar

Four types of boundary conditions were included in the portion of the model for Iron Gate Dam to Turwar. They are upstream inflow boundary conditions, tributary boundary conditions, downstream outflow boundary conditions, and surface boundary conditions.

Upstream Inflow Boundary Conditions: The upstream inflow boundary conditions for Iron Gate Dam to Turwar were based on PacifiCorp's original model, which used PacifiCorp's measured releases from Iron Gate Dam during 2000.

Upstream water quality constituent boundary conditions were the simulated outflow values from the Iron Gate reservoir.

Tributary Boundary Conditions: There are 23 tributaries to this segment of the Klamath River, including four major tributaries (Shasta, Scott, Salmon, and Trinity rivers). Five tributaries to this reach are actively gauged, including the Shasta, Scott, Salmon, and Trinity rivers, and Indian Creek. Inflows for minor tributaries were defined and quantified as daily accretion/depletions.

The Scott and Trinity rivers were assigned by summing USGS-gaged flows and daily accretion/depletions. The daily accretion/depletions were determined on the basis of a USGS

methodology. Monthly average values were used to determine accretions and depletions for each river segment on the basis of differences in gage readings, and these accretions and depletions were then assigned to individual tributaries according to the estimated basin area. Appendix D presents the USGS methodology for estimating these flows for tributaries (PacifiCorp 2004). Model node and element numbers and type of flow record employed for each tributary are summarized in Table 2-2.

Table 2-2. Element flow information for the Iron Gate to Turwar simulation

Location	Node	Element	Flow Type
Bogus Creek	7	4	7 day average
Willow Creek	55	28	7 day average
Cottonwood Creek	86	43	7 day average
Shasta River	144	72	Daily measured
Humbug Creek	204	102	7 day average
Beaver Creek	319	160	7 day average
Horse Creek	468	234	7 day average
Scott River	513	257	Daily measured + A/D Ft. Jones to Klamath
Grider Creek	656	328	7 day average (A/D Scott to Seiad)
Thompson Creek	735	368	7 day average
Indian Creek	906	453	Daily measured
Elk Creek	925	463	7 day average
Clear Creek	1000	500	7 day average
Ukonom Creek	1098	549	7 day average
Dillon Creek	1162	581	7 day average
Salmon River	1357	679	Daily measured
Camp Creek	1466	733	7 day average
Red Cap Creek	1511	756	7 day average
Bluff Creek	1547	774	7 day average
Trinity River	1609	805	Daily measured + A/D Hoopa to Klamath
Pine Creek	1644	822	7 day average
Tectah Creek	1850	925	7 day average
Blue Creek	1908	954	7 day average

Shasta River daily flows were taken from USGS Gage 11517500 (Shasta River near Yreka). Scott River daily flows were calculated from USGS Gage 11519500 (Scott River near Ft Jones) and accretion/depletions. Daily Indian Creek flows were taken from USGS Gage 11521500 (Indian Creek near Happy Camp). Salmon River daily flows were from USGS Gage 11522500 (Salmon River at Somes Bar). Trinity River daily flows were calculated from USGS Gage 11530000 (Trinity River at Hoopa) and accretion/depletions.

Water quality constituent concentrations in the tributaries for all parameters except DO were based on U.S. Fish and Wildlife Service (USFWS), USBR, and EPA data.

Temperature data for the tributaries were very limited, therefore the temperature boundary conditions for all the tributaries were configured on the basis of the USGS-estimated temperature for 2002. It was found that by directly using the USGS-estimated temperatures in these tributaries, the model reproduced observe temperatures in the Klamath River quite well. This is not surprising since the USGS study did show that there is no significant inter-year variation in the predicted in-stream temperature.

There were very little to no water quality data available for most tributaries. The only tributaries with sufficient data to represent seasonal variations for 2000 were the Shasta and Scott rivers (USBR 2003 data). For the other two major rivers, Salmon and Trinity, NCRWQCB derived representative data to approximate the boundary conditions for 2000, based on statistical analysis of historical tributary data. Several historical datasets (1960s to 1980s) with water quality data from CDWR, STORET, USBR, and USGS were supplemented with more recent data (1990s to 2007) from the Yurok Tribe and the USFWS (NCRWQCB, 2008). The data were split into two seasonal periods – Wet (November – April) and Dry (May – October), and years which had similar hydrologic conditions to the year 2000 were selected based on statistical measures. The median water quality values for the two seasonal periods were used for boundary conditions. Details of the steps involved in the derivation of the boundaries can be found in the document *Analysis of Klamath Tributary Data* (NCRWQCB 2008).

In addition to the USBR data, the NCRWQCB compiled nutrient data for several minor tributaries including Beaver Creek, Bluff Creek, Clear Creek, Dillon Creek, Elk Creek, Red Cap Creek, Indian Creek, and Bogus Creek for the period from 2001 to 2006. These data were divided into two categories. The first was data for Bogus Creek that were used to derive the boundary condition for Bogus Creek. Bogus Creek exhibits significantly higher nutrient concentrations than the other tributaries. The second was data for all other minor tributaries, which were combined to derive values representing all the minor tributaries. Because of a lack of sufficient data to characterize temporal variability, it was deemed appropriate to use an annual average value to represent the boundary conditions from the minor tributaries.

DO in all minor tributaries was estimated using 90 percent saturated conditions, except for the Shasta and Scott rivers, where DO data were available. DO saturation concentrations were based on the temperature data and atmospheric pressure corrected for elevation. Plots of all boundaries, including DO, can be found in Appendix C. A more detailed description of the boundary conditions for each of the tributaries is provided in Table 2-3.

Table 2-3. Description of Boundary Conditions for Tributaries within the Irongate to Turwar Segment

Tributary name	Temperature	Nutrients	DO	TIC/ALK
Bogus Creek	Based on USGS estimated data	Based on USGS estimated data	90 % saturation value	Based on the data in the Scott River
Willow Creek	Based on USGS estimated data	Based on USGS estimated data	90% saturation value	Based on the data in the Scott River
Cottonwood Creek	Based on USGS estimated data	Based on USGS estimated data	90% saturation value	Based on the data in the Scott River
Shasta River	Based on USGS estimated data	Nutrients were set based on USFWS data at the mouth of the Shasta River	DO was based on observed data, except for the period without monitoring data, which was set to be 90% of the saturation value.	Based on the data at the Shasta River mouth
Humbug Creek	Based on USGS estimated data	Based on USGS estimated data	90% saturation value	Based on the data in the Scott River
Beaver Creek	Based on USGS estimated data	Based on USGS estimated data	90% saturation value	Based on the data in the Scott River, except for August and September, when

Tributary name	Temperature	Nutrients	DO	TIC/ALK
				limited data were available for 2006. For August and September, the data were used directly.
Horse Creek	Based on USGS estimated data	Based on USGS estimated data	90% saturation value	Set to be the same as in the Beaver Creek boundary condition
Scott River	Based on USGS estimated data	Based on USFWS data at the mouth of the Scott River	Based on USFWS data at the mouth of the Scott River. Periods without data were set at 90% saturation value	Based on USFWS data at the mouth of the Scott River
Grider Creek	Based on USGS estimated data	Based on USGS estimated data	90% saturation value	Set based on the observed data at the mouth of Salmon River.
Thompson Creek	Based on USGS estimated data	Based on USGS estimated data	90% saturation value	Set based on the observed data at the mouth of Salmon River
Indian Creek	Based on USGS estimated data	Based on USGS estimated data	90% saturation value	Based on the observed data at the mouth of the Salmon River, except for August and September, when 2006 data were available at Indian Creek. For this period, the data were used directly.
Elk Creek	Based on USGS estimated data	Based on USGS estimated data	90% saturation value	Based on the observed data at the mouth of the Salmon River except for August and September, when 2006 data were available at Elk Creek. For this period, the data were used directly.
Clear Creek	Based on USGS estimated data	Based on USGS estimated data	90% saturation value	Based on the observed data at the mouth of the Salmon River, except for August and September, when 2006 data were available at Clear Creek. For this period, the data were used directly.
Ukonom Creek	Based on USGS estimated data	Based on USGS estimated data	90% saturation value	Based on the observed data at the mouth of the Salmon River.
Dillon Creek	Based on USGS estimated data	Based on USGS estimated data	90% saturation value	Based on the observed data at the mouth of the Salmon River, except for August and

Tributary name	Temperature	Nutrients	DO	TIC/ALK
				September, when 2006 data were available at Dillon Creek. For this period, the data were used directly.
Salmon River	Based on USGS estimated data	Based on USGS estimated data	90% saturation value	Based on the observed data at the mouth of the Salmon River.
Camp Creek	Based on USGS estimated data	Based on USGS estimated data	90% saturation value	Based on the observed data at the mouth of the Trinity River.
Red Cap Creek	Based on USGS estimated data	Based on USGS estimated data	90% saturation value	Based on the observed data at the mouth of the Trinity River, except for August and September, when 2006 data were available. For this period, the data were used directly.
Bluff Creek	Based on USGS estimated data	Based on USGS estimated data	90% saturation value	Based on the observed data at the mouth of the Trinity River, except for August and September, when 2006 data were available. For this period, the data were used directly.
Trinity River	Based on USGS estimated data	Based on USGS estimated data	90% saturation value	Based on the observed data in the Trinity River.
Pine Creek	Based on USGS estimated data	Based on USGS estimated data	90% saturation value	Based on the observed data at the mouth of the Trinity River.
Tectah Creek	Based on USGS estimated data	Based on USGS estimated data	90% saturation value	Based on the observed data at the mouth of the Trinity River.
Blue Creek	Based on USGS estimated data	Based on USGS estimated data	90% saturation value	Based on the observed data at the mouth of the Trinity River.

Downstream Outflow Boundary Conditions: The downstream boundary condition for this section is a stage-discharge condition. No water quality boundary condition is needed because only outflow is represented at the downstream.

2.3.3.9 Model Segment 9: Klamath Estuary (Turwar to the Pacific Ocean)

The estuarine portion of the Klamath River (Turwar to the Pacific Ocean) was modeled using EFDC and was not included in the original PacifiCorp Model. This model segment was ultimately calibrated using data from the year 2004 because it had the most available data for all parameters. Insufficient data were available to calibrate for the year 2000 in the estuarine portion of the Klamath River. Boundary conditions were thus prepared using monitoring data at Turwar or another appropriate location. No calibrated or validated model simulation was available for

the upstream reach of the Klamath River for the 2004 estuary calibration (since it was calibrated for 2000 as described in Section 3.0). Three types of boundary conditions were included in the Klamath Estuary portion of the model. They are upstream inflow boundary conditions, downstream open boundary conditions, and surface boundary conditions.

Upstream Inflow Boundary Conditions: The portion of the Klamath River represented by EFDC was delineated from the USGS 11530500 streamflow gage at Klamath to the Klamath River's intersection with the Pacific Ocean (Figure 2-4). Streamflow data from the Klamath River at Klamath USGS gage (11530500) were used as the upstream inflow boundary for model calibration (described in Section 3.0). Model results from the Iron Gate Dam to Turwar portion of the model are used as input for the modeling scenarios.

The upstream boundary condition for water quality was configured using the USFWS/Yurok's 2004 water quality monitoring data at Turwar (for model calibration). The USFWS station was sampled five times from June to September 2004. A time series was generated for water quality using linear interpolation of the five available data points. The following constituents were configured as state variables in the upstream boundary water quality input file using the Turwar data:

1. Phytoplankton
2. LPOC
3. LDOC
4. LPOP
5. LDOP
6. PO_4
7. LPON
8. LDON
9. NH_4
10. NO_3/NO_2
11. DO

Not all data were available to be directly used in the EFDC water quality input file. The following assumptions were made to derive parameters to create the water quality input file:

- The majority of OM was assumed to be labile, therefore no refractory component was modeled.
- The LPOM:LDOM ratio was assumed to be 0.8:0.2. This ratio was used to derive the labile particulate and dissolved components of phosphorus, nitrogen and carbon. It is based on the CE-QUAL-W2 algae partition coefficient (APOM = 0.8).
- Organic phosphorous was derived by subtracting PO_4 from total phosphorous.
- The ON:OM and OP:OM ratios were assumed to be the same as in upstream reaches, which are ON:OM=0.07, and OP:OM=0.0055. These ratios were used to derive ON from OP data.
- The algae biomass to chlorophyll ratio was assumed to be 0.067 mg algae/ug Chla, which is the same as those in the upstream reaches.

Diel DO and temperature data were not readily available at the Turwar gage for 2004 when this model was developed. Thus, daily average values were computed on the basis of the diel data for

the Upper Estuary monitoring site (at Highway 101) and specified as the upstream boundary condition at Turwar. The model can be updated to reflect additional monitoring data as these data become available.

For modeling scenarios, model output from the Iron Gate Dam to Turwar segment are used. OM conversion from the RMA model to EFDC is presented in Figure 2-5 of Section 2.3.2.

Two tributaries to the Klamath Estuary, Hunter Creek and Salt Creek, were also initially considered as part of the boundary conditions but were later eliminated. Flow estimates were available for Hunter Creek, based on drainage area for the period May 1 through September 30, and were found to be relatively insignificant (median value of 5.9 cfs in 2004) in comparison to the Klamath River flows. Salt Creek flows were smaller than those for Hunter Creek.

Downstream Open Boundary Conditions: The outlet of the Klamath River at the Pacific Ocean is characterized by a widening of approximately 1,400 meters. Depending on the conditions, the outlet may be largely closed off by a transient sand bar. The opening through this sand bar was set to approximately 200 meters in width for the model, based on measured 2004 bathymetry data. To reduce the influence of boundary reflection, the downstream open boundary of the model was set well into the Pacific Ocean, beyond the physical opening in the sand bar (Figure 2-4). To allow for flexibility in evaluating the effect of different locations of the sand bar opening, the sand bar is included in the grid system as a column of active cells. It has an internal barrier that blocks the water from penetrating all cells except those representing the opening.

Tidal data from the **National Oceanic and Atmospheric Administration** (NOAA) gage at Crescent City (9419750) were used to represent the tidal boundary of the model. Tidal elevation data from the Crescent City gage are referenced to a mean lower low water (MLLW) vertical datum, while bathymetry data obtained from the NCRWQCB use the NAVD88. The difference between the two data at this location is approximately 0.38 feet, or 0.116 meters. Tidal elevation data from the Crescent City gage station were adjusted to correspond to the bathymetry datum obtained for the lower portion of the Klamath River.

Surface Boundary Conditions: The surface boundary conditions are based on meteorological conditions. The meteorological data required by the EFDC model are specified in two separate files (*aser.inp* and *wser.inp*). The *aser.inp* file is used to specify the atmospheric pressure, air temperature, relative humidity, precipitation, evaporation, solar radiation and cloud cover. The *wser.inp* file is used to specify the wind speed and direction. Meteorological data from the Arcata Eureka Airport (WBAN 24283), approximately 35 miles downstream of the estuary along the Pacific coastline, were used. Hourly, unedited local meteorological data (atmospheric pressure, air temperature, relative humidity, precipitation, cloud cover, wind speed and direction) were available from this NOAA-NCDC station and were used in creation of the *aser.inp* and *wser.inp* files for the estuary model. These data provided the most complete data set of required surface airways parameters for the EFDC model meteorological file. Solar radiation data were not available. Clear sky solar radiation was computed on the basis of the latitude and longitude and corrected using cloud cover to generate the solar radiation data.

2.3.4 Initial Conditions

The Klamath River model requires specifying initial conditions in the input files. The initial conditions from the original PacifiCorp Model (Model Segments 1 through 8) were maintained for all segments, except BOD was eliminated from the initial condition setting for Link River,

Keno Reach, and Bypass/Full Flow Reach (see Section 2.2.1). Where field data were unavailable, the conditions of the first day of available field data were applied. In general, the impact of the initial conditions was insignificant and lasted for less than 10 days in the winter period. The initial condition for Model Segment 9 was set to values similar in magnitude as observed data. Because of the relatively large flow from the Klamath River, however, the impact of initial conditions is noticeable only for a very short (insignificant) time period.

2.4 Modeling Assumptions, Limitations, and Sources of Uncertainty

2.4.1 Assumptions

The major underlying assumptions associated with Klamath River model development are as follows:

- The initial condition and the boundary conditions set for the winter and early spring period do not have a significant effect on the simulated water quality during the critical summer and early fall periods. This assumption permits assigning the initial conditions and winter/early spring boundary conditions using best professional judgment, without impairing the model performance for the critical period.
- Time series flow data were not available for all tributaries and withdrawals. Reliable time series flow data were also not available for many monitoring locations along the length of the Klamath River. In light of the limitations, it was assumed that tributary flows could be reasonably represented through interpolation on the basis of limited flow measurements.
- One phytoplankton species and one periphyton species were assumed to be sufficient for representing the overall primary production and nutrient interactions in the system given no data is available to support multiple species modeling.
- Alkalinity is conservative (as stated in CE-QUAL-W2 manual). Therefore, no internal sources or sinks were considered.
- All the OM in the water column (and that from other sources) has the same stoichiometric ratio unless data are available to derive site-specific ratio.
- The effect of zooplankton and benthic creatures do not have a significant impact on the algal/periphyton dynamics and nutrient recycling.
- A stage-discharge relationship was applied at the Link River boundary to enable predictive simulation downstream. This adjustment was made on the basis of previous peer review comments for the Klamath River Model. Although this configuration does not explicitly simulate backwater effects, it was deemed suitable for TMDL development scenarios. The magnitude of the Link River flow is significant. And because Link River is fairly steep, flow velocities into Lake Ewauna are relatively high. If backwater flow exists, it would not have a significant effect on the nutrient budget downstream.
- The majority of OM in the boundary conditions is assumed to be labile. This assumption is justified, in part, by the model calibration and sensitivity analysis conducted for Lake Ewauna/Keno Reservoir. The analysis demonstrates that reducing the labile portion of the OM does not enable reproduction of observed anoxic conditions in the reservoir.
- Denitrification in the riverine sections is not simulated due to the fact that the majority of the river bed is rocky and DO in the water column is high. Neither of these conditions are favorable for denitrification bacteria and corresponding denitrification processes.

This assumption may potentially cause over-prediction of NO_2/NO_3 in the riverine sections, however the impact is expected to be minimal.

- The sand bar opening at the mouth of the Klamath Estuary has relatively constant dimensions and physical characteristics for a period of time; thus, a fixed grid configuration can be used for a simulation.
- The impact of sediment transport and siltation on channel geometry is not significant; therefore, the same bathymetric configuration can be used for different scenario simulations. Additionally, insufficient data are available to dynamically simulate the time-dependent effect of sediment transport on bathymetry.

2.4.2 Limitations

Potential limitations that have been identified include the following:

- The model's capabilities are constrained by the limited availability and quality of monitoring data. This is particularly the case for boundary conditions to the model, but it is also the case for in-stream model calibration data. The Klamath River model is not expected to be able to mimic the exact timing and location of all water quality conditions. The model can be used to represent the overall water quality trends in response to external loading and internal system dynamics.
- While the multi-model framework might be efficient for calibration, it is also cumbersome in terms of data management and transfer between models. Additionally, because of differences in algorithms and state parameters for RMA, CE-QUAL-W2, and EFDC (e.g., for organic components), conversion of pollutant loads between models could result in slight inaccuracies.
- The model does not simulate multiple species of phytoplankton and periphyton. Therefore, this model is currently not suitable for evaluating competition among multiple species or evolution of the aquatic algal communities and their interaction with nutrients.
- Because of the lack of a direct linkage between OM loading and SOD and benthic nutrient flux, the model in its present stage cannot fully evaluate the long-term effect of load reductions on SOD.
- Neither zooplankton nor benthic animals are simulated in the model; therefore, there could be some uncertainty in the simulation of algal dynamics and nutrient cycling.
- In the estuarine portion of the model, the sand bar opening is fixed. Although this is a reasonable assumption, it can introduce uncertainty in simulating the dynamic features of the system, particularly over an extended period of time.
- Because of a lack of data and the seasonal variability in sand bar location, it is infeasible to configure a long-term simulation model for the estuary. Therefore, the sediment diagenesis model is not activated in EFDC for predicting the sediment-water interaction.
- Algae are represented as one lumped state variable, thus interspecies differences are not simulated. The nitrogen fixing process was not explicitly represented in the model. In general, nitrogen concentrations are high in the water column. Under these circumstances, N-fixing algae tend to uptake dissolved nitrogen directly from the water column as opposed to the air since nitrogen fixation is a highly energy-demanding process.
- Denitrification is not included in the riverine models. This might result in slight over-prediction of nitrogen in the water column.

- Some fine scale nutrient patterns might not be accurately represented due to limitations in model formulations related to nutrient-periphyton interaction. RMA-11, for example, assumes periphyton uptakes only inorganic nitrogen and phosphorus and releases only organic nitrogen and phosphorus through respiration. In reality, both the uptake and release processes may involve both the inorganic and organic forms.

2.4.3 Sources of Uncertainty

As with virtually every hydrodynamic and water quality model, uncertainty is present with regard to various aspects of the Klamath River Model. These uncertainties were minimized to the extent possible in this effort, and thus the model reproduces general trends in the observed data both temporally and spatially. Further reduction of uncertainty is possible through collection of more systematic and accurate data within and external to the system and a more in-depth scientific understanding of the physical, chemical, and biological processes occurring in this unique system.

Some of the major sources of uncertainty include the following:

- *Uncertainty Associated with Boundary Conditions.* Boundary conditions for the Klamath River Model include time series flow, temperature, water quality, and atmospheric conditions. They provide the driving force for the hydrodynamic and water quality simulations. Therefore, accurate definition of boundary conditions is critical to reducing uncertainty. In developing the Klamath River Model, boundary conditions were defined using available monitoring data or were derived using different techniques (e.g., interpolation). Unfortunately, data are not available for all boundary conditions, and where data are available, they generally do not represent high temporal resolution (i.e., every point in time). Although techniques such as interpolation are a reasonable way to represent general trends in a system, precise prediction of water quality at every single point in time and every location is not possible.
- *Uncertainty in Spatial Representation.* The governing partial differential equations of hydrodynamic and water quality models are solved using the finite difference method (FDM) in CE-QUAL-W2 and finite element method (FEM) in RMA-2 and RMA-11. For both FDM and FEM, the waterbodies need to be discretized into different computational cells or nodes on the basis of topographical data. The accuracy in representing the true bathymetry of a waterbody has a significant effect on model performance. Thus, any uncertainty associated with the data sets used to discretize the waterbodies in the Klamath River has a direct effect on the model's predictive capabilities. Additionally, all the impoundments are represented using a laterally averaged system. This inherently assumes that lateral variability is insignificant, though this might not be the case. Also, all rivers are represented in a single, longitudinal dimension.
- *Uncertainty in Process Representation.* Water quality prediction for the Klamath River involves representing numerous dynamic interactions (including many physical, chemical, and biological processes). Mathematical models offer a simplified representation of these processes. Although the current state of knowledge with respect to fully understanding all the detailed interactions in the Klamath River is somewhat limited, the Klamath River modeling effort takes full advantage of all information amassed and understood to date. Major simplifications associated with the Klamath River Model that introduce uncertainty include representing the entire phytoplankton community as a single algae group, representing the entire periphyton community as a single periphyton group, representing SOD using a zero-order formulation, and representing OM with only four components based on solubility and degradability.

- *Uncertainty in Kinetic Structures.* Both CE-QUAL-W2 and RMA-11 represent major water quality decay and transformation with first-order kinetics. These kinetics are widely tested and accepted with regard to reasonably representing the dynamic interaction between water quality constituents. There is, however, uncertainty introduced in using these formulations because these processes are of higher order in reality.

3.0 MODEL TESTING

Once the Klamath River model was configured, a calibration was performed at multiple locations throughout the system. Calibration refers to the adjustment or fine-tuning of modeling parameters to produce an adequate fit of the simulated output to the field observations. The sequence of calibration for the Klamath River model involved calibrating flow and water surface elevation first and then calibrating water quality using available monitoring data. Since the original PacifiCorp Model was already calibrated for hydrodynamics (see Section 3.2), this section of the report mainly focuses on the hydrodynamic calibration of the EFDC portion of the model and the water quality calibration of the entire model.

The upper Klamath River model (Model Segments 1 through 8) was calibrated using data from the year 2000. This year was selected for calibration because relatively good boundary condition data and in-stream data were available in the upper portion of the system (particularly for Lake Ewauna). Data were available, but not to the same extent, for the lower portion of the system (particularly downstream of Iron Gate Dam). Selection of this year was deemed appropriate because water quality conditions in the upper portion of the system drive the response downstream. The model was also corroborated (validated) using data from the year 2002 for Model Segments 1 through 5. Again, considerably more data were available for the upper portion of the system in 2002 than for other years. The estuarine portion (Model Segment 9) was calibrated using data from the year 2004, because bathymetric data and data for key water quality parameters were available. Water quality data were collected as part of an intensive monitoring effort. Insufficient data were available to calibrate for the year 2000 or 2002 in the estuarine portion of the Klamath River.

3.1 Monitoring Locations

The water quality monitoring stations with relevant data used for the 2000 model calibration are presented in Table 3-1 and Figures 3-1 and 3-2.

Table 3-1. Monitoring stations used for Klamath River model calibration (2000)

Station/Location	Site ID	Source
Klamath River at Miller Island boat ramp	KR24589/ KR24594	City of Klamath Falls/ODEQ/ USBR/PacifiCorp
Klamath River at Keno Bridge (Hwy 66)	KR23490	USBR/PacifiCorp/ODEQ
J.C. Boyle Reservoir at deepest point	KR22505	USBR/STORET/ODEQ/ BEAK
Klamath River u/s Shovel Creek	KR20642	NCRWQCB
Copco Lake near Copco	KR19874	USBR/STORET
Iron Gate Reservoir	KR19021	USBR/STORET
Klamath River below Iron Gate Dam	KR18952	USBR/STORET/SWAM/ KRIS/USGS
Klamath River above Shasta River	KR17608	USBR
Klamath River above Scott River	KR14260	USBR/USFWS
Klamath River near Seiad Valley	KR12858	USBR/STORET
Klamath River at Youngs Bar	KR04036	USBR
Upper Estuary	UE	NCRWQCB/Yurok Tribe
Middle Estuary	ME	NCRWQCB/Yurok Tribe
Lower Estuary	LE	NCRWQCB/Yurok Tribe

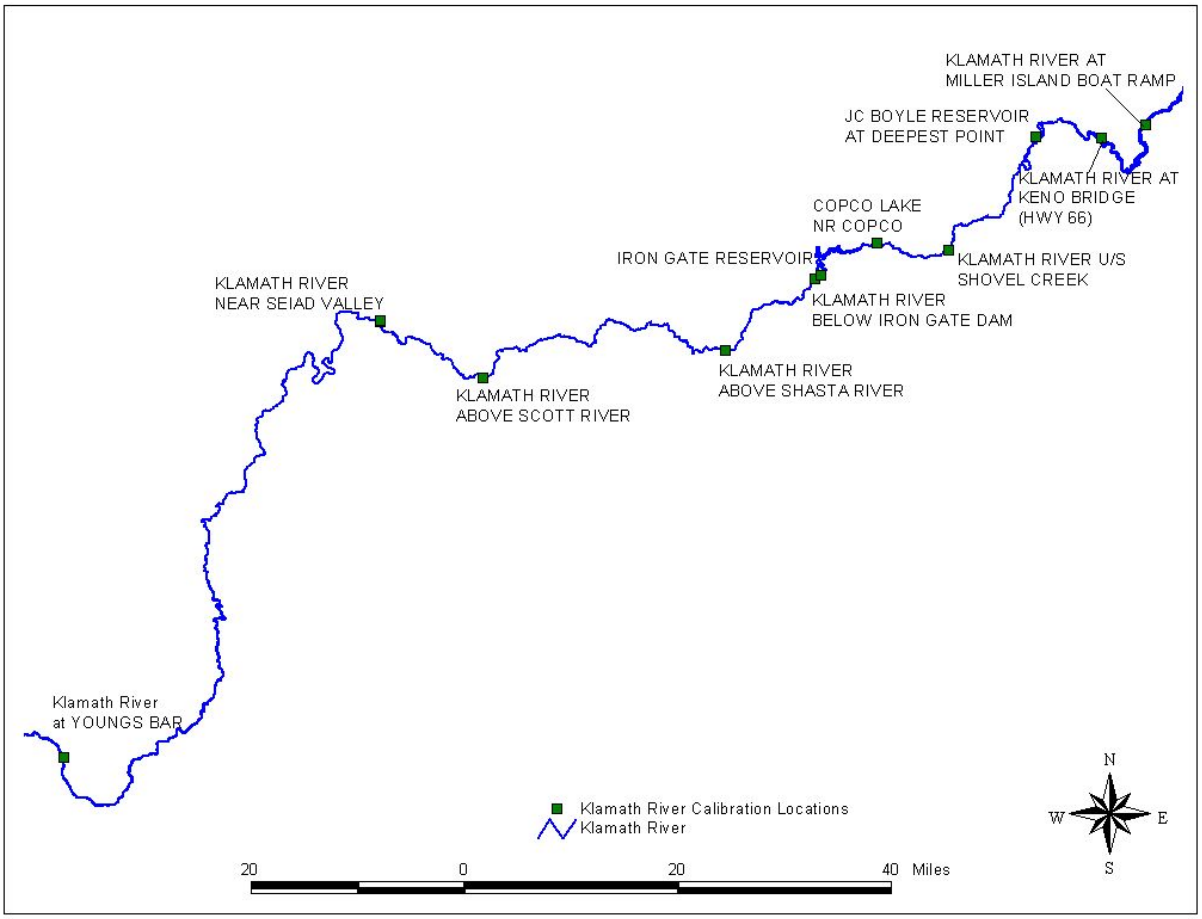


Figure 3-1. Calibration locations for Klamath River modeling (above the Klamath Estuary)

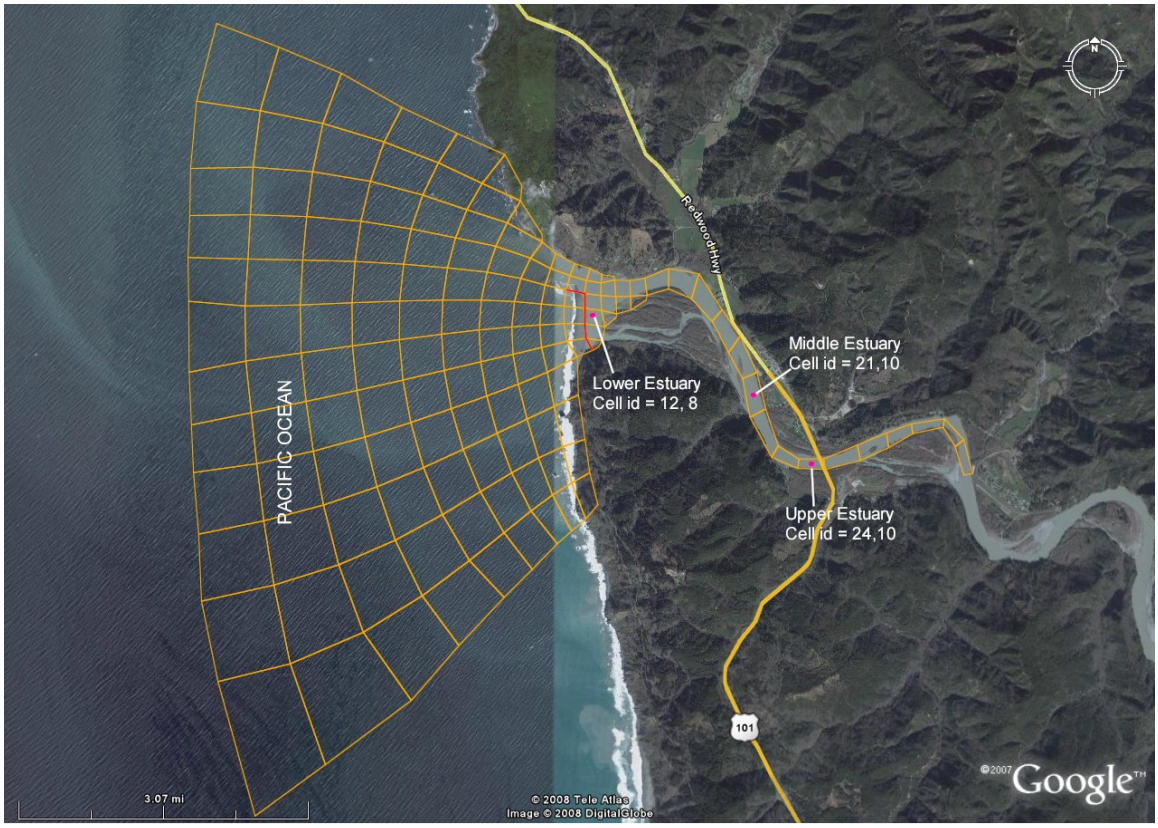


Figure 3-2. Location of water quality monitoring locations in the estuarine portion of the Klamath River (source of image: Google)

In 2002 data were collected at several additional stations in the upper portion of the river. Model results were therefore also evaluated at these stations (Table 3-2). Figure 3-3 shows the locations of the additional stations in the Lake Ewauna to Keno Dam modeling segment. Figure 3-4 shows the locations of the additional stations in the Keno Dam to J.C. Boyle modeling segment.

Table 3-2. Additional monitoring stations used for Klamath River model calibration (2002)

Station/Location	Site ID	Source
Lake Ewauna at Railroad Bridgespan	KR25173	City of Klamath Falls
Klamath River at South-Side Bypass Bridge	KR25079	City of Klamath Falls/ ODEQ/USBR/PacifiCorp
Lost River Diversion	LK	City of Klamath Falls/PacifiCorp
Klamath River at HWY 97 BR NE	KR24901	City of Klamath Falls
Klamath River below Boyle Dam	KR22129/KR22460	PacifiCorp
Klamath River u/s of Boyle Powerhouse Tailrace	KR22128	USFWS
Klamath River d/s of Boyle Powerhouse Tailrace	KR22127	ODEQ
J.C. Boyle Powerhouse Tailrace	BTR	USFWS
Klamath River near Stateline	KR20932	PacifiCorp/SWAMP

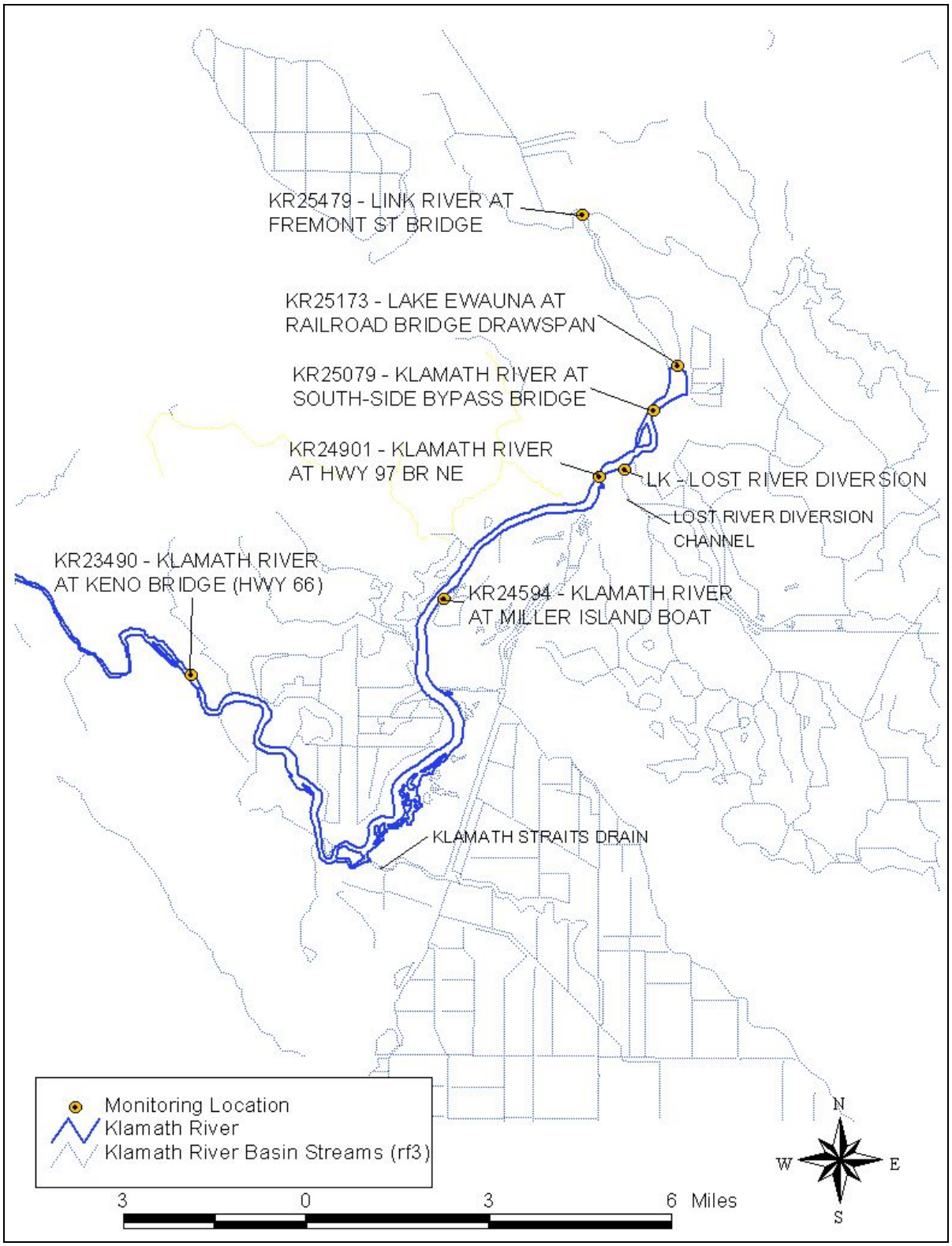


Figure 3-3. Additional calibration locations for Klamath River modeling—Lake Ewauna to Keno Dam modeling segment (2002)

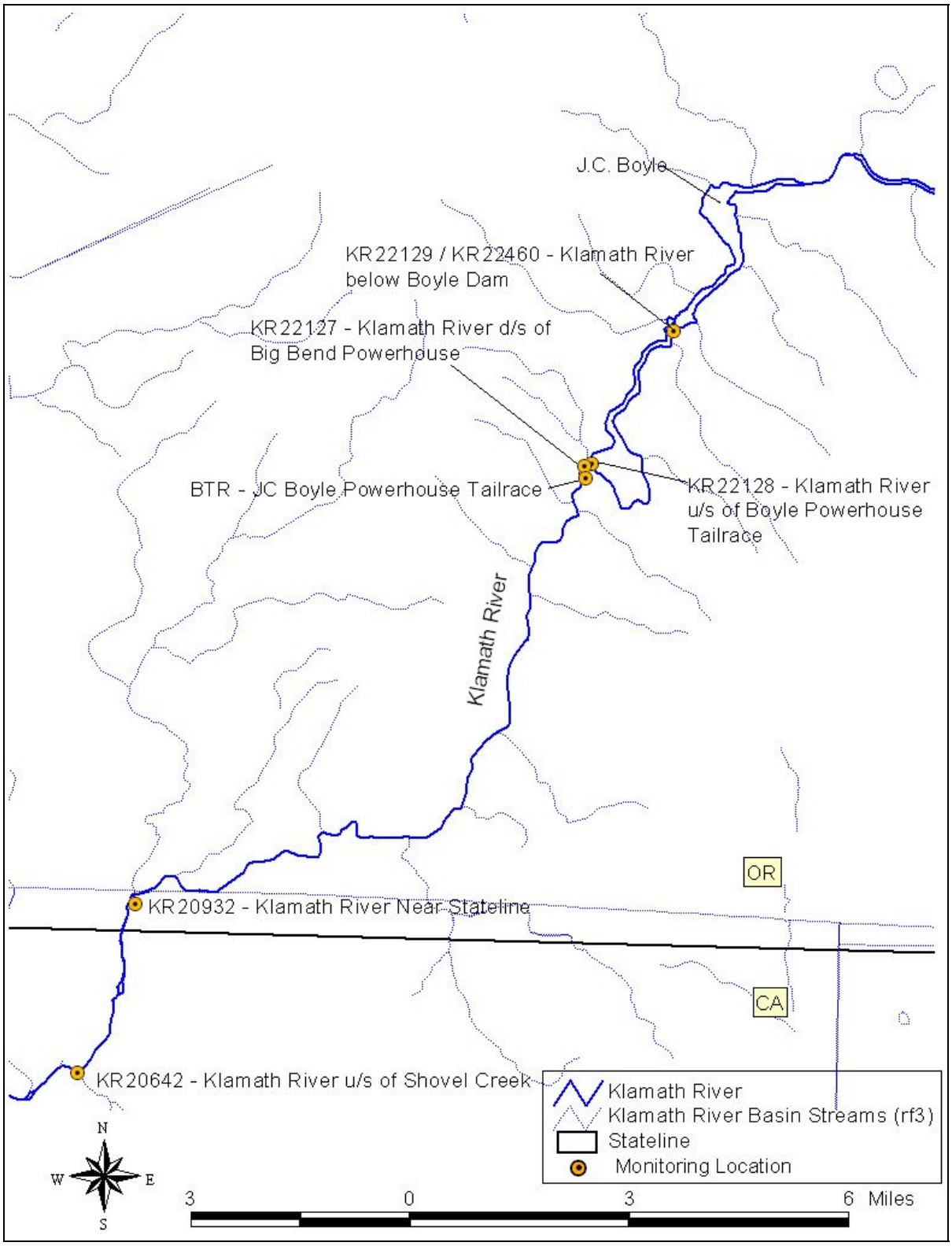


Figure 3-4. Additional calibration locations for Klamath River modeling—Keno Dam to J.C. Boyle modeling segment (2002)

3.2 Hydrodynamic Calibration

It was not necessary to perform additional hydrodynamic calibration for Klamath River Model Segments 1 through 8. The grid layout and hydrodynamic configuration and calibrations for the PacifiCorp Model were found to be reasonable, and no better information is available to further refine this component of the model. Therefore, there has been no additional effort to further refine the hydrodynamic model. Hydrodynamic calibration results (for temperature) are presented in Appendices E through K for Lake Ewauna through Turwar.

Hydrodynamic results for temperature and salinity in the estuarine portion of the Klamath River (Model Segment 9) are presented in Appendix L (Figures L-1 through L-6). Temperature and salinity sonde data collected at the surface and bottom were compared to model output for three separate locations. In general, the temperature calibration followed the observed data trend fairly well with the model underpredicting slightly at the Upper Estuary site. The model was able to simulate lower temperatures at the bottom and higher temperatures at the surface in the lower estuary where salinity and temperature stratification exist. It was able to capture the magnitude of peaks and trends and the spatial variability of salinity fairly well (i.e., high salinity at the Lower Estuary site and low salinity at the Upper Estuary site). Also, the model predicted high salinity at the bottom and lower salinity at the surface, which matches the observed salinity profile. The model also predicts significant temporal variability of salinity as a result of the complex interactions between upstream freshwater inflows and downstream tidal impact. Some disparity does exist between the simulated and observed salinity, and this can be explained by uncertainty in physical representation. For example, the exact dimensions of the sand bar opening for the simulated period are not known. Also, representation of downstream tidal characteristics is limited by available data. Overall, the model predicts the observed temperature and salinity trends and thus reasonably represents physical circulation and transport in the estuary.

3.3 Water Quality Calibration

The Klamath River modeling system was developed in a piecewise manner, where models for different sections of the river were linked by routing flow and constituent mass from upstream to downstream. The water quality calibration process involved first calibrating the upstream waterbodies and then using the resulting flow and predicted concentration time series (together with the watershed and other tributary inputs) to drive the downstream waterbody simulations.

Calibration of the water quality model was implemented by fine-tuning major kinetic parameters such as algal growth rate, death rate, nitrification/denitrification rates, OM decay rates, and SOD rates. The calibration process started with the existing PacifiCorp Model and continued with fine-tuning of parameter values after the major boundary conditions were set. This entailed comparison of model predictions to monitoring data and iterative adjustment of parameter values. The overall goal was to most accurately match observed data while maintaining consistency among all the waterbodies. In the following sections, the water quality calibration results at each station and in each modeled segment are presented. The major calibrated parameters in the CE-QUAL-W2 models are presented in Tables 3-3 and 3-4, and those for the RMA-11 models are presented in Table 3-5. Because the water quality parameters in all the riverine sections are the same, they are not listed by modeling segment.

It should be noted that while most CE-QUAL-W2 parameters were consistent from one model segment to the next, those associated with algae dynamics and OM dynamics differed somewhat. The algae parameters include growth rate, respiration rate, and death rate and are used to characterize the algae communities in each lake. In different waterbodies, algae communities can

consist of different algae species, and each of these species exhibits different characteristics. Additionally, a single algae species can exhibit different characteristics in different waterbodies because of a variety of factors. Since the model represents a single, lumped algae species and represents only the impacts of temperature, light, and nutrients on algae, algae-related parameters differ from one waterbody to the next. Although these parameter values differ, they are all within the range of literature values.

The OM parameters differ among model segments in that the decay rates for labile particulate and dissolved OM are higher in the upstream modeling segments than in the lower segments. This is justified by the fact that as a significant portion of the more labile OM upstream in the system is lost through degradation, the remaining OM downstream becomes less labile. This less labile OM is characterized by a lower decay rate.

Table 3-3. Nutrient input parameters used in the CE-QUAL-W2 Models

Parameter	Description	Units	Lake Ewauna-Keno Dam (Model Segment 2)	J.C. Boyle Reservoir (Model Segment 4)	Copco Reservoir (Model Segment 6)	Iron Gate Reservoir (Model Segment 7)	Typical literature values[a]
PO4R	Sediment release rate of phosphorus	fraction of SOD	0.01	0.01	0.01	0.01	0.001 to 0.03
ORGP	Fraction of phosphorus in OM	--	0.0055	0.0055	0.0055	0.0055	0.005 to 0.011
ORGN	Fraction of nitrogen in OM	--	0.07	0.07	0.07	0.07	0.08
NO3DK	Nitrate decay rate	day^{-1}	0.1	0.1	0.1	0.1	0.05 to 0.15
NO3T1	Lower temperature for nitrate decay	^{o}C	5	5	5	5	5
NO3T2	Upper temperature for nitrate decay	^{o}C	25	25	25	25	25
NO3K1	Lower temperature rate multiplier for nitrate decay	--	0.1	0.1	0.1	0.1	0.1
NO3K2	Upper temperature rate multiplier for nitrate decay	--	0.99	0.99	0.99	0.99	0.99
NH4DK	Ammonium decay rate	day^{-1}	0.10	0.1	0.1	0.1	0.00 to 0.80
NH4R	Sediment release rate of ammonium	fraction of SOD	0.05	0.05	0.05	0.05	0.00 to 0.40
NH4T1	Lower temperature for ammonium decay	^{o}C	5	5	5	5	5
NH4T2	Upper temperature for ammonium decay	^{o}C	25	25	25	25	25
NH4K1	Lower temperature rate multiplier for ammonium decay	--	0.1	0.1	0.1	0.1	0.1
NH4K2	Upper temperature rate multiplier for ammonium decay	--	0.99	0.99	0.99	0.99	0.99
LDOMDK	LDOM decay rate	day^{-1}	0.25	0.20	0.10	0.05	0.01 to 0.63

Parameter	Description	Units	Lake Ewauna-Keno Dam (Model Segment 2)	J.C. Boyle Reservoir (Model Segment 4)	Copco Reservoir (Model Segment 6)	Iron Gate Reservoir (Model Segment 7)	Typical literature values[a]
RDOMDK	RDOM decay rate	day^{-1}	0.001	0.001	0.001	0.001	0.001
LPOMDK	LPOM decay rate	day^{-1}	0.25	0.20	0.10	0.05	0.001 to 0.11
RPOMDK	RPOM decay rate	day^{-1}	0.001	0.001	0.001	0.001	0.001

[a] Literature values are from the CE-QUAL-W2 Users Manual which compiled data from a range of sources. The only exception is the stoichiometric coefficient, which was derived from Chapra 1997.

Table 3-4. Phytoplankton input parameters used in the CE-QUAL-W2 Models

Parameter	Description	Units	Lake Ewauna-Keno Dam (Model Segment 2)	J.C. Boyle Reservoir (Model Segment 4)	Copco Reservoir (Model Segment 6)	Iron Gate Reservoir (Model Segment 7)	Typical literature values[a]
AG	Growth rate	day^{-1}	2.3	1.2	1.2	1.2	0.2 to 9.0
AR	Dark respiration rate	day^{-1}	0.18	0.1	0.1	0.1	0.01 to 0.92
AE	Excretion rate	day^{-1}	0.05	0.05	0.02	0.02	0.01 to 0.044
AM	Mortality rate	day^{-1}	0.15	0.06	0.06	0.06	0.03 to 0.30
AS	Settling rate	day^{-1}	0.3	0.3	0.3	0.3	0.001 to 13.20
AHSP	Phosphorous half-saturation coefficient	$g.m^{-3}$	0.003	0.003	0.003	0.003	0.001 to 1.520
AHSN	Nitrogen half-saturation coefficient	$g.m^{-3}$	0.014	0.014	0.014	0.014	0.01 to 4.32
ASAT	Light saturation	$W.m^{-3}$	75	75	75	100	10 to 150
AT1	Lower temperature for minimum algal rates	^oC	5	5	5	5	N/A
AT2	Lower temperature for maximum algal rates	^oC	17	17	17	17	N/A
AT3	Upper temperature for minimum algal rates	^oC	35	35	35	35	N/A
AT4	Upper temperature for maximum algal rates	^oC	45	45	45	45	N/A
AK1	Lower temperature rate multiplier for minimum algal rates	--	0.1	0.1	0.1	0.1	N/A
AK2	Lower temperature rate multiplier for maximum algal rates	--	0.99	0.99	0.99	0.99	N/A
AK3	Upper temperature rate multiplier for minimum algal rates	--	0.99	0.99	0.99	0.99	N/A
AK4	Upper temperature rate multiplier for maximum algal rates	--	0.1	0.1	0.1	0.1	N/A
ALGP	Phosphorus to biomass ratio	--	0.0055	0.0055	0.0055	0.0055	0.005 to 0.08
ALGN	Nitrogen to biomass ratio	--	0.07	0.07	0.07	0.07	0.08

Parameter	Description	Units	Lake Ewauna-Keno Dam (Model Segment 2)	J.C. Boyle Reservoir (Model Segment 4)	Copco Reservoir (Model Segment 6)	Iron Gate Reservoir (Model Segment 7)	Typical literature values[a]
ALGC	Carbon to biomass ratio	--	0.45	0.45	0.45	0.45	0.45

[a] Literature values are from the CE-QUAL-W2 Users Manual Cole and Wells (2003); which compiled data from a range of sources. The only exception is the stoichiometric coefficient, which includes information derived from Chapra 1997.

Table 3-5. Parameters used in the RMA-11 Models

Variable	Description, units	Value	Typical literature values
ALP0	Chl a to algal biomass conversion factor, phytoplankton, mg Chl_a to mg-A	67	22 to 220
ALP1	Fraction of algal biomass that is nitrogen, phytoplankton, mg-N/mg A	0.07	0.08
ALP2	Fraction of algal biomass that is phosphorous, phytoplankton, mg-P/mg A	0.0055	0.005 to 0.08
MUMAX	Maximum specific growth rate, phytoplankton, 1/d	1.00	0.2 to 9.0
RESP	Local respiration algae, phytoplankton, 1/d	0.18	0.01 to 0.92
MORT	Local mortality rate of algae, phytoplankton, 1/d	0.05	0.03 to 0.30
KLIGHT	Half saturation coefficient for light, phytoplankton, KJ m-2 s-1	0.10	N/A
PREFN	Preference factor for NH3-N, phytoplankton	0.60	N/A
ABLP0	Chl a to algal biomass conversion factor, bed algae, mg Chl_a to mg-A	67	22 to 220
BMUMAX	Maximum specific growth rate, bed algae, 1/d	1.15	0.45 to 2.0
BRESP	Local respiration rate of algae, bed algae, 1/d	0.20	N/A
BMORT	Local mortality rate of algae, bed algae, 1/d	0.20	N/A
KBLIGHT	Half-saturation coefficient for light, bed algae, KJ m-2 s-1	0.05	N/a
PBREFN	Preference factor for NH3-N, bed algae	0.75	N/A
BET1	Rate constant: biological oxidation NH3-N, 1/d	0.30	0.0 to 0.8
BET2	Rate constant: biological oxidation NO2-N, 1/d	0.50	N/A
BET3	Rate constant: hydrolysis OM to NH3-N, 1/d	0.20	0.001 to 0.63
KNITR	Michaelis-Menton half saturation constant: nitrogen, phytoplankton, mg/l	0.014	0.01 to 4.32
KPHOS	Michaelis-Menton half saturation constant: phosphorous, phytoplankton, mg/l	0.003	0.001 to 1.52
KBNITR	Half-saturation coefficient for nitrogen, bed algae, mg/lFraction of algal biomass that is phosphorus, bed algae, mg/l	0.014	N/A
KBPHOS	Half-saturation coefficient for phosphorus, bed algae, mg/lHalf-saturation coefficient for nitrogen, bed algae, mg/l	0.003	N/A

Variable	Description, units	Value	Typical literature values
ALP3	Rate O2 production per unit of algal photosynthesis, phytoplankton, mg-O/mg-AHalf-saturation coefficient for phosphorus, bed algae, mg/l	1.40	1.40
ALP4	Rate O2 uptake per unit of algae respired, phytoplankton, mg-O/mg-ARate O2 production per unit of algal photosynthesis, phytoplankton, mg-O/mg-A	1.4	1.4
ALP5	Rate O2 uptake per unit NH3-N oxidation, mg-O/mg-NRate O2 uptake per unit of algae respired, bed algae, mg-O/mg-A	3.43	3.43

3.3.1 Link River (Model Segment 1)

Link River is a short, 1.31 mile segment that is characterized by a steep slope and rapid flow. With an average flow velocity at the end of Link River equivalent to approximately 0.9 m/s, it takes less than an hour for water to flow from Link Dam to Lake Ewauna (the next downstream segment). In this short time frame, significant water quality variation is not expected to occur. Model simulation results demonstrate this characteristic and show that the segment's outflow water quality is nearly the same as the inflow conditions.

The Link River model was developed on the basis of the RMA-11 modeling framework. In the original PacifiCorp Model, nine water column constituents and one benthic constituent were simulated. The nine water column constituents include an arbitrary constituent, BOD, DO, OM, NH_4, NO_2/NO_3, PO_4, phytoplankton, and temperature. The benthic constituent is used to simulate benthic algae such as periphyton. In this system configuration, the simultaneous representation of the BOD and OM provide some redundancy because BOD essentially is a surrogate for OM. Therefore, BOD and OM were combined into one constituent (see Section 2.2.1).

The model was run for the period from January 1, 2000, to December 31, 2000 for the 2000 calibration run. I It was also run from January 1, 2002 to December 31, 2002 for the corroboration run. No data were available to calibrate the model for the reach.

3.3.2 Lake Ewauna-Keno Dam (Model Segment 2)

The Lake Ewauna segment was developed on the basis of the CE-QUAL-W2 modeling framework. In the original PacifiCorp Model, 18 water column constituents were simulated, which included four OM components and one BOD component. In the present study, BOD was eliminated from the active constituent list in the model input data file (see Section 2.2.1).

In the calibration run, the model was run for the period from January 1, 2000, to December 31, 2000. The model output was compared to observations at two water quality stations in Lake Ewauna: Klamath River at Miller Island boat ramp and Klamath River at Keno bridge (Hwy 66). As shown in Figures E-1 through E-16 in Appendix E, the model reproduced the observed water quality pattern reasonably well. The final calibrated parameters for Lake Ewauna are presented in Tables 3-3 and 3-4.

The model reproduces the supersaturation of DO during early summer well, as well as the anoxic period. The model also was able to reproduce DO recovery in August, although the timing is shifted by more than 10 days. This shift can be explained by the fact that the current CE-QUAL-

W2 model does not simulate sediment diagenesis, and thus does not track changes in SOD with regard to preceding OM loading. Additionally, the data show low DO in late fall and winter while the model simulates relatively high DO concentrations. During this period, water temperature dropped rapidly, and algae and OM loading from UKL decreased. One would expect that these monitored phenomena would result in DO recovery from the summer anoxic conditions (as predicted by the model). Lake Ewauna, however, is a unique system that is highly dynamic and subject to tremendous OM loading from UKL. The lake also experiences an extended period of summer hypoxic/anoxic conditions which result in quick removal of algae and generation of extra OM. Therefore, it is highly likely that the lake's late fall and winter DO concentrations do not respond similarly to most other impoundments.

One possible cause is that the extremely high organic loading from both UKL and algae death during the summer result in a tremendous amount of OM being settled into the sediment layer during the summer (forming a highly enriched bed). During late fall and winter, even though OM and algae loadings cease, this highly enriched bed may provide significant oxygen consumption potential (preventing high DO concentrations). This could be the ultimate cause of the observed low DO during the late fall and winter. Without a predictive sediment diagenesis module, however, the existing CE-QUAL-W2 model is not capable of fully representing the dynamic interaction and feedback between the sediment and water column. This limitation is not expected to impact TMDL determination, because the critical, anoxic period occurs during the summer.

The predicted phytoplankton biomass matches the observed trends very well, especially the decline during the summer anoxic period and the difference in magnitude between Miller Island and Hwy 66. In general, the model also predicts nutrients well, except for timing in some situations. This is likely because of limitations surrounding the definition of boundary conditions (i.e., the use of limited data to derive the boundaries). The model was able to predict relatively high NH_4 and PO_4 concentrations during the anoxic period likely due to a combination of multiple water quality processes including OM decay, algae die-off, and release from sediment.

It should be noted that simulated NO_2/NO_3 is relatively low compared with the observed values. The accuracy of these data, however, is unknown. Measurements made by different agencies during this time period were found to be significantly different (approximately an order of magnitude). It is suspected that the high NO_2/NO_3 measurements are incorrect. During the summer anoxic period, nitrification, which is the source of NO_2/NO_3, is inhibited by the low DO concentration. NO_2/NO_3 levels are expected to be high only if an external source is supplying a significant amount. This, however, is not supported by currently available data. The model predicts the trends exhibited by the lower level NO_2/NO_3 measurements. The model results also show good agreement between the simulated and observed pH at the two locations, indicating a reasonable representation of the fate and transport of pH related constituents and their interactions.

The Lake Ewauna to Keno Dam model segment was further tested using monitoring data in 2002. In 2002 the city of Klamath Falls collected a significant amount of data in this reach. Data were collected at six monitoring stations, including Lake Ewauna at Railroad Bridge Drawspan (LERBD), Klamath River at South Side Bypass Bridge (KRSSBB), Klamath River at Highway 97 (KR97), Klamath River at Lost River Diversion Channel (KRLRDC), Klamath River at Miller Island (KRMI), and Klamath River at Highway 66 (KR66). The simulated temperature, nutrients, DO, and algae biomass are plotted against the observed data at these locations in Figures E-17 to E-57. The model reproduces the observed water quality conditions and trends well. The only exception is for DO simulation between March and June. During this period, the model tends to underpredict DO. This suggests that the estimated OM boundary condition at Link River might

be too high for this period and results in excessive deoxygenation potential in the water column. More representative monitoring data characterizing this boundary would improve the model predictions. Fortunately this time period is outside the critical summer months of July and August. Thus, it was not deemed necessary to further refine the upstream boundary.

3.3.3 Keno Dam to J.C. Boyle Reservoir (Model Segment 3)

The portion of the water quality model for Keno Reach was developed using the RMA-11 modeling framework. In the original PacifiCorp Model, nine water column constituents and one benthic constituent were simulated as in the Link River model. In this river segment, BOD and OM were combined to form a unified constituent as was done for the Link River model segment.

The model was run for the period from January 1, 2000, to December 31, 2000. No data were available to calibrate the model for the year 2000. Data were available for the year 2002 at a location downstream of Keno Dam. Figures F-1 through F-7 in Appendix F show the calibration results for the year 2002. The model reproduces the observed nutrients and pH well. The model results for DO also match the observed magnitudes and trends well. It does, however, predict a lower DO than the data show during the summer (Figure F-2). One reason might be that the model is not representing sufficient reaeration downstream of the dam. This would result in slower recovery from the low DO conditions seen upstream of the dam. It should also be noted that only one monitoring point is available during this extended time period. A single DO sample is insufficient to reflect the likely range of DO levels that would occur over a day and throughout this critical time period.

3.3.4 J.C. Boyle Reservoir (Model Segment 4)

The J.C. Boyle Reservoir segment was developed using the CE-QUAL-W2 modeling framework. The model was run for the period from January 1, 2000, to December 21, 2000. The constituent configuration in this river segment is the same as the Lake Ewauna segment (Section 3.3.2).

For calibration, the model was run for the period from January 1, 2000, to December 31, 2000, and the simulated water quality is compared with observed profiles in the reservoir at water quality monitoring station J.C. Boyle Reservoir (at deepest point). Major parameters adjusted during the calibration process included algae growth rate, algae respiration rate, algae death rate, particulate OM settling velocity, OM decay rate, and suspended solids settling velocity. As shown in Figures G-1 through G-7 in Appendix G, the model reproduced the observed pattern reasonably well.

Although the model overpredicts NH_4 and NO_3 concentrations on some dates and underpredicts them on other dates, it predicts concentrations within the range of observed data. These differences are caused by uncertainty in boundary conditions. To more accurately represent such fine-scale variability (both temporally and on a depth basis), higher resolution data (i.e., temporally and spatially) are necessary to configure boundary conditions representing major tributaries and inflows. The final calibrated parameters for J.C. Boyle Reservoir are presented in Tables 3-3 and 3-4.

The J.C. Boyle model was further tested using the 2002 data. Only three constituents are available for 2002: temperature, pH, and DO. Model predictions for these constituents follow the overall trends well and suggest a reasonable calibration (Figures G-8 through G-10 in Appendix G).

3.3.5 Bypass Reach and Full Flow Reach (Model Segment 5)

The Bypass/Full Flow Reach segment was developed using the RMA-11 modeling framework. In the original PacifiCorp Model, nine water column constituents and one benthic constituent were simulated as in the Link River model. In this study, BOD and OM are combined to form a unified constituent as was done for Link River model (see Section 2.2.1).

For calibration, the model was run for the period from January 1, 2000, to December 31, 2000. Data for temperature, nutrients, DO, Alkalinity, and pH were available to calibrate the model. Figures H-26 through H-33 in Appendix H show the comparison of model results versus observed data for the Klamath River upstream of Shovel Creek. The results indicate reasonable agreement between predictions and monitoring data. The model was further tested using data collected in 2002 and was run from January 1, 2002, through December 31, 2002. Figures H-34 through H-40 in Appendix H show the comparison of model results versus observed data for the Klamath River upstream of Shovel Creek.

In 2002 data were collected at more locations, including at the Klamath River downstream of J.C. Boyle Dam (KRJCB), Klamath River upstream of J.C. Boyle Powerhouse Tailrace (KRUPT), Klamath River downstream of Big Bend Powerhouse (KRBP), Klamath River at J.C. Boyle Powerhouse Tailrace (KRPT), and Klamath River near Stateline (KRS). The sampled constituents include temperature, DO, pH, and nutrients. The predicted results are plotted against the data in Appendix H (Figures H-1 to H-25).

As shown, the model accurately reproduces the general trends and magnitudes observed in the data. There are, however, some discrepancies between model results and data at various locations and times. For example, Figure H-1 shows that although the model has reproduced the seasonal variability and peaks of temperature downstream of J.C. Boyle, it underpredicts the diurnal fluctuation. This discrepancy may be due to local conditions not entirely captured by the model. Data collection could have occurred in a shallow region that exhibits a wider range of temperature variability while the model segment represents laterally and vertically averaged. This particular location is almost immediately downstream of J.C. Boyle, and thus a significant amount of water is always discharged from the dam (greater than 100 cfs). Under these conditions, the average temperature in this segment should be primarily controlled by the discharge temperature from the dam. The model reflects this condition while the data could actually represent a highly localized condition. The discrepancy between model results and DO data in Figure H-2 is likely also caused by this same condition. Figures H-7 and H-10 show that the model slightly overpredicts and underpredicts, respectively, the observed temperatures. This is likely due to differences between the modeled bathymetry and actual bathymetry in these portions of the river data. The model's overprediction of chlorophyll *a* in Figure H-17 is likely caused by inaccurate boundary conditions from UKL during this period, which propagate to this location in the system.

3.3.6 Copco Reservoir (Model Segment 6)

The water quality model for Copco Reservoir was developed on the basis of the CE-QUAL-W2 modeling framework. The water quality constituent configuration in this segment is the same as for the Lake Ewauna segment.

The model was run for the period from January 1, 2000, to December 31, 2000. The simulated water quality output was compared with the observed profile data in the reservoir at water quality station Copco Lake near Copco. Key parameters that were changed through this calibration process include algae growth rate (AG), particulate organic matter settling velocity (POMS), labile dissolved organic matter decay rate (LDOMDK), and labile particulate organic matter decay rate (LPOMDK).

The final calibrated parameters for Copco Reservoir are presented in Tables 3-3 and 3-4. The model results are plotted against observed data at water quality station Copco Lake near Copco in Figures I-1 through I-7 in Appendix I. The model matched the monitoring data reasonably well. It overpredicts NH_4 and NO_3 concentrations on some dates and underpredicts them on other dates for Copco Reservoir. In general, however, it predicts concentrations within the range of observed data. Differences are caused by uncertainty in boundary conditions.

3.3.7 Iron Gate Reservoir (Model Segment 7)

The model was run for the period from January 1, 2000, to December 21, 2000. The simulated water quality output was compared with the observed profile data in the reservoir at the water quality station in Iron Gate Reservoir (Figures J-1 through J-7 in Appendix J).

The Iron Gate Reservoir portion of the model was updated using the new parameter values from Copco Reservoir. It was found that by changing the values of several kinetic parameters, including SOD, AG, LDOMDK, and LPOMDK (as shown in Table 3-3 and 3-4), the model was able to predict a reasonable water quality response for DO, phytoplankton, and nutrients. Additional parameters that were fine-tuned include sediment NH_4 release rate in proportion to SOD (NH4R), sediment PO_4 release rate in proportion to SOD (PO4R), algae death rate (AM). The detailed parameter values are listed in Tables 3-3 and 3-4. The model achieves a reasonable agreement with the data, indicating that the water quality dynamics in the reservoir are reasonably represented.

3.3.8 Iron Gate Dam to Turwar (Model Segment 8)

The water quality model for this segment was developed using the RMA-11 modeling framework, and the water quality constituent configuration is the same as for the upstream riverine RMA models.

The model was run for the period from January 1, 2000, to December 31, 2000. Data for only temperature and DO were available at five stations along this reach to calibrate the model. The five stations are the Klamath River downstream of Iron Gate Dam, Klamath River above Shasta River, Klamath River above Scott River, Klamath River above Seiad Valley, and Klamath River at Youngs Bar. Model results are plotted against observed data in Appendix K. The model reproduces the observed temperature very well at these locations.

DO predictions are also accurate. The Klamath River is steep and is generally characterized by a large flow. This results in significant reaeration along the length of the river. Modeling results for this segment show that DO in the river increases to approximately saturation because of this reaeration effect. At monitoring locations farther downstream, e.g., above Seiad Valley and at Youngs Bar, data however do not show this expected trend. In fact, the model tends to overpredict DO. Observed DO can reach very low levels, e.g., 2.5 to 4.0 mg/L; however, the model tends to predict concentrations around 8.0 mg/L (close to saturation).

This disparity in results was investigated through a number of sensitivity analyses. The following adjustments to the model were made to determine their potential effect on in-stream DO concentrations:

- Tributary DO boundary conditions were reduced to 0.0 mg/L (for all incoming tributaries where monitoring data are available, except major rivers such as Shasta and Scott).
- SOD was increased to an extremely high value along the length of the river (to 5.0 g/m2/day). This represents a highly enriched substrate, although this is uncharacteristic of this region of the Klamath River.
- A very high OM concentration was set for all boundary conditions. The value used is equivalent to roughly 45.0 mg/L of CBOD.
- A series of different reaeration equations were used.

None of these adjustments was able to sufficiently lower the DO concentrations to the monitoring levels. It should also be noted that the low DO does not appear to be caused by biological activity (e.g., periphyton), based on the minimal observed diel DO fluctuation range. After running these sensitivity analyses, the quality of the DO monitoring data were further explored.

Upon further review, DO data for this time period were found to be inaccurate. The Klamath River Water Quality 2000 Monitoring Program—Project Report (Watercourse Engineering, Inc. 2003) indicated that biofouling of the DO membrane was an issue at nearly all monitoring locations (including the locations identified in this section for model calibration). Biofouling refers to the impact of biological activity on instrumentation, and it results in inaccurate DO measurements. It typically occurred within 24 to 96 hours of probe deployment and resulted in degradation (i.e., reduction) of DO concentrations. The extent of degradation in measurements is apparent from the DO monitoring data plotted in Appendix K. Sudden step increases of 2 to 3 mg/L occur multiple times over the course of the summer (e.g., beginning of August and September at the Youngs Bar station). These increases occur when a probe is removed, cleaned, and re-deployed. Subsequent to these increases, the DO concentrations again decline. Because of these inaccuracies, the model predictions should not (and do not) closely match the measured DO levels. The model predictions do tend to follow the trend in maximum DO concentrations measured for this period (where biofouling is not an issue).

3.3.9 Klamath Estuary - Turwar to the Pacific Ocean (Model Segment 9)

EFDC was used to model this portion of the Klamath River. The simulated water quality output was compared with grab sample data and measured DO sonde data at three water quality stations in the lower, middle, and upper estuary for the year 2004 (Figures L-7 through L-9 in Appendix L).

Calibration data were available for 2004 because of an intensive monitoring effort conducted by the NCRWQCB and Yurok Tribe in June, July, August, and September. Data were collected as grab samples at the surface and bottom of the estuary for a suite of nutrients along with algae measurements and sonde data measurements for temperature, DO, and salinity. Continuous sonde data were collected each month for 4–5 days at the surface and bottom.

The water quality constituents evaluated for calibration include chlorophyll *a*, DO (surface and bottom), PO_4, NH_4, and NO_2/NO_3 at each of the three locations in the estuary. Although the grab sampling data provided a variety of data for calibration, all except one data point for PO_4, NH_4 and NO_2/NO_3 were reported as non-detects. It was noted that these samples were diluted, and

that the reporting limit was raised on the basis of different dilutions (e.g., 5x, 10x and 20x) for different sampling days (because of matrix interference, possibly chloride). To consider this uncertainty in the data, error bars were provided in the calibration figures in Appendix L. These bars show the potential range of the laboratory measurements. The lower bound was estimated as half of the lowest report limit.

The water quality predictions follow the overall trends and magnitudes at the calibration locations fairly well. DO concentrations at the surface and bottom locations are replicated, and the model is able to predict low DO conditions during the summer period. The model also reproduces the observed diel fluctuation of DO in both the surface and bottom water. Since the model and observed data both show very low algae concentrations in the estuary, significant diel fluctuations of DO do not occur as a result of phytoplankton. Periphyton biomass, however, is predicted at high levels in the shallow regions of the estuary. This is likely a key contributor to diel DO fluctuation. Periphyton also influences diel fluctuation of nutrients. Table 3-6 presents the calibration parameter values for the EFDC model.

Table 3-6. Calibrated parameter values for the Klamath Estuary

Parameter	Description	Value
PMc	Algae growth rate (1/day)	1.8
BMRc	Algae respiration rate (1/day)	0.1
PRRc	Algae mortality rate (1/day)	0.05
WSc	Algae settling velocity (m/day)	0.2
TMc1	Lower optimal temperature for algae growth (^oC)	20
TMc2	Upper optimal temperature for algae growth (^oC)	25
rNitM	Nitrification rate (1/day)	0.06
KLN	Minimum hydrolysis rate for LPON (1/day)	0.07
KDN	Minimum hydrolysis rate for LDON (1/day)	0.1
KLP	Minimum hydrolysis rate for LPOP (1/day)	0.07
KDP	Minimum hydrolysis rate for LDOP (1/day)	0.1
KLC	Minimum hydrolysis rate for LPOC (1/day)	0.07
KDC	Minimum hydrolysis rate for LDOC (1/day)	0.1
SOD	Sediment oxygen demand (g O_2/m^2/day)	2.0
FPO4	Benthic flux rate of PO4 (g/ m^2/day)	0.002
FNH4	Benthic flux rate of NH4 (g/m^2/day)	0.01
PMM	Periphyton growth rate (1/day)	1.5
BMRM	Periphyton respiration rate (1/day)	0.1
PRRM	Periphyton mortality rate (1/day)	0.05
TMM1	Lower optimal temperature for periphyton growth (^oC)	20
TMM2	Upper optimal temperature for periphyton growth (^oC)	25

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Appendix A

pH Simulation Module Equations - From Chapra, 1997

In natural waters, much of the buffering capacity for modulating pH is provided by dissolved inorganic carbon species, including CO_2, HCO_3^-, and CO_3^{2-} that satisfy the chemical equilibrium equation:

$$H_2CO_3^* \Leftrightarrow HCO_3^- + H^+$$

where $H_2CO_3^*$ is approximately equal to the concentration of CO_2.

The first dissociation constant of carbonic carbon is thus:

$$K_1 = [H^+][HCO_3^-] / [H_2CO_3^*]$$

which can be calculated using equation:

$$pK_1=3404.71 / T_a + 0.032786\ T_a – 14.8435$$

where T_a = water temperature (in Kelvin)

The bicarbonate ion, HCO_3^-, further dissociates to produce CO_3^{2-} and H^+:

$$HCO_3^{-1} \Leftrightarrow CO_3^{2-} + H^+$$

The equilibrium coefficient, i.e., the second dissociation constant of carbonic acid, is thus:

$$K_2 = [H^+][CO_3^{2-}] / [HCO_3^-]$$

which can be calculated as:

$$pK_2=2902.39 / T_a +0.02379\ T_a – 6.498$$

TIC is defined as the summation of all the carbonic species:

$$TIC = [H_2CO_3^*] + [HCO_3^-] + [CO_3^{2-}]$$

And Alk is defined as the acid-neutralizing capacity of the system, and is formulated as:

$$Alk = [HCO_3^-] + 2[CO_3^{2-}] + [OH^-]+ [H^+]$$

Three intermediate ratios are defined as:

$$a = K_1 \times [H^+] / ([H^+]^2 + K_1 \times [H^+] + K_1 \times K_2)$$
$$b = K_1 \times K_2 / ([H^+]^2 + K_1 \times [H^+] + K_1 \times K_2)$$

Then, a nonlinear equation can be derived to solve for the $[H^+]$:

$$a \times TIC + 2 \times b \times TIC + Kw / [H^+] – [H^+] – Alk = 0$$

where K_w is the dissociation coefficient of water.

While Alk is generally assumed to be conservative in water quality models, this assumption does not apply to TIC because the carbonic system is under constant impact from multiple environmental factors, including biological activities, atmospheric impact, organic decay, and benthic impact. The interaction between TIC and the major impacting factors can be represented using the following differential equation:

$$\frac{dc_{TIC}}{dt} = W_{bio} + W_{atm} + W_{om} + W_{ben}$$

where c_{TIC} is the concentration of TIC (mg/L), W_{bio} is the biological contribution term (mg/L/day); W_{atm} is the atomospheric exchange term (mg/L/day); W_{om} is the organic matter contribution term (mg/L/day); and W_{ben} is the benthic contribution term (mg/L/day). The biological term represents the consumption of CO_2 through the photosynthesis process and the production of CO_2 through the respiration process, which can be represented as:

$$W_{bio} = R - P$$

where R is the CO_2 production rate by respiration (mg/L/day), and P is the CO_2 consumption rate by photosynthesis (mg/L/day).

The atmospheric exchange can be represented as:

$$W_{atm} = Kc \times ([H_2CO_3^*]s - [H_2CO_3^*])$$

where, Kc is the water-air surface exchange coefficient for CO_2 (d^{-1}) and $[H_2CO_3^*]s$ is equivalent to the saturation concentration of CO_2 (mg/L)

The organic matter decay term can be represented as:

$$W_{om} = k \times C_{om}$$

where k is organic matter decay rate (d^{-1}) and C_{om} is the concentration of organic matter as carbon (mg/L).

The benthic contribution term is formulated following the general idea in CE-QUAL-W2:

$$W_{ben} = Cc \times SOD / H$$

where Cc is a dimensionless conversion factor; SOD is the local sediment oxygen demand (g/m2/day); and H is the water depth (m).

Appendix B

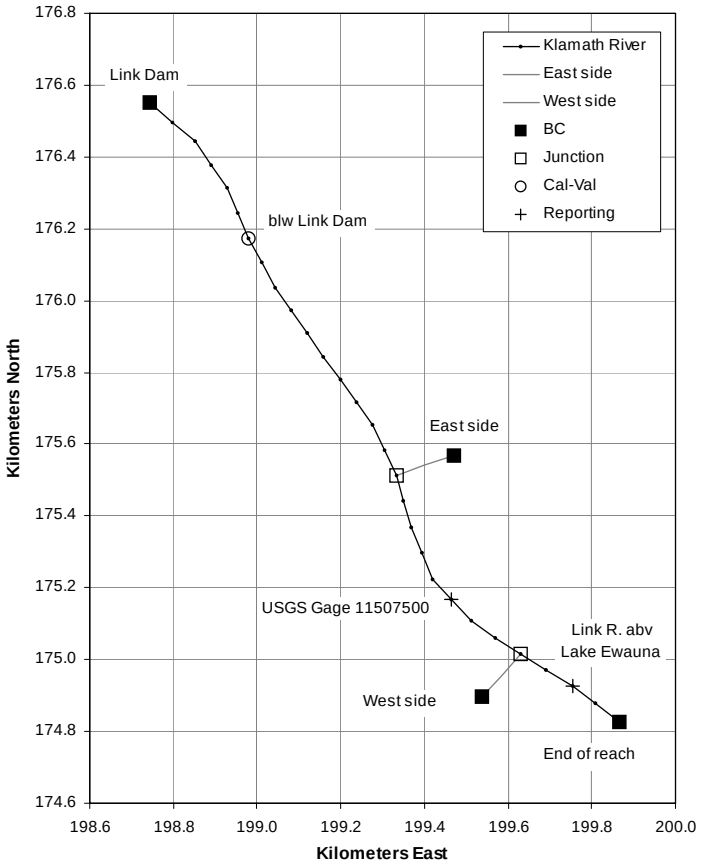
System Geometry - Excerpt from the PacifiCorp, 2005 Report

Link River Reach

The Link River reach starts at Link dam (RM 254) and terminates 1.3 miles downstream at Lake Ewauna (RM 253). The Link River reach is simulated with two junctions, representing separate powerhouse discharges into the reach and no element side flows. This reach is modeled with the RMA-2 and RMA-11 models.

Link River Reach Geometry Summary

Node spacing	75 meters
Number of nodes	29 nodes in length; 37 nodes total including junctions
Length	1.31 miles from RM 252.57-253.88
Elevations	Range: 1245-1259 meters
Widths	Constant widths: 5 meters mainstem; 20 meters junction elements
Side slopes	20:1 mainstem; 1:1 junctions
Data sources	UTM coordinates from CH2M HILL; Elevations estimated from USGS topographic maps
Notes	2 junctions: East side, West side; Nodes 30-33 at East side; 34-37 at West side



Map of Link River Representation

Geometry Information for Link River

Location	Node	Element	x-coord	y-coord	Site type
Link Dam	1	1	198.8	176.6	BC
East Side	17	9	199.9	174.8	BC
West Side	25	13	199.5	175.6	BC
End Link R reach	29	14	199.5	174.9	BC
East Side	30	15	199.3	175.5	Junction, inflow
West Side	34	16	199.6	175.0	Junction, inflow
USGS Gage 11507500	22	--	199.5	175.2	Reporting Point
Link River above Lake Ewauna	27	--	199.8	174.9	Reporting Point

BED ELEVATIONS/SLOPE

Bed slope for the Link River reach was estimated from USGS topographic maps and assumed Lake Ewauna elevations. Elevations were estimated from topographic contours to preserve the general slope of the river. Upstream reach elevation was set at 4131 ft (1259 m) MSL and downstream reach elevation was set at 4085 ft (1245 m) MSL.

CROSS-SECTIONS

Link River widths were obtained from 1:7,500-scale aerial photos taken July 21, 1988. Daily average flow for that day was 920 cfs. For numerical stability in this short and steep reach, bottom width of the mainstem was set to a constant 5 meters. These widths were assumed to represent bottom widths of trapezoidal cross-sections with twenty-to-one side slopes on the mainstem and one-to-one side slopes in tributaries.

Lake Ewauna-Keno Reservoir

The Lake Ewauna to Keno dam reach extends from the headwaters of Lake Ewauna (RM 253) 20 miles downstream to Keno dam (RM 233). The impoundment (i.e., Keno reservoir) is generally a broad, shallow body of water. Widths range from several hundred to over 1,000 feet (a range of about 90 to 300 meters), and depths range to a maximum of roughly 20 feet (approximately 6 meters). A total of 18 discharges and 7 withdrawals were represented in the model. This reach is modeled with CE-QUAL-W2.

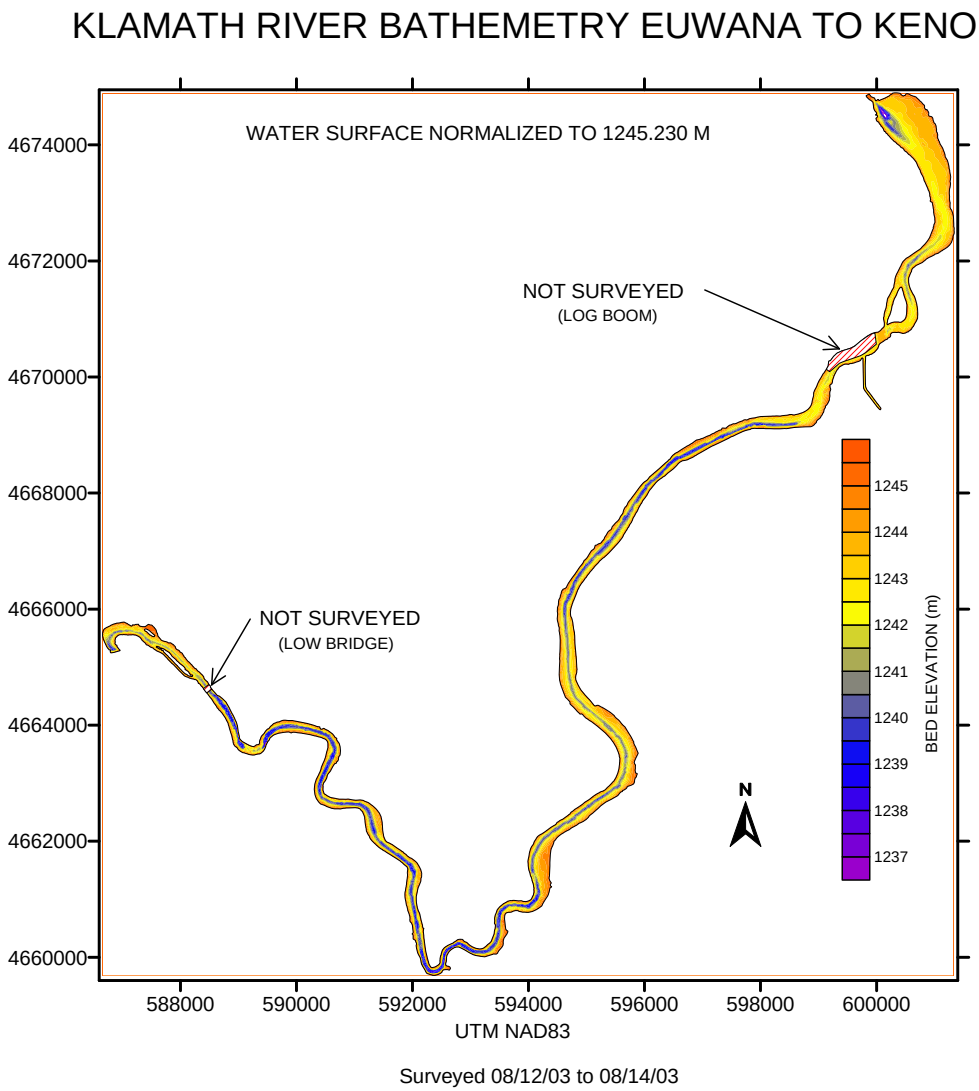
KENO DAM FEATURES

The Keno dam spillway, with an invert elevation of 4,070 feet, contains six Taintor gates. Three additional outlets include a sluice conduit, the fish attraction outlet, and a fish ladder.

Keno Dam Outlet Features

Outlet	Invert Elevation	Dimension	Operation
Sluice Conduit	4,073.0 ft	36 inch diameter	Manual gate
Fish Attraction Outlet	4,075.0 ft	30 inch diameter	Manual gate
Fish Ladder	4,078.5 ft	60 inch width	Stop logs
Spillway	4,070.0 ft	6 gates @ 40 ft width each	Remote control on three gates

Sources: PacifiCorp (2002), PacifiCorp (2000)

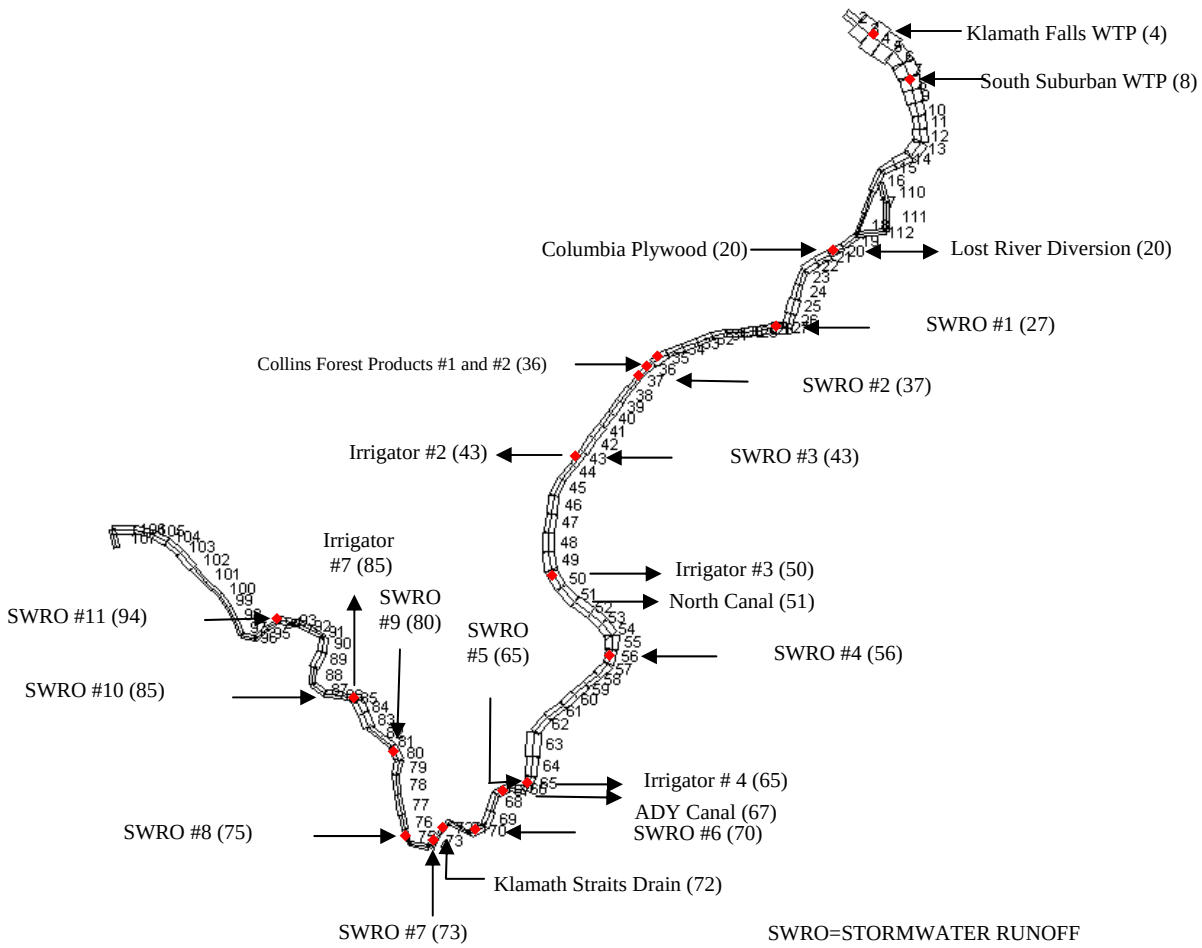


Keno Reservoir Bathymetry (PacifiCorp, 2004a)

RESERVOIR BATHYMETRY

The Lake Ewauna to Keno dam model was originally implemented with bathymetry derived from an earlier model of this reach created by Wells (ODEQ, 1995). This original representation was replaced with data from a recent bathymetric survey of the entire reservoir (PacifiCorp, 2004a).

The number of segments, number of layers, segment lengths, layer widths per segment and water surface elevation were largely retained from the previous CE-QUAL-W2 modeling of the reach by ODEQ (1995), but were supplemented with new segment orientations calculated from x-y coordinates obtained from digitized versions of 1:24,000 USGS topographic quadrangles. River segment orientations were updated because the original orientations (ODEQ 1995) contained discrepancies when applied to the newer versions of CE-QUAL-W2 used in this study.



Map of Lake Ewauna to Keno Dam CE-QUAL-W2 Representation, Identifying Inputs and Withdrawals

The CE-QUAL-W2 representation of Lake Ewauna to Keno dam reach consists of two connected reservoir sections, or branches. The main branch, Branch 1, spans the entire length of the reach and is comprised of 106 active segments, all 1,000 ft (304.8 m) in length. A second, smaller branch, Branch 2, provides an alternate flow path from segment 14 to segment 18 of Branch 1. Branch 2 has no external inflows or outflow and is comprised of three active segments, each 800 ft (243.8 m) in length. A total of 18 discharges and 7 withdrawals were represented in the model. The 15 active layers of this reach are all 2.00 ft (0.61 m) thick. Total volume generated by this model representation was consistent with volume calculated from reservoir bathymetry available from PacifiCorp.

Modeled Inflows and Outflows in the Lake Ewauna to Keno Dam Reach

Name	Type	River Bank [a]	Approximate RM [b]	Model Segment
Klamath Falls Wastewater Treatment Plant	Inflow	Left	253	4
South Suburban Sanitation District	Inflow	Left	252	8
Columbia Plywood	Inflow	Right	250	20
Lost River Diversion	Inflow/Outflow	Left	250	20
Collins Forest Products #1	Inflow	Right	247	36
Collins Forest Products #2	Inflow	Right	247	36
Klamath Straits Drain	Inflow	Left	240	72
Stormwater Runoff #1	Inflow	NA	249	27
Stormwater Runoff #2	Inflow	NA	247	37
Stormwater Runoff #3	Inflow	NA	246	43
Stormwater Runoff #4	Inflow	NA	243	56
Stormwater Runoff #5	Inflow	NA	242	65
Stormwater Runoff #6	Inflow	NA	241	70
Stormwater Runoff #7	Inflow	NA	240	73
Stormwater Runoff #8	Inflow	NA	240	75
Stormwater Runoff #9	Inflow	NA	239	80
Stormwater Runoff #10	Inflow	NA	238	85
Stormwater Runoff #11	Inflow	NA	236	94
North Canal	Outflow	Left	247	35
ADY Canal	Outflow	Left	241	67
Irrigator #2 [c]	Inflow/Outflow	NA	246	43
Irrigator #3 [c]	Inflow/Outflow	NA	244	50
Irrigator #4 [c]	Inflow/Outflow	NA	242	65
Irrigator #7 [c]	Inflow/Outflow	NA	238	85

[a] River bank is given for reference only. The model does not discriminate between banks when simulating flows.

[b] River miles are approximate as each model segment is 1000 ft in length.

[c] Nomenclature after Wells (ODEQ, 1995)

Placement of stormwater runoff and irrigator flows is as per ODEQ (1995).

Klamath River from Keno Dam to J.C. Boyle Reservoir Reach

The Keno reach extends 5.4 miles from Keno dam (RM 233) downstream to the headwaters of J.C. Boyle reservoir (RM 227). No appreciable tributary inflows occur in this reach. This reach is modeled with the RMA models.

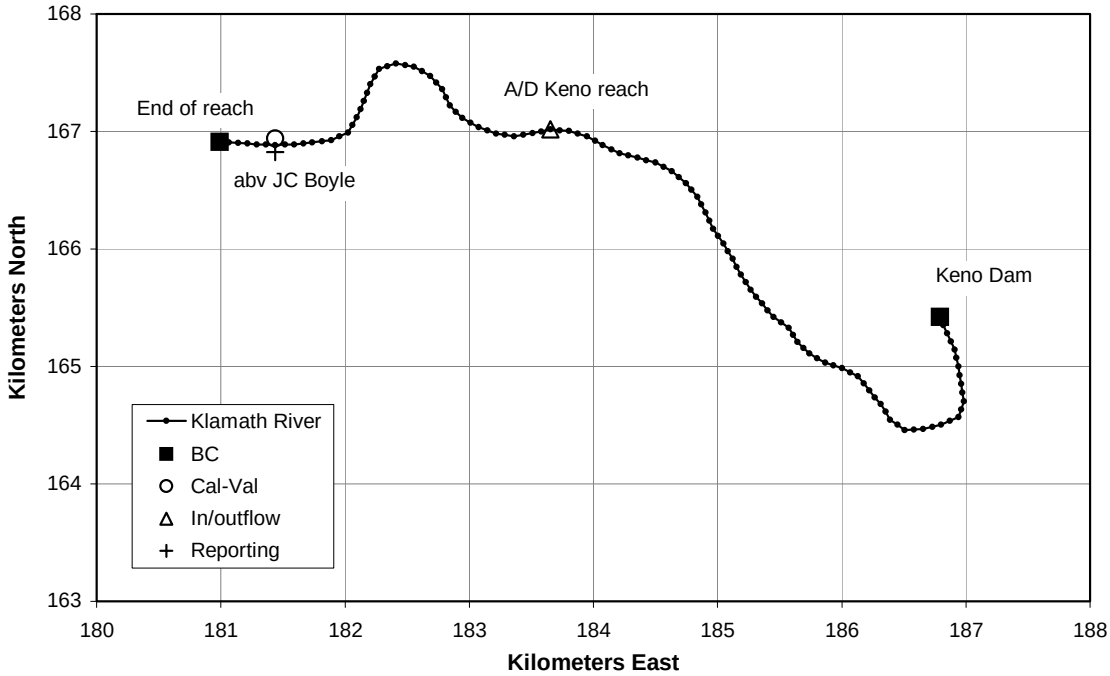
Klamath River, Keno Reach Geometry Information for the RMA-2 and RMA-11 Models

Location	Node	Element	x-coord	y-coord	Site type
Keno Dam	1	1	186.8	165.4	BC, upper
End Keno R reach	117	58	181.0	166.9	BC, lower
A/D Keno reach	73	37	183.7	167.0	A/D
1/4 mi abv J.C. Boyle	110	56	181.4	166.9	Cal/Val and Reporting

BC – boundary condition (flow, constituent concentration, stage)

A/D – accretion/depletion location

Reporting – model output location



Klamath River, Keno Reach Representation

BED ELEVATION/SLOPE

Bed slope for the Keno reach was estimated from USGS topographic maps, known elevations at Keno Dam, and estimated water surface elevations downstream in J.C. Boyle reservoir. Estimated reach elevations range from approximately 3796 ft (1158 m) MSL to 4019 ft (1225 m) MSL.

CROSS-SECTIONS

Keno reach widths were obtained from habitat surveys conducted by Thomas R. Payne and Associates (TRPA) (PacifiCorp, 2004b). Measurements were completed at roughly eight locations per mile. Because measurement locations did not always coincide with the x-y coordinates of the model, field data were linearly interpolated to determine widths for model cross sections. Extreme variations in measured widths were smoothed with a seven-times running average to produce estimates of bottom width. Using these estimates of bottom width, trapezoidal river cross-sections were constructed for each node of the reach at evenly spaced intervals of 75 meters, assuming 1:1 side slopes.

Klamath River, Keno Reach Geometry Summary

Node spacing	75 meters
Number of nodes	117 nodes in length
Length	5.37 miles from RM 228.69-234.06
Elevations	Range: 1158-1225 meters
Widths	Range: 28-78 meters
Side slopes	1:1
Data sources	UTM coordinates from CH2M HILL; Elevations estimated from USGS topographic maps
Notes	n/a

J.C. Boyle Reservoir

The J.C. Boyle reservoir reach extends 3.3 miles from the headwaters of J.C. Boyle reservoir (RM 228) to J.C. Boyle dam (RM 224). This reservoir primarily serves to regulate flows for the J.C. Boyle powerhouse located downstream at RM 220. The one significant tributary to this reach, Spencer Creek, is represented in the model as inflow added to Klamath River inflows at the headwater of the reservoir.

J.C. BOYLE DAM FEATURES

J.C. Boyle dam has four primary outlets: a spillway, a fish ladder, and two outlets into the waterway intake (a fish screen bypass and a waterway pipeline). This reach is modeled with CE-QUAL-W2.

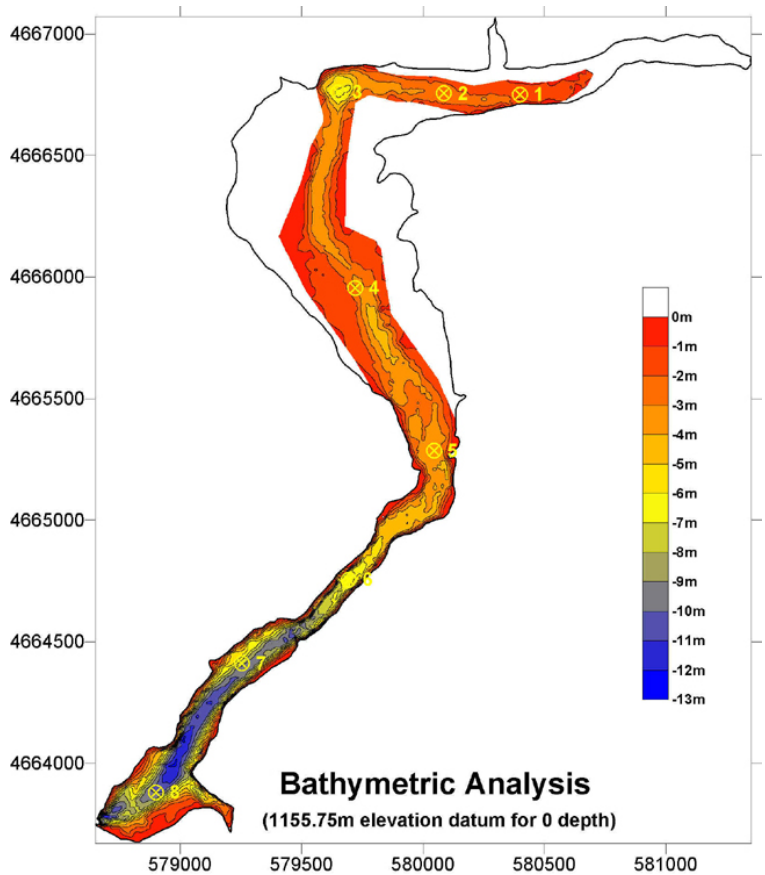
J.C. Boyle Dam Outlet Features

Outlet	Invert Elevation	Dimension	Operation
Fish ladder	3780.0 ft	24 inch diameter	Manual
Fish Screen Bypass	3757.0 ft	24 inch diameter	Manual
Waterway pipeline	3775.0 ft	14 foot diameter	**
Spillway	3782.0 ft	3 radial gates @ 35 ft width each	Remote control on one gate

Sources: PacifiCorp (2002), PacifiCorp (2000), PacifiCorp drawing: Exhibit L-4

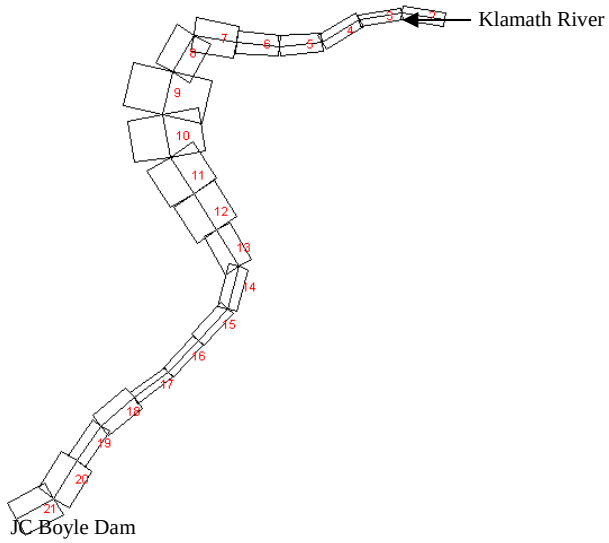
RESERVOIR BATHYMETRY

Unlike the Lake Ewauna to Keno dam reach, J.C. Boyle reservoir has never been modeled with CE-QUAL-W2. Reservoir geometry was derived from bathymetric data (PacifiCorp, 2004a). Segment length, segment orientation, layer thickness and width were required for the reservoir model. Based on the variation in the reservoir morphology and widths, the reservoir was divided into 20 active segments 887 ft (270m) in length. Segments were chosen to capture both the general shape of J.C. Boyle reservoir and pertinent features.



J.C. Boyle Reservoir Bathymetry (PacifiCorp, 2004a)

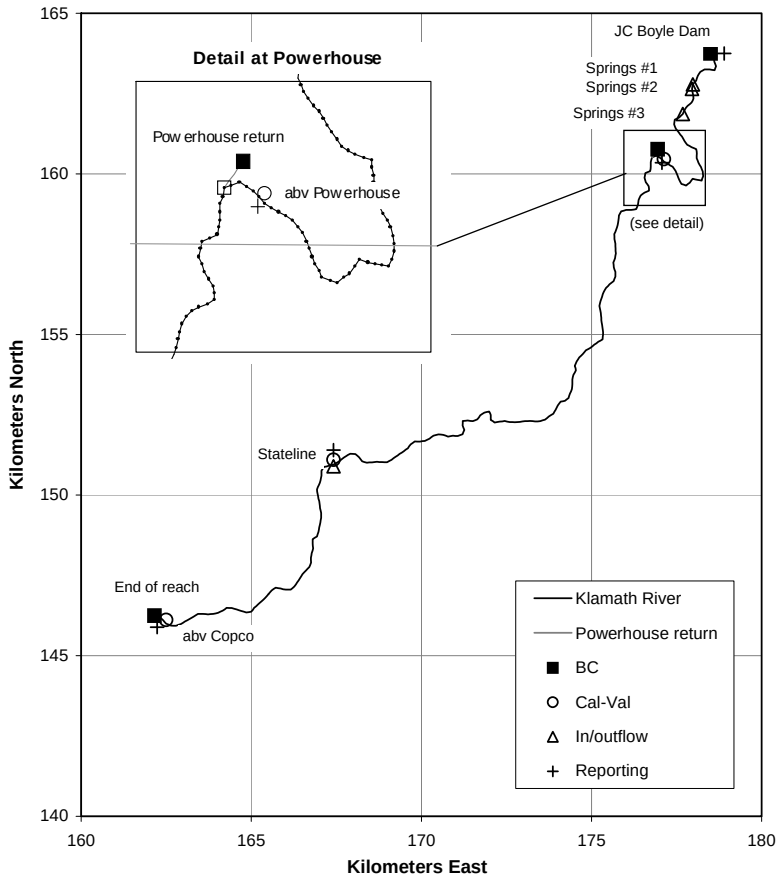
Layer thickness was set to 3.28 feet (1.0 meter). Layer widths were determined from cross-sectional information taken at the middle of each segment. Twelve active layers of varying widths were determined for each segment from this method. Although a representation using finer resolution (i.e., smaller layer thickness less than 1 meter) was attempted, models using these refined cross-sections took an uncommonly long time (on the order of a day) to run for each one-year simulation period). The model was continually adding and subtracting both layers and segments to account for the dynamic water surface elevations imposed by hydropower operations. A layer thickness of 1 meter produced reasonable results, and one-year simulation times were appreciably reduced to approximately 10 minutes.



Representation of J.C. Boyle Reservoir in CE-QUAL-W2

J.C. Boyle Bypass and Peaking Reaches

The J.C. Boyle bypass and peaking reaches extend 20.8 miles from J.C. Boyle dam (RM 224) to the headwaters of Copco reservoir (RM 204). Noteworthy features of the reaches include diversion of mainstem flows at J.C. Boyle dam for hydropower production, the powerhouse penstock return marking the beginning of the peaking reach roughly 4 miles downstream from J.C. Boyle dam (RM 220), a large springs complex in the bypass reach, and hydropower peaking operations downstream of the powerhouse. A few small streams enter the reach, the most significant of which is Shovel Creek. Important locations within the bypass and peaking reaches are presented in the table below. These reaches are modeled with the RMA models.



J.C. Boyle Bypass and Peaking Reach Representation

Geometry Information for J.C. Boyle Bypass and Peaking Reach EC Simulation

Location	Node	Element	x-coord	y-coord	Site type
J.C. Boyle Dam	1	1	178.7	163.7	BC, upper
End Peaking reach	453	226	162.2	146.2	BC, lower
J.C. Boyle Powerhouse	95	48	176.9	160.8	BC
Simulated Powerhouse Return	97	49	176.8	160.5	Junction, inflow
1/4 mi abv Powerhouse	91	46	177.1	160.4	Cal-Val
1/4 mi abv Shovel Cr	389	195	166.3	147.2	Cal-Val
1/4 mi abv Copco	447	224	162.5	146.00	Cal-Val
CA-OR Stateline	331	166	167.4	151.1	Cal-Val, A/D
Springs #1	21	11	178.0	162.8	A/D
Springs #2	23	12	178.0	162.6	A/D
Springs #4	35	18	177.7	161.9	A/D

BC – boundary condition

A/D – accretion/depletion location

Cal-Val – calibration and validation location

BED ELEVATION/SLOPE

Bed slope for these reaches was estimated from USGS topographic maps and reported elevations at J.C. Boyle dam and Copco reservoir water surface elevations. Reach elevations range from approximately 2592 ft (790 m) MSL to 3760 ft (1146 m) MSL.

CROSS-SECTIONS

J.C. Boyle bypass and peaking reach widths were obtained from habitat surveys completed by TRPA (PacifiCorp, 2004b). Measurements were completed at roughly eight locations per mile. Because measurement locations did not always coincide with the 1:24,000 x-y coordinates of the model, field data were linearly interpolated to provide widths for cross-sections of the model. Extreme variations in measured widths were smoothed with a seven-times running average to produce estimates of bottom width. Using these estimates of bottom width, trapezoidal river cross-sections were constructed for each node of the reach at evenly spaced intervals of 75 meters, assuming 1:1 side slopes. Widths and other geometric characteristics of the bypass and peaking reaches are summarized in the table below.

J.C. Boyle Bypass and Peaking Reach Geometry Summary

Node spacing	75 meters
Number of nodes	459 nodes in length
Length	20.81 miles from RM 204.72-225.53
Elevations	Range: 790-1146 meters
Widths	Range: 12-66 meters
Side slopes	1:1
Data sources	UTM coordinates from CH2M HILL; Elevations estimated from USGS topographic maps
Notes	1 junction: J.C.B Powerhouse; Nodes 97, 458, 459

Copco Reservoir

The Copco reservoir reach extends 5.0 miles from Copco reservoir headwaters (RM 204) downstream to Copco dam (RM 199). No tributaries are represented in this section of the model. Physical data for the Copco reservoir model are outlined below. This reach is modeled with CE-QUAL-W2.

COPCO DAM FEATURES

Copco dam has three primary outlets: a spillway and two penstocks that provide flows to the Copco No. 1 powerhouse. The two penstocks, fed by three intakes, are treated as a single outlet with an average centerline elevation of 2,581 feet. Because of the close proximity and similar invert elevations, the outlet works were represented in the reservoir as a single withdrawal with a midline elevation of 2,581 ft (786.6 m).

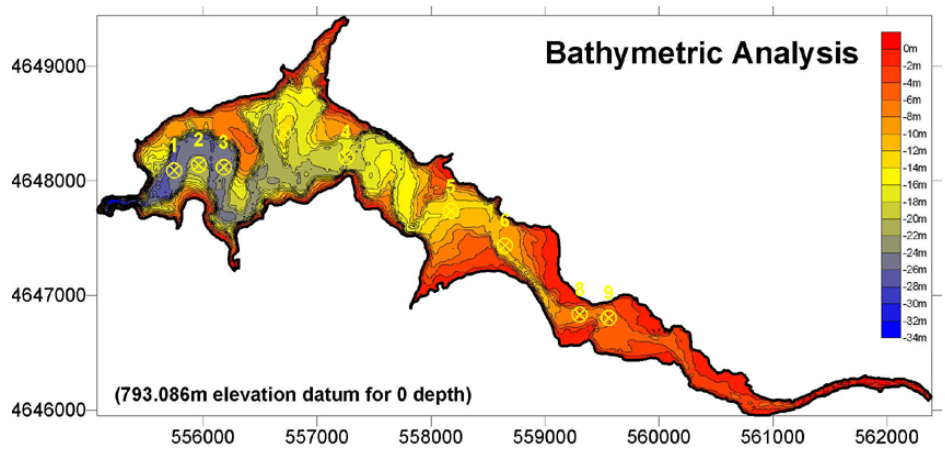
Copco Dam Outlet Features

Outlet	Invert Elevation	Dimension	Operation
Penstock Intake (Unit 1)	2575 ft	Two intakes @ 10-foot diameter each	Remote Operation
Penstock Intake (Unit 2)	2575 ft	14 foot diameter	Remote Operation
Spillway	2594 ft	3 radial gates @ 35 ft width each	Remote control on one gate, others by motorized hoist

Sources: PacifiCorp (2002), PacifiCorp (2000)

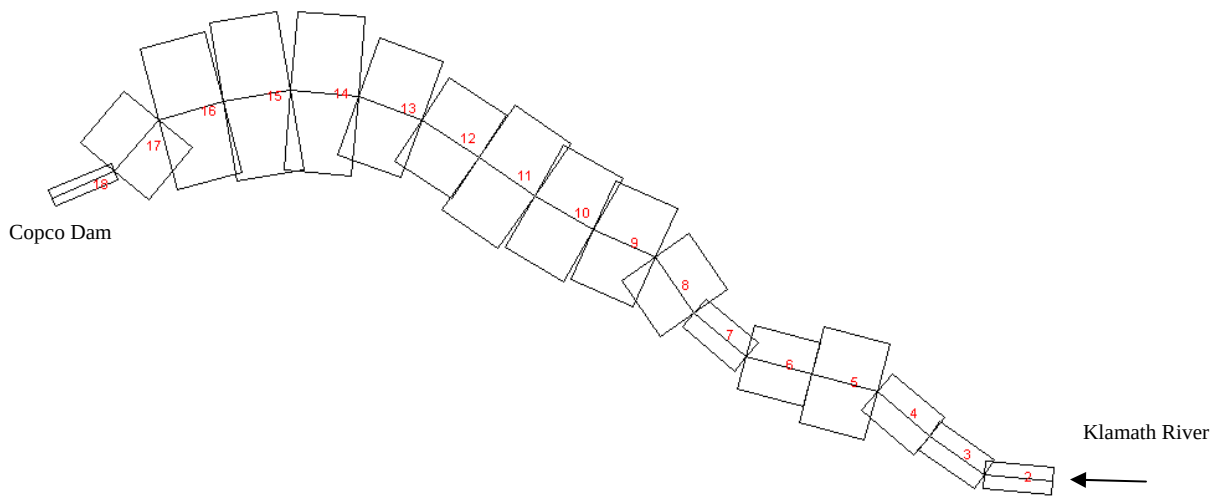
RESERVOIR BATHYMETRY

Copco reservoir geometry was derived from bathymetric data of Copco reservoir (PacifiCorp, 2004a). Segment length, segment orientation, layer thickness and width were required for the reservoir model. Segments were identified based on changes in reservoir morphology and widths. The reservoir was divided into 17 active segments 1,329 ft (405.4 m) in length. Segments were chosen to capture both the general shape of Copco reservoir and pertinent features, such as the submerged features near the dam. Due to the large bedrock outcrop in the vicinity of the Copco dam, a submerged weir was implemented in the model from layer 20 to 32.



Copco Reservoir Bathymetry (PacifiCorp, 2004a)

Layer thickness was set to 3.28 ft (1.0 m). Layer widths were determined from cross-sectional information taken at the middle of each segment. Thirty-two active layers of varying widths were determined for each segment from this method. The 3.28 ft (1.0 m) layer thickness produced reasonable results and resulted in reasonable execution times. One-year simulation times were approximately 15 minutes. Final CE-QUAL-W2 representation of Copco reservoir is shown in the Figure below.



Representation of Copco Reservoir in CE-QUAL-W2

Iron Gate Reservoir

Iron Gate reservoir extends 6.4 miles from the headwaters of Iron Gate reservoir (RM 197) to Iron Gate dam (RM 190). Except in "Without Project" scenarios, the small Copco #2 Reservoir and short river reach between Copco and Iron Gate reservoirs are not represented in the model. Instead, Copco reservoir runs directly into Iron Gate reservoir. Three tributaries to Iron Gate reservoir are represented in this CE-QUAL-W2 model: Camp Creek, Jenny Creek, and Fall Creek. The spillway for the dam is modeled as a withdrawal in the last active segment because the spillway structure draws water to the side of the dam, not over or through the dam itself. Due to its dendritic shape, Iron Gate reservoir is represented by two branches, including a main branch that receives water released from Copco Reservoir and a Camp Creek branch that represents a sizeable arm of the reservoir running up to Camp Creek. Geometry of the reservoir is outlined below.

IRON GATE DAM FEATURES

Iron Gate dam has four primary outlets: a spillway, a penstock, and two outlets that supply fish hatchery intakes. The details of these outlets are summarized in the table below.

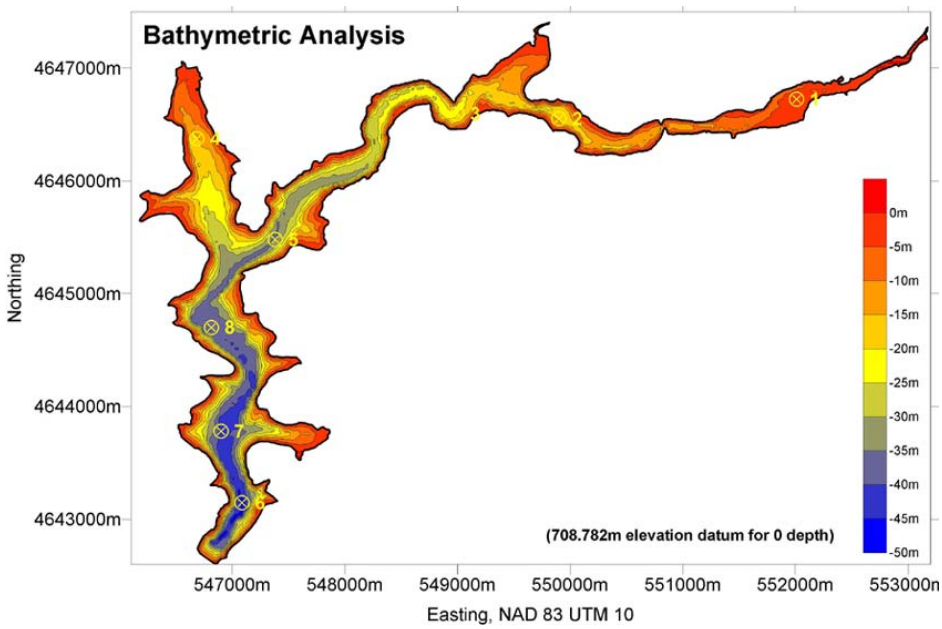
Iron Gate Dam Outlet Features

Outlet	Invert Elevation	Dimension	Operation
Upper Fish Hatchery	2293 ft	24 inch diameter	Manual
Penstock Intake	2309 ft	12 foot diameter	Remote operation
Lower Fish Hatchery	2253 ft	24 inch diameter	Manual
Spillway	2328 ft	Side channel (727 feet in length)	Overflow

Sources: PacifiCorp (2002), PacifiCorp (2000)

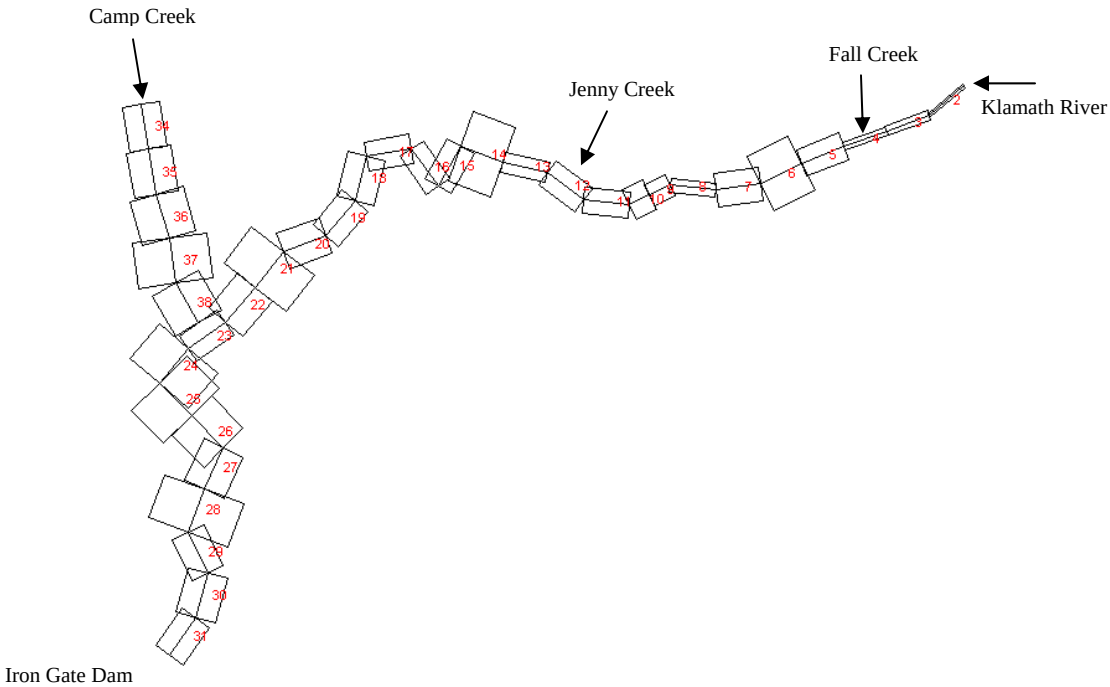
RESERVOIR BATHYMETRY

Reservoir geometry was derived from bathymetric data of Iron Gate reservoir (PacifiCorp, 2004a). Segments were laid out on the basis of changes in reservoir orientation and width. The main branch, Branch 1, has 30 active segments and the Camp Creek Branch, Branch 2, has five active segments. Segment lengths were 1,204 ft (367 m), with the exception of the narrows near the upper end of the reservoir, where half element lengths were used. Branch 2 has an external upstream boundary (Camp Creek) and connects with Branch 1, Segment 23.



Iron Gate Bathymetry (PacifiCorp, 2004a)

Based on cross-sectional information from the mid-point of each segment, Iron Gate Reservoir is represented by 50 active layers, each 3.28 ft (1 m) in thickness.



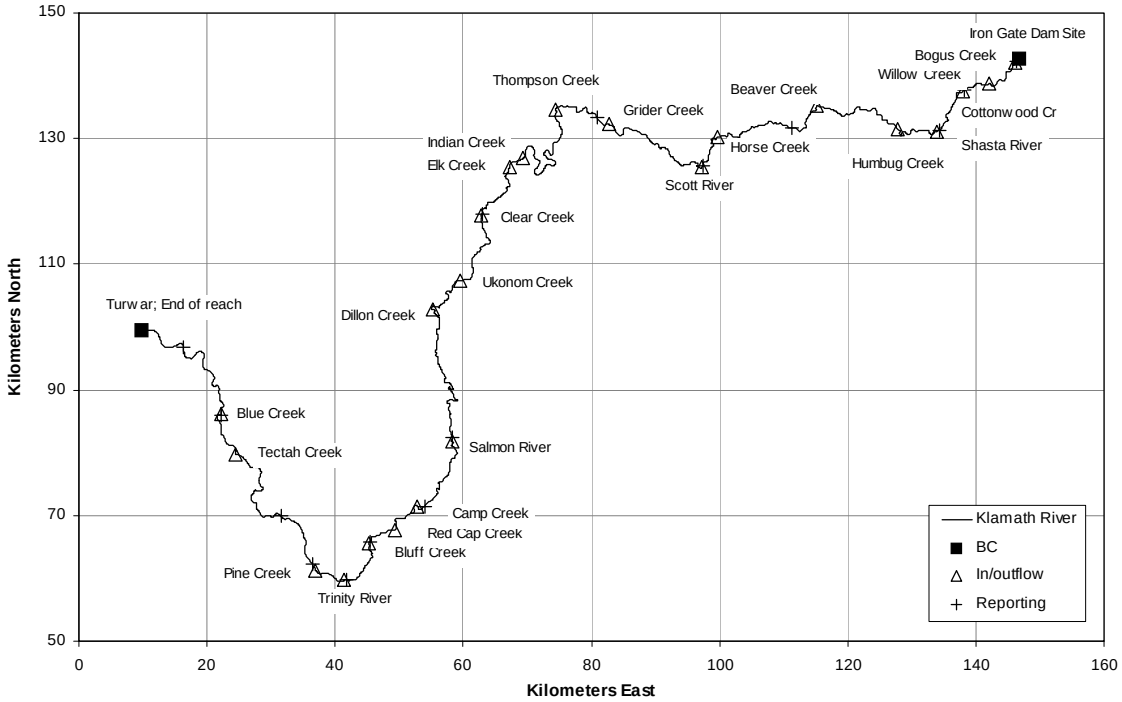
Representation of Iron Gate Reservoir for CE-QUAL-W2

Iron Gate to Turwar Reach

The Iron Gate dam to Turwar reach extends 185 miles from Iron Gate dam (RM 190) to Turwar near the mouth of the Klamath River (RM 5). Several main tributaries flow into the reach: Shasta River, Scott River, Salmon River, and Trinity River. Many smaller creeks contribute significant flow to the river along this reach and these creeks are also included in the simulation. Geometry of this reach is outlined below.

MAP COORDINATES

X-Y coordinates describing the course of the river were taken from a digitized version of the 1:24,000 USGS topographic quadrangles. This information was translated into a series of nodes and elements for use by the numerical model. The model network is shown with simulated tributaries in the figure below. Important locations within the reach, including tributaries and output locations, are presented in the Table below. Nodal spacing for the numerical grid was roughly 490 feet (150 meters). Sensitivity analyses showed model results to be relatively insensitive to a reduction in grid spacing.



Iron Gate Dam to Turwar Reach Representation Showing Tributary Names

Geometry Information for the IG-Turwar reach (150-meter grid)

Location	Node	Element	x-coord	y-coord	Site Type
Iron Gate Dam	1	1	146.747	142.634	BC, upper
End IG-Turwar reach	2081	1040	9.821	99.506	BC, lower
Bogus Creek	7	4	146.141	142.022	A/D
Willow Creek	55	28	142.035	138.739	A/D
Cottonwood Creek	86	43	137.904	137.535	A/D
Shasta River	144	72	133.963	131.178	A/D
Humbug Creek	204	102	127.848	131.402	A/D
Beaver Creek	319	160	115.190	135.232	A/D
Horse Creek	468	234	99.597	130.180	A/D
Scott River	513	257	97.299	125.428	A/D
Grider Creek (A/D Scott to Seiad)	656	328	82.714	132.246	A/D
Thompson Creek	735	368	74.440	134.626	A/D
Indian Creek	906	453	69.371	126.831	A/D
Elk Creek	925	463	67.209	125.507	A/D
Clear Creek	1000	500	62.733	117.818	A/D
Ukonom Creek	1098	549	59.559	107.347	A/D
Dillon Creek	1162	581	55.209	102.905	A/D
Salmon River	1357	679	58.333	81.788	A/D
Camp Creek	1466	733	52.865	71.474	A/D
Red Cap Creek	1511	756	49.403	67.773	A/D
Bluff Creek	1547	774	45.339	65.584	A/D
Trinity River	1609	805	41.415	59.672	A/D
Pine Creek	1644	822	36.954	61.269	A/D
Tectah Creek	1850	925	24.557	79.833	A/D
Blue Creek	1908	954	22.306	86.220	A/D
1/4 mi bl Iron Gate	4	2	146.419	142.345	reporting
1/4 mi ab Cottonwood	84	42	138.117	137.743	reporting
1/4 mi ab Shasta	142	71	134.262	131.198	reporting
Walker Bridge	369	185	111.329	131.759	reporting
1/4 mi ab Scott	511	256	97.348	125.720	reporting
USGS Gage at Seiad Valley	672	336	80.887	133.289	reporting
1/4 mi ab Clear Cr.	998	499	62.908	118.058	reporting
1/2 mi ab Salmon (Ishi Pishi)	1352	676	58.231	82.372	reporting
USGS Gage at Orleans	1454	727	54.016	71.457	reporting
1/4 mi ab Bluff Cr.	1545	773	45.357	65.876	reporting
1/4 mi ab Trinity	1607	804	41.692	59.692	reporting
Martin's Ferry	1651	826	36.505	62.187	reporting
Young's Bar	1722	861	31.541	69.894	reporting
1/4 mi ab Blue Cr.	1906	953	22.177	85.992	reporting
USGS Gage nr Turwar	2024	1012	16.341	96.868	reporting

RIVER BED ELEVATION

Bottom elevations along the reach were estimated from USGS topographic maps and reported elevations at Iron Gate dam. These elevations determined bed slope. Reach elevations range from approximately sea level to roughly 2200 ft (671 m) MSL.

CROSS-SECTIONS

Klamath River widths for the Iron Gate dam to Turwar reach were estimated from meso-habitat surveys compiled by US Fish and Wildlife Service (1997). This dataset included a reach-by-reach description of 1,741 units, or sections of the river, by habitat type, width, and maximum depth. Measurements were not uniformly spaced. Because measurement locations did not always coincide with the 1:24,000 x-y coordinates, field data were linearly interpolated to produce widths for model cross-sections. Large variations in river width were smoothed with a seven-point running average to provide estimates of bottom width for the model. From these estimated bottom widths, trapezoidal cross-sections were constructed at each node assuming 1:1 side slopes. Widths and other geometric characteristics for the Iron Gate to Turwar reach are summarized in the table below.

Klamath River, Iron Gate Dam to Turwar Reach Geometry Summary

Node spacing	150 meters
Number of nodes	2082 nodes in length
Length	190.54 miles from RM 0.00-190.54
Elevations	Range: 0-671 meters
Widths	Range: 17-340 meters
Side slopes	1:1
Data sources	UTM coordinates from CH2M HILL; Elevations estimated from USGS topographic maps
Notes	n/a

Appendix C

Electronic Appendix: Boundary Condition Spreadsheet Files

(Summary.xls, LEwauna_boundaries_2000.xls, LEwauna_boundaries_2002.xls, JCB_boundaries_2000.xls, JCB_boundaries_2002.xls, Copco_boundaries_2000.xls, IG_boundaries_2000.xls, getGRHdata_link_2000.xls, getGRHdata_link_2002.xls, getGRHdata_KenoR_2000.xls, getGRHdata_KenoR_2002.xls, getGRHdata_Fullflow_2000.xls, getGRHdata_Fullflow_2002.xls, getGRHdata_IG-Turwar_2000.xls, and Klamath Estuary_2004.xls)

Appendix D

Determination of Accretions for Tributaries from Iron Gate Dam to Turwar - Excerpt from the PacifiCorp, 2004 Report

Determination of Flow for Tributaries from Iron Gate Dam to Turwar using USGS Methodology

Accretions from Iron Gate Dam to Turwar were defined and quantified according to the methodology identified by USGS (1995, 1997). In sum, the river was divided into multiple segments (reaches) based on available gages with full coverage between 1961 and 1922. USGS used monthly averages to determine accretions and depletions for each reach based on the differences in gage readings. These accretions and depletions were then assigned to individual tributaries based on estimated basin area (individual sub-basin contributions were obtained from personal communication with Mr. M. Flug). Not all tributaries to the Klamath River were included.

For this exercise, 7-day average values were used to identify accretions and depletions for identified tributaries. The same tributaries identified by USGS (1997) were used herein.

The methodology is outlined below.

Total Accretion from Iron Gate Dam to Seiad Valley.
Accretion value is equal to the flow at gage 11520500 (Klamath River nr. Seiad Valley) minus the sum of the flows at gages 11516530 (KR below Iron Gate Dam), 11517500 (Shasta River nr. Yreka), 11519500 (Scott River nr Fort Jones). This reach accretion is further subdivided into shorter sub-reaches by according to the following criteria.

Klamath River from Iron Gate Dam to the confluence of the Shasta River.
Accretion equals 24.2% of the total area accretion. This accretion is distributed between the following creeks as determined by watershed area:

Bogus Creek – 41%
Willow Creek – 22%
Cottonwood Creek – 37%

Klamath River from the confluence of the Shasta River to the confluence of the Scott River.
Accretion equals 38.2% of the total area accretion. This accretion is distributed between the following creeks as determined by watershed area:
Humbug Creek – 28%
Beaver Creek – 32%
Horse Creek – 40%

Scott River from Ft. Jones to the confluence of the Klamath River.
Accretion equals 29.0% of the total area accretion.

Klamath River from the confluence of the Scott River to Seiad valley.
Accretion equals 8.6% of the total area accretion. This accretion is applied at Grider Creek.

Total Accretion from Klamath River from Seiad Valley to Orleans.
Accretion equals the flow at gage 11523000 (Klamath River at Orleans) minus the sum of the flows at gages 11520500 (Klamath River nr Seiad Valley), 11522500 (Salmon River at Somes Bar, Ca), and 11521500 (Indian Cr nr Happy Camp).
This accretion is distributed between the following creeks as determined by watershed area:

Thompson Creek – 16.6%
Elk Creek – 16.6%
Clear Creek – 21.4%
Ukonom Creek – 12.9%
Dillon Creek – 32.5%

Total Accretion from Klamath River from Orleans to the Mouth.
Accretion equals gage 15530500 (KR nr Klamath (Turwar), CA) minus gage 15523000 (KR at Orleans) and 11530000 (Trinity River at Hoopa).

Klamath River from Orleans to the confluence of the Trinity River.
Accretion equals 29.3% of the total area accretion. This accretion is distributed between the following creeks as determined by watershed area:

Camp Creek – 33.3%
Red Cap Creek – 33.3%
Bluff Creek – 33.3%

Trinity River from Hoopa to the confluence with the Klamath River.
Accretion equals 12.3% of the total area accretion.

Klamath River from the confluence of the Trinity River to the mouth.
Accretion equals 58.4% of total area accretion. This accretion is distributed between the following creeks as determined by watershed area:

Pine Creek – 33.3%
Tectah Creek – 33.3%
Blue Creek – 33.3%

Appendix E

Calibration Results for Lake Ewauna to Keno Dam (Modeling Segment 2)

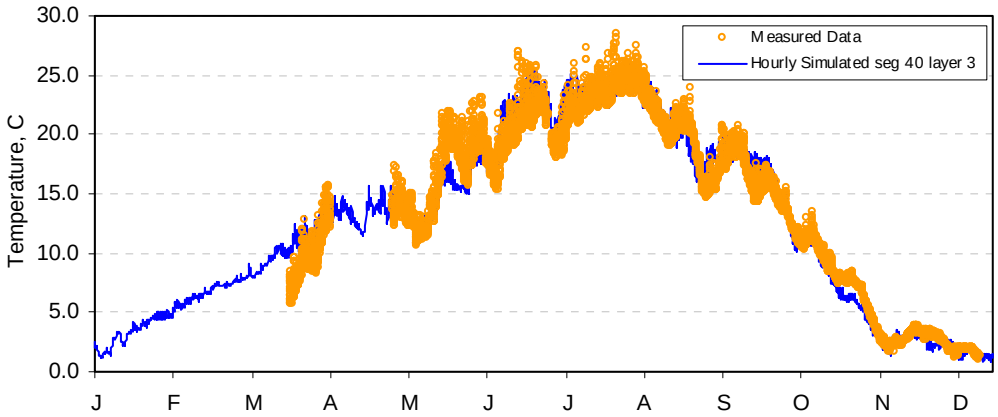


Figure E-1. Temperature calibration results for Lake Ewauna—Keno Dam at Klamath River at Miller Island boat ramp (2000).

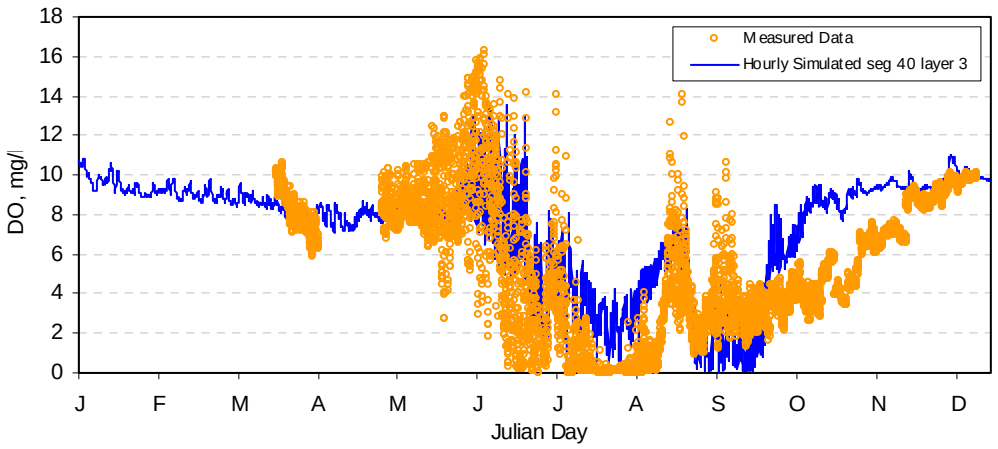


Figure E-2. Dissolved oxygen calibration results for Lake Ewauna—Keno Dam at Klamath River at Miller Island boat ramp (2000).

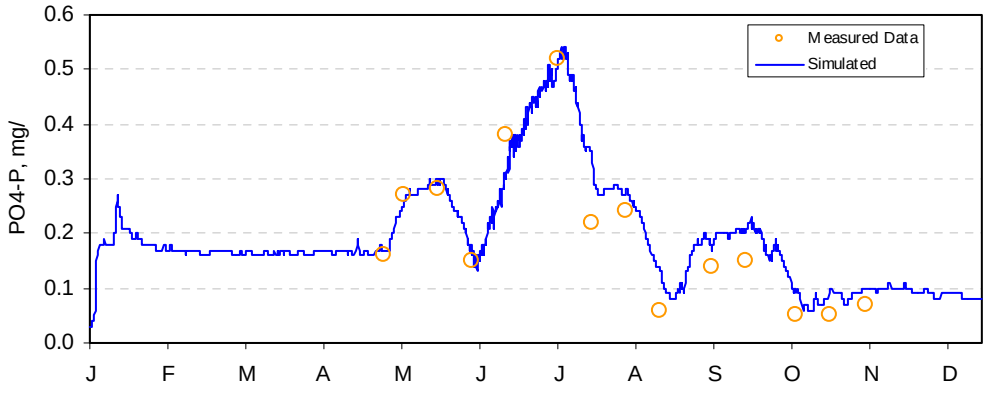


Figure E-3. PO4-P calibration results for Lake Ewauna—Keno Dam at Klamath River at Miller Island boat ramp (2000).

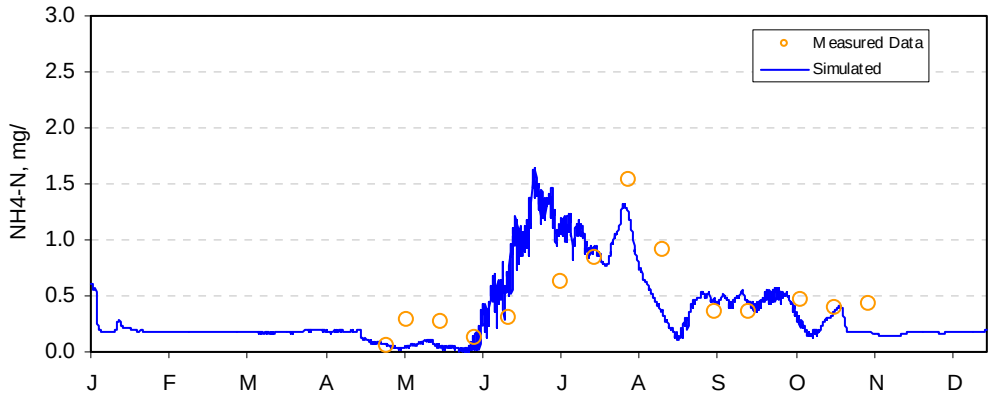


Figure E-4. NH3-N calibration results for Lake Ewauna—Keno Dam at Klamath River at Miller Island boat ramp (2000).

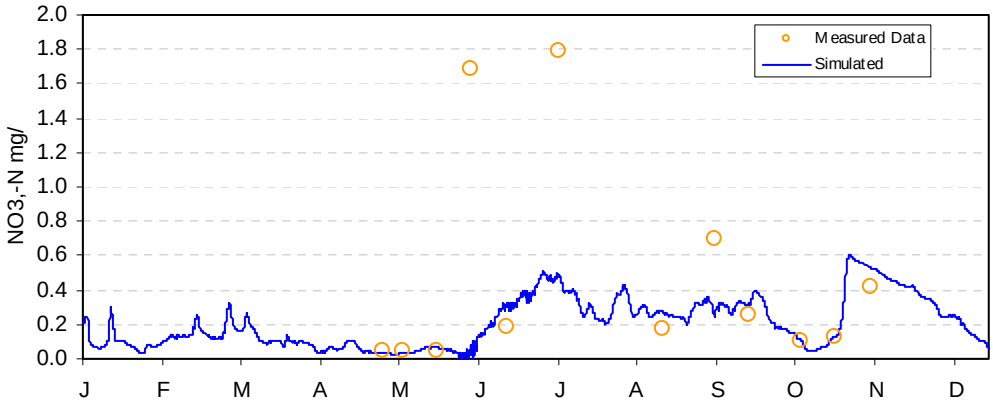


Figure E-5. NO3-N calibration results for Lake Ewauna—Keno Dam at Klamath River at Miller Island boat ramp (2000).

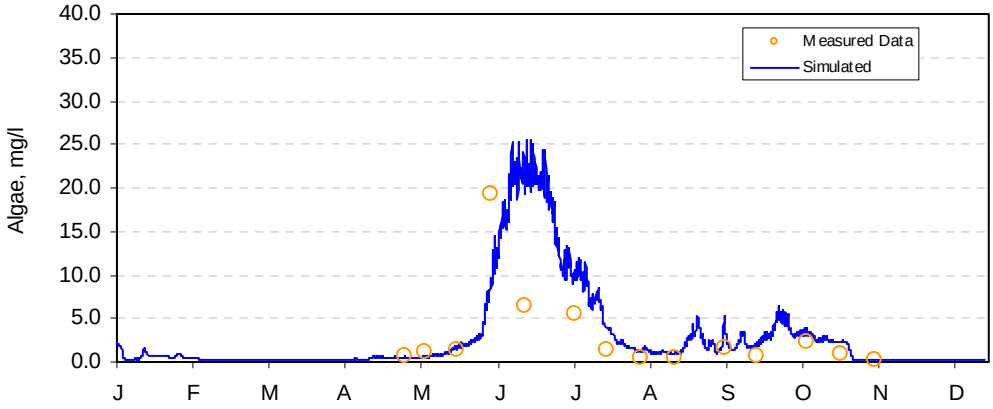


Figure E-6. Algae calibration results for Lake Ewauna—Keno Dam at Klamath River at Miller Island boat ramp (2000).

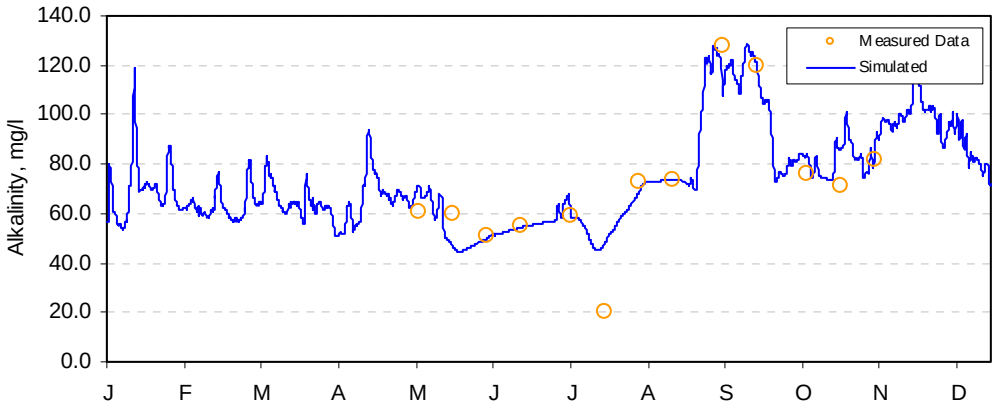


Figure E-7. Alkalinity calibration results for Lake Ewauna—Keno Dam at Klamath River at Miller Island boat ramp (2000).

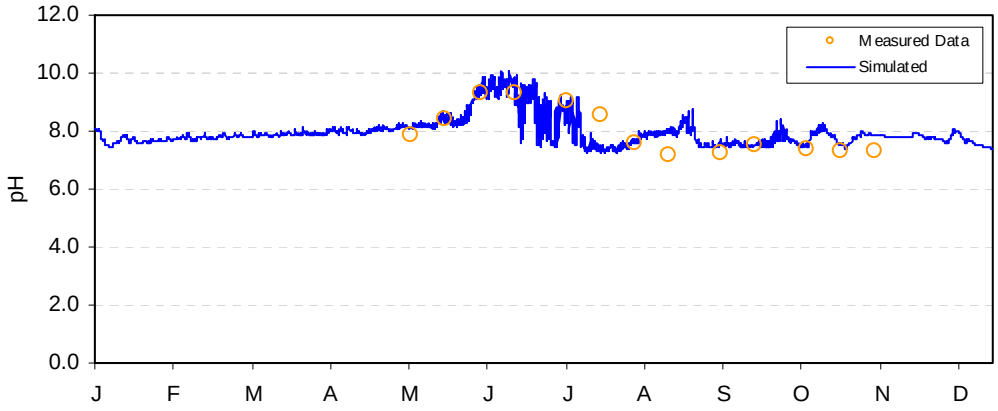


Figure E-8. pH calibration results for Lake Ewauna—Keno Dam at Klamath River at Miller Island boat ramp (2000).

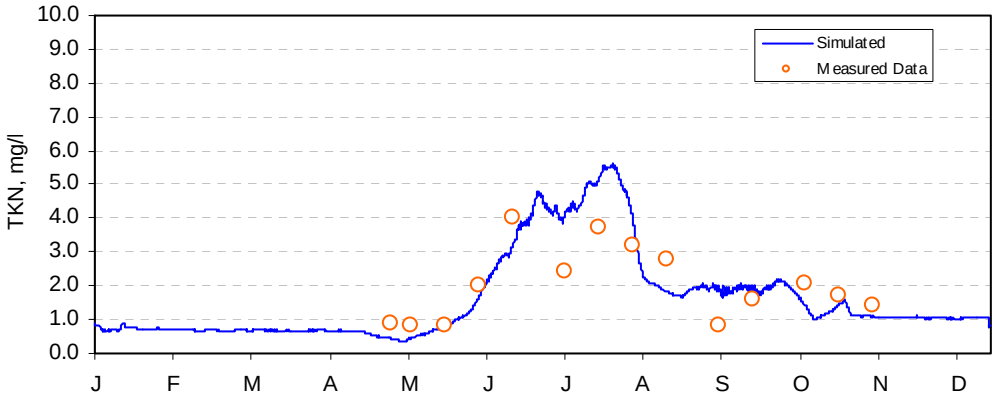


Figure E-9. TKN calibration results for Lake Ewauna—Keno Dam at Klamath River at Miller Island boat ramp (2000).

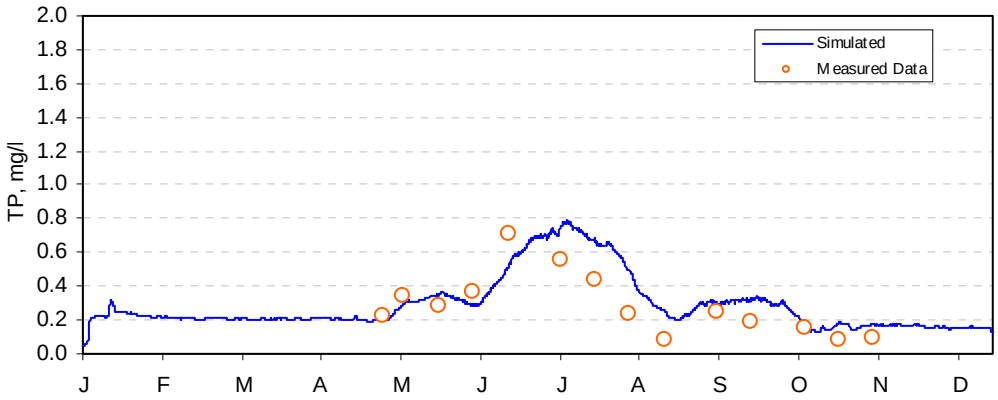


Figure E-10. TP calibration results for Lake Ewauna—Keno Dam at Klamath River at Miller Island boat ramp (2000).

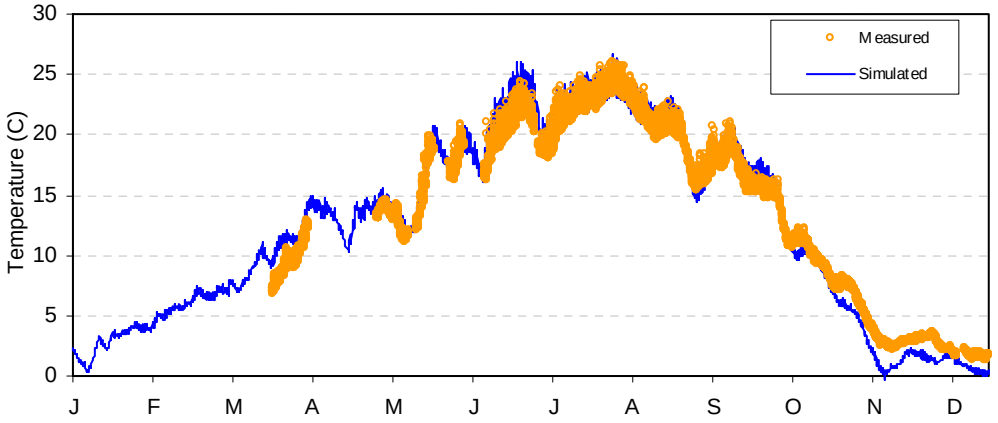


Figure E-11. Temperature calibration results for Lake Ewauna—Keno Dam at Klamath River at Keno Bridge (Hwy 66) (2000).

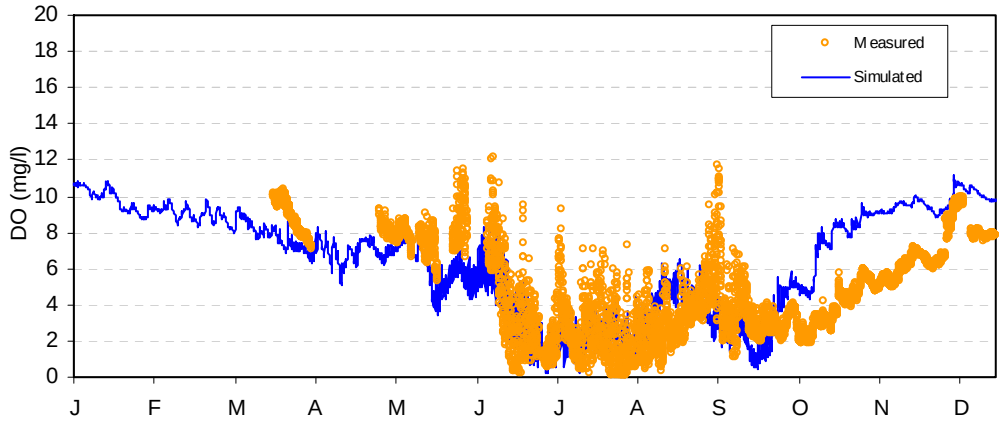


Figure E-12. Dissolved oxygen calibration results for Lake Ewauna—Keno Dam at Klamath River at Keno Bridge (Hwy 66) (2000).

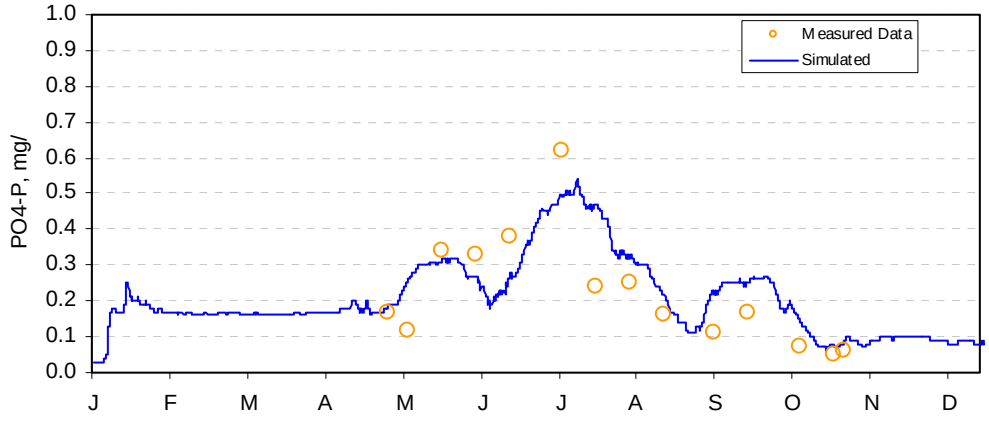


Figure E-13. PO4-P calibration results for Lake Ewauna—Keno Dam at Klamath River at Keno Bridge (Hwy 66) (2000).

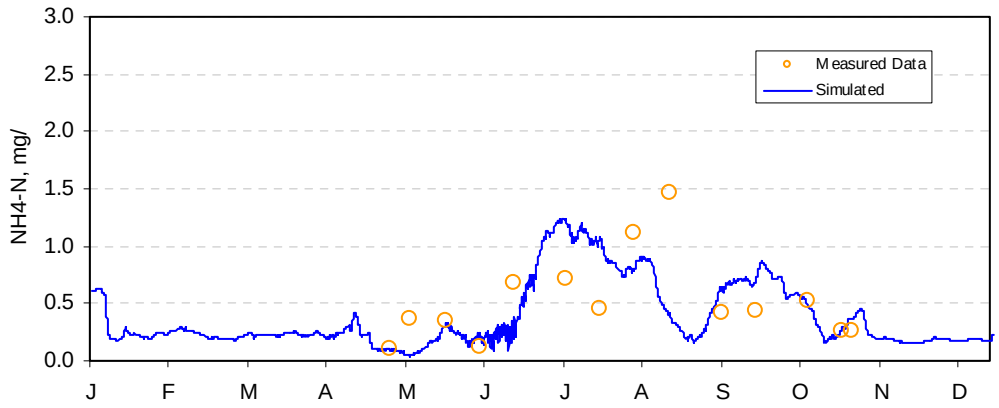


Figure E-14. NH3-N calibration results for Lake Ewauna—Keno Dam at Klamath River at Keno Bridge (Hwy 66) (2000).

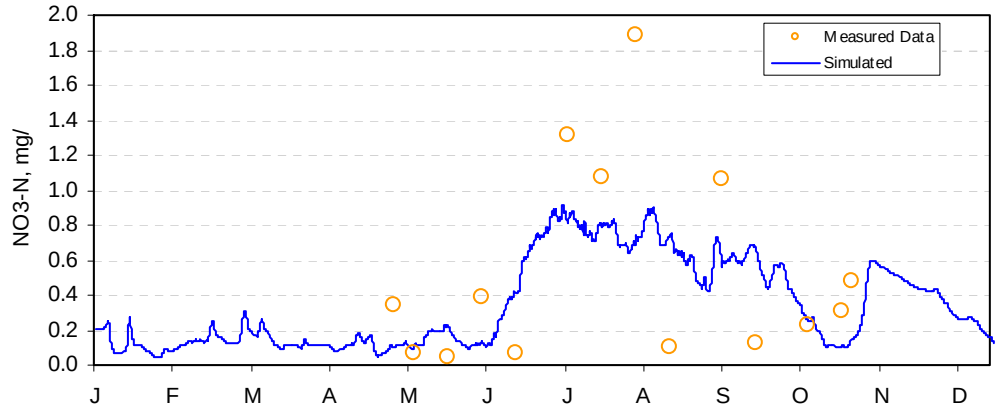


Figure E-15. NO3-N calibration results for Lake Ewauna—Keno Dam at Klamath River at Keno Bridge (Hwy 66) (2000).

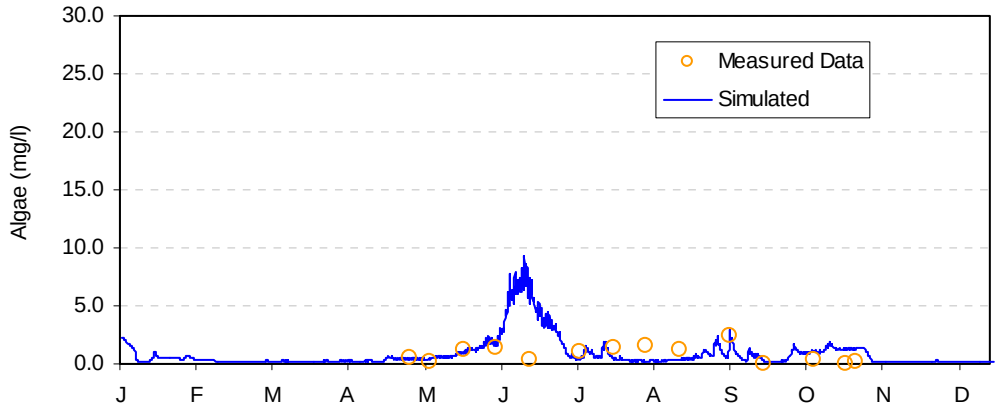


Figure E-16. Algae calibration results for Lake Ewauna—Keno Dam at Klamath River at Keno Bridge (Hwy 66) (2000).

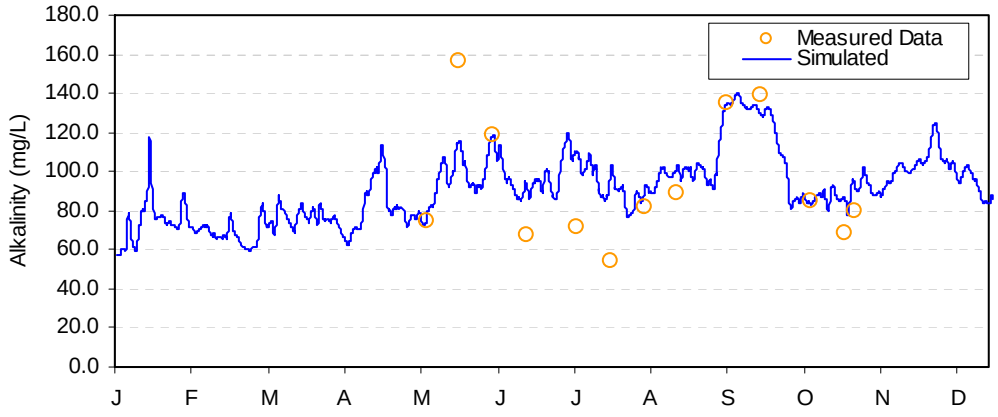


Figure E-17. Alkalinity calibration results for Lake Ewauna—Keno Dam at Klamath River at Keno Bridge (Hwy 66) (2000).

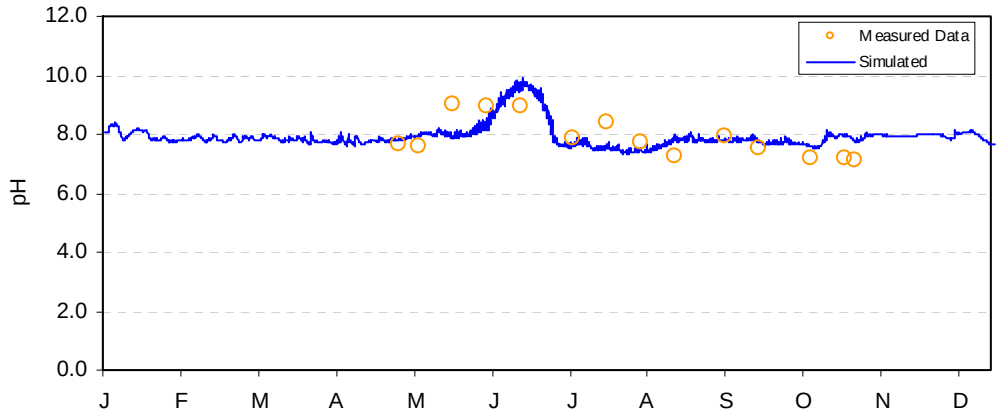


Figure E-18. pH calibration results for Lake Ewauna—Keno Dam at Klamath River at Keno Bridge (Hwy 66) (2000).

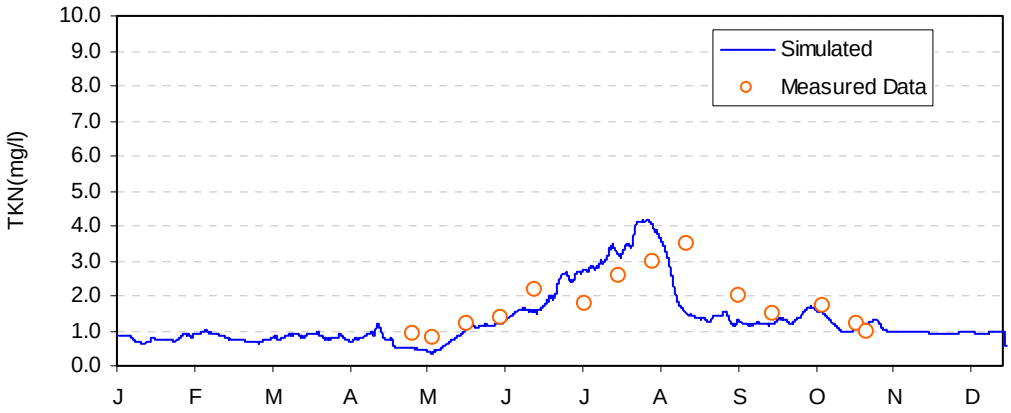


Figure E-19. TKN calibration results for Lake Ewauna—Keno Dam at Klamath River at Keno Bridge (Hwy 66) (2000).

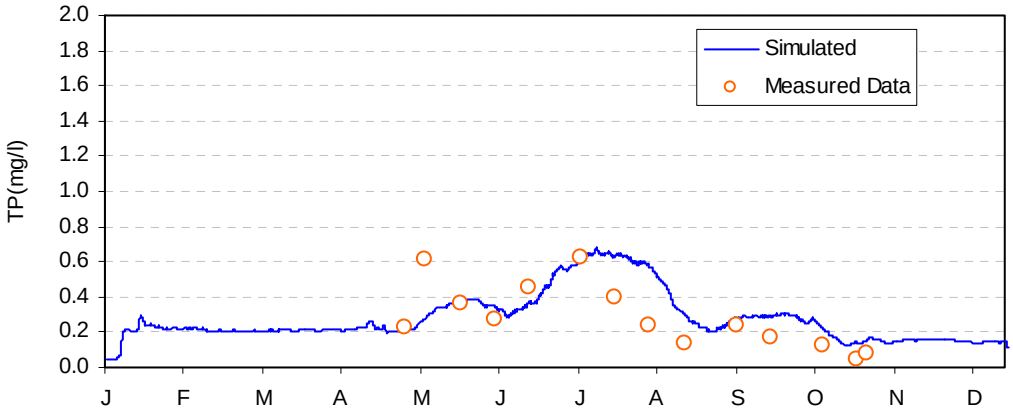


Figure E-20. TP calibration results for Lake Ewauna—Keno Dam at Klamath River at Keno Bridge (Hwy 66) (2000).

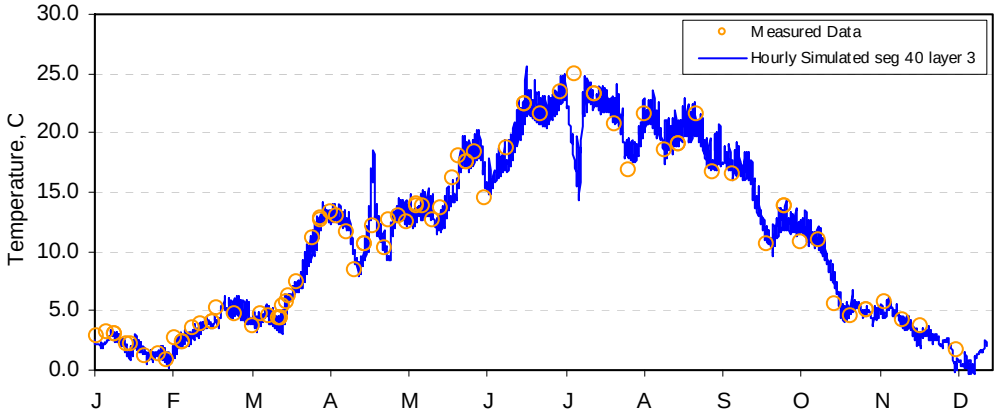


Figure E-21. Temperature calibration results for Lake Ewauna—Railroad Bridge Drawspan (2002).

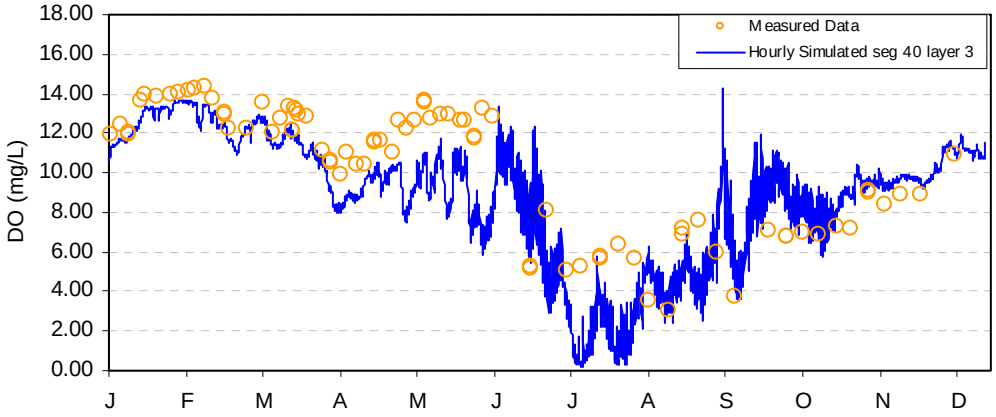


Figure E-22. Dissolved Oxygen calibration results for Lake Ewauna—Railroad Bridge Drawspan (2002).

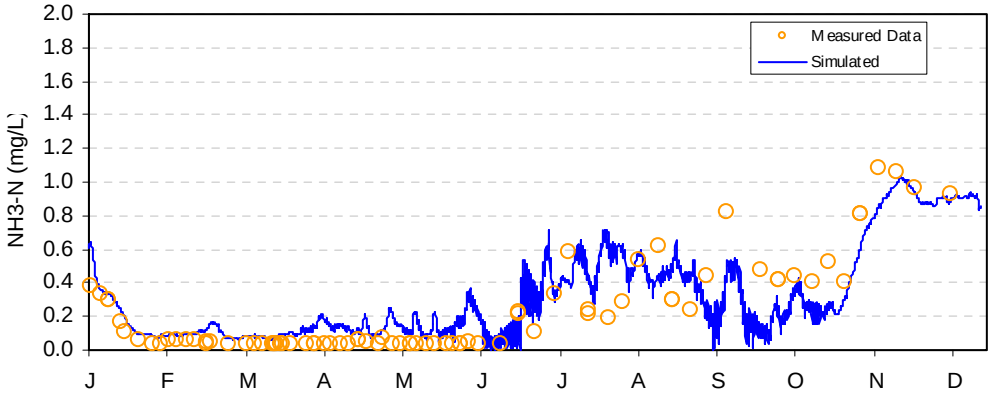


Figure E-23. NH3-N calibration results for Lake Ewauna—Railroad Bridge Drawspan (2002).

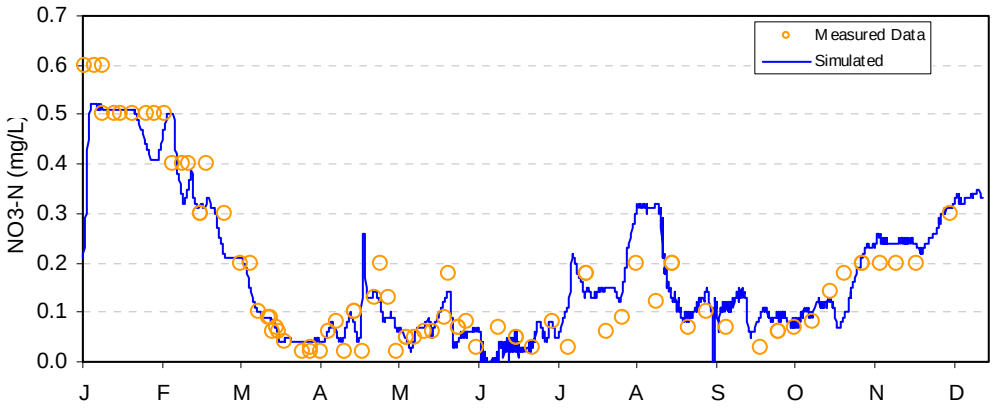


Figure E-24. NO3-N calibration results for Lake Ewauna—Railroad Bridge Drawspan (2002).

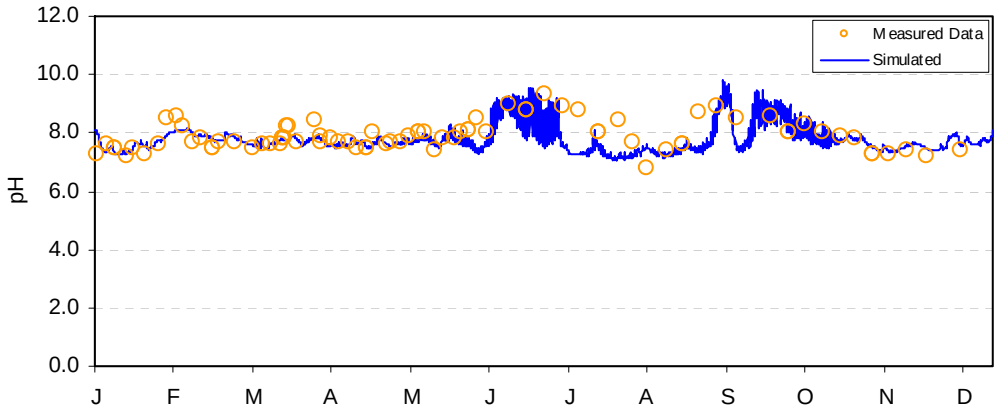


Figure E-25. pH calibration results for Lake Ewauna—Railroad Bridge Drawspan (2002).

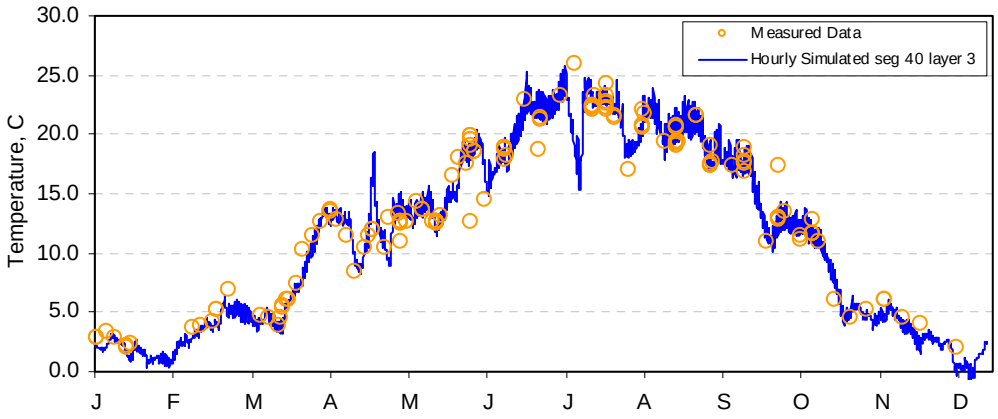


Figure E-26. Temperature calibration results for Lake Ewauna—South Side Bypass Bridge (2002).

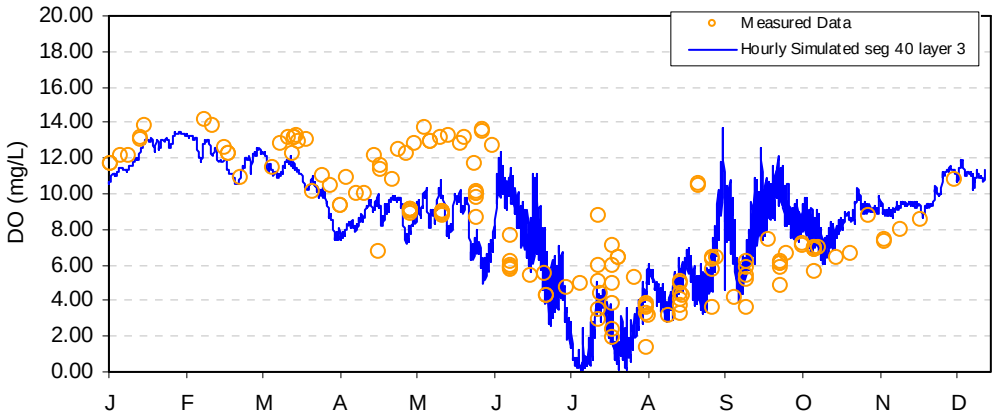


Figure E-27. Dissolved Oxygen calibration results for Lake Ewauna—South Side Bypass Bridge (2002).

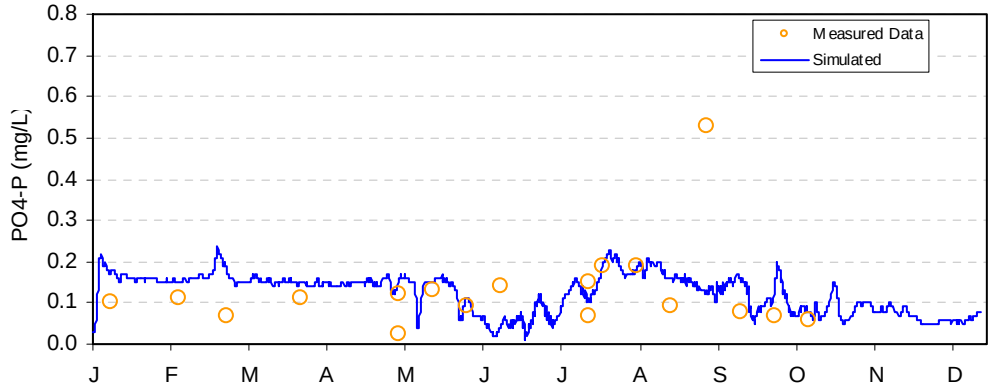


Figure E-28. PO4-P calibration results for Lake Ewauna—South Side Bypass Bridge (2002).

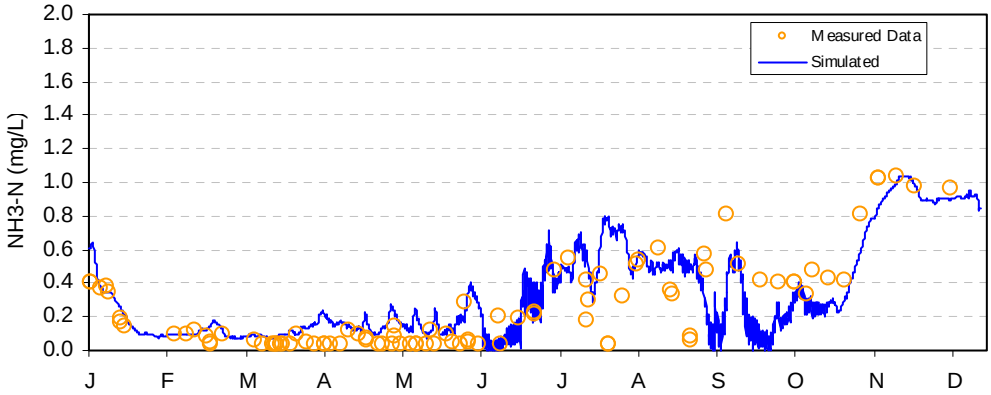


Figure E-29. NH3-N calibration results for Lake Ewauna—South Side Bypass Bridge (2002).

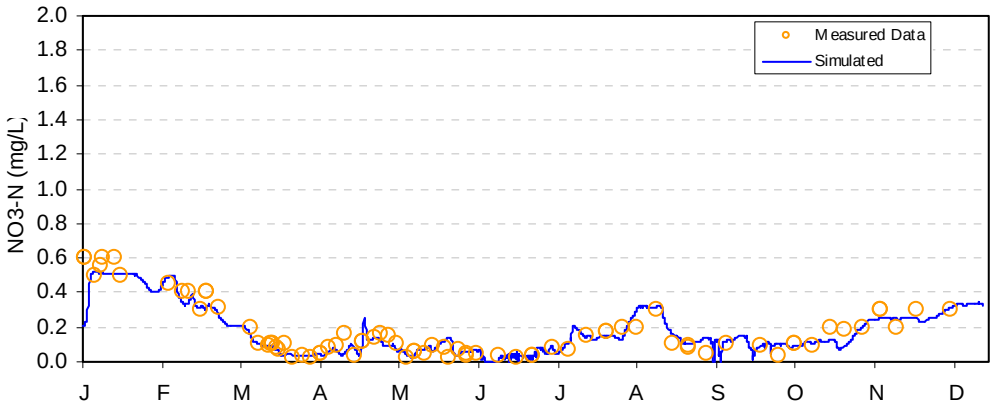


Figure E-30. NO3-N calibration results for Lake Ewauna—South Side Bypass Bridge (2002).

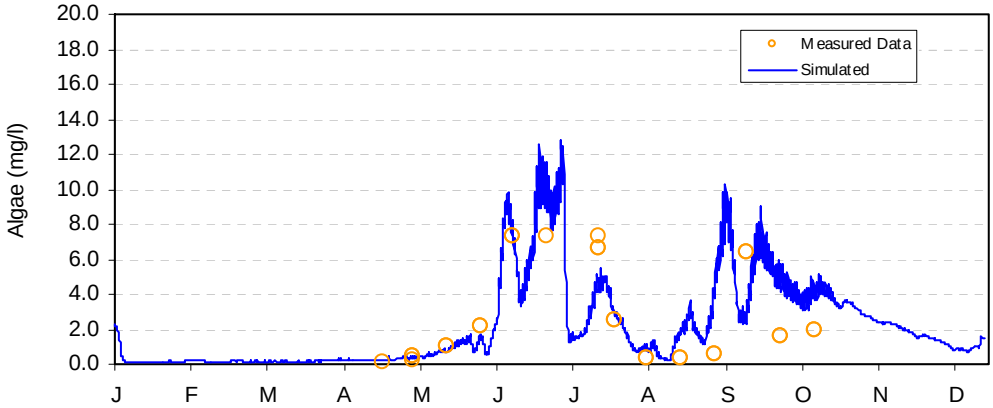


Figure E-31. Algae calibration results for Lake Ewauna—South Side Bypass Bridge (2002).

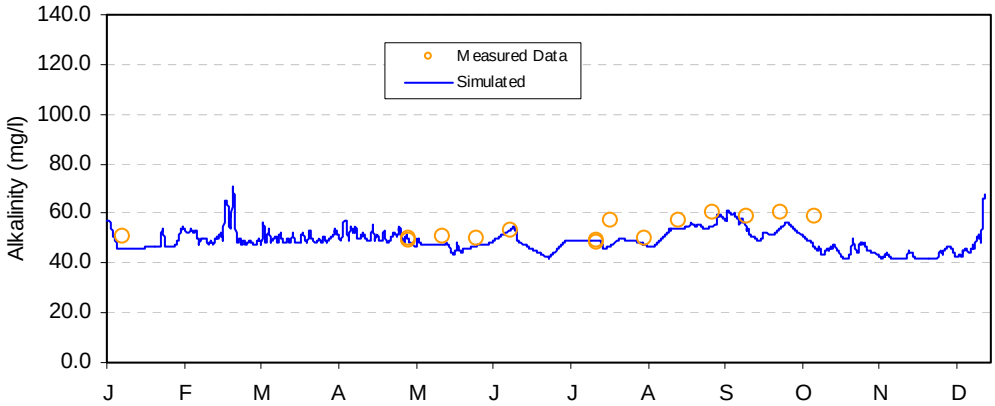


Figure E-32. Alkalinity calibration results for Lake Ewauna—South Side Bypass Bridge (2002).

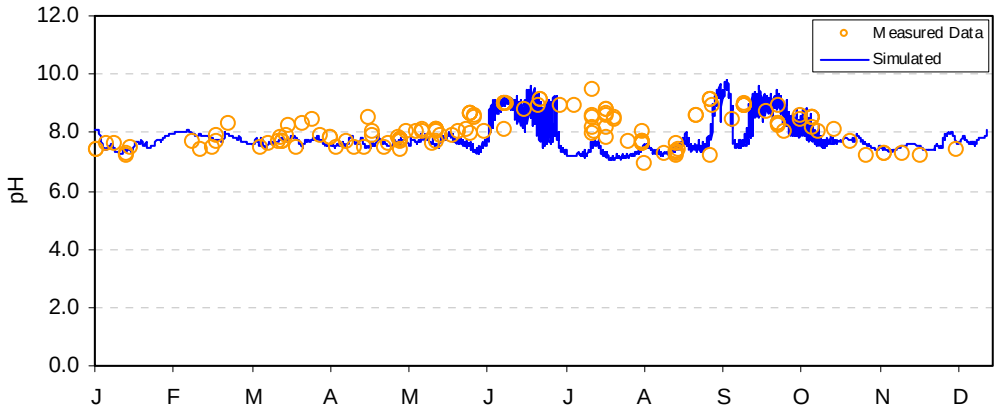


Figure E-33. pH calibration results for Lake Ewauna—South Side Bypass Bridge (2002).

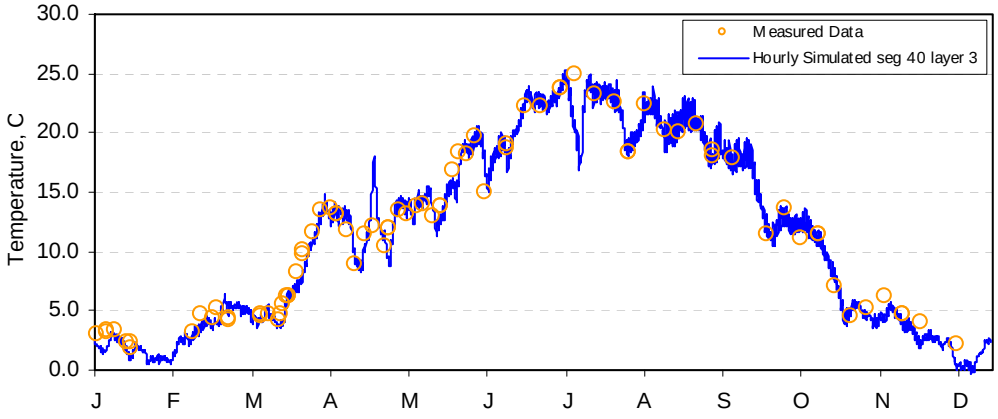


Figure E-34. Temperature calibration results for Lake Ewauna—Klamath River at Hwy 97 (2002).

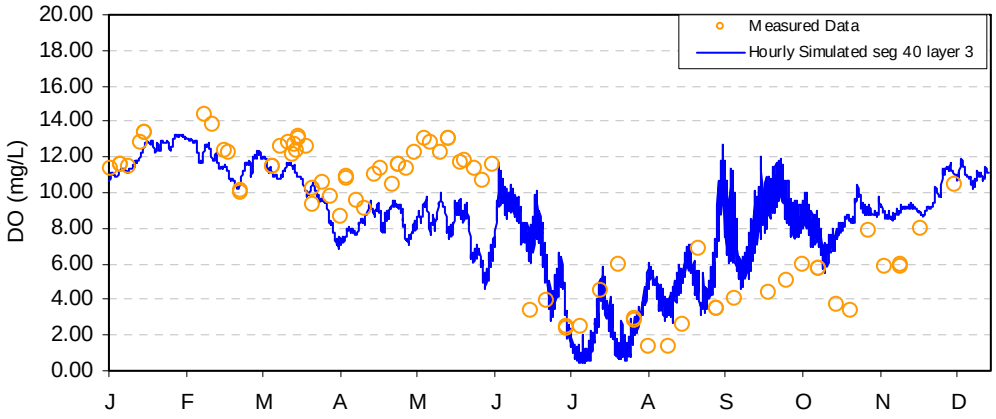


Figure E-35. Dissolved Oxygen calibration results for Lake Ewauna—Klamath River at Hwy 97 (2002).

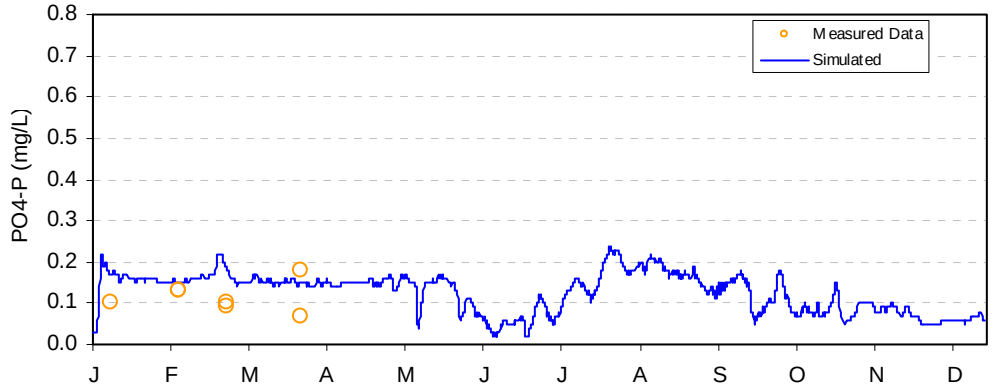


Figure E-36. PO4-P calibration results for Lake Ewauna—Klamath River at Hwy 97 (2002).

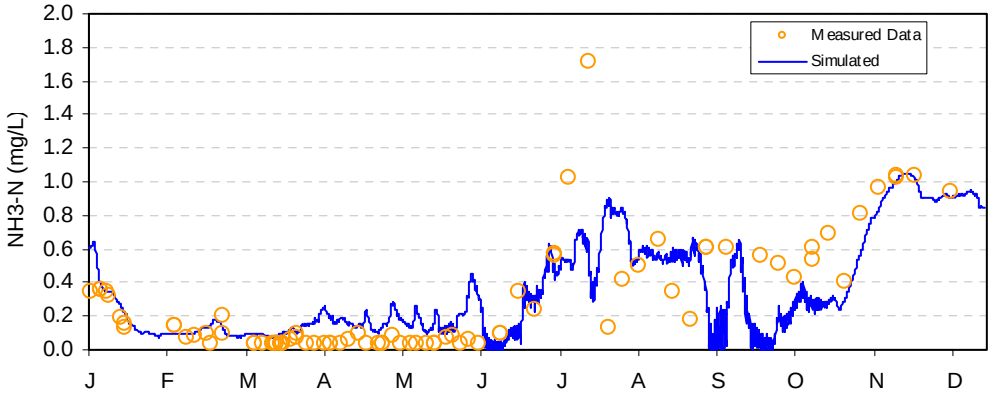


Figure E-37. NH3-N calibration results for Lake Ewauna—Klamath River at Hwy 97 (2002).

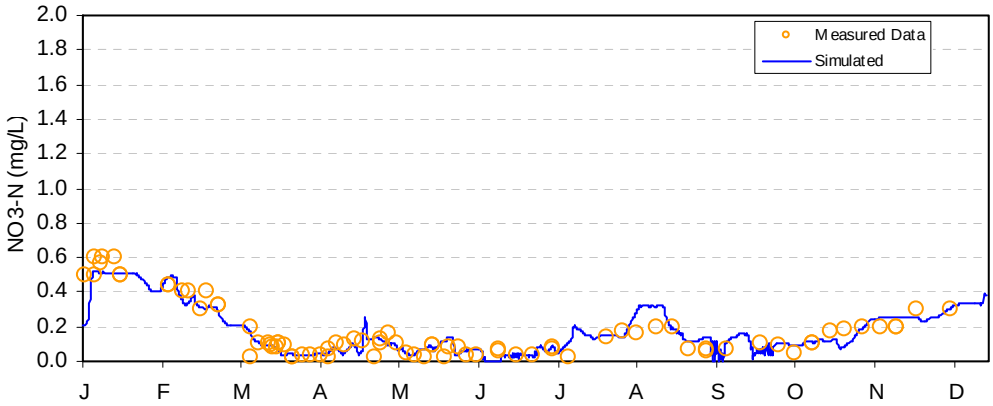


Figure E-38. NO3-N calibration results for Lake Ewauna—Klamath River at Hwy 97 (2002).

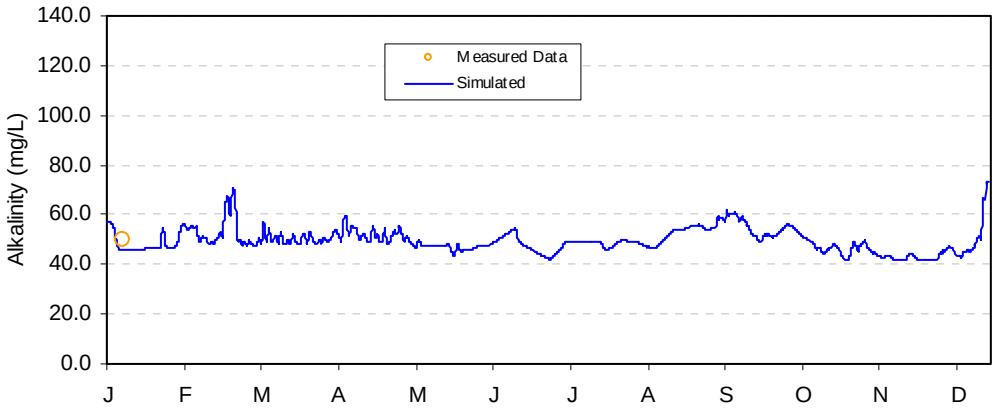


Figure E-39. Alkalinity calibration results for Lake Ewauna—Klamath River at Hwy 97 (2002).

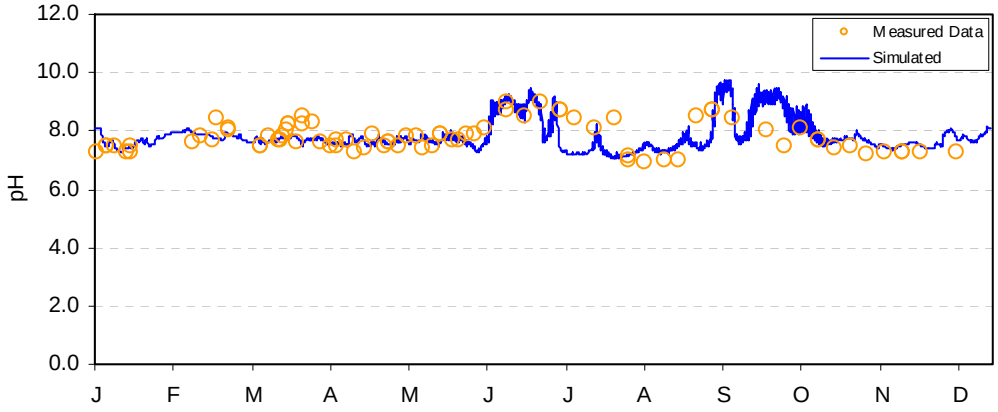


Figure E-40. pH calibration results for Lake Ewauna—Klamath River at Hwy 97 (2002).

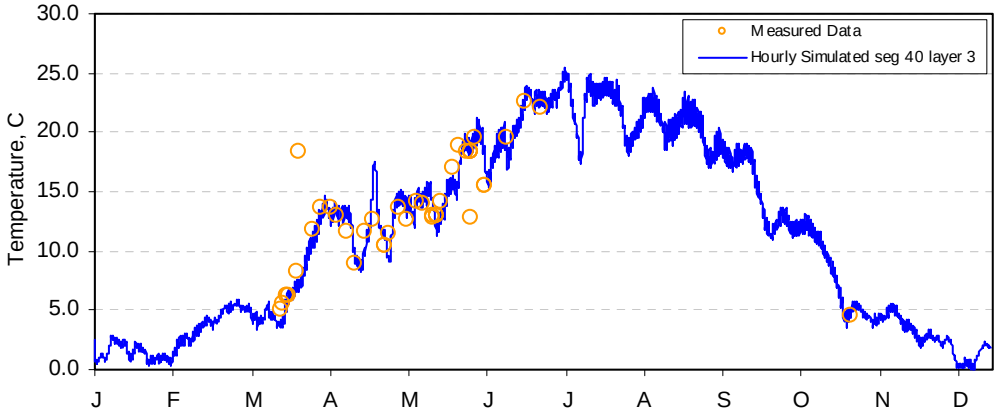


Figure E-41. Temperature calibration results for Lake Ewauna—Lost River Diversion at Klamath River (2002).

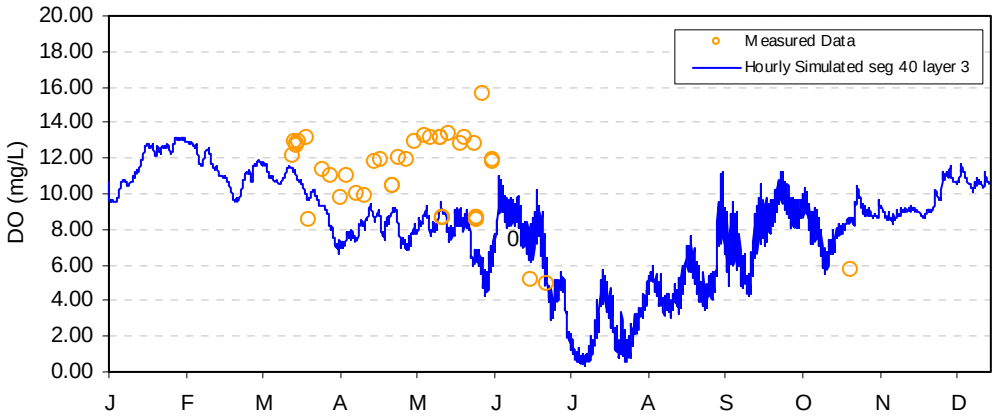


Figure E-42. Dissolved Oxygen calibration results for Lake Ewauna—Lost River Diversion at Klamath River (2002).

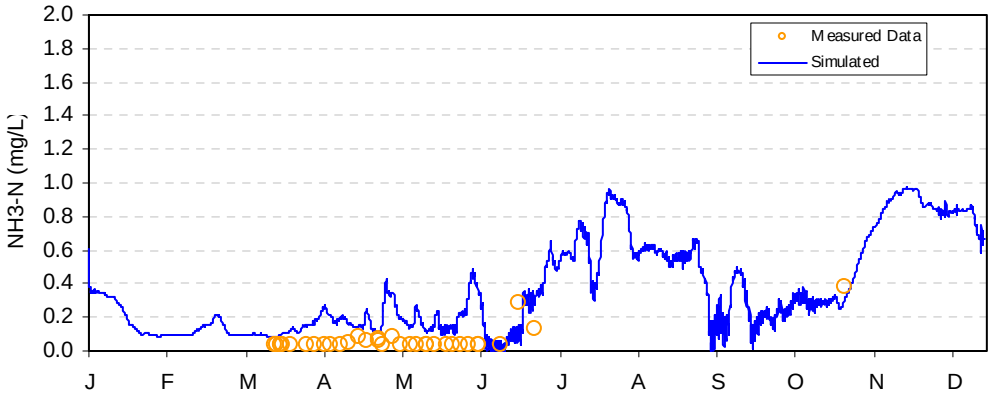


Figure E-43. NH3-N calibration results for Lake Ewauna—Lost River Diversion at Klamath River (2002).

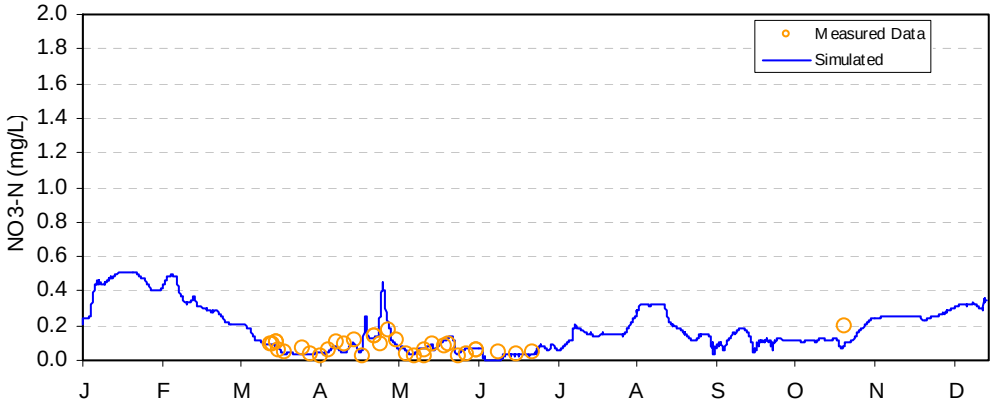


Figure E-44. NO3-N calibration results for Lake Ewauna—Lost River Diversion at Klamath River (2002).

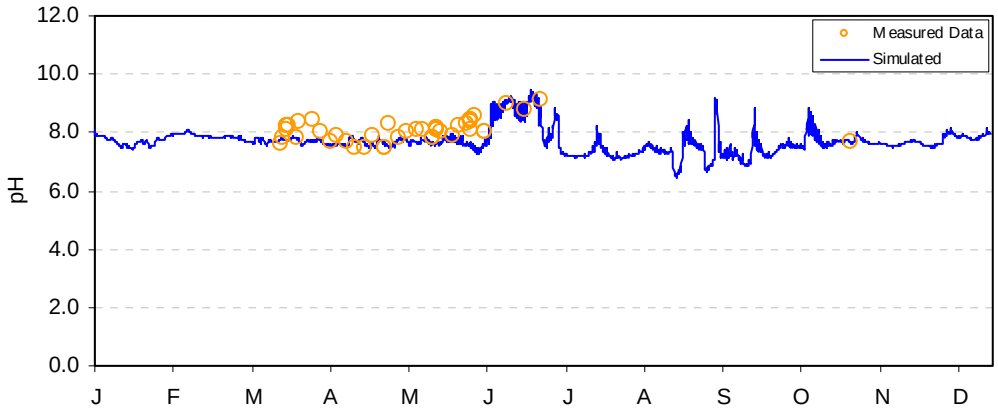


Figure E-45. pH calibration results for Lake Ewauna—Lost River Diversion at Klamath River (2002).

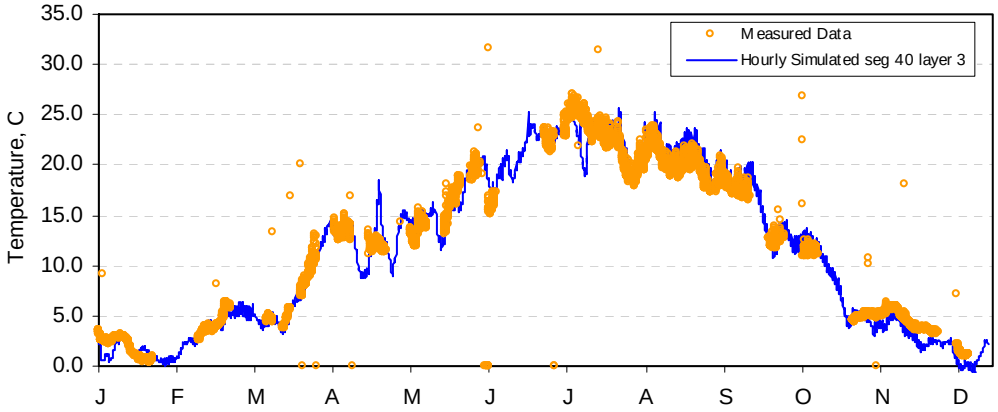


Figure E-46. Temperature calibration results for Lake Ewauna—Klamath River at Miller Island (2002).

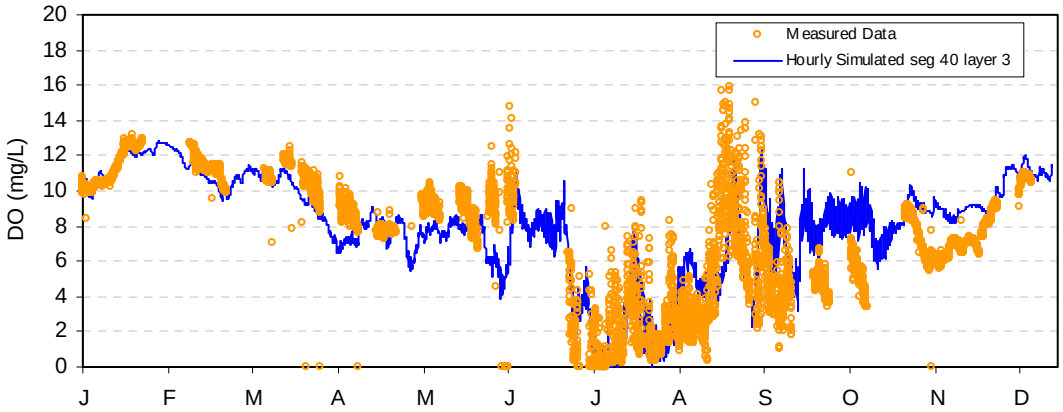


Figure E-47. Dissolved Oxygen calibration results for Lake Ewauna—Klamath River at Miller Island (2002).

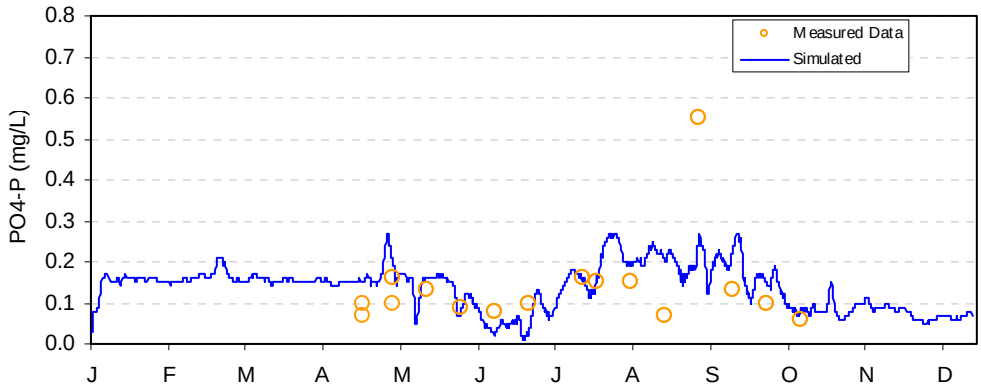


Figure E-48. PO4-P calibration results for Lake Ewauna—Klamath River at Miller Island (2002).

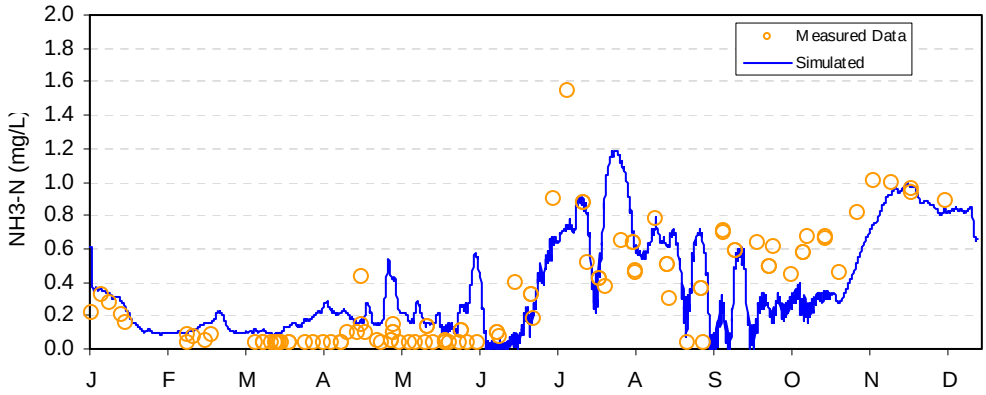


Figure E-49. NH3-N calibration results for Lake Ewauna—Klamath River at Miller Island (2002).

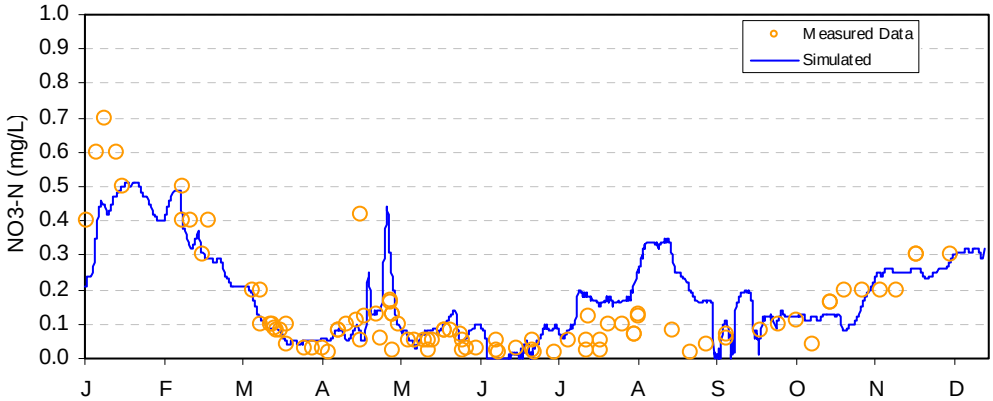


Figure E-50. NO3-N calibration results for Lake Ewauna—Klamath River at Miller Island (2002).

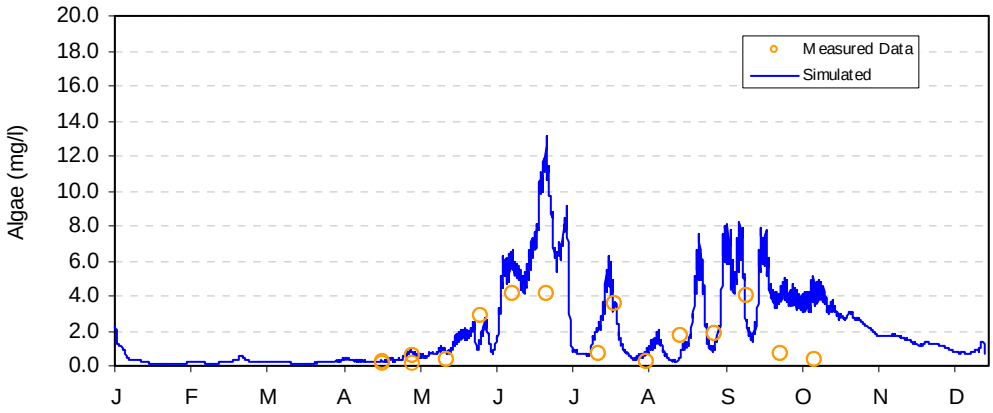


Figure E-51. Algae calibration results for Lake Ewauna—Klamath River at Miller Island (2002).

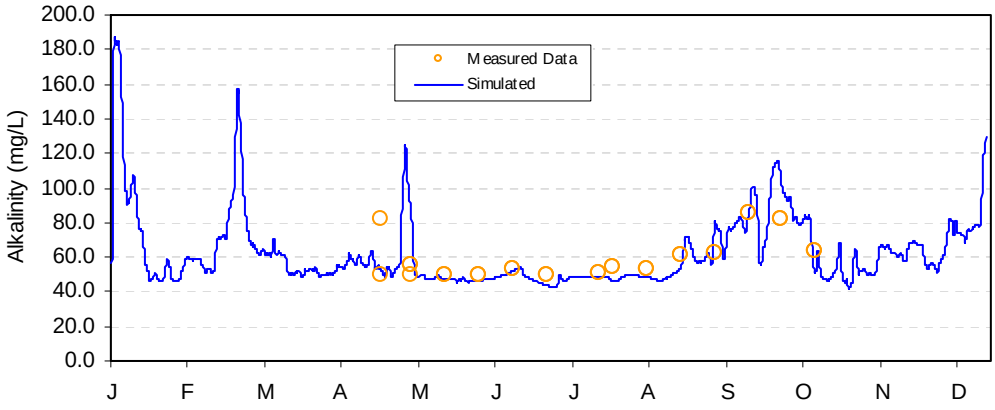


Figure E-52. Alkalinity calibration results for Lake Ewauna—Klamath River at Miller Island (2002).

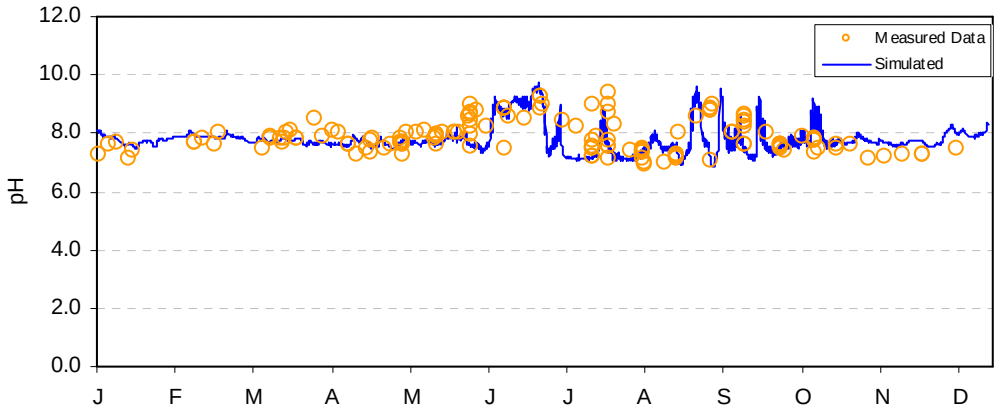


Figure E-53. pH calibration results for Lake Ewauna—Klamath River at Miller Island (2002).

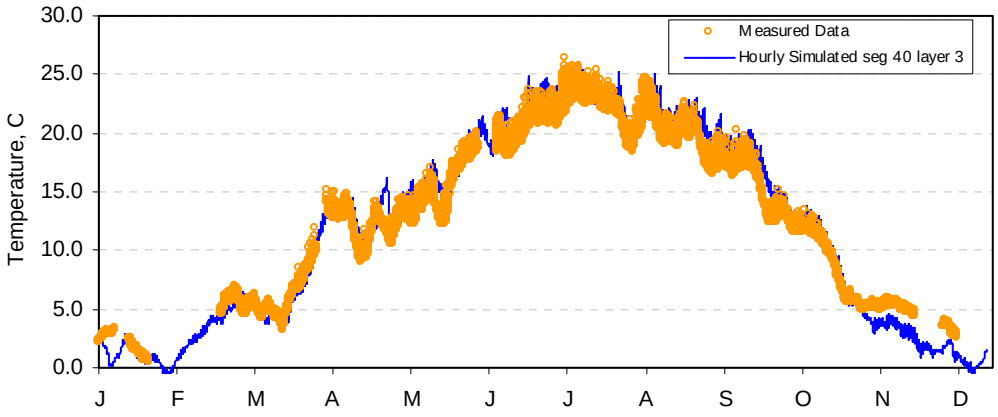


Figure E-54. Temperature calibration results for Lake Ewauna—Klamath River at Keno Bridge (Hwy 66) (2002).

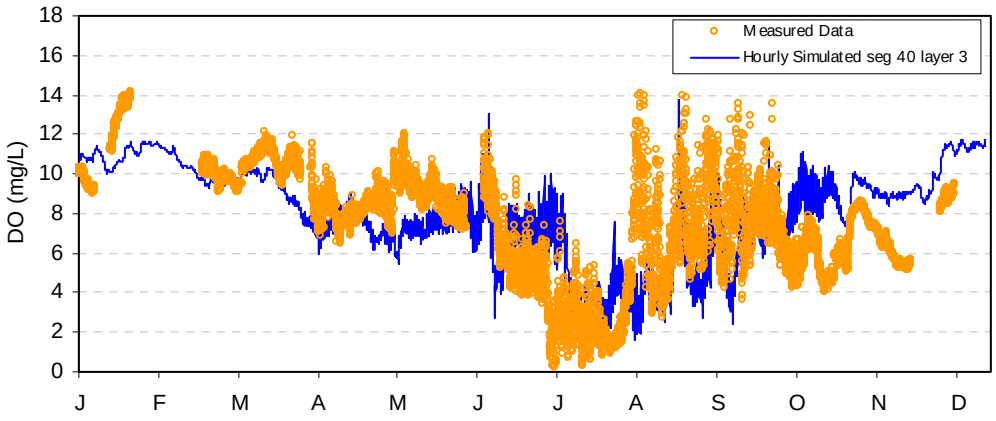


Figure E-55. Dissolved Oxygen calibration results for Lake Ewauna—Klamath River at Keno Bridge (Hwy 66) (2002).

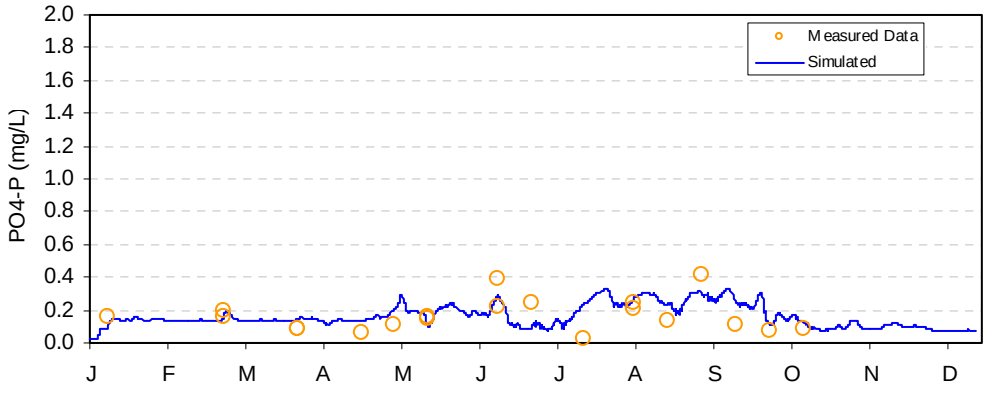


Figure E-56. PO4-P calibration results for Lake Ewauna—Klamath River at Keno Bridge (Hwy 66) (2002).

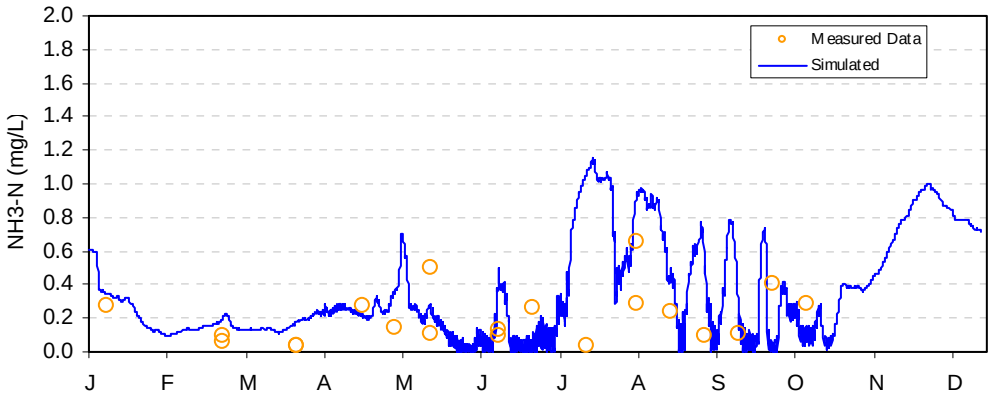


Figure E-57. NH3-N calibration results for Lake Ewauna—Klamath River at Keno Bridge (Hwy 66) (2002).

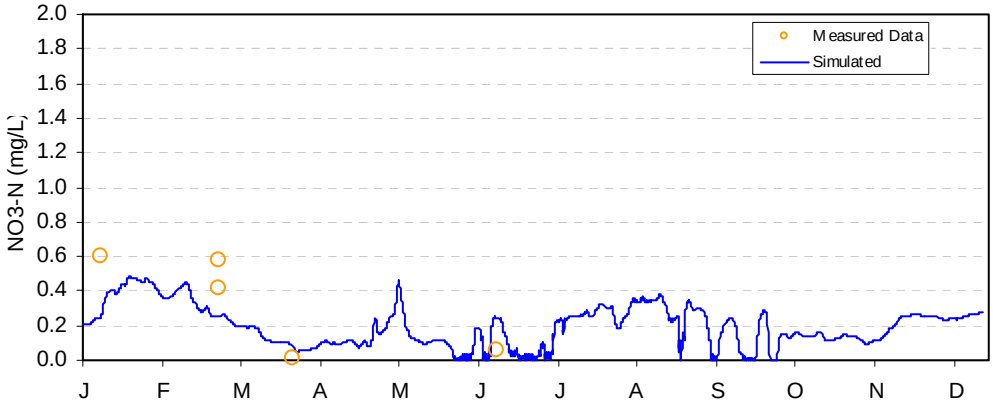


Figure E-58. NO3-N calibration results for Lake Ewauna—Klamath River at Keno Bridge (Hwy 66) (2002).

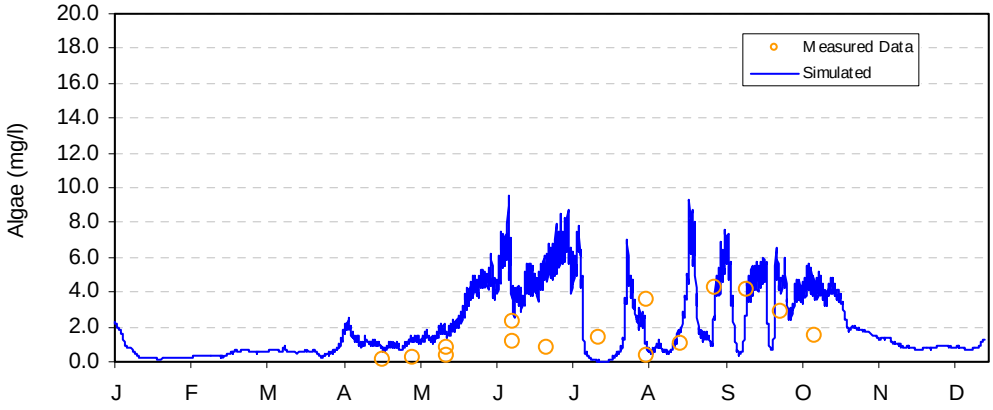


Figure E-59. Algae calibration results for Lake Ewauna—Klamath River at Keno Bridge (Hwy 66) (2002).

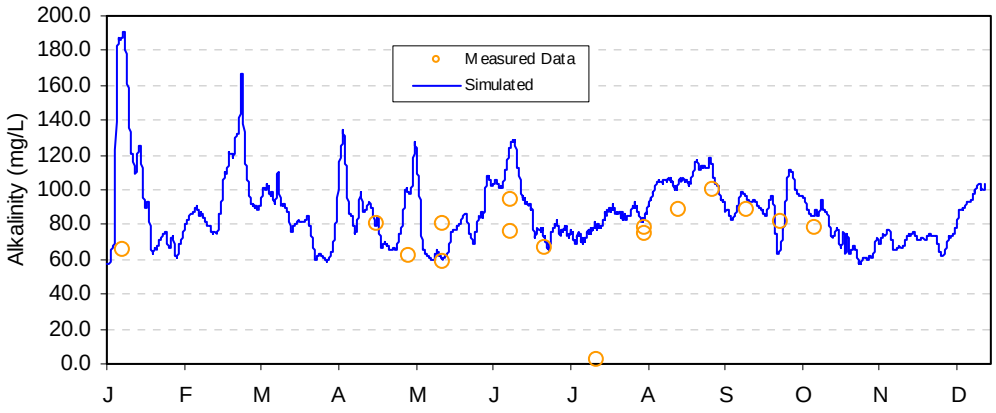


Figure E-60. Alkalinity calibration results for Lake Ewauna—Klamath River at Keno Bridge (Hwy 66) (2002).

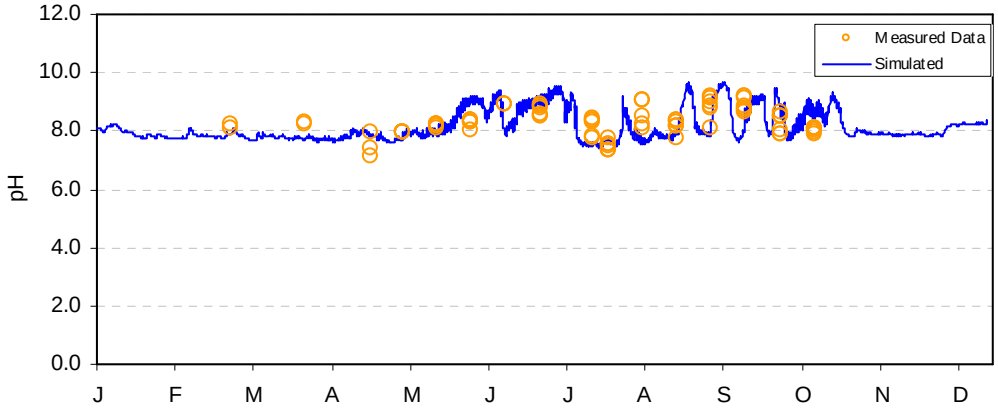


Figure E-61. pH calibration results for Lake Ewauna—Klamath River at Keno Bridge (Hwy 66) 2002).

Appendix F

Calibration Results for Keno Dam to J. C. Boyle Reservoir (Modeling Segment 3)

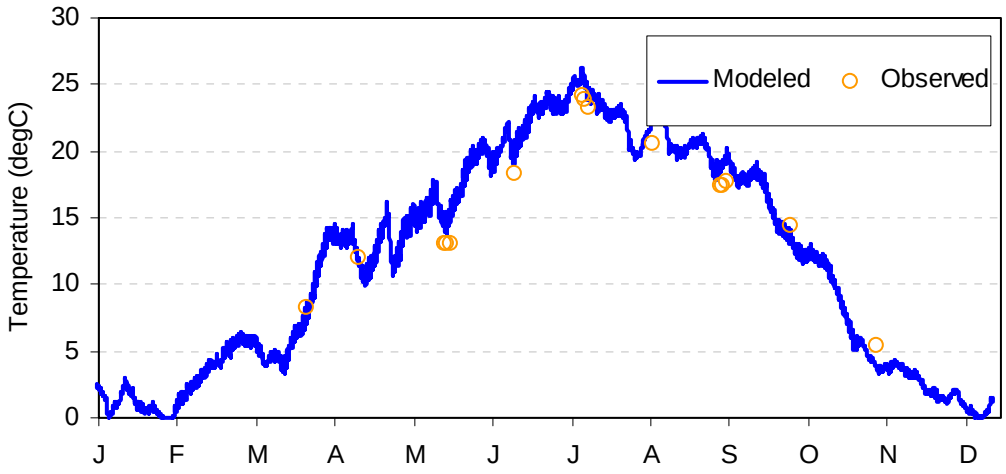


Figure F-1. Temperature calibration results for Klamath River downstream of Keno Dam (2002).

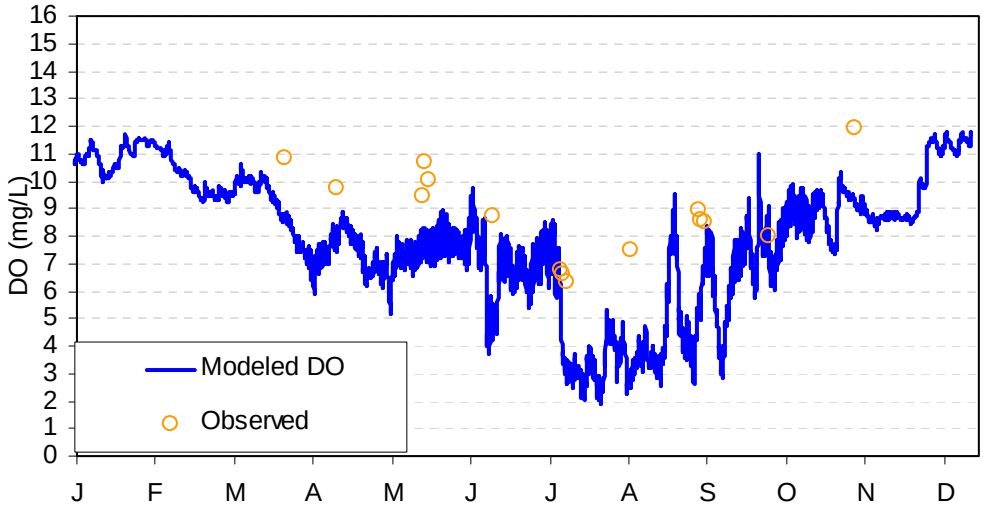


Figure F-2. Dissolved oxygen calibration results for Klamath River downstream of Keno Dam (2002).

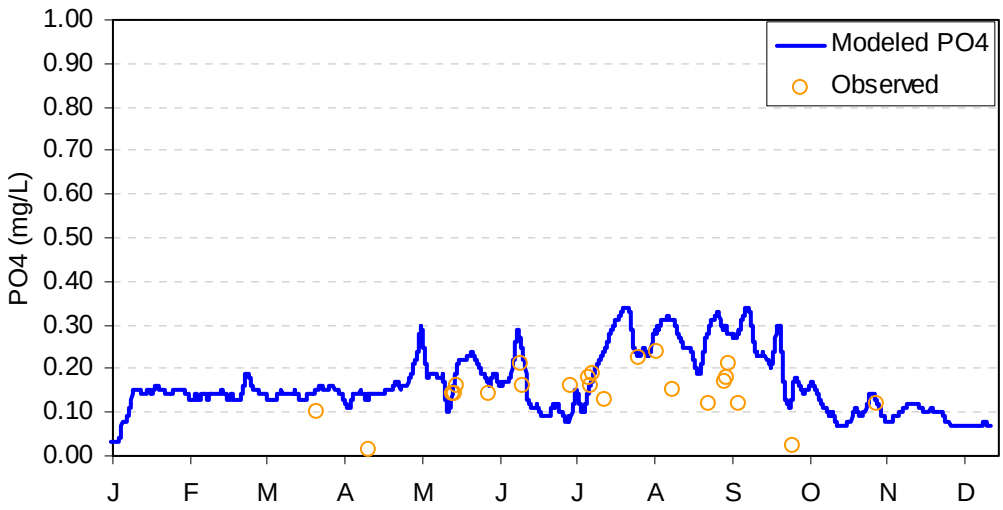


Figure F-3. PO_4 calibration results for Klamath River downstream of Keno Dam (2002).

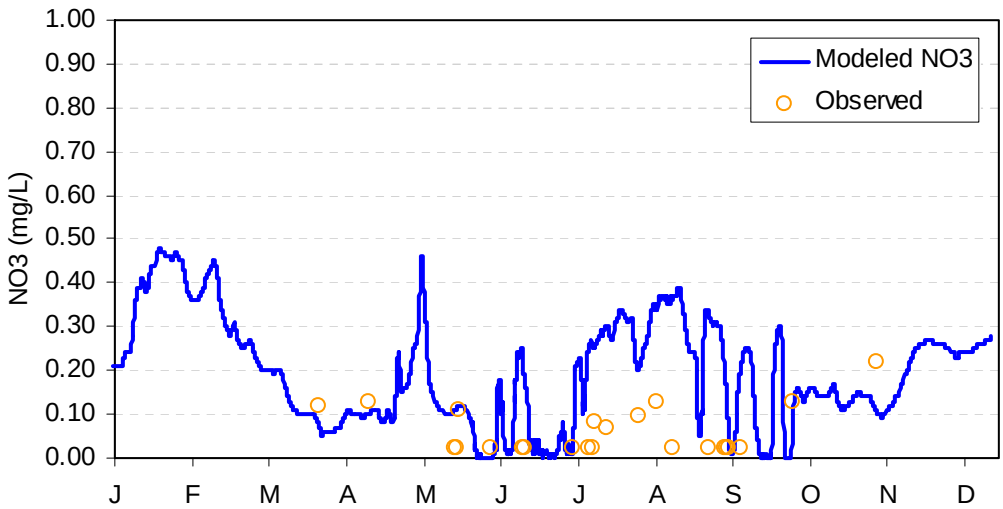


Figure F-4. NO_3 calibration results for Klamath River downstream of Keno Dam (2002).

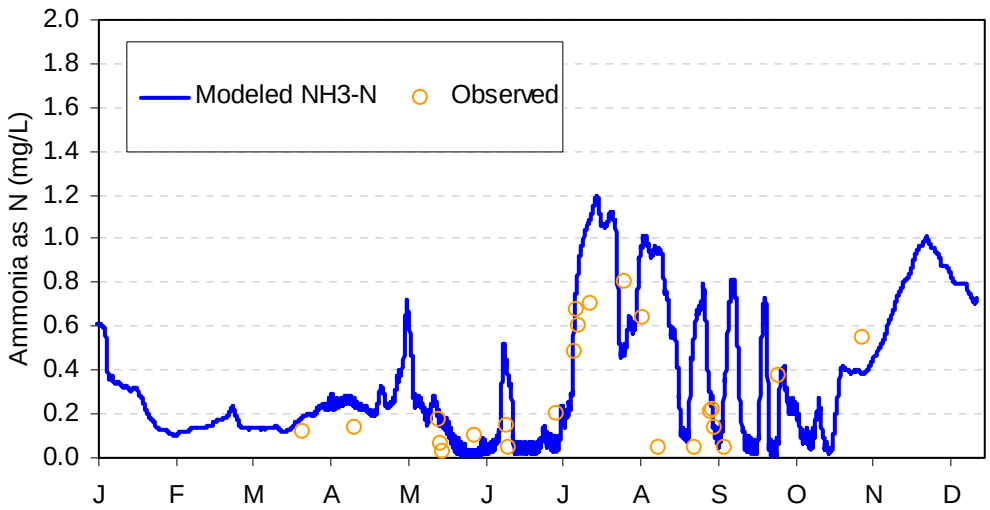


Figure F-5. NH_3-N calibration results for Klamath River downstream of Keno Dam (2002).

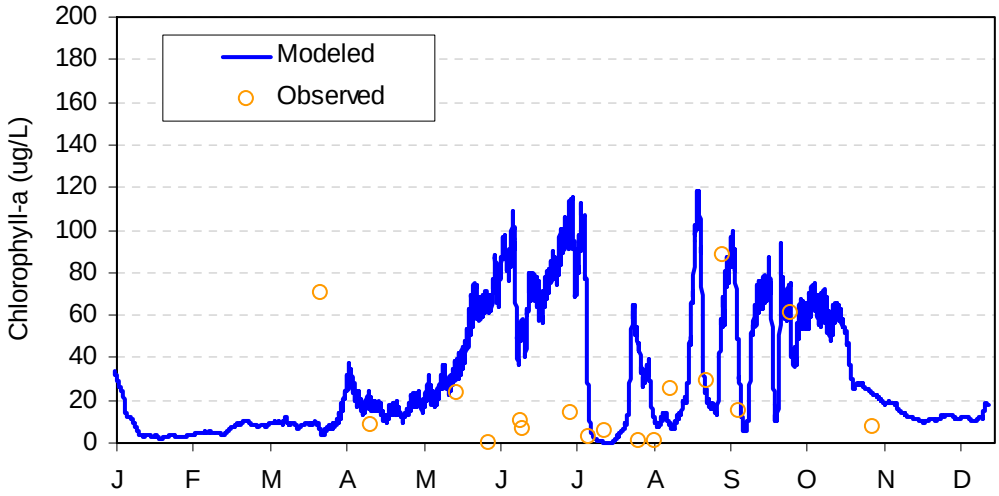


Figure F-6. Chlorophyll *a* calibration results for Klamath River downstream of Keno Dam (2002).

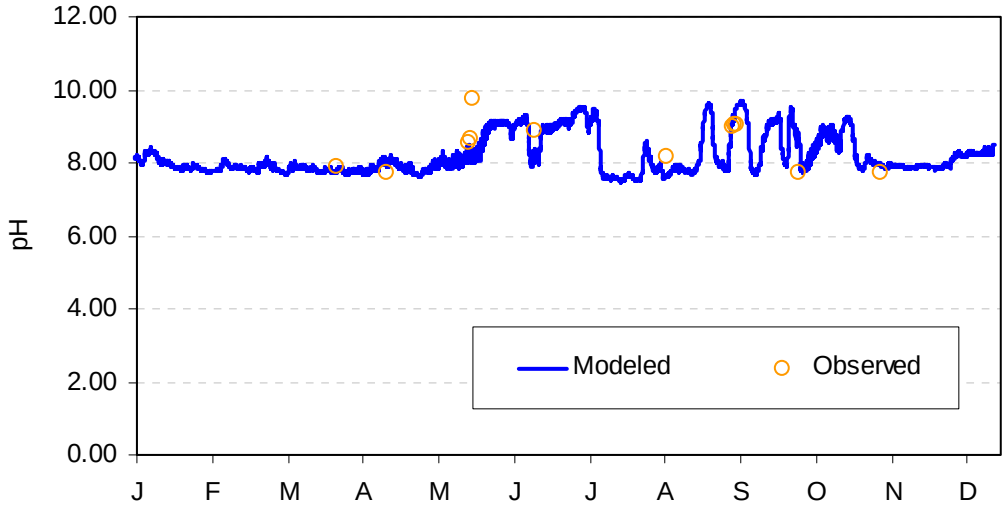


Figure F-7. pH calibration results for Klamath River downstream of Keno Dam (2002).

Appendix G

Water Quality Calibration Results for J.C. Boyle Reservoir (Modeling Segment 4)

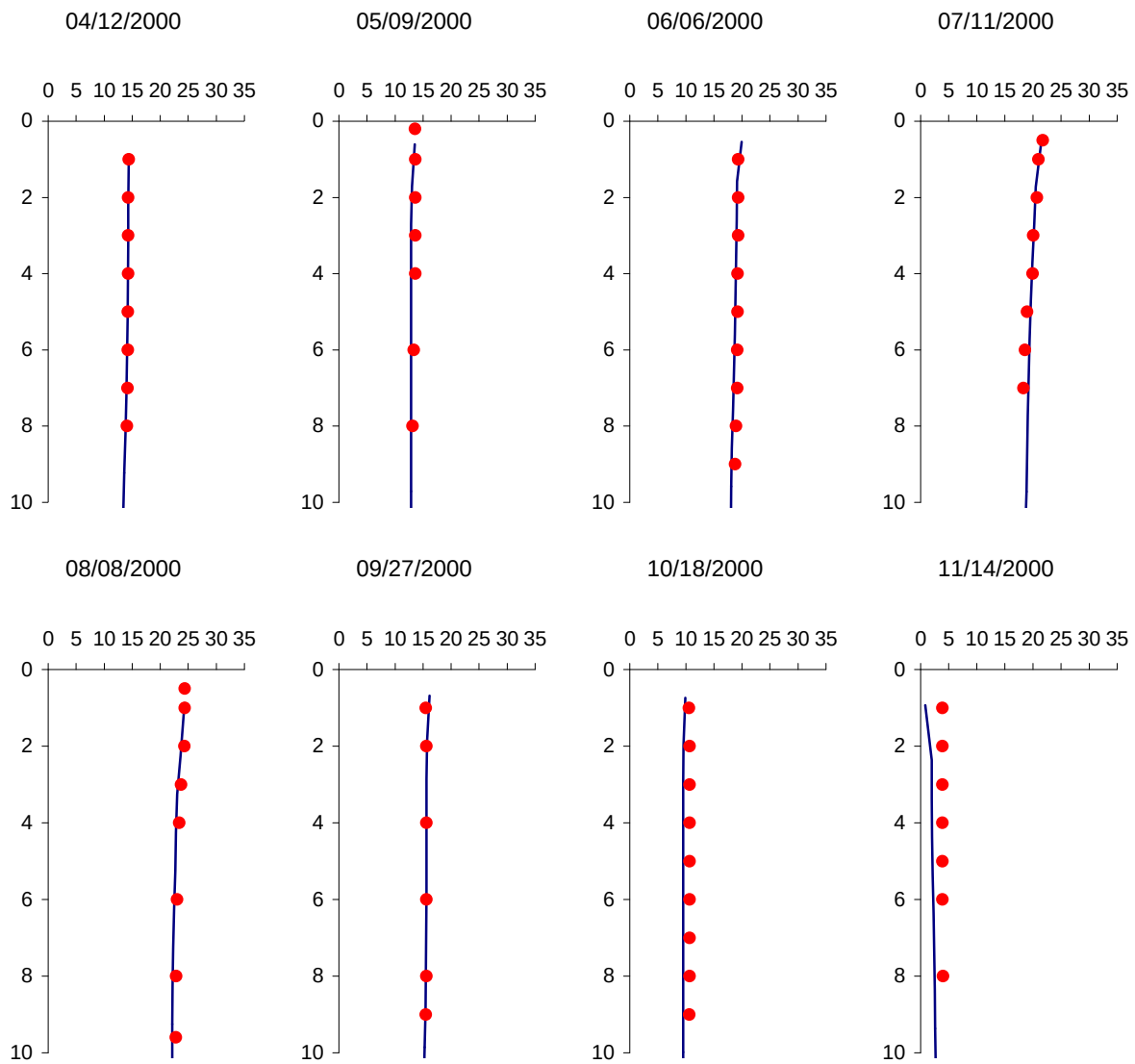


Figure G-1. Temperature profile calibration results at J.C. Boyle Reservoir at deepest point - 2000 [X-axis Temperature (degC) vs. Y-axis Depth (m)].

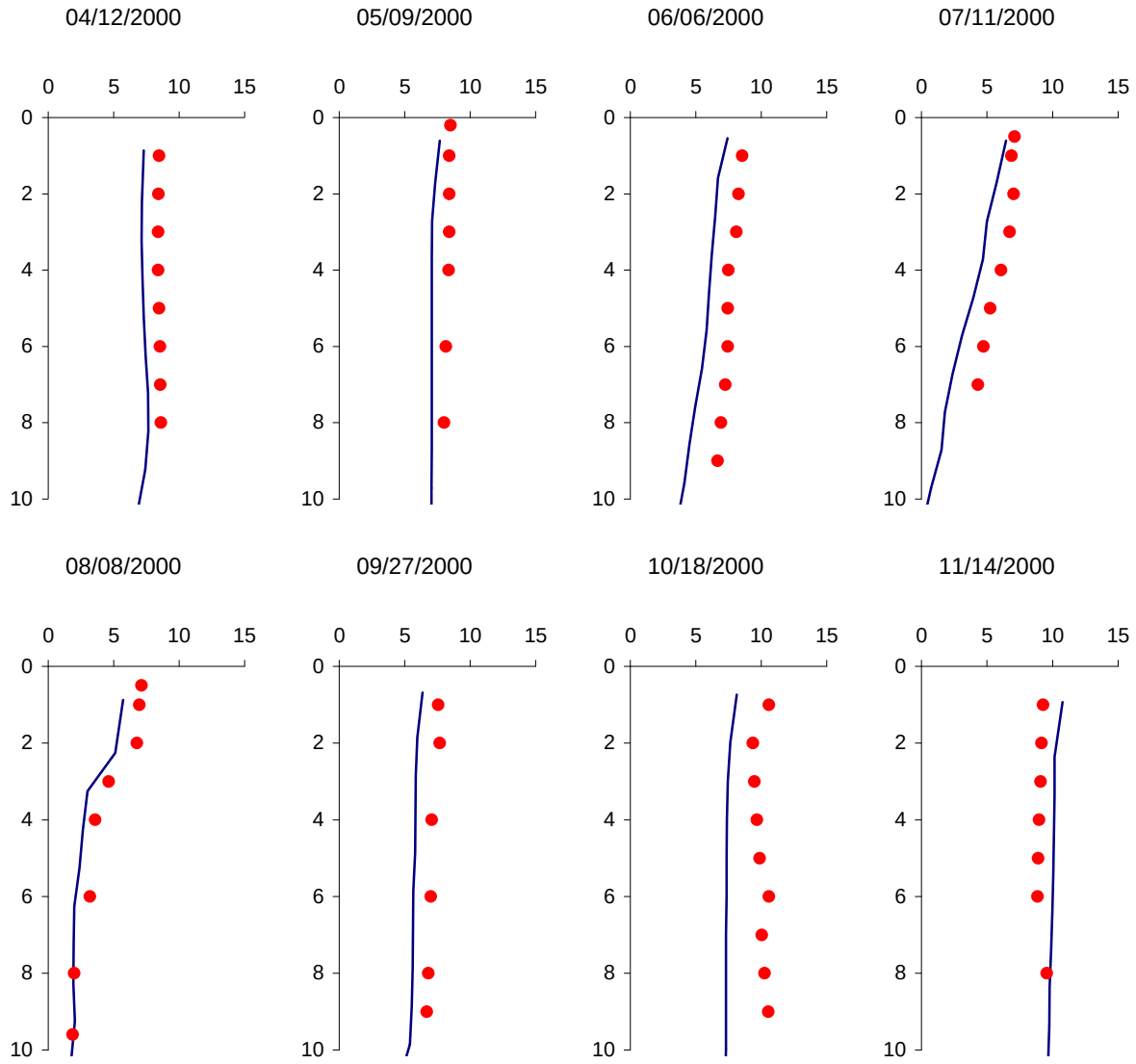


Figure G-2. Dissolved oxygen profile calibration results at J.C. Boyle Reservoir at deepest point - 2000 [(X-axis Dissolved oxygen (mg/L) vs. Y-axis Depth (m)].

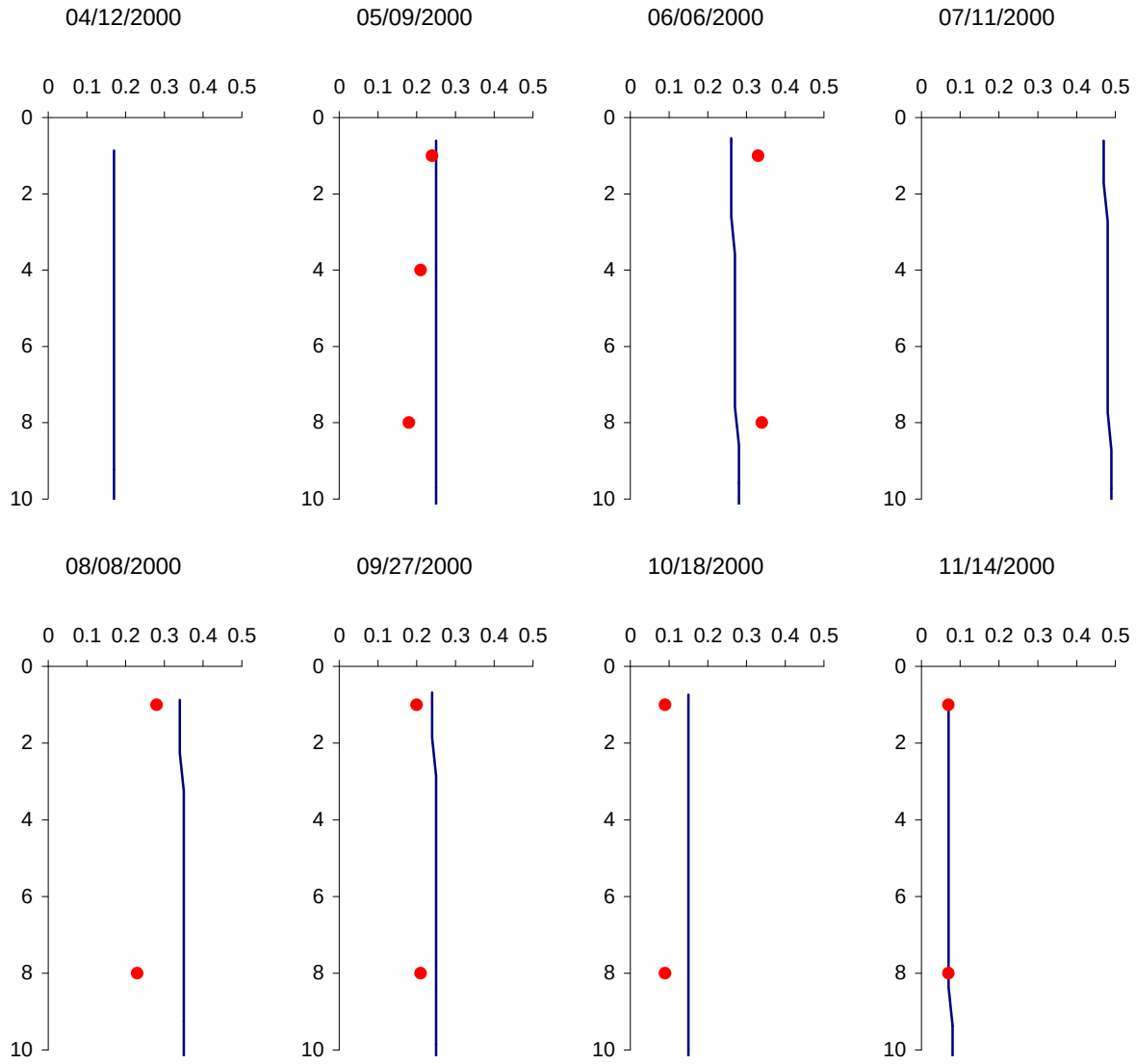


Figure G-3. PO_4 profile calibration results at J.C. Boyle Reservoir at deepest point - 2000 [(X-axis PO_4 (mg/L) vs. Y-axis Depth (m)].

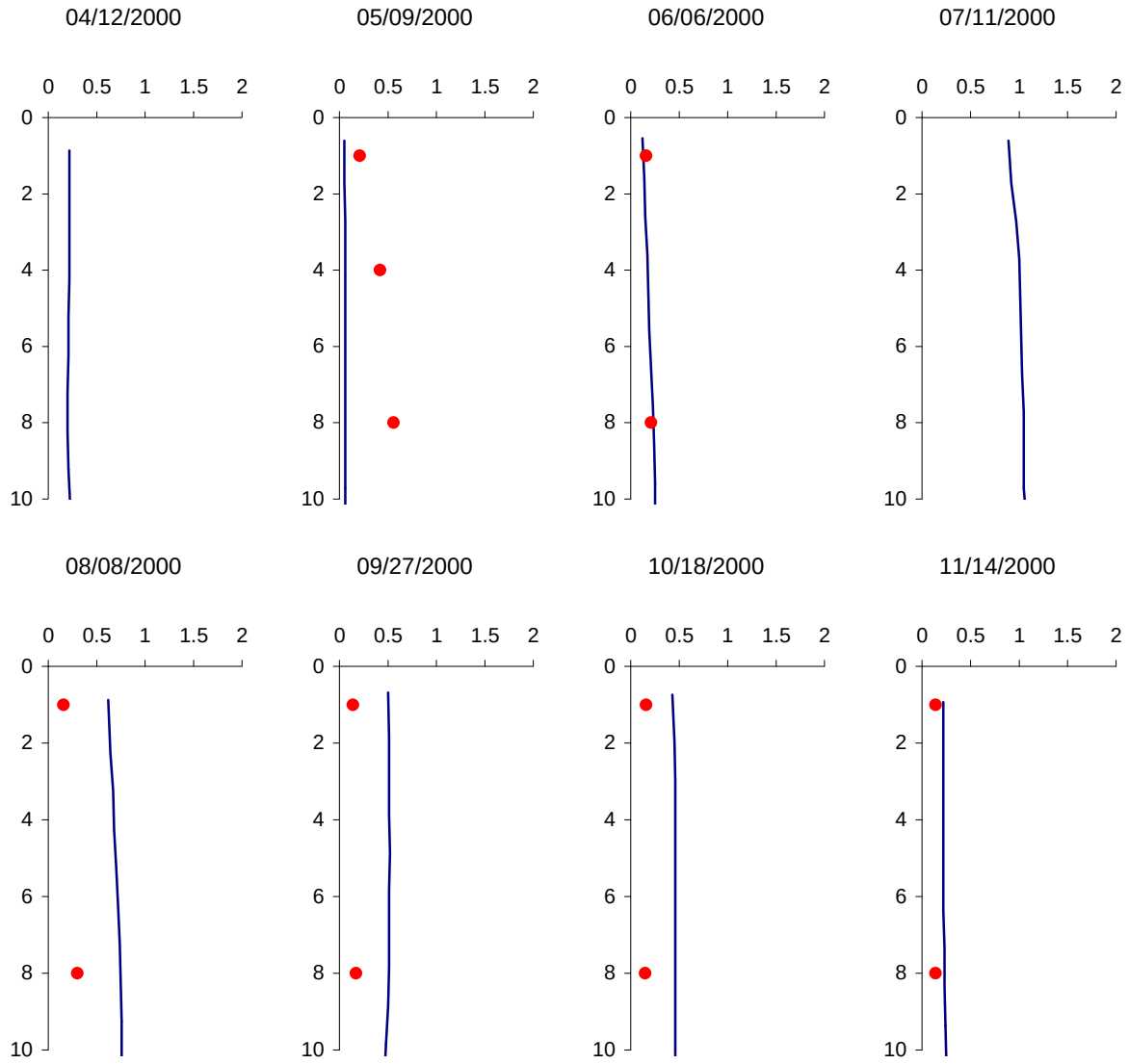


Figure G-4. NH_4 profile calibration results at J.C. Boyle Reservoir at deepest point - 2000 [(X-axis NH_4 (mg/L) vs. Y-axis Depth (m)].

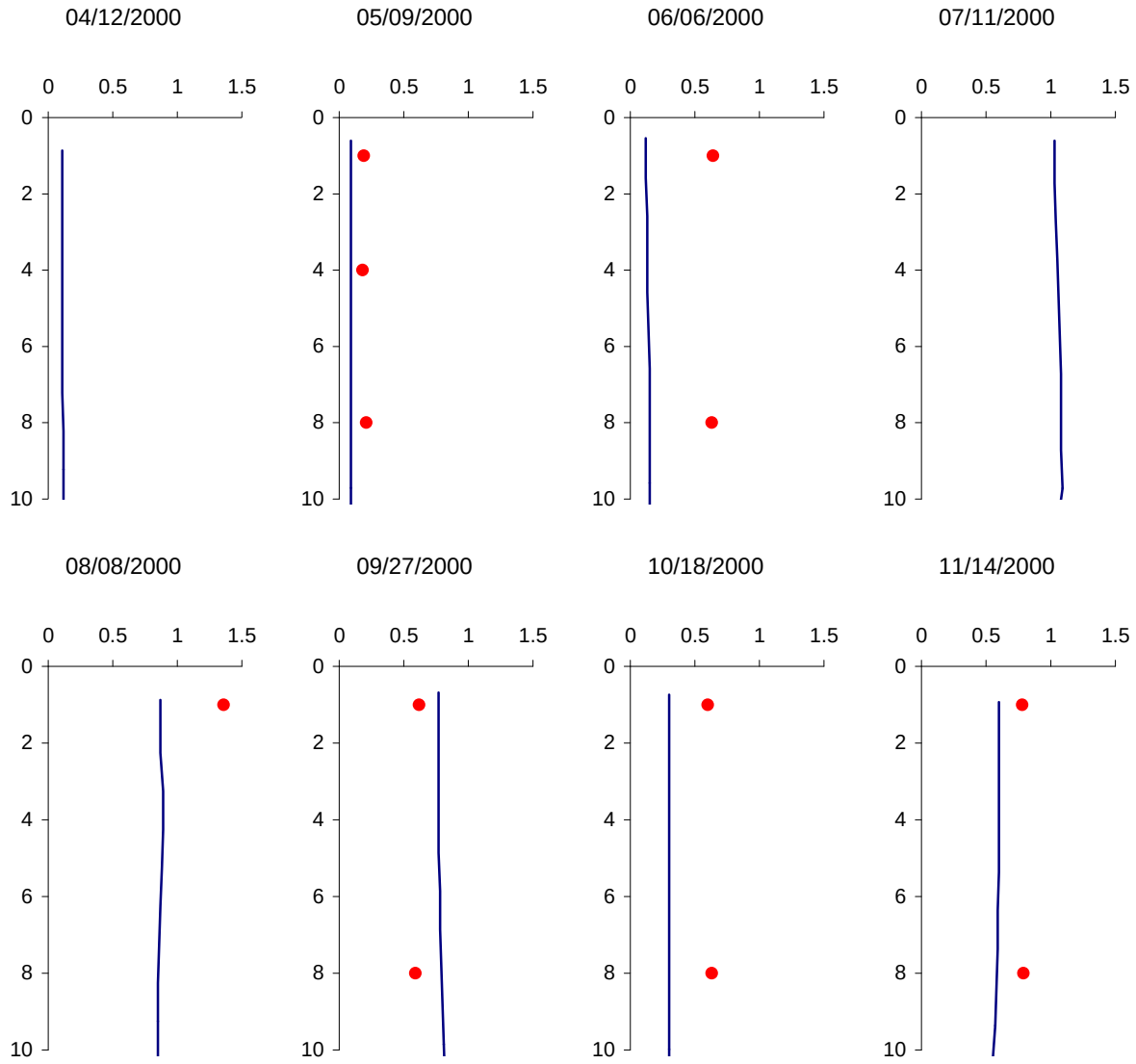


Figure G-5. NO_3 profile calibration results at J.C. Boyle Reservoir at deepest point - 2000 [(X-axis NO_3 (mg/L) vs. Y-axis Depth (m)].

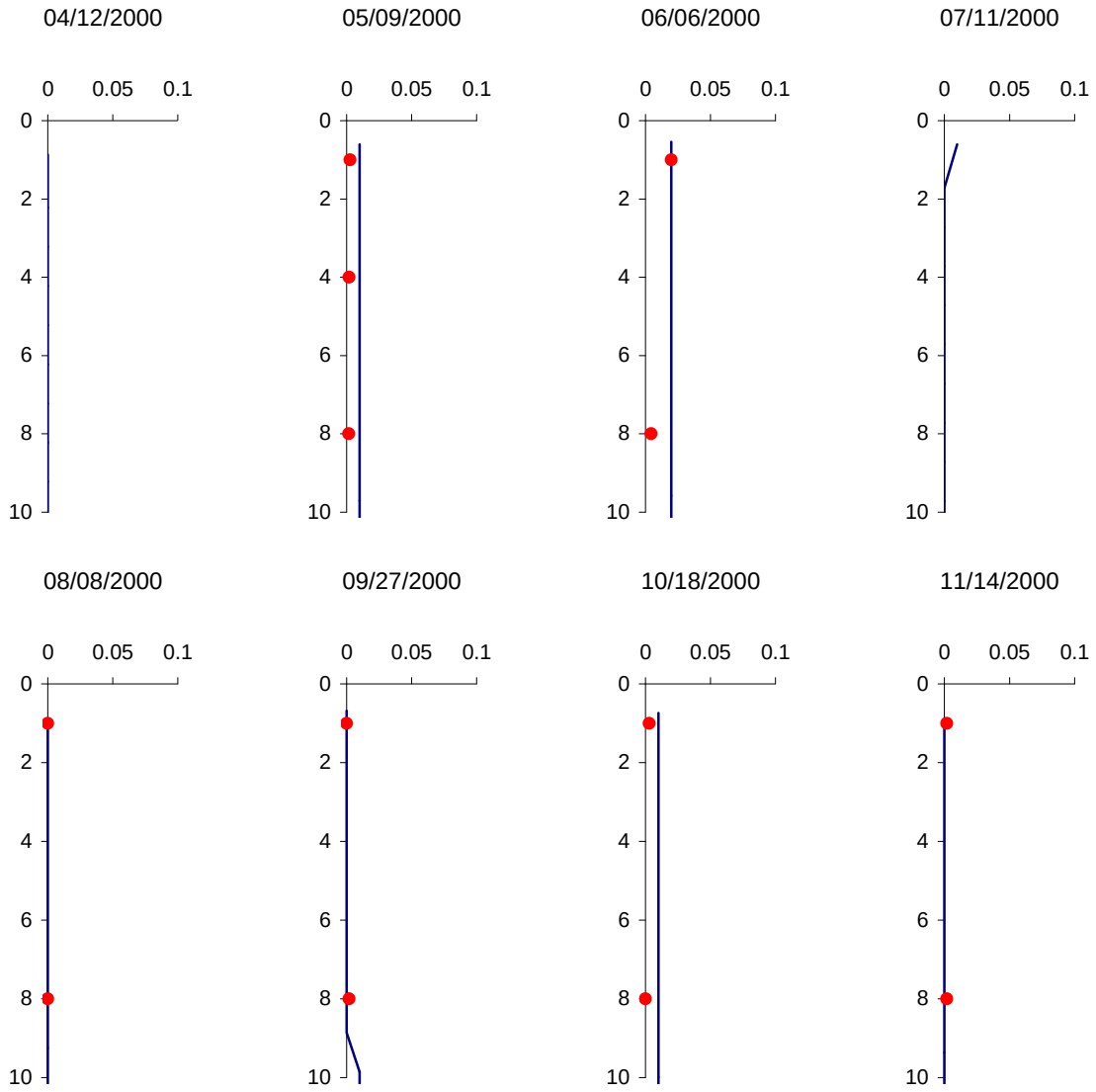


Figure G-6. Chlorophyll-a profile calibration results at J.C. Boyle Reservoir at deepest point - 2000 [(X-axis Chlorophyll-a (mg/L) vs. Y-axis Depth (m)].

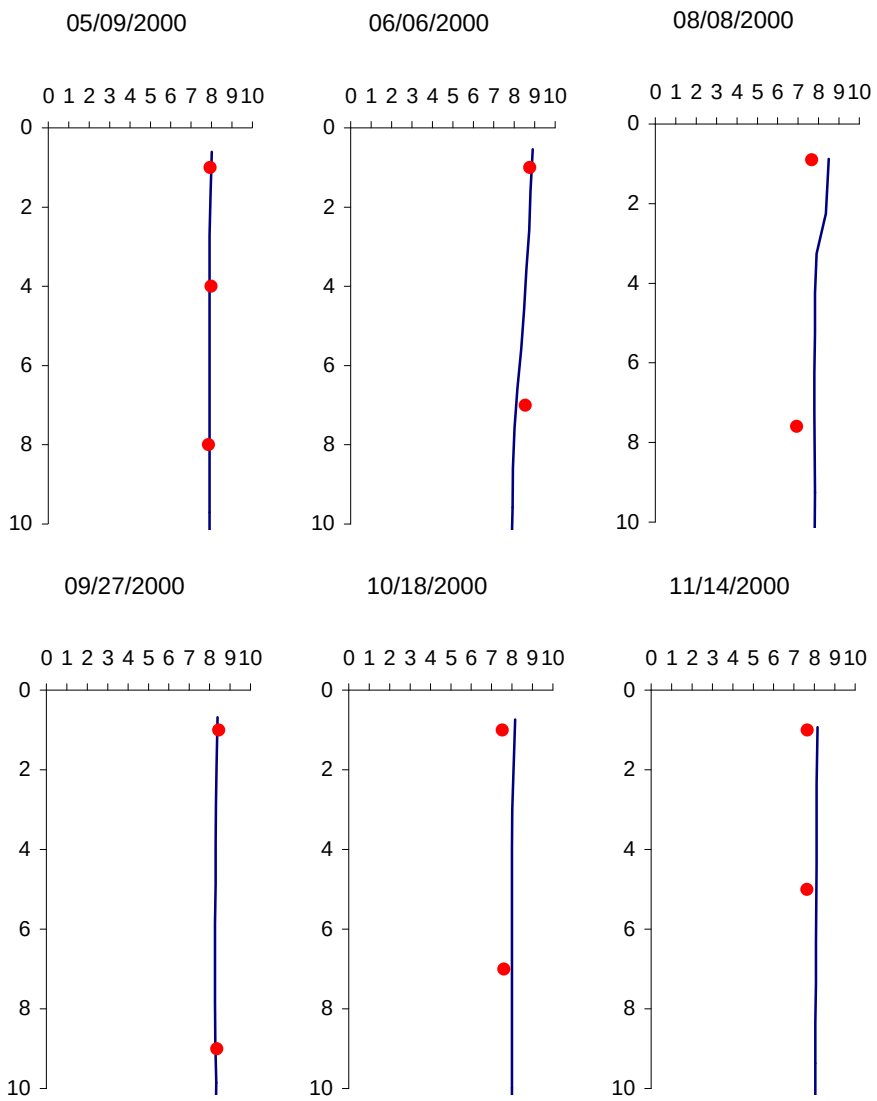


Figure G-7. pH profile calibration results at J.C. Boyle Reservoir at deepest point - 2000 [(X-axis pH vs. Y-axis Depth (m)].

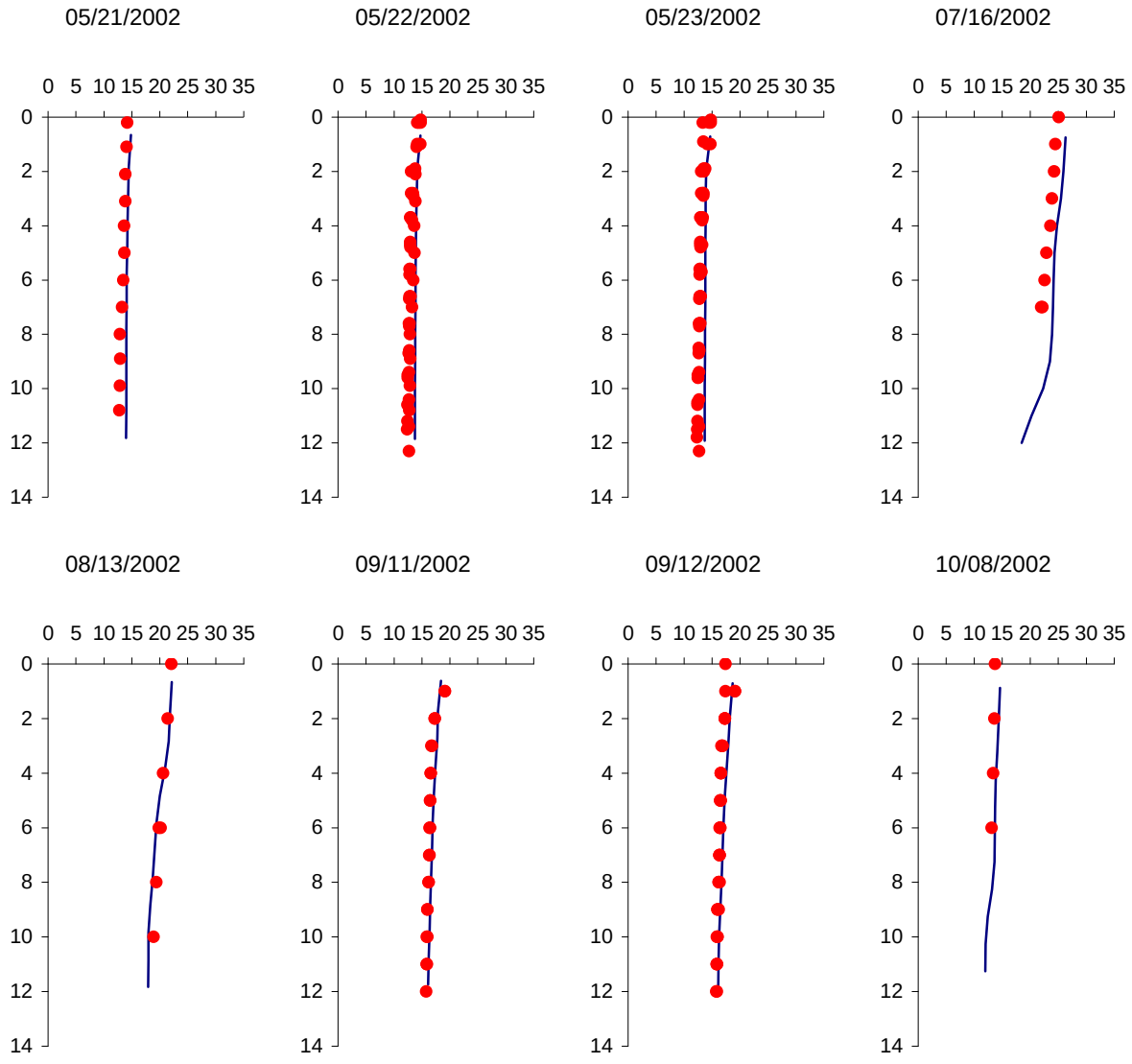


Figure G-8. Temperature profile calibration results at J.C. Boyle Reservoir at deepest point - 2002 [(X-axis Temperature (degC) vs. Y-axis Depth (m)].

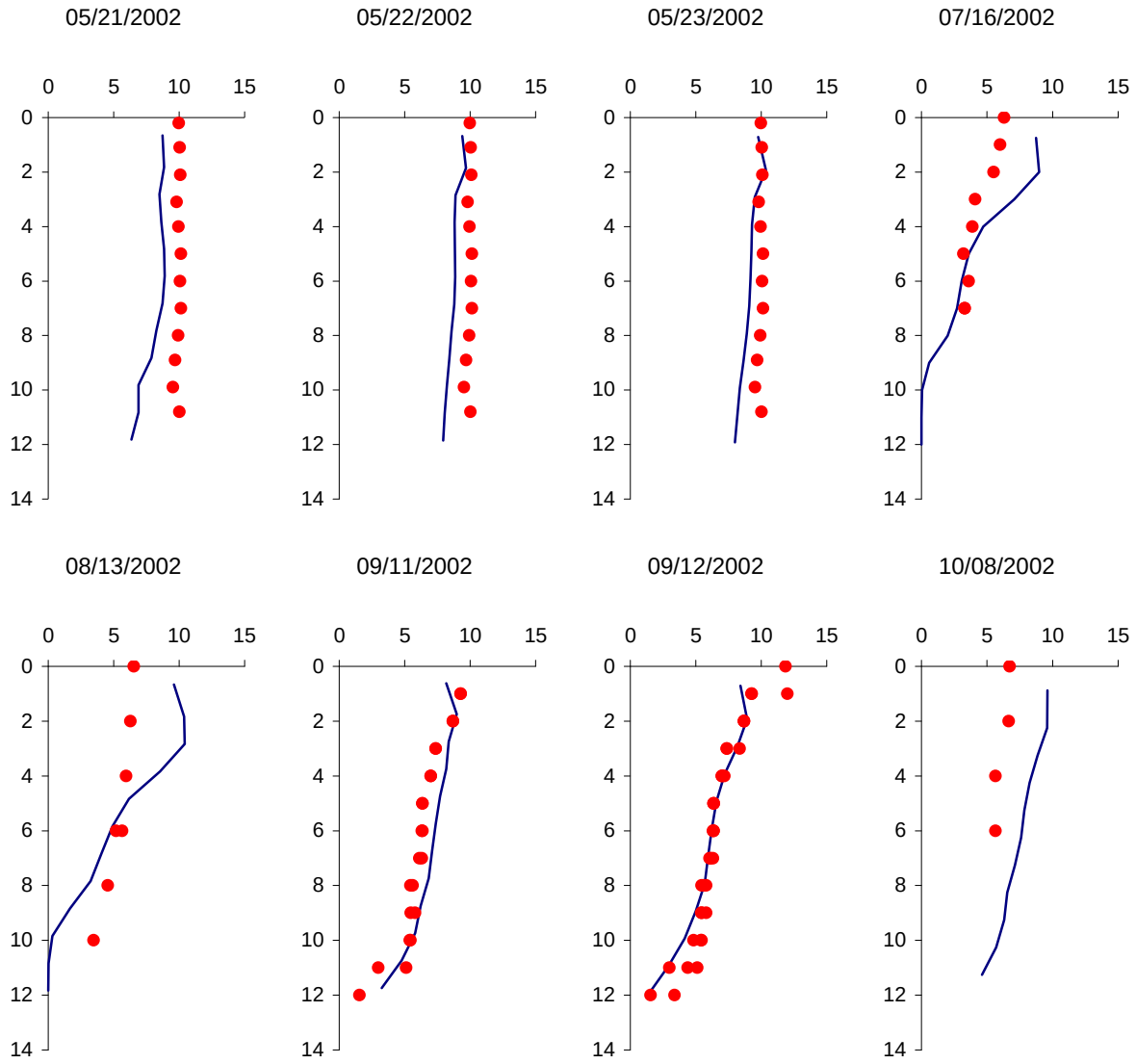


Figure G-9. Dissolved oxygen profile calibration results at J.C. Boyle Reservoir at deepest point - 2002 [(X-axis Dissolved oxygen (mg/L) vs. Y-axis Depth (m)].

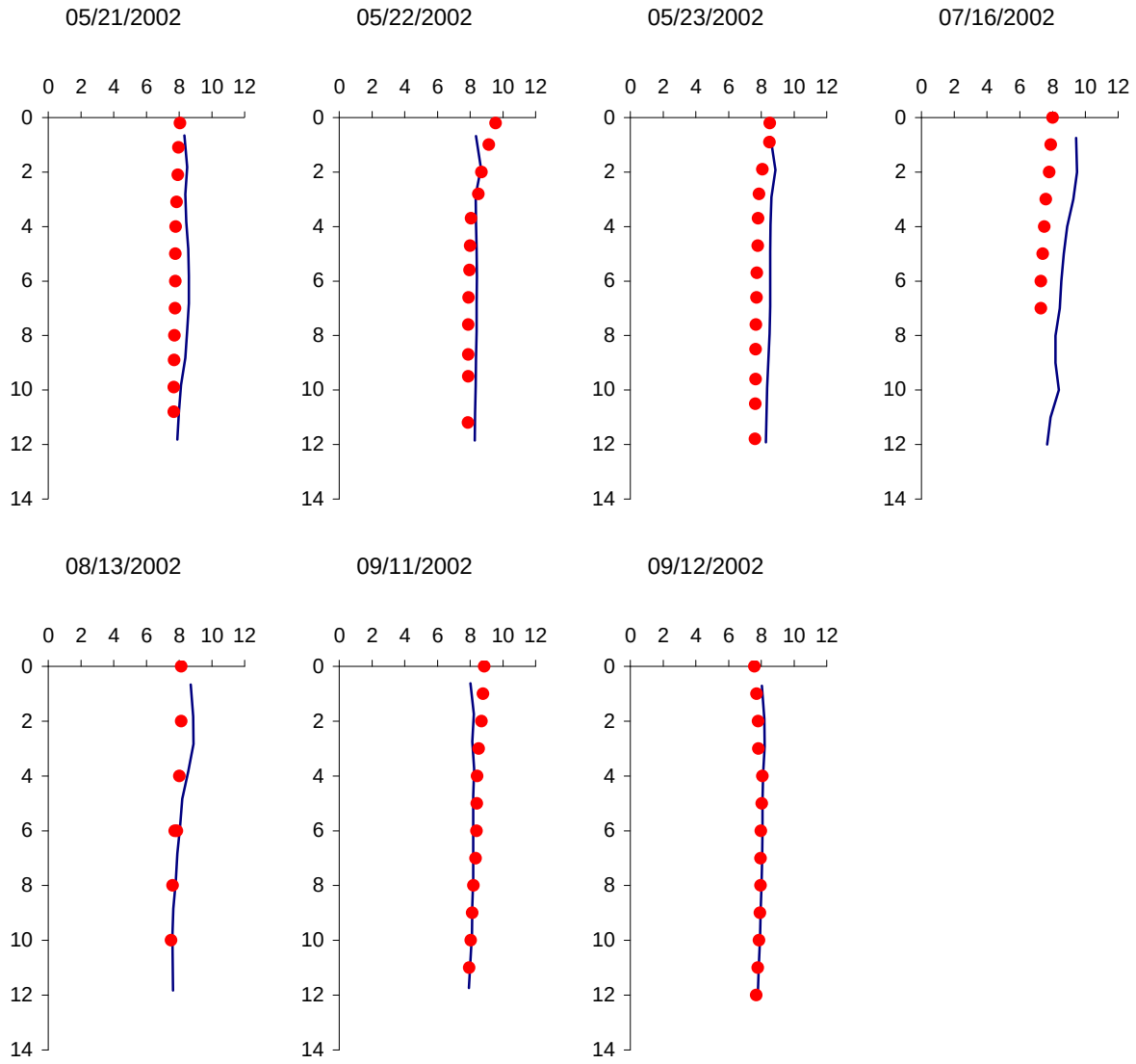


Figure G-10. pH profile calibration results at J.C. Boyle Reservoir at deepest point - 2002 [(X-axis pH vs. Y-axis Depth (m)].

Appendix H

Water Quality Calibration Results for Bypass/Full Flow Reach (Modeling Segment 5)

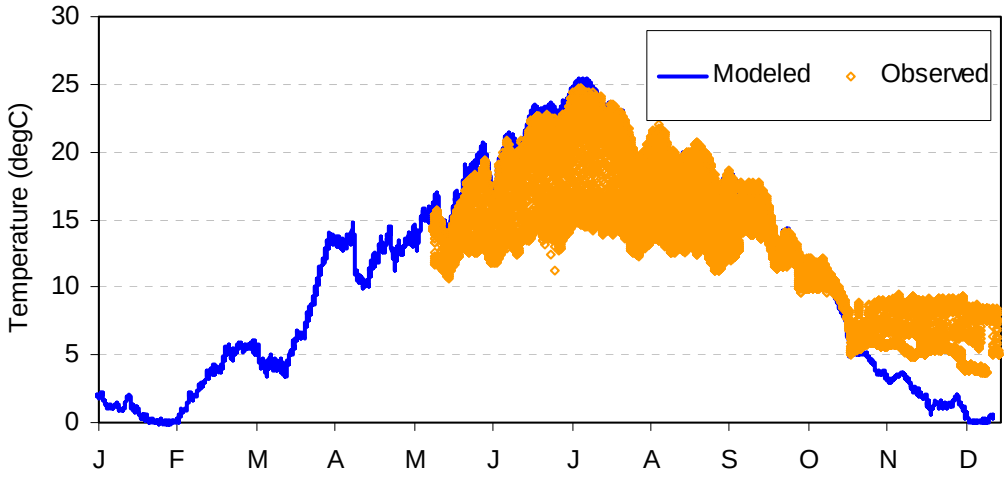


Figure H-1. Temperature calibration results for Klamath River downstream of J. C. Boyle Dam (2002).

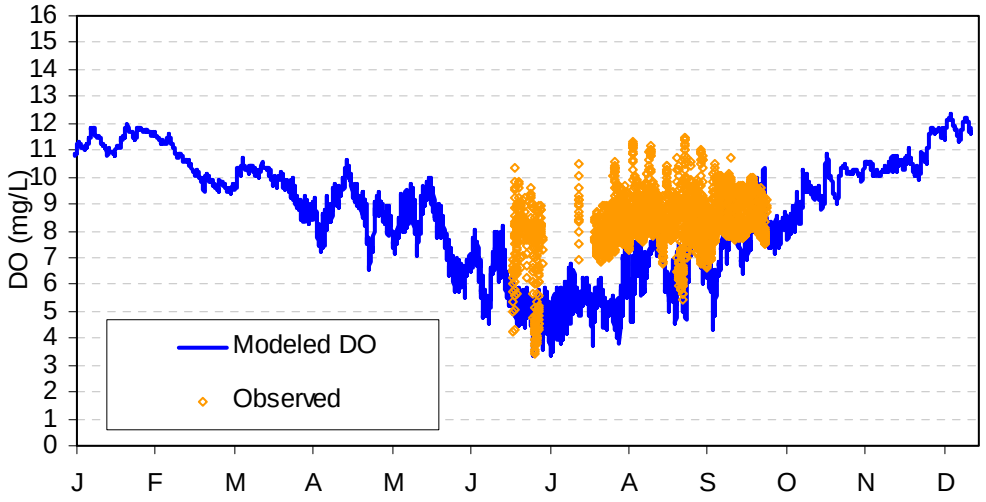


Figure H-2. Dissolved oxygen calibration results for Klamath River downstream of J. C. Boyle Dam (2002).

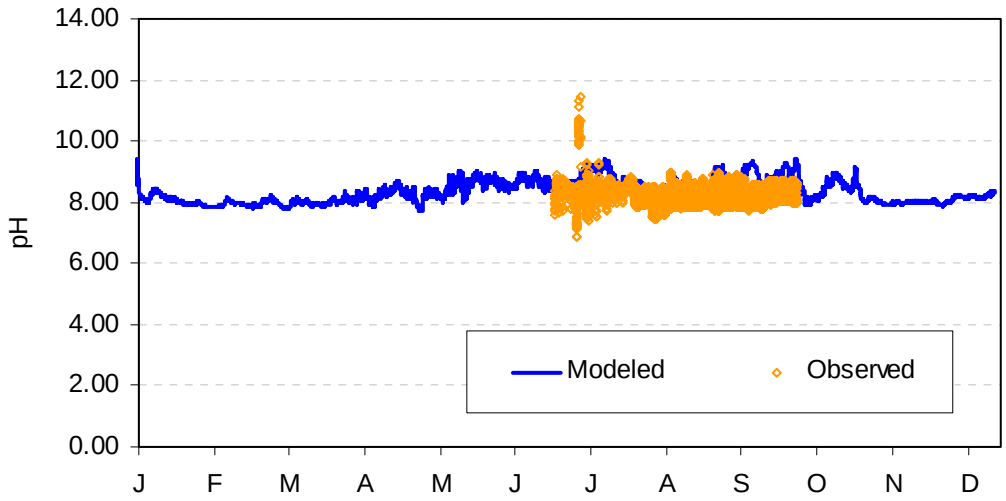


Figure H-3. pH calibration results for Klamath River downstream of J. C. Boyle Dam (2002).

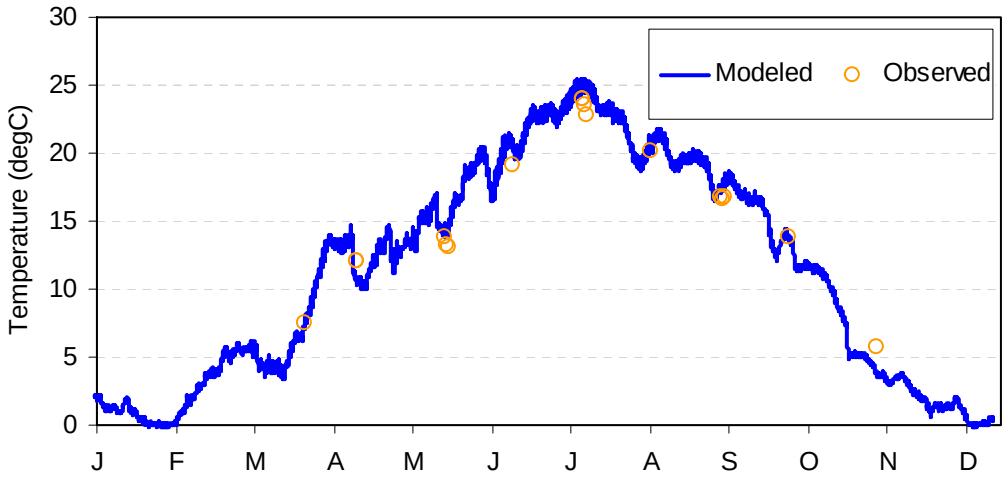


Figure H-4. Temperature calibration results for Klamath River below J. C. Boyle Dam – KR2 (2002).

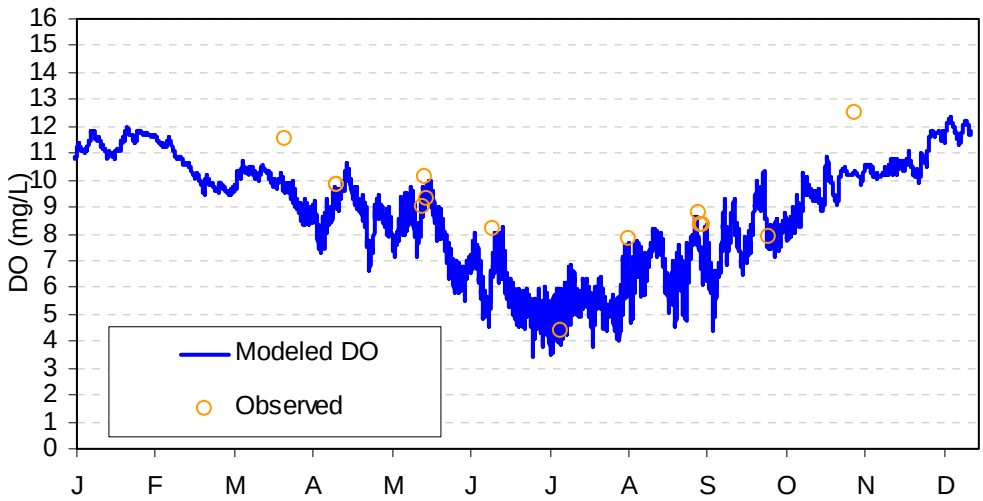


Figure H-5. Dissolved oxygen calibration results for Klamath River below J. C. Boyle Dam – KR2 (2002).

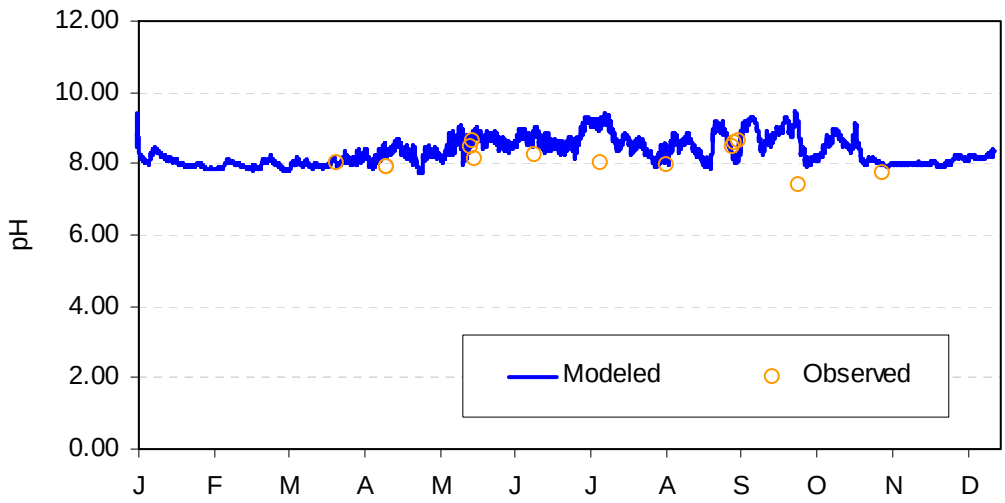


Figure H-6. pH calibration results for Klamath River below J. C. Boyle Dam – KR2 (2002).

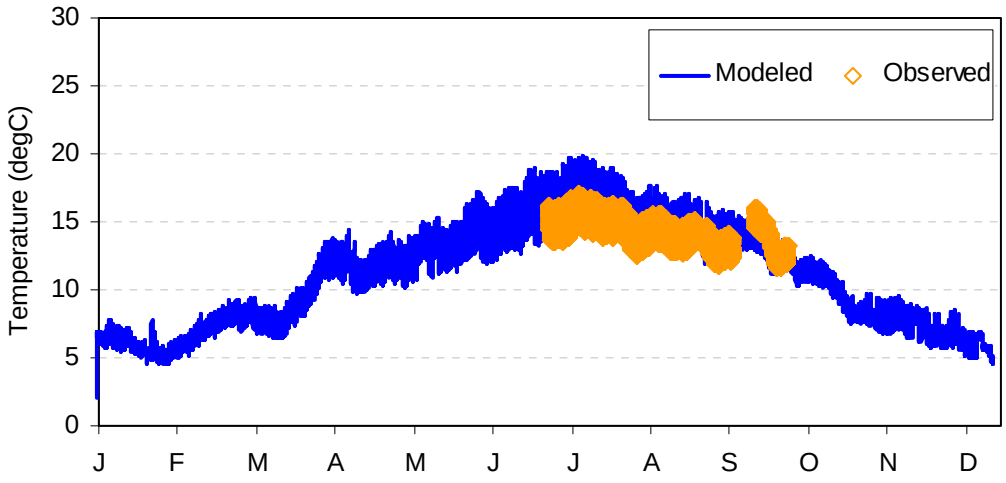


Figure H-7. Temperature calibration results for Klamath River upstream of J. C. Boyle Powerhouse tailrace (2002).

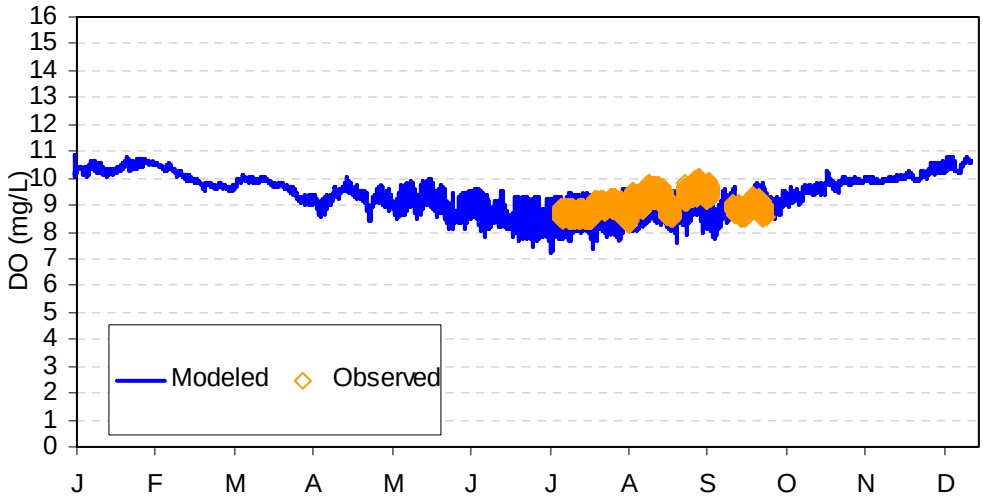


Figure H-8. Dissolved oxygen calibration results for Klamath River upstream of J. C. Boyle Powerhouse tailrace (2002).

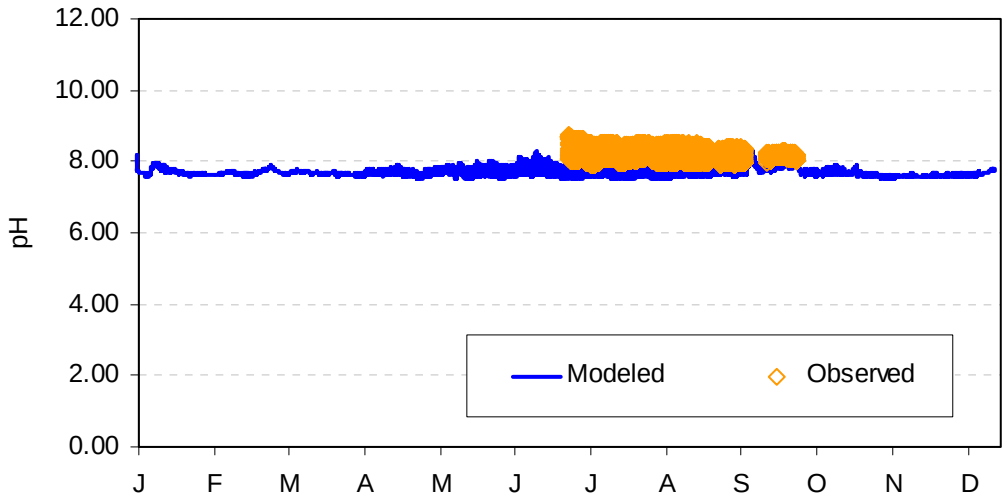


Figure H-9. pH calibration results for Klamath River upstream of J. C. Boyle Powerhouse tailrace (2002).

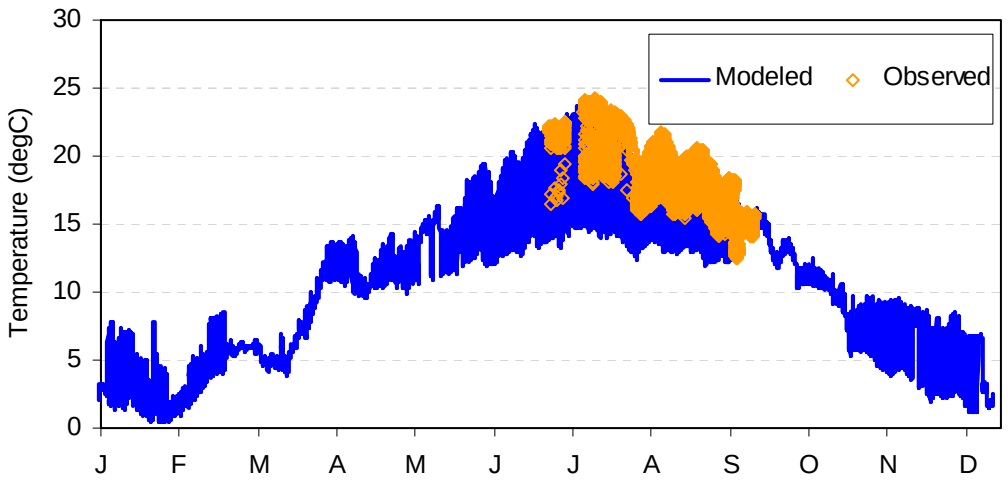


Figure H-10. Temperature calibration results for Klamath River downstream of big bend Powerhouse tailrace (2002).

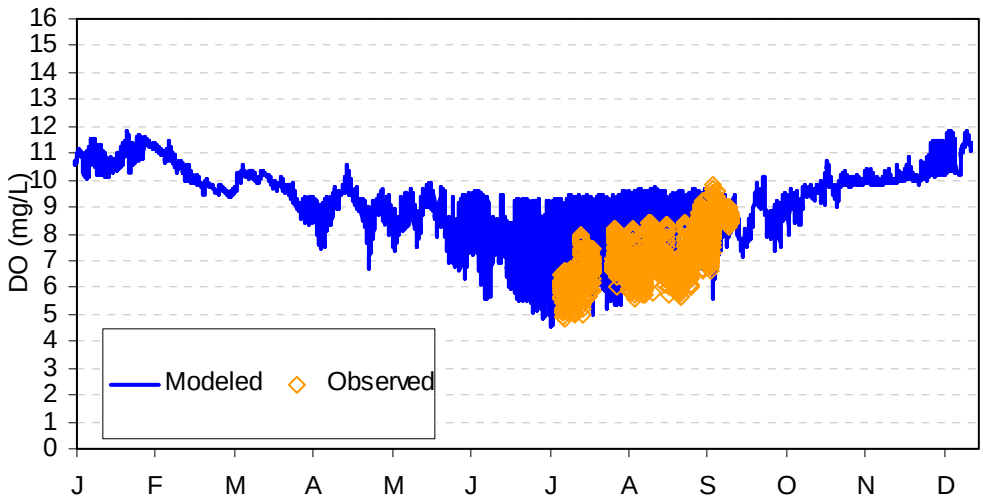


Figure H-11. Dissolved oxygen calibration results for Klamath River downstream of big bend Powerhouse tailrace (2002).

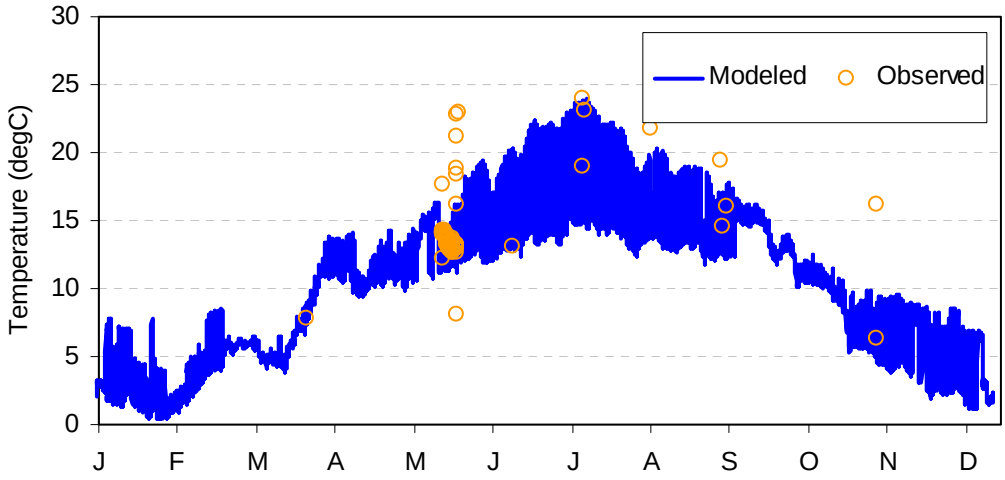


Figure H-12. Temperature calibration results for J. C. Boyle Powerhouse tailrace (2002).

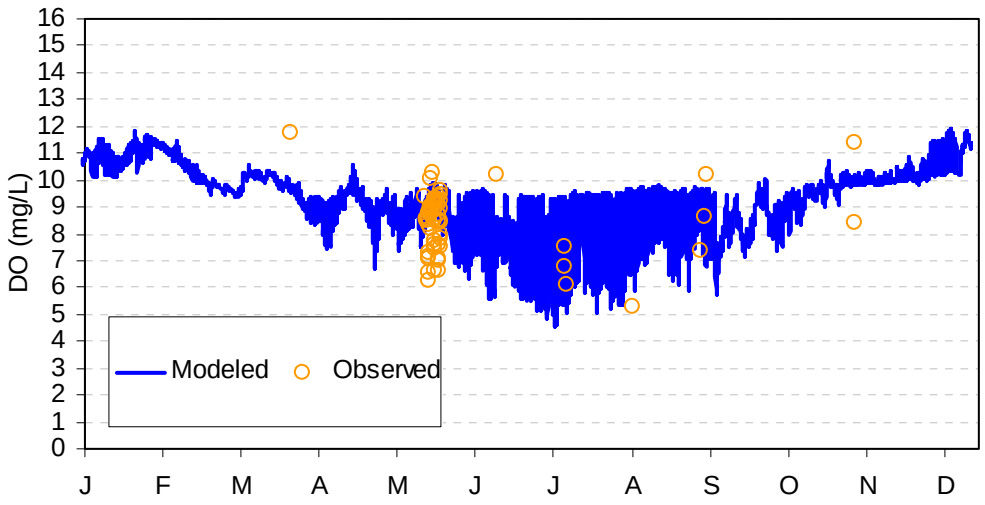


Figure H-13. Dissolved oxygen calibration results for J. C. Boyle Powerhouse tailrace (2002).

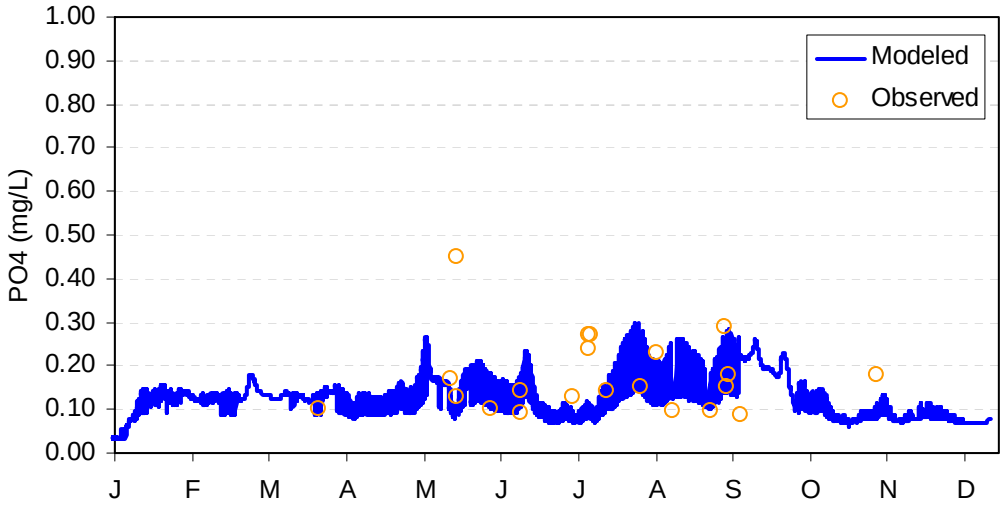


Figure H-14. PO_4 calibration results for J. C. Boyle Powerhouse tailrace (2002).

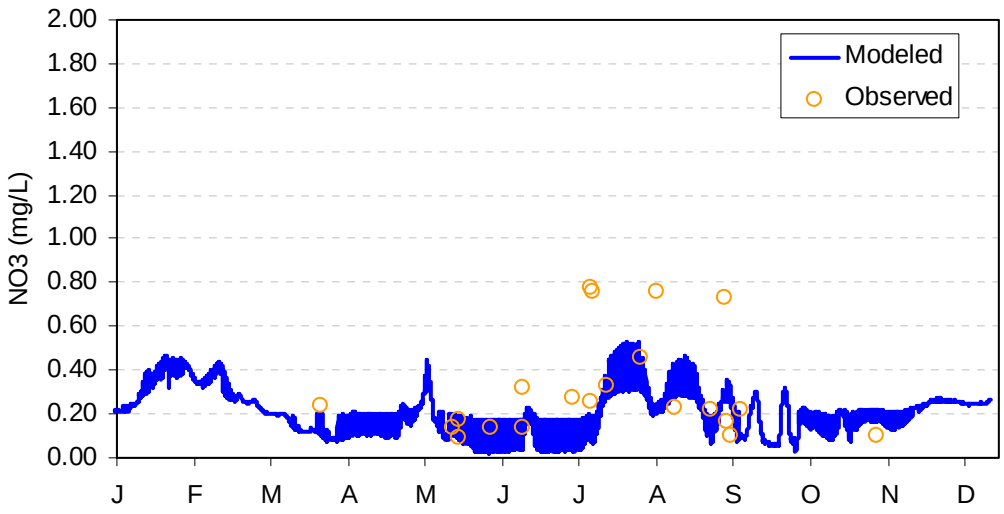


Figure H-15. NO_3 calibration results for J. C. Boyle Powerhouse tailrace (2002).

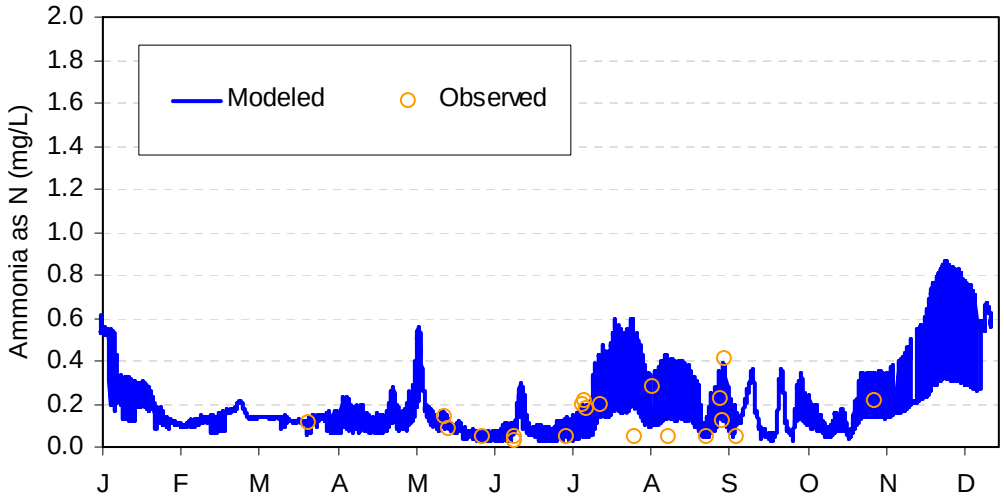


Figure H-16. NH_3-N calibration results for J. C. Boyle Powerhouse tailrace (2002).

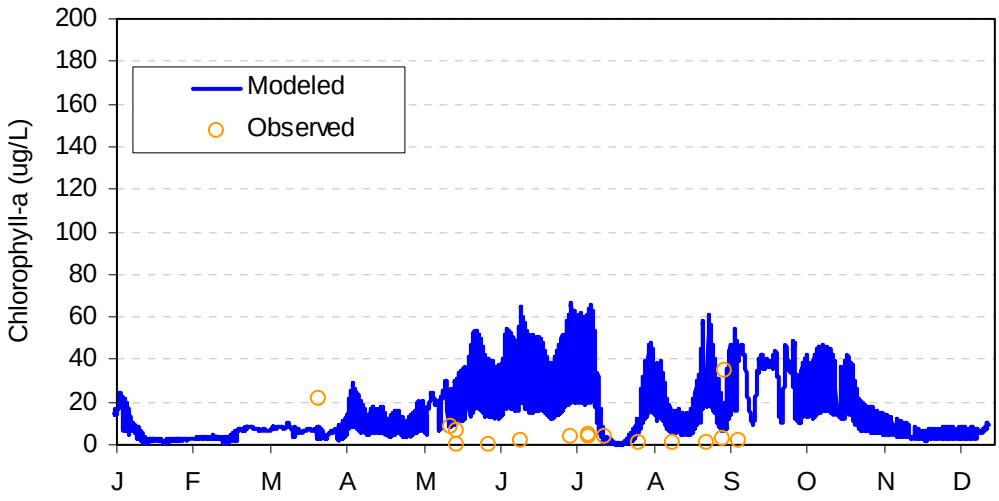


Figure H-17. Chlorophyll *a* calibration results for J. C. Boyle Powerhouse tailrace (2002).

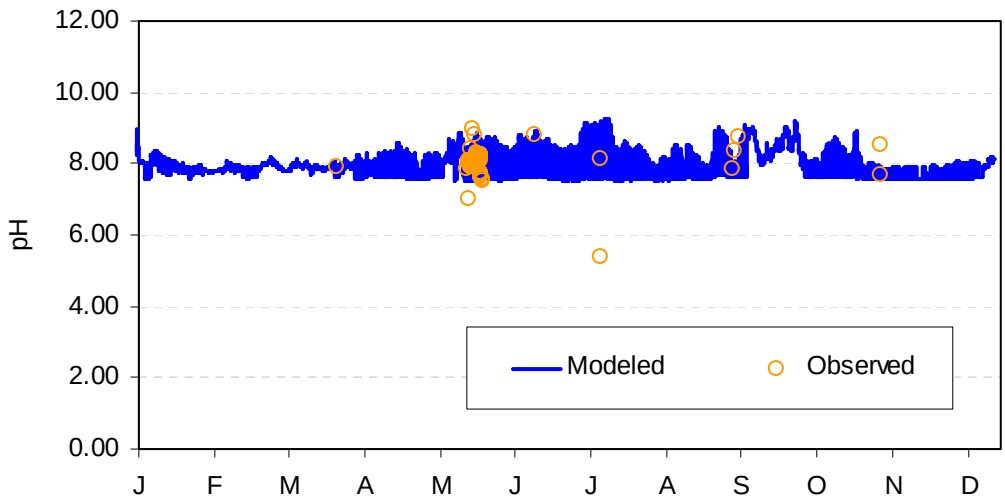


Figure H-18. pH calibration results for J. C. Boyle Powerhouse tailrace (2002).

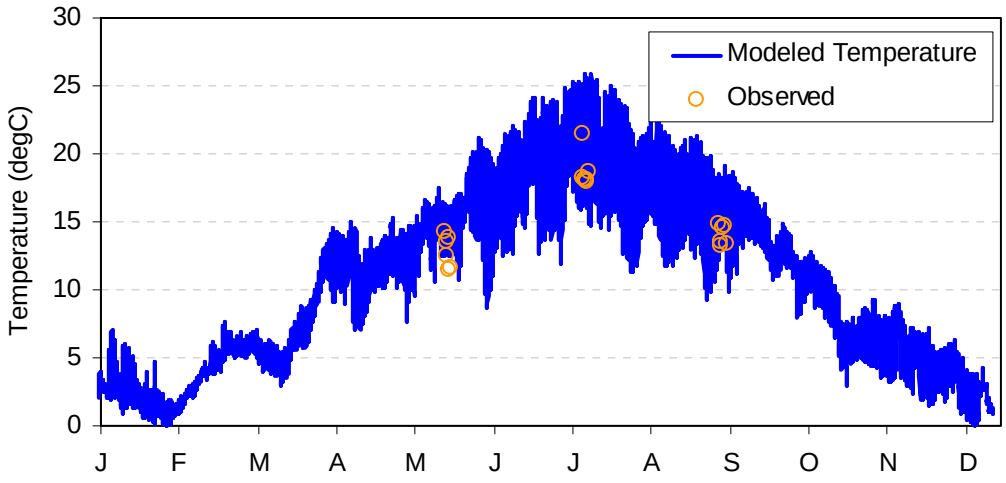


Figure H-19. Temperature calibration results for Klamath River near Stateline—KR4 (2002).

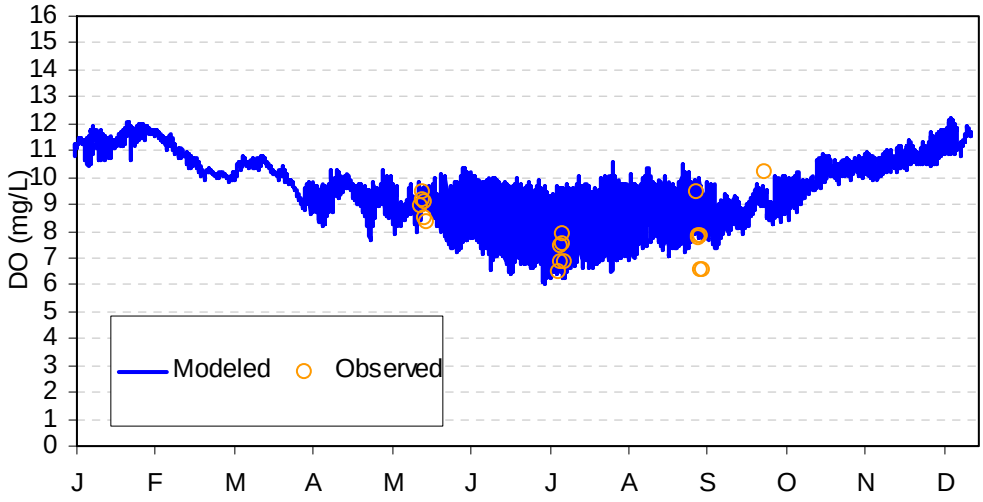


Figure H-20. Dissolved oxygen calibration results for Klamath River near Stateline—KR4 (2002).

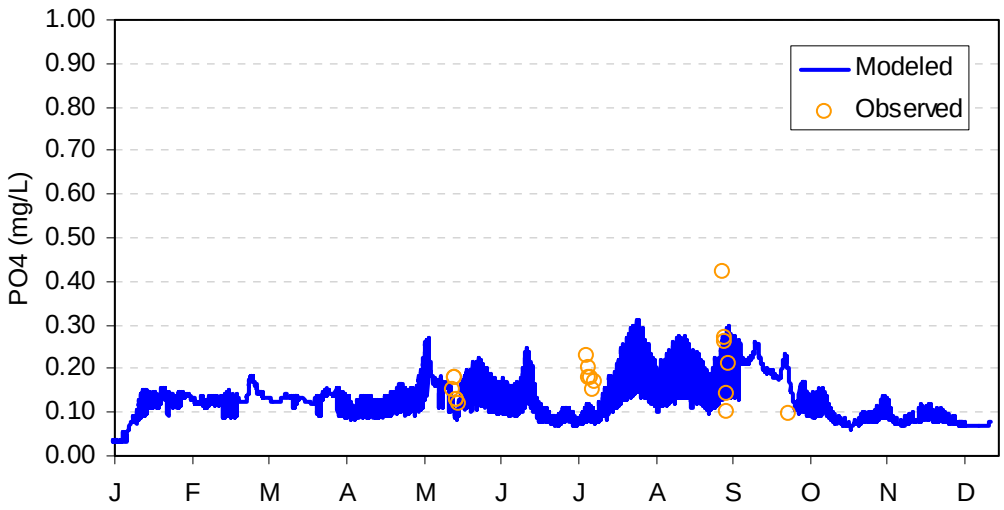


Figure H-21. PO_4 calibration results for Klamath River near Stateline—KR4 (2002).

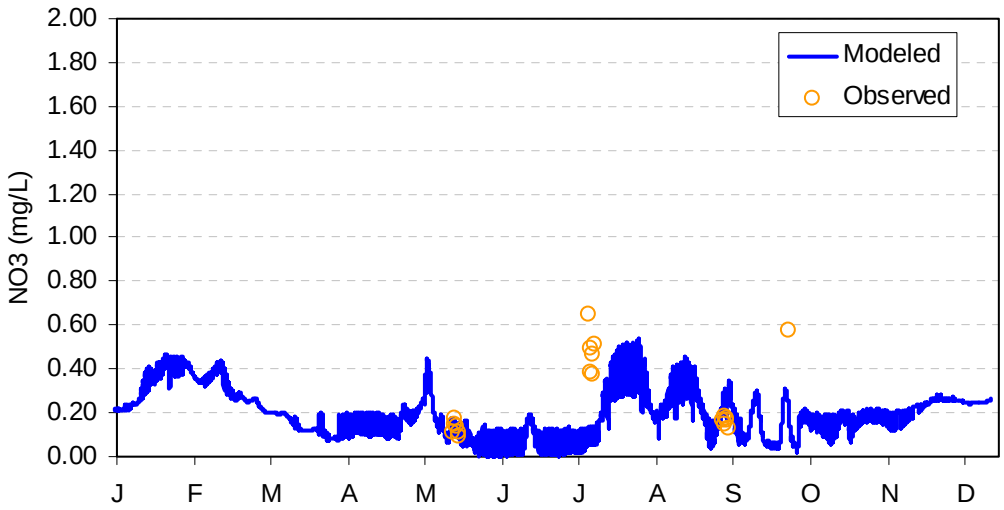


Figure H-22 NO_3 calibration results for Klamath River near Stateline—KR4 (2002).

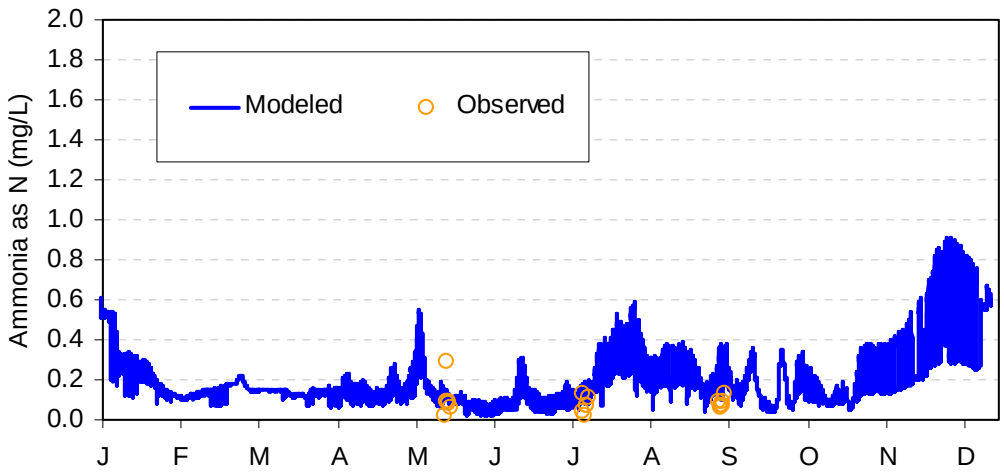


Figure H-23 NH_3-N calibration results for Klamath River near Stateline-KR4 (2002).

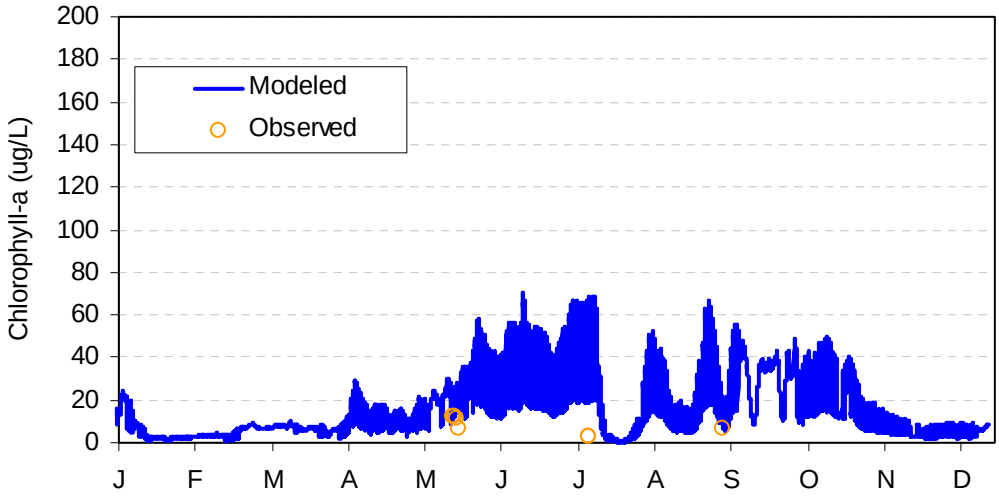


Figure H-24 Chlorophyll *a* calibration results for Klamath River near Stateline—KR4 (2002).

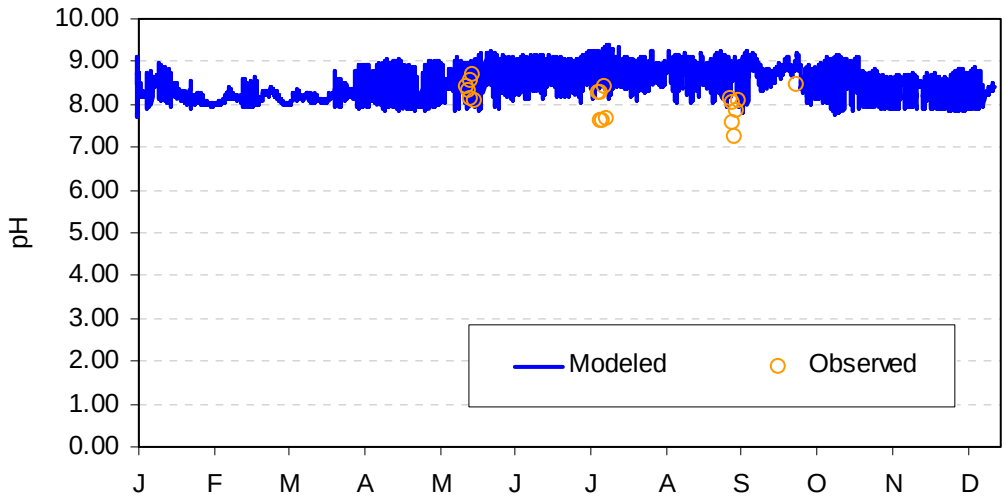


Figure H-25 pH calibration results for Klamath River near Stateline—KR4 (2002).

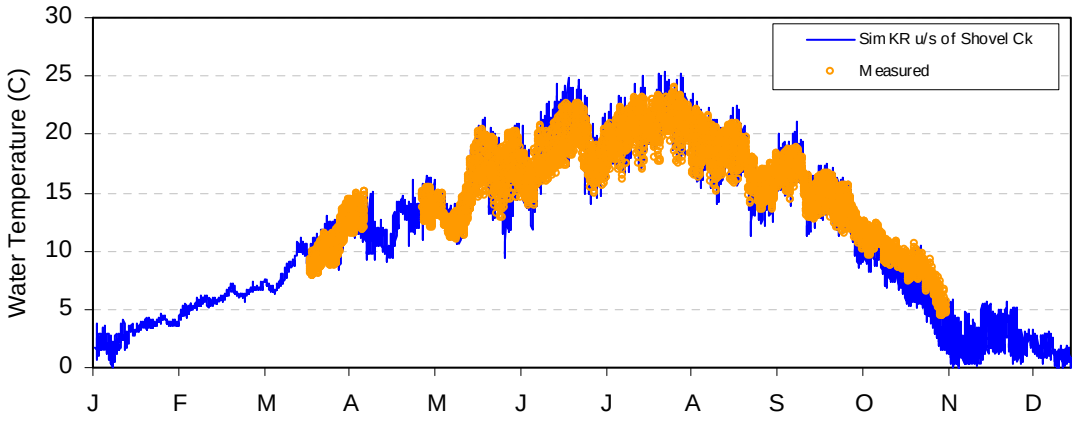


Figure H-26. Temperature calibration results for the Klamath River at Shovel Creek (2000).

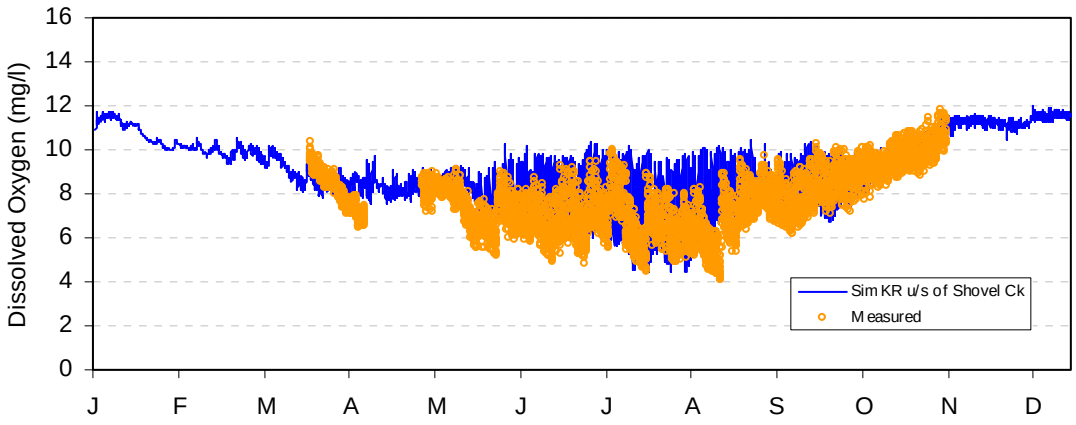


Figure H-27. Dissolved oxygen calibration results for the Klamath River at Shovel Creek (2000).

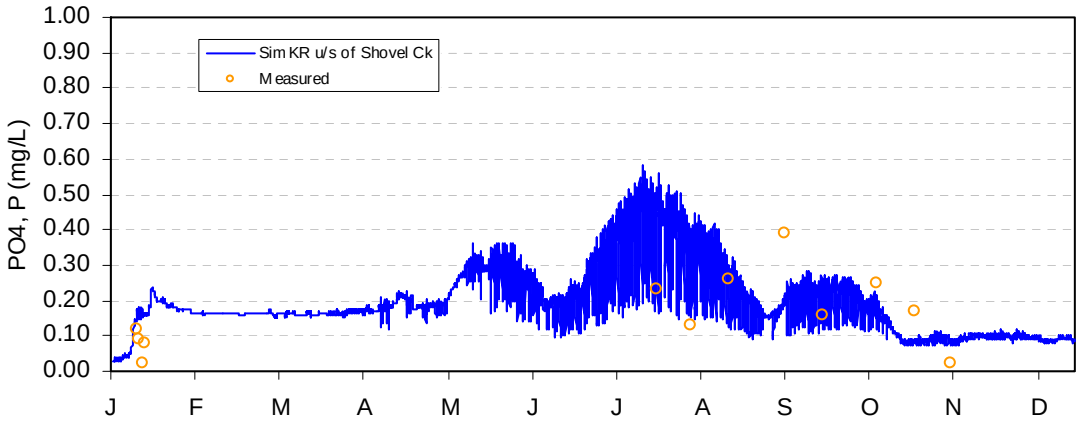


Figure H-28. PO_4-P calibration results for the Klamath River at Shovel Creek (2000).

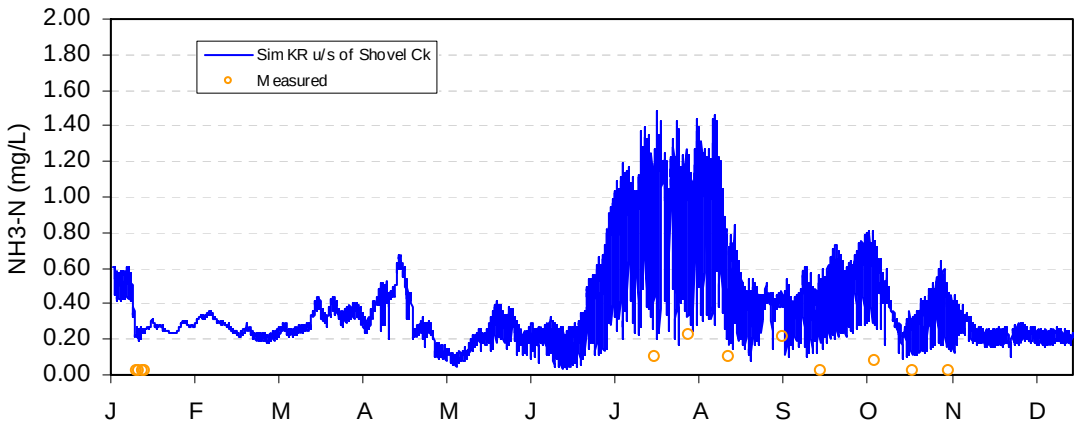


Figure H-29. NH_3-N calibration results for the Klamath River at Shovel Creek (2000).

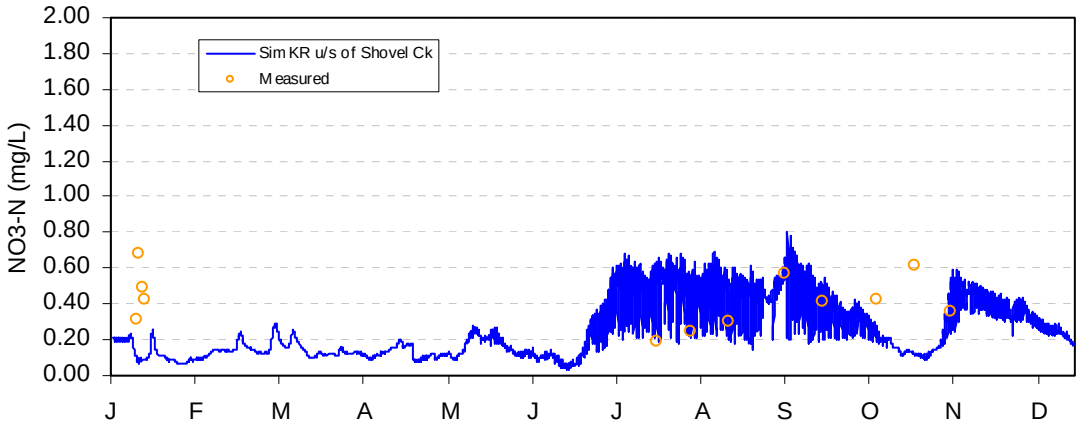


Figure H-30. NO_3-N calibration results for the Klamath River at Shovel Creek (2000).

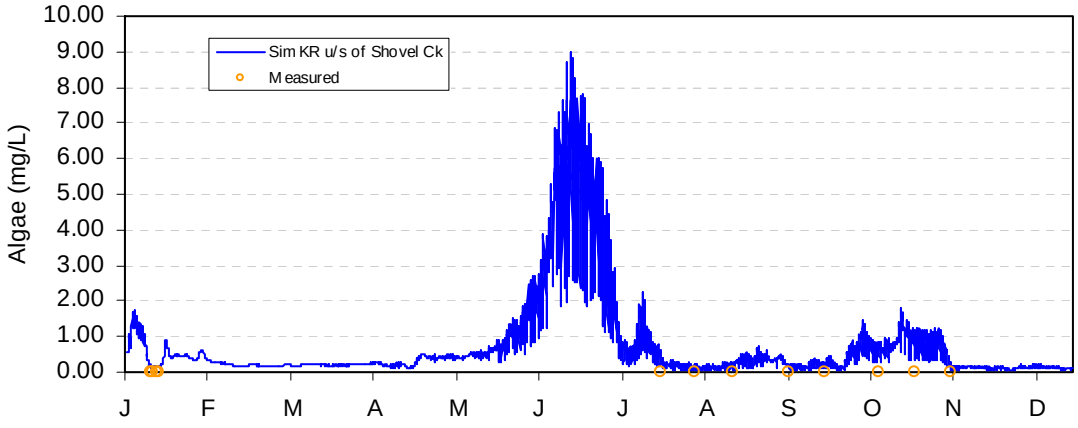


Figure H-31. Algae calibration results for the Klamath River at Shovel Creek (2000).

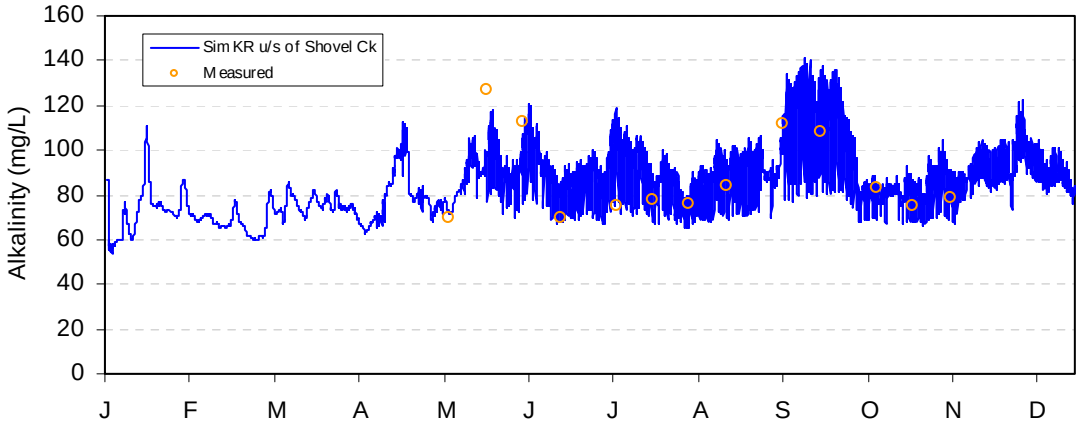


Figure H-32. Alkalinity calibration results for the Klamath River at Shovel Creek (2000).

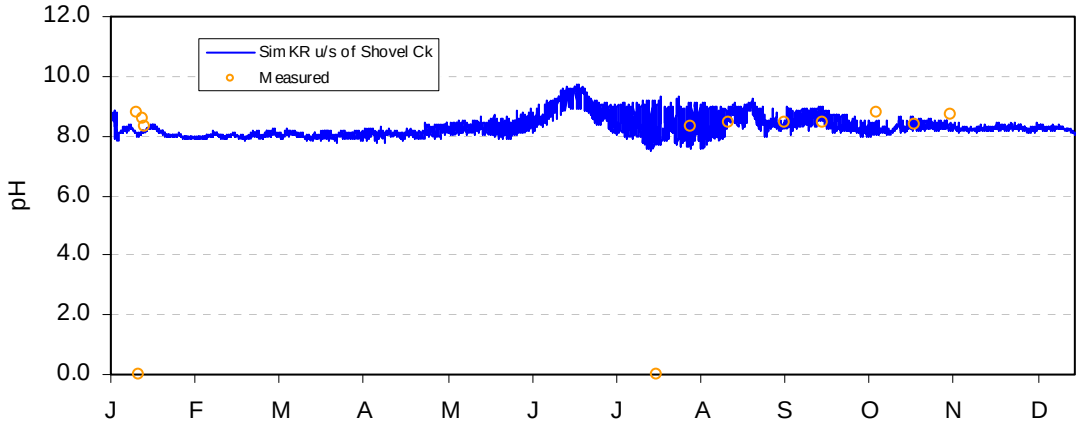


Figure H-33. pH calibration results for the Klamath River at Shovel Creek (2000).

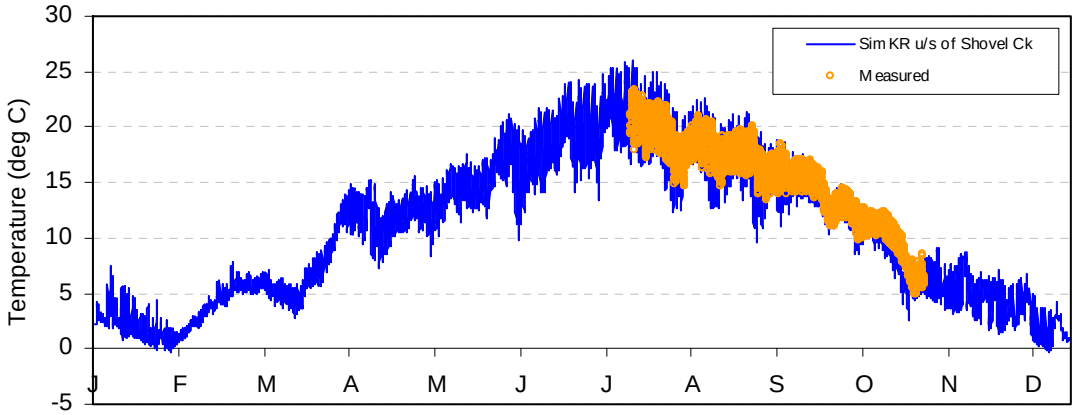


Figure H-34. Temperature calibration results for the Klamath River at Shovel Creek (2002).

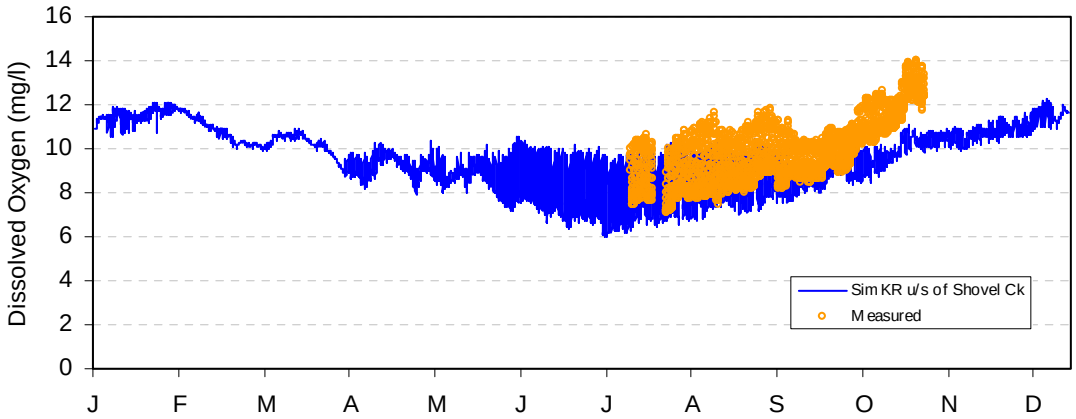


Figure H-35. Dissolved oxygen calibration results for the Klamath River at Shovel Creek (2002).

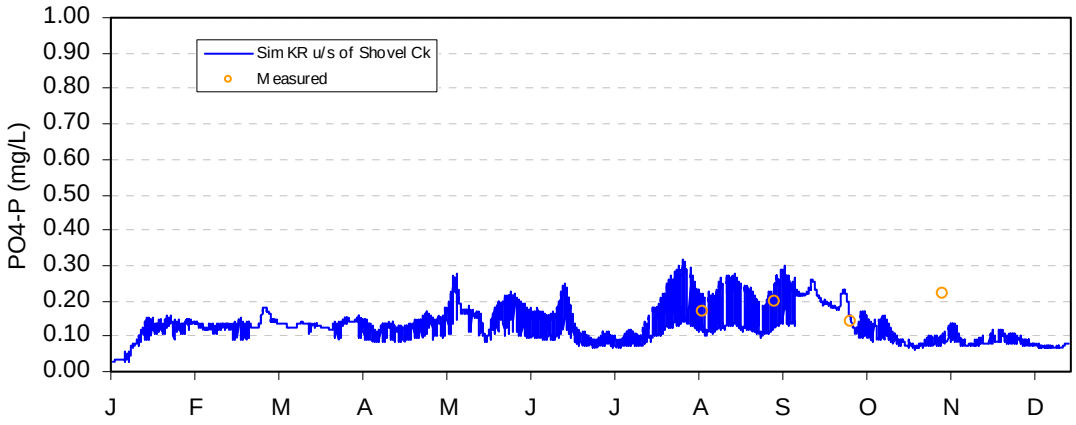


Figure H-36. PO_4-P calibration results for the Klamath River at Shovel Creek (2002).

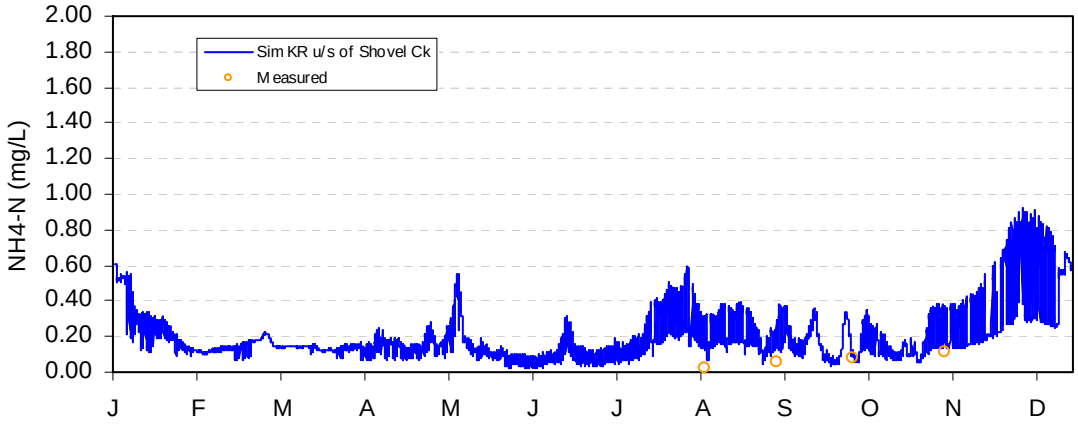


Figure H-37. NH_4-N calibration results for the Klamath River at Shovel Creek (2002).

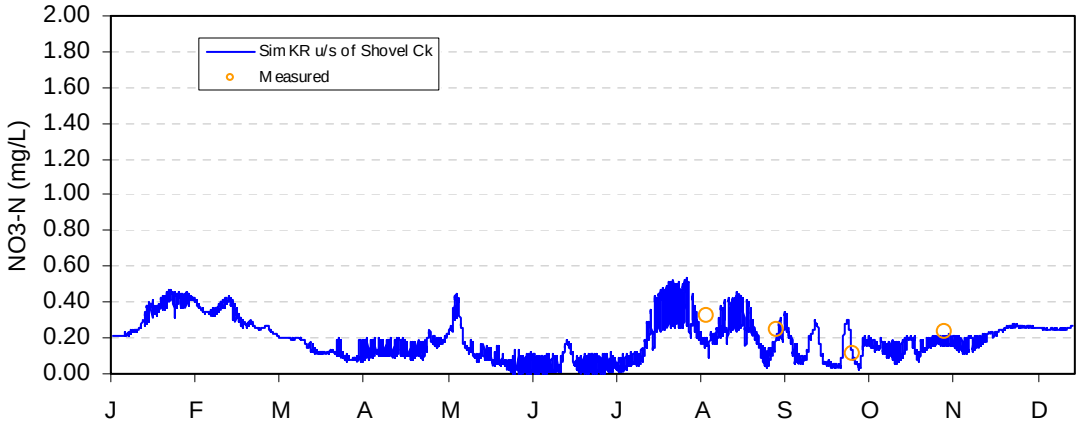


Figure H-38. NO_3-N calibration results for the Klamath River at Shovel Creek (2002).

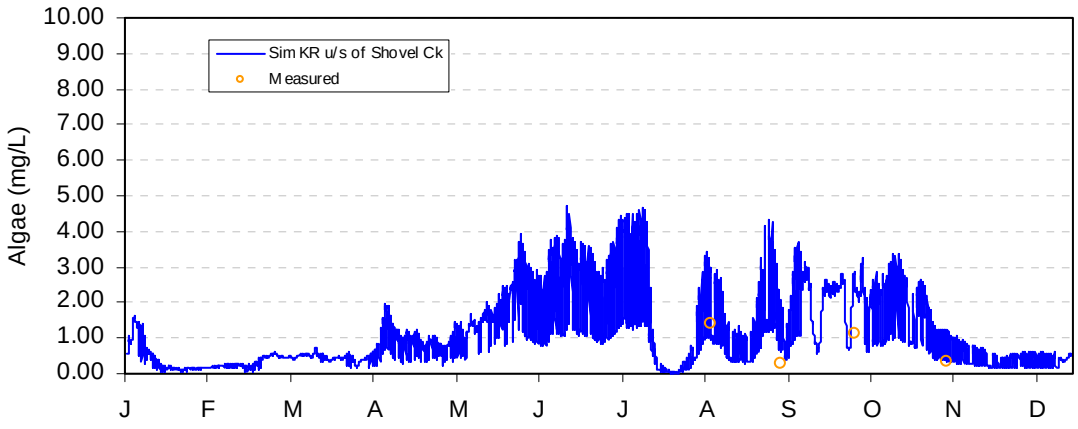


Figure H-39. Algae calibration results for the Klamath River at Shovel Creek (2002).

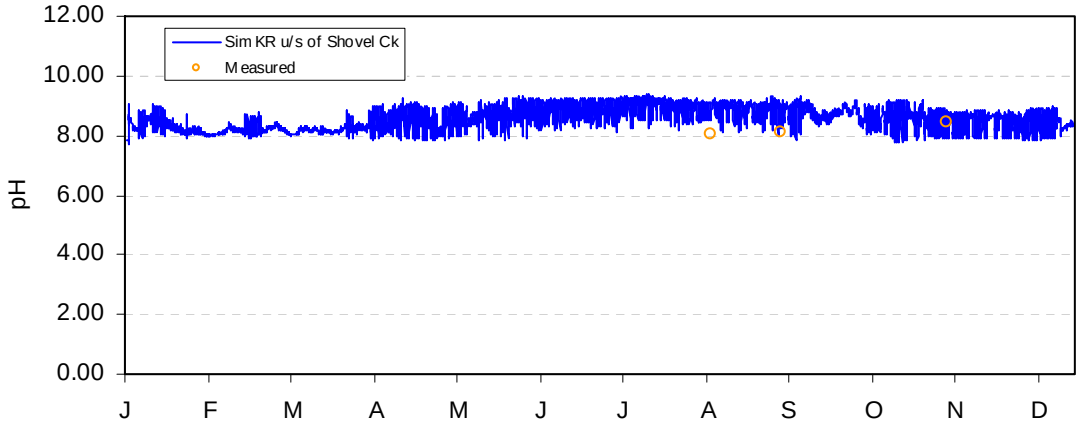


Figure H-40. pH calibration results for the Klamath River at Shovel Creek (2002).

Appendix I

Water Quality Calibration Results for Copco Reservoir (Modeling Segment 6)

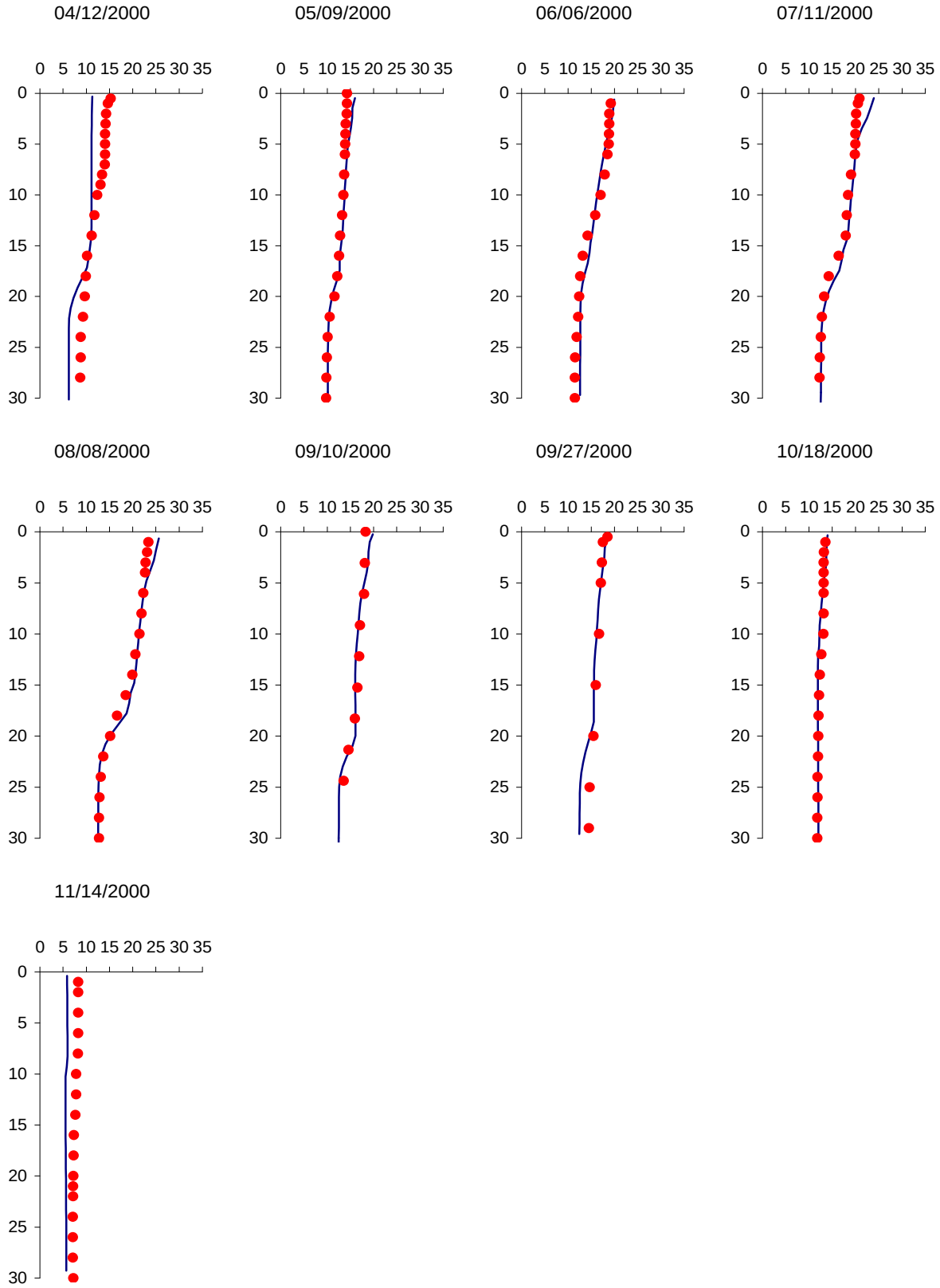


Figure I-1. Temperature profile calibration results for Copco Reservoir at Copco Lake near Copco (2000) [X-axis Temperature (degC) vs. Y-axis Depth (m)].

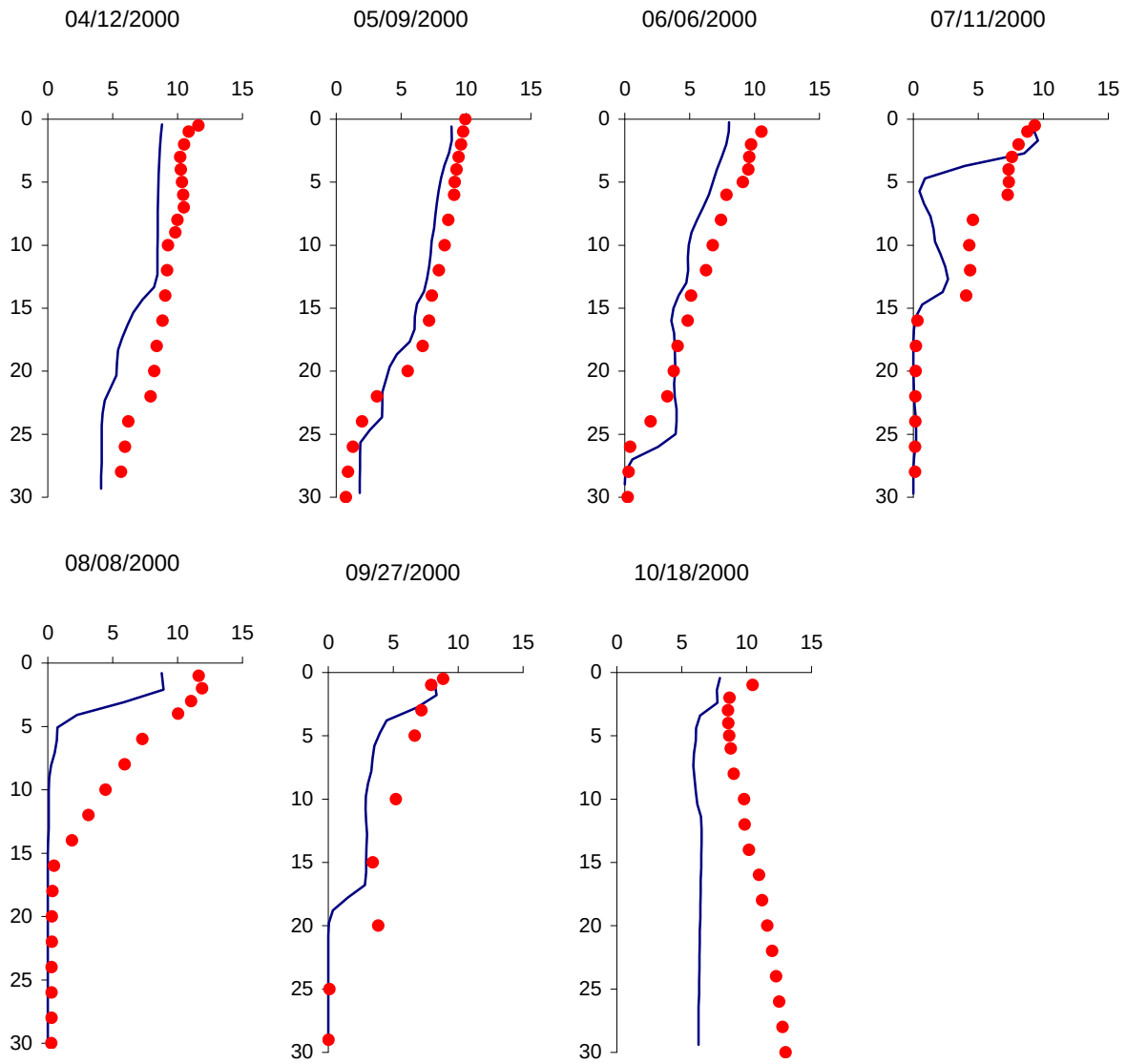


Figure I-2. Dissolved oxygen profile calibration results for Copco Reservoir at Copco Lake near Copco (2000) [X-axis Dissolved oxygen (mg/L) vs. Y-axis Depth (m)].

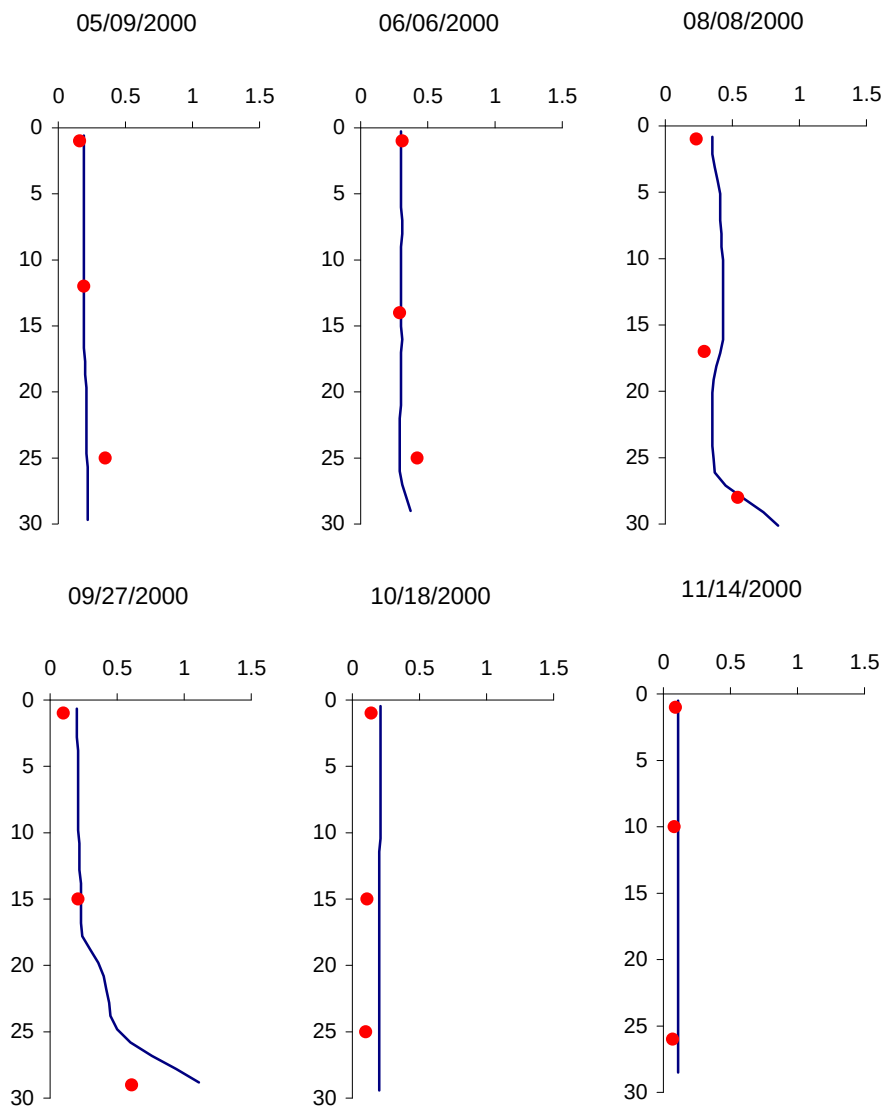


Figure I-3. PO_4 profile calibration results for Copco Reservoir at Copco Lake near Copco (2000) [X-axis PO_4-P (mg/L) vs. Y-axis Depth (m)].

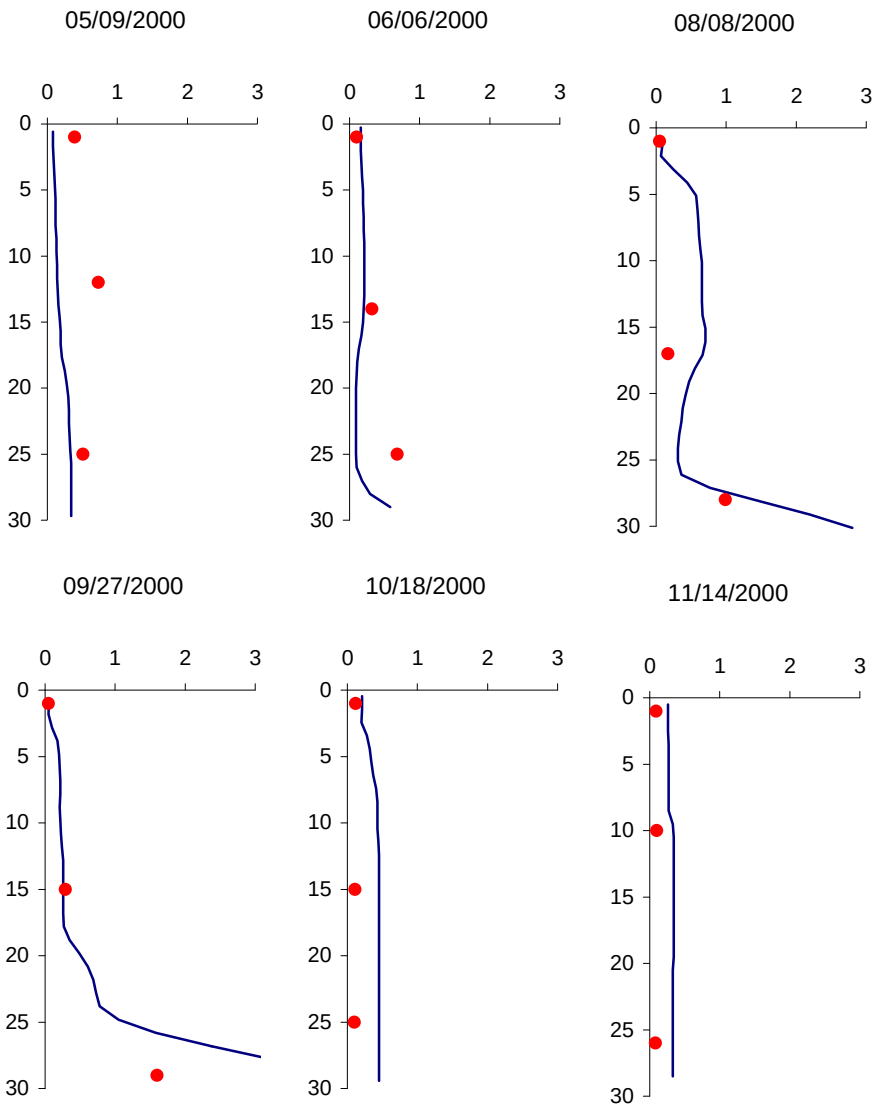


Figure I-4. NH_4 profile calibration results for Copco Reservoir at Copco Lake near Copco (2000) [X-axis NH_4 (mg/L) vs. Y-axis Depth (m)].

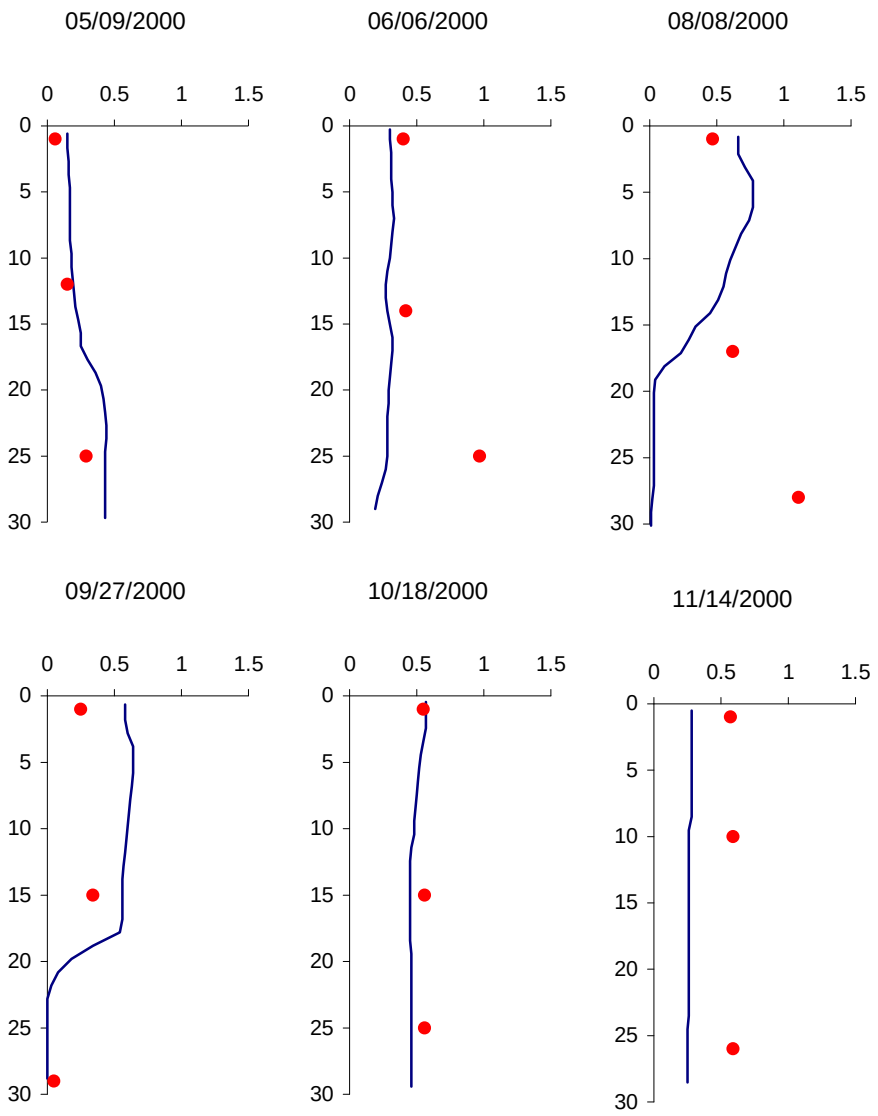


Figure I-5. NO_3 profile calibration results for Copco Reservoir at Copco Lake near Copco (2000) [X-axis NO_3 (mg/L) vs. Y-axis Depth (m)].

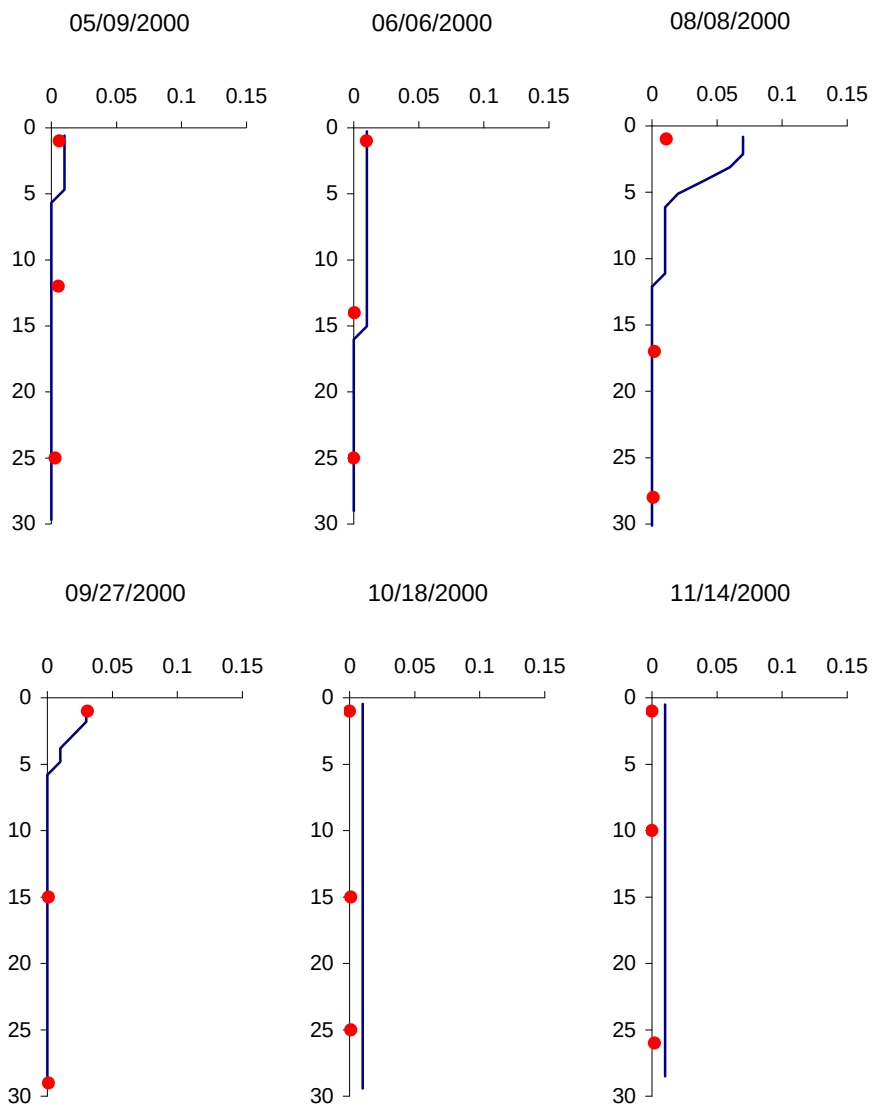


Figure I-6. Chlorophyll *a* profile calibration results for Copco Reservoir at Copco Lake near Copco (2000) [X-axis Chlorophyll *a* (mg/L) vs. Y-axis Depth (m)].

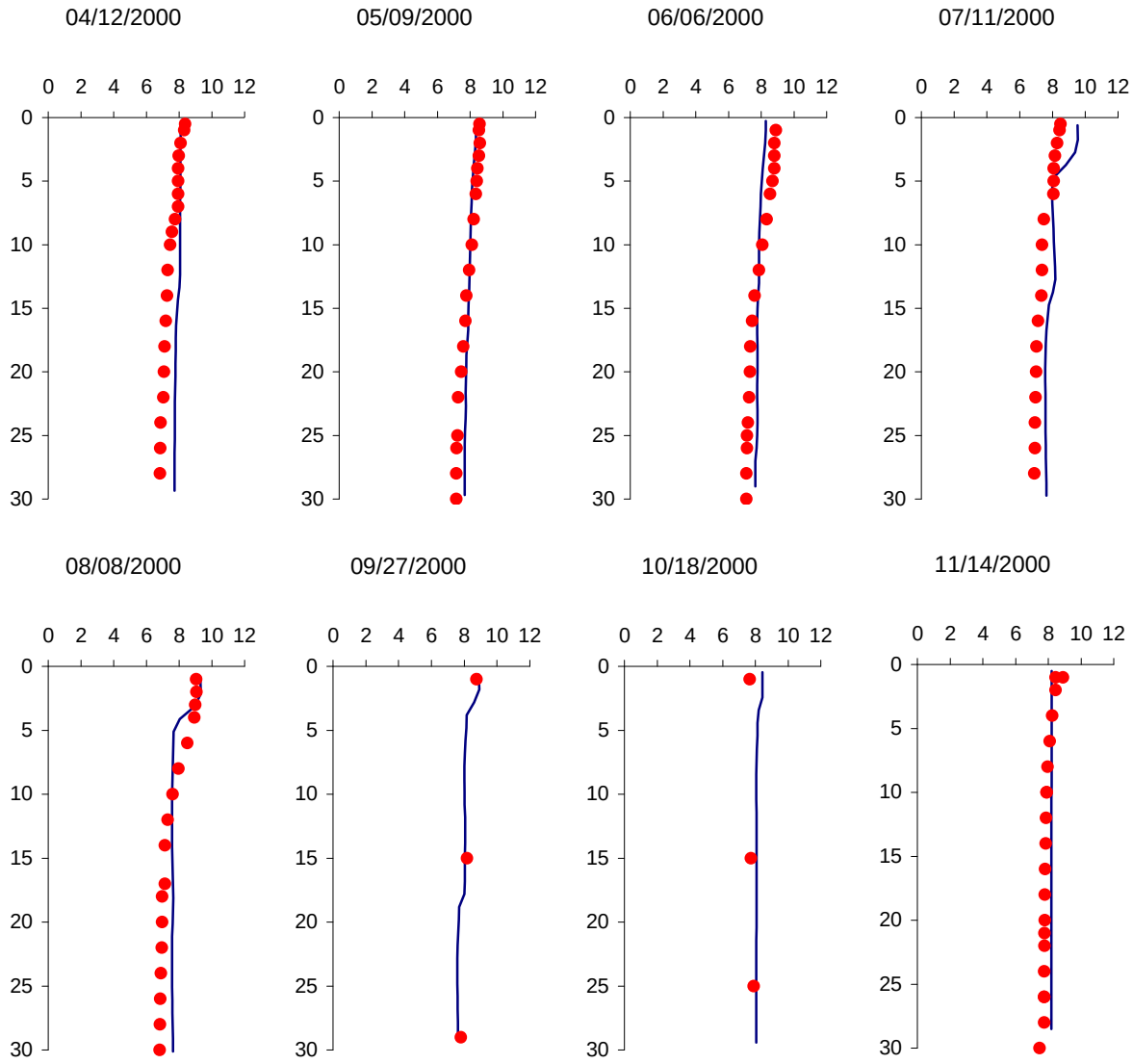


Figure I-7. pH profile calibration results for Copco Reservoir at Copco Lake near Copco (2000) [X-axis pH vs. Y-axis Depth (m)].

Appendix J

Water Quality Calibration Results for Iron Gate Reservoir (Modeling Segment 7)

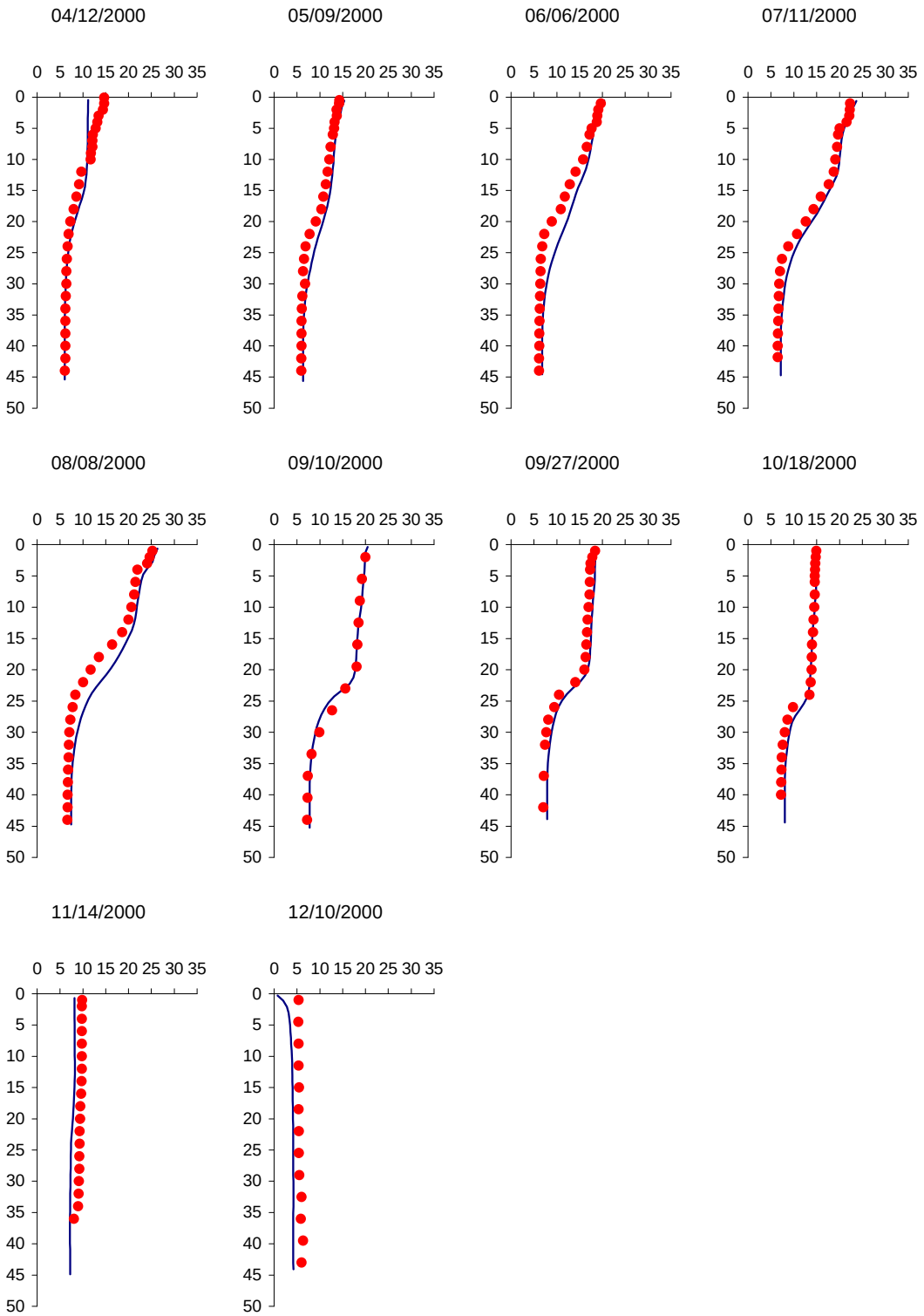


Figure J-1. Temperature profile calibration results at Iron Gate Reservoir (2000) [X-axis Temperature (deg C) vs. Y-axis Depth (m)].

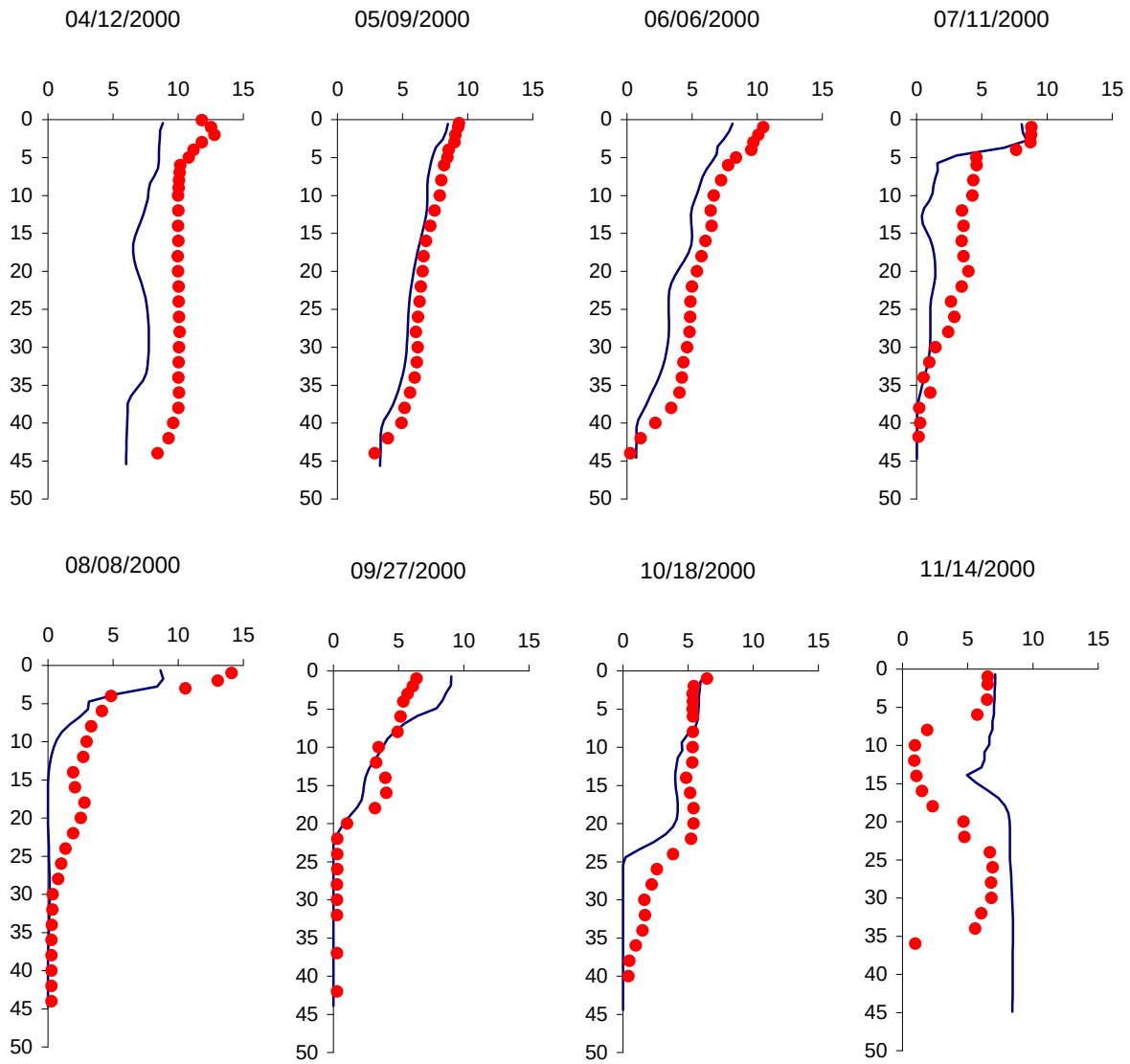


Figure J-2. Dissolved oxygen profile calibration results at Iron Gate Reservoir (2000) [X-axis Dissolved oxygen (mg/L) vs. Y-axis Depth (m)].

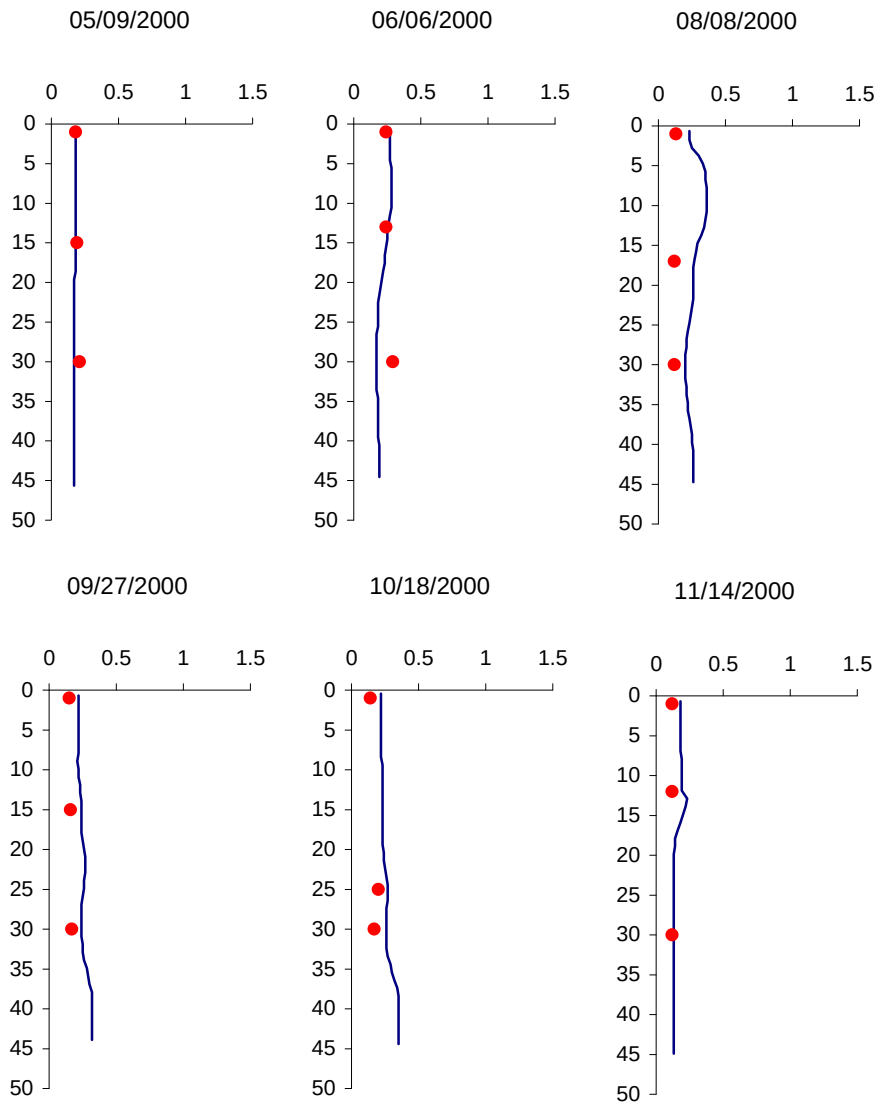


Figure J-3. PO_4 profile calibration results at Iron Gate Reservoir (2000) [X-axis PO_4 (mg/L) vs. Y-axis Depth (m)].

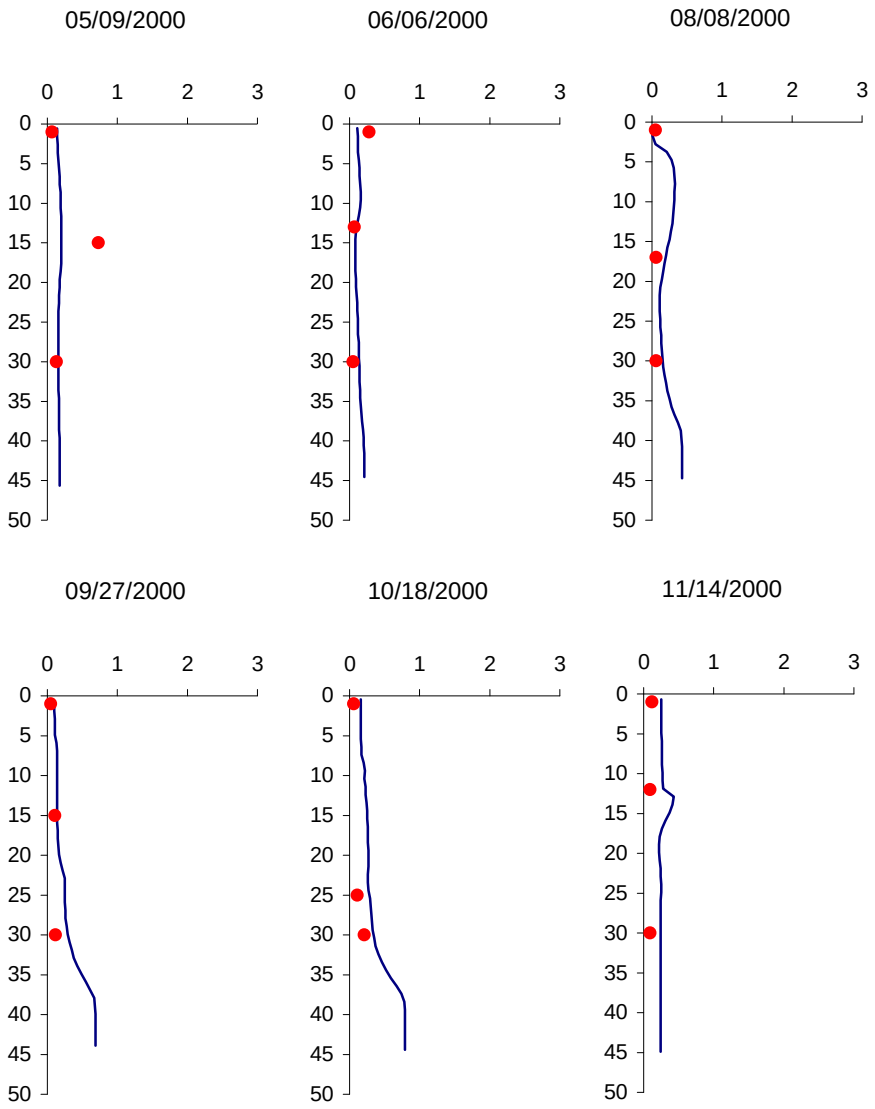


Figure J-4. NH_4 profile calibration results at Iron Gate Reservoir (2000) [X-axis NH_4 (mg/L) vs. Y-axis Depth (m)].

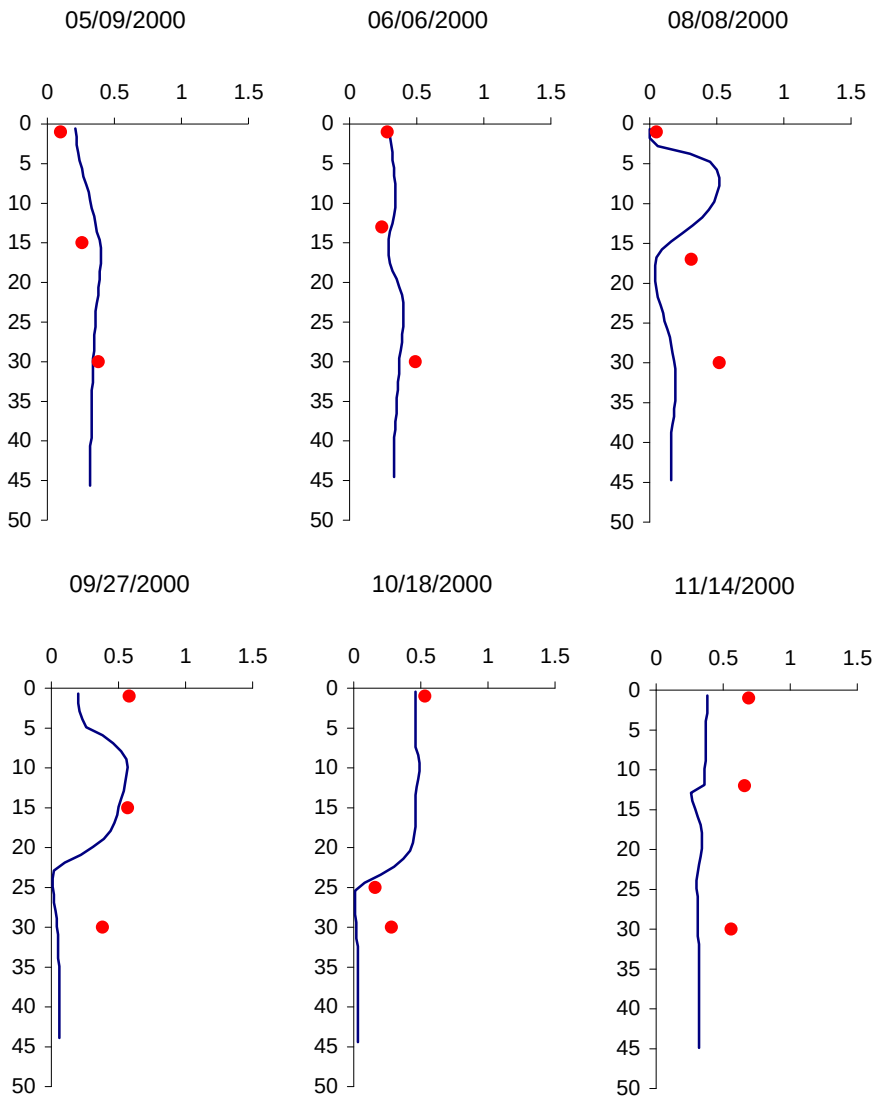


Figure J-5. NO_3 profile calibration results at Iron Gate Reservoir (2000) [X-axis NO_3 (mg/L) vs. Y-axis Depth (m)].

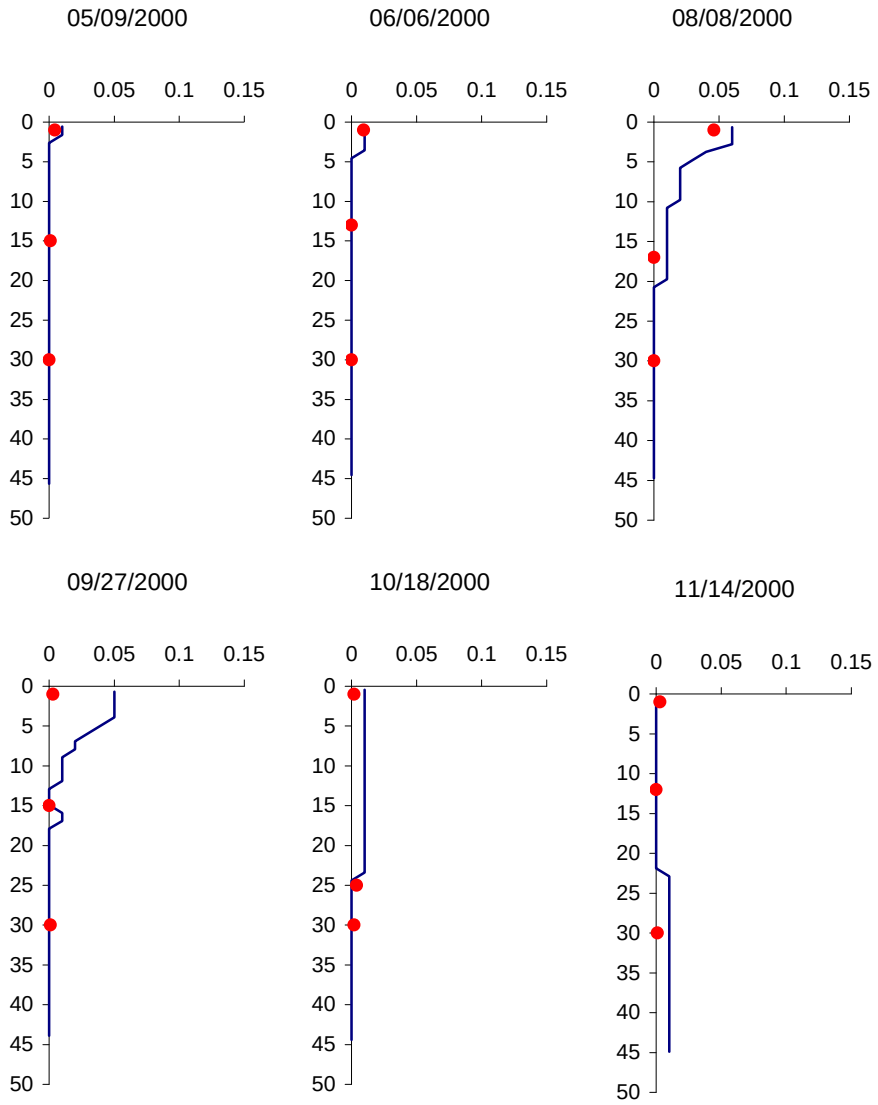


Figure J-6. Chlorophyll *a* profile calibration results at Iron Gate Reservoir (2000) [X-axis Chlorophyll *a* (mg/L) vs. Y-axis Depth (m)].

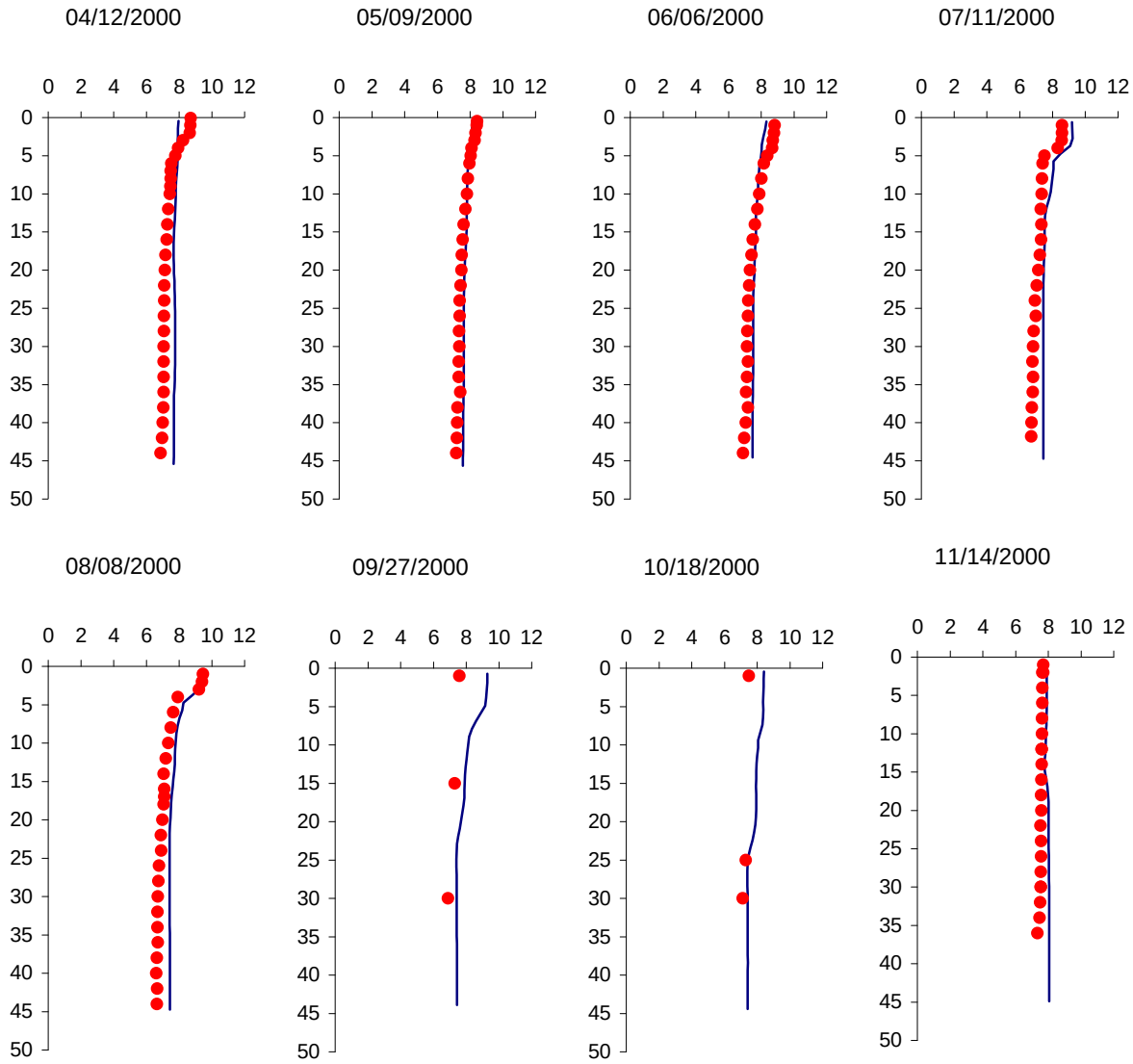


Figure J-7. pH profile calibration results at Iron Gate Reservoir (2000) [X-axis pH vs. Y-axis Depth (m)].

Appendix K

Water Quality Calibration Results for Iron Gate Dam to Turwar (Modeling Segment 8)

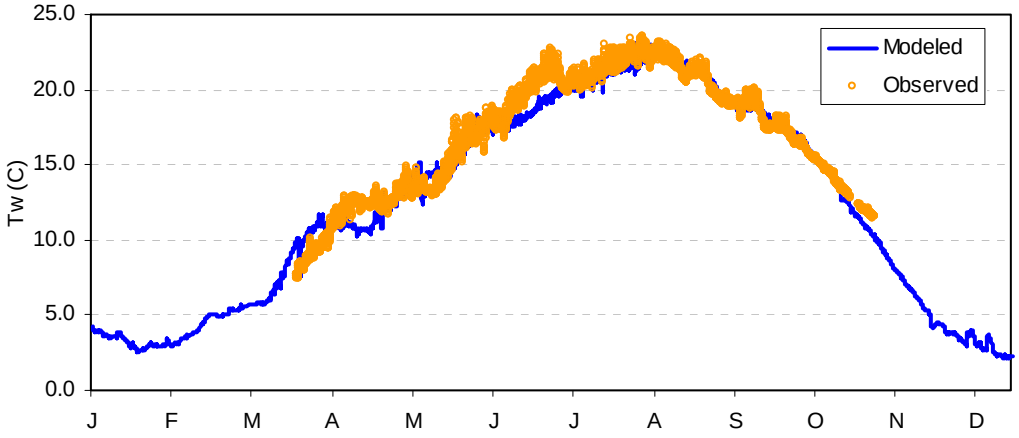


Figure K-1. Temperature calibration at Klamath River below Iron Gate Dam (Hatchery Br.) (2000).

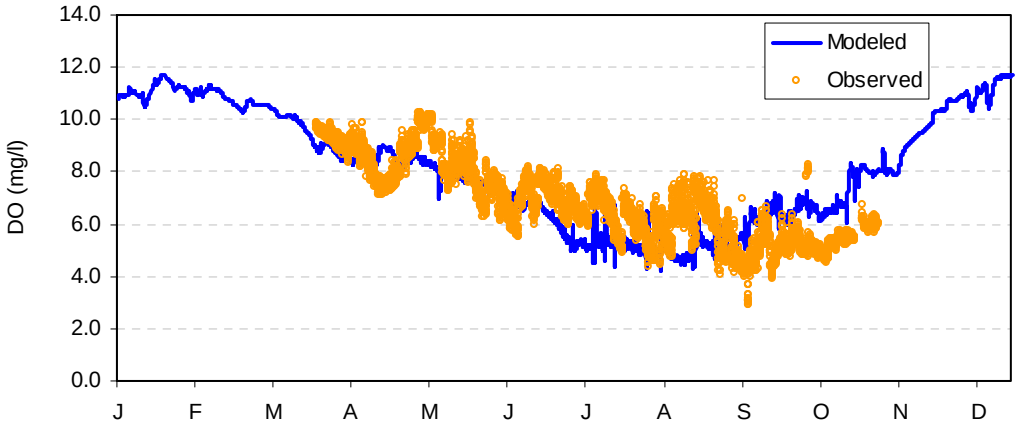


Figure K-2. Dissolved oxygen calibration at Klamath River below Iron Gate Dam (Hatchery Br) (2000).

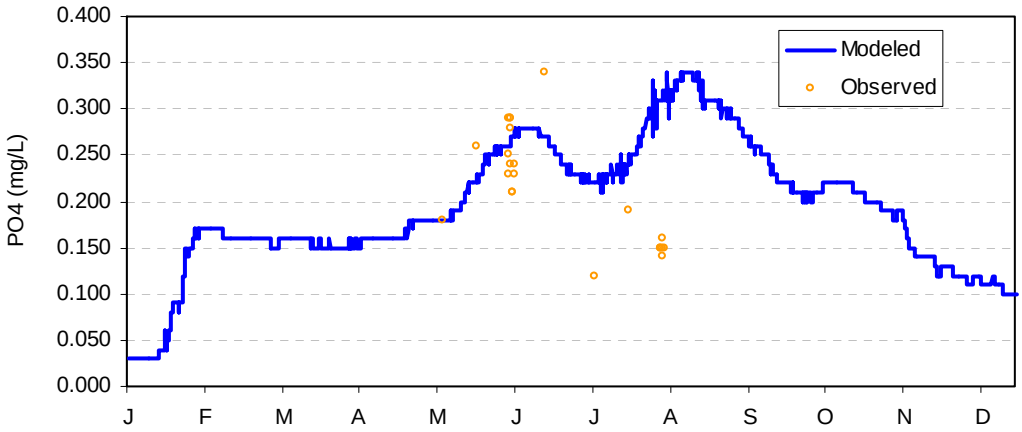


Figure K-3. PO_4 calibration at Klamath River below Iron Gate Dam (Hatchery Br) (2000).

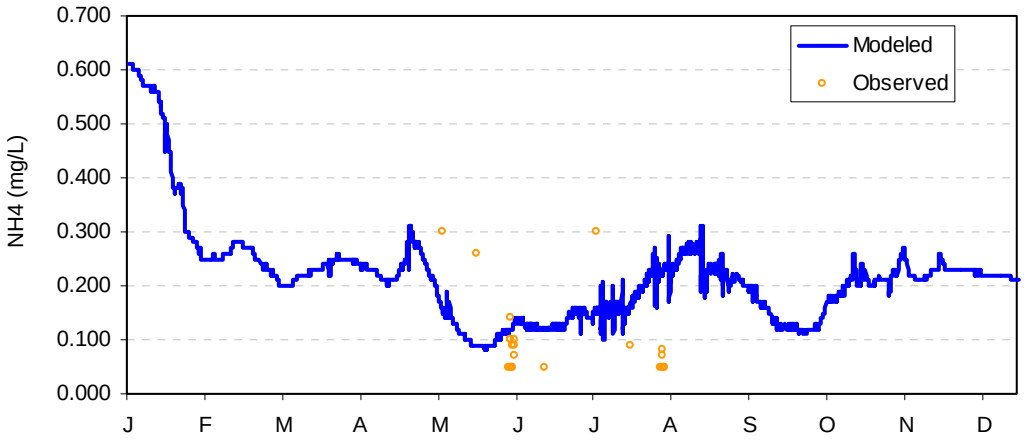


Figure K-4. NH_4 calibration at Klamath River below Iron Gate Dam (Hatchery Br) (2000).

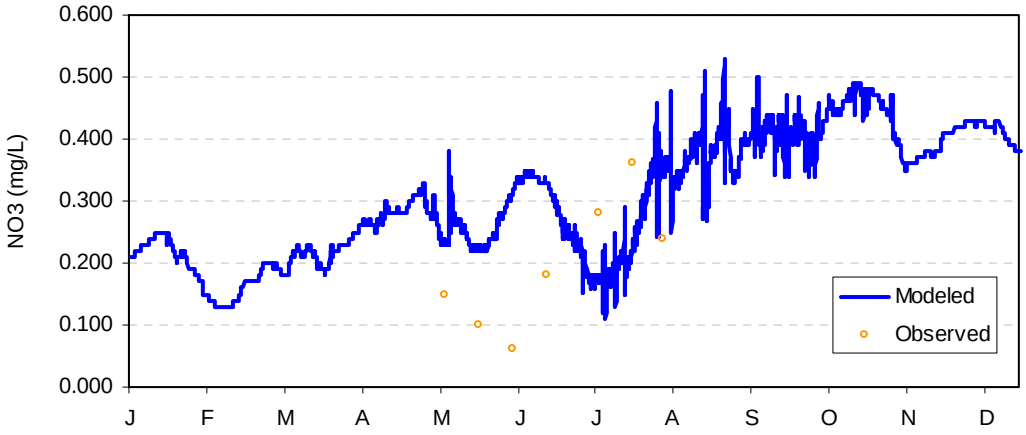


Figure K-5. NO_3 calibration at Klamath River below Iron Gate Dam (Hatchery Br) (2000).

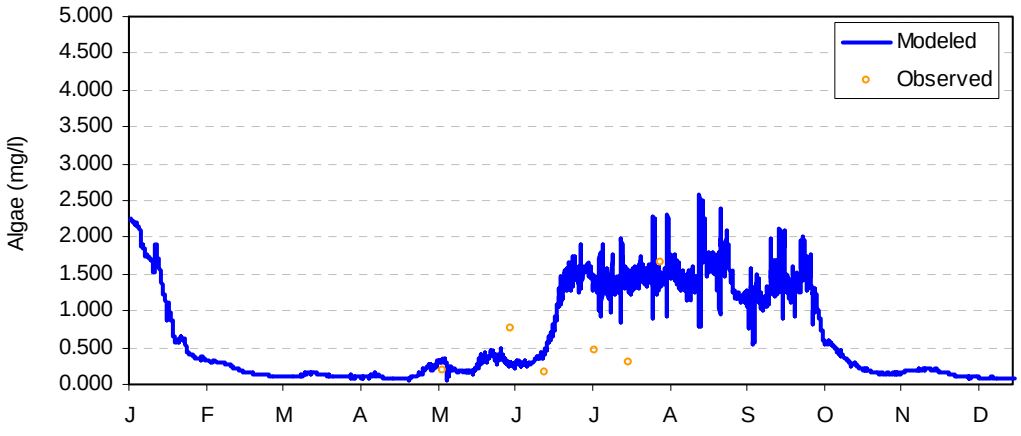


Figure K-6. Algae calibration at Klamath River below Iron Gate Dam (Hatchery Br) (2000).

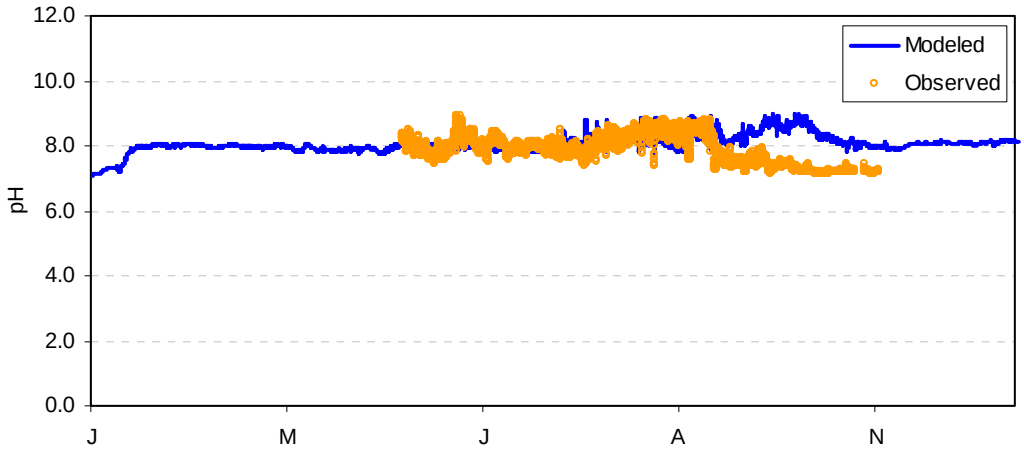


Figure K-7. pH calibration at Klamath River below Iron Gate Dam (Hatchery Br) (2000).

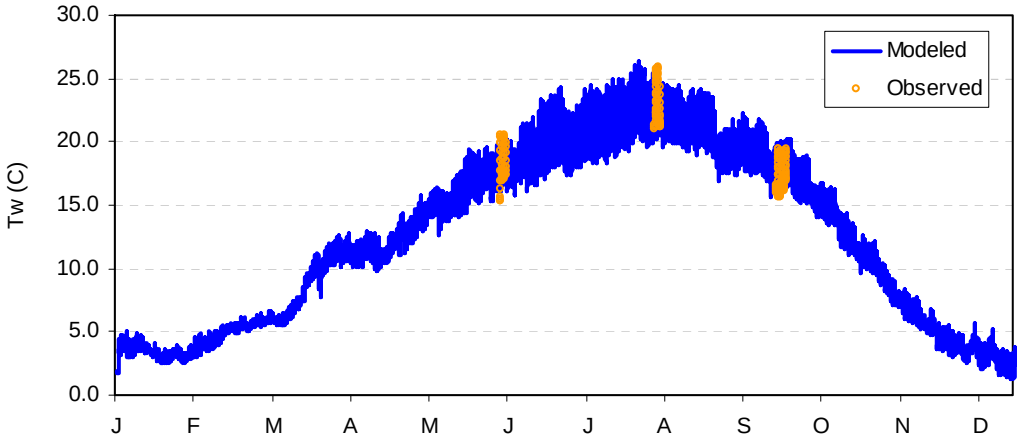


Figure K-8. Temperature calibration at Klamath River above Shasta River (2000).

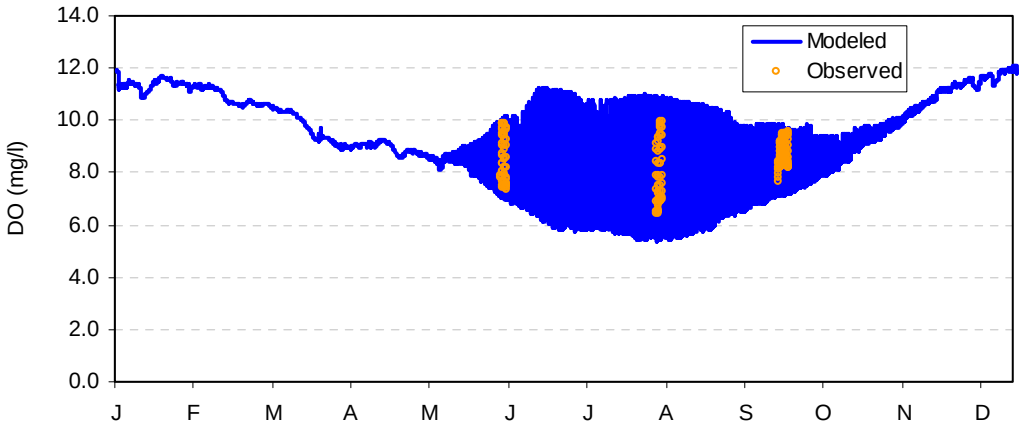


Figure K-9. Dissolved oxygen calibration at Klamath River above Shasta River (2000).

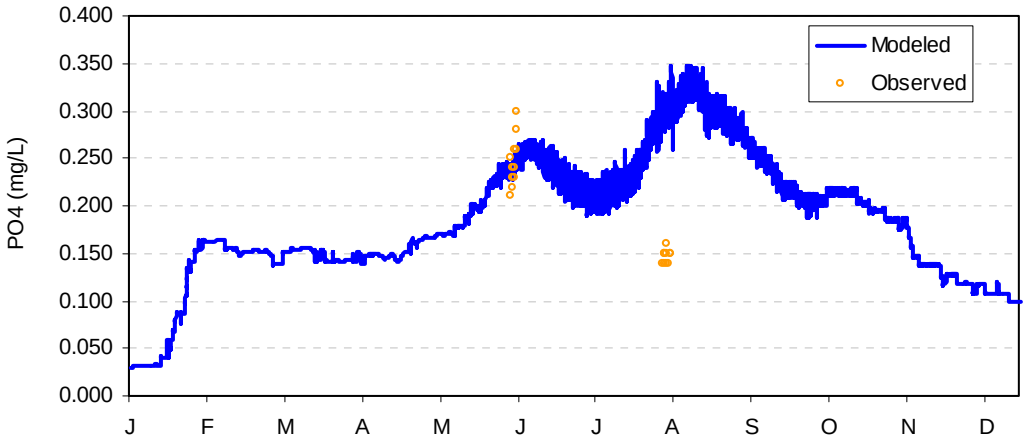


Figure K-10. PO_4 calibration at Klamath River above Shasta River (2000).

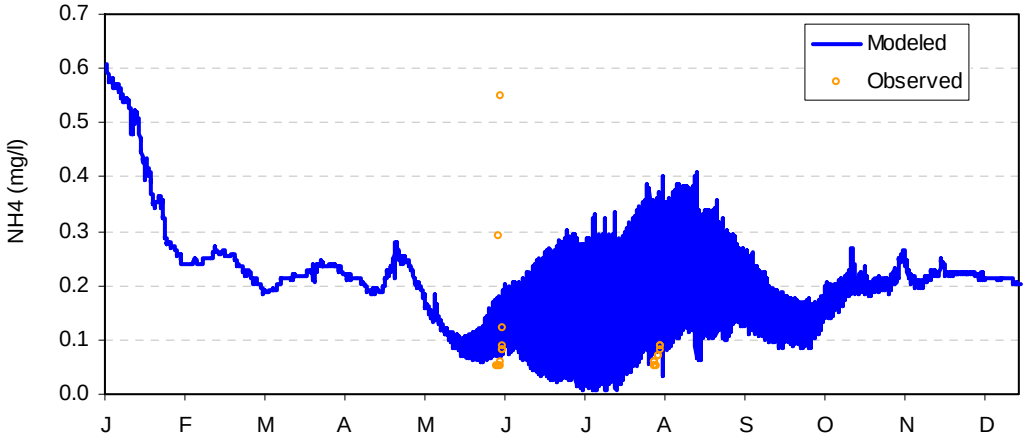


Figure K-11. NH_4 calibration at Klamath River above Shasta River (2000).

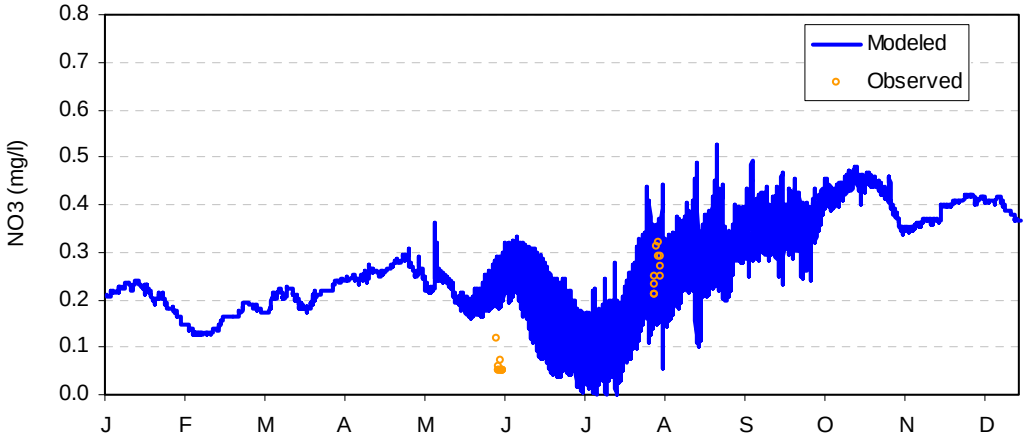


Figure K-12. NO_3 calibration at Klamath River above Shasta River (2000).

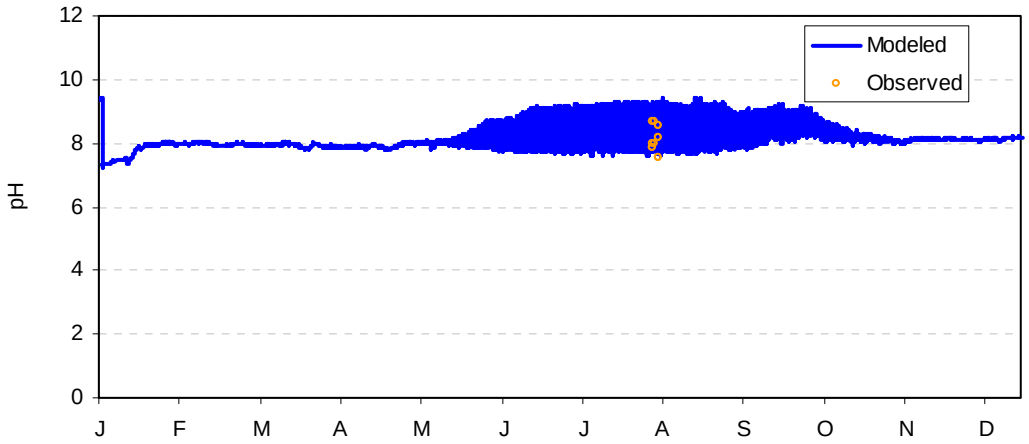


Figure K-13. pH calibration at Klamath River above Shasta River (2000).

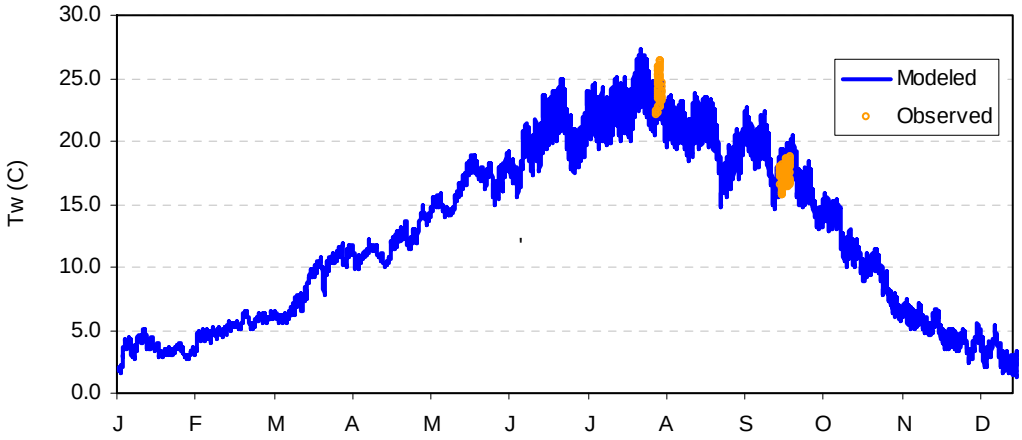


Figure K-14. Temperature calibration at Klamath River above Scott River (2000).

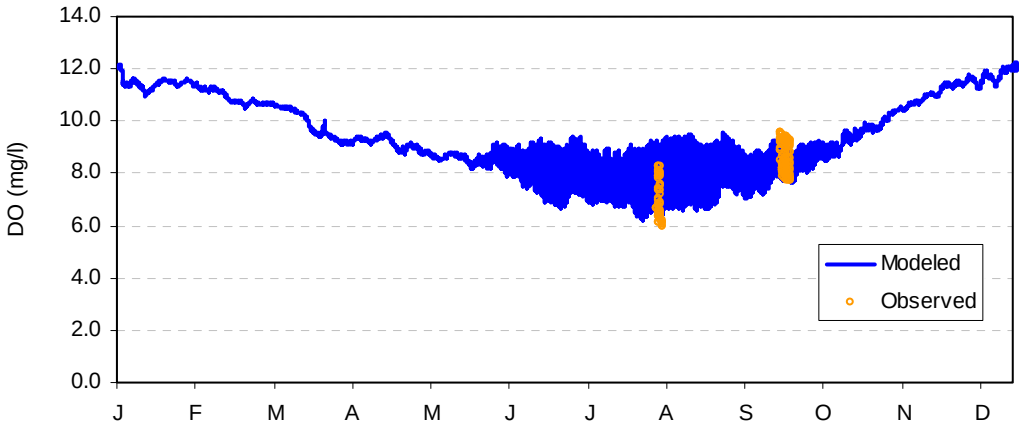


Figure K-15. Dissolved oxygen calibration at Klamath River above Scott River (2000).

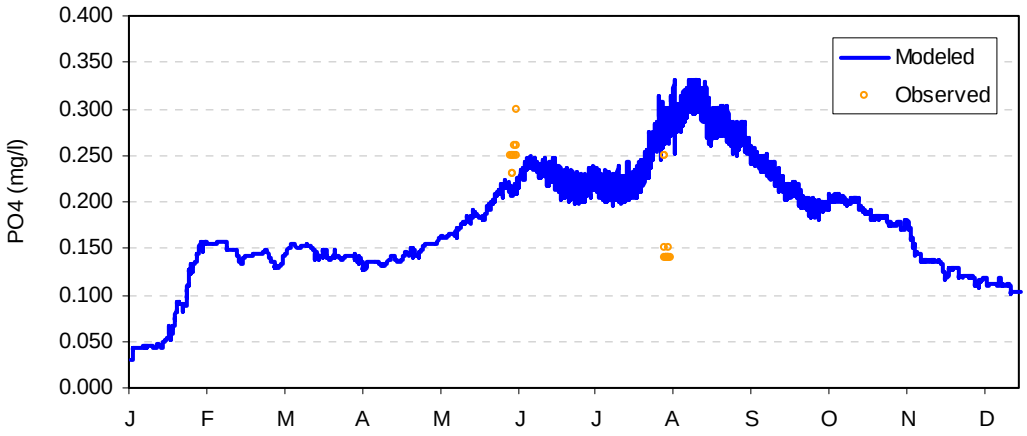


Figure K-16. PO_4 calibration at Klamath River above Scott River (2000).

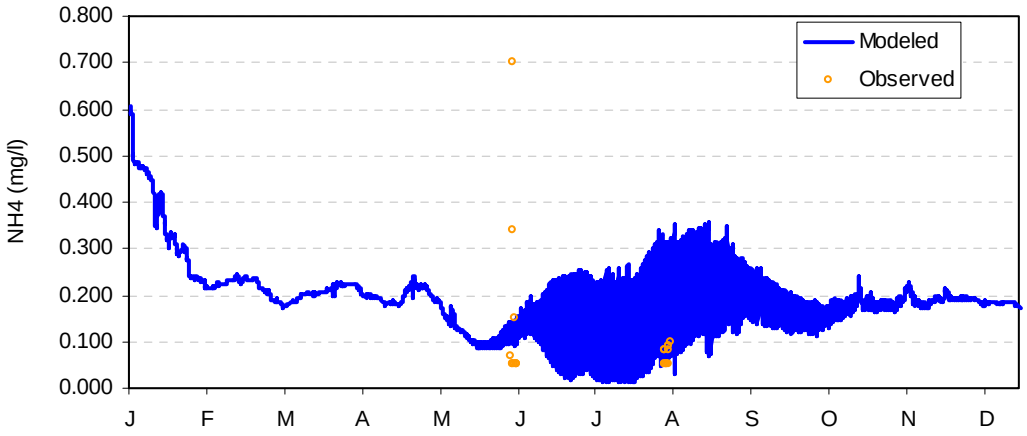


Figure K-17. NH_4 calibration at Klamath River above Scott River (2000).

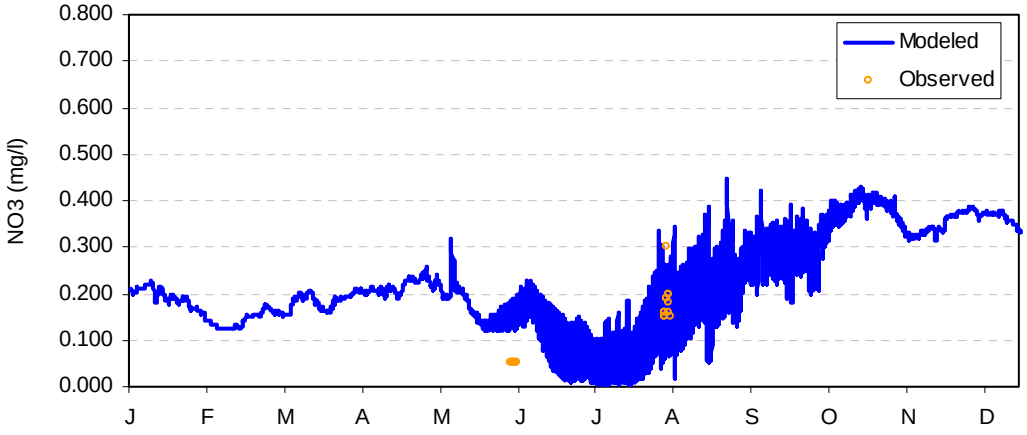


Figure K-18. NO_3 calibration at Klamath River above Scott River (2000).

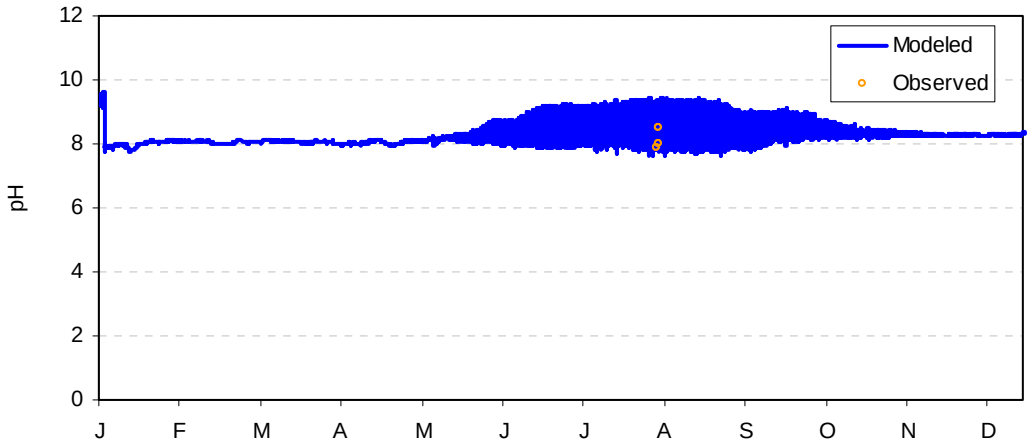


Figure K-19. pH calibration at Klamath River above Scott River (2000).

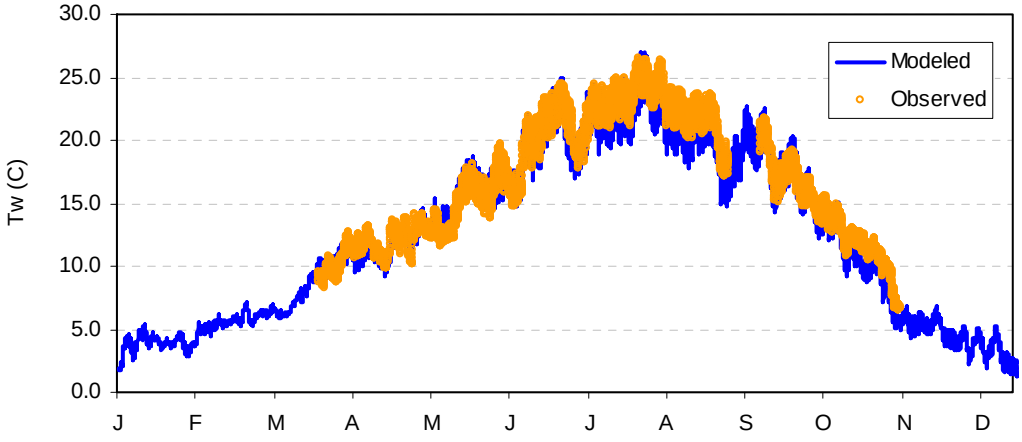


Figure K-20. Temperature calibration at Klamath River above Seiad Valley (2000).

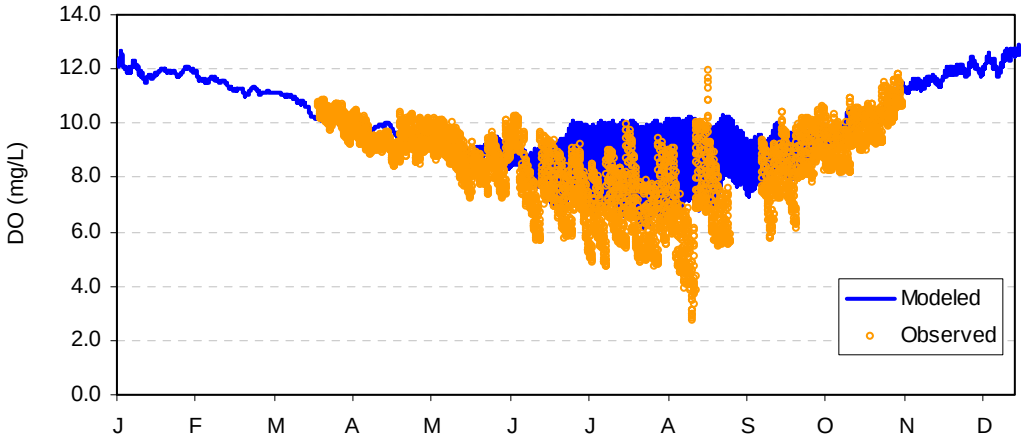


Figure K-21. Dissolved oxygen calibration at Klamath River above Seiad Valley (2000).

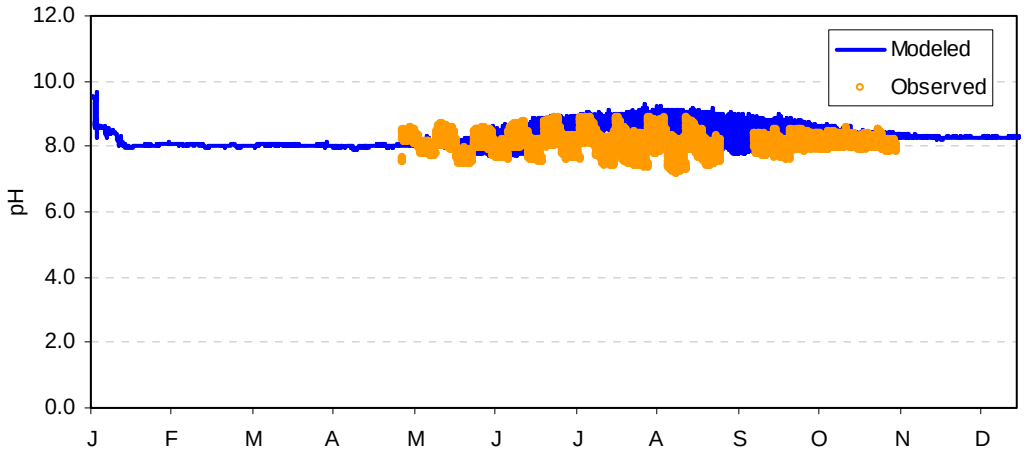


Figure K-22. pH calibration at Klamath River above Seiad Valley (2000).

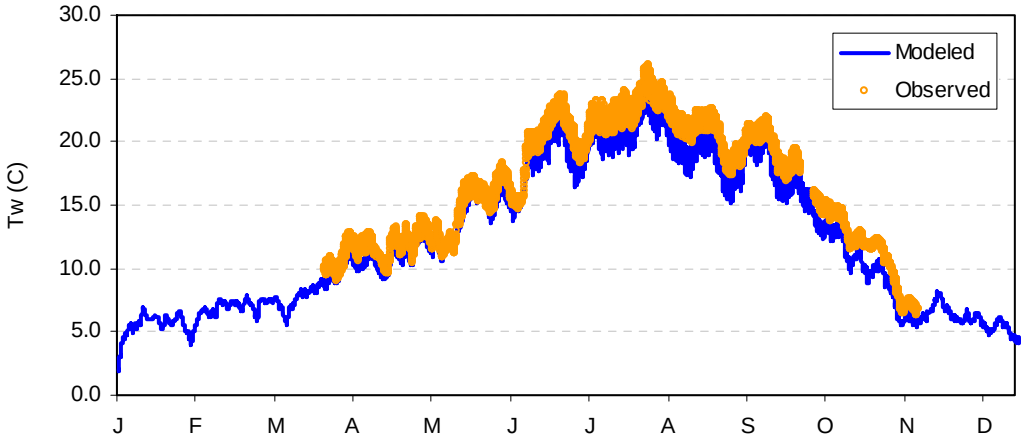


Figure K-23. Temperature calibration at Klamath River at Youngs Bar (2000).

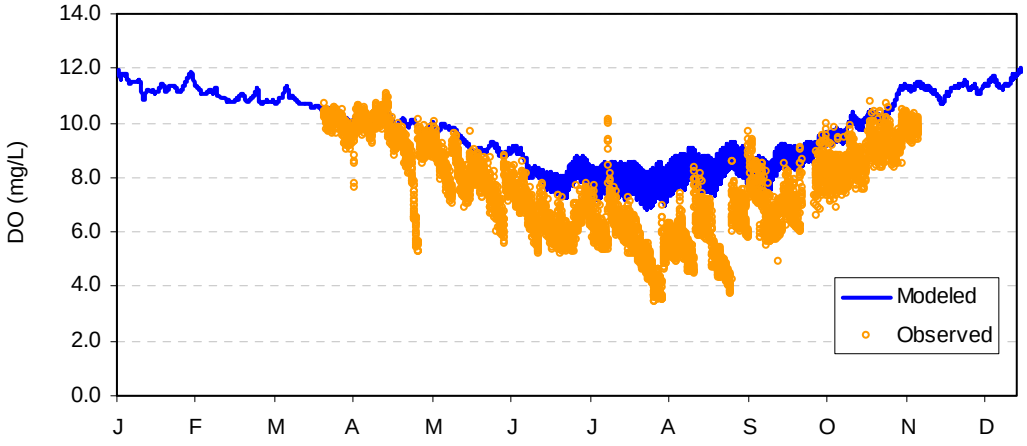


Figure K-24. Dissolved oxygen calibration at Klamath River at Youngs Bar (2000).

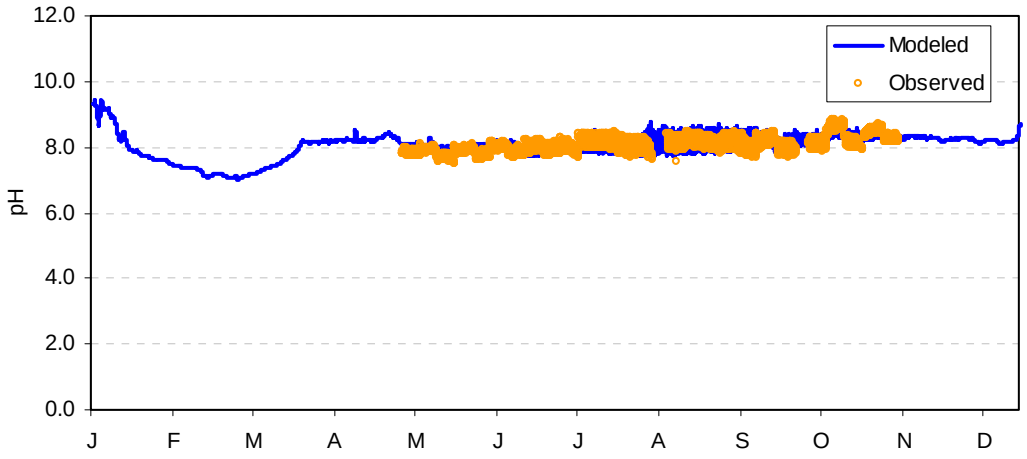


Figure K-25. pH calibration at Klamath River at Youngs Bar (2000).

Appendix L

Calibration Results for Turwar to the Pacific Ocean (Modeling Segment 9)

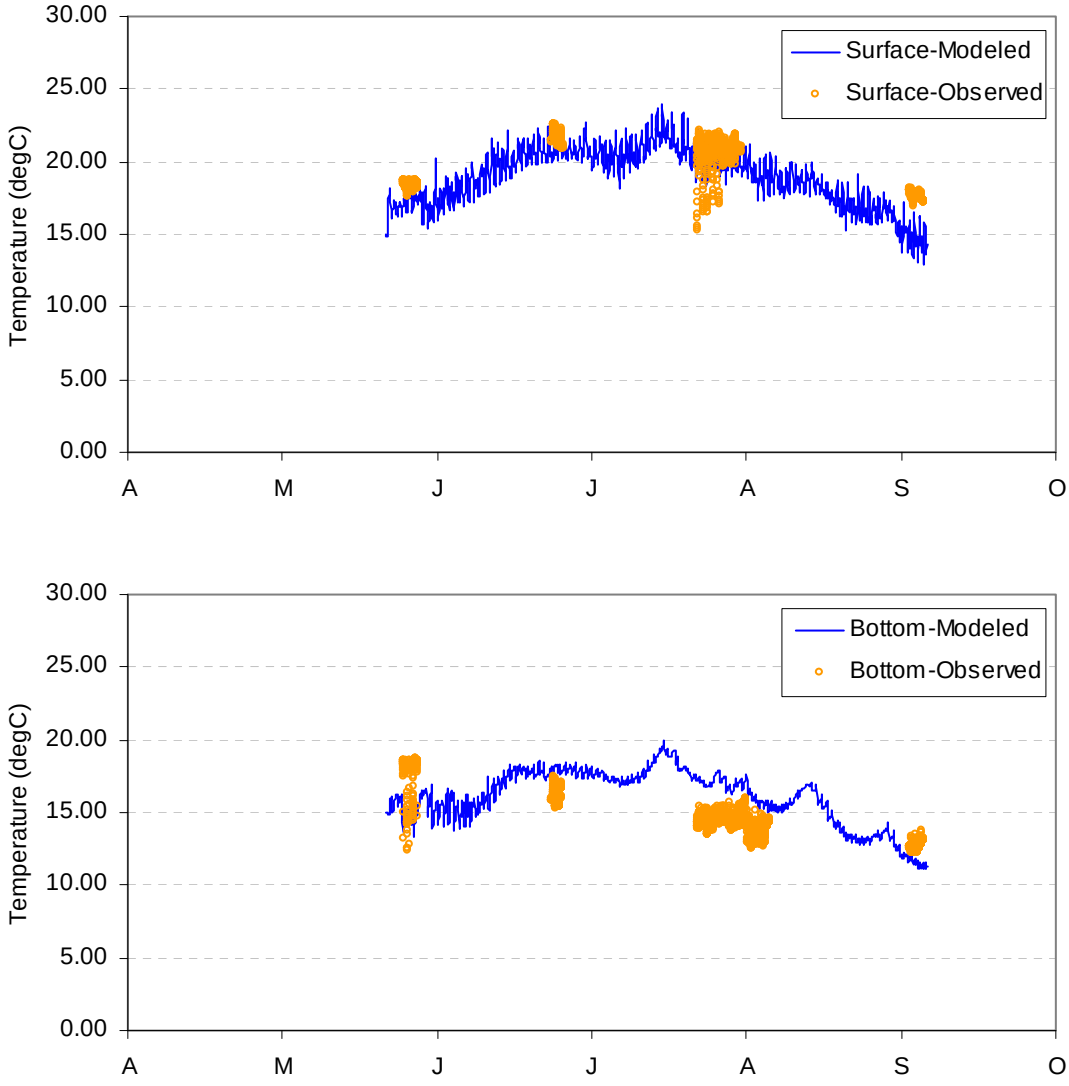


Figure L-1. Lower estuary temperature calibration (2004).

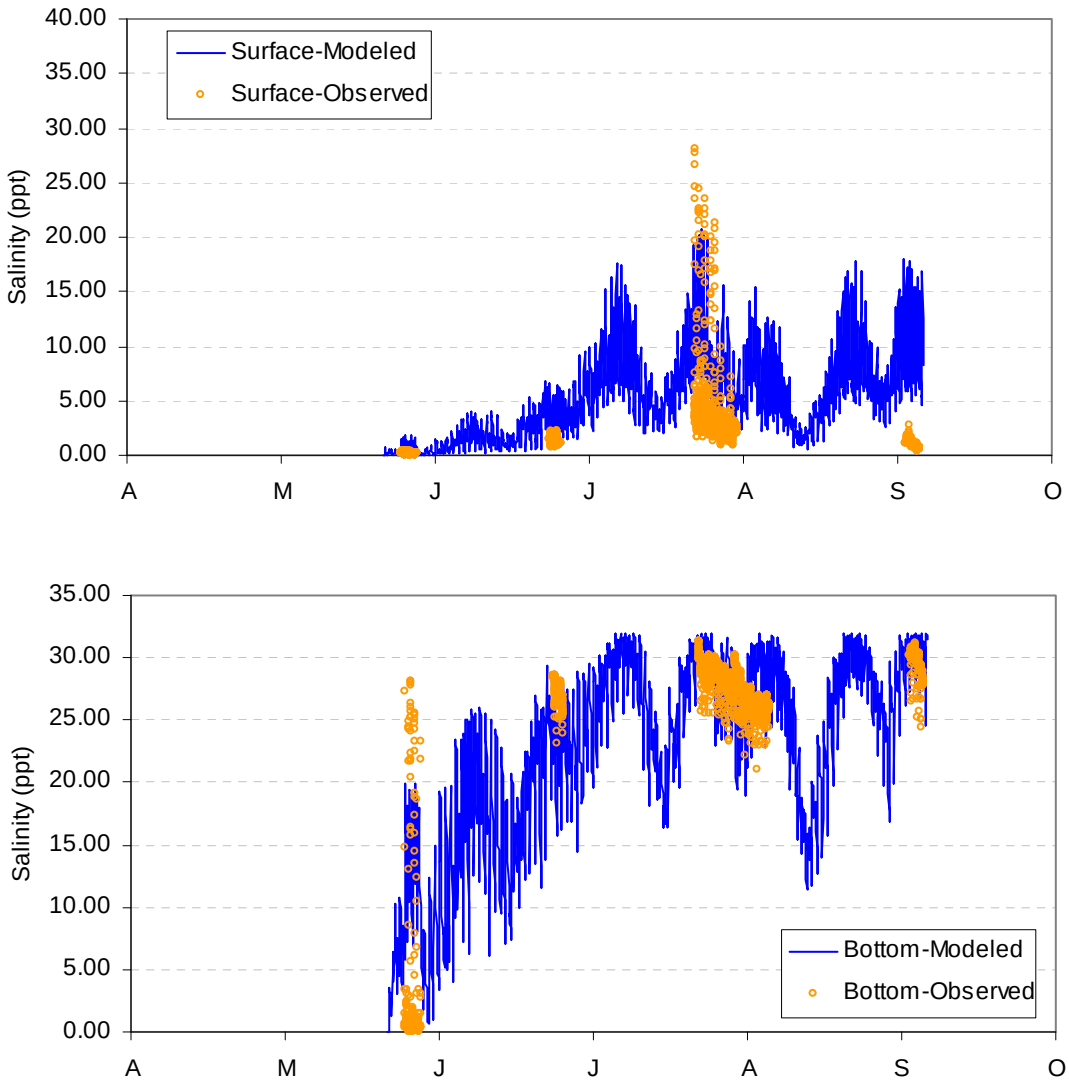


Figure L-2. Lower estuary salinity calibration (2004).

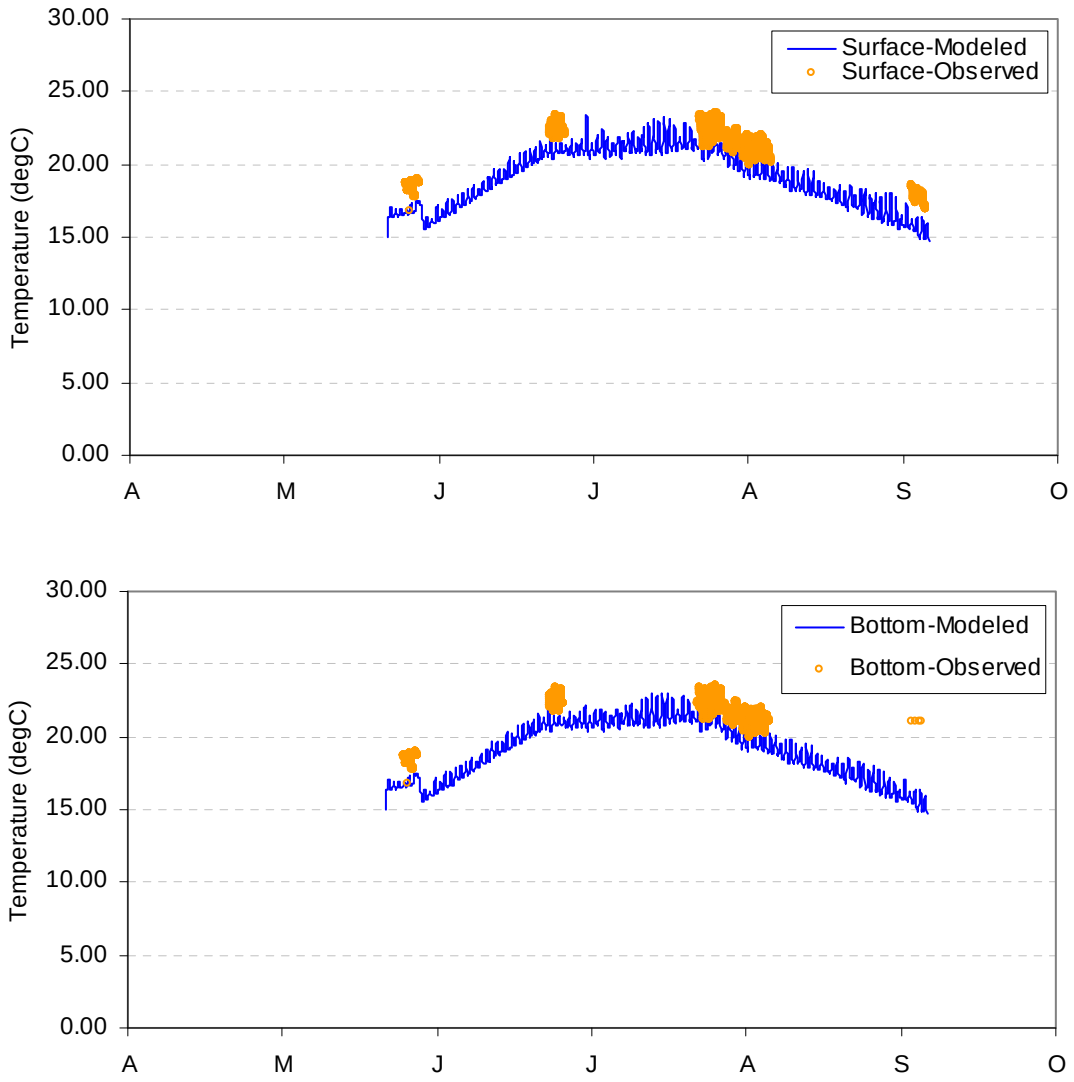


Figure L-3. Middle estuary temperature calibration (2004).

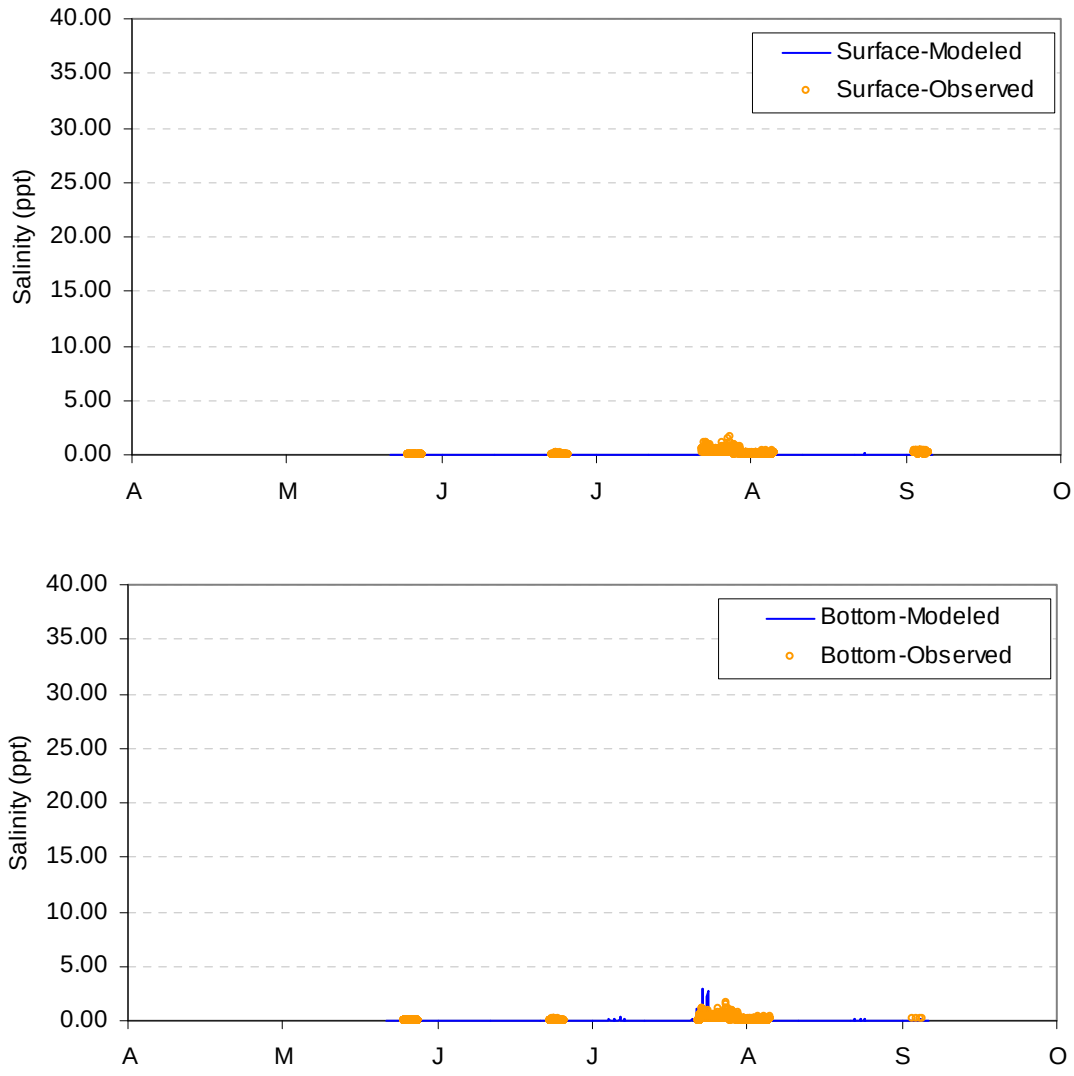


Figure L-4. Middle estuary salinity calibration (2004).

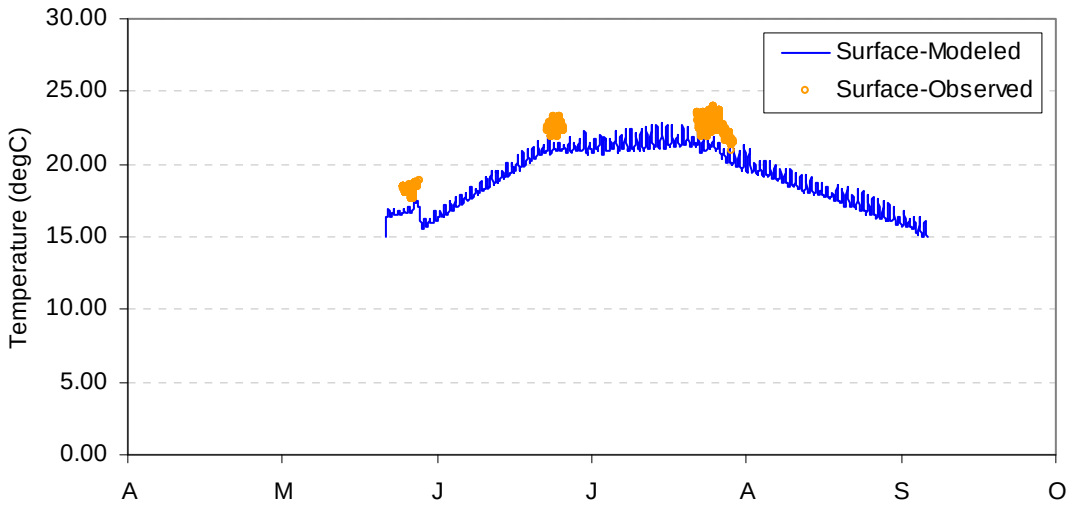


Figure L-5. Upper estuary temperature calibration (2004).

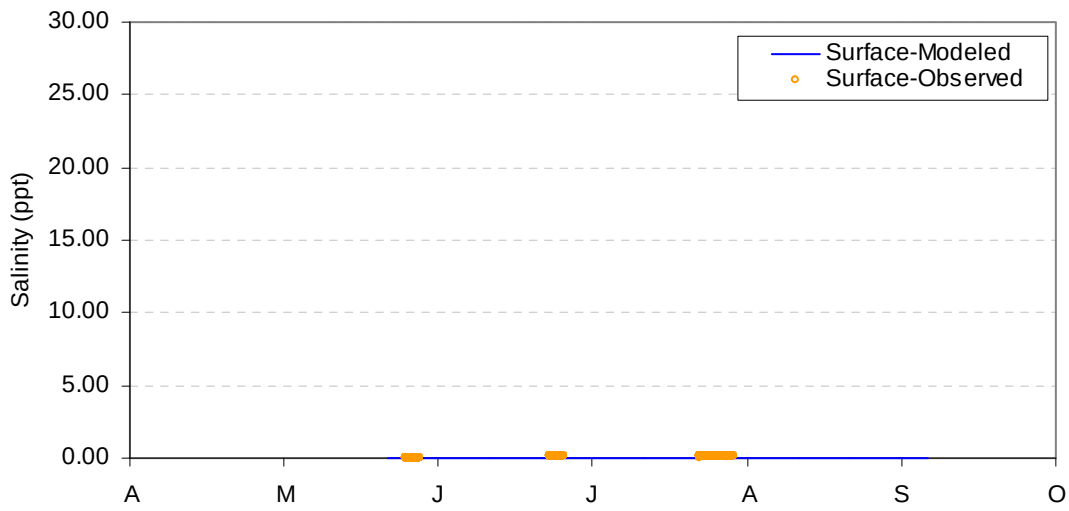


Figure L-6. Upper estuary salinity calibration (2004).

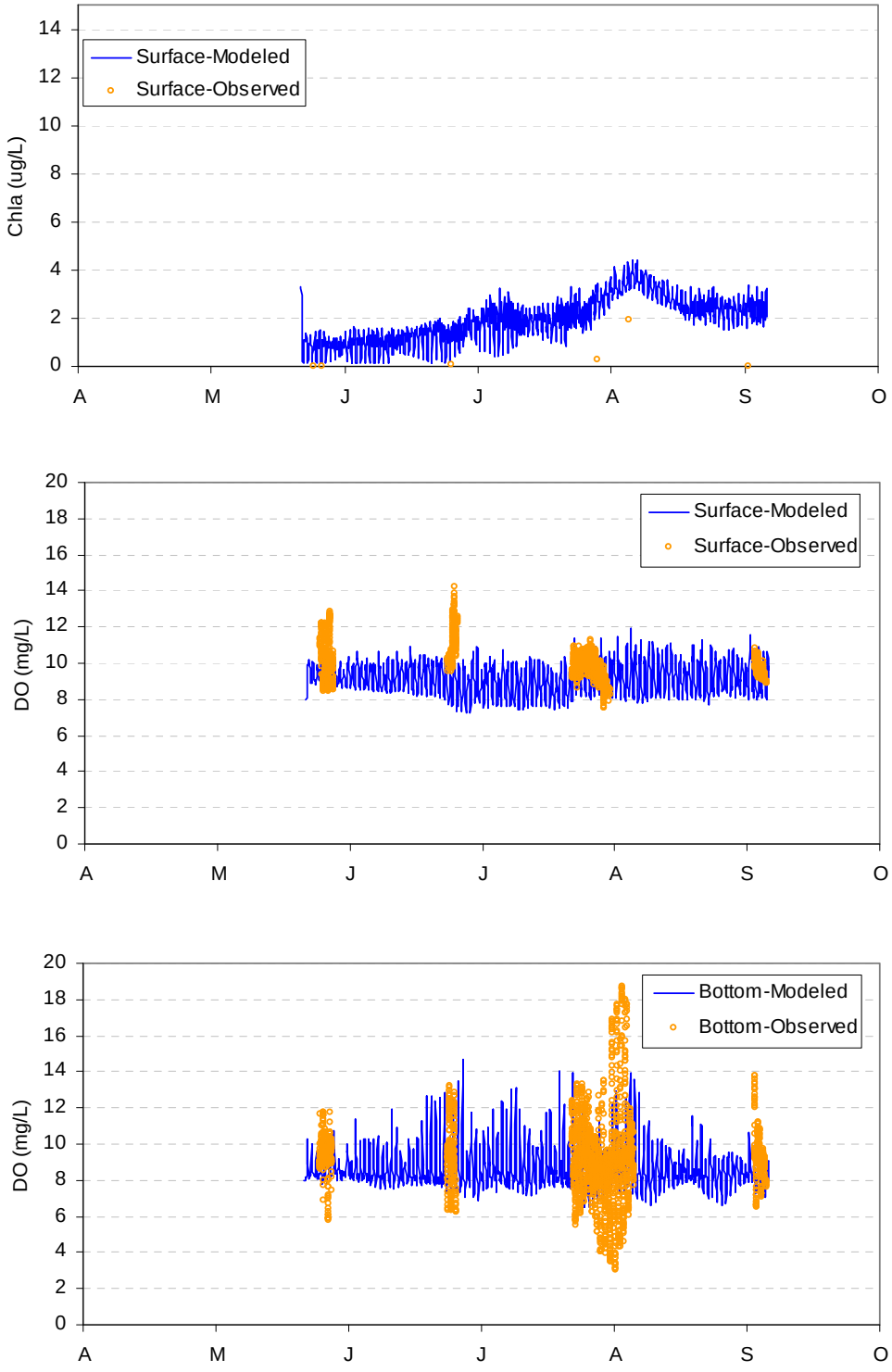


Figure L-7. Chlorophyll *a* and dissolved oxygen calibration results at the lower estuary water quality station (2004).

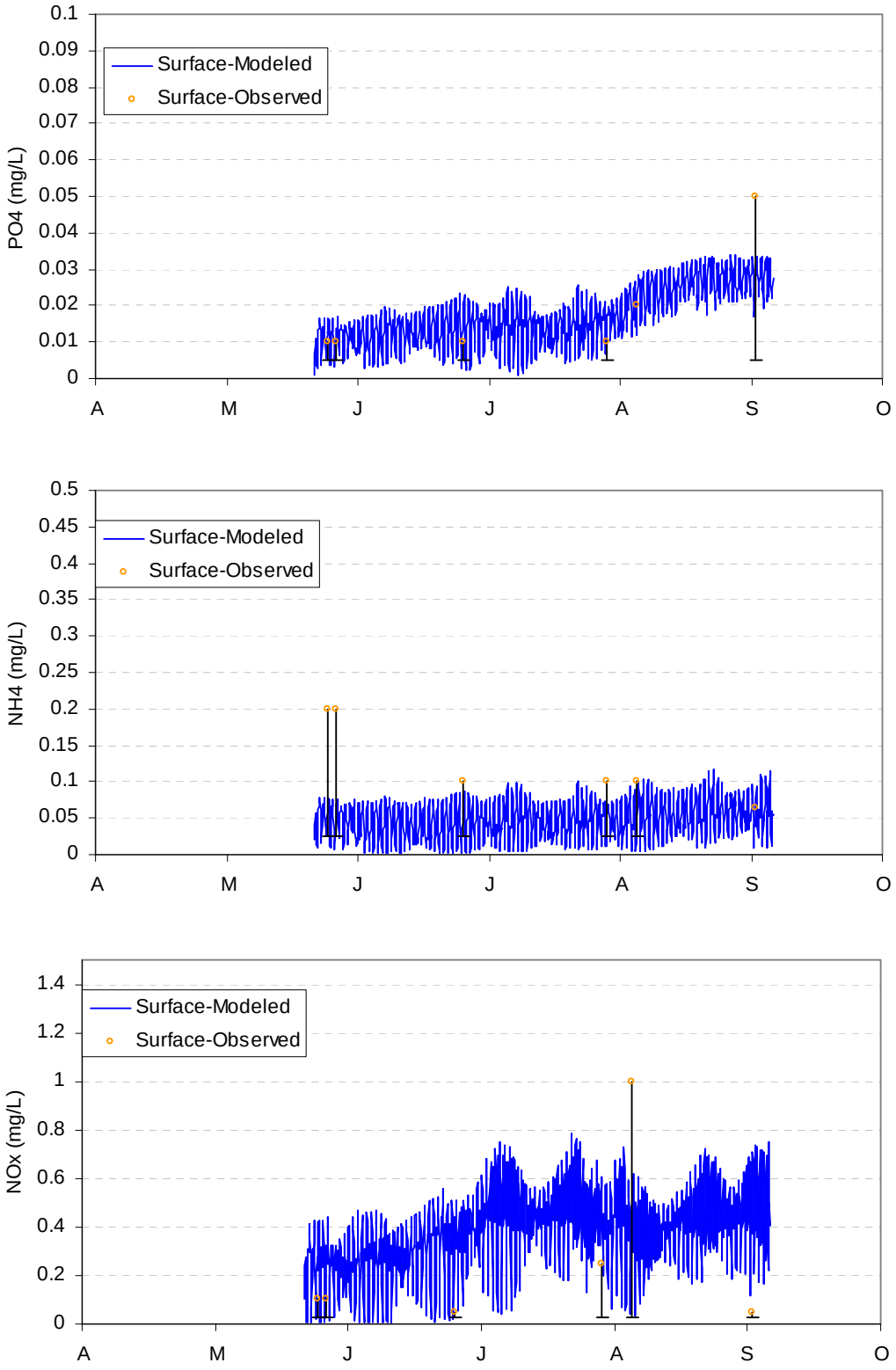


Figure L-7 (continued). PO_4, NH_4, and NO_x calibration results at the lower estuary water quality station (2004).

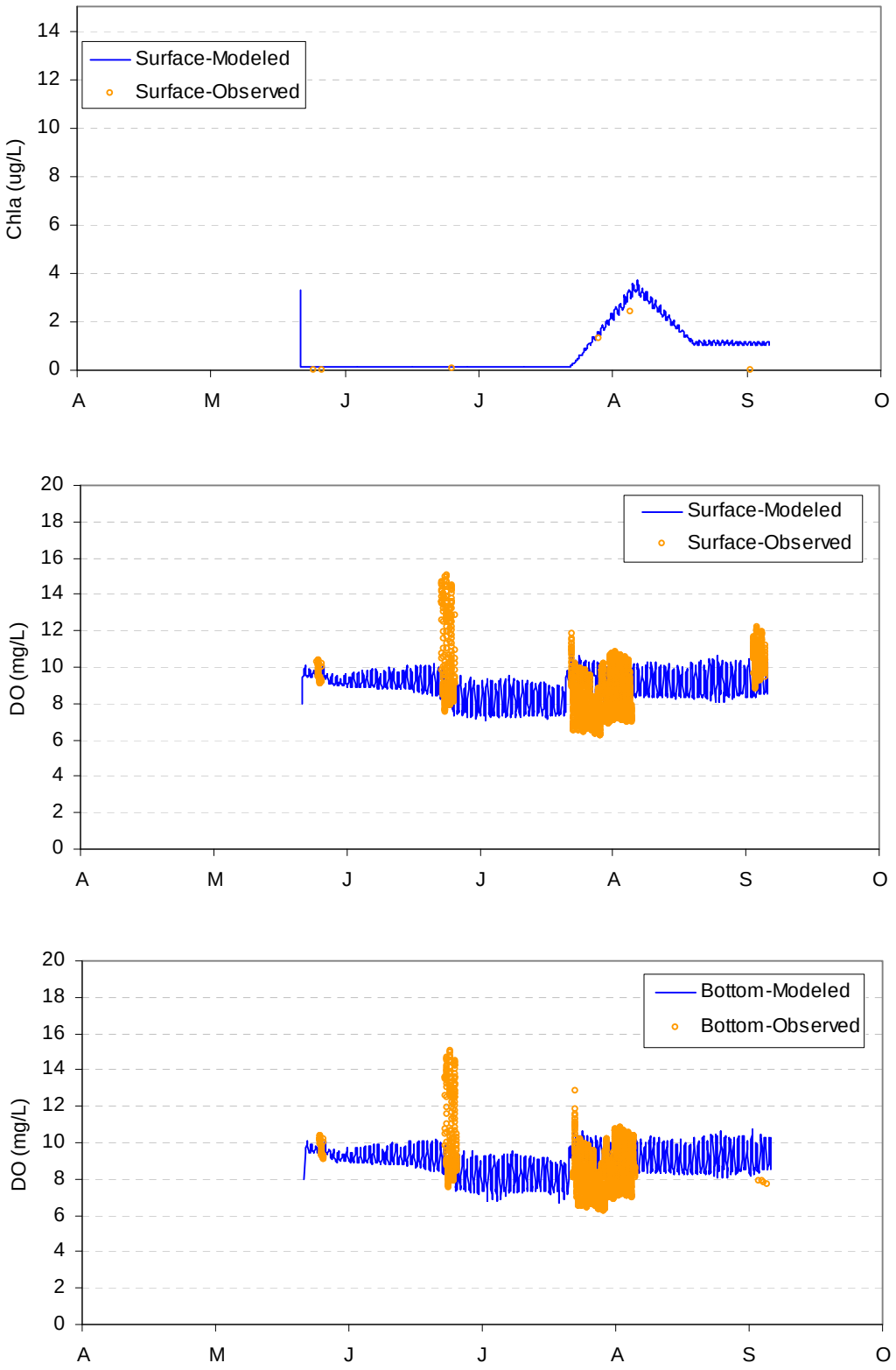


Figure L-8. Chlorophyll *a* and dissolved oxygen calibration results at the middle estuary water quality station (2004).

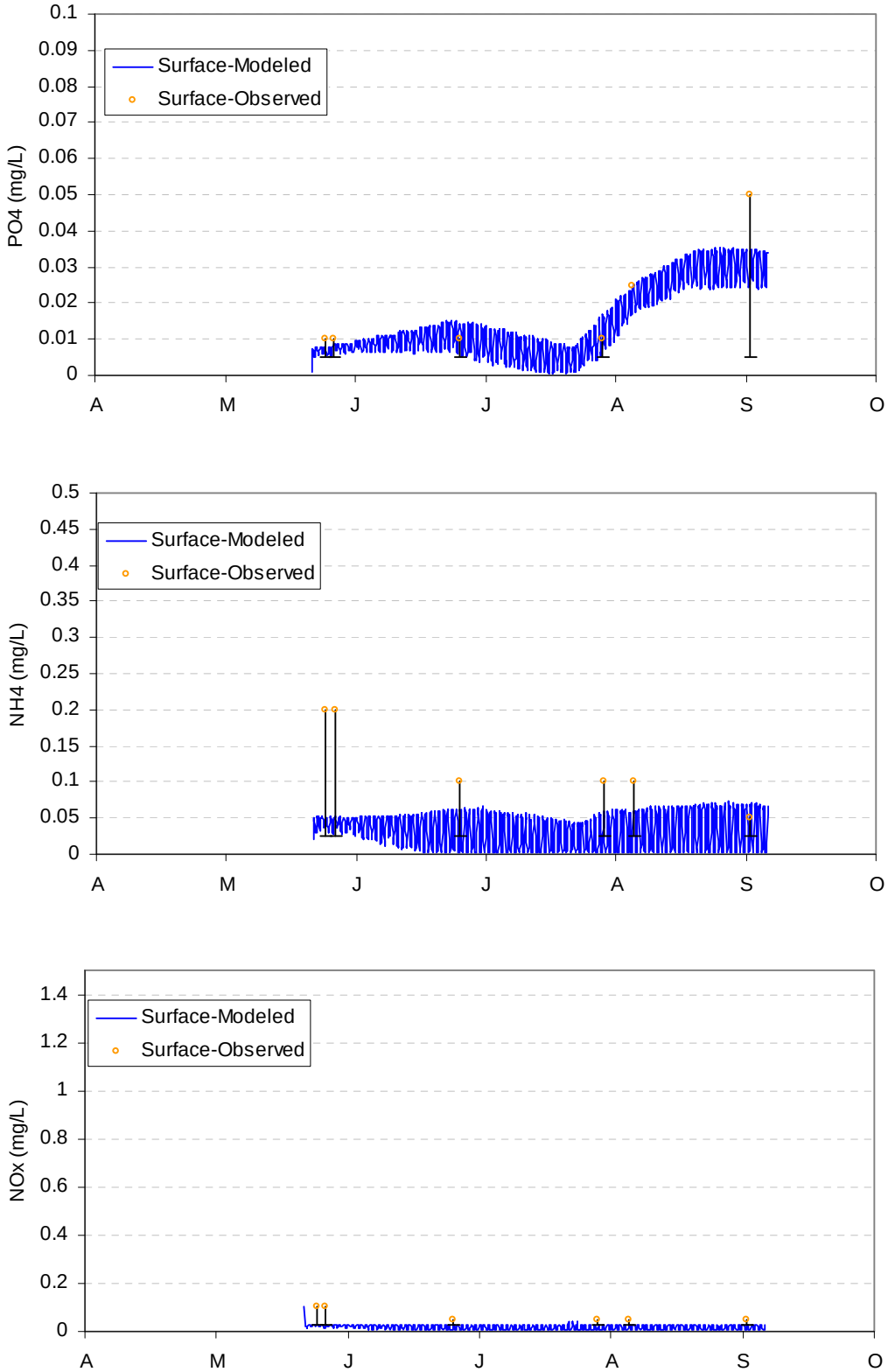


Figure L-8 (continued). PO_4, NH_4, and NO_x calibration results at the middle estuary water quality station (2004).

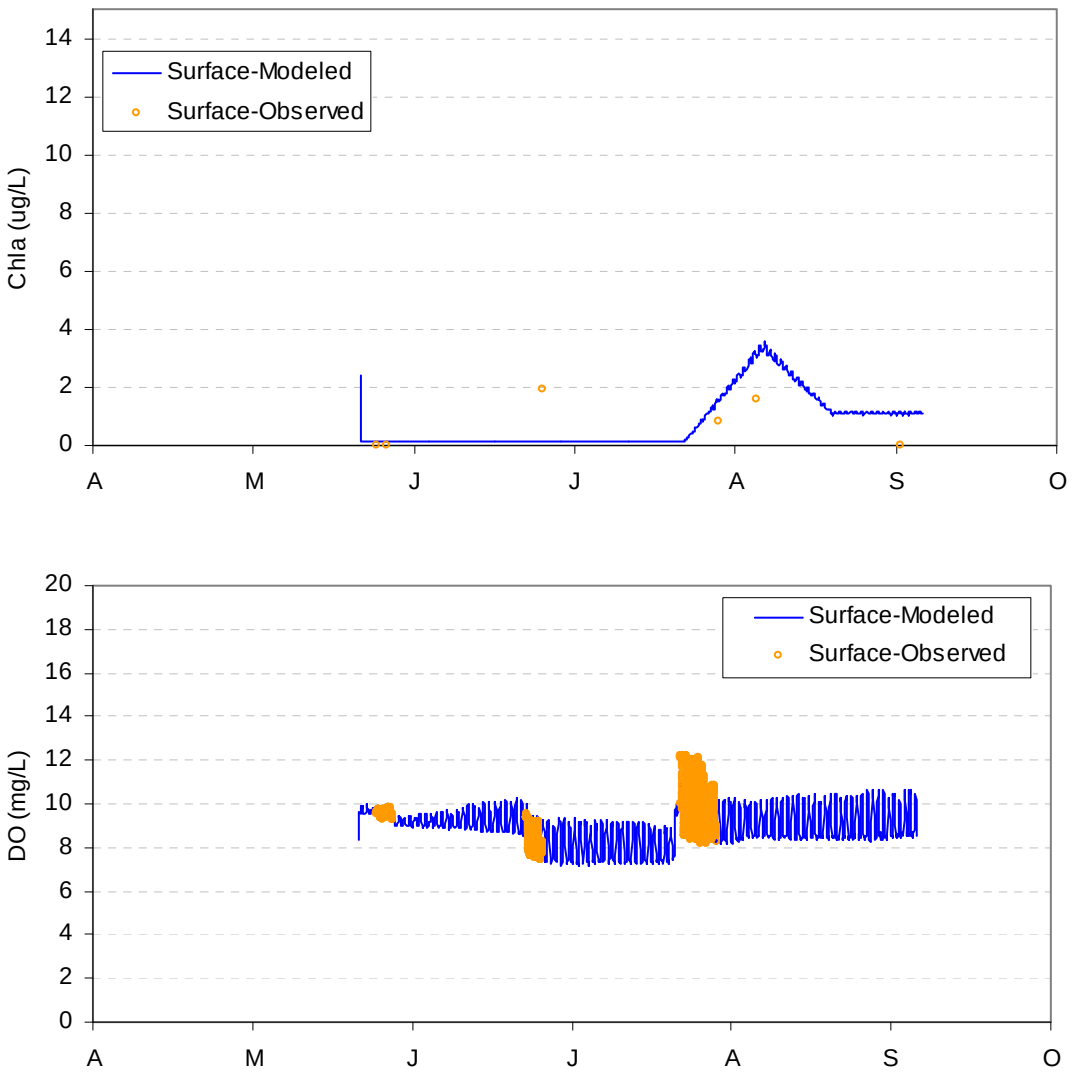


Figure L-9. Chlorophyll *a* and dissolved oxygen calibration results at the upper estuary water quality station (2004).

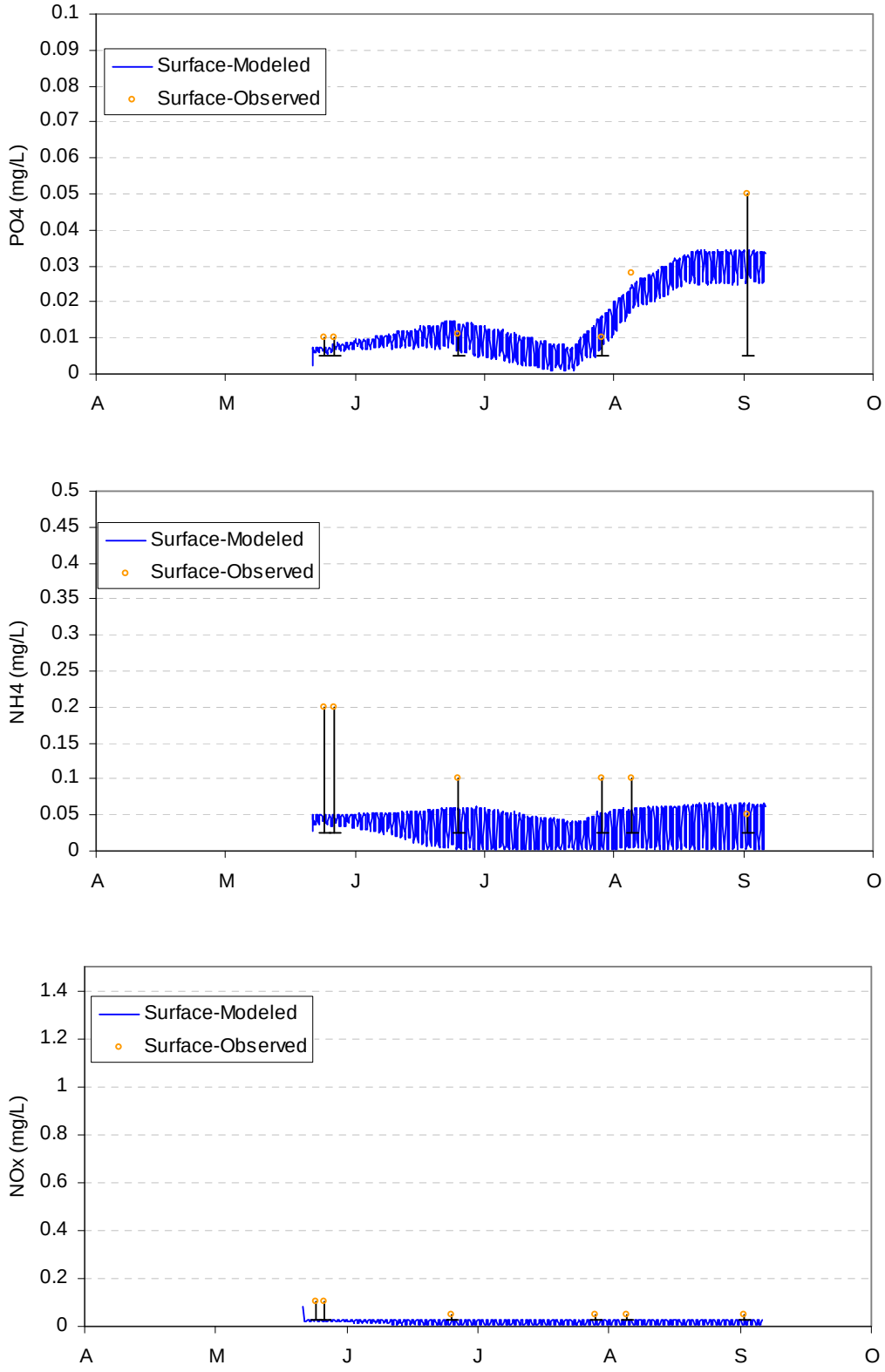


Figure L-9 (continued). PO_4, NH_4, and NO_x calibration results at the upper estuary water quality station (2004).