Approaches to Water Quality Treatment by Wetlands in the Upper Klamath Basin

Prepared for

PacifiCorp Energy
A DIVISION OF PACIFICORP

PACIFICORP
A MIDAMERICAN ENERGY HOLDINGS COMPANY

August 2012

CH2MHILL®

ES021512103604PDX

CH2MHILL®

ES021512103604PDX

Approaches to Water Quality Treatment by Wetlands in the Upper Klamath Basin

Prepared for

PacifiCorp Energy

A Division of PacifiCorp

in consultation with the **Interim Measures Implementation Committee**
for the Klamath Hydroelectric Settlement Agreement

August 2012

CH2MHILL®

Recommended Citation:

CH2M HILL. 2012. Approaches to Water Quality Treatment by Wetlands in the Upper Klamath Basin. Prepared for PacifiCorp Energy, Portland, OR. Prepared by CH2M HILL, Inc., Portland, OR. August 2012.

Contents

List of Figures

List of Figures (continued)

Page

List of Tables

List of Appendices

(This page intentionally blank)

Acronyms and Abbreviations

ac	acre
ac-ft	acre-feet
BAT	best available technology
BMP	best management practice
BOD	biological oxygen demand
C*	wetland equilibrium background concentration (mg/L)
cfs	cubic feet per second
cm	centimeters
CO_2	carbon dioxide
COD	chemical oxygen demand
CY	cubic yards
DIP	dissolved inorganic phosphorus
DO	dissolved oxygen
DON	dissolved organic nitrogen
DOP	dissolved organic phosphorus
DP	dissolved phosphorus
DRP	dissolved reactive phosphorus
EPA	U.S. Environmental Protection Agency
ESA	Endangered Species Act
ET	evapotranspiration
$FeCl_3$	ferric chloride
ft	feet
ft^3	cubic feet
FWS	free water surface
g	grams
GIS	Geographical Information System
gpd	gallons per day
gpm	gallons per minute
ha	hectares
HDPE	high-density polyethylene
HLR	hydraulic loading rate
HWTT	Hybrid Wetland Treatment Technology
k	first-order, area-based rate constant (m/yr)
kg	kilograms
L	liter
lbs/day	pounds per day
lbs/yr	pounds per year
LOW	Lake Okeechobee Watershed
mg	milligrams
mg/kg	milligram per kilogram
mg/L	milligram per liter (equivalent to parts per million, or ppm)
mgd	million gallons per day
mi	mile

mL	milliliter
mt	Metric Tons
N	nitrogen
$N0_2$-N	nitrite-nitrogen
$N0_3$-N	nitrate-nitrogen
N_20	nitrous oxide
NH_3	ammonia
NH_3-N	ammonia-nitrogen
NH_4^+	ammonium
NH_4-N	ammonium-nitrogen
NMFS	National Marine Fisheries Service
NO_2	nitrite
NO_3	nitrate
NPDES	National Pollutant Discharge Elimination System
NRCS	Natural Resources Conservation Service
NuRF	Nutrient Reduction Facility (located in Lake County, Florida)
Org-N	organic nitrogen
ortho-P	ortho-phosphorus
P	phosphorus
$P0_4$-P	phosphate phosphorus
pH	potential of hydrogen (measurement of acidity of a water sample)
PO_4	orthophosphate
PP	particulate organic phosphorus
ppb	parts per billion
ppt	parts per thousand
psi	pounds per square inch
SAV	submerged aquatic vegetation
SF	surface flow
SFWMD	South Florida Water Management District
SRP	soluble reactive phosphorus
SSF	subsurface flow
STA	Stormwater Treatment Area
TDS	total dissolved solids
TIN	total inorganic nitrogen (ammonia + nitrite + nitrate)
TKN	total Kjeldahl nitrogen
TMDL	total maximum daily load
TN	total nitrogen
TP	total phosphorus
TSS	total suspended solids
USACE	United States Army Corps of Engineers
USFWS	United States Fish and Wildlife Service
USGS	United States Geological Survey
VSS	volatile suspended solids
µg/g	microgram per gram
µg/L	microgram per liter (equivalent to parts per billion, or ppb)

Executive Summary

This report contains the findings of the constructed treatment wetlands evaluation as conducted pursuant to a study plan developed by PacifiCorp and the Interim Measures Implementation Committee related to Interim Measure 11 of the Klamath Hydroelectric Settlement Agreement (KHSA). The purpose of this study is to provide information for use in considering and planning approaches to possible development of engineered treatment wetland systems in the Upper Klamath basin. As discussed further in the report, treatment wetlands can be designed in a variety of configurations and with various augmentation technologies. The planning information included in this report specifically includes: (1) an overview of treatment wetlands and their effectiveness; (2) estimates of treatment wetland sizes and effectiveness at basin-scale to achieve specific water quality improvement objectives; (3) descriptions of potential supplemental technologies to enhance treatment wetland effectiveness; and (4) descriptions of the status of relevant treatment wetland case studies elsewhere.

The approximate magnitude of wetland acres that would be needed to achieve nutrient load reductions in the Upper Klamath basin were estimated using a first-order area-based treatment wetland model. These estimates are primarily useful as conservative “side boards” for further planning purposes, since it is not considered realistic that reductions of nutrient loads in the Klamath River would be accomplished only by restoration and construction of wetlands. Nonetheless, these calculations suggest that wetlands treatment offers a potentially important tool in a broader overall strategy for reducing nutrient loads that would also necessarily involve other technologies.

This report presents detailed information on several potential supplemental technologies to enhance treatment by wetlands (Chapter 5), including constructed emergent vegetation surface flow wetland systems, submerged aquatic vegetation (SAV) systems, periphyton-based treatment systems, various supplemental chemical treatment approaches, and systems combining chemical, settling and solids separation, and filtration. Each of these supplemental technologies are described, including their relative effectiveness, advantages and disadvantages, costs, and potential for application to potential application in the Upper Klamath basin.

A number of treatment wetland case studies from elsewhere in the region and across the country are described that serve as important potential analogies to the Klamath Basin situation. As analogies, the implementation experiences and level-of-effectiveness of these other projects provide insights for use in evaluation and planning for potential treatment wetlands in the Klamath Basin. Detailed summaries of each example are provided, including the implementation experiences and level-of-effectiveness of these other projects, as well as the situational similarities relative to potential application to the Upper Klamath basin.

The report’s final chapter includes other considerations and recommendations for potential next steps in the planning and implementation of treatment wetlands in the Upper Klamath basin. An important first step is to define the specific goals for treatment by wetlands in the Upper Klamath basin. These goals will drive most, if not all, aspects of wetland planning, design, construction, operation, and maintenance. Several important factors and constraints are then discussed that should be anticipated and addressed in future planning, including water use and water rights, land availability and suitability, regulatory permitting and approvals, complex local conditions (e.g., climatology and hydrology), and dealing with uncertainty. Finally, several steps or action items are recommended in consideration of implementing a system to support decision-making going forward. These steps include: (1) development of a “toolbox” of potential measures; (2) ongoing refinement of research and data-gathering on potential measures; (3) subbasin planning, including potential development of basin-scale nutrient budgets, to determine where on the landscape measures would be implemented; (4) wetland and supplemental treatment pilot studies; and (5) expert and public involvement.

(This page intentionally blank)

CHAPTER 1

Introduction

1.1 Context for this Study

The Klamath River flows about 254 miles from Upper Klamath Lake in south-central Oregon into northern California where the river eventually empties into the Pacific Ocean near the town of Klamath (Figure 1). The Klamath Hydroelectric Project (Project) is located on the upper Klamath River below Upper Klamath Lake between River Mile (RM) 190 and 254. The Project is owned and operated by PacifiCorp. On February 18, 2010, the United States, the States of California and Oregon, PacifiCorp, Tribes, 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 Project dams (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 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 – titled Interim Measure 11: Interim Water Quality Improvements – emphasizes nutrient reduction projects in the basin to enhance water quality in the Klamath River, while also addressing water quality, algal and public health issues in Klamath Hydroelectric Project (Project) reservoirs and dissolved oxygen in J.C. Boyle reservoir. The purpose of Interim Measure 11 is to improve water quality in the Klamath River during the interim period leading up to potential dam removal.

Upper Klamath Lake and the Klamath River in Oregon do not meet certain water quality standards as specified by the State of Oregon, and the Klamath River in California does not meet certain standards as specified by the State of California. Both States have placed these waters on their lists of “impaired waters” as required under section 303(d) of the Clean Water Act. As a result of these listings, the States and the U.S. Environmental Protection Agency (EPA) are in the process of creating Total Maximum Daily Load (TMDL) plans to bring the water quality of these waters into compliance with standards. Impaired water quality is one of the factors that have been identified as primary limiting conditions for existing fish populations (NRC 2008).

Constructed treatment wetlands evaluation is one of four categories of studies specified for Interim Measure 11. Constructed treatment wetlands have been identified as a potentially viable means of improving water quality conditions in the upper Klamath River (ODEQ 2010, Regional Water Board 2010, Lyon et al. 2009, Deas and Vaughn 2006). Target water quality improvements include reduced total suspended solids, organic matter (biochemical oxygen demand), nutrients and local water temperature improvements.

In the context of this study, “treatment wetlands” include existing wetlands that are restored or enhanced, or newly-constructed wetlands, that are designed to utilize the natural functions of wetlands (such as, vegetation, soils, and their microbial populations) to improve water quality conditions in the upper Klamath River. An additional aspect of this study’s consideration of treatment wetlands includes the potential use of ancillary treatment technologies (such as, wetland inflow pretreatment systems) to further enhance effectiveness in improving water quality.

1.2 Purpose of the Study

This report contains the findings of the constructed treatment wetlands evaluation as specified in Interim Measure 11 of the KHSA. The purpose of this study is to provide information for use in considering and planning approaches to possible use of treatment wetlands in the Upper Klamath basin. This planning information is intended to inform stakeholders regarding the potential effectiveness of treatment wetlands in the Klamath

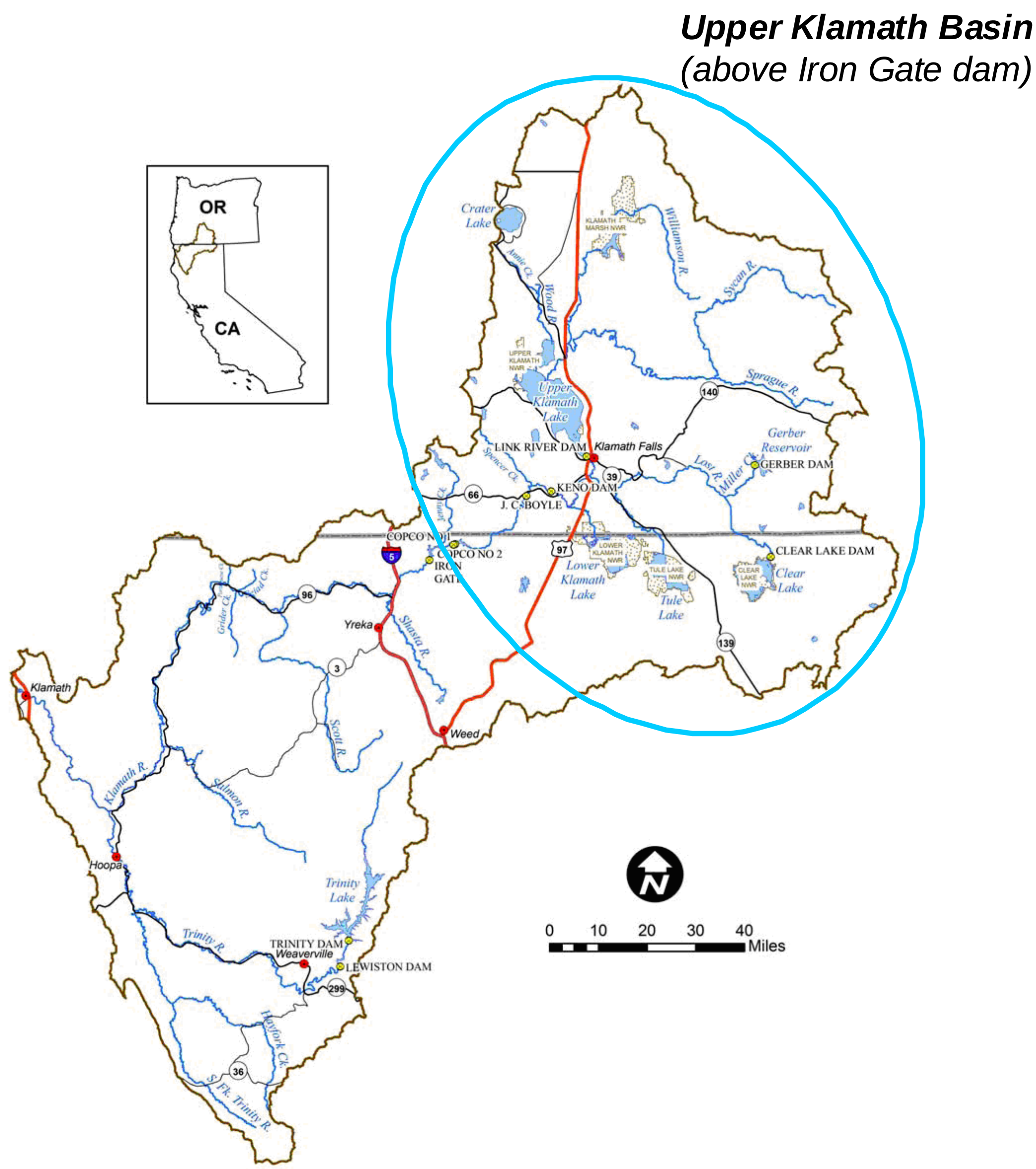

FIGURE 1
Map showing location of Upper Klamath Basin.

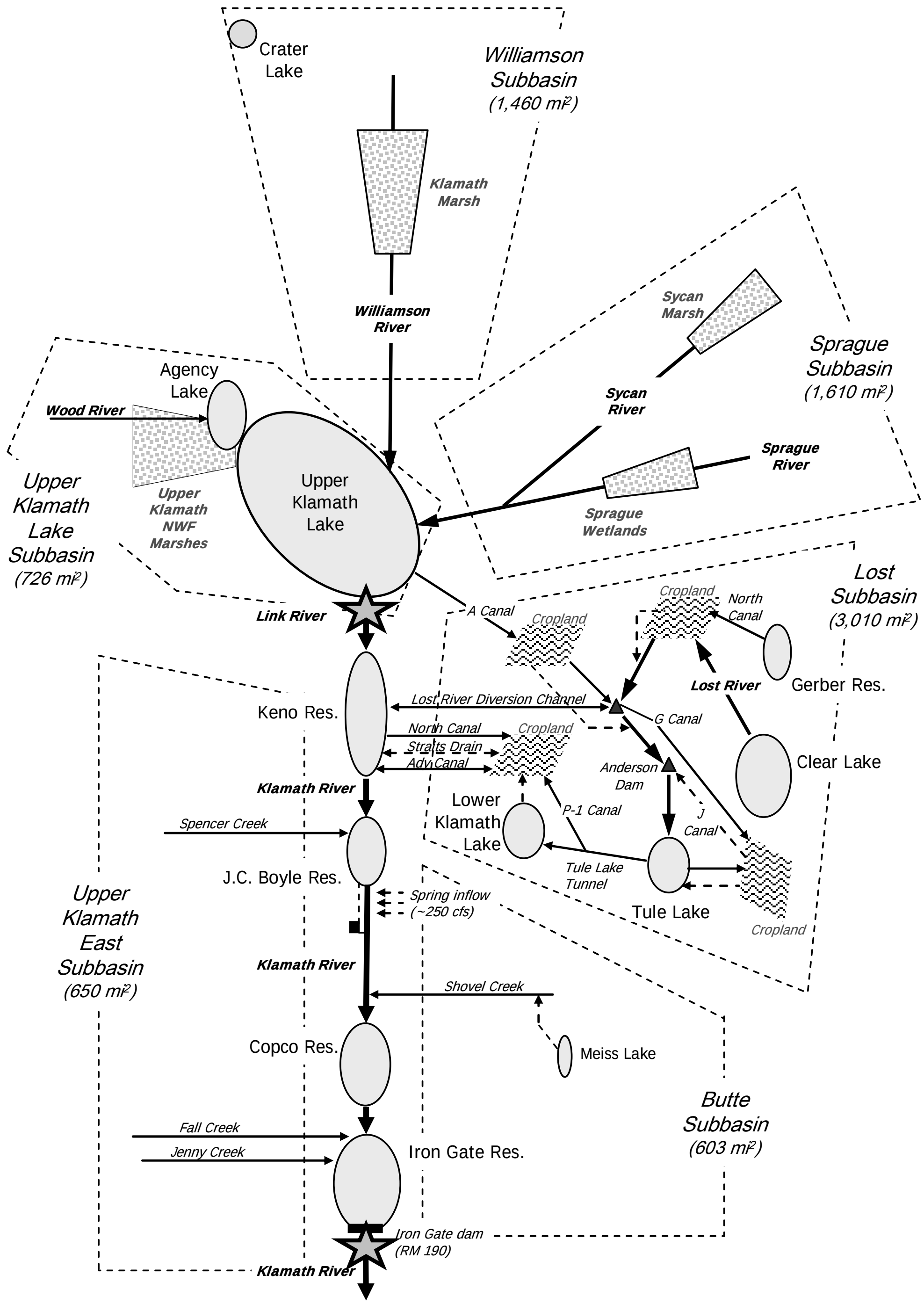

FIGURE 2

Schematic of subbasins that comprise the Upper Klamath Basin. Stars indicate Basin locations related to estimates of wetlands area (as discussed in Chapter 4) in the upstream basin areas contributing to these locations.

Basin, and to assist planning of further evaluation and potential pilot tests of constructed wetland pre-treatment techniques.

The planning information included in this report specifically includes:

1. an overview of treatment wetlands and their effectiveness;
2. estimates of treatment wetland sizes and effectiveness at basin-scale;
3. descriptions of potential supplemental technologies to enhance treatment wetland effectiveness; and
4. descriptions of the status of relevant treatment wetland case studies elsewhere.

As referred to in this study, the Upper Klamath basin is defined as the upper portion of the Klamath Basin above Iron Gate dam (Figure 1), and consists of the Williamson, Sprague, Lost, Upper Klamath Lake, Upper Klamath East, and Butte subbasins (Figure 2).

1.3 Terminology

Several different terms are used in this report to describe particular types of wetlands. In general, "wetlands" are defined in Federal regulations as "those areas that are inundated or saturated by surface or ground water at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions. Wetlands generally include swamps, marshes, bogs and similar areas." (40 CFR 230.3(t))

In general, "natural wetlands" as referred to in this report are the freshwater wetlands and marshes that occur naturally in the Upper Klamath basin (Figure 3). Akins (1970) refers to three types of naturally-occurring wetlands in the Upper Klamath basin: (1) "fresh meadows-wetland type" in river valley flats, such as occupied by the Klamath and Sycan Marshes; (2) "shallow-fresh marsh type" in regularly-flooded river valley flats, such as occurs in much of Lower Klamath Lake and the shallow margins of Upper Klamath Lake; and (3) "deep-fresh marsh type" that includes the standing water areas of Tule Lake, Lower Klamath Lake, and the bays of Upper Klamath Lake.

FIGURE 3
Example of Upper Klamath basin natural marsh-type wetlands. (Source: Oregon Wild).

For the most part, the subject matter in this report is focused on “treatment wetlands” or “constructed treatment wetlands”. In general, use of the terms “treatment wetlands” or “constructed treatment wetlands” in this report follows the definition of the Interstate Technology & Regulatory Council (ITRC) (2003) of “engineered systems, designed and constructed to utilize the natural functions of wetland vegetation, soils and their microbial populations to treat contaminants in surface water, groundwater or waste streams”. As discussed further in the report, treatment wetlands can be designed in a variety of configurations and with various augmentation technologies.

The two main types of constructed wetlands are: (1) surface flow (SF) systems (also known as free water surface [FWS] systems); and (2) subsurface flow (SSF) systems (also known as submerged aquatic vegetation [SAV] or vegetated submerged bed [VSB] systems). SF (or FWS) wetlands consist of shallow basins that have a soil bottom, emergent vegetation, and a water surface exposed to the atmosphere (Figure 4). This type of constructed wetland most closely resembles a natural wetland, with areas of open water, emergent plants, and floating vegetation. The wastewater destined for SF (or FWS) wetlands treatment may be released directly into the wetland system or may travel through a treatment train. As the wastewater passes through these systems the contaminants are reduced by various physical, chemical and biological mechanisms in the water column and the soil matrix.

SSF (or SAV/VSB) wetland systems consist of gravel, sand, or soil beds planted with wetland vegetation in which wastewater flows below ground through the soil/root zone beneath the surface of the porous soil, sand, or gravel substrate (Figure 5). The SSF systems can provide increased treatment efficiencies over SF systems because the substrate provides more surface area for bacterial biofilm growth over an SF wetland. As such, an advantage of SSF wetland systems over SF systems is a smaller land footprint, although SSF wetland systems can cost significantly more than equivalent SF wetlands to design and build.

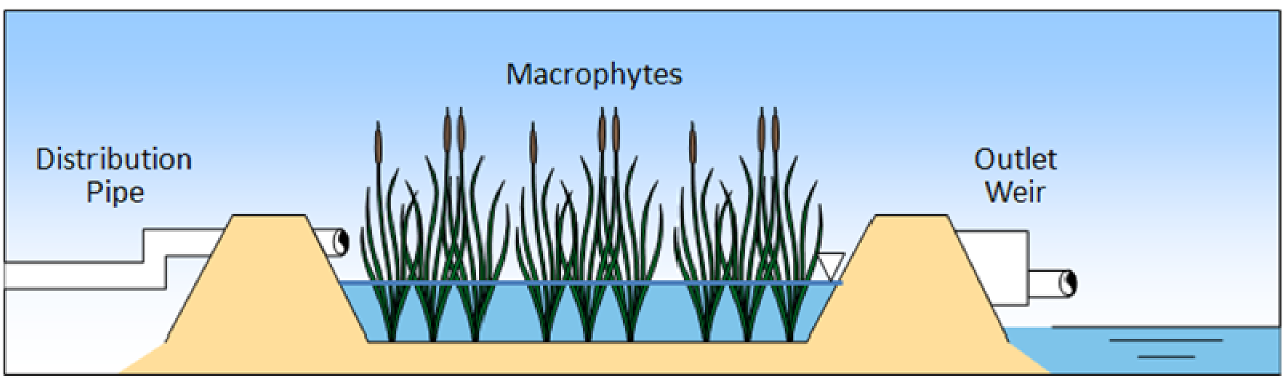

FIGURE 4
Diagram of a surface flow (SF) or free water surface (FWS) wetland system.

Another important term used in this report is “conventional treatment”. Conventional wastewater treatment consists of engineered facilities, such as sewage treatment plants, that use combinations of physical, chemical, and biological processes and operations to remove solids, organic matter and, sometimes, nutrients from wastewater. Although the subject of this report is treatment by wetlands, and not “conventional treatment” per se, some discussion of “conventional treatment” technologies occurs – both as a means to help explain treatment by wetlands, and because certain treatment wetland approaches may involve ancillary treatment using conventional techniques (such as, wetland inflow pretreatment systems).

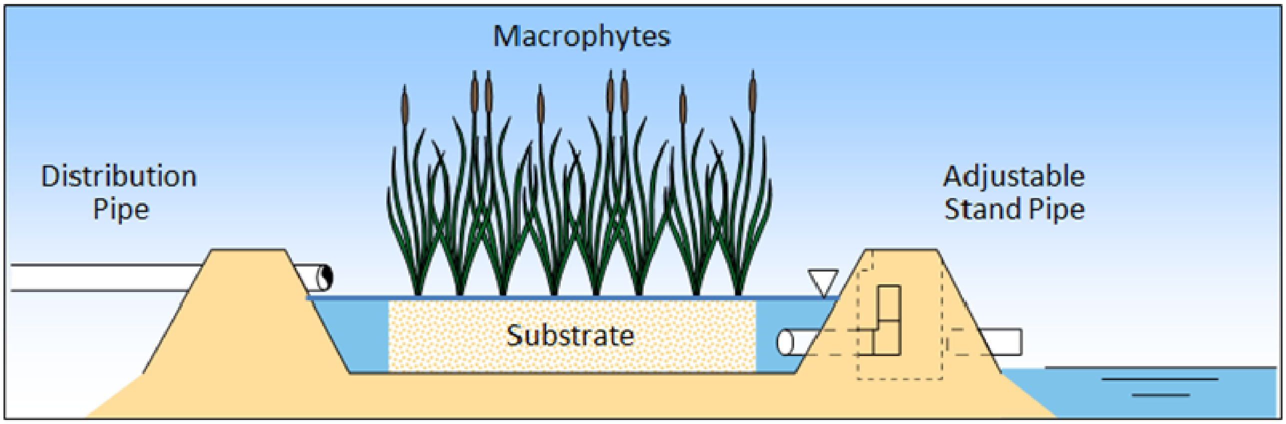

FIGURE 5
Diagram of a subsurface flow (SSF) wetland system.

General terms used to describe different degrees of conventional treatment, in order of increasing treatment level, are primary, secondary, and tertiary or advanced wastewater treatment. Primary treatment refers to removal of settleable organic and inorganic solids by sedimentation, and the removal of materials that will float (scum) by skimming. Secondary treatment refers to the further treatment of the effluent from primary treatment to remove the residual organics and suspended solids. Tertiary and/or advanced wastewater treatment is employed when specific wastewater constituents which cannot be removed by secondary treatment must be removed, such as nitrogen (N), phosphorus (P), additional suspended solids, refractory organics, heavy metals, and dissolved solids. Depending on the level of treatment, conventional wastewater treatment can include a variety of types of facilities, such as sedimentation tanks, sludge lagoons, clarifiers, digesters, reactors, sludge drying beds, in-process storage (or stabilization) ponds, and effluent application lands.

CHAPTER 2

Review of Previous Wetland Studies in the Upper Klamath Basin

2.1 Previous Studies on Wetland Conditions in the Basin

Akins (1970) evaluated the effects of land use and management on the wetlands of the Upper Klamath basin. Akins (1970) assumed the Upper Klamath basin as including the drainage area of the Klamath River upstream of Keno, Oregon, and the closed basins of the Lost River and Swan Lake Valleys of Oregon and a part of the Butte Valley of California – the total drainage area that Akins (1970) assumed comprised about 1,000,000 acres (about 1,562 square miles).

Akins (1970) estimated that wetlands of the Upper Klamath basin originally extended over approximately 350,000 acres in the basin prior to agricultural development (beginning in the early 1900s). These wetlands included: (1) "fresh meadows-wetland type" in river valley flats, such as occupied by the Klamath and Sycan Marshes; (2) "shallow-fresh marsh type" in regularly-flooded river valley flats, such as occurs in much of Lower Klamath Lake and the shallow margins of Upper Klamath Lake; and (3) "deep-fresh marsh type" that includes the standing water areas of Tule Lake, Lower Klamath Lake, and the bays of Upper Klamath Lake. Akins (1970) estimated that subsequent agricultural development resulted in the drainage and conversion (to farmland) of about 200,000 acres from development activities sponsored by the U.S. Bureau of Reclamation (Reclamation) and another 50,000 acres from other private farming enterprises. Akins (1970) estimated that approximately 10,000 acres of converted lands had been subsequently re-flooded to restore wetlands areas.

2.2 Previous Studies on Effects of Wetlands on Water Quality

Carlson (1993) estimated that about 73,000 acres of undrained wetlands existed adjacent to Upper Klamath Lake and in portions of the Wood River Valley in 1940. Carlson (1993) estimated that about 20,000 acres of the wetlands adjacent to the lake were subsequently drained for agriculture by 1989. IMST (2003) indicated that wetlands surrounding Upper Klamath Lake and Agency Lake have declined from an historic level of 51,150 acres in wetlands to a current level of 17,370 acres as a result of dike construction and marsh drainage. IMST (2003) indicated that these wetlands were important filters that modify the form, amounts, and timing of nutrients delivered into the lake from the surrounding watershed, and the loss of these wetlands is a major reason for more nutrients entering the lake.

Gearheart et al. (1995) considered the reestablishment of wetlands on agricultural lands adjacent to Upper Klamath Lake to be one of the most effective watershed management strategies for reducing P loading to the lake, both as a means of eliminating an agricultural P source and creating habitat with the potential to trap incoming sediment and P. Gearheart et al. (1995) ranked the use of wetlands elsewhere in the watershed to treat nonpoint source agricultural pollution as moderately effective for P load reductions. Beyond their role in P storage, Gearheart et al. (1995) indicated that lakeshore wetlands likely serve as sources of dissolved organic carbon to Upper Klamath Lake.

Perdue et al. (1981) observed that Klamath Marsh waters rich in humic acids inhibited the growth of the blue-green algae *Aphanizomenon flos-aquae*, which is largely responsible for water quality impairments in Upper Klamath Lake. Geiger (2001) suggested that restored marshes around Upper Klamath Lake may thus serve to retard the onset of blooms of *A. flos-aquae* in the lake.

Snyder and Morace (1997) examined the nutrient loading to Upper Klamath Lake from adjacent drained wetlands. These drained wetland areas were reclaimed from the lake by building dikes to isolate them from the lake, constructing a series of drainage ditches, and installing pumps to drain the water and maintain a lowered water table. A consequence of lowering the water table is the increased ability of air and oxygenated water to move through the subsurface and facilitate the rapid aerobic decomposition of the organic peat soils. Nutrients, N and

P, are then liberated, leach into adjacent ditches, and are subsequently pumped to the lake or its tributaries. Snyder and Morace (1997) determined that, for individual drained wetlands, the yield of N and P lost from the organic soils for the study period ranged from 27 to 540 lbs/acre/yr and from 0 to 15 lbs/acre/yr, respectively. The total mass of N and P loss during this period was 3,000 tons/yr and 60 tons/yr, respectively, for all drained wetlands that were sampled. Implications of this study are that decomposition of organic peat soils of exposed wetlands within the lake's perimeter can result in release of nutrients to the lake, and suggests that the timing and duration of wetland inundation is an important consideration for managing nutrient loads.

Mayer (2005) conducted a study to evaluate water use at the Lower Klamath National Wildlife Refuge (NWR) in the Klamath Basin and the impacts of refuge wetland management on N and P concentrations and loads as water enters and leaves different wetland habitats on the refuge. Mayer (2005) determined that outflow nutrient concentrations of N and P generally increased relative to inflow concentrations, but nutrient loads were reduced. From 55 to 77 percent of the mass of N, and 19 to 51 percent of the mass of P entering the refuge wetlands was retained. Seasonal wetlands retained less P than permanent wetlands, possibly because of the annual drying cycle, and predominance of annual vegetation. For all refuge wetlands, dissolved inorganic N was retained more efficiently than particulate N, and particulate P was retained more efficiently than SRP. Mayer (2005) concluded that the ultimate effect of refuge wetland management was to decrease net N and P loads but increase the proportion of bioavailable P in the refuge outflow.

The Nature Conservancy (TNC) has been monitoring water quality at the Williamson River Delta (the Delta) since its restoration in 2007 (Wong and Hendrixson 2011). The 7,500-acre Delta straddles the last four miles of the Williamson River before it discharges into Upper Klamath Lake. Historically a naturally-functioning wetlands system, the wetlands were drained and converted for agricultural use in the 1940s. TNC completed the restoration by breaching levees on the perimeter of the Delta west of the Williamson River (known as Tulana) in 2007 and on the perimeter of the Delta east of the river (known as Goose Bay) in 2008 (Wong and Hendrixson 2011). Approximately 5,500 acres were re-flooded, restoring the wetlands as an open and passively managed system.

A key goal of this wetland restoration is to facilitate improvement in water quality in Upper Klamath and Agency Lakes by nutrient removal from surface waters through wetland ecosystem processes (Wong et al. 2011, Wong and Hendrixson 2011). TNC has conducted post-restoration effectiveness water quality monitoring in the Delta to determine the extent to which the wetlands provide a source or sink of nutrients, and to assess the effects of the restoration on surface water chemistry in the wetlands and adjacent lakes. In 2007, Wong et al. (2011) documented the actual P and N concentrations and loads in surface waters after initial flood restoration of the Delta, and compared results to those from prior experiments by Aldous et al. (2007), which used mesocosms to predict that P would be released into the water column upon initial flooding. Aldous et al. (2007) determined that 1 to 9 grams of P per square meter (g P/m^2) were released from the soils to the water column over the course of the experiment, which amounted to 1 to 16 percent of total soil P. Scaling up to the entire wetland, Aldous et al. (2007) determined this equates to a total of approximately 64 tons of TP released over 7,400 acres. Aldous et al. (2007) concluded that, even though they determined that P was released while undertaking this study, it is likely a temporary process, and once the wetland begins to resume more natural hydrological and biogeochemical functions and vegetation structure, it will re-start the process of soil accretion and P sequestration.

The subsequent field sampling in the Delta by Wong et al. (2011) corroborated the prior Aldous et al. (2007) mesocosm-based predictions that wetlands would release P upon flooding. Actual P concentrations in the wetlands were up to six times greater than at adjacent Upper Klamath Lake and Agency Lake sampling sites. However, the corresponding amount of P release from the wetlands estimated by Wong et al. (2011) was 0.2 g P/m^2, an order of magnitude less than the range of 1 to 9 g P/m^2 predicted by Aldous et al. (2007). Wong et al. (2011) found that P concentrations in the wetlands increased from the first post-flooding sampling event to the second event that occurred seven days later, which also differed from the laboratory experiments by Aldous et al. (2007), where the majority of P release occurred within 2-3 days of flooding and then rapidly declined.

Wong and Hendrixson (2011) provide three additional years (2008, 2009, and 2010) of post-restoration effectiveness water quality monitoring in the Delta. Wong and Hendrixson (2011) conclude that the increase in P concentrations that occurred with the initial flood restoration of the Delta has diminished since 2007. Seasonal trends and ranges in P concentrations in 2009 and 2010 were lower than in 2007 and 2008. For example, P concentrations at wetlands sampling sites ranged up to about 0.7 mg P/L in 2007 and 2008, and ranged up to about 0.4 mg P/L in 2009 and 2010. The data in Wong and Hendrixson (2011) also indicate that P concentrations at wetlands sampling sites in 2009 and 2010 were more similar to adjacent lake sampling sites that in 2007 and 2008, indicating that that the wetlands are continuing to transition toward a nutrient condition more in equilibrium with the surrounding lakes. Wong and Hendrixson (2011) concluded that quantifying an accurate nutrient load from the wetlands as a whole since restoration flooding occurred may be difficult or impossible because of the hydrologic connectivity and spatial complexity of the wetlands.

Concurrently with the TNC monitoring (Wong et al. 2011, Wong and Hendrixson 2011), the USGS (Kuwabara et al. 2012) deployed porewater profilers in the flood-restored wetland area in the Delta to assess spatial and temporal variation in benthic flux of solutes over an approximately 4-year period beginning 3 days after initial flooding, including macronutrient parameters dissolved organic carbon (DOC), ammonia (NH3), and SRP.

Benthic fluxes for DOC calculated by Kuwabara et al. (2012) were highest at the newly flooded wetland sites during the initial sampling in November 2007 (as high as 712 mg/m^2/day), compared to negative or insignificant DOC fluxes at adjacent lake sites. Over the next several sampling events during 2008-2011, DOC benthic fluxes dissipated in the reconnected wetlands, converging to values similar to those for established wetlands and to the adjacent lake. In contrast to DOC, benthic sources of SRP from within the reconnected wetlands were consistently elevated throughout the study period (that is, significant in magnitude relative to adjacent lake and established-wetland sites). Kuwabara et al. (2012) indicate that these results suggest a multi-year time scale associated with restoring wetlands to provide natural, seasonal function and processes.

TNC has also conducted sampling of water quality at the inlets and outlets of Sycan Marsh during 2004-2006 (Aldous 2009, as cited in Wong and Bienz 2011) and 2010-2011 (Wong and Bienz 2011). Sycan Marsh lies within the headwaters of the Sprague subbasin of the Upper Klamath Basin (Figure 2). The Sycan River is the single outlet from the marsh and eventually flows into the Sprague River. The Sprague subbasin contributes roughly 33 percent of the inflow volume to Upper Klamath Lake and the highest portion of the total external P load to the lake (about 27 percent) (Boyd et al. 2002). The marsh itself is fed by several streams (Coyote, Long, Pole, Chocktoot, and Shake Creeks), and by a groundwater fen on the northern end of the marsh. The marsh is drained on the southern end where the Sycan River leaves the marsh.

The main goal of the sampling by TNC was to investigate the potential effects of TNC's recent management and restoration activities on the water quality at the inlets and outlets of Sycan Marsh (Wong and Bienz 2011). Historically a natural, unaltered wetlands system, the flow regime and vegetation within Sycan Marsh and along its inlet streams were affected by irrigation system development and agricultural use beginning in the early 20th century (Wong and Bienz 2011). In 1980, 1999, and 2001, TNC acquired tracts totally 30,539 acres at Sycan Marsh to establish a preserve to enhance the ecological health of the marsh. In 2005, TNC initiated a strategic interagency watershed restoration program in the marsh and surrounding watershed to enhance the marsh's natural hydrologic patterns and processes (Wong and Bienz 2011).

In 2004-2005, Aldous (2009, as cited in Wong and Bienz 2011) assessed TP and total nitrogen (TN) concentrations and loads at the inlets and outlets of Sycan Marsh. Aldous (2009) found that both concentrations and loads of TP and TN were greater at the Sycan outlet than any of the individual inlets to the marsh, and stated that the Sycan Marsh may be a significant contributor to N and P loading in the Sprague subbasin. For example, in August 2006, instantaneous TP and TN load estimates, summed for the inlets, increased by about 55 percent and by almost 200 percent, respectively, upon leaving the marsh (Aldous 2009). Aldous (2009) also found both TN and TP instantaneous loads at the outlet were much greater than at the inlet in June 2005 and May 2006.

In 2012-2011, Wong and Bienz (2011) conducted sampling of water quality at the inlets and outlets of Sycan Marsh during a total of three sampling events (November 2010, May 2011, and August 2011). The results of the sampling by Wong and Bienz (2011) suggest a contrast to the observations by Aldous (2009). Whereas Aldous (2009, as cited in Wong and Bienz 2011) reported a substantial increase in instantaneous loads of TN and TP at the outlet, Wong and Bienz (2011) reported substantial decreases in dissolved N and P and TN and TP instantaneous loads at the outlet in the November 2010 and August 2011 sampling events, with the exception of TN in November. Wong and Bienz (2011) were not able to sample the Sycan River inlet site during the May 2011 event, and thus were not make inlet-outlet comparison for that event. Based on the contrasting results with Aldous (2009), Wong and Bienz (2011) indicated that an overarching assessment of long term changes in nutrient water chemistry at the marsh could not yet be made and warranted further investigation.

Studies have occurred in recent years in the Wood River Wetland to assess the effects on water quality, hydrology, and habitat conditions of restoring previously-drained wetland areas (Carpenter et al. 2009; A. Hamilton, BLM, pers. comm.). In 1994, the BLM completed acquisition of the Wood River Ranch, a 3,200-acre cattle ranch on the north side of Agency Lake (Figure 2) that was drained and converted to irrigated pastureland in the 1950s (Carpenter et al. 2009). Starting in 1995, BLM implemented a resources management plan for the area, renamed as the Wood River Wetland, which included cessation of grazing and implementation of seasonal flooding and water management aimed at improving water quality, increasing water availability, and providing wildlife and fish habitat (Carpenter et al. 2009).

In 2003-2005, the U.S. Geological Survey (USGS), in cooperation with BLM and with subsequent support from the U.S. Fish and Wildlife Service, began a study to characterize the hydrologic and water-quality conditions within the Wood River Wetland to help address questions regarding possible consequences of future management options (Carpenter et al. 2009). The results of the 2003–2005 study indicated that N and P levels, primarily as dissolved organic N (DON), ammonium (NH_4^+), and SRP were high in wetland surface waters. DOC concentrations also were elevated in surface water. Despite the high NH4 concentrations, the nitrate (NO_3) levels were moderate to low in wetland surface waters.

Carpenter et al. (2009) found that the surface-water concentrations of NH_4^+ and SRP increased in spring and summer to very high values by the end of summer. These high levels were assumed indicative of evaporative concentration (referred to as “evapo-concentration”), since the NH_4^+ and SRP concentrations outpaced those for chloride (a conservative tracer). Carpenter et al. (2009) also conducted in-situ chamber experiments in June and August 2005 that indicated a positive flux of NH4 and SRP from the wetland sediments, which were thought to derive from diffusion of nutrients from decomposed peat, decomposing aquatic vegetation, or upwelling ground water. Water-column SRP and TP levels decreased during autumn and winter following inputs of irrigation water and precipitation, which have lower nutrient concentrations. The SRP concentrations, however, decreased faster than the dilution rate alone, possibly due to precipitation of P with iron, manganese, or calcium (Carpenter et al. 2009).

Carpenter et al. (2009) suggested that the high concentrations of dissolved N and P during the growing season gave rise to abundant macrophytes, phytoplankton, and benthic and epiphytic algae in the wetland that produced large observed effects on DO and pH. Carpenter et al. (2009) measured midday values of surface-water DO during summer that were often supersaturated (greater than 115 percent saturation) with elevated pH (up to 9.2 units), indicative of high rates of photosynthesis.

Carpenter et al. (2009) developed a water budget for the Wood River Wetland over 2 water years (2004 and 2005). Outflows exceeded inflows by about 22 percent over the 2-year period. Inflows for the entire wetland consisted of precipitation (43 percent), regional ground-water discharge (40 percent), applied water for irrigation from adjacent surface-water bodies (12 percent), ground-water seepage through dikes (4 percent), and discharge from artesian wells (1 percent). Outflows consisted of open-water evaporation (64 percent) and evapotranspiration from emergent vegetation (36 percent). Changes in surface- and ground-water storage during this period amounted to losses of 1 and 2 percent relative to the total inflow, respectively. A water-budget residual consisting of the errors in measurement of all water-budget components and the sum of any unaccounted for components indicated a deficit of water of 19 percent relative to the total inflow. The monthly patterns in water-budget residuals closely resembled the pattern expected for soil-moisture storage, which was not estimated, but indicated that soil moisture could represent a significant part of the water budget for the wetland.

Since 2006, BLM has implemented a carefully managed water regime within the Wood River Wetland to optimize vegetation establishment, water storage and discharge, water quality discharge, and the accumulation of new organic soil (A. Hamilton, BLM, pers. comm.). In addition, artesian wells with high nutrient concentrations in the area were decommissioned in 2008. During 2007-2011, BLM conducted nutrient and water quality sampling at several sites in the Wood River Wetland, and the results indicate a trend of declining nutrient discharge concentration over this 5-year sampling period (A. Hamilton, BLM, pers. comm.).

Substantial reductions in P concentrations, particularly compared to previous 2003-2005 values, suggest that the newer water management regime has helped to ameliorate the effects of evapo-concentration, which led to high nutrient concentrations (Carpenter et al. 2009) under prior conditions. Potential mechanisms for improved water quality conditions include: depletion through pumping of the legacy nutrient pool; increased uptake of nutrients due to expanded emergent vegetation cover; lower decomposition rates of peat soils resulting from increased anaerobic (reducing) conditions; and increased long-term organic sediment accumulation (A. Hamilton, BLM, pers. comm.). In addition, during 2008-2011, BLM showed consistent year-to-year net retention of nutrients in the Wood River Wetland (referred to as "sequestration"), including sequestration rates up to 1.79 $g/m^2/yr$ of TN and up to 0.83 $g/m^2/yr$ of TP (A. Hamilton, BLM, pers. comm.). The groundwater load of the nutrient budget was estimated by multiplying shallow piezometer well nutrient concentrations (believed to be a conservative but uncertain measure of groundwater inflow concentration) to the groundwater volume inflow estimates (A. Hamilton, BLM, pers. comm.).

BLM developed a water balance estimate for 2007-2011 whereby the residual term of the annual water budget was attributed to groundwater and levee seepage after accounting for surface water inflows/outflows, open water evaporation, and evapo-transpiration (A. Hamilton, BLM, pers. comm.). Inflows for the entire wetland consisted of precipitation (21 percent), ground-water discharge (59 percent), and applied water for irrigation and leakage from adjacent surface-water bodies (20 percent). Outflows consisted of open-water evaporation and evapo-transpiration from emergent vegetation (69 percent) and pumping (31 percent).

In 2010-2011, BLM also conducted a study to determine accumulation of new organic soil in the Wood River Wetland, referred to as "subsidence reversal" (A. Hamilton, BLM, pers. comm.). The study was based on 22 plots established in 2010 in various wetland habitat types that were subsequently measured in 2011 based on sampling of clay horizon cores. Initial results indicate a mean soil accumulation rate of 1.4 inches per year (in depth) and a range from 0.5 to 4.7 inches per year (A. Hamilton, BLM, pers. comm.). Actual elevation change will be measured from deep set benchmarks starting in 2012. At the initial results rate, the wetland area would accumulate about 1.7 feet of new soil after 15 years. Soil accumulation estimates are being used by BLM to predict the potential effects of subsidence reversal on growth and expansion of emergent vegetation and other habitat types in the Wood River Wetland under future water management scenarios including reconnection Upper Klamath Lake to the wetland. Build-up of organic soils, along with on-going plant uptake and biomass accumulation, is also anticipated to result in a continued long-term trend of nutrient sequestration in the wetland (A. Hamilton, BLM, pers. comm.).

2.3 Previous Studies on Potential Treatment Wetlands

Lytle (2000) prepared a technical memorandum for the Bureau of Reclamation to provide information about the potential use of wetlands treatment as a water quality improvement option for water entering the Klamath River from the Straits Drain (that discharges into the Klamath River at Keno reservoir as depicted in Figure 2). Lytle (2000) used a modeling approach from Kadlec and Knight (1996) to estimate wetland size requirements for reducing TP from an influent value of 0.41 mg/L to a target effluent value of 0.16 mg/L at an assumed input flow of 70 to 133 cfs. This modeling approach from Kadlec and Knight (1996) is similar to that used in the estimates of wetlands area in CH2M HILL's analysis as discussed in Chapter 4 of this report.

Lytle (2000) used TP as the "parameter of concern" because of its role as the potential limiting nutrient for algal growth, and to conservatively estimate wetland size requirements because as Lytle (2000) states "treatment of total P would require the largest wetland surface area". The influent value of 0.41 mg/L was chosen because it represented the highest TP concentration in the 1991-1999 dataset for the Straits Drain used by Lytle (2000). The target effluent value of 0.16 mg/L was chosen because it represented the mean TP concentration (in the 1991-

1999 dataset) at the monitoring site in Keno reservoir upstream of the Straits Drain. The assumed input flow of 70 to 133 cfs was chosen to represent the range of daily flow values from the Straits Drain.

Lytle (2000) estimated from this modeling analysis that an emergent surface-flow wetland system would require a treatment surface area ranging from 1,633 to 3,115 acres provided the wetland received a daily flow between 70 and 130 cfs. With this estimated treatment surface area, it would be anticipated that the wetland system could achieve a 61 percent reduction in TP concentration (i.e., from the assumed 0.41 mg/L to 0.16 mg/L values as described above). Lytle (2000) recommended that, if a wetland system was considered, a wetland pilot-study be implemented to develop site and water-quality specific parameters prior to design and construction of a full-scale system.

Deas and Vaughn (2006) characterized organic matter and associated water quality constituents in the reach between Link Dam and Keno Dam, and used this information to assess the feasibility of improving the water quality in this reach using treatment wetlands. Deas and Vaughn (2006) determined carbonaceous oxygen demand (COD) and biochemical oxygen demand (BOD) in water samples from the reach. COD measures the ultimate oxygen demand on the water from all carbon in the water, and BOD measures the oxygen demand over several days (in this case, five days) on the water from biologically available organic matter (e.g., algae biomatter). The combined COD and BOD information assists in estimation of labile and refractory fractions of organic matter.

The results of this organic matter characterization were used to estimate a representative range of wetland sizes and specifications for organic matter treatment. Deas and Vaughn (2006) performed calculations indicating that wetlands treatment may be a viable option for notably reducing organic loads from Upper Klamath Lake.

Using a flow rate of 25 percent of monthly average inflow[1] to Keno Reservoir as a baseline for system design, Deas and Vaughn (2006) estimated that 1,054 acres of treatment wetland area would on average reduce BOD by 26 percent, and 2,192 acres of treatment wetland area would on average reduce BOD by 82 percent (scaling this up to 100 percent of river flows would translate to approximately four times these acres of wetlands). However, during periods of high BOD in influent waters, this wetland area would be incapable of processing the assumed 25 percent of river flow and diversions to the wetland would have to be reduced to less than half – a considerable reduction in overall treatment capacity during a critical water quality period. Deas and Vaughn (2006) indicated that the reduction of BOD in the wetland water is largely dependent on the influent BOD, the desired BOD effluent, depth, flow rate, wetland size, and desired level of reliability.

As follow-up to the evaluation by Deas and Vaughn (2006), Mahugh et al. (2008) conducted a feasibility study to assess potential full- and pilot-scale treatment wetlands adjacent to the Klamath River near Klamath Falls, and to estimate construction and operation costs for a pilot-scale treatment wetland. Mahugh et al. (2008) stated that the purpose of the treatment wetlands would be to reduce the organic load delivered into Keno Reservoir from Upper Klamath Lake to improve water quality (principally dissolved oxygen) for fish habitat in the Klamath River. This feasibility study focused on land parcels encompassing more than 84,000 acres along Keno Reservoir (from RM 235 to RM 253), and including lands to the east of Keno Reservoir with hydrologic connections through major irrigation canals and drains. Potential constructed treatment wetland sites were identified, and top sites were ranked for potential pilot-scale implementation. To identify the range of treatment levels associated with a pilot wetland design, a series of wetland treatment models were developed and applied to a pre-design configuration to assess BOD and total suspended solids (TSS) removal capabilities. For the modeling purposes, a 63-acre site located on the west shoreline of Keno Reservoir ("Site 1") was used for calculations. Overall, the modeling indicated that wetland treatment is a viable approach to reducing TSS and BOD in the Keno reservoir reach, with reductions ranging from approximately 60 to 90 percent and 50 to 70 percent for BOD and TSS, respectively, depending on wetland design features. Data from the modeling analysis were used to develop pre-design cost

[1] Assumed to be 25 percent of the average flow at USGS gage 11507500, Link River at Klamath Falls, OR, calculated from May through October 2005. The monthly average flows at this gage ranged from 968 cfs to 2,487 cfs by month, with an average of 1,289 cfs for the May through October 2005 period. Therefore, the assumed 25 percent flow values used in the calculations by Deas and Vaughn (2006) were 242 cfs to 622 cfs by month, with an average of 322 cfs.

estimates for pilot project design and construction. Total construction cost was estimated at $252,400 (not including operations and management costs).

Lyon et al. (2009) conducted a feasibility assessment to examine the potential for treatment wetlands to provide improved water quality in the vicinity of PacifiCorp's Klamath Hydroelectric Project (from RM 190 to RM 253). Lyon et al. (2009) identified potential sites and conceptual layouts for locations both upstream and within the Project reservoirs. The upstream sites were considered as "preventative" sites, because upstream sites would be intended for treatment of water quality upstream of the reservoirs (and below Upper Klamath Lake) to remove nutrients and algae (i.e., the "cause" component). The "reservoir" sites would be intended for treatment of accumulations of algae biomass within the reservoirs, such as in reservoir coves (i.e., the "effect" component). A number of candidate sites for upstream "preventative" treatment wetlands were identified, especially in the river upstream of Copco reservoir between about RM 205 to RM 209. Conceptual layouts for constructed treatment wetlands were developed for these sites, along with calculated estimates of potential treatment effectiveness (e.g., nutrient reductions). The sites were generally low-lying and directly adjacent to the river. A major constraint identified in this assessment was that the potential sites for constructed wetlands on PacifiCorp-owned lands in the Project area could receive and treat only a minor fraction of the total flow of the Klamath River, and would be unlikely to provide measureable improvements to downstream river or reservoir water quality. Instead, to achieve a demonstrable river nutrient and organic matter load reduction, it would be necessary to develop more and larger wetland sites at a more basin-wide scale that could collectively or in the aggregate treat a substantial portion of the river flow. Lyon et al. (2009) estimated that costs of constructed treatment wetlands would vary greatly depending on size and site conditions. Lyon et al. (2009) estimated that costs could range from $15 to $45 million for construction of 300 acres of treatment wetlands on PacifiCorp-owned sites, including about $150,000 to $2 million per year for operation and maintenance.

Lyon et al. (2009) also evaluated several candidate sites for "reservoir" treatment wetlands, especially within Copco reservoir (located on the Klamath River from RM 198.6 to RM 203.2). Potential sites were identified and design concepts were developed to evaluate vegetated swale ("bioswale") filtration as a possible treatment of localized accumulations of algae biomass resulting from summertime blooms in the reservoirs.

Site conditions adjacent to reservoir cove areas are amenable to potential construction of vegetated swales for removal and filtering of accumulated algae biomatter. However, a key constraint to the use of this system is how the algae would be efficiently collected from the reservoir and pumped to the swales. In addition, the use of vegetated swales for treatment of algae would need additional pilot-scale or demonstration testing to determine operating conditions and removal efficiencies. Until such determinations are made, the ultimate effectiveness of implementing vegetated swales for reducing algae biomatter remains uncertain. As with constructed treatment wetlands, Lyon et al. (2009) estimated the cost of installing and maintaining vegetated swales varies widely with design and site variability. Lyon et al. (2009) estimated that costs for vegetated swale systems at two sites identified in the assessment (about 50 acres) would be on the order of $2.5 to $3.8 million for construction, and about $125,000 to $275,000 per year for operation and maintenance.

The Regional Water Board (2010) identified the restoration and construction of wetlands as an important strategy for reducing the large loads of nutrients and organic matter in the Klamath River to meet TMDL allocations. TMDL allocations established by the Regional Water Board (2010) for the Klamath River will require reductions in the current nutrient loads in the river (as measured at Stateline) of about 90 percent for TP and 70 percent for TN. The Regional Water Board (2010) mentions wetland treatment systems and other "nutrient reduction technologies" as potential actions to address these substantial nutrient reductions.

(This page intentionally blank)

CHAPTER 3
Overview of Treatment by Wetlands

3.1 Background

Treatment wetlands are natural or constructed wetlands engineered for water quality improvement. Over the past five decades, treatment wetlands have grown to become an accepted technology for improving water quality with minimal energy requirements and a more natural, "environmentally-friendly" profile (Vymazal 2011). Historically, wetlands have been most frequently used for removal of conventional wastewater contaminants, such as BOD, TSS, and nutrients. More recently, treatment wetland technologies have been applied to a diversity of polluted waters (Kadlec and Wallace 2009). Treatment wetlands have been successfully used for a variety of wastewater problems, not only to augment conventional wastewater treatment plants (Kadlec and Wallace 2009), but to treat agricultural wastewaters (Bays and Jordahl 2010, Harris et al. 2007), heavy metals removal in drainage affected by mines (Nairn et al. 2009), pesticides reduction in runoff (Budd et al. 2009, Gregoire et al. 2009, Imfeld et al. 2009), aircraft deicing fluid discharges at airports (Van der Tak et al. 2005), membrane concentrate (Frank et al. 2010), and for wastewater temperature reduction for protection of fisheries habitats (Madison et al. 2008). In addition, recent and ongoing research is helping to optimize the design and treatment capacity of these engineered natural systems in terms of hydraulics (Lightbody et al. 2009), physico-chemical processes (Austin et al. 2007), and microbial community structure (Austin and Sun 2007).

Treatment wetlands are ecosystems engineered to improve water quality by mimicking natural wetland ecosystem physical, biological, and chemical processes. Planted with vegetation adapted to life in saturated soil, treatment wetlands range from large open water marshes to subsurface gravel bed filters. Regardless of how they are designed or constructed, their intent is to utilize natural microbial, physical, and chemical processes found in highly productive wetland environments. Such processes have been proven to remove or transform many kinds of contaminants from a variety of water sources, and are fundamentally the same processes applied in conventional wastewater treatment practices, just at natural rates. In doing so, the systems also create aquatic vegetation communities similar to those found in nature, often in landscapes where such communities have been historically eliminated by land use conversion and development. Treatment wetlands can also provide other benefits that conventional water treatment processes lack, such as educational opportunities, wildlife habitat, and public opportunities for recreation (e.g., Bays et al. 2000).

3.2 Mechanisms of Treatment in Wetlands

It is well-established that interception and removal of nutrients and particulates (including algae) can be accomplished using constructed wetlands (EPA 2000a, EPA 2000b, ITRC 2003, Kadlec and Wallace 2009). The wetland-related mechanisms and processes that are utilized to improve water quality include: (1) settling of suspended particulate matter; (2) filtration and chemical precipitation through contact of the water with the substrate and plant materials; (3) chemical transformation; (4) adsorption and ion exchange on the surfaces of plant materials, substrate, and sediment; and (5) uptake and transformation of nutrients by microorganisms and plants. The most effective treatment wetlands are those that include and successfully utilize these mechanisms and processes.

In constructed treatment wetlands, effective interception and removal of nutrients and particulates requires careful wetland system design and management. For example, the removal of nutrients and particulates require that the average residence time of water in the wetland – referred to as "hydraulic residence time" or "hydraulic retention time" (HRT) is of sufficient duration (on the order of several days) for wetland-related mechanisms and processes to occur. In addition to HRT, another important factor that determines the removal rate of nutrients and particulates in wetlands is the influent (or inflow) concentrations or loads of these constituents. Wetland treatment efficiency is dependent upon inflow concentration and mass loading. Treatment efficiencies decline to zero as the wetland inflows approach background constituent concentrations – that is, those concentrations that would occur as a result of natural (i.e., unimpaired) hydrologic, climatologic, and parent geochemical conditions.

3.2.1 Particulates – Suspended Solids and Organic Matter

Removal of particulates, including suspended inorganic materials and organic biomatter, is primarily due to the physical processes of settling, sedimentation, filtration, and interception (EPA 2000a, EPA 2000b, Kadlec and Wallace 2009). Water flowing into a wetland containing particulates slows and spreads through a large area of shallow water and emergent vegetation. Particulates (typically measured as total suspended solids [TSS]) tend to settle and are trapped due to lowered flow velocities and sheltering from wind. These particulates contain biochemical oxygen demanding (BOD) components, fixed forms of TN and TP, and trace levels of metals and organics.

Typically in wetland systems, total suspended solids and biochemical oxygen demand require the smallest treatment areas for effective removal. In contrast, TN (NH3, NO_3 and organic N) and TP require much larger wetland areas for adequate treatment (Kadlec and Wallace 2009). Normally, an HRT of about 2 days is needed to remove approximately 80 to 90 percent of total suspended solids (TSS) typically found in lake and river waters. TSS removal is most pronounced in the inlet region of a FWS constructed wetland. Generally, the influent TSS from oxidation pond systems are removed in the first 2 to 3 days of the nominal hydraulic retention time in fully vegetated zones near the inlet (Gearheart et al. 1989, Reed et al. 1995, Kadlec and Knight 1996). Enhanced settling and flocculation processes account for most of this removal, and the overall removal efficiency is a function of the terminal settling velocity of the influent and flocculated solids. Crites et al. (2006) report that more buoyant algae biomatter may require 6 to 10 days of HRT in natural treatment systems for removal.

Using Stokes' Law to approximate discrete settling velocity, particles ranging from 1 to 10 µm with a specific gravity ranging from 1.01 to 1.10 will settle at a rate of from 0.3 to 4×10^{-4} m/day. Typical hydraulic loads to wetlands are in the range of 0.01 to 0.5 m/day (note that the hydraulic load is equivalent to the mean settling velocity of a particle that will be removed exactly at that loading). Assuming the higher settling velocity of 0.3 m/day and a typical system velocity of 50 m/day and depth of 0.8 m, the larger particles would settle by gravity in approximately 2.7 days, or 133 m along the wetland longitudinal axis.

3.2.2 Nutrients

3.2.2.1 Wetland Processes Affecting Phosphorus

Phosphorus typically enters wetlands with suspended solids or as dissolved P. Wetland systems can remove P through processes of assimilation (i.e., direct uptake into tissues), adsorption (i.e., attachment to mineral surfaces), and accumulation (i.e., buildup and storage) of P in various abiotic and biotic components, including organic and inorganic soil and sediment particles, above-ground and below-ground plant tissues, detritus, periphyton, microorganisms, and other organic matter (Johnston 1991, Walbridge and Struthers 1993, Reddy et al. 2011). The key pathways for P in wetland systems are shown in Figure 6.

Significant quantities of P associated with suspended solids are deposited in wetlands (Walbridge and Struthers 1993). Soils and sediments serve as long-term sinks for P and store the majority of P in the ecosystem. The main long-term sink for P in a treatment wetland system is burial in wetland sediments (Kadlec and Wallace 2009). Adsorption is only a temporary sink as sorption sites are often saturated within a few years of operation (Kadlec and Wallace 2009). The potential for long-term storage of P through adsorption to wetland soil is greater than the maximum rates of P accumulation possible in plant biomass (Johnston 1991, Walbridge and Struthers 1993, Kadlec and Wallace 2009). By comparison, storage of P in vegetation and other biotic communities of wetlands tend to be small and short term (EPA 2000a, EPA 2000b, Kadlec and Wallace 2009). Uptake occurs during the growth phase of these organisms and subsequent release into surface water occurs during subsequent senescence and death in the late summer and fall, as organic matter decomposes.

Phosphorus is stored both in organic and inorganic forms in wetlands soils and sediments (Kadlec and Wallace 2009, Reddy et al. 2011). A large proportion of the P in wetland soils, especially in peats, occurs in organic form, suggesting the importance of organic P sequestration in the long-term stabilization of P in wetlands. The efficiency of any wetland to store P on a long-term basis is determined by the peat or sediment accretion rate times the net increase of P stored by these processes each year. To retain as much P as possible, the input rates should be limited to the long-term storage capacity, which is controlled by peat/sediment accretion. In typical North American freshwater wetlands, accretion rates average from 1 to 2 mm/year (Craft and Richardson 1993). Wetland soils can, however, reach a state of P saturation, after which P may be released from the system (Richardson 1985).

Eutrophic lakes and nutrient-enriched wetlands typically exhibit high rates of recently accreted organic material, consisting of partially decomposed detrital matter originating from microbes, periphyton, and macrophytes, and particulate inorganic material (Reddy et al. 2011). This accreted organic matter in productive wetland systems ultimately forms peat over time that has different physical and biological characteristics than the underlying soil. This recently accreted organic material (also referred to as “floc”) can act as a sink or source of nutrients to the overlying water column and serves as an indicator of the nutrient retention characteristics of a wetland (Reddy et al. 2011).

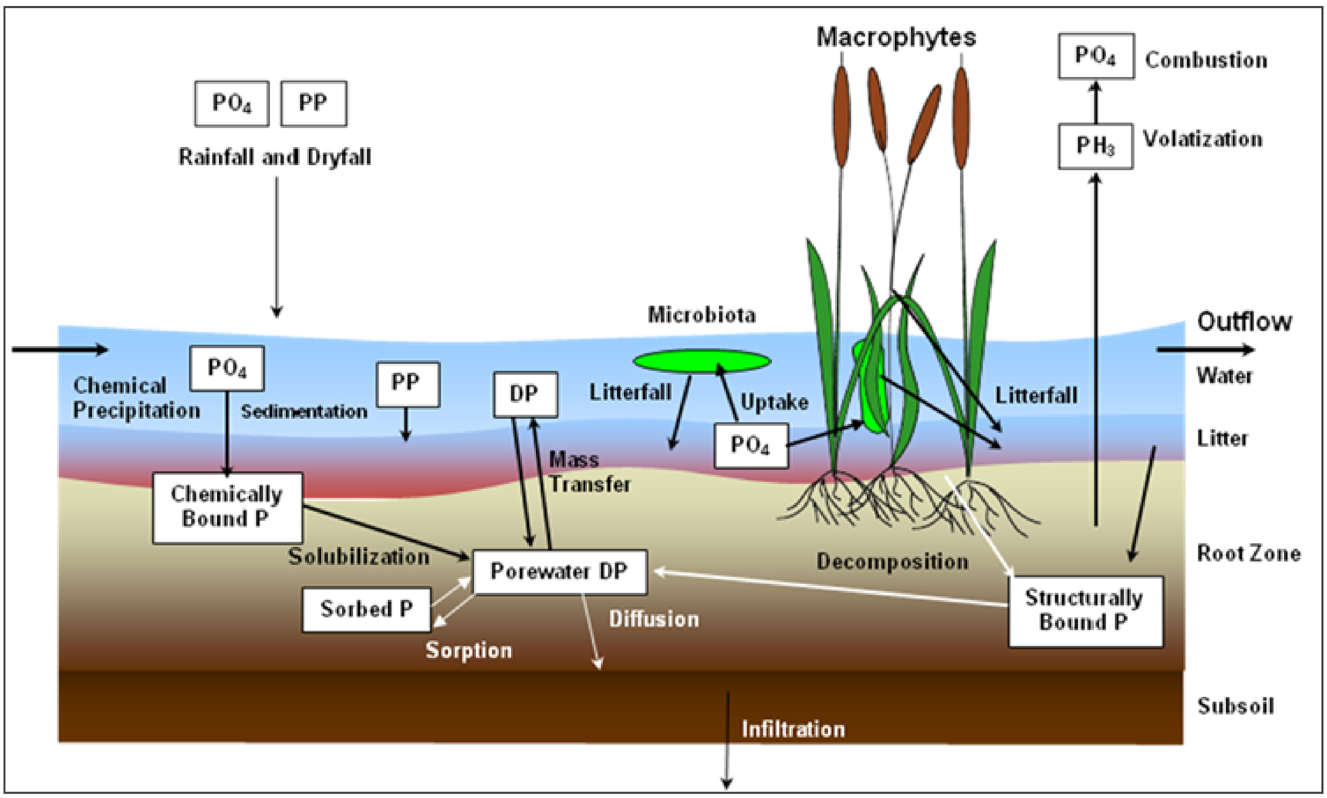

FIGURE 6

Major pathways for phosphorus (P) in wetland systems. (Source: Kadlec and Knight 1996). PO_4=orthophosphate, PP = particulate organic P; DP = dissolved phosphorus; PH_3 = phosphine inorganic P; DIP = dissolved inorganic P; DOP = dissolved organic P.

Dissolved P is processed by wetland soil microorganisms, plants, and geochemical mechanisms (Walbridge and Struthers 1993, Kadlec and Wallace 2009). Microbial removal of P from wetland soil or water is rapid and highly efficient; however, following cell death, the P is released again. Similarly, for plants, litter decomposition causes a release of P.

Most inorganic P compounds in soils fall into one of two groups: those containing calcium and those containing iron and aluminum (Walbridge and Struthers 1993, Kadlec and Wallace 2009). In more alkaline wetlands, such as found in the Klamath Basin, P precipitates with calcium as calcium phosphate (Novotny and Olem 1994). However, the presence of aluminum is the significant predictor of dissolved P sorption and removal from water in most wetland systems (Walbridge and Struthers 1993). The capacity for P adsorption by a wetland, however, can be saturated in a few years if it has low amounts of aluminum and iron or calcium (Richardson 1985).

Wetlands along rivers have a high capacity for P adsorption because as clay is deposited in the floodplain, iron and aluminum in the clay accumulate as well (Gambrell 1994). Thus floodplains tend to be important sites for P removal from the water column, beyond that removed as sediments are deposited (Walbridge and Struthers 1993).

3.2.2.2 Wetland Processes Affecting Nitrogen

Forms of N in wetlands include organic N, oxidized N (NO_3/nitrite), and NH3/NH_4^+. Traditionally recognized pathways are illustrated in Figure 7. Pathways include: (1) physical transfers of N within wetlands without changes in species of N; (2) microbially-mediated transformations of N species; and (3) transformations in species of N between organic and inorganic forms (Kadlec and Wallace 2009). The physical transfers include settling of suspended particles and resuspension of these particles if there is some physical disturbance. Dissolved forms diffuse over very small distances to areas of lower concentration. Plants translocate N from roots to shoots during rapid growth, and from shoots to roots late in the growing season. Wetland plant shoots containing N reach maturity and accumulate at the base of the water column. Ammonium is charged, and may sorb onto negatively charged surfaces on organic materials and soil particles.

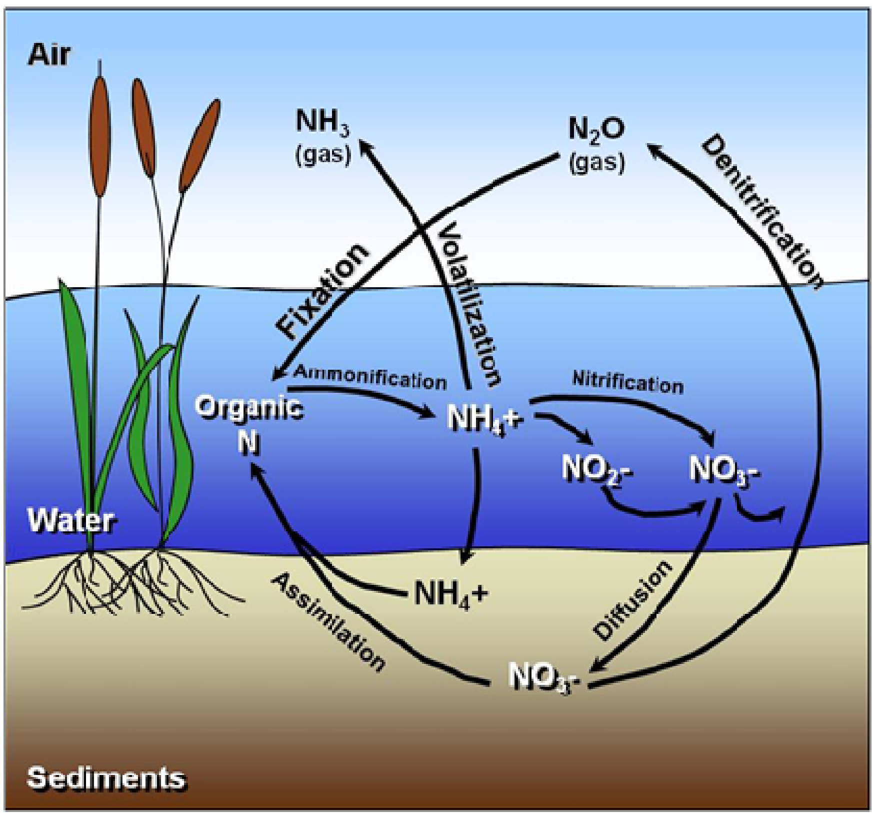

FIGURE 7
Major pathways for nitrogen in wetland systems. (Source: Kadlec and Knight 1996). NO_{3-}=nitrate; NO_{2-}=nitrite; NH_3=ammonia; NH_{4+}=ammonium; N_20 = nitrous oxide.

Nitrogen species are subject to a number of chemical and biological transformations that affect sources, sinks, and ultimately the fate of N in wetlands. These transformations include assimilation (microbial or plant uptake), nitrification, denitrification, ammonification (or mineralization), anaerobic NH3 oxidation (ANNAMOX), and dissimilatory NO_3 reduction to NH3 (DNRA). When NO_3 is the dominant form of N entering a wetland, assimilation (plant uptake) and denitrification and are the dominant processes (Kadlec and Wallace 2009). Denitrification occurs primarily in soil sediments and in periphyton films, and is temperature dependent. It is sometimes assumed as an operative mechanism for N removal in the absence of proof otherwise (Kadlec and Wallace 2009).

The primary transformation processes are defined as follows:

- **Plant Uptake or Assimilation**. Wetland plants will take up or assimilate N as an important part of their metabolism. Inorganic N forms are reduced by the plant to organic N compounds used for plant structure. During the growing season, there can be a high rate of uptake of N by wetland plants. Estimates of net annual N uptake by emergent wetland plant species vary from 0.5 to 3.3 gN/m^2/yr (Kadlec and Wallace 2009).

- **Nitrification**. Nitrification is a process in which wetland microbes convert NH_4^+ to nitrite (NO_2), and subsequently to NO_3 in the presence of dissolved oxygen. NH3 will require approximately 10 times the wetland area (or HRT) of that needed to process NO_3 since it must be nitrified to NO_3 under oxidized conditions. Temperature affects nitrification so that rates can be significantly reduced during the colder months, which can affect treatment wetland design requirements.

- **Denitrification**. Denitrification is a process in which wetland microbes convert NO_3 to N gas ($NO_3^- \rightarrow N_2$ or N_2O) under anaerobic conditions. Denitrification reactions usually occur in wetland sediments where dissolved oxygen is low and available carbon is high. When NO_3 is the dominant form of N entering a wetland, denitrification and plant uptake are the dominant processes (Kadlec and Wallace 2009). Like nitrification, temperature affects denitrification so that rates can be significantly reduced during the colder months, which can affect design requirements.

- **Mineralization or Ammonification**. Mineralization or ammonification is the biological transformation of organic forms of N to NH_4^+. This process occurs both in aerobic and anaerobic conditions.

- **Anaerobic Ammonia Oxidation (ANNAMOX)**. ANNAMOX is the microbial oxidation of NH_4^+ via NO_2 under anaerobic conditions, producing N_2 (i.e., $NH_4^+ + NO_2^- \rightarrow N_2 + 2H_2O$) – a transformation process that was not recognized until relatively recently (Kadlec and Wallace 2009). The end result is therefore similar to denitrification, in that N_2 is released to the atmosphere, but NH_4^+ and NO_2, rather than NO_3, are consumed in the process. The process is increasingly being considered in wastewater treatment plant designs, engineered treatment wetland systems, and has been found to occur in natural wetland systems.

- **Dissimilatory Nitrate Reduction to Ammonia (DNRA)**. DNRA can occur under similar conditions as denitrification (Tiedje 1982), and is also conducted by bacteria. The organisms use NO_3 as an electron acceptor, and the end result is the reduction of NO_3 to NH_4^+ (Tiedje 1982). The existence of DNRA therefore means that NO_3 disappearance does not necessarily mean that N_2O and N_2 (products of denitrification) are the only products (Tiedje 1982).

- **Nitrogen Fixation**. Nitrogen fixation is the transformation of atmospheric N (N_2) to NH3/ NH_4^+. Once fixed as NH3, the N can be incorporated by many organisms as organic N. Certain aerobic and anaerobic microorganisms are capable of fixing N. Abiotic fixation is also accomplished naturally lightning, and by humans through the use of natural gas in the Haber-Bosch process to create N fertilizers and other products.

3.2.2.3 Effectiveness of Treatment for Nutrients in Wetlands

The research literature on constructed treatment wetlands has shown success in reducing nutrients (Kadlec and Wallace 2009, Crumpton et al. 2008, Woltemade 2000). These include examples of constructed treatment wetlands for river diversions that have shown effective nutrient reductions. River diversion treatment wetlands are especially applicable to the Klamath River application.

Relevant examples of constructed treatment wetlands that have shown effective nutrient reductions include the Everglades Construction Project FL (South Florida Water Management District [SFWMD] 2011), Des Plaines River IL (Kadlec 2010), Prado CA (Reilly et al. 2000), Richland Chambers TX (Kadlec et al. 2011), and Olentangy River OH (Mitsch et al. 1998). These are engineered wetlands ranging in size from about 2 to 45,000 acres that provide significant reduction in N and P reduction of river flows. Similarly, wetlands constructed or restored within historic lake littoral zones and perimeter marshes have demonstrated the feasibility of the use of treatment wetlands for treating eutrophic lake water (such as, Lake Apopka in Florida reported by Coveney et al. 2001) and lake inflows (such as, Lake Balaton Hungary as reported by Tátrai et al. 2000, and in China reported by Lu et al.

2009). Such examples, and their relevance to potential use of treatment wetlands in the Klamath Basin, are discussed further below in Chapter 6 (*Relevant Treatment Wetland Case Studies)*.

The performance of operating constructed treatment wetlands reveals the range of effluent quality and the variability of performance possible with these types of systems. Given the wide variety of treatment wetland sizes, configuration, inlet/outlet (I/O) placement, and also the variety of hydraulic and nutrient loadings to which treatment wetlands are subjected, considerable variation in performance occurs. This wide variation in performance clearly indicates that effective removal of nutrients requires careful wetland system design and management.

The removal of mineral N will require ample HRT (approximately 3 to 7 days) for plant uptake and nitrification/denitrification processes to occur. Wetland plants will take up or assimilate N as an important part of their metabolism. Inorganic N forms are transformed by the plant to organic N compounds used for plant structure. During the growing season, there can be a high rate of uptake of N by wetland plants. Estimates of net annual N uptake by emergent wetland plant species vary from 0.5 to 3.3 $gN/m^2/yr$ (Kadlec and Wallace 2009).

In addition to HRT, another important factor that determines the removal rate of nutrients in wetlands is the influent concentrations or loads of these constituents. For example, several studies have shown that higher influent loads result in higher area-specific N removal (kilogram per hectare and year) (EPA 2000a, Kadlec and Wallace 2009, Crites et al. 2006). In cases of high influent loads, the addition of pretreatment of the inflow to the wetland with chemicals (e.g., alum, sulfate) is often considered (EPA 2000a) to augment nutrient removal rates. Ancillary treatment technologies, such as inflow pretreatment, for potential use in treatment wetlands in the Klamath Basin are discussed further below in Chapter 5 (*Potential Supplemental Technologies to Enhance Treatment by Wetlands)*.

A summary of the range in N removal efficiency (by N species) in surface flow treatment wetlands is presented in Table 1. Many of the wetlands listed in Table 1 are designed to treat or polish wastewater, and therefore influent concentrations are considerably higher than found in the Klamath River. Table 2 provides a summary of nutrient removal data from wetlands receiving flow diverted from a river. Influent concentrations are more relevant to Klamath River flows than the comparatively high influent levels in Table 2, which includes a number of wetlands treating secondary wastewater.

TABLE 1
Summary of Annual Nitrogen Removal Data in Surface Flow Treatment Wetlands by Nitrogen Species
Source: Kadlec and Wallace (2009)

	Median Influent (mg/L)	Median Effluent (mg/L)	Load Removed (50th Percent) (g/m^2-yr)	Rate Coefficient (50th Percent) (g/m^2-yr)	Removal Efficiency (%)	Comments
Total N	17.4	10.2	129	12.6	41.4	Data from 116 wetlands
Organic N	10.8	5.7	90	17.3	47.2	Data from 60 wetlands
TKN	32.4	20	207	9.8	38.3	Data from 101 wetlands
NH4-N	15.5	7.3	127	14.7	52.9	Data from 118 wetlands
NO3-N	4	1.4	51	26.5	65.0	Data from 72 wetlands

Notes:
NH4-N - restricted to wetlands receiving > 1 mg/L NH4-N
Organic N - restricted to wetlands receiving > 5 mg/L Organic N
TN - restricted to wetlands receiving > 5 mg/L TN
NO3-N - restricted to wetlands receiving high proportion NO_3
TKN - restricted to wetlands receiving > 5 mg/L TKN

TABLE 2
Summary of Nitrogen and Phosphorus Removal Data in Wetlands Receiving Flow from River Diversions and Other Large Systems
Sources: As Listed in "Reference" Column

Location	Median Influent (mg/L)			Removal Efficiency (%)			Reference
	TP	TN	NO3-N	TP	TN	NO3-N	
Caernarvon Diversion, Mississippi River, Louisiana	0.15	1.9	1.46-2.14	62	44	57	Day et al. 2009
Fourleague Bay, Atchafalaya River, Louisiana	0.13	1.4	0.69	20	43	51	Perez et al. 2011
Richland Chambers Wetlands, Trinity River, Texas	0.97	NR	3.17	45	NR	77	Kadlec et al. 2011
Prado Wetlands, Santa Ana River, California	NR	NR	10	NR	NR	90	OCWD 2008
Des Plaines River, Illinois	NR	NR	2.3	NR	NR	67	Kadlec 2010
Everglades Construction Project, Florida	0.145	NR	NR	72	NR	NR	SFWMD 2010

Notes:
NR=not reported

3.2.3 Water Temperature

Constructed treatment wetland systems have focused mostly on TSS and nutrients as the primary constituents of treatment interest. However, constructed treatment wetlands can be effective in reducing water temperature in certain cases, particularly where the wetland receives discharges that are warmer than natural ambient water temperature conditions. Several factors affect the thermal treatment capacity of a wetland system including hydraulic retention time, emergent vegetation density, climatic conditions, topographic and bank vegetation shading, channel cross section geometry, and influent temperatures. Emergent wetland vegetation provides shading of the water surface to minimize solar heating while radiant heat loss and evaporative cooling help to dissipate energy and reduce wetland effluent temperatures. On average, with 2 days of detention time through a densely vegetated wetland, effluent temperatures can be reduced to approximately average daily air temperatures during the summer months.

The ability of constructed wetlands to effectively reduce effluent temperature has been documented in the Willamette Valley, Oregon. Emond et al. (2007) documented reductions in effluent temperature from treatment wetlands receiving warm discharge from the Salem (Oregon) Wastewater Treatment Plant. Monitoring data indicates that water flowing at a depth of 1 foot through 4.1 acres of treatment wetlands can cool effluent temperatures by up to 5°C in August and up to 10°C in December with 4 to 8 days and nights of detention time (Emond et al. 2007).

Smesrud et al. (2007) similarly documented reductions in effluent temperature from treatment wetlands (known as the “Talking Water Gardens”) receiving warm discharge at the Albany-Millersburg (Oregon) Municipal Water Reclamation Facility (WRF). On average, with 2 nights of detention time through a densely vegetated wetland, effluent temperatures are reduced to approximately average daily air temperatures during the summer months. Water temperature is reduced by as much as 2.8°C in July and August. The heat energy removed from the combined effluent flow is about 80 Mkcal/day in July and August when ambient temperatures are highest, and about 150 million kilo-calories per day in October when ambient temperatures have cooled. During the cooler weather in the fall the temperature of the water discharging from the wetlands is up to 5.6°C cooler than the water from the treatment facilities.

Gregory (2009) reported that a pilot treatment wetland constructed at the City of Woodburn (Oregon) WWTP showed that temperature reduction in the pilot wetland was relatively modest. Monitoring in September indicated that significantly more thermal reduction actually occurred in the WWTP storage lagoon than in the wetland (about 4°C cooling compared to 1.2°C in the wetland). Decreasing the hydraulic retention time from 2.5 to 0.5 days in mid-September did not change the average discharge temperature.

Energy inputs to the wetland include solar radiation, convective heat transfer from the air, vertical and lateral ground heat transfer, and thermal energy from wetland inflows (Kadlec and Wallace 2009, Kadlec 2009). Energy outputs from the wetland include solar back radiation, evapotranspiration (the combination of evaporation and transpiration, or ET), convective heat transfer from the air, vertical and lateral ground heat transfer, and energy exiting in wetland outflows. In wetlands that are above freezing, the surface water temperature is most strongly related to air temperature (Kadlec and Wallace 2009). This relationship largely depends on the relative humidity, which correlates with the rate of energy loss to ET. When relative humidity is around 50 percent, such as often observed in summer, the water temperature is usually driven by atmospheric conditions toward the ambient air temperature (Kadlec and Wallace 2009).

As water enters the wetland at higher or lower temperatures than the wetland system, energy absorbed or lost by the water will be changed until it gradually reaches a balanced or equilibrium temperature (i.e., the water temperature reached where thermal energy absorbed and lost are equal). The initial change in water temperature occurs in the “accommodation zone” of the wetland (a term used by Kadlec and Wallace [2009]). After reaching the balanced temperature of the wetland, the water will no longer continue to change temperature with increased retention times. Data compiled by Kadlec and Wallace (2009) show that the HRT required to overcome the accommodation zone and achieve a balanced temperature is on the order of one to three days. This depends on several factors including the incoming water temperature, water depth, and climatic conditions (Kadlec and

Wallace 2009). Where incoming water is considerably warmer than the wetland water, relative humidity is high, and the ambient air temperature is unusually warm, the HRT necessary to reach balance temperatures is longer.

Whether or not water temperature itself is a focus of treatment, the temperature of incoming and outgoing effluent is of great importance in treatment wetland systems (Kadlec and Reddy 2001, Kadlec and Wallace 2009). One reason for this is that water temperature in wetlands has been shown to strongly influence the rate of microbial processes leading to water quality treatment, such as NH3 N processing (Kadlec and Wallace 2009, Picard et al. 2005, Werker et al. 2002, Faulwetter et al. 2009). This is important for wetland designers trying to both calculate accurate rates of treatment as well as optimize their systems for removal. Wetland systems in very hot or cold climates can be concerned about evaporative losses where water is in short supply (Kadlec 2006) or freezing of the wetland at various times of the year.

Whether or not constructed treatment wetlands would be effective in reducing water temperature in the Upper Klamath basin will depend on various site-specific conditions. As described above in this section, several intrinsic wetland factors (e.g., wetland configuration, vegetation density, or influent temperatures) affect water temperature conditions at the local wetland scale. Other important factors that determine local water temperature conditions also occur at the larger basin-wide scale, such as climatic and hydrologic conditions. These various factors can display considerable variability in the Upper Klamath basin, which can result in equally large variability in wetland temperature conditions.

As an illustration, potential wetland water temperatures were calculated based on the heat budget formulations of Martin and McCutcheon (1999) for various hypothetical wetland depths and meteorological conditions in the Upper Klamath basin. Figure 8 shows calculated maximum, minimum, and mean water temperatures for various hypothetical wetland depths, and the associated range in air temperature (based on hourly observations from Klamath Falls for a representative August day) used to calculate the water temperatures. Data and information supporting these calculations are included in Appendix A.

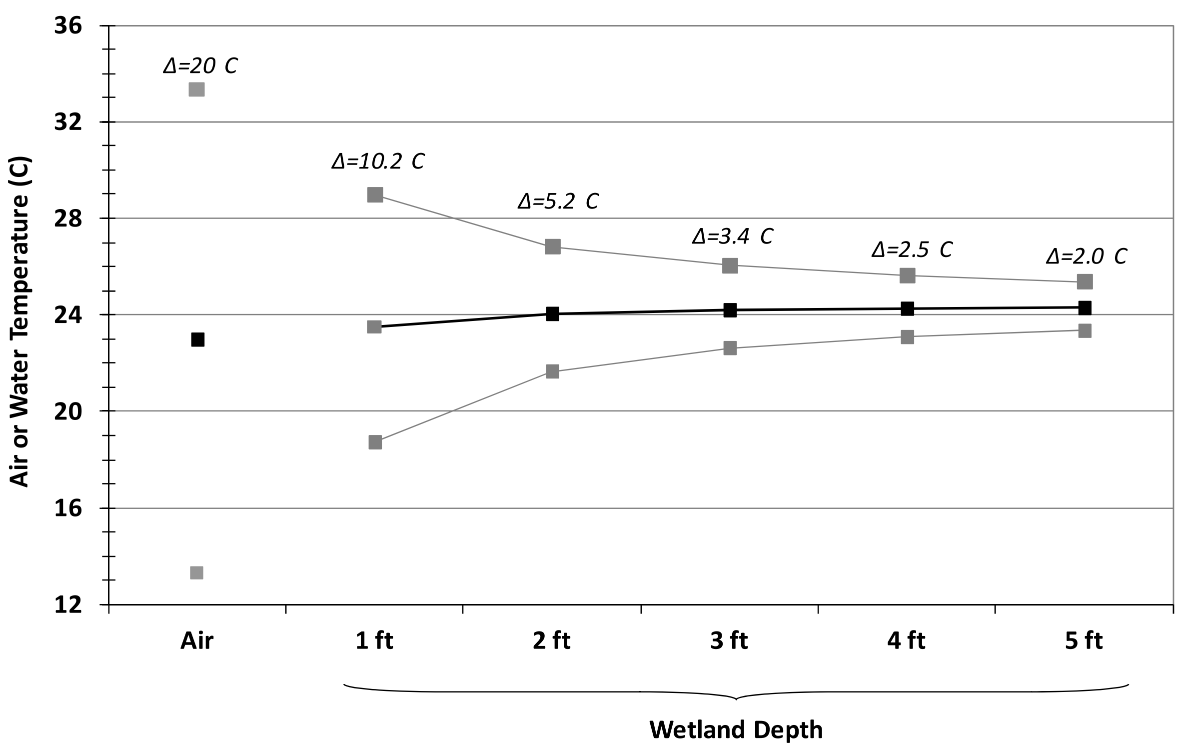

FIGURE 8

Calculated maximum, minimum, and mean water temperatures for various hypothetical wetland depths, and the associated range in air temperature (based on hourly observations from Klamath Falls for a representative August day).

The results shown in Figure 8 indicate that air temperatures vary considerably more than the wetland water temperatures. As wetland depth increases, the water temperature range diminishes, although the mean water temperature varies little. These results are consistent with Kadlec and Knight (2009), who note that diurnal variation of temperatures in wetlands can be considerable, with ranges as large as 8 to 10°C.

Due to such substantial variability, more specifically assessing water temperature effects of wetlands in the Upper Klamath basin is beyond the scope of this analysis. The effects of treatment wetlands on water temperature will require evaluation on a case-by-case basis, including steps to determine site-specific conditions or use of treatment wetlands pilot studies (as discussed further below in section 7.3.5).

CHAPTER 4

Preliminary Estimates of Wetland Area Needed for Treatment in the Upper Klamath Basin

4.1 Background

When considering the potential use of treatment wetlands to reduce nutrient loads in the Klamath River (along with organic matter and total suspended solids), it is important to first determine the approximate magnitude of wetland acres that would be needed to achieve nutrient load reductions and concomitant water quality improvements. This determination would help to frame planning discussion and guide further activities (as described below) to assess locations, configurations, water rights requirements, and types of treatment wetlands (including associated ancillary treatment techniques) that might need to be implemented to achieve desired nutrient reductions.

4.2 Approach

The approximate magnitude of wetland acres corresponding to various levels of treatment performance were estimated using the first-order area-based treatment wetland model developed by Kadlec and Knight (1996), and updated in Kadlec and Wallace (2009), based on developments summarized in Kadlec (2003). The model estimates the potential wetland outflow concentration for a given inflow rate and concentration for a given wetland area, as adjusted for water temperature and system water balance.

Known as the P-k-C* model, the terms are defined as follows:

$$A = \frac{QP}{k} * \left(\frac{(C_e - C^*)}{(C_i - C^*)}\right)^{-\left(\frac{1}{P}\right)}$$

Where:

A = wetland area (square meters [m^2])
Q = flow (cubic meters per year [m^3/yr])
C_i = inflow (influent) concentration (mg/L)
C_e = wetland outflow (effluent) concentration (mg/L)
C^* = wetland equilibrium background concentration (mg/L)
k = first-order, area-based rate constant (m/yr), which varies depending upon pollutant
P = weathering factor that takes into account the estimated number of hydraulic tanks-in-series and the number of component compounds for a particular parameter (dimensionless)

In this model, the value of *k* can vary for temperature-dependent parameters, such as N, and is adjusted using the Arrhenius equation as follows:

$$k = k_{20}\theta^{(T - 20)}$$

Where:

θ = temperature coefficient
T = temperature (degree Celsius [°C])

The model was used to determine the wetland outflow concentrations (in mg/L) associated with increasing wetland area for given inflow concentrations. The model also was used to determine the reduction in wetland outflow concentrations as a percent of given inflow concentrations associated with increasing wetland area.

Two specific river locations were used as focal points for this evaluation. These points represent the terminus of their contributing basin areas. The first location is at Link River dam just below the Upper Klamath Lake (in Oregon), which represents the source of the Klamath River. The second location is at the Klamath River below Iron

Gate dam (in California), which represents the outflow point from the Upper Klamath basin as defined in this report (and shown in Figure 1). These two locations are shown (using star symbols) in Figure 2, which includes a schematic of the contributing subbasins to these locations.

Estimates from the model were derived using a "sensitivity analysis-type" approach. In this manner, ranges in or alternative settings of assumed parameters and inputs were subjected to various model runs or scenarios. This allowed the analysis to depict the variability or uncertainty associated with the assumed parameters and inputs as described further in the next section.

Estimates from the model were produced for the period May through October, which corresponds to the portion of the year when river flows are lowest and primary production conditions are highest in the basin. This period also corresponds to the time of highest concentrations of nutrient and organic matter loads in the river. Estimates assumed average monthly temperature and water flow conditions by month for each location.

4.2.1 Treatment Model Parameter Selection

For the purposes of this wetland modeling analysis, values were selected to represent the specific parameters and rate constants contained in the P-k-C* model equation described above. Under ideal circumstances, data from on-site pilot or demonstration studies are used to parameterize the model through calibration of removal rate constants (*k* values) and the background concentrations (*C** values). However, in the absence of such available data (as in this case), values were selected from the research literature and other available datasets.

Values for the first-order, area-based removal rate constant (*k*) were selected using information from Kadlec and Wallace (2009) for *k* values derived from calibrations to actual performance data from 282 treatment wetland projects. Kadlec and Wallace (2009) indicate that, for a subset of wetlands in colder climates with a seasonal growth cycle such as in the Upper Klamath basin, *k* values for TP exhibit a seasonal pattern. For example, *k* values are relatively high in May through October (range: 15 to 24 m/yr) and relatively low in December through March (range: 7 to 12 m/yr). Based on the Kadlec and Wallace (2009) information, monthly *k* values were used in the modeling of TP for the warmer May-October period when most wetland production would occur. In addition, a single annual value was assumed for modeling of TP to represent the year in whole. Table 3 lists the *k* values used in the modeling analyses for TP.

For TN, a single annual *k* value is assumed for modeling purposes (Table 3). Kadlec and Wallace (2009) report a significant temperature dependence of *k* values for TN. Annual *k* values were assumed for modeling the component forms of TN (Table 3), and were further adjusted by the Arrhenius equation to approximate seasonal changes in performance and converted to a monthly value. For example, the NO3-N k value of 27 m/yr is adjusted using the temperature correction factor in June (16.9°C) to 21.1 m/yr and then divided by 12 to yield the monthly rate constant of 1.8 m/mo.

TABLE 3
Assumed First-Order Area-Based Removal Rate Constants (*k*) in the P-k-C* Model

Constant	Unit	May	June	July	August	September	October	Annual
k (TP)	m/yr	21	19	15	18	20	24	10
k (TN)	m/yr	-	-	-	-	-	-	13
k (Org-N)	m/yr	-	-	-	-	-	-	17
k (NH_3-N)	m/yr	-	-	-	-	-	-	15
k (NO_3-N)	m/yr	-	-	-	-	-	-	27

The background concentration (*C**) for use in the P-k-C* model represents the irreducible background nutrient concentrations of wetland systems when a steady-state balance is reached of atmospheric input, internal cycling, and burial in the absence of external input (Kadlec and Knight 1996). Primary production of wetland materials yields trace amounts of microbial floc, plant fragments, dissolved and colloidal organic compounds, algal cells and related biological matter, all which contain some amount of P and N. Diffusion of P and organic N from flooded soils can occur in response to physical and chemical changes in the water column. Animals can have an effect locally through either disturbance of sediments, consumption and cycling of plant matter, or modification of the hydraulics leading to short-circuiting. The sum of these site-specific processes yields an effective lower limit to performance in treatment wetlands, which may be different for lakes and flowing water bodies.

Values of the wetland equilibrium background concentration ($C*$) for use in this analysis were selected based upon review of minimum concentrations from existing seasonal data sets for the two specific Klamath River locations used as focal points for this evaluation. For the purposes of this wetland modeling analysis, two background levels of TP were selected – 0.002 mg/L and 0.02 mg/L – which bracket minimum concentrations observed at the Link River location in Oregon and the Klamath River below Iron Gate dam in California. These values are lower than minimum TP values observed for Williamson River Delta wetlands (0.05 mg/l to 0.1 mg/L), but are similar in range to observed minimum values for SRP (0.00 to 0.04 mg/L) (TNC 2011). Emergent marsh wetlands may only be expected to achieve these P concentrations under extremely low hydraulic and mass loading conditions (i.e., very large areas), as demonstrated later in this report and as seen in the Everglades Stormwater Treatment Areas (SFWMD 2011). However, these values are typical of the range used as a basis for conceptual wetland sizing.

For TN and its component organic N (Org-N), two background levels were selected: 0.5 mg/L and 1.0 mg/L. A review of the surface water quality data for N at the Link River location in Oregon and the Klamath River below Iron Gate dam in California indicated that 1.0 mg/L would be a reasonable assumption for minimum observed concentration levels. Kadlec and Wallace (2009) report a lower end of $C*$ values for TN and Org-N of about 0.5 mg/L. Lower values may possibly occur but cannot be assumed without specific pilot data or other wetland performance data.

For NH_3-N, a $C*$ value of zero was assumed for use in this analysis. NH_3-N values are below detection limits of 0.01 mg/L from May-July in the Williamson River delta in Oregon (TNC 2011). Actual measured background concentrations can be expected to vary in response to hydraulic loading and the particular form of N prevalent in a wetland. For conceptual modeling purposes, zero is commonly selected for this parameter (Kadlec and Wallace 2009). Kadlec and Wallace (2009) indicate that treatment wetland systems can be presumed to have a theoretical background of zero for NH_3-N. For example, a similar application of calibrating and sizing a riverine treatment wetland system on the St. Johns River assumed zero as the background concentration parameter (National Research Council 2012).

A $C*$ value of zero also was assumed for use in this analysis for NO_3-N. For conceptual modeling purposes, zero is commonly selected for this parameter (Kadlec and Wallace 2009). Similar to NH_3-N, background NO_3-N concentrations can theoretically decrease to zero, in the absence of external inputs (Kadlec and Wallace 2009). NO_3-N concentrations in wetlands within the Williamson River Delta in Oregon decrease to values less than the method detection limit of 0.01 mg/L in May-July (TNC 2011). However, measured apparent background will vary in response to hydraulic loading and prevalent N form and concentration, as well as seasonally in response to temperature.

The Arrhenius factors for the different N forms, taken from Kadlec and Wallace (2009), were 1.05, 1.05, and 1.11 for Org-N, NH_3-N, and NO_3-N, respectively. TN was estimated using the transformation equations of Kadlec and Wallace (2009), assuming a sequential mineralization of Org-N, nitrification of NH_3-N, and denitrification of NO_3-N to N gas. Table 4 summarizes the monthly temperature values used to adjust the rate constants.

TABLE 4
Assumed Water Temperatures (*T*) in the P-k-C* Model

Location	Unit	May	June	July	August	September	October	Annual
Link Site	°C	10.5	14.6	19.4	18.4	14.2	8.3	8.1
Iron Gate Site	°C	13.9	17.9	23.4	21.7	17.3	10.9	11.1

4.2.2 Assumed Influent Water Quality and Flow Conditions

For the purposes of this wetland modeling analysis, inflow (influent) water quality and flow conditions were assumed to represent treatment wetland influent conditions in the P-k-C* model equation described above. To represent assumed influent water quality concentrations at the Link River location in the wetlands treatment model, water quality data were used from PacifiCorp's sampling location KR 25312 on the Klamath River below Upper Klamath Lake for the months of May to October in years 2000 to 2007[2]. To represent flows at this location, average monthly flow data were used from the USGS 11507500 gaging station for Link River at Klamath Falls, Oregon in years 2000 to 2007. The resulting average seasonal and monthly water quality data for TP, TN, and flow are summarized in Table 5. The values of other water quality parameters used in the analysis are presented in Appendix B, Table A-1.

For the modeling analysis at the Link River location, scenarios were run assuming all (100 percent) of the river flow is diverted into and through (and therefore receive treatment by) the modeled wetland areas. However, it is not considered realistic to expect that 100 percent of the river's flow would be diverted into treatment wetlands in actual practice. Also, it is likely not possible to deliver a monthly average flow to the wetland consistently on a daily basis, due to variable flow conditions within the Upper Klamath basin system. Therefore, scenarios were also run assuming half (50 percent) of the river flow is diverted into and through (and therefore receive treatment by) the modeled wetland areas. These 50 and 100 percent flow values provide a conservative "side board" of the higher end of wetlands effectiveness in treating river flows. Users of this report are invited to estimate corresponding values of wetlands areas for lesser flows by estimating an approximate proportional reduction. For example, the wetlands areas for an assumed flow through the wetlands that is 10 or 20 percent of the full (100 percent) river flow would be approximately 10 or 20 percent of the model's estimates of wetlands area for full (100 percent) river flow scenarios.

TABLE 5
Assumed Water Quality and Flow Conditions for the Link River Location Below Upper Klamath Lake

Parameter	Unit	Seasonal Mean (May-October)	May	Jun	Jul	Aug	Sep	Oct
TP	mg/L	0.19	0.16	0.19	0.28	0.18	0.15	0.15
TN	mg/L	2.32	0.91	1.53	2.44	4.00	2.74	2.32
Flow	cfs	1,097	1,660	1,500	1,020	941	679	781

To represent assumed influent water quality concentrations at the Klamath River below Iron Gate dam in the wetlands treatment model, water quality data were used from PacifiCorp's sampling location KR 190 (located at the Iron Gate Hatchery Bridge) for the months of May to October in years 2000 to 2007[2]. To represent flows at this location, average monthly flow data from the USGS 11516530 gaging station on the Klamath River below Iron

[2] Source: Klamath Water Quality Data 2000-2003, 2004, 2005, 2007 available at http://www.pacificorp.com/es/hydro/hl/kr.html#.

Gate dam in California in years 2000 to 2007 were used. The resultant average seasonal and monthly water quality data for TP, TN, and flow are summarized in Table 6. The values of other water quality parameters used in the analysis are presented in Appendix B, Table A-2.

For the modeling analysis at the Klamath River below Iron Gate dam location, scenarios were run assuming half (50 percent) of the river flow is diverted into and through (and therefore receive treatment by) the modeled wetland areas. Full (100 percent) flow values were not modeled at this location (unlike the Link River location) for two main reasons. First, 50 percent flow values are considered sufficient to provide a conservative estimate of the maximum effectiveness of wetlands in treating river flows at this location. Second, the modeling of both 50 and 100 percent flow values at the upstream Link River site are already considered sufficient to assess the relative treatment effectiveness at differing flow levels.

TABLE 6
Assumed Water Quality and Flow Conditions for the Klamath River Location Below Iron Gate Dam

Parameter	Unit	Seasonal Mean (May-October)	May	Jun	Jul	Aug	Sep	Oct
TP	mg/L	0.16	0.16	0.18	0.16	0.15	0.19	0.12
TN	mg/L	1.18	0.67	1.10	1.38	1.48	1.24	1.20
Flow	cfs	1,340	2,280	1,540	969	942	1,070	1,240

4.2.3 Modeling Scenarios

As discussed above, estimates from the P-k-C* model were derived using a "sensitivity analysis-type" approach to portray a range of scenarios that depict the variability or uncertainty associated with assumed modeling parameters and inputs. A total of 20 scenarios were run to produce estimates from the P-k-C* model of wetland acreages to treat the contributing basin area to the Link River location below Upper Klamath Lake. Table 7 summarizes the model parameters for each scenario, including the percentage of river flow assumed to be treated, a brief description of the scenario, and the comparisons possible with other model scenarios to evaluate the relative sensitivity of parameter value selection. Scenarios 1A through 4L include model runs related to TP, in which various differences or ranges in *k* values, *C** values, and flow are assumed. Scenarios 1A through 3 are model runs based on the annual *k* value for TP (of 10 m/yr) that differ according to *C** values (i.e., assuming either 0.002 or 0.020 mg/L) or flow (i.e., assuming either 50 or 100 percent of flow is treated). Scenarios 4A through 4F are based on the month-by-month *k* values for May through October (per Table 3) at the *C** value of 0.002 mg/L, and scenarios 4G through 4L are based on the same monthly *k* values at the higher *C** value of 0.020 mg/L. The model outputs for scenarios 1A through 4L are contained in Appendix C.

Scenarios 5A through 6B include model runs related to TN based on the annual *k* value for TN and its component N forms (per values in Table 3) that differ according to *C** values (i.e., assuming either 0.5 or 1.0 mg/L) or flow (i.e., assuming either 50 or 100 percent of flow is treated). The model outputs for scenarios 5A through 6B are contained in Appendix C.

A total of 6 scenarios were run to produce estimates from the P-k-C* model of wetland acreages to treat the contributing basin area to the Klamath River at Iron Gate dam location (Table 8). Fewer scenarios were run for this location because the overall number of scenarios, including for the Link River location, are sufficient to assess relative differences or "sensitivity" of the P-k-C* model to changes in assumed *k* values, *C** values, and flow.

Scenarios 7 through 10 include model runs related to TP, in which differences in *k* values are assumed, including the annual *k* value and the monthly *k* values for July and October, which bracket the high and low ends of the range of *k* values for the May-October period. Scenario 7 assumes the *C** value of 0.002 mg/L and scenario 8 assumes the higher *C** value of 0.020 mg/L. The model outputs for scenarios 7 through 10 are contained in

Appendix D. Scenarios 11 and 12 include model runs related to TN based on the annual *k* value for TN and its component N forms (per values in Table 3) that differ according to *C** values (i.e., assuming either 0.5 or 1.0 mg/L). The model outputs for scenarios 11 through 12 are contained in Appendix D.

TABLE 7

Modeling Scenarios for Calculations of Wetlands Area in the Upper Klamath Basin Area Draining to the Link River Location

Scenario Number	TP		TN	Percent Flow	Scenario Descriptor[1]	Comparisons[2]			
	k	*C**	*C**			*C**	*k*	Flow	Area[3]
1A	10	0.020	-	50	TP, Annual k, C* .02, Q50	2		1B	7
1B	10	0.020	-	100	TP, Annual k, C* .02, Q50	3		1A	
2	10	0.002	-	50	TP, Annual k, C* .002, Q50	1A		3	8
3	10	0.002	-	100	TP, Annual k, C* .002, Q100	1B		2	
4A-4F	15-24	0.002	-	50	*(C* .002, Q50 By Month)*	4G-4L	4A-4F		9-10
4G-4L	15-24	0.02	-	50	*(C* .02, Q50 By Month)*	4A-4F	4G-4L		
5A	-	-	1.0	50	TN, Annual k, C* 1.0, Q50	6A		5B	11
5B	-	-	1.0	100	TN, Annual k, C* 1.0, Q100	6B		5A	
6A	-	-	0.5	50	TN, Annual k, C* 0.5, Q50	5A		6B	12
6B	-	-	0.5	100	TN, Annual k, C* 0.5, Q100	5B		6A	

[1] This descriptor is used in graphs of model results presented in section 4.3.

[2] The results from scenarios listed in these Comparisons columns can be compared to the scenario listed in the first column (Scenario Number) to assess relative differences or “sensitivity”.

[3] The scenario numbers listed in the Area column are for scenarios related to the Upper Klamath basin area draining to the Klamath River at Iron Gate location as presented in Table 8.

TABLE 8

Modeling Scenarios for Calculations of Wetlands Area in the Upper Klamath Basin Area Draining to the Klamath River at Iron Gate Location

Scenario Number	TP		TN	Percent Flow	Scenario Descriptor[1]	Comparisons[2]			
	k	*C**	*C**			*C**	*k*	Flow	Area[3]
7	10	0.020	-	50	TP, Annual k, C* .02, Q50	8			1A
8	10	0.002	-	50	TP, Annual k, C* .002, Q50	7			2
9	15	0.002	-	50	TP, July k, C* .002, Q50		10		4C
10	24	0.002	-	50	TP, Oct k, C* .002, Q50		9		4F
11	-	-	1.0	50	TN, Annual k, C* 1.0, Q50	12			5A
12	-	-	0.5	50	TN, Annual k, C* 0.5, Q50	11			6A

[1] This descriptor is used in graphs of model results presented in section 4.3.

[2] The results from scenarios listed in these Comparisons columns can be compared to the scenario listed in the first column (Scenario Number) to assess relative differences or "sensitivity".

[3] The scenario numbers listed in the Area column are for scenarios related to the Upper Klamath basin area draining to the Link River location as presented in Table 7.

4.3 Results: Model Estimates Nutrient Reduction as a Function of Wetlands Area

4.3.1 Link River Location (Source of the Klamath River)

4.3.1.1 Phosphorus

Calculated TP concentrations (mg/L) as a function of wetlands area (ac) in the Upper Klamath basin area draining to the Link River location are shown in Figures 9 through 12. Figure 9 shows the results for scenarios 1A through 3, which all assume the annual *k* value for TP of 10 m/yr (per Table 3). These scenarios differ according to *C** values (i.e., assuming either 0.002 or 0.020 mg/L) or flow (i.e., assuming either 50 or 100 percent of flow is treated).

The curves shown in Figure 9 conform to first-order expectations whereby calculated TP concentrations approach a baseline horizontal asymptote (defined by the *C** values) as wetland area increases. As such, these curves indicate that progressively more wetlands area would be required to achieve successive increments of reduction in TP concentration. For example, under scenario 1B, a 50 percent reduction (from the initial concentration of 0.19 mg/L to 0.095 mg/L) corresponds to a treatment wetlands area of approximately 20,000 acres (Figure 9). By comparison, an 80 percent reduction to 0.035 mg/L (i.e., an additional 30 percent reduction) corresponds to a wetlands area of approximately 90,000 acres (Figure 9). Wetlands areas in the range of 20,000 to 40,000 acres appear to be an inflection point of TP reduction for the scenario 1B curve (and the other scenario curves) shown in Figure 9. These curves suggest that a treatment wetland sizing strategy for the Upper Klamath basin may be to maximize efficiency (and minimize the point of diminishing returns) of nutrient reduction such as indicated in these results, even if that may mean accepting a lesser "target" concentration reduction.

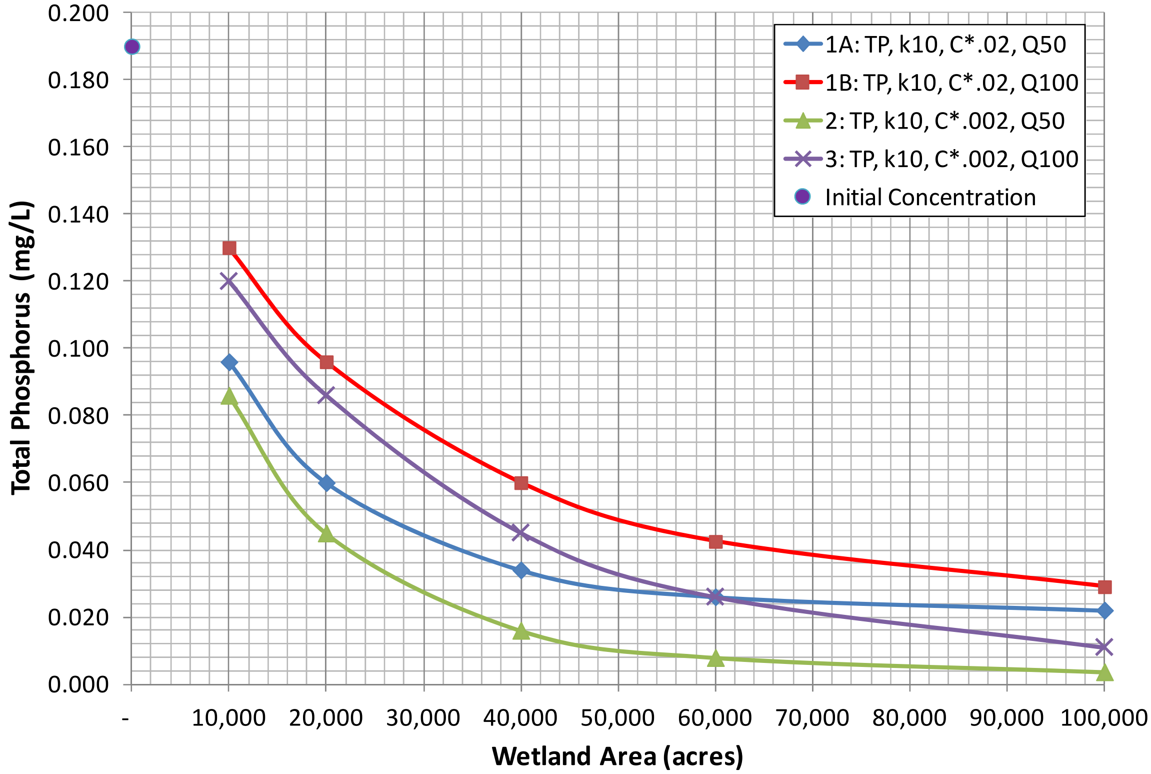

FIGURE 9
Calculated TP concentrations (mg/L) as a function of wetlands area (acres) in the Upper Klamath basin area draining to the Link River location for scenarios 1A, 1B, 2, and 3. The Initial Concentration (shown on the y-axis) represents the "untreated" wetland influent (inflow) concentration.

Comparison of the curves for scenarios 1A and 1B (Figure 9)[3] indicate the effects of assuming treatment of 100 percent of flow versus 50 percent of flow. Comparison of the curves for scenarios 2 and 3 (Figure 9) also indicate the effects of the differing flow assumptions. These comparisons indicate that the scenarios which assume 50 percent of flow have lower TP concentrations than the scenarios assuming 100 percent flow throughout the modeled range of wetlands area. For example, estimated TP concentration for scenario 1A is 0.032 mg/L for a treatment wetlands area of 40,000 acres, which is equal to about an 83 percent reduction from the initial concentration of 0.19 mg/L. By comparison, estimated TP concentration for scenario 1B is 0.060 mg/L for a wetlands area of 40,000 acres, which is a 68 percent reduction from the initial concentration. In addition, the scenarios which assume 50 percent of flow have slightly steeper curves, particularly in the range between 10,000 and 50,000 acres. These model results indicate that, for a given wetlands size, better reduction efficiency would occur at lower flow levels, which would be expected if lower flows result in greater hydraulic residence times, and thus allow more time for wetland processes to provide treatment.

Although the model calculations indicate better reduction efficiencies at lower flow levels, it is important to recognize that this reduction efficiency pertains specifically to wetlands effluent (outflow) and not necessarily to the river after wetlands outflow is received and mixed. The assumed treatment of 100 percent of flow in scenarios 1B and 3 means that the wetlands outflow and river concentrations would be the same since the wetlands are assumed to be treating the entire river flow. As previously discussed, treating the entire river flow is likely an unrealistic assumption for actual implementation in the Upper Klamath basin, but reasonable to include in this modeling analysis for the purpose of bounding treatment wetlands area relationships. By contrast, the assumed

[3] In Figure 8 (and subsequent similar figures of model results in this chapter), the Initial Concentration (shown on the y-axis) represents the "untreated" wetland influent (inflow) concentration. Given the high implied hydraulic loading rate of wetlands areas less than 10,000 acres for these flow rates, the curve lines are not extended between the Initial Concentration and the first modeled point (at 10,000 acres) due to uncertainty over the probable trend of such an extended line at lower wetland area levels less than 10,000 acres.

treatment of 50 percent of flow in scenarios 1A and 2 means that the wetlands are assumed to be treating half the river flow, with the other half remaining in the river at initial (inflow) "untreated" concentrations.

Figure 10 compares the wetland effluent (outflow) curves for scenarios 1A and 1B with a third curve representing the TP concentrations calculated for scenario 1A when remixed with an equal (50 percent) fraction of the initial (inflow) "untreated" TP concentration of 0.19 mg/L. As discussed above, estimated TP concentration of treatment wetlands effluent for scenario 1A is 0.032 mg/L for a treatment wetlands area of 40,000 acres, which is equal to about an 83 percent reduction from the initial concentration of 0.19 mg/L. By comparison, the third curve in Figure 10 indicates that estimated TP concentration in the river would be 0.096 mg/L when the effluent is subsequently mixed back in with the river, which is equal to about a 49 percent reduction from the assumed "untreated" TP concentration of 0.19 mg/L. Therefore, although a given area of treatment wetlands would likely have better reduction efficiencies at lower flow levels, these better reduction efficiencies would be offset by the proportion of total river flow that is not subject to wetlands treatment.

Using the approach for estimating the third curve in Figure 10, Table 9 provides calculated "river mixture" TP concentrations and percent-reductions for various levels of treatment wetlands area under scenarios 1A and 2 (which both assume a 50-percent treatment flow fraction). The percent-reduction values in Table 9 indicate that percent-reductions are in proportion to the treatment flow fraction. For example, wetland outflow (effluent) TP concentration for scenario 1A would be reduced by 40 percent for a treatment wetlands area of 10,000 acres. When subsequently mixed into the river, the estimated TP concentration in the river would be reduced by 20 percent from the assumed "untreated" TP concentration of 0.19 mg/L. Likewise, the percent-reduction values achieved in the "river mixture" for other levels of treatment wetlands area under scenarios 1A and 2 are about half (or 50 percent) of the percent-reduction values in the wetland outflow (effluent).

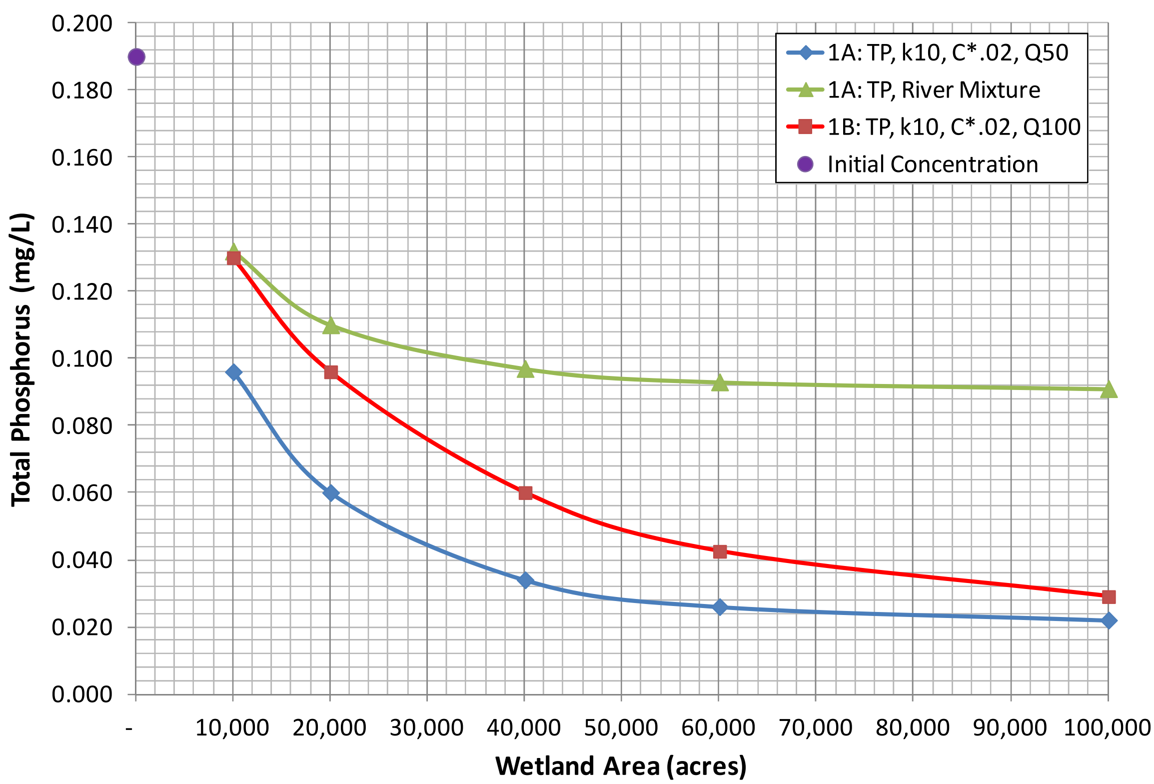

FIGURE 10

Calculated TP concentrations (mg/L) as a function of wetlands area (acres) in the Upper Klamath basin area draining to the Link River location for scenarios 1A and 1B, and the calculated River Mixture for scenario 1A. The Initial Concentration (shown on the y-axis) represents the "untreated" wetland influent (inflow) concentration.

TABLE 9

Calculated Total Phosphorus (TP) Concentrations (mg/L) and Reductions (%) as a Function of Wetlands Area (ac) in the Upper Klamath Basin Area Draining to the Link River Location for Scenarios 1A and 2.

Scenario	Flow Proportion	Assumed C*	Wetland Area (ac)	TP in Wetland Outflow		TP in River Mixture	
				mg/L	% Reduction	mg/L	% Reduction
Initial Value			*0*	*0.19*			
	0.5	0.020	10,000	0.096	40%	0.132	20%
	0.5	0.020	20,000	0.060	63%	0.110	31%
1A	0.5	0.020	40,000	0.034	79%	0.097	39%
	0.5	0.020	60,000	0.026	84%	0.093	42%
	0.5	0.020	100,000	0.022	86%	0.091	43%
	0.5	0.002	10,000	0.086	46%	0.123	23%
	0.5	0.002	20,000	0.045	72%	0.103	36%
2	0.5	0.002	40,000	0.016	90%	0.088	45%
	0.5	0.002	60,000	0.008	95%	0.084	48%
	0.5	0.002	100,000	0.004	98%	0.082	49%

Figure 11 shows the wetland effluent (outflow) curves for scenarios 4A through 4F based on the monthly *k* values for May through October (per Table 3) at the *C** value of 0.002 mg/L. Figure 12 shows the wetland effluent (outflow) curves for scenarios 4G through 4L based on the same monthly *k* values at the higher *C** value of 0.020 mg/L. These curves illustrate that treatment wetlands would have better reduction efficiencies during May through October (as compared to curves assuming annual k values as shown in Figures 9 and 10). These better efficiencies are expected since wetland production processes (that act to transform and/or sequester nutrients) would be greatest during the May-October period. These sets of monthly curves also converge toward the two background levels of TP assumed in the analysis – 0.002 mg/L in Figure 11 and 0.02 mg/L in Figure 12 – for wetlands areas exceeding about 40,000 acres. In other words, these results suggest that treatment wetlands areas exceeding about 40,000 acres would provide little or no additional nutrient reduction benefits during the months of greatest wetland production.

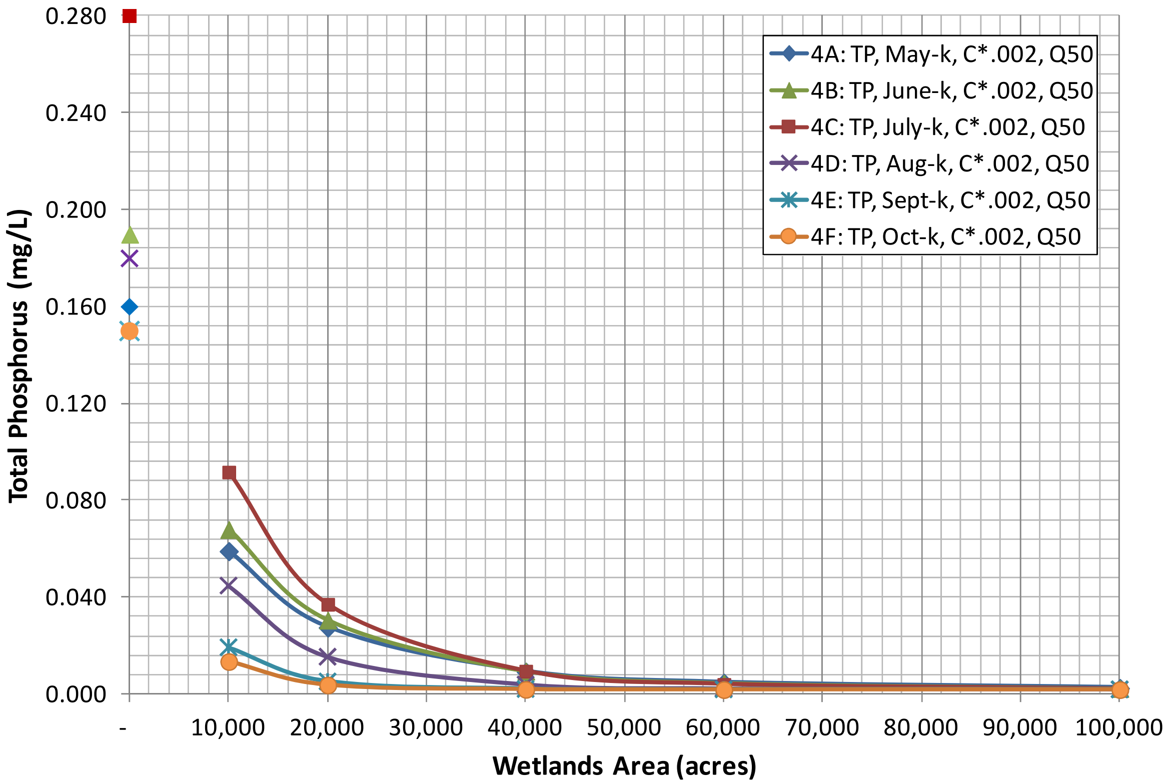

FIGURE 11

Calculated TP concentrations (mg/L) as a function of wetlands area (acres) in the Upper Klamath basin area draining to the Link River location for scenarios 4A through 4F (assuming a C* value of 0.002 mg/L). Initial Concentrations (shown on the y-axis) represent the "untreated" wetland influent (inflow) concentrations.

Overall, the curves in Figures 9 through 12 demonstrate that the results of this modeling analysis are sensitive to changes in assumed *k* and *C** values, particularly in the range of treatment wetlands areas up to about 40,000 acres. As such, these results suggest that an important future step would be to determine more site-specific *k* and *C** values for potential treatment wetlands in the Upper Klamath basin, particularly in areas where wetland construction may be likely. This would help account for site-specific factors such as surface water-groundwater interactions. The use of treatment wetlands pilot studies for such determinations is discussed further below in section 7.3.5.

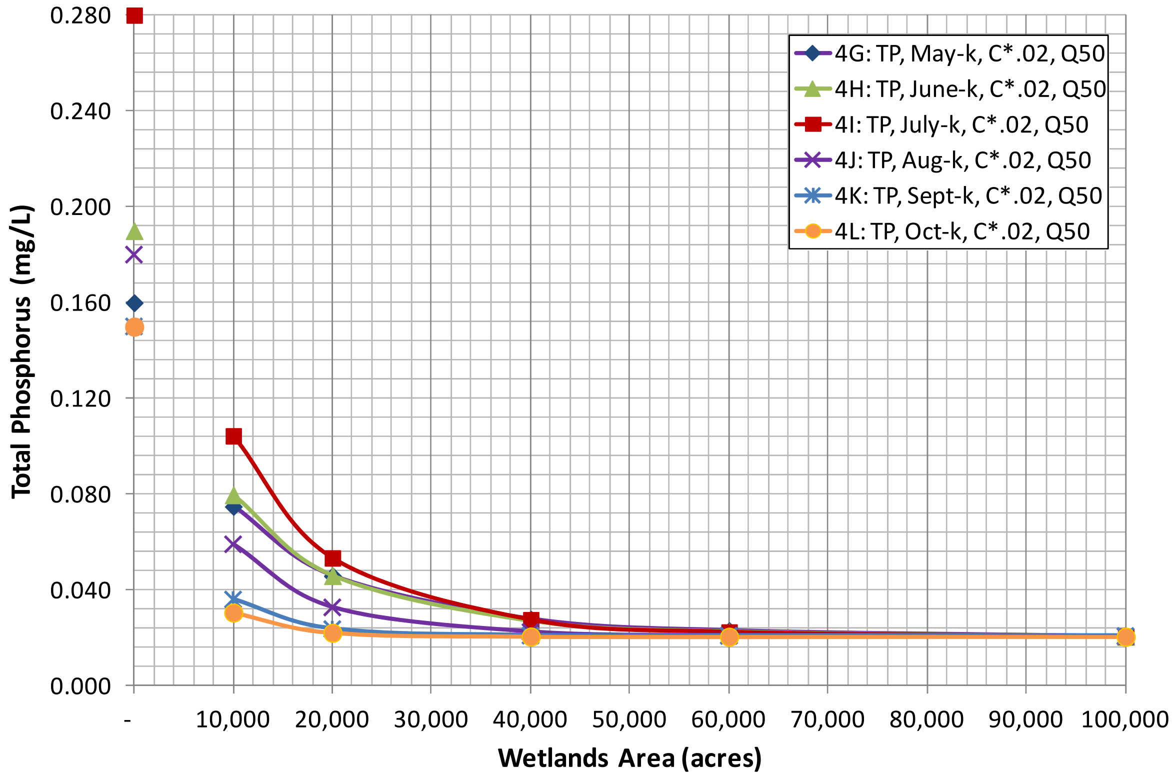

FIGURE 12
Calculated TP concentrations (mg/L) as a function of wetlands area (acres) in the Upper Klamath basin area draining to the Link River location for scenarios 4G through 4L (assuming a C* value of 0.020 mg/L). Initial Concentrations (shown on the y-axis) represent the "untreated" wetland influent (inflow) concentrations.

4.3.1.2 Nitrogen

Calculated TN concentrations (mg/L) as a function of wetlands area (ac) in the Upper Klamath basin area draining to the Link River location are shown in Figure 13 based on the results for scenarios 5A through 6B. These four scenarios all assume the same *k* values for TN and component N forms as listed in Table 3, but differ according to *C** values (i.e., assuming either 0.5 or 1.0 mg/L) or flow (i.e., assuming either 50 or 100 percent of flow is treated).

As with TP discussed above, the TN curves shown in Figure 13 follow first-order shapes whereby calculated TN concentrations approach a baseline horizontal asymptote (defined by the *C** values) as wetland area increases. As such, these curves indicate that progressively more wetlands area would be required to achieve successive increments of reduction in TN concentration. Wetlands areas of about 40,000 to 60,000 acres appear to be inflection points of TN reduction for the curves shown in Figure 13. As with TP discussed above, these TN curves suggest that a treatment wetland sizing strategy for the Upper Klamath basin may be to minimize the point of diminishing returns of nutrient reduction beyond such inflection points, even if that may mean accepting a lesser "target" concentration reduction.

Comparison of the curves in Figure 13 indicate that the scenarios which assume 50 percent of flow (scenarios 5A and 6A) have lower TN concentrations than the other scenarios assuming 100 percent flow throughout the modeled range of wetlands area. In addition, the scenarios which assume 50 percent of flow have slightly steeper curves, particularly in the range between 10,000 and 40,000 acres. As with TP discussed above, these model results indicate that, for a given wetlands size, better TN reduction efficiency in wetland effluent (outflow) would occur at lower flow levels (which would allow relatively more time for wetland processes to provide treatment). However, with regard to ultimate effects on TN concentrations in the river, these better reduction efficiencies at lower flow levels would be offset by the proportion of total river flow that is not subject to wetlands treatment.

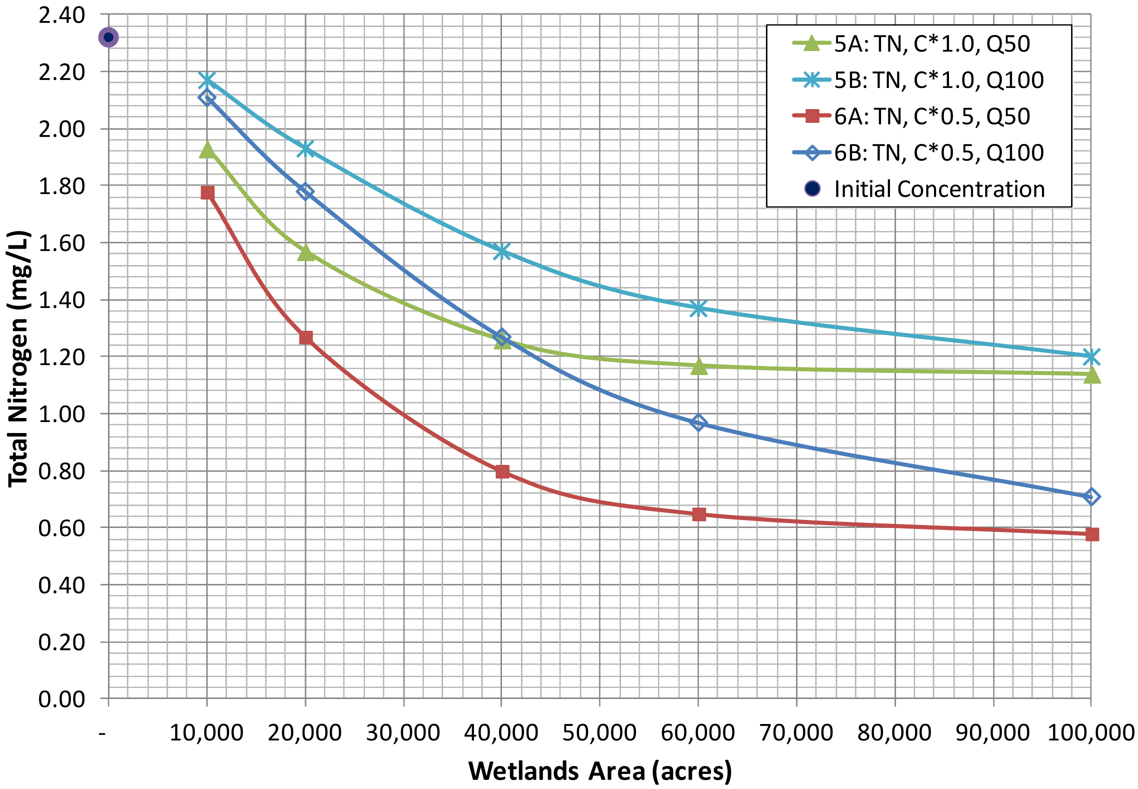

FIGURE 13

Calculated TN concentrations (mg/L) as a function of wetlands area (acres) in the Upper Klamath basin area draining to the Link River location for scenarios 5A through 6B. The Initial Concentration (shown on the y-axis) represents the "untreated" wetland influent (inflow) concentration.

Table 10 provides calculated "river mixture" TN concentrations and percent-reductions for various levels of treatment wetlands area under scenarios 5A and 6A (which both assume a 50-percent treatment flow fraction). As with TP discussed above, the percent-reduction values of TN in Table 10 indicate that percent-reductions are approximately in direct proportion to the treatment flow fraction. That is, the percent-reduction values achieved in the "river mixture" for all levels of treatment wetlands area under scenarios 5A and 6A are about half (or 50 percent) of the percent-reduction values in the wetland outflow (effluent).

4.3.2 Klamath River below Iron Gate Dam (Outflow from Upper Klamath Basin)

4.3.2.1 Phosphorus

Calculated TP concentrations (mg/L) as a function of wetlands area (ac) in the Upper Klamath basin area draining to the Klamath River below Iron Gate dam location are shown in Figure 14 based on the results for scenarios 7 and 8, which assume *C** values of 0.002 or 0.020 mg/L, respectively. As discussed in section 4.2.2 above, both scenarios 7 and 8 were run assuming half (50 percent) of the river flow is diverted into and through (and therefore receive treatment by) the modeled wetland areas.

As with TP curves discussed in section 4.3.1.1 above for the Link River location, the curves shown in Figure 14 for the Iron Gate location follow first-order shapes whereby calculated TP concentrations approach a baseline horizontal asymptote (defined by the *C** values) as wetland area increases. As such, these curves indicate that progressively more wetlands area would be required to achieve successive increments of reduction in TP concentration. A wetlands area of about 40,000 acres appears to be approximate inflection points of TP reduction for the curves shown in Figure 14. As with TP curves discussed above for the Link River location, these curves tend to support a treatment wetland sizing strategy for the basin that would minimize the point of diminishing returns of nutrient reduction beyond such an inflection point.

TABLE 10
Calculated Total Nitrogen (TN) Concentrations (mg/L) and Reductions (%) as a Function of Wetlands Area (ac) in the Upper Klamath Basin Area Draining to the Link River Location for Scenarios 5A and 6A.

Scenario	Flow Proportion	Assumed C*	Wetland Area (ac)	TN in Wetland Outflow		TN in River Mixture	
				mg/L	% Reduction	mg/L	% Reduction
Initial Value			*0*	*2.32*			
5A	0.5	1.0	10,000	1.93	17%	2.13	8%
	0.5	1.0	20,000	1.57	32%	1.95	16%
	0.5	1.0	40,000	1.26	46%	1.79	23%
	0.5	1.0	60,000	1.17	50%	1.75	25%
	0.5	1.0	100,000	1.14	51%	1.73	25%
6A	0.5	0.5	10,000	1.78	23%	2.05	12%
	0.5	0.5	20,000	1.27	45%	1.80	23%
	0.5	0.5	40,000	0.80	66%	1.56	33%
	0.5	0.5	60,000	0.65	72%	1.49	36%
	0.5	0.5	100,000	0.58	75%	1.45	38%

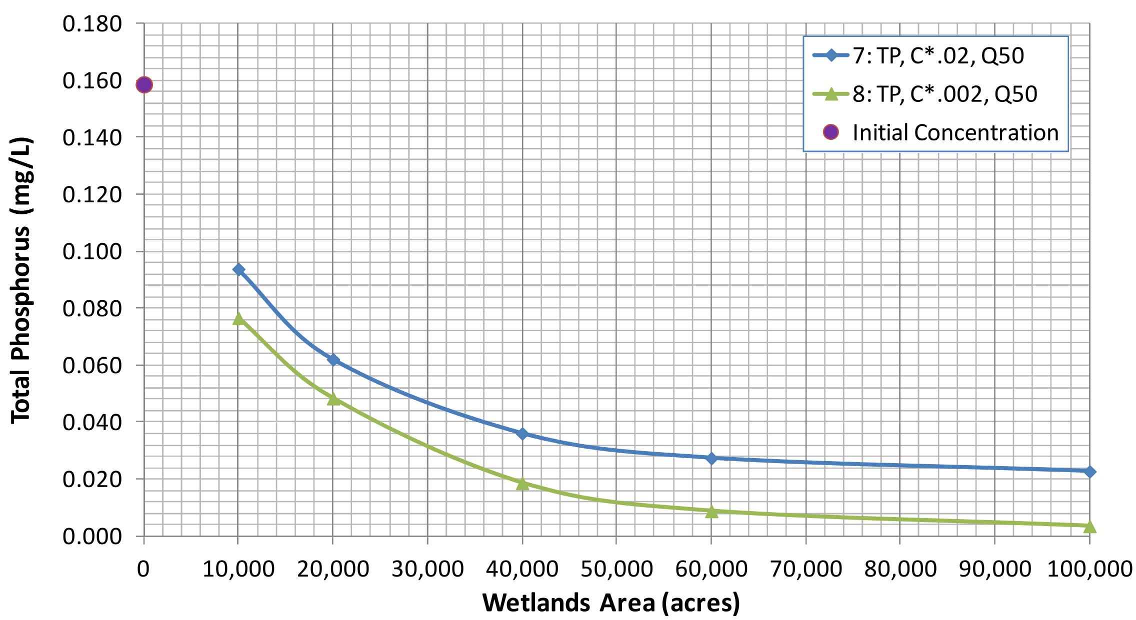

FIGURE 14
Calculated TP concentrations (mg/L) as a function of wetlands area (acres) in the Upper Klamath basin area draining to the Iron Gate location for scenarios 7 and 8. The Initial Concentration (shown on the y-axis) represents the “untreated” wetland influent (inflow) concentration.

Figure 14 also includes "river mixture" curves representing the TP concentrations calculated for scenarios 7 and 8 when remixed with an equal (50 percent) fraction of the initial (inflow) "untreated" TP concentration of 0.16 mg/L. For example, estimated TP concentration of treatment wetlands effluent for scenario 7 is 0.062 mg/L for a treatment wetlands area of 20,000 acres, which is equal to about a 61 percent reduction from the initial concentration of 0.16 mg/L. By comparison, the estimated TP concentration in the river would be 0.110 mg/L when the effluent is subsequently mixed back in with the river, which is equal to about a 30 percent reduction from the assumed "untreated" TP concentration of 0.19 mg/L. Therefore, although a given area of treatment wetlands would likely have better reduction efficiencies at lower flow levels, these better reduction efficiencies would be offset by the proportion of total river flow that is not subject to wetlands treatment.

Table 11 provides calculated "river mixture" TP concentrations and percent-reductions for various levels of treatment wetlands area under scenarios 7 and 8 (which both assume a 50-percent treatment flow fraction). As with TP discussed above for the Link River location, the percent-reduction values of TP in Table 11 for the Iron Gate location indicate that the percent-reduction values achieved in the "river mixture" for all levels of treatment wetlands area under scenarios 7 and 8 are about half (or 50 percent) of the percent-reduction values in the wetland outflow (effluent).

TABLE 11

Calculated Total Phosphorus (TP) Concentrations (mg/L) and Reductions (%) as a Function of Wetlands Area (ac) in the Upper Klamath Basin Area Draining to the Iron Gate Location for Scenarios 7 and 8.

Scenario	Flow Proportion	Assumed C*	Wetland Area (ac)	TP in Wetland Outflow		TP in River Mixture	
				mg/L	% Reduction	mg/L	% Reduction
Initial Value			*0*	*0.16*			
7	0.5	0.020	10,000	0.094	41%	0.126	20%
	0.5	0.020	20,000	0.062	61%	0.110	30%
	0.5	0.020	40,000	0.036	77%	0.097	39%
	0.5	0.020	60,000	0.027	83%	0.093	41%
	0.5	0.020	100,000	0.023	86%	0.091	43%
8	0.5	0.002	10,000	0.077	52%	0.118	26%
	0.5	0.002	20,000	0.048	69%	0.104	35%
	0.5	0.002	40,000	0.019	88%	0.089	44%
	0.5	0.002	60,000	0.009	94%	0.084	47%
	0.5	0.002	100,000	0.004	98%	0.081	49%

Figure 15 shows the wetland effluent (outflow) curves for scenarios 9 and 10 based on the monthly *k* values for July and October (per Table 3), which bracket the high and low ends of the range of *k* values for the May-October period, at the *C** value of 0.002 mg/L. These curves illustrate that treatment wetlands would have better reduction efficiencies during May-October (as compared to curves assuming annual k values as shown in Figure 14), which would be expected since wetland production processes (that act to transform and/or sequester nutrients) are greatest during these months. These curves also illustrate a convergence toward the background levels of TP assumed in the analysis (i.e., 0.002 mg/L) for treatment wetlands areas exceeding about 40,000 acres. In other words, these curves suggest that treatment wetlands areas exceeding about 40,000 acres would not provide additional nutrient reduction benefits during the months of greatest wetland production.

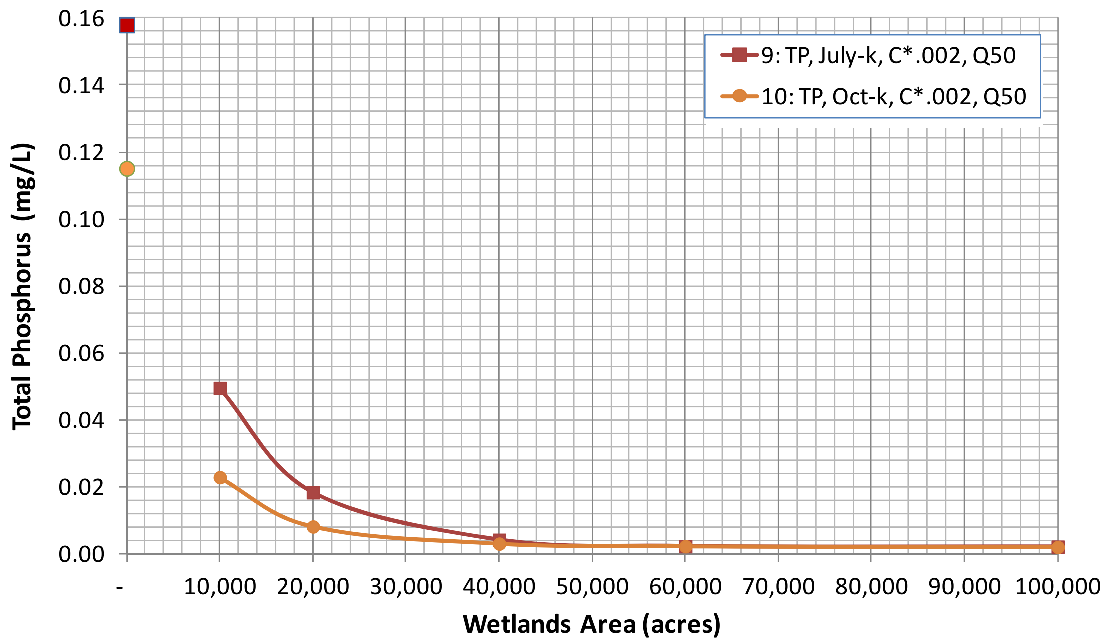

FIGURE 15
Calculated TP concentrations (mg/L) as a function of wetlands area (acres) in the Upper Klamath basin area draining to the Iron Gate location for scenarios 9 and 10 (assuming a C* value of 0.002 mg/L). Initial Concentrations (shown on the y-axis) represent the "untreated" wetland influent (inflow) concentrations.

As with TP curves discussed in section 4.3.1.1 above for the Link River location, the curves in Figures 14 and 15 for the Iron Gate location demonstrate that the results of this modeling analysis are sensitive to changes in assumed *k* and *C** values, particularly in the range of treatment wetlands areas up to about 40,000 acres. As such, these results lends additional support for the need to determine more site-specific *k* and *C** values for potential treatment wetlands in the Upper Klamath basin. The use of treatment wetlands pilot studies for such determinations is discussed further below in section 7.3.5.

4.3.2.2 Nitrogen

Calculated TN concentrations (mg/L) as a function of wetlands area (ac) in the Upper Klamath basin area draining to the Iron Gate location are shown in Figure 16 based on the results for scenarios 11 through 12. These two scenarios both assume the same *k* values for TN and component N forms as listed in Table 3), but differ according to *C** values (i.e., assuming 1.0 or 0.5 mg/L, respectively). As discussed in section 4.2.2 above, both scenarios 11 and 12 were run assuming half (50 percent) of the river flow is diverted into and through (and therefore receive treatment by) the modeled wetland areas.

The TN curve shown in Figure 16 for scenario 12 follows shallow curvilinear shape whereby calculated TN concentrations approach a baseline horizontal asymptote (defined by the *C** values) as wetland area increases. Wetlands area of about 40,000 acres appears to be the inflection points of TN reduction for the scenario 12 curve. The trend line for scenario 11 is basically flat, indicating little change in TN concentrations in wetland effluent (outflow) as a function of treatment wetland size. This flat line is the result of an assumed wetland background TP concentration (i.e., *C** value of 1.0 mg/L) that is not much lower than the wetland influent (inflow) concentration (of 1.18 mg/L).

Table 12 provides calculated "river mixture" TN concentrations and percent-reductions for various levels of treatment wetlands area under scenarios 11 and 12 (which both assume a 50-percent treatment flow fraction). As with TP discussed above, the percent-reduction values of TN in Table 12 indicate that percent-reductions are approximately in direct proportion to the treatment flow fraction. That is, the percent-reduction values achieved in the "river mixture" for all levels of treatment wetlands area under scenarios 11 and 12 are about half (or 50 percent) of the percent-reduction values in the wetland outflow (effluent).

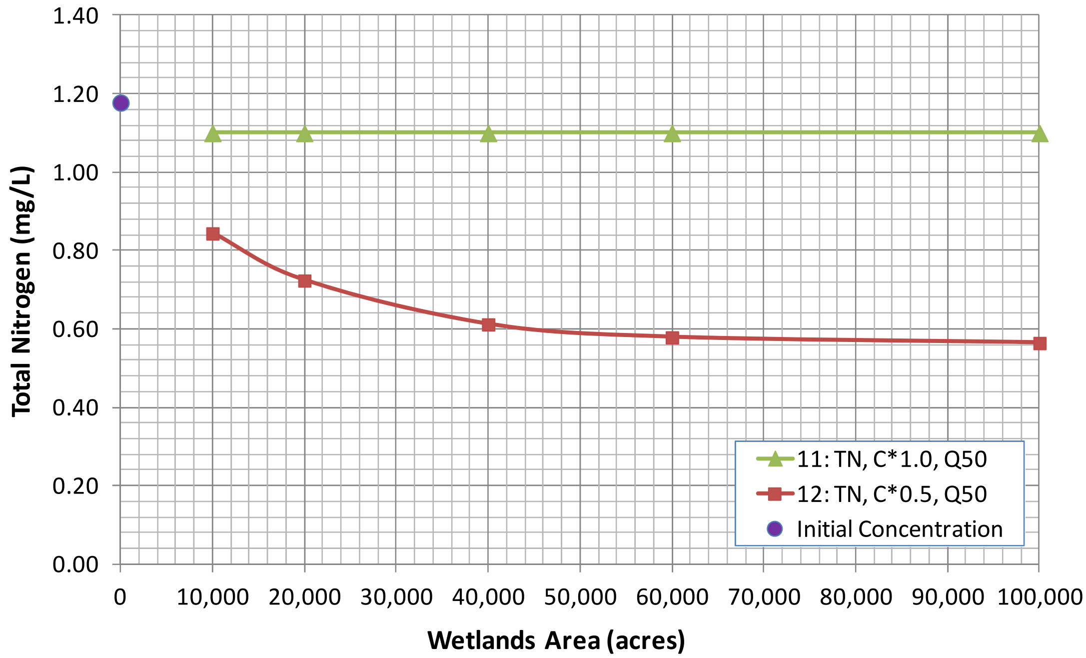

FIGURE 16
Calculated TN concentrations (mg/L) as a function of wetlands area (acres) in the Upper Klamath basin area draining to the Iron Gate location for scenarios 11 and 12. The Initial Concentration (shown on the y-axis) represents the "untreated" wetland influent (inflow) concentration.

TABLE 12
Calculated Total Nitrogen (TN) Concentrations (mg/L) and Reductions (%) as a Function of Wetlands Area (ac) in the Upper Klamath Basin Area Draining to the Iron Gate Location for Scenarios 11 and 12.

Scenario	Flow Proportion	Assumed C*	Wetland Area (ac)	TN in Wetland Outflow		TN in River Mixture	
				mg/L	% Reduction	mg/L	% Reduction
Initial Value			*0*	*1.18*			
11	0.5	1.0	10,000	1.10	7%	1.14	3%
	0.5	1.0	20,000	1.10	7%	1.14	3%
	0.5	1.0	40,000	1.10	7%	1.14	3%
	0.5	1.0	60,000	1.10	7%	1.14	3%
	0.5	1.0	100,000	1.10	7%	1.14	3%
12	0.5	0.5	10,000	0.84	28%	1.01	14%
	0.5	0.5	20,000	0.72	39%	0.95	19%
	0.5	0.5	40,000	0.61	48%	0.90	24%
	0.5	0.5	60,000	0.58	51%	0.88	25%
	0.5	0.5	100,000	0.56	52%	0.87	26%

4.4 Potential Implications of Wetland Area Estimates

The Klamath River TMDL developed by the Regional Water Board (2010) includes TMDL allocations and related targets for TP (and other TMDL parameters) at several locations, including “Stateline” (the Oregon-California border) and the “Iron Gate tailrace” (consistent with the Klamath River location below Iron Gate dam as assumed in this wetlands evaluation). As described by the Regional Water Board (2010), the numeric targets for TP (and other TMDL parameters) in the Klamath River TMDL “serve as the goal post from which TMDLs and associated load and waste load allocations are developed”. The values presented by the Regional Water Board (2010) suggest that the current loads of TP in the Klamath River below Iron Gate dam would need to be reduced by about 80 to 90 percent to meet TMDL targets. The Regional Water Board (2010) identified the restoration and construction of wetlands as an important strategy for reducing the large loads of nutrients and organic matter in the Klamath River to meet TMDL allocations.

The calculations of wetlands sizing developed in the preceding sections provide context for the levels of nutrient reduction that may be possible with various amounts of treatment wetlands area in the Upper Klamath basin. As discussed above, estimates of nutrient reduction effectiveness in this modeling are sensitive to assumed model parameters and values (e.g., *C** and *k*), particularly in the range of treatment wetlands areas up to about 40,000 acres. Future additional assessment of treatment wetlands feasibility and design in the Upper Klamath basin should seek to determine more precise site-specific values for these important parameters.

Despite the model’s sensitivities and range of results, a relatively consistent outcome from the results of this analysis (as seen in Figures 9 through 16) is that a treatment wetlands area of roughly 40,000 acres appears to represent an inflection point beyond which nutrient reduction effectiveness diminishes. This level or magnitude of wetland acres should help to provide context for further planning and evaluation activities (as discussed below in Chapter 7 *Other Considerations and Recommendations*), such as the assessment of potential wetlands locations, water rights requirements, types of treatment wetlands, and use of possible supplemental treatment techniques.

Development of wetland systems, whether for water treatment or habitat creation, is a land intensive proposition, as the value of 40,000 acres of treatment wetlands would attest. Therefore, land availability, and other land-based constraints (such as water availability, water rights, soils and topographic suitability), can be an impediment to the construction of treatment wetlands. Nonetheless, these calculations suggest that treatment wetlands offers a potentially important tool in a broader overall strategy for reducing nutrient loads that likely also would involve other water treatment processes and technologies. Moreover, treatment wetlands are often used in conjunction with other supplemental treatment technologies, such as those presented below in Chapter 5 *Potential Supplemental Technologies to Enhance Treatment by Wetlands*. These supplemental treatment technologies can substantially reduce the amount of land required, while leveraging the capital, energy and maintenance cost savings commonly associated with passive systems like treatment wetlands.

CHAPTER 5

Potential Supplemental Technologies to Enhance Treatment by Wetlands

This chapter describes several potential supplemental technologies to enhance treatment by wetlands. These supplemental technologies are described (in sections that follow), including their relative effectiveness, advantages and disadvantages, costs, and potential for application in the Upper Klamath basin. The specific potential technologies described in this chapter include:

Constructed Treatment Wetlands with Specialized Biological Features

- Constructed Emergent Vegetation Surface Flow Wetland System
- Submerged Aquatic Vegetation (SAV) Systems
- Periphyton Treatment Systems

Supplemental Chemical Treatment Approaches

- Application of Alum
- Application of Ferric Chloride
- Application of Calcium-Based Amendments
- Application of Lanthanum-Modified Bentonite Clay (Phoslock™)
- Application of Zeolites
- Application of Polymers: Polyaluminium Chloride (PACl) and Polyacrylamide (PAM)
- Low Intensity Chemical Dosing (LICD)
- Wetlands Soils Amendment

Combined Chemical/Physical Treatment Approaches

- Chemical Treatment Combined with Settling and Solids Separation
- Hybrid Wetland Treatment Technology
- Large-Scale Alum Injection and Treatment Wetland Settling System
- Advanced Treatment Using Chemical Treatment Combined with Filtration

The specific potential technologies described in this chapter do not comprise a fully comprehensive suite of all technologies that are potentially applicable for water quality treatment in the Upper Klamath basin. Rather, the technologies described in this chapter are those that in our judgment could be considered for potential implementation as an integrated part of, or in combination with, constructed treatment wetlands in the Upper Klamath basin, which is the specific subject of this report. Consideration of other potentially-applicable water quality treatment technologies (e.g., wastewater treatment facilities, algal filtration, other stormwater BMPs) is beyond the scope of this report.

The reader is advised that the cost information provided in this chapter is not necessarily comparable between or among the technologies presented. For example, cost information can differ between technologies in the costing units (e.g., $/acre vs. $/gallon) available from source materials. In addition, for some technologies (e.g., wetlands soils amendment), costs are so variable and site-dependent that only qualitative information is presented. However, the cost information contained in this chapter provides informative planning-level information and context for potential costs of these technologies.

5.1 Constructed Treatment Wetlands with Specialized Biological Features

5.1.1 Constructed Emergent Vegetation Surface Flow Wetland System

5.1.1.1 Description

The most typical treatment wetland systems involve surface flow (SF) wetlands that use emergent vegetation to promote nutrient uptake and removal. Constructed emergent vegetation SF wetland systems are designed as shallow, flow-through wetlands with pumped or gravity inflows and outflows, with emergent macrophytes as the dominant vegetation (Figure 4). Constructed emergent vegetation SF wetland systems provide water quality enhancement through natural wetland processes that include trapping and sedimentation of solids, biological update and resulting settling and soil accretion, and chemical and biological transformations of pollutants. The wastewater destined for SF wetlands treatment may be released directly into the wetland system or may travel through other supplemental treatment processes (such as described in other sections of this chapter). As the wastewater passes through these systems the contaminants are reduced by various physical, chemical and biological mechanisms in the water column and the soil matrix.

This type of constructed wetland most closely resembles a natural wetland, with areas of open water, emergent plants, and floating vegetation. Although primarily intended for water quality enhancement, these systems also provide significant functions for water volume storage and for wildlife habitat. Typical species of emergent macrophytes used in these treatment wetlands include cattails (*Typha spp.*), bulrush (*Scirpus spp.*), and other subdominant plant species (Kadlec and Knight 1996). The North American Treatment Wetland Database lists hundreds of species of emergent macrophytes occurring in constructed treatment wetlands. Emergent macrophytes such as cattails have specific growth requirements including a range of allowable water depths (about 0 to 4 ft), a range of hydroperiods (up to 100 percent inundation), and a range of water quality conditions in terms of oxygen-demanding pollutants (not totally anaerobic) and TP (typically greater than 50 µg/L). Optimal growth of these plants does not occur at the extreme ends of these ranges.

5.1.1.2 Effectiveness

The effectiveness of constructed emergent vegetation SF wetland systems to enhance water quality is described above in section 3.2.2.3 (*Effectiveness of Treatment for Nutrients in Wetlands*).

5.1.1.3 Advantages and Disadvantages

The key advantage of constructed emergent vegetation SF wetland systems is that they would be expected to cost significantly less than other more-complex constructed wetland systems to design, build, and maintain. An ancillary benefit of these systems is that they also provide significant functions for water volume storage and for wildlife habitat.

The key disadvantage of emergent vegetation SF systems over other wetland systems is that a larger land footprint would be needed for an equivalent level of treatment (e.g., about 50 percent more land area than a submerged aquatic vegetation [SAV] wetland of equivalent effectiveness).

5.1.1.4 Associated Costs

The most complete database of cost information for constructed emergent vegetation SF wetland systems comes from Florida (SFWMD 2004). SFWMD (2004) reports capital costs that range from about $230 to $2,500 per acre and average about $1,000 per acre (ac). SFWMD (2004) reports annual operation and maintenance (O&M) costs that range from about $60/ac/yr to $750/ac/yr and average about $260/ac/yr. SFWMD (2004) also calculated costs on the basis of TP mass removal that range from about $50/kg to $1,600/kg of TP removal and an average about $400/kg of TP removal.

In 2001, the California Department of Transportation (Caltrans) conducted a review to determine the cost-effectiveness of permanent structural storm water Best Management Practices (BMPs). One of the 15 BMP technologies included in the review was constructed wetlands. Based on information from several entities, Caltrans (2001) reported the median cost per acre treated for constructed wetlands was $3,667 (presumably in 2001 dollars), with a 90-percentile range in cost per acre of $760 to $13,800.

5.1.1.5 Applicability in the Upper Klamath Basin

Emergent vegetation SF wetland treatment systems have been successfully constructed and operated in a variety of settings as indicated by the various examples described below in Chapter 6 (*Relevant Treatment Wetland Case Studies*). For planning purposes, there are no particular technical reasons why this type of treatment wetland system could not be successfully applied in the Upper Klamath basin. Rather, the successful construction and operation of this type of treatment wetland system would be dependent on site-specific factors and constraints, such as land availability and suitability, water availability and use, permitting and approvals, soils conditions, climatologically-related wetland performance variations, and other factors (see section 7.2 *Anticipating Important Factors and Constraints*). However, these site-specific factors and constraints apply not just to emergent vegetation SF wetland treatment systems, but also to all other treatment wetland technologies as described in the following sections. The applicability of the above-reported costs to the Upper Klamath basin may be affected by regional differences in prices of materials, labor, climatologically-related wetland performance variations, and other factors.

5.1.2 Submerged Aquatic Vegetation (SAV) Systems

5.1.2.1 Description

The most typical constructed treatment wetland systems involve SF wetlands that use emergent vegetation to promote nutrient uptake and removal (as described in the previous section). Instead of emergent vegetation, SAV systems involve wetland treatment cells that are comprised of submerged aquatic vegetation, sometimes in concert with crushed limestone (Figure 17). SAV systems provide water quality enhancement through similar natural wetland processes as those described above for emergent macrophyte communities.

SAV wetland systems can provide increased treatment efficiencies over SF systems because SAV systems have deeper water depth, greater plant surface area for microbial processes, and aerobic water column. These advantages can provide increased treatment efficiencies because of two key P removal mechanisms: (1) direct uptake from the water column through the stems, leaves, and associated periphyton of submerged plants; and (2) co-precipitation of P with $CaCO_3$ due to the photosynthesis-driven large swings in pH near these surfaces (Dierberg et al. 2002). As such, an advantage of SAV wetland systems over SF systems is a smaller land footprint, although SAV wetland systems can cost significantly more than equivalent SF wetlands to design and build.

5.1.2.2 Effectiveness

Rate constants for P removal in SAV systems are roughly two to three times that of SF wetland systems (Dierberg et al. 2002, Gumbricht 1993, Knight et al. 2003 cited in Kadlec and Wallace 2009). Much of the SAV performance research has been conducted at small experimental scales in mesocosms and for relatively short durations (less than 2 years), although the results appear scalable. In a mesocosm-scale SAV study, Dierberg et al. (2002) found that inflow TP was reduced by 50 to 70 percent at HRTs of 1.5 to 7 days. Dierberg et al. (2002) noted that results in a 363-acre SAV system in Florida were similar, although the levels of concentration reduction were somewhat less for the larger-scale system.

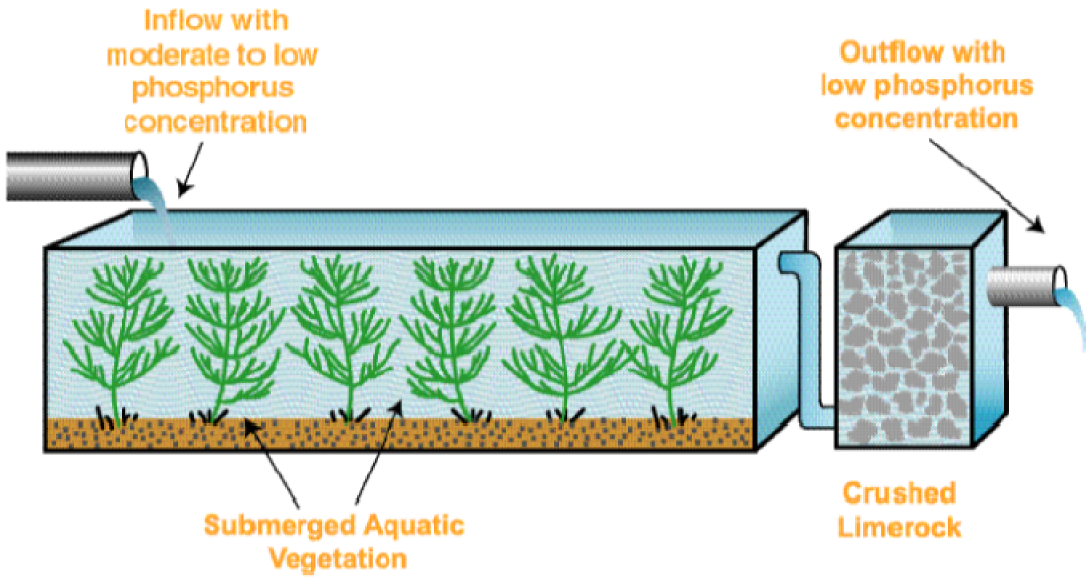

FIGURE 17
Submerged aquatic vegetation (SAV) system with crushed limerock outlet filter (SFWMD 2002).

5.1.2.3 Advantages and Disadvantages

The key advantage of SAV systems over SF wetland systems is a smaller land footprint (about 50 percent less land area than a SF wetland of equivalent effectiveness). The key disadvantage is that SAV systems can cost significantly more than equivalent SF wetlands to design, build, and maintain.

5.1.2.4 Associated Costs

No costs from full-scale constructed SAV systems were available for this analysis. SFWMD (2004) reports that the surface area for an SAV system for a given TP removal goal is expected to be smaller than for an emergent vegetation SF system due to the higher expected k-value. However, SFWMD (2004) indicates that a given area of an SAV system is likely to be more expensive to build and operate than a comparable emergent vegetation SF system. SFWMD (2004) assumes a capital cost to construct an SAV system value at about 20 percent higher than a comparable emergent vegetation SF system, and annual O&M costs per acre are expected to be 3 to 4 times higher (SFWMD 2004).

5.1.2.5 Applicability for Potential Use in the Upper Klamath Basin

SAV systems for P removal have largely been used in Florida for treatment of agricultural runoff. In the Upper Klamath basin, the co-precipitation of P with $CaCO_3$ and overall P removal may not be as significant as shown in Florida SAV, because Ca and alkalinity are generally higher in Florida waters. Nevertheless, the capacity of SAV systems for direct uptake of P from the water may still provide appreciably more P removal than SF wetland systems in the Upper Klamath basin. However, before SAV systems can be applied to the Upper Klamath basin, additional research and pilot testing would be needed to assess SAV wetland system performance under the climatological, soils, and other conditions specific to the Upper Klamath basin.

5.1.3 Periphyton Treatment Systems

5.1.3.1 Description

Periphyton are essentially ubiquitous in healthy wetland systems, and are attractive for nutrient removal because of their rapid growth and nutrient assimilation rates. Traditional SF treatment wetland systems have been coupled with periphyton-based cells to achieve low P levels. Because daily fluctuations in the productivity of algae can readily consume CO2 and induce an alkaline shift in pH, the nucleation process during the resulting calcite precipitation is thought to include calcium phosphate, and provide an enhanced process of removal that supplements biological uptake of P. Chemical adsorption to carbonate shells of diatoms within the algae also removes P (SFWMD 2002).

Periphyton based cells can be placed following emergent wetland cells, or in a treatment train such as emergent-to-SAV-to-periphyton cells. Periphyton systems typically include some emergent wetland plants (i.e., sparse macrophytes), but at a much lower density than emergent treatment wetlands. Periphyton systems are typically 0.5 to 1 foot in depth, graded with limestone gravel, which serves as a substrate for a thin (0.5 to 1 inch) mat of various species of algae, bacteria, and fungi, while slowing or reducing the growth of emergent macrophytes. A schematic of a periphyton system is shown in Figure 18.

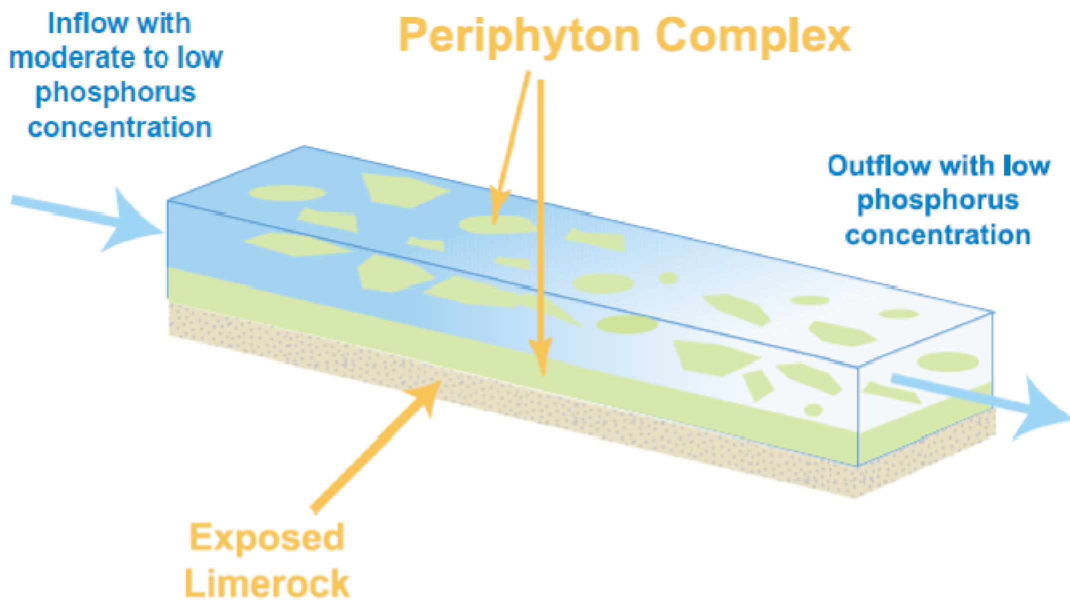

FIGURE 18
Periphyton-based treatment system schematic (SFWMD 2002).

5.1.3.2 Effectiveness

Periphyton treatment systems have been tested by several researchers (SFWMD 2002, CH2M HILL 2000, CH2M HILL 2002, CH2M HILL 2003, Debusk et al. 2004). These systems have shown the potential for reduction to low concentrations of P. For example, SFWMD 2002 reported that periphyton-dominated systems implemented as a natural system of stormwater improvement in Florida achieved effluent concentrations of TP near 10 µg /L compared to inflow TP concentrations of about 29 µg/L.

Substrates used in periphyton treatment systems can also affect performance levels. Of substrates evaluated (i.e., peat, sand, and shellrock), sand substrate performs the best, favoring periphyton growth and P removal comparable to shellrock substrate. Outflow concentrations were slightly higher for the peat substrate systems (SFWMD 2002).

Similar to SAV systems, the rate constant for P removal in periphyton systems is about two to three times that of emergent treatment wetland systems, therefore requiring approximately 50 to 70 percent of the area (SFWMD 2002, CH2M HILL 2003, Debusk et al. 2004).

5.1.3.3 Advantages and Disadvantages

Because relatively few large scale active periphyton systems are in operation, this technology is best considered an emerging technology. Further research is needed to better establish treatment cost effectiveness. Long-term operating costs have been estimated, but are not yet well established. Further research also is needed to better understand the N mass balance in periphyton-based systems, including the level of nitrification, denitrification and NH3 conversions (e.g., volatilization) that may be occurring.

Periphyton cells require relatively little management but are land-intensive treatment options that depend on environmental energy inputs from the sun and the atmosphere. Because these are solar-driven systems, it must have a large area to grow enough periphyton and other plants to capture very low TP concentrations through biological uptake and to sequester that TP in the form of calcium- and carbon-bound accreted sediments. No harvesting of biomass or sediments is envisioned for this process, so TP must be effectively stored within a periphyton system footprint to achieve a useful project life.

5.1.3.4 Associated Costs

No costs from full-scale periphyton treatment systems were available for this analysis. As with SAV systems (discussed above), SFWMD (2004) reports that the surface area for a periphyton treatment system for a given TP removal goal is expected to be smaller than for an emergent vegetation SF system due to the higher expected k-value. However, SFWMD (2004) indicates that a given area of periphyton treatment system is likely to be more expensive to build and operate than a comparable emergent vegetation SF system. SFWMD (2004) assumes a capital cost to construct an SAV system value at about 50 percent higher than a comparable emergent vegetation SF system, and annual O&M costs per acre are expected to be 3 to 4 times higher (SFWMD 2004).

5.1.3.5 Applicability in the Upper Klamath Basin

Similar to SAV systems, the best data and information on periphyton-based systems is from Florida. Calcium and alkalinity are likely lower in the Klamath River region than in Florida, and the lack of extensive calcite precipitation would likely reduce overall P removal as compared to Florida results. Additional research would be needed to adequately evaluate and design a periphyton-based system for the Upper Klamath basin, and to ensure that implementation and operation of these systems would be reliable and predictable in the long term.

5.2 Supplemental Chemical Treatment Approaches

Chemical treatment involves application of chemicals for pre-treating of water prior to entering treatment wetlands or as post-treatment to polish the water leaving the treatment wetlands. Chemical treatment is successfully used to reduce turbidity and nutrient concentrations in drinking water and wastewater, and has been tested with varying levels of success in conjunction with treatment wetlands as discussed further below. The specific chemical treatment technology that will work best at any given location will primarily depend upon the quality and the quantity of water that has to be treated.

5.2.1 Application of Alum

5.2.1.1 Description

Alum, the common name for aluminum sulphate ($Al_2(SO_4)_3 \cdot 18\ H_2O$)[4], has been used in a variety of settings to reduce the P status of water. Removal of phosphate using alum is through the formation of either aluminum hydroxide, which subsequently adsorbs P, or precipitation of aluminum phosphate (Metcalf & Eddy 1991, Bottcher et al. 2009). Alum has been used extensively in lakes as a management technique to reduce the amount of P in the water and limit the availability of this nutrient for algae production. Alum also is commonly used in water treatment plants to clarify drinking water, and in the treatment of wastewaters to precipitate and remove P. Due to the success of alum in the above mentioned settings and its wide availability, it could also be useful in a constructed wetland setting.

When alum is added to wastewater, the metal phosphate $Al_{0.8}H_2PO_4(OH)_{1.4}$ is formed (Sedlak 1991). When only moderate P removal is required, and relatively small alum dosages used, the metal phosphate is the predominant complex formed. However, when lower soluble P concentrations are required, larger alum dosages are required, and the formation of aluminum hydroxide precipitate [$Al(OH)_3$)] becomes an important reaction. The aluminum

[4] The number of water molecules may vary from 14 to 18.

hydroxide precipitate constitutes a gelatinous "floc". As the floc settles, the associated bound P is removed from the water. The floc also tends to collect suspended particles in the water and carry them down to the bottom. On the bottom, the floc forms a layer that also can act as a P barrier by combining with P as it is released from the sediments. Alum addition also results in Al^{3+} ions in solution, which can combine with dissolved phosphate to form an $AlPO_4$ precipitate. In addition to reducing P, alum treatment can reduce algal blooms in two ways: enmeshment and precipitation of algae in the floc and through reductions in available P for algal growth.

Variables that are typically considered when using alum are cost, alkalinity consumption, amount of sludge generated, and safety issues. Alum is typically available as a liquid with 4.4 percent Al, or as a 'lump', with 17 percent Al_2O_3 (Bottcher et al. 2009). The unit costs for alum are about $1/gal for the liquid form, or about $350/dry ton. Bench scale 'jar tests' are typically performed to determine alum dosages and application requirements (Bottcher et al. 2009). The formation of aluminum hydroxide precipitate with large doses of alum can result in a significant production of additional solids or sludge, and consumes alkalinity and decreases pH (Bottcher et al. 2009).

An additional concern for alum additions is the effect on biological components of the wetland, for example, Pilgrim and Brezonik (2005) noted that nearly all the invertebrates were eliminated from a settling pond of lake influent which had been treated with alum due to floc accumulation. Potential toxicity can be a concern with alum use, including potential buildup and development of toxic concentrations of aluminum, and the association of sulfate from the alum with potential methylation of mercury. To manage toxicity risks, careful management and monitoring of dosing levels and resultant water quality effects is needed.

5.2.1.2 Effectiveness

The form of P influences both the efficiency with which materials are removed from solution as well as floc settling characteristics (Bottcher et al. 2009). Particulate P is readily removed, and settles quickly. Dissolved or soluble P reacts rapidly, but tends to form $AlPO_4$ rather than sorbing to $Al(OH)_3$ floc, and the milky $AlPO_4$ does not settle well. Floc that contains significant organic P also does not settle well. Complete removal of dissolved or soluble P tends to require higher doses of alum. In general, floc settling times range from less than 3 hours to more than 24 hours. In addition to P removal by absorption onto metal hydroxides, alum can also precipitate SRP directly as aluminum phosphate ($AlPO_4$).

The controlling factor in the effectiveness, and also the toxicity, of alum is the pH of the system (Malecki-Brown et al. 2007). Alum has a pH of 2.4 (Beecroft et al. 1995) and therefore tends to decrease the pH of the system to which it is added. As long as the pH remains between 6 and 8, P inactivation will result, however, if the pH decreases to between 4 and 6, bound P will be released. Below pH 4 and above pH 8 soluble Al^{3+} dominates which may result in aluminium toxicity (Cooke et al. 1993). The minimum solubility for aluminum phosphate is approximately $10^{-6.5}$ M which occurs at a pH value of approximately 6-6.5. In general, the coagulation process is maximized, and residual metal concentrations minimized, when the coagulated water is maintained within the pH range of minimum solubility for the applied coagulant. For alum, this pH zone is approximately 6-8, since freshly precipitated alum floc has a minimum solubility of approximately 10^{-5} M which occurs in the pH range of 6.2-8.0. However, over a period of several months, the alum floc ages, eventually forming gibbsite, which has a minimum solubility of approximately 10^{-9} M in the pH range of approximately 5-7.

When alum used, alkalinity is consumed as a result of the coagulation process which can result in a decrease in solution pH, depending upon the applied alum dose and the available buffering capacity of the source water. However, the alkalinity consumption during coagulation is less with alum than with ferric coagulants (discussed below). This suggests that at equal doses the addition of alum will have less impact on pH than would be observed using iron-based coagulants.

Temperature may also have a significant impact on the coagulation process. Under cold temperatures, floc formation and the removal efficiency achieved using metal salts for coagulation decreases substantially. Colder temperatures often require a change in coagulant or change in dose to maintain acceptable settling characteristics and removal efficiencies. In the seasonally variable climate conditions in the Upper Klamath basin, temperature likely is an insignificant parameter impacting coagulation processes during the warmer period of the year, but may be a significant factor during colder months.

There are a number of case studies of lakes that have undergone nutrient inactivation with alum. Welch and Cooke (1995) evaluated the effectiveness and longevity of alum treatments (based on reduction in TP) on 21 lakes across the U.S. In stratified lakes, percent reduction in controlling internal P loading was continuously above 80 percent over an average of 13 years. Internal loading rate of TP was reduced in six of nine non-stratified lakes by an average of 66 percent over an average of 8 years. However, Welch and Cooke (1995) did find that alum treatment of lakes with high external loading was not very effective. Welch and Schrieve (1994) also found that alum may be completely ineffective, or effectiveness may be short-lived, if much of the lake is covered with macrophytes that senesce during summer and contribute P to the water.

Alum has been tested in a wetland treating municipal wastewater in Florida by Malecki-Brown et al. (2007, 2010) in Florida. Wetland cells treated with alum had significantly lower SRP than their control counterparts, with removals ranging from 77 to 86 percent depending on the wetland vegetation. On average, alum reduced SRP concentrations to one third of that in the controls.

Ann et al. (2000a; 2000b) investigated the efficacy of alum to increase P retention in highly organic Lake Apopka (Florida) marsh soils under flooded conditions. They found that 12 g of alum was needed for each kilogram of soil to minimize P release from soil to overlying water in the Lake Apopka marsh that had previously been drained and used for intensive agriculture.

Bottcher et al. (2009) provide an extensive summary and evaluation of alum treatment of agricultural runoff to reduce P loading to marsh and wetland areas in Florida. Table 13 provides a general summary of changes in water chemistry of agricultural runoff with increasing alum dose reported by Bottcher et al. (2009). Based on these data, P removal in agricultural runoff is most significant at alum dosages greater than 10 mg Al/L.

TABLE 13
Typical Changes in Water Quality Characteristics Resulting From Alum Treatment of Agricultural Runoff
(Adapted from Bottcher et al. [2009]; original data from Harper [1987])

Parameter (units)	Raw Water	Alum Treatment (mg Al/L)			
		5	10	15	20
pH (units)	7.24	6.88	6.59	6.40	6.10
Alkalinity (mg/L)	186	153	128	102	80.6
Total N (µg /L)	5066	3689	3689	2913	2816
Total P (µg /L)	853	696	642	257	80
Ortho-P (µg /L)	666	244	100	24	11

5.2.1.3 Advantages and Disadvantages

An advantage of alum treatment is that is provides rapid, highly efficient removal of P and TSS. In comparison to other potential coagulants (such as, iron and calcium), aluminum compounds are more suitable for coagulation processes under near-neutral pH conditions. For example, aluminum coagulation occurs in a neutral pH environment, while iron precipitation occurs under more alkaline conditions. Alum is relatively inexpensive compared to other treatment approaches on the basis of unit costs per mass of P removed. The floc of precipitated P from alum treatment is stable and chemically inert, even if there are fluctuations in redox or pH, and it becomes even more stable as it ages (Harper 2007).

Potential adverse impacts of alum include potential buildup and development of toxic concentrations of aluminum, and excessive acidification in systems with low buffering capacity (Malecki-Brown et al. 2007). Unlike iron, aluminum compounds do not undergo reduction-oxidation reactions under anaerobic conditions. However, there is some concern regarding the effect of aluminum and sulfate ions on the plant and animal communities. Potential toxicity can be a concern with alum use. Moreover, the association of sulfate from the alum with methylation of mercury in wetland systems warrants further examination prior to implementation. To minimize toxicity risks, careful management and monitoring of dosing levels and resultant water quality effects is needed.

Another potential disadvantage of using alum is that the alum floc produced from alum treatment can over time produce a sludge that may require removal and disposal. For example, Bottcher et al. (2009) indicate that sludge production is about 0.28 percent of the volume of the treated flow (or 374 ft^3 alum sludge per mgd of water treated) after a 30-day settling period at a dosage of 10 mg Al/L. Settled fresh floc will dewater naturally over a period of 30-60 days, with a 95 percent reduction in volume compared to fresh floc (Bottcher et al. 2009).

5.2.1.4 Associated Costs

Bottcher et al. (2009) present detailed cost information for 22 existing alum stormwater treatment facilities in Florida with treated watershed areas ranging from 64 to 1450 acres. The average capital cost for these existing alum stormwater treatment facilities is $307,627, ranging from $75,000 to $786,585 depending upon facility size and features. In general, the capital cost of constructing a facility is independent of the watershed size since the capital cost for constructing a treatment system for a 100-acres watershed at one location is identical to the cost of constructing a system to treat 1000 acres at the same location, although annual operation and maintenance (O&M) costs would increase. Estimated O&M costs range from $8,731 to $38,874 per year, including chemical, power, manpower for routine inspections, and equipment renewal and replacement costs.

Bottcher et al. (2009) report that the current cost of approximately $350/dry ton for aluminum sulfate is equivalent to a unit cost of approximately $1/gallon. Bottcher et al. (2009) further estimate that P removal costs using alum treatment range from approximately $75-250/kg of P removed over a 20-year life-cycle cost. In terms of nutrient removal, alum was determined to be a substantially more cost-effective method of removing P than wet detention systems, and that the mass pollutant removal cost with alum treatment decreases as the size of the treatment facility increases.

5.2.1.5 Applicability in the Upper Klamath Basin

Use of alum is a proven technology for reducing P levels, notably in eutrophic conditions such as occur in the Upper Klamath basin. The upper Klamath River has relatively low alkalinity (around 60 mg/L as $CaCO_3$), and therefore excessive acidification could be an issue with alum addition. This relevance of this issue could be determined through pre-application assessments, such as bench tests, and, if necessary, can be addressed actively with addition of other chemicals (albeit at higher cost and with additional residuals), or passively with appropriately designed wetlands.

The large reduction of nutrients in the Upper Klamath basin that will be needed to meet regulatory objectives (i.e., TMDLs) is a challenge that will continue over the long-term. Alum addition is a potential tool to improve P removal performance of wetland cells (for example, see the discussion in section 5.3.2 on *Hybrid Wetland Treatment Technology*). However, it is important to note that most P removal over the long term is associated with burial of wetland detritus, and is not related to sorption. As such, a well-designed wetland system may not need additional treatment with alum or other coagulant addition to achieve effective net P removal over the long term.

If alum were to be considered for use in the Upper Klamath basin, consideration would be needed to be given to potential removal and disposal of sludge produced over time from alum floc. Assuming that alum sludge production estimates of Bottcher et al. (2009) (as discussed above) apply to the Upper Klamath basin, it can be estimated that approximately 10,420 CY/day of sludge would be produced in the hypothetical scenario that the entire mean flow (1164 cfs) from the Upper Klamath basin (at a basin location corresponding to Link River dam) is subjected to alum treatment.

5.2.2 Application of Ferric Chloride

5.2.2.1 Description

Ferric chloride ($FeCl_3$) is the most widely used iron salt in North America, and is second only to alum for use in chemical coagulation. Ferric chloride has been widely used in wastewater treatment processes to reduce P concentrations. Ferric chloride has been successfully used to remove phosphates from sewage effluent by precipitation with a mixture of ferric salt and lime, but precipitated P is strongly affected by changes in redox potential (Ann et al. 2000b). The addition of this iron compound is also a method that is used to regulate P availability and control eutrophication in lakes and reservoirs (Cooke et al. 1993). More recently, ferric chloride is garnering more attention as a potential treatment within wetland and marshes, and associated soils (Sherwood and Qualls 2001, Faithfull et al. 2005).

The main mechanism of P removal upon addition of ferric chloride involves the precipitation of metal oxyhydroxides and subsequent adsorption of P by ligand exchange. The precipitates remove P by similar primary mechanisms previously discussed for alum. Removal of suspended solids, algae, P, heavy metals, and bacteria occurs primarily by enmeshment and adsorption onto the metal hydroxide precipitate. Removal of additional dissolved P occurs as a result of formation of $FePO_4$.

In the cases of lakes and wetlands, the water body itself serves as a settling basin, not only removing P from the settling of precipitates though the water column, but also forming a blanket of precipitated metal oxyhydroxides covering the top layer of sediment, blocking the release of P from the sediment.

Iron hydroxides have been shown to be very sensitive to changes in redox potential (Sherwood and Qualls 2001). Under anaerobic conditions, phosphate adsorbed to iron oxyhydroxide complexes or precipitated ferric phosphate complexes may redissolve as $Fe+^3$ to $Fe+^2$ reduction occurs. However, the potential release of P under reducing conditions depends not only on redox conditions, but also on the solubility of the various iron oxyhydroxide-phosphate complexes formed. Lack of easily mineralized organic matter could retard the development of reduced conditions and subsequent Fe reduction.

5.2.2.2 Effectiveness

When using ferric chloride, the impact of pH on the coagulation and floc-forming process is significant, as it is with other common coagulants. In general, the coagulation process is maximized, and residual metal concentrations minimized, when the coagulated water is maintained within the pH range of minimum solubility for the applied coagulant. For ferric chloride, this pH zone is approximately 8-10, since the resultant floc has a minimum solubility of approximately 10^{-9} M which occurs in the pH range of approximately 8-10. The stability of the floc decreases substantially and the solubility of Fe^{+3} increases substantially at pH values both lower and higher than this range. Unlike floc produced by alum, the iron oxyhydroxide floc does not undergo a significant aging process or shift in solubility characteristics over time. In addition to P removal by absorption onto metal hydroxides, ferric chloride can also precipitate SRP directly as ferric phosphate ($FePO_4$). The minimum solubility for ferric phosphate is approximately $10^{-5.8}$ M which occurs at a pH value of approximately 4-5.

A significant factor in the effectiveness of different iron-containing absorbents is their high content of Fe (III) oxyhydroxides. Under reducing conditions, which exist in conditions where dissolved oxygen is low or absent, Fe (III) is reduced microbially to Fe (II), with the release of adsorbed P. Thus, Fe (III) absorbents are likely to be ineffective unless oxic conditions are present and maintained. Addition of ferric iron will also result in reduced water alkalinity with precipitation of ferric hydroxide and the efficiency of water column phosphate removal may be low with both reactions occurring. Addition of a suitable buffer may be necessary in situations where alkalinity is low. The alkalinity consumption during coagulation is higher with ferric coagulants than with alum. This suggests that at equal doses the addition of ferric chloride will have a more significant impact on pH than would be observed using alum. In contrast, alkalinity is added to the source water during coagulation with alkaline coagulants, such as lime, sodium hydroxide, or sodium aluminate.

Temperature may also have a significant impact on the coagulation process. Under cold temperatures, floc formation and the removal efficiency achieved using metal salts for coagulation decreases substantially. Colder temperatures often require a change in coagulant or change in dose to maintain acceptable settling characteristics and removal efficiencies. In the seasonally variable climate conditions in the Upper Klamath basin, temperature likely is an insignificant parameter impacting coagulation processes during the warmer period of the year, but may be a significant factor during colder months.

Sherwood and Qualls (2001) evaluated ferric chloride additions to agricultural runoff entering the northern Everglades in Florida as a means for enhancing natural mechanisms of P removal from wetlands. In this study, ferric chloride was added to Everglades water spiked with soluble phosphate in microcosms simulating the Everglades ecosystem. Results indicated that on average less than 1 percent of the added soluble phosphate was measured in the water column during the 139-day testing period. Based on these results, Sherwood and Qualls (2001) suggested that ferric chloride addition thus might prove an effective means of long-term P retention in the Florida Everglades and perhaps other wetland systems.

Wisniewski (1999) undertook laboratory and in situ experiments in two hypertrophic lakes in Poland to determine the possibilities for the effective SRP precipitation by means of $FeCl_3$, applied directly to organic sediments (at a specific depth) previously subject to resuspension. Mesocosms treated with $FeCl_3$ showed either negligible or slight reduction of SRP in surface water, but SRP in interstitial water in lake sediments and water overlying sediments was reduced by 59 to 69 percent.

Browne et al. (2004) found that a 10 mg/L dose of $FeCl_3$ removed 79 and 73 percent of SRP and total P, respectively, from laboratory tests of enriched stream waters in New Zealand. Higher doses of 100 mg/L $FeCl_3$ removed 92 and 95 percent of SRP and total P, respectively. These tests relied on maintaining oxic conditions for Fe to bind to P.

5.2.2.3 Advantages and Disadvantages

The primary advantages of using ferric iron are: (1) it is considered a “natural” product; (2) it is readily available and potentially of lower cost compared to other chemical treatment alternatives; and (3) provides hydrogen sulfide binding and precipitation (in addition to phosphates). The primary disadvantages of using ferric iron are: (1) binding efficiency may be lower than anticipated based on stoichiometry, including the potential for reversible binding of P (redox-dependent); (2) reduction in alkalinity associated with dosing (may require alkalinity addition); and (3) low floc retention in turbulent conditions.

Iron precipitation occurs under more alkaline conditions compared to coagulation with alum, which occurs in a neutral pH environment. Iron is unstable in a reduced environment, and the collected floc must be stored in an aerobic environment at all times. Because of this instability, use of $FeCl_3$ in wetlands may not be desirable since precipitated P could potentially become soluble as water levels and redox fluctuate.

The addition of either ferric chloride for P precipitation can have a significant impact on other biological processes. The formation of metal hydroxides consumes alkalinity, and maintaining pH stability is important in biological systems. Also, ferric chloride is produced by dissolving iron ore in hydrochloric acid. As a result, heavy metals are common contaminants. Strict control of chemical characteristics of ferric chloride is necessary when using this compound in treating surface or drinking waters.

5.2.2.4 Associated Costs

Information is not readily available on costs associated with use of ferric chloride in constructed treatment wetlands. However, reviews of cost-effectiveness of chemical coagulants to treat storm water indicate that ferric chloride is similar in cost to alum (Narayanan and Pitt 2006, Peluso and Marshall 2002). Ferric chloride is slightly more expensive than the same volume of alum; however, the iron-based coagulants are more concentrated, therefore the cost to treat water is similar.

5.2.2.5 Applicability in the Upper Klamath Basin

For planning purposes, there are no particular technical reasons why use of ferric chloride in constructed treatment wetlands could not be considered for use in the Upper Klamath basin. However, the potential for redox-dependent reversible binding of P (as described above) could release P removed with ferric chloride. For this reason, ferric chloride would be less suitable than other chemical coagulant options (e.g., alum) for use in a wetland system implemented in the Upper Klamath basin.

5.2.3 Application of Calcium-Based Amendments

5.2.3.1 Description

Calcium-based amendments are potential tools to reduce P concentrations due to the limited solubility of Ca-P compounds and lack of redox effects. Finely ground calcium carbonate (calcite) or calcium-magnesium carbonate (dolomite) has also been used to successfully facilitate the formation and precipitation of calcium and magnesium phosphate, removing and sequestering P (Prochaska and Zouboulis 2006, Song et al. 2006).

Major potential sources of calcium include the following:

- **Limestone ($CaCO_3$)** - Crushed or powdered limestone is widely used as a calcium-rich soil amendment to reduce soil acidity and enhance P retention. It may also be a suitable material for adsorbing P in a wetland, because of its high Ca content and associated ability to facilitate P precipitation. Limestone is inexpensive, especially in some regions, but has limited solubility and cannot generally raise the soil pH to above about pH 7. Limestone, especially finely ground limestone, provides a surface on which P will adsorb, and then transform to stable minerals like hydroxyapatite (Reddy et al. 2005). Limestone provides a long-term effect. "Overdosing" with limestone will not occur, as solubility limits will control concentrations. There is relatively little impact of changes in redox potential on Ca-P forms (Ann et al. 2000b).
- **Hydrated Lime ($Ca(OH)_2$)** - Lime is more costly than limestone, but is much more soluble, and can raise pH to levels above 7. An important effect that must be considered is that high rates of lime can make organic forms of P more soluble (Ann et al. 2000a). Excessive increases in pH along with high levels of Ca can lead to the formation of layers of caliche, which can reduce permeability.
- **Gypsum ($CaSO_4$)** - Gypsum is somewhat more soluble than limestone, but includes sulfate, which once reduced in wetlands can enhance the loss of P sorbed to iron through precipitation of iron sulfides.

5.2.3.2 Effectiveness

There have been several studies that have demonstrated effective use of calcium-based amendments to enhance P retention in wetlands and soils. Zurayk et al. (1997) added crushed lime to wetlands in Lebanon and found that P removal was rapid and positively correlated with the amount of lime added. They found that P fixation was highest in the soil with the highest lime addition (49 percent CaCO3/ 51 percent soil) with 99 percent P removal observed. However, even at the lowest addition (1.5 percent CaCO3/ 98.5 percent soil), P retention was higher than for the soil without lime added (93 percent compared with 88 percent).

Ann et al. (2000a; 2000b) identified lime and slaked lime as preferred amendments to increase P retention in re-flooded P-rich organic soils. This was not only because of their effectiveness in immobilizing P under heavily reduced conditions, but also because of the low solubilities and low desorption potential of Ca-P compounds formed in this soil. The effective amounts of lime and slaked lime required to minimize P release from soil to the overlying floodwater were 7 to 15 g kg^{-1} soil.

In field studies in Quebec, Canada, Comeau et al. (2001) reported 92 percent P removal from trout farm effluents in subsurface-flow constructed wetlands with crushed limestone media. Shilton et al. (2005) tested the feasibility of using a New Zealand limestone from the Tararua region in a wetland setting to adsorb P. P removal in a batch experiment with a hydraulic retention time of 12 hours was 64 percent, but when tested in the field, P removal decreased to an average of 18 percent. Batch experiments included tests at different temperatures to determine the effect of higher temperatures on P removal. These showed that, as temperature increased, rates of P removal also increased.

Precipitation of phosphate with calcite commonly occurs in hardwater lakes during summer, and is favored by increased pH (Stumm and Morgan 1995). The use of calcite barriers was compared for three forms of material (crushed limestone and two forms of precipitated calcite, SoCal™ and ESCal™) in laboratory reactors for natural lake sediment in Australia (Hart et al. 2003). These trials demonstrated that the two precipitated calcites could effectively reduce P release from sediments under anaerobic conditions, whereas the limestone was ineffective. The maximum binding capacity for SoCal™ and ESCal™ were 3 percent and 1 percent (by weight), respectively, at high external P concentrations (Hart et al. 2003).

The long-term effectiveness is determined by the stability of calcite barriers and the binding capacity for P. The calcite barriers dissolve in systems where the water is undersaturated with $CaCO_3$. A speciation computer modeling program (e.g., PHREEQC; Parkhurst 1995) may be used to calculate the saturation indices for lake waters at various pH and temperature conditions, which indicate whether the calcite would dissolve or precipitate under these ambient conditions. If natural waters were theoretically at equilibrium with hydroxyapatite, at pH 8, the P concentration would be limited to 0.000037 µg /L, but this level of complete removal is not observed due to a number of complicating factors (Snoeyink and Jenkins 1980).

5.2.3.3 Advantages and Disadvantages

The primary advantages of calcium-based amendments are: (1) they are considered "natural" products; and (2) they are readily available and potentially of lower cost compared to other chemical treatment alternatives. The primary disadvantages of calcium-based amendments are: (1) the low binding efficiency of some products; and (2) turbidity associated with dosing and resultant periods of water column turbidity.

Like alum, calcium compounds are also relatively stable. However, calcium precipitation occurs under very alkaline conditions (pH levels in excess of 10-11) compared to coagulation with alum that occurs in a neutral pH environment. Unlike solutions of aluminum or iron which consist of dissolved ions in solution, calcium is typically supplied as a slurry of calcium hydroxide solids (lime) in water, which must be stirred continuously to prevent separation. Because calcium precipitation typically occurs at a pH range of 10-12, treatment processes that use calcium as a precipitate often requires pH neutralization as a second step. The precipitate must be separated from the treated water prior to pH neutralization to avoid dissolution of the precipitate and release of undesirable compounds as the pH is lowered. In view of the additional steps and equipment required, calcium hydroxide is seldom used for coagulation processes designed to remove P.

5.2.3.4 Associated Costs

Information is not readily available on costs associated with use of calcium-based amendments in constructed treatment wetlands. However, reviews of cost-effectiveness of calcium-based amendments associated with storm water BMPs indicate that the technique is relatively inexpensive. Taylor and Wong (2002) estimated costs of approximately \$25 - \$30 per m^3 of area treated, including purchase, transportation and application of the amending agents, and replacement of amended soil if its pollutant removal capacity diminishes over time. Taylor and Wong (2002) also cited a case study in Seattle, where the total estimated amended soil cost was approximately \$11 to \$33 per m^3 of area treated (in 1996 dollars).

5.2.3.5 Applicability in the Upper Klamath Basin

For planning purposes, there are no particular technical reasons why use of calcium-based amendments in constructed treatment wetlands could not be considered for use in the Upper Klamath basin. However, as described above, use of calcium would likely require additional treatment process steps involving separation of calcium precipitate from the treated water followed by pH neutralization. In view of the additional steps and equipment required, the use of calcium-based amendments would be less suitable than other chemical coagulant options (e.g., alum) for use in a wetland system implemented in the Upper Klamath basin.

5.2.4 Application of Lanthanum-Modified Bentonite Clay (Phoslock™)

5.2.4.1 Description

Phoslock™ is lanthanum-modified bentonite clay developed by the Australian Federal Government-owned Commonwealth Scientific and Industrial Research Organization (CSIRO) and the Australian Water and Rivers Commission (AWRC) to remove P from natural water bodies and waste water streams (Robb et al. 2003). Phoslock™ is a relatively new commercial product that is emerging as an effective eutrophication and/or blue-green algae management tool. Although information on the use of Phoslock™ in treatment wetlands is lacking, it is included in this report because the developers of Phoslock™ consider it an effective alternative to use of alum in natural water bodies (Afsar and Groves 2009). Compared to alum, Phoslock™ is considered to operate effectively over a wider range of pH and alkalinity conditions, and have less risk of potential toxicity in aquatic organisms (Afsar and Groves 2009).

Phoslock™ contains the rare earth element lanthanum in a matrix of bentonite clays and acts to remove dissolved phosphate from the water column by binding it into an insoluble precipitate. The lanthanum ions sorbed to the clay matrix react preferentially with free phosphate compounds in water (removing SRP) and rapidly form a highly stable insoluble species of lanthanum phosphate, or rhabdophane (National Industrial Chemicals Notification and Assessment Scheme 2001, Douglas et al. 2004). This resulting rhabdophane complex has a very low solubility ($K_{sp} < 10^{-27}$) and is not influenced by changes in pH and redox reactions in waterbody sediments, thus is not bio-available.

As a general rule, Phoslock™ is applied at the rate of 100:1; that is, 100 g Phoslock™ is required to remove 1 g of bioavailable P (Afsar and Groves 2009). Phoslock™ is generally supplied in a granular form (in 25kg bags) and can be applied to the water body both as a slurry and as granules.

5.2.4.2 Effectiveness

In laboratory tests, Douglas et al. (2000) found that Phoslock™ removed 87 to 98 percent of bioavailable P (or SRP) over the range of pH 6 – 8. When the solution pH was raised above 9, the SRP removal rate of Phoslock™ was slowed, with 40 percent of the SRP removed after the first hour of treatment and 60 percent removed after 24 hrs (Douglas et al. 2000). The observed decline was attributed to the formation of the hydroxyl species of the lanthanum ions. Douglas et al. (2000) further comments that if the observed rate of the P uptake at pH 9 were to continue, 99 percent of the SRP could be removed in about 4 days.

The laboratory tests by Douglas et al. (2000) also examined the effect of dissolved organic matter on the P-removal capacity of Phoslock™ using solutions containing humic acid with the concentration of 100 ppm. The results demonstrated that in the presence of 100 ppm humic acid, the performance of Phoslock™ was not reduced significantly up to pH 7. However, at pH 9, performance was reduced. Douglas et al. (2000) also mentioned that the kinetic study of the P uptake at pH 9 appeared to continue after 24 hrs. The authors suggested that if the study was to continue, quantitative removal would have occurred after 9 days.

Flapper (2003) reported on field trials of Phoslock™ applied to mesocosms at Fyshwick Lagoon in Australia. Total P in treated mesocosms was reduced by 83 to 96 percent compared with control mesocosms. The reduction in total P lasted about 9 weeks before total P concentrations began to increase and return to the same level as untreated mesocosms. Flapper (2003) also speculated that Phoslock™ application prevented a bloom of *Microcystis aeruginosa*, which developed in the control mesocosms after the start of the trials.

Robb et al. (2003) reported on two full-scale Phoslock™ applications (in a slurry from a small boat) undertaken in 2001/2002 in impounded sections of the Vasse and Canning Rivers in Australia. Following the first Phoslock™ treatment in the Vasse River, dissolved P concentrations were reduced from 50 µg/L to 20 µg/L. After the second treatment was applied several weeks later, dissolved P concentrations at the control site had reached almost 200 µg/L, but dissolved P concentrations at the treatment site remained low, reaching the detection limit of 5 µg/L. After 194 days, dissolved P concentrations at both the control and treatment sites were of similar magnitude. Similar applications in the Canning River also resulted in reductions in dissolved P concentrations, but of lesser magnitude, thought to be attributable to greater proportional amounts of runoff-related nutrient inputs to the Canning River.

Australia Water Quality Center (2008) reported on a comprehensive single-day application of Phoslock™ in Torrens Lake, Australia in 2008. Concentrations of TP before Phoslock™ treatment ranged from 0.095 to 0.155 mg/L at all sites. Concentrations of TP declined to minima of 0.045-0.061 mg/L within 2 to 3 weeks following the application, then subsequently began to increase again to pretreatment level after 4 to 5 weeks. Some, but not all of these increases were thought to be attributable to increases in runoff to the lake from rain events that occurred 3 to 4 weeks following the application.

Australia Water Quality Center (2008) also reported that the 2008 Phoslock™ treatment in Torrens Lake, Australia appeared to decrease the total algal biomass and cyanobacterial abundance in the water column soon after application. This observation was thought to be attributable to a direct flocculation and sedimentation effect of the bentonite clay on particulate matter, including algal cells, more so than an indirect response on algal growth from nutrient limitation; a process that can reasonably be expected to have a lag time of a few weeks.

Douglas et al. (2004) consider that the use of Phoslock™ to remove phosphate is superior to the use of more conventional use of alum and ferric chloride in several ways: (1) Phoslock™ can achieve greater total removal of phosphates; (2) Phoslock™ is effective over a wider pH range (c. 4.5–8.5) than Fe (III) (c. 3.5-4.5) or Al (c. 5.0–6.5); (3) Phoslock™ precipitates polyphosphates equally well as orthophosphates; and (4) the solubility product of La-phosphate is extremely low, and therefore is not bio-available with less potential for toxicity.

5.2.4.3 Advantages and Disadvantages

The advantages of Phoslock™ include: (1) a relatively high affinity for P compared with conventional treatments; (2) an ability to bind P under both aerobic and anaerobic conditions and over a broad range of pH conditions; (3) it does not readily re-release P when physical and chemical conditions change; and (4) a relatively low toxicity, resulting in safe handling, application and disposal.

The disadvantages of Phoslock™ include: (1) relatively high cost compared with conventional treatments; (2) slow settling of fine particles, and resultant turbidity in waters during the application period; and (3) eventual burial in sediment that reduces efficacy and smothers benthos.

5.2.4.4 Associated Costs

As a proprietary product, cost of treatment using Phoslock™ is only readily available by specific request from the manufacturer or designated representatives. Such a request was not made for this study given unknowns regarding if and how Phoslock™ might be used in conjunction with potential constructed treatment wetlands in the study area.

5.2.4.5 Applicability in the Upper Klamath Basin

Use of Phoslock™ is a technology reported to be effective for reducing P levels, including in eutrophic conditions such as occur in the Upper Klamath basin. Phoslock™ is touted by Douglas et al. (2004) and in other manufacturer materials as a superior alternative to conventional use of alum and ferric chloride. Phoslock™ does not have an extensive track record of use as with alum and ferric chloride, and has a higher relative cost compared with these conventional coagulants (although specific cost information for Phoslock™ was not readily available for this report). The relevance of these issues could be evaluated through pre-application assessments, such as bench tests, to more definitively determine the potential costs and effectiveness of Phoslock™ for use in the Upper Klamath basin.

5.2.5 Application of Zeolites

5.2.5.1 Description

Zeolites belong to a family of naturally occurring volcanic minerals with unique physical and chemical characteristics. There are over forty-eight varieties of natural zeolite minerals with similar structures and molecular makeup, each with its own particular attributes – some subtle and some more obvious (Coombs et al. 1998, Flanigen et al. 2010). Natural zeolites consist of a negatively charged three-dimensional aluminosilicate lattice which forms a network of open channels and internal surface area. The channels, typically 0.3 to 0.7 nanometers in diameter (3 to 7 angstroms, slightly larger than a water molecule), selectively screen molecules according to size and exchangeable cations, thus giving rise to the term “molecular sieve”.

Zeolites provide a substantial cation exchange capacity (CEC). Positively charged cations (sodium, calcium, potassium, and magnesium) are loosely bound at the junctures of the negatively charged aluminosilicate lattice structure. Cation exchange occurs when two or more positively-charged compounds or elements exchange places on a negatively charged host. As a cation exchange agent, zeolites have been widely investigated and applied to remove cation contaminants in waters and wastewaters (Widiastutia et al. 2008), and for capping chemically contaminated sediments (Vopel et al. 2008). A classic example of cation exchange in zeolite is the removal of NH3 from water and wastewater. When a molecule of NH3 is hydrated, the reaction produces NH4+, which is readily exchanged for all or part of the calcium, potassium and magnesium cations contained in zeolite and adsorbed on to its stable aluminosilicate lattice.

Candidate zeolites include clinoptilolites and chabazites, and are available in many commercially-available powder or granular forms. They have a high surface area for binding of cations, typical packed bed porosity of 30 to 50 percent, and are stable across a wide range of pH. The ion exchange capacity is of high significance and is typically within the range of 1.65 to 2.5 milliequivalents per gram (mEq/gm)[5]. These properties make natural zeolites excellent candidates for water, wastewater, and sediment treatment media (Nyugen and Tanner 1998, Bowman 2003, Vopel et al. 2008, Widiastutia et al. 2008).

"Modified zeolites" have been developed that allow for more efficient and effective targeting of nutrient or contaminant removal. One form of modified zeolite includes modification with a surfactant to reverse the surface properties of natural zeolite from negative to positive. Surfactant-modified zeolite (SMZ) is capable of simultaneous sorption of anions, cations, and non-polar organic molecules from water, making it amenable to various water treatment applications (Bowman 2003). These include applications of SMZ as a sub-surface permeable reactive barrier to control contamination of groundwater, removal of petroleum hydrocarbons from oilfield wastewaters, and removal of nutrients and pathogens from wastewater (Bowman 2003).

Another form of modified zeolite includes an aluminum-amended, proprietary zeolite (Z2G1 or Aqual P™ by Minsorb) that is marketed specifically for nutrient removal from lakes and other waterways. The aluminium amendment gives the zeolite mineral, which is a strong cation absorber, a strong affinity for, and thus the ability to sequester phosphate. These products were originally developed as a sediment capping material to assist in the reduction of internal P loading in lakes from anoxic sediments during seasonal stratification, and are the only known capping agents that are capable of inactivating both P and N (Gibbs and Ozkundakci 2011, Ozkundakci et al. 2011). Although the modified zeolites have anion exchange properties, the zeolite is still accessible to sorb inorganic cations. It is because the modifying agents are relatively large molecules that remain on the external surface of the zeolite crystal and do not enter the zeolite channels. The internal cation-exchange site of the zeolite remains accessible to sorb inorganic cations.

5.2.5.2 Effectiveness

Most of the research regarding the effectiveness of zeolites, particularly with regard to nutrients, has occurred relative to applications to remove nutrient in waters and wastewaters (Bowman 2003, Widiastutia et al. 2008), and for capping nutrient releases from sediments (Vopel et al. 2008). Research by Widiastutia et al. (2008) on the use of SMZ to remove phosphate (PO_4^{3-}) from greywater indicates that SMZ using cethylpyridinium chloride could remove phosphate ranging from 50 to 90 percent depending upon variables such as the initial phosphate concentration, contact time, and initial pH.

Gibbs and Ozkundakci (2011) tested the aluminum-amended, proprietary P-inactivation zeolite Z2G1 on sediment cores from Lake Okaro, New Zealand for P removal efficacy prior to a whole lake trial to manage internal P loading. Sediment core study results showed that a thin layer of Z2G1 (about 2 mm) could completely block the release of P from the sediment under aerobic and anoxic conditions, and remove P from the overlying water in contact with the capping layer. However, subsequent results from the whole lake treatment study indicated that the sediment P release occurred after the application of Z2G1 (Ozkundakci et al. 2011). Similarly, reduction in N release was lower than expected when compared to the laboratory incubation study. Ozkundakci et al. (2011) concluded that these differences illustrate the difficulty of extrapolating laboratory results to a whole lake, and are likely to be a number of factors, such as the timing of the Z2G1 application after the lake had stratified and the

[5] The mEq is one-thousandth of a compound's or an element's equivalent weight. The equivalent weight is the amount of a substance that combines with or displaces 8.0 g of oxygen (or 1.008 g of hydrogen); it is the ratio of the molecular weight to the number of protons (acid/base reactions) or electrons (redox reactions) involved in the reaction.

release of SRP from the sediment had begun, and the uneven coating of the capping material on the sediment surface.

Gibbs et al. (2011) assessed the effectiveness of the modified zeolite Z2G1 in comparison to alum and Phoslock^{TM} as sediment capping agents to manage P release from sediments. Gibbs et al. (2011) determined that all three products are capable of blocking the release of P from the sediments depending on dosage rates and other factors, such as pH and water hardness. At all levels of treatment the alum and Phoslock^{TM} completely blocked the release of dissolved reactive P (DRP) from the sediments and had the capacity to remove additional DRP from the overlying water column. The Z2G1 at the 50 and 100 percent treatments was not able to block the release of DRP from the sediment. At the 200 percent treatment, which had complete sediment coverage, Z2G1 did completely block the DRP release and had the capacity to remove additional DRP from the overlying water column comparable with the other capping materials. For NO_3-N, the greatest reduction (48 percent) occurred in the Phoslock^{TM} treatments although similar levels of reduction (43 percent) also occurred in the 100 and 200 percent alum treatments, while Z2G1 caused a 37 percent reduction at the 200 percent treatment.

Use of zeolites in treatment wetlands has been more limited than the applications described above, but research to date indicates considerable effectiveness of zeolite use in conjunction with treatment wetlands. Shuib et al. (2011) assessed the removal of nutrients and TSS from moderate strength wastewater in a horizontal subsurface flow constructed wetland using natural zeolite (unmodified) as a substrate. During a study period of 39 weeks, the constructed wetland with zeolite achieved significant removal of COD, NH_4-N, TN, and TSS compounds. Removals for COD, NH_4-N, TN, and TSS were found to be 89, 99, 96, and 95 percent, respectively, at 4 days hydraulic residence time (HRT) in the wetlands, and 85, 99, 91, and 91 percent, respectively, at 3 days HRT. Based on these results, Shuib et al. (2011) concluded that zeolite as a substrate media can substantially improve the effluent quality of the constructed wetlands.

Stefanakis and Tsihrintzis (2012) tested gravity filters containing zeolite (unmodified), bauxite, and carbonate material that were operated for 3 years to treat the effluent of a pilot-scale vertical flow constructed wetland. Results showed a significant improvement of effluent quality at an HRT of 1 day. Zeolite was more effective in N and organic matter removal, while bauxite in P retention. The carbonate material had the lowest efficiency among all filter materials used. The filter containing a 50:50 mixture of zeolite and bauxite showed the highest efficiency in pollutant removals. The increase of the residence time from 1 to 2 days did not show a respective statistically significant increase in removal rates.

Hillsborough County, Florida operated a zeolite (unmodified) filter pilot plant over a 216-day period to evaluate its ability to enhance N removal in stormwater from a pre-sedimentation pond prior to discharge to a post-filtration wetland (Smith et al. 2006). When operated at steady or simulated storm-event flow rates, the zeolite filter was highly effective at removing NH3, producing an effluent about a 90 percent or greater reduction in NH3-N concentration and removing more than 95 percent of the applied NH3 mass. The effluent also had about a 32 percent reduction in total inorganic N concentration. These results demonstrated that the zeolite filter performance was much superior to a parallel sand filter operated under similar conditions. The zeolite filter had limited or no removal effectiveness for dissolved organic N (DON) and NO3-N. DON removal in the zeolite filter was 25 percent or less. Nitrate-N was actually increased in the zeolite filter effluent (compared to influent), which Smith et al. (2006) indicate would require treatment or containment by other measures if TN removal were to be the goal.

5.2.5.3 Advantages and Disadvantages

The primary advantages of natural zeolites include: (1) they are relatively inexpensive and abundant natural products; and (2) a relatively high affinity for cation removal, particularly NH4+. The primary disadvantages of natural zeolites include: (1) they lack affinity for anion removal, including phosphate; (2) effectiveness is dependent on the initial contaminant concentrations, contact time, pH, permeability, and structural stability of the zeolite; and (3) effectiveness may differ among various zeolite forms and commercial products.

The primary advantages of modified zeolites include: (1) more efficient and effective targeting of nutrient or contaminant removal than natural unmodified zeolites; and (2) the only known nutrient treatment agent that is capable of inactivating both P and N. The primary disadvantages of modified zeolites include: (1) relatively higher costs compared with natural zeolites; (2) effectiveness is dependent on the initial contaminant concentrations, contact time, pH, permeability, and structural stability of the zeolite; and (3) effectiveness may differ among modified zeolite forms and commercial products.

5.2.5.4 Associated Costs

Natural zeolites are relatively inexpensive, costing in bulk about $200.00 per ton from large manufacturers, such as Bear River Zeolite Co. in Idaho (http://www.bearriverzeolite.com/) or the St. Cloud Mining Co. in New Mexico (http://www.stcloudmining.com/). Modified zeolites are more expensive due to the additives and processing required. For example, Bowman (2003) reports that surfactant-modified zeolites are about cost 7-10 times as much as unmodified zeolites.

Costs of treatment using aluminum-amended, proprietary modified zeolites (Z2G1 or Aqual P™ by Minsorb) are only readily available by specific request from the manufacturer or designated representatives. Such a request was not made for this study given unknowns regarding if and how proprietary modified zeolites might be used in conjunction with potential constructed treatment wetlands in the study area.

5.2.5.5 Applicability in the Upper Klamath Basin

Use of modified zeolites is a technology reported to be effective for reducing nutrient levels, including in eutrophic conditions such as occur in the Upper Klamath basin. Modified zeolites do not have an extensive track record of use as with alum, and are expected to have higher relative cost. The relevance of these issues could be evaluated through pre-application assessments, such as bench tests, to more definitively determine the potential costs and effectiveness of modified zeolites for use in the Upper Klamath basin.

5.2.6 Application of Polymers: Polyaluminium Chloride (PACl) and Polyacrylamide (PAM)

5.2.6.1 Description

Polyaluminum Chlorides (PACl)

Polyaluminum chlorides (PACl) are synthetic polymers that serve as an inorganic aluminum-based coagulant. PACls are a potential substitute for alum because: (1) PACls tend to be more robust than alum with regard to achieving coagulation goals because their precipitates vary less under changing environmental conditions; and (2) PACl has less potential toxicity-related concerns than alum.

PACL products react to form insoluble aluminum polyhydroxides which precipitate in big volumetric flocs similar to those formed with alum to absorb and precipitate suspended pollutants in the water. PACl compounds contain supplemental hydroxide $(OH)^-$ ions which cause lower pH depression and alkalinity impacts during coagulation processes. PACL products generally do not require the addition of acid or base to control the pH within the natural water range.

PACl has less toxicity potential than sulfate-containing alum. Sulfate loading from alum can be a concern because of the link of sulfur cycling in wetland environments with microbially-mediated methylation of mercury. Methylmercury is toxic, and once formed can bioaccumulate among trophic levels.

Polyacrylamide (PAM)

Polyacrylamide (PAM) is a degradable, synthetic, potassium-based long-chain polymer that has been used for many different purposes, including water treatment and erosion and stormwater control. We are unaware of the use of PAM in conjunction with constructed treatment wetlands, but include discussion here because of its use to reduce P runoff when applied to fields. In addition, while not a coagulant, PAM can be used with coagulants (such as alum) to enhance flocculation and settling to remove fine particles.

Water treatment versions of PAM in the anionic form have shown very low aquatic toxicity potential to the environment. Cationic PAMs have better binding potential directly to clay surfaces but compared to anionic forms do exhibit high toxicity potential to aquatic organisms are usually prohibited from use within stormwater and erosion control practices. Anionic PAM can adsorb metal ions, both ferrous and non-ferrous via electrostatic interaction and chelation, which aids particulate settling via flocculation, particularly when used with a coagulant.

5.2.6.2 Effectiveness

Polyaluminium Chlorides (PACl)

PACls are synthetic polymers designed for coagulation based on optimum charge neutralization and bridge binding. Precipitates formed by alum and ferric salt application are amorphous hydroxides and the exact characteristics of those products and the efficiency of the chemicals used are dependent upon a number of variables such as temperature and mixing energy (Edzwald and Van Benschoten 1990). Engineered polymers like PACls tend to be more efficient and robust with regard to achieving coagulation goals because their precipitates are less variable.

PACls have very good performance over a broad dosing range, and inorganic/organic polymer blends appear to be the most difficult to overdose. Even so, however, more optimal dosing of PACls improves coagulant performance. Thus, more optimal dosing of PACls leads to more efficient coagulant utilization and better performance.

Compared to alum and ferric chloride, the performance of PACls with regard to P and turbidity removal is minimally affected by changes in temperature, mixing regimes, storm water quality and dose. The performance of PACls is also less affected by different rapid or slow mixing specifications.

PACL modified with silica or sulfate provided consistent removal of P and fine particles in a stormwater treatment study at Tahoe Basin CA (Trejo-Gaytan et al. 2006). The effectiveness of this approach is described in the following section 5.2.6 *Low Intensity Chemical Dosing (LICD)*.

Polyacrylamide (PAM)

Land applications are the most common way in which PAM is used for nutrient control. Entry and Sojka (2003) report that land applications of PAM alone as a soil stabilizer may reduce ortho-P and total-P in runoff water at the source up to 92 percent. Stieber and Bahr (1996) report that PAM applied to the soil directly or used in combination with filter strips resulted in consistent 80-95+ percent reductions in P concentration.

Once the P enters a water course, the chemistry and particulates change and PAM by itself is less reliable for P binding (CRWQCB 2005). The addition of a coagulant (such as alum) is usually required to destabilize the electrostatic charge of the colloidal or suspended particles with the P, which is similar to what is used in water treatment plants. The coagulant concentration needs to be determined by jar testing and will vary in proportion to the P concentration. Once the electrostatic charge has been destabilized, a PAM polymer specific for the target particle, after the coagulate reaction occurs, may be used to rapidly settle the solids.

CRWQCB (2005) demonstrated that the use of PAM alone was unable to remove the solubilized P in runoff from California farmland feeding into the Salton Sea, but was able to remove the particulate P very effectively. PAM added to water can reduce TP when P exists as the organic particulate material or soil particulate form by standard flocculation processes. Thus, the effectiveness of PAM for P removal from water appears to increase with increasing particulate mass within the water. Bottcher et al. (2009) also report that the effectiveness of PAM to remove P in runoff waters is best in the presence of existing organic particulates or soil particulates.

CRWQCB (2005) demonstrated that the use of PAM in combination with alum substantially increased effectiveness in reducing P in runoff to the Salton Sea. Alum treatment alone was effective in binding virtually all of the P, but resulted in significant floc travel through the ditch systems with alum floc ending up in the Salton Sea. CRWQCB (2005) determined that use of alum alone would require installation of additional ponds or basins when treating larger flows due to accommodate the retention and settling times required for alum floc settling. Using alum together with PAM was still highly effective at 93 percent in soluble P reduction, but substantially enhanced floc settling and retention by creating a heavier floc that settles more rapidly.

5.2.6.3 Advantages and Disadvantages

Polyaluminum Chlorides (PACl)

The advantages of PACls include: (1) they reduce total dissolved P as well as alum; (2) typically produce less flocculate than alum; (3) they have a broader range of pH over which they are effective (i.e., optimally effective for waters with pH ranging from 6 – 8, and relatively effective up to a pH of 10); and (4) they reportedly have ten to twenty times less dissolved aluminum in solution than does alum.

The disadvantages of PACls are: (1) they are substantially more expensive than alum, and distributors are more limited; (2) they are available in a wide range of products that are not all equal in performance, which requires an understanding and assessment of their properties in the context of water treatment goals in order to select a preferred PACl coagulants; and (3) it is reported that some PACls are less effective at removing P than alum.

Polyacrylamide (PAM)

The advantages of PAM are: (1) when land applied, it can substantially reduce nutrients in runoff waters, which could significantly reduce the costs of other treatment systems downstream as reduced nutrient values would require less to treat; (2) it can enhance the effectiveness of other coagulants by aiding flocculation and settling; and (3) it is biodegradable and non-toxic.

The disadvantages of PAM are: (1) its effectiveness is temporary due to the environmental biodegradation of the PAM molecule, so needs to be used in combination with other coagulants to result in long term binding of P; and (2) the polymers are specific to the lithology targeted and must be tested before application (improper application or incorrect PAM material can result in ineffective binding of the target material).

5.2.6.4 Associated Costs

Generally, PACl coagulants are more costly than alum on an equivalent Al basis (Pernitsky and Edzwald 2006). On the other hand, PACls are pre-neutralized and are less acidic than alum. For low and moderate alkalinity waters, lower amounts of base would be used to control coagulation pH, thus reducing costs. Sludge quantities, dewatering characteristics and treatment and disposal are also important considerations in evaluating the overall cost of PACl versus alum (Pernitsky and Edzwald 2006). For example, Gebbie (2001) reported that chemical costs using PACl were about 10 percent higher than alum per unit volume of treated water. However, Gebbie (2001) also reported that treatment effectiveness of PACl was more robust than alum and produced less sludge requiring disposal. As a result, the overall treatment costs using PACl were about 14 percent less than alum per unit volume of treated water.

Land applications of PAM to soil may be the least expensive method for reducing P releases into runoff water (Blonco-Canqui et al. 2004). Sojka and Lentz (1997) reported that the typical costs for using PAM are $15 to $35/acre, and that these costs are partially or entirely retrieved by savings in erosion-related field operations, improving infiltration, water conservation, or crop responses.

The California Regional Water Quality Control Board (CRWQCB 2005) indicates that costs for PAM land applications can vary greatly based on P loading, lithology and rain event cycles. Application rates can typically vary from 4.5 kilograms (10 pounds)/acre to 23 kilograms (50 pounds)/acre with PAM product costs averaging $17.60/kilogram. CRWQCB (2005) estimated that the cost of land-applied PAM to treat the Imperial and Coachella Valleys at the source would be 11-18 million dollars per year. It was also noted that the land-applied PAM would greatly reduce the sediment loads to receiving waters, thus reducing maintenance costs.

CRWQCB (2005) also estimated costs for use of alum and PAM in combination that were quite costly in comparison to land-applied PAM treatment. The estimated cost of alum and PAM in combination to treat the three tributaries that feed to the Salton Sea was estimated to be 44 million dollars per year.

5.2.6.5 Applicability in the Upper Klamath Basin

PACls are an alternative to the use of alum as a potential supplemental technology to enhance treatment by wetlands to reduce P in the Upper Klamath basin. It is possible that PACls may be more effective than use of alum in the Upper Klamath basin given that PACls are effective over a broader range of water temperatures and pH.

However, PACls have a higher relative cost compared with alum, and are available in a wide range of products that can vary appreciably in performance. The relevance of these issues could be evaluated through pre-application assessments, such as bench tests, to more definitively determine the potential costs and effectiveness of PACls for use in the Upper Klamath basin.

As noted above, we are unaware of the use of PAM in conjunction with constructed treatment wetlands, but include discussion here because of its use to reduce P runoff when land applied. In addition, while not a coagulant, PAM could be used with coagulants (such as alum) to enhance flocculation and settling to remove fine particles.

5.2.7 Low Intensity Chemical Dosing (LICD)

5.2.7.1 Description

Low Intensity Chemical Dosing (LICD) is defined as the carefully-metered addition of low concentrations of chemicals coagulants to enhance and accelerate the rate of P removal from the water column through precipitation and settling of chemically-formed and naturally-occurring particulate P. Under LICD, coagulant doses are minimized in order that chemicals are most efficiently used to limit potential environmental effects, such as toxicity due to potential overdosing.

Recent studies have also been conducted in the Tahoe Basin to assess the feasibility of using LICD to reduce fine particles (<20 µm in diameter) that cause turbidity and P inputs responsible for impacting the treasured clarity of Lake Tahoe and regional water quality goals (Bachand et al. 2010, Bachand et al. 2009, Trejo-Gaytan et al. 2006). These studies used settling columns to show the feasibility of coagulant dosing to target fine particle removal from storm water in shallow treatment basins and wetlands. From screening tests of a wide range of available coagulant types, four were subjected to detailed testing, including a polymer blend (JenChem 1720), polyaluminum chloride (PACL; Pass-C and PAX-XL9), and an aluminum chlorohydrate (ACH; SumaChlor 50). The four selected coagulants did not necessarily represent the most effective coagulants in the screening tests, but they did represent diverse coagulant chemistries that provided relatively robust performance for different dosing levels with regard to turbidity and P removal. The specific products used in these studies did not constitute an endorsement, and coagulants with similar chemistries are assumed to perform similarly.

An LICD approach has also been tested in the Everglades program (SFWMD 2002). The purpose of LICD in the Everglades case was to minimize chemical requirements, and to use downstream wetlands to remove pin floc associated with low doses of coagulant (Bottcher et al. 2009). Less than 5 mg/L of ferric or aluminum chloride salts was added to wetland influent with a total P concentration of 130 µg /L. Aluminum chloride was used rather than alum because some research showed that the sulfate in the alum could increase methylation of mercury.

5.2.7.2 Effectiveness

The results of the Tahoe Basin testing indicated that the PACL coagulants were the most effective, performing well at low concentrations under a wide range of conditions, even at low doses. In Tahoe Basin tests reported by Trejo-Gaytan et al. (2006), both actual stormwater from the area and synthetic stormwater was used, with TP of the influent ranging from 48 to 340 µg /L. A wide range of coagulants were tested, and the largest differences were found in suspended solids removal, rather than for TP removal.

Although the PACL coagulants were the most effective, all four types tested were effective in reducing turbidity and P. Overall, coagulation reduced mean turbidity and P by 85–95 percent within 10 hours of dosing, compared to 20 and 55 percent reductions in turbidity and P, respectively, for nontreated water over the same amount of time. To achieve equivalent treatment levels, an order of magnitude increase in time was required for the nontreated water.

Bachand et al. (2010) concluded that, whereas most treatment basins and wetlands will not effectively remove fines and TP within a 24-hour hydraulic residence time, those which utilize coagulant dosing should effectively remove fines and TP. Bachand et al. (2010) further concluded that, given the paucity of available acreage in the Tahoe Basin and its high cost, coagulant dosing systems could be retrofitted to existing treatment basins and wetlands, enabling these treatment areas to be more effective in targeting P and fines, providing effective treatment of runoff waters from areas two or three times greater than currently exists.

The low intensity chemical dosing system in the Everglades program using ferric or aluminum chloride did not perform better than an ordinary emergent wetland. In separate but related studies, Bachand et al. (1999) and Coffelt et al. (2001) (as cited in SFWMD 2002) showed that higher doses (20 mg/L as Al) achieved 20 to 30 µg /L, but would still not achieve a 10 µg /L target.

5.2.7.3 Advantages and Disadvantages

The key principal of LICD is to minimize coagulant dosing while still achieving robust coagulant performance for varying operational and environmental conditions. Thus, the principal advantage of LICD is that coagulant dosing is effective but minimized, which should reduce potential environmental effects because flocculate will be minimized and efficiently incorporated into sediments and soil. Lower dosing means less chemical coagulants are used, and because less flocculate is formed, related maintenance to manage or remove the flocculate is also reduced.

The minimizing of dosing also presents the principal disadvantage of LICD. That is, minimal dosing implies that the injected amount of coagulant is nearer to a threshold of effectiveness. In LICD, management of selected coagulants is more challenging and demanding, requiring that the most appropriate chemicals are selected and that the requirements for their application (e.g. mixing rate and duration, dose, pH) be met. It also necessitates that dose is regulated based upon flow or other parameters, and adjusted when water conditions such as quality, temperature, and hydraulics change.

5.2.7.4 Associated Costs

Information is not readily available on costs associated with use of LICD in conjunction with constructed treatment wetlands. However, it is expected that overall costs would be equivalent to, if not less than, costs of various chemical treatments as previously described. For example, Bachand et al. (2000) indicates that implementation of LICD requires higher absolute capital costs for incorporating facilities and practices to optimize chemical usage in this technology. However, because of much lower dosing levels, LICD systems produce much less flocculate and thus have lower maintenance costs. In other chemical application technologies, chemical utilization is much less efficient, which results in increased sludge production and associated management costs. Also, the higher metal dosing levels in other chemical application technologies do not necessarily lead to better P removal and thus may provide no added function aside from increasing costs of materials and sludge management. As a result of these observations, Bachand et al. (2000) concluded that the costs for P mass removal using LICD likely is lower than the costs associated with other chemical application technologies (assuming use of equivalent coagulant chemical types).

5.2.7.5 Applicability in the Upper Klamath Basin

The Tahoe Basin studies are relevant to Klamath River system in that the climate is roughly similar to the Upper Klamath River system. Phosphorus levels faced by the Everglades program are similar to those in the Upper Klamath basin. The Bottcher et al. (2009) study concluded that low concentrations of coagulant in wetland influent would not provide much reduction in total P, while the Bachand et al. (1999) and Coffelt et al. (2001) studies showed that four-times higher doses (20 mg/L as Al) could produce total P reductions of up to about 80 percent. This indicates that large dosages (beyond those anticipated in an LICD approach) may be necessary for effective augmentation of treatment by coagulants in such eutrophic environments. Moreover, the association of sulfate from the alum with methylation of mercury in wetland systems warrants further examination prior to implementation. The aluminum chloride coagulant used in the Everglades program could be considered in the Upper Klamath basin, but increased chloride levels may not be a desirable water quality effect, considering irrigation users downstream.

5.2.8 Wetlands Soils Amendment

5.2.8.1 Description

Wetland soils and more specifically, their organic matter content have a key role in a wetland's ability to reduce pollution (Kadlec and Wallace 2009). As a predominant soil characteristic, organic matter content and quality influences a number of physical, biological, and chemical properties in wetland soils. Organic matter has a high water holding capacity, which may perpetuate the saturated conditions that enhanced organic matter accumulation. Organic matter also retains exchangeable nutrients such as NO_3 and phosphate.

The ability of wetland soils to retain P in particular can vary depending on the net equilibrium P concentration – defined as the P concentration where the amount of P sorption is equal to the amount of P desorption, resulting in a net P adsorption of zero. If the net equilibrium P concentration is greater than the P concentration in solution, the sediment (soil) will release P (House and Denison 2002, Zhou et al. 2005).

Chemical treatments can be used prior to construction of treatment wetland systems if wetland soils are already high in P. For example, the addition of alum to agricultural land prior to construction of treatment wetlands in Ireland was successful in binding the labile P and preventing it from entering the water column (Babatunde et al. 2008). In addition, aluminum-containing residuals from drinking water plants have been examined as a tool to reduce releases from soils that have accumulated high levels of P (Rew 2006). The residuals are effective in reducing P releases, but Rew (2006) noted that to provide maximum effectiveness, alum residuals would need to be incorporated to the full depth of soil with elevated levels of P.

Also, as described in section 5.2.3 (*Application of Calcium-Based Amendments*), limestone and lime applications to wetlands soils could potentially be used to enhance P removal. Liming of acid soils provides a rich source of calcium to reduce soil acidity and enhance P retention. In general, increasing soil pH (from acidic to more neutral) can lead to increased mineralization of P due to increased soil microbial activity (Follett et al. 1981). Phosphorus can react with increased Ca in solution, increased Ca sorbed on soil surfaces, or directly onto particles of $CaCO_3$. For the latter process, very small particle size limestone is a clear advantage (Follett et al. 1981). Resultant amorphous Ca-P solids gradually transition to the more stable and much less soluble forms (e.g., hydroxyapatite) (Snoeyink and Jenkins 1980).

5.2.8.2 Effectiveness

Studies at Lake Apopka in Florida have examined the effectiveness of a number of soil amendments to limit P mobilization from former agricultural soils converted to treatment wetlands (Ann et al. 2000a). The focus was on P released to the water column above the soil under anaerobic conditions. Important findings of this study were as follows:

- Ferric chloride and alum were the most effective amendments, followed by lime, then limestone. Dolomite was ineffective.
- High rates of amendments were required because of binding of amendment cations (Ca, Fe, Al) with organic matter.
- With Ca amendments, the optimal effect was acheived at about 150 mmol Ca/kg soil. This was achieved with the equivalent of about 15.3 tons/ac of limestone alone, or 7.6 tons/ac limestone plus 3.8 tons/ac hydrated lime ($Ca(OH)_2$). These rates are very high compared to typical agricultural rates. For long-term treatment, achieving pH 8 may be necessary to form more stable Ca-P compounds, but increased solubility/dispersion of organic P at these pH levels is a risk. At least within the time frame of the study, amorphous Ca-P compounds dominated—there was little conversion to hydroxyapatite. Organic ligands such as organic acids, as well as Mg and carbonate can prevent crystal growth.
- With alum, the optimal effect was achieved at about with the equivalent of about 3 tons/ac, assuming a soil density of 0.28 g/cm^3 and a soil depth of 20 cm

In a parallel study, Ann et al. (2000b) examined changes in soil redox potential on P solubility for $CaCO_3$, alum, and ferric chloride amendments. Although ferric chloride was very effective in removing P in the 2000a study, the 2000b study found that sorption was strongly affected by changes in redox. In contrast, P sorption was minimally affected by changes in redox with the $CaCO_3$ and alum amendments.

In general, the liming of acid soils can result in the solubility of all of the calcium phosphates decreasing as pH increases (Lindsay 1979). High inorganic P retention is observed especially where alkalinity and Ca are high (Reddy et al. 2005). Apatite can provide a gradual source of P in acid soils, and so a slow release of P would be expected if soils amended with lime, creating hydroxyapatite, are allowed to acidify by natural processes.

5.2.8.3 Advantages and Disadvantages

The principal advantage of wetlands soils amendment is that it can substantially improve the soil and water quality functions of constructed treatment wetlands, particularly processing, retention, or sequestration of nutrients. Under the right circumstances, wetlands soil amendment can accelerate water quality functioning in constructed wetlands by improving initial soil conditions.

Disadvantages of wetlands soils amendment mainly relate to uncertainties related to effectiveness of these amendments to inactivate P in wetlands, the longevity of treatment, and the potential effects on flora and fauna. The long-term efficacy of a one-time application of amendments is unknown. Repeated applications may be needed, which could be costly and result in soil buildup of chemicals contained in the amendment formulation.

5.2.8.4 Associated Costs

Specific cost information on wetlands soils amendment is not provided in this report since costs can vary substantially depending on specific nature of the amendments, the amounts involved, site characteristics, and other factors. Costs associated with soil amendments will include the amending materials and the methods of integrating the amending materials into the existing soil. Costs associated with applying the soil amendments will include transport and site access, site preparation (e.g., grading and tilling), application (e.g., blower application), and maintenance activities.

5.2.8.5 Applicability in the Upper Klamath Basin

Soils in the Upper Klamath basin are naturally high in P. Amending the soils to reduce available P prior to wetland construction may be helpful to prevent P release upon flooding, and ensure net P removal. Alum or alum water treatment residuals appear to be preferred as compared to Ca or Fe amendments, as there is minimal pH or redox effects on P sorbed to alum.

Limestone and lime applications could potentially be used to enhance P removal. However, aluminum-based approaches (alum, PACL, $AlCl_3$), such as described above, have a better established track record for P removal. The large increase in acidity associated with aluminum-based approaches would not be a factor for calcium based amendments, but very high pH may be needed to obtain P removal.

5.3 Combined Chemical/Physical Treatment Approaches

Although the Klamath River is not “wastewater”, methods used for P removal in wastewater applications provide a useful basis for considering alternatives. Wastewater treatment plants (WWTPs) often use a combination of chemical, biological, and physical methods to achieve very low levels of P. The most likely approach for flows in the Klamath Basin would be chemical/physical, injecting alum or $FeCl_3$ into the flow from a river diversion, followed by a settling pond. Both continuous amendment addition and batch treatments applied from a boat have been used to remove P in lagoon systems. Effluent from the settling pond would be the influent to the wetland system. Biological, filtration, and membrane-based methods are often used in advanced wastewater treatment, but they are not likely practical for the large flows and low P concentrations in Klamath Basin surface waters. Therefore, they are only briefly described here for completeness.

5.3.1 Chemical Treatment Combined with Settling and Solids Separation

5.3.1.1 Description

Chemical pretreatment and settling upstream of a treatment wetland system can be used to enhance efficiency and effectiveness of treatment wetland systems. Chemical pretreatment approaches borrow from technologies used for conventional wastewater treatment. Chemical processes for P removal commonly rely on the formation of sparingly soluble orthophosphates (Sedlak 1991). The use of a liquid-solid separation unit process (primary or secondary clarifiers; lagoons) allows settling and removal of precipitated P. The most common chemicals used for the precipitation of P are alum ($Al_2(SO_4)_3{\cdot}18\ H_2O$)[6] and ferric chloride ($FeCl_3$). Variables that are typically considered are cost, alkalinity consumption, amount of sludge generated, and safety issues. Two major decisions must be made when implementing chemical P removal: (1) selection of the chemical to be used; and (2) deciding at which point(s) in a treatment process the chemical is to be added.

When metal salts are added to wastewater, the precipitated phosphate is in the form of a metal phosphate. For ferric chloride addition, the metal phosphate has a composition of $Fe_{1.6}H_2PO_4(OH)_{3.8}$ and when alum is used, $Al_{0.8}H_2PO_4(OH)_{1.4}$ is formed (Sedlak 1991). Alum is more commonly used (Metcalf & Eddy 1991, Bottcher et al. 2009). When only moderate P removal is required, the above species are the predominant complexes formed. However, when lower soluble P concentrations are required, the formation of metal hydroxides becomes an important reaction, and larger chemical dosages are required. The formation of metal hydroxides also results in a significant production of additional solids. Both iron and aluminum coagulants consume alkalinity and decrease pH (Bottcher et al. 2009).

5.3.1.2 Effectiveness

The actual required dosages of chemical necessary to achieve a desirable effluent soluble P concentration should be determined with jar tests with the water or wastewater to be treated. The results from jar tests provide a reasonable indication of the required dosages, but only full-scale application allows selecting the optimal dosage. Dosage commonly needs to be adjusted to allow for variability in the water to be treated. For stormwater in Florida, the minimum dosage of alum is about 5 mg Al/L to produce an acceptable floc (Bottcher et al. 2009). Alum as a liquid is 48.5 percent aluminum sulfate by weight, and the typical dosage is 5 to 10 mg Al/L to achieve adequate floc formation and settling (Harper 2007). Dosage requirements of up to 20 to 25 mg Al/L for stormwater are sometimes considered, based on data summarized in Bottcher et al. (2009).

As a rule of thumb, to achieve effluent levels lower than 0.5 mg/L, values as high as 6 moles metal (Al or Fe)/mole P may be required (Sedlak 1991), in addition to filtration (Metcalf & Eddy 1991). Since peak P concentrations in the Klamath River near Upper Klamath Lake are less than 0.5 mg/L, high chemical doses would likely be required to achieve treatment. In wastewater applications, anionic polymer is often needed to aid in flocculation and to avoid the formation of “pin-floc” that stay suspended. Operations that are favorable for the formation of “sweep floc” are favored for P removal (Metcalf & Eddy 1991).

Chemical pretreatment and settling upstream of a treatment wetland system has been the subject of studies in the Florida Everglades (SFWMD 2002). An example application of this approach involved chemical addition, flocculation, and settling of preceding inflow to a cattail-dominated wetland. A schematic of this system is shown in Figure 19. The downstream constructed wetland did not provide additional P removal, but did provide desirable adjustments to water chemistry (pH, alkalinity, etc.)(Bottcher et al. 2009).

[6] The number of water molecules may vary from 14 to 18.

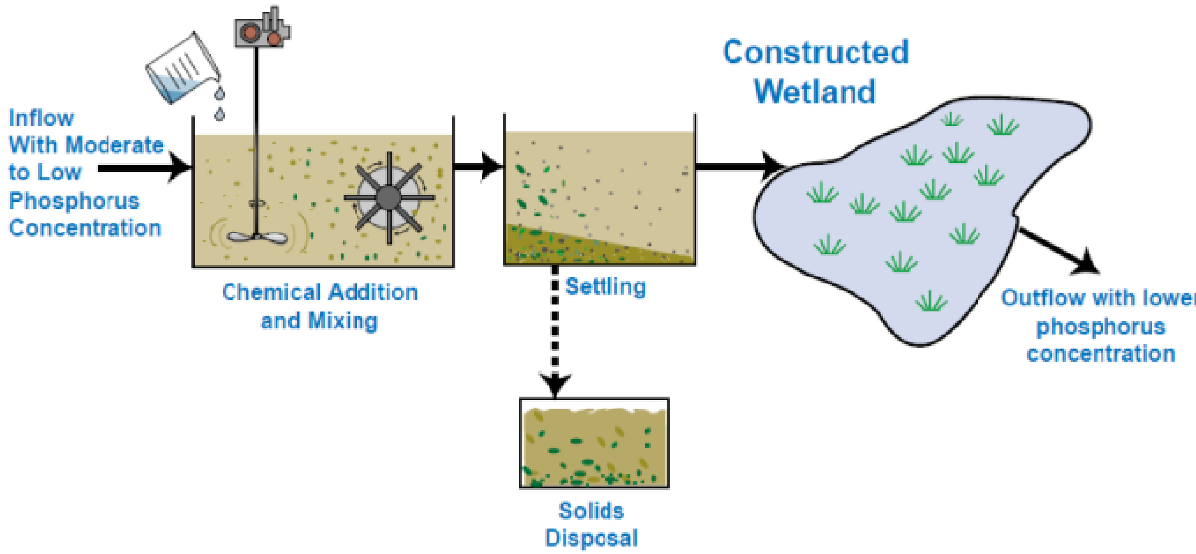

FIGURE 19
Approach to chemical pretreatment and settling upstream of a treatment wetland system used in Florida (SFWMD 2002).

In testing this system, the HLR was 10 cm/d, water depth was 30 cm, and the nominal HRT was 2.6 days (SFWMD 2002). Ferric chloride and poly-aluminum chloride (PACL) were the coagulants tested. Influent concentrations averaged 112 µg TP/L at the north test cell, and 55 µg TP/L at the south cell. Results were variable. None of the systems tested achieved the target effluent level of 10 µg/L. The control system (no chemical addition, wetland only) resulted in 33 µg /L discharge, whereas $FeCl_3$ and PACL treatments resulted in 18 and 16 µg/L respectively (SFWMD 2002). Therefore, chemical treatment improved P removal as compared to an emergent treatment wetland alone.

Capture of floc by the settling tank was only partial, due to system design and operational constraints, but the downstream wetland effectively captured the fugitive floc. Residual solids were determined to be non-hazardous, and wetland outflows did not have apparent biotoxicity (SFWMD 2002).

In a second case in the Florida Everglades (SFWMD 2002), the addition of Fe or Al salts, along with organic polymers, followed by high-rate solids separation techniques were applied at locations both upstream and downstream of emergent vegetation SF wetland systems. Figure 20 illustrates the approach.

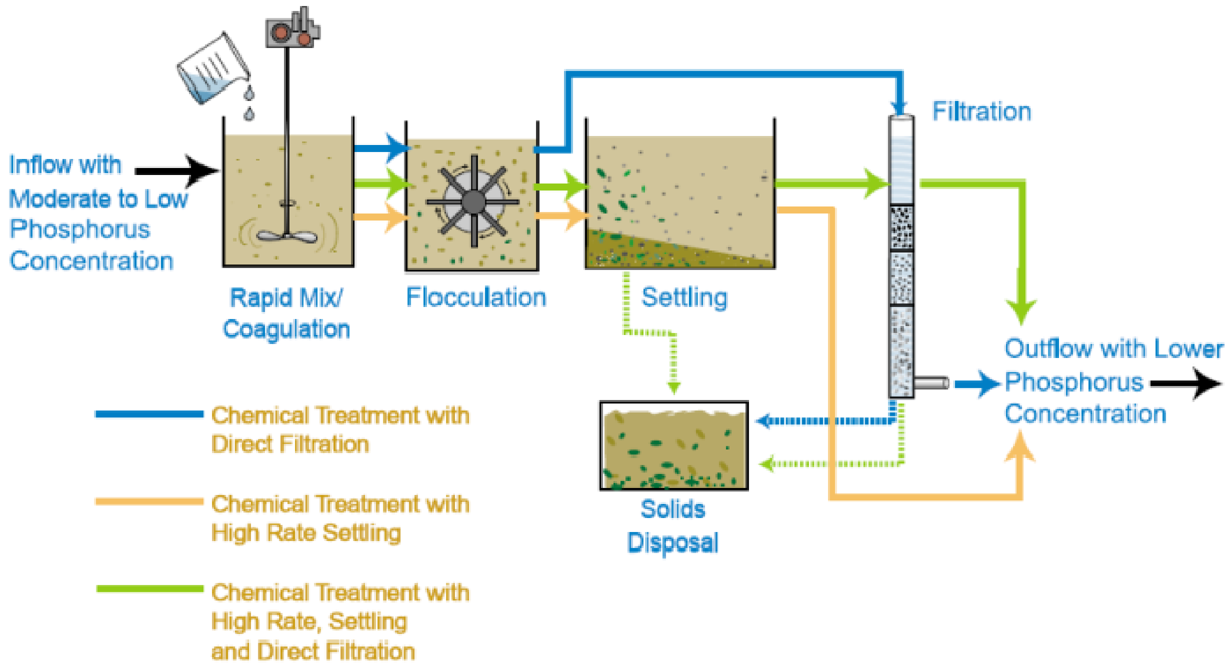

FIGURE 20
Approach to chemical pretreatment and solids separation with a treatment wetland system used in Florida (SFWMD 2002).

Aluminum chloride, alum, $FeCl_3$, and PACL all provided good results (i.e., met 10 µg TP/L standard), but alum was removed from consideration due to a concern with elevated sulfate favoring the formation of methylated mercury (Hg) (SFWMD 2002). $FeCl_3$ was eliminated due to the potential for redox fluctuations to release P, and PACL was eliminated because $AlCl_3$ was less costly.

The plan was that settled solids would be land applied. Treatment scenarios were prepared for 80 to 380 mgd of flow, depending on the fraction of total flow treated. The estimated area required for land application of solids was 1150 to 1680 acres, based on an assumed annual loading rate of 28 dry tons/ac/yr.

5.3.1.3 Advantages and Disadvantages

The principal advantages of chemical treatment combined with solids separation is relatively low land requirements, operational flexibility, reliability, and the ability to reduce nutrient levels substantially lower that can be achieved using other technologies.

The principal disadvantages of chemical treatment combined with solids separation are relatively high capital, operations, and maintenance costs, as well as solids management and disposal requirements.

5.3.1.4 Associated Costs

In the second case from the Florida Everglades (SFWMD 2002) as discussed above, costs were estimated for full-scale implementation of the chemical treatment with solids separation technology assuming an objective of reducing the Total P content of treated surface waters to concentrations of 10 µg/L or less. This facility would be capable of treating an average of 380 million gallons per day (mgd) of surface waters with continuous production of a 10 µg/L Total P effluent with no flow diversion or by-pass.

SFWMD (2002) estimated that such a system (including the associated emergent vegetation SF wetland system, but not including a dedicated land application area) could be constructed for a total cost of $428 million (estimated in 2002 dollars based on 50-year present worth calculations). SFWMD (2002) estimated that capital costs would be about $204 million, consisting mainly of land cost, and construction and other civil work. Operation and maintenance costs were estimated to comprise the other $224 million. Chemical costs were estimated to be 70 percent of total operating costs. Other operating costs included energy consumption, residuals management, sampling, and monitoring.

SFWMD (2002) estimated costs of some other facility scenarios to assess economy of scale. For example, the cost for a 120 mgd facility producing a 20 µg/L Total P effluent and with a 20 percent flow diversion was estimated at $304 million (with $145 million in initial capital costs). Capital costs range from $1.40/gallon for 100 mgd plant to $0.50/gallon for a 350 mgd treatment plant. Figure 21 illustrates the general lack of economy of scale per pound of P removed. The study found that the cost effectiveness of chemical treatment is greater with higher influent concentrations.

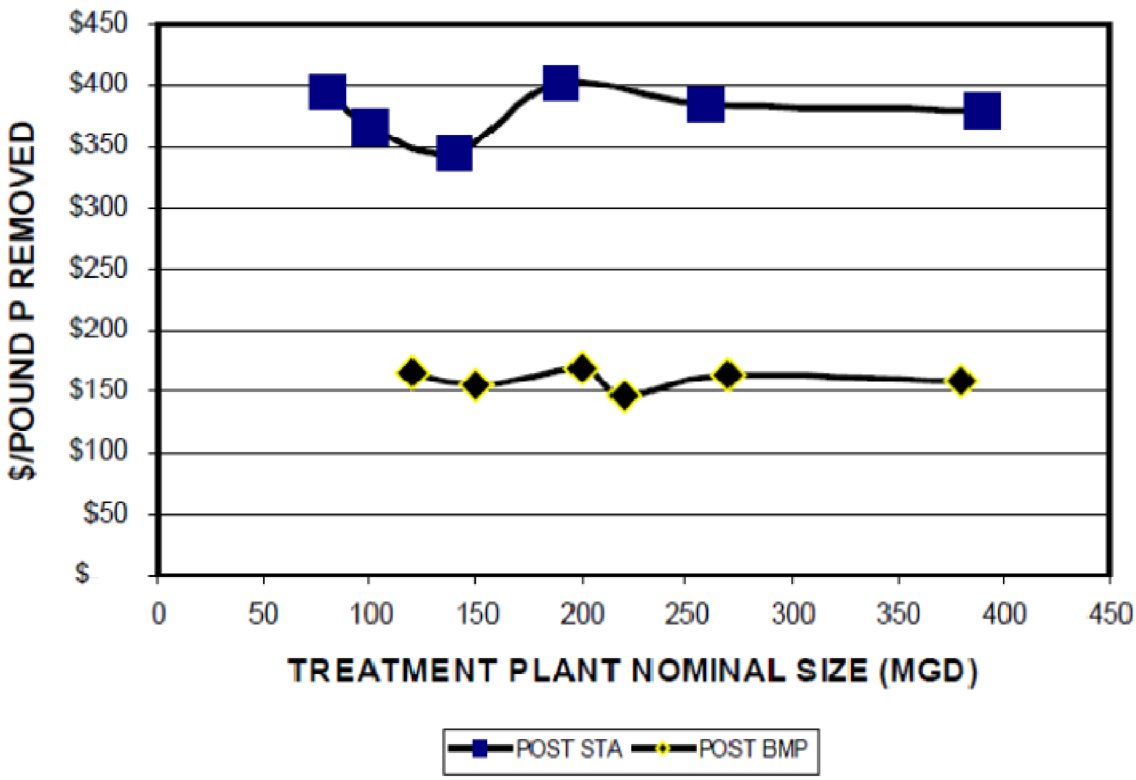

FIGURE 21
Present worth cost per pound of P removed in large scale chemical treatment with solids separation (SFWMD 2002).
(Post-STA represents placement of the treatment facility downstream of treatment wetlands, and Post-BMP represents placement of the treatment facility upstream of treatment wetlands, and downstream of agricultural areas implementing BMPs).

5.3.1.5 Applicability in the Upper Klamath Basin

Chemical treatment combined with settling and solids separation could be considered for implementation in the upper Klamath Basin. Wetlands downstream of an alum treatment system would be desirable, even if all of the P removal were to occur in the alum system. Downstream wetlands help to capture alum flock that does not settle well (pin flock) and to provide some adjustment in pH and alkalinity before the flow returns to the river.

Extrapolating the present worth calculations above to the mean Klamath River flow (at the location represented by Link River dam) of 1164 cfs, the 50-year present worth of this kind of system would be on the order of $1.3 billion. In addition, these studies show that these adjunct chemical/physical treatment techniques would likely be more cost effective upstream from wetlands, rather than downstream from wetlands.

5.3.2 Hybrid Wetland Treatment Technology

5.3.2.1 Description

Hybrid Wetland Treatment Technology (HWTT) involves a treatment wetland system that uses a combination of wetland and chemical treatment approaches in a series of wetland zones or cells. Chemical coagulants are added, either continuously or intermittently, to the front end of the treatment system, which contains one or more deep zones or cells to capture the resulting floc material. A fundamental concept of the HWTT is that the floc resulting from coagulant addition generally remains active and has the capability of additional P sorption. Both passive and active reuse of floc material is practiced in HWTT. Passive reuse refers to the settling of active flocs of plant roots and stems, where it can contact additional untreated aliquots of water. Active reuse refers to the mechanical resuspension of previously settled floc.

HWTT using alum has been tested in the northern Everglades watershed, near Lake Okeechobee (Watershed Technologies 2008). The process is covered by two U.S. Patents (7,014,776 and 7,179,387). As an example, the HWTT system at Nubbin Slough in Florida is shown in Figure 22. The basic layout of an HWTT involves an initial contact cell (deep zone), where alum is mixed with the inflow, a shallow “filtration cell”, partially covered by floating aquatic vegetation (FAV) (water hyacinth), and a final settling zone (deep zone). Deep zones at some sites incorporate submerged aquatic vegetation. Floating mat designs, contained by a circular floating boom, have also been tested (Bottcher et al. 2009). The FAV cell in the HWTT approach is designed to provide quiescent conditions and extensive root surface area in the water column to promote settling of alum floc and removal of particulate P. Floating booms and barriers are used as needed to force flow by cell to the surface, to the bottom, or to contain floating vegetation (Watershed Technologies 2008).

FIGURE 22
Hybrid Wetland Treatment Technology (HWTT) facility at Nubbin Slough in Florida (source: South Florida Water Management District).

Floc reuse is a component of the HWTT system to further reduce chemical requirements. Settled floc still has significant P removal capacity, and can be reused (Bottcher et al. 2009). During the dry season, water levels are lowered, and excess settled floc from the contact cell is removed and applied to the shallow filtration zone, and is incorporated into the soil along with FAV debris prior to reflooding. Periodic resuspension of settled floc for additional P removal is described in DeBusk's U.S. Patent 7,014, 776. The vegetation in the FAV zone is also used to inhibit phytoplankton growth, and thereby limit the formation of additional particulate P.

5.3.2.2 Effectiveness

In general, soluble reactive P is the easiest to remove in the HWTT system, followed by particulate P, followed by dissolved organic P (Bottcher et al. 2009). In the HWTT system tested near Lake Okeechobee, inflow and outflow concentrations and percentage removals at four test sites were as follows (Watershed Technologies 2008):

- Test Site 1 – From 754 to 118 µg P/L (84 percent reduction, alum at 4 mg Al/L and sodium aluminate at 8 mg Al/L)
- Test Site 2 – From 151 to 21 µg P/L (86 percent reduction, alum at 20 to 25 mg Al/L, but also found that intermittent dosing reduced total alum usage by 1/3 but provided similar removals)
- Test Site 3 – From 492 to 0.35 µg P/L (93 percent reduction, alum at 20 to 25 mg Al/L)
- Test Site 4 – From 16,700 to 950 µg P/L (94 percent reduction, alum at 200 mg Al/L; recycling floc produced greater reductions in final P)
- Ideal #2 Grove, St. Lucie County FL – From 202 to 15 µg P/L (continuous dosing) or 17 µg P/L (intermittent dosing) (Bottcher et al. 2009)

To mitigate the large drop in pH associated with alum addition, two different approaches were used. Injection of sodium aluminate (8 mg/L) at the upstream end was one approach, and the other was a final pH buffering cell consisting of submerged aquatic vegetation and limestone (referred to in the reports as 'limerock') (SAV/LR). Both approaches were successful, but the chemical approach was more costly than the limerock. The SAV/LR approach has also been used in the Everglades stormwater treatment areas (STAs) (Watershed Technologies 2008). In that application, the limerock component has been controversial, as it would be extremely expensive on a large-scale. However, SAV systems alone can reliably alkalize the water by 0.5 to 1 pH unit, at least partially reversing the acidification caused by alum addition.

5.3.2.3 Advantages and Disadvantages

The principal advantage of HWTT systems is the combining of the strengths of treatment wetlands and conventional chemical treatment systems to provide removal efficiencies beyond those attainable by either separate technology. Phosphorus removal in HWTT systems is markedly higher than in treatment wetlands due to the use of chemical amendments. Unlike conventional chemical treatment systems, however, HWTT systems incorporate several design and operational components that minimize amendment use. These include: (1) passive and active recycling/reuse of chemical flocs; (2) sequencing and configuring of the wetland unit processes to provide desirable P species transformations; (3) use of wetland components, rather than chemical amendments, for pH buffering; and (4) utilization of the wetland biota to transform/remove additional contaminants, such as N.

The principal disadvantage of HWTT systems is similar to the disadvantages previously described for chemical treatment systems, since HWTT systems employ a chemical treatment component. Of note, however, is that HWTT systems are typically designed to operate on a range of coagulant types. As such, chemical use can be selective and more flexibly managed.

5.3.2.4 Associated Costs

Specific cost information for HWTT systems was not obtained for this report. It is noted that the State of Florida and SFWMD reported expenditures totaling $10.5 million from 2008 to 2011 for construction and operation of six HWTT facilities for the Northern Everglades and Estuaries Protection Program (South Florida Ecosystem Restoration Task Force 2011)

5.3.2.5 Applicability in the Upper Klamath Basin

If alum if used as a means of further reduced P in the Upper Klamath basin, alum recycling may be a useful approach to limit residuals and cost of alum inputs. The implementation and effectiveness of FAV and SAV systems has not been demonstrated at a watershed or basin-level scale. Test Site 2 results from Lake Okeechobee as discussed above is most similar in concentrations to the TP along the Klamath River, where values range from a seasonal average of 160 µg /L TP up to 280 µg /L TP in July, just below Upper Klamath Lake, and a seasonal average of 150 µg /L TP up to 191 µg /L TP in September at Hatchery Bridge below Iron Gate Dam. However, additional research and development would be needed before implementation in the Upper Klamath basin. The limestone often used for these systems is not readily available in the region, and transport likely would be costly. The amendments often used for these systems could cause undesirable increases in sodium levels in the river.

5.3.3 Large-Scale Alum Injection and Treatment Wetland Settling System

5.3.3.1 Description

A noteworthy large-scale application of chemical treatment combined with solids settling and solids removal in the context of a wetland treatment system is Lake County Water Authority's Nutrient Reduction Facility (NuRF) in Florida. The NuRF system uses off-line liquid alum injection, followed by pond settling to treat the outflow from the hypereutrophic Lake Apopka, upstream of Lake Beauclair in central Florida (Harper 2007). Discharge from Lake Apopka is the single largest controllable source of pollution in Lake County. Alum was selected because of its reliability and history of successful use in many different water treatment applications. It is the largest alum treatment system for runoff treatment in Florida as of 2009 (Bottcher et al. 2009).

Figure 23 shows an aerial overview of the NuRF system and its facilities. Flows from Lake Apopka are diverted into the NuRF system's inflow canal (in quantities ranging from 5 to 300 cfs) and injected with an alum solution pumped from a large alum storage facility (as shown in Figure 23). Alum floc is formed as the flow enters two 9-acre treatment ponds where the floc settles out. Detention time is about 3 hrs (Bottcher et al. 2009). Alum floc is removed from the ponds with a dredge about once or twice per year, and is dewatered using a centrifuge (at the floc dewatering facility shown in Figure 23) to a water content of 40 percent (Bottcher et al. 2009). Following centrifuge dewatering, the dewatered alum sludge (Figure 24) is placed on permeable soils were it continues to dry and consolidate (Bottcher et al. 2009). The sludge finally is land applied to take advantage of the additional P sorption capacity of the residual material (Harper 2007).

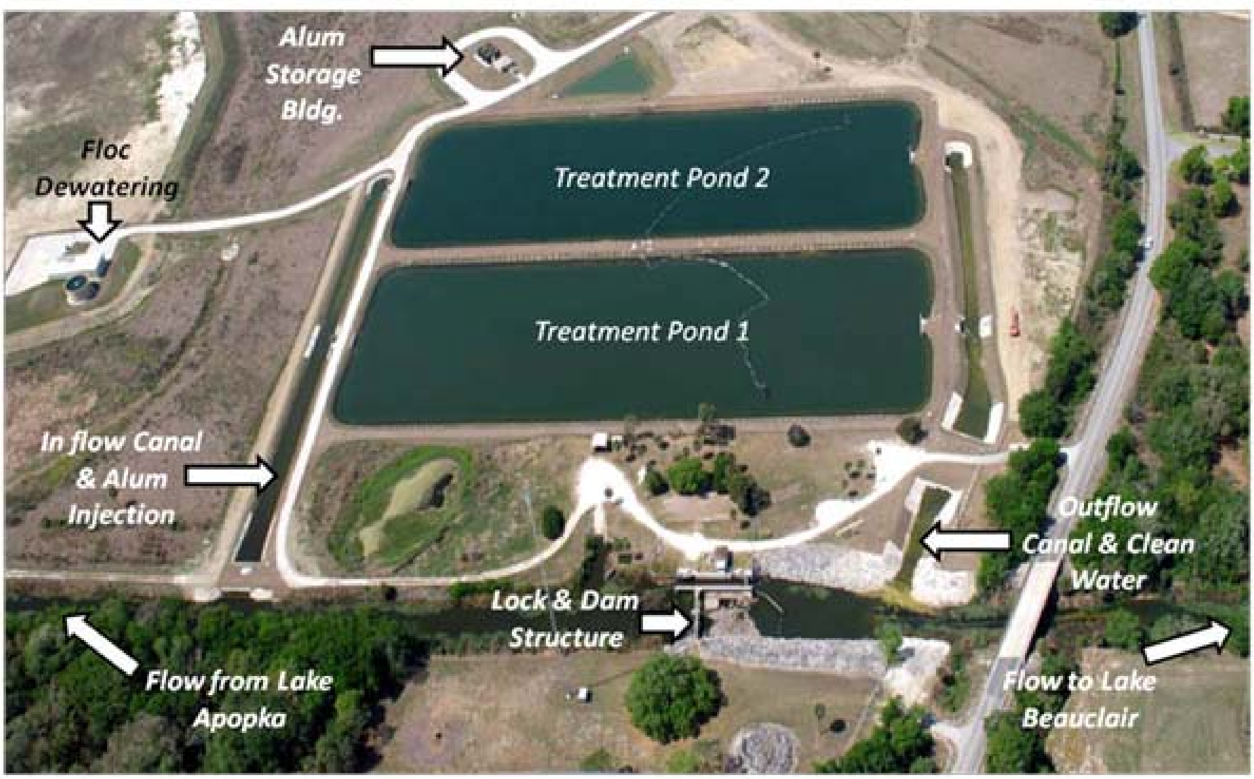

FIGURE 23
Aerial photograph of the Nutrient Reduction Facility (NuRF) in Florida (source: Lake County Water Authority).

FIGURE 24
Dewatered alum sludge from the Nutrient Reduction Facility (NuRF) in Florida (source: Lake County Water Authority).

5.3.3.2 Effectiveness

Treatment reliability and ability to treat substantial flow volumes were key reasons this alum system is used. Nutrient removal is approximately 67 percent. Approximately 90 percent of the flow in the canal is treated by the system. At maximum design flow of 300 cfs, that facility uses about 36,000 gal of alum per day (Bottcher et al. 2009). Flows in excess of 300 cfs are allowed to bypass the system (Bottcher et al. 2009). Elevation differences eliminate the need for pumps.

5.3.3.3 Advantages and Disadvantages

The NuRF provides a large-scale scale example of chemical treatment combined with solids settling and solids removal. As such, the advantages and disadvantages of the NuRF system are similar to those described in section 5.3.1.3 above. Given its relatively large scale, the principal advantage of the NuRF system is to provide mass removal of P, and reductions in effluent P concentrations, that are beyond those attainable by other treatment wetland technologies. The principal disadvantages are relatively high capital, operations, and maintenance costs, as well as solids management and disposal requirements.

5.3.3.4 Associated Costs

The capital cost for the NuRF system was about $7.1 million. Annual operation and maintenance costs are approximately $1 million.

5.3.3.5 Applicability in the Upper Klamath Basin

If the same dosage were applied to the entire ~1000 cfs flow of the Klamath River, it would require ~120,000 gal of alum per day of operation. Scaling these costs to Klamath River flows, assuming no economies of scale, the cost would be on the order of $20M, not accounting for inflation.

To check these results with Klamath River flows and chemistry, CH2M HILL's wastewater treatment plant design model Pro2D[7] was used. Figure 25 shows results for TP, and Figure 26 show results for TKN (most river TN is organic and NH4-N, and NO3-N is not significantly affected by alum treatment).

These results confirm general Florida recommendations for treatment of P in stormwater, which suggest at least 5 mg/L as Al will be required. Dosages of 20 mg/L or more may be required to come close to the 10 µg /L goal required for inflows to the Everglades. If the 10 mg/L as Al dose used in the NuRF is incorporated into the model, the model suggests that significantly less solids would be produced (3500 CY/d at 4.5 percent TS in the model vs. 10,400 CY per day after 30 days settling for NuRF). However, it should be noted that the TS after 30 days of settling is not defined in Bottcher et al. (2009), and that the estimated volumes are of roughly the same order of magnitude.

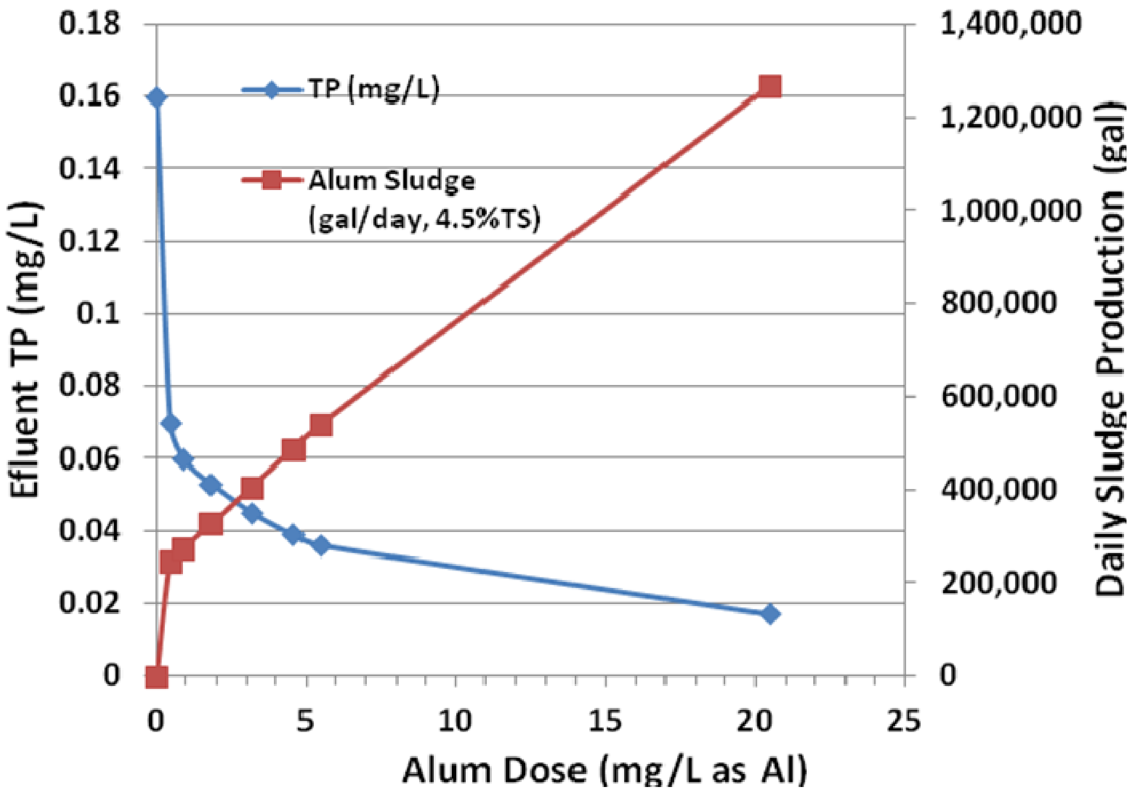

FIGURE 25
Summary of expected total phosphorus (TP) removal and sludge production as a function of alum dose.
(Note: Results assume treatment of full Klamath River flow [1164 cfs], and are derived from CH2M HILL's Pro2D Wastewater Treatment Plant Design Tool, with typical Klamath River chemistry.)

Results in Table 14 highlight the large volume of sludge generated at higher dosages of alum. Model results suggest significant improvements in water quality can be achieved at relatively modest doses of alum. The model suggests a 5 mg/L as Al dose will reduce TP to 0.036 mg/L and TKN to 1.03 mg/L. The estimated volume of dewatered sludge (85 percent TS) at this dosage that would need to be land applied, assuming a 180 day period of operation, would be 25,380 CY, or roughly 19,000 tons of alum residuals, assuming a specific gravity of about 0.9 g/cm^3. If the land application rate were 2 tons/ac, roughly 8500 acres would be required. Alum dosages could potentially be reduced significantly by placing treatment cells in series, and recycling alum floc upstream.

[7] Pro2D (**Pro**fessional **Pro**cess **D**esign) is a steady state wastewater treatment plant simulator that has been developed by CH2M HILL to perform complete treatment plant simulations. PRO2D is used to develop the overall mass balance for a treatment plant and to size individual unit processes. Influent wastewater constituents are divided into fractions which facilitate mass balance calculations. Liquid/solids separation unit process performance parameters (e.g., TSS removal efficiencies) are entered into PRO2D, which then performs all of the recycle calculations and automatically returns a completed process flow and mass balance for the wastewater treatment plant. Flow and mass balances are computed for both average and diurnal peak operating conditions, as defined by specified process loading and operating conditions.

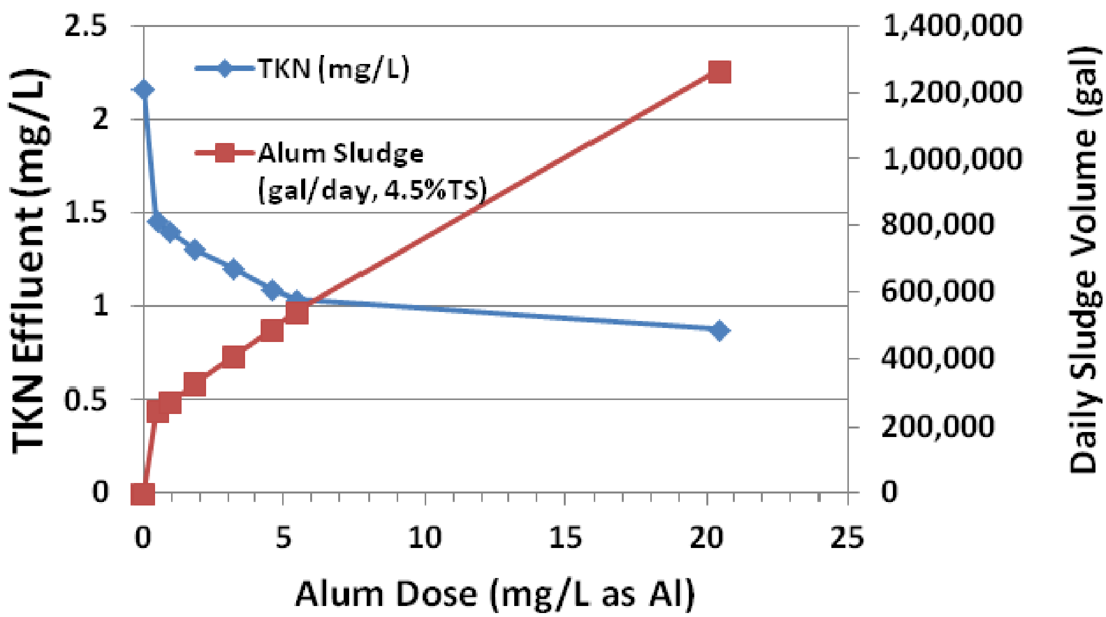

FIGURE 26
Summary of expected TKN removal and sludge production as a function of alum dose.
(Note: Results assume treatment of full Klamath River flow [1164 cfs], and are derived from CH2M HILL's Pro2D Wastewater Treatment Plant Design Tool, with typical Klamath River chemistry.)

TABLE 14
Summary of Estimated Alum Dosages, Sludge Volumes, and Effluent Water Quality
(Based on average upper Klamath River concentrations, and Pro2D Model results)

Parameter	Liquid Alum Dose (mg/L)							
	0	**5**	**10**	**20**	**35**	**50**	**60**	**225**
	Liquid Alum Dose (mg/L as Al)							
	0	**0.5**	**0.9**	**1.8**	**3.2**	**4.5**	**5.4**	**20.4**
Alum Sludge (gal/day, 4.5% TS)	0	246,000	273,000	326,000	406,000	486,000	539,479	1,264,880
Alum Sludge (CY/day, 4.5% TS)	0	1,218	1,352	1,614	2,010	2,406	2,671	6,263
Alum Sludge (CY/day, 35% TS, centrifuge)	0	157	174	208	258	309	343	805
Alum Sludge (CY/day, 85% TS, drying beds)	0	64	72	85	106	127	141	332
TKN (mg/L)	2.16	1.45	1.4	1.3	1.2	1.09	1.03	0.87
TP (mg/L)	0.16	0.07	0.06	0.053	0.045	0.039	0.036	0.017

5.3.4 Advanced Treatment Using Chemical Treatment Combined with Filtration

5.3.4.1 Description

Filtration, in combination with chemical treatment, has been examined as an alternative advanced treatment tool to remove particles and suspended solids from runoff to the Florida Everglades (SFWMD 2002). Use of advanced treatment technologies is most applicable to situations where it is necessary to achieve very low P (of around 10 μg/L) or to limit the presence of suspended solids in the final effluent. However, the use of advanced treatment technologies is likely to be cost-limiting or cost-prohibitive for treatment of river diversions at large scale.

There are many filtration technology alternatives, including:

- ***Automatic continuous backwash package filters*** – Commonly referred by the supplier name "Dynasand™", although the patent on the technology has expired and other suppliers are available in the marketplace, this deep bed filter is a standard technology for smaller plants. Each filter has a surface area of 4.5 m^2 and a peak capacity of approximately 1.5 megaliters per day (MLD).
- ***Blue Pro®*** – This technology is a significant advancement over the above-mentioned continuous backwash filters, although they are very similar in unit size and configuration. The primary difference with Blue Pro® is that the sand is impregnated with ferric oxide, which improves removal of soluble P.
- ***Deep Bed Mono Media Filters*** – The standard technology of the drinking water industry, these filters have also been used at some of the largest tertiary wastewater treatment plants to provide high levels of TP removal. The technology is often coupled with tertiary clarification (e.g. Actiflo™) ahead of the filters to allow for very low effluent TP performance. The Rock Creek WWTP in Oregon is an example where this treatment train has been used with success.
- ***Tertiary Clarification and Actiflo™/Densadeg™*** – As the name suggests, tertiary clarification is a gravity separation technology used on secondary effluent. High chemical doses are used to drive down the soluble effluent TP concentration. This technology, on its own, cannot reliably achieve effluent TP levels below 0.1 mg/L. Rather, it is typically coupled with other technologies.
- ***Cloth/Fabric Media Filtration*** – This filtration technology is relatively new in the industry, and has been in the marketplace for less than 10 years. The technology has seen wider acceptance in recent years even for large plants, with the largest facility being the Pine Creek WWTP in Calgary, AB (100 MLD first phase capacity currently in operation). Operating experience shows good performance in capturing secondary solids; however, it may be less suitable (due to media fouling) for secondary effluents that have a high proportion of chemical solids.
- ***Membrane Filtration*** – Rather than using sand or fabric as the separation media, this technology uses polymeric membranes to remove solids and TP. Supplemental soluble P removal is provided through alum addition ahead of the membranes.
- ***Membrane Bioreactor (MBR)*** – MBRs integrate membrane technology into the activated sludge process to provide compact secondary and tertiary treatment in one step. MBR technology is the basis for several large treatment facilities in North America, including the 160 MLD Brightwater WWTP (Washington), 120 MLD North Las Vegas WRF (Nevada), 90 MLD Yellow River (Gwinnett County, Georgia), and 75 MLD Frederick County (Maryland) projects.
- ***Tertiary Nitrifying MBR*** – This innovative technology was conceived for the Woodward Avenue WWTP (Hamilton, Ontario) expansion project, which is currently in the pre-design phase. This technology couples nitrification and tertiary P removal in one integrated step. When constructed, the Woodward Ave. WWTP T-MBR will have a peak capacity in excess of 600 MLD.

5.3.4.2 Effectiveness

Testing for the Lake Okeechobee Watershed Project in Florida has shown that treatment wetland outflow subjected to membrane filtration (microfiltration) combined with chemical treatment can consistently produce an outflow TP concentration of around 10 μg /L, with the filtration accounting for roughly half of the removal (SFWMD 2002). Conestoga-Rovers (1998) reported on trials of filtration and chemical treatment to "polish" the outflow from treatment wetlands that had an average total P concentration of 24 μg /L. Membrane filtration units that were tested produced average filtrate concentrations of 11 to 13 μg /L with no chemical addition. Addition of chemical coagulants (i.e., alum, ferric chloride) resulted in final effluent concentration of 8 to 11 μg /L.

Based on these tests, a full-scale wetland treatment system with membrane filtration (microfiltration) combined with chemical treatment was designed that could handle an average daily flow of 175 mgd (about 270 cfs)

(SFWMD 2002). The design assumed that approximately 10 percent of the influent P mass would not be treated to a concentration of 10 µg /L as P during extreme peaks in flow. During these time periods, the “non-treated” wetland discharge (that portion not subjected to filtration and chemical treatment) would be blended with the treated discharge to achieve a concentration of 28 µg /L as P during the peak flow periods. The remaining 90 percent of the P mass into the filtration and chemical treatment system would be treated to yield a blended effluent concentration of equal to or less than 10 µg /L as P (SFWMD 2002).

5.3.4.3 Advantages and Disadvantages

The principal advantages of wetland treatment system with combined microfiltration and chemical treatment are relatively low land requirements, operational flexibility, reliability, and the ability to reduce nutrient levels substantially lower that can be achieved using other technologies. The principal disadvantages are relatively high capital, operations, and maintenance costs, as well as solids management and disposal requirements .

5.3.4.4 Associated Costs

Specific cost information for filtration systems was not obtained for this report given the large assortment of filter system types, sizes and features. SFWMD (2002) estimated costs for the 175-mgd wetland treatment system with combined microfiltration and chemical treatment described above. Their estimates of the 50-year present worth for the full-scale system ranged from $259 million to $308 million. Costs per million gallons treated were estimated to be about $115 to $137 (SFWMD 2002). Costs per pound of TP removed were estimated to be $462 to $550 (SFWMD 2002). These costs per gallon treated and per pound P removed were roughly similar to estimates for chemical treatment and solids separation (as previously discussed in the section 5.3.1 *Chemical Treatment Combined with Settling and Solids Separation*).

5.3.4.5 Applicability in the Upper Klamath Basin

A full-scale wetland treatment system with combined microfiltration and chemical treatment could be considered for implementation in the upper Klamath Basin. Such a system could take advantage of a combination of engineered technologies to achieve reduction of nutrients to relatively low levels, such as approaching levels called for in basin TMDLs. However, such a system would be costly. Extrapolating the present worth calculations above to the mean Klamath River flow (at the location represented by Link River dam) of 1164 cfs, the 50-year present worth of this kind of system would be on the order of $1.3 billion.

CHAPTER 6

Relevant Treatment Wetland Case Studies

Actual treatment wetlands used elsewhere to address similar requirements and objectives serve as important potential analogies to the Klamath Basin situation. As analogies, the implementation experiences and level-of-effectiveness of these other projects provide valuable insights for use in evaluation and planning for potential treatment wetlands in the Klamath Basin. Detailed summaries of each example are described on the following pages, including the implementation experiences and level-of-effectiveness of these other projects, as well as the situational similarities relative to potential application to the Upper Klamath basin.

6.1 Arcata Marsh and Wildlife Sanctuary, California

Background	The Arcata Marsh and Wildlife Sanctuary was one of the first man-made marsh systems built to combine the functions of wastewater reuse of domestic sewage effluent, a wildlife sanctuary, and an urban recreational area. The Arcata Marsh and Wildlife Sanctuary includes constructed treatment wetlands and marshes for treatment and recycling of domestic effluent from the City of Arcata. Construction of this wetland system began about 30 years ago with pilot-scale wetland studies that then expanded to a full-scale system of constructed treatment wetlands that was completed in about 1986. The Arcata Marsh and Wildlife Sanctuary has become an international model of appropriate and successful wastewater reuse and wetland enhancement technologies. Over 150,000 people a year use the facility for passive recreation, birdwatching, or scientific study. The Arcata Marsh and Wildlife Sanctuary includes three freshwater wetlands (Gearheart, Allen, and Hauser Marshes) that were constructed to receive treated wastewater, thereby treating the wastewater further and enhancing the receiving water at the same time. These three freshwater wetlands total 31 acres, have an average depth of 1.5 ft, an average retention time of 9 days, and are operated in series. Three additional wetland treatment cells are used to process oxidation pond effluent to standards prior to release. These total 7.5 acres in size, have an average depth of 2 ft, an average retention time of 2 days, and operating in parallel. The facilities also include ten 200-ft by 20-ft pilot wetland cells that have been used since construction began to demonstrate the effectiveness of constructed wetlands to achieve water quality and habitat goals.
Size	The Arcata Marsh and Wildlife Sanctuary covers approximately 150 acres.
Effectiveness	The City of Arcata currently operates 35.5 acres of surface flow wetlands to provide TSS, BOD, and N removal following treatment of municipal wastewater using oxidation ponds. Of the total system, 7.5 acres are termed “treatment” wetlands followed by the remaining 28 acres of so-called “enhancement” wetlands. The 7.5-acre subsystem receives an average flow of 2.9 mgd, and the BOD and TSS inflows from upstream ponds average 52 and 59 milligrams per liter (mg/L), respectively (EPA 2000a). The products of the flow and concentrations are wetland loading rates of 76 kg/ac/d BOD and 86 kg/ac/d TSS (EPA 2000a). Available performance data for the Arcata Marsh (EPA 2000a) indicate that from 1988 to 2000, wetland influent TSS averaged 59 mg/L and effluent averaged 28 mg/L, representing a 53 percent reduction. During the same time period, wetland influent BOD averaged 52 mg/L and effluent averaged 36 mg/L, representing a 31 percent reduction. The wetlands and marshes also support a diverse community of epiphytes, invertebrates, mammals, aquatic macrophytes, and bird life.
Cost	The construction cost to develop the wastewater-marsh system was $5,300,000 (in 1985 dollars) and $500,000 annually to maintain.
Applicability	This case is the oldest and nearest wetland treatment project to the Upper Klamath basin. It serves as demonstration of a successful constructed wetlands project that addresses water quality goals, as well as other valued objectives for habitat creation, recreation, and scientific study.

6.2 Albany-Millersburg Integrated Treatment Wetlands System, Oregon

Background	The Albany-Millersburg Talking Water Gardens are the first public/private engineering project of its kind in the United States: an integrated wetlands system designed to provide an additional level of natural treatment for a combined municipal and industrial treated wastewater flow. The cities of Albany and Millersburg have joined with metals manufacturer ATI Wah Chang to create this unique combined municipal-industrial water reclamation system inspired by the surrounding environment: an engineered wetland that mimics the cleansing and cooling characteristics that occur in nature. It is the final step in returning treated water safely to the Willamette River. Construction of the Albany-Millersburg engineered wetlands facility was initiated in 2010 and completed during 2011. The Albany-Millersburg engineered wetlands facility is designed to: (1) reduce water temperature, or excess thermal load (ETL), by up to 150 million kilocalories per day to meet state guidelines and protect sensitive fish habitat; (2) naturally aerate and treat water to improve water quality by reducing pollutant levels, including the removal of 2,000 pounds per day of N and 40 pounds per day of P; (3) restore riparian forest and wetlands through plantings of native species; (4) promote wildlife habitat in a former industrial area by reclaiming treated wastewater for use in healthy wetlands environments; (5) create a natural attraction and educational facility for the public.
Size	The Talking Water Gardens includes nine emergent treatment wetland cells covering 39 acres on a 50-acre site.
Effectiveness	The wetland system is designed for the primary function of cooling. The 9.6 million gallons per day (mgd) of Albany/Millersburg effluent and the 3 mgd of Wah Chang effluent are both fully treated to meet river discharge standards and could be blended and discharged directly to the outfall if they were cooler. Analysis of system performance indicates that water temperature of wetlands effluent (at a flow of about 1 mgd) can be cooled from about 23°C to 18°C during the hottest month of July (City of Albany 2010). For TSS, nutrients, and BOD, the wetlands system provides additional treatment functions beyond what is required for river discharge. Preliminary analysis of system performance indicates that mass loading of TSS, NH_3, and BOD in the influent to the system is reduced in the effluent by about 33, 50, and 36 percent, respectively.
Cost	The total public-private investment in the project is $19 million.
Applicability	This case is a unique wetland treatment project located in the region near to the Upper Klamath basin. It serves as demonstration of a constructed wetlands project with similar climatological conditions, and that addresses similar water quality treatment goals, including nutrients, organic matter, and water temperature.

6.3 Prado Wetlands, Santa Ana River, California

Background	The Prado Wetlands are located on lands behind Prado Dam near Riverside, California. The Prado Wetlands system consists of 50 shallow wetland ponds that have been utilized to remove N in Santa Ana River water since 1992 (OCWD 2008). The Santa Ana River is the main source of recharge for the vast Orange County groundwater basin, and consists primarily of tertiary treated wastewater from upstream dischargers. The river also receives stormwater flows and natural runoff, especially during the winter months. A U.S. Army Corps of Engineers permit allows up to 50 percent of the base flow in the Santa Ana River to be diverted through the Prado Wetlands system, which include flows up to about 200 cfs. Because of the importance of the wetlands system to improving water quality in the river below Prado Dam, reconstruction of the wetlands system was completed in 1997 to accommodate increased river flows and to improve the operation flexibility of the system. The modifications to the system included dividing the system into four subsystems, dividing some of the larger ponds into smaller ponds using levees, and adding conveyance channels to allow manifold of water to combinations of ponds.
Size	The Prado Wetlands include nearly 465 acres of constructed wetlands.
Effectiveness	The Prado Wetlands system removes approximately 20 tons of NO_3 per month, and during summer months reduces NO3 concentration from 10 mg/L to less than 1 mg/L (OCWD 2008). The primary mechanism for the NO_3 removal is denitrification. Research indicates that the key to denitrification in the Prado Wetlands are the plant regimes, which determine the quality and quantity of organic matter that provides both a short-term and a long-term organic source for denitrifiers (Ibekwe et al. 2006). Research on the Prado Wetlands has further assessed the effects of HRT and wetland age on NO_3 removal rates (Ibekwe et al. 2006, OCWD 2008). Results determined that longer HRT did lower outflow NO_3 concentrations, but varying HRT did not affect removal rates. Nitrate removal rates were seasonally dependent. The ponds are most effective during the warm summer months, due to warm temperature effects on denitrifiers. Nitrate removal rates also increase with wetland age. More mature wetlands provide increased litter which in turn provide additional organic carbon, greater anoxic zones, and improved habitat for the denitrifiers and for the entire microbial community.
Cost	Total costs for the Prado Wetlands system are on the order of $5 million.
Applicability	This case provides the example nearest to the Upper Klamath basin of a treatment wetlands associated with river diversion, and that addresses similar water quality treatment goals for nutrients and organic matter.

6.4 New River Wetlands Project, Salton Sea, California

Background	The New River originates in Mexico, flows into Calexico, California and through Imperial County before emptying into the Salton Sea. The New River acquires nutrients and heavy metals from sewage; nutrients, silt, selenium and pesticides from agricultural drainage. Excessive nutrients, including phosphates and nitrates from runoff of agricultural fertilizers and municipal wastewater carried by the New River contribute to eutrophication of the Salton Sea, which has no outlets. The flow in the New River at the border is about 150- 200 cfs, increasing to 600 cfs where it enters the Salton Sea. To assess the potential of improving water quality in the New River using treatment wetlands, two pilot wetlands – the Imperial and Brawley Wetlands – were constructed in 2000, and their performance monitored for several years. The goal of the wetlands was to decrease high loads of suspended sediments, nutrients (N and P), selenium, and pathogens. While research has documented the effectiveness of wetlands in improving water quality in other parts of North America, the Salton Sea region has a unique climate, with extreme heat (an annual average daily maximum temperature of 88 degrees) and little precipitation (less than three inches per year).
Size	The Imperial wetland has a total area of 43 acres, of which 22 acres are wetted. The Brawley wetland has a total area of 9 acres, of which 6 acres are wetted.
Effectiveness	Several years of water quality monitoring data (2001-2007) are available from the Brawley and Imperial wetlands to quantify the removal of the monitored pollutants. Both wetlands are very effective at removing suspended sediments and pathogens, with over 90 percent removal. The bulk of the removal occurs in the sedimentation basins of both systems, and the high infiltration rates at both wetlands enhance removal capability. For example, infiltration rates often exceed 50 percent of influent volume in the Brawley wetland. Both wetlands also are effective at removing nutrients. For TN, the monitoring data indicated an average 49 percent removal occurred in the Imperial wetland, and an average 72 percent removal occurred in the Brawley wetland. For TP, an average 38 percent removal occurred in the Imperial wetland, and an average 49 percent removal occurred in the Brawley wetland. The areal loading (total mass of constituent entering the wetland divided by the wetland area) to the Imperial wetland was around twice the loading to Brawley. Therefore, although more TN and TP were retained in Brawley as a fraction of inflow, the absolute magnitude of TN and TP retention was higher in Imperial. These observations for TN and TP reflect the longer hydraulic residence time for Brawley as opposed to Imperial. For both TN and TP, losses to infiltration were approximately one quarter of the input loading.
Cost	The capital costs for the Imperial and Brawley Wetlands, constructed in 2000, were approximately $2.04 million. The O&M costs for the two pilot wetlands over five years (2001 through 2005) were about $700,000 (Tetra Tech and WMS 2007).
Applicability	This case provides an example in California (albeit of different climatological conditions) of constructed wetlands treating runoff that includes eutrophic nutrient conditions similar to the Upper Klamath basin. Also, this case also provides an pertinent example of pilot investigations leading to master planning of a network of treatment wetlands being considered for construction along the New and Alamo Rivers (Tetra Tech and WMS 2007).

6.5 Tahoe City Wetland Treatment System, California

Background	The Tahoe City Wetland Treatment System (TCWTS) was constructed in 1997 to treat stormwater runoff from 57 acres of commercial, highway, and residential land use in the Lake Tahoe Basin. This subalpine, constructed, surface flow wetland treatment system consists of two cells in series, with a design water surface area of about 1.5 acres. Construction started in 1997, and the wetland started to operate in 1998. Water is conveyed to the site from the drainage area as subsurface (groundwater) flow into an upper detention basin. From the upper basin, water then flows through a control pipe to a lower wetland cell where most of the natural treatment occurs.
Size	About 1.5 acres.
Effectiveness	Heyvaert et al. (2006) reported on the nutrient and suspended sediment removal capability of this system. Monitoring data indicate an improvement of 49 percent or greater in effluent concentrations of dissolved P, NO_3, SRP, and total suspended solids. On average, event mean concentrations of TP were reduced from a median 279 µg/L at the inflow to 94 µg/L at the outflow. Event mean concentrations of TN were reduced from a median 1,599 µg/L at the inflow to 810 µg/L at the outflow. Almost four metric tons of suspended sediment were captured in this wetland during a period of one year. Overall effluent quality from the TCWTS was relatively consistent with, or better than, the results from other monitored stormwater treatment practices (Strecker et al. 2005). In this relatively cold, subalpine environment, little treatment effectiveness is expected during winter because of low temperature and dormant wetland plants. However, monitoring indicates that this system is effective throughout the year, although it is less effective in winter and spring than in summer. It is assumed that some effectiveness is maintained in winter because enough water is available to the system to keep it from freezing under the ice, and a build-up of snow on top of the ice also acts to insulate the system. Some of the submerged plants remain active, and sufficient oxygen is maintained to provide microbial activity in the wetland sediment.
Cost	Capital costs for the treatment wetland are estimated at about $6 million.
Applicability	This case serves as demonstration of a constructed wetlands project in the region near to the Upper Klamath basin. This case has similar nutrient treatment goals, and operates under cold wintertime climatological conditions that can be similar to the Upper Klamath basin. These wintertime conditions are an important consideration and potential constraint related to constructed treatment wetlands and their effectiveness in the Upper Klamath basin.

6.6 Richland-Chambers Treatment Wetlands, Trinity River, Texas

Background	The Tarrant Regional Water District (TRWD) operates a treatment wetland system that consists of a series of sedimentation ponds and wetland cells that naturally treat water diverted from the Trinity River and discharged back to Richland-Chambers Reservoir in Texas. The concept is to convey discharge from the wetlands to the reservoir for storage, and ultimately to use the stored water as a supplemental source for Dallas-Ft. Worth. Development of the system began with an original pilot-scale treatment wetland constructed in 1992. The pilot-scale system was designed for a flow of 0.1 million gallons per day, had two settling basins followed by three parallel wetland trains, each with three free-water surface (FWS) wetland cells in series. The two settling ponds were approximately 0.07 acre and 0.2 acre in size and provided a minimum 24-hour detention time each for the design flow. Each wetland cell was about 0.25 acre with a total wetland area of about 2.25 acres in the three trains. The detention time within the wetland trains was approximately 7 to 10.5 days. TRWD spent eight years (1992-2000) studying the pilot-scale system. Based on the effectiveness of the pilot-scale constructed wetland treatment, TRWD proceeded with construction of a 243-acre field-scale constructed wetland project in 2002, with a treatment capacity of 12-15 MGD. In 2008, an additional 200 acres of wetland cells were added. TRWD plans to construct additional wetland cells in three more phases in the future to achieve a total treatment wetland surface area of about 1,500 acres.
Size	The Richland-Chambers treatment wetlands are 443 acres in size, with plans to reach 1,500 acres.
Effectiveness	Studies have been conducted over several years on both the pilot-scale and field-scale wetland cells (Kadlec et al. 2011). Results from the pilot-scale studies showed that target levels for nutrient and sediment concentrations for the “finished” water from the treatment wetland were as good as, or better than, the levels coming into the Richland-Chambers Reservoir from its natural tributaries. Nutrient removal from the river water was above 65 percent for P and more than 80 percent for N, while sediment removal was about 95 percent. For the field-scale system, removals were 77, 45, and 96 percent for NO3-N, TP, and TSS, respectively (at inflow concentrations of 3.17, 0.97, and 212 mg/L, respectively). Results from both pilot-scale and full-scale studies showed that the bulk of TSS removal and a substantial fraction of TP removal were in the initial sedimentation basins. Several other conclusions resulted from the studies. A diverse wetland plan community provides a more robust treatment system. Selective planting is targeted in critical zones designed to achieve good distribution of flow to optimize treatment across wetland cells. The ability to vary the water depth within the wetland cells also is important to achieve desired treatment results. It is beneficial to have both shallow and deep zones within the wetland cells.
Cost	Capital costs for the 443 acres of treatment wetlands are estimated at about $16 million, with full build-out of the 1,500 acres estimated at about $27 million (in 2003 dollars).
Applicability	This case provides another example of treatment wetlands associated with a large river diversion, with water quality treatment goals for nutrients that are relevant to the Upper Klamath basin.

6.7 Caernarvon Freshwater Diversion Project, Louisiana

Background	Large diversions of the Mississippi River have been used to re-establish flows of sediments and nutrients to the marshes in the Mississippi Delta that have been impeded by channelization and other anthropogenic disturbances for decades. Such diversions include the Caernarvon Freshwater Diversion Project into Breton Sound, downstream from New Orleans. The Caernarvon Project diversion control structure began diverting river water into coastal bays and marshes in Breton Sound in 1991. The project aims to enhance emergent marsh vegetation, reduce marsh loss, and increase significant commercial and recreational fisheries and wildlife productivity in the Breton Sound estuary. Prior to the diversion, the area was losing about 1,000 acres of marshland annually. The introduction of fresh water and alluviums from the Mississippi River via the Caernarvon Project diversion control structure serves to reduce this degenerating trend in the local area. The Caernarvon diversion is an important source of nutrients to the bays and marshes in Breton Sound. The Caernarvon diversion introduces 1 to 3 billion cubic meters of Mississippi River water into the receiving marshland and Breton Sound estuary annually. This riverine input contains 2 to 5 million kg of NO_3 and 3 to 7 million kg of TN. Considerable research is underway to determine the main sources and sinks of nutrients from the Caernarvon diversion to the marshes of Breton Sound and to examine nutrient removal potential.
Size	A total of 16,000 acres of marshland are preserved and 77,000 acres of marshes and bays are enhanced by the project.
Effectiveness	Day et al. (2009) examined large scale effects of pulsed flows in 2001-02 from Caernarvon diversions into Breton Sound. Pulses ranged from 183 m3/sec down to 15 m3/sec. Influent NO_3-N was 2.5-17.7 mg/L, and by mid-point of the estuary was reduced to 0.06-4.7 mg/L. N removal was attributed to a combination of dilution with Gulf water, rainfall, uptake by biota, denitrification, and burial. Lane et al. (2004) examined a 16-day pulse in spring 2001 that peaked at 226 m^3/s. Influent concentrations were 1.9 mg/L TN, 1.46 to 2.14 mg/L dissolved inorganic N (DIN), and 0.15 mg/L TP. Removal efficiencies (inflow concentration vs. outflow concentration) were 44, 57, and 62 percent, respectively. Although there is potential for regeneration of nutrients during the summer with warm water temperatures, denitrification losses are a permanent sink for NO_3. Pulsed flows favor both increased denitrification and nitrification. Dissolved inorganic P (DIP), in contrast, tends to be released under low loading and taken up under high loading in estuarine systems (Patrick and Khalid 1974).
Cost	Construction of the project was completed in 1991 at a cost of $26.1 million.
Applicability	This case provides one of the largest-scale examples in the U.S. of treatment of nutrients by wetlands associated with river diversion.

6.8 Des Plaines River Wetlands Demonstration Project, Illinois

Background	The Des Plaines River Wetlands Demonstration Project in Illinois is a complex of seven wetlands constructed in 1988-1991 (Hey et al. 1994a, 1994b). Operational pumping of inflow from the Des Plaines River to four of the wetlands began in 1989 and the remaining three beginning in 1993. The project has been the subject of numerous investigations (see WRI 1992, Hey et al. 1994a, 1994b), including in experimental wetland cells that are 5 to 8.5 acres in size. Hydraulic loading rates ranged from 1.3 to 5.7 cm/day. One long-term, full-scale example of restored wetlands treating agricultural runoff has yielded findings of relevance to wetland planners worldwide.
Size	The Des Plaines River Wetlands Demonstration Project totals about 550 acres. Experimental wetland cells are 5 to 8.5 acres in size.
Effectiveness	Monitoring reported by WRI (1992) indicated that average removal efficiencies in Des Plaines River diversion wetland cells were 92, 84, and 85 percent for TSS, NO3-N, and TP, respectively. Subsequent analysis by Hey et al. (1994) suggested TSS removal ranged from 76-99 percent, NO3-N removal ranged from 39-99 percent, and TP removal ranged from 52-99 percent for the 1990 and 1991 growing seasons. Modeling by Wang and Mitsch (2000) suggested TP was retained in the sediments at a rate of 1.08 to 2.47 g/m2/yr. Alvord and Kadlec (1996) reported that, during continuous flow operation, significant removals were found for NO_3-N, TP, TSS and other parameters, including atrazine. Kadlec (2010) subsequently reported that episodic loading to the wetlands showed average NO_3-N concentrations were reduced from 2.3 mg/L to 0.9 mg/L over 28 filling events, for an average mass removal of 67 percent. First-order removal rate constants calibrated using a dynamic mass balance model were found to be greater for the events than for previous steady state performance for the wetlands (k20 = 107 vs. 37 m/yr). Nitrogen removal was by displacement of antecedent treated water and treatment during and after the event.
Cost	[*Cost information was not obtained*]
Applicability	This case provides an example of constructed wetlands treating agricultural runoff within a moderate-sized river basin – conditions similar to those of the Upper Klamath basin.

6.9 Everglades Construction Project, Florida

Background	The Everglades Construction Project comprises six large-scale stormwater treatment wetlands constructed to treat agricultural runoff that had caused significant loading of P and disruption of flow volume and timing to the Everglades. These wetlands are managed using an adaptive approach incorporating information on water levels, flows and quality to remain within an established design operational "envelope" unique to each STA to prevent dryout and maintain a minimum of 15 cm water depth, avoid keeping the water stage too deep for too long by limiting depth to a maximum of 137 cm for 10 days, maintain target depths between storm events and have frequent field observations by site managers (SFWMD 2010).
Size	The Everglades Construction Project treatment wetlands total over 65,000 acres (SFWMD 2010).
Effectiveness	Since 1994, these wetlands have received over 12 million megaliters (ML) of inflow and retained 1,403 Mt of P, reducing P loads by 74 percent and levels from an overall annual flow-weighted mean (FWM) TP concentration of 145 to 40 µg/L (SFWMD 2010). During 2009-2010, these wetlands received a total of 1.7 million ML of water, equating to an average hydraulic loading rate of 2.83 cm/d. The inflow P load was 253 Mt, equating to an average TP loading rate of 1.52 g/m^2/yr. An estimated 76 percent load reduction was achieved, with the wetlands retaining 192 Mt of TP and reducing inflow TP concentrations from 147 to 33 µg/L. Outflow TP concentrations ranged from 15 to 94 µg/L (SFWMD 2010). Research in these wetlands on reducing outflow P to 10 ppb led to the understanding that submersed aquatic vegetation (SAV) and native calcareous periphyton exhibit greater removal rates for P compared to emergent aquatic vegetative systems, leading to a greater areal efficiency for treatment (DeBusk et al. 2001, Dierberg et al. 2002, Pietro et al. 2010). As a result of this research, a 100-acre periphyton and SAV wetland cell has been in operation for over five years, yielding geometric, flow-weighted mean P concentrations of around 8 to 10 µg/L (SFWMD 2010).
Cost	Approximately $1.2 billion (SFWMD 2010)
Applicability	This case provides an example of constructed wetlands treating runoff that includes eutrophic nutrient conditions similar to those of the Upper Klamath basin. This case has also generated considerable data and information as presented in previous sections of this report.

6.10 Mississippi-Ohio-Missouri Basin Nutrient Control Implementation Initiative (NCII)

Background	The Mississippi–Ohio–Missouri (MOM) Basin covers 3.2 million km2 or about 40 percent of the lower 48 United States. The Ohio, Missouri, and Upper Mississippi Rivers account for 80 percent of the Mississippi River Basin and contribute 90 percent of the NO_3–N flux to the Gulf of Mexico, which is a key cause of hypoxia in the Gulf of Mexico (Mitsch and Day 2006). The Nutrient Control Implementation Initiative (NCII) is underway to evaluate and implement actions aimed at improving scientific understanding of nutrient inputs and nutrient control actions, and their downstream impacts. The NCII proposes to implement a network of nutrient control pilot projects in the basin to assess and demonstrate the relative effectiveness of various nutrient control actions. The pilot projects would be long-term capital investments, rather than short-term trials, and would represent initial steps toward a larger, basinwide network of land management practices aimed at protecting water quality in the Mississippi River and northern Gulf of Mexico. Examples of the types of practices that would be implemented and tested include: constructed wetlands; erosion and sediment control; crop scheduling and nutrient management; vegetative buffers; runoff interception; and other already established best management practices for nutrients and improving nutrient efficiency.
Size	Creation or restoration 2.2 million ha of wetlands will be required to remove 40 percent of the TN discharging to the Gulf of Mexico (Mitsch and Day 2006). This estimate is based on relationships between loading and retention rates for wetlands determined from several years of data from various wetland studies in the MOM Basin.
Effectiveness	Wetlands constructed on farms could intercept and treat runoff (Fink and Mitsch 2004) and river diversion wetlands such as the Des Plaines River wetlands described below, and other wetland diversions constructed in the Mississippi River in Louisiana could reduce river NO_3 loads. For example, Hey et al. (2005a, 2005b) estimated that 320,000 acres of nutrient farm wetlands adjacent to the upper Illinois River would provide treatment equivalent to an advanced wastewater treatment (AWT) plant. Kadlec (2005) estimated hydraulic loadings of 2-7 cm/day would reduce NO_3 loads by 30 percent. Multiple other benefits would result, including flood mitigation, recreation, and restored wildlife habitat and biodiversity (Mitsch et al. 1998).
Cost	Hey et al. (2005a) valued the removal of metric ton of N in a wetland at $2,500 on farmlands adjacent to the upper Illinois River. Hey et al. (2005a, 2005b) estimated that the cost of 320,000 acres of nutrient farm wetlands adjacent to the upper Illinois River would cost $103 million, which would be 50 percent less than the cost of an AWT of equivalent treatment potential. In addition, Hey et al. (2005a, 2005b) indicate that additional revenue could be generated through the sale of nutrient reduction “credits” during the summer or fall when the wetlands have excess capacity to remove nutrients, conceptually totaling 26,100 Mt of N and 2000 Mt of P valued at approximately $56.3 million.
Applicability	This case includes a large basin-wide nutrient control program in which treatment wetlands are planned at a large scale. Such basin-wide nutrient control will likely be necessary to achieve long-term nutrient reduction goals in the Upper Klamath basin.

6.11 Clayton County Constructed Treatment Wetlands, Georgia

Background

Just south of Atlanta, Georgia, the Clayton County Water Authority (CCWA) provides water, sewer, and stormwater services to more than 280,000 county residents and portions of adjacent counties. In the early 1980's, CCWA was one of the first utilities in the world to begin using spray irrigation of highly treated wastewater onto forested areas to complete the purification process. This land application system (LAS) was a way to increase water supplies for its growing population while minimizing the stream impact of wastewater discharges.

CCWA operated the LAS system for almost 30 years as the County matured into a densely developed urbanized area. In response to the need for additional wastewater treatment capacity and as part of CCWA's mater planning process, CCWA began converting the land application system to constructed treatment wetlands as the most reliable and sustainable option for both treatment and water supply augmentation. The constructed treatment wetlands provide more capacity on less land, are more cost effective, and enhance greenspace and wildlife habitat.

In 2002, the Shoal Creek LAS was converted into a series of treatment wetlands (Panhandle Road Constructed Wetlands) and the existing treatment plant was replaced with an advanced, biological treatment plant. Following this success, CCWA began developing a larger wetlands complex on the E.L. Huie Jr. Site. The Huie Constructed Treatment Wetlands site was completed in four phases spread out over the past six years (2005-2010) so that CCWA could continue to maintain its treatment capacity on the site.

Size

The Panhandle Road Constructed Wetlands include 53 acres of wetland area, and the Huie Constructed Treatment Wetlands include 264 acres of wetland area.

Effectiveness

The Panhandle Road Constructed Wetlands consists of three multi-cell treatment "trains" with a combined treatment capacity of 4.4 million gallons per day (mgd). The E.L. Huie Constructed Wetlands consist of nine multi-cell treatment "trains" that were built in four phases and have a total treatment capacity of 17.4 mgd. Even though a portion of the water in the wetlands is expected to infiltrate into the groundwater supply, the vast majority flows into two of CCWA's water supply reservoirs, Shoal Creek and Blalock Reservoirs. Water typically takes two years under normal conditions to filter through wetlands and reservoirs before being reused and takes less than a year under drought conditions.

Both wetland systems polish highly treated effluent from primary and secondary wastewater treatment facilities that include nutrient removal followed by disinfection. In addition, the constructed wetlands buffer the reservoirs in the unlikely event of a treatment plant upset. A National Pollutant Discharge Elimination System (NPDES) permit was received for the constructed wetlands. Both constructed wetlands systems (Panhandle and Huie) are required to comply with the discharge limits established in their NPDES permit waste load allocations.

Cost

This $30 million project was funded through municipal bonds, a GEFA loan and funds designated from CCWA.

Applicability

This case provides an example of treatment wetlands associated with a river-reservoir system, and that includes water quality treatment goals for nutrients similar to the Upper Klamath basin.

6.12 Philip Morris Engineered Wetlands, James River, Virginia

Background	Philip Morris USA (PM USA) has created 48 acres of engineered wetlands on their Park 500 property in Chester, Virginia, adjacent to the James River. The engineered wetlands system was designed to further reduce N, P, and suspended solids from the traditional on-site wastewater treatment process at this tobacco processing facility. The 1.53 mgd of treated wastewater previously sent to the James River is now diverted to the constructed wetlands for additional treatment. The engineered wetlands system relies on natural physical, and biological processes such as uptake and chemical synthesis to remove nutrients such as N and P. The reclaimed water from the wetlands then is returned to the James River. The system was modeled after an existing one in Clayton County, Georgia (described above). The engineered wetlands system configuration uses two parallel north-south flow paths. Each flow path contains a series of three wetland cells, for a total of six separate wetlands encompassing 48 acres of wetlands on 70 acres of land. Flow from the existing wastewater treatment plant is pumped to the inlet of the wetland system. From there, water moves through the natural treatment system by gravity. The parallel treatment paths add operational flexibility to the system while the multiple cells in series improve treatment efficiencies. The system includes a series of small, deep water zones interspersed with shallow marsh zones. The marsh zone is covered with grasses and plants that grow in shallow water. It takes an average of 9 to 14 days for the water to traverse the entire wetland system.
Size	The system encompasses 48 acres of wetlands on 70 acres of land.
Effectiveness	The system was designed to further reduce TN discharge by 13 percent and phosphorous discharge by 34 percent. Initial results actually exceed these figures. The first few years of monitoring data indicate that TN concentrations in the wastewater were reduced by 36 percent, from an average inflow concentration of 9.6 ppm to an outflow concentration of 6.2 ppm. The data indicated that total phosphorous concentrations were reduced by 81 percent, from an average inflow concentration of 0.52 ppm to an outflow concentration of 0.1 ppm. However, this preliminary data represents a period of rapid plant growth and pollutant uptake levels that are not likely to be sustained over time. The wetlands have also created new habitat for several wildlife species.
Cost	The entire system cost $7.175 million and was commissioned in June 2008.
Applicability	This case provides another example of treatment wetlands associated with a large river diversion, with water quality treatment goals for nutrients that are relevant to the Upper Klamath basin.

6.13 Brighton Wastewater Treatment Wetland, Ontario, Canada

Background	The Brighton Wastewater Treatment Wetland in Ontario, Canada is currently the largest approved constructed wetland in Ontario (15 acres) treating municipal wastewater treatment plant effluent. The Municipality of Brighton is located along the north shore of Lake Ontario and has a two-cell lagoon system as part of a conventional wastewater treatment plant (WWTP). In 1999, rather than expanding the lagoon system to augment its rated capacity, the Municipality opted to construct the treatment wetland on land that they already owned, at a cost of about $0.5M (considerably less than the cost of a conventional treatment system expansion). The wetland increased the rated capacity of the existing lagoon system from 3,864 m3/d to 4,600 m3/d. The treatment wetland was designed to provide acceptable effluent concentrations for total suspended solids, biochemical oxygen demand, TP, and NH3-N.
Size	15 acres
Effectiveness	The wetland quickly provided measurable improvements in water quality, with complete nitrification of NH3 occurring throughout the summer months of the first year. The wetland has further reduced the loading to Presqu'ile Bay over the past 10 years by up to about 20 T/year of combined 5-day biochemical oxygen demand (BOD5), total suspended solids, TP, and N. Trends in the concentrations of the constituents of concern are seasonal and are also affected by operational changes in the wetland system. From a seasonal perspective, a significant issue for winter treatment is NH3. With the onset of winter, temperatures drop in the wetland, and nitrification slows to near zero in January and February. A secondary issue limiting nitrification and winter removals of NH3 is the formation of ice on the surface of the wetland, which results in reduced oxygen transfer. Fortunately, the impact of NH3 on the receiving water is significantly less in winter. From an operational perspective, trends in monitoring data indicate that treatment efficiency of the wetland changes little even after almost total removal of standing wetland vegetation by muskrats for hut and feeding mound construction, and no reduced efficiency at low wetland water levels during re-vegetation periods.
Cost	Constructed at a cost of about $0.5M.
Applicability	This case provides another example of a constructed wetlands project that operates under cold wintertime climatological conditions. Seasonally cold conditions are an important consideration and potential constraint related to constructed treatment wetlands in the Upper Klamath basin.

CHAPTER 7

Other Considerations and Recommendations

7.1 Defining Goals

Defining the specific goals for treatment by wetlands in the Upper Klamath basin to improve water quality is a critical and all-important first step. The specific goals and objectives will drive most, if not all, aspects of wetland planning, design, construction, operation, and maintenance. Establishing clearly defined and specific goals is essential for any significant technology or engineering project, and such is the case for the planning and implementation of constructed treatment wetlands in the Upper Klamath basin. Defining specific performance goals is important not only for the design and operation of the individual wetland or wetland system, but also for the process of planning and managing a program of wetland treatment actions across the basin (or subbasins) (Figure 27).

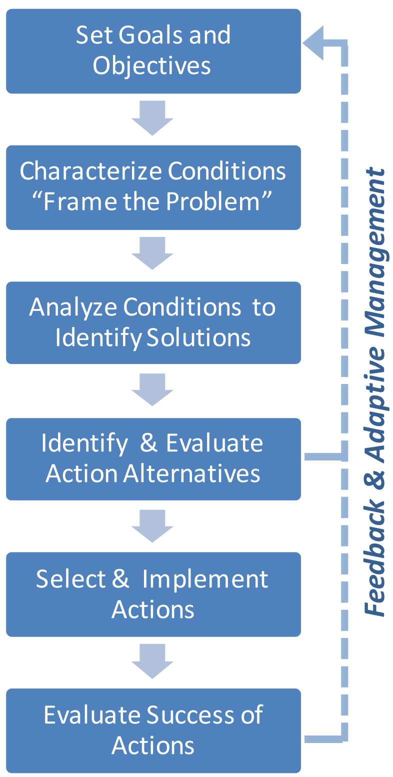

FIGURE 27
Decision-making process for planning and implementation of management actions

Specific goals will be instrumental in determining wetland sizes and area (land) requirements that would be needed for the project. For example, depending on the specified level of treatment defined in the goal, the area (land) requirements could vary substantially or indicate the need for appreciable areas of land. Also, depending on land availability, the goal's specified level of treatment will dictate whether supplemental treatment technologies are needed in conjunction with the wetlands in order to meet goals. Moreover, the wetland features and ancillary components that are needed to meet the goals will ultimately determine costs, including costs for land, site investigation and engineering design, wetland construction and operation, any pre- and/or post-treatment components, and other related facilities or infrastructure needs.

In previous studies on potential use of treatment wetlands in the Upper Klamath basin, specific goals for treatment have been assumed in order to estimate wetland treatment sizes and conceptual designs. For example, in modeling of potential use of treatment wetlands in the Klamath River reach between Link Dam and Keno Dam, Deas and Vaughn (2006) assumed an effluent BOD_5 from the wetland that would not exceed 6 mg/L. To estimate wetlands area needed for water quality improvement of water entering the Klamath River from the Straits Drain, Lytle (2000) used TP as the "parameter of concern" with a target effluent value of 0.16 mg/L, which was chosen because it represented the mean TP concentration in data from the Keno reservoir upstream of the Straits Drain. In other examples of treatment wetlands across the country (such as described in Chapter 6 *Relevant Treatment Wetland Case Studies*), the specific goals for treatment are frequently tied to regulatory requirements, such as dictated by NPDES permits or TMDL allocations.

7.2 Anticipating Important Factors and Constraints

7.2.1 Water Use

Hydrology, or water availability and use, are arguably the most important factors in the planning and design of constructed wetlands in the Upper Klamath basin. Wetland water balance is crucial for determining wetland size and likely treatment effectiveness, and therefore the wetland's land location and size requirements. Wetland water balance is also crucial for indentifying inflows, outflows, and internal gains and losses of water to a wetland, and therefore the wetland's water use and potential water rights requirements.

A logical use of constructed wetlands in the Upper Klamath basin would be to treat surface waters discharging to or diverted from streams or rivers in the basin. As such, an important question driving wetland designs and regulatory approvals will be "are flows gained or lost in the wetlands"? The gains or sources of water to a constructed wetland include precipitation, snowmelt, and direct runoff from the wetland catchment. Water losses from a constructed wetland occur through the outlet discharge, evapotranspiration, and infiltration (including bank storage).

Water rights requirements in particular are likely to be a major constraint to design and implementation of constructed wetlands in the Upper Klamath basin, especially if net water consumption occurs in wetlands (that is, net flow losses exceed flow gains). For example, in modeling of potential use of treatment wetlands in the Klamath River reach between Link Dam and Keno Dam, Deas and Vaughn (2006) assumed that a 20 percent loss of flow would occur within the wetlands (from an average wetland inflow of 323 cfs to an average wetland outflow of 259 cfs). Mahugh et al. (2008) estimated the water balance of a pilot-scale treatment wetland adjacent to the Klamath River near Klamath Falls, which indicated a 60 percent loss in flow (from an inflow to the wetland of 9 cfs to an outflow of 3.6 cfs). In modeling potential treatment wetland performance at sites adjacent to the Klamath River upstream of Copco reservoir, Lyon et al. (2009) estimated a 42 percent loss of flow would occur within a wetlands system designed to treat about 10 percent of the total flow in the Klamath River (from an average wetland inflow of 128 cfs to an average wetland outflow of 75 cfs). Lyon et al. (2009) estimated that about three-quarters of the loss of flow would be due to infiltration, and the other quarter from net evapotranspiration (assuming a hydraulic residence time in the wetland of 2 days). This range in magnitude of water losses can be found in other large-scale treatment wetlands such as the Everglades constructed treatment wetlands (referred to as "Stormwater Treatment Areas"). For example, over a ten-the period of record, seepage losses from individual treatment cells in those peat-soil dominated systems have ranged from about 3 to 42 percent (SFWMD 2011).

An assessment of potential water rights requirements for constructed treatment wetlands in the Upper Klamath basin was beyond the scope of our evaluation for this report. However, experience with constructed treatment wetlands elsewhere suggest that options may exist for gaining need water use/water rights approvals. For example, irrigation water rights are the most common type of use in the Klamath Basin and may be transferrable to a wetland under certain circumstances. The irrigation water rights would have to be proofed or adjudicated to be eligible for transfer. The transfer would be for place of use and type of use, but would not need to change the time of use or the rate and duty. Farm land with water rights could be purchased to acquire access to the water rights for use to construct and irrigate wetlands on the land, or potentially to transfer the water rights on a permanent or temporary time basis to other lands. Farm land with water rights could also be leased on a long term contract and the water rights could be used on the land for wetlands, or transferred on a temporary lease to other lands for use in wetlands in a more favorable location. At the end of the lease, the water rights would revert back to the original place of use and type of use. Transfers of irrigation water rights to other properties or new water rights for wetland development may be possible, but are less likely given the current extent of water rights allocations in the basin and the lengthy review and approval process.

7.2.2 Land Availability and Suitability

The preliminary estimates of wetland area needed for treatment in the Upper Klamath basin (see Chapter 4) indicate that relatively large areas of land would be needed to site treatment wetlands, particularly if treatment wetlands are to be used as a key means of appreciable nutrient reduction in the basin. Because of the potential of such large land requirements, land availability and suitability would be major constraints facing the implementation and use of treatment wetlands in the Upper Klamath basin. Most of the land would have to be acquired from current land owners, and therefore land availability would depend heavily on landowner cooperation. Even if land is available, land acquisition costs can be a major cost that may render a particular site prohibitive for implementing a constructed wetland. Public agencies with lands in the Upper Klamath Basin may be interested in developing treatment wetlands. Such opportunities on public lands could be an expedient means of acquiring potential wetland sites and is worthy of further investigation.

Aside from availability of land/real estate, the siting of constructed wetlands can be highly constrained by land suitability, including site topography and soils, environmental resources, surrounding land use activities, and land use requirements (more on this latter item is discussed below under *Permitting and Approvals*). In general, level land with low-permeability soils affords the optimal physical setting for a constructed wetland. Potential wetland sites with other physical conditions can be used, but the construction and cost requirements of a constructed wetland will also increase proportionally as the wetland site further deviates from optimal site conditions. Level land for treatment wetlands better accommodates proper water retention time. Sloped sites require more extensive grading and potential additional facilities to move water onto or off of the site. Low-permeability soils allow more effective retention of water within a treatment wetland. Soils in the Upper Klamath basin range from silt and clay loam to sandy loam. The most suitable sites for treatment wetlands would target sites with soils containing high concentrations of silt and/or clay.

Other concerns regarding soils include possible elevated concentrations of organic carbon, organic N, or P, which may result in increasing concentrations, and negative removal efficiencies, in the wetland system. Another concern is the potential presence of elevated mercury in the soils (as residual from previous use during historical mining practices), which could be subject to mercury methylation or leaching in the wetland system. These potential problems can be assessed during design and managed as necessary, such as through avoidance or soil amendments and pre-treatments.

It is logical to assume that constructed wetlands in the Upper Klamath basin would be sited to avoid sensitive environmental resources and that minimize impacts to surrounding land use activities and their social and economic values. For example, siting of constructed wetlands presumably would avoid locating project features in areas with high ecological value, such as existing wetlands or marshland, habitats of particular importance to sensitive species, or land areas containing important cultural or archeological resources. Siting of constructed wetlands also presumably would avoid areas with high intensity or economically valuable land uses, or lands occupied by infrastructure such as large drainage canals, roads, highways, or utility corridors. However, the constraints on siting of constructed wetlands to avoid these effects can be mitigated or balanced by recognizing that constructed wetland technology also provides an opportunity to create or restore wetlands for environmental and socio-economic enhancement, such as wildlife habitat, greenbelts, passive recreation associated with ponds, and other environmental and public use amenities.

7.2.3 Permitting and Approvals

Many federal, state, and local regulatory permits and approvals are potentially “in play” and will require consideration before initiating a constructed wetland project. Construction and implementation of treatment wetlands could require a variety of permits and regulatory approvals. Types of permits and approvals needed are likely to vary with wetland type and location, and may not be evident until a substantial level of detail has been developed during planning and design. Depending on specific aspects of a particular project, the permitting and approval process could be time-consuming and result in implementation of additional measures or requirements, or even conditions that may prevent projects from proceeding. Early determination of specific regulatory requirements, including consultation with the regulatory agencies during the planning process, would be prudent to help in the formulation of a strategy that proactively addresses regulatory permits and approvals.

At the federal level, the key regulatory drivers are likely to include the Clean Water Act (CWA) Section 402 National Pollutant Discharge Elimination System (NPDES) program, the CWA Section 404 Wetland Fill Program, and the Endangered Species Act (ESA). The Section 402 NPDES program is administered by the U.S. Environmental Protection Agency (EPA) or delegated to the appropriate state agency, which in the Upper Klamath basin is the Department of Environmental Quality (DEQ) in Oregon and the North Coast Regional Water Quality Control Board (Regional Board) in California. The NPDES program requires a permit for all discharges into regulated water bodies. Constructed wetlands associated with a point source discharge (e.g., a constructed wetland used as the final effluent polishing from a wastewater treatment plant) are regulated as point source discharges under the

NPDES program. Constructed wetlands functioning as pollution treatment devices for nonpoint source discharges or industrial stormwater, and then discharging as point sources into regulated waters, must be covered by an individual, general, or multisector NPDES permit.

The CWA Section 404 Wetland Fill Program is administered by the U.S. Army Corps of Engineers (USACE) and EPA, and covers fill activities into wetlands. Natural wetlands, created wetlands, and natural wetlands used for treatment are all considered regulated waters. Any fill and most construction activities in these wetlands require a fill permit from USACE unless the fill activity is deemed exempt. USACE also will ask for concurrence from the following agencies before issuing the permit:

- U.S. Fish and Wildlife Service (USFWS) for ESA-listed terrestrial and resident aquatic species
- National Marine Fisheries Service (NMFS) for ESA-listed marine and anadromous fish species
- DEQ (for activities in Oregon) and the Regional Board (for activities in California) for water quality certification (under CWA Section 401)
- State historical preservation offices for cultural resources
- Oregon Department of Fish and Wildlife (ODFW) and California Department of Fish and Game (CDFG) for fish and wildlife resources (for activities in Oregon and California, respectively).

If a wetland were to be constructed in an area affecting ESA-listed species or their habitat, construction could be prevented or restricted under the ESA as directed by USFWS or NMFS. It is recommended that siting of a proposed constructed wetland avoid such situations. If use by ESA-listed species is suspected on or near a proposed site, additional documentation and consultation likely will be needed to identify and assess alternatives or protection measures that would be required to avoid or minimize effects.

Of these measures, a particularly important and potentially costly measure would be installation and operation of fish screens for situations involving flow diversions to treatment wetlands. USFWS and NMFS have specific fish screening requirements, including fish screen design and operations criteria, for all diversions that are located within the habitat of ESA-listed fish species. ODFW and CDFG also require installation of fish screens in many circumstances, or otherwise encourage the installation of fish screens on the intakes of diversions for protection of fish species. ODFW offers a cost sharing program or tax credits for the installation of screening and bypass devices for individuals required to install these devices for a diversion of under 30 cfs. The cost of fish screens that may be necessary for diversions of water to constructed treatment wetland systems, or to other water quality treatment systems, is a potentially significant cost and technical issue that must be considered and evaluated as part of planning efforts related to water quality improvement in the Upper Klamath basin.

Other federal regulations may apply that seek to protect cultural resources, including historical and archeological sites. These include the National Historic Preservation Act (NHPA), Archeological Resources Protection Act (ARPA), American Indian Religious Freedom Act (AIRFA), and Native American Graves Protection and Repatriation Act (NAGPRA). These laws direct federal agencies, or private projects on federal lands, to protect historical, cultural, and archeological treasures. Constructed wetland projects would need to address and comply with procedures and requirements under these laws.

Important state regulations will require consideration for the planning and implementation of constructed wetland projects in the Upper Klamath basin. For example, in Oregon, the Division of State Lands (DSL) is jointly involved with USACE in the wetland-fill approval process to administer Oregon´s Removal-Fill Law (ORS 196.795-990), which regulates fill activities in “waters of the state” in similar manner as described above with regard to the CWA Section 404 Wetland Fill Program. DSL will issue a fill permit in conjunction with the USACE 404 Wetland Fill Permit unless the fill activity is deemed exempt.

As discussed above under Water Use, water rights requirements and approvals will be a major potential constraint to design and implementation of constructed wetlands in the Upper Klamath basin. Water Right Permits are granted in Oregon by the Oregon Water Resources Department (WRD) and in California by the State

Water Resources Control Board (State Board), Division of Water Rights. In granting Water Right Permits, WRD and the State Board must determine under what conditions water may be taken and used.

In Oregon, ODFW typically makes mitigation recommendations on wetland creation or restoration projects that may affect fish and wildlife and their habitat. ODFW makes these recommendations as part of its review of other agencies' permit application processes, such as the Removal-Fill Permit and Water Use Permit discussed above. The Fish and Wildlife Habitat Mitigation Policy (Oregon Administrative Rules, Division 415) provides the basis for ODFW's mitigation-related comments on these permit applications.

In California, CDFG requires a Streambed Alteration Agreement (SAA) for projects that will divert or obstruct the natural flow of water, change the bed, channel or bank of any stream, or propose to use any material from a streambed. The SAA basically specifies the requirements of what will and will not be done in the riparian zone and streamcourse, including the streambed sloping upwards to the top of the bank.

Local land use plans and zoning requirements are other potential constraints on the siting, design, and implementation of constructed wetlands in the Upper Klamath basin. Local government planning departments will need to be consulted regarding land use plans and zoning requirements for potential use and locations of constructed wetlands. A more specific analysis of these plans and requirements is important to guide further planning and to identify possible permits needed for constructed wetlands in the Upper Klamath basin. In California, the local government planning department also is often responsible for project review under the requirements of the California Environmental Quality Act (CEQA). The main purpose of CEQA review is to identify and prevent significant potential environmental impacts from proposed projects.

7.2.4 Complex Local Climatological and Hydrologic Conditions

The Upper Klamath basin has seasonally distinct and varied climatological conditions, which are important factors that could pose challenges to constructed wetlands design and operation. The upper Klamath Basin climate is characterized by hot, dry summers and wet winters with moderate to low temperatures. The air temperature variations will affect the treatment performance of constructed wetlands, although not consistently for all constituents. Treatment performance for nutrients, particularly N, tends to decrease with colder temperatures, but BOD and TSS removal through flocculation, sedimentation, and other physical mechanisms is less affected.

In colder months, ice and snow is a common occurrence in the Upper Klamath basin. Wetland biological processes slow in response to colder temperatures. Wetland plants are dormant and production of new plant biomass stops below freezing temperatures. Ice cover can further affect constructed wetlands by altering wetland hydraulics and restricting solar insolation and atmospheric reaeration. However, the insulating layer provided by ice cover would slow down the rate and degree of cooling in the water column but would not affect physical processes such as settling, filtration, and flocculation. It is noteworthy that many existing treatment wetland systems are located in Canada and the northern United States that continue to provide many water quality improvement functions, even during the winter. These systems have shown that removal of BOD_5, TSS, and TP is not significantly affected by the cold temperatures that exist in treatment wetlands covered by ice and snow. Water flowing under the ice will have temperatures a few degrees above freezing, and although most of the microbial N transformation processes are slowed, some N treatment continues.

The dry summers and wet winters, together with extensive groundwater-surface water interaction in the Upper Klamath basin, results in substantial hydrologic complexity related to wetland conditions. The basin's semi-arid climate means that evapotranspiration exceeds precipitation, particularly during the period April through September. For example, of the average annual precipitation of 27 inches at Klamath Marsh, about 8 inches (30 percent) occurs from April through October (Mayer et al. 2006). During the same period, evapotranspiration from the marsh is estimated to be about 19 to 22 inches (Bidlake 2000). The substantially higher evapotranspiration during this extensive period of the year sets up the expectation of net flow loss from wetlands (as discussed above under *Water Use*) and the concentration of nutrients and other constituents of interest that can occur from the loss of water. For example, Gannett et al. (2010) indicate that flow of the Williamson River at the outlet of Klamath Marsh near Kirk (USGS station 11493500) ceases during most summers due to the large amount of evapotranspiration in the marsh. In addition, Duff et al. (2009) found that high concentrations of total P (22 mg/ L) and total N (30 mg/L) accumulated in a restoration wetland along Upper Klamath Lake during summer when water temperature, air temperature, and evapotranspiration were highest.

However, in other cases, wetlands in the Upper Klamath basin have been observed to actually augment flows as a result of groundwater infiltration and wetland storage. For example, Mayer et al. (2006) conclude that the fact that there is still a considerable outflow in many years from the Klamath Marsh even after evapotranspiration needs are met suggests there is a sizeable groundwater contribution to the marsh in addition to the surface water inflows. Gannett et al. (2010) estimate that groundwater recharge from precipitation is about 20 percent of the total precipitation basinwide, although the exact percentage varies spatially and temporally. Recharge from precipitation in high elevation areas of the basin occurs during spring snowmelt, and recharge also occurs from irrigation during the irrigation season. Porter et al. (2001) estimate that under 'natural' conditions, large wetland areas in the Upper Klamath basin served to increase flows in the dry season (supplementing baseflows) because the wetlands store water during high flow events. Porter et al. (2001) indicate that, while some of the detained water would likely be consumed by evapotranspiration, the large wetland areas provided substantial net addition of flow to the Klamath River flows during the early (and perhaps late) summer.

These conditions highlight that the planning and implementation of constructed wetlands for water quality treatment in the Upper Klamath basin must contend with complex, obscure, and varied local climatological and hydrologic conditions. These conditions will complicate and challenge assumptions regarding the hydrology of locations where constructed wetlands may be sited or the assumptions regarding flow conditions (and resultant treatment effectiveness) that will be required to support the design, construction, and operation of treatment wetlands.

7.2.5 Uncertainty

The planning for treatment by wetlands in the Upper Klamath basin to improve water quality must proceed with the understanding that the achievable levels of effectiveness are inherently uncertain and dependent on multiple factors. As discussed above in Chapter 3 (*Overview of Treatment by Wetlands*), constructed treatment wetlands have shown to be effective in reducing target constituents. In addition, as discussed in Chapter 5 (*Potential Supplemental Technologies to Enhance Treatment by Wetlands*), effectiveness of constructed treatment wetlands can be further enhanced with the addition of supplemental treatment technologies. Nonetheless, the body of evidence on wetlands generally, and constructed treatment wetlands more specifically, indicates that the effectiveness of treatment using wetlands is mixed and varied, which introduces a level of uncertainty that should be taken into account.

In concept, uncertainly regarding the effectiveness of wetlands can be depicted as expected response curves, which show a relatively wide range in treatment 'performance' for natural, non-engineered wetlands that narrows as site suitability and engineering intensifies (including use of supplemental treatment technologies)(Figure 28) . For example, Fisher and Acreman (2004), in a review of evidence for nutrient removal in 'natural wetlands' from around the world, concluded that the majority of wetlands studied (80 percent) reduced nutrient loadings. However, the levels of reductions reported varied from quite small to large percentages. Moreover, some wetlands actually increased nutrient loadings by increasing the loading of soluble N and P species (this is depicted in Figure 28 as the lower curve being in the "negative" part of the range).

As the curves move to the right, increasing levels of applied engineering to treatment wetlands are represented. As these levels increase, it is presumed that treatment performance is incrementally enhanced and can be maintained within a more predictably narrow range. The higher levels of engineering in the design, construction, and operation of the treatment wetlands systems increase the certainty and reliability that targeted levels of nutrient reduction will be and are achieved. For example, a detailed description is provided in section 5.3.3 of the Lake County Water Authority's Nutrient Reduction Facility (NuRF) in Florida, which represents a large-scale treatment system comprised of chemical treatment (using alum injection), solids settling, and solids removal. The NuRF was designed to provide substantial reduction in P in water coming from Lake Apopka and entering into Lake Beauclair over a range of inflow conditions. The NuRF provides relatively consistent reductions of around 60 to 80 percent in TP that are typically within 5 to 10 ppb of the target TMDL target of 32 ppm established for Lake Beauclair (Bottcher et al. 2009; Lake County Water Authority website).

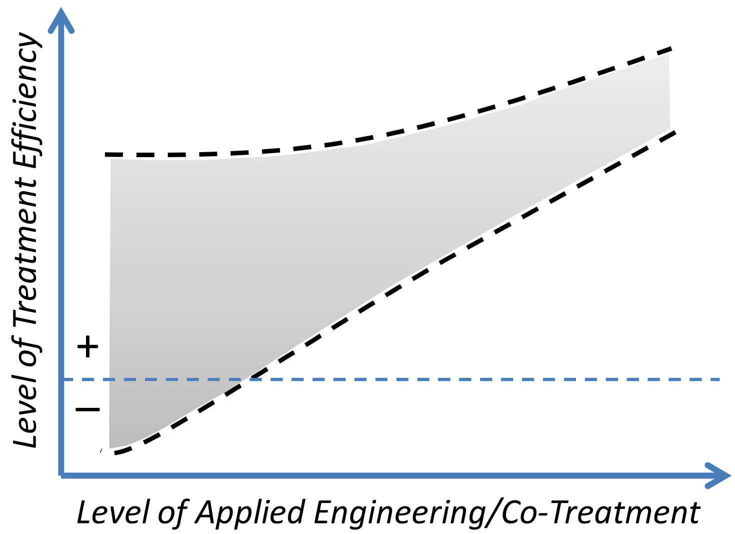

FIGURE 28
Conceptual expectation response curves of wetland treatment efficiency as a function of level of applied engineering and co-treatment technologies.

As evident in Figure 28 and the examples above, the range of uncertainty is narrowed, or conversely the degree of certainty heightened, by the extent to which engineered designs and technologies are included in treatment wetland system construction and operation. As such, the desired degree of certainty should be an important consideration in goal-setting (see the first section of this chapter) and in the resultant design, construction, and operation of treatment wetlands. In the case of the Upper Klamath basin, the degree of certainty should perhaps be an explicitly defined factor, given that TMDLs are targeting large nutrient reductions in the basin, and treatment wetlands are intended as an important tool for contributing to substantial nutrient reductions.

7.3 A Decision Support System

Basin-wide implementation of measures to achieve significant nutrient reduction, including treatment wetlands and other treatment technologies, will be a complex undertaking requiring several steps to plan, identify, and implement actions (such as indicated in Figure 27). In anticipation of this complex process, a decision-making framework should be developed and used to support planning and implementation. Such a decision-making framework would outline the process for making and communicating decisions on the planning, siting, design, construction, and operation of treatment technologies for the purpose of meeting water quality goals for the Upper Klamath basin. For example, a decision-making framework could be developed using steps, milestones, and tools (e.g., spreadsheets; GIS analysis; models) built around the sort of process outlined in Figure 27 (under section 7.1 *Defining Goals* above).

The term Decision Support System (DSS) has been used to describe various permutations of software products and analytical tools that are linked together to help users make decisions for planning and administration of wide variety of complex actions, including river basin and wetlands management (e.g., Aschmann et al. 1999, Hickey and Gibbs 2009, Klang et al. 2010). However, in the following discussion, specific recommendations on a decision support system products and tools are not provided. Rather, the discussion is focused on suggested steps toward developing a decision support system at a more generic planning level. As a starting point, we recommend the following elements be considered:

- "Toolbox" to Screen and Rank Potential Measures
- Technology Research and Refinement

- Subbasin Planning
- Development of Basin-Scale Nutrient Budgets
- Wetland and Supplemental Treatment Pilot Studies
- Expert and Public Involvement

7.3.1 "Toolbox" of Potential Measures

A useful initial planning task would be to develop a "toolbox" that could be used to identify and prioritize an extensive list of potential water quality treatment measures that could potentially be used to meet goals (including, but not limited to, wetland treatment technologies presented in this report). The toolbox product would be a prioritization of measures, including the "best" candidate measures that should be considered in the subsequent planning and implementation of actions. The toolbox would support an evaluation process that compares the cost effectiveness, level of certainty of performance, and potential beneficial and adverse effects of all the listed measures.

As a first step, the toolbox would be populated (perhaps in spreadsheet form) with the extensive list of potential water quality treatment measures. The list could be developed from several sources, including specific wetland treatment and co-treatment technologies discussed in this report, and additional water quality treatment technologies as identified in other reports, the research literature, and commercial product information.

The toolbox would then be used to screen and rank the candidate list of measures. Screening would determine if a measure would be applicable for use. For example, candidate measures could be screened out if: (1) significant uncertainty exists regarding their potential performance; (2) information is lacking to estimate benefits and costs; (3) the measure lacks acceptability by stakeholders; (4) cost of the measure is prohibitive or dependent upon factors that cannot be controlled; and (5) another similar measure is substantially more effective.

Measures that have passed the screening process would then be ranked or prioritized. Ideally, this ranking would be based on objective criteria that would capture effectiveness and certainty of performance relative to goals for water quality treatment. Until goal and objectives are specifically defined (see section 7.1 *Defining Goals* above), definitions for such objectives are premature. However, such criteria should be objective, function-based, and, if possible, quantitative. For example, comprehensive screening of water quality treatment measures for the Lake Okeechobee Watershed Project in Florida used two primary criteria: (1) "expected performance for TP mass load reduction in dollars per pound of TP removed"; and (2) "proof-of-concept in terms of the estimated number of years of operational experience and data from comparable systems" (SFWMD 2004).

Three additional secondary criteria were used in the ranking: (1) water storage volume in dollars per acre-foot of available storage per year; (2) a qualitative assessment of habitat creation benefit; and (3) a qualitative assessment of public acceptance of applying the measure. In this example, the five primary and secondary criteria were ranked on a scale of 1 to 5 points with 5 being the highest ranking, indicating the highest potential as a measure. Scores for primary criteria were multiplied by 3 and added to the actual scores for the three secondary criteria to obtain a total score for each measure (SFWMD 2004). All measures were then ranked and prioritized by total score.

7.3.2 Technology Research and Refinement

On-going technology research and refinement efforts will be needed to support planning and implementation of constructed treatment wetlands (or other water quality treatment measures) in the Upper Klamath basin to improve water quality. This would include on-going efforts to: (1) identify new technologies that can be used either to accelerate achievement of treatment objectives or increase cost-effectiveness of implementation; (2) update existing data sets and monitoring or modeling tools; and (3) develop new analytical tools (e.g., see section 7.3.4 *Development of Basin-Scale Nutrient Budgets* below).

As discussed under section 7.2.5 *Uncertainty* above, uncertainties exist in estimating performance of constructed treatment wetland technologies, and also for other water quality measures and nutrient reduction BMPs. Some uncertainties associated with the performance of these technologies and other BMPs include the impacts of land availability, different soils and hydrologic conditions, interaction with other lands uses and operations, and nutrient conditions and sources currently within the basin. For example, the constructed treatment wetland technologies performance estimates used in this report were based on information available in the research literature, or from monitoring data and reports from applications elsewhere as cited. These performance estimates should continue to be refined over time as ongoing and future research provides additional information.

The planning and implementation of water treatment measures in the Upper Klamath basin will likely use various sources of data and assessment tools to evaluate various measures and programs at the basin or subbasin level to meet water quality goals and objectives. Use of these data and assessment tools will likely require commitment and effort to refine these data and tools. Specifically, this includes effort to update input data sets; refine and integrate hydrologic and water quality modeling tools; evaluate the effectiveness of existing and on-going measures or BMPs at the field level; and evaluate the effect on existing and on-going nutrient load reductions to the Klamath River.

7.3.3 Subbasin Planning

A decision support system should include a process and procedures for landscape-level (e.g., subbasin) planning to determine the applicability and siting of various measures (or suites of measures) in the Upper Klamath basin. In effect, this process would determine where on the landscape measures would be implemented, taking into account desired attributes or important constraints, such as land availability, water availability, soils and hydrologic conditions, potential effects on sensitive species and habitats, presence of culturally significant resources, and other factors deemed important.

Because specific goals and potential measures are not yet specifically defined, it is premature to recommend specific subbasin planning process steps and methods. However, a basic approach should start with the entire Upper Klamath basin and focus on areas that can be determined to be most suitable for implementation of wetland restoration, constructed treatment wetland, or other water quality treatment measures based on selected screening factors. Areas identified as being suitable for these actions can then be prioritized and ranked for further consideration in the planning process.

Mahugh et al. (2008) conducted a feasibility study to assess potential treatment wetlands adjacent to the Klamath River near Klamath Falls. The study included a process for identifying and ranking potential treatment wetland sites that provides an example to consider for subbasin planning. The process ranked all lands within the study area based on several attributes that represent desirable treatment wetland characteristics. The attributes included:

- wetland opportunity (former or degraded wetland sites were preferred over existing wetland sites);
- longitudinal distance from Link River mouth (sites closer to the mouth were preferred to better capture hypereutrophic Upper Klamath Lake water);
- bordering Klamath River and lateral distance to Klamath River bank (sites adjacent to the river were preferred to better capture the hypereutrophic water);
- soil chemistry and composition (non-alkaline soils with low permeability and high silt/clay content were preferred);
- presence of existing water control structures (site with existing flow control were preferred for cost reduction);
- ground slope (0-3 percent areas preferred to allow proper retention time and minimize grading);
- elevation above Klamath River (lower elevation site preferred to avoid or minimize pumping);

- land parcel size (effectiveness of treatment wetland generally assumed to increase with site size);
- regulatory compliance (sites preferred with fewer potential regulatory or permitting issues);
- property owner willingness to participate (sites preferred with known or likely landowner cooperation); and
- existing vegetation (site preferred that already contain wetland vegetation as source to establish new wetland vegetation).

Mahugh et al. (2008) used a quantitative geographic information system (GIS) framework to apply the attributes to land parcels over the study area, and identify and ranks specific locations for potential siting of treatment wetlands. Various GIS layers and topographic maps were used to spatially characterize attributes on a property-by-property basis. Individual attribute categories were assigned relative values on a scale of 0 to 10, where higher values represent desirable conditions and low values represent undesirable conditions for the implementation of treatment wetlands. The information above was embedded in the GIS and a final ranking value for a site was calculated as the sum of the product of individual attribute values and attribute weighting factor for each attribute. Potential full-scale wetland treatment sites were defined as those greater than 100 acres because smaller sites were assumed not likely to provide the desired level of water quality improvement based on a prior feasibility analysis by Deas and Vaughn (2006).

Landscape analysis of potential wetland restoration for the Lake Okeechobee Watershed Project (described earlier) provides another example to consider for subbasin planning. In this case, a two-phase process was used to identify “planning area alternatives” for five planning areas in the watershed. Each planning area alternative was configured as a combination of one or more management measures (i.e., water quality treatment features or activities) to be sited at given locations within a planning area.

The first phase of the process (Phase 1) involved the use of two primary screening factors: land use (only certain land use categories were deemed suitable for wetland restoration and other features and activities); and soils (certain soil conditions were considered essential or favorable). These primary factors were used as “coarse filters” to identify and focus planning on areas with desirable existing land use and soil type conditions. The second phase of the process (Phase 2) involved the use of five secondary screening factors: ecological value (used to avoid existing high quality ecological lands); land disturbance (target lands that are not highly disturbed or have high contaminants); economic value (minimize real estate costs and other economic impacts); cultural resources (avoid areas of cultural resource significance); and environmental and economic equity (EEE; avoid or minimize disproportionate impacts on any particular community). These secondary factors were used as a series of “fine filters” to the results of Phase 1 to further focus on areas that were deemed to be suitable.

The two-phase process was performed using a GIS system to identify, characterize, evaluate, and rank potential locations within the study area. At the completion of Phase 1 and Phase 2 analyses, the GIS system generated a composite summary score for each 30-m^2 pixel within the project study area, and generated polygons around higher valued areas. These polygons were further ranked based on composite score, and used to represent best options for siting restoration and treatment features and activities.

7.3.4 Development of Basin-Scale Nutrient Budgets

Given the importance and emphasis on reducing substantial nutrient loading to Upper Klamath Lake and the Klamath River, another important subbasin planning consideration may be a more thorough understanding and perhaps development of basin-scale nutrient budgets. These basin-scale nutrient budgets are intended to define the quantities of nutrients present in the system, and provide spatially-explicit maps of where key nutrient sources are located. Development of landscape-level nutrient budgets and analyses of nutrient sources and sinks have proven to be valuable and essential tools for planning of nutrient abatement strategies in other basins and watersheds (e.g., Fitz and Sklar 1999, Windhausen et al. 2004, Wang et al. 2004, SFWMD 2007, SFWMD 2008, Chebud et al. 2011). These tools provide a valuable understanding of the fate and transport of nutrients within the basin – where nutrients are coming from and how they move through the system. These tools can also help to identify and evaluate nutrient abatement or reduction strategies, such as the relative effectiveness of upland source controls or downstream treatment systems.

Various example applications and modeling products are available that can serve as building blocks for developing tools that may be applicable to the Upper Klamath basin for basin-scale nutrient budgets and analyses. Applicable examples of development and use of landscape-level nutrient budget and strategy analysis tools include Chesapeake Bay watersheds (Correll et al. 1994, Sprague et al. 2000, Boynton et al. 2008), the Everglades basin and Lake Okeechobee watershed in Florida (Fitz and Sklar 1999, Hiscock et al. 2003, SFWMD 2007, SFWMD 2008, Chebud et al. 2011), the Neuse River basin in North Carolina (Stow et al. 2001, Paerl et al. 2006), several northwestern Ohio watersheds (Baker and Richards 2002), and the Laplatte River and Lake Champlain Basins in Vermont (Windhausen et al. 2004, Wang et al. 2004).

A host of applicable modeling tools are available, such as BASINS (Better Assessment Science Integrating point and Nonpoint Sources), Pollutant Load (PLOAD), the Soil and Water Assessment Tool (SWAT), the Watershed Assessment Model (WAM), the Watershed Ecosystem Nutrient Dynamics (WEND) model , the Sediment Nutrient Assessment Program (SNAP) model, the Long-Term Hydrologic Impact Assessment and Non Point Source Pollutant Model (L-THIA), the Watershed Analysis Risk Management Framework (WARMF), and the Spreadsheet Tool for Estimating Pollutant Load (STEPL). Information on these (and other) models is available from EPA's model website (http://water.epa.gov/scitech/datait/models/) and the Ecosystem-Based Management (EBM) Tools Network (http://ebmtoolsdatabase.org/tools). Selection of appropriate modeling tools should be based on several important considerations, including decisions on objectives and desired output from the models, relevance of specific models to conditions in the Upper Klamath basin, the extent of the model's data requirements, ease of model use, and whether modifications would be needed for application to the Upper Klamath basin.

Landscape-level nutrient budget analysis in the Lake Okeechobee Watershed (LOW) in Florida (Hiscock et al. 2003, SFWMD 2007, SFWMD 2008) offers a pertinent example of how such an analysis might be used in the Upper Klamath basin. In the LOW analysis, P budgets were developed for various subbasins of the watershed within a GIS analysis framework. The analysis included surveys of land use practices (as they relate to P import), which were used to develop net P import coefficients (e.g., kg/ha/yr). These coefficients were spatially applied to land use to quantify the annual mass of P in the watershed. The methodology was used to prepare maps showing the distribution of P throughout the watershed. The analysis also was used to analyze nutrient abatement and reduction scenarios by applying controls to specified land uses to assess the effects on P import, export and net import.

An important aspect of the LOW nutrient budget analysis was to distinguish between P within the watershed that is present as the result of anthropogenic activities (termed "legacy P") and naturally-occurring P (termed "antecedent P"). Further, the LOW analysis assessed the recent temporal trend (history) of legacy P in the watershed based on changes in GIS land use coverages dating back to 1988. The ability to model legacy P required extensive watershed-wide study and testing of soil-nutrient land use-nutrient relationships (see SFWMD 2007 and SFWMD 2008 for additional details). However, the understanding and quantification of legacy P has provided the basis for setting objectives for nutrient abatement strategies that focus realistically on sources of P that have an anthropogenic cause.

7.3.5 Wetland and Supplemental Treatment Pilot Studies

Pilot projects are important to support planning and implementation of constructed treatment wetlands in the Upper Klamath basin. Pilot project and facilities will help to determine the site-specific requirements, effectiveness, feasibility, and costs of constructed treatment wetland technologies in the Upper Klamath basin. This information would be used in the more detailed planning, design, and implementation of treatment technologies needed to improve water quality in the Upper Klamath basin.

Example of pilot studies that should be considered include: (1) constructed wetland treatment pilot cells; (2) chemical treatment bench tests and demonstrations; and (3) hybrid wetland treatment systems. Constructed wetland treatment pilot cells would have several important purpose, including to: (1) ensure that the proposed treatment processes will meet project goals and objectives, and applicable regulatory requirements; (2) test wetland performance under climatic influences and real-world conditions specific to the Upper Klamath basin; (3) test and evaluate specific types of wetland components (e.g., soils and vegetation types) and their relative performances; (4) test and verify important assumptions that will be used to design and construct full-scale wetlands; and (4) assist in estimating overall capital and operation costs.

The chemical treatment pilot study would investigate the effectiveness of chemical treatment technologies as applicable to conditions specific to the Upper Klamath basin. As discussed in Chapter 5 (*Potential Supplemental Technologies to Enhance Treatment by Wetlands)*, there are several optional chemical treatment approaches that could be used to augment wetland treatment. Among these options, treatment costs are likely to vary depending upon influent water quality, volume of water treated and level of treatment desired. The location of chemical addition (pre-treatment vs. post-treatment or “polishing”) also plays a major part in determining total treatment cost. Selection of potential technologies appropriate for use in the Upper Klamath basin should be based on results of the pilot study which will evaluate the comparative effectiveness of these differing applications.

Hybrid wetland treatment systems would combine beneficial attributes of wetland treatment cells and chemical treatment systems. The technology capitalizes on many of the positive attributes of treatment wetlands (effective N and suspended solids removal), and on the effective P removal capability of chemical additive or injection systems. Pilot study of hybrid wetland treatment systems would evaluate the technical efficacy and cost-effectiveness of implementing this combined technology approach within the Upper Klamath basin. Results of pilot studies would help to determine feasibility, siting, design, and implementation of hybrid wetland treatment systems on a larger scale in the basin.

7.3.6 Expert and Public Involvement

Stakeholder outreach and involvement will be a crucial component to implementation of constructed treatment wetlands, or other water quality treatment measures, in the Upper Klamath basin. Without public acceptance and support, and especially the participation by willing landowners, this endeavor would have little chance for success. It is reasonable to anticipate that economic incentives will be necessary for landowners to participate. Without substantial landowner willingness, it will be difficult to place wetlands strategically within the landscape.

Economic and social incentives will likely be needed to gain sufficient landowner participation. Other potential public disincentives would need to be alleviated, such as potentially cumbersome regulatory procedures, a lack of understanding of the programs and actions, frustration and/or mistrust of the government, and the possibility of removing land from production. Education and outreach could help solve these issues over time. It is important that regulators and advocates of the programs and actions allocate time and effort to increase awareness within the basin that the actions are considered essential for the larger objective of basin water-quality improvement. This outreach may necessitate that agency staff become acquainted with local landowners and how they perceive the program.

Involvement of technical experts also will be a crucial component to implementation of constructed treatment wetlands, or other water quality treatment measures, in the Upper Klamath basin. Given the complexity of conditions in the Upper Klamath basin and the targeting of large nutrient reductions, the implementation of eventual solutions will require a “state-of-the-science” level of expertise.

CHAPTER 8

References

Afsar, A. and S. Groves. 2009. Comparison of P-inactivation efficacy and ecotoxicity of Alum and Phoslock™. PWS Report Number: IR 015/09. Phoslock Water Solutions Limited. Frenchs Forest, Australia. July 2009.

Akins, G.J. 1970. The effects of land use and land management on the wetlands of the Upper Klamath Basin. M.S. Thesis. Western Washington State College, Bellingham, Washington. 122 p.

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. Vol. 27, No. 4, December 2007, pp. 1025–1035.

Aldous, A. 2009. Nitrogen and phosphorus loading to and from Sycan Marsh, Oregon. The Nature Conservancy.

Alvord, H.H. and R.H. Kadlec. 1996. Atrazine fate and transport in the Des Plaines Wetlands. Ecological Modelling 90:97-107

Ann, Y., K.R. Reddy, and J.J. Delfino. 2000a. Influence of chemical amendments on phosphorus immobilization in soils from a constructed wetland. Ecological Engineering 14(1-2): 157-167.

Ann, Y., K.R. Reddy, and J.J. Delfino. 2000b. Influence of redox potential on phosphorus solubility in chemically amended wetland organic soils. Ecological Engineering 14(1-2): 169-180.

Aschmann, S., D. Anderson, R. Croft, and E. Cassell. 1999. Using a watershed nutrient dynamics model, WEND, to address watershed-scale nutrient management challenges. Journal of Soil and Water Conservation 54(4): 630-635.

Austin, D. and G. Sun. 2007. Completely autotrophic nitrogen-removal over nitrite in lab-scale constructed wetlands: Evidence from a mass balance study. Chemosphere 68, 1120-1128.

Austin, D., D. Maciolek, S. Wallace, and B. Davis. 2007. Damköhler number design method to avoid clogging of subsurface flow constructed wetlands by heterotrophic biofilms. Water Science and Technology 56 7-14.

Australia Water Quality Center. 2008. Torrens Lake Phoslock™ Trial February 2008. Report of Water Quality Results to 23 April 2008. Prepared by Australian Water Quality Centre for Adelaide City Council. April 2008.

Babatunde, A.O., Y.Q. Zhao, M. O'Neill, and B. O'Sullivan. 2008. Constructed Wetlands for Environmental Pollution Control: A Review of Developments, Research and Practice in Ireland. Environment International. 34 (1):116-126.

Bachand, P., A. Heyvaert, S. Prentice and T. Delaney. 2010. Feasibility study and conceptual design for using coagulants to treat runoff in the Tahoe Basin. ASCE Journal of Environmental Engineering 136: 1218-1231.

Bachand, P., P. Vaithiyanathan., and C.J. Richardson. 2000. Phase II Low Intensity Chemical Dosing (LICD): Development of Management Practices. Final Report. Prepared for Florida Department of Environmental Protection . December 2000.

Bachand, P., P. Vaithiyanathan., R.G. Qualls, and C.J. Richardson. 1999. Low Intensity Chemical Dosing of Stormwater Treatment Areas: An Approach to Enhance Phosphorus Removal Capacity of Stormwater Treatment Areas. Phase I Final Report for FDEP Contract WM 694. Duke University.

Bachand, P., S. Bachand, S. Lopus, A. Heyvaert and I. Werner. 2009. Treatment with Chemical Coagulants at Different Dosing Levels Changes Ecotoxicity of Stormwater from the Tahoe Basin, California, USA. Journal of Environmental Science and Health, Part A: Toxic/Hazardous Substances and Environmental 45(2): 137–154.

Baker, D., and R. Richards. 2002. Phosphorus budgets and riverine phosphorus export in northwestern Ohio watersheds. Journal of Environmental Quality 31(1): 96-108.

Bays, J. and J. Jordahl. 2010. From research to restoration: Global progress in wetlands treatment of agricultural runoff. 12th IWA International Conference on the Use of Wetland Systems for Water Pollution Control. Venice, IT.

Bays, J., G. Dernlan, H. Hadjimiry, K. Vaith, and C. Keller. 2000. Treatment Wetlands for Multiple Functions: Wakodahatchee Wetlands, Palm Beach County, Florida. Proceedings of the Water Environment Federation, WEFTEC 2000: Session 51 through Session 60, pp. 15-37(23).

Beecroft, J.R., M.C. Koether, and G.W. van Loon. 1995. The chemical nature of precipitates formed in solutions of partially neutralized aluminum sulfate. Water Research 29(6): 1461-1464.

Bidlake, W.R. 2000. Evapotranspiration from a Bulrush-Dominated Wetland in the Klamath Basin, Oregon. Journal American Water Resources Association 36 (6): 1309–1320.

Blanco-Canqui, H., Gantzer, C.J., Anderson, S.H., and Thompson, A.L. 2004. Soil Berms as an Alternative to Steel Plate Borders for Runoff Plots. Soil Science Society Am. J., (68), pp. 1689-1694.

Bottcher. D., T. DeBusk, H. Harper, S. Iwinski, G. O'Connor, and M. Wanielista. 2009. Technical Assistance for the Northern Everglades Chemical Treatment Pilot Project. Technical Report. South Florida Water Management District, Project PS 100093. July 2009.

Boynton, W., J. Hagy, J. Cornwell, W. Kemp, S. Greene, M. Owens, J. Baker, and R. Larsen. 2008. Nutrient Budgets and Management Actions in the Patuxent River Estuary, Maryland. Estuaries and Coasts 31(4):623-651.

Bowman, R. 2003. Review: Applications of surfactant-modified zeolites to environmental remediation. Microporous and Mesoporous Materials 61: 43–56.

Boyd, M., S. Kirk, M. Wiltsey, and B. Kasper. 2002. Upper Klamath Lake drainage total maximum daily load (TMDL) and water quality management plan (WQMP). Oregon Department of Environmental Quality, Portland.

Browne, P., M. Evans, and D. Ho. 2004. Second stage phosphorus removal from Rotorua lakes/streams. Technical Report. Prepared for Environment Bay of Plenty. Prepared by URS New Zealand Ltd.

Budd, R., A. O'Geen, and K.S. Goh. 2009. Efficacy of constructed wetlands in pesticide removal from tailwaters in the Central Valley, California. Environ. Sci. Technol. 43: 2925-2930.

California Department of Transportation (Caltrans). 2001. Third Party Best Management Practice Retrofit Pilot Study Cost Review. Prepared by Caltrans Environmental Program, Office of Environmental Engineering. Sacramento, CA. May 2001.

California Regional Water Quality Control Board (CRWQCB). 2005. Final Report for Reducing Eutrophic Conditions of the Salton Sea. California Regional Water Quality Control Board, Colorado River Basin Region.

Carlson, J.R. 1993. The evaluation of wetland changes around Upper Klamath Lake, Oregon, using multitemporal remote sensing techniques, in Campbell, S.G., ed., Environmental research in the Klamath Basin, Oregon-1991 annual report: Denver, Colorado, Bureau of Reclamation Report No. R-93-13, p. 197-202.

Carpenter, K.D., Snyder, D.T., Duff, J.H., Triska, F.J., Lee, K.K., Avanzino, R.J., and Sobieszczyk, Steven, 2009, Hydrologic and water-quality conditions during restoration of the Wood River Wetland, upper Klamath River basin, Oregon, 2003–05: U.S. Geological Survey Scientific Investigations Report 2009-5004, 66 p.CH2M HILL. 2000. PSTA Research and Demonstration Project Phase 1 Summary Report. Prepared for the South Florida Water Management District (SFWMD). August 2000.

CH2M HILL. 2002. Final Literature Search and Summary Technical Report. Periphyton and Submerged Aquatic Vegetation Type Stormwater Treatment Area Field Scale Test Facility. Prepared for U.S. Army Corps of Engineers, Jacksonville District. April.

CH2M HILL. 2003. PSTA Research and Demonstration Project Phase 1, 2, and 3 Summary Report. Prepared for the South Florida Water Management District (SFWMD). March 2003.

Chebud, Y., M. Ghinwa, and R. Rivero. 2011. Phosphorus run-off assessment in a watershed. J. Environ. Monitoring 13 (1): 66-73.

City of Albany. 2010. Albany-Millersburg Talking Water Gardens. August 2010. 19 pp. Available at: twg.cityofalbany.net/.

Coffelt, G., T. Aziz, D. Campbell, S. Gray, B. Gu, J. Jorge, J.M. Newman, K. Pietro and L. Wenkert. 2001. Chapter 8: Advanced treatment Technologies for Treating Stormwater Discharges into Everglades Protection Area. In: SFWMD. Everglades Consolidated Report. South Florida Water Management District, West Palm Beach, FL. pp. 8-1-8-46.

Comeau, Y., Brisson, J., Réville, J.-P., Forget, C. and Drizo, A. 2001. Phosphorus removal from trout farm effluents by constructed wetlands. Wat. Sci. Tech., 44(11–12), 55–60.

Conestoga-Rovers and Associates. 1998. Microfiltration Supplemental Technology Demonstration Report: Final Report. Prepared for the South Florida Water Management District. May 1998.

Cooke, G.D., E.B. Welch, S.A. Peterson, and P.R. Newroth. 1993. Phosphorus activation and sediment oxidation. In: Restoration and management of lakes and reservoirs, pp. 161-209, Lewis Publications, Boca Raton, Florida.

Coombs, D.S., and 18 others. 1998. Recommended nomenclature for zeolite minerals: report of the subcommittee on zeolites of the International Mineralogical Association, Commission on New Minerals and Mineral Names. Mineralogical Magazine, August 1998, Vol. 62(4), pp. 533-571.Correll D.L., T.E. Jordan, and D.E Weller. 1994. The Chesapeake Bay watersheds: Effects of land use and geology on dissolved nitrogen concentrations. Pages 639-648 in Toward a Sustainable Coastal Watershed: The Chesapeake Experiment. Smithsonian Environmental Research Center, Norfolk, VA.

Coveney, M. F., D.L. Sites, E.F. Lowe, L.E. Battoe, and R. Conrow. 2002. Nutrient Removal from Eutrophic Lake Water by Wetland Filtration. Ecological Engineering. 19(2): 141-159.

Coveney, M., E. Lowe, and L. Battoe. 2001. Performance of a recirculating wetland filter designed to remove particulate phosphorus for restoration of Lake Apopka (Florida, USA). Wat. Sci. Technol. 44 (11-12): 131–136.

Craft, C. B. and C. J. Richardson. 1993. Peat accretion and N, P and organic C accumulation in nutrient-enriched and unenriched Everglades peatlands. Ecol. Applications 3: 446–458.

Crites, R.W., E.J. Middlebrooks, and S.C. Reed. 2006. Natural Wastewater Treatment Systems. CRC Press.

Crumpton, W.G., D. Kovacic, D. Hey, and J. Kostel. 2008. Potential of wetlands to reduce agricultural nutrient export to water resources in the Corn Belt. pp. 29-42 in Gulf Hypoxia and Local Water Quality Concerns Workshop , ASABE Pub #913C0308.

Day, J., R. Lane, M. Moerschbaecher, R. DeLaune, R. Twilley, I. Mendelssohn, and J. Baustian. 2009. The impact of the Caernarvon diversion on above and belowground marsh biomass in the Breton Sound estuary after Hurricane Katrina. Final Report submitted to the Louisiana Department of Natural Resources. Project Number 2512-07-01.

Deas, M.L. 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, Inc. for U.S. Bureau of Reclamation Klamath Area Office. September 30.

DeBusk, T.A., K.A. Grace, F.E., Dierberg, S.D., Jackson, M.J. Chimney, and B. Gu. 2004. An investigation of the limits of phosphorus removal in wetlands: a mesocosm study of a shallow periphyton-dominated treatment system. Ecol. Eng. 23(1):1-14.

Dierberg, F.E., T.A. DeBusk, S.D. Jackson, M.J. Chimney, and K. Pietro. 2002. Submerged aquatic vegetation-based treatment wetlands for removing phosphorus from agricultural runoff: response to hydraulic and nutrient loading. Water Research 36(6): 1409-1422.

Douglas, G.B., J.A. Adeney, and L.R. Zappia. 2000. Sediment Remediation Project: 1998/9 Laboratory Trials Report. CSIRO Land and Water. Report number: 6/00.

Douglas, G.B., M.S. Robb, D.N. Coad, and P.W. Ford. 2004. A review of solid phase absorbants for the removal of phosphorus from natural and waste waters. In: Valsami-Jones E ed. Phosphorus in environmental technology: principles and applications. London, IWA Publishing. Pp. 291–320.

Duff, J.H., K.D. Carpenter, D.T. Snyder, K.K. Lee, R.J. Avanzino, and F.J. Triska. 2009. Phosphorus and nitrogen legacy in a restoration wetland, Upper Klamath Lake, Oregon. Wetlands 29(2):735-746.

Edzwald, J.K. and J.B. Van Benschoten, J.B. 1990. Aluminum Coagulation of Natural Organic Matter. In Chemical Water and Wastewater Treatment, H. Hahn and R. Klute, Springer Verlag, NY, pp. 34I-359.

Emond, H., M. Madison, F. Sinclair, and C. Stultz. 2007. Natural Treatment System Alternatives for Effluent Thermal (Temperature) Management in Western Oregon. Proceedings of the Water Environment Federation, WEFTEC 2007: Session 51 through Session 60. Pp. 3985-3986.

Entry, J.A., and Sojka, R.E. 2003. The Efficiency of Polyacrylamide to Reduce Nutrient Movement From an Irrigated Field. Transactions of the American Society of Agricultural Engineers. 46(1), pp. 75-83.

Faithfull, C.L., D.P. Hamilton, D.F. Burger, and I. Duggan. 2005. Waipato Peat Lakes Sediment Nutrient Removal Scoping Exercise. Technical Report. Prepared for Environment Waipato by the University of Waikaito, School of Science and Technology. Hamilton, New Zealand. 96 pp.

Faulwetter J.L., Gagnon, V., Sundberg, C., Chazarenc, F., Burr, M.D., Brisson, J., Camper, A.K., and O.R. Stein. 2009. Microbial processes influencing performance of treatment wetlands: a review. Ecol. Eng. 35: 987–1004.

Fink, D.F., and W. Mitsch. 2004. Seasonal and storm event nutrient removal by a created wetland in an agricultural watershed. Ecol. Eng. 23, 313–325.Fisher and Acreman 2004 nutrient removal in wetlands

Fitz, H., and F. Sklar. 1999. Ecosystem analysis of phosphorus impacts and altered hydrology in the Everglades: a landscape modeling approach. Pages 585-620 in K. R. Reddy, G. A. O'Connor, and C. L. Schelske, editors. Phosphorus Biogeochemistry in Subtropical Ecosystems. Lewis Publishers, Boca Raton, FL.

Flanigen, E.M., R.W. Broach, and S.T. Wilson. 2010. “Introduction” Chapter 1. Zeolites in Industrial Separation and Catalysis. Edited by S. Kulprathipanja. Wiley Verlag GmbH & Co. KGaA, Weinheim. ISBN: 978-3-527-32505-4.

Flapper, T. 2003. Phoslock trials at Fyshwick STP lagoon. Technical Report. Research and Development Program. ECOWISE.

Follet, R., R. Danahue, and L. Murphy. 1981. Soil and Soil Amendments. Prentice-Hall, Inc., New Jersey.

Frank. P., J. Bays, and K. Ortega. 2010. Oxnard Use of Wetlands for Treatment and Reuse of Membrane Concentrate at Oxnard, California. 12th IWA International Conference on the Use of Wetland Systems for Water Pollution Control. Venice, IT.

Gambrell, R.P. 1994. Trace and toxic metals in wetlands – a review. J. Environ. Qual. 23: 883-891.

Gannett, M.W., K.E. Lite Jr., J.L. La Marche, B.J. Fisher, and D.J. Polette. 2010. Ground-Water Hydrology of the Upper Klamath Basin, Oregon and California. U.S. Geological Survey Water-Resources Investigations Report 2007–5050. Prepared in cooperation with the Oregon Water Resources Department. April 2010.

Gearhart, R.A., F. Klopp, and G. Allen. 1989. Constructed free surface wetlands to treat and receive wastewater: pilot project to full scale. In: D.A. Hammer (Editor), Constructed Wetlands for Wastewater Treatment. Lewis Publishers, Chelsea, MI, pp. 121-138.

Gearheart, R.A., Anderson, J.K., Forbes, M.G., Osburn, M., and Oros, D. 1995. Watershed strategies for improving water quality in Upper Klamath Lake, Oregon. Volume II. Report to the U.S. Bureau of Reclamation. 288 p. plus appendices.

Gebbie, P. 2001. Using Polyaluminium Coagulants in Water Treatment. Proceedings 64th Annual Water Industry Engineers and Operators Conference. September 2001. Pages 40-47.

Geiger, S.N. 2001. Reassociating wetlands with Upper Klamath Lake to improve water quality: Klamath Fish and Wildlife Management Symposium, Arcata, Calif., May 22–25, 2001.

Gibbs, M., and D. Ozkundakci. 2011. Effects of a modified zeolite on P and N processes and fluxes across the lake sediment–water interface using core incubations. Hydrobiologia 661:21–35.

Gibbs, M., C. Hickey, and D. Ozkundakci. 2011. Sustainability assessment and comparison of efficacy of four P-inactivation agents for managing internal phosphorus loads in lakes: sediment incubations. Hydrobiologia 658:253–275.

Gregoire, C., Elsaesser, D., Huguenot, D., Lange, J., Lebeau, T., Merli, A., Mose, R., Passeport, E., Payraudeau, S., Schütz, T., Schulz, R., Tapia-Padilla, G., Tournebize, J., Trevisan, M., Wanko, A. 2009. Mitigation of agricultural nonpoint-source pesticide pollution in artificial wetland ecosystems. Environ. Chem. Lett. 7 205-231.

Gregory, C.T. 2009. Temperature and infiltration characterization of a constructed wetland for wastewater treatment. MS Thesis. Oregon State University. December 2009.

Gumbricht , T. 1993. Nutrient removal capacity in submersed macrophyte pond systems in a temperate climate. Ecol. Eng. 2(1): 49-62.

Harper, H.H. 2007. Current research and trends in alum treatment of stormwater runoff. 9th Biennial Conference on Stormwater Research & Watershed Management.

Harris, K., Watson, F., Brown, K., Burton, R., Carmichael, S., Casagrande, J.M., Casagrande, J.R., Daniels, M., Earnshaw, S., Frank, D., Hanson, E., Lienk, L.L., Martin, P., Travers, B., Watson, J., Wiskind, A. 2007. Agricultural Management Practices and Treatment Wetlands for Water Quality Improvement in Southern Monterey Bay Watersheds: Final Report. Report to California State Water Resources Control Board. The Watershed Institute, California State Monterey Bay, Publication No. WI-2007-01. 162 pp.

Hart B.T., Roberts S., James R., O'Donohue M., Taylor J., Donnert D., Furrer R. 2003. Effectiveness of calcite barriers in preventing phosphorus release from sediments. Australian Journal of Chemistry 56: 207–217.

Hey, D., K. Barrett, and C. Biegen. 1994a. The hydrology of four experimental constructed marshes. Ecological Engineering. 3: 319-343.

Hey, D.L., Barrette, R.K., Biegen, C. 1994b. Water quality improvement by four experimental wetlands. Ecological Engineering. 3(4): 381-397.

Hey, D.L., L.S. Urban and J.A. Kostel. 2005a. Nutrient farming: The business of environmental management. Ecological Engineering 24: 279-287.

Hey, D.L., J.A. Kostel, A.P. Hurter and R.H. Kadlec. 2005b. Comparing Economics of Nitrogen Farming with Traditional Removal. WERF 03-WSM-6CO. Water and Environment Research Foundation, Alexandria, VA.

Heyvaert, A.C., J.E. Reuter and C.R. Goldman. 2006. Subalpine, cold climate, stormwater treatment with a constructed surface flow wetland. Journal of the American Water Resources Association, 42(1): 45-54.

Hickey, C.W., and M. Gibbs. 2009. Lake sediment phosphorus release management – Decision support and risk assessment framework. New Zealand Journal of Marine and Freshwater Research 43: 819-856.

Hiscock, J., C. Thourot and J. Zhang. 2003. Phosphorus budget—land use relationships for the northern Lake Okeechobee watershed. Florida. Ecol. Eng. 21(1):63-74.

House, W.A and F.H. Denison. 2002. Exchange of inorganic phosphate between river waters and bed-sediments. Environ. Sci. Technol. 36:4295-4301.

Ibekwe, A.M., S.R. Lyon, M. Leddy, and M. Jacobson-Meyers. 2006. Impact of plant density and microbial composition on water quality from a free water surface constructed wetland. Journal of Applied Microbiology 102 (2007) 921–936.

Imfeld, G., Braeckevelt, M., Kuschk, P., Richow, H.H. 2009. Monitoring and assessing processes of organic chemicals removal in constructed wetlands. Chemosphere 74 349-362.

Independent Multidisciplinary Science Team (IMST). 2003. IMST Review of the USFWS and NMFS 2001 Biological Opinions on Management of the Klamath Restoration Project and Related Reports. Technical Report 2003-1 to the Oregon Plan for Salmon and Watersheds. Oregon Watershed Enhancement Board. Salem, Oregon.

International Boundary and Water Commission (IBWC). 2008. The New River Wetlands Project, Imperial, California. International Boundary and Water Commission (IBWC). http://ponce.sdsu.edu/brawley_imperial_wetlands_doc.html

Interstate Technology & Regulatory Council (ITRC). 2003. Technical and Regulatory Guidance Document for Constructed Treatment Wetlands. Prepared by the Interstate Technology & Regulatory Council. December 2003.

Johnston, C. A. 1991. Sediment and nutrient retention by freshwater wetlands: effects on surface water quality. Crit. Rev. Environ. Control 21: 491–565.

Kadlec, R. H. 2003. Effects of pollutant speciation in treatment wetlands design. Ecol. Eng. 20 (1):1-16.

Kadlec, R.H. 2005. Nitrogen farming for pollution control. Journal of Environmental Science and Health 40: 1307–1330.

Kadlec, R.H. 2006. Water temperature and evapotranspiration in surface flow wetlands in hot arid climate. Ecol. Eng. 26 (4):328–340.

Kadlec, R. H. 2009.Wastewater treatment at the Houghton Lake wetland: Temperatures and the energy balance. Ecol. Eng. 35:1349-1356.

Kadlec, R.H. 2010. Nitrate dynamics in event-driven wetlands. Ecol Eng 36 (4): 503-516.

Kadlec, R. H. and R L. Knight. 1996. Treatment Wetlands. CRC Press, Boca Raton, Florida.

Kadlec, R.H. and K.R. Reddy. 2001. Temperature effects in treatment wetlands. Water Environ. Res. 73:543–557.

Kadlec, R.H. and S. Wallace. 2009. Treatment Wetlands. 2nd edition. CRC Press. Boca Raton, FL.

Kadlec, R.H., J.S. Bays, L.E. Mokry, D. Andrews, M.R. Ernst. 2011. Performance analysis of the Richland-Chambers treatment wetlands. Ecol. Eng. 37: 176–190

Klang, P., J. Nangia, and V. Wymar. 2010. Evaluation of a GIS-based watershed modeling approach for sediment transport. International Journal of Agricultural and Biological Engineering 3(3): 43-53.

Knight, R.L., Gu, B., Clarke, R.A., and Newman, J.M. 2003. Long-term phosphorus removal in a Florida aquatic system dominated by submerged aquatic vegetation. Ecol. Eng. 20(45):63.

Kuwabara, J.S., Topping, B.R., Carter, J.L., Wood, T.M.., Parchaso, F., Cameron, J.M., Asbill, J.R., Carlson, R.A., and Fend, S.V., 2012, Time scales of change in chemical and biological parameters after engineered levee breaches adjacent to Upper Klamath and Agency Lakes, Oregon: U.S. Geological Survey Open-File Report 2012-1057, 26 p. http://pubs.usgs.gov/of/2012/1057

Lane, R., Day, J.W., Justic, D., Reyes, E., Marx, B., Day, J.N., Hyfield, E. 2004. Changes in stoichiometric Si, N, and P ratios of Mississippi River water diverting through coastal wetlands to the Gulf of Mexico. Estuarine, Coastal Shelf Sci. 60, 1–10.

Lightbody, A., H. Nepf, and J. Bays. 2009. Circulation in wetland deep-zones, and its impact on wetland performance. Ecological Modelling, 35(5):754-768.

Lindsay, W.L. 1979. Chemical Equilibria in Soils. John Wiley & Sons, New York.

Lu, S.Y., Wu, F.C.,Lu, Y.F.,Xiang, C.S.,Zhang, P.Y.,Jin, C.X. 2009. Phosphorus removal from agricultural runoff by constructed wetland. Ecol. Eng. 35(3): 402-409.

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 CH2M HILL and Alex Horne Associates. Prepared for PacifiCorp Energy. January 2009.

Lytle, C.M. 2000. Water Quality Data Review and Wetland Size Estimate for the Treatment of Wastewaters from the Klamath Straits Drain. Draft Technical Memorandum. Prepared for United States Bureau of Reclamation, Klamath Project Office, Klamath Falls. Prepared by David Evans and Associates. July 28, 2000. 15 pp.

Madison, M., D. Taniguchi-Dennis, L. Macpherson, H. Emond, and E. Callaway. 2008. Enhanced Wetlands & Water Quality - the Albany-Millersburg Integrated Wetland Project. Oregon Insider. March 2008.

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 for U.S. Bureau of Reclamation, Klamath Basin Area Office. Proposal No. 07SF200051. June 18. 49 pp.

Malecki-Brown, J.R. White, K.R. Reddy. 2007. Soil biogeochemical characteristics influenced by alum application in a municipal treatment wetland. J. Environ. Qual. 36:1904-1913.

Malecki-Brown, L. M., J.R. White, and H. Brix. 2010. Alum application to improve water quality in a municipal wastewater treatment wetland: effects on macrophyte growth and nutrient uptake. Chemosphere 79:186-192.

Martin, J.L. and S.C. McCutcheon. 1999. Hydrodynamics and Transport for Water Quality Models. Lewis Publishers, London.

Mayer, T., F. Wurster, and D. Craver. 2006. Klamath Marsh Hydrology and Water Rights. Water Resources Branch, U.S. Fish and Wildlife Service. November 2006.

Mayer, T.D. 2005. Water-quality impacts of wetland management in the Lower Klamath National Wildlife Refuge, Oregon and California, USA. Wetlands. Vol. 25, No.3, September 2005, pp. 697-712.

Metcalf & Eddy. 1991. Wastewater Engineering: Treatment, Disposal, and Reuse. McGraw Hill, Third Edition, New York.

Mitsch, W.J. and J. Gosselink. 2000. Wetlands. 3rd Ed. John Wiley & Sons, Inc. New York, NY.

Mitsch, W.J. and J.W. Day, Jr. 2006. Restoration of wetlands in the Mississippi-Ohio-Missouri (MOM) River Basin: Experience and needed research. Ecological Engineering 26: 55-69.

Mitsch, W.J., Wu, X., Nairn, R.W., Weihe, P.E., Wang, N., Deal, R., Boucher, C.E. 1998. Creating and restoring wetlands. BioScience 48, 1019–1030.

Nairn, R.W., T. Beisel, R.C. Thomas, J.A. LaBar K.A. Strevett, D. Fuller, W.H. Strosnider, W.J. Andrews, J. Bays, and R.C. Knox RC. 2009. Challenges in design and construction of a large multi-cell passive treatment system for ferruginous lead-zinc mine waters. In: Proceedings of the 26th National Meeting of the American Society of Mining and Reclamation. Billings, MT. May 30 – June 5, 2009. Pp. 871—892.

Narayanan, A. and R. Pitt. 2006. Costs of Urban Stormwater Control Practices Department of Civil, Construction, and Environmental Engineering The University of Alabama. Tuscaloosa, AL. June 2006.

National Industrial Chemicals Notification and Assessment Scheme. 2001. Full public report: Lanthanum modified clay. File no: NA/899. Australian Government, Department of Health and Aging.

National Research Council (NRC). 2008. Hydrology, Ecology, and Fishes of the Klamath River Basin. National Academies Press, Washington D.C. 272 p.

National Research Council (NRC). 2012. Review of the St. Johns River Water Supply Impact Study: Final Report. Committee to Review the St. Johns River Water Supply Impact Study. National Academies Press, Washington D.C.

North Coast Regional Water Quality Control Board. 2010. Final Staff Report for the Klamath River Total Maximum Daily Loads (TMDLs) and Action Plan 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. North Coast Regional Water Quality Control Board, Santa Rosa CA. March 2010.

Novotny, V. and H. Olem. 1994. Water Quality: Prevention, Identification, and Management of Diffuse Pollution. Van Nostrand Reinhold Publ., New York.

Nguyen, L. and C. Tanner, 1998. Ammonium removal from wastewaters using natural New Zealand zeolites. New Zealand Journal of Agricultural Research 41: 427–446.Orange County Water District (OCWD). 2008. Prado constructed wetlands. Orange County Water District. http://www.ocwd.com/_html/prado.htm

Oregon Department of Environmental Quality (DEQ). 2010. Draft Upper Klamath and Lost River Subbasins Total Maximum Daily Load (TMDL) and Water Quality Management Plan (WQMP). Oregon Department of Environmental Quality (DEQ). February 2010.

Ozkundakci, D., D. Hamilton, and M. Gibbs. 2011. Hypolimnetic phosphorus and nitrogen dynamics in a small, eutrophic lake with a seasonally anoxic hypolimnion. Hydrobiologia 661:5–20.

Paerl, H., L. Valdes, M. Piehler, and C. Stow. 2006. Assessing the effects of nutrient management in an estuary experiencing climatic change: the Neuse River Estuary, North Carolina. Environmental Management 37(3): 422-436.

Parkhurst, D.L. 1995. User's Guide to PHREEQC-A computer program for speciation, reaction-path, advective-transport, and inverse geochemical calculations. U.S. Geological Survey Water Resources Investigations Report 95-4227, 143p.

Patrick, W. H. and R. A. Khalid. 1974. Phosphate release and sorption by soils and sediments: Effect of aerobic and anaerobic conditions. Science 186:53-55.

Peluso, V. and A. Marshall. 2002. Best Management Practices for South Florida Urban Stormwater Management Systems. Appendix A - Typical Costs Associated with Structural BMPs. Everglades Stormwater Program South Florida Water Management District, West Palm, FL.

Perdue, E.M., Lytle, C.R., Sweet, M.S., and Sweet, J.W. 1981. The chemical and biological impact of Klamath marsh water on the Williamson River, Oregon: U.S. Department of the Interior Office of Water Research and Technology, Project A-047-ORE, Portland, Ore., Portland State University, 199 p.

Perez, B.C., J.W. Day, D. Justic, R.R. Lane and R.R. Twilley. 2011. Nutrient stoichiometry, freshwater residence time, and nutrient retention in a river-dominated estuary in the Mississippi Delta. Hydrobiologia. 658 (1): 41-54.

Pernitsky, D. and J. Edzwald. 2006. Practical Paper. Selection of alum and polyaluminum coagulants: principles and applications. 2006 Journal of Water Supply: Research and Technology—AQUA. 55.2 (2006): 121-141.

Picard C.R., Fraser, L.H., and D. Steer. 2005. The interacting effects of temperature and plant community type on nutrient removal in wetland microcosms, Biores. Technol. 96:1039–1047.

Pietro, K., G. Germain, R. Bearzotti, and N. Iricanin. 2010. Chapter 5: Performance and Optimization of the Everglades Stormwater Treatment Areas. 2010 South Florida Environmental Report, Vol. I.

Pilgrim, K.M., and P.L. Brezonik. 2005. Evaluation of the potential adverse effects of lake inflow treatment with alum. Lake and Reservoir Management 21: 78-88.

Porter, S., B. Mallory, and B. Hecht. 2002. Review of Phase II Instream "Unimpaired" Flows in the Klamath River. Prepared by Balance Hydrologics, Inc. for the Yurok Tribe. March 2002.

Pychra, C. and E. Lopez. 2003. Municipal Wastewater Lagoon Phosphorus Removal. USEPA Technical Support Section, Water Compliance Branch, Chicago, IL. www.lagoonsonline.com/phosphorus.

Reddy, K.R., R.G. Wetzel, and R.H. Kadlec. 2005. Biogeochemistry of Phosphorus in Wetlands. Pp. 263-316. In Phosphorus: Agriculture and the Environment. Agronomy Monograph 46, American Society of Agronomy, Madison, WI.

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 Rev. Environ. Sci. Technol. 41: 149-186.

Reed, S.C., R.W. Crites, and E.J. Middlebrooks. 1995. Natural Systems for Waste Management and Treatment, McGraw-Hill, Inc., New York. 433pp.

Reilly, J.F., Horne, A.J., and C.D. Miller. 2000. Nitrate removal from a drinking water supply with a large-scale, free surface constructed wetlands prior to groundwater recharge. Ecol. Eng. 14: 33-47.

Rew, T.J. 2006. Aluminum water treatment residuals for reducing phosphorus loss from manure-impacted, high water table soils. MS Thesis, University of Florida.

Richardson, C. J. 1985. Mechanisms controlling phosphorus retention capacity in freshwater wetlands. Science 228: 1424–1426.

Robb, M., B. Greenop, Z. Goss, G. Douglas, and J. Adeney. 2003. Application of Phoslock (TM), an innovative phosphorus binding clay, to two Western Australian waterways: preliminary findings. Hydrobiologia. 494 (1-3), 237-243.

Sedlak, R.I. 1991. Phosphorus and Nitrogen Removal From Municipal Wastewater: Principles and Practice, Second Edition, Lewis Publishers, Boca Raton, Florida.

Sherwood, L. and R. Qualls. 2001. Stability of Phosphorus within a Wetland Soil following Ferric Chloride Treatment to Control Eutrophication. Environ. Sci. Technol. 35:4126-4131.

Shilton, A.; Pratt, S.; Drizo, A.; Mahmood, B.; Banker, S.; Billings, L.; Glenny, S.; Luo, D. 2005. 'Active' filters for upgrading phosphorus removal from pond systems. Water Science and Technology 51(12): 111-116.

Shuib, N., K. Baskaran, W. Davies, and S. Muthukumaran. 2011. Effluent quality performance of horizontal subsurface flow constructed wetlands using natural zeolite (escott). Proceedings of the 2011 International Conference on Environment Science and Engineering (IPCBEE).8:19-23.Smesrud, J., M. Boyd, B. Kasper, and S. Eisner. 2007. An Energy Balance Model for Effluent Heat Reduction through Wetlands. Presented at the 2007 Conference of the Pacific Northwest Clean Water Association.

Smith, D., and Berryman & Henigar, Inc. 2006. Hillsborough Filter Pilot Demonstration Final Report. Taliaferro Stormwater Research Facility. Hillsborough County, Florida. SWFWMD Permit No. 44024189.002. Prepared for Hillsborough County Department of Public Works and the Florida Department of Environmental Protection. July 2006.Snoeyink, V.L. and D. Jenkins. 1980. Water Chemistry. John Wiley & Sons, New York.

Snyder, D.T., and Morace, J.L. 1997. Nitrogen and phosphorus loading from drained wetlands adjacent to Upper Klamath and Agency Lakes, Oregon: U.S. Geological Survey Water-Resources Investigations Report 97–4059, 67 p.

Sojka, R.E. and Lentz, R.D. 1997. A PAM Primer: A Brief History of PAM and PAM Related Issues. U.S. Department of Agriculture, Agricultural Research Service, Northwest Irrigation and Soils Research Lab. Posted on USDA-ARS website. http://www.nwisrl.ars.usda.gov/research/PAM/primer.

Song, Y., P.T. Weidler, U. Berg, R. Nuesch, and D. Donnert. 2006. Calcite-Seeded Crystallization of Calcium Phosphate for Phosphorus Recovery. Chemosphere. 63(2): 236-243.

South Florida Ecosystem Restoration Task Force. 2011. 2011 Integrated Financial Plan. Submitted to the U.S. Congress, Florida Legislature, Seminole Tribe of Florida, and Miccosukee Tribe of Indians of Florida.

South Florida Water Management District (SFWMD). 2002. Everglades Consolidated Report: Chapter 4C: Advanced Treatment Technologies. South Florida Water Management District.

South Florida Water Management District (SFWMD). 2004. Central and Southern Florida Project Comprehensive Everglades Restoration Plan. Development of Alternative Plans. Part 1 – Storage and Water Quality. Lake Okeechobee Watershed Project.

South Florida Water Management District (SFWMD). 2007. Final Report. Task 1. Search and review of existing information. Technical assistance in review and analysis of existing data for evaluation of legacy phosphorus in the Lake Okeechobee Watershed. South Florida Water Management District. September 2007.

South Florida Water Management District (SFWMD). 2008. Task 3 Report. Legacy P Abatement Plan. Technical assistance in review and analysis of existing data for evaluation of legacy phosphorus in the Lake Okeechobee Watershed. South Florida Water Management District. August 2008.

South Florida Water Management District (SFWMD). 2010. 2010 South Florida Environmental Report – Volume I, South Florida Water Management District, West Palm Beach, FL.

South Florida Water Management District (SFWMD). 2011. South Florida Environmental Report. West Palm Beach, FL. www.sfwmd.gov

Sprague, L., M. Langland, S. Yochum, R. Edwards, J. Blomquist, S. Phillips, G. Sherik, and S. Preston. 2000. Factors Affecting Nutrient Trends in Major Rivers of the Chesapeake Bay Watershed. U.S. Geological Survey Water Resources Investigations Report 00-4218. 98 pp.

Stefanakis, A. and V. Tsihrintzis. 2012. Use of zeolite and bauxite as filter media treating the effluent of Vertical Flow Constructed Wetlands. Microporous and Mesoporous Materials 155: 106–116.

Stieber, T.D., and Bahr, G.L. 1996. Reduction of Nutrient and Pesticide Losses Through the Application of Polyacrylamide in Surface Irrigated Crops. Proceedings of the 1996 PAM Conference, Twin Falls, ID. pp. 41-48.

Stow, C., M. Borsuk, and D. Stanley. 2001. Long-term changes in watershed nutrient inputs and riverine exports in the Neuse River, North Carolina. Water Research 35(6): 1489-99.

Strecker, E., Howell, J., Thayumanavan, A. and Leisenring, M. 2005. Lake Tahoe basin Stormwater BMP Evaluation and Feasibility Study. Prepared for Lahontan Regional Water Quality Control Board and UCD Tahoe Research Group, by GeoSyntec Consultants, July 2005.

Stumm, W., and J. Morgan. 1995. Aquatic Chemistry, Chemical Equilibria and Rates in Natural Waters. Wiley-Interscience, New York, pp. 325-334.

Tátrai, I., K. Mátyás, J. Korponai, G. Paulovits and P. Pomogyi. 2000. The role of the Kis-Balaton Water Protection System in the control of water quality of Lake Balaton. Ecol. Eng. 16: 73-78.

Taylor, A. and T. Wong. 2002. Non-structural Stormwater Quality Best Management Practices – A Literature Review of Their Value and Life-cycle Costs. Technical Report No. 02/13. Cooperative Research Centre for Catchment Hydrology. Melbourne, Victoria, Australia.

Tetra Tech and WMS. 2007. New and Alamo River Wetlands Master Plan. Revised Final Report. Prepared by Tetra Tech, Inc., Lafayette, CA, and Wetlands Management Services, Chelsea, MI. May 21, 2007.

The Nature Conservancy (TNC). 2011. Water Quality Conditions on the Williamson River Delta, Oregon: Three Years Post-Restoration. 2010 Annual Data Report. Klamath Falls, Oregon.

Tiedje, J.M. 1982. Denitrification. Pp. 1011-1026. In A.L. Page, R.H. Miller and D.R. Keeney (eds.), Methods of Soil Analysis, Part 2. Agronomy Monograph No. 9., Amer. Soc. Agron., Madison, WI.

Trejo-Gaytan, J., P. Bachand, and J. Darby. 2006. Treatment of Urban Runoff at Lake Tahoe: Low-Intensity Chemical Dosing. Water Environment Research 78(13): 2487-2500.

U.S. Environmental Protection Agency (EPA). 2000a. Constructed Wetlands Treatment of Municipal Wastewaters. U.S. Environmental Protection Agency, Office of Research and Development. EPA/625/R-99/010, September 2000.

U.S. Environmental Protection Agency (EPA). 2000b. Guiding Principles for Constructed Treatment Wetlands. U.S. Environmental Protection Agency, Office of Research and Development. EPA 843-B-00-003. October 2000.

Van der Tak, L., J. Bays, T. Beatty, and G. Fuselier. 2005. Biological Treatment of De-Icing Agents and Stormwater for Runway 11-19r and Associated Taxiways, Washington Dulles International Airport. Proceedings of the Water Environment Federation, WEFTEC 2005: Session 31 through Session 40, pp. 3109-3123 (15).

Vopel, K., M. Gibbs, C. Hickey, and J. Quinn. 2008. Modification of sediment–water solute exchange by sediment-capping materials: effects on O2 and pH. Marine and Freshwater Research 59: 1101–1110.Vymazal, J. 2011. Constructed Wetlands for Wastewater Treatment: Five Decades of Experience. Environ. Sci. Technol. 2011, 45, 61–69.

Walbridge, M. R. and J. P. Struthers. 1993. Phosphorus retention in non-tidal palustrine forested wetlands of the mid-Atlantic region. Wetlands 13: 84–94.

Wang, D., J. Dorioz, D. Trevisan, D. Braun, L. Windhausen, and J. Vansteelant. 2004. Using a landscape approach to interpret diffuse phosphorus pollution and assist with water quality management in the basins of Lake Champlain (Vermont) and Lac Léman (France). pp. 159-190 IN T.O. Manley, P.L. Manley, and T.B. Mihuc. (eds.) Lake Champlain: Partnerships and Research in the New Millennium. Kluwer Academic Publishers: New York.

Wang, N. and W.J. Mitsch. 2000. A detailed ecosystem model of phosphorus dynamics in created riparian wetlands. Ecological Modeling 126:101-130.

Watershed Technologies. 2008. Implementation of hybrid wetland treatment technology in the Northern Everglades Watershed. Task 45 Deliverable: Final Report. Prepared for: Florida Department of Agriculture and Consumer Services (FDACS).

Welch, E.B. and G.D. Cooke. 1999. Effectiveness and longevity of phosphorus inactivation with alum. J. Lake and Reserv. Manag. 15:5-27.

Welch, E.B. and G.D. Schrieve. 1994. Alum treatment effectiveness and longevity in shallow lakes. Hydrobiologia. Volume 275-276, Number 1, 423-431.

Werker, A.G., Dougherty, J.M., McHenry, J.L., and W.A. Van Loon. 2002. Treatment variability for wetland wastewater treatment design in cold climates, Ecol. Eng. 19 (1):1–11.

Widiastutia, N., H. Wua, M. Ang, and D. Zhang. 2008. The potential application of natural zeolite for greywater treatment. Desalination 218: 271–280.Windhausen, L., D. Braun, and D. Wang. 2004. A landscape scale evaluation of phosphorus retention in wetlands of the Laplatte River Basin, Vermont, USA. Pp. 221-240 in T.O. Manley, P.L. Manley, and T.B. Mihuc. (eds.) Lake Champlain: Partnerships and Research in the New Millennium. Kluwer Academic Publishers: New York.

Wisniewski, R. 1999. Phosphate inactivation with iron chloride during sediment resuspension. Lakes and Reservoir: Research and Management. 4:65-73.

Woltemade, C. J. 2000. Ability of restored wetlands to reduce nitrogen and phosphorus concentrations in agricultural drainage water. Journal of Soil and Water Conservation, 55(3): 303-309.

Wong, S. and C. Bienz. 2011. Summary of Water Quality Sampling at Sycan Marsh, Oregon, 2010–2011. The Nature Conservancy, Klamath Falls, OR. 17 pp.

Wong, S. and H. Hendrixson. 2011. Water Quality Conditions on the Williamson River Delta, Oregon: Three Years Post-Restoration. 2010 Annual Data Report. July 22, 2011. 38 pp.

Wong, S., M. Barry, N. Rudd, H. Hendrixson and C. Doehring. 2011. Nutrient Release from a Recently Flooded Delta Wetland: Comparison of Field Measurements to Laboratory Results. Wetlands 31(2): 433-443.

Wong, T. and W. Geiger. 1997. Adaptation of wastewater surface flow wetland formulae for application in constructed storm water wetlands. Ecological Engineering 9:187-202.

WRI. 1992. Intensive Studies of Wetland Functions: 1990-1991 Research Summary of the Des Plaines River Wetlands Demonstration Project. Technical Paper No. 2, December 1992.

Zhou, A., H. Tang, and D. Wang. 2005. Phosphorus adsorption on natural sediments: Modeling and effects of pH and sediment composition. Water Research 39:1245-1254.

Zurayk, R., Nimah, M., Geha, Y., Rizk, C. 1997. Phosphorus retention in the soil matrix of constructed wetlands. Communications in Soil Science and Plant Analysis 28(6-8): 521-535.

APPENDIX A
Wetlands Temperature Information

APPENDIX A
Wetlands Temperature Information

As described in section 3.2.3 of the report, potential wetland water temperatures were calculated for various hypothetical wetland depths and meteorological conditions in the Upper Klamath basin. Water temperature conditions for the hypothetical wetland setting were estimated using the heat budget formulations of Martin and McCutcheon (1999)[1] in a spreadsheet model made in Microsoft Excel by Watercourse Engineering. The governing equation for this model is a simplification of the advection diffusion equation:

$$\frac{\partial T_w}{\partial t} = S = \frac{H_n A_p}{C_p \rho V_p}$$

Where:

T_w = water temperature,

t = hourly time step,

S = sources/sinks,

H_n = net heat flux,

A_p = surface area,

C_p = specific heat of water,

r = density of water, and

V_p = volume.

The net heat flux *(H_n)* expresses the net heat transfer to the water body as:

$$H_n = H_{sw} + H_{atm} - H_b + H_L + H_s$$

Where:

H_{sw} = shortwave radiation,

H_{atm} = downwelling longwave radiation,

H_b = upwelling longwave radiation,

H_L = latent heat flux, and

H_s = sensible heat flux.

The net heat flux (H_n) was calculated on an hourly basis using meteorological data from the Klamath Falls Agrimet Station (KFLO)[2] for a representative August day. The hourly H_n values were then used to calculate hourly changes in water temperature based on surface area, and given water properties such as density and specific heat capacity (using the advection diffusion equation listed above). Depth is user-specified, and was set from 1 to 5 ft to represent various hypothetical wetland depths. Table A-1 list the constants and coefficients used in the temperature calculations .Table A-2 provides the hourly meteorological data used in the calculations. Figure 8 (as discussed in section 3.2.3 of the report) shows the maximum, minimum, and mean water temperatures (among the calculated hourly values) that were determined for various hypothetical wetland depths, and the associated

[1] Martin, J. and S. McCutcheon. 1999. Hydrodynamics and Transport for Water Quality Models. Lewis Publishers, London.

[2] KFLO is a station in the U.S. Bureau of Reclamation's meteorological network.

range in air temperature (based on hourly meteorological observations from Klamath Falls for a representative August day).

TABLE A-1
Constants and Coefficients Used in Temperature Calculations

Constant or Coefficient	Value
Swinbank's Coefficient	0.94×10^{-5} K^{-2}
Stefan-Boltzman Constant	5.669×10^{-8} Wm^{-2} K^{-4}
Reflectivity (atmospheric)	0.03
Emissivity	0.97
Water density	997.33 kg m^{-3}
Evaporative heat flux coefficient, a	1.7222×10^{-9} m s^{-1} mb^{-1}
Evaporative heat flux coefficient, b	1.5277×10^{-9} mb^{-1}

TABLE A-2
Hourly Meteorological Data Used in Calculations, Including Cloud Cover (CLD), Dry Bulb Temperature (DBT), Dew Point Temperatures (DPT), Wet Bulb Temperature (WetB), Atmospheric Pressure (APR), Wind Speed (WND), and Shortwave Solar Radiation (SWS). Data from the Klamath Falls Agrimet Station (KFLO) for August 1, 2001.

Date & Time	CLD (Fraction)	DBT (°C)	DPT (°C)	WetB (°C)	APR (mb)	WND (m/s)	SWS (W/m2)
8/1/2001 0:00	0.0	17.22	5.60	11.00	930	0.00	0.00
8/1/2001 1:00	0.0	16.67	5.68	10.90	930	1.34	0.00
8/1/2001 2:00	0.0	16.11	5.77	11.10	930	0.45	0.00
8/1/2001 3:00	0.0	14.44	6.29	9.70	930	0.00	0.00
8/1/2001 4:00	0.0	15.00	6.29	10.20	930	1.34	0.00
8/1/2001 5:00	0.0	15.00	4.99	10.30	930	0.00	0.00
8/1/2001 6:00	0.0	13.33	5.77	8.80	930	0.00	0.00
8/1/2001 7:00	0.0	14.44	6.62	8.80	930	0.00	0.00
8/1/2001 8:00	0.0	17.78	7.03	10.90	930	2.68	0.00
8/1/2001 9:00	0.0	20.00	8.20	11.80	930	3.13	0.00
8/1/2001 10:00	0.0	23.33	8.30	14.20	930	2.68	0.00
8/1/2001 11:00	0.0	23.89	9.98	14.10	930	1.79	0.00
8/1/2001 12:00	0.0	27.22	11.39	16.80	930	2.68	0.00
8/1/2001 13:00	0.0	28.33	11.48	17.00	930	3.58	0.00
8/1/2001 14:00	0.0	30.00	11.96	18.20	930	3.58	0.00
8/1/2001 15:00	0.0	30.56	11.43	18.00	930	3.58	0.00
8/1/2001 16:00	0.0	31.67	12.42	19.00	930	4.47	0.00

TABLE A-2

Hourly Meteorological Data Used in Calculations, Including Cloud Cover (CLD), Dry Bulb Temperature (DBT), Dew Point Temperatures (DPT), Wet Bulb Temperature (WetB), Atmospheric Pressure (APR), Wind Speed (WND), and Shortwave Solar Radiation (SWS). Data from the Klamath Falls Agrimet Station (KFLO) for August 1, 2001.

Date & Time	CLD (Fraction)	DBT (°C)	DPT (°C)	WetB (°C)	APR (mb)	WND (m/s)	SWS (W/m2)
8/1/2001 17:00	0.0	32.22	9.99	17.90	930	1.79	0.00
8/1/2001 18:00	0.0	33.33	9.88	19.40	930	3.58	0.00
8/1/2001 19:00	0.0	31.11	9.08	18.50	930	3.13	0.00
8/1/2001 20:00	0.0	28.33	7.15	17.00	930	3.58	0.00
8/1/2001 21:00	0.0	25.00	6.18	14.60	930	2.68	0.00
8/1/2001 22:00	0.0	23.89	6.09	14.10	930	3.13	0.00
8/1/2001 23:00	0.0	22.78	5.35	13.80	930	1.79	0.00

APPENDIX B
Treatment Wetlands Model Input Data

APPENDIX B
Treatment Wetlands Model Input Data

TABLE B-1
P-k-C* Model Water Quality and Flow Input Data (as Discussed in Chapter 4) for the Link River Location

Parameter	Unit	Mean (May-October)	May	Jun	Jul	Aug	Sep	Oct
BOD	mg/L	7.12	2.94	6.20	9.71	12.20	10.33	7.44
TSS	mg/L	13.30	9.40	3.60	13.00	21.80	16.53	12.80
N02-3 N	mg/L	0.10	0.06	0.06	0.10	0.11	0.05	0.12
NH3 N	mg/L	0.18	0.11	0.09	0.27	0.33	0.14	0.20
TP	mg/L	0.16	0.16	0.19	0.28	0.18	0.15	0.15
TN	mg/L	2.16	0.91	1.53	2.44	4.00	2.74	2.32
Flow	cfs	1,164	1,660	1,500	1,020	941	679	781

TABLE B-2
P-k-C* Model Water Quality and Flow Input Data (as Discussed in Chapter 4) for the Iron Gate Location

Parameter	Unit	Mean (May-October)	May	Jun	Jul	Aug	Sep	Oct
BOD	mg/L	3.09	2.63	2.75	3.00	5.00	2.20	4.78
TSS	mg/L	7.93	7.98	7.87	8.04	8.53	8.00	7.28
N02-3 N	mg/L	0.28	0.17	0.17	0.21	0.20	0.42	0.43
NH3 N	mg/L	0.10	0.13	0.08	0.08	0.06	0.07	0.13
TP	mg/L	0.15	0.16	0.18	0.16	0.15	0.19	0.12
TN	mg/L	1.17	0.67	1.10	1.38	1.48	1.24	1.20
Flow	cfs	1,513	2,280	1,540	969	942	1,070	1,240

APPENDIX C
Treatment Wetlands Model Output (Link River Location)

APPENDIX C
Treatment Wetlands Model Output (Link River Location)

As described in section 4.3.1 of the report, this appendix contains output from the P-k-C* model for the Link River location for scenarios 1A, 1B, 2, 3, 4A-4F, 4G-4L, 5A, 5B, 6A, and 6B as listed in Table 7 (in section 4.2.3).

Scenarios: 1A and 5A

Surface Flow Monthly PkC* Treatment Wetland Design Model w/ Water Balance (Wastewater Parameters)

User inputs indicated by white boxes.
Pop-up notes indicated by red triangles.

Project Name: ath Wetlands - 2012 nitrogen runs- 50% Klamath River at Link River
Project Number: WQ-KR253:ppt,tempKlamathFallsAirp

Flow Rate Converter

	Value	Units
Flow (Enter monthly flowrate below under "General and Hydrologic Input Data"; use this cell simply for converstion between units)		cfs
Converted Flow		m^3/d

Add Results to NADB Charts

General and Hydrologic Input Data

Wetland Hydrology

Month	Days in Month	Air Temp (°C)	Precip (m/mo)	Inflow (m3/d)	ET (m/mo)	Infiltration (m/mo)	Outflow (m3/d)
January	31	-0.7	0.04	1513207	0.06	0.37	515257
February	28	1.0	0.03	1362743	0.02	0.37	321866
March	31	3.6	0.02	1591497	0.01	0.37	637667
April	30	5.9	0.03	1925455	0.02	0.37	950875
May	31	10.5	0.03	2028211	0.03	0.37	1058753
June	30	14.6	0.01	1832485	0.06	0.37	710129
July	31	19.4	0.00	1247754	0.08	0.37	75944
August	31	18.4	0.01	1151114	0.12	0.37	0
September	30	14.2	0.01	830612	0.15	0.37	0
October	31	8.3	0.02	955388	0.18	0.37	0
November	30	2.8	0.05	937038	0.15	0.37	0
December	31	-1.0	0.04	976184	0.11	0.37	0
Average		8.1	0.025	1362641	0.08	0.37	355874

Water Quality Input Data

Month	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
January								
February								
March								
April								
May			0.74	0.11	0.06		0.16	
June			1.39	0.09	0.06		0.19	
July			2.07	0.27	0.10		0.28	
August			3.57	0.33	0.11		0.18	
September			2.57	0.14	0.05		0.15	
October			2.05	0.20	0.12		0.15	
November								
December								
Annual Average			2.07	0.19	0.08		0.18	

		BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Target Effluent Conc., mg/L	C_e =								
Desired Confidence Percentile		0.5		0.5	0.5	0.5	0.5	0.5	0.5
Max Month/Annual Factor		1.7	1.9	1.8	2.5	2.5	1.6	1.8	3.0
Design Target Conc., mg/L	C_d =								
Wetland Background Limit, mg/L	C* =	Influent Dependent	Influent Dependent	1	0	0	1	0.02	300
P-Factor	P =	1.0	3.0	3.0	3.0	3.0	3.0	4.0	3.0
Confidence-based Rate Constant, 20°C, m/y	k_{20} =	0	200	17	18	27	13	10	75
Temperature Factor	θ =	1.00	1.065	1.05	1.05	1.11	1.06	1.00	1.00

Wetland Size and Output Predictions

Wetland Area

20,000	ac
8097.17	ha
80971660	m2

Effluent Concentrations

Month	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
January								
February								
March								
April								
May			0.91	0.00	0.02	0.98	0.06240	
June			1.1	0.12	0.07	1.28	0.06262	
July			1.1	0.15	0.09	1.33	0.04858	
August			1.2	0.26	0.16	1.58	0.03145	
September			1.0	0.01	0.05	1.11	0.01620	
October			1.1	0.12	0.17	1.40	0.01869	
November								
December								
Annual Average			*1.07*	*0.11*	*0.09*	*1.28*	*0.04*	

	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Avg Percent Reduction (by concentration)			48%	42%	-11%	45%	78%	

Scenarios: 1A and 5A

Surface Flow Monthly PkC* Treatment Wetland Design Model w/ Water Balance (Wastewater Parameters)

User inputs indicated by white boxes.
Pop-up notes indicated by red triangles.

Project Name: nath Wetlands - 2012 nitrogen runs- 50% Klamath River at Link River
Project Number: WQ-KR253:ppt,tempKlamathFallsAirp

Average Mass Loading (lb/day)				6202.8	572.7	251.4	7026.8	553	
Monthly Mass Loading (kg/ha/d)	Jan								
	Feb								
	Mar								
	Apr								
	May			0.2	0.0	0.0	0.2	0.0	
	Jun			0.3	0.0	0.0	0.3	0.0	
	Jul			0.3	0.0	0.0	0.4	0.0	
	Aug			0.5	0.0	0.0	0.6	0.0	
	Sep			0.3	0.0	0.0	0.3	0.0	
	Oct			0.2	0.0	0.0	0.3	0.0	
	Nov								
	Dec								
Average Mass Loading (kg/ha/d)				0.3	0.0	0.0	0.4	0.0	
Average Mass Out (lb/day)				837	87	73	1003	31	
Monthly Mass Out (kg/ha/d)	Jan								
	Feb								
	Mar								
	Apr								
	May			0.1	0.0	0.0	0.1	0.0	
	Jun			0.1	0.0	0.0	0.1	0.0	
	Jul			0.0	0.0	0.0	0.0	0.0	
	Aug			0.0	0.0	0.0	0.0	0.0	
	Sep			0.0	0.0	0.0	0.0	0.0	
	Oct			0.0	0.0	0.0	0.0	0.0	
	Nov								
	Dec								
Average Mass Out (kg/ha/d)				0.0	0.0	0.0	0.1	0.0	
Percent Reduction (by mass)				87%	85%	71%	86%	94%	

Hydraulic Properties Based on Area and Flow

						Override
Percent Open Water	10%	Transition Side Slopes (H:V) - (10:1)	10			
Marsh Zone Depth (m)	0.46	Berm Side Slopes (H:V) - (4:1)	4	# of Deep Zones	40	
Deep Zone Depth (m)	1.4	Approx Aspect Ratio (L:W) - (5:1)	5	Deep Zone Bot Length	50	
Volume (m3)	36108397					

Hydraulic Loading Rate, q	HLR =	1.7	cm/d
Nominal Hydraulic Residence Time, days	HRT =	26.5	days

Nitrogen Species Calculations

Nitrogen Models
per K&K Eqns 13-28. 13-29, 13-39:
Adapted for Monthly TIS model
Organic Nitrogen (ON)

$$C_{ON_{OUT}} = C_{ON}^{*} + (C_{ON_{IN}} - C_{ON}^{*})\left[1 + \frac{k_{ON}A}{NQ}\right]^{-N}$$

Ammonia Nitrogen (AN)

$$C_{AN_{OUT}} = C_{AN}^{*} + (C_{AN_{IN}} - C_{AN}^{*})\left[1 + \frac{k_{AN}A}{NQ}\right]^{-N} + \left(\frac{k_{ON}}{k_{AN} - k_{ON}}\right)(C_{ON_{IN}} - C_{ON}^{*})\left(\left[1 + \frac{k_{ON}A}{NQ}\right]^{-N} - \left[1 + \frac{k_{AN}A}{NQ}\right]^{-N}\right)$$

Nitrate Nitrogen (NN)

$$C_{NN_{OUT}} = C_{NN_{IN}}\left[1 + \frac{k_{NN}A}{NQ}\right]^{-N} + \Psi\left[\begin{array}{l} \left(\frac{k_{AN}}{k_{NN} - k_{AN}}\right)C_{AN_{IN}}\left(\left[1 + \frac{k_{AN}A}{NQ}\right]^{-N} - \left[1 + \frac{k_{NN}A}{NQ}\right]^{-N}\right) \\ + \left(\frac{k_{ON}}{k_{AN} - k_{ON}}\right)\left(\frac{k_{AN}}{k_{NN} - k_{ON}}\right)(C_{ON_{IN}} - C_{ON}^{*})\left(\left[1 + \frac{k_{ON}A}{NQ}\right]^{-N} - \left[1 + \frac{k_{NN}A}{NQ}\right]^{-N}\right) \\ - \left(\frac{k_{ON}}{k_{AN} - k_{ON}}\right)\left(\frac{k_{AN}}{k_{NN} - k_{AN}}\right)(C_{ON_{IN}} - C_{ON}^{*})\left(\left[1 + \frac{k_{AN}A}{NQ}\right]^{-N} - \left[1 + \frac{k_{NN}A}{NQ}\right]^{-N}\right) \end{array}\right]$$

where Ψ = fraction of ammonium nitrified, assumed to be 100% = 100%

Scenarios: 1A and 6A

Surface Flow Monthly PkC* Treatment Wetland Design Model w/ Water Balance (Wastewater Parameters)

User inputs indicated by white boxes.
Pop-up notes indicated by red triangles.

Project Name: ath Wetlands - 2012 nitrogen runs- 50% Klamath River at Link River
Project Number: WQ-KR253:ppt,tempKlamathFallsAirp

Flow Rate Converter

	Value	Units
Flow (Enter monthly flowrate below under "General and Hydrologic Input Data"; use this cell simply for converstion between units)	1237	cfs
Converted Flow	3,026,444	m^3/d

Add Results to NADB Charts

General and Hydrologic Input Data

Wetland Hydrology

Month	Days in Month	Air Temp (°C)	Precip (m/mo)	Inflow (m3/d)	ET (m/mo)	Infiltration (m/mo)	Outflow (m3/d)
January	31	-0.7	0.04	1513207	0.06	0.37	515257
February	28	1.0	0.03	1362743	0.02	0.37	321866
March	31	3.6	0.02	1591497	0.01	0.37	637667
April	30	5.9	0.03	1925455	0.02	0.37	950875
May	31	10.5	0.03	2028211	0.03	0.37	1058753
June	30	14.6	0.01	1832485	0.06	0.37	710129
July	31	19.4	0.00	1247754	0.08	0.37	75944
August	31	18.4	0.01	1151114	0.12	0.37	0
September	30	14.2	0.01	830612	0.15	0.37	0
October	31	8.3	0.02	955388	0.18	0.37	0
November	30	2.8	0.05	937038	0.15	0.37	0
December	31	-1.0	0.04	976184	0.11	0.37	0
Average		8.1	0.025	1362641	0.08	0.37	355874

Water Quality Input Data

Month	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
January								
February								
March								
April								
May			0.74	0.11	0.06		0.16	
June			1.39	0.09	0.06		0.19	
July			2.07	0.27	0.10		0.28	
August			3.57	0.33	0.11		0.18	
September			2.57	0.14	0.05		0.15	
October			2.05	0.20	0.12		0.15	
November								
December								
Annual Average			2.07	0.19	0.08		0.18	

		BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Target Effluent Conc., mg/L	C_e =								
Desired Confidence Percentile		0.5		0.5	0.5	0.5	0.5	0.5	0.5
Max Month/Annual Factor		1.7	1.9	1.8	2.5	2.5	1.6	1.8	3.0
Design Target Conc., mg/L	C_d =								
Wetland Background Limit, mg/L	C* =	Influent Dependent	Influent Dependent	0.5	0	0	0.5	0.02	300
P-Factor	P =	1.0	3.0	3.0	3.0	3.0	3.0	4.0	3.0
Confidence-based Rate Constant, 20°C, m/y	k_{20} =	0	200	17	18	27	13	10	75
Temperature Factor	θ =	1.00	1.065	1.05	1.05	1.11	1.06	1.00	1.00

Wetland Size and Output Predictions

Wetland Area

20,000	ac
8097.17	ha
80971660	m2

Effluent Concentrations

Month	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
January								
February								
March								
April								
May			0.57	0.10	0.06	0.73	0.06240	
June			0.69	0.24	0.12	1.05	0.06262	
July			0.61	0.19	0.11	0.92	0.04858	
August			0.66	0.28	0.18	1.12	0.03145	
September			0.50	0.00	0.00	0.50	0.01620	
October			0.57	0.09	0.12	0.77	0.01869	
November								
December								
Annual Average			*0.60*	*0.15*	*0.10*	*0.85*	*0.03999*	

	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Avg Percent Reduction (by concentration)			71%	21%	-18%	64%	78%	

Scenarios: 1A and 6A

Surface Flow Monthly PkC* Treatment Wetland Design Model w/ Water Balance (Wastewater Parameters)

User inputs indicated by white boxes.
Pop-up notes indicated by red triangles.

Project Name: hath Wetlands - 2012 nitrogen runs- 50% Klamath River at Link River
Project Number: WQ-KR253:ppt,tempKlamathFallsAirp

Average Mass Loading (lb/day)				6202.8	572.7	251.4	7026.8	553	
Monthly Mass Loading (kg/ha/d)	Jan								
	Feb								
	Mar								
	Apr								
	May			0.2	0.0	0.0	0.2	0.0	
	Jun			0.3	0.0	0.0	0.3	0.0	
	Jul			0.3	0.0	0.0	0.4	0.0	
	Aug			0.5	0.0	0.0	0.6	0.0	
	Sep			0.3	0.0	0.0	0.3	0.0	
	Oct			0.2	0.0	0.0	0.3	0.0	
	Nov								
	Dec								
Average Mass Loading (kg/ha/d)				0.3	0.0	0.0	0.4	0.0	
Average Mass Out (lb/day)				470	118	77	665	31	
Monthly Mass Out (kg/ha/d)	Jan								
	Feb								
	Mar								
	Apr								
	May			0.1	0.0	0.0	0.1	0.0	
	Jun			0.1	0.0	0.0	0.1	0.0	
	Jul			0.0	0.0	0.0	0.0	0.0	
	Aug			0.0	0.0	0.0	0.0	0.0	
	Sep			0.0	0.0	0.0	0.0	0.0	
	Oct			0.0	0.0	0.0	0.0	0.0	
	Nov								
	Dec								
Average Mass Out (kg/ha/d)				0.0	0.0	0.0	0.0	0.0	
Percent Reduction (by mass)				92%	79%	69%	91%	94%	

Hydraulic Properties Based on Area and Flow

Percent Open Water	10%	Transition Side Slopes (H:V) - (10:1)	10			Override
Marsh Zone Depth (m)	0.46	Berm Side Slopes (H:V) - (4:1)	4	# of Deep Zones	40	
Deep Zone Depth (m)	1.4	Approx Aspect Ratio (L:W) - (5:1)	5	Deep Zone Bot Length	50	
Volume (m3)	36108397					

Hydraulic Loading Rate, q	HLR =	1.7	cm/d
Nominal Hydraulic Residence Time, days	HRT =	26.5	days

Nitrogen Species Calculations

Nitrogen Models
per K&K Eqns 13-28. 13-29, 13-39:
Adapted for Monthly TIS model
Organic Nitrogen (ON)

$$C_{ON_{OUT}} = C_{ON}^{*} + (C_{ON_{IN}} - C_{ON}^{*})\left[1+\frac{k_{ON}A}{NQ}\right]^{-N}$$

Ammonia Nitrogen (AN)

$$C_{AN_{OUT}} = C_{AN}^{*} + (C_{AN_{IN}} - C_{AN}^{*})\left[1+\frac{k_{AN}A}{NQ}\right]^{-N} + \left(\frac{k_{ON}}{k_{AN}-k_{ON}}\right)(C_{ON_{IN}} - C_{ON}^{*})\left(\left[1+\frac{k_{ON}A}{NQ}\right]^{-N} - \left[1+\frac{k_{AN}A}{NQ}\right]^{-N}\right)$$

Nitrate Nitrogen (NN)

$$C_{NN_{OUT}} = C_{NN_{IN}}\left[1+\frac{k_{NN}A}{NQ}\right]^{-N} + \Psi \left[\begin{array}{l} \left(\frac{k_{AN}}{k_{NN}-k_{AN}}\right)C_{AN_{IN}}\left(\left[1+\frac{k_{AN}A}{NQ}\right]^{-N} - \left[1+\frac{k_{NN}A}{NQ}\right]^{-N}\right) \\ + \left(\frac{k_{ON}}{k_{AN}-k_{ON}}\right)\left(\frac{k_{AN}}{k_{NN}-k_{ON}}\right)(C_{ON_{IN}} - C_{ON}^{*})\left(\left[1+\frac{k_{ON}A}{NQ}\right]^{-N} - \left[1+\frac{k_{NN}A}{NQ}\right]^{-N}\right) \\ - \left(\frac{k_{ON}}{k_{AN}-k_{ON}}\right)\left(\frac{k_{AN}}{k_{NN}-k_{AN}}\right)(C_{ON_{IN}} - C_{ON}^{*})\left(\left[1+\frac{k_{AN}A}{NQ}\right]^{-N} - \left[1+\frac{k_{NN}A}{NQ}\right]^{-N}\right) \end{array}\right]$$

where Ψ = fraction of ammonium nitrified, assumed to be 100% = 100%

Scenarios: 1B and 6B

Surface Flow Monthly PkC* Treatment Wetland Design Model w/ Water Balance (Wastewater Parameters)

User inputs indicated by white boxes.
Pop-up notes indicated by red triangles.

Project Name	ath Wetlands - 2012 nitrogen runs- 100% Klamath River at Link River
Project Number	WQ-KR253:ppt,tempKlamathFallsAirp

Flow Rate Converter

	Value	Units
Flow (Enter monthly flowrate below under "General and Hydrologic Input Data"; use this cell simply for converstion between units)		cfs
Converted Flow		m^3/d

Add Results to NADB Charts

General and Hydrologic Input Data

Wetland Hydrology

Month	Days in Month	Air Temp (°C)	Precip (m/mo)	Inflow (m3/d)	ET (m/mo)	Infiltration (m/mo)	Outflow (m3/d)
January	31	-0.7	0.04	3026414	0.06	0.37	2776927
February	28	1.0	0.03	2725485	0.02	0.37	2465266
March	31	3.6	0.02	3182995	0.01	0.37	2944537
April	30	5.9	0.03	3850910	0.02	0.37	3607265
May	31	10.5	0.03	4056422	0.03	0.37	3814058
June	30	14.6	0.01	3664970	0.06	0.37	3384381
July	31	19.4	0.00	2495507	0.08	0.37	2202555
August	31	18.4	0.01	2302228	0.12	0.37	1988745
September	30	14.2	0.01	1661225	0.15	0.37	1314555
October	31	8.3	0.02	1910776	0.18	0.37	1564084
November	30	2.8	0.05	1874077	0.15	0.37	1556201
December	31	-1.0	0.04	1952367	0.11	0.37	1670104
Average		8.1	0.025	2725281	0.08	0.37	2440723

Water Quality Input Data

Month	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
January								
February								
March								
April								
May			0.74	0.11	0.06		0.16	
June			1.39	0.09	0.06		0.19	
July			2.07	0.27	0.10		0.28	
August			3.57	0.33	0.11		0.18	
September			2.57	0.14	0.05		0.15	
October			2.05	0.20	0.12		0.15	
November								
December								
Annual Average			2.07	0.19	0.08		0.18	

		BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Target Effluent Conc., mg/L	C_e =								
Desired Confidence Percentile		0.5		0.5	0.5	0.5	0.5	0.5	0.5
Max Month/Annual Factor		1.7	1.9	1.8	2.5	2.5	1.6	1.8	3.0
Design Target Conc., mg/L	C_d =								
Wetland Background Limit, mg/L	C^* =	Influent Dependent	Influent Dependent	0.5	0	0	0.5	0.02	300
P-Factor	P =	1.0	3.0	3.0	3.0	3.0	3.0	4.0	3.0
Confidence-based Rate Constant, 20°C, m/y	k_{20} =	0	200	17	18	27	13	10	75
Temperature Factor	θ =	1.00	1.065	1.05	1.05	1.11	1.06	1.00	1.00

Wetland Size and Output Predictions

Wetland Area

5,000	ac
2024.29	ha
20242915	m2

Effluent Concentrations

Month	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
January								
February								
March								
April								
May			0.7	0.12	0.06	0.89	0.14	
June			1.2	0.21	0.06	1.51	0.16	
July			1.6	0.56	0.12	2.30	0.23	
August			2.7	0.97	0.18	3.81	0.15	
September			1.9	0.64	0.13	2.66	0.12	
October			1.7	0.48	0.15	2.34	0.12	
November								
December								
Annual Average			*1.64*	*0.50*	*0.12*	*2.25*	*0.15*	

	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Avg Percent Reduction (by concentration)			21%	-161%	-40%	4%	17%	

Scenarios: 1B and 6B

Surface Flow Monthly PkC* Treatment Wetland Design Model w/ Water Balance (Wastewater Parameters)

User inputs indicated by white boxes.
Pop-up notes indicated by red triangles.

Project Name: ath Wetlands - 2012 nitrogen runs- 100% Klamath River at Link River
Project Number: WQ-KR253:ppt,tempKlamathFallsAirp

Average Mass Loading (lb/day)				12405.6	1145.3	502.8	14053.7	1106	
Monthly Mass Loading (kg/ha/d)	Jan								
	Feb								
	Mar								
	Apr								
	May			1.5	0.2	0.1	1.8	0.3	
	Jun			2.5	0.2	0.1	2.8	0.3	
	Jul			2.6	0.3	0.1	3.0	0.3	
	Aug			4.1	0.4	0.1	4.6	0.2	
	Sep			2.1	0.1	0.0	2.3	0.1	
	Oct			1.9	0.2	0.1	2.2	0.1	
	Nov								
	Dec								
Average Mass Loading (kg/ha/d)				2.8	0.3	0.1	3.2	0.2	
Average Mass Out (lb/day)				8802	2673	630	12105	821	
Monthly Mass Out (kg/ha/d)	Jan								
	Feb								
	Mar								
	Apr								
	May			1.3	0.2	0.1	1.7	0.3	
	Jun			2.1	0.4	0.1	2.5	0.3	
	Jul			1.8	0.6	0.1	2.5	0.3	
	Aug			2.6	0.9	0.2	3.7	0.1	
	Sep			1.2	0.4	0.1	1.7	0.1	
	Oct			1.3	0.4	0.1	1.8	0.1	
	Nov								
	Dec								
Average Mass Out (kg/ha/d)				2.0	0.6	0.1	2.7	0.2	
Percent Reduction (by mass)				29%	-133%	-25%	14%	26%	

Hydraulic Properties Based on Area and Flow

Percent Open Water	10%	Transition Side Slopes (H:V) - (10:1)	10			Override
Marsh Zone Depth (m)	0.46	Berm Side Slopes (H:V) - (4:1)	4	# of Deep Zones	20	
Deep Zone Depth (m)	1.4	Approx Aspect Ratio (L:W) - (5:1)	5	Deep Zone Bot Length	50	
Volume (m3)	9074303					

Hydraulic Loading Rate, q	HLR =	13.5	cm/d
Nominal Hydraulic Residence Time, days	HRT =	3.3	days

Nitrogen Species Calculations

Nitrogen Models
per K&K Eqns 13-28. 13-29, 13-39:
Adapted for Monthly TIS model
Organic Nitrogen (ON)

$$C_{ON_{OUT}} = C_{ON}^{*} + (C_{ON_{IN}} - C_{ON}^{*})\left[1+\frac{k_{ON}A}{NQ}\right]^{-N}$$

Ammonia Nitrogen (AN)

$$C_{AN_{OUT}} = C_{AN}^{*} + (C_{AN_{IN}} - C_{AN}^{*})\left[1+\frac{k_{AN}A}{NQ}\right]^{-N} + \left(\frac{k_{ON}}{k_{AN}-k_{ON}}\right)(C_{ON_{IN}} - C_{ON}^{*})\left(\left[1+\frac{k_{ON}A}{NQ}\right]^{-N} - \left[1+\frac{k_{AN}A}{NQ}\right]^{-N}\right)$$

Nitrate Nitrogen (NN)

$$C_{NN_{OUT}} = C_{NN_{IN}}\left[1+\frac{k_{NN}A}{NQ}\right]^{-N} + \Psi\left[\begin{array}{l} \left(\frac{k_{AN}}{k_{NN}-k_{AN}}\right)C_{AN_{IN}}\left(\left[1+\frac{k_{AN}A}{NQ}\right]^{-N} - \left[1+\frac{k_{NN}A}{NQ}\right]^{-N}\right) \\ +\left(\frac{k_{ON}}{k_{AN}-k_{ON}}\right)\left(\frac{k_{AN}}{k_{NN}-k_{ON}}\right)(C_{ON_{IN}} - C_{ON}^{*})\left(\left[1+\frac{k_{ON}A}{NQ}\right]^{-N} - \left[1+\frac{k_{NN}A}{NQ}\right]^{-N}\right) \\ -\left(\frac{k_{ON}}{k_{AN}-k_{ON}}\right)\left(\frac{k_{AN}}{k_{NN}-k_{AN}}\right)(C_{ON_{IN}} - C_{ON}^{*})\left(\left[1+\frac{k_{AN}A}{NQ}\right]^{-N} - \left[1+\frac{k_{NN}A}{NQ}\right]^{-N}\right) \end{array}\right]$$

where Ψ = fraction of ammonium nitrified, assumed to be 100% = 100%

Scenarios: 1B and 5B

Surface Flow Monthly PkC* Treatment Wetland Design Model w/ Water Balance (Wastewater Parameters)

User inputs indicated by white boxes.
Pop-up notes indicated by red triangles.

Project Name: ath Wetlands - 2012 nitrogen runs- 100% Klamath River at Link River
Project Number: WQ-KR253:ppt,tempKlamathFallsAirp

Flow Rate Converter

	Value	Units
Flow (Enter monthly flowrate below under "General and Hydrologic Input Data"; use this cell simply for converstion between units)		cfs
Converted Flow		m^3/d

Add Results to NADB Charts

General and Hydrologic Input Data

Wetland Hydrology

Month	Days in Month	Air Temp (°C)	Precip (m/mo)	Inflow (m3/d)	ET (m/mo)	Infiltration (m/mo)	Outflow (m3/d)
January	31	-0.7	0.04	3026414	0.06	0.37	0
February	28	1.0	0.03	2725485	0.02	0.37	0
March	31	3.6	0.02	3182995	0.01	0.37	0
April	30	5.9	0.03	3850910	0.02	0.37	0
May	31	10.5	0.03	4056422	0.03	0.37	178589
June	30	14.6	0.01	3664970	0.06	0.37	0
July	31	19.4	0.00	2495507	0.08	0.37	0
August	31	18.4	0.01	2302228	0.12	0.37	0
September	30	14.2	0.01	1661225	0.15	0.37	0
October	31	8.3	0.02	1910776	0.18	0.37	0
November	30	2.8	0.05	1874077	0.15	0.37	0
December	31	-1.0	0.04	1952367	0.11	0.37	0
Average		8.1	0.025	2725281	0.08	0.37	14882

Water Quality Input Data

Month	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
January								
February								
March								
April								
May			0.74	0.11	0.06		0.16	
June			1.39	0.09	0.06		0.19	
July			2.07	0.27	0.10		0.28	
August			3.57	0.33	0.11		0.18	
September			2.57	0.14	0.05		0.15	
October			2.05	0.20	0.12		0.15	
November								
December								
Annual Average			2.07	0.19	0.08		0.18	

Parameter	Symbol	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Target Effluent Conc., mg/L	C_e =								
Desired Confidence Percentile		0.5		0.5	0.5	0.5	0.5	0.5	0.5
Max Month/Annual Factor		1.7	1.9	1.8	2.5	2.5	1.6	1.8	3.0
Design Target Conc., mg/L	C_d =								
Wetland Background Limit, mg/L	C^* =	Influent Dependent	Influent Dependent	1	0	0	1	0.02	300
P-Factor	P =	1.0	3.0	3.0	3.0	3.0	3.0	4.0	3.0
Confidence-based Rate Constant, 20°C, m/y	k_{20} =	0	200	17	18	27	13	10	75
Temperature Factor	θ =	1.00	1.065	1.05	1.05	1.11	1.06	1.00	1.00

Wetland Size and Output Predictions

Wetland Area

Value	Unit
80,000	ac
32388.66	ha
323886640	m2

Effluent Concentrations

Month	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
January								
February								
March								
April								
May			1.0	0.00	0.01	0.98	0.0284	
June			1.0	0.04	0.03	1.09	0.0230	
July			1.0	0.00	0.02	1.17	0.0173	
August			1.0	0.00	0.00	1.86	0.0292	
September			1.1	0.19	0.42	1.71	-0.0242	
October			1.1	0.00	2.44	3.52	-0.0623	
November								
December								
Annual Average			*1.03*	*0.04*	*0.49*	*1.72*	*0.0019*	

	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Avg Percent Reduction (by concentration)			50%	80%	-480%	26%	99%	

Scenarios: 1B and 5B

Surface Flow Monthly PkC* Treatment Wetland Design Model w/ Water Balance (Wastewater Parameters)

User inputs indicated by white boxes.
Pop-up notes indicated by red triangles.

Project Name: ath Wetlands - 2012 nitrogen runs- 100% Klamath River at Link River
Project Number: WQ-KR253:ppt,tempKlamathFallsAirp

Average Mass Loading (lb/day)				12405.6	1145.3	502.8	14053.7	1106	
Monthly Mass Loading (kg/ha/d)	Jan								
	Feb								
	Mar								
	Apr								
	May			0.1	0.0	0.0	0.1	0.0	
	Jun			0.2	0.0	0.0	0.2	0.0	
	Jul			0.2	0.0	0.0	0.2	0.0	
	Aug			0.3	0.0	0.0	0.3	0.0	
	Sep			0.1	0.0	0.0	0.1	0.0	
	Oct			0.1	0.0	0.0	0.1	0.0	
	Nov								
	Dec								
Average Mass Loading (kg/ha/d)				0.2	0.0	0.0	0.2	0.0	
Average Mass Out (lb/day)				34	1	16	56	0	
Monthly Mass Out (kg/ha/d)	Jan								
	Feb								
	Mar								
	Apr								
	May			0.0	0.0	0.0	0.0	0.0	
	Jun			0.0	0.0	0.0	0.0	0.0	
	Jul			0.0	0.0	0.0	0.0	0.0	
	Aug			0.0	0.0	0.0	0.0	0.0	
	Sep			0.0	0.0	0.0	0.0	0.0	
	Oct			0.0	0.0	0.0	0.0	0.0	
	Nov								
	Dec								
Average Mass Out (kg/ha/d)				0.0	0.0	0.0	0.0	0.0	
Percent Reduction (by mass)				100%	100%	97%	100%	100%	

Hydraulic Properties Based on Area and Flow

Percent Open Water	10%	Transition Side Slopes (H:V) - (10:1)	10			Override
Marsh Zone Depth (m)	0.46	Berm Side Slopes (H:V) - (4:1)	4	# of Deep Zones	80	
Deep Zone Depth (m)	1.4	Approx Aspect Ratio (L:W) - (5:1)	5	Deep Zone Bot Length	50	
Volume (m3)	144054832					

Hydraulic Loading Rate, q	HLR =	0.8	cm/d
Nominal Hydraulic Residence Time, days	HRT =	52.9	days

Nitrogen Species Calculations

Nitrogen Models
per K&K Eqns 13-28. 13-29, 13-39:
Adapted for Monthly TIS model
Organic Nitrogen (ON)

$$C_{ON_{OUT}} = C^*_{ON} + (C_{ON_{IN}} - C^*_{ON})\left[1+\frac{k_{ON}A}{NQ}\right]^{-N}$$

Ammonia Nitrogen (AN)

$$C_{AN_{OUT}} = C^*_{AN} + (C_{AN_{IN}} - C^*_{AN})\left[1+\frac{k_{AN}A}{NQ}\right]^{-N} + \left(\frac{k_{ON}}{k_{AN}-k_{ON}}\right)(C_{ON_{IN}} - C^*_{ON})\left(\left[1+\frac{k_{ON}A}{NQ}\right]^{-N} - \left[1+\frac{k_{AN}A}{NQ}\right]^{-N}\right)$$

Nitrate Nitrogen (NN)

$$C_{NN_{OUT}} = C_{NN_{IN}}\left[1+\frac{k_{NN}A}{NQ}\right]^{-N} + \Psi\left[\begin{array}{l} \left(\frac{k_{AN}}{k_{NN}-k_{AN}}\right)C_{AN_{IN}}\left(\left[1+\frac{k_{AN}A}{NQ}\right]^{-N} - \left[1+\frac{k_{NN}A}{NQ}\right]^{-N}\right) \\ + \left(\frac{k_{ON}}{k_{AN}-k_{ON}}\right)\left(\frac{k_{AN}}{k_{NN}-k_{ON}}\right)(C_{ON_{IN}} - C^*_{ON})\left(\left[1+\frac{k_{ON}A}{NQ}\right]^{-N} - \left[1+\frac{k_{NN}A}{NQ}\right]^{-N}\right) \\ - \left(\frac{k_{ON}}{k_{AN}-k_{ON}}\right)\left(\frac{k_{AN}}{k_{NN}-k_{AN}}\right)(C_{ON_{IN}} - C^*_{ON})\left(\left[1+\frac{k_{AN}A}{NQ}\right]^{-N} - \left[1+\frac{k_{NN}A}{NQ}\right]^{-N}\right) \end{array}\right]$$

where Ψ = fraction of ammonium nitrified, assumed to be 100% = 100%

Scenarios: 2 and 6A

Surface Flow Monthly PkC* Treatment Wetland Design Model w/ Water Balance (Wastewater Parameters)

User inputs indicated by white boxes.
Pop-up notes indicated by red triangles.

Project Name: ath Wetlands - 2012 nitrogen runs- 50% Klamath River at Link River
Project Number: WQ-KR253:ppt,tempKlamathFallsAirp

Flow Rate Converter

	Value	Units
Flow (Enter monthly flowrate below under "General and Hydrologic Input Data"; use this cell simply for converstion between units)		cfs
Converted Flow		m^3/d

Add Results to NADB Charts

General and Hydrologic Input Data

Wetland Hydrology

Month	Days in Month	Air Temp (°C)	Precip (m/mo)	Inflow (m3/d)	ET (m/mo)	Infiltration (m/mo)	Outflow (m3/d)	cfs	MGD
January	31	-0.7	0.04	1513207	0.06	0.37	515257	210.60	136.12
February	28	1.0	0.03	1362743	0.02	0.37	321866	131.56	85.03
March	31	3.6	0.02	1591497	0.01	0.37	637667	260.64	168.45
April	30	5.9	0.03	1925455	0.02	0.37	950875	388.66	251.20
May	31	10.5	0.03	2028211	0.03	0.37	1058753	432.75	279.69
June	30	14.6	0.01	1832485	0.06	0.37	710129	290.25	187.60
July	31	19.4	0.00	1247754	0.08	0.37	75944	31.04	20.06
August	31	18.4	0.01	1151114	0.12	0.37	0	0.00	0.00
September	30	14.2	0.01	830612	0.15	0.37	0	0.00	0.00
October	31	8.3	0.02	955388	0.18	0.37	0	0.00	0.00
November	30	2.8	0.05	937038	0.15	0.37	0	0.00	0.00
December	31	-1.0	0.04	976184	0.11	0.37	0	0.00	0.00
Average		8.1	0.025	1362641	0.08	0.37	355874		

Water Quality Input Data

Month	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
January								
February								
March								
April								
May			0.74	0.11	0.06		0.16	
June			1.39	0.09	0.06		0.19	
July			2.07	0.27	0.10		0.28	
August			3.57	0.33	0.11		0.18	
September			2.57	0.14	0.05		0.15	
October			2.05	0.20	0.12		0.15	
November								
December								
Annual Average			2.07	0.19	0.08		0.18	

		BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Target Effluent Conc., mg/L	C_e =							0.16581	
Desired Confidence Percentile		0.5		0.5	0.5	0.5	0.5	0.5	0.5
Max Month/Annual Factor		1.7	1.9	1.8	2.5	2.5	1.6	1.8	3.0
Design Target Conc., mg/L	C_d =							0.09	
Wetland Background Limit, mg/L	C^* =	Influent Dependent	Influent Dependent	0.5	0	0	0.5	0.002	300
P-Factor	P =	1.0	3.0	3.0	3.0	3.0	3.0	4.0	3.0
Confidence-based Rate Constant, 20°C, m/y	k_{20} =	0	200	17	18	27	13	10	75
Temperature Factor	θ =	1.00	1.065	1.05	1.05	1.11	1.06	1.00	1.00

Wetland Size and Output Predictions

Wetland Area

20,000	ac
8097.17	ha
80971660	m2

Effluent Concentrations

Month	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
January								
February								
March								
April								
May			0.6	0.10	0.06	0.73	0.05018	
June			0.7	0.24	0.12	1.05	0.04892	
July			0.6	0.19	0.11	0.92	0.03174	
August			0.7	0.28	0.18	1.12	0.01354	
September			0.5	0.00	0.00	0.50	-0.00424	
October			0.6	0.09	0.12	0.77	-0.00148	
November								
December								
Annual Average			*0.60*	*0.15*	*0.10*	*0.85*	0.02311	

	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Avg Percent Reduction (by concentration)			71%	21%	-18%	64%	87%	

Scenarios: 2 and 6A

Surface Flow Monthly PkC* Treatment Wetland Design Model w/ Water Balance (Wastewater Parameters)

User inputs indicated by white boxes.
Pop-up notes indicated by red triangles.

Project Name: ath Wetlands - 2012 nitrogen runs- 50% Klamath River at Link River
Project Number: WQ-KR253:ppt,tempKlamathFallsAirp

Average Mass Loading (lb/day)				6202.8	572.7	251.4	7026.8	553	
Monthly Mass Loading (kg/ha/d)	Jan								
	Feb								
	Mar								
	Apr								
	May			0.2	0.0	0.0	0.2	0.0	
	Jun			0.3	0.0	0.0	0.3	0.0	
	Jul			0.3	0.0	0.0	0.4	0.0	
	Aug			0.5	0.0	0.0	0.6	0.0	
	Sep			0.3	0.0	0.0	0.3	0.0	
	Oct			0.2	0.0	0.0	0.3	0.0	
	Nov								
	Dec								
Average Mass Loading (kg/ha/d)				0.3	0.0	0.0	0.4	0.0	
Average Mass Out (lb/day)				470	118	77	665	18	
Monthly Mass Out (kg/ha/d)	Jan								
	Feb								
	Mar								
	Apr								
	May			0.1	0.0	0.0	0.1	0.0	
	Jun			0.1	0.0	0.0	0.1	0.0	
	Jul			0.0	0.0	0.0	0.0	0.0	
	Aug			0.0	0.0	0.0	0.0	0.0	
	Sep			0.0	0.0	0.0	0.0	0.0	
	Oct			0.0	0.0	0.0	0.0	0.0	
	Nov								
	Dec								
Average Mass Out (kg/ha/d)				0.0	0.0	0.0	0.0	0.0	
Percent Reduction (by mass)				92%	79%	69%	91%	97%	

Hydraulic Properties Based on Area and Flow

Percent Open Water	10%	Transition Side Slopes (H:V) - (10:1)	10			Override
Marsh Zone Depth (m)	0.46	Berm Side Slopes (H:V) - (4:1)	4	# of Deep Zones	40	
Deep Zone Depth (m)	1.4	Approx Aspect Ratio (L:W) - (5:1)	5	Deep Zone Bot Length	50	
Volume (m3)	36108397					

Hydraulic Loading Rate, q	HLR =	1.7	cm/d
Nominal Hydraulic Residence Time, days	HRT =	26.5	days

Nitrogen Species Calculations

Nitrogen Models
per K&K Eqns 13-28. 13-29, 13-39:
Adapted for Monthly TIS model
Organic Nitrogen (ON)

$$C_{ON_{OUT}} = C_{ON}^{*} + (C_{ON_{IN}} - C_{ON}^{*})\left[1+\frac{k_{ON}A}{NQ}\right]^{-N}$$

Ammonia Nitrogen (AN)

$$C_{AN_{OUT}} = C_{AN}^{*} + (C_{AN_{IN}} - C_{AN}^{*})\left[1+\frac{k_{AN}A}{NQ}\right]^{-N} + \left(\frac{k_{ON}}{k_{AN}-k_{ON}}\right)(C_{ON_{IN}} - C_{ON}^{*})\left(\left[1+\frac{k_{ON}A}{NQ}\right]^{-N} - \left[1+\frac{k_{AN}A}{NQ}\right]^{-N}\right)$$

Nitrate Nitrogen (NN)

$$C_{NN_{OUT}} = C_{NN_{IN}}\left[1+\frac{k_{NN}A}{NQ}\right]^{-N} + \Psi\left[\begin{array}{l} \left(\frac{k_{AN}}{k_{NN}-k_{AN}}\right)C_{AN_{IN}}\left(\left[1+\frac{k_{AN}A}{NQ}\right]^{-N} - \left[1+\frac{k_{NN}A}{NQ}\right]^{-N}\right) \\ +\left(\frac{k_{ON}}{k_{AN}-k_{ON}}\right)\left(\frac{k_{AN}}{k_{NN}-k_{ON}}\right)(C_{ON_{IN}} - C_{ON}^{*})\left(\left[1+\frac{k_{ON}A}{NQ}\right]^{-N} - \left[1+\frac{k_{NN}A}{NQ}\right]^{-N}\right) \\ -\left(\frac{k_{ON}}{k_{AN}-k_{ON}}\right)\left(\frac{k_{AN}}{k_{NN}-k_{AN}}\right)(C_{ON_{IN}} - C_{ON}^{*})\left(\left[1+\frac{k_{AN}A}{NQ}\right]^{-N} - \left[1+\frac{k_{NN}A}{NQ}\right]^{-N}\right) \end{array}\right]$$

where Ψ = fraction of ammonium nitrified, assumed to be 100% = 100%

Scenarios: 3 and 6B

Surface Flow Monthly PkC* Treatment Wetland Design Model w/ Water Balance (Wastewater Parameters)

User inputs indicated by white boxes.
Pop-up notes indicated by red triangles.

Project Name: ath Wetlands - 2012 nitrogen runs- 100% Klamath River at Link River
Project Number: WQ-KR253:ppt,tempKlamathFallsAirp

Flow Rate Converter

	Value	Units
Flow (Enter monthly flowrate below under "General and Hydrologic Input Data"; use this cell simply for converstion between units)	1	cfs
Converted Flow	2,446.6	m^3/d

Add Results to NADB Charts

General and Hydrologic Input Data

Wetland Hydrology

Month	Days in Month	Air Temp (°C)	Precip (m/mo)	Inflow (m3/d)	ET (m/mo)	Infiltration (m/mo)	Outflow (m3/d)	cfs	MGD
January	31	-0.7	0.04	3026414	0.06	0.37	0	0.00	0.00
February	28	1.0	0.03	2725485	0.02	0.37	0	0.00	0.00
March	31	3.6	0.02	3182995	0.01	0.37	0	0.00	0.00
April	30	5.9	0.03	3850910	0.02	0.37	0	0.00	0.00
May	31	10.5	0.03	4056422	0.03	0.37	178589	73.00	47.18
June	30	14.6	0.01	3664970	0.06	0.37	0	0.00	0.00
July	31	19.4	0.00	2495507	0.08	0.37	0	0.00	0.00
August	31	18.4	0.01	2302228	0.12	0.37	0	0.00	0.00
September	30	14.2	0.01	1661225	0.15	0.37	0	0.00	0.00
October	31	8.3	0.02	1910776	0.18	0.37	0	0.00	0.00
November	30	2.8	0.05	1874077	0.15	0.37	0	0.00	0.00
December	31	-1.0	0.04	1952367	0.11	0.37	0	0.00	0.00
Average		8.1	0.025	2725281	0.08	0.37	14882		

Water Quality Input Data

Month	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
January								
February								
March								
April								
May			0.74	0.11	0.06		0.16	
June			1.39	0.09	0.06		0.19	
July			2.07	0.27	0.10		0.28	
August			3.57	0.33	0.11		0.18	
September			2.57	0.14	0.05		0.15	
October			2.05	0.20	0.12		0.15	
November								
December								
Annual Average			2.07	0.19	0.08		0.18	

		BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Target Effluent Conc., mg/L	C_e =								
Desired Confidence Percentile		0.5		0.5	0.5	0.5	0.5	0.5	0.5
Max Month/Annual Factor		1.7	1.9	1.8	2.5	2.5	1.6	1.8	3.0
Design Target Conc., mg/L	C_d =								
Wetland Background Limit, mg/L	C* =	Influent Dependent	Influent Dependent	0.5	0	0	0.5	0.002	300
P-Factor	P =	1.0	3.0	3.0	3.0	3.0	3.0	4.0	3.0
Confidence-based Rate Constant, 20°C, m/y	k_{20} =	0	200	17	18	27	13	10	75
Temperature Factor	θ =	1.00	1.065	1.05	1.05	1.11	1.06	1.00	1.00

Wetland Size and Output Predictions

Wetland Area

80,000	ac
32388.66	ha
323886640	m2

Effluent Concentrations

Month	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
January								
February								
March								
April								
May			0.5	0.03	0.03	0.58	0.012	
June			0.5	0.05	0.04	0.63	0.00	
July			0.5	0.00	0.00	0.70	0.00	
August			0.5	0.00	0.00	1.48	0.01	
September			0.6	0.20	0.39	1.17	-0.05	
October			0.5	0.00	0.39	0.90	-0.10	
November								
December								
Annual Average			0.52	0.05	0.14	0.91		

	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Avg Percent Reduction (by concentration)			75%	75%	-69%	61%	#VALUE!	

Scenarios: 3 and 6B

Surface Flow Monthly PkC* Treatment Wetland Design Model w/ Water Balance (Wastewater Parameters)

User inputs indicated by white boxes.
Pop-up notes indicated by red triangles.

Project Name: ath Wetlands - 2012 nitrogen runs- 100% Klamath River at Link River
Project Number: WQ-KR253:ppt,tempKlamathFallsAirp

Average Mass Loading (lb/day)				12405.6	1145.3	502.8	14053.7	1106	
Monthly Mass Loading (kg/ha/d)	Jan								
	Feb								
	Mar								
	Apr								
	May			0.1	0.0	0.0	0.1	0.0	
	Jun			0.2	0.0	0.0	0.2	0.0	
	Jul			0.2	0.0	0.0	0.2	0.0	
	Aug			0.3	0.0	0.0	0.3	0.0	
	Sep			0.1	0.0	0.0	0.1	0.0	
	Oct			0.1	0.0	0.0	0.1	0.0	
	Nov								
	Dec								
Average Mass Loading (kg/ha/d)				0.2	0.0	0.0	0.2	0.0	
Average Mass Out (lb/day)				17	2	5	30		
Monthly Mass Out (kg/ha/d)	Jan								
	Feb								
	Mar								
	Apr								
	May			0.0	0.0	0.0	0.0	0.0	
	Jun			0.0	0.0	0.0	0.0	0.0	
	Jul			0.0	0.0	0.0	0.0	0.0	
	Aug			0.0	0.0	0.0	0.0	0.0	
	Sep			0.0	0.0	0.0	0.0	0.0	
	Oct			0.0	0.0	0.0	0.0	0.0	
	Nov								
	Dec								
Average Mass Out (kg/ha/d)				0.0	0.0	0.0	0.0		
Percent Reduction (by mass)				100%	100%	99%	100%	#VALUE!	

Hydraulic Properties Based on Area and Flow

Percent Open Water	10%	Transition Side Slopes (H:V) - (10:1)	10			Override
Marsh Zone Depth (m)	0.46	Berm Side Slopes (H:V) - (4:1)	4	# of Deep Zones	80	
Deep Zone Depth (m)	1.4	Approx Aspect Ratio (L:W) - (5:1)	5	Deep Zone Bot Length	50	
Volume (m3)	144054832					

Hydraulic Loading Rate, q	HLR =	0.8	cm/d
Nominal Hydraulic Residence Time, days	HRT =	52.9	days

Nitrogen Species Calculations

Nitrogen Models
per K&K Eqns 13-28. 13-29, 13-39:
Adapted for Monthly TIS model
Organic Nitrogen (ON)

$$C_{ON_{OUT}} = C_{ON}^{*} + (C_{ON_{IN}} - C_{ON}^{*})\left[1+\frac{k_{ON}A}{NQ}\right]^{-N}$$

Ammonia Nitrogen (AN)

$$C_{AN_{OUT}} = C_{AN}^{*} + (C_{AN_{IN}} - C_{AN}^{*})\left[1+\frac{k_{AN}A}{NQ}\right]^{-N} + \left(\frac{k_{ON}}{k_{AN}-k_{ON}}\right)(C_{ON_{IN}} - C_{ON}^{*})\left(\left[1+\frac{k_{ON}A}{NQ}\right]^{-N} - \left[1+\frac{k_{AN}A}{NQ}\right]^{-N}\right)$$

Nitrate Nitrogen (NN)

$$C_{NN_{OUT}} = C_{NN_{IN}}\left[1+\frac{k_{NN}A}{NQ}\right]^{-N} + \Psi\left[\begin{array}{l} \left(\frac{k_{AN}}{k_{NN}-k_{AN}}\right)C_{AN_{IN}}\left(\left[1+\frac{k_{AN}A}{NQ}\right]^{-N} - \left[1+\frac{k_{NN}A}{NQ}\right]^{-N}\right) \\ + \left(\frac{k_{ON}}{k_{AN}-k_{ON}}\right)\left(\frac{k_{AN}}{k_{NN}-k_{ON}}\right)(C_{ON_{IN}} - C_{ON}^{*})\left(\left[1+\frac{k_{ON}A}{NQ}\right]^{-N} - \left[1+\frac{k_{NN}A}{NQ}\right]^{-N}\right) \\ - \left(\frac{k_{ON}}{k_{AN}-k_{ON}}\right)\left(\frac{k_{AN}}{k_{NN}-k_{AN}}\right)(C_{ON_{IN}} - C_{ON}^{*})\left(\left[1+\frac{k_{AN}A}{NQ}\right]^{-N} - \left[1+\frac{k_{NN}A}{NQ}\right]^{-N}\right) \end{array}\right]$$

where Ψ = fraction of ammonium nitrified, assumed to be 100% = 100%

Scenarios: 2 and 5A

Surface Flow Monthly PkC* Treatment Wetland Design Model w/ Water Balance (Wastewater Parameters)

User inputs indicated by white boxes.
Pop-up notes indicated by red triangles.

Project Name: ath Wetlands - 2012 nitrogen runs- 50% Klamath River at Link River
Project Number: WQ-KR253:ppt,tempKlamathFallsAirp

Flow Rate Converter

	Value	Units
Flow (Enter monthly flowrate below under "General and Hydrologic Input Data"; use this cell simply for converstion between units)		cfs
Converted Flow		m^3/d

Add Results to NADB Charts

General and Hydrologic Input Data

Wetland Hydrology

Month	Days in Month	Air Temp (°C)	Precip (m/mo)	Inflow (m3/d)	ET (m/mo)	Infiltration (m/mo)	Outflow (m3/d)
January	31	-0.7	0.04	1513207	0.06	0.37	0
February	28	1.0	0.03	1362743	0.02	0.37	0
March	31	3.6	0.02	1591497	0.01	0.37	0
April	30	5.9	0.03	1925455	0.02	0.37	0
May	31	10.5	0.03	2028211	0.03	0.37	0
June	30	14.6	0.01	1832485	0.06	0.37	0
July	31	19.4	0.00	1247754	0.08	0.37	0
August	31	18.4	0.01	1151114	0.12	0.37	0
September	30	14.2	0.01	830612	0.15	0.37	0
October	31	8.3	0.02	955388	0.18	0.37	0
November	30	2.8	0.05	937038	0.15	0.37	0
December	31	-1.0	0.04	976184	0.11	0.37	0
Average		8.1	0.025	1362641	0.08	0.37	

Water Quality Input Data

Month	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
January								
February								
March								
April								
May			0.74	0.11	0.06		0.16	
June			1.39	0.09	0.06		0.19	
July			2.07	0.27	0.10		0.28	
August			3.57	0.33	0.11		0.18	
September			2.57	0.14	0.05		0.15	
October			2.05	0.20	0.12		0.15	
November								
December								
Annual Average			2.07	0.19	0.08		0.18	

		BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Target Effluent Conc., mg/L	C_e =								
Desired Confidence Percentile		0.5		0.5	0.5	0.5	0.5	0.5	0.5
Max Month/Annual Factor		1.7	1.9	1.8	2.5	2.5	1.6	1.8	3.0
Design Target Conc., mg/L	C_d =								
Wetland Background Limit, mg/L	C^* =	Influent Dependent	Influent Dependent	1	0	0	1	0.002	300
P-Factor	P =	1.0	3.0	3.0	3.0	3.0	3.0	4.0	3.0
Confidence-based Rate Constant, 20°C, m/y	k_{20} =	0	200	17	18	27	13	10	75
Temperature Factor	θ =	1.00	1.065	1.05	1.05	1.11	1.06	1.00	1.00

Wetland Size and Output Predictions

Wetland Area

80,000	ac
32388.66	ha
323886640	m2

Effluent Concentrations

Month	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
January								
February								
March								
April								
May			1.0	0.00	0.01	1.00	0.0014	
June			1.0	0.00	0.02	1.28	0.0128	
July			1.0	0.06	0.06	1.28	-0.05	
August			1.1	0.29	0.33	3.03	0.03	
September			1.5	2.52	7.80	11.83	0.66	
October			-0.8	106.93	-0.12	106.00	0.36	
November								
December								
Annual Average			*0.81*	*18.30*	*1.35*	*20.73*	*0.17*	

	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Avg Percent Reduction (by concentration)			61%	-9496%	-1512%	-786%	7%	

8/16/2012 3:29 PM C:\Documents and Settings\kcarlso2\My Documents\Mail\OL Temp Attachments\CH2MHILLTrtWetlandToolkit_v2011e_2012KlamathLinkRiver_11 (3).xlsm Avg_N-C1_P-C.002_50%Q_253Link

Scenarios: 2 and 5A

Surface Flow Monthly PkC* Treatment Wetland Design Model w/ Water Balance (Wastewater Parameters)

User inputs indicated by white boxes.
Pop-up notes indicated by red triangles.

Project Name: hath Wetlands - 2012 nitrogen runs- 50% Klamath River at Link River
Project Number: WQ-KR253:ppt,tempKlamathFallsAirp

Average Mass Loading (lb/day)				6202.8	572.7	251.4	7026.8	553	
Monthly Mass Loading (kg/ha/d)	Jan								
	Feb								
	Mar								
	Apr								
	May			0.0	0.0	0.0	0.1	0.0	
	Jun			0.1	0.0	0.0	0.1	0.0	
	Jul			0.1	0.0	0.0	0.1	0.0	
	Aug			0.1	0.0	0.0	0.1	0.0	
	Sep			0.1	0.0	0.0	0.1	0.0	
	Oct			0.1	0.0	0.0	0.1	0.0	
	Nov								
	Dec								
Average Mass Loading (kg/ha/d)				0.1	0.0	0.0	0.1	0.0	
Average Mass Out (lb/day)									
Monthly Mass Out (kg/ha/d)	Jan								
	Feb								
	Mar								
	Apr								
	May			0.0	0.0	0.0	0.0	0.0	
	Jun			0.0	0.0	0.0	0.0	0.0	
	Jul			0.0	0.0	0.0	0.0	0.0	
	Aug			0.0	0.0	0.0	0.0	0.0	
	Sep			0.0	0.0	0.0	0.0	0.0	
	Oct			0.0	0.0	0.0	0.0	0.0	
	Nov								
	Dec								
Average Mass Out (kg/ha/d)									
Percent Reduction (by mass)									

Hydraulic Properties Based on Area and Flow

Percent Open Water	10%	Transition Side Slopes (H:V) - (10:1)	10			Override
Marsh Zone Depth (m)	0.46	Berm Side Slopes (H:V) - (4:1)	4	# of Deep Zones	80	
Deep Zone Depth (m)	1.4	Approx Aspect Ratio (L:W) - (5:1)	5	Deep Zone Bot Length	50	
Volume (m3)	144054832					

Hydraulic Loading Rate, q	HLR =	0.4	cm/d
Nominal Hydraulic Residence Time, days	HRT =	105.7	days

Nitrogen Species Calculations

Nitrogen Models
per K&K Eqns 13-28. 13-29, 13-39:
Adapted for Monthly TIS model
Organic Nitrogen (ON)

$$C_{ON_{OUT}} = C_{ON}^{*} + (C_{ON_{IN}} - C_{ON}^{*})\left[1 + \frac{k_{ON}A}{NQ}\right]^{-N}$$

Ammonia Nitrogen (AN)

$$C_{AN_{OUT}} = C_{AN}^{*} + (C_{AN_{IN}} - C_{AN}^{*})\left[1 + \frac{k_{AN}A}{NQ}\right]^{-N} + \left(\frac{k_{ON}}{k_{AN} - k_{ON}}\right)(C_{ON_{IN}} - C_{ON}^{*})\left(\left[1 + \frac{k_{ON}A}{NQ}\right]^{-N} - \left[1 + \frac{k_{AN}A}{NQ}\right]^{-N}\right)$$

Nitrate Nitrogen (NN)

$$C_{NN_{OUT}} = C_{NN_{IN}}\left[1 + \frac{k_{NN}A}{NQ}\right]^{-N} + \Psi\left[\begin{array}{l} \left(\frac{k_{AN}}{k_{NN} - k_{AN}}\right)C_{AN_{IN}}\left(\left[1 + \frac{k_{AN}A}{NQ}\right]^{-N} - \left[1 + \frac{k_{NN}A}{NQ}\right]^{-N}\right) \\ + \left(\frac{k_{ON}}{k_{AN} - k_{ON}}\right)\left(\frac{k_{AN}}{k_{NN} - k_{ON}}\right)(C_{ON_{IN}} - C_{ON}^{*})\left(\left[1 + \frac{k_{ON}A}{NQ}\right]^{-N} - \left[1 + \frac{k_{NN}A}{NQ}\right]^{-N}\right) \\ - \left(\frac{k_{ON}}{k_{AN} - k_{ON}}\right)\left(\frac{k_{AN}}{k_{NN} - k_{AN}}\right)(C_{ON_{IN}} - C_{ON}^{*})\left(\left[1 + \frac{k_{AN}A}{NQ}\right]^{-N} - \left[1 + \frac{k_{NN}A}{NQ}\right]^{-N}\right) \end{array}\right]$$

where Ψ = fraction of ammonium nitrified, assumed to be 100% = 100%

Scenarios: 3 and 5B

Surface Flow Monthly PkC* Treatment Wetland Design Model w/ Water Balance (Wastewater Parameters)

User inputs indicated by white boxes.
Pop-up notes indicated by red triangles.

Project Name: ath Wetlands - 2012 nitrogen runs- 100% Klamath River at Link River
Project Number: WQ-KR253:ppt,tempKlamathFallsAirp

Flow Rate Converter

	Value	Units
Flow (Enter monthly flowrate below under "General and Hydrologic Input Data"; use this cell simply for converstion between units)		cfs
Converted Flow		m^3/d

Add Results to NADB Charts

General and Hydrologic Input Data

Wetland Hydrology

Month	Days in Month	Air Temp (°C)	Precip (m/mo)	Inflow (m3/d)	ET (m/mo)	Infiltration (m/mo)	Outflow (m3/d)
January	31	-0.7	0.04	3026414	0.06	0.37	2028464
February	28	1.0	0.03	2725485	0.02	0.37	1684608
March	31	3.6	0.02	3182995	0.01	0.37	2229164
April	30	5.9	0.03	3850910	0.02	0.37	2876330
May	31	10.5	0.03	4056422	0.03	0.37	3086964
June	30	14.6	0.01	3664970	0.06	0.37	2542614
July	31	19.4	0.00	2495507	0.08	0.37	1323698
August	31	18.4	0.01	2302228	0.12	0.37	1048299
September	30	14.2	0.01	1661225	0.15	0.37	274547
October	31	8.3	0.02	1910776	0.18	0.37	524010
November	30	2.8	0.05	1874077	0.15	0.37	602573
December	31	-1.0	0.04	1952367	0.11	0.37	823314
Average		8.1	0.025	2725281	0.08	0.37	1587049

Water Quality Input Data

Month	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
January								
February								
March								
April								
May			0.74	0.11	0.06		0.16	
June			1.39	0.09	0.06		0.19	
July			2.07	0.27	0.10		0.28	
August			3.57	0.33	0.11		0.18	
September			2.57	0.14	0.05		0.15	
October			2.05	0.20	0.12		0.15	
November								
December								
Annual Average			2.07	0.19	0.08		0.18	

		BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Target Effluent Conc., mg/L	C_e =								
Desired Confidence Percentile		0.5		0.5	0.5	0.5	0.5	0.5	0.5
Max Month/Annual Factor		1.7	1.9	1.8	2.5	2.5	1.6	1.8	3.0
Design Target Conc., mg/L	C_d =								
Wetland Background Limit, mg/L	C* =	Influent Dependent	Influent Dependent	1	0	0	1	0.002	300
P-Factor	P =	1.0	3.0	3.0	3.0	3.0	3.0	4.0	3.0
Confidence-based Rate Constant, 20°C, m/y	k_{20} =	0	200	17	18	27	13	10	75
Temperature Factor	θ =	1.00	1.065	1.05	1.05	1.11	1.06	1.00	1.00

Wetland Size and Output Predictions

Wetland Area

20,000	ac
8097.17	ha
80971660	m2

Effluent Concentrations

Month	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
January								
February								
March								
April								
May			0.8	0.06	0.06	0.96	0.1376	
June			1.3	0.14	0.05	1.51	0.1603	
July			1.8	0.44	0.11	2.32	0.2276	
August			2.8	0.85	0.17	3.83	0.1456	
September			2.1	0.51	0.11	2.69	0.1107	
October			1.8	0.38	0.14	2.35	0.1117	
November								
December								
Annual Average			1.76	0.40	0.11	2.28	0.1489	

	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Avg Percent Reduction (by concentration)			15%	-108%	-27%	3%	19%	

Scenarios: 3 and 5B

Surface Flow Monthly PkC* Treatment Wetland Design Model w/ Water Balance (Wastewater Parameters)

User inputs indicated by white boxes.
Pop-up notes indicated by red triangles.

Project Name: ath Wetlands - 2012 nitrogen runs- 100% Klamath River at Link River
Project Number: WQ-KR253:ppt,tempKlamathFallsAirp

Average Mass Loading (lb/day)				12405.6	1145.3	502.8	14053.7	1106	
Monthly Mass Loading (kg/ha/d)	Jan								
	Feb								
	Mar								
	Apr								
	May			0.4	0.1	0.0	0.5	0.1	
	Jun			0.6	0.0	0.0	0.7	0.1	
	Jul			0.6	0.1	0.0	0.8	0.1	
	Aug			1.0	0.1	0.0	1.1	0.1	
	Sep			0.5	0.0	0.0	0.6	0.0	
	Oct			0.5	0.0	0.0	0.6	0.0	
	Nov								
	Dec								
Average Mass Loading (kg/ha/d)				0.7	0.1	0.0	0.8	0.1	
Average Mass Out (lb/day)				6169	1386	372	7966	521	
Monthly Mass Out (kg/ha/d)	Jan								
	Feb								
	Mar								
	Apr								
	May			0.3	0.0	0.0	0.4	0.1	
	Jun			0.4	0.0	0.0	0.5	0.1	
	Jul			0.3	0.1	0.0	0.4	0.0	
	Aug			0.4	0.1	0.0	0.5	0.0	
	Sep			0.1	0.0	0.0	0.1	0.0	
	Oct			0.1	0.0	0.0	0.2	0.0	
	Nov								
	Dec								
Average Mass Out (kg/ha/d)				0.3	0.1	0.0	0.4	0.0	
Percent Reduction (by mass)				50%	-21%	26%	43%	53%	

Hydraulic Properties Based on Area and Flow

Percent Open Water	10%	Transition Side Slopes (H:V) - (10:1)	10			Override
Marsh Zone Depth (m)	0.46	Berm Side Slopes (H:V) - (4:1)	4	# of Deep Zones	40	
Deep Zone Depth (m)	1.4	Approx Aspect Ratio (L:W) - (5:1)	5	Deep Zone Bot Length	50	
Volume (m3)	36108397					

Hydraulic Loading Rate, q	HLR =	3.4	cm/d
Nominal Hydraulic Residence Time, days	HRT =	13.2	days

Nitrogen Species Calculations

Nitrogen Models
per K&K Eqns 13-28. 13-29, 13-39:
Adapted for Monthly TIS model
Organic Nitrogen (ON)

$$C_{ON_{OUT}} = C^*_{ON} + (C_{ON_{IN}} - C^*_{ON})\left[1+\frac{k_{ON}A}{NQ}\right]^{-N}$$

Ammonia Nitrogen (AN)

$$C_{AN_{OUT}} = C^*_{AN} + (C_{AN_{IN}} - C^*_{AN})\left[1+\frac{k_{AN}A}{NQ}\right]^{-N} + \left(\frac{k_{ON}}{k_{AN}-k_{ON}}\right)(C_{ON_{IN}} - C^*_{ON})\left(\left[1+\frac{k_{ON}A}{NQ}\right]^{-N} - \left[1+\frac{k_{AN}A}{NQ}\right]^{-N}\right)$$

Nitrate Nitrogen (NN)

$$C_{NN_{OUT}} = C_{NN_{IN}}\left[1+\frac{k_{NN}A}{NQ}\right]^{-N} + \Psi\left[\begin{array}{l} \left(\frac{k_{AN}}{k_{NN}-k_{AN}}\right)C_{AN_{IN}}\left(\left[1+\frac{k_{AN}A}{NQ}\right]^{-N} - \left[1+\frac{k_{NN}A}{NQ}\right]^{-N}\right) \\ +\left(\frac{k_{ON}}{k_{AN}-k_{ON}}\right)\left(\frac{k_{AN}}{k_{NN}-k_{ON}}\right)(C_{ON_{IN}} - C^*_{ON})\left(\left[1+\frac{k_{ON}A}{NQ}\right]^{-N} - \left[1+\frac{k_{NN}A}{NQ}\right]^{-N}\right) \\ -\left(\frac{k_{ON}}{k_{AN}-k_{ON}}\right)\left(\frac{k_{AN}}{k_{NN}-k_{AN}}\right)(C_{ON_{IN}} - C^*_{ON})\left(\left[1+\frac{k_{AN}A}{NQ}\right]^{-N} - \left[1+\frac{k_{NN}A}{NQ}\right]^{-N}\right) \end{array}\right]$$

where Ψ = fraction of ammonium nitrified, assumed to be 100% = 100%

Scenario 4A

Surface Flow Simple PkC* Design Model (Wastewater Parameters)

User inputs indicated by white boxes.
Pop-up notes indicated by red triangles.
Reference: Kadlec and Wallace, 2006
Treatment Wetlands. Boca Raton: CRC Press, Inc.

Project Name	Klamath Wetlands - MAY 2012 P runs- 50% flow at link Riv
Project Number	2000-2007 WQ sampling KR253; temp

General Inflow Data

Parameter	Value	Units
Annual Average Daily Flow	830	cfs
Converted Flow	2,030,678	m^3/d
Wastewater Temperature	10.5	°C

Add Results to NADB Plots

Water Quality Characteristics

Parameter		BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Influent Concentration, mg/L	C_i =							0.16	
Average Target Effluent Conc., mg/L	C_e =								
Desired Confidence Percentile		0.5		0.5	0.5	0.5	0.5	0.5	0.5
Max Month/Annual Factor		1.7	1.9	1.8	2.5	2.5	1.6	1.8	3.0
Design Target Conc., mg/L	C_d =								
Wetland Background Limit, mg/L	C^* =	2	5.10	1.5	0	0	1.5	0.002	40
Reduction fraction to target	$F_e = 1 - C_e/C_i$=	No Value	No Value	No Value	No Value	No Value	No Value	1.000	No Value
Reduction fraction to background	$F_b = 1 - C^*/C_i$ =							0.987	
Confidence-based Rate Constant, 20°C, m/y	k_{20} =	33	200	17	15	27	13	21	83
Temperature Factor	θ =	1.00	1.000	1.05	1.05	1.11	1.06	1.00	1.00
P-Factor	P =	1	3	3	3	3	3	3.4	3
Areal Rate Constant, m/y	k_T =	33	200	10.7	9.5	10.0	7.7	21	83
Area required for each parameter, ac		C*>Cd	C*>Cd	C*>Cd			C*>Cd	C*>Cd	C*>Cd

Required Treatment Wetland Area

Required Treatment Wetland Area	A_{max} =	0.0	acres	Displays minimum wetland area to treat all pollutants down to desired targets
		0.0	ha	
User Defined Area	A_{user} =	10,000	acres	User specified wetland area; leave blank if you wish to use A_{max} (above) for effluent calculations below.
		4048.6	ha	

Final Effluent Concentrations and Percent Removal

Area (ha) used for Calculations =	4048.6 ha	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Design Target Conc., mg/L	C_d =								
Influent concentrations, mg/L	C_i =							0.2	
Confidence-based Effluent concentration, mg/L	C_e =							0.060	
Percent Reduction (by concentration)								62%	
Mass Loading (lb/day)								705	
Mass Loading (kg/ha/d)								0.1	
Mass Out (lb/day)								268	
Mass Out (kg/ha/d)								0.0	
Percent Reduction (by mass)								62%	

Hydraulic Properties Based on Area and Flow

						Override
Percent Open Water	10%	Transition Side Slopes (H:V) - (10:1)	10			
Marsh Zone Depth (m)	0.46	Berm Side Slopes (H:V) - (4:1)	4	# of Deep Zones	28	
Deep Zone Depth (m)	1.4	Approx Aspect Ratio (L:W) - (5:1)	5	Deep Zone Bot Length	50	
Volume (m3)	18093349					

Hydraulic Loading Rate, q	HLR =	5.0	cm/d
Nominal Hydraulic Residence Time, days	HRT =	8.9	days

8/16/2012 1:33 PM
\\Rosa\proj\PacificCorp\370910WQTechAna\WetlandsAssessment\2012WetlandsModeling\CH2MHILLTrtWetlandToolkit_v2011e_2012KlamathLinkRiver_17jb_19.xlsm
P_May_253LinkR_C.002_50%Q

Scenario 4B

Surface Flow Simple PkC* Design Model (Wastewater Parameters)

Project Name	amath Wetlands - 2012 P runs- Jun -50% flow at Link River	**User inputs indicated by white boxes.** **Pop-up notes indicated by red triangles.**
Project Number	2000-2007 WQ sampling KR253; temp	**Reference: Kadlec and Wallace, 2006 Treatment Wetlands. Boca Raton: CRC Press, Inc.**

General Inflow Data

Add Results to NADB Plots

Parameter	Value	Units
Annual Average Daily Flow	750	cfs
Converted Flow	1,834,950	m^3/d
Wastewater Temperature	14.6	°C

Water Quality Characteristics

Parameter		BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Influent Concentration, mg/L	C_i =							0.19	
Average Target Effluent Conc., mg/L	C_e =							0.04	
Desired Confidence Percentile		0.5		0.5	0.5	0.5	0.5	0.5	0.5
Max Month/Annual Factor		1.7	1.9	1.8	2.5	2.5	1.6	1.8	3.0
Design Target Conc., mg/L	C_d =							0.02	
Wetland Background Limit, mg/L	C* =	2	5.10	1.5	0	0	1.5	0.002	40
Reduction fraction to target	$F_e = 1 - C_e/C_i$=	No Value	No Value	No Value	No Value	No Value	No Value	0.785	No Value
Reduction fraction to background	$F_b = 1 - C^*/C_i$ =							0.989	
Confidence-based Rate Constant, 20°C, m/y	k_{20} =	33	200	17	15	27	13	19	83
Temperature Factor	θ =	1.00	1.000	1.05	1.05	1.11	1.06	1.00	1.00
P-Factor	P =	1	3	3	3	3	3	3.4	3
Areal Rate Constant, m/y	k_T =	33	200	13.1	11.6	15.4	9.7	19	83
Area required for each parameter, ac		C*>Cd	C*>Cd	C*>Cd			C*>Cd	27108.7	C*>Cd

Required Treatment Wetland Area

Required Treatment Wetland Area	A_{max} =	27108.7	acres	Displays minimum wetland area to treat all pollutants down to desired targets
		10975.2	ha	
User Defined Area	A_{user} =	20,000	acres	User specified wetland area; leave blank if you wish to use A_{max} (above) for effluent calculations below.
		8097.2	ha	

Final Effluent Concentrations and Percent Removal

Area (ha) used for Calculations =	8097.2 ha	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Design Target Conc., mg/L	C_d =							0.022	
Influent concentrations, mg/L	C_i =							0.2	
Confidence-based Effluent concentration, mg/L	C_e =							0.034	
Percent Reduction (by concentration)								82%	
Mass Loading (lb/day)								753	
Mass Loading (kg/ha/d)								0.0	
Mass Out (lb/day)								137	
Mass Out (kg/ha/d)								0.0	
Percent Reduction (by mass)								82%	

Hydraulic Properties Based on Area and Flow

Percent Open Water	10%	Transition Side Slopes (H:V) - (10:1)	10			Override
Marsh Zone Depth (m)	0.46	Berm Side Slopes (H:V) - (4:1)	4	# of Deep Zones	40	
Deep Zone Depth (m)	1.4	Approx Aspect Ratio (L:W) - (5:1)	5	Deep Zone Bot Length	50	
Volume (m3)	36108397					

Hydraulic Loading Rate, q	HLR =	2.3	cm/d
Nominal Hydraulic Residence Time, days	HRT =	19.7	days

8/16/2012 2:18 PM
\\Rosa\proj\PacificCorp\370910WQTechAna\WetlandsAssessment\2012WetlandsModeling\CH2MHILLTrtWetlandToolkit_v2011e_2012KlamathLinkRiver_17jb_19.xlsm
P_Jun_KR253LinkR_C.002_50%Q

Scenario 4C

Surface Flow Simple PkC* Design Model (Wastewater Parameters)

Project Name	- 2012 P runs-July _ 50% flow Klamath River at Link River
Project Number	2000-2007 WQ sampling KR253; temp

User inputs indicated by white boxes.
Pop-up notes indicated by red triangles.
Reference: Kadlec and Wallace, 2006
Treatment Wetlands. Boca Raton: CRC Press, Inc.

General Inflow Data

Parameter	Value	Units
Annual Average Daily Flow	510	cfs
Converted Flow	1,247,766	m^3/d
Wastewater Temperature	19.4	°C

Add Results to NADB Plots

Water Quality Characteristics

Parameter		BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Influent Concentration, mg/L	C_i =							0.28	
Average Target Effluent Conc., mg/L	C_e =							0.04	
Desired Confidence Percentile		0.5		0.5	0.5	0.5	0.5	0.5	0.5
Max Month/Annual Factor		1.7	1.9	1.8	2.5	2.5	1.6	1.8	3.0
Design Target Conc., mg/L	C_d =							0.02	
Wetland Background Limit, mg/L	C^* =	2	5.10	1.5	0	0	1.5	0.002	40
Reduction fraction to target	$F_e = 1 - C_e/C_i$=	No Value	No Value	No Value	No Value	No Value	No Value	0.858	No Value
Reduction fraction to background	$F_b = 1 - C^*/C_i$ =							0.993	
Confidence-based Rate Constant, 20°C, m/y	k_{20} =	33	200	17	15	27	13	15	83
Temperature Factor	θ =	1.00	1.000	1.05	1.05	1.11	1.06	1.00	1.00
P-Factor	P =	1	3	3	3	3	3	3.4	3
Areal Rate Constant, m/y	k_T =	33	200	16.5	14.6	25.5	12.6	15	83
Area required for each parameter, ac		C*>Cd	C*>Cd	C*>Cd			C*>Cd	29687.9	C*>Cd

Required Treatment Wetland Area

Required Treatment Wetland Area	A_{max} =	29687.9	acres	Displays minimum wetland area to treat all pollutants down to desired targets
		12019.4	ha	
User Defined Area	A_{user} =	50,000	acres	User specified wetland area; leave blank if you wish to use A_{max} (above) for effluent calculations below.
		20242.9	ha	

Final Effluent Concentrations and Percent Removal

Area (ha) used for Calculations =	20242.9 ha	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Design Target Conc., mg/L	C_d =							0.022	
Influent concentrations, mg/L	C_i =							0.3	
Confidence-based Effluent concentration, mg/L	C_e =							0.009	
Percent Reduction (by concentration)								97%	
Mass Loading (lb/day)								773	
Mass Loading (kg/ha/d)								0.0	
Mass Out (lb/day)								25	
Mass Out (kg/ha/d)								0.0	
Percent Reduction (by mass)								97%	

Hydraulic Properties Based on Area and Flow

						Override
Percent Open Water	10%	Transition Side Slopes (H:V) - (10:1)	10			
Marsh Zone Depth (m)	0.46	Berm Side Slopes (H:V) - (4:1)	4	# of Deep Zones	63	
Deep Zone Depth (m)	1.4	Approx Aspect Ratio (L:W) - (5:1)	5	Deep Zone Bot Length	50	
Volume (m3)	90097026					

Hydraulic Loading Rate, q	HLR =	0.6	cm/d
Nominal Hydraulic Residence Time, days	HRT =	72.2	days

8/16/2012 2:19 PM
\\Rosa\proj\PacificCorp\370910WQTechAna\WetlandsAssessment\2012WetlandsModeling\CH2MHILLTrtWetlandToolkit_v2011e_2012KlamathLinkRiver_17jb_19.xlsm
P_July_KR253LinkR_C.002_50%Q

Scenario 4D

Surface Flow Simple PkC* Design Model (Wastewater Parameters)

Project Name: Klamath Wetlands - Aug 2012 P runs- 50% flow at link Rive

Project Number: 2000-2007 WQ sampling KR253; temp

User inputs indicated by white boxes.
Pop-up notes indicated by red triangles.
Reference: Kadlec and Wallace, 2006
Treatment Wetlands. Boca Raton: CRC Press, Inc.

General Inflow Data

Parameter	Value	Units
Annual Average Daily Flow	471	cfs
Converted Flow	1,151,125	m^3/d
Wastewater Temperature	18.4	°C

Add Results to NADB Plots

Water Quality Characteristics

Parameter		BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Influent Concentration, mg/L	C_i =							0.18	
Average Target Effluent Conc., mg/L	C_e =							0.04	
Desired Confidence Percentile		0.5		0.5	0.5	0.5	0.5	0.5	0.5
Max Month/Annual Factor		1.7	1.9	1.8	2.5	2.5	1.6	1.8	3.0
Design Target Conc., mg/L	C_d =							0.02	
Wetland Background Limit, mg/L	C^* =	2	5.10	1.5	0	0	1.5	0.002	40
Reduction fraction to target	$F_e = 1 - C_e/C_i$=	No Value	No Value	No Value	No Value	No Value	No Value	0.781	No Value
Reduction fraction to background	$F_b = 1 - C^*/C_i$ =							0.989	
Confidence-based Rate Constant, 20°C, m/y	k_{20} =	33	200	17	15	27	13	18	83
Temperature Factor	θ =	1.00	1.000	1.05	1.05	1.11	1.06	1.00	1.00
P-Factor	P =	1	3	3	3	3	3	3.4	3
Areal Rate Constant, m/y	k_T =	33	200	15.7	13.9	22.8	11.9	18	83
Area required for each parameter, ac		C*>Cd	C*>Cd	C*>Cd			C*>Cd	17718.0	C*>Cd

Required Treatment Wetland Area

		Value	Units	
Required Treatment Wetland Area	A_{max} =	**17718.0**	acres	Displays minimum wetland area to treat all pollutants down to desired targets
		7173.3	ha	
User Defined Area	A_{user} =		acres	User specified wetland area; leave blank if you wish to use A_{max} (above) for effluent calculations below.
			ha	

Final Effluent Concentrations and Percent Removal

Area (ha) used for Calculations =	**7173.3 ha**	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Design Target Conc., mg/L	C_d =							0.022	
Influent concentrations, mg/L	C_i =							0.2	
Confidence-based Effluent concentration, mg/L	C_e =							0.02	
Percent Reduction (by concentration)								88%	
Mass Loading (lb/day)								463	
Mass Loading (kg/ha/d)								0.0	
Mass Out (lb/day)								56	
Mass Out (kg/ha/d)								0.0	
Percent Reduction (by mass)								88%	

Hydraulic Properties Based on Area and Flow

						Override
Percent Open Water	10%	Transition Side Slopes (H:V) - (10:1)	10			
Marsh Zone Depth (m)	0.46	Berm Side Slopes (H:V) - (4:1)	4	# of Deep Zones	37	
Deep Zone Depth (m)	1.4	Approx Aspect Ratio (L:W) - (5:1)	5	Deep Zone Bot Length	50	
Volume (m3)	31998897					

Hydraulic Loading Rate, q	HLR =	1.6	cm/d
Nominal Hydraulic Residence Time, days	HRT =	27.8	days

Scenario 4E

Surface Flow Simple PkC* Design Model (Wastewater Parameters)

Project Name	Klamath Wetlands - Sep 2012 P runs- 50% flow at link Rive	**User inputs indicated by white boxes.**
Project Number	2000-2007 WQ sampling KR253; temp	**Pop-up notes indicated by red triangles.**
		Reference: Kadlec and Wallace, 2006
		Treatment Wetlands. Boca Raton: CRC Press, Inc.

General Inflow Data

Parameter	**Value**	**Units**
Annual Average Daily Flow	340	cfs
Converted Flow	830,621	m^3/d
Wastewater Temperature	14.2	°C

Add Results to NADB Plots

Water Quality Characteristics

Parameter		**BOD5**	**TSS**	**Organic N**	**NH_4-N**	**$NO_{2/3}$-N**	**TN**	**TP**	**FC**
Influent Concentration, mg/L	C_i =							0.15	
Average Target Effluent Conc., mg/L	C_e =							0.04	
Desired Confidence Percentile		0.5		0.5	0.5	0.5	0.5	0.5	0.5
Max Month/Annual Factor		1.7	1.9	1.8	2.5	2.5	1.6	1.8	3.0
Design Target Conc., mg/L	C_d =							0.02	
Wetland Background Limit, mg/L	C^* =	2	5.10	1.5	0	0	1.5	0.002	40
Reduction fraction to target	$F_e = 1 - C_e/C_i$=	No Value	No Value	No Value	No Value	No Value	No Value	0.737	No Value
Reduction fraction to background	$F_b = 1 - C^*/C_i$ =							0.987	
Confidence-based Rate Constant, 20°C, m/y	k_{20} =	33	200	17	15	27	13	20	83
Temperature Factor	θ =	1.00	1.000	1.05	1.05	1.11	1.06	1.00	1.00
P-Factor	P =	1	3	3	3	3	3	3.4	3
Areal Rate Constant, m/y	k_T =	33	200	12.8	11.4	14.8	9.5	20	83
Area required for each parameter, ac		C*>Cd	C*>Cd	C*>Cd			C*>Cd	10239.6	C*>Cd

Required Treatment Wetland Area

Required Treatment Wetland Area	A_{max} =	**10239.6**	acres	Displays minimum wetland area to treat all pollutants down to desired targets
		4145.6	ha	
User Defined Area	A_{user} =		acres	User specified wetland area; leave blank if you wish to use A_{max} (above) for effluent calculations below.
			ha	

Final Effluent Concentrations and Percent Removal

Area (ha) used for Calculations =	**4145.6 ha**	**BOD5**	**TSS**	**Organic N**	**NH_4-N**	**$NO_{2/3}$-N**	**TN**	**TP**	**FC**
Design Target Conc., mg/L	C_d =							0.022	
Influent concentrations, mg/L	C_i =							0.2	
Confidence-based Effluent concentration, mg/L	C_e =							0.02	
Percent Reduction (by concentration)								85%	
Mass Loading (lb/day)								279	
Mass Loading (kg/ha/d)								0.0	
Mass Out (lb/day)								41	
Mass Out (kg/ha/d)								0.0	
Percent Reduction (by mass)								85%	

Hydraulic Properties Based on Area and Flow

Percent Open Water	10%	Transition Side Slopes (H:V) - (10:1)	10			Override
Marsh Zone Depth (m)	0.46	Berm Side Slopes (H:V) - (4:1)	4	# of Deep Zones	28	
Deep Zone Depth (m)	1.4	Approx Aspect Ratio (L:W) - (5:1)	5	Deep Zone Bot Length	50	
Volume (m3)	18525215					

Hydraulic Loading Rate, q	HLR =	2.0	cm/d
Nominal Hydraulic Residence Time, days	HRT =	22.3	days

8/16/2012 2:21 PM
\\Rosa\proj\PacificCorp\370910WQTechAna\WetlandsAssessment\2012WetlandsModeling\CH2MHILLTrtWetlandToolkit_v2011e_2012KlamathLinkRiver_17jb_19.xlsm
P_Sep_KR253LinkR_C.002_50%Q

Scenario 4F

Surface Flow Simple PkC* Design Model (Wastewater Parameters)

Project Name	Klamath Wetlands - Oct 2012 P runs- 50% flow at link Rive
Project Number	2000-2007 WQ sampling KR253; temp

User inputs indicated by white boxes.
Pop-up notes indicated by red triangles.
Reference: Kadlec and Wallace, 2006
Treatment Wetlands. Boca Raton: CRC Press, Inc.

General Inflow Data

Parameter	Value	Units
Annual Average Daily Flow	391	cfs
Converted Flow	955,397	m^3/d
Wastewater Temperature	8.3	°C

Add Results to NADB Plots

Water Quality Characteristics

Parameter		BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Influent Concentration, mg/L	C_i =							0.15	
Average Target Effluent Conc., mg/L	C_e =							0.04	
Desired Confidence Percentile		0.5		0.5	0.5	0.5	0.5	0.5	0.5
Max Month/Annual Factor		1.7	1.9	1.8	2.5	2.5	1.6	1.8	3.0
Design Target Conc., mg/L	C_d =							0.02	
Wetland Background Limit, mg/L	C^* =	2	5.10	1.5	0	0	1.5	0.002	40
Reduction fraction to target	$F_e = 1 - C_e/C_i$=	No Value	No Value	No Value	No Value	No Value	No Value	0.726	No Value
Reduction fraction to background	$F_b = 1 - C^*/C_i$ =							0.986	
Confidence-based Rate Constant, 20°C, m/y	k_{20} =	33	200	17	15	27	13	24	
Temperature Factor	θ =	1.00	1.000	1.05	1.05	1.11	1.06	1.00	1.00
P-Factor	P =	1	3	3	3	3	3	3.4	3
Areal Rate Constant, m/y	k_T =	33	200	9.6	8.6	7.9	6.9	29	83
Area required for each parameter, ac		C*>Cd	C*>Cd	C*>Cd			C*>Cd	7885.2	C*>Cd

Required Treatment Wetland Area

Required Treatment Wetland Area	A_{max} =	**7885.2**	acres	Displays minimum wetland area to treat all pollutants down to desired targets
		3192.4	ha	
User Defined Area	A_{user} =		acres	User specified wetland area; leave blank if you wish to use A_{max} (above) for effluent calculations below.
			ha	

Final Effluent Concentrations and Percent Removal

Area (ha) used for Calculations =	**3192.4 ha**	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Design Target Conc., mg/L	C_d =							0.022	
Influent concentrations, mg/L	C_i =							0.1	
Confidence-based Effluent concentration, mg/L	C_e =							0.02	
Percent Reduction (by concentration)								85%	
Mass Loading (lb/day)								307	
Mass Loading (kg/ha/d)								0.0	
Mass Out (lb/day)								47	
Mass Out (kg/ha/d)								0.0	
Percent Reduction (by mass)								85%	

Hydraulic Properties Based on Area and Flow

						Override
Percent Open Water	10%	Transition Side Slopes (H:V) - (10:1)	10			
Marsh Zone Depth (m)	0.46	Berm Side Slopes (H:V) - (4:1)	4	# of Deep Zones	25	
Deep Zone Depth (m)	1.4	Approx Aspect Ratio (L:W) - (5:1)	5	Deep Zone Bot Length	50	
Volume (m3)	14280192					

Hydraulic Loading Rate, q	HLR =	3.0	cm/d
Nominal Hydraulic Residence Time, days	HRT =	14.9	days

Scenario 4G

Surface Flow Simple PkC* Design Model (Wastewater Parameters)

Project Name	Klamath Wetlands - MAY 2012 P runs- 50% flow at Link Ri	User inputs indicated by white boxes.
Project Number	2000-2007 WQ sampling KR253; temp	Pop-up notes indicated by red triangles.
		Reference: Kadlec and Wallace, 2006
		Treatment Wetlands. Boca Raton: CRC Press, Inc.

General Inflow Data

Parameter	Value	Units
Annual Average Daily Flow	830	cfs
Converted Flow	2,030,678	m^3/d
Wastewater Temperature	10.5	°C

Add Results to NADB Plots

Water Quality Characteristics

Parameter		BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Influent Concentration, mg/L	C_i =							0.16	
Average Target Effluent Conc., mg/L	C_e =							0.04	
Desired Confidence Percentile		0.5		0.5	0.5	0.5	0.5	0.5	0.5
Max Month/Annual Factor		1.7	1.9	1.8	2.5	2.5	1.6	1.8	3.0
Design Target Conc., mg/L	C_d =							0.02	
Wetland Background Limit, mg/L	C^* =	2	5.10	1.5	0	0	1.5	0.02	40
Reduction fraction to target	$F_e = 1 - C_e/C_i$=	No Value	No Value	No Value	No Value	No Value	No Value	0.746	No Value
Reduction fraction to background	$F_b = 1 - C^*/C_i$ =							0.873	
Confidence-based Rate Constant, 20°C, m/y	k_{20} =	33	200	17	15	27	13	21	83
Temperature Factor	θ =	1.00	1.000	1.05	1.05	1.11	1.06	1.00	1.00
P-Factor	P =	1	3	3	3	3	3	3.4	3
Areal Rate Constant, m/y	k_T =	33	200	10.7	9.5	10.0	7.7	21	83
Area required for each parameter, ac		C*>Cd	C*>Cd	C*>Cd			C*>Cd	70121.9	C*>Cd

Required Treatment Wetland Area

Required Treatment Wetland Area	A_{max} =	**70121.9**	acres	Displays minimum wetland area to treat all pollutants down to desired targets
		28389.4	ha	
User Defined Area	A_{user} =	**80,000**	acres	User specified wetland area; leave blank if you wish to use A_{max} (above) for effluent calculations below.
		32388.7	ha	

Final Effluent Concentrations and Percent Removal

Area (ha) used for Calculations =	**32388.7 ha**	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Design Target Conc., mg/L	C_d =							0.022	
Influent concentrations, mg/L	C_i =							0.2	
Confidence-based Effluent concentration, mg/L	C_e =							0.0216	
Percent Reduction (by concentration)								86%	
Mass Loading (lb/day)								705	
Mass Loading (kg/ha/d)								0.0	
Mass Out (lb/day)								97	
Mass Out (kg/ha/d)								0.0	
Percent Reduction (by mass)								86%	

Hydraulic Properties Based on Area and Flow

						Override
Percent Open Water	10%	Transition Side Slopes (H:V) - (10:1)	10			
Marsh Zone Depth (m)	0.46	Berm Side Slopes (H:V) - (4:1)	4	# of Deep Zones	80	
Deep Zone Depth (m)	1.4	Approx Aspect Ratio (L:W) - (5:1)	5	Deep Zone Bot Length	50	
Volume (m3)	144054832					

Hydraulic Loading Rate, q	HLR =	0.6	cm/d
Nominal Hydraulic Residence Time, days	HRT =	70.9	days

8/16/2012 2:18 PM
\\Rosa\proj\PacificCorp\370910WQTechAna\WetlandsAssessment\2012WetlandsModeling\CH2MHILLTrtWetlandToolkit_v2011e_2012KlamathLinkRiver_17jb_19.xlsm
P_May_253LinkR_C.02_50%Q

Scenario 4H

Surface Flow Simple PkC* Design Model (Wastewater Parameters)

Project Name	amath Wetlands - 2012 P runs- Jun -50% flow at Link River	**User inputs indicated by white boxes.** **Pop-up notes indicated by red triangles.**
Project Number	2000-2007 WQ sampling KR253; temp	**Reference: Kadlec and Wallace, 2006** **Treatment Wetlands. Boca Raton: CRC Press, Inc.**

General Inflow Data

Parameter	Value	Units
Annual Average Daily Flow	750	cfs
Converted Flow	1,834,950	m^3/d
Wastewater Temperature	14.6	°C

Add Results to NADB Plots

Water Quality Characteristics

Parameter		BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Influent Concentration, mg/L	C_i =							0.19	
Average Target Effluent Conc., mg/L	C_e =							0.04	
Desired Confidence Percentile		0.5		0.5	0.5	0.5	0.5	0.5	0.5
Max Month/Annual Factor		1.7	1.9	1.8	2.5	2.5	1.6	1.8	3.0
Design Target Conc., mg/L	C_d =							0.02	
Wetland Background Limit, mg/L	C^* =	2	5.10	1.5	0	0	1.5	0.02	40
Reduction fraction to target	$F_e = 1 - C_e/C_i$=	No Value	No Value	No Value	No Value	No Value	No Value	0.785	No Value
Reduction fraction to background	$F_b = 1 - C^*/C_i$ =							0.893	
Confidence-based Rate Constant, 20°C, m/y	k_{20} =	33	200	17	15	27	13	19	83
Temperature Factor	θ =	1.00	1.000	1.05	1.05	1.11	1.06	1.00	1.00
P-Factor	P =	1	3	3	3	3	3	3.4	3
Areal Rate Constant, m/y	k_T =	33	200	13.1	11.6	15.4	9.7	19	83
Area required for each parameter, ac		C*>Cd	C*>Cd	C*>Cd			C*>Cd	75752.4	C*>Cd

Required Treatment Wetland Area

Required Treatment Wetland Area	A_{max} =	75752.4	acres	Displays minimum wetland area to treat all pollutants down to desired targets
		30669.0	ha	
User Defined Area	A_{user} =	80,000	acres	User specified wetland area; leave blank if you wish to use A_{max} (above) for effluent calculations below.
		32388.7	ha	

Final Effluent Concentrations and Percent Removal

Area (ha) used for Calculations =	32388.7 ha	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Design Target Conc., mg/L	C_d =							0.022	
Influent concentrations, mg/L	C_i =							0.2	
Confidence-based Effluent concentration, mg/L	C_e =							0.02	
Percent Reduction (by concentration)								88%	
Mass Loading (lb/day)								753	
Mass Loading (kg/ha/d)								0.0	
Mass Out (lb/day)								89	
Mass Out (kg/ha/d)								0.0	
Percent Reduction (by mass)								88%	

Hydraulic Properties Based on Area and Flow

Percent Open Water	10%	Transition Side Slopes (H:V) - (10:1)	10			Override
Marsh Zone Depth (m)	0.46	Berm Side Slopes (H:V) - (4:1)	4	# of Deep Zones	80	
Deep Zone Depth (m)	1.4	Approx Aspect Ratio (L:W) - (5:1)	5	Deep Zone Bot Length	50	
Volume (m3)	144054832					

Hydraulic Loading Rate, q	HLR =	0.6	cm/d
Nominal Hydraulic Residence Time, days	HRT =	78.5	days

8/16/2012 2:19 PM
\\Rosa\proj\PacificCorp\370910WQTechAna\WetlandsAssessment\2012WetlandsModeling\CH2MHILLTrtWetlandToolkit_v2011e_2012KlamathLinkRiver_17jb_19.xlsm
P_Jun_KR253LinkR_C.02_50%Q

Scenario 4l

Surface Flow Simple PkC* Design Model (Wastewater Parameters)

Project Name	- 2012 P runs-July _ 50% flow Klamath River at Link River	**User inputs indicated by white boxes.** **Pop-up notes indicated by red triangles.**
Project Number	2000-2007 WQ sampling KR253; temp	**Reference: Kadlec and Wallace, 2006** **Treatment Wetlands. Boca Raton: CRC Press, Inc.**

General Inflow Data

Parameter	Value	Units
Annual Average Daily Flow	510	cfs
Converted Flow	1,247,766	m^3/d
Wastewater Temperature	19.4	°C

Add Results to NADB Plots

Water Quality Characteristics

Parameter		BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Influent Concentration, mg/L	C_i =							0.28	
Average Target Effluent Conc., mg/L	C_e =							0.04	
Desired Confidence Percentile		0.5		0.5	0.5	0.5	0.5	0.5	0.5
Max Month/Annual Factor		1.7	1.9	1.8	2.5	2.5	1.6	1.8	3.0
Design Target Conc., mg/L	C_d =							0.02	
Wetland Background Limit, mg/L	C^* =	2	5.10	1.5	0	0	1.5	0.02	40
Reduction fraction to target	$F_e = 1 - C_e/C_i$=	No Value	No Value	No Value	No Value	No Value	No Value	0.858	No Value
Reduction fraction to background	$F_b = 1 - C^*/C_i$ =							0.929	
Confidence-based Rate Constant, 20°C, m/y	k_{20} =	33	200	17	15	27	13	15	83
Temperature Factor	θ =	1.00	1.000	1.05	1.05	1.11	1.06	1.00	1.00
P-Factor	P =	1	3	3	3	3	3	3.4	3
Areal Rate Constant, m/y	k_T =	33	200	16.5	14.6	25.5	12.6	15	83
Area required for each parameter, ac		C*>Cd	C*>Cd	C*>Cd			C*>Cd	78115.9	C*>Cd

Required Treatment Wetland Area

Required Treatment Wetland Area	A_{max} =	78115.9	acres	Displays minimum wetland area to treat all pollutants down to desired targets
		31625.9	ha	
User Defined Area	A_{user} =	15,000	acres	User specified wetland area; leave blank if you wish to use A_{max} (above) for effluent calculations below.
		6072.9	ha	

Final Effluent Concentrations and Percent Removal

Area (ha) used for Calculations =	6072.9 ha	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Design Target Conc., mg/L	C_d =							0.022	
Influent concentrations, mg/L	C_i =							0.3	
Confidence-based Effluent concentration, mg/L	C_e =							0.07	
Percent Reduction (by concentration)								74%	
Mass Loading (lb/day)								773	
Mass Loading (kg/ha/d)								0.1	
Mass Out (lb/day)								204	
Mass Out (kg/ha/d)								0.0	
Percent Reduction (by mass)								74%	

Hydraulic Properties Based on Area and Flow

						Override
Percent Open Water	10%	Transition Side Slopes (H:V) - (10:1)	10			
Marsh Zone Depth (m)	0.46	Berm Side Slopes (H:V) - (4:1)	4	# of Deep Zones	34	
Deep Zone Depth (m)	1.4	Approx Aspect Ratio (L:W) - (5:1)	5	Deep Zone Bot Length	50	
Volume (m3)	27103242					

Hydraulic Loading Rate, q	HLR =	2.1	cm/d
Nominal Hydraulic Residence Time, days	HRT =	21.7	days

8/16/2012 2:20 PM
\\Rosa\proj\PacificCorp\370910WQTechAna\WetlandsAssessment\2012WetlandsModeling\CH2MHILLTrtWetlandToolkit_v2011e_2012KlamathLinkRiver_17jb_19.xlsm
July_P_KR253LinkR_C.02_50%Q

Scenario 4J

Surface Flow Simple PkC* Design Model (Wastewater Parameters)

Project Name: Klamath Wetlands - Aug 2012 P runs- 50% flow at link Rive
Project Number: 2000-2007 WQ sampling KR253; temp

User inputs indicated by white boxes.
Pop-up notes indicated by red triangles.
Reference: Kadlec and Wallace, 2006
Treatment Wetlands. Boca Raton: CRC Press, Inc.

General Inflow Data

Parameter	Value	Units
Annual Average Daily Flow	471	cfs
Converted Flow	1,151,125	m^3/d
Wastewater Temperature	18.4	°C

Add Results to NADB Plots

Water Quality Characteristics

Parameter		BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Influent Concentration, mg/L	C_i =							0.18	
Average Target Effluent Conc., mg/L	C_e =							0.04	
Desired Confidence Percentile		0.5		0.5	0.5	0.5	0.5	0.5	0.5
Max Month/Annual Factor		1.7	1.9	1.8	2.5	2.5	1.6	1.8	3.0
Design Target Conc., mg/L	C_d =							0.02	
Wetland Background Limit, mg/L	C^* =	2	5.10	1.5	0	0	1.5	0.02	40
Reduction fraction to target	$F_e = 1 - C_e/C_i$=	No Value	No Value	No Value	No Value	No Value	No Value	0.781	No Value
Reduction fraction to background	$F_b = 1 - C^*/C_i$ =							0.890	
Confidence-based Rate Constant, 20°C, m/y	k_{20} =	33	200	17	15	27	13	18	83
Temperature Factor	θ =	1.00	1.000	1.05	1.05	1.11	1.06	1.00	1.00
P-Factor	P =	1	3	3	3	3	3	3.4	3
Areal Rate Constant, m/y	k_T =	33	200	15.7	13.9	22.8	11.9	18	83
Area required for each parameter, ac		C*>Cd	C*>Cd	C*>Cd			C*>Cd	49682.2	C*>Cd

Required Treatment Wetland Area

		Value	Units	
Required Treatment Wetland Area	A_{max} =	**49682.2**	acres	Displays minimum wetland area to treat all pollutants down to desired targets
		20114.2	ha	
User Defined Area	A_{user} =	-	acres	User specified wetland area; leave blank if you wish to use A_{max} (above) for effluent calculations below.
			ha	

Final Effluent Concentrations and Percent Removal

Area (ha) used for Calculations =	**20114.2 ha**	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Design Target Conc., mg/L	C_d =							0.022	
Influent concentrations, mg/L	C_i =							0.2	
Confidence-based Effluent concentration, mg/L	C_e =							0.02	
Percent Reduction (by concentration)								88%	
Mass Loading (lb/day)								463	
Mass Loading (kg/ha/d)								0.0	
Mass Out (lb/day)								56	
Mass Out (kg/ha/d)								0.0	
Percent Reduction (by mass)								88%	

Hydraulic Properties Based on Area and Flow

						Override
Percent Open Water	10%	Transition Side Slopes (H:V) - (10:1)	10			
Marsh Zone Depth (m)	0.46	Berm Side Slopes (H:V) - (4:1)	4	# of Deep Zones	63	
Deep Zone Depth (m)	1.4	Approx Aspect Ratio (L:W) - (5:1)	5	Deep Zone Bot Length	50	
Volume (m3)	89525233					

Hydraulic Loading Rate, q	HLR =	0.6	cm/d
Nominal Hydraulic Residence Time, days	HRT =	77.8	days

Scenario 4K

Surface Flow Simple PkC* Design Model (Wastewater Parameters)

Project Name	Klamath Wetlands - Sep 2012 P runs- 50% flow at link Rive	User inputs indicated by white boxes.
Project Number	2000-2007 WQ sampling KR253; temp	Pop-up notes indicated by red triangles.
		Reference: Kadlec and Wallace, 2006
		Treatment Wetlands. Boca Raton: CRC Press, Inc.

General Inflow Data

Parameter	Value	Units
Annual Average Daily Flow	340	cfs
Converted Flow	830,621	m^3/d
Wastewater Temperature	14.2	°C

Add Results to NADB Plots

Water Quality Characteristics

Parameter		BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Influent Concentration, mg/L	C_i =							0.15	
Average Target Effluent Conc., mg/L	C_e =							0.04	
Desired Confidence Percentile		0.5		0.5	0.5	0.5	0.5	0.5	0.5
Max Month/Annual Factor		1.7	1.9	1.8	2.5	2.5	1.6	1.8	3.0
Design Target Conc., mg/L	C_d =							0.02	
Wetland Background Limit, mg/L	C^* =	2	5.10	1.5	0	0	1.5	0.02	40
Reduction fraction to target	$F_e = 1 - C_e/C_i$=	No Value	No Value	No Value	No Value	No Value	No Value	0.737	No Value
Reduction fraction to background	$F_b = 1 - C^*/C_i$ =							0.869	
Confidence-based Rate Constant, 20°C, m/y	k_{20} =	33	200	17	15	27	13	20	83
Temperature Factor	θ =	1.00	1.000	1.05	1.05	1.11	1.06	1.00	1.00
P-Factor	P =	1	3	3	3	3	3	3.4	3
Areal Rate Constant, m/y	k_T =	33	200	12.8	11.4	14.8	9.5	20	83
Area required for each parameter, ac		C*>Cd	C*>Cd	C*>Cd			C*>Cd	29632.4	C*>Cd

Required Treatment Wetland Area

Required Treatment Wetland Area	A_{max} =	29632.4	acres	Displays minimum wetland area to treat all pollutants down to desired targets
		11996.9	ha	
User Defined Area	A_{user} =	-	acres	User specified wetland area; leave blank if you wish to use A_{max} (above) for effluent calculations below.
			ha	

Final Effluent Concentrations and Percent Removal

Area (ha) used for Calculations =	11996.9 ha	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Design Target Conc., mg/L	C_d =							0.022	
Influent concentrations, mg/L	C_i =							0.2	
Confidence-based Effluent concentration, mg/L	C_e =							0.02	
Percent Reduction (by concentration)								85%	
Mass Loading (lb/day)								279	
Mass Loading (kg/ha/d)								0.0	
Mass Out (lb/day)								41	
Mass Out (kg/ha/d)								0.0	
Percent Reduction (by mass)								85%	

Hydraulic Properties Based on Area and Flow

						Override
Percent Open Water	10%	Transition Side Slopes (H:V) - (10:1)	10			
Marsh Zone Depth (m)	0.46	Berm Side Slopes (H:V) - (4:1)	4	# of Deep Zones	48	
Deep Zone Depth (m)	1.4	Approx Aspect Ratio (L:W) - (5:1)	5	Deep Zone Bot Length	50	
Volume (m3)	53448856					

Hydraulic Loading Rate, q	HLR =	0.7	cm/d
Nominal Hydraulic Residence Time, days	HRT =	64.3	days

Scenario 4L

Surface Flow Simple PkC* Design Model (Wastewater Parameters)

Project Name	Klamath Wetlands - Oct 2012 P runs- 50% flow at link Rive
Project Number	2000-2007 WQ sampling KR253; temp

User inputs indicated by white boxes.
Pop-up notes indicated by red triangles.
Reference: Kadlec and Wallace, 2006
Treatment Wetlands. Boca Raton: CRC Press, Inc.

General Inflow Data

Parameter	Value	Units
Annual Average Daily Flow	391	cfs
Converted Flow	955,397	m^3/d
Wastewater Temperature	8.3	°C

Add Results to NADB Plots

Water Quality Characteristics

Parameter		BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Influent Concentration, mg/L	C_i =							0.15	
Average Target Effluent Conc., mg/L	C_e =							0.04	
Desired Confidence Percentile		0.5		0.5	0.5	0.5	0.5	0.5	0.5
Max Month/Annual Factor		1.7	1.9	1.8	2.5	2.5	1.6	1.8	3.0
Design Target Conc., mg/L	C_d =							0.02	
Wetland Background Limit, mg/L	C^* =	2	5.10	1.5	0	0	1.5	0.02	40
Reduction fraction to target	$F_e = 1 - C_e/C_i$=	No Value	No Value	No Value	No Value	No Value	No Value	0.726	No Value
Reduction fraction to background	$F_b = 1 - C^*/C_i$ =							0.863	
Confidence-based Rate Constant, 20°C, m/y	k_{20} =	33	200	17	15	27	13	24	
Temperature Factor	θ =	1.00	1.000	1.05	1.05	1.11	1.06	1.00	1.00
P-Factor	P =	1	3	3	3	3	3	3.4	3
Areal Rate Constant, m/y	k_T =	33	200	9.6	8.6	7.9	6.9	29	83
Area required for each parameter, ac		C*>Cd	C*>Cd	C*>Cd			C*>Cd	23007.6	C*>Cd

Required Treatment Wetland Area

Required Treatment Wetland Area	A_{max} =	**23007.6**	acres	Displays minimum wetland area to treat all pollutants down to desired targets
		9314.8	ha	
User Defined Area	A_{user} =	-	acres	User specified wetland area; leave blank if you wish to use A_{max} (above) for effluent calculations below.
			ha	

Final Effluent Concentrations and Percent Removal

Area (ha) used for Calculations =	**9314.8 ha**	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Design Target Conc., mg/L	C_d =							0.022	
Influent concentrations, mg/L	C_i =							0.1	
Confidence-based Effluent concentration, mg/L	C_e =							0.02	
Percent Reduction (by concentration)								85%	
Mass Loading (lb/day)								307	
Mass Loading (kg/ha/d)								0.0	
Mass Out (lb/day)								47	
Mass Out (kg/ha/d)								0.0	
Percent Reduction (by mass)								85%	

Hydraulic Properties Based on Area and Flow

						Override
Percent Open Water	10%	Transition Side Slopes (H:V) - (10:1)	10			
Marsh Zone Depth (m)	0.46	Berm Side Slopes (H:V) - (4:1)	4	# of Deep Zones	43	
Deep Zone Depth (m)	1.4	Approx Aspect Ratio (L:W) - (5:1)	5	Deep Zone Bot Length	50	
Volume (m3)	41523604					

Hydraulic Loading Rate, q	HLR =	1.0	cm/d
Nominal Hydraulic Residence Time, days	HRT =	43.5	days

APPENDIX D
Treatment Wetlands Model Output (Iron Gate Location)

APPENDIX D

Treatment Wetlands Model Output (Iron Gate Location)

As described in section 4.3.1 of the report, this appendix contains output from the P-k-C* model for the Iron Gate location for scenarios 7 through 12 as listed in Table 8 (in section 4.2.3).

Scenarios: 7 and 12

Surface Flow Monthly PkC* Treatment Wetland Design Model w/ Water Balance (Wastewater Parameters)

User inputs indicated by white boxes.
Pop-up notes indicated by red triangles.

Project Name	Vetlands - 2012 nitrogen runs- 50% Klamath River at Iron Gate Flows
Project Number	WQ-KR190_ppt,temp 2001-2008; Etc

Flow Rate Converter

	Value	Units
Flow (Enter monthly flowrate below under "General and Hydrologic Input Data"; use this cell simply for converstion between units)	1065	cfs
Converted Flow	2,605,629	m^3/d

Add Results to NADB Charts

General and Hydrologic Input Data

Wetland Hydrology

Month	Days in Month	Air Temp (°C)	Precip (m/mo)	Inflow (m3/d)	ET (m/mo)	Infiltration (m/mo)	Outflow (m3/d)
January	31	1.9	0.04	2593399	0.02	0.00	2858777
February	28	3.6	0.03	2605632	0.03	0.00	2579924
March	31	6.3	0.02	2935923	0.07	0.00	2332188
April	30	8.4	0.03	3119419	0.10	0.00	2135640
May	31	13.9	0.03	2789127	0.14	0.00	1336182
June	30	17.9	0.00	1883884	0.16	0.00	0
July	31	23.4	0.00	1185379	0.18	0.00	0
August	31	21.7	0.01	1152350	0.16	0.00	0
September	30	17.3	0.01	1308933	0.12	0.00	0
October	31	10.9	0.02	1516894	0.07	0.00	816959
November	30	5.0	0.05	1504661	0.03	0.00	1857724
December	31	2.3	0.05	1798253	0.01	0.00	2269299
Average		11.1	0.024	2032821	0.09		1348891

Water Quality Input Data

Month	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
January								
February								
March								
April								
May			0.39	0.13	0.17		0.16	
June			0.83	0.08	0.17		0.18	
July			1.08	0.08	0.21		0.16	
August			1.23	0.06	0.20		0.15	
September			0.75	0.07	0.42		0.19	
October			0.64	0.13	0.43		0.12	
November								
December								
Annual Average			0.82	0.09	0.27		0.16	

		BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Target Effluent Conc., mg/L	C_e =								
Desired Confidence Percentile		0.5		0.5	0.5	0.5	0.5	0.5	0.5
Max Month/Annual Factor		1.7	1.9	1.8	2.5	2.5	1.6	1.8	3.0
Design Target Conc., mg/L	C_d =								
Wetland Background Limit, mg/L	C^* =	Influent Dependent	Influent Dependent	0.5	0	0	0.5	0.02	300
P-Factor	P =	1.0	3.0	3.0	3.0	3.0	3.0	4.0	3.0
Confidence-based Rate Constant, 20°C, m/y	k_{20} =	0	200	17	18	27	13	19	75
Temperature Factor	θ =	1.00	1.065	1.05	1.05	1.11	1.06	1.00	1.00

Wetland Size and Output Predictions

Wetland Area

100,000	ac
40485.83	ha
404858300	m2

Effluent Concentrations

Month	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
January								
February								
March								
April								
May			0.5	0.01	0.02	0.55	0.02	
June			0.5	0.03	0.03	0.59	0.02	
July			0.5	0.03	0.02	0.57	0.02	
August			0.5	0.03	0.02	0.57	0.02	
September			0.5	0.02	0.02	0.56	0.02	
October			0.5	0.02	0.02	0.55	0.02	
November								
December								
Annual Average			*0.52*	*0.02*	*0.02*	*0.56*	*0.0210*	

	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Avg Percent Reduction (by concentration)			36%	75%	93%	52%	87%	

8/17/2012 8:49 AM C:\Documents and Settings\kcarlso2\My Documents\Mail\OL Temp Attachments\CH2MHILLTrtWetlandToolkit_v2011e_2012KlamathIronGate_17.xlsm Avg_N-C.5_P-C.02_50%Q_KR19

Surface Flow Monthly PkC* Treatment Wetland Design Model w/ Water Balance (Wastewater Parameters)

User inputs indicated by white boxes.
Pop-up notes indicated by red triangles.

Project Name: Wetlands - 2012 nitrogen runs- 50% Klamath River at Iron Gate Flows
Project Number: WQ-KR190_ppt,temp 2001-2008; Etc

Average Mass Loading (lb/day)				3672.4	415.0	1189.0	5276.4	711	
Monthly Mass Loading (kg/ha/d)	Jan								
	Feb								
	Mar								
	Apr								
	May			0.0	0.0	0.0	0.0	0.0	
	Jun			0.0	0.0	0.0	0.1	0.0	
	Jul			0.0	0.0	0.0	0.0	0.0	
	Aug			0.0	0.0	0.0	0.0	0.0	
	Sep			0.0	0.0	0.0	0.0	0.0	
	Oct			0.0	0.0	0.0	0.0	0.0	
	Nov								
	Dec								
Average Mass Loading (kg/ha/d)				0.0	0.0	0.0	0.1	0.0	
Average Mass Out (lb/day)				1553	69	56	1678	62	
Monthly Mass Out (kg/ha/d)	Jan								
	Feb								
	Mar								
	Apr								
	May			0.0	0.0	0.0	0.0	0.0	
	Jun			0.0	0.0	0.0	0.0	0.0	
	Jul			0.0	0.0	0.0	0.0	0.0	
	Aug			0.0	0.0	0.0	0.0	0.0	
	Sep			0.0	0.0	0.0	0.0	0.0	
	Oct			0.0	0.0	0.0	0.0	0.0	
	Nov								
	Dec								
Average Mass Out (kg/ha/d)				0.0	0.0	0.0	0.0	0.0	
Percent Reduction (by mass)				58%	83%	95%	68%	91%	

Hydraulic Properties Based on Area and Flow

						Override
Percent Open Water	10%	Transition Side Slopes (H:V) - (10:1)	10			
Marsh Zone Depth (m)	0.46	Berm Side Slopes (H:V) - (4:1)	4	# of Deep Zones	89	
Deep Zone Depth (m)	1.4	Approx Aspect Ratio (L:W) - (5:1)	5	Deep Zone Bot Length	50	
Volume (m3)	180018502					

Hydraulic Loading Rate, q	HLR =	0.5	cm/d
Nominal Hydraulic Residence Time, days	HRT =	88.6	days

Nitrogen Species Calculations

Nitrogen Models
per K&K Eqns 13-28. 13-29, 13-39:
Adapted for Monthly TIS model
Organic Nitrogen (ON)

$$C_{ON_{OUT}} = C_{ON}^{*} + (C_{ON_{IN}} - C_{ON}^{*})\left[1+\frac{k_{ON}A}{NQ}\right]^{-N}$$

Ammonia Nitrogen (AN)

$$C_{AN_{OUT}} = C_{AN}^{*} + (C_{AN_{IN}} - C_{AN}^{*})\left[1+\frac{k_{AN}A}{NQ}\right]^{-N} + \left(\frac{k_{ON}}{k_{AN}-k_{ON}}\right)(C_{ON_{IN}} - C_{ON}^{*})\left(\left[1+\frac{k_{ON}A}{NQ}\right]^{-N} - \left[1+\frac{k_{AN}A}{NQ}\right]^{-N}\right)$$

Nitrate Nitrogen (NN)

$$C_{NN_{OUT}} = C_{NN_{IN}}\left[1+\frac{k_{NN}A}{NQ}\right]^{-N} + \Psi\left[\begin{array}{l} \left(\frac{k_{AN}}{k_{NN}-k_{AN}}\right)C_{AN_{IN}}\left(\left[1+\frac{k_{AN}A}{NQ}\right]^{-N} - \left[1+\frac{k_{NN}A}{NQ}\right]^{-N}\right) \\ +\left(\frac{k_{ON}}{k_{AN}-k_{ON}}\right)\left(\frac{k_{AN}}{k_{NN}-k_{ON}}\right)(C_{ON_{IN}} - C_{ON}^{*})\left(\left[1+\frac{k_{ON}A}{NQ}\right]^{-N} - \left[1+\frac{k_{NN}A}{NQ}\right]^{-N}\right) \\ -\left(\frac{k_{ON}}{k_{AN}-k_{ON}}\right)\left(\frac{k_{AN}}{k_{NN}-k_{AN}}\right)(C_{ON_{IN}} - C_{ON}^{*})\left(\left[1+\frac{k_{AN}A}{NQ}\right]^{-N} - \left[1+\frac{k_{NN}A}{NQ}\right]^{-N}\right) \end{array}\right]$$

where Ψ = fraction of ammonium nitrified, assumed to be 100% = 100%

8/17/2012 8:49 AM C:\Documents and Settings\kcarlso2\My Documents\Mail\OL Temp Attachments\CH2MHILLTrtWetlandToolkit_v2011e_2012KlamathIronGate_17.xlsm Avg_N-C.5_P-C.02_50%Q_KR19

Scenarios: 8 and 11

Surface Flow Monthly PkC* Treatment Wetland Design Model w/ Water Balance (Wastewater Parameters)

User inputs indicated by white boxes.
Pop-up notes indicated by red triangles.

Project Name: Vetlands - 2012 nitrogen runs- 50% Klamath River at Iron Gate Flows
Project Number: WQ-KR190_ppt,temp 2001-2008; Etc

Flow Rate Converter

	Value	Units
Flow (Enter monthly flowrate below under "General and Hydrologic Input Data"; use this cell simply for converstion between units)	2130	cfs
Converted Flow	5,211,258	m^3/d

Add Results to NADB Charts

General and Hydrologic Input Data

Wetland Hydrology

Month	Days in Month	Air Temp (°C)	Precip (m/mo)	Inflow (m3/d)	ET (m/mo)	Infiltration (m/mo)	Outflow (m3/d)
January	31	1.9	0.04	2593399	0.02	0.00	2619937
February	28	3.6	0.03	2605632	0.03	0.00	2603061
March	31	6.3	0.02	2935923	0.07	0.00	2875550
April	30	8.4	0.03	3119419	0.10	0.00	3021041
May	31	13.9	0.03	2789127	0.14	0.00	2643833
June	30	17.9	0.00	1883884	0.16	0.00	1674446
July	31	23.4	0.00	1185379	0.18	0.00	953837
August	31	21.7	0.01	1152350	0.16	0.00	947345
September	30	17.3	0.01	1308933	0.12	0.00	1156395
October	31	10.9	0.02	1516894	0.07	0.00	1446900
November	30	5.0	0.05	1504661	0.03	0.00	1539967
December	31	2.3	0.05	1798253	0.01	0.00	1845358
Average		11.1	0.024	2032821	0.09		1943973

Water Quality Input Data

Month	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
January								
February								
March								
April								
May			0.39	0.13	0.17		0.16	
June			0.83	0.08	0.17		0.18	
July			1.08	0.08	0.21		0.16	
August			1.23	0.06	0.20		0.15	
September			0.75	0.07	0.42		0.19	
October			0.64	0.13	0.43		0.12	
November								
December								
Annual Average			0.82	0.09	0.27		0.16	

Parameter		BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Target Effluent Conc., mg/L	C_e =								
Desired Confidence Percentile		0.5		0.5	0.5	0.5	0.5	0.5	0.5
Max Month/Annual Factor		1.7	1.9	1.8	2.5	2.5	1.6	1.8	3.0
Design Target Conc., mg/L	C_d =								
Wetland Background Limit, mg/L	C^* =	Influent Dependent	Influent Dependent	1	0	0	1	0.02	300
P-Factor	P =	1.0	3.0	3.0	3.0	3.0	3.0	4.0	3.0
Confidence-based Rate Constant, 20°C, m/y	k_{20} =	0	200	17	18	27	13	10	75
Temperature Factor	θ =	1.00	1.065	1.05	1.05	1.11	1.06	1.00	1.00

Wetland Size and Output Predictions

Wetland Area

10,000	ac
4048.58	ha
40485830	m2

Effluent Concentrations

Month	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
January								
February								
March								
April								
May			0.6	0.00	0.10	0.91	0.12	
June			1.0	0.00	0.05	1.09	0.11	
July			1.1	0.06	0.04	1.16	0.07	
August			1.1	0.10	0.05	1.24	0.07	
September			1.0	0.00	0.09	1.08	0.09	
October			0.8	0.00	0.20	1.07	0.07	
November								
December								
Annual Average			*0.92*	*0.03*	*0.09*	*1.09*	*0.09*	

	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Avg Percent Reduction (by concentration)			-13%	71%	66%	7%	45%	

Surface Flow Monthly PkC* Treatment Wetland Design Model w/ Water Balance (Wastewater Parameters)

User inputs indicated by white boxes.
Pop-up notes indicated by red triangles.

Project Name: Vetlands - 2012 nitrogen runs- 50% Klamath River at Iron Gate Flows
Project Number: WQ-KR190_ppt,temp 2001-2008; Etc

Average Mass Loading (lb/day)				3672.4	415.0	1189.0	5276.4	711	
Monthly Mass Loading (kg/ha/d)	Jan								
	Feb								
	Mar								
	Apr								
	May			0.3	0.1	0.1	0.5	0.1	
	Jun			0.4	0.0	0.1	0.5	0.1	
	Jul			0.3	0.0	0.1	0.4	0.0	
	Aug			0.4	0.0	0.1	0.4	0.0	
	Sep			0.2	0.0	0.1	0.4	0.1	
	Oct			0.2	0.1	0.2	0.4	0.0	
	Nov								
	Dec								
Average Mass Loading (kg/ha/d)				0.4	0.0	0.1	0.6	0.1	
Average Mass Out (lb/day)				3961	116	382	4678	372	
Monthly Mass Out (kg/ha/d)	Jan								
	Feb								
	Mar								
	Apr								
	May			0.4	0.0	0.1	0.6	0.1	
	Jun			0.4	0.0	0.0	0.4	0.0	
	Jul			0.2	0.0	0.0	0.3	0.0	
	Aug			0.3	0.0	0.0	0.3	0.0	
	Sep			0.3	0.0	0.0	0.3	0.0	
	Oct			0.3	0.0	0.1	0.4	0.0	
	Nov								
	Dec								
Average Mass Out (kg/ha/d)				0.4	0.0	0.0	0.5	0.0	
Percent Reduction (by mass)				-8%	72%	68%	11%	48%	

Hydraulic Properties Based on Area and Flow

Percent Open Water	10%	Transition Side Slopes (H:V) - (10:1)	10			Override
Marsh Zone Depth (m)	0.46	Berm Side Slopes (H:V) - (4:1)	4	# of Deep Zones	28	
Deep Zone Depth (m)	1.4	Approx Aspect Ratio (L:W) - (5:1)	5	Deep Zone Bot Length	50	
Volume (m3)	18093349					

Hydraulic Loading Rate, q — HLR = 5.0 cm/d
Nominal Hydraulic Residence Time, days — HRT = 8.9 days

Nitrogen Species Calculations

Nitrogen Models
per K&K Eqns 13-28. 13-29, 13-39:
Adapted for Monthly TIS model
Organic Nitrogen (ON)

$$C_{ON_{OUT}} = C_{ON}^{*} + (C_{ON_{IN}} - C_{ON}^{*})\left[1+\frac{k_{ON}A}{NQ}\right]^{-N}$$

Ammonia Nitrogen (AN)

$$C_{AN_{OUT}} = C_{AN}^{*} + (C_{AN_{IN}} - C_{AN}^{*})\left[1+\frac{k_{AN}A}{NQ}\right]^{-N} + \left(\frac{k_{ON}}{k_{AN}-k_{ON}}\right)(C_{ON_{IN}} - C_{ON}^{*})\left(\left[1+\frac{k_{ON}A}{NQ}\right]^{-N} - \left[1+\frac{k_{AN}A}{NQ}\right]^{-N}\right)$$

Nitrate Nitrogen (NN)

$$C_{NN_{OUT}} = C_{NN_{IN}}\left[1+\frac{k_{NN}A}{NQ}\right]^{-N} + \Psi\left[\begin{array}{l} \left(\frac{k_{AN}}{k_{NN}-k_{AN}}\right)C_{AN_{IN}}\left(\left[1+\frac{k_{AN}A}{NQ}\right]^{-N} - \left[1+\frac{k_{NN}A}{NQ}\right]^{-N}\right) \\ + \left(\frac{k_{ON}}{k_{AN}-k_{ON}}\right)\left(\frac{k_{AN}}{k_{NN}-k_{ON}}\right)(C_{ON_{IN}} - C_{ON}^{*})\left(\left[1+\frac{k_{ON}A}{NQ}\right]^{-N} - \left[1+\frac{k_{NN}A}{NQ}\right]^{-N}\right) \\ - \left(\frac{k_{ON}}{k_{AN}-k_{ON}}\right)\left(\frac{k_{AN}}{k_{NN}-k_{AN}}\right)(C_{ON_{IN}} - C_{ON}^{*})\left(\left[1+\frac{k_{AN}A}{NQ}\right]^{-N} - \left[1+\frac{k_{NN}A}{NQ}\right]^{-N}\right) \end{array}\right]$$

where Ψ = fraction of ammonium nitrified, assumed to be 100% = 100%

Scenario 9

Surface Flow Simple PkC* Design Model (Wastewater Parameters)

Project Name: 2 P runs-July _ 50% flow Klamath River at Iron Gate Flows
Project Number: 2000-2007 WQ sampling from Iron Ga

User inputs indicated by white boxes.
Pop-up notes indicated by red triangles.
Reference: Kadlec and Wallace, 2006
Treatment Wetlands. Boca Raton: CRC Press, Inc.

General Inflow Data

Parameter	Value	Units
Annual Average Daily Flow	484.5	cfs
Converted Flow	1,185,378	m^3/d
Wastewater Temperature	23.4	°C

Add Results to NADB Plots

Water Quality Characteristics

Parameter		BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Influent Concentration, mg/L	C_i =							0.16	
Average Target Effluent Conc., mg/L	C_e =							0.04	
Desired Confidence Percentile		0.5		0.5	0.5	0.5	0.5	0.5	0.5
Max Month/Annual Factor		1.7	1.9	1.8	2.5	2.5	1.6	1.8	3.0
Design Target Conc., mg/L	C_d =							0.02	
Wetland Background Limit, mg/L	C^* =	2	5.10	1.5	0	0	1.5	0.002	40
Reduction fraction to target	$F_e = 1 - C_e/C_i$ =	No Value	No Value	No Value	No Value	No Value	No Value	0.747	No Value
Reduction fraction to background	$F_b = 1 - C^*/C_i$ =							0.987	
Confidence-based Rate Constant, 20°C, m/y	k_{20} =	33	200	17	15	27	13	15	83
Temperature Factor	θ =	1.00	1.000	1.05	1.05	1.11	1.06	1.00	1.00
P-Factor	P =	1	3	3	3	3	3	3.4	3
Areal Rate Constant, m/y	k_T =	33	200	20.1	17.6	38.5	15.6	15	83
Area required for each parameter, ac		C*>Cd	C*>Cd	C*>Cd			C*>Cd	19962.2	C*>Cd

Required Treatment Wetland Area

Required Treatment Wetland Area	A_{max} =	**19962.2**	acres	Displays minimum wetland area to treat all pollutants down to desired targets
		8081.9	ha	
User Defined Area	A_{user} =		acres	User specified wetland area; leave blank if you wish to use A_{max} (above) for effluent calculations below.
			ha	

Final Effluent Concentrations and Percent Removal

Area (ha) used for Calculations =	**8081.9 ha**	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Design Target Conc., mg/L	C_d =							0.022	
Influent concentrations, mg/L	C_i =							0.2	
Confidence-based Effluent concentration, mg/L	C_e =							0.02	
Percent Reduction (by concentration)								86%	
Mass Loading (lb/day)								413	
Mass Loading (kg/ha/d)								0.0	
Mass Out (lb/day)								58	
Mass Out (kg/ha/d)								0.0	
Percent Reduction (by mass)								86%	

Hydraulic Properties Based on Area and Flow

Percent Open Water	10%	Transition Side Slopes (H:V) - (10:1)	10			Override
Marsh Zone Depth (m)	0.46	Berm Side Slopes (H:V) - (4:1)	4	# of Deep Zones	40	
Deep Zone Depth (m)	1.4	Approx Aspect Ratio (L:W) - (5:1)	5	Deep Zone Bot Length	50	
Volume (m3)	36040331					

Hydraulic Loading Rate, q	HLR =	1.5	cm/d
Nominal Hydraulic Residence Time, days	HRT =	30.4	days

8/17/2012 8:42 AM C:\Documents and Settings\kcarlso2\My Documents\Mail\OL Temp Attachments\CH2MHILLTrtWetlandToolkit_v2011e_2012KlamathIronGate_17.xlsm
P_July_KR190IrnGt_C.002_50%Q

Scenario 10

Surface Flow Simple PkC* Design Model (Wastewater Parameters)

Project Name: 12 P runs- Oct_50% flow Klamath River at Iron Gate Flows
Project Number: 2000-2007 WQ sampling from Iron Ga

User inputs indicated by white boxes.
Pop-up notes indicated by red triangles.
Reference: Kadlec and Wallace, 2006
Treatment Wetlands. Boca Raton: CRC Press, Inc.

General Inflow Data

Parameter	Value	Units
Annual Average Daily Flow	620	cfs
Converted Flow	1,516,892	m^3/d
Wastewater Temperature	10.9	^oC

Add Results to NADB Plots

Water Quality Characteristics

Parameter		BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Influent Concentration, mg/L	C_i =							0.12	
Average Target Effluent Conc., mg/L	C_e =							0.04	
Desired Confidence Percentile		0.5		0.5	0.5	0.5	0.5	0.5	0.5
Max Month/Annual Factor		1.7	1.9	1.8	2.5	2.5	1.6	1.8	3.0
Design Target Conc., mg/L	C_d =							0.02	
Wetland Background Limit, mg/L	C^* =	2	5.10	1.5	0	0	1.5	0.002	40
Reduction fraction to target	$F_e = 1 - C_e/C_i$=	No Value	No Value	No Value	No Value	No Value	No Value	0.653	No Value
Reduction fraction to background	$F_b = 1 - C^*/C_i$ =							0.983	
Confidence-based Rate Constant, 20°C, m/y	k_{20} =	33	200	17	15	27	13	24	83
Temperature Factor	θ =	1.00	1.000	1.05	1.05	1.11	1.06	1.00	1.00
P-Factor	P =	1	3	3	3	3	3	3.4	3
Areal Rate Constant, m/y	k_T =	33	200	10.9	9.7	10.4	7.9	24	83
Area required for each parameter, ac		C*>Cd	C*>Cd	C*>Cd			C*>Cd	12779.2	C*>Cd

Required Treatment Wetland Area

Required Treatment Wetland Area	A_{max} =	**12779.2**	acres	Displays minimum wetland area to treat all pollutants down to desired targets
		5173.8	ha	
User Defined Area	A_{user} =		acres	User specified wetland area; leave blank if you wish to use A_{max} (above) for effluent calculations below.
			ha	

Final Effluent Concentrations and Percent Removal

Area (ha) used for Calculations =	**5173.8 ha**	BOD5	TSS	Organic N	NH_4-N	$NO_{2/3}$-N	TN	TP	FC
Design Target Conc., mg/L	C_d =							0.022	
Influent concentrations, mg/L	C_i =							0.1	
Confidence-based Effluent concentration, mg/L	C_e =							0.02	
Percent Reduction (by concentration)								81%	
Mass Loading (lb/day)								385	
Mass Loading (kg/ha/d)								0.0	
Mass Out (lb/day)								74	
Mass Out (kg/ha/d)								0.0	
Percent Reduction (by mass)								81%	

Hydraulic Properties Based on Area and Flow

						Override
Percent Open Water	10%	Transition Side Slopes (H:V) - (10:1)	10			
Marsh Zone Depth (m)	0.46	Berm Side Slopes (H:V) - (4:1)	4	# of Deep Zones	32	
Deep Zone Depth (m)	1.4	Approx Aspect Ratio (L:W) - (5:1)	5	Deep Zone Bot Length	50	
Volume (m3)	23102206					

Hydraulic Loading Rate, q	HLR =	2.9	cm/d
Nominal Hydraulic Residence Time, days	HRT =	15.2	days

CH2MHILL

ES021512103604PDX