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Klamath River Pollutant Reduction Workshop–Information Packet

PREPARED FOR

California State Coastal Conservancy

With Support From
PacifiCorp

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Images of the Klamath Basin (from upper left to lower right) courtesy of Jan Tik, Kimberly Trimiew, Stillwater Sciences, and Kunal Mukherjee.

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1 WORKSHOP OVERVIEW AND INTRODUCTION

On February 18, 2010, the United States, the States of California and Oregon, PacifiCorp, tribal nations, and a number of other stakeholder groups signed the Klamath Hydroelectric Settlement Agreement (KHSA). The KHSA lays out the process for additional studies, environmental review, and a determination by the Secretary of the Interior regarding whether removal of four dams owned by PacifiCorp on the Klamath River (i.e., Iron Gate, J.C. Boyle, Copco 1, and Copco 2 dams) will advance restoration of the salmonid fisheries of the Klamath Basin, and is in the public interest (which includes effects on local communities and tribes).

The KHSA includes provisions and detailed actions (called Interim Measures) for the interim operation of the dams and mitigation activities prior to removal of the dams or the termination of KHSA. One of the measures–Interim Measure 10 (IM 10)–states that PacifiCorp shall provide one-time funding to convene a basin-wide technical workshop to evaluate approaches for improving water quality in the basin. The Klamath River Pollutant Reduction workshop (hereafter called workshop) will inform decision-making for IM 11, with a focus on nutrient reduction approaches in the basin including constructed wetlands and other treatment technologies and water quality accounting. The workshop is a joint effort, with PacifiCorp meeting its IM 10 commitment, the California Coastal Conservancy providing matching funds and project oversight, and the California Water Board Academy providing the workshop venue and logistics.

PacifiCorp, the North Coast Regional Water Quality Control Board (NCRWQCB), and Oregon Department of Environmental Quality (ODEQ), have formed a steering committee (project Steering Committee) to guide workshop development, including the format, agenda, composition of expert panels, and follow-up activities. The purpose of the workshop is to identify the technologies and strategies that will provide a clear working framework to reduce nutrient and organic matter loads to the Klamath River and improve water quality conditions within the Klamath Basin. Based upon the outputs from the workshop, highly ranked options and projects will subsequently be presented in a planning level document as a conceptual level feasibility analysis, which will serve as a guide for further development of more formal engineering feasibility analyses. While the number of potential solutions to water pollution challenges in the Klamath Basin is large, the project Steering Committee has chosen to evaluate a subset of possible solutions for the workshop; projects within this subset represent technologies and improvement strategies that have demonstrated success in other systems challenged by nutrient pollution and that would be applied at a similar scale in the Klamath Basin.

This pre-workshop information packet provides background information on the basin necessary to inform the design of water quality improvement projects, criteria for evaluating projects, and brief descriptions of large scale pollutant removal technologies to be evaluated at the workshop. For broader context, a summary of relevant nutrient reduction programs that have been applied either regionally or nationally, as well as estimated costs of these large-scale programs, are provided in Appendix A.

Invited experts and workshop participants are strongly encouraged to review the information presented in this information packet prior to the workshop. Invited experts are also encouraged to bring to the workshop additional nutrient reduction alternatives. Some of the invited experts will be asked to prepare presentations on pollutant reduction scenarios and related topics. Following the presentations by experts, workshop participants will be organized into small break out groups

that will be charged with evaluating and ranking the different project types and, ultimately, using different project types to design a basin-scale pollutant removal system as part of "design charrette" (i.e., a collaborative session in which a group of experts drafts a solution to a design problem and then shares the outcome with the larger group to facilitate broader dialog). Outcomes of the small group evaluations and the design charrette will help inform potential funding decisions regarding future development of engineering feasibility analyses.

2 PROJECT EVALUATION CRITERIA

Project evaluation criteria for the workshop have been developed in an attempt to provide a consistent basis for evaluating and ranking water quality improvement projects selected for conceptual-level consideration by the Steering Committee. Pollutant reduction project evaluation criteria are relatively simple due to the following:

- IM 10 is the first step in a multi-step project development process;
- the candidate project information is somewhat limited and may not uniformly support complex evaluation queries;
- time and effort that can be reasonably asked of volunteer experts;
- some of the proposed projects are innovative, with few comparable examples.

Context is extremely important when anticipating the degree of future success of any proposed solution to poor water quality conditions. The ultimate performance of the water quality improvement projects considered in this workshop is necessarily linked to the reach-scale or water-body-specific locations. Each nutrient reduction strategy will be evaluated and ranked at these locations (i.e., Upper Klamath Lake, Keno Reach) within the Klamath Basin. It is likely that the current state of the science and our understanding of the water quality conditions, dynamics and improvement potential in the Klamath Basin will be the limiting factors in the evaluation process.

Evaluating pollution control strategies is a challenge that can be met using multiple criteria analysis (MCA), which is "a framework for ranking or scoring the overall performance of decision options against multiple objectives." See Hajkowicz and Collins (2007). Stated simply, MCA rates various 'solutions' against project objectives, and delivers a 'score' that enables a rational and defensible decision. MCA analysis proceeds by creating a matrix of potential solutions or decision options transposed against a set of evaluation criteria. Structurally, the approach lends itself well to a simple spreadsheet format.

Evaluation criteria are typically derived from project objectives, with workshop objectives including the following:

- Reduce seasonal concentrations of nutrients (nitrogen, phosphorus)
- Improve overall water quality (e.g., dissolved oxygen, pH, temperature, turbidity/ total suspended solids)
- Allow the system to support designated beneficial uses

Evaluation criteria can be qualitative or quantitative. An example of an explicit quantitative criterion is cost per unit nutrient removed. The units, and the overall rank for any given quantitative criterion, should be consistent and explicitly expressed across technology comparisons. The advantage of quantitative criteria is the inherent numeric ranking reflected in the results. However, workshop participants should avoid spending valuable discussion time

discerning the small, and often meaningless, difference between exact numerical values, since the criteria are necessarily derived using a set of assumptions possessing some degree of uncertainty.

Alternatively, qualitative metrics such as 'improved', 'no change', or 'de minimis' may be more appropriate, especially when the criteria are more global and less easily quantified. For the purposes of the workshop, the Steering Committee has more often chosen a semi-quantitative scoring approach using high/medium/low (H/M/L) rankings, or a simple +/-/0 ranking. Though this scheme does not provide numeric rankings, it does allow for clear differentiation between water quality improvement projects and supports the streamlined group discussions envisioned for the workshop breakout sessions.

Individual evaluation criteria are rarely equally important to the overall project rank and therefore must be appropriately weighted. Both subjective and objective factors affect the weighting. For example, improved sucker recruitment may be a strongly weighted criterion for individuals focused on improvements to fisheries populations, while reduced nutrient concentrations would be higher for individuals focused more directly on water quality improvements, even though both criteria are related in the ecosystem. There are a variety of methods that can be used to assign weights to individual criteria. The approach adopted for the workshop is to tabulate and analyze the raw results, with a possible weighting step imposed later by the Steering Committee with expert input.

Finally, we recognize that decision is ultimately a human judgment that can be informed, but not replaced, by an MCA approach. There are inevitably relevant issues that a matrix decision making approach cannot adequately capture or address. To address this, the Steering Committee has provided an 'other' category for unforeseen or unique project qualities that have no explicitly stated criterion. The Steering Committee has also included a short set of open-ended questions that encourage brief narratives to more fully explain project qualities, identify bias, and assist with weighting results. Some of the open-ended questions may not be in the expert's fields or may be uncomfortably subjective; participants may skip these questions if they wish.

As with all decision-making frameworks, the process is iterative and the journey through evaluation and ranking will almost certainly identify new possible solutions, approaches and values.. To conserve effort, the pre-workshop background information provides objective performance data for a subset of possible evaluation criteria, which limits the number of qualitative criteria evaluations.

The first project objective, nutrient reduction, can be evaluated with objective and quantitative criteria. Estimated nutrient reduction performance is provided in Sections 4.1 to 4.6 and can be used by workshop participants provided the units of removal are the same. For example, a particular treatment wetlands option might provide an anticipated removal rate of 1.9 metric tons (MT) phosphorus over the lifetime of the project. Workshop participants can question the derivation of these values and provide their own confidence intervals and narrative explanations, as needed.

For the second objective, improvement of overall water quality, the criteria are subjective (e.g., overall effect on dissolved oxygen, pH, water temperature, total suspended solids [TSS]/turbidity algal growth,) and can be ranked as high/medium/low. Participants can refine responses to subjective criteria through narrative expansion. In this way, attendees can address the more subjective criteria in the workshop, and help the Steering Committee assign appropriate weights to criteria.

The third objective, support of existing beneficial uses, has objective and subjective elements that can be handled in a manner similar to the second objective. Participants can provide objective metrics about the effects on existing beneficial uses and/or provide explanatory comments, which will provide more focus and improve the overall evaluation under this objective.

Some other important concerns are globally applicable to the objectives (e.g., what it costs, how fast it can be implemented, etc.). The Steering Committee has considered a set of 'Global Criteria' that allow workshop participants to take a broader view of a particular technology, outside of the workshop objectives. As mentioned above, a set of open-ended questions will invite a narrative response from each working group. These narratives will be useful to capture unexplored aspects of a project and other considerations that are tangential to the core project objectives, for example, what is the 'green score' of the chosen technology? The global considerations and narrative sections are in some ways duplicative of the more targeted evaluation criteria, but they offer a summary view of the technologies and projects and can assist the Steering Committee in collating results and weighting individual concerns. Criteria considered by the Steering Committee are presented in Appendix B and will be further refined prior to the workshop.

3 BASIN OVERVIEW

3.1 Geographic and Physical Setting

The mainstem Klamath River traverses approximately 409 km (254 mi), originating in Upper Klamath and Agency Lakes within the Cascade Mountains of Southern Oregon and flowing southwest through the Northern California Coast range, to its terminus at the Pacific Ocean (Figure 1). With an overall watershed area of 40,720 km^2 (approximately 15,722 mi^2), the Klamath River is second only to the Sacramento River on the basis of annual flow volume in California (Kruse and Scholz 2006). It supports important runs of anadromous fish species including coho and Chinook salmon and steelhead trout, as well as resident rainbow trout (NRC 2004).

The Klamath River is unique to the Pacific Northwest region because the upper-most portion of the watershed is comprised of a low-relief volcanic plateau, while the lower portion of the watershed traverses steeper, mountainous terrain. The Upper Klamath and Agency Lake system, Oregon (collectively referred to as the Upper Klamath Lake system), is located in the uppermost portion of the Klamath River Basin in the Cascade Mountains, where it drains an interconnected series of marshes and streams from a catchment of approximately 9,842 km^2 (3,800 mi^2). Much of the upper basin is covered with volcanic pumice deposits derived from the formation of the Crater Lake caldera (Johnson et al. 1985). The area is also dominated by basalt flows and north to north west-trending fault lines which have formed large, shallow, graben lakes (i.e., formed by tectonic faulting) such as Upper Klamath Lake (Dicken 1980).

The Upper Klamath Lake system serves as a focal point of the Klamath River Basin due to its function as the receiving body of water for upper watershed areas, and as the headwaters for the Klamath River proper. Major tributaries to the Upper Klamath Lake system include the Wood, Williamson, and Sprague rivers (Figure 1). Link River Dam is located at the outlet of Upper Klamath Lake and controls water elevation in the lake (FERC 2007). From Link River Dam, water flows through the short stretch of Link River (1.9 km [1.2 mi]) and into Lake Ewauna and the Upper Klamath River at the Keno Impoundment.

Downstream of Lake Ewauna, the mainstem Klamath River has five major dams, including Keno Dam, J.C. Boyle Dam, Copco 1 and 2 Dams, and Iron Gate Dam. These dams are part of the Klamath Hydroelectric Project (see Section 3.5). The Klamath River downstream of Iron Gate Dam and extending to the Klamath River estuary is unregulated. The largest tributaries to the mainstem Klamath River in the lower basin are the Shasta, Scott, Salmon, and Trinity rivers (Figure 1).

3.2 Climate and Climate Change

Annual precipitation in the Klamath Basin ranges from 15–150 inches per year, with drier conditions (15–40 inches per year) at the higher elevations of the upper basin (i.e., greater than 1,219 m [4,000 ft]) and wetter conditions (40–150 inches per year) at lower elevations and in coastal areas of the lower basin (Figure 2). The upper basin receives rain and snow during the late fall, winter and spring, with most winter precipitation falling as snow. The Upper Klamath Lake system freezes over intermittently from November through February, and can remain frozen for several months duration. Midwinter rains can occur in the lower-elevation areas (i.e., 914–1,219 m [3,000–4,000 ft]) of the upper basin, but due to a rain shadow effect of the Cascade Mountains annual rainfall is variable throughout this portion of the Klamath Basin (Risley and Laenen 1999), and ranges from a mean annual precipitation (1961–1990) level of 166.1 cm (65.4 in) at Crater Lake National Park in the Cascade Range to 34.3 cm (13.5 in) at Klamath Falls (Gannett et al. 2007) (see also Figure 2).

In the lower basin, precipitation is also variable, although it tends to be dominated by rainfall during winter months, except at the higher elevations of the Klamath and Siskiyou mountain ranges where wintertime snowfall occurs. Throughout the basin, peak stream flows generally occur during snowmelt runoff periods in late spring/early summer. After the runoff period, flows decrease in the late summer/early fall.

Climate change is expected to result in a wide variety of effects in the Klamath River Basin. In general, climate model predictions for the Pacific Northwest and Northern California include the following (Barr et al. 2010, OCCRI 2010, USBR 2011a):

- Increased average ambient air temperature
- Increased number of extreme heat days
- Changes to annual and seasonal precipitation, including increased frequency and length of drought, diminished snow pack and more winter rain
- Increased heavy precipitation
- Changes to annual stream flow and groundwater hydrology
- Changes in water quality
- Vegetation changes

Numerous climate change models predict that air temperatures in the Pacific Northwest and the Klamath Basin will increase over the next 50–80 years at a rate of 0.04 to 0.06ºF per year, such that by the end of the 21st century, average annual air temperatures will increase by 4–7ºF (USBR 2011a, OCCRI 2010, Barr et al. 2010) (see also Appendix C, Figure C-1). Mean precipitation is also projected to change gradually from existing precipitation averages. By the mid-21st century (2035–45), annual precipitation projections in the Klamath Basin exhibit a wide range, from an 11% reduction to a 24% increase overall, depending on the climate model (Barr et al. 2010). Changes in seasonal precipitation are anticipated to result in winter rain replacing winter snow,

and earlier and higher winter and spring (December–March) stream flows and lower late spring and summer (April–July) stream flows (Barr et al. 2010, USBR 2011a). Simulated changes in projected decade-mean runoff in the Klamath Basin follow this same pattern, but vary by sub-watershed (USBR 2011a). Projected changes to groundwater hydrology under climate change may also decrease late summer stream flows in the Klamath River Basin, including alterations of the timing and amount of recharge, increases in evapotranspiration, lowering of the groundwater table, and increased pumping demand (OCCRI 2010, USBR 2011a).

As with stream flow predictions, groundwater effects are expected to vary by sub-watershed (USBR 2011a).

Changes to air temperatures and precipitation and flow patterns will result in changes to water temperatures in the Klamath Basin. Increasing air temperatures and decreasing summer flows in the Klamath River Basin would be expected to cause general increases in summer and fall water temperatures on the order of 3.6–5.4°F for the period 50 years into the future (Bartholow 2005, Perry et al. 2010).

3.3 Vegetation

Forestland covers much of the Klamath Basin (Figure 3). In the upper basin, vegetation is predominantly forest (~70%) and scrub/grassland vegetation types (~14%), with the latter often used as rangelands (Figure 4). Relatively large extents of cultivated croplands occur in low-lying former wetland areas that surround much of the present Upper Klamath Lake system (see also Section 4.1.1), including the Wood River, Lost River, and Upper Klamath River downstream of Upper Klamath Lake. Cultivated croplands also occur in valleys along the Shasta River and Scott River in the lower Klamath Basin (Figure 3).

3.4 Land Use and Ownership

The largest area of current land use in the Klamath Basin is private and public forest (ODEQ 2002). Other major land uses include rangeland and agriculture, with relatively smaller extents of urbanized (developed) areas, the primary one being the city of Klamath Falls on the southeastern shore of Upper Klamath Lake (Figure 4). In the upper Klamath Basin as whole, agriculture currently occurs on roughly 5% of the drainage (ODEQ 2002); however, the extent of agricultural land use in the Lost River sub-basin is much greater, and in the case of the Wood River, agriculture is the primary land use (Figure 4).

Historical land use activities included agriculture, ranching, logging, wetland draining and water diversions, all of which have altered seasonal stream flows and water temperatures, increased concentrations of nutrients (nitrogen and phosphorus) and suspended sediment in watercourses, and degraded other water quality parameters such as pH and dissolved oxygen concentrations. Historical diking and draining of wetlands has been extensive in the Klamath Basin; approximately 80% of the natural wetlands have been lost to other land uses, including agriculture. The current extent of wetlands surrounding Upper Klamath Lake and Lower Klamath Lake is significantly diminished over historical conditions (Figure 5). In the lower Klamath Basin, historical hillslope and in channel gold mining and extensive logging have also occurred, affecting water quality in many of the lower tributary basins.

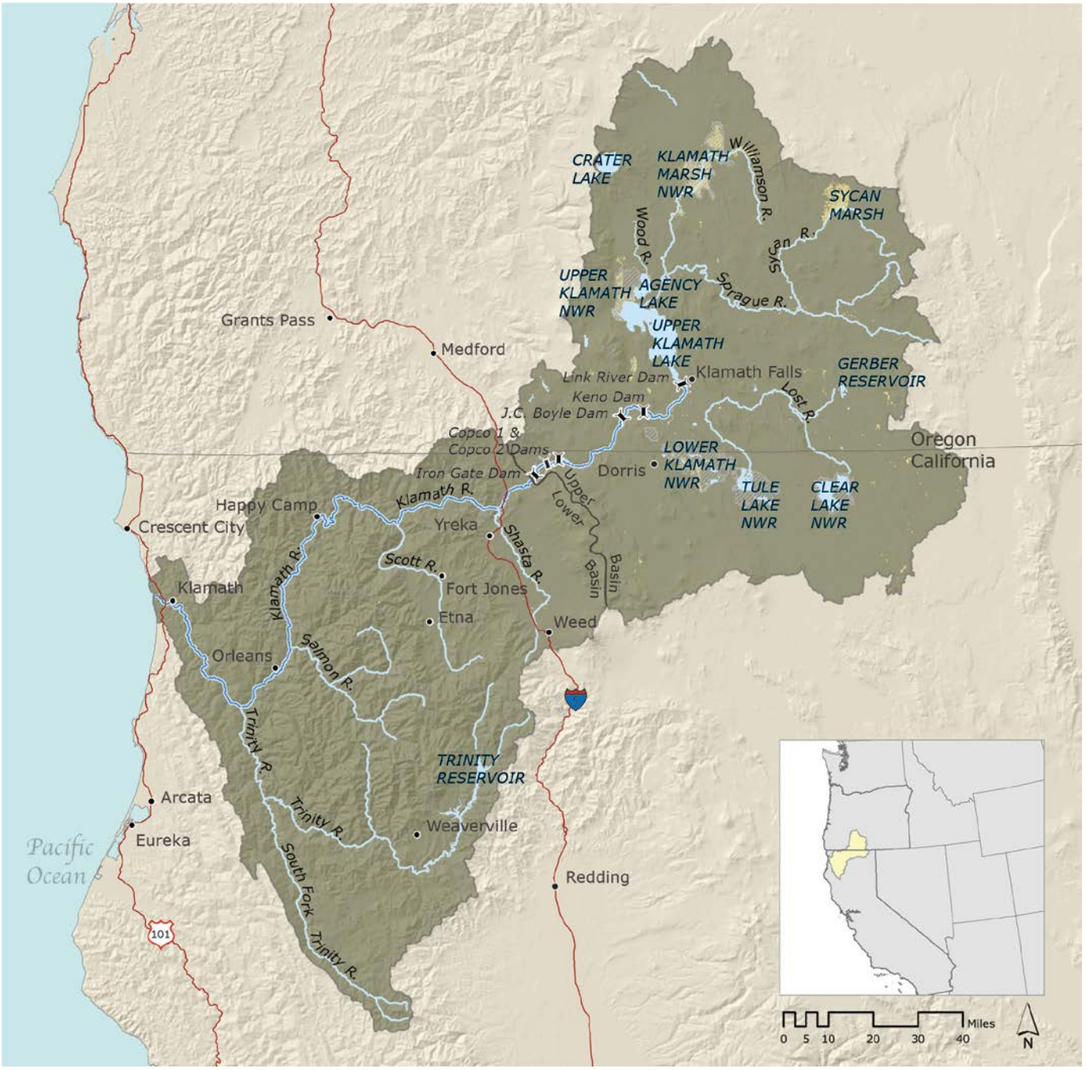

Figure 1. Klamath Basin major rivers, lakes, and Klamath River dams. Source: National Hydrography Dataset/USGS, Bureau of Transportation, National Atlas, USFWS.

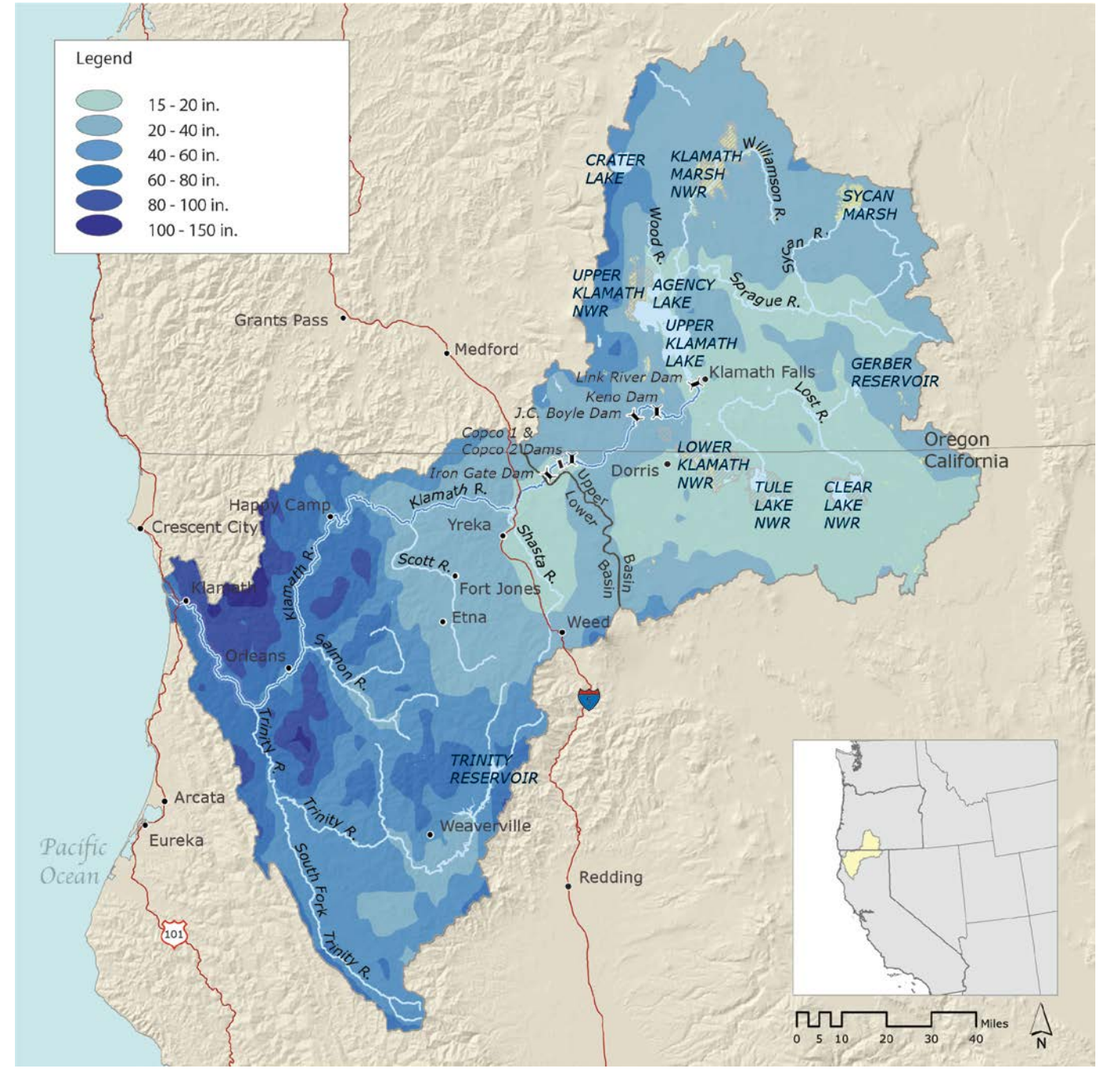

Figure 2. Klamath Basin average annual precipitation. Source: PRISM, National Hydrography Dataset/USGS, Bureau of Transportation, National Atlas, USFWS.

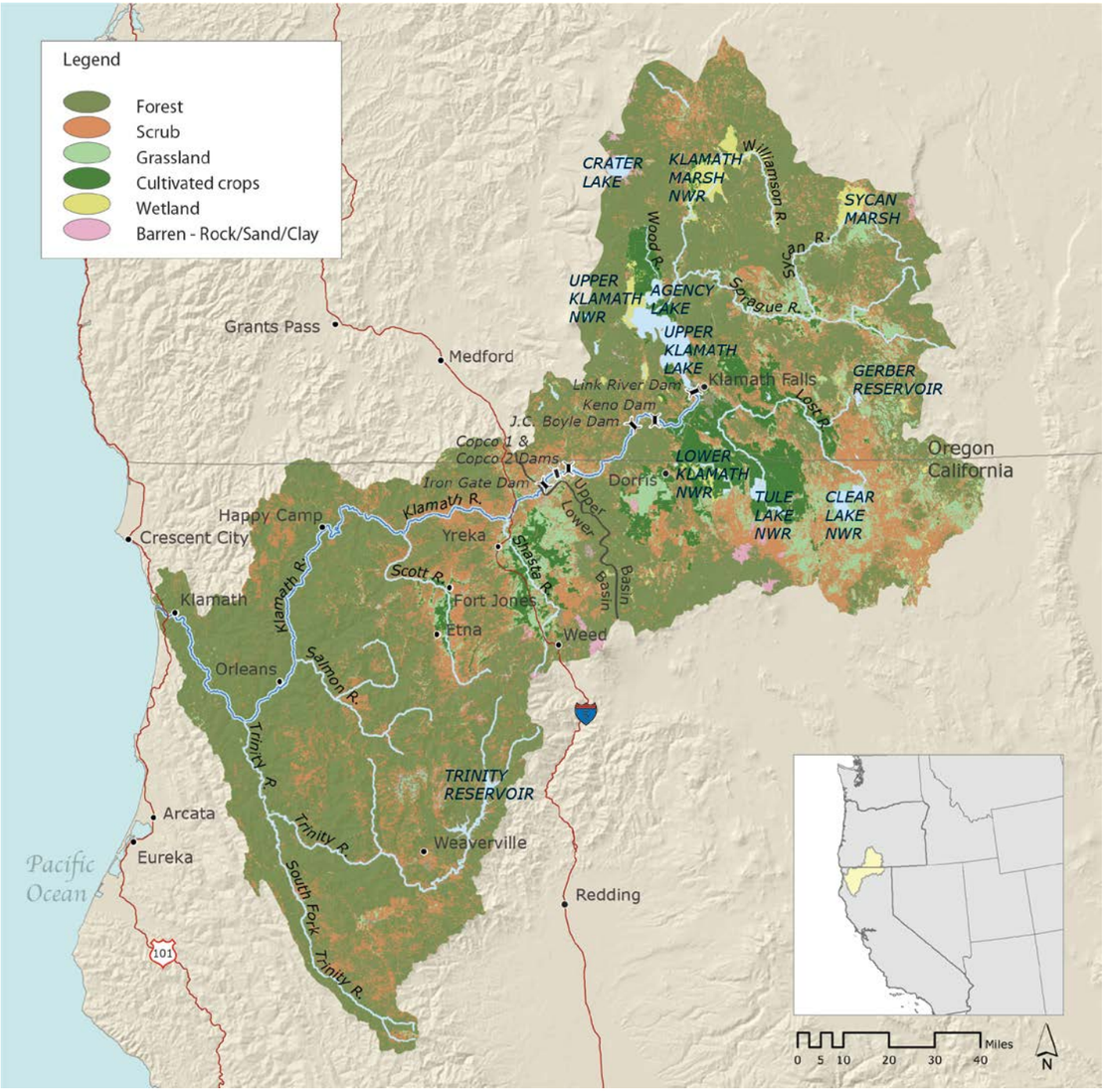

Figure 3. Klamath Basin vegetation. Source: National Hydrography Dataset / USGS, Bureau of Transportation, National Atlas.

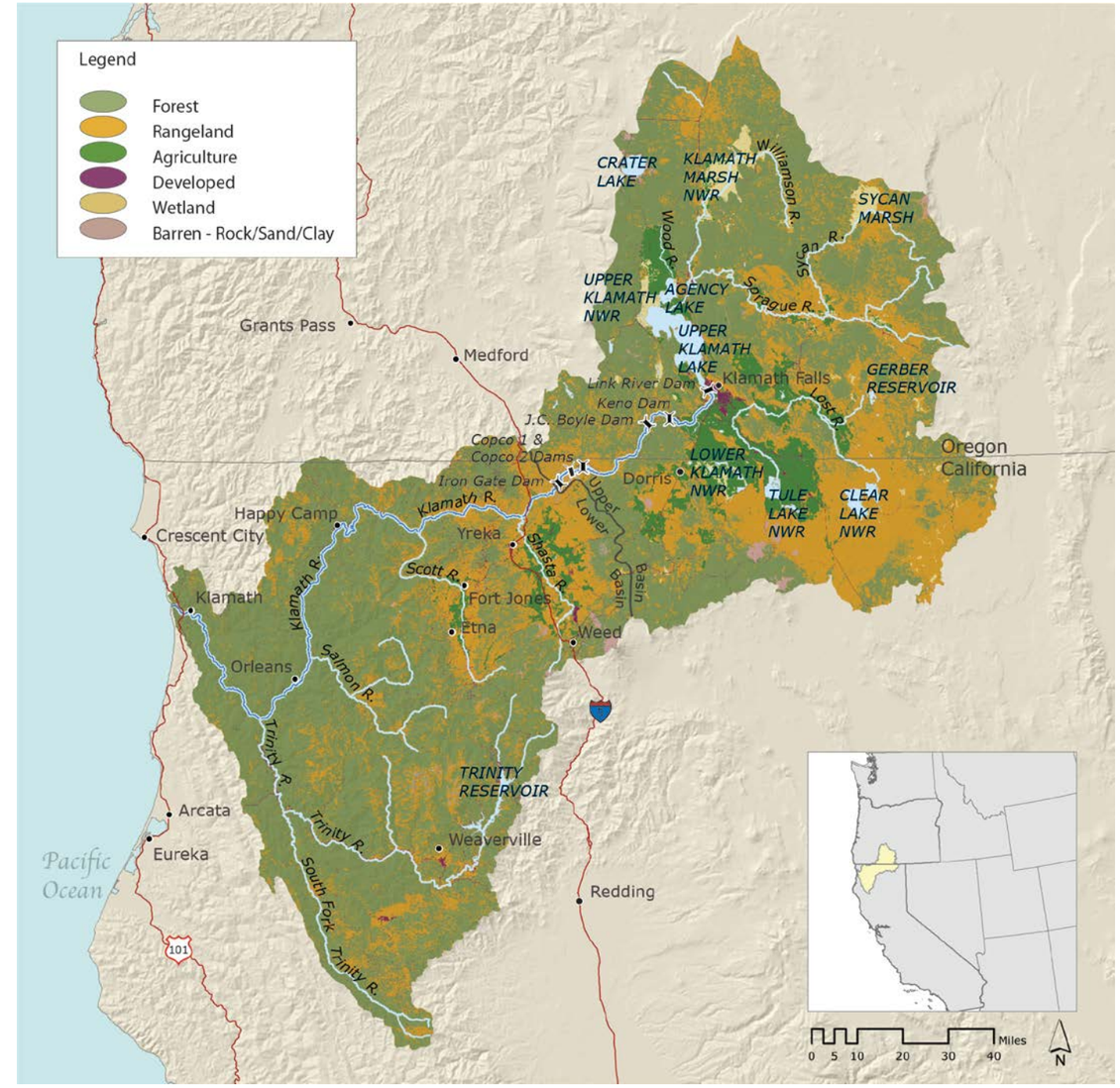

Figure 4. Klamath Basin land use. Source: USDA NRCS, USGS, National Hydrography Dataset / USGS, Bureau of Transportation, National Atlas, USFWS.

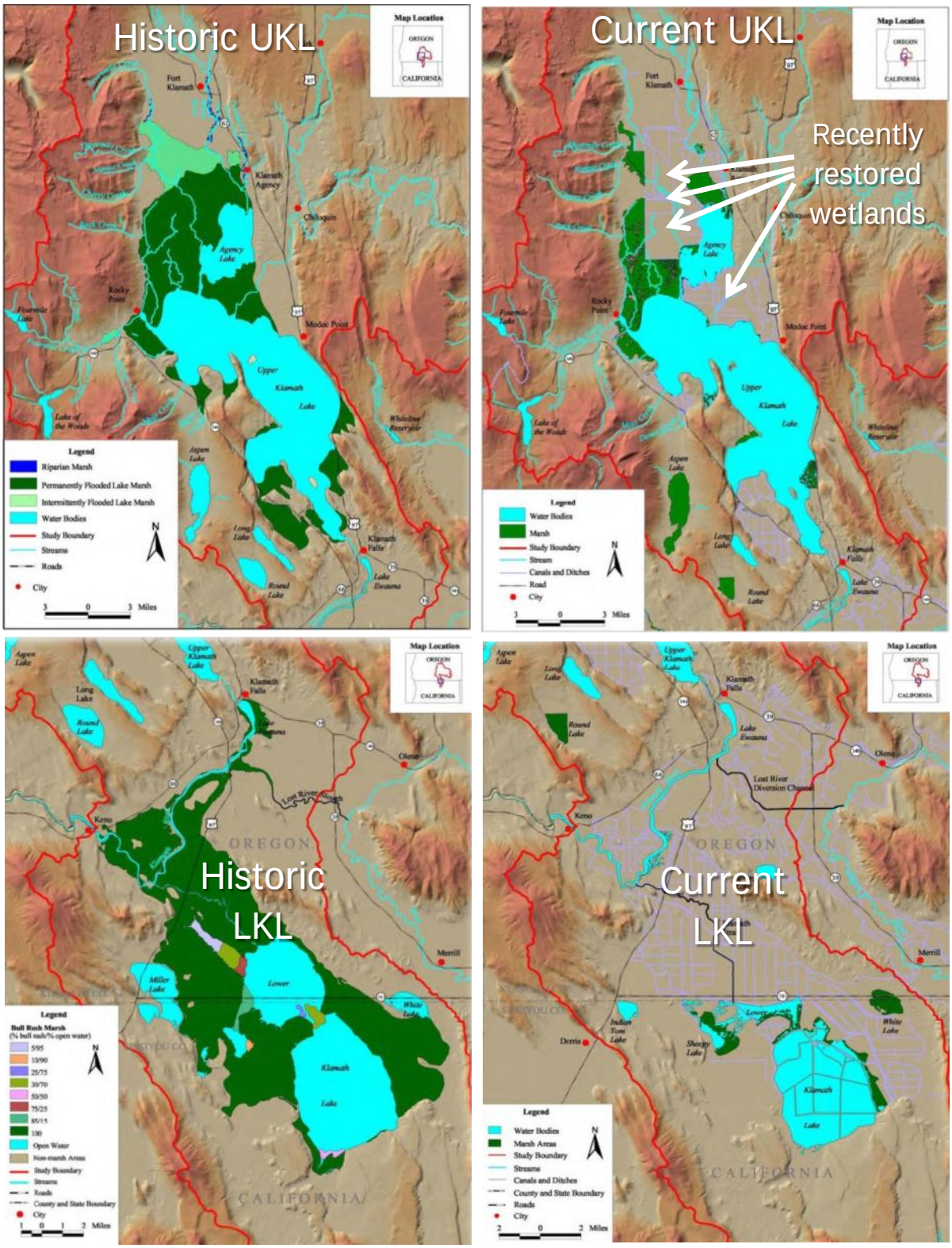

Figure 5. Historic Upper Klamath Lake (UKL) and Lower Klamath Lake (LKL) vs. current Upper Klamath Lake and LKL wetlands. Source: USBR (2005).

Currently, land ownership in the Klamath Basin is predominantly private and United States Forest Service (USFS). In the Upper Klamath Lake drainage, private lands account for roughly 42% and USFS accounts for 53% of the land area, respectively (see Figure 6 in ODEQ 2002). National Park and National Monument land accounts for roughly 3% of the total basin area, primarily Crater Lake National Park at the northern-western edge of the basin, but also Redwood National Park at the Klamath Estuary. Several National Wildlife Refuges make up just under 1% of land area, all of which are owned and managed by US Fish & Wildlife Service (USFWS). The Bureau of Land Management (BLM) and Tribal lands comprise 0.5% or less of the total basin area (Figure 6).

3.5 Irrigation and Hydroelectric Project Infrastructure

Water supply to the Klamath River is affected by historical conversion of wetlands to agricultural land uses, irrigation diversions, and importation of water from other basins (Hardy 1999, Hecht and Kamman 1996). During the spring and summer months, a significant amount of water from the Klamath Basin upstream of J. C. Boyle Dam is diverted for irrigation purposes by the USBR Klamath Project. The USBR Klamath Project supplies irrigation water for over 200,000 acres on approximately 1,400 farms locally in Oregon and California, and it includes a large network of irrigation canals and pumps as well as multiple dams and reservoirs (
Figure 7).

Computer models and historical data indicate that the construction and operation of the USBR Klamath Project has had a substantial effect on Klamath River flows (NMFS 2010). For example, at Iron Gate Dam for the 1961–2006 period, USBR calculated that the Klamath Project reduced median April flow from approximately 3,300 cfs to approximately 2,200 cfs (roughly 30% reduction), and median July flow from approximately 1,400 cfs to approximately 700 cfs (roughly 50% reduction) (NMFS 2010).

The Klamath Hydroelectric Project (Project), owned by PacifiCorp, is located within the high-gradient reaches of the downstream portion of the Upper Klamath Basin and generates an average of 716,820 megawatt-hours of electricity annually through the operation of a series of dams (FERC 2007). These dams include Keno Dam (River Mile [RM] 233.0), J.C. Boyle Dam (RM 224.7), Copco 1 (RM 198.6) and Copco 2 Dams (RM 198.3), and Iron Gate Dam (RM 190.1) (Figure 8). These dams were developed largely for hydropower generation and provide limited storage capacity, mainly in Copco 1 and Iron Gate Reservoirs (total capacity 1.09×10^8 m^3 [88,160 acre-ft], active capacity 1.45×10^7 m^3 [11,749 acre-ft]) (FERC 2007).

3.6 Hydrology

The primary sources of water to the Klamath River and the Upper Klamath Lake system are the Williamson River (including its major tributary, the Sprague River) and the Wood River (Figure 1). The Williamson River and its tributaries provide between 46–53% of the total water inflow to the lake, and the Wood River contributes approximately 16–20% of the inflow and drains directly into the northern end of Agency Lake (USACE 1982, ODEQ 2002, Kann and Walker 1999, Walker et al. 2012). The Sprague River has larger flow seasonal fluctuations than the Wood and Williamson rivers (Figure 9). The remaining hydrologic inputs are from precipitation on the lake surface, springs, and inflows from several smaller streams and canals; Appendix C provides a detailed accounting of annual average inflows and outflows to Upper Klamath Lake over the 1992 to 2010 period (data from Walker et al. 2012). Season hydrological attributes for the Upper

Basin are contained in Walker et al. 2012). Groundwater dynamics are discussed in detail in Gannett et al. (2007).

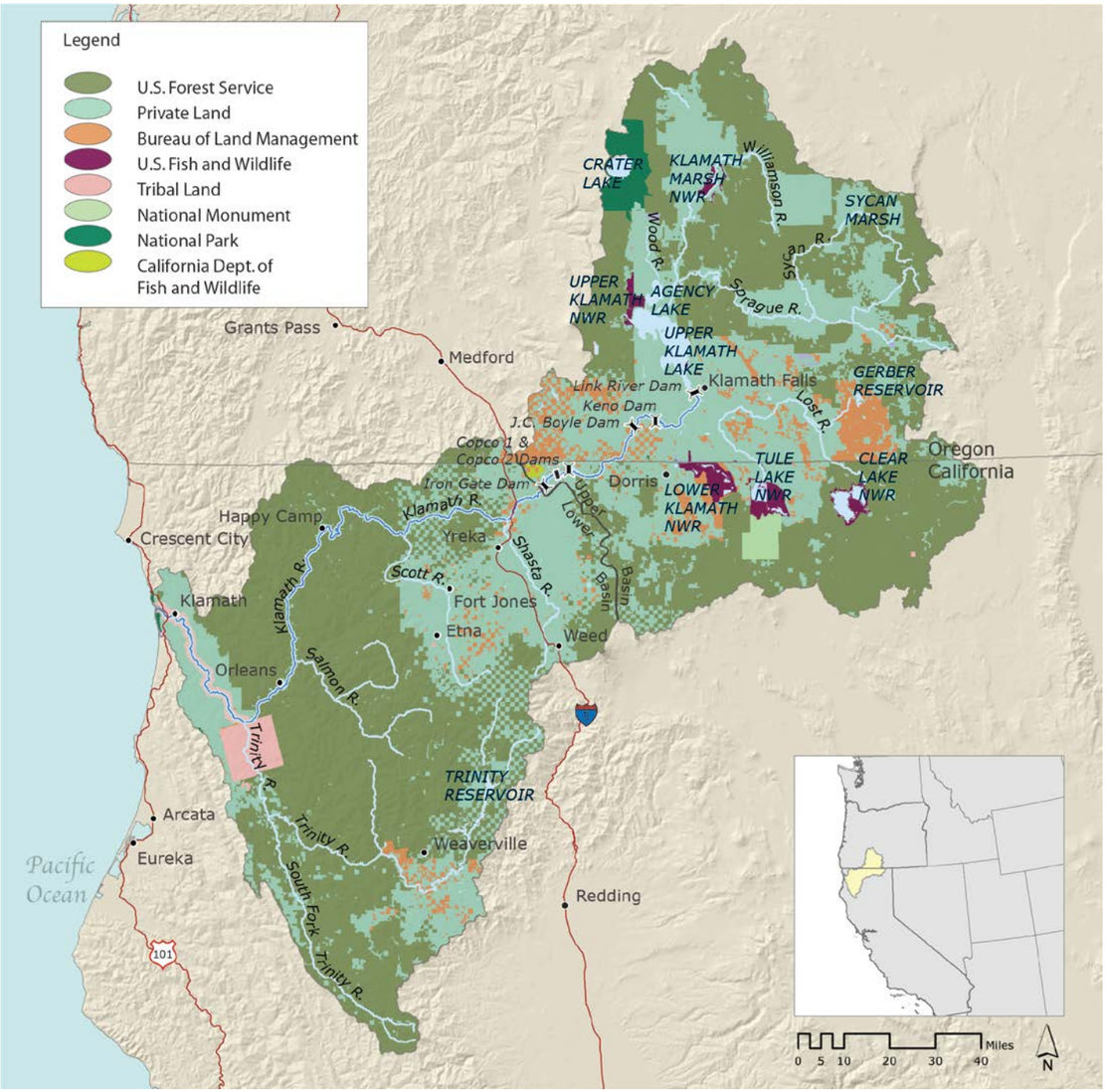

Figure 6. Klamath Basin land ownership. Source: OR BLM, California Resources Agency, National Hydrography Dataset / USGS, Bureau of Transportation, National Atlas.

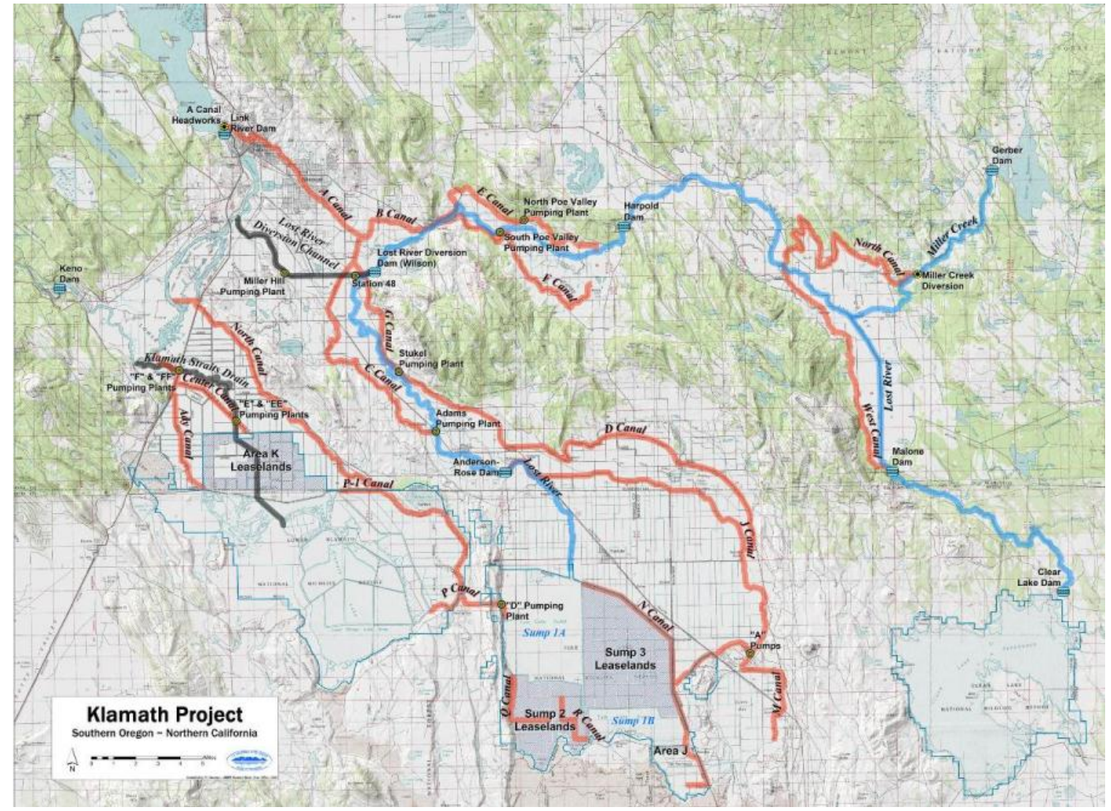

Figure 7. Network of irrigation canals in the USBR Klamath Project.

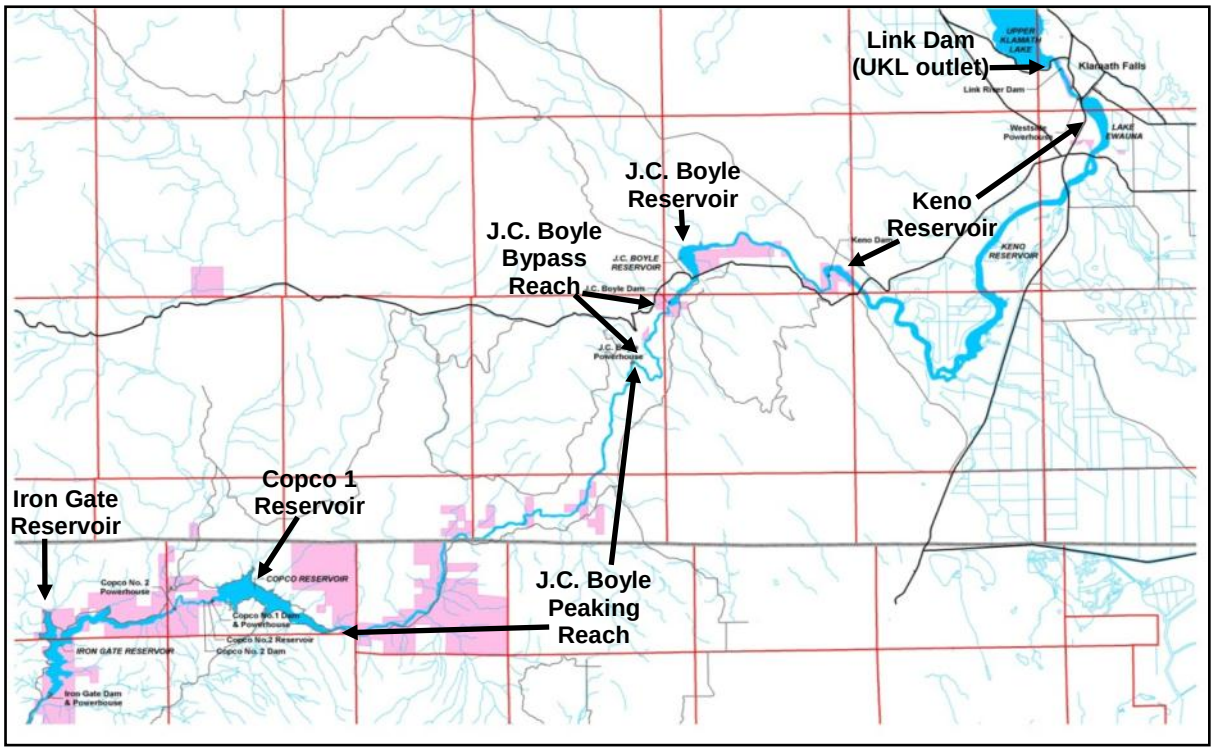

Figure 8. Klamath Hydroelectric Project from Link Dam to Iron Gate Dam. Source: PacifiCorp.

Natural streamflow patterns into and out of Upper Klamath Lake have been significantly altered by irrigation and hydroelectric power diversions, irrigation returns, deforestation, and by the construction of the Link River Dam at the lake outlet. Water is released from the dam for hydropower production, to provide flows for the Link and Klamath Rivers, and for irrigation uses through the A-Canal (see Figure 7). Concurrent with the construction of the Link River Dam, channels cut into the natural basalt sill at the outlet of Upper Klamath Lake (Figure 10) allow greater lake level fluctuation and use of additional water volume to supply irrigation water to USBR Klamath Project farmland. Unlike typical dams which increase water level and storage, the Link River Dam does not increase the maximum pre-dam elevations, but permits greater control over storage by allowing higher lake elevations to be maintained longer into the early summer period, and allows lake levels to be lowered below pre-dam natural elevations (see Appendix C, Figure C-2).

There are numerous agricultural diversions and drains along Keno Reservoir (Figure 7). The two primary drains returning water to Keno Reservoir are the Klamath Straits Drain (KSD), which operates year-round, and the Lost River Diversion Channel, which diverts flow away from the reservoir during irrigation season but discharges to the reservoir for the remainder of the year. These drains carry irrigation return flows from agricultural lands in the Lost River watershed, including the Lost River, Tule Lake, and Lower Klamath Lake (Figure 7). Water is also diverted from the Klamath River into the North and Ady Canals (Figure 7).

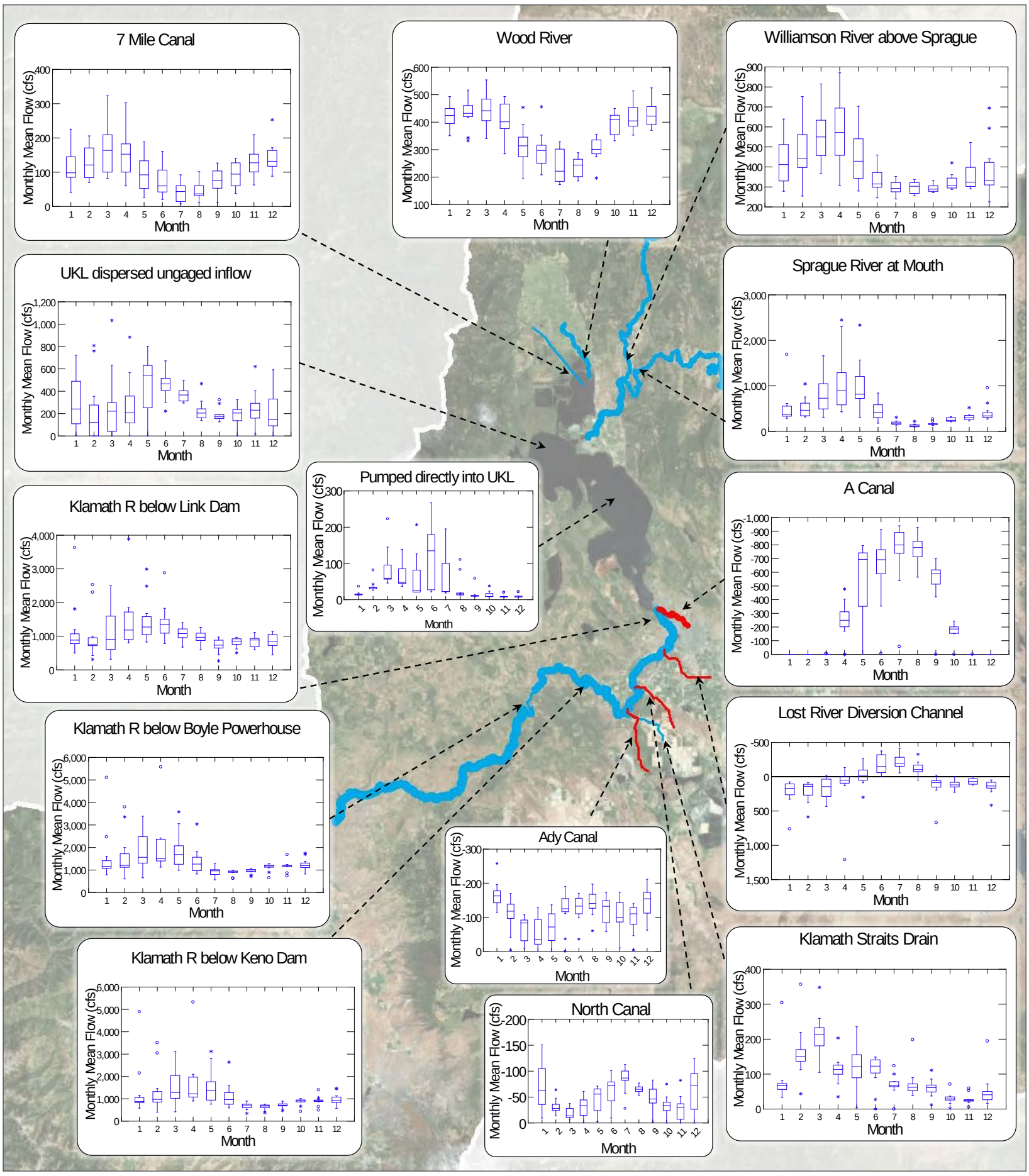

Figure 9. Box plots showing mean flows by month for hydrologic years 2000-2010 for key locations in the Upper Klamath Basin. In box plots, negative values indicate diversions from the Klamath River. In map, mainstem Klamath River and tributaries are blue, diversions are red, and link thickness is approximately proportional to mean May through September flow. Data from Walker et al. (2012), USGS gages, and USBR.

Figure 10. Photo of Link River Dam looking upstream towards Upper Klamath Lake showing the natural rock reef and two artificial channels that were cut through the reef to allow Upper Klamath Lake to be lowered below the historic reef elevation. Source: USFWS 2008.

With the exception of the J.C. Boyle bypass reach where water is temporarily diverted into a canal before being returned, the Klamath River's flow increases with distance downstream of Keno Dam due to tributary inputs and a lack of substantial mainstem agricultural diversions. The river receives water from Spencer Creek in J.C. Boyle Reservoir, high-volume springs below J.C. Boyle Dam, and other tributaries including Shovel, Jenny, and Fall creeks. Many additional tributaries enter the river between Iron Gate Dam and the Pacific Ocean. Some of these tributaries, including the Shasta, Scott, and Trinity rivers, have major water diversions while other tributaries are more pristine such the Salmon River and the many smaller streams that flow primarily through U.S. Forest Service lands (Figure 1).

3.7 Water Quality

The water quality concerns in the Klamath Basin are predominately distributed nonpoint sources of pollution rather than discrete point source pollution (ODEQ 2002). The following is a brief summary of reach-scale water quality summaries for the reaches that are the focus of the workshop.

3.7.1 Reach-scale water quality summaries

3.7.1.1 Upper Klamath Lake and major tributaries (Wood, Williamson, and Sprague Rivers)

Water quality in many tributaries to Upper Klamath Lake does not meet Oregon water quality standards (ODEQ 2002). Water temperature, pH and dissolved oxygen issues are related to riparian vegetation disturbance that reduces stream surface shading, channel widening (increased width to depth ratios due to factors such as loss of riparian vegetation), reduced flow volumes

(chiefly from irrigation) or increased high temperature discharges, disconnected floodplains which prevent/reduce groundwater discharge into the river, and increased erosional/nutrient inputs (ODEQ 2002).

Water quality in Upper Klamath Lake is poor during the summer and early fall, with large algal blooms that lead to high pH and un-ionized ammonia concentrations, depressed dissolved oxygen, and problematic levels of algal toxins. Land use within the tributary watersheds and management of lakeside wetlands has substantially increased the input of phosphorus into the lake, providing an essential nutrient for algal proliferation (ODEQ 2002). Anthropogenic land use activities are estimated to have contributed 38% of the lake's total external phosphorus load from 1992–1998 and 31% from 2008–2010 (Walker et al. 2012). The following paragraphs provide additional detail regarding nutrient inputs and water quality in Upper Klamath Lake, and further information is available in Appendix C.

By surface area, Upper Klamath Lake is one of the largest freshwater lakes (~267 km^2 [~66,000 ac]) in the western United States (Colman et al. 2004). It is also extremely shallow. The maximum depth of Upper Klamath Lake (ca. 17.7 m [58 ft]) occurs in a narrow trough extending along the west central shoreline of the lake between Eagle Ridge and Howard Bay; however, the mean summer depth of the lake is approximately 2 m (7 ft). Agency Lake is connected to Upper Klamath Lake at the north by the Agency Lake narrows and areas flooded during recent wetland restoration projects (Figure 11). The maximum depth of Agency Lake is ~4.5 m (~15 ft) (USACE 1982). The broad, shallow morphology affects water quality in the Upper Klamath Lake system, since it affects water temperatures, circulation, and vertical mixing patterns.

Figure 11. Wetland restoration activities on the Williamson River Delta allowing for greater connectivity between Agency Lake and Upper Klamath Lake.

Upper Klamath and Agency Lakes are hypereutrophic and are seasonally dominated by large blooms of the nitrogen-fixing cyanobacterium *Aphanizomenon flos-aquae* (AFA) (Kann 1998, Kann and Smith 1999). Numerous studies have documented that recurring algal blooms and their decline (Figure 12) are associated with periods of elevated pH, toxic levels of un-ionized ammonia, and depressed dissolved oxygen concentrations (Kann and Smith 1999, Perkins et al. 2000, Loftus 2001, ODEQ 2002, Kann and Welch 2005, Wood et al. 2006, Hoilman et al. 2008, Lindenberg et al. 2009, Kannarr et al. 2010, Kann 2011, Kann 2012a). Based on exceedances of water quality standards for dissolved oxygen, pH, and chlorophyll (algal biomass) (e.g., Loftus 2001), both lakes were designated as water quality limited for resident fish and aquatic life. More specifically these conditions have been linked to large die-offs and redistribution of the federally listed shortnose (*Chasmistes brevirostris*) and Lost River (*Deltistes luxatus*) suckers (Perkins et al. 2000, Kann and Welch 2005, Wood et al. 2006, Banish et al. 2009).

Eutrophic conditions in Upper Klamath Lake were described by observers in the 19th century and from an early limnological survey in 1913 (Kemmerer et al. 1923); however, increased eutrophication has occurred since that time (Bortelson and Fretwell 1993). Paleolimnological evidence indicates that AFA (as indicated by AFA akinetes preserved in lake sediments) did not appear in Upper Klamath Lake until the latter part of the 19th century and increased substantially

after that time (Bradbury et al. 2004, Eilers et al. 2004). During the past approximately 75 years, AFA has dominated the summer phytoplankton (Phinney et al. 1959, Miller and Tash 1967, Bond et al. 1968, Gahler 1969, Klamath Consulting Service 1983).

Watershed activities beginning in the late 1800s and accelerating though the 1900s included: (1) timber harvest, (2) drainage of wetlands, (3) agricultural activities associated with livestock grazing and irrigated cropland, and (4) hydrologic modifications such as water diversions and channelization (Snyder and Morace 1997, ODEQ 2002, Bradbury et al. 2004, Eilers et al. 2004). These activities are the main causes for the increased erosion and loading of nutrients (particularly phosphorus) from the watershed that are generally contemporaneous with the increase in Upper Klamath Lake's trophic state and shift to dominance by large blooms of blue-green algae (ODEQ 2002).

Figure 12. Satellite photo of Upper Klamath Lake showing typical summer lake-wide blooms of AFA. Source: NASA Earth Observatory http://earthobservatory.nasa.gov/IOTD/view.php?id=1743. Inset photo credit: Jacob Kann.

Excessive phosphorus loading linked to watershed development has been determined to be a key factor driving the massive AFA blooms that dominate Upper Klamath Lake. Based upon analysis of extensive water quality monitoring datasets and mathematical modeling of the lake phosphorus, algal bloom, and pH dynamics, the ODEQ (2002) determined that reduction of phosphorus loads from anthropogenic sources would be the most effective means of improving water quality conditions in the lake. This approach was supported by the elevated phosphorus concentrations and unit area phosphorus loads observed in discharges from pumped agricultural areas and tributary outflows, as compared with background levels observed in springs and tributary headwaters. This approach is further supported by recent pasture-level monitoring showing that first-flush irrigation events and storm events have the potential to export large

quantities of phosphorus from irrigated grazing land in the Upper Klamath Lake basin (Ciotti et al. 2010).

While other researchers hypothesized that reduced humic substances associated with the loss of wetlands may also play a role in the shift to AFA dominance, they ultimately concluded that the ODEQ (2002) approach of reducing algal biomass and improving water quality through phosphorus reduction was complementary but not inconsistent with their hypothesis (Milligan et al. 2009). The more recent conclusion by Milligan et al. (2009) is in contrast to the NRC (2004) who proposed the humic hypothesis as a distinct competing alternative to land-use mediated changes in the phosphorus environment and subsequent dominance by AFA.

Given the importance of phosphorus in controlling the severe algal blooms and adverse water quality conditions, phosphorus budgets (mass balance accounting of the various external phosphorus sources, outflows, storage, and net retention within the lake) have been computed to evaluate effects of land and water uses on water quality and to develop effective control programs for achieving water quality standards (Kann and Walker 1999, ODEQ 2002, Walker et al. 2012). The historical phosphorus budgets and phosphorus mass balance model were utilized by ODEQ (2002) to establish management targets for lake phosphorus concentrations and external phosphorus loads consistent with achieving lake water quality standards for algal biomass (expressed as chlorophyll a), and pH, as required under the Total Maximum Daily Load (TMDL) regulations established under the Clean Water Act. Achieving the loading target (109 metric tons/year of TP) would require a 40% reduction in external phosphorus load relative to the historical baseline, which is assumed to be 66 ppb (the average value measured in springs and other relatively un-impacted sources in the watershed).

Studies show that sediment recycling of phosphorus accounted for 61% of the total phosphorus loading to Upper Klamath Lake (Kann and Walker 1999, ODEQ 2002). Efforts to understand internal phosphorus recycling mechanisms have been made by a variety of researchers and indicates that mechanisms are a combination of algal translocation, diffusion, pH or anaerobic mediated release, microbial and macroinvertebrate metabolic cycling, bioturbation, and resuspension that operate at varying temporal and spatial scales (Barbiero and Kann 1994; Laenen and LeTourneau 1996; Kuwabara et al. 2007, et al. 2009, et al. 2012; Simon et al. 2009; Simon and Ingle 2011). Although this source of phosphorus plays an important role in driving AFA dynamics in Upper Klamath Lake, the ultimate sediment phosphorus source is watershed derived, reflecting prior loading inputs (Walker et al. 2012). Additional detail regarding sediment phosphorus recycling is provided in Appendix C.

Hydrologic and nutrient balances for Upper Klamath Lake were recently updated, indicating that the three major tributaries to the lake (Wood River, Sprague River, Williamson River) each contribute roughly 20% of the lake's external TP load, despite their relative differences in drainage area and somewhat lesser differences in flow volume to the lake (Figure 13). The updated hydrologic and nutrient balances provide context for nutrient management in the Upper Basin (Walker et al. 2012). These updated balances reflect watershed management and wetland restoration efforts including re-flooding of large drained wetlands on the periphery of Upper Klamath Lake (e.g., Wong et al. 2011, USBLM 2005, Carpenter et al. 2009, Duff et al. 2011) and watershed conservation projects to reduce water use and decrease nutrient export from grazed areas (e.g., GMA 2011a, 2011b). These updated balances indicate: internally generated TN loads (primarily atmospheric fixation of nitrogen by AFA) exceeded external inflow loads by more than 3-fold on an annual basis; a one-year lag in the response of the annual outflow TP loads to variations in the inflow TP loads; large amounts of internal nutrient generation in June and July

reflect high rates of phosphorus recycling from bottom sediments and nitrogen fixation (N-fixation) by blue-green algae; and, TP concentration ranges generally decreased in water years (WY) 2008–2010 and may reflect decreased transient nutrient releases from antecedent agricultural soils in the restored wetland areas (e.g., Wong et al. 2011). Walker et al. (2012) also show generally increasing flow-weighted-mean (FWM) TP concentrations from tributary headwaters to mouths that indicate anthropogenic loading (e.g., see Figure C-4, Appendix C), but also that several tributaries show a decreasing trend in the FWM TP during the 1992–2010 period. Current trends also reflect an expected lag in the response time of nutrients to restoration efforts, especially lake peripheral wetlands for which restoration of function is incremental (e.g., Wong et al. 2011). Additional detail regarding nutrient and hydrologic budgets is provided in Appendix C.

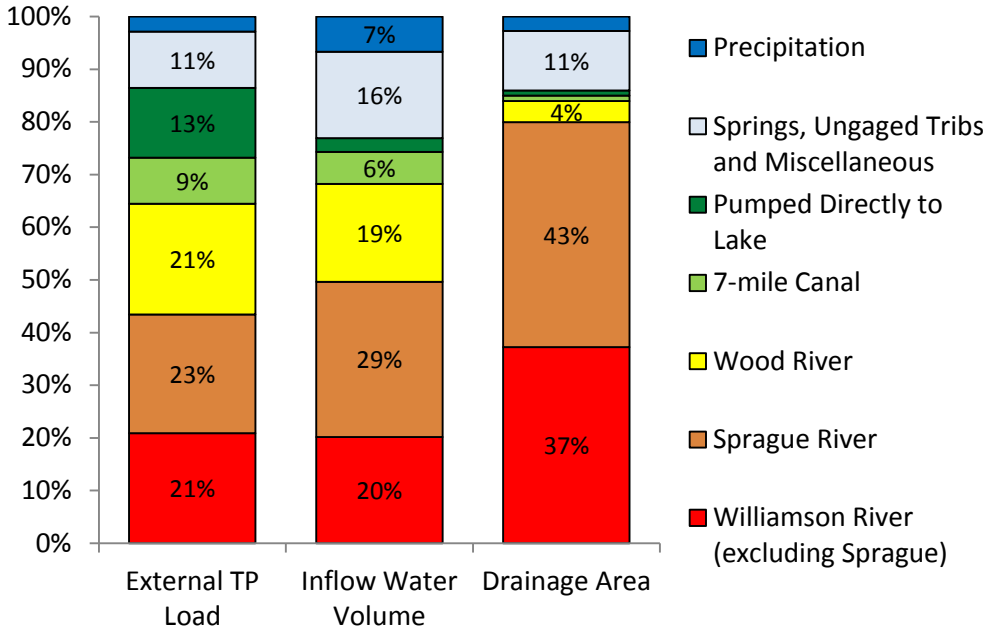

Figure 13. Relative contributions of tributaries and other sources to external total phosphorus (TP) load, inflow water volume, and drainage area to Upper Klamath Lake for hydrologic years 1992-2010. Data from Walker et al. (2012) Appendix E.

Watershed concentrations of nitrogen tend to be very low leading to low total and bioavailable N:P ratios in inflows to Upper Klamath Lake (<1 for bioavailable forms) with increased ratios in the Upper Klamath Lake outflow due partially to N-fixation occurring in the lake (e.g., Kann 2012b, Walker et al. 2012). More recent studies show that the range of N:P ratios in Upper Klamath Lake are clearly in the range (generally less than 10:1 by weight during the algal growing season for the total forms of these nutrients) that indicates dominance by N-fixing blue-green algae should occur with subsequent control of algal biomass by phosphorus (Kann 1998, 2011, 2012a; Lindenberg et al. 2009; Jassby and Kann 2010). Large net-negative retention of nitrogen in Upper Klamath Lake (especially during the summer bloom period; Walker et al. 2012), low TN:TP and TIN:SRP ratios, as well as the seasonal timing of Upper Klamath Lake nitrogen increases, demonstrate the ability of AFA to overcome nitrogen limitation, and grow to large levels based on the amount of available phosphorus. Phosphorus limitation of the initial AFA bloom has been indicated by a variety of lake studies (Kann 2012a, Hoilman et al. 2008, Lindenberg et al. 2009).

It is generally during the influx of available nitrogen (chiefly ammonia; see Appendix C for additional detail) occurring when the initial AFA bloom declines that the non-nitrogen fixing blue-green alga *Microcystis aeruginosa* appears in Upper Klamath Lake (Eldridge et al. 2012). Although never reaching the biomass levels of AFA, these secondary *M. aeruginos*a occurrences are responsible for production of the hepatotoxin microcystin in Upper Klamath Lake (Eldridge et al. 2012). Thus, N-fixation by AFA during the early summer appears to supply nitrogen for growth of toxigenic *M. aeruginos*a later in the summer.

As is typical for many shallow lake ecosystems, the concentration of nutrients, their ratios, the underwater light climate, and climatic variables (e.g., temperature and wind speed) are important determinants of annual bloom dynamics of AFA in Upper Klamath Lake (e.g., Kann and Welch 2005, Jassby and Kann 2010, Wood et al. 2006, Morace 2007, Kann 2012a). In addition, hydrodynamic modeling has shown the influence of wind-driven lake currents on algal and water quality spatial patterns (Wood et al. 2006, et al. 2008). Additional detail regarding nutrient algal interactions is provided in Appendix C.

3.7.1.2 Link River to Keno Dam

The internal nutrient dynamics of Link River have not been studied intensively, but available data indicate that most water quality constituents remain relatively unchanged though the reach, with the exception of dissolved oxygen which re-aerates towards (but not reaching) saturation when it enters the reach below saturation.

The 20 mile long Keno Reservoir is the subject of ongoing investigations by the U.S. Bureau of Reclamation, USGS and Watercourse Engineering. Many available reports present water quality information (Sullivan et al. 2008, 2009; Watercourse Engineering 2003, 2011) including detailed characterization of organic matter in the reach (Deas and Vaughn 2006, Sullivan et al. 2010) and development of predictive water quality models (Sullivan et al. 2011).

Dissolved oxygen concentrations are acutely low during the summer and fall throughout much of Keno Reservoir (Figure 14). pH can also exceed water quality standards during this period, increasing potential for ammonia toxicity. During the late spring and summer, huge quantities of blue-green algae are discharged from Upper Klamath Lake into Keno Reservoir. For unknown reasons, they do not fare well in Keno Reservoir and their biomass declines with increasing distance downstream of Link Dam (Sullivan et al. 2011). Particulate organic matter, derived from the upstream-generated blue-green algae that then die and decay, settles and becomes reservoir sediments. Modeling results indicate that sediment oxygen demand (derived from particulate organic matter settled in the current year) is the largest contributor to oxygen depletion in this reach, followed in importance by suspended particulate organic matter and dissolved organic matter (Sullivan et al. 2011) (Figure 15).

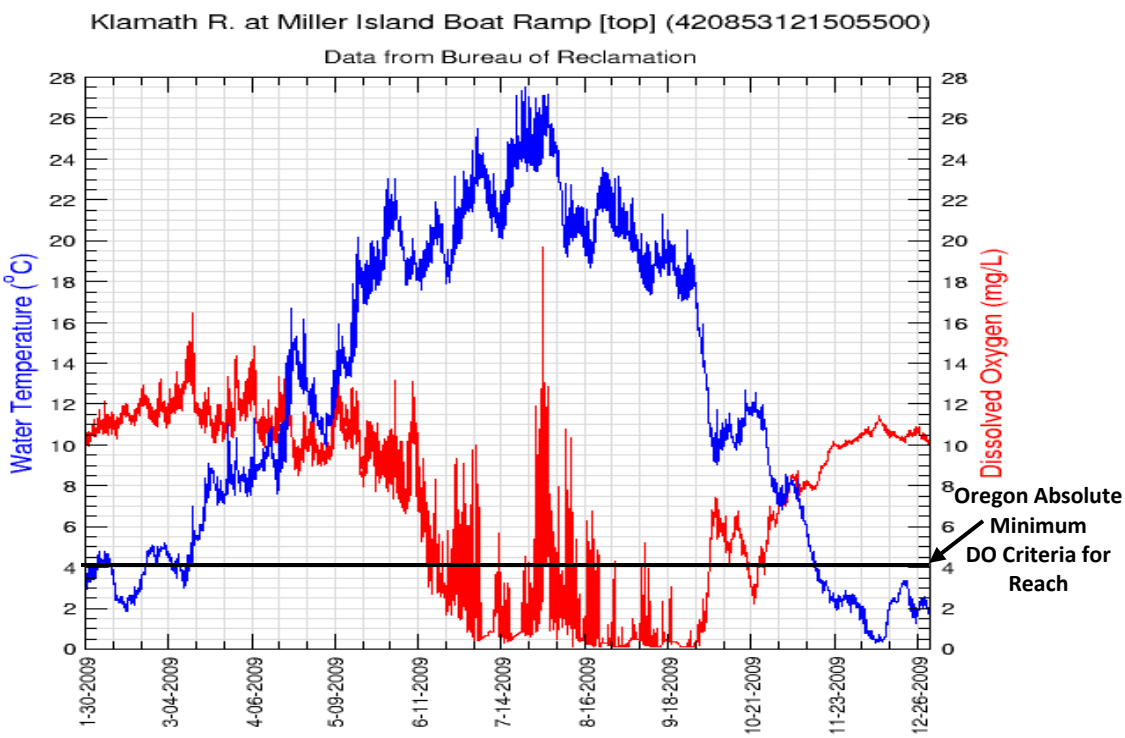

Figure 14. Water temperature and dissolved oxygen concentration near water surface at Miller Island in Keno Reservoir (RM 246). Data from U.S. Bureau of Reclamation.

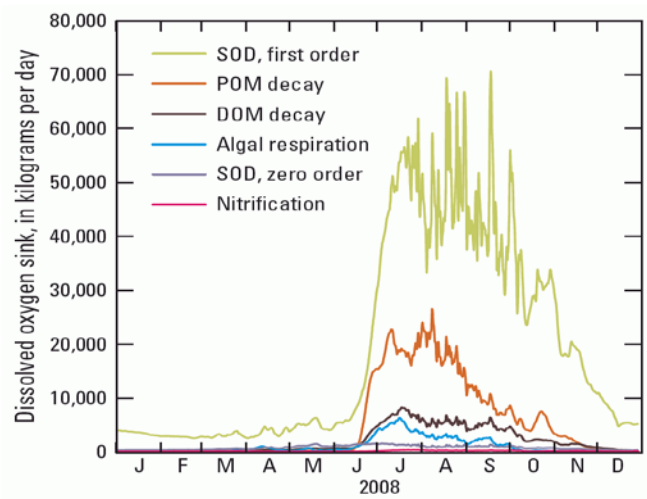

Figure 15. Simulated total annual dissolved-oxygen production consumption (sinks) in the Link–Keno reach of the Klamath River, Oregon, for calendar year 2008. DOM, dissolved organic matter; POM, particulate organic matter; SOD, sediment oxygen demand. Figure from Sullivan et al. (2011).

In the summer, labile particulate organic matter decays rapidly (80% of its oxygen demand was expressed in roughly 8 days during laboratory tests) (Sullivan et al. 2010). Particulate matter in winter, and dissolved organic matter throughout the year, is slower to decay. This slower decay

rate results in less oxygen demand over short periods, but it can also be transported downstream of Keno Dam and exert oxygen demand in the Klamath River (Sullivan et al. 2012).

There are numerous drains flowing into Keno Reservoir (Section 3.6). These include the Klamath Straits Drain (KSD), which operates year-round, and the Lost River Diversion Channel, which diverts flow away from the reservoir during irrigation season but discharges to the reservoir for the remainder of the year. Relative to mainstem monitoring stations upstream (i.e., Link River and Miller Island), KSD outflow has substantially higher concentrations of total and dissolved forms of phosphorus throughout the year and further increases the already high phosphorus concentrations in the river. The water quality effects of the KSD and the Lost River Diversion Channel on the Keno Reservoir nutrient concentrations are variable by year, depending on their relative flow contributions, but their effect on dissolved oxygen is not nearly as great as that of Upper Klamath Lake (additional detail is provided in Appendix C).

3.7.1.3 Keno Dam to upstream of Copco Reservoir

Downstream of Keno Dam, the Klamath River enters a canyon and drops approximately 1,460 feet in 30 miles (49 ft/mile, 1% gradient) before reaching Copco Reservoir. Within this overall steep reach, there are approximately 14.3 miles with lower (20–27 ft/mile) gradient, including the area impounded by J.C. Boyle Reservoir. J.C. Boyle Reservoir has a hydrologic residence time of approximately one day at an average flow of 1,600 cfs, and about 2.5 days at 700 cfs (PacifiCorp 2004). Given its small volume and shallow depth, it does not thermally stratify, does not host substantial blue-green algal blooms, and has relatively little effect on nutrient concentrations (PacifiCorp 2006). Portions of the reservoir do experience hypoxia at times during the summer months (ODEQ 2010), and PacifiCorp is currently exploring engineered approaches for increasing dissolved oxygen levels.

Water quality trends evident from Keno Dam to upstream of J.C. Boyle Reservoir include nitrification, continued reduction in phytoplankton and breakdown of organic matter (additional detail is provided in Appendix C).

3.7.1.4 Iron Gate and Copco 1 Reservoirs

Water quality in Iron Gate and Copco 1 Reservoirs has been intensively studied for many years by PacifiCorp and the Karuk Tribe. Available reports include nutrient budgets (Kann and Asarian 2007; Asarian et al. 2009, 2010), water quality summaries (PacifiCorp 2004), phytoplankton community analyses (Kann and Asarian 2006, Asarian and Kann 2011), toxigenic phytoplankton species (discussed in Section 3.7.2), and annual monitoring reports (Raymond 2008a, 2008b, 2009a, 2009b, 2010a, 2010b).

Iron Gate and Copco 1 reservoirs thermally stratify during the warm summer months, with the deeper waters (hypolimnion) in both reservoirs having low levels of dissolved oxygen as well as high concentrations of ammonia and soluble reactive phosphorus (Asarian et al. 2009). During the stratified period, the upper water column layers (epilimnion) in both reservoirs host nuisance blooms of green algae and cyanobacteria and have elevated pH (Asarian and Kann 2011). High levels of microcystin also occur during summer months in Iron Gate and Copco 1 Reservoirs and have prompted the posting of public health advisories around the reservoirs and, during certain years, along the length of the Klamath River.

3.7.1.5 Iron Gate Dam to Klamath Estuary

The Klamath River flows freely for 190 miles between Iron Gate Dam and the Pacific Ocean. The primary water quality concerns in this reach are those that affect anadromous salmonids in the summer, including high water temperature, low dissolved oxygen, and high pH. Another water quality issue in this reach is the presence of toxic blue-green algae during the late summer and early fall (discussed in Section 3.7.2 below).

Data collected by the USFWS (unpublished) for 2001–2010 indicate that the highest daily mean water temperatures for the months of July and August occur in middle portion of the reach, near Happy Camp (RM 100), with lower temperatures near Iron Gate Dam and downstream of the confluence with the Trinity River. Photosynthesis and respiration by periphyton and macrophytes (rooted aquatic plants) cause diel (24 hour) cycles in dissolved oxygen and pH, with high pH during late afternoon and low dissolved oxygen during night and early morning (NCRWQCB 2010) (see Appendix C for a summary of periphyton growth in the Klamath River). Dissolved oxygen and pH conditions generally improve (i.e., reduced daily minimum dissolved oxygen, reduced daily maximum pH, and reduced magnitude of diel cycles for both parameters) with distance downstream of Iron Gate Dam, although there are exceptions and variability within the overall pattern (HVTEPA 2008).

On its path from Iron Gate Dam to the Klamath Estuary, nutrient concentrations in the Klamath River decline substantially, primarily due to dilution from several large tributaries and secondarily due to seasonal nutrient retention (Asarian et al. 2010). Potential mechanisms for this retention include temporary seasonal storage in periphyton biomass as well as permanent removal through microbial denitrification.

3.7.2 Phytoplankton

As noted above for Upper Klamath Lake, blue-green algae, also known as cyanobacteria, are plentiful in the Klamath River watershed. These algal blooms are often associated with fish kill events due to the oxygen demand of algal communities during dark respiration, and many cyanobacteria are also known to produce algal toxins (cyanotoxins) that can be harmful to animals and humans et al. (e.g., Carmichael 1994, Carmichael et al. 2000; Chorus and Bartram 1999).

One of the main aspects of longitudinal phytoplankton changes in the Klamath River system is the transition from AFA dominated Upper Klamath Lake to toxigenic *Microcystis* dominance in Copco and Iron Gate Reservoirs (Figure 16). Longitudinal increases in the amount of available nitrogen may to be responsible for the shift from the N-fixing AFA in Upper Klamath Lake to the non-N-fixing *Microcystis* in Copco and Iron Gate Reservoirs. For example, as noted in the Upper Klamath Lake section, N-fixation by AFA during the early summer appears to supply nitrogen for growth of toxigenic *M. aeruginos*a later in the summer, and these secondary non-dominant *M. aeruginos*a occurrences are responsible for production of the hepatotoxin microcystin in Upper Klamath Lake (Jacoby and Kann 2007, Eldridge et al. 2012). Such blooms consistently produce cell densities and microcystin toxin that exceed public health guideline levels (e.g., Jacoby and Kann 2007, Kann and Corum 2009, Raymond 2010), and that are associated with public health exceedances downstream of the reservoirs (e.g., Kann and Bowman 2011, Fetcho 2008), and the potential for bioaccumulation of microcystin in a variety of fish species and freshwater mussels (Fetcho 2006, 2011; Kann 2008, et al. 2010, et al. 2011; Mekebri et al. 2009, CH2M Hill 2009a, 2009b; Prendergast and Foster 2010).

The ratios of N:P (the overall nutrient balance summary shows Upper Klamath Lake inflow TN:TP ratios of ~3) are very low in Upper Klamath Lake tributaries but a large influx of nitrogen occurs within the lake system (Kann and Walker 1999, Walker et al. 2012) causing both the ratio of TN:TP (~15) and the absolute concentration and load of total nitrogen to increase greatly in the outflow relative to the inflow. TIN:SRP ratios in Upper Klamath Lake tributaries are generally <1 but increase to values >10 in the Upper Klamath Lake outflow (Kann 2012b). Analyses of nutrient and phytoplankton data in Copco and Iron Gate reservoirs show that the timing of *Microcystis* blooms is related to an influx of nitrate from the Klamath River that is tied to the timing of the Upper Klamath Lake AFA bloom and subsequent increase in outflow nitrogen (Kann and Asarian 2007, Asarian and Kann 2011). Although much of the nitrogen released from Upper Klamath Lake is either organic or occurs as ammonia, conversion to nitrate takes place prior to entering the reservoirs (Asarian et al. 2010).

The presence of nitrate as well as increasing N:P ratios was a prerequisite for the transition from early season N-fixers such as AFA and *Anabaena* to *Microcystis* in Copco and Irongate reservoirs, likely explaining the increased presence and dominance of *Microcystis* in Copco 1 and Iron Gate reservoirs relative to Upper Klamath Lake upstream (Asarian and Kann 2011). In addition both higher nitrate and higher SRP were associated with increased July dominance of *Microcystis* in Copco Reservoir, indicating that not only was nitrate necessary for the non-nitrogen fixing *Microcystis* to increase in importance, but that increased SRP was further associated with increased *Microcystis* dominance (Asarian and Kann 2011). In-reservoir control of *Microcystis* biomass by both nitrogen and phosphorus was also noted by Moisander et al. 2009), who showed frequent co-limitation by these nutrients. Thus, while nitrogen clearly influences relative dominance, phosphorus control is still essential for reduction of overall algal biomass.

Figure 16. Aerial photo of *Microcystis aeruginosa* bloom in Copco (top) and Irongate (bottom) Reservoirs in September 2007. Photo Credit: Jacob Kann.

Longitudinal trends show the importance of the Upper Klamath Lake and the downstream reservoirs for providing habitat conducive for growth of blue-green algae (Kann 2006, Kann and Asarian 2006). Although high algal biovolume dominated by AFA occurs in the outflow of Upper Klamath Lake during the summer months, phytoplankton biomass levels rapidly decline as the system changed from the lacustrine environment of Upper Klamath Lake to the riverine environment of the Klamath River (Kann and Asarian 2006). In general, diatoms increased in prevalence downstream before decreasing again in the Copco/Iron Gate Reservoir complex as blue-green algae again dominated (Kann and Asarian 2006). Riverine diatoms then tend to again dominate the phytoplankton below Copco 1 and Iron Gate reservoirs.

3.7.3 Water quality effects on fish populations

Water quality problems in Upper Klamath Lake negatively affect native fish populations, including the Shortnose sucker (*Chasmistes brevirostris*), Lost River sucker (*Deltistes luxatus*), and interior redband trout (*Oncorhynchus mykiss ssp.*) (ODEQ 2002). Both sucker species were listed as endangered in 1988, with excessive algal blooms and associated poor water quality (i.e., low dissolved oxygen, high pH, high ammonia) identified as primary factor in their declines (ODEQ 2002). Fish kills occurring in 1995-1997 in Upper Klamath Lake included high numbers of these three species, as well as native blue and tui chubs (Perkins et al. 2000). The Shortnose

and Lost River sucker species have experienced substantial declines in the abundance of spawning fish because losses from mortality have not been balanced by recruitment of new individuals (Hewitt et al. 2011; Janney et al. 2009). Algal toxins may also negatively affect sucker populations in Upper Klamath Lake. During 2007 and 2008, USGS conducted a reconnaissance study in the lake to evaluate the presence, concentration, and dynamics of exposure to microcystin toxin by Lost River sucker and shortnose sucker. Water and juvenile sucker samples at multiple lake sites suggested possible exposure to microcystin and resulting gastro-intestinal lesions in the fish. The authors hypothesized that the lesions were caused by algal toxins, and that the route of exposure to toxins was an oral route through the food chain, rather than exposure to dissolved toxins at the gills (VanderKooi et al. 2010). However, there were other possible explanations for the lesions, including the potential for an undetected viral infection, and additional work to describe the observed pathologies is ongoing. Overall, degraded water quality resulting from algal blooms is a significant threat to the long-term viability of the endangered suckers and other aquatic life in Upper Klamath Lake, not only because of catastrophic mortality events, but also because of reduced fitness and survival as result of chronic stress (ODEQ 2002) and possibly exposure to algal toxins.

Water quality problems also affect fish and other aquatic species living downstream of Upper Klamath Lake. In Keno Reservoir, poor water quality during summer and fall (i.e., mid-June through October), including high water temperatures and low dissolved oxygen, led NOAA Fisheries Service and the US Department of the Interior to prescribe interim trap-and-haul measures to transport primarily adult fall-run Chinook salmon past Keno Reservoir during periods when conditions would be harmful to salmonids. Currently salmonid access to Keno Reservoir (and ultimately Upper Klamath Lake) is blocked by the downstream dams (i.e., Iron Gate, Copco 1 & 2, J.C. Boyle); however these dams are being considered for removal.

Numerous fish species use the Klamath River and tributaries during all or some portion of their lives, including salmonids, lamprey, sturgeon, suckers, minnows, and sculpin. Many other species are present in the Klamath Estuary. Of the five species of anadromous salmonids in the Klamath River (i.e., fall- [including late-fall] and spring-run Chinook salmon; coho salmon; fall-, winter-, and summer-run steelhead; and coastal cutthroat trout), all but coastal cutthroat have experienced significant reductions (i.e., greater than 50 percent) from historical numbers. Stress associated with high water temperatures can make cold water species more vulnerable to disease and parasites, and have been associated with fish kills in the Klamath River downstream of Iron Gate Dam during low flow periods in late summer (Hardy and Addley 2001). Seasonally low dissolved oxygen and high diurnal variation in pH downstream of the dams can also be stressful to fish. In Copco 1 and Iron Gate Reservoirs and the Klamath River downstream of Iron Gate Dam, the occurrence of microcystin toxin in fish and mussel tissue has been studied since 2005 (Fetcho 2006; Kann 2008; CH2M Hill 2009a, 2009b; Prendergast and Foster 2010; Kann et al. 2011; Fetcho 2011). Results have been variable, with some results indicating microcystin in reservoir fish tissue (i.e., yellow perch), and salmonid (Kann et al. 2011) and mussel tissue downstream of Iron Gate Dam (Kann 2008).

3.7.4 Summary of water quality and algae monitoring efforts

There are many entities collecting water quality data in the Klamath Basin (Figure 17). The Klamath Basin Monitoring Program (KBMP, www.kbmp.net) is a multi-agency organization created to coordinate water quality monitoring in the Klamath Basin. KBMP members represent various organizations, including government agencies (federal, tribal, state, and local) as well as non-profit and for-profit entities. Member organizations conduct, fund, or have an interest in

water quality monitoring in the Klamath Basin. Most, but not all, entities collecting water quality data in the Klamath Basin participate in KBMP.

Monitoring most relevant to workshop includes those efforts in the Upper Klamath Basin. One long-term dataset of particular note is the Klamath Tribes 1990 to present sampling of nutrients, phytoplankton, and zooplankton in Upper Klamath Lake (Kann 2010a) and nutrients in tributaries to Upper Klamath Lake (Kann 2010b). The ODEQ also has long-term monitoring sites with samples collected at least four times per year near Link Dam, Keno Dam, Klamath Straits Drain, and Lost River. In collaboration with the USBR, the USGS has done intensive research and monitoring in Upper Klamath Lake and adjacent wetlands as well as Keno Reservoir (project list and reports available at http://or.water.usgs.gov/klamath/). USBR also has its own water quality monitoring program including continuous water quality probes at between Link and Keno Dams (data available at http://or.water.usgs.gov/cgibin/grapher/graph_setup.pl?basin_id=klamath#step1). In addition, USBR is currently conducting a nutrient budget study in the Klamath Irrigation Project (R. Carlson, Physical Scientist, U.S. Bureau of Reclamation, Klamath Falls, Oregon, pers. comm., 2012). The Nature Conservancy is monitoring nutrients and other parameters in the restored wetlands in the Williamson River Delta (Wong and Hendrickson 2011). U.S. BLM is monitoring water quality, vegetation, and peat accretion rates in the Wood River Wetland on Agency Lake (Hamilton 2012). Since 2001, PacifiCorp has collected nutrient (Raymond 2008a, 2009a, 2010a) and phytoplankton (Raymond 2008b, 2009b, 2010b) samples in the vicinity of the Klamath Hydroelectric Project. In the mainstem Klamath River and tributaries between Iron Gate Dam and the Klamath Estuary, the Karuk and Yurok Tribes have collected nutrient samples and operated continuous water quality probes since 2001 and sampled phytoplankton since 2005.

Since 2009, USBR, PacifiCorp, Karuk Tribe, Yurok Tribe, NCRWQCB, ODEQ, and U.S. EPA have been collaborating in water quality monitoring under KHSA Interim Measure 15, funded by PacifiCorp (Watercourse Engineering 2011).

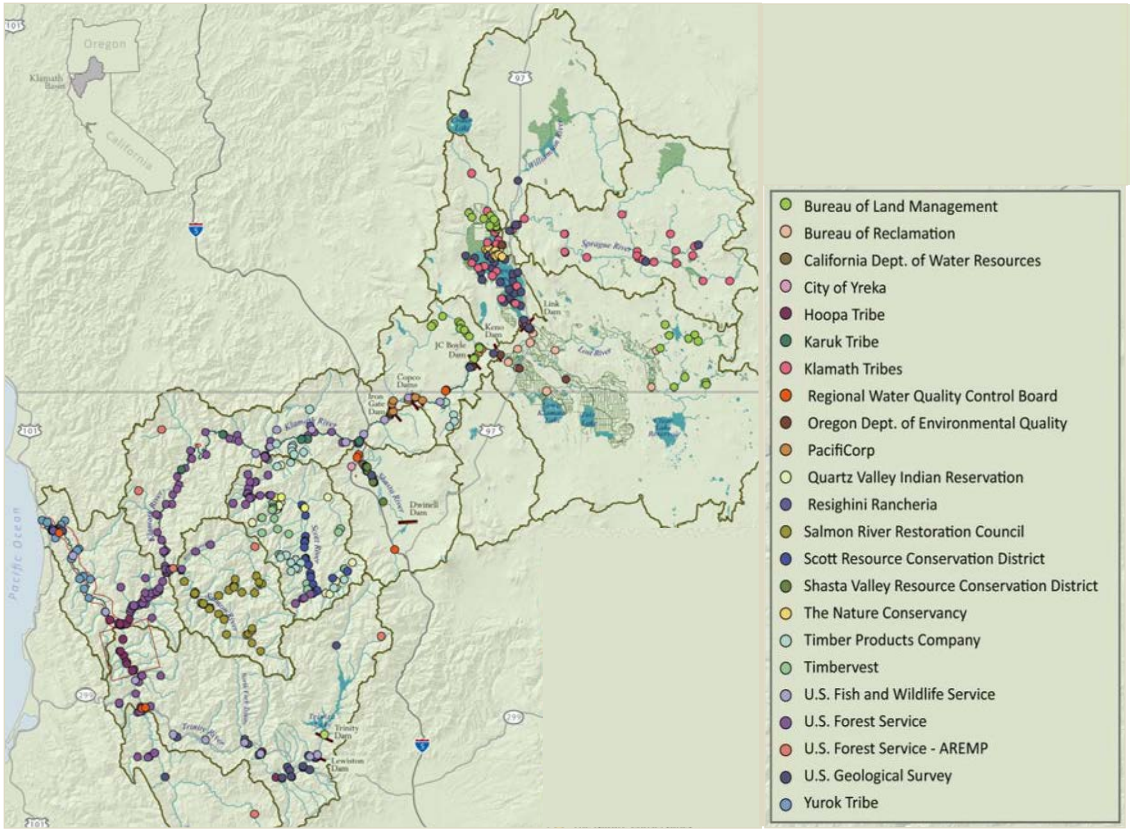

Figure 17. Map showing locations and entities for water quality monitoring in the Klamath Basin for 2009. Parameters and frequency vary by site and entity. Figure adapted from Royer and Stubblefield (2010).

3.7.5 Regulatory considerations

3.7.5.1 Beneficial uses

Beneficial uses of water in the Klamath River basin are designated by the Oregon Department of Environmental Quality, the California State and Regional Water Quality Control Boards, and the Hoopa Valley Tribe. Other tribal water quality programs, including the development and adoption of beneficial uses, are underway by the Karuk Tribe, the Resighini Rancheria, and the Yurok Tribe. These tribes have not yet completed processes for United States Environmental Protection Agency (USEPA) approved delegation under the Clean Water Act (CWA) (NCRWQCB 2010). There are numerous approved beneficial uses within the Klamath River basin including uses for aesthetics and cultural purposes, agriculture, commercial fishing, fish and wildlife habitat, potable and industrial water supply, navigation, groundwater recharge, and recreation.

3.7.5.2 Water quality standards

Water quality standards for fresh surface waters in the Klamath River basin have been established by ODEQ, NCRWQCB, and the Hoopa Valley Tribe to protect designated beneficial uses. Water quality objectives adopted by the Hoopa Valley Tribe establish water quality objectives for those portions of the Klamath and Trinity rivers under the jurisdiction of the tribe. The Yurok and Karuk Tribes have also adopted water quality objectives for the Klamath River, as has the Resighini Rancheria; however, the associated water quality plans have not yet been approved by

USEPA (NCRWQCB 2010). Adopted water quality standards for the Klamath River basin are available for download at the following links:

Oregon: http://arcweb.sos.state.or.us/pages/rules/oars_300/oar_340/340_041.html
California: http://www.waterboards.ca.gov/northcoast/water_issues/programs/basin_plan/
Hoopa Valley Tribe: http://www.hoopa-nsn.gov/documents/WQCP.pdf

3.7.5.3 TMDLs

Section 303(d) of the CWA requires states to identify water bodies that do not meet water quality objectives and are not supporting their designated beneficial uses. These water bodies are considered to be impaired with respect to water quality. ODEQ and NCRWQCB have both included the Klamath Basin and specifically, the Klamath and Lost Rivers on their CWA Section 303(d) lists of water bodies with water quality impairments. In the upper Klamath Basin, where water from the Klamath River and its tributaries is heavily influenced by agricultural irrigation withdrawals and returns from southern Oregon and northern California, impairments include dissolved oxygen, chlorophyll a, water temperature, pH, and ammonia (ODEQ 2010). The entire Klamath River and its tributaries are currently listed as impaired under section 303(d) of the Clean Water Act for water temperature and organic enrichment/low dissolved oxygen (USEPA 2010). Potential sources of impairment include hydroelectric operations, upstream impoundment and flow regulation, among others. The Lower Klamath River is also listed for sediment impairment from the Trinity River confluence (RM 40.0) to the estuary mouth (RM 0.0). Klamath River from Copco 1 Reservoir (RM 203.1) to Iron Gate Dam (RM 190.1) is listed as impaired for toxicity due to the presence of microcystin, a toxin produced by the blue green alga *Microcystis aeruginosa* present in the Project reservoirs (USEPA 2010). Because of potential concerns regarding these water quality impairments and for fish passage considerations, four PacifiCorp dams (Iron Gate, Copco 1 and 2, and J.C. Boyle) are being considered for removal from the Klamath River.

For water bodies included on the 303(d)-list, total maximum daily loads (TMDLs) must be developed by the state with jurisdiction over the water body to protect and restore beneficial uses of water. TMDLs (1) estimate the water body's capacity to assimilate pollutants without exceeding water quality standards; and, (2) set limits on the amount of pollutants that can be added to a water body while still protecting identified beneficial uses. ODEQ and the NCRWQCB cooperated on the development of TMDLs for the impaired water bodies of the Klamath Basin. Additional information regarding the Oregon TMDLs can be found on ODEQ's website (http://www.deq.state.or.us/WQ/TMDLs/klamath.htm) and for the California TMDLs on the NCRWQCB website (http://www.waterboards.ca.gov/northcoast/water_issues/programs/tmdls).

3.8 Water Rights

Water rights are a critical factor in the consideration of pollutant reduction approaches in the Klamath River Basin. In Oregon, a water right is required for any project that would store water, which can include wetland restoration and/or treatment projects. According to the Oregon Division of Water Rights, storage occurs if a project includes a structure that does not have a permanent, open outlet at original natural grade and/or if the project's -use of the water would be wetland enhancement or some other beneficial use. In California, if a project takes water from a lake, river, stream, or creek, or from underground supplies for a beneficial use, a water right is required.

The Upper Klamath Basin is fully allocated, meaning there are no available water rights to prescribe. The Klamath Basin Adjudication is an ongoing legal process whereby water rights that vested before adoption of Oregon's water code in 1909 are quantified and documented through an adjudication proceeding in the Klamath County Circuit Court. There are currently numerous water rights claims in the Upper Klamath Basin that have not yet been resolved.

4 POTENTIAL WATER TREATMENT TECHNOLOGIES

The following section provides summary information regarding a number of possible large-scale water treatment technologies and improvement strategies for the Klamath Basin. The technologies and strategies have been selected by the Steering Committee because they have demonstrated success in other systems similar in size and the degree of nutrient pollution. Additional technical detail for each technology is presented in Appendix C, as needed.
Note that the units in this section are mixed (i.e., English and metric); this is because units are presented as encountered in engineering and planning (typically English units), regulatory (can be either unit system) or the scientific literature (typically metric units), whichever is most applicable. Since workshop participants will span the engineering, regulatory, and scientific disciplines, our objective is to present of the most “familiar” units where possible. Appendix D presents common unit conversions.

4.1 Wetland Restoration

4.1.1 Goals and capabilities

Interest in wetland restoration typically stems from the fact that many of the ecosystem functions provided by wetlands in the U.S. and other parts of the world have been lost as humans have drained or otherwise negatively impacted millions of acres of these natural systems (Mitsch and Gosselink 2007). These functions include reductions in the peak hydrograph, carbon sequestration, nutrient transformation, supporting rich breeding grounds for waterfowl, and providing food and habitat for numerous species of fish and wildlife. In the Klamath Basin, approximately 80% of the natural wetlands have been lost to other land uses, including agriculture. Increasing the extent of wetlands is a recommended strategy for increasing resiliency to climate change in the built environment, the economy, and human and tribal systems in the Klamath Basin (Barr et al. 2010).

In this context, wetland restoration refers to the rehabilitation of natural systems that are degraded and often involves reestablishing wetland hydrology and vegetation (Mitsch and Gosselink 2007). Wetland restoration goals generally include improving or re-establishing the following:

- flood control
- water storage
- water treatment (wastewater, stormwater or non-point source pollution, surface water)
- coastal habitat
- fish and wildlife habitat
- mitigation (replacement of similar habitat lost elsewhere)
- research opportunities

If multiple goals are desired for a particular wetland restoration project, selecting a primary goal is important because it will largely dictate how the wetland is managed. Wetlands that are designed with the primary goal of water quality treatment are generally referred to as "treatment wetlands" and have specific design and operation criteria. These types of wetlands are a subset of general wetlands and are covered in more detail in Section 4.2. The remainder of this section focuses on wetlands restored/rehabilitated for water storage and fish and wildlife habitat, as these have been identified as important ecosystem functions in the Upper Klamath Basin.

4.1.2 Similar applications

Throughout the U.S., there are numerous examples of natural wetlands that are currently managed for water resources and/or wildlife habitat, and possess the secondary goal of water treatment. There are also several examples in the Upper Klamath Basin itself, including large, agency-managed projects and smaller projects spearheaded by private landowners (Table 1 and Figure 18).

Table 1. Examples of previously drained and re-flooded or natural wetlands that are currently managed for water storage and/or wildlife habitat in the Upper Klamath Basin.

Location	Name	Approximate wetland size (acres)	Management entity	Purpose	Additional information
Sprague River	Ridgeway Project	257	Private	Habitat	Lev (2001)
	Sycan Marsh	30,539	The Nature Conservancy	Habitat	The Nature Conservancy website http://www.nature.org/ourinitiatives/regions/northamerica/unitedstates/oregon/placesweprotect/sycan-marsh.xml
Williamson River	Williamson River Delta	7,440	The Nature Conservancy	Habitat + water storage	David Evans and Associates, Inc. (2005), Wong and Hendrickson (2011)
	Klamath Marsh National Wildlife Refuge	13,021 (emergent) 1,008 (open water) 13,889 (sedge meadows)	USFWS	Habitat	Table 3-2 of USFWS (2010)
Wood River	Wood River Wetlands	3,200	BLM	Habitat + future water storage	US BLM (1996), Hamilton (2012)
Upper Klamath Lake	Upper Klamath Lake NWR	15,000	USFWS	Habitat	--
	Barnes and Agency Lake	9,884	USFWS	Water storage	USFWS (2008)

Location	Name	Approximate wetland size (acres)	Management entity	Purpose	Additional information
	Ranches				
Lost River	Circle 5 Ranch	1,011	Private	Habitat	Lev (2001)
Lower Klamath Lake/Tule Lake	Lower Klamath Lake NWR	21,500 (seasonal) 5,500 (emergent) 4,500 (open water)	USFWS	Habitat	Areas are averages for 2003–2005, from Table 2 of Risley and Gannett (2006).
	Tule Lake NWR	1,700 (seasonal) 2,000 (emergent) 10,500 (open water)	USFWS	Habitat	
Keno Reservoir	Miller Island Wildlife Refuge	1,420	ODFW	Habitat	Table 1 of ODFW (2008)

Note: The units in this table may be mixed (i.e., English and metric); however, the units are presented as typically encountered in engineering, planning, or regulatory areas, or the scientific literature, whichever is most applicable. Appendix D presents common unit conversions.

4.1.3 Basic design elements

Individual wetland restoration designs must be tailored to local conditions and constraints (i.e., seasonal range of water temperature, layout, key habitat to be provided, etc.). However, general design criteria for restored/rehabilitated wetlands include the following:

- Wetland hydrology
- Hydric soils
- Gradual slope for maintaining low water velocity
- Varied depth to support a variety of vegetation types and habitats
- Inlet and outlet structures if hydrology is managed

An additional emerging technique is the use of beavers as a cost-effective biological tool for promoting wetland restoration (Pollock et al. 2011). Beavers build dams that form ponds, create wetland habitat, elevate groundwater tables, reduce water velocity, increase floodplain connectivity, and promote sediment deposition within stream channels and floodplains (Pollock et al. 2011, Westbrook et al. 2011). Beavers and their ponds are present in the Klamath Basin (Friedrichsen 1996, USFWS 2010) and there may be opportunities to increase local beaver populations by encouraging landowner tolerance of beavers and their activities, including assistance to prevent and resolve beaver-human conflicts such as unwanted flooding.

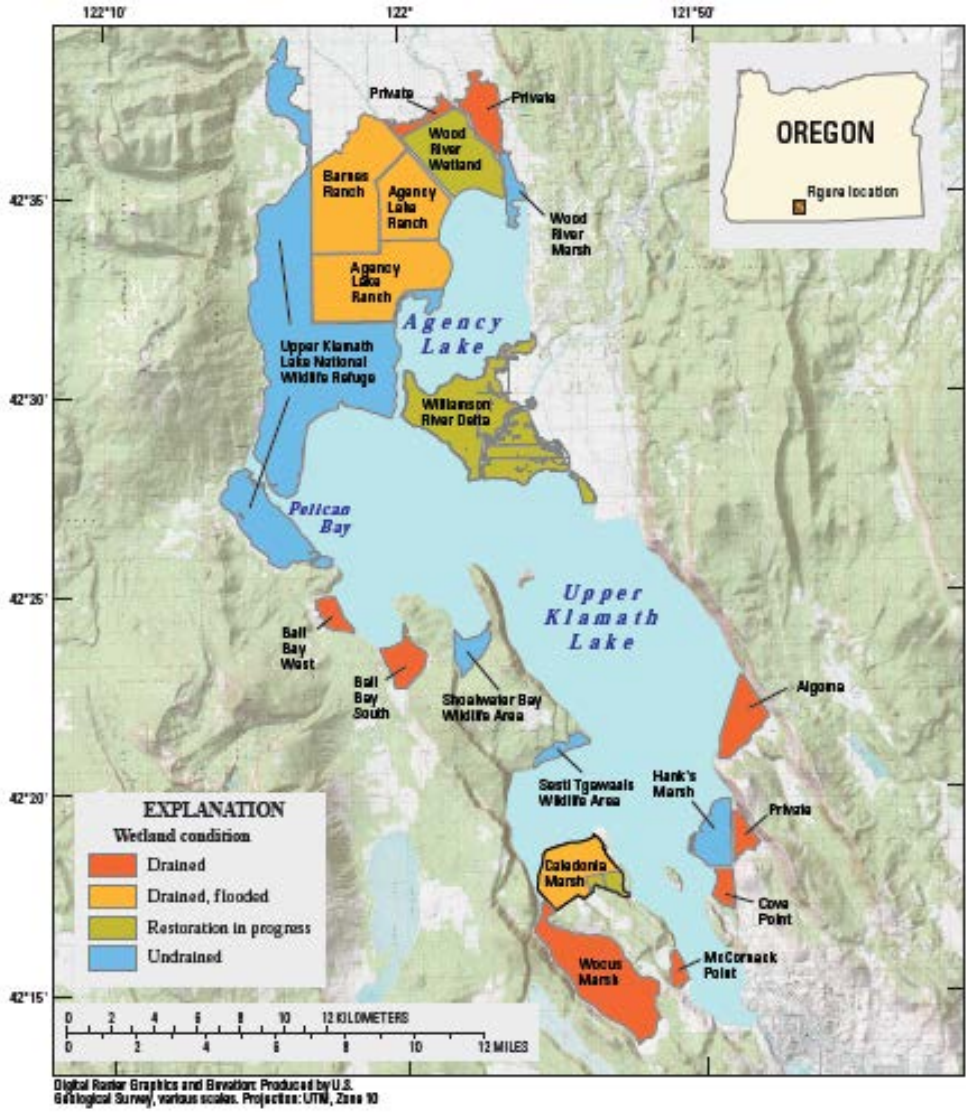

Figure 18. Map showing wetlands surrounding Upper Klamath Lake and Agency Lake, Oregon, 2006. Source: Wood et al. 2009

4.1.4 General siting requirements

The following are siting requirements for wetlands that are expected to be generally applicable in the Upper Klamath Basin (Mitsch and Gosselink 2007):

- Proximity to water body of interest (if the wetland will be used for water storage)
- Elevation compared to water body of interest (if the wetland will be used for water storage)
- Ground slope
- Soil properties
- Size
- Presence of existing water control structures
- Regulatory compliance
- Land ownership
- Existing vegetation

4.1.5 Treatment cost estimates

The cost of wetland restoration/rehabilitation at a given site varies depending on a variety of factors, including size, site conditions and habitat requirements. General cost elements for wetland restoration are anticipated to include some or all of the following (Table 2):

- Land acquisition
- Permitting and water rights acquisition
- Site investigation and engineering design
- Construction
- Perimeter fencing
- Recreational/educational/research facilities
- Operation and maintenance costs (O&M)

Nutrient removal estimates are difficult to estimate for wetland restoration projects due to site specific variability in conditions such as hydrology (seasonal flow patterns), hydraulics (how water distributes/moves though the wetland), land acquisition costs, and operation and maintenance needs. Additional detail regarding cost assumptions is provided in Appendix C.

Table 2. Summary of cost estimates for restored wetlands in the Upper Klamath Basin based upon general size, land acquisition, and construction costs for the Wood River and Williamson River Delta projects adjacent to Agency Lake/Upper Klamath Lake (additional detail in Appendix C).

Cost considerations	Cost estimates for project life
Capital costs (construction and land acquisition)	$15M–$27.5M
O&M	$16M–$128M
Total cost over project life (50 yrs)	$31M– $155M
Unit cost TN removal ($ per kg)	$1–$13
Unit cost TP removal ($ per kg)	$30–$480

Note: The units in this table may be mixed (i.e., English and metric); however, the units are presented as typically encountered in engineering, planning, or regulatory areas, or the scientific literature, whichever is most applicable. Appendix D presents common unit conversions.

4.1.6 Possible treatment locations/general opportunities and constraints

4.1.6.1 Wood River

The Wood River flows just 18 miles from its inception in natural springs in the Jackson F. Kimball State Recreation Site, to its confluence with Agency Lake. Its relatively small watershed consists of 220 square miles (570 km^2) and is dominated by low gradient rural pasture lands and marsh. Historically, much of the watershed was seasonal or permanent wetlands. Currently, numerous agricultural canals criss-cross the landscape, including the 7-mile canal that flows from the northwest area of the watershed toward Agency Lake, and pockets of wetlands exist as seasonal wet meadows or permanently flooded areas in pastures (Figure 19). The Wood River contributes roughly 20% of inflow TP concentration to Upper Klamath Lake and 5% of TN (Appendix C, Table C-1).

The general number, distribution, and size of land parcels located proximal to (i.e., within 1 to 3 miles of) the Wood River are presented in Table 3. Potential opportunities for additional wetland restoration in the Wood River watershed include expansion of the current 3,200-acre Wood River Wetland project and partial or full conversion of other nearby parcels to wetlands. Given the prior

status of much of the low-lying areas in this watershed as wetlands, the general proximity of most parcels to the Wood River itself, the presence of multiple water conveyance structures (i.e., canals), and the presence of several large (> 1,000 acres) parcels that are likely to have a single landowner, this watershed appears to offer relatively good possibilities for further wetland restoration. It is assumed that wetland restoration projects involving water diversions during the irrigation season would require a water right. While conversion of agricultural water rights to wetland use is permissible in Oregon, further investigation of the status of existing water rights in the Wood River watershed is necessary to determine whether this represents a major constraint to wetland restoration opportunities in this tributary to Upper Klamath Lake.

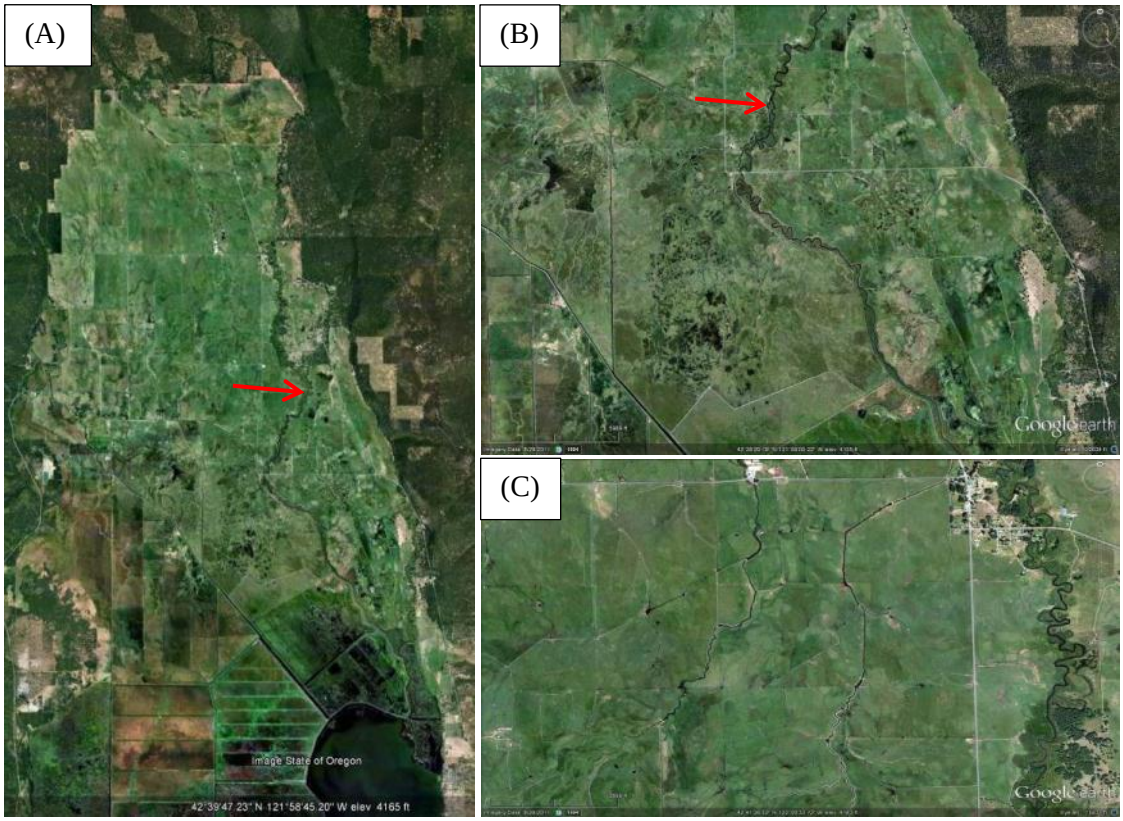

Figure 19. (A) Wood River watershed with Agency Lake and Wood River Wetlands in the lower right hand corner. (B) Meandering reaches of the Wood River flowing through areas with pockets of seasonal wet meadows or permanently flooded areas. (C) Numerous agricultural canals adjacent to the Wood River. Wood River shown by red arrows.

Table 3. General parcel information for parcels located within 1 to 3 miles of the Wood River.

	Within 1 mile of river	Within 3 miles of river
Total number of parcels	349	1,520
Parcel size range	< 1 acre to 7,102 acres	< 1 acre to 19,903 acres
Average parcel size	111.2 acres	85.4 acres
Parcels ≥ 5,000 acres	1	5
Parcels ≥ 1,000 acres	7	12
Parcels ≥ 500 acres	8	21

Note: The units in this table may be mixed (i.e., English and metric); however, the units are presented as typically encountered in engineering, planning, or regulatory areas, or the scientific literature, whichever is most applicable. Appendix D presents common unit conversions.

4.1.6.2 Williamson River

Unlike the Wood River watershed, the 3,000-square mile Williamson River watershed is primarily comprised of forested uplands and mountainous terrain. Low-lying areas of the watershed cover roughly 15% of the total watershed area and include rangelands, wetlands, and urban areas. The Williamson River contributes approximately 45% of inflow TP concentration to Upper Klamath Lake and 55% of TN (Appendix C, Table C-1).

Existing wetland areas include the Klamath Marsh National Wildlife Refuge (40,646 acres), which is currently managed for the conservation and recovery of endangered, threatened, sensitive species and their habitats and to provide and enhance habitat for fall and spring migrant waterfowl. The general number, distribution, and size of land parcels located within 1 to 3 miles of the Williamson River are presented in Table 4.

Table 4. General parcel information for parcels located within 1 to 3 miles of the Williamson River.

	Within 1 mile of river	Within 3 miles of river
Total number of parcels	2,213	3,472
Parcel size range	< 1 acre to 22,165 acres	< 1 acre to 22,674 acres
Average parcel size	144.5 acres	141.3 acres
Parcels ≥ 5,000 acres	18	25
Parcels ≥ 1,000 acres	24	31
Parcels ≥ 500 acres	19	25

Note: The units in this table may be mixed (i.e., English and metric); however, the units are presented as typically encountered in engineering, planning, or regulatory areas, or the scientific literature, whichever is most applicable. Appendix D presents common unit conversions.

Potential opportunities for additional wetland restoration in the Williamson River watershed include expansion of the current Williamson River Delta project, expansion of the Klamath Marsh National Wildlife Refuge, and partial or full conversion of other nearby parcels to wetlands. Although several of the existing larger (>1,000 acre) parcels are already managed as wetlands, the relatively high number of larger parcels (24 within 1 mile of the river, and 31 within 3 miles of the river for parcels > 1,000 acres) suggests that there may still be good possibilities for further wetland restoration along the Williamson River. Further investigation of specific parcels is needed, similar to that conducted by Mahugh et al. (2009). It is assumed that wetland restoration projects involving water diversions during the irrigation season would require a water right. While conversion of agricultural water rights to wetland use is permissible in Oregon, further investigation of the status of existing water rights in the Williamson River watershed is necessary to determine whether this represents a major constraint to wetland restoration opportunities in this tributary to Upper Klamath Lake.

4.1.6.3 Sprague River

The Sprague River (watershed 1,565 square miles) alternatingly flows through constrained reaches and small river valleys on its path to the Upper Klamath Lake. In the small valleys, the river is sinuous, containing multiple bends and oxbows (Rasmussen 2012). Here, given the flashy seasonal hydrology of the Sprague River (Section 3.6), the river has opportunities to overflow its banks and sustain seasonal wetlands and wet meadows, sequestering natural phosphorus transported downstream with sediments during snowmelt periods. The Sprague River contributes approximately 23% of inflow TP concentration to Upper Klamath Lake and 33% of TN (Table 1).

The general number, distribution, and size of land parcels located within 1 to 3 miles of the Williamson River are presented in Table 5. The moderate number of larger parcels (13 within 1 mile of the river, and 16 within 3 miles of the river for parcels > 1,000 acres) suggests that there may be possibilities for further wetland restoration along the Sprague River. Further investigation of specific parcels is needed, similar to that conducted by Mahugh et al. (2009). Wetland restorations have taken place in the Sprague River watershed involving private landowners that converted existing agricultural lands to wetlands by allowing winter/spring time flooding of lands adjacent to the river banks (Table 1). In this type of situation, water rights may not be required by ODWR if the diversion takes place outside of the irrigation season. While summertime treatment largely does not occur in the wetlands due to seasonal drying, the converted land area is neither grazed nor farmed at any point during the year, reducing nutrient additions (nitrogen and phosphorus) and soil oxidation that would have taken place under the prior agricultural land use. Some oxidation and subsequent nutrient release can occur due to seasonal drying of these wetlands; however, based on related studies on other seasonally dried wetlands in the Upper Klamath Basin it is anticipated that the net effect is improved water quality on an annual basis (see Section 3.7.1.1). It is assumed that wetland restoration projects that require water diversions during the irrigation season would require a water right.

Table 5. General parcel information for parcels located within 1 to 3 miles of the Sprague River.

	Within 1 mile of river	Within 3 miles of river
Total number of parcels	4,113	7,530
Parcel size range	< 1 acre to 19,865 acres	< 1 acre to 19,865 acres
Average parcel size	72.6 acres	58.2 acres
Parcels ≥ 5,000 acres	3	21
Parcels ≥ 1,000 acres	13	16
Parcels ≥ 500 acres	29	43

Note: The units in this table may be mixed (i.e., English and metric); however, the units are presented as typically encountered in engineering, planning, or regulatory areas, or the scientific literature, whichever is most applicable. Appendix D presents common unit conversions.

4.1.6.4 Upper Klamath Lake

Several relatively large parcels of land along the shores of Upper Klamath Lake (e.g., Algoma, Caledonia Marsh, Wocus Marsh) are previously drained wetlands that are currently used for agriculture and/or grazing (Figure 18). In the long term, the reconnection of these wetlands to the lake would provide benefits in terms of restored habitat for larval and juvenile suckers, as well as the eventual restored function of nutrient sequestration through peat accumulation (Lindenberg et al. 2009). Based on preliminary review of the Oregon Division of Water Rights (ODWR) on-line water rights information system (WRIS) (http://www.wrd.state.or.us/owrd/pages/wr/wris.aspx, accessed 24 June 2012), these locations appear to currently possess surface water diversions for irrigation use and given landowner willingness, these sites may be eligible for water rights transfer to wetland use during the irrigation season.

Expansion of the USFWS “walking wetlands” program is another potential opportunity to improve water quality in the Upper Klamath Basin using wetlands. This program is currently in use in the Lower Klamath NWR and involves rotating areas of agricultural production with areas of marsh and/or treatment wetlands on refuge lands. Program proponents indicate that higher crop yields are maintained in farmed areas with lower inputs of fertilizers and pesticides and at the

same time, high-quality wetlands are available for wildlife. While the two land uses (i.e., agricultural and wildlife habitat) are traded, such that the net habitat area remains the same at any given point, the decrease in use of fertilizers and pesticides in the watershed would be an overall benefit to water quality.

4.1.6.5 Hydroelectric Reach

As part of the Klamath Facilities Removal Secretarial Determination studies, the Lead Agencies investigated restoration potential for wetland/riparian zones in the river reaches replacing the reservoirs as part of the reservoir area management plan, should the dams at J.C. Boyle, Copco 1 and 2, and Iron Gate be removed in 2020. The analysis identified a total of 272 acres in J.C. Boyle, Copco 1, and Iron Gate Reservoirs that would be returned to wetland/riparian habitat following reservoir drawdown. In these areas, native wetland species would passively be allowed to re-vegetate and native riparian trees would be actively planted (USBR 2011b). Although not quantitatively examined in the reservoir area management plan, these wetland/riparian zones would be expected to provide fish and wildlife habitat, improve assimilatory capacity of upstream nutrients, and result in improvement of other water quality parameters such as water temperature and dissolved oxygen in the Hydroelectric Reach (USBR 2011b).

4.2 Treatment Wetlands

4.2.1 Goals and capabilities

Treatment wetlands are natural or constructed wetlands designed to improve water quality by utilizing the natural physical, biological and chemical processes that occur in wetlands to remove or deactivate pollutants. Treatment wetlands have been shown to effectively reduce levels of a wide range of point and non-point source pollutants including total suspended solids [TSS], nutrients, metals, trace organic compounds including pesticides and herbicides, and pathogens (Kadlec and Wallace 2009). Due to their proven pollutant reduction behavior, relatively low maintenance cost, simplicity of operation, and aesthetic and ecological value, treatment wetlands are increasingly being used for water quality improvement in a variety of settings, including agricultural and urban applications (USEPA 2000a, 2000b).

Treatment wetland performance is sensitive to particle settling dynamics for sediment-associated pollutants (i.e., TSS, biochemical oxygen demand [BOD], pathogens, total metals, total phosphorus), and to reaction time for biochemical oxidation and reduction processes (e.g., nitrification, denitrification), sorption of dissolved metals and trace organic compounds onto sulfide and/or mineral precipitates, and sorption and/or uptake of dissolved phosphorus. Thus, under typical operating conditions, estimated pollutant removals in treatment wetlands are generally based upon particle settling and hydraulic residence time (HRT) (Kadlec and Wallace 2009). Depending on the pollutant removal desired, typical HRTs vary from a small number of days (2–3) for TSS, BOD, and bacteria, to 7–10 days or more for nutrients (i.e., nitrogen and phosphorus) (Kadlec and Wallace 2009). Inlet concentrations of pollutants also influence the treatment performance of a wetland, with greater inflow concentrations and mass loadings generally resulting in greater treatment efficiency. When inlet concentrations approach background concentrations of a particular pollutant, treatment wetland removal efficiencies approach zero. Further information regarding the removal mechanisms for each of the water quality parameters of interest can be found in Reddy et al. (2012), Kadlec and Wallace (2009), and USEPA (2000a, 2000b).

Treatment wetlands are a type of natural treatment system, and as such offer additional benefits including habitat creation and enhancement, groundwater recharge, aesthetics, recreation, and education (Kadlec and Wallace 2009). However, since the primary anticipated benefit is improved water quality, ecological impacts such as habitat creation and enhancement, while complementary, are usually secondary in importance. Treatment wetland design features tend to maximize water treatment, with dense stands of emergent vegetation rather than expanses of open water, where the latter would provide more suitable habitat for migrating waterfowl. However, this is not always the case; the Irvine Ranch Water District's San Joaquin Wildlife Sanctuary in Irvine, CA, was constructed to maximize nitrate removal rates while maintaining 90% open water and episodically exposed shoreline for avian habitat. Given the required habitat components, enhanced nitrate removal was addressed through carbon enhancements consisting of seasonal plantings of barnyard grass (Fleming-Singer and Horne 2006). The habitat value of the IRWD system was estimated to be relatively high, with avian species richness ranging between 65 and 76 species of shorebirds and waterfowl (65–83 birds per hectare per month) during April–October 1999–2002, and 80% removal efficiency for total inorganic nitrogen (TIN) and 60% for total nitrogen (TN) (Fleming-Singer and Horne 2006).

As with general wetland restoration, the ecological benefits of treatment wetlands are also expected to offset impacts of climate change. Carbon sequestration in the peat layers of constructed tule wetlands is being investigated as a mitigation strategy for agricultural soil oxidation, greenhouse gas emissions, and land subsidence in multiple locations, including the Sacramento-San Joaquin River Delta, California. Consideration of the potential benefits and risks of large scale treatment wetland use in the Delta indicated that these systems could result in total annual reductions in carbon and other greenhouse gas emissions and soil accretion could help improve levee stability through reducing hydrostatic pressure, improving water supply reliability and benefiting native wildlife species (Merrill et al. 2011).

4.2.2 Similar applications

There are numerous examples of treatment wetlands that have been used for nutrient and organic matter removal in the United States, including systems associated with a large river diversion and/or nutrient treatment at scales relevant to the Upper Klamath Basin. CH2MHill provided a review and short summary of several project examples in their recent draft report on approaches to treatment wetlands in the Upper Klamath Basin (CH2MHill, *in prep*). While the report is largely a theoretical boundary setting exercise for nutrient removal assuming significant diversion of water from the Keno Impoundment into treatment wetlands, Chapter 6 of the report provides short summaries of 13 treatment wetland examples that could be similar to those implemented in the Upper Klamath Basin. Several of these are listed below as examples.

- Arcata Marsh and Wildlife Sanctuary, Arcata, CA
- Albany-Millersburg Integrated Treatment Wetlands System, OR
- Prado Wetlands, Santa Ana River, CA
- New River Wetlands Project, Salton Sea, CA
- Des Plains River Wetlands Demonstration Project, IL
- Everglades Construction Project, FL
- Mississippi-Ohio-Missouri Basin Nutrient Control Implementation Initiative, NCII

4.2.3 Basic design elements

Individual treatment wetland designs must be tailored to local conditions and constraints (i.e., seasonal range of water temperature, layout, constituent to be removed, etc.). However, general design criteria for free water surface treatment wetlands (i.e., wetlands where water flows above ground) include the following:

- Configuration supporting a tortuous flow path for water;
- Gradual slope for maintaining low water velocity;
- Varied depth to support a variety of vegetation types and related treatment functions;
- Multiple cells to prevent short circuiting; and,
- Inlet and outlet structures.

4.2.4 General siting requirements

The following are siting requirements for free water surface treatment wetlands that are expected to be generally applicable in the Upper Klamath Basin (Mahugh et al. 2009):

- Proximity to water body of interest
- Elevation compared to water body of interest
- Ground slope
- Soil properties
- Size
- Presence of existing water control structures
- Regulatory compliance
- Land ownership
- Proximity to dredged soils disposal sites (if needed)
- Existing vegetation

4.2.5 Treatment cost estimates

The cost of treatment wetland creation at a given site varies depending on a variety of factors, including size, site conditions and water treatment requirements. General cost elements for treatment wetlands are similar to those for habitat-focused wetland restoration and are anticipated to include some or all of the following:

- Land acquisition
- Permitting and water rights acquisition
- Site investigation and engineering design
- Construction
- Planting
- Perimeter fencing
- Pre-treatment components, if needed
- Recreational/educational/research facilities
- Operation and maintenance costs (O&M)

Summary costs are provided in Table 6. Additional detail regarding cost assumptions and estimates is provided in Appendix C.

Table 6. Summary of cost estimates for a treatment wetland (or series of wetlands) on the order of 1,000–2,000 acres in size, treating a 70-cfs flow, for approximately 60% TP removal and 90% TN removal, assuming a project life of 50 years (additional detail in Appendix C)

Cost considerations	Cost estimates for project life
Capital costs (construction and land acquisition)	$17M
O&M	$21M–$64M
Total cost over project life (50 yrs)	$38M– $82M
Unit cost TN removal ($ per kg)	$10–$48
Unit cost TP removal ($ per kg)	$47–$162

Note: The units in this table may be mixed (i.e., English and metric); however, the units are presented as typically encountered in engineering, planning, or regulatory areas, or the scientific literature, whichever is most applicable. Appendix D presents common unit conversions.

4.2.6 Possible treatment locations/general opportunities and constraints

4.2.6.1 Upper Klamath Lake

Existing wetland restoration projects along Upper Klamath Lake, Agency Lake, Lower Klamath Lake, and Tule Lake (Figure 18) are generally managed for water storage and/or wildlife habitat, with water quality improvement as a secondary, albeit important, management goal (see Section 4.1.1.). Nutrient retention has been observed as a result of the Wood River Wetland restoration project (Hamilton 2012) and management of the Upper and Lower NWRs, and it may be possible to expand the footprint of these projects. It may also be possible to enhance water quality treatment potential in a portion of these wetlands through altered flooding regimes (e.g., flood frequency, water depth), changes in the ratio of open water to vegetated stands, and/or use of supplemental technologies to enhance nutrient removal. The primary advantage to enhancing water quality treatment potential in existing wetlands is that these systems already meet the majority (if not all) of the physical siting requirements for treatment wetlands and may not require additional land acquisition or water rights allocations. However, water storage and wildlife habitat benefits may be decreased with conversion of a portion of the existing wetlands to treatment wetlands and these potential impacts would need to be assessed prior to conversion. Consideration of rotating cells, similar to the USFWS Walking Wetlands program, whereby portions of the existing wetlands would be managed as water quality treatment wetlands then returned to use as water storage and wildlife habitat wetlands on a rotating basis, may be informative.

Regarding the use of supplemental technologies to enhance water treatment in existing wetlands, CH2MHill provided a review and short summary of several potential options, including specialized biological features such as submerged aquatic vegetation (SAV) systems and periphyton treatment systems, chemical treatment approaches (e.g., alum, ferric chloride, and other soil amendment and dosing options), and hybrid wetland systems including pre-filtration cells (CH2MHill, *in prep*). For the most part, application of the specialized biological filters requires further research and pilot testing to determine efficacy in the Upper Klamath Basin. The same is true of the chemical treatment and soil amendments, which would require bench-scale tests to determine applicability and cost-effectiveness in the low-alkalinity, weakly buffered waters of the upper basin (CH2MHill, *in prep*). Consideration of the potential impacts to aquatic species (i.e., suckers) and wildlife through possible toxicity or floc build-up would also be needed if this approach is chosen.

4.2.6.2 Keno Reservoir

CH2MHill conducted a theoretical boundary setting exercise for nutrient removal assuming significant diversion of water from the Keno Impoundment into treatment wetlands (CH2MHill, *in prep*). The draft report indicates that 20,000 acres of treatment wetlands would reduce phosphorus and nitrogen concentrations by 50% and 15%, respectively, assuming 100% diversion of the Klamath River at Link River Dam. As stated in their draft report, these assumptions are conservative and can be further refined based upon realistic expectations of flow diversions and nutrient reduction targets. Additional work is ongoing in that regard and, if available, the final report will be provided to workshop participants prior to September 2012.

The USBR undertook the Keno Reservoir Wetlands Feasibility Study, Phase II project, which identified and described key site attributes for treatment wetlands on lands along Lake Ewauna and the Klamath River between the mouth of Link River and Keno Dam. The project assessed and prioritized potential pilot- and full-scale treatment wetland sites and included a pre-design wetland configuration evaluation at the selected priority site (Mahugh et al. 2009). Over 500 parcels were included in the study area, encompassing more than 84,000 acres of predominantly agricultural lands. A number of general site attributes were considered, including whether or not the site was a degraded or former wetland, proximity to the Link River mouth and the Klamath River, soil chemistry and general properties, presence of existing water control structures, ground slope, elevation above the Klamath River, size, regulatory compliance (i.e., sites containing jurisdictional waters, such as wetlands and streams, that would require additional permitting), landowner willingness to participate, proximity to dredged spoils disposal sites, and presence/absence of existing vegetation (Mahugh et al. 2009).

Based on the GIS analysis, 20 sites were identified as viable candidates for full-scale treatment wetlands in the vicinity of Keno Reservoir. Most high ranking full-scale wetland sites were located along an approximately north-south corridor adjacent to the Klamath River and Highway 97 from near Wild Horse Butte in the south to the upper extent of the Miller Island Wildlife Refuge in the north (Figure 20). Properties ranged in size from approximately 100 acres to over 1,420 acres. The largest parcel was Miller Island Wildlife Refuge (1,420 acres), located immediately adjacent to the river, approximately 5 miles from Link Dam (Figure 20).

Ten sites were identified as viable pilot-scale sites, based on their smaller size and proximity to Link River. An expanded set of site attributes was applied to the potential pilot-scale sites, with some of the information provided through field verification (Mahugh et al. 2009). Two sites ranked favorably (i.e., a score of ≥ 8 out of 13 possible points), including a narrow 63-acre site located along the west shore of Lake Ewauna (Figure 20, “Pilot-scale Site 1” in Table 7), and a site containing multiple smaller parcels at the downstream end of the Keno Impoundment (Figure 20, “Pilot-scale Site 6” in Table 7) . Additionally, as part of a preliminary feasibility assessment of treatment wetlands in the vicinity of PacifiCorp’s Klamath Hydroelectric Project, Lyon et al. (2009) identified two treatment wetland sites upstream of Keno Dam that could be used to treat water quality upstream of the Project reservoirs by removing nutrients before they are transported downstream (Sites 020 and 022 in Table 7).

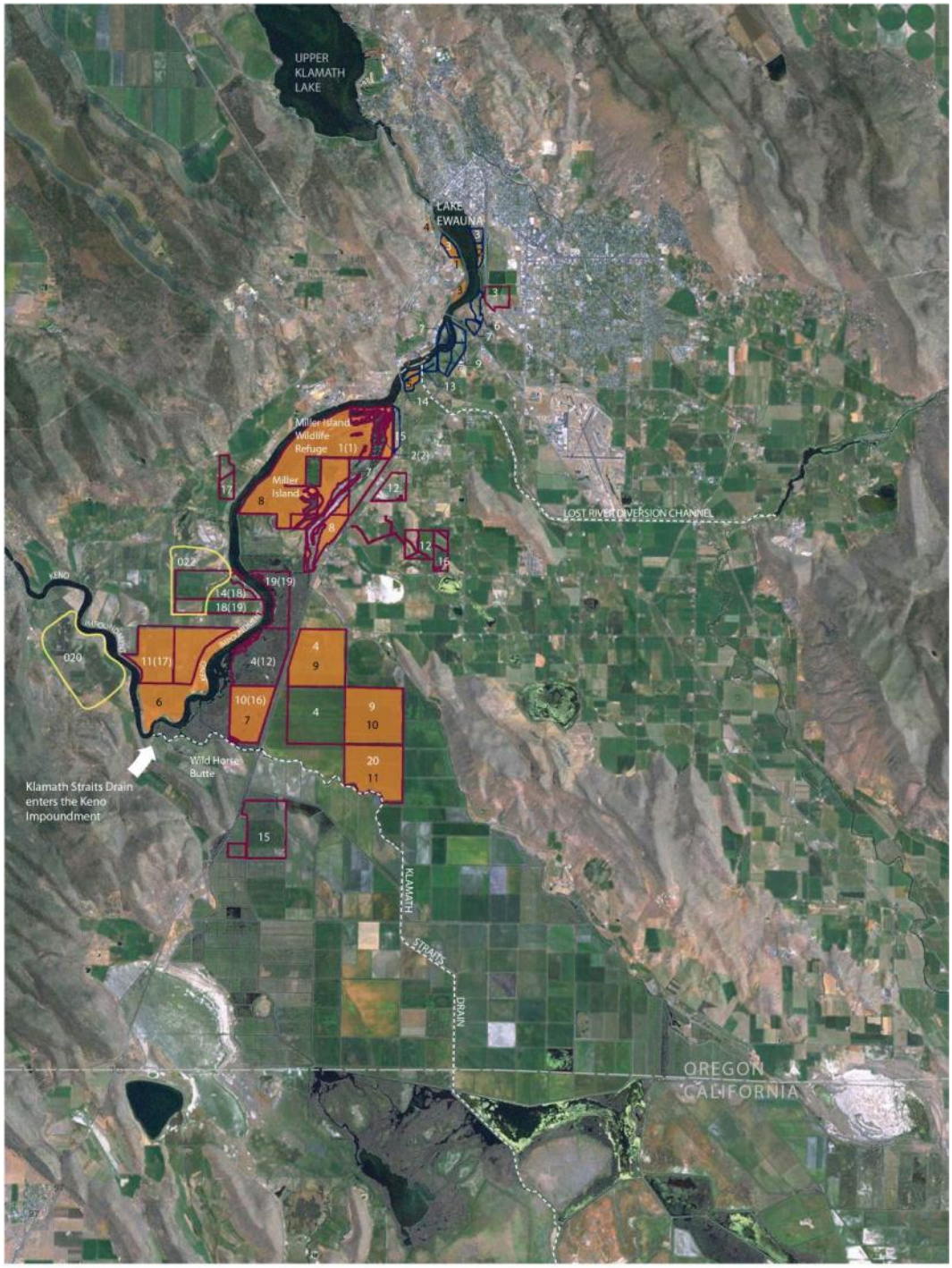

Figure 20. Potential treatment wetland sites for Keno Reach.

Table 7. Potential treatment wetland sites identified between Link River Dam and Keno Dam.

Site ID	Features	Water rights information		Sources
		WRIS basic status	Eligible for WRA or transfer?	
Pilot-scale Site 1	• 63 acres • privately owned • some water conveyance structures (no pumps) • road access • configured favorably for linear treatment cells • supports wetland soils and plants • pre-design modeling exercise indicated 60-90% BOD reductions and 50-70% TSS reductions using parallel treatment cells • total estimated design and construction cost = $252,400	• located within a region of primary ground water diversions • proximal to a single surface water diversion for irrigation use	pending adjudication	Mahugh et al. (2009) WRIS
Pilot-scale Site 6	• 1,300 acres consisting of 11 individual parcels • privately owned • water conveyance structures • road access • supports wetland soils and plants • located near the downstream end of the Keno Impoundment	• located within region of primary surface water diversions for irrigation	pending adjudication	Mahugh et al. (2009) WRIS
Site 020 Site 022	• 600 acres each • located directly adjacent to the Keno Impoundment • water conveyance structures • supports wetland soils	• Site 022 located within region of primary surface water diversions for irrigation	• Site 022 pending adjudication • Site 020 existing primary surface water right for irrigation use	Lyon et al. (2009) WRIS
Site 4 Site 9 Site 10 Site 15	• total approximately 3,300 acres • achieve (in combination) the 1,600 to 3,100 acres recommended for 60% reduction in TP and 90% reduction in TN from KSD flows • close proximity to Keno Impoundment permits possible augmentation of winter and/or spring flows in the wetlands to avoid seasonal periods of soil oxidation	• located within region of primary surface water diversions for irrigation	pending adjudication	Mahugh et al. (2009) Lytle (2000) WRIS

Note: The units in this table may be mixed (i.e., English and metric); however, the units are presented as typically encountered in engineering, planning, or regulatory areas, or the scientific literature, whichever is most applicable. Appendix D presents common unit conversions.

4.2.6.3 Klamath Straits Drain

Lytle (2000) summarized available water quality data during the period 1991–1999 and investigated the potential for treatment wetlands to effectively treat water in the KSD prior to entering the Keno Impoundment. Using a first order removal model, Lytle (2000) estimated required wetland treatment areas that range between approximately 1,600 and 3,100 acres in order to achieve a roughly 60% reduction in total phosphorus concentrations (0.4 to 0.2 mg/L) and a 90% reduction in total nitrogen including ammonium. Given the generally low TSS and high refractory DOC content of KSD flows (see section 3.7.1.4 above), treatment wetlands along the KSD would not be expected to improve water quality in the downstream Keno Impoundment with respect to TSS and DOC. Additionally, given the high seasonal flow variability in the KSD, with summer flows typically 300 cfs or more and winter flows less than 100 cfs, it may be necessary to impound or to manage flows so that treatment wetlands receive a more consistent seasonal flow and do not routinely dry out (Lytle 2000).

While Lytle (2000) did not identify specific treatment wetland sites proximal to the KSD, a subset of the sites identified for the Keno Reservoir Wetlands Feasibility Study, Phase II project (see above section) near the mouth of the KSD and Keno Reservoir may serve as suitable locations for treating KSD water (Sites 4, 9, 10, and 15 in Figure 20 and Table 7). The sites identified by Mahugh et al. (2009) average 550 acres in size and combined they total approximately 3,300 acres, achieving the roughly 1,600 and 3,100 acres recommended by Lytle (2000) for 60% reduction in total phosphorus and 90% reduction in total nitrogen from KSD flows. Given the proximity of these sites to the Keno Impoundment, it may also be possible to augment winter and/or spring flows in the wetlands when KSD flow is significantly lower, thereby avoiding seasonal periods of soil oxidation. Acquiring water rights for diversion of the Klamath River during the winter and/or spring season may not present significant challenges, since diversion would not occur during the irrigation season.

4.2.6.4 Hydroelectric Reach

As described above, PacifiCorp conducted a preliminary feasibility assessment of treatment wetlands in the vicinity of the Klamath Hydroelectric Project in order to improve water quality. Lyon et al. (2009) identified 10 sites for potential treatment wetlands ranging in size from 8 to 63 acres and located in the Hydroelectric Reach from the upstream end of Iron Gate Reservoir to the downstream end of J.C. Boyle Reservoir that could be used to remove nutrients, organic matter, and algal cells from reservoir water. Most of the sites are currently owned by PacifiCorp. Potentially important design features for these sites include multiple-cell surface flow treatment wetlands using existing irrigation and river diversion canals, and, if necessary, alum treatment to enhance phosphorus and particulate removal (Lyon et al. 2009). Preliminary estimates of available acreage for these sites total 254 acres, accounting for berms, roads, and setbacks, and on the order of 150 cfs, or roughly 6–17% of the overall river flow (Lyon et al. 2009).

Lastly, as part of the Klamath Facilities Removal Secretarial Determination studies, the Lead Agencies identified a total of 272 acres in J.C. Boyle, Copco 1, and Iron Gate Reservoirs that would be expected to be returned to wetland/riparian areas if the reservoirs were removed (see Section 4.1.5). Some portion of this area could be considered for treatment wetland development to enhance anticipated nutrient removal. Further consideration of these sites is pending the Secretarial Determination of whether dam removal should occur.

4.3 Decentralized (Diffuse) Source Treatment Systems

4.3.1 Goals and capabilities

The proven ability to remove pollutants in large-scale treatment wetlands (see Section 4.2) opened the door to the possibility of additional treatment occurring throughout a given catchment rather than at the bottom of the catchment or just prior to discharge into a large receiving water body. Subsequently, design and implementation of a network of small-scale diffuse source (decentralized) treatment wetlands has been undertaken in order to achieve the benefits of wetland ecosystem functioning in multiple locations throughout a watershed.

The goals for diffuse source (decentralized) treatment systems include the following:

- Reduction in peak hydrograph *throughout* the watershed
- Reduced land subsidence and soil carbon oxidation *throughout* the watershed
- *On-site* removal of sediment, nutrient, and herbicides/pesticides, particularly for *first flush* runoff events
- Increase in extent of wetland habitat and associated wildlife (e.g., benthic macroinvertebrates, amphibians, fish, birds) *throughout* the watershed

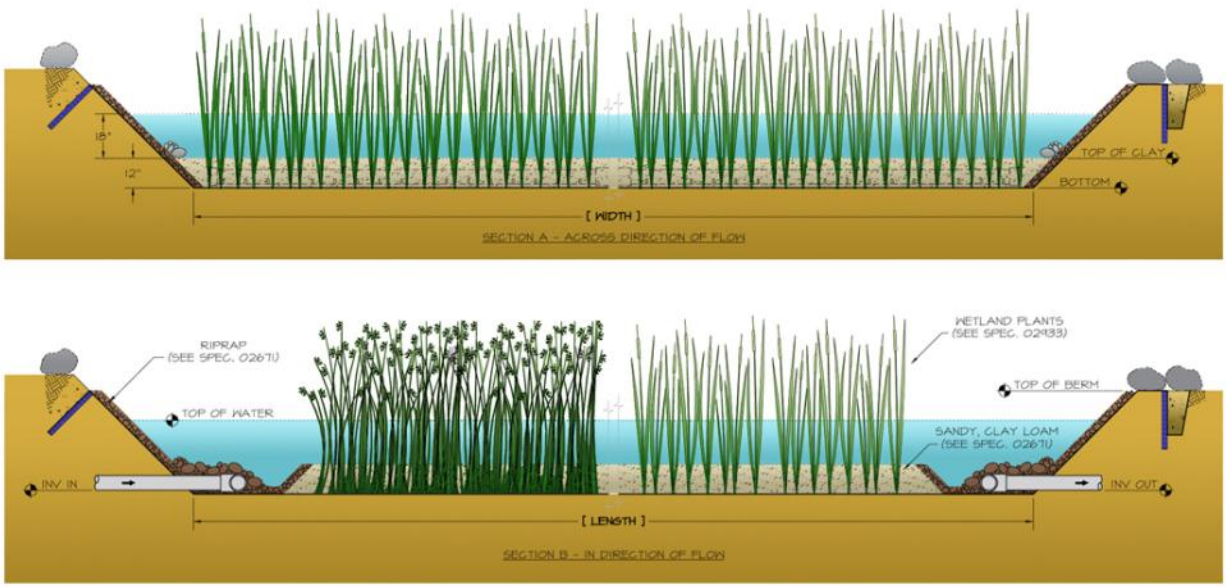

Figure 21. Basic schematic of a decentralized (diffuse) source treatment wetland. Top tile is a cross-section perpendicular to the direction of flow and the bottom tile is a cross-section in the direction of flow. Widths and lengths for these small systems vary, but are typically on the order of 100-300 feet such that an individual wetland comprises roughly 1 to 2 % of the treated area.

4.3.2 Basic design elements

As decentralized (diffuse) source treatment systems are essentially small-scale wetlands (Figure 21) scattered throughout the watershed, the basic design elements are similar to those presented for larger natural wetlands and treatment wetlands (Sections 4.1 and 4.2). However, rather than being sized based on treatment efficiency, as is often the case with larger-scale treatment wetlands (see Section 4.2), the decentralized (diffuse) source systems are designed to accommodate an estimated amount of stormwater runoff from the landscape or a particular HRT

given adjacent agricultural canal flow. Additionally, specific design elements are needed to allow these systems to function at a smaller scale in a setting such as the Klamath Basin. These elements are presented below for treating runoff 1) within pastures and agricultural fields and 2) adjacent to small drainage ditches. While these systems can also be used to treat wastewater and runoff from small-to medium-sized housing developments, these situations are not known to be prevalent in the Upper Klamath Basin. A possible exception to this is the City of Tulelake wastewater treatment plant, located in the adjacent Lost River watershed (see Figure 1), which is in the process of upgrading its treatment facility and has plans to include wetland treatment.

4.3.2.1 Systems in pastures and agricultural fields

Diffuse source (decentralized) treatment systems in pastures and agricultural fields are not intended to be *continuous* flow-thru systems. Rather, they are *intermittent* flow-through systems in that they possess a designated outflow while still mimicking the natural variability in water depth and areal extent that is expected in wetlands dependent on runoff. For this type of application, diffuse source (decentralized) treatment systems can be conceived of as vegetated detention basins, designed on the basis of estimated runoff. Given the importance of downstream water use in the Klamath Basin, it is assumed that these systems would not be designed as terminal wetlands, but rather they would treat on-site runoff such that there would be an outflow of water from each site. The required wetland area is determined using the following formula:

$$A_w = \frac{A_p \times I \times C}{ET_w} \times f$$

Where A_w = wetland area (acres)
I = annual rainfall (feet)
A_p = parcel area (acres)
C = watershed run-off coefficient assumed to be 0.25
ET_w = wetland annual evapotranspiration (feet) = 0.7x ET_{pan}
f = decimal fraction of desired total ET_w loss is assumed to be 0.1

The resulting wetland area tends to be on the order of 1 to 2% of the parcel area.

Example calculations for a 100-acre parcel in the Upper Klamath Basin

Assumptions:

- I = 15 inches per year (1.25 feet per year) (see Section 3.2 for typical rainfall patterns)
- ET_{pan} = 60 inches per year (5 feet per year) for Klamath Falls (Western Regional Climate Center data accessed July 10, 2012 http://www.wrcc.dri.edu/htmlfiles/westevap.final.html)

$$A_w = \frac{100\ acres\ \times 1.25\ ft \times 0.25}{0.7 \times 5\ ft} \times 0.1 = 0.9\ acres$$

Additional design elements include the following:

- *Elevation*—locate in naturally occurring depressions, low lying areas, or former wetland areas in a given parcel.
- *Earthen berms*—generally to be avoided, since the site is likely to be wet and difficult to work with typical earth moving equipment. Berms should have two feet of freeboard and should be higher at the discharge end of the wetlands.

- *Diversion structures*—divert run-off from parcel using one or more vegetated swales from higher elevations to the lowest area in the field or pasture.
- *Level control structure*—install at the lowest elevation in the field or pasture to maintain a maximum water depth of 2 feet in the small wetland.
- *Vegetation*—primary treatment species (e.g., *Typha spp.*, *Scirpus spp.*, and *Sparganium eurycarpum*, *Eleocharis spp.*), secondary food and habitat species (e.g., *Nymphae spp.*).
- *Exclusion fencing*—to keep grazing animals out of the wetlands. A 50-foot barrier is the suggested minimum.

4.3.2.2 Systems located proximal to drainage ditches and agricultural canals

By installing overflow weirs in appropriate locations, it is possible to divert periodic storm events and snowmelt runoff into low-lying areas adjacent to drainage ditches and agricultural canals. Small areas set aside for treating a portion of ditch flows are similar to those developed for in-pasture or in-field wetlands (see above); however, the drainage ditch systems are designed to be *continuous* flow-through systems, in contrast to the *intermittently* flow-through in-pasture or in-field wetlands.

As with larger treatment wetlands, the required wetland area is linked to HRT (see Section 4.2). Although rather than designing based on pollutant removal efficiency, it is assumed that HRTs in these systems are typically on the order of 2–3 days, in order to reduce ET losses while still maintaining basic treatment and wildlife habitat value.

$$A_w = \frac{Q \times HRT}{d}$$

Where A_w = wetland area (acres)
Q = inlet flow (cfs)
HRT = hydraulic residence time (days)
d = average water depth (feet)

Example calculations for a 0.01-cfs canal in the Upper Klamath Basin

Assumptions:

- Q = 0.01 cfs
- HRT = 2 days
- d = 2 feet

$$A_w = \frac{0.01\ cfs \times 2\ days \times \frac{86{,}400s}{day}}{2\ feet} = 864ft^2 = 0.02\ acres$$

Additional design elements for these systems include the following:

- *Elevation*—locate in naturally occurring depressions, low lying areas, or former wetland areas alongside drainage ditches or agricultural canals.
- *Overflow weir and diversion box*—as flow increases in the ditch or canal, the water level should rise and flow out of the ditch and into the diversion box. The diversion box should have the ability to restrict the volume of flow into the wetlands and should be designed to shut off completely, if needed.
- *Distribution trench constructed at the head of the wetland*—the trench should be designed so that water in the trench is 4 feet deep and at right angles to the direction of flow.

- *Adjustable weir at discharge end of the wetland*—water levels in the vegetated area of the wetlands should be maintained at 2 feet or less. Treated water is returned to the ditch at this point.
- *Earthen berms*—if needed, see above note regarding earthwork.

4.3.3 General siting requirements

The following are general siting requirements for diffuse source (decentralized) treatment systems:

- Remnant wetlands or low lying areas in pastures or agricultural areas that possess hydric soils and should readily return to wetlands if not drained, plowed, or grazed
- Locations that support the use of gravity flow, rather than pumps
- Existing drainage canals/ditches adjacent to or near the proposed site (i.e., within 0.25 mile)
- Existing flow control structures up-gradient from the wetland location that could be used for diversion
- Locations that have a single land owner and where the land owner is amenable to the creation of wetlands and associated habitat in lieu of the loss of pasture or field
- Locations where the land owner can readily see habitat benefits (i.e., not tucked away and forgotten)
- Road access
- Lack of pervasive shade—wetlands plants are typically phototropic and will not grow well in the shade. At the small scale of these systems, too much shade is problematic.

4.3.4 Treatment cost estimates

There are relatively few requirements and hence, relatively low costs, for building decentralized (diffuse) source treatment systems (Table 8). Unlike habitat and treatment wetlands, it is assumed that land acquisition is unnecessary as the wetlands can be located on a fraction of an existing parcel, assuming there is a willing landowner.

- Site survey
- Construction
- Planting
- Exclusion fencing
- Operation and maintenance costs (O&M)

Additional detail regarding cost assumptions and estimates is provided in Appendix C.

Table 8. Summary of cost estimates for decentralized (diffuse) source treatment systems, assuming a 100-acre parcel with a 0.9-acre in-pasture wetland and a project life of 50 years (additional details in Appendix C).

Cost considerations	Cost estimates for Project life (50 yrs)
Capital costs (land acquisition and construction)	$17,630
O&M	$11,700
Total cost over project life (50 yrs)	$29,330
Unit cost TN removal ($ per kg)	$4–$9
Unit cost TP removal ($ per kg)	$161–$322

Note: The units in this table may be mixed (i.e., English and metric); however, the units are presented as typically encountered in engineering, planning, or regulatory areas, or the scientific literature, whichever is most applicable. Appendix D presents common unit conversions.

4.3.5 Possible treatment locations/general opportunities and constraints

Examination of Google Earth imagery for the three main tributaries to Upper Klamath Lake, the Wood River, Sprague River, and Williamson River, indicates possibilities for decentralized (diffuse) source treatment systems in all three watersheds. However, in the Wood River watershed in particular, the predominance of agricultural lands containing low-lying areas and former wetlands, and the preponderance of agricultural canals and drainage ditches crossing numerous land parcels (Figure 19), makes this watershed a potentially ideal location for this approach to water treatment. As discussed in Section 4.1.5, there are roughly 350 parcels in the Wood River watershed located within 1 mile of the creek, and the vast majority are less than 500 acres in size (Table 3), providing potential opportunities for numerous <0.5 acre to 1 acre decentralized (diffuse) source treatment wetlands. The Sprague and Williamson Rivers also possess meandering reaches with potential for small decentralized (diffuse) source treatment wetlands to be supported (see Figure 22 for Sprague River example). There are roughly 2,200 and 4,100 parcels in the Sprague and Williamson River watersheds, respectively, located within 1 mile of the creek, and the vast majority are less than 500 acres in size (Table 4, Table 5). The large number of parcels provides potential opportunities for numerous <0.5 acre to 1 acre decentralized (diffuse) source treatment wetlands in these tributary watersheds, although a more focused investigation of individual parcels is needed.

As with larger treatment wetlands (Section 4.2) and habitat wetlands (Section 4.1), water rights represent a general constraint for the implementation of decentralized (diffuse) source treatment wetlands. If water is diverted from combination irrigation and drainage ditches into the small wetland, then downstream users would likely need to agree. Although return flow from the wetlands is included as a design parameter, some water would be lost to evapotranspiration. However, since the majority of decentralized (diffuse) source treatment wetlands would likely store far less than 9.2 acre-feet, the projects would be eligible to apply for an “alternate-reservoir storage permit” in the state of Oregon, which receives expedited processing.

Figure 22. Example meandering reach along the Sprague River flowing through areas with pockets of seasonal wet meadows or permanently flooded areas.

4.4 Algae/Biomass Removal from the Water Column via Filtration

4.4.1 Goals and capabilities

Algal biomass is biologically labile, meaning that when algal cells senesce or die, the organic material contained within breaks down rapidly and creates a pulse of bioavailable nutrients which can promote subsequent algal blooms (see also Section 3.7). Algal biomass removal from water bodies results in the reduction of phosphorus and nitrogen concentrations in the water column. While there can still be appreciable nutrient content in sediments and inflow waters, the continued harvest of algal biomass from the water column is a non-invasive (no chemical application) mechanism to remove nutrients from the system. Further, reductions in algae-induced BOD and a potential source of cyanotoxins are recognized ecological benefits of this approach. The following are goals of direct removal of algal biomass from the water column of Upper Klamath Lake and other affected water bodies in the system:

1. Nutrient removal
2. Reduction in biological oxygen demand (BOD) associated with senescent algal blooms
3. Reduction in cyanotoxin levels
4. Improve habitat for aquatic species

4.4.1.1 Currently available technologies

Several technologies are currently available to remove algal biomass from waters of the Klamath River Basin. For the purposes of this discussion, the technologies presented here are all forms of

filtration or separation of algal biomass from water. While several technologies exist for algae removal from lakes and reservoirs, technologies such as bioreactors (Tanaka et al. 2001) that only decompose algal biomass and do not remove the associated nutrients are not included. Also omitted are technologies such as centrifugation, which have been proven to cause cell lysis and thus, could release cyanotoxins into the environment, and procedures which require massive chemical additives, such as magnetic filtration. The suitable technologies that remove both algal biomass and the associated nutrients include land-based filtration, land-based separation, and *in situ* filtration techniques.

Land-based filtration

Land based filtration via screen filters is currently a common practice on Upper Klamath Lake for the harvest of AFA for commercial sale as a dietary supplement (Carmichael et al. 2000). This process involves diversion of lake water through a canal where debris screens block passage of large debris into a series of algal screens. The algal screens capture algal biomass as water is then pumped over the screens (Figure 23).

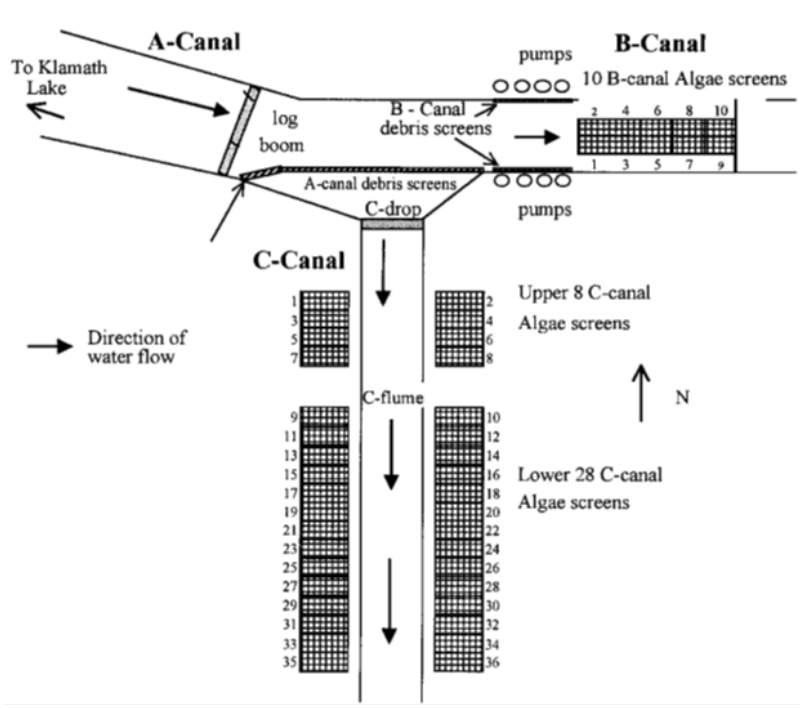

Figure 23. Schematic representation of land based algal filtration site (Cell Tech Inc.). Water flows down the A-canal to split into the B and C canals. In the C-canal, water is gravity fed over a series of screens. In the B-canal, water is pumped over a series of screens. At both locations, water is initially screened to restrict entrance of fish and debris into the site. Source: Carmichael et al. 2000.

Rapid sand filtration is a proven technology that is often applied for removal of particulates from waste waters (Figure 24) and can be adapted to filter algal biomass from lake and reservoir water. The sand filter consists of a containment vessel with coarse sand overlying a gravel bed with drain panels. Water is pumped in over the sand filter and algal biomass is retained on the sand filter surface. To remove biomass, surface sand and biomass is scraped off. When compared to screen filtration, sand filtration is less efficient due to the added weight of sand removed from the filter when removing biomass and the tendency to lose filtration speed as pores clog with algae.

Another land based filtration mechanism is cross flow filtration (Figure 25) where lake water and algal biomass are passed over a membrane with specific pore size that allows passage of water but retains algal cells. This cross flow filtration technology is superior to screen filtration in that the cross flow movement of water continuously cleans the membrane from algal cell fouling, allowing for reduced maintenance of the filter apparatus. While this technology requires less mechanical assistance in screen cleaning, it is also more expensive and requires more infrastructure to implement.

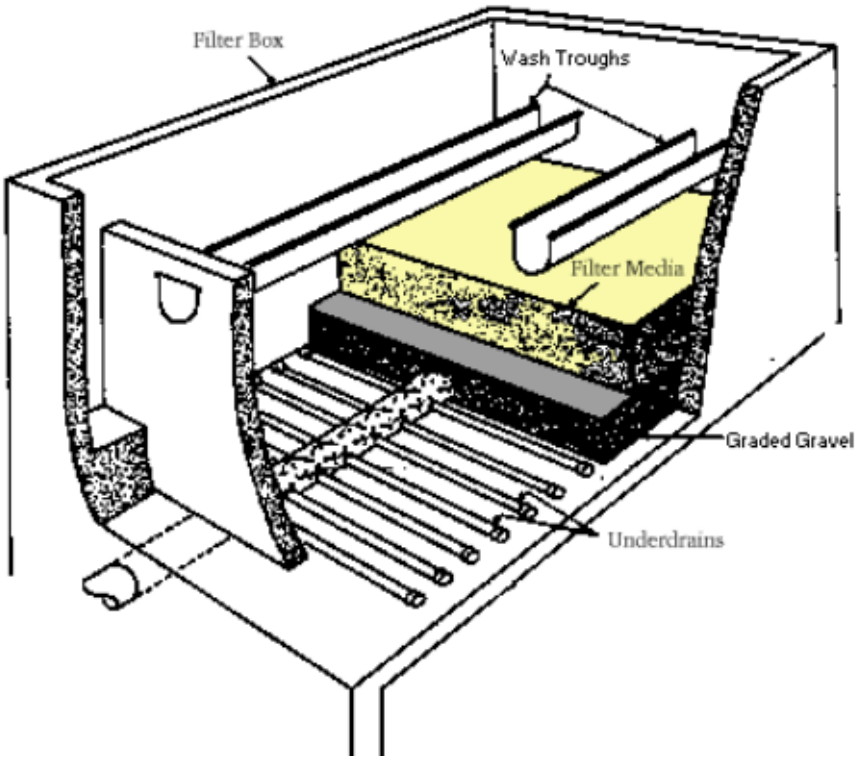

Figure 24. Schematic of rapid sand filtration unit for algal biomass removal. Source: Mountain Empire Community College, Virginia.

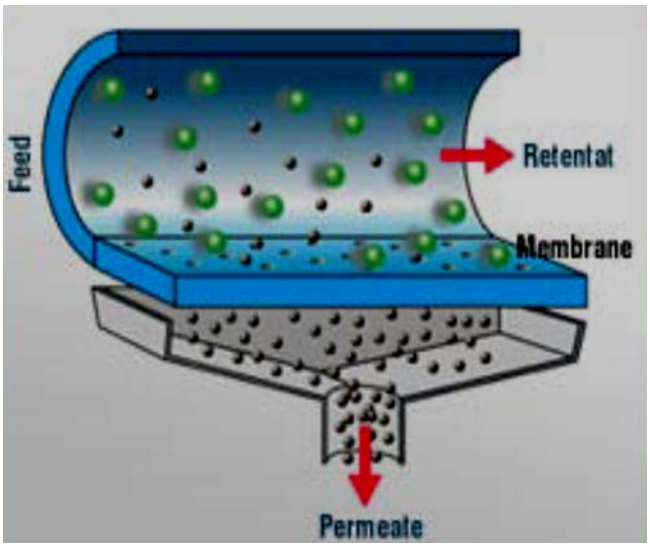

Figure 25. Schematic of cross flow membrane filtration. Source: Alting Membrane Filtration and Process Technology, http://www.alting.fr/en_US/produits/filtration-tangentielle.

Land-based flotation/separation

The final land based technique does not use filtration mechanisms, but rather uses the principles of flocculation and flotation to separate and remove algal biomass. The principle of flotation uses air, combined with flocculation, to bring the algal biomass to the surface of the water. Dissolved air flotation (Arora et al. 1995) uses a compressor to supersaturate the water with air, and then moves the medium into a pressurized chamber, where the air forms tiny bubbles adhering to algae cells to bring them to the surface. A similar process, suspended air flotation, uses surfactants to create tiny bubbles, making the process less expensive with less mechanical components, and using less energy than dissolved air; however, the addition of surfactants would degrade water quality and is therefore not considered further. Ozoflotation is another use of the flotation principle, in which the water is agitated with ozone to denature the algal toxins while floating the biomass to the surface for removal (Benoufella et al. 1994). All of these flotation technologies utilize pumps and aeration, combined with chemical flocculants to increase removal efficiency (up to 95%). A final version of this technology is the adaptation of the Jameson cell, a method utilized in industrial processing to separate particles from solution, to the removal of algal biomass (Yan and Jameson 2004). The premise is a gravity fed cell that induces air mixing via travel down an induction column to a mixing reservoir where algae then float to the surface for mechanical removal. This process also uses chemical flocculants but is 98% efficient in removing *Microcystis* cells.

One significant drawback to all of the flotation techniques is the need for a chemical additive to aid flocculation. The potential exists for cyanobacteria to be removed with up to 90% efficiency without flocculation if algal cells are producing mucilage (Yan and Jameson 2004).

In situ barge filtration

The concept of *in situ* algal removal has been presented and utilized by private industry on Upper Klamath Lake for many years. Simplexity Health Inc. designed and constructed large floating harvesting barges (Figure 26) that can screen filter up to 200,000 lbs of wet algae per day. At 110 feet wide, the harvester has nine rotating screens that filter out AFA for commercial resale as a dietary supplement. Several smaller vessels operated by other companies pump water up to a set

of fixed screens onboard, which retain algal biomass and allow water to pass back into the lake. Some clear advantages of this technology include mobility, which enables maximization of incurring heavy bloom areas, and minimal land needs onshore for infrastructure.

Because private industry has developed economical algae removal techniques and is utilizing cyanobacteria on Upper Klamath Lake, there is potential for a partnership with private industry for algae removal via cost structure per unit removal. This would negate expenditures for land or infrastructure and utilize resources and assets already in place.

Coupling the filtration and subsequent removal of algal biomass with beneficial uses of the material provides benefits to both the economic and environmental aspects of this process (Spolaore et al. 2006). Several options for use or disposal of this material exist and are listed in order of decreasing economic and environmental benefits:

1) Dietary supplement[1]
2) Biofuels production (biodiesel, methane)
3) Soil amendment (may need to be tested for algal toxins prior to soil application)
4) Composting/landfill

Figure 26. *In situ* algae harvesting barge. Note nine screen panels which rotate through water column to capture algae. Mechanical rakes then remove algal biomass and shunt algae to holding tanks within the barge. Source: Simplexity Health Inc.

While harvest of algae, specifically AFA, for processing and sale as a dietary supplement is established commercially (Carmichael et al. 2000), it may be advantageous to utilize existing commercial entities to conduct harvesting year round for a fee or utilize the harvest model currently employed to remove additional algae from the system. Within the Klamath Basin, a system of algae removal credits could encourage intense harvest activity by private contractors and negate the need for investment in significant infrastructure. Also, the use of algal biomass in biofuels production can provide a steady, consumable market which may offset filtration and removal costs.

[1] Commercial harvest and sale of algae from Upper Klamath Lake is an established private enterprise with an estimated $100M in annual sales (Carmichael et al. 2000).

4.4.2 Similar applications

Land based screen filtration and open water vessel based screen filtration are both conducted routinely on Upper Klamath Lake by private industry who harvest AFA for refinement and sale as a human dietary supplement. Currently, private industry harvesting is conducted only intermittently, when conditions are optimal to produce a pure culture for subsequent human use. Expansion of the land based screen filtration and open water vessel based screen filtration to include all forms of algae for a variety of uses (i.e., dietary supplement, biofuels, soil amendment, composting/landfill) would presumably increase the amount of time spent harvesting and the associated nutrient removal and improvements to water quality and support of beneficial uses.

Land based filtration by sand column is routinely used nationwide to reduce particulates in wastewater. Both the cross-flow filtration technique and the flotation techniques have been used in food preparation and industrial processes; however, only proof-of-concept pilot scale studies have been performed to confirm their utility in algal removal. These techniques have not been employed on lakes or reservoirs at scales relevant to the Klamath River restoration project needs.

4.4.3 Basic design elements

Several design elements are common to all filtration options:

- Identification of target algal bloom hot spots (dominant bloom locations)
- Determination of target algal cell size / screen / filter size to capture multiple species of algae (i.e., would expand beyond the current focus on pure cultures of AFA to all nuisance blooms)
- Selection of criteria for exclusion of biota/ endangered species from filtration using fish barriers or fencing
- Debris exclusion using debris fencing and floatable booms
- Minimization of biomass compaction to avoid cell damage and subsequent algal toxin release
- Utilization of air drying to dewater biomass
- Biomass storage and transportation/ disposal/ secondary utilization

There are also several specific design elements for each of the filtration options; these are presented in Appendix C.

4.4.4 General siting requirements

The following are general siting considerations for land-based filtration or flotation/separation treatment options:

- Close proximity to water body where treatment is to occur
- Potential slope for gravity feed to minimize pumping
- Size and land availability (existing holdings–public/private)
- Existing water control and conveyance systems/structures
- Proximity to disposal/secondary application

The following are general siting considerations for *in situ* barge filtration options:

- Existing commercial operations (no land acquisition necessary)
- Dockage
- Opportunities for biomass off -loading

- Proximity to disposal/secondary application

4.4.5 Treatment cost estimates

Several cost considerations are common to all filtration options:

- Water body volume
- Range and mean of algal biomass per unit volume
- Algal nutrient concentration in biomass
- Potential waste disposal issues if elevated algal toxin levels are present and would remain viable over time

Specific cost considerations for *in situ* barge filtration are given below:

- Vessel costs
- Filtration unit costs and maintenance estimates per unit
- Dockage facilities and biomass off-loading costs

Cost considerations for land-based filtration and flotation/separation are presented in Appendix C, because cost estimates for these two technologies are still being developed. Additional information will be presented at the workshop, if available.

4.4.5.1 Cost estimates

In situ barge filtration

Using information gathered from private enterprise harvesting algae on Upper Klamath Lake, the following cost estimates are provided for *in situ* barge based removal of algal biomass (Table 9). These estimates are expected to be viable for any of the possible treatment locations. Additional detail regarding cost assumptions and estimates is provided in Appendix C.

Table 9. Summary of cost estimates for *in situ* barge filtration of algae/biomass from the water column assuming one barge unit operating for 100 days per year in Upper Klamath Lake with a filtration capacity of roughly 900 kg (2,000 lbs) per day (dry weight) (additional details in Appendix C).

Cost considerations	Cost estimates for project life
Capital costs	$300,000
O&M	$3.4M
Total cost for project life (10 years)	$3.7M
Unit cost TN removal ($ per kg)	$8
Unit cost TP removal ($ per kg)	$61
Unit cost TN transportation and disposal for landfill ($/kg)	$7
Units cost TP transportation and disposal for landfill ($/kg)	$53

Note: The units in this table may be mixed (i.e., English and metric); however, the units are presented as typically encountered in engineering, planning, or regulatory areas, or the scientific literature, whichever is most applicable. Appendix D presents common unit conversions.

4.4.6 Possible treatment locations/general opportunities and constraints

Upper Klamath Lake, by far the largest water body in the basin, is the site of current active algal harvest (commercial) for dietary supplement products. Several private companies operate here due to the large size and seasonal abundance of AFA. The presence of significant water control structures in the lake is also beneficial with respect to possible retrofitting for filtration. Upper

Klamath Lake is also a prime location for *in situ* removal by filtration barges, given the large size of the lake and resulting ease of mobility for large watercraft. Keno Reservoir is also a prime location for application of these technologies, although given that algal blooms have been observed to originate in Upper Klamath Lake and senesce in Keno Reservoir, filtration and harvest of live algae as a dietary supplement is potentially a less attractive option. Several pre-existing canals and water control structures are located in or proximal to Link River Dam (see Figure 10) and Keno Reservoir; these would be potentially available for co-location of land-based technologies. Finally, a history of seasonal *Microcystis aeruginosa* blooms in Copco 1 and Iron Gate reservoirs makes them prime locations for both land-based and *in situ* technologies. For any of the approaches (i.e., land- or water-based), potential hazardous waste disposal issues may need to be considered, such as batch testing of biomass, if elevated concentrations of algal toxins are present and would remain viable over time in materials to be landfilled. Similar considerations would likely apply for land application of biomass.

4.5 Sediment Removal (Dredging)

4.5.1 Goals and capabilities

Dredging is the removal of accumulated lake sediments (i.e., muck) from lake bottoms with the goal of improving water quality, recreation, navigation or other uses (IEPA and NIPC 1998). Dredging can improve water quality by removing nutrient-rich sediments and decomposing organic plant matter, thereby reducing the amount of nutrients available. In some cases, dredging can improve water quality by deepening a lake or other water body enough to create thermal stratification and limit nutrient movement from the deeper areas to the upper waters. The entire lake bottom can be dredged or dredging can be conducted in certain zones where it may be most beneficial (e.g., areas with the thickest sediment layer or greatest concentration of nutrients).

There are two primary methods used for lake dredging: mechanical dredging and hydraulic dredging. Dry mechanical dredging involves either partially or completely draining a lake to expose the sediments to freezing or drying conditions. Then earthmoving equipment, such as bulldozers, scrapers, backhoes and draglines, is used to remove the sediment and transport it to a disposal site. Wet mechanical dredging does not require draining the lake; in this type of dredging, backhoes, draglines or grab buckets are used to scoop sediment either from the shoreline or from a floating barge. Wet mechanical dredging typically causes severe sediment resuspension and turbidity. Mechanical dredging is generally not well-suited for removing lake sediments because of drawdown requirements (dry) and turbidity impacts (wet).

Hydraulic dredging is the preferred method for dredging lake sediments because it does not require drawing down lake levels, is faster than mechanical dredging, creates less turbidity and can effectively remove loose, watery sediments. Thus, hydraulic dredging is the focus of this write-up. Two primary types of hydraulic dredges are horizontal auger dredges and cutterhead dredges. Horizontal auger dredges use vertical “knives” on a rotating auger, which is mounted on a boom at the front of the dredge hull. The dredge moves across the lake by a winch attached to a cable and loosens and cuts the sediments. This type of dredge works well for silty sediments, causing minimal turbidity, and can remove up to 120 yd^3 per hour. The cutterhead dredge uses a rotating “basket” of smooth or toothed blades, mounted to a boom at the front of the hull, to dislodge sediments which are then sucked up a pipe using a centrifugal pump (Figure 27). Cutterhead dredges can dredge at deeper depths and remove higher quantities of sediments (150–350 yd^3 per hour) than horizontal auger dredges.

Hydraulic dredging requires the design and construction of a settling basin to which the slurry (sediment/water mix) is piped (Figure 28). There the sediments settle from the water column. In some cases the sediment slurry can be decanted prior to being transported to the settling basin, which significantly decreases the amount of land required for the basin (CDM 2011). After settling (and treatment, in some cases), the water is pumped back into the lake and the sediments are left in the basin to dry. Selecting a suitable site for these settling basins requires the consideration of many factors. Example factors include:

- Land ownership and availability to accommodate the amount of material to be dredged (e.g., several acres to tens of acres) at an upland location
- Amount of land to be cleared
- Depth of water table below the surface (e.g., 4 feet or more)
- Road access
- Distance from actual dredging operations

Ultimately the sediments are utilized or disposed of in a variety of ways, including agricultural soil augmentation, fill for planned projects or, if contaminated, hauled to a landfill.

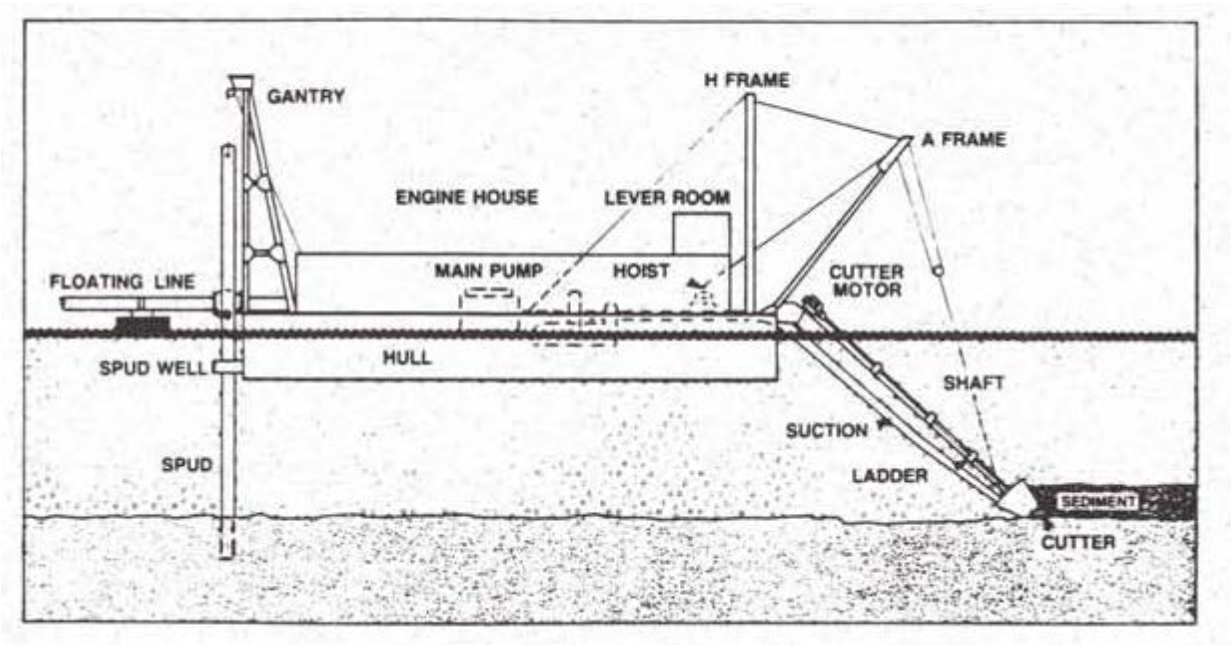

Figure 27. Configuration of a typical cutterhead dredge (IEPA and NIPC 1998).

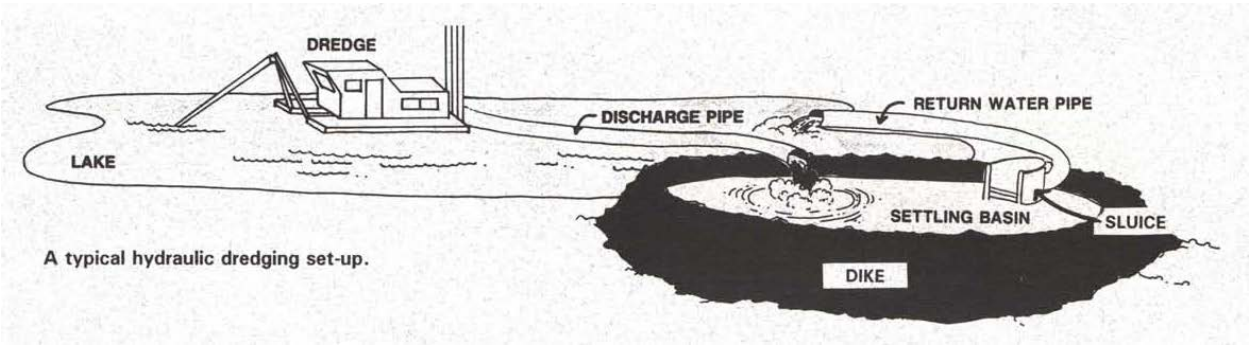

Figure 28. A typical hydraulic dredging set-up (IEPA and NIPC 1998).

Potential ecological and environmental impacts associated with dredging include impacts to sensitive aquatic species, loss of benthic organisms and submerged aquatic vegetation, temporary water quality impacts during dredging operations (e.g., increased turbidity) and temporary impacts to terrestrial systems due to noise and disturbance from operations (USACE 2009). Impacts to sensitive aquatic species can be avoided by selecting a dredge type that reduces entrainment and temporarily relocating less mobile organisms during dredging operations. Impacts to benthic organisms and submerged aquatic vegetation are unavoidable; however, muck does not provide a suitable habitat for desired species and nearby organisms are expected to recolonize the area following dredging operations. Adult fish generally avoid areas where dredging is taking place; however, dredging operations should be designed to avoid certain windows of time when fish are performing critical life history functions (e.g., spawning). Temporary water quality impacts can be lessened by using equipment that includes turbidity barriers (e.g., silt curtains) and selectively targets specific sediment layers. Finally, noise and other disturbances to wildlife are unavoidable, but will be temporary in nature.

Dredging requires a permit from the U.S. Army Corps of Engineers (USACE). USACE consults with the National Marine Fisheries Service (NMFS) and the U.S. Fish and Wildlife Service (USFWS) on permit applications to ensure that species are protected; permits may include conditions to avoid, minimize and/or mitigate impacts to species. An "Incidental Take Permit" may be issued by the NMFS or USFWS for projects that may only incidentally "take" or harm species during the course of work (http://cms.oregon.gov/OWEB/docs/pubs/permitguide.pdf, Accessed July 2012).

4.5.2 Similar applications

Recent lake hydraulic dredging operations that may be similar to Upper Klamath Lake include the following:

- Burnaby Lake, British Columbia
- Lake Okeechobee, FL
- Lake Trafford, FL
- Vancouver Lake, WA

Summaries of these projects are provided in Appendix C. In addition to providing information on basic design elements, siting requirements and treatment costs, several key lessons learned can be gleaned from these case studies, including (1) the benefits of conducting pilot dredging operations prior to full-scale application, (2) the potential to obtain equal or greater benefits at a lower cost by targeting certain areas, (3) the need to control external nutrient sources to fully address impacts and (4) the need for well-planned operations and maintenance (O&M) activities after dredging is complete to ensure long-term benefits.

4.5.3 Basic design elements

Large lake dredging operations, while having many characteristics in common, must be tailored to local conditions and constraints (e.g., thickness of sediment layer, presence of known contaminants, final means of disposal). General design considerations for large lake dredging and disposal of dredged material include the following:

- Identification of area(s) to be dredged
- Sediment composition
- Presence of woody debris, which can inhibit hydraulic dredging

- Transportation options
- Selection of dredging methodology
- Determination of presence or absence of contaminants
- Siting of settling basins
- Possibility of using gravity flow for sediment slurry transport
- Sediment settling rates
- Selection of dredged material disposal
- Presence of endangered species; identification of species to be relocated during dredging operations
- Selection of barrier material to minimize sediment resuspension

4.5.4 General siting requirements

In the instance that dredging an entire lake area is not feasible, certain areas where the potential benefit of dredging is greatest could be considered. The following are some general siting requirements for selecting dredge locations:

- Presence of highest nutrient concentrations
- Presence of a thick target mud layer if nutrient concentrations are evenly distributed
- Area generally free of debris and large obstructions
- Presence/absence of contaminated sediments (could be problematic for disposal)
- Location of sediment inputs (may want to conduct deeper dredging near input areas to create a sediment sink)
- Areas with low presence of special status species and associated habitat

4.5.5 Treatment cost estimates

The following list includes information needed to estimate direct costs of dredging operations; this list does not include associated indirect costs (e.g., disposal costs) (Farnham 2010).

- Volume of sediment to be removed
- Selection of most appropriate dredging technology (hydraulic vs. mechanical)
- Required dredging equipment (including dredge head)
- Time required to remove total sediment volume
- Required dredge operating hours
- Dredge operating costs (i.e., fuel, personnel, routine maintenance and repairs, production rates and equipment depreciation)

A summary of cost estimates is provided for sediment dredging in Upper Klamath Lake in Table 10. Additional detail regarding cost assumptions and estimates is provided in Appendix C.

Table 10. Summary of cost estimates for sediment removal (dredging) of the top 10 cm of sediment in Upper Klamath Lake and Keno Reservoir (additional detail in Appendix C). These estimates do not include disposal costs.

Cost considerations	Cost estimates for project life
Capital and O&M costs	$5–15/yd^3
Total cost for project life (Upper Klamath Lake)	$153M–$458M
Total cost for project life (Keno Reservoir)	$490,000–$1.5M
Unit cost TN removal ($ per kg)	Insufficient information to estimate
Unit cost TP removal ($ per kg)*	$110–$329

*Assumes TP concentrations in sediments in Keno Reservoir are the same as those in Upper Klamath Lake (see also Appendix C).

Note: The units in this table may be mixed (i.e., English and metric); however, the units are presented as typically encountered in engineering, planning, or regulatory areas, or the scientific literature, whichever is most applicable. Appendix D presents common unit conversions.

4.5.6 Possible treatment locations/general opportunities and constraints

4.5.6.1 Upper Klamath Lake

Upper Klamath Lake traps coarse sediment (sand and coarser such that downstream impoundments (i.e., Keno, J.C. Boyle, Copco and Iron Gate) contain a very small proportion of coarse sediment and are dominated by silt and clay (Eilers and Gubala 2003, Stillwater Sciences 2004). In addition, the water content of sediments in Upper Klamath Lake is considerably higher than the reservoirs, ranging from 84% to over 97% at the sediment-water interface (Eilers et al. 2004). Sediment accumulation rates in Upper Klamath Lake have increased significantly during the 20th century, with an average accumulative rate of 22 g/m^2/year for the upper 10 cm of sediment (Eilers et al. 2004).

The primary intended ecological benefits of dredging Upper Klamath Lake are to reduce the lake's nutrient (primarily phosphorus) loads concentrations and decrease associated algal blooms, which are severely degrading the lake's water quality conditions. Two-thirds of the lake's phosphorus load comes from internal sources, which include lake sediments (Simon and Ingle 2011). Total phosphorus concentrations are highest in the top 1 cm of sediment; however, the total concentration of phosphorus per weight of sediment (median = 0.61 ± 0.19 mg/g) is actually lower than that of other lakes throughout the country (Simon et al. 2009). The issue with Upper Klamath Lake is the amount of phosphorus stored in its sediments. Given the large area of the lake (approximately 232 km^2) and the median phosphorus concentration value, the calculated amount of phosphorus in the top 10 cm of sediment is approximately 1,392 metric tons (Simon et al. 2009). Simon and Ingle (2011) found that total phosphorus was highest in the northern portions, Howard Bay and the southernmost portion of the lake, whereas carbon and nitrogen concentrations were highest in the southern and northern portions of the lake. Based on the knowledge that total phosphorus is highest in the top layers of sediment and is spatially concentrated in certain areas of the lake, dredging operations could potentially target those areas and depths and reduce costs.

Eilers et al. (2004) found elevated nitrogen (^{14}N and ^{15}N) concentrations in the upper sediment layers (i.e., 20% enrichment in the top 17 cm). Whereas phosphorus concentrations drop off considerably below 20 cm, nitrogen concentrations gradually taper off, such that N:P ratios are generally above 20 in the sediment layers below 17 cm and below 20 in the layers above 17 cm.

This indicates that deeper dredging may be required to remove additional nitrogen; however, more information is needed on the contribution of sediments to in-lake nitrogen loading. Disposal of dredge material from Upper Klamath Lake could involve in-basin or out-of-basin facilities. Studies conducted for the KHSA focusing on potential dredging opportunities in J.C. Boyle, Copco 1, and Iron Gate reservoirs can help inform potential disposal options for dredged materials from nearby Upper Klamath Lake. The KHSA studies indicated that the nearest landfills are at least 20 miles away from the hydroelectric reservoirs, posing a significant transportation and sediment disposal constraint (CDM 2011, Lynch 2011). Nearby settling basins are the preferred alternative for these reservoirs, particularly if the slurry can be decanted prior to transfer, thus reducing the amount of land required for disposal. This finding more than likely applies to Upper Klamath Lake as well. Thus, the working assumption is that all dredge material from Upper Klamath Lake would be utilized or disposed of within the upper Klamath Basin, and relatively proximal to Upper Klamath Lake.

4.5.6.2 Keno Reservoir

Studies of Keno Reservoir indicate that there are approximately 144,000 $yd^3$of sediment, 46,000 yd^3 of which are sand and coarser and the remainder (98,000 yd^3) is composed of silt and clay (PacifiCorp 2003). In their bathymetric study, Eilers and Gubala (2003) found that much of the reservoir was characterized by reflective, irregular substrate, indicating rock interspersed with depositional material. In the forebay (i.e., the deeper area immediately upstream of the dam) of Keno Reservoir, the study indicated significant quantities of soft, flocculent material (Eilers and Gubala 2003). Dredging was conducted in the forebay area in 2002 to remove approximately 10,000 yd^3 of sediment and enhance flow to the fish ladder intake (PacifiCorp 2004).

4.6 Water Column Oxidation/Sediment Sequestration (Phosphorus Inactivation)

Often the implementation of sediment phosphorus sequestration (inactivation) and aeration/oxygenation of a water column management have approaches that are in common or complimentary relative to achieving a specific water quality goal. Because of this shared potential, Section 4.6 is intended to address water quality objectives, sediment phosphorus sequestration and water column aeration/oxygenation.

4.6.1 Goals and capabilities

4.6.1.1 Sediment phosphorus sequestration (inactivation)

Alum is the most widely used in-lake technique to inactivate sediment phosphorus and reduce internal phosphorus loading in lakes. There were 150 recorded alum treatments to lakes by 2005–most of these occurred in the United States (Welch and Gibbons, NALMS 2005). There have been many more since and many more have presumably gone unrecorded. Alum is also the favored coagulant to remove phosphorus from wastewater and suspended solids from drinking water.

Alum added to water forms an aluminum hydroxide ($Al(OH)_3$) commonly called “floc” onto and into which phosphorus is sorbed and this polymeric matrix settles out rather quickly (Figure 29). This polymeric matrix is incorporated into the sediment and sinks until it reaches a sediment depth of similar bulk density over time. This process does not form a sediment cap and is not a biological barrier. The appropriate alum dose to a lake is based on the quantity of mobile phosphorus in sediment and overlying water and is usually around 50-100 g/m^2 of lake surface

area (Cooke et al., 2005). The advantage of alum application is its insensitivity to redox conditions, such that phosphorus remains sorbed to aluminum hydroxide polymer even during seasonal periods of anoxia in the sediments and/or water column. The main precaution associated with alum use is the prevention of low pH (< 6.0) that can directly or indirectly (solubilize the free aluminum ion [Al^{3+}]) and harm aquatic life. To maintain appropriate pH, sodium aluminate is added along with alum as a buffer. Any repeat or continuous phosphorus inactivation treatment would require buffering in the relatively low alkalinity waters of the Klamath Basin.

Aluminum is one of the most abundant elements on earth. It is constantly solubilized from soil and bedrock through weathering by precipitation that is naturally acidic. However, calcium and magnesium are also naturally weathered, producing alkalinity and pH ranges in natural waters that render aluminum non-toxic. While some inorganic forms of aluminum can be toxic to aquatic animals at high and low pH, at circumneutral pH (6–8) and equilibrium conditions the insoluble and non-toxic form of aluminum prevails. Aluminum toxicity was an important reason that low alkalinity, poorly buffered lakes in the northeastern United States and Scandinavia became fishless due to strongly acid precipitation (i.e., relatively low pH).

While toxic effects have been produced under laboratory conditions at concentrations from 1 to a few mg/L aluminum and circumneutral pH (USEPA 1988), these conditions do not prevail during or after a buffered alum treatment in natural waters. As opposed to laboratory conditions where exposure to continuous renewal of concentrations of aluminum over days artificially maintains aluminum concentration in the water, the alum floc forms quickly during treatment and settles to the lake bottom. This process removes aluminum from the water column. Alum treatments ranging from 5–26 mg/L, in which fish and aquatic life were studied before and after treatment, have shown very few negative, and usually positive effects, to aquatic biota (see Chapter 8 in Cooke et al. [2005]). This is because (1) residual aluminum concentration remaining in the water column are relatively low (0.1–0.2 mg/L); (2) pH remains above 6, due to buffering; (3) only a fraction of a given waterbody is treated each day allowing avoidance of the immediately treated area by fish and other non-benthic aquatic species, and; (4) much of the residual aluminum is likely to be chemically complexed with dissolved organic matter, which is abundant in eutrophic lakes, rendering the aluminum non-bioavailable and non-toxic (see Chapter 8 in Cooke et al. [2005]). None of the studied alum treatments resulted in fish kills. In fact, effects on benthic animals were usually beneficial, increasing diversity and abundance, because oxygen levels increased as a result of lower phosphorus and algal produced oxygen demand. A thorough review of alum effects on the treated aquatic environment is given in Cooke et al. (2005).

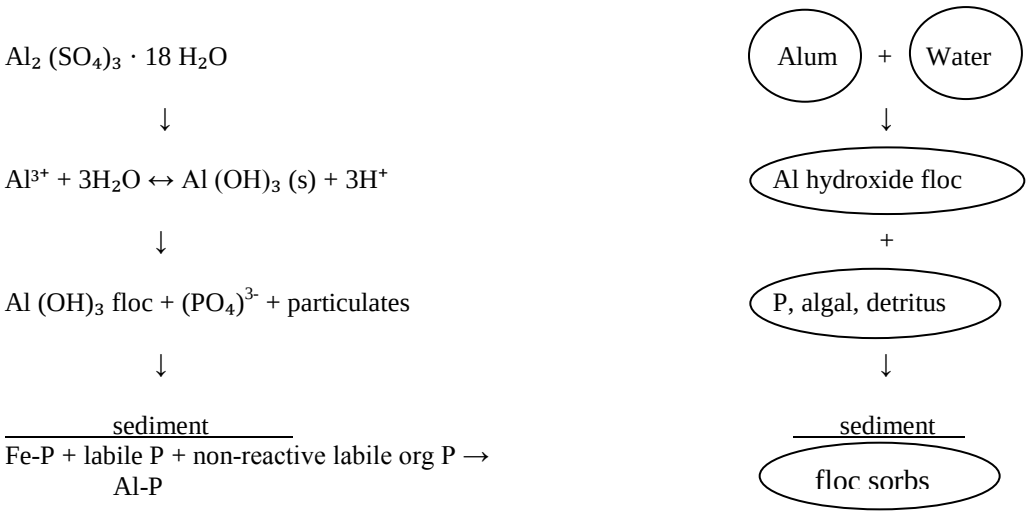

Figure 29. Process of sediment phosphorous sequestration (inactivation).

Alum treatments have increased over the past four decades, such that the procedure is now considered to be routine and one of the most frequently used in-lake methods of restoration. Monitoring of pH and dissolved oxygen at frequent intervals following application has indicated that these constituents remain in ranges safe to aquatic life and aluminum does not occur in its toxic form. Therefore, there is widespread consensus among lake scientists that alum is effective and safe at sequestering and inactivating phosphorus (Osgood et al., 2011).

Effectiveness of sediment phosphorous sequestration (inactivation)

Treatment effectiveness/longevity of sediment phosphorus inactivation using alum was evaluated in 21 lakes in 1999 (Cooke et al. 2005). Reduction in sediment phosphorus release rate (internal loading) initially averaged about 70% for the shallow, thermally unstratified lakes and 85% for stratified lakes. Summer total phosphorus (TP) concentration in the water was reduced by about half in stratified and unstratified lakes alike, and chlorophyll and the biomass fraction as cyanobacteria decreased similarly. Longevity of treatments varied, but usually about 10 years can be expected with effectiveness waning over time as the alum floc layer sinks and new sediment with un-complexed phosphorus settles and covers the alum layer.

Alum injection

Alum has been directly injected into inflows to lakes or into stormwater retention ponds on a continual basis in several states. While the published literature on the subject is sparse, the general opinion from users is that it can be effective. Injection into lakes through aeration/circulation systems is also an option. Doses are relatively low to avoid problems of low pH and accumulations of floc in flowing water, although a collection system may be necessary. Unless the water body is small, alum injection without aeration to inflows is usually not effective at inactivating sediments throughout a lake due to low sedimentation rates resulting in limited floc distribution. Injecting alum through an aeration system, creating a continuous micro-alum floc during certain times of the year, can be more effective at distributing alum to sediments throughout the lake while simultaneously inactivating phosphorus in the water column carried into the lake from external sources.

4.6.1.2 Aeration/oxygenation

There are two principal techniques used to increase dissolved oxygen (DO) in lakes and reservoirs; 1) Complete circulation and destratification, which aerates/oxygenates the whole water body, and 2) Oxygenation of the entire hypolimnion, or only a horizontal layer of the water column (Cooke et al, 2005). Increasing the aerobic habitat is a goal with both techniques, as well as phosphorus reduction with hypolimnetic aeration, if iron is controlling phosphorus release from the sediments. However, phosphorus removal is usually not a principal objective with complete circulation because destratification and water column mixing can increase phosphorus availability to algae.

Complete circulation

The most frequently used aeration technique is the addition of compressed air through diffuser hoses placed along the lake bottom (Figure 30). The resulting unconfined plume of air bubbles rises through the water column causing the entrained water mass to circulate throughout the lake. Oxygenation occurs when the under saturated-mass is exposed to atmospheric exchange. Circulation has also been achieved with pumps or jets.

In addition, to oxygenating the water column, complete circulation can cause light limitation and reduce algal biomass if the lake is sufficiently deep. Also, mixing can reduce cyanobacteria by neutralizing their buoyancy regulation. However, circulation poses a risk of even worse algal problems if the air flow rate is too low and only weak mixing of high nutrient water occurs (Cooke et al. 2005).

Hypolimnetic aeration/oxygenation

This technique is used to oxygenate only the hypolimnion while maintaining water column stratification. Hypolimnetic aeration/oxygenation can provide a cool-water habitat for cold-water fish, a daily predation refuge for zooplankton and a domestic water supply by adding oxygen to lake or reservoir bottom waters without mixing the water column. Oxygenation is achieved through either full or partial air lift units (Figure 30), or by injecting pure oxygen at depth with a pump (deep oxygen injection system–DOIS) or into water pumped through a down flow bubble contact system (DBCS). Also, hypolimnetic water can be pumped to the surface, where it obtains air bubbles, and then pumped back to the hypolimnion. Naturally oxygenated epilimnetic water can also be pumped into the hypolimnion to provide the needed oxygen. Hypolimnetic phosphorus and internal phosphorus loading from anoxic sediments are typically reduced with oxygenation if sufficient iron is available for complexation of phosphorus. Layer aeration can also reduce phosphorus (Figure 30).

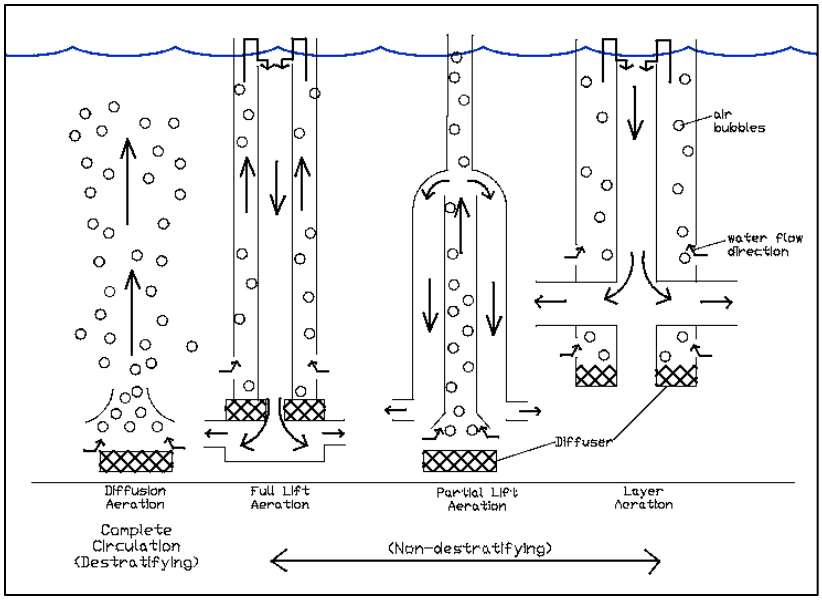

Figure 30. Aeration schematics for complete circulation and non-mixing hypolimnetic aeration.

The aforementioned aeration techniques have been widely applied to lakes and reservoirs throughout the world for over sixty years. Cooke et al. (2005) lists 51 cases of artificial circulation that were studied, mostly in the 1960s and 1970s, and 28 of hypolimnetic aeration in the 1970s, to 1990s. However, most aeration applications have gone unreported in the peer reviewed literature.

4.6.2 Basic design elements

Individual project designs must be tailored to local conditions and constraints. However, basic design consideration for phosphorus sequestration and water column aeration include the following outline.

4.6.2.1 Sediment phosphorous sequestration (inactivation)

Design of sediment phosphorus sequestration approach is based upon the five following basic design considerations:

- The area of sediment and/or volume of water to be treated can directly define costs and implementation strategy;
- The amount of material needed to inactivate or remove phosphorus (dose) defines both cost and approach;
- The implementation strategy defines both effectiveness and longevity of management impact on phosphorus sequestration. Strategy has three basic alternatives. The first strategy is maintenance, which is a repeated treatment or continuous treatment, such as water – column stripping of phosphorus or continuous injection of alum to remove phosphorus from the water column and build-up aluminum concentration to inactivate sediment phosphorus over time. The second strategy is sediment inactivation, which is directly adding a sufficient dose to both strip the water column of phosphorus and to provide a direct infusion of aluminum into the sediments to inactivate sediment phosphorus. The third strategy is interception of phosphorus before it is of available for biological uptake;

- Alum supply logistics both relative to transportation cost and amount of alum needed for the treatment; and,
- Treatment preparation area availability and mitigation relative to treatment staging can be temporary or permanent depending upon the implementation strategy.

The following summarizes the five basic design element considerations:

1. Size of water body to treat
2. Alum dose required (typically 50-100 gAl/m² of lake surface area)
3. Application strategy (maintenance–water column stripping or injection), sediment inactivation, or inception
4. Logistical constraints posed by alum volume required and proximity to supply
5. Availability/location of application staging area

4.6.2.2 Aeration/oxygenation

Aeration/Oxygenation design depends upon the water body morphological conditions that define both the water volume and the physical parameters to be addressed, volume and rate of air or oxygen needed to increase dissolved oxygen in the water column, the demand for oxygen that reduces the dissolved oxygen concentration in the water column, characteristics of the air or oxygen circulation system, the amount of air or oxygen needed to overcome both demand and distribution needs, and air or oxygen delivery system. The following summarize the design elements basic considerations:

1. Water body area/width/depth
2. Compressed air capacity for complete circulation method–adhere to criterion for air flow per area per time
3. DO demand within the sediments and water column for hypolimnetic aeration/ oxygenation
4. Hose length and pore size for air transport
5. DO demand for hypolimnetic aeration/oxygenation and air/oxygen needed to exceed that rate
6. Choice of air/oxygen injection device

4.6.3 General siting requirements

The following are general siting requirements for phosphorus sequestration and water column aeration:

Sediment phosphorous sequestration (inactivation)

1. Availability of appropriate staging area for alum storage and barge handling–depending on the size of the application this typically ranges from 5,000-10,800 ft^2 but can be as large as 100,000 ft^2
2. Barge/truck access to staging area
3. Source of electrical power for pumps, injectors and compressor(s)

Aeration/oxygenation

1. Source of electrical power for compressor(s)
2. Near shore space for compressor(s)/oxygenating device(s)–typically 500 ft^2
3. Iron (Fe) content of water is below 15:1 Fe:P, Fe will not control P concentration regardless of dissolved oxygen concentrations

4. Access for heavy equipment

4.6.4 Treatment cost estimates

The following are general items for which unit costs and counts are needed for estimating the treatment costs of phosphorus sequestration. Given available information, cost estimates are provided in Section 4.6.5.

Sediment phosphorous sequestration (inactivation)

1. Sediment phosphorus fraction analysis
2. Water column phosphorus, alkalinity and pH
3. Water body area, volume and depth
4. Jar test data to determine application ratio of alum-to-sodium aluminate
5. Alum and sodium aluminate cost

The following are general information requirements for estimating the treatment costs of aeration/oxygenation of the entire water column, the hypolimnion only, or a horizontal layer of the water column. Given available information, cost estimates are provided in Section 4.6.5.

Aeration

1. Compressor(s)
2. Airlines / supports
3. Power
4. Installation
5. Maintenance of equipment
6. Monitoring DO / temperature

4.6.5 Possible treatment locations/general opportunities and constraints

4.6.5.1 Application of sediment phosphorous sequestration to Upper Klamath Basin waters

Upper Klamath Lake

Treatment of the entire Upper Klamath Lake to inactivate sediment phosphorus is likely to reduce outflow TP by 50–70% during summer when 80% of phosphorus loading is internal. This estimate is based on studies of shallow lakes where internal loading rates are reduced 70% on average, despite low alum doses of 11-30 g Al/m² (Welch and Gibbons 2005). Doses determined from more recent sediment data are now higher; Green Lake, in Seattle, Washington, was treated in 2004 at 98 g Al/m². Thus, using a higher dose in Upper Klamath Lake may result in greater effectiveness than observed in previous cases.

However, there are logistical and cost issues to treating Upper Klamath Lake itself. Until 2011, the largest known lake treated with alum is the 700-ha (1,730-acre) Grand Lake, in St. Marys, Ohio (Gibbons, 2012 unpublished data). Upper Klamath is 26,800 ha (66,000 acres), or roughly 13 times larger than any treatment to date. The shallow lakes cited above ranged from approximately 45–140 ha (approximately 110–350 acre). Treating a large lake is possible by up-scaling equipment (i.e., sizes and number of barges and storage tanks). Tetra Tech has supervised treating the middle 40% of a 5,000-ha (12,360-acre) shallow lake in Ohio with low doses in 2011 and 2012 at a cost of $3.5 million per year. Tetra Tech recently estimated the cost of treating one third of the 233,000-ha (575,760-acre) Lake Okeechobee to be $274 million. Given that the

proposed treatment area of Lake Okeechobee was approximately 80,000 ha (approximately 198,000 acres) or about three times the area of Upper Klamath Lake, and adjusting for current price of materials, alum treating of the entire Upper Klamath Lake would cost an estimated $91 to $180 million. The effective longevity of this type of phosphorus inactivation treatment in Upper Klamath Lake would be from 8 to 15 years, assuming that external loading would continue to add new phosphorus to the lake sediments. This means that Upper Klamath Lake would need to be treated 3 to 6 times in a 50 year period to meet management goals. This is equivalent to a unit removal cost of approximately $260 per kg phosphorus.

Keno Reservoir

A more cost-effective approach to reducing phosphorus concentrations downstream of Upper Klamath Lake and through Copco 1 and Iron Gate Reservoirs may be injection of alum into the Klamath River and Lost River Diversion canal prior to their entering Keno Reservoir or as it enters Keno Reservoir (Lake Ewauna). The reservoir would allow settling time and a deposition site for the alum floc and its sorbed phosphorus. This approach would direct treatment at the major source of phosphorus to downstream waters. Furthermore, there should be some benefit from the settled floc inactivating sediment phosphorus in Keno Reservoir. The settled inactivated Al-P would not be resolubilized in the anoxic waters of Keno Reservoir, because aluminum is not sensitive to low redox conditions. Such treatment should have the benefit of reducing *Microcystis aeruginosa* blooms in Copco 1 and Iron Gate Reservoirs, thereby reducing high summertime oxygen demand. An added benefit of employing alum injection through microfloc technology would be that it also uses aeration to create the microfloc, thus adding dissolved oxygen to the system.

The cost of a microfloc injection system that would be operated from April through October would be approximately $5 to $8 million in capital costs for injection facilities and installation, with annual operation and maintenance including costs of $2 to $4.5 million per year for materials, electricity, and equipment maintenance, assuming a 20 year project life. This is equivalent to a unit removal cost of approximately $49 per kg phosphorus.

4.6.5.2 Application of aeration/oxygenation to Upper Klamath Basin waters

There are three water bodies where aeration/oxygenation techniques would be likely to increase seasonal dissolved oxygen concentrations and result in beneficial phosphorus reduction:

- Keno Reservoir–complete circulation
- Copco 1 and Iron Gate Reservoirs–hypolimnetic aeration / oxygenation

The entire Keno Reservoir water column experiences anoxia, or near anoxic conditions, for approximately 8 miles during summer months when high inflow concentrations of TSS and nutrients originate largely from large seasonal algal blooms and internal loading of phosphorus in Upper Klamath Lake. Complete circulation of the water column at one or more points in the reservoir would increase dissolved oxygen during critical summer periods and may enhance phosphorus retention. Phosphorus retention using complete circulation will depend on how much iron is present in the water column and HRT of Keno Reservoir. If alum injection in the inflows to Keno Reservoir is employed, circulation may not be necessary to retain phosphorus. In that event, the benefit of circulation would be to enhance aquatic life and increase the decomposition rate of the high inflow BOD, which can be 10-20 mg/L.

Aerating or oxygenating the hypolimnion of Copco 1 and Iron Gate Reservoirs would increase dissolved oxygen concentrations in these waterbodies during critical summer periods and could also increase phosphorus retention. While available dissolved oxygen data indicate that anoxia is not consistently reached in bottom waters, concentrations do reach 1 mg/L in the bottom 10 m or so of Copco 1 and Iron Gates Reservoirs and surficial sediments may be anoxic enhancing internal loading under those conditions. However, since some release of phosphorus from the reservoirs can occur at times that may stimulate downstream periphyton growth (see Section 3.7.1.4.1), retention of phosphorus in reservoir sediments due to hypolimnetic oxygenation may be beneficial.

Circulation of Keno Reservoir appears to offer the most benefit for sequestering phosphorus and reducing BOD. Data on iron concentration is needed to determine the feasibility and benefits from circulation.

5 REFERENCES

Ahn, H. and R. T. James. 2001. Variability, uncertainty, and sensitivity of phosphorus deposition load estimates in South Florida. Water, Air, and Soil Pollution 126: 29–32.

Aldous, A., P. McCormick, C. Ferguson, S. Graham, and C. Craft. 2005. Hydrologic regime controls soil phosphorus fluxes in restoration and undisturbed wetlands. Restoration Ecology 13: 341–347.

Aldous, A. R., C. B. Craft, C. J. Stevens, M. J. Barry, and L.B. Bach. 2007. Soil phosphorus release from a restoration wetland, Upper Klamath Lake, Oregon. Wetlands 27: 1025–1035.

Arora, H., J. R. DeWolfe, R. G. Lee, and T. P. Grubb. 1995. Evaluation of dissolved air floatation process for water clarification and sludge thickening. Water Science Technology 31: 137–147

Asarian, E., and J. Kann. 2011. Phytoplankton and nutrient dynamics in Iron Gate and Copco Reservoirs 2005–2010. Prepared by Kier Associates and Aquatic Ecosystem Sciences for Klamath Basin Tribal Water Quality Work Group, Klamath, California.

Asarian, E., J. Kann, and W. Walker 2009. Multi-year nutrient budget dynamics for Iron Gate and Copco Reservoirs, California. Prepared by Riverbend Sciences, Kier Associates, Aquatic Ecosystem Sciences, and William Walker for Karuk Tribe Department of Natural Resources, Orleans,
California. http://www.klamathwaterquality.com/documents/asarian_et_al_2009_Cop_IG_Budget_may05dec07_report.pdf

Asarian, E., J. Kann, and W. Walker. 2010. River nutrient loading and retention dynamics in free-flowing reaches, 2005–2008. Final Technical Report. Yurok Tribe Environmental Program, Klamath,
California. http://www.klamathwaterquality.com/documents/asarian_et_al_2010_klam_nutr_dynamics_final_report_revised.pdf

ACEC (Association of Consulting Engineering Companies, British Columbia). 2012. Award of Merit Burnaby Lake Rejuvenation Project–Urban Jewel. http://www.acec-bc.ca/media/11443/acecbcAwards12-2M.pdf

Banish, N. P, B. J. Adams, R. S. Shively, M. M. Mazur, D. A. Beauchamp, and T. T. Wood. 2009. Distribution and habitat associations of radio-tagged adult Lost River suckers and shortnose suckers in Upper Klamath Lake, Oregon. Transactions of the American Fisheries Society 138: 153–168.

Barbiero, R. P., and J. Kann. 1994. The importance of benthic recruitment to the population development of *Aphanizomenon flos-aquae* and internal loading in a shallow lake. Journal of Plankton Research 16: 1581–1588.

Barr, B. R., M. E. Koopman, C. D. Williams, S. J. Vynne, R. Hamilton, and B. Doppelt. 2010. Preparing for climate change in the Klamath basin. National Center for Conservation Science and Policy and The Climate Leadership
Initiative. http://www.geosinstitute.org/images/stories/pdfs/Publications/ClimateWise/KlamathBasinCFFReport_Final_Long_20100901.pdf

BARR. 2007. Kohlman Lake dredging feasibility study. Final Report. Prepared for Ramsey-Washington Metro Watershed District by BARR, Minneapolis, Minnesota. http://rwmwd.org/vertical/Sites/%7BAB493DE7-F6CB-4A58-AFE0-56D80D38CD24%7D/uploads/%7BD770596B-10F3-4F9E-AC20-322E7BC2FE7A%7D.PDF

BARR. 2009. Fountain Lake sediment dredging assessment. Draft Report. Prepared for Shell Rock River Watershed District by BARR, Minneapolis, Minnesota. https://wiki.umn.edu/pub/Albert_Lea/SRRWDdocs/Fountain_Lake_Dredging_Assessment_DRAFT.pdf

Bartholomew J. L., M. J. Whipple, D. G. Stevens, and J. L. Fryer. 1997. The life cycle of *Ceratomyxa shasta*, a myxosporean parasite of salmonids, requires a freshwater polychaete as an alternate host. Journal of Parasitology 83: 859–868.

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

Benoufella, F., A. Laplanche, V. Boisdon, and M. M. Bourbigot. 1994. Elimination of microcystis cyanobacteria (blue-green algae) by an ozoflotation process: a pilot plant study. Water Science and Technology 30: 245-257.

Bond, C. E., C. R. Hazel, and D. Vincent. 1968. Relations of nuisance algae to fishes in Upper Klamath Lake. Terminal progress report for Research Grant WP 00625. Prepared by Oregon State University, Department of Fisheries and Wildlife, Corvallis for U. S. Federal Water Pollution Control Administration.

Bortelson, G. C., and M. O. Fretwell. 1993. A review of possible causes of nutrient enrichment and decline of endangered sucker populations in Upper Klamath Lake, Oregon. Water-Resources Investigations Report 93-4087. U. S. Geological Survey.

Boyle, J. C. 1987. Fifty years on the Klamath. Journal of the Shaw Historical Society 1: 13–43.

Bradbury, J. P., S. M. Colman, and R. L. Reynolds. 2004. The history of recent limnological changes and human impact on Upper Klamath Lake, Oregon. Journal of Paleolimnology 31: 151–161.

Burnaby Parks, Recreation, and Cultural Services. 2011. City of Burnaby Fact Sheet. 6 April. http://lin.ca/Files/20998/City%20of%20Burnaby%20Fact%20Sheet.PDF

DWR (California Department of Water Resources). 2011a. Salton Sea Ecosystem Restoration Program. http://www.water.ca.gov/saltonsea

DWR. 2011b. Salton Sea Ecosystem Restoration Program. Salton Sea Species Conservation Habitat Project Frequently Asked Questions.. http://www.water.ca.gov/saltonsea/docs/faqs_schproject.pdf

Carmichael, W. W. 1994. The toxins of cyanobacteria. Scientific American 270: 78–86.

Carmichael, W. W., C. Drapeau, and D. M. Anderson. 2000. Harvesting of *Aphanizomenon flos-aquae* Ralfs ex. Born. & Flah. Var. *flos-aquae* (cyanobacteria) from Klamath Lake for human dietary use. Journal of Applied Phycology 12: 585–595.

Carpenter, K. D., D. T. Snyder, J. H. Duff, F. J. Triska, K. K. Lee, R. J. Avanzino, and S. Sobieszczyk. 2009. Hydrologic and water-quality conditions during restoration of the Wood River Wetland, Upper Klamath River Basin, Oregon, 2003-05. Scientific Investigations Report 2009-5004. U.S. Geological Survey. http://pubs.usgs.gov/sir/2009/5004

CDM. 2011. Klamath settlement process sediment management in the reservoirs. Unpublished report. http://klamathrestoration.gov/sites/klamathrestoration.gov/files/Sediment%20White%20Paper.pdf

CH2M Hill. 2009a. Analysis of microcystin in resident fish and mussel tissues in the vicinity of the Klamath Hydroelectric Project in 2008. Prepared for PacifiCorp, Portland, Oregon by CH2M Hill, Redding, California.

CH2M Hill. 2009b. Occurrence of microcystin in Chinook salmon and steelhead in the Klamath River in 2007. Prepared for PacifiCorp, Portland, Oregon by CH2M Hill, Redding, California.

CH2MHill. In preparation. Approaches to water quality treatment by wetlands in the upper Klamath Basin. Prepared for PacifiCorp Energy, in consultation with the Interim Measures Implementation Committee for the Klamath Hydroelectric Settlement Agreement.

Chesapeake Bay Program. 2004. What's the status of point source nitrogen reduction in the Chesapeake Bay watershed? http://archive.chesapeakebay.net/pubs/waterqualitycriteria/doc-pointsource_for_tribtools.pdf

Chorus, I.,and J. Bartram, editors. 1999. Toxic cyanobacteria in water. E & FN Spon, London.

Chow, W. 2011. Burnaby Lake dredging completed. Burnaby NewsLeader online. http://www.bclocalnews.com/news/121349114.html

Ciotti, D., S. M. Griffith, J. Kann, and J. Baham. 2010. Nutrient and sediment transport on flood irrigated pasture in the Klamath Basin, Oregon. Rangeland Ecology & Management 63: 308–316.

City of Burnaby Environment and Waste Management Committee. 2000. Options analysis for the full scale Burnaby Lake Dredging Project. https://burnaby.civicweb.net/Documents/DocumentList.aspx?ID=3602

Coleman, M. E., J. Kann, and G. G. Scoppettone. 1988. Life history and ecological investigations of catostomids from the Upper Klamath Lake basin, Oregon. Annual draft report. U.S. Fish and Wildlife Service, National Fisheries Research Center, Seattle, Washington.

Colman, S. M., J. P. Bradbury, and J. G. Rosenbaum. 2004. Paleolimnology and paleoclimate studies in Upper Klamath Lake, Oregon. Journal of Paleolimnology 31: 129–138.

Cooke, G. D., E. B. Welch, and S. Nichols. 2005. Restoration and management of lakes and reservoirs, Third edition. Taylor and Francis Group. New York, New York.

Connelly M., and L.Lyons. 2007. Upper Sprague watershed assessment. Klamath Basin Ecosystem Foundation, Oregon State University, Klamath Basin Research and Extension Center. https://nrimp.dfw.state.or.us/web%20stores/data%20libraries/files/OWEB/OWEB_916_2_Upper%20Sprague%20WS%20Assessment.pdf

David Evans and Associates, Inc. 2005. Williamson River Delta Restoration. Final Environmental Impact Statement. Prepared for Natural Resources Conservation Service, The Nature Conservancy, and U. S. Bureau of Reclamation. http://conserveonline.org/workspaces/Williamson.River.Delta.Preserve/wrdp-public-info/EIS%20Final%20June%202005.pdf

Dean, A.P., and D. C. Sigee. 2006. Molecular heterogeneity in *Aphanizomenon flos-aquae* and *Anabaena flos-aquae* (Cyanophyta): a synchrotron-based Fourier-transform infrared study of lake micropopulations. European Journal of Phycology 41: 201-212.

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

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

Dicken, S. N. 1980. Pluvial Lake Modoc, Klamath County, Oregon, and Modoc and Siskiyou Counties, California. Oregon Geology 42:179–187.

Duff, J. H., K. D. Carpenter, D. T. Snyder, K. K. Lee, R. J. Avanzino, and F.J. Triska. 2011. Phosphorus and nitrogen legacy in a restoration wetland, Upper Klamath Lake, Oregon. Wetlands 29: 735–746.

EA Engineering Science and Technology. 2002. Lake Okeechobee pilot dredging project report. http://www.sfwmd.gov/portal/page/portal/pg_grp_sfwmd_watershed/sediment_dredging190/tab744041/pilotdredgingreport.pdf

Eilers, J. M. and C. P. Gubala. 2003. Bathymetry and sediment classification of the Klamath Hydropower Project impoundments. Prepared by J. C. Headwaters, Bend for PacifiCorp, Portland, Oregon. http://www.pacificorp.com/content/dam/pacificorp/doc/Energy_Sources/Hydro/Hydro_Licensing/Klamath_River/Bathymetry_and_Sediment_Classification_Report_Final_April_2003.pdf

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

Eldridge, S. L. C., T. M. Wood, and K. R. Echols. 2012. Spatial and temporal dynamics of cyanotoxins and their relation to other water quality variables in Upper Klamath Lake, Oregon, 2007–09. Scientific Investigations Report 2012–5069. U.S. Geological Survey.

Elsroad, A., N. Rudd, and H. Hendrickson. 2011. Williamson River Delta Preserve vegetation monitoring: Tulana third-year post-breaching results. The Nature Conservancy, Klamath Basin Field Office, Klamath Falls, Oregon. http://conserveonline.org/library/williamson-river-delta-preserve-vegetation-2/@@view.html

Farnham, G. 2010. Comparative cost analyses and considerations of sediment dredging. Prepared for Board of Commissioners, Lake Sinissippi Improvement District. http://www.lakesinissippi.org/2010-02-10%20LSID%20Dredging%20Report.pdf

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

Fetcho, K. 2006. Klamath River blue-green algae bloom report: Water Year 2005. Prepared for Yurok Tribe Environmental Program, Klamath, California. http://www.yuroktribe.org/departments/ytep/Water.htm

Fetcho, K. 2008. 2007 Klamath River blue-green algae summary report. Yurok Tribe Environmental Program. Klamath, California.

Fetcho, K. 2011. 2009 Klamath River blue-green algae summary report. Yurok Tribe Environmental Program, Klamath, California.

Fleming-Singer, M. S., and A. J. Horne. 2006. Balancing wildlife needs and nitrate removal in constructed wetlands: the case of the Irvine Ranch Water District's San Joaquin Wildlife Sanctuary. Ecological Engineering 26: 147-166.

Friedrichsen, P. T. 1996. Summertime stream temperatures in the North and South Forks of the Sprague River, South Central Oregon. Master's thesis. Department of Forest Engineering, Oregon State University, Corvallis, Oregon. http://ir.library.oregonstate.edu/xmlui/handle/1957/9502

Gahler, A. R. 1969. Field studies on sediment-water algal nutrient interchange processes and water quality of Upper Klamath and Agency lakes. Working Paper 66. Federal Water Pollution Control Administration, Northwest Region, Pacific Northwest Water Laboratory, Corvallis, Oregon.

Gannett, M.W., K. E. Lite, Jr., J. L. La Marche, B. J. Fisher, and D. J. Polette. 2007. Groundwater hydrology of the upper Klamath Basin, Oregon and California. Scientific Investigations Report 2007-5050. U. S. Geological Survey.

GLDDC (Great Lakes Dredge and Dock Company). n. d. Lake Trafford restoration. http://www.gldd.com/ProjectItem.aspx?type=12&id=78

GMA (Graham Matthews & Associates). 2011a. 2007–2010 KBRT monitoring report, Volume 1: Surface water. Prepared for Klamath Basin Rangeland Trust, Klamath Falls, Oregon.

GMA. 2011b. 2007–2010 KBRT monitoring report, Volume 2: Water quality monitoring. Prepared for Klamath Basin Rangeland Trust, Klamath Falls, Oregon.

Golueke, C. G., W. J. Oswald, and H. B. Goates. 1957. Anaerobic digestion of algae. Applied Microbiology 5: 47-55.

Gorini, R. F. 1987. Lake restoration by dredging *in* T. R. Patin, editor. Management of bottom sediments containing toxic substances. Proceedings of the 13th U.S./Japan Experts Meeting, Baltimore, Maryland. http://www.dtic.mil/cgi-bin/GetTRDoc?AD=ADA253003#page=366

Graham, S. A., C. B. Craft, P. V. McCormick, and A. Aldous. 2005. Forms and accumulation of soil P in natural and recently restored peatlands—Upper Klamath Lake, Oregon, USA. Wetlands 25: 594–606.

Guiry, M. D. and G. M. Guiry. 2012. AlgaeBase. National University of Ireland, Galway. http://www.algaebase.org

Hajkowicz, S. A., and K. Collins. 2007. A review of multiple criteria analysis for water resource planning and management. Water Resources Management 21: 1553–1566.

Hamilton, A. 2012. Wood River wetland monitoring. Presentation at the Klamath Basin Monitoring Program conference in Yreka, California, April 5, 2012. U. S. Bureau of Land Management, Klamath Falls, Oregon. http://www.kbmp.net/images/stories/pdf/KBMP_minutes/meeting_10/Wood_River_Management_Studies_4_6_2011.pdf

Hardy, T. B. 1999. Evaluation of interim in-stream flow needs in the Klamath River: Phase I. Final Report. Prepared for U.S. Department of the Interior, Washington, D. C. by Institute of Natural Systems Engineering, Utah Water Research Laboratory, Utah State University, Logan, Utah. http://www.usbr.gov/mp/kbao/esa/10_HardyPhaseIReport_4_6_01.pdf

Hardy T., and C. Addley. 2001. Evaluation of interim instream flow needs in the Klamath River. Phase II. Final Report. Prepared for U.S. Department of the Interior, Washington, D.C. by Institute for Natural Systems Engineering, Utah Water Research Laboratory, Utah State University, Logan.

Hecht, B. and G. R. Kamman. 1996. Initial assessment of pre- and post- Klamath Project hydrology on the Klamath River and impacts of the project on instream flows and fishery habitat. Prepared by Balance Hydrologics, Berkeley for The Yurok Tribe, Eureka, California. http://www.balancehydro.com/pdf/9506Klamath.pdf

Hewitt, D. A., B. S. Hayes, E. C. Janney, A. C. Harris, J. P. Koller, and M. A. Johnson. 2011. Demographics and run timing of adult Lost River (*Deltistes luxatus*) and shortnose (*Chasmistes brevirostris*) suckers in Upper Klamath Lake, Oregon, 2009. U.S. Geological Survey Open-File Report 2011-1088.

Hoilman, G. R., M. K. Lindenberg, and T. M. Wood. 2008. Water quality conditions in Upper Klamath and Agency Lakes, Oregon, 2005. Scientific Investigations Report 2008–5026. U.S. Geological Survey. http://pubs.usgs.gov/sir/2008/5026

Hossain, A. B.M. S, A. Salleh, A. N. Boyce, P. Chowdhury, and M. Naquiddin. 2008. Biodiesel fuel from algae as renewable energy. American Journal of Biochemistry and Biotechnology: 4: 250-254

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

IEPA and NIPC (Illinois Environmental Protection Agency and Northern Illinois Planning Commission). 1998. Lake notes: lake dredging. http://www.epa.state.il.us/water/conservation/lake-notes/lake-dredging.pdf

IMS (Innovative Materials Systems). n. d. Lake and pond dredging picture gallery. IMS, Prairie Village, Kansas. http://www.imsdredge.com/lake-pond-Print-gallery.html

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

Janney, E. C., B. S. Hayes, D. A. Hewitt, P. M. Barry, A. Scott, J. Koller, M. Johnson, and G. Blackwood. 2009. Demographics and 2008 run timing of adult Lost River (*Deltistes luxatus*) and shortnose (*Chasmistes brevirostris*) suckers in Upper Klamath Lake, Oregon, 2008. U.S. Geological Survey Open-File Report 2009-1183.

Jassby, A. and J. Kann. 2010. Upper Klamath Lake monitoring program: preliminary analysis of status and trends for 1990–2009. Prepared for Klamath Tribes Natural Resources Department, Chiloquin, Oregon.

Johnson, D. M., R. R. Petersen, D. R. Lycan, J. W. Sweet, M. E. Neuhaus, and A. L. Schaedel. 1985. Atlas of Oregon lakes. Oregon State University Press, Corvallis.

Kadlec, R. H. and S. D. Wallace. 2009. Treatment wetlands. Second edition. CRC Press. Boca Raton, Florida.

Kann, J. 1998. Ecology and water quality dynamics of a shallow hypereutrophic lake dominated by cyanobacteria (*Aphanizomenon flos-aquae*). Doctoral Dissertation. University of North Carolina, Chapel Hill.

Kann, J. 2006. *Microcystis aeruginosa* occurrence in the Klamath River system of southern Oregon and northern California. Technical Memorandum. Prepared for the Yurok Tribe Environmental and Fisheries Programs, Klamath, California.

Kann, J. 2008. Microcystin bioaccumulation in Klamath River fish and freshwater mussel tissue: preliminary 2007 results. Technical Memorandum. Prepared for Karuk Tribe Department of Natural Resources, Orleans, California.

Kann. J. 2010a. Upper Klamath Lake 2009 data summary report. Technical Memorandum. Prepared by Aquatic Ecosystem Sciences for Klamath Tribes Natural Resources Department, Chiloquin, Oregon.

Kann. J. 2010b. Upper Klamath Lake tributary loading: 2009 data summary report. Technical Memorandum. Prepared by Aquatic Ecosystem Sciences for Klamath Tribes Natural Resources Department, Chiloquin, Oregon

Kann, J. 2011. Upper Klamath Lake 2010 data summary report. Prepared by Aquatic Ecosystem Sciences, Ashland, Oregon for Klamath Tribes Natural Resources Department, Chiloquin, Oregon.

Kann, J. 2012a. Upper Klamath Lake 2011 data summary report. Prepared by Aquatic Ecosystem Sciences, Ashland, Oregon for Klamath Tribes Natural Resources Department, Chiloquin, Oregon.

Kann, J. 2012b. Upper Klamath Lake tributary loading: 2011 data summary report. Prepared by Aquatic Ecosystem Sciences, Ashland, Oregon for Klamath Tribes Natural Resources Department, Chiloquin, Oregon.

Kann, J., and V.H. Smith. 1999. Estimating the probability of exceeding elevated pH values critical to fish populations in a hypereutrophic lake. Canadian Journal of Fisheries and Aquatic Sciences 56: 2262–2270.

Kann, J., and W. W. Walker. 1999. Nutrient and hydrologic loading to Upper Klamath Lake, Oregon, 1991–1998. Draft report submitted to Klamath Tribes, Chiloquin, Oregon and U. S. Bureau of Reclamation, Klamath Falls, Oregon.

Kann, J., and E. B. Welch. 2005. Wind control on water quality in shallow, hypereutrophic Upper Klamath Lake, Oregon. Lake and Reservoir Management 21: 149–158.

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

Kann, J., and E. Asarian. 2007. Nutrient budgets and phytoplankton trends in Iron Gate and Copco Reservoirs, California, May 2005–May 2006. Final Technical Report to State Water Resources Control Board,Sacramento, California.

Kann, J., and S. Corum. 2009. Toxigenic *Microcystis aeruginosa* bloom dynamics and cell density/chlorophyll a relationships with microcystin toxin in the Klamath River, 2005–2008. Technical Memorandum. Prepared for Karuk Tribe, Orleans, California.

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

California. http://www.klamathwaterquality.com/documents/2009_Klamath_River_FreshwaterMussel_%20Microcystin_%20Bioaccumulation.pdf

Kann, J., and C. Bowman. 2011. Middle Klamath River toxic cyanobacteria trends, 2010. Technical Memorandum. Prepared for Karuk Tribe Department of Natural Resources, Orleans, California.

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

Kannarr, K. E., D. Q. Tanner, M. K. Lindenberg, and T. M. Wood. 2010. Water-quality data from Upper Klamath and Agency Lakes, Oregon, 2007–08. Open-File Report 2010–1073. U. S. Geological Survey. http://pubs.usgs.gov/of/2010/1073

Karlsson-Elfgren, I. and A. K. Brunberg. 2004. The importance of shallow sediments in the recruitment of *Anabaena* and *Aphanizomenon* (Cyanophyceae). Journal of Phycology 40: 831-836.

Kemmerer, G., J. F. Bovard, and W. R. Boorman. 1923. Northwestern lakes of the United States: biological and chemical studies with reference to possibilities in production of fish. Bulletin of the United States Bureau of Fisheries 39.

Klamath Consulting Service. 1983. The Upper Klamath Lake. EPA 314 Clean Lakes Program 1981–1983, Phase 1: Diagnostic/feasibility study. Prepared by Klamath Consulting Service, Klamath Falls for Klamath County, Oregon.

Klinkenberg, C. E. 2001. Environmental and public use inspection report for Klamath Basin. Prepared by Federal Energy Regulatory Commission, Portland Regional Office, Portland, Oregon. http://digital.library.ucr.edu/cdri/documents/p2082_EnvironmentalandPublicInspectionReport.txt

Kruse, S. A. and A. J. Scholz. 2006. Preliminary economic assessment of dam removal: The Klamath River. Ecotrust, Portland, Oregon.

Kuwabara, J. S., D. D. Lynch, B. R. Topping, F. Murphy, J. L. Carter, N.S. Simon, F. Parchaso, T. M. Wood, M. K. Lindenberg, K. Wiese, and R. J. Avanzino. 2007. Quantifying the benthic source of nutrients to the water column of Upper Klamath Lake, Oregon. Open File Report 2007–1276. U. S. Geological Survey, Reston, Virginia.

Kuwabara, J. S., B.R. Topping, D. D. Lynch, J. L. Carter, and H. I. Essais. 2009. Benthic nutrient sources to hypereutrophic Upper Klamath Lake, Oregon, USA. Environmental Toxicology and Chemistry 28: 516–524.

Kuwabara, J. S., B. R. Topping, J. L. Carter, T. M. Wood, F. Parchaso, J. M. Cameron, J. R. Asbill, R. A. Carlson, and S. V. Fend. 2012. Time scales of change in chemical and biological parameters after engineered levee breaches adjacent to Upper Klamath and Agency Lakes, Oregon. Open-File Report 2012-1057. U. S. Geological Survey. http://pubs.usgs.gov/of/2012/1057

Laenen, A., and A. P. LeTourneau. 1996. Upper Klamath Basin nutrient-loading study–estimate of wind-induced resuspension of bed sediment during periods of low lake elevation. Open-File Report 95-414. U. S. Geological Survey, Portland, Oregon.

Lev, E. 2001. Heroic tales of wetland restoration. The Wetlands Conservancy. Tualatin, Oregon.

Lind P. 2009. Holocene floodplain development of the lower Sycan River, Oregon. Master's thesis, University of Oregon, Department of Geography, Eugene, Oregon. http://scholarsbank.uoregon.edu/jspui/bitstream/1794/10171/1/Lind_Pollyanna_ms2009fa.pdf

Lindenberg, M. K., G. Hoilman, and T.M. Wood. 2009. Water quality conditions in Upper Klamath and Agency Lakes, Oregon, 2006. Scientific Investigations Report 2008-5201. U.S. Geological Survey, Reston, Virginia. http://pubs.usgs.gov/sir/2008/5201

Loftus, M. E. 2001. Assessment of potential water quality stress to fish. Supplement to: effects of water quality and lake level on the biology and habitat of selected fish species in Upper Klamath Lake. Prepared for U. S. Bureau of Indian Affairs, Portland, Oregon.

Lynch, D. D. 2011. Subject: infeasibility of the mechanical removal of reservoir bottom sediments if Klamath River dams are removed in 2020. Memorandum to the Record from Dennis D. Lynch, Program Manager, Klamath Secretarial Determination.

Lyon, S., A. Horne, J. Jordahl, H. Emond, and K. Carlson. 2009. Preliminary feasibility assessment of constructed treatment wetlands in the vicinity of the Klamath Hydroelectric Project. Draft report. Prepared by CH2MHill for PacifiCorp Energy, Portland, Oregon.

Lytle, C. M. 2000. Water quality data review and wetland size estimate for the treatment of wastewaters from the Klamath Straits Dam. Draft Technical Memorandum. Prepared for U. S. Bureau of Reclamation, Klamath Project Office, Klamath Falls, Oregon.

Madrid Engineering Group. 2005. Lake Hancock dredging feasibility study. Final Report. Prepared for Polk County Natural Resources Division by Madrid Engineering Group, Bartow, Florida. http://www.polk.wateratlas.usf.edu/upload/documents/final%20rpt%20section%201.pdf

Mahugh, S., M. L. Deas, R. A. Gearheart, J. Vaughn, R. Piaskowski, and A. Rabe. 2008. Keno Reservoir Feasibility Study, Phase II – Identification and assessment of potential treatment wetland sites in the upper Klamath River. Prepared by Rabe Consulting, Klamath Falls, Oregon; GeoEngineers, Inc., Portland, Oregon; and Watercourse Engineering, Inc., Davis, California for U.S. Bureau of Reclamation, Klamath Basin Area Office, Klamath Falls, Oregon.

Mekebri A, G. J. Blondina, and D. B. Crane. 2009. Method validation of microcystins in water and tissue by enhanced liquid chromatography tandem mass spectrometry. Journal of Chromatography A 1216: 3147–3155.

Merrill, A., H. Shepley, M. Singer, N. Hume, S. Siegel, P. Bachand, B. Morris, J. Feingold, A. Ferguson, G. Young, and C. Ingram. 2011. Greenhouse gas reduction and environmental benefits in the Sacramento-San Joaquin Delta: Advancing carbon capture wetland farms and exploring potential for low carbon agriculture. Prepared for The Nature Conservancy, Sacramento, California.

Miller, W. E., and J. C. Tash. 1967. Interim report on Upper Klamath Lake studies, Oregon. Water Pollution Control Research Series, Publication WP-20-8. Federal Water Pollution Control Administration, Pacific Northwest Water Laboratory, Corvallis, Oregon.

Milligan A., P. Hayes, S. Geiger, K. Haggard, and M. Kavanaugh. 2009. Use of aquatic and terrestrial plant decomposition products for the control of *Aphanizomenon flos-aquae* at Upper Klamath Lake, Oregon. Prepared for U. S. Fish and Wildlife Service, Klamath Basin Ecosystem Restoration Office by Wetland Research Consortium. Oregon State University, Corvallis.

Mitsch, W. J., and J. G. Gosselink. 2007. Wetlands. Fourth edition. John Wiley & Sons, Hoboken, New Jersey.

Morace, J. L. 2007. Relation between selected water-quality variables, climatic factors and lake levels in Upper Klamath and Agency Lakes, Oregon, 1990–2006. U.S. Geological Survey, Scientific Investigation Report 2007-5117.

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

NRC (National Research Council). 2004. Endangered and threatened fishes in the Klamath River Basin. National Academies Press, Washington, D.C. http://www.nap.edu/catalog.php?record_id=10838

NRC. 2010. Progress toward restoring the Everglades: The third biennial review–2010. National Academies Press,Washington, D.C. http://www.nap.edu/catalog.php?record_id=12988

NMFS (National Marine Fisheries Service). 2010. Biological opinion for the operation of the Klamath Project between 2010 and 2018. File Number 151422SWR2008AR00148. Consultation conducted by National Marine Fisheries Service, Southwest Region.

OCCRI (Oregon Climate Change Research Institute). 2010. Oregon climate assessment report. K. D. Dello and P. W. Mote, editors. College of Oceanic and Atmospheric Sciences, Oregon State University, Corvallis, Oregon www.occri.net/OCAR

ODEQ (Oregon Department of Environmental Quality). 1998. Oregon's 1998 303(d) list. ODEQ, Portland. http://www.deq.state.or.us/wq/assessment/rpt98.htm

ODEQ. 2002. Upper Klamath Lake drainage total maximum daily load (TMDL) and water quality management plan (WQMP). Oregon Department of Environmental Quality, Portland. http://www.deq.state.or.us/wq/tmdls/docs/klamathbasin/ukldrainage/tmdlwqmp.pdf

ODEQ. 2010. Upper Klamath and Lost River subbasins total maximum daily load (TMDL) and water quality management plan (WQMP). Oregon Department of Environmental Quality, Bend, Oregon.

ODFW (Oregon Department of Fish and Wildlife (ODFW). 2008. Klamath Wildlife Area Management Plan. Salem, Oregon.

Opdyke, M.R., M.B. David, and B.L. Rhoads. 2006. Influence of geomorphological variability in channel characteristics on sediment denitrification in agricultural streams. Journal of Environmental Quality 35: 2103–2112.

Osgood, D., H. Gibbons, P. Eberhardt, A. Shortelle. 2011. Alum for phosphorus inactivation and interception. North American Lake Management Society Workshop Manual. Spokane, Washington.

PacifiCorp. 2004. Exhibit C–Construction history and proposed construction: Klamath Hydroelectric Project (FERC Project No. 2082). Portland, Oregon. http://www.pacificorp.com/content/dam/pacificorp/doc/Energy_Sources/Hydro/Hydro_Licensing/Klamath_River/Exhibit_C_Construction_History_and_Proposed_Construction.pdf

PacifiCorp. 2005. Response to FERC AIR AR-2, anadromous fish restoration for the Klamath Hydroelectric Project (FERC Project No. 2082). Final Technical Report. Portland, Oregon.

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

Perez M., C. Cox, and K. Cook. 2009. Facing facts in the Chesapeake Bay. Environmental Working Group. http://www.ewg.org/files/chesapeake-bay-pollution.pdf

Perkins, D. L., J. Kann, and G. G. Scoppettone. 2000. The role of poor water quality and fish kills in the decline of endangered Lost River and shortnose suckers in Upper Klamath Lake. Prepared by U.S. Geological Survey, Biological Resources Division for U. S. Bureau of Reclamation, Klamath Falls Project Office, Klamath Falls, Oregon. http://www.usbr.gov/mp/kbao/esa/Fish_Kill.pdf

Perry R. W., J. C. Risley, S. J. Brewer, E. C. Jones, and D.W. Rondorf. 2011. Simulating water temperature of the Klamath River under dam removal and climate change scenarios. Open File Report 2011-1243. U.S. Geological Survey.

Phinney, H. K., C. A. Peek, and J. L. McLachlan. 1959. A survey of the phytoplankton problems in Klamath Lake. Report to the supporting agencies: Oregon Klamath River Commission, California Klamath River Commission, Klamath County Court, City of Klamath Falls, and Klamath Chamber of Commerce. Corvallis, Oregon.

Pollock, M. M., J. M. Wheaton, N. Bouwes, and C. E. Jordan. 2011. Working with beaver to restore salmon habitat in the Bridge Creek intensively monitored watershed: Design rationale and hypotheses. Interim Report. National Marine Fisheries Service, Northwest Fisheries Science Center, Seattle, Washington. http://www.gis.usu.edu/~jwheaton/et_al/BridgeCreek/NOAA/BDSS_Tech_Memo_6.07.11.pdf

Prendergast L. and K. Foster. 2010. Analysis of microcystin in fish in Copco and Iron Gate reservoirs in 2009. Technical Memorandum. PacifiCorp, Portland, Oregon.

Rasmussen C. G. 2012. Geomorphology, hydrology and biology of floodplain vegetation in the Sprague basin. Doctoral dissertation. University of Oregon, Department of Geography, Eugene. http://gradworks.umi.com/3490795.pdf

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

Raymond R. 2008b. Results of 2007 phytoplankton sampling in the Klamath River and Klamath Hydroelectric Project (FERC Project No. 2082). Final Report. Prepared by E &S Environmental Chemistry, Corvallis, for PacifiCorp, Portland, Oregon.

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

Raymond R. 2009b. Phytoplankton species and abundance observed during 2008 in the vicinity of the Klamath Hydroelectric Project. Prepared by E&S Environmental Chemistry, Corvallis, for CH2MHill and PacifiCorp, Portland, Oregon.

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

Raymond R. 2010b. Phytoplankton species and abundance observed during 2009 in the vicinity of the Klamath Hydroelectric Project. Prepared by E&S Environmental Chemistry, Corvallis, for PacifiCorp, Portland, Oregon.

Reddy, K. R., S. Newman, T. Osborne, R. White, and H. Fitz. 2011. Phosphorus cycling in the Everglades ecosystem: Legacy phosphorus implications for management and restoration. Critical Reviews in Environmental Science and Technology 41: 149–186.

Richardson, C. J. 2008. Everglades experiments: Lessons for ecosystem restoration. Ecological Studies Volume 201. Springer, New York.

Risley J. C., and Gannett M.W. 2006. An evaluation and review of water-use estimates and flow data for the Lower Klamath and Tule Lake National Wildlife Refuges, Oregon and California. http://pubs.usgs.gov/sir/2006/5036

Risley, J. C., and A. Laenen. 1999. Upper Klamath Lake Basin nutrient-loading study—assessment of historic flows in the Williamson and Sprague rivers. Water-Resources Investigations Report 98-4198. U. S. Geological Survey, in cooperation with U. S. Bureau of Reclamation, Portland, Oregon.

Rosenberg, J. N., G. A.Oyler, L. Wilinson, and M. J. Betenbaugh. 2008. A green light for engineered algae: redirecting metabolism to fuel a biotechnology revolution. Current Opinions in Biotechnology 19:430-436.

Royer, C. F., and A. P. Stubblefield. 2010. Klamath Basin water quality monitoring plan. Prepared for Klamath Basin Monitoring Program by Klamath Watershed Institute, Humboldt State University, Department of Forestry and Wildland Resources, Arcata, California.

San Diego County Water Authority. 2011. Salton Sea update, March 16, 2011. Pages 58-64 *in* San Diego County Water Authority Board of Directors and Standing Committees meeting documents, March 24, 2011. San Diego, California. http://www.sdcwa.org/sites/default/files/files/board/2011-monthly/2011_03_24/boardpacket-2011-03-24.pdf

Sander, K., and G. S. Murthy. 2010. Life cycle analysis of algae biodiesel. International Journal of Life Cycle Assessment 15:704-714.

Scheidt, D. J. and P. I. Kalla. 2007. Everglades ecosystem assessment: water management, water quality, eutrophication, mercury contamination, soils and habitat. Monitoring for adaptive management: A RE-MAP status report, EPA 904-R-07-001. Environmental Protection Agency, Atlanta, Georgia.

SFWMD (South Florida Water Management District). 2004. Central and Southern Florida Project Comprehensive Everglades Restoration Plan. Development of Alternative Plans. Part 1—Storage and water quality. Lake Okeechobee Watershed Project. SFMWD, West Palm Beach, Florida.

SFWMD. 2011a. Dispersed water management program: shallow water retention benefits, water resources and the environment. Just the Facts, October 2011. West Palm Beach, Florida. http://www.sfwmd.gov/portal/page/portal/xrepository/sfwmd_repository_pdf/jtf_dispersed_water_mgmt.pdf

SFWMD. 2011b. Quick facts on Lake Trafford restoration. SFWMD Splash Factsheet, January 2011. West Palm Beach, Florida. http://www.sfwmd.gov/portal/page/portal/xrepository/sfwmd_repository_pdf/spl_lake_trafford.pdf

Simon, N. S., and S. N. Ingle. 2011. Physical and chemical characteristics including total and geochemical forms of phosphorus in sediment from the top 30 centimeters of cores collected in October 2006 at 26 sites in Upper Klamath Lake, Oregon. Open File Report 2011-1168. U. S. Geological Survey. http://pubs.usgs.gov/of/2011/1168

Simon, N. S., D. Lynch, and T. N. Gallaher. 2009. Phosphorus fractionation in sediment cores collected in 2005 before and after onset of an *Aphanizomenon flos-aquae* bloom in Upper Klamath Lake, OR, USA. Water Air Soil Pollution. 204: 139–153.

Snyder, D. T., and J. L. Morace. 1997. Nitrogen and phosphorus loading from drained wetlands adjacent to Upper Klamath and Agency lakes, Oregon. Water Resources Investigations Report 97-4059. U.S. Geological Survey. http://or.water.usgs.gov/pubs_dir/Pdf/97-4059.pdf

Spolaore P., C. Joannis-Cassan, E. Duran, and A. Isambert. 2006. Commercial applications of microalgae. Journal of Bioscience and Bioengineering 101: 87–96.

Stillwater Sciences. 2004. A preliminary evaluation of the potential downstream sediment deposition following the removal of Iron Gate, Copco, and J. C. Boyle dams, Klamath River, CA. Final Report. Prepared for American Rivers, Nevada City, California, by Stillwater Sciences, Berkeley,
California. http://act.americanrivers.org/site/DocServer/Stillwater_Final_Report.pdf?docID=5421&JServSessionIda004=q6hbem6fq1.app19c

Stocking, R. W. and J. L. Bartholomew. 2007. Distribution and habitat characteristics of *Manayunkia Speciosa* and infection prevalence with the parasite *Ceratomyxa Shasta* in the Klamath River, Oregon, California. Journal of Parasitology 93: 78–88.

Sullivan, A. B., M. L. Deas, J. Asbill, J. D. Kirshtein, K. Butler, M. A. Stewart, R. E. Wellman, and J. Vaughn. 2008. Klamath River water quality and acoustic doppler current profiler data from Link River Dam to Keno Dam, 2007. Open-File Report 2008-1185. U. S. Geological Survey.

Sullivan, A. B., M. L. Deas, J. Asbill, J. D. Kirshtein, K. Butler, and J. Vaughn. 2009. Klamath River water quality data from Link River Dam to Keno Dam, Oregon, 2008. U. S. Geological Survey Open File Report 2009-1105. U. S. Geological Survey.

Sullivan, A. B., D. M. Snyder, and S. A. Rounds. 2010. Controls on biochemical oxygen demand in the upper Klamath River, Oregon. Chemical Geology 269: 12-21.

Sullivan A. B., S. A. Rounds, M. L. Deas, J. R. Asbill, R. E. Wellman, M. A. Stewart, M. W. Johnston, and I.E. Sogutlugil. 2011. Modeling hydrodynamics, water temperature, and water quality in the Klamath River upstream of Keno Dam, Oregon, 2006-09. Scientific Investigations Report 2011-5105. U. S. Geological Survey.

Tanaka, T., K. Tsuzuki, N. Nishijima, and T. Takagi. 2001. Algae removal performance of a fluidized bed biofilm reactor system for lake water treatment. Water Science and Technology 43: 277-283.

Tervita. n. d. Reclamation Services Burnaby Lake dredging and sediment removal case study. Factsheet. http://www.tervita.com/knowledge-centre/case-studies/Documents/burnaby_lake.pdf

Tetra Tech. 2006. Performance evaluation of the New River Demonstration Wetlands. Final Report. Prepared for Citizen's Congressional Task Force on the New River, Brawley, California by Tetra Tech, Pasadena, California, in association with Wetlands Management Services, Chelsea, Michigan. http://www.newriverwetlands.com/images/pilotfinalreport.pdf

Tetra Tech. 2007. New and Alamo River Wetlands Master Plan. Revised Final Report. Prepared for Wildlife Conservation Board, Sacramento, California and Salton Sea Authority, La Quinta, California by Tetra Tech, Lafayette, California, in association with Wetlands Management Services, Chelsea,
Michigan http://rd.tetratech.com/waterquality/projects/doc/mp_wetland_development_imperial_valley/New_and_Alamo_River_Master_Plan_v2_print.pdf

USACE (U. S. Army Corps of Engineers). 1982. Potential eutrophication control measures for Upper Klamath Lake, Oregon: data evaluation and experimental design. Upper Klamath Lake, Oregon—Water Resources Development Project. Prepared by Entranco Engineers, Bellevue, Washington for USACE, San Francisco District, San Francisco, California.

USACE. 2007. Review of biological research on juvenile and adult salmonid use of Vancouver Lake. Draft Final Report. USACE, Portland District, Portland, Oregon. http://www.cityofvancouver.us/publicworks/vancouverlake/MapsMaterials/USACEBioResearch_VanLake_1107.pdf

USACE. 2009. Operations and maintenance dredging and placement of dredged material, Toledo Harbor, Lucas County, Ohio. Finding of No Significant Impact and Environmental Assessment. USACE, Buffalo District, Buffalo, New York. http://www.westernlakeerie.org/corps_404_fonsi_open_lake_0409.pdf

USBLM (U. S. Bureau of Land Management). 1996. Upper Klamath Basin and Wood River Wetland Record of Decision and Resource Management Plan. U. S. Bureau of Land Management, Klamath Falls, Oregon. Available from: http://www.blm.gov/or/plans/files/Wood_River_RMP.pdf

USBLM. 2005. Wood River wetland monitoring report, 2003–2005. U. S. Bureau of Land Management, Klamath Falls, Oregon. http://nrimp.dfw.state.or.us/web%20stores/data%20libraries/files/BLM/BLM_661_2_Klamath-WoodRWetlands_Monitoring_Report_03-05.pdf

USBR (U. S. Bureau of Reclamation). 2000. Imperial Valley drainwater reclamation and reuse study. USBR, Lower Colorado Region, Southern California Area Office, Temecula. http://www.usbr.gov/lc/region/g2000/publications/Imperial.pdf

USBR. 2005. Natural flow of the upper Klamath River – Phase I. Natural inflow to, natural losses from, and natural outfall of Upper Klamath Lake to the Link River and the Klamath River at Keno. Prepared by USBR, Technical Service Center for USBR, Klamath Basin Area Office, Klamath Falls, Oregon. http://www.usbr.gov/mp/kbao/docs/undepleted_klam_fnl_rpt.pdf

USBR. 2007. Restoration of the Salton Sea. Summary Report. USBR Lower Colorado Region, Boulder City, Nevada. http://www.usbr.gov/lc/region/saltnsea/FinalSummaryRpt.pdf

USBR. 2011a. SECURE Water Act, Section 9503(c) – Reclamation climate change and water. U.S. Bureau of Reclamation, Policy and Administration, Denver, Colorado. Report to Congress, 2011.

USBR. 2011b.. Reservoir area management plan for the Secretary's Determination on Klamath River dam removal and basin restoration. Technical Report No. SRH-2011-19. Prepared by USBR, Technical Service, Center, Denver, Colorado for USBR, Mid-Pacific Region.

USEPA (U.S. Environmental Protection Agency). 1988. Ambient water quality criteria for aluminum. EPA 440/5-86-008. U. S. Environmental Protection Agency, Office of Water Regulations and Standards Criteria, Washington, D. C.

USEPA. 2000a. Constructed wetlands treatment of municipal wastewaters. EPA/625/R-99/010. U. S. Environmental Protection Agency, Office of Research and Development.

USEPA. 2000b. Guiding principles for constructed treatment wetlands. EPA 843-B-00-003. U. S. Environmental Protection Agency, Office of Research and Development.

USEPA. 2009. The next generation of tools and actions to restore water quality in the Chesapeake Bay. A Revised Report Fulfilling Section 202a of Executive Order 13508, November 24, 2009. http://executiveorder.chesapeakebay.net/file.axd?file=2009%2F11%2F202a+Water+Quality+Report.pdf

USEPA. 2010. Review of California's 2008-2010 Section 303(d) list. Enclosure to letter from Alexis Strauss, U. S. Environmental Protection Agency, Region 9, San Francisco, California to Thomas Howard, State Water Resources Control Board, Sacramento, California.

USEPA. 2011. The Great Waters Program: Chesapeake Bay. U. S. Environmental Protection Agency website: http://www.epa.gov/airquality/gr8water/index.html

USFWS (U.S. Fish and Wildlife Service). 2008. Biological/Conference Opinion regarding the effects of the U.S. Bureau of Reclamation's proposed 10-Year Operation Plan (April 1, 2008–March 31, 2018) for the Klamath Project and its effects on the endangered Lost River and shortnose suckers. Klamath Falls Fish and Wildlife Office, Klamath Falls, Oregon and Yreka Fish and Wildlife Office, Yreka, California. http://www.usbr.gov/mp/kbao/operations/klamath_project_2008_fws_bo_final.pdf

USFWS. 2010. Klamath Marsh National Wildlife Refuge Comprehensive Conservation Plan. U.S. Fish and Wildlife Service, Sacramento, California and Tule Lake, California. http://www.fws.gov/klamathbasinrefuges/KlamathMarshCCP/kmarshccpFINAL.html

VanderKooi S. P., S. M. Burdick, K. R. Echols, C. A. Ottinger, B. H. Rosen, and T. M. Wood. 2010. Algal toxins in upper Klamath Lake, Oregon: Linking water quality to juvenile sucker health. Fact Sheet 2009-3111. US Geological Survey, Western Fisheries Research Center, Seattle, Washington. http://pubs.usgs.gov/fs/2009/3111/pdf/fs20093111.pdf

Walker, W. W. 2001. Development of a phosphorus TMDL for Upper Klamath Lake, Oregon. Prepared for Oregon Department of Environmental Quality, Bend, Oregon. March 2001. http://www.wwwalker.net/pdf/klamath_tmdl_final_march_7_2001.pdf

Walker, W., J. Walker, and J. Kann. 2012. Evaluation of water and nutrient balances for the upper Klamath Lake basin in water years 1992-2010. Prepared for Klamath Tribes Natural Resources Department, Chiloquin, Oregon by Environmental Engineers, Concord, Massachusetts and Aquatic Ecosystem Sciences, Ashland, Oregon.

Watercourse Engineering, Inc. 2003. Klamath River water quality 2000 monitoring program. Project Report. Prepared for U. S. Bureau of Reclamation, Klamath Falls Area Office, Klamath Falls, Oregon, with support from PacifiCorp, Portland, Oregon.

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

Welch, G., and H. Gibbons. 2009. Aeration of lakes and reservoirs. Lake Line 29: 8-10.

Westbrook, C. J., D. J. Cooper, and B. W. Baker. 2011. Beaver assisted river valley formation. River Research and Applications 27:247–256.

Wong, S., and H. Hendrickson. 2011. Water quality conditions in the Williamson River Delta, Oregon: three years post-restoration. 2010 Annual Data Report. The Nature Conservancy Klamath Basin Field Office, Klamath Falls, Oregon. http://conserveonline.org/workspaces/Williamson.River.Delta.Preserve/documents/water-quality-conditions-on-the-williamson-river-0/view.html

Wong, S.W., H. A. Hendrickson, and C. M. Doehring. 2010. Post-restoration water quality conditions at the Williamson River Delta, Upper Klamath Basin, Oregon, 2007–2009. The Nature Conservancy Klamath Basin Field Office, Klamath Falls, Oregon. http://conserveonline.org/workspaces/Williamson.River.Delta.Preserve/documents/postrestoration-water-quality-conditions-at-the/view.html

Wong, S.W., M. J. Barry, A. R. Aldous, N. T. Rudd, H. A. Hendrickson, and C.M. Doehring. 2011. Nutrient release from a recently flooded delta wetland: comparison of field measurements to laboratory results. Wetlands 31: 433–443.

Wood, T. M., Hoilman, G.R. and M.K. Lindenberg. 2006. Water-quality conditions in Upper Klamath Lake, Oregon, 2002-2004. Scientific Investigations Report 2006-5209. U.S. Geological Survey. http://pubs.usgs.gov/sir/2006/5209

Wood, S. A., J. P. Rasmussen, P. T. Holland, R. Campbell, and A. L. M. Crowe. 2007. First report of the cyanotoxin Anatoxin-A from *Aphanizomenon issatschenkoi* (cyanobacteria). Journal of Phycology 43:356-365.

Wood, T. M., R. T. Cheng, J. W. Gartner, G. R. Hoilman, M. K. Lindenberg, and R. E. Wellman. 2008. Modeling hydrodynamics and heat transport in Upper Klamath Lake, Oregon, and implications for water quality. Scientific Investigations Report 2008–5076. U. S. Geological Survey. http://pubs.usgs.gov/sir/2008/5076

Yan, Y., and G. J. Jameson. 2004. Application of the Jameson Cell technology for algae and phosphorus removal from maturation ponds. International Journal of Mineral Processing 73: 23-28.

Yen, H., and D. E. Burne. 2007. Anaerobic co-digestion of algal sludge and waster paper to produce methane. Bioresource Technology 98: 130-134

Appendices

Appendix A

Relevant national/regional examples of watershed-scale pollutant reduction

1 CHESAPEAKE BAY

1.1 Overview

The Chesapeake Bay is the largest estuarine system in the contiguous United States with a watershed area of nearly 64,000 square miles traversing the states of New York, Pennsylvania, Delaware, Maryland, Virginia, and West Virginia (Figure A-1). As a major resting ground along the Atlantic Flyway, it also supports more than 3,600 species of plants, fishes, and other animals. More than 500 million pounds of seafood are extracted from the Bay each year, and economists have estimated its overall value at more than $1 trillion (USEPA 2009). In addition, its 1,500 square miles of wetlands provide valuable ecosystem services, such as critical habitat for fish, shellfish, and wildlife; filtration and treatment of residential, agricultural, and industrial wastes; and coastal storm and wave damage reduction (USEPA 2009).

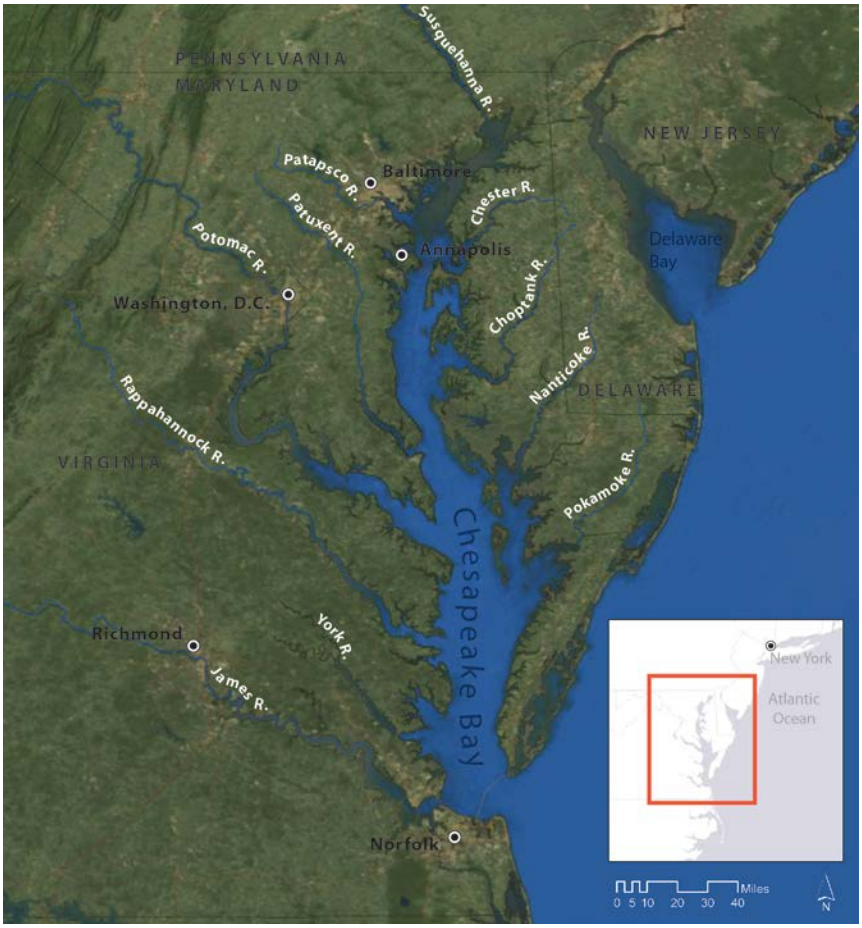

Figure A-1. Location of Chesapeake Bay and major tributaries. Source: ESRI.

1.2 Identification of water quality problems

In 1975, Chesapeake Bay became first estuary to be targeted for protection and restoration when Congress directed the USEPA to initiate a study investigating the causes of the observed environmental declines. In the process, the USEPA concluded that increased nitrogen and phosphorus loads were causing the Bay to become nutrient enriched, leading to excessive

seasonal growth of phytoplankton and algae followed by decomposition and prolonged anoxic conditions during the summer months, killing bottom-dwelling organisms such as oysters, clams, and worms (USEPA 2011). The main sources of nutrient pollution are agricultural, urban, and suburban runoff, wastewater outfalls, and atmospheric deposition. Agricultural land use covers approximately 8.5 million acres, or 25%, of the total watershed area and is, by far, the number one source of nutrient and sediment pollution to the Bay (USEPA 2009). However, pollution carried in urban and suburban runoff is of increasing concern due to the region's continued population growth and related construction (USEPA 2009).

Since 1983, the state-federal Chesapeake Bay Program has coordinated and conducted Chesapeake Bay watershed restoration efforts through a series of voluntary agreements from the partners–Maryland, Pennsylvania, Virginia, the District of Columbia, and the USEPA. However, not all of the states contributing to the water quality problems in the Bay signed on to the agreements and there was little in the way of consequences for not reaching the restoration goals. By 2009, water quality improvement had only reached 21% of the goals (USEPA 2009). After years of insufficient progress, continued impaired water quality designation, and an Executive Order by President Obama, the USEPA developed nutrient and sediment TMDLs for all six watershed states and the District of Columbia in December 2010. The Chesapeake Bay TMDL sets watershed limits of 185.9 million pounds of nitrogen (a 25% reduction), 12.5 million pounds of phosphorus (a 24% reduction), and 6.45 billion pounds of sediment (a 20% reduction) per year. In order to achieve these reductions, the USEPA requires documented "reasonable assurance" through the development of Watershed Implementation Plans (WIPs) and stop-gap measures that detail how and when each of the six states and the District of Columbia will meet their pollution allocations.

1.3 Proposed solutions and challenges

Both point and nonpoint nutrient sources have been targeted for reduction in the Chesapeake Bay watershed. Municipal and industrial wastewater treatment plant discharges currently contribute 20% of the total nitrogen and phosphorus loadings to the system (USEPA 2009). Since 1985, these point sources have achieved a 32% reduction (26 million pounds per year) in total nitrogen being delivered to the Bay (Chesapeake Bay Program 2004). Three elements of the Chesapeake Bay Program's point source control strategy are responsible for the present-day reductions in nutrient loading:

- Prohibition of consumer detergents containing phosphorus,
- Wastewater treatment plant tertiary upgrades, and
- Enforcement of permit requirement compliance.

Because the majority of municipal treatment plants discharge into fresh waters where phosphorus is the limiting nutrient, nitrogen concentrations received little attention until recent years.

Nonpoint sources of nutrients contribute about 50% of the total nitrogen and 75% of the total phosphorus that reaches the Bay, the largest single source originating from agricultural runoff (USEPA 2009). To address nonpoint sources of nutrients to the Bay, the following programs are underway:

- *Implementation of best management practices (BMPs).* The Chesapeake Bay Watershed Initiative (CBWI) offers technical and financial cost-share assistance to landowners for the implementation of best management practices that reduce erosion and nutrient pollution. This includes fencing off streams, planting stream buffers, rotating crops and cattle, and

using manure management technology. However, to date, not all farmers have participated in the cost-share programs. Further, annual funding available for the voluntary cost-share programs is grossly insufficient to achieve pollution prevention goals, and those geographic areas and agricultural operations responsible for most of the nutrient and sediment load have not been the focus of available funding (Perez et al. 2009).

- *Wetland restoration and protection.* Despite the valuable water quality, habitat improvement, and shoreline protection services wetlands provide, Bay states had, by 2009, achieved little more than half of the 25,000 acres restoration goal set to be completed by 2010, with some states making more progress than others (Chesapeake Bay Foundation 2010). Permitted losses of wetlands have slowed in recent years, likely due to the economic down-turn.
- *Conservation and restoration of riparian corridors, forest buffers, and resource lands*. This includes, but is not limited to pavement removal, expansion of the urban tree canopy, preservation of undeveloped farm land and natural areas, stream buffer zones, and river restoration for channelized rivers. Many of the recent watershed-wide gains, particularly in Pennsylvania, have been due to the increased conservation dollars provided by the federal Farm Bill and the Chesapeake Bay Foundation (CBF) investments in restoration (Chesapeake Bay Foundation 2010). Conservation of resource lands has been very progressive in recent years, partially due to the slowed pace of development.
- *Urban stormwater management*. This includes, but is not limited to, general sewer system upgrades to prevent combined sewer overflow into the waterways during storm events, bioswales for urban storm water management, and regulation of commercial fertilizer application and consumer lawn fertilizer formulation. Only those municipalities with the foresight and funding to tackle this management problem before regulatory requirements were in place made significant gains in this arena.

To date, nonpoint source pollution reduction strategies have achieved less than 50% of the goals set by the voluntary Chesapeake Bay Program agreements (Perez et al. 2009). Identified reasons for this short-coming include gaps in the regulatory framework regarding controls on soil erosion from croplands, CWA permitting programs for livestock animals (dairy, beef, swine) and poultry operations, land application of manure, and the widespread use of agricultural chemical fertilizers (Perez et al. 2009).

The Chesapeake Bay TMDLs require Watershed Implementation Plans (WIPs), with final phase II WIPs in place by March 30, 2012, and Phase III WIPs submitted by 2017. Each successive WIP provides increasingly more specific detail on load goals and actions to achieve those goals. All pollution control measures needed to fully restore the Bay and its tidal rivers are to be in place by 2025, with at least 60% of the actions completed by 2017. If the partner states do not meet this schedule, the USEPA has identified the following consequences that may include, but are not limited to the following:

- Revising the draft or final pollutant WLAs in the Bay TMDL to assign more stringent pollutant reduction responsibilities to point sources of nutrient and sediment pollution
- Objecting to state-issued CWA National Pollutant Discharge Elimination System (NPDES) permits
- Acting to limit or prohibit new or expanded discharges of nutrients and sediments
- Withholding, conditioning, or reallocating federal grant funds
- Taking other actions as appropriate

1.4 Costs

Available data from the Chesapeake Bay Foundation indicate that for the period 2007-2010, funding for the Chesapeake Bay Program from all sources (i.e., federal, state, non-governmental organizations [NGOs]) has been in excess of $3.8 billion, or approximately $950 million per year (ChesapeakeStat data accessed August 3, 2012 http://stat.chesapeakebay.net/?q=node/127). Roughly $2.8 billion or 75% of the total amount has been allocated to protecting and restoring water quality, with another roughly $550 million or 14% allocated to maintaining healthy watersheds. The states of Delaware, Maryland, New York, Pennsylvania, Virginia, West Virginia, and the District of Columbia have contributed approximately 72% of the total funding, the federal government has contributed 27%, and NGOs have contributed 1%.

2 EVERGLADES

2.1 Overview

The original (pre-drainage) Everglades ecosystem covered approximately 11,000 square miles of south Florida, including the Kissimmee River watershed to Lake Okeechobee, the central Everglades marshes and the southern estuaries (Biscayne Bay, Florida Bay and the southwest Florida coast) (Figure A-2). Water flowed through the marshes as a sheet, a "river of grass," driven by, generally seasonal, but regionally stochastic rainfall and periodic overflows from Lake Okeechobee. Draining of the Everglades for agriculture and development began in the late 1800s and early 1900s and culminated in 1948 with the Central and South Florida (C&SF) Project, an expansive water control system of levees and canals, designed to provide flood control and water supply for urban and agricultural development. These large-scale changes to the landscape severely altered the natural timing, distribution, quantity and quality of water in south Florida, and resulted in many negative consequences (e.g., overdrying/ponding in different areas, eutrophication, algal blooms, altered salinity regimes). The Comprehensive Everglades Restoration Plan (CERP) was initiated in 2000 to restore the Everglades ecosystem by reestablishing those hydrologic characteristics. This summary focuses on water quality in the central Everglades, including natural characteristics, primary issues and restoration methods.

2.2 Identification of water quality problems

The pre-drainage Everglades ecosystem was an oligotrophic system, with low total phosphorus concentrations estimated to be at or below 10 ppb. Prior to drainage for agricultural development, the primary source of water in the Everglades system was rainfall, with periodic overflow events from Lake Okeechobee and minimal groundwater input (NRC 2010). Rainfall is notably low in nitrogen and phosphorus, with phosphorus concentrations measured at 9–10 parts per billion (ppb) (Ahn and James 2001, Richardson 2008). With implementation of the C&SF Project and construction of a dike around Lake Okeechobee, sources of water to the Everglades changed significantly; water now flowed from the lake through canals and was supplied to the system through engineered means. Much of the water was directed through canals to the east and west coasts and released "to tide" into estuaries. Also, increasing agricultural development in the Everglades Agricultural Area (EAA) required large quantities of water, which took precedence over the ecosystem's needs to the south. The EAA is a 700,000-acre area immediately south of Lake Okeechobee (27% of the historical Everglades) where sugar cane and winter vegetables are the dominant crops (Figure A-2). Nutrient runoff from the EAA and other agriculturally-dominated basins is the primary source of phosphorus loading south into the central Everglades.

Such nutrient loading has resulted in water quality problems such as high phosphorus concentrations in the soil and expansion of phosphorus-limited cattails. Unnatural water levels exacerbate this problem by causing certain areas to dry out and phosphorus in the soils to be released. Areas considered phosphorus-enriched by the State of Florida have soil concentrations of greater than 500 ppb; these areas constituted 24% of the total area in 2005, with 49% of the area having concentrations over 400 ppb (Scheidt and Kalla 2007).

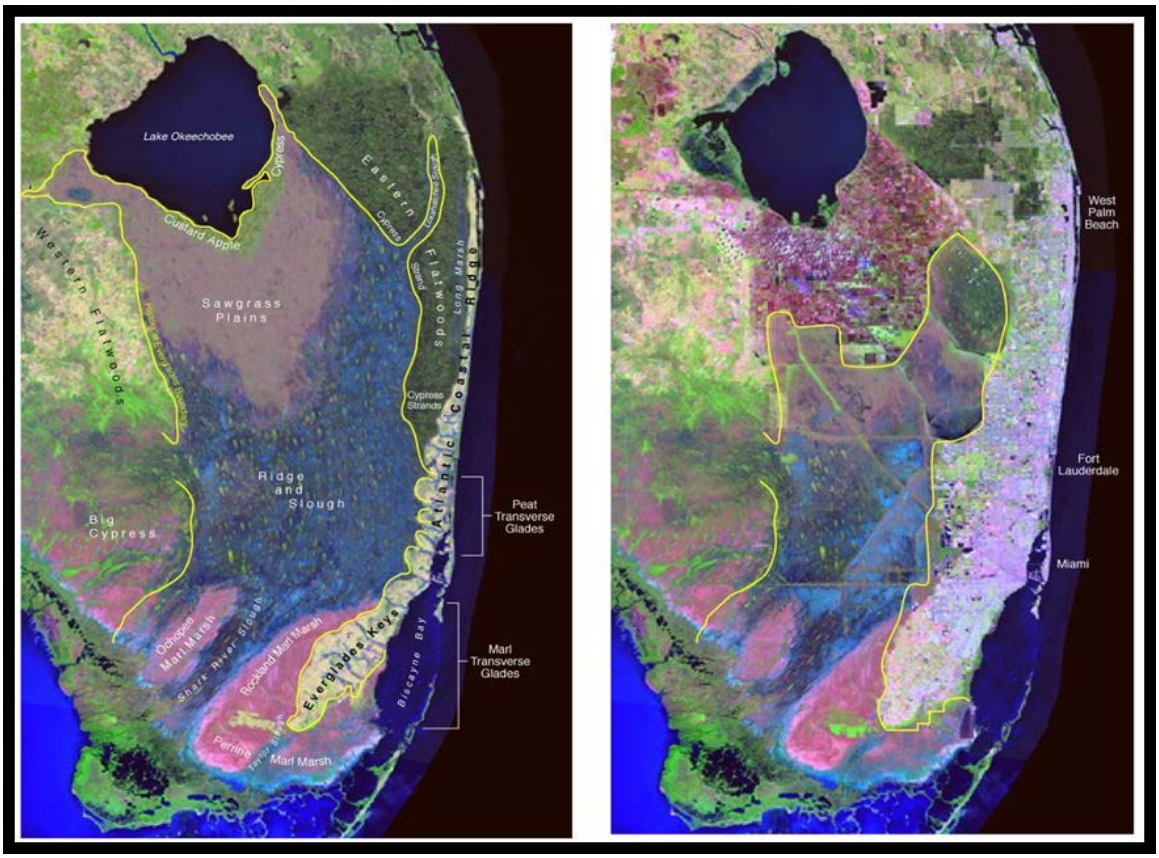

Figure A-2. Pre-drainage and current Everglades system.

2.3 Proposed solutions and challenges

To address identified water quality problems in the Everglades ecosystem, several strategies have been implemented as part of CERP and other water quality programs.

- **Best Management Practices (BMPs):** BMPs are applied to both agricultural and non-agricultural lands north and south of Lake Okeechobee. Examples of BMPs include improved nutrient management practices, fencing cattle out of waterways, sediment and erosion control measures, use of conservation and riparian buffers, increased wetland and ditch water retention, improved irrigation management and controlled drainage. Implementation of the BMP program in the EAA has resulted in an average total phosphorus load reduction of 54% over a 14-year period (1996–2009), more than double the targeted goal of 25% (NRC 2010).
- **Stormwater Treatment Areas (STAs):** STAs are constructed wetlands used to retain nutrients and other contaminants using vegetation and microbial communities (Figure A-

3). Research has shown that such wetlands are most effective at removing inorganic forms of phosphorus and nitrogen, and less effective at removing organic forms of those nutrients (Kadlec and Wallace 2009). To date, a total of 65,000 acres of STAs have been constructed on former agricultural lands along the southern boundary of the EAA. Each STA consists of several cells, composed of plant communities, through which nutrient-rich water is fed. Inflow and outflow rates are controlled and plant communities are managed to optimize nutrient removal. Between WY 1994 and 2009, the first six STAs reduced the load of inflow phosphorus by 72%; over this time period there was severe hurricane damage to the STAs, as well as three consecutive years of drought. Still, yearly performance efficiencies for phosphorus removal for this period ranged from 64 to 88%. The South Florida Water Management District (SFWMD) tries to keep the seasonal inflows as consistent as possible, but during extremely wet periods inflow loads have exceeded design loads and removal efficiencies have decreased (NRC 2010); however, outflows typically have low phosphorus concentrations.

- **Dispersed Water Management Program:** North of Lake Okeechobee a program to retain water on private, public and tribal lands for water quality and storage benefits has been put in place. This reduces direct runoff and inflow into Lake Okeechobee and other waterways, thereby reducing the amount of water discharged to the canals and estuaries during the wet season. Detaining water also reduces nutrient concentrations in the runoff as it steadily flows across the landscape. A total of 131,500 acre-feet of storage is now provided through this method (SFWMD 2011a).

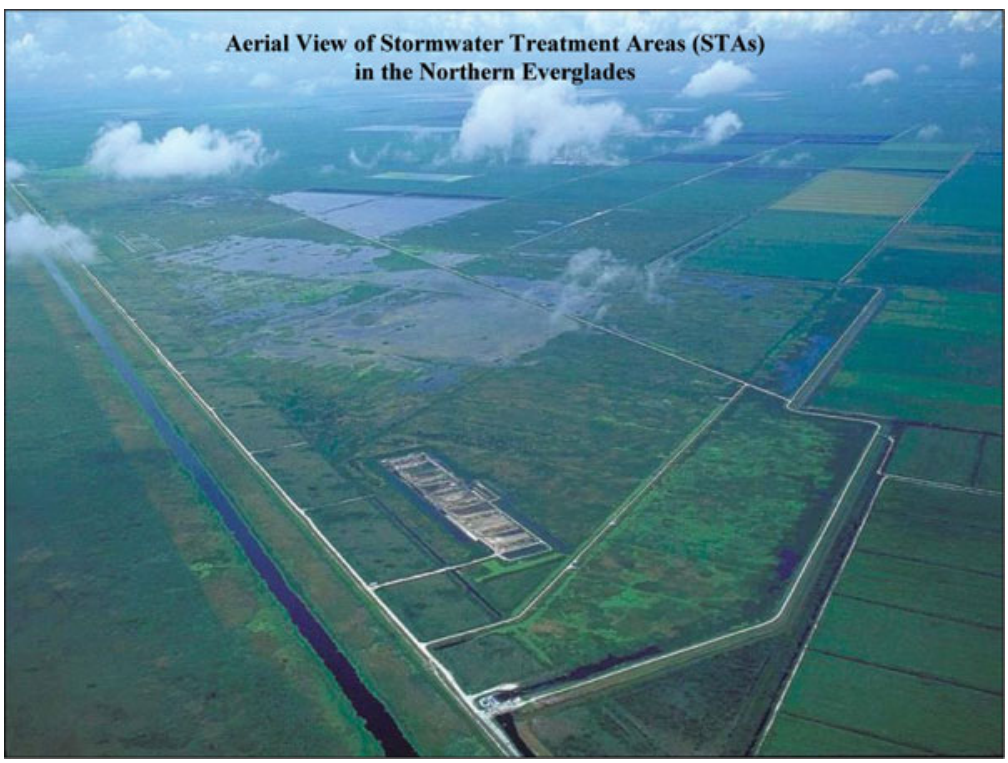

Figure A-3. Aerial view of Stormwater Treatment Areas (STAs) in the Northern Everglades. Photo courtesy of USGS.

Despite the progress that has been made to improve water quality conditions in the central Everglades, phosphorus loading continues to be a significant issue. An additional 50,000 acres of

STAs are in the planning stages, which will provide needed treatment and temporary storage; however, optimizing phosphorus removal efficiencies and managing inflow loads remain challenging (NRC 2010). Additionally, legacy phosphorus in Everglades peat soils and Lake Okeechobee sediments remains a significant source of loading. Studies on the efficacy of dredging Lake Okeechobee sediments and/or using alum to treat the lake's water have found that such efforts would be very costly and time consuming and do not address the issue of limiting source inputs (EA Engineering Science and Technology 2002). Establishing more natural hydrologic regimes in the central Everglades to prevent overdrying and subsequent phosphorus release from the soils is a goal of restoration projects in the central Everglades; however, progress has been slow and impeded by lawsuits regarding the impacts of adding water that does not meet the 10 ppb phosphorus threshold. In summary, efforts to limit phosphorus inputs have been successful to varying degrees, but considerable work is still needed and the issue remains contentious.

2.4 Costs

To date (i.e., since approximately 2000), funding for the Everglades program from all sources has been in excess of $5.2 billion, or approximately $433 million per year over 12 years. Roughly $2.8 billion or 54% of the total funding has been provided by the federal government, including funds from the U.S. Department of Interior and U.S. Army Corps of Engineers (including the Herbert Hoover Dike Rehabilitation project to strengthen a 143-mile series of levees and structures surrounding Lake Okeechobee). The remaining $2.4 billion or 46% has been provided by the State of Florida (South Florida Water Management District) and has primarily been for land acquisition.

3 SALTON SEA

3.1 Overview

The Salton Sea is the largest inland body of water in California. It is located on the San Andreas Fault at an elevation of more than 220 feet below sea level in between the Coachella and Imperial Valleys of southern California (Figure A-4). This is one of the most arid regions in the United States with an average annual rainfall of only 3 inches and more than 110 days of temperatures that exceed 100°F each year. The Salton Sea was formed during a breach of the Colorado River levee system in 1905. Prior to this event, the Colorado River periodically emptied into the northwest portion of the Salton Basin, forming the much larger Lake Cahuilla (USBR 2007). The current sea is a terminal lake fed by the New, Whitewater, and Alamo rivers, as well as various agricultural drainage systems and creeks (USBR 2007). Evaporative water loss has been estimated at 5.78 feet per year, and the surface area fluctuates around an average of 362 square miles, depending on inflow volumes (USBR 2000).

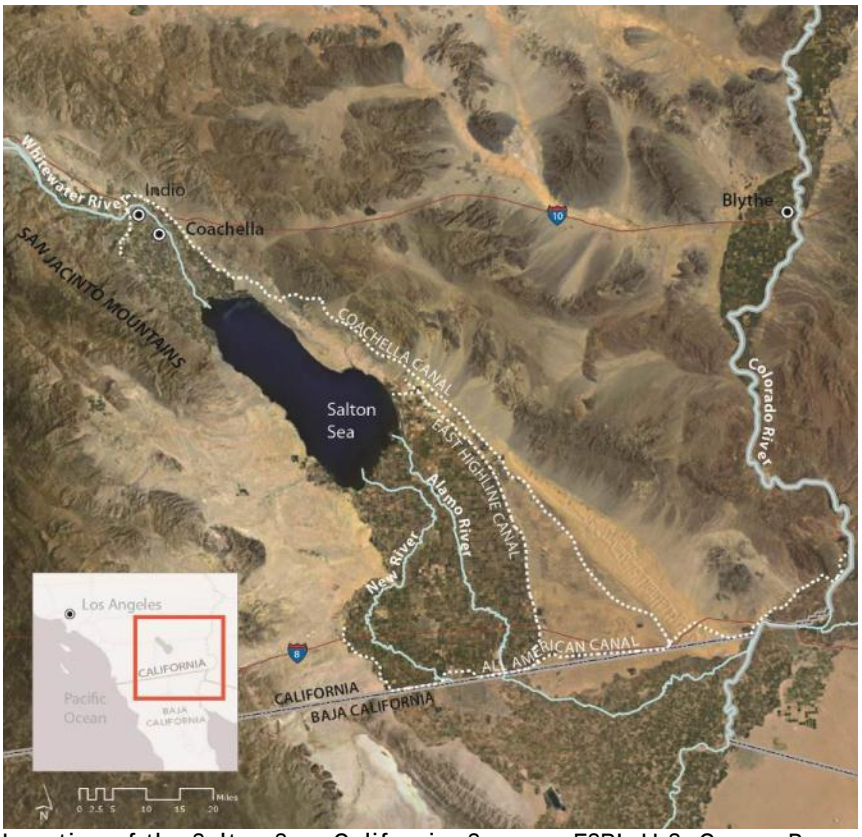

Figure A-4. Location of the Salton Sea, California. Source: ESRI, U.S. Census Bureau Geography Division.

Beginning in 1929, the California Department of Fish and Game (CDFG) introduced more than 30 marine fish species to the Salton Sea; three of which, the sargo, Gulf croaker, and orangemouth corvina, adapted and became established (USBR 2007). Around 1965, tilapia was unintentionally introduced from agricultural drains, and by the early 1970s, it dominated the Sea's fish community (USBR 2007). The only native fish to call the Sea home is the endangered desert pupfish, a key driver for restoration actions (USBR 2007). As part of the Pacific Flyway, the Salton Sea is also an important link that sustains the migratory cycles for many species of birds. It provides habitat for all seasons as an important way station for seasonal resting and feeding, wintering, spring conditioning, and breeding habitat (USBR 2007). The Salton Sea ecosystem currently supports more than 400 bird species, approximately 70% of all the bird species recorded in California (USBR 2007).

3.2 Identification of water quality problems

Tiled farmland drainage and wastewater treatment plant effluent contribute most of the nutrients and dissolved salts to the waters of the Salton Sea. With no outlet, evaporative losses of water concentrate these dissolved salts and nutrients, making the Sea hypereutrophic (USBR 2007). Increasing salinity and decreasing dissolved oxygen levels currently pose the greatest threat to the future of the Salton Sea ecosystem, although temperature fluctuations may become of increasing concern as water levels drop (USBR 2007). The salinity of the Sea has been recently measured at about 48,000 mg/L (USBR 2007). If the salinity reaches 60,000 mg/L, the majority of the existing fishery is projected to be lost (USBR 2007). Future water conservation, recycling, and transfers

are anticipated to further reduce water delivery to the Salton Sea, which will accelerate the rate of salinity increase, reduce the suitability of habitat for fish and wildlife, and expose a greater area of seabed playa. This leads to a human health concern when the high winds and arid conditions of the Salton Sea basin erode the exposed seabed playa and increase the concentration of airborne particles with a diameter of less than 10 microns (PM10) (USBR 2007).

3.3 Proposed solutions and challenges

The Salton Sea Reclamation Act of 1998 directed the Secretary of the Interior to study the feasibility of possible actions that would allow the continuation of the various environmental and recreational uses for this waterbody. Initial studies focused on options for reducing salinity and stabilizing the water surface elevation of the Sea. By 2007, under the direction of the California State Legislature, the California Department of Water Resources (DWR) released the Final Programmatic EIR which described the Preferred Alternative for the restoration of the Salton Sea (described below) based upon years of interagency collaboration and public comment. The Report identified the need for additional environmental data, the collection of which was to be completed by the Departments of Water Resources and Fish and Game, in coordination with the U. S. Geological Survey in 2008 (DWR 2011a). In 2011, the Army Corps of Engineers and the California Natural Resources Agency released a Draft EIS/EIR for the Salton Sea Species Conservation Habitat Project. This state project is intended to serve as a "proof of concept" for the restoration of shallow water habitat as described by the preferred alternative (DWR 2011b). Depending on funding, the project will create up to 3,770 acres of shallow water habitat, to be completed by 2015 (DWR 2011b).

The Preferred Alternative, as described in the Final Programmatic EIR, includes a saline habitat complex and marine sea formed by barriers, air quality management facilities to reduce particulate emissions from the exposed playa, a brine sink for discharge of salts, conveyance facilities, and sedimentation/distribution facilities (DWR 2007). The alternative consists of a series of three concentric independent lakes separated by dikes, with deep pools, and habitat islands (DWR 2007). Each lake would receive water directly from the New and Alamo Rivers and would operate at increasingly higher salinities, with evaporation concentrating salinities from 20,000 to 60,000 mg/L (DWR 2007). A brine pool would occupy the area of the innermost dike. The brine pools would be formed in the lakes at up to 20 feet in depth, enough to support a sustainable fishery (DWR 2007).

Since the size of the sea would be reduced under all feasible alternatives, there is a concern that intense and persistent thermal stratification may result at depths exceeding 10 meters (USBR 2007). This would cause the sea to shift from a system with several mixing events per year, to one that is mixed for a short period in the winter (USBR 2007). This stability coupled with the expected continuing eutrophication would make the hypolimnion of the sea anoxic for most of the year. During that time, hydrogen sulfide and ammonia concentrations could increase, reaching unprecedented levels (USBR 2007). The sudden redistribution of anoxia, hydrogen sulfide gas, and ammonia gas throughout the water column and their release to the air would likely occur during the single winter mixing event, potentially causing annual fish kills and odor problems (USBR 2007). The 2007 Restoration of the Salton Sea Summary Report describes five approaches aimed at reducing these risks: (1) reduce nutrient inputs, (2) avoid deep water to improve the efficiency of wind mixing, (3) mechanical circulation, (4) aeration/oxygenation/ozonation, and (5) pump water out of the Sea and treat it by ozonation/oxygenation before returning the treated water to the Sea.

In the interim, habitat values at the Salton Sea have continued to decline as salinity increased and water levels receded. To address these concerns, DWR and DFG are establishing the Salton Sea Financial Assistance Program, which will provide grants to eligible applicants for projects that conserve fish and wildlife habitat within the Salton Sea ecosystem. Further, the San Diego County Water Authority, the Imperial Irrigation District, and the Coachella Valley Water District have collaborated since 2003 to ensure the delivery of 165,000 acre-feet of water to the Salton Sea (San Diego County Water Authority 2011).

In an effort to decrease the loads of suspended sediments, nutrients, selenium, and pathogens into the Salton Sea, numerous TMDLs have been developed in recent years. While the Salton Sea Nutrient TMDL is currently still being developed, the following TMDLs for tributaries to the sea are already in place:

- New River Dissolved Oxygen TMDL (2010)
- New River Trash TMDL (2007)
- Imperial Valley Drains Sedimentation/Siltation TMDL (2005)
- New River Sedimentation/Siltation TMDL (2003)
- Alamo River Sedimentation/Siltation TMDL (2002)
- New River Pathogen TMDL (2002)

Additionally, two pilot treatment wetlands (Brawley and Imperial) were constructed on the New River in 2000, and their performance was monitored for several years (Tetra Tech 2006). The Imperial wetland receives water solely from agricultural drainage, and has a total area of 43 acres, 22 of which are wetted (Tetra Tech 2006). It has two sedimentation basins, operating in parallel, followed by four wetland cells in series (Tetra Tech 2006). The Brawley wetland has a total area of 9 acres, of which 6 acres are wetted, and receives water solely from the New River (Tetra Tech 2006). It has a single sedimentation basin and two wetland cells in series (Tetra Tech 2006). According to the 2006 Performance Evaluation report, "both wetlands were able to remove a substantial fraction of the influent load of the key contaminants monitored." They were found to remove more than 90% of influent suspended sediments and pathogens. Bioaccumulation of toxic chemicals, such as selenium, is a concern when using wetlands to enhance water quality in the region. The 2007 follow-up report concluded that, while the treatment wetlands pose some small level of ecological risk, they provide better habitat than the agricultural drains, the New and Alamo Rivers, and the Salton Sea (Tetra Tech 2007).

3.4 Costs

Available information indicates that for a 50-year lifespan, the total present worth cost of the Preferred Alternative is estimated to be between $241 and $262 million, based on an annual interest rate of 3.5%. The estimate breaks down as roughly $183 million in capital costs (i.e., land acquisition, engineering and design, permitting, biological and cultural surveys, and construction) and an estimated annual O&M and monitoring cost (present worth) of $1.2 to $1.6 million per year (Tetra Tech 2006).

Appendix B

Suggested evaluation criteria for pollutant reduction projects

The following are suggestions for project evaluation criteria developed by the Klamath River Pollutant Reduction Workshop Steering Committee. It is beyond the scope of the pre-workshop information packet to develop all of the information needed to apply the below criteria. Invited experts are encouraged to review the suggested criteria along with the technology summaries provided in Sections 4.1 to 4.6, and, based on their knowledge and experience, to bring any associated additional information to the workshop for broader group discussion. The set of criteria below will be further refined prior to the workshop.

Objective 1—Reduce nutrients

- quantity per unit volume water
- unit removal rate
- seasonal variations in rate
- nutrient form affected (e.g., total nitrogen [TN], ammonium [NH4+], nitrate [NO3-], total inorganic nitrogen [TIN], total phosphorus [TP], ortho-phosphorus [PO43-], soluble reactive phosphorus [SRP], particulate organic carbon [POC], dissolved organic carbon [DOC], total organic carbon [TOC])
- effective project life time
- implementation time frame
- cost per unit reduction
- energy density per unit reduction
- disposal cost
- others

Objective 2—Improve overall water quality

For dissolved oxygen, pH, water temperature, total suspended solids/turbidity, chlorophyll-a, and algal toxins:

- Change in parameter
- Seasonal max/min
- Steady state range
- Cost per unit change
- Energy per unit change
- Implementation time frame
- Effective life time
- Other
- Near shore water quality improvement (ocean)—factors changed (list), season of change (list), areal extent (shore miles)
- Subjective water quality ‘improvement’—reduction in algae (%), aesthetic improvement (H/M/L), recreation opportunity (+/-/0), season of greatest improvement (months), durability of result (H/M/L)

Objective 3—Support Designated Beneficial Uses

- Fisheries—effects on salmonids, suckers, eulachon, lamprey, game fish, others (+/-/0)
- Irrigation supply—surface water availability, surface water quality, ground water availability, ground water quality, return flow/ recharge, ET/consumptive use, other (+/-/0)

- Domestic Use—public entity/effect (name/effect), private entity/effect (name/effect)
- Industrial Water Supply—entity/effect (name, size, effect)
- Livestock Watering—entity/effect (name, size, effect)
- Wildlife and Hunting —species/effect (name, effect), area (acres)
- Fishing—tribal Harvest (species, effect), sport harvest (species, effect), commercial harvest (species, effect)
- Boating—change in opportunity (+/-/0)
- Water Contact Recreation—change in opportunity (+/-/0)
- Aesthetic Quality—visual change (+/-/0), olfactory change (+/-/0), taste change (+/-0)
- Hydro Power—change in flow regime (describe shift in average hydrograph), volumetric change (+/-/0), change in ramp rates (cfs/hour), change in fish passage (+/-/0), change in revenue (+/-/0), Federal Power Act compliance (+/-/0)
- Commercial Navigation and Transportation—change in opportunity (+/-/0)
- Overall ecosystem function—change (+/-/0), effects (list)
- Others

Global criteria

Some concerns are globally applicable to all of the objectives (e.g., what it costs, how fast it can be implemented, etc.). The Steering Committee has considered included a set of 'Global Criteria' that allow workshop participants to take a broader view of a particular technology, outside of the workshop objectives. The global considerations are in some ways duplicative of the more targeted evaluation criteria, but they offer a summary view of the technologies and projects and can assist the Steering Committee in collating results and weighting individual concerns:

- Demonstrable benefits to WQ, fisheries and T&Es (H/M/L)
- Cost (H/M/L)
- Energy use (H/M/L)
- CO2 load (H/M/L)
- Consumptive water use (H/M/L)
- Speed/rate (H/M/L)
- Effectiveness
    - Nutrient reduction (H/M/L), water quality improvement (H/M/L), support of BUs (H/M/L)
    - Legal feasibility (water rights, licensing and permitting, nuisance, insurance) (+/-/0)
    - Engineering feasibility (technology well developed, Infrastructure feasibility (power grid, materials, technological barriers) (+/-/0)
    - Cultural/social effects (noise, odor, aesthetics, displacement, environmental justice) (H/M/L)
    - Potential for unintended consequences (H/M/L)
    - Compatibility/synergy—with other interim measures (+/-/0), with restoration measures (+/-/0), with dam removal (+/-/0)
    - Job creation—short term (+/-/0), long term (+/-/0)

Narrative questions

A set of open-ended questions will invite a narrative response from each working group. These narratives will be useful to capture unexplored aspects of a project and other considerations that are tangential to the core project objectives, for example, what is the 'green score' of the chosen technology?

- How will this technology/measure interact with dam removal, should there be an affirmative Secretarial Determination on the removal of J.C. Boyle, Copco 1 and 2, and Iron Gate Dams?
- Is this technology/measure likely to facilitate improvement in juvenile sucker recruitment in Upper Klamath Lake?
- Does this technology/measure provide an acceptable cost to benefit ratio?
- Is this technology/approach a long-term solution or improvement?
- Does this technology/measure address multiple water quality problems? Is it a global solution?
- Are there readily identifiable legal constraints on this technology/measure?
- Are there likely to be opportunities for funding for this technology/measure?
- Are there readily identifiable political ramifications for this technology/measure?
- How does this technology/measure 'fit' with other approaches? Is there a hybrid of several approaches that would enhance overall treatment effectiveness?
- How 'green' is this technology/measure?
- Others?

Appendix C

Additional technical detail for potential projects

1 ADDITIONAL FIGURES FOR CLIMATE CHANGE AND HYDROLOGY

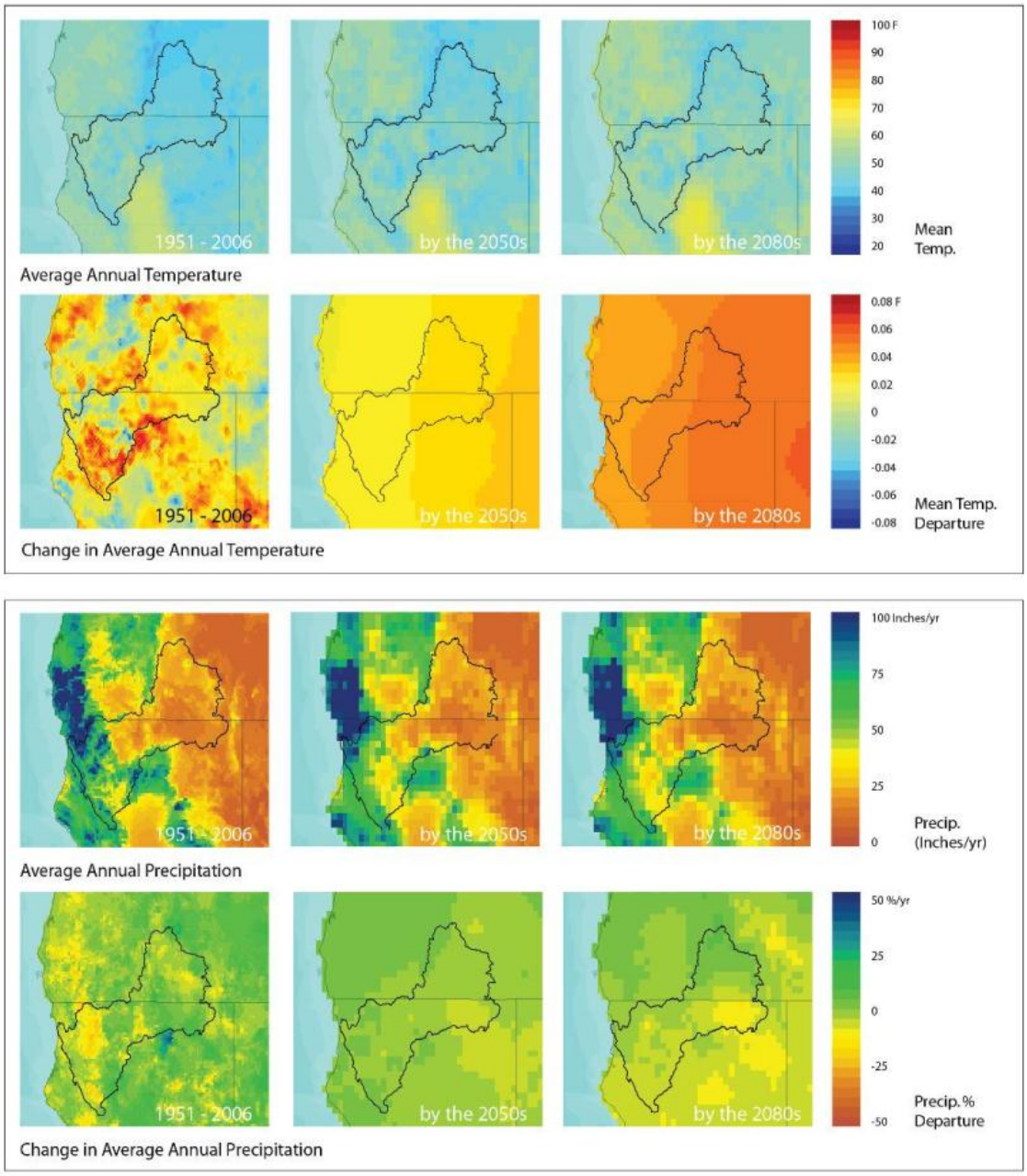

Figure C-31. Projected changes in annual temperature and precipitation for the Klamath Basin. Climate change model: Ensemble average (half in a suite of international climate monitoring models project a greater amount of change, and half of the models project less change as compared to the 1951–1990 baseline average). Source: Climate Wizard.

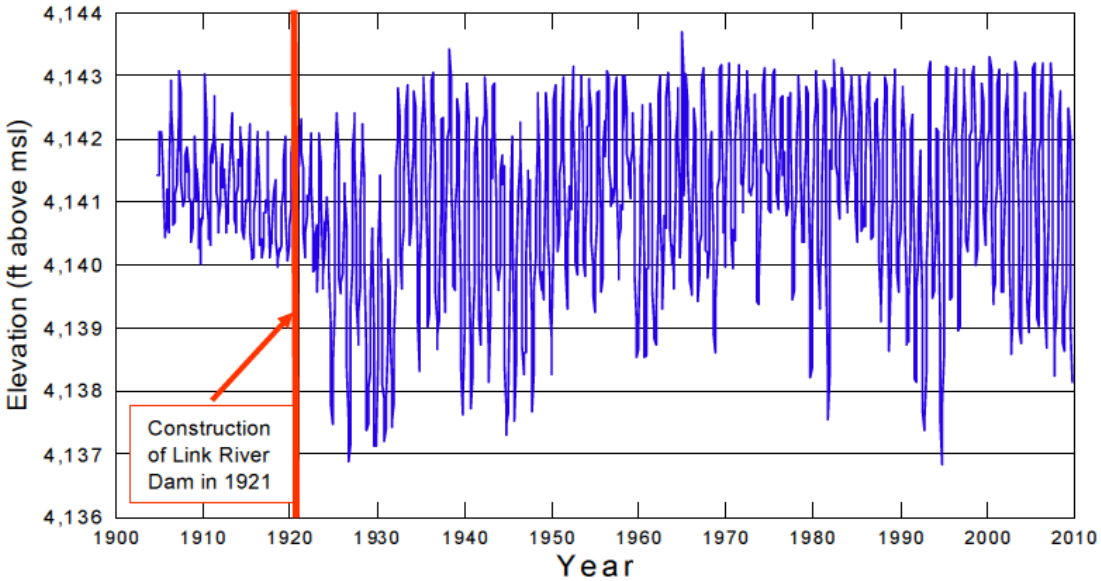

Figure C-32. Upper Klamath Lake surface elevation time series; end of month elevations for Water Years 1905-2009. Data source: www.usbr.gov/mp/kbao/operations/water/korep1.cfm?lakeid = ukldata1

2 REACH-SCALE WATER QUALITY SUMMARIES

2.1 Upper Klamath Lake and major tributaries (Wood, Williamson, and Sprague Rivers)

External phosphorus sources in Upper Klamath Lake include: (1) fluvial inputs from streams draining the catchment, (2) runoff and irrigation-return waters pumped from agricultural areas adjacent to the lake and tributaries, (3) diffuse sources such as springs and marshes, and (4) atmospheric deposition. Lake and outflow phosphorus levels are driven by external phosphorus loads and "internal phosphorus loads" cycling between the water column and bottom sediments (ODEQ 2002). While "internal phosphorus loads" released from bottom sediments in early summer contributes to algal blooms, they ultimately reflect anthropogenic antecedent external phosphorus loads stored and recycled from the bottom sediments since the early 1900's. Relative to the historical baseline, external phosphorus loads in Upper Klamath Lake have been increased due to anthropogenic activities (ODEQ 2002, Walker et al. 2012). Anthropogenic contributions were estimated to be 38% of the total external phosphorus load in 1992-1998 and 31% in 2008-2010 (Walker et al. 2012).

A phosphorus mass-balance model was subsequently developed for simulating the linkage between external phosphorus load, internal phosphorus cycling processes, and spatially-averaged lake water quality, expressed in terms of means and frequency distributions of total phosphorus (TP), chlorophyll-*a*, and pH (Walker 2001). The historical phosphorus budgets and model were utilized by ODEQ (2002) to establish targets for lake phosphorus concentrations and external to achieve lake water quality standards for algal biomass (expressed as chlorophyll a), and pH, as required under the Total Maximum Daily Load (TMDL) for Upper Klamath Lake (2002). In the TMDL the phosphorus-load was expressed in terms of a long-term-average under hydrologic conditions that occurred in that 7-year historical period of record (October 1991-September

1998). Achieving the loading target (109 metric tons/year of TP) would require a 40% reduction in external phosphorus load relative to the period of record loads. The average inflow TP concentration (66 parts per billion [ppb]) corresponding to the TMDL target was similar to the average value measured in springs and other relatively un-impacted sources in the watershed, which were assumed to represent natural background conditions.

Similar to an earlier phosphorus budget effort by Miller and Tash (1967), who estimated sediment recycling of phosphorus (internal phosphorus loading) to be 57% of the total load, Kann and Walker (1999) and ODEQ (2002) estimated that phosphorus recycling from lake sediments accounted for 61% of the total phosphorus loading to Upper Klamath Lake, with much of this recycling occurring during the summer algal growing season. Efforts to understand internal phosphorus recycling indicate a consistently positive benthic flux for soluble reactive phosphorus (SRP) (Kuwabara et al. 2007, et al. 2009). Kuwabara et al. (2009) also concluded that diffusive flux alone could not account for water column phosphorus increases and concluded that sediment resuspension and bioturbation were also important mechanisms of nutrient release. Another recent study by Simon et al. (2009) confirm that measurable losses of phosphorus from surficial sediments of Upper Klamath Lake were a source of phosphorus during an algal bloom, but also indicate that the internal loading mechanisms they evaluated could not account for the total internal load of phosphorus in Upper Klamath Lake. Sampling of the benthic invertebrate community in the lake has revealed high densities of benthic invertebrates in the sediments (Kuwabara et al. 2012) that could contribute to internal loading through metabolic cycling as well as bioturbation. Simon et al. (2009) indicate that iron plays a role in SRP release (dissolution of phosphorus complexed to iron) at varying spatial and temporal scales; but also that this type of phosphorus loss could not account for all of the phosphorus release for the lake as a whole. Translocation of sediment phosphorus through AFA akinete migration also could not account for a majority of sediment phosphorus recycling in Upper Klamath Lake (Barbiero and Kann 1994).

All of the above mechanisms can be enhanced through wind-driven resuspension in the shallow Upper Klamath Lake (Laenen and LeTourneau 1996). Notwithstanding the precise internal phosphorus loading mechanism (diffusion, pH or anaerobic mediated release, microbial and macroinvertebrate metabolic cycling, bioturbation), it is clear that many of the mechanisms operate at varying temporal and spatial scales and play an important role in driving AFA dynamics in Upper Klamath Lake. However, the ultimate sediment phosphorus source is watershed-derived, reflecting prior loading inputs (Walker et al. 2012).

Watershed concentrations of nitrogen tend to be low leading to low total and bioavailable N:P ratios in inflows to Upper Klamath Lake (<1 for bioavailable forms) with increased ratios in the Upper Klamath Lake outflow due partially to N-fixation occurring in the lake (e.g., Kann 2012b, Walker et al. 2012). Although observations of nitrogen and phosphorus are unavailable for Upper Klamath Lake prior to ca. 1956, data suggest that water column N:P ratios have been low for a least the past 40 years (cf. Phinney et al. 1959, Klamath Consulting Service 1983). More recent studies show that N:P ratios in Upper Klamath Lake are clearly in the range (generally less than 10:1 by weight during the algal growing season for the total forms of these nutrients) that indicates dominance by N-fixing blue-green algae should occur (Kann 1998, Kann 2011, Kann 2012a, Lindenberg et al. 2009, Jassby and Kann 2010). While N:P ratio influences algal species composition, phosphorus concentration is a major factor controlling algal biomass (Kann 1998, Kann and Walker 1999, Lindenberg et al. 2009). Analyses indicate that when TIN and SRP ratios were less than 5:1 in Upper Klamath Lake, the percentage of AFA containing heterocysts (the specialized cells where N-fixation takes place) increased (Kann 1998). Large net-negative retention of nitrogen in Upper Klamath Lake (especially during the summer bloom period;

Walker et al. 2012), low TN:TP and TIN:SRP ratios, as well as the seasonal timing of Upper Klamath Lake nitrogen increases, demonstrate the ability of AFA to overcome nitrogen limitation, and grow to large levels based on the amount of available phosphorus.

Seasonal patterns of low dissolved phosphorus during bloom growth, increasing dissolved phosphorus after bloom decline, and negative correlations between dissolved phosphorus and algal biomass provide evidence for early season phosphorus limitation (Kann 2012a, Hoilman et al. 2008). Potential phosphorus-limitation characterized by chlorophyll-*a* to phosphorus ratios greater than 1 were also observed during initial AFA blooms (generally in late June and early July), but to a lesser degree during late-season blooms (Lindenberg et al. 2009). In addition, as the season progresses, growth becomes increasingly restricted by light, as indicated by the increased attenuation coefficients and the decreasing water column light intensity during mid-May to early July which coincides with the primary growth and increase in AFA biomass (Hoilman et al. 2008, Kann 2010a). USGS studies have also found a correlation between chlorophyll-*a* concentrations and photic zone depth (Hoilman et al. 2008, Lindenburg et al. 2009, Wood et al. 2006) and noted the potential for algal self-shading, especially at deeper sites (Hoilman et al. 2008, Lindenburg et al. 2009). There is some indication that a “clear water” phase may precede annual AFA increases and may serve to promote nitrogen fixation generally which generally has a high energy/light requirement (Kann 2012a).

The typical seasonal pattern of nitrogen and phosphorus in Upper Klamath Lake is shown in

Figure C-33 (Kann 2012a). The pattern in the amount of available nitrogen appears to influence bloom dynamics (higher levels in 2011 were associated with a depressed AFA bloom); and the pattern in SRP, which remains low even as algal biomass increases, indicates uptake of available phosphorus as the AFA blooms actively grow. Although the reason for the bloom decline is not clearly understood, both TIN and SRP increase as the bloom declines (crashes), indicating that these nutrients are not limiting algal growth during the bloom decline period (Figure C-33).

The most recent Upper Klamath Lake hydrologic and nutrient balance computations that encompass water years 1992-2010 provide the context for nutrient management in the Upper Basin (Walker et al. 2012). For example, substantial research, water quality monitoring, watershed management, and wetland restoration management efforts have occurred since development of the WY 1992–1998 nutrient budgets (Kann and Walker 1999) that provided a baseline for the TMDL (ODEQ 2002). Several of the large drained wetlands on the periphery of Upper Klamath Lake are no longer grazed or farmed, and are in varying stages of restoration (e.g., Wong et al. 2011, USBLM 2005, Carpenter et al. 2009, Duff et al. 2011). Watershed conservation projects such as those implemented by the Klamath Basin Rangeland Trust (KBRT) in the Wood River and Sevenmile Creek drainages have sought to reduce water use and decrease nutrient export from grazed areas (e.g., GMA 2011a, GMA 2011b).

The annual water and nutrient mass balances for the overall period are summarized in Table C-1; however, the interannual and seasonal trends are also of primary importance and these are shown in detail in Walker et al. (2012). Major conclusions include: changes in TP storage and net retention are highly variable from year-to-year but of lower magnitude than the inflows and outflows; the negative TN retention values primarily reflect atmospheric fixation of nitrogen by AFA and exceed the average inflow load by more than 3-fold; data suggest there is a one-year lag in the response of the annual outflow TP loads to variations in the inflow TP loads; large negative spikes in net retention occur in June and July and reflect high rates of phosphorus recycling from bottom sediments and N-fixation by blue-green algae; TP concentration ranges generally

decreased in WY 2008–2010 and may reflect decreased transient nutrient releases from antecedent agricultural soils in the restored wetland areas (e.g., Wong et al. 2011).

Table C-1. Water and nutrient mass balances for water years 1992–2010. Source: Walker et al. 2012.

Upper Klamath Lake Water and Nutrient Balances, Water Years: 1992–2010

	Flow	Nutrient Loads		Percent of Inflow			Nutrient Concs		Dr. Area	Runoff	P Export	N Export
Term	hm^3/yr	TP mt/y	TN mt/y	Flow	TP	TN	TP ppb	TN ppb	km^2	m/yr	kg/km^2	kg/km^2
Major Gauged Sites												
Wood River @ Weed Road	246.9	21.0	28.0	16%	13%	5%	85	114	333	0.74	63	84
Wood River @ Dike Road	317.7	35.6	55.7	20%	22%	10%	112	175	394	0.81	90	141
7-Mile Canal	103.4	14.8	49.2	6%	9%	9%	143	476	96	1.07	153	510
Sprague River	501.9	38.1	177.3	32%	23%	33%	76	353	4171	0.12	9	43
Williamson River	845.9	73.4	296.3	53%	45%	55%	87	350	7812	0.11	9	38
Klamath L Outlet	1439.2	162.4	2364.2	90%	99%	440%	113	1643	9771	0.15	17	242
Agency Lake Inflows												
Wood River above Weed Rd	246.9	21.0	28.0	56%	34%	18%	85	114	333	0.74	63	84
Wood River below Weed Rd	70.9	14.5	27.7	16%	24%	18%	205	391	61	1.17	239	455
7-Mile Canal	103.4	14.8	49.2	23%	24%	32%	143	476	96	1.07	153	510
Agency Lake Ranch	21.0	11.0	49.0	5%	18%	32%	525	2330	46	0.45	238	1056
Total Agency Inflow	442.1	61.4	153.9	100%	100%	100%	139	348	537	0.82	114	287
Klamath Lake Inflows												
Sprague River	501.9	38.1	177.3	58%	45%	50%	76	353	4171	0.12	9	43
Williamson - Sprague	344.0	35.3	118.9	40%	42%	33%	103	346	3641	0.09	10	33
Pumped Inflows to KL	23.8	11.4	59.9	3%	13%	17%	479	2520	50	0.47	227	1194
Total Klamath Inflow	869.7	84.8	356.2	100%	100%	100%	98	410	7862	0.11	11	45
Overall Balance												
Total Tributaries	1267.0	123.8	401.3	80%	75%	75%	98	317	8302	0.15	15	48
Total Pumped to Lake	44.8	22.4	108.9	3%	14%	20%	501	2431	97	0.46	232	1128
Ungauged Inflows	279.1	18.1	27.6	18%	11%	5%	65	99	1105	0.25	16	25
Total External Inflows	1590.8	164.3	537.7	100%	100%	100%	103	338	9504	0.17	17	57
Precipitation	114.2	4.8	28.7	7%	3%	5%	42	251	267	0.43	18	107
Evaporation	255.6			16%					267	0.96		
Net Inflow	1449.4	169.1	566.4	91%	103%	105%	117	391	9771	0.15	17	58
Lake Outflow	1439.2	162.4	2364.2	90%	99%	440%	113	1643	9771	0.15	17	242
Storage Increase	2.6	-0.9	0.1	0%	-1%	0%						
Retention	7.6	7.6	-1797.8	0%	5%	-334%						
Natural Background vs. Anthropogenic Loads												
Background / Natural	1590.8	103.4	157.5	100%	63%	29%	65	99	9504	0.17	11	17
Anthropogenic	1590.8	60.9	380.2	0%	37%	71%	38	239			6	40

Morphometry	Mean	Min	Max
Volume (hm^3)	546.1	224.2	743.5
Area (km^2)	267.1	217.4	270.9
Elevation (ft)	4140.9	4136.8	4143.3
Mean Depth (meters)	2.0	1.0	2.7

Phosphorus Model Parameters		
Hydraulic Residence Time	0.38	years
Net Water Load	5.96	m/yr
Areal Total P Load	0.62	g/m^2-yr
Total P Retention Coefficient	5%	

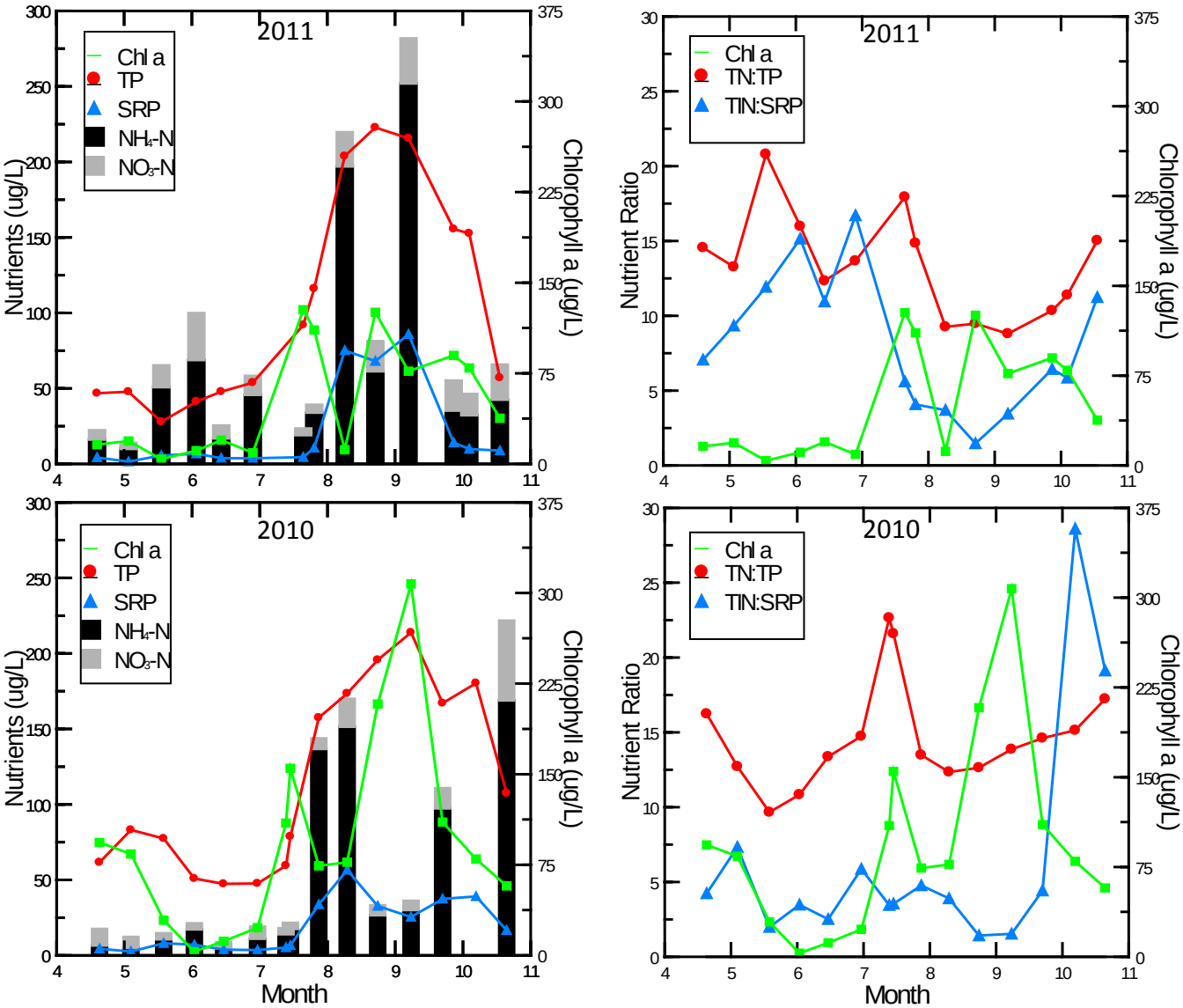

Figure C-33. Lake-wide mean chlorophyll, SRP, and TIN time-series for Upper Klamath Lake in 2010 and 2011. Source: Figure 14 from Kann 2012a.

Time series trend analyses from Walker et al. (2012) indicate:

- increasing trends in the total inflow volume to Agency Lake and discharges from the Wood River basin;
- increases in Wood River flows may be related to decreased agricultural withdrawals and management programs aimed at increasing Wood River flow;
- an apparent decreasing trend in flow-weighted mean (FWM) concentrations in the Wood River likely reflect KBRT watershed management and grazing strategies aimed at reducing nutrient input in Agency Lake tributaries as well as implementation of wetland restoration projects (Agency Lake Ranch and Wood River Ranch);

- significant decreasing trends in FWM TP concentrations in discharges from the Williamson River excluding the Sprague River, lower Wood River (below Weed Road), pumped inflows, and in the total inflows;
- no trends in FWM TP concentrations were observed in the upper Wood River (above Weed Road), Sevenmile Creek, or the Sprague River;
- results for TN were qualitatively similar, with the exception that trend magnitudes were slightly lower; and
- despite long-term decreases in the total inflow nutrient concentrations and loads, significant trends in outflow loads or net retention were not observed over the 1992–2010 period.

In addition, apparent shifts in the monthly lake TP retention and concentration trends were observed where retention increased in June and decreased in July while lake concentrations decreased in June but increased in August and September indicating a forward-shift in the timing of internal loading and bloom dynamics consistent with Jassby and Kann (2010), who indicated a shift towards a later maximum in the timing of peak algal biomass (Walker et al. 2012).

Estimates of TP load reduction in WY 2008–2010 relative to the TMDL baseline (31%) reflect both reductions in flow (22%) attributed to variations in precipitation and reductions in FWM concentration (11%) attributed to restoration and management efforts or other factors (Walker et al. 2012). Corresponding reductions in TN load and concentration relative to the TMDL baseline were 36% and 18%, respectively. The combined inflow TP concentration from external sources averaged 94 ppb in WY 2008–2010, as compared with 113 ppb in WY 1992–1994, 105 ppb in WY 1992–1998 (TMDL baseline), and the TMDL goal of 66 ppb (ODEQ 2002). Average TP loads in those periods were 124, 145, and 180 mt/yr, respectively, as compared with the TMDL goal of 109 mt/yr (Walker et al. 2012).

Spatial gradients in TP and TN concentrations reflect cumulative impacts of anthropogenic inputs along each tributary (moving downstream from relatively un-impacted headwaters in each basin; Figure C-34. Spatial variations in FWM nutrient concentrations, WY 2002–2010. Source: Figure 16, Walker et al. 2012.). Although changes in response to nutrient management are being detected in the 1992–2010 data set, increasing downstream FWM nutrient concentrations indicate the potential for additional management opportunities, and current trends also reflect an expected lag in the response time of nutrients to restoration efforts.

For example, restoration of deltaic wetlands surrounding the mouths of the Wood and Williamson Rivers is ongoing, and maturation of vegetation, soil accretion, as well as wetland soil-water column equilibration occurs at varying timescales (Aldous et al. 2005, et al. 2007; Carpenter et al. 2011; Duff et al. 2011; Elsroad et al. 2011; Graham et al. 2005; Kuwabara et al. 2012; Wong et al. 2009; USBLM 2005). Recent research indicates that phosphorus released from the Williamson River Delta restoration project initially increased after flooding, but declined rapidly to a fraction of the annual load prior to reconnection with the lake (Wong et al. 2011), and additional measurements in the wetland also indicated a declining nutrient flux (Wong and Hendrickson 2011).

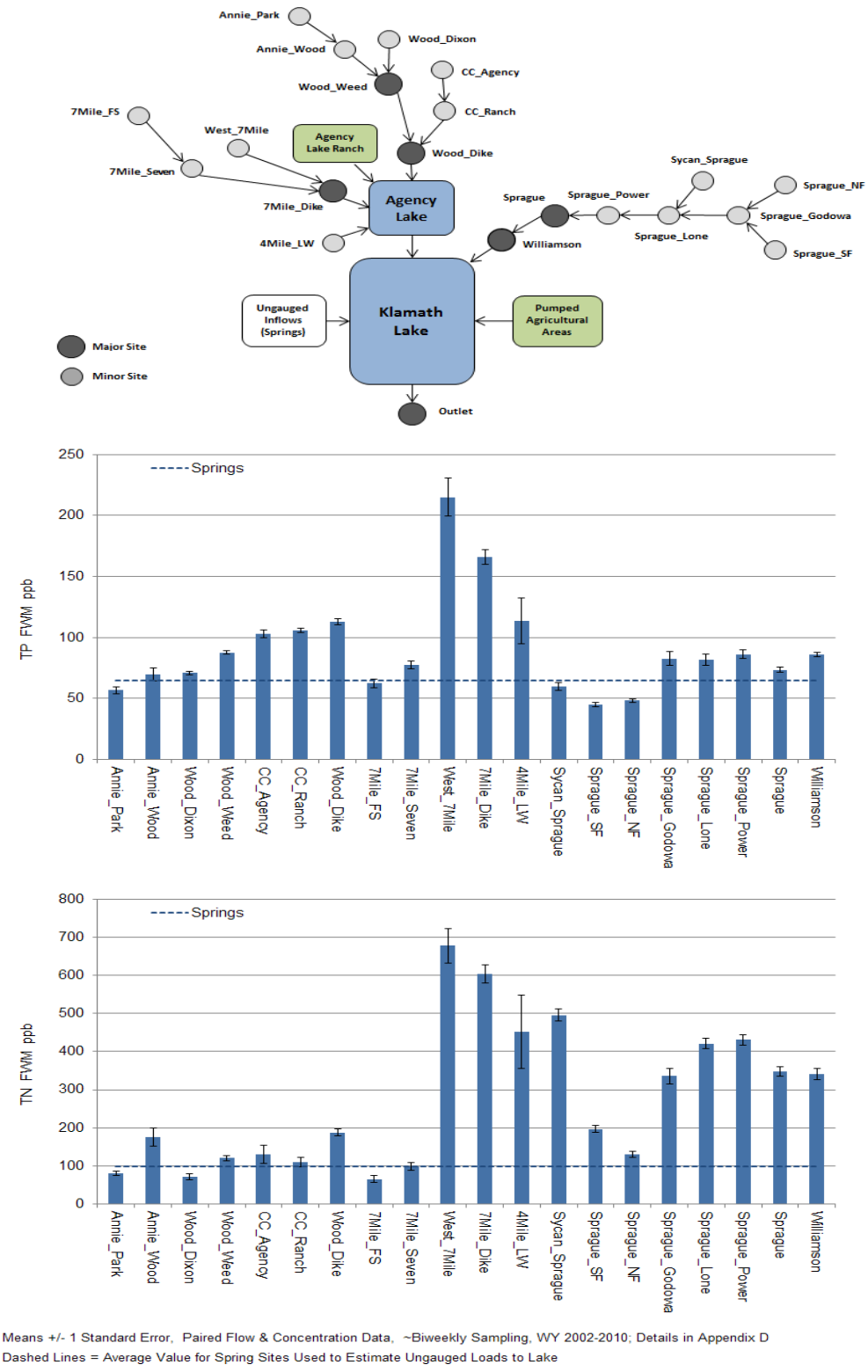

Figure C-34. Spatial variations in FWM nutrient concentrations, WY 2002–2010. Source: Figure 16, Walker et al. 2012.

2.2 Link River to Keno Dam

Relative to monitoring stations at the upstream end of this reach (i.e., Link River and Miller Island), the KSD has substantially higher concentrations of total and dissolved forms of phosphorus and carbon through the entire year; and total nitrogen concentrations are higher in the KSD except during July–September, and estimated particulate phosphorus (PP) and total inorganic nitrogen (TIN) concentrations are higher in January–June (Figure C-35). In contrast, chlorophyll concentrations in the KSD are lower (Figure C-35) indicating low levels of live algae. Limited available data from the Lost River Diversion Channel suggest that while this canal it has lower nutrient concentrations than the KSD, it has higher total phosphorus (TP) and soluble reactive phosphorus (SRP) concentrations than Klamath River mainstem stations (Figure C-35). The effect of these two drains on mainstem river nutrient concentrations is variable by year, depending on their relative flow contribution. For 1990–2000, the average monthly relative contribution of the KSD to the river during the irrigation season was 20% or less (Mayer 2001), but in August 2002, a year with very low mainstem flows, the KSD's flow contribution was 52% (ODEQ 2010). Modeling indicates that the drains have relatively little effect on dissolved oxygen under current conditions because dissolved oxygen is already so low in the summer and fall, but that the drains could have a much larger effect in the future if the organic matter load from Upper Klamath Lake were to be reduced (Sullivan et al. 2012).

As water flows through Keno Reservoir, there is a shift in the composition of nutrients and organic matter from particulate to dissolved forms (Figure C-35 through Figure C-37) for the July–December period. This is primarily due to internal processes (e.g., settling, decomposition, and release of dissolved nutrients from the sediments), but also due to inputs from the KSD. For January through September, concentrations of total phosphorus are higher at Keno Dam than Link River (Figure C-35 and Figure C-36) and similar for October–December. In December–June, total nitrogen concentrations rise slightly between Link River and Keno Dam, while the opposite generally occurs during the rest of the year (Figure C-35 and Figure C-37).

2.3 Keno Dam to upstream of Copco 1 Reservoir

Deas (2008) conducted a detailed study of the 4.7 mile reach between Keno Dam and J.C. Boyle Reservoir for the months of June–September in 2007. Total nitrogen and total phosphorus concentrations were nearly unchanged from the top and bottom of the reach, but there were substantial changes in form of nutrients and organic matter. The majority of ammonia was converted to nitrate through nitrification, chlorophyll decreased as phytoplankton cells died in the turbulent reach, and PO_4 increased due to breakdown of organic matter. The same trends are evident in a monthly summary of data from the years 2000–2011 (Figure C-36 and Figure C-37).

In the bypass reach between J.C. Boyle Dam and the J.C. Boyle Powerhouse, most of the Klamath River's flow is diverted into a canal. Approximately 1 mile downstream of the dam, high-volume springs contribute to the Klamath River. Nutrient concentrations in these springs have never been directly sampled, but IFR and PCFFA (2009) estimated long-term average nutrient concentrations for these springs using mixing equations and empirical data and determined that the springs substantially dilute concentrations of dissolved and particulate constituents such as phosphorus, nitrogen, and carbon. The springs also cool the river in summer.

The full flow of the Klamath River is returned at the J.C. Boyle Powerhouse. The river between the J.C. Boyle Powerhouse and Copco Reservoir is referred to as the "peaking reach" and the Klamath River's flow fluctuates on a daily cycle due to hydropower peaking operations. In

conjunction with the springs upstream, these peaking operations result in high intra-daily variability in nutrient concentrations. Thus, depending on the timing of when samples are collected within the daily hydrograph, sampled concentrations can be either higher or lower than daily flow-weighted average concentrations, which should be taken into account during data analysis and interpretation.

Water quality trends evident from Keno Dam to upstream of J.C. Boyle Reservoir (i.e., nitrification, continued reduction in phytoplankton and breakdown of organic matter) continue between J.C. Boyle Dam and Copco Reservoir (Figure C-36 and Figure C-37). In the summer months, the Klamath River upstream of Copco Reservoir exhibits the highest nitrate concentrations (and highest percent of total nitrogen in the form of nitrate) of anywhere on the mainstem Klamath (Figure C-37).

2.4 Iron Gate and Copco 1 Reservoirs

Nutrient budgets for Copco and Iron Gate Reservoirs are dominated by mainstem inflows and outflows, with other tributaries being only relatively minor contributors due to the combination of both lower flows and lower nutrient concentrations in the tributaries (Asarian and Kann 2009). Concentrations of total nitrogen (TN) are consistently lower in the Klamath River downstream of Iron Gate Reservoir than upstream of Copco Reservoir (Figure C-37), with occasional exceptions. This is likely due to 1) settling of organic matter and algal material, 2) nutrient storage in the water column and sediments of the reservoirs, 3) penstock intakes that draw water from intermediate depths where concentrations are lower, and 4) possible atmospheric losses through denitrification (Asarian and Kann 2009). On an annual basis, the reservoirs retain approximately 12% of incoming TN load (Asarian et al. 2009). During the warmer months their effect is greater; for July–September, TN concentrations at Iron Gate Dam have been measured at 39% lower than the reservoirs' combined flow-weighted inflow (Asarian et al. 2010).

Relative to the Klamath River upstream of Copco Reservoir, total phosphorus (TP) concentrations in the Klamath River downstream of Iron Gate Reservoir are generally equal or lower from January through August or September (varies by year), but then exhibit an opposite pattern until approximately December (Asarian et al. 2011) (Figure C-36). On a seasonal basis, reservoir sediments can release phosphorus to the water column during periods of seasonal hypolimnetic anoxia; however, most of the phosphorus released from the reservoir sediments during the anoxic period appears to remain within the hypolimnion until the reservoirs begin to turn over in the fall, and therefore is primarily not released into the river during the summer period of peak primary productivity downstream. An exception to this is that in many years TP concentrations are higher downstream of Iron Gate Dam than upstream of Copco Reservoir during the months of August through October when peak in-reservoir algal blooms occur. The reservoirs retain approximately 13% of incoming TP load on an annual basis (Asarian et al. 2009), and during the months of July–September TP concentrations at Iron Gate Dam are have measured at 13% lower than the reservoirs' combined flow-weighted inflow (Asarian et al. 2010).

During the late summer and fall when reservoir algal blooms senesce and the thermal stratification begins to break down, water with low dissolved oxygen is released into the Klamath River from Iron Gate Dam (PacifiCorp 2005, NCRWQCB 2010). The reservoirs also cause a thermal lag in which releases from Iron Gate Dam are cooler in the spring and warmer in the late summer and early fall than they would be without the reservoirs (PacifiCorp 2005, NCRWQCB 2010, Perry et al. 2011).

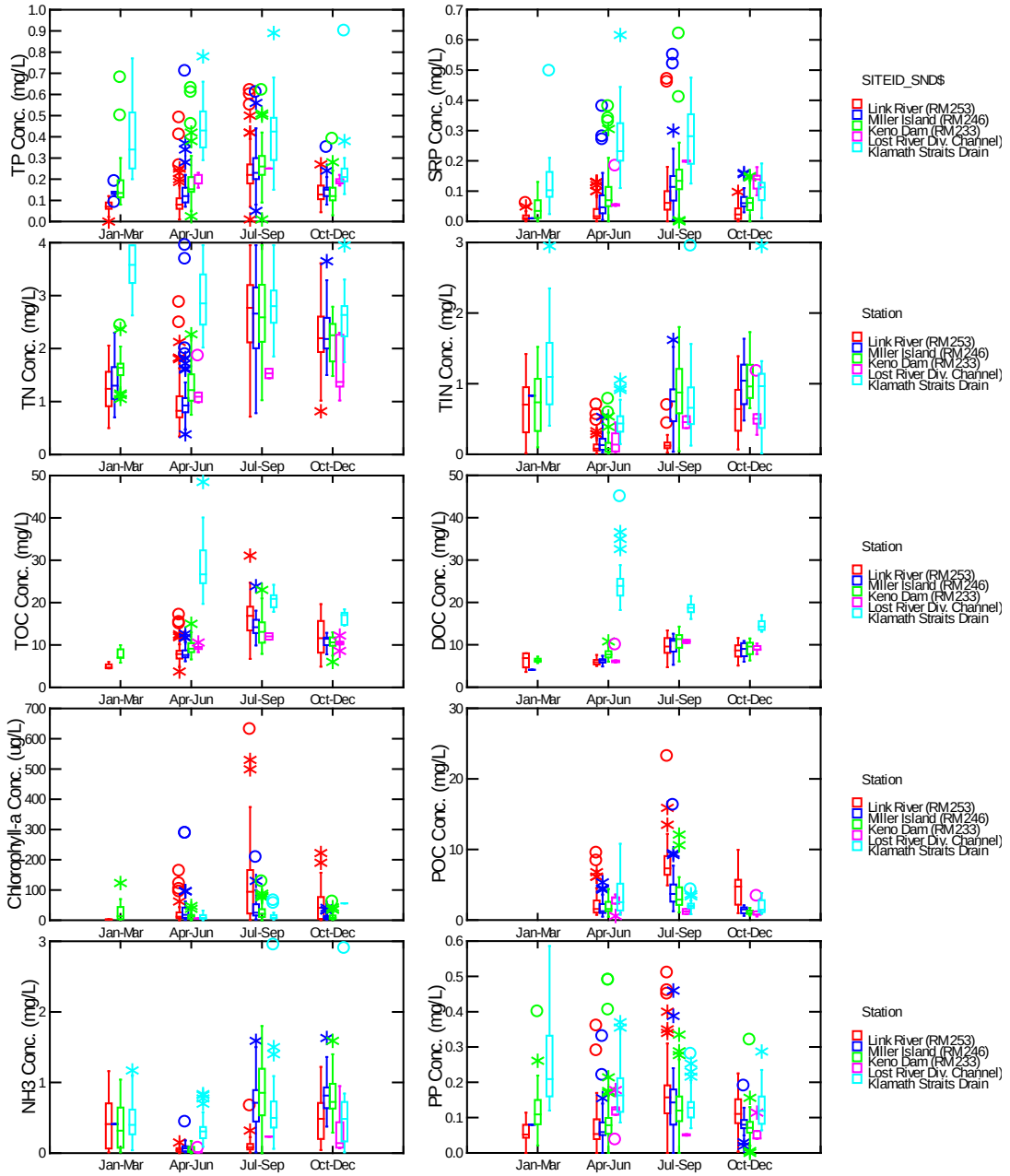

Figure C-35. Box slots showing seasonal summary of phosphorus, nitrogen, carbon and chlorophyll parameters for three mainstem locations between Link and Keno dams as wells as two drains than enter Keno Reservoir. Data span 2000–2011 and are from USGS, ODEQ, USBR, and PacifiCorp. TP= total phosphorus, SRP=soluble reactive phosphorus, PP= estimated particulate phosphorus (calculated as TP minus SRP), TOC = total organic carbon, DOC = dissolved organic carbon, and POC = particulate organic carbon, TN= total nitrogen, TIN = total inorganic nitrogen, and BOD5= 5-day biological oxygen demand. Sample size varies by season (fewer samples in Oct–Dec and Jan–Mar), parameter (far fewer samples for POC), and location (far fewer samples at Lost River Diversion Channel). Link River combines several stations between Link Dam to the mouth of Link River, and Keno Dam includes stations upstream and downstream of the dam.

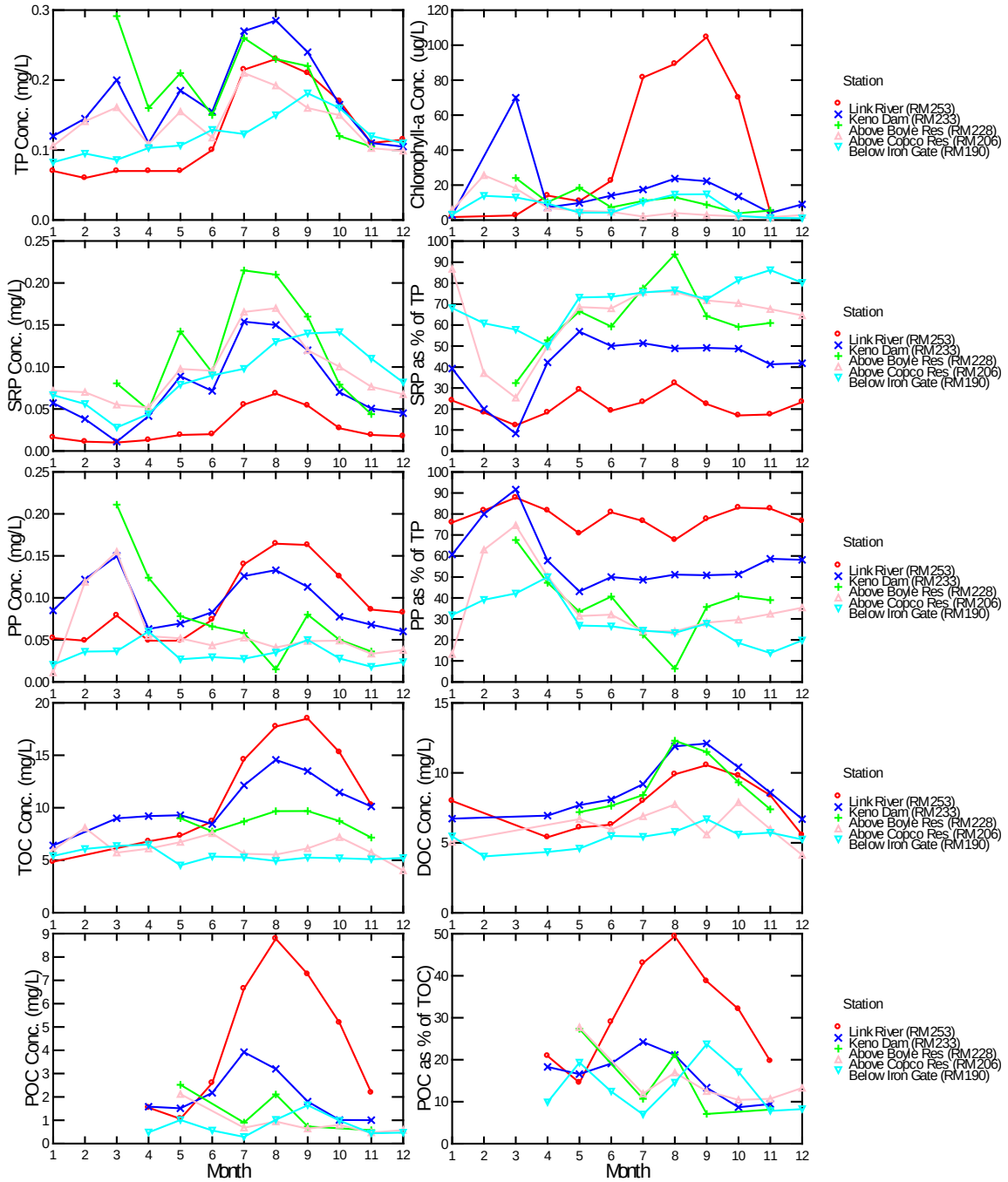

Figure C-36. Monthly summary of phosphorus, chlorophyll, and carbon parameters for five locations between Link and Iron Gate dams for 2000–2011. Data are from USGS, ODEQ, USBR, PacifiCorp, Karuk Tribe, and USFWS. TP= total phosphorus, SRP=soluble reactive phosphorus, PP= estimated particulate phosphorus (calculated as TP minus SRP), TOC = total organic carbon, DOC = dissolved organic carbon, and POC = particulate organic carbon. Values are medians by month (all years combined). Sample size varies (minimum=2, maximum=125) by month (far fewer samples during November–April than May–October), parameter (far fewer samples for POC), and location. Link River combines several stations between Link Dam to the mouth of Link River, and Keno Dam includes stations upstream and downstream of the dam.

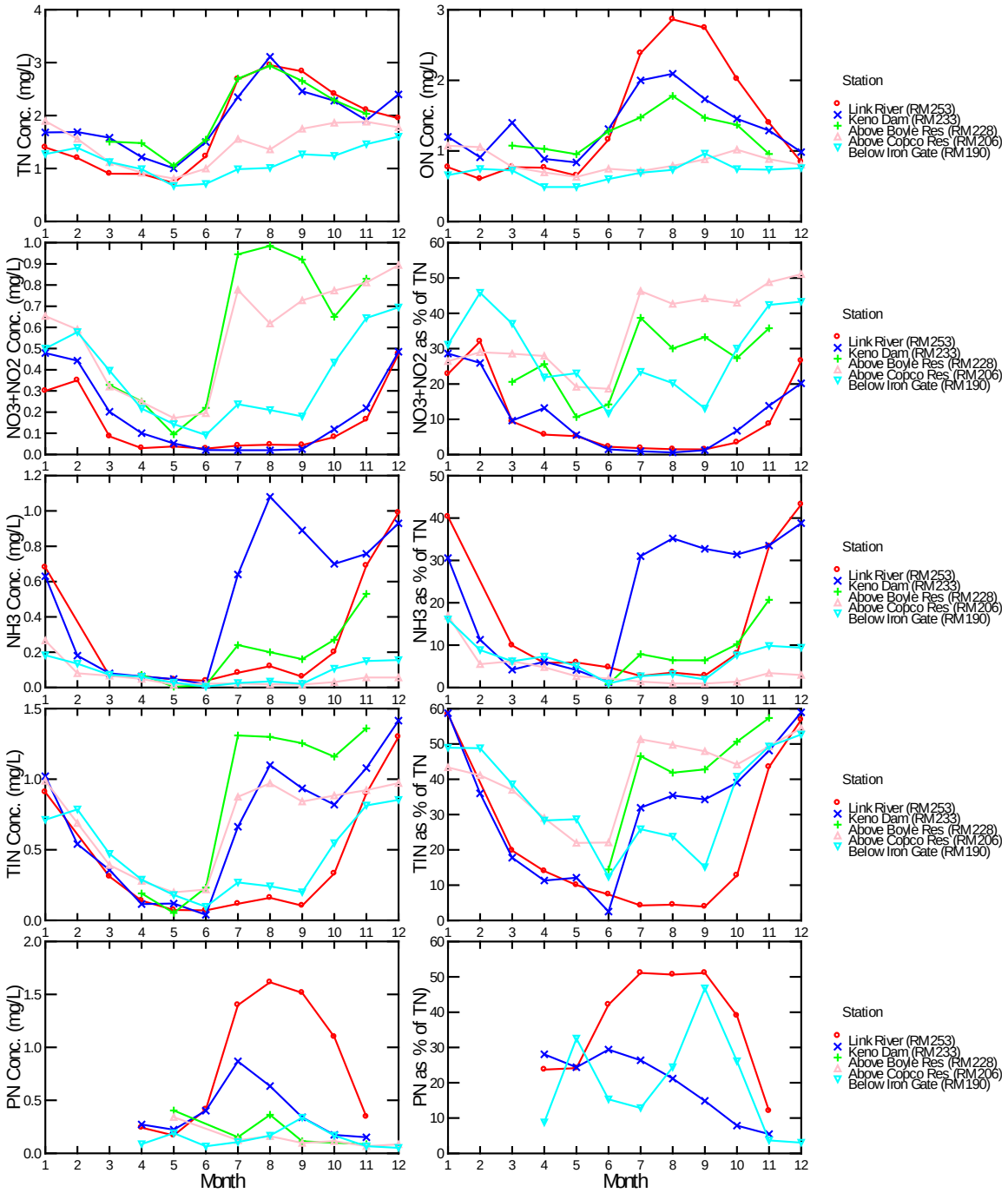

Figure C-37. Monthly summary of nitrogen parameters for five locations between Link and Iron Gate dams for 2000–2011. Data are from USGS, ODEQ, USBR, PacifiCorp, Karuk Tribe, and USFWS. TN= total nitrogen, ON=organic nitrogen, NO3+NO2= nitrate plus nitrite, NH3= ammonia, TIN = Total inorganic nitrogen, and PN = particulate nitrogen. Values are medians by month (all years combined). Sample size varies (minimum=2, maximum=126) by month (many more samples during May–October than November–April), parameter (far fewer samples for PN), and location. Link River combines several stations between Link Dam to the mouth of Link River, and Keno Dam includes stations upstream and downstream of the dam.

2.5 Periphyton

There have been almost no surveys of periphyton in the reach from Keno Dam to upstream of Copco 1 Reservoir. A substantial portion of the reach is confined with high gradient and less than ideal physical conditions for periphyton growth. In addition, it is likely that daily hydropower peaking operations reduce periphyton populations by scouring during peaking flows, drying during low flows, and reduced light availability (due to increased depth and turbidity) during peaking flows (PacifiCorp 2005b).

Despite hydropower peaking and an overall steep gradient, several lines of evidence suggest active periphyton populations with this reach. The elevation loss between Keno Dam and Copco Reservoir is a relatively steep (1% gradient); however, approximately 10.7 miles of that length possesses lower (25–27 ft/mile) gradient, not including the area impounded by J.C. Boyle Reservoir (Karuk Tribe 2006). The lower gradient areas provide potential habitat for periphyton. The reach from Keno Dam to upstream of Copco Reservoir also exhibits higher absolute and relative retention rates for nitrogen and phosphorus on a per-mile basis than river reaches further downstream (i.e., between Iron Gate Dam and Turwar) (Asarian et al. 2010), and periphyton uptake is the most likely explanation for these high retention rates. In addition, continuous water quality probes deployed at the Klamath River upstream of Copco in 2000 (Watercourse Engineering 2003) and 2002 (USFWS, unpublished data) indicate diel cycles in pH and D.O. with similar magnitude to that observed at sites downstream of Iron Gate Dam such as Seiad Valley. These diel cycles are likely due to a combination of hydropower peaking and periphyton photosynthesis/respiration but it is extremely difficult to ascertain the relative contribution of each.

Periphyton and macrophytes (rooted aquatic plants) thrive in the Klamath River from Iron Gate Dam to the Klamath Estuary due to an ample supply of nutrients and low stable summer flows. Relatively few periphyton samples have been collected in the Klamath River, but available data show a longitudinal shift in summer periphyton species composition. N-fixing periphyton species are absent or rare in the upper portion of the reach (i.e., from Iron Gate Dam to approximately Seiad Valley) but the periphyton community transitions toward domination by species capable of fixing nitrogen (e.g., diatoms in the genera *Epithemia* and *Rhopalodia* and cyanobacteria in the genera *Calothrix* and *Rivularia*) in the lower portions of the reach (i.e., from Orleans to Turwar) (HVTEPA 2008, Asarian et al. 2010).

In addition to affecting dissolved oxygen and pH, periphyton also play a role in the life cycle of the parasite *Ceratomyxa shasta*, which causes the often fatal disease ceratomyxosis in salmonids. The actinospore stage of *C. shasta* infects salmonids and the myxospore stage infects the alternate host, a polychaete worm *Manayunkia speciosa* (Bartholomew et al. 1997). The Klamath River between the Shasta River and Seiad Valley is referred to as the “infectious zone” where high percentage of juvenile Chinook and coho salmon become infected with *C. Shasta* due to high concentrations of *C. shasta* actinospores in the water column (Bartholomew 2011). In the Klamath River, *M. speciosa* is often associated with two microhabitats: (1) sand-silt with fine benthic organic matter, and (2) *Cladophora*, a filamentous green algal periphyton species that thrives in nutrient-rich waters (Stocking and Bartholomew 2007). The presence of *Cladophora* allows *M. speciosa* to occupy higher velocity habitats (Stocking and Bartholomew 2007). The distribution of *Cladophora* in the Klamath River has not been systematically mapped.

3 POTENTIAL WATER TREATMENT TECHNOLOGIES

The following is additional technical information for potential water treatment technologies considered for the Klamath River Pollutant Reduction Workshop.

3.1 Wetland Restoration

3.1.1 Treatment cost estimates

Examples of wetland restoration costs from the general literature and those specific to the Klamath Basin are provided in Table C-2. Some of the costs are specific to habitat-focused wetland restoration, while others are based on costs developed for treatment wetlands. While there are likely many similarities in costs between the two types of wetlands, treatment wetlands are more engineered systems and may have higher design, earthwork, and hydrologic control costs due to the specific sizing and plumbing of treatment cells.

Table C-2. General cost considerations and example cost estimates for wetland restoration projects.

Cost element	Cost estimates	Source
Land acquisition	$750/acre for BLM Wood River wetland in 1992-1994	A. Hamilton pers. comm. (2012)
	$700/acre for Agency Lake Ranch in 1998	David Evans and Associates, Inc.)2005)
	$5,000/acre rough estimate for lands around Keno Reservoir	Deas (2011)
Permitting and water rights (if needed)	Variable	--
Site investigation and engineering design	$7,500-$ 22,500 (assuming 15% of capital costs as a rule of thumb)	Lyon et al. (2009)
Construction: - Earthwork (e.g., berm construction, excavation) - Hydrologic control and diversion structures (e.g., weirs, diversion boxes, pumps) - Planting	$50,000-$150,000 per acre	Lyon et al. (2009)
	$5,600 per acre (for pilot scale)	Mahugh et al. (2009)
	$230-$2,500 per acre per year (average $1000). Note: unclear whether engineering/design is included in cost.	SFWMD (2004) as cited in CH2MHill (in prep)
Perimeter fencing (if needed)	$5,600 per acre (for pilot scale)	Mahugh et al. (2009)
Pre-treatment components (if needed)	Variable	--
Recreational/educational/research facilities (if needed)	Variable	--
Operation and maintenance	Variable, but median value for surface flow treatment wetlands = $800 per acre per year	Kadlec and Wallace (2008)
	$60-$750 per acre per year (average $260)	SFWMD (2004) as cited in CH2MHill (*in prep*)

As another relevant Klamath Basin example, approximate costs for the Wood River Wetland Restoration project, located adjacent to Upper Klamath Lake (see Section 4.1), from 1992 through 2006 are as follows (A. Hamilton, pers. comm., 2012):

- Land acquisition $2,400,000 (1992–1994) (approximately $750/acre for 3,200 acres)

- Phase 1: pump station, new dike, meandered channels (1996–1997) $680,000
- Phase 2: middle levee and settling ponds (1997) $540,000
- Phase 3: Wood River stream channel restoration (1997–2000) $1,580,000

Nutrient removal estimates are difficult to estimate for wetland restoration projects due to site specific variability in conditions such as hydrology (seasonal flow patterns), hydraulics (how water distributes/moves though the wetland), geology, and vegetation. One wetland restoration project in the Klamath Basin where nutrient reductions can be roughly quantified is the re-flooding of the Williamson River Delta. Re-flooding the 3,200 acres on the north/west side of the Williamson River Delta (Tulana Farms) initially released 2 MT of phosphorus, but that is only a small fraction of the 21-25 MT of phosphorus loaded to the lake from the Delta annually prior to flooding (Wong et al. 2011), indicating that the re-flooding of wetlands fringing Upper Klamath Lake resulted in reduced loading to the lake even in the first year (i.e., 23 MT - 2 MT = 21 MT). The 21 MT of "avoided" phosphorus loading is not actually the amount of phosphorus removed or sequestered by the wetlands; rather, it is a comparison of phosphorus loading given wetland land use relative to the former land use (agriculture). The diked/farmed former wetlands around the lake were, and in some cases still are, the "hot spots" of phosphorus export to the lake (Snyder and Morace 1997). Nutrient concentrations in the Williamson River Delta wetlands have declined since the initial year, indicating the nutrient export from the wetlands is continuing to slow (Wong and Hendrickson 2011).

Estimated unit removal costs for nitrogen and phosphorus in restored wetlands in the Upper Klamath Basin based on costs for the Wood River wetlands and the Williamson River Delta are presented in Table C-3.

Table C-3. Cost estimates for restored wetlands in the Upper Klamath Basin based upon general size, land acquisition, and construction costs for the Wood River and Williamson River Delta projects adjacent to Agency Lake/Upper Klamath Lake.

Cost consideration	Cost scenario[1]	
	Lower	**Higher**
Wetland area (acres)	3,200	3,200
Project life (years)	50	50
Capital costs (construction and land acquisition)[2]		
Per unit area ($/acre)	$4,700	$8,600
Sub-total Capital ($)	$15,040,000	$27,520,000
Operation and Maintenance (O&M) costs[3]		
Per unit area ($/acre/yr)	$100	$800
Annual ($/yr)	$320,000	$2,560,000
Sub-Total O&M ($)	$16,000,000	$128,000,000
Total cost for Project life		
Capital + O&M ($)	$31,040,000	$155,520,000

Cost consideration	Cost scenario[1]	
	Lower	Higher
Unit removal costs		
TN removal rate (mg N/m^2/d)	100	50
TP removal rate (g P/m^2/yr)	--	0.5
TP "avoided" loading rate (g P/m^2/yr) [4]	1.6	0.5
Total TN removed for project life (kg)	23,634,480	11,817,240
Total TP removed for project life (kg)	1,048,982	323,760
TN unit removal cost ($/kg)	$1	$13
TP unit removal cost ($/kg)	$30	$480

[1] Lower estimate is a "best case scenario" that includes lower capital and/or O&M costs coupled with higher removal rates. Higher estimate is a "worst case scenario" that includes higher capital and/or O&M costs coupled with lower removal rates.

[2] Per-acre land costs assumed to be $3,000 (i.e., mid-way between $700-750 cost for 1990s Wood River land acquisitions and the $5,000 current small-parcel estimate from Deas (2011). Lower estimate construction costs assumed to be $1,700/acre based on 2000-2010 construction costs for entire Williamson River Delta project (i.e., $10M/5,800 wetland acres = $1,700/acre) http://www.fws.gov/klamathfallsfwo/suckers/sucker_pub/oct08posters/RestoringWetlands-SternHendrickson.pdf. Higher estimate construction costs assumed to be $5,600/acre from Mahugh et al. (2009). See also Table C-2.

[3] Lower O&M costs are unknown, but assumed to be less than the average value from SFWMD (2004). Higher O&M costs are median value from Kadlec and Wallace (2009) (see also Table C-2).

[4] Re-flooding north/west side of the Williamson River Delta (Tulana Farms) initially released 2 MT of phosphorus (Wong et al. 2011), whereas previous annual phosphorus export to lake was 21-25 MT (Synder and Morace 1997), indicating 21 MT reduced loading to the lake in the first year. Release of phosphorus should be lower than 2 MT in subsequent years. 21 MT/yr/3,200 acres=1.6 g P/m^2/yr.

Note: The units in this table may be mixed (i.e., English and metric); however, the units are presented as typically encountered in engineering, planning, or regulatory areas, or the scientific literature, whichever is most applicable. Appendix D presents common unit conversions.

3.2 Treatment Wetlands

3.2.1 Treatment cost estimates

Kadlec and Wallace (2008) provide an algorithm to determine capital costs for free water surface constructed wetlands based on empirical data (Table C-4).

Table C-4. Capital cost for free water surface constructed wetlands as a function of wetland size, based on equation from Kadlec and Wallace (2008) derived from 84 wetlands in the United States (r^2 value of 0.79). Total capital cost = 194 A $^{0.690}$, where A = wetland area in hectares.

Wetland area (acres)	Wetland area (ha)	Total capital cost ($)	Capital cost per unit area ($/acre)
10	4	$509,002	$50,900
50	20	$1,545,286	$30,906
100	40	$2,492,984	$24,930
200	81	$4,021,890	$20,109
500	202	$7,568,483	$15,137
1,000	405	$12,210,109	$12,210
5,000	2,023	$37,068,827	$7,414
10,000	4,047	$59,802,527	$5,980
20,000	8,094	$96,478,431	$4,824

Wetland area (acres)	Wetland area (ha)	Total capital cost ($)	Capital cost per unit area ($/acre)
50,000	20,235	$181,555,261	$3,631
100,000	40,469	$292,900,111	$2,929

To estimate treatment costs per unit of phosphorus removed, we applied cost information presented in Table C-2 and Table C-4 to wetland sizing calculations developed for the Klamath Straits Drain (i.e., a treatment wetland 1,633 acres in size, treating a 70-cfs flow, for approximately 90% TN and 60% TP removal) (Lytle 2000) (see Table C-5). Two cost scenarios were evaluated assuming two different values for O&M costs: in the lower cost scenario, per-acre capital (construction and land acquisition) costs are based upon the scale-dependent regression equation in Table C-4 and O&M costs apply the median value from Kadlec and Wallace (2009). In the higher cost scenario, per-acre capital costs are also from Table C-4, but O&M costs are the average value presented in SFWMD (2004).

Table C-5. Cost estimates for treatment wetlands in the Upper Klamath Basin based upon size and nutrient removal assumptions in Lytle (2000) for a system treating water from the Klamath Straits Drain.

Cost consideration	Cost scenario[1]	
	Lower	Higher
Wetland area (acres)	1,600	1,600
Project life (years)	50	50
Capital costs (construction and land acquisition)[2]		
Per unit area ($/acre)	$10,488	$10,488
Sub-total Capital ($)	$17,126,949	$17,126,949
Operation and Maintenance (O&M) costs[3]		
Per unit area ($/acre/yr)	$260	$800
Annual ($/yr)	$424,580	$1,306,400
Sub-Total O&M ($)	$21,229,000	$65,320,000
Total cost for Project life		
Capital + O&M ($)	$38,355,949	$82,446,949

Cost consideration	Cost scenario[1]	
	Lower	Higher
Unit removal costs		
Average flow (cfs) [4]	70	70
Days operating per year	365	365
Mean inflow TN concentration (mg/L) [4]	1.35	1.35
Mean outflow TN concentration (mg/L) [4]	0.14	0.81
Mean inflow TP concentration (mg/L) [4]	0.41	0.41
Mean outflow TP concentration (mg/L) [4]	0.16	0.25
Annual TN load removed (MT/yr)	76	34
Annual TP load removed (MT/yr)	16	10
Total TN load removed (kg)	3,800,000	1,700,000
Total TP load removed (kg)	800,000	500,000
TN unit removal cost ($/kg)	$10	$48
TP unit removal cost ($/kg)	$47	$162

[1] Lower estimate is a "best case scenario" that includes lower capital and/or O&M costs coupled with higher removal rates. Higher estimate is a "worst case scenario" that includes higher capital and/or O&M costs coupled with lower removal rates.
[2] Per-acre capital costs based upon scale-dependent regression equation (Kadlec and Wallace 2009).
[3] Lower O&M costs are the average value from SFWMD (2004). Higher O&M costs are median value from Kadlec and Wallace (2009) (see also Table C-2).
[4] From Lytle (2000) for TP and TKN + NH3-N in KSD at Stateline Highway.
Note: The units in this table may be mixed (i.e., English and metric); however, the units are presented as typically encountered in engineering, planning, or regulatory areas, or the scientific literature, whichever is most applicable. Appendix D presents common unit conversions.

3.3 Decentralized (Diffuse) Source Treatment Systems

3.3.1 Treatment cost estimates

Cost assumptions for building decentralized (diffuse) source treatment systems are listed below.

- Site survey—essential for accurately determining design water levels ($1,000–1,500)
- Water control and diversion structures built into berms
  - Diversion box—$1,500–$2,500
  - Level control—$2,000–$3,000
- Earthwork—generally to be avoided, since the site is likely to be wet and difficult to work with typical earth moving equipment. If earthwork is necessary, the following cost estimates generally apply:
  - Excavation—$2.50 per cubic yard
  - Berms (two -four feet high, 3:1 slopes, 3 ft top)—$5 per cubic yard.
- Planting ($5,000–$10,000 per acre)
  - Primary treatment species (e.g., *Typha spp., Scirpus spp., and Sparganium*)
  - Secondary food and habitat species (e.g., *Nymphae spp.*)
- Exclusion fencing—$1.23 per linear foot for three strand barbed wire around the perimeter of the system. Building directly adjacent to a canal can eliminate fencing on the canal side.

Using the above assumptions, cost estimates for a 100-acre parcel in the Upper Klamath Basin with minimal earthwork and full perimeter fencing for small (0.9 acre) in-pasture wetland are presented in Table C-6.

Table C-6. Cost estimates for decentralized (diffuse) source treatment systems in the Upper Klamath Basin.

Cost consideration	Cost scenario	
	Lower	**Higher**
Wetland area (acres)	0.9	0.9
Project life (yrs)	50	50
Capital costs (assumes no land acquisition needed)		
Site survey	$1,500	$1,500
Diversion box	$2,500	$2,500
Level control	$3,000	$3,000
Pumps	$—	$—
Earthwork	$250	$250
Planting	$9,700	$9,700
Exclusion fencing @ $1.25 per foot	$680	$680
Sub-total capital costs	$17,630	$17,630
Operation & maintenance (O&M) costs		
Per unit area ($/acre/yr)	$260	$260
Annual ($/yr)	$234	$234
Sub-total O&M ($)	$11,700	$11,700
Total cost for project life		
Capital + O&M ($)	$29,330	$29,330
Unit removal costs		
TN removal rate (mg N/m^2/d)	100	50
TP removal rate (g P/m^2/yr)	1	0.5
Total TN removed for project life (kg)	6647	3324
Total TP removed for project life (kg)	182	91
TN unit removal cost ($/kg)	$4	$9
TP unit removal cost ($/kg)	$161	$322

Note: The units in this table may be mixed (i.e., English and metric); however, the units are presented as typically encountered in engineering, planning, or regulatory areas, or the scientific literature, whichever is most applicable. Appendix D presents common unit conversions.

3.4 Algae/Biomass Removal from the Water Column via Filtration

3.4.1 Basic design elements

Design elements for land-based filtration units:

- Prioritization of gravity-fed canal flow options

- Determination of diversion pumping capacity for current rotation screen methods
- Identification of existing water control structures
- Determination of retrofitting feasibility of existing water control structures

Design elements for land-based flotation/separation units:
- Evaluation of flocculants and chemical characteristics (cost & environmental impact)
- Identification of water control structures with substantial infrastructure to house floatation cells
- Calculation of pumping capacity and cell treatment volumes for lake scale filtration (up scaling)
- Determination of biomass dewatering methodologies suitable for algal slurries

Design elements for *in situ* barge filtration units:
- Identification of fabricators/ engineers for barge production
- Determination of efficient off-loading methods (dock vs. on-lake off-loading procedures)
- Selection of dockage / storage options for harvest barges
- Identification of potential indirect effect on private industry

3.4.2 Treatment cost estimates

Cost considerations for algae/biomass removal from the water column via filtration are listed below.

Land-based filtration:
- Retrofitting of existing structures
- Filtration unit costs and maintenance estimates per unit volume
- Disposal or secondary use (transport, storage)

Land-based floatation/separation:
- Land costs
- Construction and operation costs
- Retrofitting of existing structures
- Chemical flocculation costs

Using information gathered from private enterprise harvesting algae on Upper Klamath Lake, cost estimates for *in situ* barge-based removal of algal biomass are provided in Table C-7. These estimates are expected to be viable for any of the possible treatment locations.

Table C-7. Cost estimates for *in situ* barge-based removal of algal biomass.

Cost consideration	Cost estimate
Mole C per mole biomass [1]	120
Mole N per mole biomass	16
Mole P per mole biomass	1
Barge filtration capacity (lbs wet weight/day)	200,000
Barge filtration capacity (lbs dry weight/day)	2,000
Project life (yrs)	10

Cost consideration	Cost estimate
Capital costs	
Filtration barge ($)	$250,000
Off-load vessel (tender) ($)	$50,000
Sub-total capital costs ($)	$300,000
Operation & maintenance (O&M) costs [2]	
Fuel for barge and tender ($/day)	$400
Maintenance for barge and tender ($/day)	$125
Personnel ($/day)	$400
Annual ($/yr)	$337,625
Sub-total O&M costs ($)	$3,376,250
Total cost for project life	
Capital + O&M costs ($)	$3,676,250
Unit removal costs	
Biomass removed (lbs/day)	2,000
Carbon removed (lbs/day)	1,699
Nitrogen removed (lbs/day)	264
Phosphorus removed (lbs/day)	37
Total TN removed for project life (kg)	437,983
Total TP removed for project life (kg)	60,614
TN unit removal cost ($/kg)	$8
TP unit removal cost ($/kg)	$61
Transportation and disposal costs assuming landfill	
75% dewatered material (lbs/day)	50,000
Landfill disposal ($/lb)	$0.010
Transportation ($/lb)	$0.008
Total landfill and transportation costs ($/day)	$875
TN unit transportation and disposal cost ($/kg)	$7
TP unit transportation and disposal cost ($/kg)	$53

[1] Adjusted Redfield ratio (120:16:1) used 1) to provide conservative pricing; 2) to account for carbon in dead/dying algal biomass that has already lost some of the nitrogen and phosphorus due to senescence, but is captured in the harvest process, and; 3) Redfield ratio was originally developed for marine plankton and there is a wide range of ratios available in the literature for different algae. In the Upper Klamath Lake system, the ratio may drop below 100 based upon the excess nitrogen and phosphorus availability.

[2] Assumes maintenance of barge and tender = 5% total cost/year. Assumes 100 days of *in situ* filtration per year.

[3] Assumes natural dewatering (air drying) is 75% efficient.

Note: The units in this table may be mixed (i.e., English and metric); however, the units are presented as typically encountered in engineering, planning, or regulatory areas, or the scientific literature, whichever is most applicable. Appendix D presents common unit conversions.

3.5 Sediment Removal (Dredging)

3.5.1 Similar applications

3.5.1.1 Burnaby Lake

Burnaby Lake, a 398-acre glacial lake in Burnaby, British Columbia, recently underwent a dredging project to remove approximately 235,431 cubic yards of accumulated sediments that had reduced the overall depth of the lake (Burnaby Parks Recreation and Cultural Services 2011, Chow 2011). The process began with a pilot dredging operation in 1999 that removed 2,500 cubic meters of sediment and confirmed the feasibility of larger scale lake dredging (City of Burnaby Environment and Waste Management Committee 2000). Between September 2009 and May 2011 the $20.5-million Burnaby Lake Rejuvenation Project dredged the lake sediments using the Model 5012 HP Versi-Dredge pump dredge (Tervita n.d.; IMS n.d.). The following process was utilized: (1) isolating a series of dredge zones, (2) relocating sensitive aquatic species (e.g., turtles), (3) dredging (using double turbidity barriers to contain mobile sediments and separate species from dredging operations) , (4) transporting the "dredgeate" to the City of Burnaby's clean fill depot (5) separating water from the sediments by centrifuge, (6) transporting and reusing the sediment as subgrade for a nearby future sports field, (7) treating the liquid stream and (8) returning the treated liquids to the lake (ACEC n.d.).

3.5.1.2 Lake Okeechobee

In 2002, a pilot dredging project was conducted for Lake Okeechobee, a 467,200-acre lake in south-central Florida, to determine the feasibility of removing approximately 200 million cubic meters of nutrient-laden sediments (EA Engineering Science and Technology 2002). A SEDCUT dredge head with a 6-inch mouth opening was used to successfully remove dredge slurry along discrete lanes with minimal resuspension; the technology was found to be suitable for larger-scale application by making modifications such as increasing the head width and adding more hydraulic pumps. Approximately 6,000 cubic yards of dredge material was relocated to a confined disposal facility (CDF) along the shore of the lake where it was allowed to settle, then the liquid fraction was skimmed from the top and treated to remove phosphorus. Full-scale application to remove the desired quantity of sediment would involve dredging, disposal and treatment over a 30-year time period at an estimated cost of $27 million per year ($813 million total).

3.5.1.3 Lake Trafford

Lake Trafford is a shallow, 1,600-acre lake in Immokalee, Florida that was dredged to remove muck that had accumulated as a result of high nutrient inputs and decomposing exotic plant material. The Lake Trafford Restoration Project was authorized by Congress in 1996 and was implemented by the South Florida Water Management District from 2006–2010. Dredging was implemented in three phases in 2006, 2007 and 2010 using a cutter suction 20-inch dredge L.W. (GLDDC n.d.); muck was initially removed from the central part of the lake (to a depth of 6 feet), followed by the littoral zone around the lake's edges (to a depth of 2 feet). A total of 6.3 million cubic yards of sediment were removed and pumped to a CDF one mile north of the lake. The total cost of the project was approximately $21 million (SFWMD 2011b).

3.5.1.4 Vancouver Lake

In 1983 dredging of Vancouver Lake, a 2,600-acre lake in Vancouver, WA, was completed as part of the $17-million Vancouver Lake Restoration Project to improve the lake's water quality

(Gorini 1987). A total of 9 million cubic yards of dredge material were removed using a modified, electrified 30-inch cutter suction dredge. The dredge material was used to construct an island at the west end of the lake, modify the shoreline of the lake and fill 600 acres of uplands. Unfortunately, operations and maintenance (O&M) activities were not executed as planned and the lake has since silted in a second time, necessitating additional restoration activities sooner than expected (USACE 2007).

3.5.2 Treatment cost estimates

Hydraulic dredging costs can vary greatly, from $5 to $15 per cubic yard (IEPA and NIPC 1998), which includes equipment operation and maintenance. Previous studies estimated hydraulic dredging cost per cubic yard at: $1.80–$2.50 (Lake Hancock; Madrid Engineering Group, Inc. 2005), $3.02 (Lake Okeechobee; EA Engineering Science and Technology 2002), $4–7 (Kohlman Lake; BARR 2007; higher than normal because of limited access to public right-of-way) and $4–6 (Fountain Lake, BARR 2009). Mechanical dredging can be more expensive, ranging from $8–30 per cubic yard (including disposal) (IEPA and NIPC 1998). The Kohlman Lake Dredging Feasibility Study estimated mechanical dredging costs at $5–11 per cubic yard (BARR 2007).

3.5.2.1 Upper Klamath Lake

Given readily available information, the following are example calculations for potential dredging costs for the Upper Klamath Lake system.

Lake Area—Area estimates vary depending on whether bordering wetlands and/or Agency Lake are included (for the purposes of this analysis, Agency Lake is included). In a prior investigation, USBR estimated the required dredging area for the Upper Klamath Lake system at 75,000 acres (J. Hicks, USBR, pers. comm. with M. Singer, Stillwater Sciences, June 6, 2012). However, a more recent study estimated the lake area at approximately 232 km^2 (57,328 acres), including Agency Lake (Simon et al. 2009).

Dredging depth—In a prior study, USBR estimated the required dredging depth for Upper Klamath Lake to be 3 feet (J. Hicks, pers. comm., June 6, 2012). However, more recent estimates indicated that phosphorus concentrations were highest in the upper 10 cm of sediments (see also Section 4.5.5 of the main report) (Simon et al. 2009).

Dredging volume—The more recent dataset is used to provide cost estimates of required dredging volume: 57,328 acres x 10 cm (= 0.33 feet) = 18,918 acre-feet = 30,521,040 cubic yards

Total costs—Assuming $5–15 per cubic yard, the total cost of dredging the entire Upper Klamath Lake/Agency Lake system to a depth of 10 cm is estimated to range from $153 million to $458 million.

Unit removal cost for phosphorus—Based on the median phosphorus concentration value reported in Simon et al. (2009), the calculated amount of phosphorus in the top 10 cm of sediment is approximately 1,392 metric tons (see also Section 4.5.5). For a range of $153 million to $458 million, this equates to $110 to $329 per kg TP. Additional information on the areal extent of elevated phosphorus concentrations in the regions mentioned above (northern, Howard Bay and southernmost areas) would allow further refinement to these estimates, and given the greater mass

of phosphorus removed per volume of sediment, would be expected to decrease the removal cost per metric ton of phosphorus in lake hotspots.

3.5.2.2 Keno Reservoir

The following information is available for Keno Reservoir:
Reservoir Area—approximately 2,300 acres (Klinkenberg 2001)

Dredging Volume—approximately 144,000 cubic yards (including sand and gravel), with approximately 98,000 cubic yards (silt and clay).

Total Costs—Assuming $5–15 per cubic yard, the total cost of dredging 144,000 cubic yards in Keno Reservoir ranges from $490,000 to $1.5 million. Assuming the same TP content for Keno Reservoir sediment deposits as those of Upper Klamath Lake, the unit removal costs would be the same (i.e., $110 to $329 per kg TP)

Appendix D

Common conversion factors

Common Conversion Factors

Multiply	By	To obtain
	Length	
inch (in.)	2.54	centimeter (cm)
inch (in.)	25.4	millimeter (mm)
foot (ft)	0.3048	meter (m)
mile (mi)	1.609	kilometer (km)
yard (yd)	0.9144	meter (m)
	Area	
Acre	4,047	square meter (m^2)
Acre	0.4047	hectare (ha)
Acre	0.004047	square kilometer (km^2)
square foot (ft^2)	0.09290	square meter (m^2)
square inch (in^2)	6.452	square centimeter (cm^2)
square mile (mi^2)	259.0	hectare (ha)
square mile (mi^2)	2.590	square kilometer (km^2)
	Volume	
gallon (gal)	3.785	liter (L)
gallon (gal)	0.003785	cubic meter (m^3)
million gallons (Mgal)	3,785	cubic meter (m^3)
cubic inch (in^3)	16.39	cubic centimeter (cm^3)
cubic inch (in^3)	0.01639	cubic decimeter (dm^3)
cubic inch (in^3)	0.01639	liter (L)
cubic foot (ft^3)	28.32	cubic decimeter (dm^3)
cubic foot (ft^3)	0.02832	cubic meter (m^3)
cubic yard (yd^3)	0.7646	cubic meter (m^3)
cubic mile (mi^3)	4.168	cubic kilometer (km^3)
acre-foot (acre-ft)	1,233	cubic meter (m^3)
acre-foot (acre-ft)	0.001233	cubic hectometer (hm^3)
	Flow rate	
acre-foot per day (acre-ft/d)	0.01427	cubic meter per second (m^3/s)
acre-foot per year (acre-ft/yr)	1,233	cubic meter per year (m^3/yr)
foot per second (ft/s)	0.3048	meter per second (m/s)
cubic foot per second (ft^3/s)	0.02832	cubic meter per second (m^3/s)
cubic foot per day (ft^3/d)	0.02832	cubic meter per day (m^3/d)
gallon per minute (gal/min)	0.06309	liter per second (L/s)
gallon per day (gal/d)	0.003785	cubic meter per day (m^3/d)
million gallons per day (Mgal/d)	0.04381	cubic meter per second (m^3/s)
inch per hour (in/h)	0.0254	meter per hour (m/h)
	Mass	
ounce (oz)	28.35	gram (g)
Pound (lb)	0.4536	kilogram (kg)
ton, short (2,000 lb)	0.9072	megagram (Mg)
ton, long (2,240 lb)	1.016	megagram (Mg)
ton per day (ton/d)	0.9072	metric ton per day
ton per year (ton/yr)	0.9072	metric ton per year

Temperature in degrees Celsius (°C) may be converted to degrees Fahrenheit (°F) as follows:
$°F=(1.8\times°C)+32$

Temperature in degrees Fahrenheit (°F) may be converted to degrees Celsius (°C) as follows:
$°C=(°F-32)/1.8$