Evaluating the Potential for Watershed Restoration to Reduce Nutrient Loading to Upper Klamath Lake, Oregon

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Abbreviations, Chemical Terms, and Units of Measure

BMP	best management practice
UKL	Upper Klamath Lake
$H_2PO_4^-$	orthophosphate
HPO_4	orthophosphate
N	nitrogen
N_2	nitrogen gas
NO_3^-	nitrate
P	phosphorus
pH	measure of hydrogen ion activity; determines acidity or alkalinity
m^2/d	square meter/day
m^2/yr	square meter/year
g	gram
ha	hectare
kg	kilogram
kg/d	kilogram/day
kg/yr	kilogram/year
lb/d	pound/day
m	meter
mg/L/yr	milligram/liter/year
mg/L	milligram/liter
mo	month
µg/L	microgram/liter
spp.	species
yr	year

Conversion Factors

SI to Inch/Pound

Multiply	By	To obtain
	Length	
kilometer (km)	0.6214	mile (mi)
meter (m)	1.094	yard (yd)
	Area	
hectare (ha)	2.471	acre
square meter (m^2)	10.76	square foot (ft^2)
hectare (ha)	0.003861	square mile (mi^2)
square kilometer (km^2)	0.3861	square mile (mi^2)
	Volume	
liter (L)	1.057	quart (qt)
milliliter	0.0338	ounce
	Mass	
gram (g)	0.03527	ounce, avoirdupois (oz)
kilogram (kg)	2.205	pound avoirdupois (lb)
metric ton	1.102	ton per
milligram	$3.5274/10^5$	ounce

Evaluating the Potential for Watershed Restoration to Reduce Nutrient Loading to Upper Klamath Lake, Oregon

Paul McCormick and Sharon G. Campbell

Abstract

A literature review of best management practices to reduce nutrient loading was performed to provide information for resource managers in the Klamath Basin, Oregon. Although BMPs have already been implemented in the watershed, some sense of their effectiveness in reducing phosphorus loading and their cost for installation and maintenance is still lacking. This report discusses both causes of nutrient loading and a wide-variety of BMPs used to treat or reduce causal factors. We specifically focused on cattle grazing as the principal land-use and causal factor for nutrient loading in the Klamath Basin above Upper Klamath Lake, Oregon. Several BMP types, including stream corridor fencing, riparian buffer strips and constructed wetlands, seem to have potential for reducing phosphorus loading that may result from cattle grazing. However, no single BMP is likely to be the most effective in all locations or situations.

Introduction

Declining populations of two endangered species, *Deltistes luxatus (*Lost River sucker) and *Chasmistes brevirostris (*Shortnose sucker), have been attributed to poor water quality in Upper Klamath Lake (UKL) (U.S. Fish and Wildlife Service, 1998). UKL is classified as hypereutrophic and the resulting water quality associated with this condition is considered a probable cause for and a major limiting factor in the decline and recovery of these native fish species. Summertime water quality in UKL is characterized by warm temperatures and diel fluctuations in both dissolved oxygen concentrations and pH that stress resident fish (Kann, 1998; Kann and Smith, 1999; Saiki and others, 1999).

A number of factors contribute to poor water quality in the Klamath Basin (U.S. Fish and Wildlife Service, 1999). Nearly all Klamath Basin streams and rivers have been degraded through geomorphic change, loss of riparian habitat, diversions and associated flow reductions in the mainstem, water returned from drained wetlands, diking, and inputs from agricultural practices and logging operations. Many Klamath Basin water bodies fail to meet State of Oregon water quality standards (Oregon Department of Environmental Quality, 2002). The loss of lake fringe habitat has seriously affected larval and juvenile sucker habitat as well as water quality functions (Braunworth and others, 2003). About 80% of the wetlands in the Klamath Basin have been lost, including about 40,000 acres along the perimeter of UKL (Campbell, ed., 1993). Historically, it is estimated that the lakeshore wetlands along UKL annually removed about 230 tons of phosphorus and 16,000 metric tons of nitrogen (Shapiro and Associates, Inc., 2000).

The Klamath Basin Ecosystem Restoration Office was established in 1994 to implement habitat restoration projects addressing ecosystem restoration and water quality, economic stability and drought impacts. Recovery of listed suckers is a priority for the restoration program. A variety of habitat restoration and water quality objectives are considered critical to the recover of these endangered fish. To date, the Ecosystem Restoration Office has funded 400 projects aimed at accomplishing sucker recovery objectives. With the establishment of total maximum daily load targets for UKL by Oregon Department of Environmental Quality, a greater emphasis on nonpoint source pollution control, particularly phosphorus loading reduction, has been identified as a priority in the UKL watershed. Watershed nonpoint source control, over the past 30 years, has been principally accomplished through a variety of best management practices.

Best management practices (BMPs) are defined as measures, sometimes structural, that are determined to be the most effective, practical means of preventing or reducing pollution inputs from nonpoint sources to water bodies (Don Chapman Consultants, 1989). In the absence of those historical landscape features that regulated sucker populations and water quality in UKL, implementation of selected BMPs in the watershed may assist resource managers in making incremental progress toward sucker recovery objectives. The question is which BMPs are most appropriate or effective in achieving phosphorus-loading reductions for the UKL watershed. To date, there has been no quantitative evaluation of the effectiveness of watershed BMPs and restoration projects in providing realized benefits to water quality. The value of wetland and riparian habitats upstream of UKL for the control of phosphorus loading to UKL is unknown. However, water quality improvement in UKL could result in increased abundance and survival for sucker populations.

A literature review, intended to focus on those BMPs widely cited in reducing nutrient loading in North America and, if possible, in the Pacific Northwest region, has been performed. The primary focus of the review was BMP literature on the major land use in the UKL watershed, cattle grazing, and includes several major categories of BMP such as restored and constructed wetlands, irrigation and cattle management, stream corridor fencing and grazing practices. Management techniques aimed at reducing P in surface runoff is the primary focus of this report.

Literature Review

Phosphorus and Nitrogen in the Watershed

Phosphorus (P) is an essential element for plant growth (Sharpley, 1995). Soil P is found in various chemical and physical forms, which differ in their availability to plants (Hansen and others, 2002). There are three classes of soil P, soluble, reactive, and stable (fig. 1). The soluble class consists of the most available and reactive forms of P, predominantly orthophosphate anions ($H_2PO_4^-$, HPO_4^{2-}). Soluble P makes up only a very small portion (commonly <1%) of the total P in soil. The P in the reactive and stable classes is associated with the solid soil phase and occurs in both organic and inorganic forms. The organic P in the reactive class is from relatively fresh organic material that is readily decomposed, while the inorganic P is found on soil exchange sites or in relatively soluble minerals. Reactive phosphorus forms are in dynamic equilibrium with solution P. When soluble P uptake or loss occurs, P from the reactive class can replace it through processes such as desorption, dissolution, and mineralization. The largest class of soil P is referred to as stable or fixed. Stable P is not biologically available and is made of organic and inorganic compounds that are occluded, insoluble, or tightly sorbed. Inorganic P forms in the stable class are dominated by crystalline aluminum and iron compounds or by calcium compounds.

Phosphorus is a macronutrient that is commonly limiting to plant (and algal) growth in both terrestrial and aquatic ecosystems. Surface soils typically contain between 0.02 and 0.10% P in inorganic and organic fractions that vary widely in terms of their availability for plant growth.

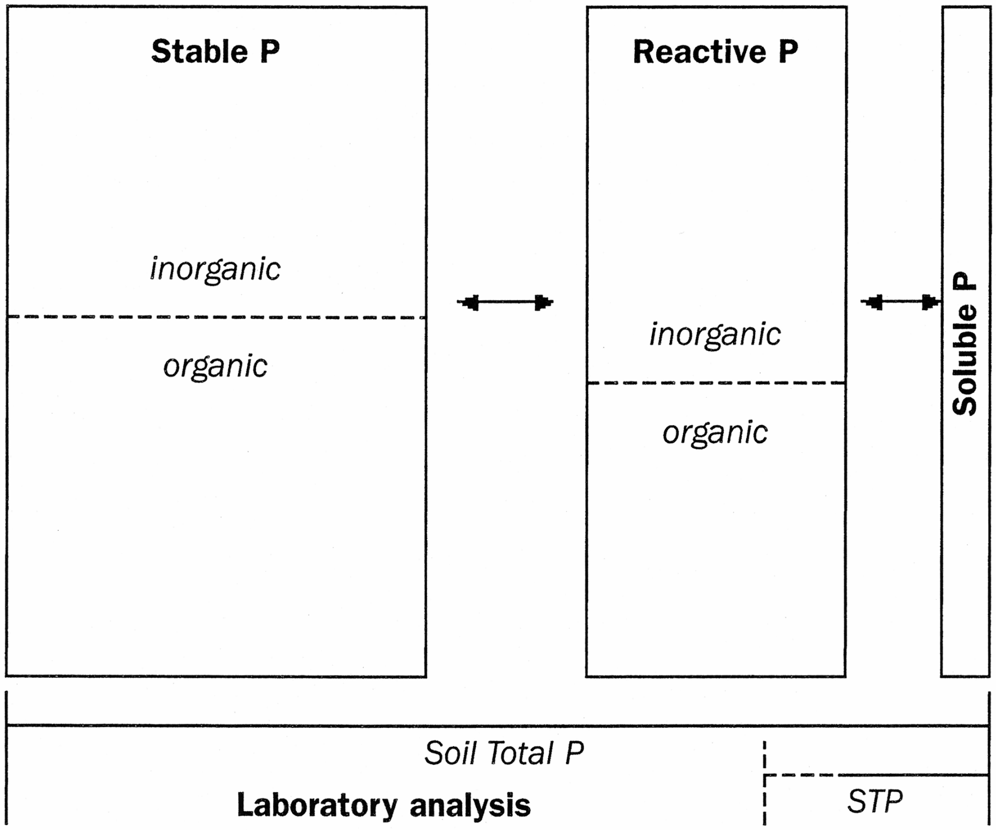

Figure 1. Classes of soil phosphorus in the landscape, from Hansen and others, 2002.

Phosphorus is only available for plant and microbial uptake in the form of orthophosphate (PO_4^{-3}). However, other P forms are easily converted to orthophosphate and are collectively referred to as biologically available P. The abundance of various fractions and the concentration of biologically available P are affected by soil and vegetation characteristics as well as management practices such as the application of inorganic P fertilizer or manure.

Soil P exists in three fractions with respect to biologically available P (fig. 1). "Solution P" consists primarily of orthophosphate dissolved in the soil porewater and is immediately available for plant uptake. This fraction is generally quite small. A much larger fraction exists as "reactive P", which includes loosely bound orthophosphate and easily mineralizable organic P. Reactive P represents the primary pool of biologically available P and, thus, is the primary determinant of soil fertility and the potential for runoff that promotes eutrophication in receiving waters. "Fixed P" includes both insoluble inorganic compounds and refractory organic compounds that may be slowly converted to reactive P. Fixed P includes primary (for instance, apatite) and secondary (occluded Fe- or Al-P) mineral complexes and refractory humic material. These three fractions are maintained in a dynamic equilibrium that is controlled both by inputs (fertilizer) and losses (plant uptake) of P from solution and by various soil characteristics such as pH and texture. The capacity of a soil to hold P refers to its ability to convert solution P to reactive and fixed forms. Soils with a high potential to hold P are less susceptible to the loss of dissolved orthophosphate (following fertilizer application) by leaching or in surface runoff, but can still contribute significant biologically available P to receiving waters in the form of reactive particulate P.

The terrestrial P cycle is controlled by various chemical reactions in the soil and a few key biological processes. Biological processes include plant and microbial uptake of orthophosphate from the soil solution, conversion of this P into organic forms (nucleic acids) and the return of this material to the soil solution through the leaching and mineralization of dead plant and microbial material. There is no significant gaseous phase to the P cycle as does exist for nitrogen (N). Inputs to the terrestrial P cycle include the weathering of parent materials - an extremely slow process -, applications of inorganic fertilizers and manure, and atmospheric deposition. Losses occur in surface and, less frequently, subsurface runoff and as a result of crop harvesting and livestock removal.

N is ubiquitous in the environment (Follett, 1995). N cycling in pastures is shown in figure 2. N accounts for 78% of the atmosphere as elemental gas (N_2). N_2 gas is inert and is not directly available for plant uptake. N taken up by plants from the soil originates from indigenous organic or inorganic forms.

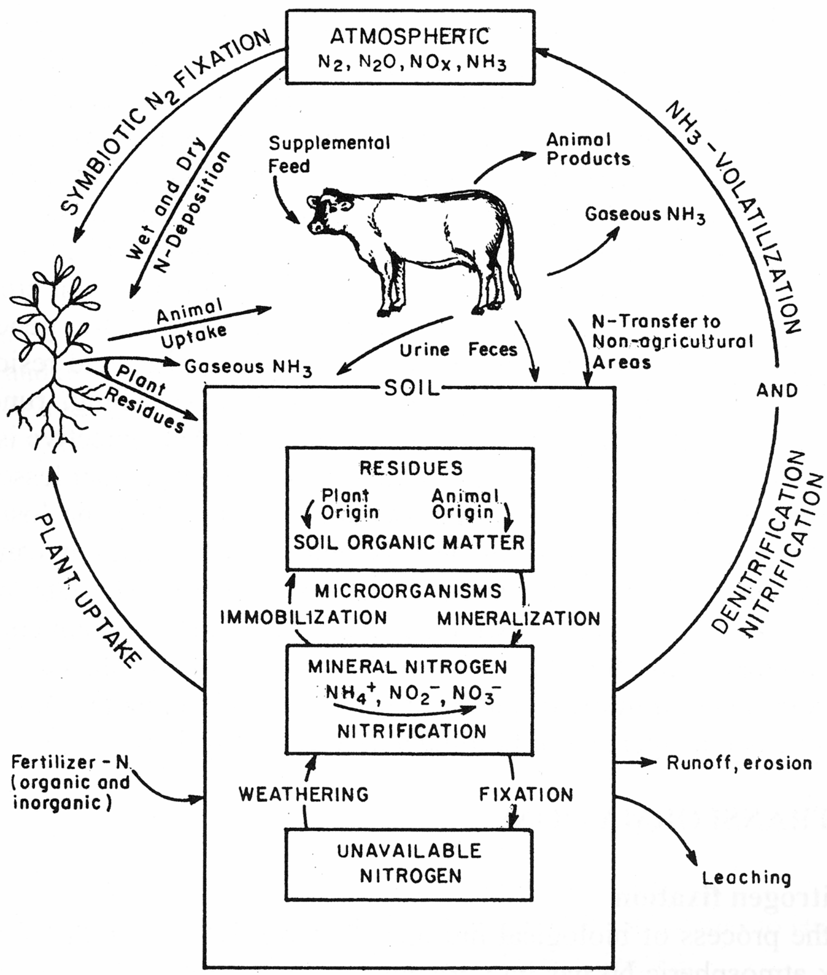

Figure 2. The nitrogen cycle in pastures, from Follett, 2001.

Organic nitrogen occurs naturally as part of the soil's organic matter fraction, but it can also be added to the soil from manure, symbiotic and nonsymbiotic nitrogen fixation, plant residues, and other sources. Nitrogen also represents the mineral fertilizer most applied to agricultural lands and in pastures. The form of nitrogen affecting water quality is nitrate (NO_3^-), a water-soluble anion that is not sorbed to the negatively charged sites on soil colloids, is very mobile, and moves readily with percolating water (leaching). Although both nitrogen and phosphorus in the watershed can contribute to undesirable surface water nutrient loading to receiving water bodies, in the Klamath Basin phosphorus is considered to be the nutrient of greatest concern and therefore is the target for total maximum daily load and general reduction.

Soil Phosphorus Losses to Surface Waters

Soil P losses can occur as dissolved P or particulate (sediment-bound) P in surface runoff or as dissolved P leached to subsurface drainage. Subsurface transport can be a major source of P loss in soils with a low P binding capacity, in waterlogged soils, from lands with tile drainage systems, and in circumstances where soils have become saturated with P (due to excessive application of inorganic fertilizer or manure). However, the majority of P loss typically occurs as surface runoff.

Phosphorus losses from undisturbed lands such as forests and native grasslands are typically low and primarily in the form of dissolved P. Phosphorus losses from agricultural lands are generally higher and primarily in the form of particulate P associated with fine soil particles. These particles are the most easily eroded and adsorb P to a greater extent than do coarser particles. Consequently, sediment in surface runoff typically is enriched with P compared with the source soils (Massey and Jackson, 1952, Sharpley, 1985). Most P export from agricultural watersheds occurs in runoff during a few storm events (Cosser, 1989). Routine water quality monitoring (monthly sampling) rarely captures such events and, thus, can grossly underestimate watershed P loads.

Biologically available P in surface runoff includes both dissolved P and bioavailable particulate P. Runoff dissolved P concentrations are determined using standard laboratory analyses. The biologically available P content of particulate P in surface runoff varies widely. Thus, runoff total P or particulate P concentration alone may not be a good predictor of biologically available P. Soil extraction procedures (bicarbonate-extractable P) and other methods (ion exchange resins) have been developed to better estimate the biologically available P content of surface runoff from different sources (Sharpley and others, 1991).

Phosphorus Bioavailability in the Aquatic Environment

Transformations between dissolved P and particulate P fractions occur once runoff enters the aquatic environment. Suspended sediments can act either as a source or sink for dissolved P depending upon relative concentrations in the sediment and water fractions. Therefore, increased total P concentrations during storm events may be accompanied by a decline in dissolved P concentrations due to elevated sediment loads. Once deposited in rivers and lakes, sediments are exposed to conditions that may enhance P bioavailability. Reducing conditions (anaerobic processes) in the sediments promote desorption of Fe-bound and other-metal-bound P that was not readily available in the source soils. Microbial mineralization of organic P, by contrast, is promoted by aerobic sediment conditions, which increase bacterial growth rates. These two processes may alternate in importance in UKL on a seasonal and even daily basis depending on wave action.

UKL is a shallow water body housed by a broad basin. Prevailing winds from the northwest exercise considerable wind-mixing potential due to the long fetch. When weather conditions are calm for one or more days in the summer, thermal stratification can occur. Biological oxygen demand from decaying organic material in the bottom sediments can rapidly decrease the oxygen content of waters located below the euphotic (lighted) zone. In addition, UKL waters are weakly buffered, a condition that allows the equilibrium state of the hydrogen ion concentration (pH) equilibrium to shift back and forth between acidic and alkaline end points very easily. Kann and Smith (1999) report that pH can shift between 5 and 10 units on a diel basis in response to algal respiration during darkness and photosynthesis during daylight hours.

Livestock Grazing Land Use and Nutrient Loading

Livestock grazing is the single largest land use in the United States, with the heaviest concentration on both private and public lands in the arid West (Krueger and others, 2002). As such, it has had a profound effect on ecosystem processes such as nutrient cycling (Bellows, 2001). While grazers are an integral part of grassland ecosystems, the introduction of cattle to lands across this region has resulted in widespread impacts to both riparian and upland habitats (George, 1996). Prior to passage of the Taylor Grazing Act in 1934, livestock grazing on federal lands were essentially unregulated. By the early 1900s, the damage to natural resources caused by overgrazing was becoming evident to land managers, politicians, and the general public (Griffiths, 1902). The Taylor Act was a first attempt to control grazing activities by limiting livestock numbers and collecting grazing fees that supported activities such as fencing and vegetation management to improve range conditions. This could be considered the first attempt to develop and implement grazing BMPs.

Modern grazing practices have improved the compatibility of grazing with the natural landscape (Krueger and others, 2002). Best management practices such as stream corridor fencing to exclude cattle access to the stream channel, manure collection, offstream watering facilities, and supplementary feeding located away from streams are just a few examples. In many cases, BMPs are implemented to achieve stream and riparian habitat improvement and to prevent stream channel sedimentation. Potential water quality benefits, such as reduction in nutrient loading, are more costly to quantify and therefore less often monitored following BMP implementation. As a result, many potential water quality improvements resulting from specific BMP implementation have yet to be quantified or documented.

Effects of Grazing on Nutrient Transport

Nutrient transport from pastures to drainage systems can occur through surface and subsurface flows (Follett, 1995; Follett, 2001; Follett and Delgado, 2002). Inputs of nitrogen include fertilizers, manure, biological N fixation, wet and dry deposition from the atmosphere, supplemental feed to livestock, and mineralization of soil organic matter. Losses of nitrogen can occur when plant or animal products are harvested, manure is removed, N fixation occurs in soils and may be released to the atmosphere, leached by subsurface or surface water, and transported by soil erosion or surface runoff. In general, nitrogen is utilized by plants for growth and is grazed by cattle where it is utilized or returned to the soil in manure or urine.

Phosphorus transport follows many of the same transport pathways as nitrogen with the exception of loss of phosphorus to the atmosphere. In general, the phosphorus cycle flows from the soil to plants that are consumed by herbivores and returned to the soil as manure or urine (Sharpley, 1995; Hansen and others, 2002). Another important source of phosphorus can be supplemental feed for livestock.

Grazing animals affect nitrogen and phosphorus transport in a number of ways. They consume plant material and convert it to body mass or release it back to the pasture as excretia and, in some cases, back to the atmosphere as gas. In addition, cattle compact the soil and, along with the removal of plant materials, may accelerate soil erosion, reduce infiltration, and increase surface runoff into nearby streams and water bodies.

Grazing livestock alter watershed nutrient cycling in various ways through their feeding and other activities (Bellows, 2001). In pastures, P is stored in plants and soils in various fractions (such as plant-available P, microbial P, metal-bound P). Cycling between these two compartments (plants and soil) occurs by plant uptake and litter decomposition, and the rate of cycling of different fractions is related to their bioavailability. Phosphorus is diverted from this cycle by grazing and some of this diverted P is returned to the litter layer in solid or liquid forms. The P in livestock fecal material is typically more readily available than that in plant tissue. Thus, both P availability and turnover rate increases as grazers are added to the landscape. Low levels of grazing can increase plant P storage whereas overgrazing reduces stored P in both above-ground and below-ground compartments.

Grazers not only mobilize plant nutrients, but they redistribute them across the landscape. Rather than being distributed evenly across a pasture, livestock deposition of feces and urine is concentrated in preferred locations such as watering spots and shaded areas used for resting (Follett, 2001). Petersen and others (1956) suggested that excrement from grazing cattle provided little nutrition to pasture vegetation because it was deposited disproportionately in only a few patches. Livestock often congregate in stream-riparian zones, resulting in preferential deposition of feces and urine in these areas. Concentration of animals in this zone has been associated with elevated stream-water nutrient concentrations (Lemly, 1982; Mosely and others, 1993).

Overstocking results in excess manure deposition and increased erosion, which accelerates nutrient losses. In particular, the concentration of cattle in riparian-stream zones results in loss of riparian vegetation, streambank destabilization, and changes in stream morphology that increase sediment and associated P inputs to the stream.

Grazing can increase surface runoff P loads from upland areas through several mechanisms. Cattle activity compacts surface soils resulting in reduced infiltration and increased surface runoff volumes. Increased overland flow rates contribute to erosion and particle transport. Reduced vegetative cover from overgrazing also promotes erosion. Phosphorus concentrations associated with eroded sediment are generally greater than that of the source soils because runoff carries smaller, more P-rich clay and silt particles rather than the heavier, nutrient-poor particles (Sharpley, 1995). The importance of sediment transport in determining P exports from irrigated lands is illustrated by the fact that total P concentrations in runoff are strongly correlated with sediment concentration (Bjorneberg and others, 2002). By contrast, dissolved P concentrations are most strongly correlated with indicators of available P (such as test P) in field soils (Westermann and others, 2001).

Land management practices such as fertilization on improved pastures can greatly increase P exports from grazed lands. Supplemental feeding also increases P input to the pasture and can increase P runoff. Grazing operations, particularly those involving seasonal grazing, also export P from the watershed when cattle are shipped to market. A portion of the P ingested by cattle is stored as new animal tissue and ultimately exported out of the watershed when the animals are sold. For example, Shewmaker (1997) modeled the export of P in the form of cattle biomass from the Cascade-Reservoir watershed in Idaho, where approximately 30,000 animals are trucked in each summer to graze 52,000 acres of mostly irrigated pasture. Using three different calculation methods, they estimated that on average, livestock grazing removed 19–39 tons of P annually from the watershed. The authors suggested that grazing might not increase P loads to downstream receiving waters if these annual exports were combined with proper grazing management.

Nutrient Transport Management

Best management practices can be classified as either non-structural or structural. Nonstructural BMPs are modifications in agricultural practices that do not require some type of construction (Don Chapman Associates, 1989). They focus on source reduction (pollution prevention) and programs and procedures for controlling agricultural nonpoint- source pollution. Structural BMPs are practices requiring that something be built and maintained after construction. There are two excellent general resources that describe BMP types, applications, and pertinent details on implementation; *National Management Measures for the Control of Nonpoint Pollution from Agriculture* (U.S. Environmental

Protection Agency, 2003) and *Stream Corridor Restoration: Principles, Processes, and Practices* (Federal Interagency Stream Restoration Working Group, 1998). Although the latter reference does not directly address water quality, the restoration of stream corridors is an integral part of practices that reduce nutrient transport and help protect the receiving water body from excess nutrients originating in the watershed.

There is a broad consensus in the literature that a primary goal of grazing management plans should be to restore and maintain a functioning riparian zone. Healthy riparian zones fulfill numerous ecological functions related to water quality including streambank stabilization, the dissipation of stream energy during high flows, and the filtering of sediments (Kauffman and Krueger, 1984). A functional riparian zone can also serve as a buffer to protect receiving waters from sediment and nutrients emanating from degraded uplands. Proper functioning of this zone requires adequate and suitable vegetation (Clifton, 1989), generally in some advanced successional state (U.S. Department of the Interior, 1990).

Although riparian zones constitute only a small percentage of rangeland in the West, livestock disproportionately concentrate within these areas. Impacts associated with overgrazing of these habitats are fairly well understood (U.S. Environmental Protection Agency, 2003). Reduction in vegetative cover, soil compaction, streambank damage, and direct fouling of streams when animals have direct access are just a few examples. These impacts affect stream water quality through increased soil loss and streambank erosion and reduced nutrient retention, all of which contribute to increased loads of sediments and associated nutrients. Stream nutrient loads are further increased by the deposition of feces and urine in the riparian zone and directly into the stream.

Numerous studies have focused on grazing in riparian zones in the West and management practices to prevent or reduce these impacts, and the most commonly recommended BMPs are discussed below. These fall into three major categories: (1) complete exclusion of cattle from the riparian zone, (2) partial exclusion, and (3) "passive exclusion" by using herding behavior to reduce grazing intensity rather than a physical barrier (U.S. Environmental Protection Agency, 2003).

A commonly employed option for protecting riparian zones from livestock damage is complete exclusion. Stream corridor fencing or, in some instances, structures such as ditches or banks can exclude livestock from direct access to streams. The total exclusion of livestock frequently produces a marked improvement in the vegetative condition and functioning of the riparian zone (Gunderson, 1968, Winegar, 1977, Claire and Storch, 1983, Duff, 1983, Prichard and others, 1993). As an example, Winegar (1997) reported that "Stream channels fenced from livestock afford the opportunity to compare sediment loads above (in livestock-grazed area) and below a fenced area. Sampling done at three runoff periods showed reduction in sediment loads of 79%, 48% and 69% respectively after flowing through 3.5 miles of protected channel." Exclusion can lead to a reversal in soil compaction, thereby increasing rates of water infiltration, and reduce the amount of sediment and manure entering streams. However, exclusion alone is not always adequate, and some streams are so degraded that they require active restoration (Federal Interagency Stream Restoration Working Group, 1998).

Pilot BMP studies conducted by the Vermont Department of Environmental Conservation on grazed lands in three watersheds (Meals, 2004) included fencing to exclude livestock from the stream-riparian zone coupled with improvements to livestock crossings and stream-bank revegetation resulted in as much as a 49% decline in total P export, a 15% decline in stream total P concentrations, and a 28% reduction in total suspended solids export. However, treatment results were variable with one treated watershed showing substantial reductions and a second showing an increase in nutrient and sediment export compared to the pretreatment period. These divergent results were attributed to excessive field runoff from new farming operations begun during the post-BMP period in the latter watershed and not to the ineffectiveness of BMPs.

Complete exclusion of livestock is not always a practical or acceptable option for ranchers.

Fencing costs per mile for riparian areas in the West have been estimated at $8,000 to $12,000 (Oppenheimer, 1996). Riparian zones typically provide some of the highest-quality grazing habitat such that livestock exclusion from these areas has an economic cost. Exclusion of cattle from streams requires that alternative water sources be provided. Cattle tend to follow fence lines and thus create a highly disturbed area adjacent to the riparian zone. Fencing also can limit movement of native wildlife in some areas and is susceptible to damage or destruction from winter or spring flooding.

Controlled Grazing

Complete exclusion of cattle by fencing is not the only strategy that has been used to return stream-riparian zones to an acceptable state. Other options include (1) fencing off the riparian zone and managing it as a separate grazing unit or (2) controlling cattle use through herding or by providing alternative sites for watering and loafing. Grazing management plans for riparian zones could include (1) limiting the time during which grazing is allowed, (2) limiting the density of cattle, or (3) excluding cattle during periods when streambanks are most susceptible to damage (Chaney and others, 1990).

Alternative Watering and Loafing Areas

Placing areas for livestock watering, salting, and shade away from streams and other sensitive habitats can more evenly distribute activity across the pasture and might be effective in reducing livestock activity in riparian-stream zones to acceptable levels in some instances without the need for fencing. The strategic deployment of watering troughs for this purpose is recommended by U.S. Environmental Protection Agency (2003).

Stream Crossings

Hardened structures allow passage with minimal disturbance to the stream and associated riparian zone (U.S. Environmental Protection Agency, 2003). The purpose is to provide a controlled crossing or watering access point for livestock, thereby controlling bank and streambed erosion, reducing sedimentation, and enhancing water quality.

Upland Areas

While grazing impacts to riparian zones have the most obvious influence on stream water quality, an effective strategy for reducing nutrient inputs from grazed lands must also consider upland management practices as well. Indeed, the condition of the riparian zone is dependent upon the proper functioning of adjacent upland areas with respect to vegetative cover and hydrologic processes such as infiltration and overland flow that affects sediment and nutrient transport (Elmore, 1992; U.S. Environmental Protection Agency, 2003). Furthermore, the paradigm of the riparian zones as a land-water buffer and, therefore, a regulator of upland influences on streams may not apply to irrigated lands where return flows or runoff (and associated sediment and nutrients) from upland pastures enter the stream directly through culverts, ditches, or other conduits. With respect to water quality, the condition and management of upland grazed areas typically focus on water (and soil) retention capacity and the spatial distribution of soil nutrients.

Irrigation Management

Flood and furrow irrigation have been identified as major sources of sedimentation in the Yakima River Basin (Lieb and others, 2002, 2004). The Washington Department of Ecology has set a sediment limit for irrigation return flows of 25 nephelometric turbidity units (56 mg/L). Experiments indicate that applying polyacrylamide to furrows, and having grass-lined ditches successfully maintains irrigation return flows below the sediment limit. The costs for polyacrylamide were $50/hectare (ha)/yr.

Perennial-grass seeding of ditches was estimated at $60/ha. Many farmers in the Yakima River Basin are converting to sprinkler or drip irrigation systems that are considerably more costly than either flood or furrow irrigation methods. However, grass-lined ditches may be an effective sediment reduction (total P source) method in the Klamath Basin. The estimated effectiveness of grass-lined ditches alone was up to 45% (U.S.D.A. Soil Conservation Service, 1992).

Animal Manure as a Source of Phosphorus in the Watershed

Solid waste is usually a mixture of dung, urine, and undigested vegetable matter. Cattle deposit wastes throughout the pasture in a variable fashion, but a greater density of wastes may accumulate near watering and feeding locations, or in loafing areas such as the riparian zones near streams where vegetation is plentiful and shade is available. Inorganic and organic P is present in significant quantities in animal manure (Chadwick and Chen, 2002). Some organic P compounds are relatively stable and mobile in soils while others may be quickly sequestered in clay minerals within the soil and accumulate in soil over time. The potential for surface water contamination from animal manure is seasonal and varies with irrigation practices, stocking densities, and meteorology (storm events). Livestock that receive supplemental feed in the form of grain may produce manure high in organic matter. If large quantities of organic matter enter watercourses adjacent to pastures through irrigation or storm event runoff, then they can create a large oxygen demand that can adversely impact aquatic life forms in the streams. Stream buffer strips that exclude cattle from close proximity to streams are one mechanism to prevent animal manures from contaminating watercourses.

Summary of Best Management Practices for Nutrient Loading Reduction

Table 1 compiles examples of BMPs for nutrient loading reduction throughout the United States. Again, the primary focus is on practices applied to pasture or grazing land. In general, BMPs were successful in reducing nutrient concentrations or loading; however, there were some exceptions.

Role of Wetlands in Watershed Phosphorus Management

Wetlands perform a number of hydrologic and water quality functions that reduce watershed P loading and associated water quality impacts. Wetlands serve as hydrologic buffers that store and slowly release water during high-flow events, thus attenuating peak discharges and associated scouring that increases stream sediment loads. The reduced velocity of water spreading across the wetland surface and the hydraulic resistance afforded by wetland vegetation promote settling of suspended sediment and its associated P that would otherwise be transported downstream. Wetlands also provide sinks for dissolved P, which is removed through various chemical and biological pathways. While the majority of the dissolved P entering a wetland may eventually be exported downstream (as opposed to entering long-term storage in the soil), the bioavailability of this P is reduced through the conversion of reactive forms such as orthophosphate into more refractory organic P.

Various types of natural and created wetlands have been used successfully to reduce nutrient loading from both point discharges and nonpoint sources to downstream waters (U.S. Environmental Protection Agency, 1993b, Mitsch, 1994, Kadlec and Knight, 1996). Wetlands are generally considered more to be more efficient at reducing nitrogen loading than at achieving significant P removal. However, in a recent review of data from 57 natural wetlands around the world, Fisher and Acreman (2004) found that 84% of these systems exhibited net total P removal ranging from 5to100% (table 1). These findings indicate that P retention is a robust property of wetland ecosystems as they encompass several wetland types (marshes, swamps, and riparian wetlands) in many different environmental settings. Wetlands are also known to exhibit a finite capacity for sustained P removal, and this capacity is easily diminished when these systems are exposed to excessive P loading rates (Richardson, 1985).

Table 1. Summary of literature review of BMP effectiveness in reducing phosphorus and nitrogen nutrients.

Source	BMP Type Implemented	P reduction, in percent except as noted	N reduction, in percent
Anderson, 1998	Temporary fencing	30	
Anderson, 1998	Permanent fencing	40	
Anderson, 1998	Permanent fencing, Riparian restoration	50	
Braskerud, 2002	Constructed wetlands	21–44	
Clausen and Meals, 1989	Filter strip	86	
Cooper and Gilliam, 1984	Forest buffer strip	50	
Coveney and others, 2001	Constructed wetlands	30–67	
Dillaha, and others, 1989	Grass buffer strip	61–79	54–73
Doyle and others, 1977	Forest buffer strip		98
Fink and Mitsch, 2004	Constructed wetlands	28–59	
Fisher and Acreman, 2004	Natural wetlands	5–100	
Gearhart, and others, 1995	Wetlands	10–40	
Gearhart, and others, 1995	Riparian fencing	20–40	
Graham and others, 2005	Lakeshore wetlands	0.4–0.45 $g/m^2/yr$	
Inamdar and others, 2001	Combined Strip cropping, Low till, and Filter strips	4	26
Meals, 1993	Combined Manure storage, Barnyard runoff, Milkhouse waste management, and Erosion control	In watershed, 2–44 4–32	
Meals, 1996	Vegetated filter strip	89	92
Meals, 2002	Riparian fencing	12–34	
Meals, 2004	Riparian exclosures, Streambank revegetation, Hardened stream crossings	21–41	
Meals and Hopkins, 2002	Combined Exclosure and fencing	21–41	
Mitsch, 1994	Natural and constructed Wetlands	1–4 $g/m^2/yr$	
Mitsch and others, 1995	Constructed wetlands	0.5–3.0 $g/m^2/yr$	
Osborne and Kovacic, 1993	Forest buffer strip	50–85	79–98
Peterjohn and Correll, 1984	Forest buffer strip	74–85	79
Plunkett and others, 2002	Constructed wetlands	2	
U.S. Department of Agriculture, 1992	Grass-lined ditches	45	
Vache and others, 2002	No till	54–75	54–75
Winegar, 1997	Stream channel fencing	Total suspended solids, 48–79	
Young and others, 1980	Grass buffer strip	83	84

Thus, rather than viewing wetland restoration and creation as a panacea for watershed P-load reduction, these efforts should be planned in conjunction with land management strategies discussed elsewhere in this report that are aimed at source reductions.

Historical Distribution of Wetlands in the Upper Klamath Lake Watershed

Historical records and reconstructions indicate that wetlands were a significant landscape feature of the predisturbance UKL watershed. Tule (*Scirpus acutus*) marshes constituted a significant fraction of the historic acreage of UKL and undoubtedly regulated several aspects of the lake's water quality including the P concentration. In particular, wetlands forming the Williamson and Wood River deltas would have acted as filters at higher river flows, trapping sediments and nutrients before they entered the lake. Soil survey maps for the Sprague River Valley delineate an extensive historic floodplain that undoubtedly contained significant wetland acreage. The collective effect of this extensive floodplain system would have been to slow the rate of transport of sediments and nutrients downstream to UKL.

Approximately 85–90% of the wetland acreage in the Upper Klamath Basin is believed to have been lost or severely degraded since the mid 1800s, primarily due to the development of farming and ranching operations (Gearheart and others, 1995b). Roughly 22,000 acres of wetland adjacent to UKL have been lost, primarily to agricultural uses (Carlson, 1993).

Riparian and floodplain habitats in the Sprague and Wood River watersheds have been lost or degraded as a result of ditching and diking to promote drainage and prevent overbank flows. These dramatic losses accompanied by changes in river and floodplain function have undoubtedly accelerated transport and reduced storage of P in UKL tributary basins, particularly the Sprague and Wood River Valleys, and reduced removal of P within the lake itself. Watershed-scale wetland losses such as this are believed to have contributed to the cultural eutrophication of other large, shallow lakes such as Lake Erie (Mitsch and Reeder, 1992).

The magnitude of wetland loss in the Upper Klamath Basin has undoubtedly had a significant impact on P transport and retention in the watershed. For example, Graham and others (2005) estimated average P storage rates in remnant lakeshore wetlands of 0.40–0.45 g/m2/yr (table 1). These figures coupled with acreage loss estimates for this wetland type (Carlson, 1993) indicate the loss of some 40 metric tons of P storage capacity each year. It is important to recognize that drained wetlands not only lose their storage function but may even become P sources as a result of both natural (organic matter oxidation) and human (fertilization) processes. Snyder and Morace (1997) estimated that lakeshore wetlands drained for farming have contributed a maximum of more than 4,000 metric tons of P to the lake since drainage, due in part to the oxidation of the peat soils. We are not aware of any information that could be used to do a comparable assessment of the effects of wetland loss on P fluxes elsewhere in the watershed.

Given the potential significance of wetlands to P retention in the UKL watershed, wetland restoration offers a potential means of reducing P loads to UKL. For example, Gearheart and others (1995b) considered the reestablishment of wetlands on agricultural lands adjacent to UKL 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. Use of wetlands elsewhere in the watershed to treat nonpoint source agricultural pollution was ranked as moderately effective for P load reductions by these investigators. Beyond their role in P storage, lakeshore wetlands likely serve as sources of dissolved organic carbon to UKL. Perdue and others (1981) observed that Klamath Marsh waters rich in humic acids inhibited the growth of

Aphanizomenon flos-aquae blooms, which are responsible for water quality impairments in UKL. It has since been suggested that restored tule marshes around UKL may thus serve to retard the onset of these blooms (Geiger, 2001), although this hypothesis has yet to be rigorously tested.

General Wetland Design Considerations

The concept of restoring wetland habitat within the Upper Klamath Basin to promote watershed P storage and other functions has been proposed by various stakeholders and is supported by available scientific evidence already presented. However, successful implementation of this watershed restoration strategy will require careful planning and project siting, especially for projects where P removal is the principal goal. Wetland creation and restoration remains an inexact science as evidenced by the unpredictable outcomes for many such projects (Zedler and Callaway, 1999). While the creation of a wet area containing indicator wetland plant species is a relatively straightforward process, efforts to restore specific ecosystem functions such as P retention have frequently proven more challenging.

Although standardized methodologies for wetland creation remain elusive, certain guiding principals can be used to increase the likelihood of successfully constructing a functional wetland ecosystem. First, and perhaps not surprisingly, there is growing evidence that restoration of a degraded or destroyed wetland is more likely to yield success than attempts to create a wetland where none previously existed (Hammer, 1992). For example, a recent study by the U.S. Geological Survey (Hunt and others, 1999) found that both the construction time and costs for a wetland restoration project were considerably less than that for an equivalent wetland creation effort and that the restored site, once established, was more likely to be delineated as a wetland. Secondly, the characteristics of natural wetlands in the same watershed and geomorphic setting should be used to guide restoration designs and expectations of attainable functions. For example, estimates of annual P storage rates in remnant wetlands adjacent to UKL (Graham and others, 2005) can be used to gauge expectations for long-term P removal in restored lakeshore wetlands. Finally, the concept of “self design” (Mitsch, 1998) emphasizes the importance of facilitating natural processes of wetland development (vegetation colonization) and minimizes more intensive and costly human intervention (extensive vegetation plantings). Certainly, engineering and construction to achieve a suitable hydrology (such as by dike removal or grading to restore historic contours and flow paths) and the presence of a suitable seedbank or other plant propagule source are necessary for successful wetland creation. However, the additive benefits of more intensive intervention are increasingly being questioned. Exceptions to this paradigm are discussed by Streever and Zedler (2000).

In the UKL watershed, the “River Bend” floodplain restoration project provides a good example of rapid wetland recovery following restoration of historic topography to restore seasonal flooding to a portion of the lower Williamson River floodplain. While some active planting of *Scirpus* and *Typha* was conducted during project construction, rapid natural establishment of submerged aquatic vegetation beds at lower elevations (long hydroperiod) and emergent vegetation (in particular *Typha* and, to a more limited extent, *Scirpus*) was observed during the first growing season following project construction (McCormick, pers. obs.). Whereas only two species (*Typha* and *Scirpus*) were actively planted in limited quantities during the construction process, a plant survey one year later found more than two dozen obligate and facultative wetland taxa (not including submerged aquatic vegetation) at the site (McCormick, unpub. data, 2005). The coverage of *Typha* had increased dramatically, and this species was abundant even in locations where it had not been planted, suggesting that active planting of this species provided no additional benefit to wetland development.

Phosphorus Removal Processes in Wetlands

An understanding of P removal processes and their limits is critical when one designs wetlands for this purpose and developing performance expectations. Three processes contribute to P removal in wetlands:

(1) Physical settling of sediments and associated P due to reduced water velocity and vegetative resistance to flow;
(2) Chemical sorption and precipitation reactions whereby dissolved inorganic phosphorus forms complexes of varying stability with iron, aluminum, and calcium under appropriate conditions;
(3) Biological uptake of dissolved inorganic phosphorus

The importance of these three processes and the overall efficiency of wetland P removal are strongly affected by inflow P composition. Wetlands that receive large volumes of sediment typically exhibit high removal rates since much of the incoming P is in particulate (sediment-bound) fractions that are retained by physical settling and burial. The capacity of wetlands to exhibit sustained removal of dissolved P through chemical and biological means is more limited. In fact, wetlands may be net sinks or sources of P depending on their chemical and biological characteristics (Richardson 1985). Removal rates can exhibit seasonality in temperate climates, and they are susceptible to stochastic events such as flooding and droughts that can promote the release of stored P to downstream ecosystems. Constructed wetland design and predictions concerning P removal capacity require an understanding of the dominant P removal processes expected to occur in a particular wetland. For example, restored floodplain wetlands in the Sprague River watershed would likely function as traps for P-rich sediments, whereas restored lacustrine wetlands around UKL would potentially remove both dissolved and particulate (algal) P from lake water through plant uptake and settling, respectively.

Newly constructed wetlands can exhibit rapid changes in P removal rates due to transient processes such as flushing of P from newly flooded soils, saturation of various short-term P storage pools, and the increase in vegetative biomass (a major P pool) to some sustainable level. Short-term performance is not necessarily indicative of, and often exceeds, long-term removal rates. Soil processes and uptake by algae and submersed vegetation, which often establish quickly following flooding, may account for high initial P removal rates. However, these storage pools are finite and are quickly saturated. Rooted macrophytes establish more slowly and typically provide more sustainable P removal. Vegetation removes dissolved inorganic phosphorus from the soil porewater and, in many species, from the surface water as well through adventitious roots or foliar uptake. Upon senescence, a portion of this P is retained either in below-ground plant structures or in dead above-ground biomass, which is eventually incorporated into the soil matrix. Net P retention through these processes of translocation and organic matter storage is typically quite low. However, it is this incremental accumulation of new soil that constitutes the major mechanism of long-term P removal in many natural and created wetlands (Kadlec and Knight, 1996).

Factors Affecting Phosphorus Removal

Several factors related to wetland location and design must be carefully considered if P removal is the primary goal of a wetland restoration or creation project. Detailed guidance on design and operational parameters for various types of treatment wetlands is provided in several textbooks (Hammer, 1992; Marble, 1992; Kadlec and Knight, 1996). We limit our discussion to the more common aspects of wetland design that affect P removal.

Position Within the Watershed

Wetlands should be located within the watershed to maximize the capture of P loads. Wetland restoration is implemented for various purposes (improved habitat value, flood control, water quality improvement) and project locations best suited for one purpose may not be optimal for meeting other objectives. Crumpton (2001) noted that most wetland restoration efforts in the Midwest corn belt have emphasized creation of wildlife habitat rather than water quality improvements and that the placement of these projects is such that the wetlands receive only 4% of the agricultural drainage entering streams within the watershed. He found that this same wetland acreage, if strategically located within the watershed, could capture 50% of the total drainage and produce significantly greater reductions in downstream nutrient loading.

Antecedent Soil Conditions

Soil concentrations of P and P-binding metals can strongly affect the capacity of newly created wetlands to function as P sinks. Fine mineral sediments such as those found in alluvial deposits are especially conducive to P sorption (Marble, 1992). The occurrence of these soil types can be delineated using Natural Resources Conservation Service soil maps. Newly flooded soils high in P (such as fallow agricultural lands) have little excess P sorption capacity and, in fact, may temporarily release P to the water following flooding. For example, wetlands in the process of being restored along the shores of UKL have peat soils rich in P as a result of decades of fertilization and soil oxidation, and they will likely function as a temporary source of P to UKL if and when they are reconnected to the lake. Several factors can affect the magnitude and duration of this residual P flux, including water residence time, the establishment of rooted vegetation, and the rate at which new soil accumulates and buries the existing P-rich layer. For example, Robinson and Reddy (1998) estimated P flux rates of almost 1 $mg/m^2/d$ for flooded agricultural soils in central Florida and predicted that restored wetlands on these sites could function as P sources for between 5 and >30 years depending upon the soil depth increment contributing to these fluxes. By contrast, wetlands reestablished on farmed peat soils in south Florida for the sole purpose of P removal exhibited net P storage within a matter of months to a few years (Newman and Pietro, 2001). Thus, it should not be assumed that wetlands reestablished on agricultural land cannot be used for P removal.

Inflow Phosphorus Conditions

The composition and rate of P inputs strongly affect the rate and efficiency of P retention. Sedimentation is an effective means of particulate P removal, and thus wetlands receiving high sediment loads typically exhibit high P removal rates. Retention of dissolved P is considerably less efficient than for particulate P due to biological cycling and the reversibility of some chemical reactions that bind dissolved inorganic phosphorus. Wetlands need to be sized to handle the anticipated P load to avoid P saturation and eventual export. Incorporating a sedimentation basin at the upstream end of the wetland can greatly increase its treatment life in situations where high sediment loads are anticipated. Increases in P loading rates beyond certain limits result in reduced efficiency of P removal. However, higher loading rates can increase absolute P load reductions, which is often a more important management goal than removal efficiency.

Hydrologic Conditions

Hydrologic conditions exert numerous effects on wetland P removal. The hydraulic loading rate determines both the P loading rate and, in conjunction with the hydraulic residence time, affects the rate and efficiency of P removal. Increased hydraulic loading rates and shortened retention times reduce the efficiency of P removal, but they can increase the absolute amount of P removed by the wetland up to the point where excessive loads begin to resuspend sediment and damage vegetation. With respect to the hydraulic path of the water, wetland design and management should promote sheet flow—an even spreading of water across the wetland surface. This condition minimizes current velocity (increased particulate settling) and maximizes contact between the water and wetland biota (increased dissolved P uptake). The presence of preferred flow paths (old drainage ditches that parallel the direction of flow) promotes "short-circuiting" of water through the wetland and results in a substantial reduction in P-removal performance. Increased hydroperiod, (the annual duration of flooding), promotes the accumulation of P as plant detritus in wetland soils, but it may also retard the consolidation of fine flocculent material into the soil matrix. Increased water depths reduce the exposure of incoming water to the soil and plant processes that contribute to P removal and also promote higher current velocities, which reduced particulate settling. If maintained for extended periods, deep-water conditions will impair emergent macrophyte growth and survival and associated P retention.

Examples of Wetlands Designed to Treat Nonpoint-Source Pollution Loads

In contrast to the widespread use of wetlands for wastewater treatment, relatively few published studies have investigated the efficiency with which wetlands reduce nonpoint-source P loading. We limit our discussion here to examples that provide quantitative estimates of P retention and are relevant to potential wetland restoration or creation scenarios in the Upper Klamath Basin.

The creation of floodplain wetlands has been a major focus of watershed nutrient management efforts given the capacity of these systems to entrain large volumes of sediment-bound P. Mitsch (1994) compared the performance three wetlands, two natural and one constructed and found similar rates of P retention (1–4 $g/m^2/yr$) despite differences in location, hydrology, and vegetation (table 1). The constructed wetland, which was designed specifically to treat nonpoint-source pollution, exhibited a greater percent retention of P. However, this comparison illustrates that floodplain wetlands generally act as P sinks, due in large part to the settling of suspended river sediments (particulate P removal). These findings are consistent with a survey of many wetlands worldwide conducted by Fisher and Acreman (2004), who concluded that riparian wetlands usually function as sinks for total P while exhibiting net export of dissolved P in many cases.

In agricultural watersheds in Norway, wetlands were created along first-order streams to trap high P sediments from fields used for grain production and dairy cattle operations (Braskerud, 2002). Wetlands were constructed by expanding the stream banks and installing low dams at the downstream end. Aquatic vegetation was either planted or allowed to colonize naturally. Inflow total P concentrations averaged between 170–430 µg/L, compared with average outflows of 100–270 µg/L, and all wetlands produced concentration reductions. Retention of total P, which was generally associated with suspended sediments, averaged 21–44% of inputs or 26–71 g/m^2 area of wetland/yr (table 1). The extremely high storage rates are likely due to the importance of sediment trapping in these watersheds as opposed to the uptake and storage of dissolved P.

Riparian wetlands were constructed to treat waters pumped from a Midwestern river in a watershed dominated by agriculture (Mitsch and others, 1995). Wetland basins ranged in size between 1.9 and 3.4 ha and were colonized primarily by *Typha* spp. following flooding. Pumping began in 1989 at different intensities to test for effects of different flow rates on P removal. Both wetland types exhibited consistent P removal during the 3-yr study period, with low flow wetland eventually exhibiting greater concentration reductions than high flow systems. Intensive sampling during year 3, for example, documented average inflow total P concentrations of 176 µg/L and average outflows of 34 and 45 µg/L for low flow and high flow wetlands, respectively. Phosphorus removal ranged between 0.5–3 $g/m^2/yr$ during the study period, and sediment trapping was identified as the major mechanism of P removal (table 1). These wetlands also functioned as P transformers, with bioavailable P (dissolved inorganic phosphorus) constituting as much as 50% of the incoming P load but no more than 10% at the discharge. These authors compared their findings with the analysis of Mitsch (1994) described earlier to further reinforce the "rule of thumb" P removal rate range of 1 and 4 g/m^2/yr for floodplain wetlands in agricultural watersheds.

Further evidence of the efficacy of P removal using created floodplain wetlands was provided by Fink and Mitsch (2004), who tracked the performance of a 1.2 ha constructed wetland receiving nutrient-rich surface and ground-water inputs from a 17 ha agricultural watershed in the Ohio River Basin. Reductions in the total P concentration of water passed through the wetland averaged 59% during the first 2 years of operation (table 1). Removal efficiency was lowest during high flows generated by precipitation events, but 28% of total P was still retained on a mass basis during these periods. On an area basis, load reductions for this newly constructed wetland were estimated to be 6.2 g/m^2/yr, a rate which is consistent with those found in previous studies.

The potential for constructed lakeshore wetlands to restore hypereutrophic lakes is illustrated by the efforts of the St. Johns Water Management District to restore Lake Apopka, a shallow (average depth, 1.7 m), 124 km^2 lake in central Florida. This lake and its associated fishery had been in decline since the 1920s as a result of excessive nutrient loading from agricultural and other sources. As part of a concerted effort to reduce external P loads and in-lake P stores, the water management district proposed the creation of a flow-through treatment wetland on muck farmland (historically sawgrass marsh) adjacent to the lake (Coveney and others, 2001). A 660-acre demonstration project consisting of four treatment cells receiving an average of 97 million gallons per day, became operational in 2003. Total P in the inflow from Lake Apopka ranged from approximately 120 and 230 μg/L and hydraulic loading rate varied from 6.5 to 42 m/yr.

The major mechanism of P removal in these cells is the settling of particulate matter, primarily phytoplankton. Water is circulated through the wetland and back to the lake with the goal of maximizing the rate rather than the efficiency of P removal. Following an initial 2-mo flushing phase resulting in net export of P, the wetland began removing P at a rate of 10 lb/d with a mass removal efficiency ranging from roughly 30–67% (table 1). Modeling was used to predict a maximal rate of P removal of 4 g /m^2/yr at a P loading 10–15 g /m^2/yr and a hydraulic loading rate of 60–90 m/yr.

Two important issues associated with the operation of this wetland include (1) the need for periodic draw-downs to consolidate the flocculent material (primarily settled plankton) that rapidly accumulated in the wetland cells in order to maintain P removal efficiency, and (2) incidents of waterfowl kills in proximity to the wetland creation sites, which were attributed to pesticide residues from past farming activities on this land. The issue of legacy contaminants has been carefully considered in major wetlands restoration efforts in the Upper Klamath Basin (The Nature Conservancy's Williamson River Delta Project).

A 3-year study was conducted to assess the efficacy of converting farm drains to wetland habitat to reduce agricultural nutrient loads near their source (Plunkett and others 2002). Reductions in particulate P accounted for most P removal, indicating that the presence of vegetation slowed the flow of water through the drain and promoted settling of suspended matter. However, this material accumulated on the drain bottom as flocculent sediment that was highly susceptible to resuspension at higher flows. Therefore, these constructed systems tended to act as P sinks during low-flow events and as P sources during high flow events. Consequently total P load reductions averaged only 2% during the study period (table 1). The investigators concluded that the inefficiency of these "in-line" wetlands as P nutrient sinks was due to the inability to control flow rates, and they recommended either "off-line" wetlands (separate from the drains and with independent hydrologic controls) or increased farm reuse of drainage water as more effective strategies for reducing agricultural nutrient loads.

Examples of Buffer Strips, Streambank Fencing, and Other Best Management Practices Designed to Treat Nonpoint Source Phosphorus Loads

Clausen and Meals (1989) studies involved paired watersheds in Vermont, the LaPlatte River and St. Albans Bay. Water quality monitoring was performed over a 7-year period from 1981 to 1986. The watersheds studied had multiple land uses that included agriculture and forests. Agricultural use included row crops and intense pasture for dairy farms. BMPs implemented in these watersheds consisted of undefined "filter strips." This summary article cited previous reports on both watersheds that could provide more explicit information on the type of vegetation, dimensions of, and other details of the filter strips. In another article (Meals, 1996), these two watersheds are also discussed and the filter strips are referred to as "vegetative filter strips." Smith (1992) reports on a variety of grasses used in vegetative filter strips and various widths in a guideline for minimum widths based on field slopes. Clausen and Meals (1989) reported P reductions of 86% in milkhouse filter strips (table 1). However, they also reported that streams still exceeded water quality criteria at certain times of years (low flow) or during runoff events (high flows). The critical controlling component in filter strips is laminar flow that spreads water slowly over a large vegetated area allowing sediment to be retained and infiltration of water into the soil to occur.

Meals (1993) again discusses one of the same Vermont watersheds, the LaPlatte River. A combination of BMPs was implemented in several subwatersheds within this basin over a period of 11 years from 1979 to 1989. BMPs included manure storage, barnyard runoff treatments by filter strips, milkhouse waste management, and erosion control. Two of the subwatersheds were compared using a statistical technique to control for hydrologic variability. In general, P reductions ranged from 32 to 44% (table 1); however, post-BMP phosphorus export was higher than expected during extreme runoff events. In addition, land-use changed and the numbers of animal units changed during the study. It appeared to the author that a high level of land-owner participation in BMP implementation may be required before significant reductions can be achieved.

Osborne and Kovacic (1993) presented a review of literature on riparian vegetated buffer strips for use in stream water quality restoration. They also presented results of investigations on the effectiveness of forested and grass-vegetated buffer strips in reducing shallow subsurface inputs of nutrients from agriculture to a stream. Their findings indicated that both forested and grass vegetated buffer strips reduced nitrogen concentrations in shallow ground water (79 - 98%, table 1). The forested vegetated buffer strips were dominated by mature cottonwoods and silver maple and were approximately 16 m wide. The grass vegetated buffer strips were reed canary grass and were approximately 39 m wide. They also report on use of rye grass and oats as grass types for use in vegetated buffer strips. Rye grass in both 10-m- and 20-m-wide plots seemed to be more effective in reducing total P concentrations in surface runoff than oats. While vegetated buffer strips do reduce nutrient loading during the growing season, when plants are dormant, they can act as nutrient sources, releasing accumulated nutrients to adjacent streams. Periodic harvesting of plant biomass may be required to reduce dormant season loading of P.

Osborne and Kovacic (1993) also compiled results of other authors listed in table 1 who reported quantitative results for both forest and grass buffer strips (Doyle and others, 1977; Peterjohn and Correll, 1984; Dillaha and others, 1989; Young and others 1980; and Cooper and Gilliam 1987). The effectiveness of vegetated buffer strips in reducing nitrogen and P ranged from 54 to 98% for nitrogen and 50 - 85% for P. Although specific vegetation types were not detailed by Osborne and Kovacic (1993) the overall width of the forest or vegetated filter strips was often included. The width of forested buffer strips ranged from 10 to 50 m; that of vegetated buffer strips from 5 27 m. Despite their overall effectiveness at reducing nitrogen and P concentrations in both surface and near-surface water draining into streams, the authors still cite a lack of information on the long-term effectiveness of vegetated buffer strips as treatments for nutrient reduction.

Gearheart and others (1995a) in an extensive report entitled "Watershed Strategies for Improving Water Quality in Upper Klamath Lake, Oregon" made recommendations for developing BMP strategies to address constructing or restoring wetlands to treat nonpoint and drainage pumpdown sources, implementing riparian fencing measures, restoring streambanks, and removing roads as an erosion-control measure. The authors estimated a 54% reduction in total P to UKL if all BMPs were implemented at their maximum potential level. The range reported was from 21 to 54% depending on the percentage of maximum potential levels. The authors did break out each of their BMPs separately and, of the potential maximum of 54% reduction, wetlands and riparian fencing were both highly ranked as BMPs for P reduction. Some caution must be used in comparing results from this report to results obtained by other authors, since the total contribution of each BMP to the in-lake total P is not clear. It is apparent that in the watersheds, particularly the Sprague River Valley watershed, that riparian fencing, streambank restoration, and constructed wetlands were the BMPs recommended to reduce P loading to UKL.

Meals (1996), again reporting on the LaPlatte and St. Albans Bay watersheds in Vermont, cites reductions of 89% in P and 92% in nitrogen on an annual basis for vegetated filter strips. However, these percentages represented measurements at the “edge of the field.” Instream measurements did not show significant decreases in P concentrations. Before adjusted for background increases, P concentrations in streams increased 0.0006 - 0.010 mg/L/yr during the study period. After adjusting for background increases, a P reduction of 26 - 44% was estimated after BMP implementation.

Anderson (1998) applied a management model (WKSLKMOD) to determine the best watershed management strategies for reducing UKL total P concentration. His thesis concluded that the "... control of total P loading from agricultural land use, grazing land use, and waste water treatment plant sources appears to be more cost effective and beneficial..." than controlling total P loading from mined, urban, or septic tank system sources. The author further concludes that riparian fencing in all subbasins should occur as "riparian fencing provided the second highest total P load removal of any management practice." The amount of existing grazed land that would need to be fenced is approximately 7% basinwide.

Meals (2002) reported on another Vermont watershed, the Champlain Valley. Lake Champlain Basin was an Environmental Protection Agency National Monitoring Program special nonpoint-source control monitoring site for a 7-year project completed in 2001. The project demonstrated that implementation of simple and inexpensive pollution control measures can yield significant improvements in water quality. During the project, livestock access to streams was identified as the major source of elevated phosphorus concentration, bacteria, and organic matter. The BMPs implemented throughout the watershed were livestock exclusion, riparian restoration and bioengineered streambank protection. Approximately 30 - 50% of pasture riparian zones were restored in the treatment watersheds, and livestock were completely excluded from streams. In addition, streambanks were protected using tree revetments and willow plantings. The total cost of these treatments was $40,000. A rapid recovery was documented in the Champlain Valley. Just 3 years after BMP implementation, the streambanks were completely stabilized and sections of the streams became narrower and deeper, offering better fish habitat. Growth of grasses, shrubs, and willows in the stream buffer increased after grazing pressure was removed and overall sediment loading to the streams decreased. Water sampling results indicated that average phosphorus, nitrogen, and sediment export from the watershed decreased by 12 - 34%, and bacteria counts were reduced by 30 - 40% compared to pre-BMP treatment levels. The stream protection measures kept nearly 1 ton of P, 2 tons of N, and 126 tons of sediment out of the stream each year.

Meals and Hopkins (2002) also discuss the Lake Champlain Basin studies. In this article, the authors are more explicit regarding the details of the BMPs implemented in the watershed. For example, in one area 450 m of fencing was installed, creating 2-5 m of protected riparian zone and eliminating three livestock crossing areas. In another area, 2,300 m of stream was fenced on both sides to exclude livestock, a livestock bridge was installed, a 300 m stabilized livestock travel lane was created, and three culvert and two armored stream crossings for livestock were installed. The protected riparian zones in this instance were from 2 to 8 m wide. Livestock watering systems were installed on two farms to replace stream access. In the protected riparian zone, streambanks were stabilized using tree revetments and brushrolls. The tree revetments used whole coniferous trees secured to exposed streambanks. Brushrolls consisted of cylindrical bundles (0.5 m by 3-5 m) of woody vegetation such as alder, staked into exposed banks. In some locations, willow plantings were done, but those areas were small. Again, the total cost for these BMPs was $40,000. When the two treatment areas were compared, one area experienced a reduction in total P per year of 3,600 kg or 68%. The other treatment area experienced a smaller reduction in total P per year of 375 kg or 16%. The treatment area with the smaller reduction in total P/yr also had a lower participation in implemented BMPs. Repeatedly throughout the literature, the need for having full participation by landowners with critical riparian zones in the program has been reinforced. In addition, the hydrologic cycle and changes that include high runoff events can impact results. Overall, in these studies, total P concentrations were reduced by 21% and total P export from the watershed was reduced by 21 - 41%.

Inamdar and others (2001) reported on BMP impact on sediment and nutrient yields for the Nomini Creek watershed in Virginia. Land use was primarily row crops; however, the BMPs included vegetated filter strips. The results of a 6-year study from 1986 to 1990 included both pre- and post-BMP implementation monitoring. The authors report that nitrogen was reduced by 26% and total P by 4%. Although total particulate P was reduced by 30%, soluble P increased by 92%.

Vaché and others (2002) used a water-quality model to evaluate water quality under three alternative land-use scenarios for two agricultural watersheds in Iowa. The land use categories included forest, grassland, row crops, pasture, riparian zones, wetlands, ponds, set-aside conservation areas, and fencerows. The SWAT model simulations varied the percentage of land use among these categories to determine the potential for BMPs to affect surface water discharge and sediment and nutrient loading in these watersheds. Their findings indicate that nutrient loads could be reduced by 54 - 75%. BMPs included riparian buffers, engineered wetlands, grassed waterways, filter strips and field borders.

Regional Best Management Practice Examples

Regional examples of BMPs that have been implemented include those from the Feather River, Stone Creek Nursery, and the Arcata Marsh. The Stone Creek Nursery wetland was constructed in 1999 by the Bureau of Reclamation, the Rogue River Valley and Medford irrigation districts, the Rogue Valley Council of Governments and the J.H. Stone Nursery (Steinfeld, 2001). The Bureau of Reclamation designed the constructed wetland and provided funding, and the irrigation districts constructed the wetlands and were supposed to provide a water right for operation. The Rogue Valley Council of Governments provided planting, monitoring, and educational outreach assistance, while the J.H. Stone Nursery donated land, produced the wetland plant seedlings, completed the environmental documents, and maintained the area over time. Although the wetland was successfully established and continues to be maintained, the water right has never been provided. The original concept was that the wetland would reduce sediment, nutrients, and pesticides from a portion of the water from Jackson Creek, control sediment produced from nursery activities, provide methods and material for growing native wetland plants from seed, establish studies and monitoring projects to evaluate plant growth, water quality, and effectiveness of the wetland system, and develop myriad educational outreach opportunities for the community. Since the water right has not been provided, only some of these components have been successful. The wetland, located near Medford, Ore., successfully treats the irrigation return flow waters for the Stone Creek Nursery in terms of reducing sediment concentrations. Some educational outreach has been accomplished. However, the water quality monitoring and effectiveness of wetland functioning assessments have not been performed because the water right to divert water from Jackson Creek through the wetland has not been accomplished.

The Feather River Coordinated Resource Management Group has a history of watershed restoration that spans nearly 20 years, from 1985 to the present (Feather River Coordinated Resource Management Group, 1996a). It cites 39 watershed projects in the Feather River watershed (Plumas County, Calif.) for a variety of resource management activities including erosion control, stream restoration, acid mine drainage, fuels reduction, and educational outreach activities. The primary effort has been erosion control and stream restoration, although two reservoirs have been impacted by sediment infill that has reduced reservoir capacity (Rock Creek and Cresta Reservoirs). The consortium of partners for the Feather River Coordinated Resource Management Group includes these Federal agencies: the Army Corps of Engineers, the Natural Resource Conservation Service, U.S.D.A. Forest Service, and the U.S. Fish and Wildlife Service. California state agencies include the Departments of Transportation, Forestry and Fire Protection, Fish and Game, and Water Resources, and the Water Quality Control Board. There are also several universities and private sector companies. Three projects that have been completed are the Greenhorn Creek Trout Enhancement, the Wolf Creek Restoration, and the Red Clover Creek Demonstration Projects.

The Greenhorn Creek Trout Enhancement Project was completed in 1992 (Feather River Coordinated Resource Management Group, 1996c). The project goals were erosion control and stream restoration to provide better brown trout habitat. Total project costs were $406,050. In a flood plain of 17.6 acres, 2,800 feet of stream were restored. Cattle were excluded for 3 years after planting and limited grazing was allowed thereafter. Monitoring of stream and biological responses has been performed. The stream particle sizes altered significantly, with sands and fines being reduced from 30.7% to 4.7% of the stream bottom after restoration construction was completed. The stream width was reduced to 30' from 50 to 275'. Meander reconstruction subsequently failed in up to 50% of the stream reaches during three major flood events in 1995, but no stream bank erosion occurred. Revegetation success was excellent, but at the time of the fact sheet, no change in bird or insect populations had been observed, although some juvenile brown trout had been seen in the project area.

The Wolf Creek Restoration Project was completed in 1992 (Feather River Coordinated Resource Management Group, 1996d). The project goals were erosion control and water quality and stream restoration. The length of stream in the project area was 9,636'. The total costs for the project were $850,990. The stream shape and characteristics were substantially changed and improvements and fish habitat were measured. Unfortunately, the 1995 floods also severely impacted the stream restoration efforts on this project. Most of the meanders were destroyed in the phase I section of the project, but the rock weirs in phases II and III of the project performed as designed, even though a new creek channel was formed by the flooding. Treated streambanks did not erode and private property adjacent to the creek was protected.

The Red Clover Creek Demonstration Project was completed in 1987 (Feather River Coordinated Resource Management Group, 1996b). The goals for the project were to reverse a down-cut channel that had resulted from overgrazing and to stabilize the stream banks to reduce further erosion and sediment transport downstream. In addition, the project would attempt to raise the ground-water table and water storage capacity to restore meadow habitat and moisture to improve range forage quality and quantity and to improve water quality and fish and wildlife habitat. The project included construction of rock check dams, streambank stabilization and revegetation, and stream corridor fencing to exclude livestock in a 70-acre stream reach. Total project costs were $172,500. The check dams successfully reduced stream velocities, which resulted in sediment deposition along the stream banks. The channel became more narrow and sinuous. Revegetation and stream-bank reinforcement decreased sloughing of vertical stream banks. Excluding livestock also had similar results but at a slower rate, indicating that grazing management alone may provide channel benefits at a lower cost than intensive restoration efforts if changes aren't needed quickly. The ground-water table was significantly increased in the vicinity of the check dams which fostered regrowth of riparian and floodplain vegetation. Both plant density and number of plant species increased, and there was a shift from sagebrush to more desirable riparian meadow vegetation. Average populations of adult trout increased, but reproductive rates did not, apparently because no spawning or rearing habitat was created. Waterfowl usage and nesting increased by 700% and deer use also increased.

No specific information on the Arcata Marsh was reviewed, but the success of watershed restoration efforts using selected BMPs such as constructed wetlands, stream corridor fencing, and stream restoration has been well documented. However, water quality improvements, particularly nutrient loading decreases associated with specific BMPs, is not widely available in the Klamath region. If sediment loading and rate of erosion are decreased by implementing a BMP, then total P loading may also decrease, although efforts to quantify the decrease are not widely reported.

Monitoring to Quantify Best Management Practice Effectiveness

Why Expend Resources on Monitoring?

Perhaps the most difficult and, therefore, neglected aspect of BMP implementation is the collection of information that can be used to reliably quantify the water-quality benefits of specific actions. Commonly, BMP implementation itself is used as the primary measure of success in watershed restoration efforts. In fact, there are numerous instances where BMPs have either not functioned as well as anticipated or have produced unexpected deleterious effects. As noted earlier, BMPs are often site specific, and studies have repeatedly shown that the effectiveness of a management action in one situation cannot be predicted from experiences elsewhere. Monitoring of water quality and other parameters relevant to farm and watershed P loading must be performed both prior to BMP implementation to characterize the baseline condition and after implementation to quantify the extent to which anticipated benefits have been achieved. Informed management decisions concerning the most cost-effective strategies for watershed restoration require reliable information on the effectiveness of individual BMPs in achieving water quality goals within the basin of interest.

Challenges to Implementing a Sound Monitoring Plan

Monitoring to assess BMP performance is rarely conducted for at least two reasons: it can be costly and requires a multiyear commitment of resources, often for a period of 5-10 years spanning both pre- and post-BMP implementation. Limited sample collection (quarterly water-quality monitoring for a year prior to and after BMP implementation) is rarely sufficient to quantify BMP effectiveness due to the high degree of temporal variability. This is especially true if the goal is to document improvements at the watershed scale. Agency initiatives and funding can wax and wane quickly, which may prevent the collection of sufficient pre-BMP (baseline) data and the sustained support of post-BMP monitoring. And, there is often a hesitancy to divert limited funds from “on-the-ground” land management and restoration activities, which produce tangible results (trees planted, streams fenced), for the collection of water quality samples or other environmental measurements. This may be particularly true for agencies that in the past have supported monitoring efforts that were poorly designed and, thus, yielded little useful information for making environmental management decisions.

Monitoring Program Design

Designing a BMP implementation and monitoring program as a rigorous “experiment” maximizes the chances of identifying water quality trends and linking them to BMP measures. This design should include sampling of control (no BMPs) and treated (BMPs applied) locations both prior to and following BMP implementation. Three general experimental designs are commonly used: (1) upstream-downstream sites, (2) paired watersheds, and (3) multiple watershed monitoring. Upstream-downstream and paired-watershed sampling designs may be most applicable to monitoring in the UKL watershed. Upstream-downstream sampling, as the name implies, involves the collection of water quality data both above and below an area subject to BMPs. Pre-BMP and post-BMP differences between collection sites are compared to quantify BMP effectiveness.

A modification to this design, referred to as the paired site-upstream downstream design, includes upstream-downstream sampling of control as well as treated areas along the same stream. Paired-watershed studies involve the monitoring of outflow water quality from two watersheds having similar land-use characteristics but differing in the presence or absence of BMPs. Monitoring is conducted prior to BMP implementation to develop a baseline relationship for water quality between the two watersheds and then continues following implementation to detect changes in this relationship, presumably a relative reduction in P loads in the treated watershed. Paired watershed studies provide the most powerful approach for assessing BMP effectiveness at the watershed scale. However, this design requires the availability of paired watersheds in close proximity with similar land use patterns and minimal change in land management in the control watershed during the monitoring period.

Examples of monitoring studies utilizing the above are provided at the following link, which describes U.S. Environmental Protection Agency's Section 319 National Monitoring Program: *http://www.epa.gov/OWOW/NPS/Section319/319over.html.*

Determining the appropriate length of monitoring associated with BMP implementation is important to assure that sufficient data are collected at minimal cost. The pre-BMP monitoring period should be sufficient to capture natural variability of climatic and hydrologic factors that affect P loading. As an example, 2 to 3 years of baseline monitoring have been recommended for paired watershed studies (U.S. Environmental Protection Agency, 1993a). Ideally, conditions during the baseline period should be representative of a range of hydrologic conditions. Water-quality conditions during consecutive dry years, for example, may provide a poor baseline data set for predicting conditions during periods of higher precipitation. The magnitude of the expected BMP effect on P loading should also be considered since shorter baseline periods may limit the ability to statistically detect small but environmentally significant treatment effects. Monitoring should include both routine (biweekly or monthly) and storm-event sampling for P concentrations as well as flow measurements in order to calculate P loads. Adequate consideration of storm events is critical, as these may contribute a large share of the annual P load.

The duration of post-BMP monitoring is affected by the same factors as for baseline sampling and by the expected timeframe for BMP effectiveness to be achieved. The recovery of lands that have been impacted by overgrazing can take several years to occur. Thus, while some immediate benefits may be detected from BMPs that, for example, exclude cattle from streams or reduce their congregation and fecal deposition in riparian zones, the restoration of more natural hydrologic and nutrient cycles in both riparian and upland areas can take years. Thus, it is essential that monitoring be maintained for a sufficient period of time to adequately gauge the success of the specific BMPs being implemented.

Although a reduction in P loading in the UKL watershed is the goal of BMP activities discussed in this document, it is valuable for BMP monitoring to include more than simply measuring stream P concentrations and loads. Quantitative information about the changes that BMPs have on landscape attributes that control watershed P transport including hydrology, soil, and vegetation is needed to show that BMP implementation produced the desired effects. Information on variables such as plant cover, erosion, or livestock relocation is also necessary to quantitatively link P load reductions to the level of watershed improvement (as opposed to traditional metrics such as the number of acres treated or miles of stream fenced). Monitoring of landscape attributes and BMP structures (where installed) is required to document BMP maintenance and longevity. Water quality benefits may be short lived if structural BMPs are not properly maintained or other land management actions are not sustained. Water quality and landscape monitoring should be coordinated to facilitate statistical analysis to detect cause-effect relationships.

Confounding influences can overwhelm the effects of BMP treatments, particularly those applied at small spatial scales or monitored for short periods of time. High P loads from untreated "hot spots" within the watershed may obscure small reductions in loading from treated lands. Watershed events (severe flooding) and upstream changes in land use can counteract or temporarily negate the beneficial effects of a BMP treatment. Runoff from degraded uplands may overwhelm BMP effects in riparian areas, particularly if the areas treated are small. Confounding effects are not always easy to predict. For example, large increases in the abundance of herbivorous insects and rodents in cattle exclosures were found to mask the effects of reduced cattle grazing intensity on vegetation (Belsky and others, 1999). Confounding influences cannot be entirely eliminated, but they must be documented to understand their potential effect on water-quality trends. The effects of such influences can be minimized by ensuring that the level of BMP implementation is sufficient to achieve substantial improvements in water quality.

The effectiveness of BMP selection and implementation is ultimately dependent upon a clear understanding of how P sources, storage, and transport within the UKL watershed have been altered by human development. Available water quality data indicate that the Sprague River is a major contributor of P to UKL, and available land-use information would suggest that cattle grazing is a major source of anthropogenic P. Based on this literature review, pilot studies of 2 or 3 types of BMPs that seem to show good promise for reducing upland P loading are stream corridor fencing, constructed wetlands, and overall stream restoration.

Conclusions

The total maximum daily load for uplands in the UKL watershed calls for a reduction of 18% in external total P loading resulting from upland hydrology and land cover restoration (Oregon Department of Environmental Quality, 2002). This value is divided between the Williamson (7.1%) and Sprague (11.1%) River Basins. Oregon Department of Environmental Quality further estimates that 12,900 kg/yr and 20,200 kg/yr of total P, respectively, are represented by these percentages. Therefore, the maximum amount of external total P loading that could be reduced is 33,100 kg/yr. Further, the total maximum daily load specifies that the annual mean total P concentration from all inflows to the lake is 66 μg/L. The issue for UKL is how to best accomplish that using BMPs that are cost effective, feasible, and can be maintained for extended time periods.

As an example of potential reductions in total P loading, average loading values for the Wood River during the period 1991-1993 were 35 kg/d at a sampling location on the Dixon Road bridge that is in close proximity to the spring-fed headwaters of the river (Campbell, 1993). At the most downstream sampling location at Agency Dike Road, average total P loading values were 80.5 kg/d. The difference between those two total P loading values represents the <u>maximum</u> amount that might be reduced through implementation of BMPs in the watershed, 45.5 kg/d. Using midrange percent reductions in total P loading values for a constructed wetland BMP of 26.3-56.7% from table 1 could conceivably result in a decrease in total P loading for the Wood River inflow to the Agency Lake subbasin of UKL of 12 – 25.8 kg/d. Converting this to the units for the total maximum daily load for UKL, the range for total P loading potential reductions could range from 4,380 to 9,419 kg/yr. Although the Wood River is not explicitly mentioned in the total maximum daily load discussed previously, it could potentially represent 13.2 – 28.5% of the desired total maximum daily load reduction for UKL. Two other BMPs such as stream channel or riparian fencing have been attributed a range of 20-50% reduction in total P loading (table 1). The range of total P reduction for the Wood River example could be 9.1 – 22.8 kg/d, 3321.2 – 8322 kg/yr, and 10 – 25% of the desired total maximum daily load reduction for UKL. Grass buffer strip BMPs are reported to cause 50 -86% reductions in total P loading (table 1). Using the same calculation, the potential range of total P loading for the Wood River example could be 22.8 – 39.1 kg/d, 8322 – 14271.5 kg/yr, and 25 – 43.1% of the desired total maximum daily load reduction for UKL. Forest buffer strips (see table 1) have similar reported values, and the single instance of grass-lined ditches is also in that category. It appears that BMPs have a strong potential for reducing nutrient loading, particularly total P loading, in the UKL watershed.

There are some significant costs associated with BMP implementation and maintenance both in terms of materials and labor and in terms of the location of and level of participation in a watershed program for UKL. If willing landowners are not identified—in particular, those with property located along an important drainage channel or downstream of other participating landowners—then the benefits produced by those upstream or off-channel may be significantly diminished. If landowners participate in various cost-share programs only as long as it takes to install some BMP that has useful features other than nutrient loading reduction potential and then drop out of the program, cease maintaining the BMP, or do not authorize water quality monitoring, then the full benefits of BMP implementation may not be realized. This becomes critical when so much of the literature and case studies do not report pre- and post-implementation water quality monitoring. This information is critical to identify specific BMPs that yield the best return for investment in the UKL watershed.

No single BMP may be universally applicable to land use in the UKL watershed. Stream corridor fencing seems to be a likely method for reducing total P loading, but in the Sprague River Valley, where there is no headwater hydraulic control, winter and spring flooding can quickly remove large sections of fencing. Keeping fencing intact in this watershed could require significant and highly variable annual labor and materials costs that are completely dependent on the meteorological conditions in that year.

Constructed and restored wetlands also appear to have a significant potential for reducing nutrient loading to UKL. However, wetlands can be sources of dissolved P that is readily available for algal growth, and trading off total P loading reductions for dissolved P loading increases may exacerbate UKL eutrophication issues rather than alleviate them. Size, location, and maintenance of wetlands have to be factored into this BMP implementation, although there are certainly instances where a wetland treatment may be both appropriate and highly effective.

It should be clear that water quality monitoring information is the greatest missing element for BMP implementation effectiveness. Sadly, less than 5% of BMPs have scientifically rigorous monitoring both prior to and after implementation to determine their ability to reduce nutrient loading. The expense of sample collection and analyses, the required expertise to interpret data results and to varying extents, the reluctance by private landowners to allow access to their property for sample collection are impediments to achieving quantified BMP effectiveness. The Klamath Basin Rangeland Trust has a program in the Wood River Valley that has been in progress since 2001 but lacks pre-BMP implementation water quality data. Historical data such as reported in Campbell and others (1993) may be valuable along with data available from other U.S. Geological Survey, Tribal, State, Federal agency, or consultant sources to partially fill this gap in water quality information. Even so, a true before and after comparison will not be available.

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