Phosphorus Sources and Sinks in Watersheds: A Review

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EXECUTIVE SUMMARY

Many regions around the world are concerned with phosphorus (P) and the risk it poses to water quality. Phosphorus is the limiting nutrient in most freshwater systems and, when in excess, it can accelerate eutrophication. Many countries have adopted some form of phosphorus management strategy to reduce the risk of phosphorus entering surface water from agricultural land. In Alberta, the Soil Phosphorus Limits Project was initiated in 1999 to develop soil phosphorus limits that will maintain or improve surface water quality by minimizing phosphorus loading from agricultural soils. With laboratory work complete, micro-watershed studies have recently been initiated to identify the relationship between dissolved phosphorus (DP) and soil-test phosphorus (STP). However, on a larger scale there are a variety of phosphorus sources and sinks within watersheds that influence the phosphorus content of surface water. A key question is what proportion of phosphorus in surface water can be attributed to agricultural land, and what factors govern inconsistencies in the various sources and sinks. To better understand this complex issue, a review of literature pertaining to phosphorus sinks and sources was conducted. Research carried out in Alberta and elsewhere that attempted to integrate phosphorus fluxes on a watershed scale was assessed, and its implications on the Soil Phosphorus Limits Project discussed.

Terrestrial sources of phosphorus to water include non-point source contributions from various land uses (cultivated land, grassland, irrigated land) and point sources from urban and industrial areas. Phosphorus losses from cultivated lands are highly variable, as they are a function of several factors. Conservation tillage reduces particulate phosphorus, but can increase dissolved fractions due to leaching from vegetation residues and surface application of fertilizers and manure. Losses from fertilized fields, especially of dissolved phosphorus forms, can be minimized by incorporation. Losses of inorganic fertilizer are often higher than manure immediately following application, but inorganic fertilizer is more readily available to plants; therefore, the risk of runoff is rapidly reduced. In general, less than 1% of applied phosphorus is lost in surface runoff, with the exception of phosphorus loss from frozen soils. While this small percentage may be agronomically insignificant, these amounts can have a large impact on receiving water.

Unimproved grassland areas tend to have low export values, compared to cultivated land, with the majority of losses occurring during snow-melt runoff. However, the surface application of commercial fertilizers and/or manure to grassland can increase phosphorus export in runoff. The timing of application, properties of the fertilizer, and risk of erosion are the main factors influencing the amount of phosphorus exported during an event.

The contributions of phosphorus from urban sources come in many different forms, such as municipal wastewater treatment plant (MWTP) effluent, stormwater effluent, and industrial effluents. The phosphorus contributions of these point sources have decreased in recent times with the advent and implementation of new, more efficient, phosphorus removal technology. In a worst-case scenario, municipal wastewater treatment plant effluent could be estimated at 2 g capita^{-1} d^{-1} phosphorus. Stormwater runoff is more dilute than the MWTP effluent. Although combined sewer overflow effluent has a higher concentration of nutrients, it accounts for only 5% of the total annual effluent discharge. The contribution of phosphorus by municipal and

industrial effluent to the load in the receiving body depends on the degree of effluent treatment, the receiving waters ability to dilute, and the morphological characteristics of the water body.

Phosphorus from septic systems can be a major contaminant to groundwater and surface water resources. Phosphorus in groundwater moves at much slower rates than groundwater itself, but can still be a threat to surface water resources. Although septic systems generally account for only a small portion of phosphorus budgets (4 to 6%), values of up to 63% have been reported for lakes in Alberta.

Atmospheric contributions may also prove to be an important source of phosphorus to surface water. However, this source has not been given much consideration until recently. Atmospheric phosphorus is associated with organic (e.g., pollen) and inorganic dust particles and will contain varying concentrations of phosphorus depending on its source. Studies have reported that dustfall and precipitation accounted for 3.1 to 4.5% of phosphorus loading to lake systems.

Little emphasis has been placed on groundwater as a source of phosphorus to surface waters. However, studies from Alberta suggest that while phosphorus concentration in groundwater is low, it may serve as a mechanism for recycling phosphorus from lakebed or riparian sediments, in addition to directly contributing phosphorus to surface water.

In addition to terrestrial and atmospheric sources of phosphorus to surface water, there are many studies that suggest much of the phosphorus loading is generated internally. Phosphorus flux within lentic and lotic environments is influenced by the physical, chemical, and biological composition of the system. There are short-term and long-term storage compartments that retain and release phosphorus at variable rates according to pH and redox conditions of overlying water and sediment. The biota (macrophytes, phytoplankton, zooplankton, and bacteria) represent short-term storage compartments that, upon senescence and death, release much of the stored phosphorus back to the water. Remaining phosphorus will be buried in the litter layer and upon decay a portion will be released to the water column. As litter is buried by sedimentation, the remaining phosphorus is placed in long-term storage.

Water management can influence the extent of phosphorus removal in aquatic ecosystems. In Alberta, there are no studies that have been conducted in flowing water in streams that relate to nutrient spiraling or uptake along a gradient**.** Based on findings of others, phosphorus retention in Alberta's lotic systems can be expected to be greatest in summer when flows are lowest and biological activity is highest. Long-term storage is more likely in wetlands and riparian zones than in rivers and streams as vegetation aids burial of phosphorus. Evidence from some watershed studies, such as the Pine Lake Study, Alberta, indicate that internal loading may contribute more to phosphorus concentrations in surface water than surrounding land use.

Although there have been many plot/small watershed and large watershed studies conducted, very few have attempted to link these two different scales. Where it has been attempted, several studies found that a large amount of soluble reactive phosphorus (SRP) from the land is rapidly attenuated instream through adsorption or dilution. While SRP may be rapidly attenuated, other authors have reported that about 80% of adsorbed phosphorus was resuspended in the first large storm event. Most studies that have tried to link the two different scales have not accounted for important sources and sinks, such as wetlands and livestock operations. Studies examining larger, multiple land use watersheds, have shown relationships between instream phosphorus concentrations and land use, but few have been able to account for phosphorus loads from specific sources. Global information system based models that take into account land use and hydrology may be better able to link these scales and processes.

Considering all the variables, it is not surprising that each watershed seems to have a unique phosphorus budget. Export coefficients of phosphorus in runoff is site-specific, and heavily influenced by factors such as vegetation, soil structure, topography, precipitation, and watershed size. Combining those variables with varying land use and management practices on tracts of land, a watershed-scale study will only produce phosphorus budgets that are suitable for the watershed that is being evaluated. Those studies that have attempted to budget phosphorus at the watershed scale have not incorporated essential watershed components or have made assumptions to compensate for gaps in our knowledge.

There are many studies that show contradictory results for the relationship between STP and DP, particularly at the watershed scale where the variability in land management is high. Watershed-scale studies have indicated that 90% of surface runoff comes from the area immediately adjacent to the stream. Therefore, targeting management practices within critical source areas may be the most effective way to address phosphorus transport to surface waters. The risk of phosphorus movement to aquatic systems can be assessed according to distance from water body, management practices, climate, soil, and topography. Rating each site individually will incorporate the variability that is inherent in phosphorus flux and provide more reasonable soil phosphorus limits for producers in Alberta.

The determination of STP and the relationship between STP and DP, alone, will not provide all the necessary information for generating phosphorus limits for agricultural land in Alberta. Particulate phosphorus also needs to be included in analyses as sediment, especially smaller fractions such as clay, may be highly enriched in phosphorus. Furthermore, atmospheric deposition and runoff from grassland in the critical source area may prove to be important sources of phosphorus to Alberta’s surface water.

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PART 1: INTRODUCTION

Phosphorus (P) is the limiting nutrient in most freshwater systems, and when found in excess, it can accelerate eutrophication. Eutrophication causes algal blooms, aquatic plant growth, and taste and odour problems, and oxygen depletion in the water column. Early research on phosphorus in surface water was stimulated by observations of increasing eutrophication in the late 1960s, in part because increased biological growth interfered with the treatment of industrial and municipal water supplies (Keup 1968). As concerns regarding point sources of phosphorus have been addressed (e.g., wastewater treatment plant effluent), more attention has been given to diffuse, non-point sources.

Phosphorus has a restricted biogeochemical cycle. Unlike nitrogen and carbon, it is closed to the atmosphere and cannot escape from a waterway by gaseous diffusion (Keup 1968). Eutrophication of lakes may be accelerated when phosphorus concentrations in the water are between 0.01 and 0.02 mg L^{-1}. This is an order of magnitude more sensitive than soil concentrations of phosphorus considered critical for plant growth, which range from 0.20 to 0.30 mg L^{-1} in soil solution (Daniel et al. 1998). Thus, small additions of phosphorus can result in large changes in aquatic systems.

Many regions around the world are concerned with phosphorus and the risk it poses to water quality. To better manage phosphorus, the United States has required all states to adopt the phosphorus index or implement environmental soil phosphorus limits. Most of the states have adopted the phosphorus index approach that rates the potential for phosphorus loss in runoff based on source (e.g., soil phosphorus, applied phosphorus type, rate, and method) and transport (e.g., runoff and erosion) characteristics of the site (Heathwaite et al. 2000). In the Netherlands, phosphorus surplus in the soil has resulted in increased phosphorus leaching losses (Van der Molen et al. 1998). Since 1991, legislation has reduced manure application rates. These standards have recently been replaced by agriculturally inevitable nutrient losses and environmentally acceptable nutrient losses. An environmentally acceptable loss of phosphorus has been defined as 1 kg P_2O_5 ha^{-1}, while inevitable losses have been measured between 25 and 75 kg P_2O_5 ha^{-1}. By 2010, an equilibrium fertilization rate is expected to be met, and is defined as the supply of fertilizers and manure that meets crop demand and compensates for inevitable losses (Van der Molen et al. 1998). Even with standards, inevitable losses will be greater than environmentally acceptable losses in 2010. In the United Kingdom, the Code of Good Agricultural Practice for the Protection of Water Quality recommends that organic manure application rates match crop phosphorus uptake for a rotation of soils containing more than 25 mg L^{-1} Olsen-extractable phosphorus. No phosphorus fertilizer is recommended for most crops at Olsen-extractable phosphorus > 45 mg L^{-1} (Withers et al. 2000).

In Alberta, a 5-yr, province-wide study was conducted to assess agricultural impacts on water quality (CAESA 1998). The study compared land use areas of varying agricultural intensity (low, medium, and high) and found the risk of water quality degradation increased in areas of the province where agricultural intensity was highest. The study did not attempt to evaluate the contributions of specific cropping practices or livestock production in relation to water quality. Further, those conducting the study reported the difficulty and time-consuming nature of integrating numerous and varied databases from significantly different project designs and monitoring methods.

In 1998 to 1999, during the development of draft regulations for confined feeding operations (CFOs), decisions were made to develop phosphorus limits that would determine appropriate rates for manure and fertilizer application to soil. In response to this request, the Soil Phosphorus Limits Project was initiated in 1999 with the objective of developing soil phosphorus limits that will maintain or improve surface water quality by minimizing phosphorus loading from agricultural soils (Olson et al. 2001). Laboratory and field results from the Phosphorus Mobility Study were used to develop a preliminary phosphorus export model, which predicts the concentration of dissolved phosphorus (DP) in runoff at the edge of field based on soil-test phosphorus (STP). The Phosphorus Mobility Study, which was part of the Soil Phosphorus Limits Project, was conducted (1) to develop a relationship between STP and potential runoff losses, and (2) to determine how landscape and climatic variables influence the movement of phosphorus (Olson et al. 2001). The Haynes Creek Watershed case-study, also part of the Soil Phosphorus Limits Project, was conducted to see how STP in the watershed compared to DP in Haynes Creek. Based on the phosphorus export model developed from the Phosphorus Mobility Study, it was estimated that about 70% of the DP in the Haynes Creek M1 sub-basin originated from field runoff.

The first phase of the Soil Phosphorus Limits Project focused mainly on laboratory experiments, with initial field-scale work completed in Haynes Creek watershed. In Phase 2, emphasis will be placed on monitoring numerous micro-watersheds to determine the STP:DP relationship at the field scale.

The main sources of phosphorus to aquatic systems are surface runoff from varying land uses (e.g., arable land, native grassland, forested, urban), septic fields, groundwater seepage, atmospheric deposition (dustfall and precipitation), internal sediment release, waterfowl and other wildlife, and the decomposition of organic matter. The relative contribution of each of these sources to an individual system is difficult to quantify since phosphorus transport is heavily influenced by the physical, chemical, and biological composition of the local environment.

The purpose of this review was to identify phosphorus sinks and sources that may impact Alberta’s water resources and, if feasible, quantify their individual contributions. Further, this report will assess research carried out in Alberta and elsewhere that has attempted to integrate phosphorus fluxes on a watershed scale. Finally, this review identified implications of the information on the Soil Phosphorus Limits Project. Part 2 provides a review of terrestrial phosphorus losses, while Part 3 focuses on phosphorus flux in the aquatic environment. Part 4 examines watershed-scale case studies.

PART 2: TERRESTRIAL PHOSPHORUS LOSSES

CULTIVATED LAND

Introduction

Phosphorus loss from cultivated land has long been regarded as a major contributor of nonpoint source pollution to aquatic environments. Surface runoff is the most widely studied mechanism of phosphorus loss. Although surface runoff is often narrowly defined as overland flow, it can also include unsaturated zone subsurface flow or near-surface interflow. However, these transport mechanisms are rarely studied, so this review will focus on overland flow. Artificial or tile drains, another potential pathway of phosphorus transport, are rare in Alberta; therefore, they are not considered here.

A large proportion of the phosphorus lost from cultivated land is associated with sediment. Therefore, it is widely believed that by controlling erosion through conservation tillage practices, phosphorus losses can be minimized. However, several studies have suggested that dissolved phosphorus is also a significant component of phosphorus lost from cultivated lands (Menzel et al. 1978), particularly in Alberta (Anderson et al. 1998; Wright et al. 2002). Moreover, dissolved phosphorus is a major contributor to eutrophication, as much of it is biologically available in aquatic ecosystems.

Diversity in crop type, soil characteristics, and management practices makes it difficult to generalize about phosphorus losses from cultivated land (Haygarth and Jarvis 1999). Furthermore, runoff volume, which is directly related to phosphorus flux, is difficult to predict as it is influenced by a number of other factors including rainfall intensity, antecedent moisture conditions, slope, seasonal variability, and residue cover. The distribution of phosphorus within the soil profile also impacts phosphorus loss as surface runoff only interacts with the top few centimeters of the soil profile (Sharpley 1985; McDowell et al. 2001a). Several different management factors, including crop type, manure addition, inorganic fertilizer addition, and tillage regime, will be examined in this section.

Methods

Most studies of phosphorus export from cultivated lands were conducted on small plots and often used rainfall simulators in an attempt to control the many factors affecting runoff volumes and phosphorus release. However, even with controlled rainfall intensity and duration, there is generally a high degree of variability among replicates (Mueller et al. 1984; Beke et al. 1989; Daniel et al. 1993). One reason for this inconsistency is that with rainfall simulators, runoff is often restricted to inter-rill regions, which vastly increases variability among replicates (Mueller et al. 1984). Along with differences in management practices discussed previously, localized influences of microtopography, rainfall intensity, and infiltration rates also contribute to the variability in rainfall simulations. Despite this inherent variability, rainfall simulators offer a practical method for comparing differences among treatments.

Several studies have also been conducted using rainfall-generated runoff on hydrologically isolated small plots (Dunigan et al. 1976; Ginting et al. 1998; Kimmell et al. 2001). Although there is greater variation in rainfall runoff amounts (Blevins et al. 1990), they may provide more realistic estimates of runoff losses of nutrients than simulated rainfall plots. Results are often quite variable among years, therefore, several years are required for study. Additionally, there is no control over rainfall intensity and duration, and there can be strong seasonal influences.

Watershed-scale studies have been limited, with the exception of studies comparing conservation-tilled and conventionally-tilled watersheds. Most of these studies have taken place on small, single management type watersheds. Some of the studies are paired watershed studies, while others compare phosphorus loss before and after beneficial management practice (BMP) implementation.

Tillage Effects on Phosphorus Losses

Conservation tillage practices are designed to minimize soil erosion by leaving more crop residue on the soil surface through reduced tillage or by reducing slopes through contour cropping or constructing terraces. As discussed previously, phosphorus loss from cropland is often dominated by sediment-associated or particulate phosphorus (PP) forms; therefore, much emphasis has been placed on conservation tillage as a method to minimize particulate and total phosphorus (TP) losses from agricultural land (Römkens et al. 1973; Angle et al. 1984; Gaynor and Findlay 1995). However, concentrations of DP and the proportion of phosphorus lost as dissolved fractions increase in runoff from conservation-tilled plots (Gaynor and Findlay 1995; Bundy et al. 2001). In some cases, the total amount of DP lost under conservation tillage can exceed losses from conventional tillage (Römkens et al. 1973; Johnson et al. 1979). There are two major causes of the increased DP concentrations: (1) reduced runoff volumes that increase the concentration of DP and proportionally decrease losses of PP, and (2) leaching from crop residues.

In raised wide beds in Texas, losses of TP were greater from chisel-plow plots than from no-till, but losses of soluble reactive phosphorus (SRP) were greater from no-till (Torbert et al. 1996). In this study, losses of SRP increased during the event and were attributed to leaching from vegetation (Torbert et al. 1996). Schreiber and McDowell (1985) reported that up to 80% of nutrients can be released from senescing vegetation and that lower rainfall intensities resulted in greater leaching due to increased contact time between rainfall and vegetation. Freezing and drying can also enhance leaching of soluble nutrients from vegetation (Timmons et al. 1970). The low slopes on the plots in this study would also have promoted rainfall contact with the vegetation.

Conversely, Blevins et al. (1990) found that conventionally-tilled plots (moldboard plowing followed by disking) tended to lose more DP than either chisel-plow or no-till plots. Chisel plowing was conducted across the plot slope and effectively reduced runoff volumes due to high surface roughness. No-till plots reduced runoff and DP losses due to higher crop residue cover. Results from this study were not significant due to large variability in rainfall among years. However, under rainfall simulations, Mostaghimi et al. (1988) reported similar SRP losses between conventional and no-till plots.

As illustrated by Blevins et al. (1990), phosphorus losses are highly dependent on runoff volume. Although conservation tillage consistently reduces sediment loss, its effect on runoff volume is variable. While most studies have reported reductions in overall runoff amounts (Blevins et al. 1990), recently-tilled soils have higher infiltration rates that can reduce overall runoff volumes from conventionally tilled lands (Wendt and Corey 1980). Angle et al. (1984) reported nine times greater runoff volumes from very small conventionally-tilled watersheds. They attributed this difference to higher vegetative cover on conservation plots, but noted that runoff losses from early spring were often lower from conventional tillage due to recent tillage activities and higher from conservation tillage due to the relatively smooth soil surface. Bundy et al. (2001) also reported similar seasonal trends in runoff volumes from conservation and conventionally-tilled plots. Differences in runoff volumes may also be enhanced by surface crust formations on the moldboard plowed plots (Blevins et al. 1990).

At the watershed scale, DP losses from conservation tillage systems are variable. Inamdar et al. (2001) found reductions in TP loss following the implementation of BMPs, including conservation tillage, in the Nomini Creek Watershed near the Chesapeake Bay, Virginia. There were increased losses of DP fractions; however, it could not be determined whether this was a result of conservation tillage practices or increased streamflow. In a paired watershed study in Iowa, Johnson et al. (1979) also reported higher DP losses from ridge tillage than from conventional tillage methods, but differences were not significant. Dissolved phosphorus losses increased with increasing residue cover, suggesting that leaching from residue cover was an important factor in the higher DP losses, but surface application of fertilizers was also cited as a possible contributor (Johnson et al. 1979).

Using different conservation tillage techniques, Burwell et al. (1974) compared a level-terraced field watershed with a contour-cropped corn watershed in Iowa. Total phosphorus loss was greater from the contour-cropped watershed, but DP loss was slightly higher from the terraced watershed, despite having lower rainfall. However, the study must be questioned as the high concentrations of DP from the terraced watershed were not attributed to lower runoff, but to the presence of a feedlot near the stream.

In contrast, Chichester and Richardson (1992) reported that at the watershed scale, DP loss tended to be lower from no-till than from conventional tillage, although the results were not significant. Therefore, the effects of phosphorus from vegetation leaching may diminish with time or may not be as critical at larger scales. The study was conducted on a poorly-drained clay soil that was subject to shrinkage and swelling; therefore, runoff volumes were not affected as much as they could have been on coarser-textured soils.

Conservation tillage reduces the amount of sediment and PP contributed to water bodies. In controlled studies, DP losses can increase due to leaching from surface residue; however, the impacts of DP fractions have yet to be determined at the watershed scale. There are also other concerns with conservation tillage as it allows phosphorus to accumulate near the soil surface as crop residues and fertilizers are not incorporated into the soils, increasing potential loss in surface runoff (Gaynor and Findlay 1995). These factors will be discussed further in the following section. Reduced tillage also promotes the formation of macropores, which allow preferential flow of nutrients through the soil profile and could augment groundwater contamination (Drury et al. 1993; Heckrath et al. 1995; Beauchemin et al. 1998).

Inorganic and Organic Fertilizer Effects

Inorganic fertilizer application can increase STP levels as well as phosphorus concentrations in surface runoff. Furthermore, inorganic fertilizer does not increase infiltration or improve other soil properties like organic fertilizers. Therefore, phosphorus loss generally increases as fertilization rate increases (Klausner et al. 1974; Daniel et al. 1993). However, a few studies have reported reductions in runoff loss due to increased dry-matter production (Muir et al. 1973; Bundy et al. 2001). Other factors in addition to application rates and STP can affect phosphorus losses, including timing of application, fertilizer placement, and incorporation method.

As with inorganic fertilizers, manure can increase STP concentrations and phosphorus concentrations in runoff. However, there are several complicating factors in addition to those cited for inorganic fertilizers. Phosphorus in manure is often high in organic phosphorus, which must be mineralized before it can be taken up by plants. Therefore, manure applications can have a longer-term impact on phosphorus loss in surface runoff (Eghball and Gilley 1999). Additionally, manure amendments tend to improve soil physical properties, such as aggregate stability and water holding capacity, by increasing organic matter content (Sommerfeldt and Chang 1985). Increased concentrations of phosphorus in runoff from manured land can be offset by reduced runoff volumes. Phosphorus losses from manure are difficult to quantify as phosphorus concentrations are highly variable according to species, rations, and storage conditions. However, this review will focus on factors affecting manure nutrient losses following land application.

Timing of application. Phosphorus in runoff from frozen ground can account for a significant proportion of phosphorus loss, especially of soluble forms (Nicholaichuk and Read 1978). In a study conducted in Saskatchewan, surface application of phosphorus fertilizer applied in late fall resulted in losses of 4.8 kg ha^{-1} from summerfallow or nearly 10% of applied phosphorus the following spring (Nicholaichuk and Read 1978). Although the conditions were extreme (large snowmelt runoff, surface application of fertilizer), snowmelt runoff accounts for greater than 85% of runoff from many northern prairie watersheds (Nicholaichuk 1967). Decreased infiltration, freeze-thaw cycles, and increased contact time between soil, vegetation, and precipitation can increase losses of soluble nutrients (Timmons et al. 1977).

Converse et al. (1976) found no difference in TP loss from manure applied in the fall, winter, and spring. However, the range of phosphorus loss was quite variable (TP loss: 0.55 to 8.09 kg ha^{-1}), and may have obscured trends. In contrast, other studies have reported much greater losses from winter-applied manure. Runoff from frozen manured land was estimated to contribute 60% (2.1 to 2.5 kg ha^{-1}) of the phosphorus budget to Lake Mendota, Wisconsin (Ryden et al. 1973). Hensler et al. (1970) reported losses of 6% (15.6 kg ha^{-1}) of the applied manure phosphorus from one winter storm event. Phosphorus concentrations in runoff from manured land were higher than from other land use types, but losses were only greater from frozen ground (Hensler et al. 1970). As with soluble nutrients, TP loss increases with high application rates of inorganic fertilizer. Runoff rates tend to be much greater due to reduced infiltration capacity of frozen soils, while nutrient concentrations are increased due to increased soil-water contact (Wendt and Corey 1980).

The greatest potential for phosphorus loss is immediately following fertilizer application. Unlike organic phosphorus-rich manures, inorganic phosphorus is readily available for plant uptake. Eghball and Gilley (1999) found that the greatest amount of nutrients from inorganic fertilizer were lost in the first rainfall event following fertilizer application, while manure and compost losses increased in subsequent events, reflecting the sustained release of nutrients.

Fertilizer application and incorporation. Tillage also plays a role in phosphorus loss from fertilized plots. Phosphorus at the top of the soil profile is more vulnerable to surface runoff (Sharpley 1985; McDowell et al. 2001a). Therefore, conservation tillage methods, which leave fertilizer at the soil surface, have greater losses of phosphorus than incorporated (Timmons et al. 1973) or injected fertilizer (Baker and Laflen 1982).

Römkens et al. (1973) studied the effects of five different tillage methods, used for fertilizer incorporation, on soluble and particulate nutrient losses in runoff. They found losses of soluble nutrients highest from the least disturbed plots (coulter plow). Particulate phosphorus losses were greatest from the conventional and till systems.

Kimmell et al. (2001) examined the effects of tillage and fertilizer phosphorus placement on runoff phosphorus losses in a grain sorghum-soybean rotation. With the chisel plow system, there were no differences in soluble phosphorus losses from broadcast or knifed fertilizer plots when compared with an unfertilized check. However, with conservation tillage plots (no-till and ridge-till), higher soluble phosphorus losses were found from broadcast fertilizer plots on grain sorghum than from knifed-plots. Previous studies have also shown significant differences between chisel-tilled and no-till when phosphorus was broadcast (Blevins et al. 1990; Seta et al. 1993). Total phosphorus losses were also affected by phosphorus placement under grain sorghum, with greater losses from broadcast plots (Kimmell et al. 2001). Tillage alone had a significant effect on TP losses under soybeans. In this case, chisel-till and ridge-till systems had greater total phosphorus losses compared with no-till under soybeans.

There are very few watershed-scale studies that examine the impact of fertilization rates and tillage on phosphorus losses. Schuman et al. (1973) compared fertilized watersheds in Iowa with various tillage practices and fertilization rates. Losses of phosphorus from contour-cropped corn were proportional to fertilizer application rates. Overall losses from terraced corn were lowest due to low runoff rates, despite a high fertilizer application rate. Therefore, hydrology plays a pivotal role in phosphorus losses from fertilized land.

Surface applied manure can increase concentrations of TP and dissolved reactive phosphorus (DRP), but some of these increases can be offset by the reduced amount of runoff due to increased infiltration rate (Wendt and Corey 1980). Differences in runoff volume, along with variability among plots and manure nutrient contents, result in few significant differences among phosphorus losses between manured and unmanured land, especially from incorporated manure (Mueller et al. 1984; Eghball and Gilley 1999; Bundy et al. 2001). These findings must still be tested at the watershed scale.

Tillage practices and incorporation of manure can, however, affect the forms of phosphorus lost. If manure is well-incorporated, PP loss tends to be greater, while DRP loss is reduced (Ginting et al. 1998; Zhao et al. 2001). However, some conservation tillage methods, such as ridge tillage, which leave a high proportion of manure on the surface, result in higher DRP loss (Zhao et al. 2001). Eghball and Gilley (1999) reported higher losses of DP and biologically available phosphorus (BAP) from no-till plots treated with compost and manure, while losses of PP and TP were higher from tilled plots, regardless of manure and compost application.

Although several studies have examined the relationship between STP and nutrient losses in runoff, this relationship may not be valid for surface-applied manure. On rainfall simulation plots in Wisconsin, no-till manure had higher DRP losses, despite lower STP levels. Therefore, the authors concluded that surface-applied manure masked the effect of STP on runoff DRP (Bundy et al. 2001).

Types of amendments. Eghball and Gilley (1999) compared runoff from sorghum and winter wheat residue amended with compost, manure, and fertilizer. On the sorghum plots, two different rates of each amendment were used, one based on crop nitrogen (N) requirements and the other on crop phosphorus requirements. Runoff from land amended with compost and manure indicated that losses of DP, BAP, and PP were not significantly different between amendments on sorghum residue. However, on wet soils, runoff losses of DP and BAP from winter wheat plots were greater from the compost treatment and were much greater than runoff from dry soils. Particulate phosphorus loss was not different among treatments.

Comparing runoff from plots amended with inorganic fertilizer, manure, and municipal sewage sludge, Bundy et al. (2001) reported greater phosphorus loss from plots amended with biosolids (stabilized sewage sludge), followed by manure, inorganic fertilizer, and the untreated control. Runoff volumes were reduced as a result of increased residue cover from straw in the manure and increased dry-matter production on fertilizer amended plots, as well as increased infiltration on manured plots.

Crop Type

Total phosphorus loss tends to be greater from crops with low surface residue cover, such as row crops (Johnes 1996). Particulate and total phosphorus losses were also greater under sorghum than winter wheat residue (Eghball and Gilley 1999).

Burwell et al. (1975) examined phosphorus loss from various crop types during snowmelt runoff, following planting, and during the remainder of the year. Snowmelt accounted for the majority of the runoff and soluble nutrient loss, while most of the sediment-associated loss was found in the 2 mo following planting. Very little runoff was recorded during the rest of the year due to increased crop cover. Crop cover played a significant role in PP losses, with the greatest losses from fields with the least crop cover. The greatest PP losses were from fallow fields, followed by continuous corn, corn rotation, oat rotation, and hay. The most DP losses followed the pattern hay (alfalfa) > continuous corn > oat rotation > corn rotation > fallow. Dissolved phosphorus loss was greater from the alfalfa due to leaching from crop residues (Timmons et al. 1973; Schreiber and McDowell 1985).

Similar results were found in Oklahoma watersheds, where DP loss from alfalfa was higher than from other crop types (Olness et al. 1975), except for irrigated cotton. Irrigated cotton had the greatest TP losses, followed by dryland cotton, wheat, and alfalfa. However, TP losses were much higher than for other studies reported in the literature, ranging from 2.48 to 5.01 kg ha^{-1} yr^{-1} for the dryland crops.

In Alberta, Beke et al. (1989) reported higher inorganic phosphorus losses from alfalfa than barley or fallow; however, results were not significant due to large variability among plots.

Other management practices that affect residue cover can also influence phosphorus losses from cultivated land. One of these practices, conservation or minimum tillage, was discussed previously. On large plots, Klausner et al. (1974) found no differences in SRP loss among rotations of corn, wheat-bean, and winter wheat; however, differences may be more apparent with TP. Poor management practices, such as burning or otherwise removing crop residue, resulted in higher losses of SRP compared to good soil management practices due to higher runoff rates.

General Watershed Studies

At larger watershed scales, there have been very few studies that relate specific agricultural management practices to phosphorus loads. The exceptions to this are watershed-scale studies examining the impact of conservation tillage and grazed lands. There have also been several studies focusing on fertilizer application and nitrate-nitrogen concentrations (Smith et al.1987; Bouraoui et al. 1999; Castillo et al. 2000). However, phosphorus dynamics are much more complex than nitrogen as soil interactions play a more important role in phosphorus release (Johnson et al. 1997). At larger scales, the variability in management practices and the inherent difficulty in relating land use to water quality at the outlet of a watershed, often prohibit specific linkages between agricultural practices and water quality. Therefore, results are more general at the watershed scale.

In north-central Alberta, export coefficients were higher from cultivated watersheds (60-99%) than from forested watersheds of tributaries to Baptiste Lake (Trew et al. 1987). Export coefficients ranged from 21.7 to 33 kg km^{-2} yr^{-1} from cropland, while forested sites were much lower during the same 3-yr period (range 10 to 17 kg km^{-2} yr^{-1}). Similarly, watersheds surrounding Lake Wabamun, Alberta that had significant proportions of cropland (>30%) contributed more TP (range: 22 to 48.5 kg km^{-2} yr^{-1}) than watersheds with only a small proportion of cropland (<12%) (range 7.5 to 12 kg km^{-2} yr^{-1}) and forested watersheds (range 10 to 17 kg km^{-2} yr^{-1}) (Mitchell 1985).

GRASSLANDS

Introduction

Prior to land being cultivated for agriculture, grassland covered much of southern Alberta. With the development of modern agricultural practices and other human activities, much of the grassland area has disappeared. Grassland within developed areas is often adjacent to surface watercourses and riparian zones or in areas that are at greater risk of impacting surface water through runoff (steep slopes, problem soils). Due to their proximity to watercourses, nutrient export from native and improved grassland areas can be an important source of phosphorus for surface water systems in Alberta.

Sources of phosphorus exported from grassland areas can include soil, living and dead plant material, animal wastes, and commercial fertilizer. Many of the studies examined involved an assessment of the impacts of added phosphorus, either in the form of commercial fertilizers or some form of livestock manure (slurry, solids) on the concentrations and amount of phosphorus exported from an area. The amount of phosphorus lost in runoff is controlled mainly by transport (runoff and erosion) and source factors (soil phosphorus content and rate, timing, method and type of phosphorus applied) (Sharpley 1995).

This section focuses primarily on phosphorus export from grassland watersheds, cattle wintering sites, and the factors that influence phosphorus losses from these landscapes. The degree of vegetative cover will influence the amount of surface runoff from an agricultural area. Bare or sparsely covered soil surfaces found in overgrazed areas are more vulnerable to surface runoff. A well-developed cover of living or dead plant material will reduce the velocity of raindrops striking the soil surface. This helps to maintain infiltration rates by maintaining the aggregation of the soil particles over a longer period of time. If surface runoff occurs, living and dead plant material will reduce runoff velocity allowing an opportunity for increased infiltration. Reduced runoff velocity will also reduce the energy available for detachment and movement of soil particles. Additionally, if flow velocity is reduced, some sediment entrained in the runoff water will be deposited (McDowell et al. 2001b).

The variety of management practices used on grassland areas as well as the diversity of the natural landscapes result in a wide range of values describing phosphorus losses from grassland. As mentioned in the previous section, runoff volume is difficult to predict, but is critical to understanding phosphorus fluxes. Phosphorus added to agricultural systems leads to an increase in the levels of phosphorus susceptible to loss to aquatic systems as only 5 to 10% of added phosphorus is taken up by agricultural crops (Haygarth and Jarvis 1999). Values for the export of surplus phosphorus on a kg ha^{-1} yr^{-1} basis are variable, with the watershed and local factors accounting for the discrepancies. A study of a catchment in Northern Ireland estimated surplus phosphorus at 24 kg ha^{-1} yr^{-1}. The amount of phosphorus lost is often a small percentage of the phosphorus applied as fertilizer. Nash et al. (2000) reported 2.3% of phosphorus applied from poultry litter was lost and 3.1% of applied inorganic phosphorus was lost. Smith et al. (1992) and Daniel et al. (1998) reported similar results.

Methods

The majority of the studies were carried out on a plot scale or within small watersheds (<10 ha). Rainfall simulators were used on the small plots to generate consistent runoff, eliminating the variability of natural rainfall over a large watershed. Some studies have been carried out on larger watersheds, including Haynes Creek (16 600 ha) and Crowfoot Creek (160 000 ha) in Alberta.

When examining the transport of phosphorus on grasslands, it is important to remember that after phosphorus application, any rainfall that causes overland flow can potentially increase solubilization and physical detachment of phosphorus. Catchment-wide losses depend on the hydrological connectivity from the phosphorus source to the watercourse (Preedy et al. 2001). Plot-scale studies, then, can only indicate rates of phosphorus mobilization and do not accurately reflect the amount of phosphorus being transferred from hillslope to watercourse as the plot area is hydrologically isolated (Edwards and Daniel 1994; Preedy et al. 2001). Processes such as deposition-suspension, adsorption-desorption, dissolution-precipitation, and biological uptake-mineralization are often ignored as they relate to transport to streams (Nelson et al. 1996).

Unimproved Native Grasslands

Although there are large areas primarily of native grassland, little has escaped the impact of livestock and/or the additions of fertilizer. As a result, there are few papers dealing with phosphorus export from unimproved grassland. Nutrient loads from native prairie are representative of natural levels of phosphorus export. As such, they are useful in determining the impact of other land uses (Timmons and Holt 1977).

There are a few studies that include phosphorus export data related to unimproved grassland areas. A plot study conducted by Timmons and Holt (1977) found that greater than 80% of runoff losses occurred during spring melt. Average TP losses during the study were 0.11 kg ha^{-1} of TP compared with 0.03 kg ha^{-1} of total DP and 0.02 kg ha^{-1} of DP. In Jawson et al. (1982), a 0.9 ha ungrazed check and a 21.5 ha grazed area were examined. The TP and DP export losses from the check area were slightly lower than those found in Timmons and Holt (1977). Schepers and Francis (1982) observed that runoff from a 0.11 ha control area had much greater phosphorus concentrations than grazed areas. Export values were not reported. The reasons for the higher values were related to wildlife and leaching of soluble nutrients from decomposing plant material. Runoff from this area contained little sediment that was predominantly organic material.

Improved Grasslands

In the context examined in this section, improved grasslands refer to those grass areas that have been modified due to the presence of livestock and the addition of fertilizers. The majority of the studies carried out have looked at grasslands under some form of management.

Livestock Impacts

Several authors have examined the effects of various grazing regimes and manure wastes on phosphorus in soil and in surface runoff. Examination of changes in phosphorus concentrations in grassland soils indicate that organic forms of phosphorus account for about half of the TP found. The percentage of TP as organic phosphorus ranges from 32 to 37% in Brown and Dark Brown Chernozemic soils to 43 to 50% in Black Chernozemic soils in Alberta (Dormaar and Webster 1963; Dormaar and Lutwick 1966). Dormaar and Willms (1998) found that TP in soils in the foothills of southwestern Alberta decreased under increased grazing pressures. Additionally, the heavy grazing treatments resulted in less thatch, root ramification, vegetative cover, soil moisture, and smaller soil aggregates, leaving these areas susceptible to water and wind erosion. The presence of livestock and wildlife can result in the compaction of the soil surface, leading to reduced infiltration rates and increased surface runoff (Van Keuren et al. 1979; Heathwaite 1995).

Regardless of the time of year, the presence of livestock in a watershed and their proximity to a watercourse affects the amount of phosphorus exported. A study by Mitchell and Hamilton (1982) in central Alberta concluded that TP export from a watershed is proportional to the concentration of cattle. Export values in this study ranged widely, with the range attributed to the location of cattle relative to the watercourse, vegetation and ponds along the course, and the nature of the stream bed. Many studies have shown that the presence of cattle caused an increase in phosphorus export. A review paper by Gillingham and Thorrold (2000) cites several studies that stated the presence of livestock results in the deterioration of stream quality due to increased phosphorus concentrations. Jawson et al. (1982) observed the highest phosphorus concentrations in samples collected when cattle were present; however, the low flows associated with these samples resulted in relatively small loads and minimal impact on receiving streams. Overall, the data indicated a trend toward increased phosphorus when animals were grazing and the authors suggested the phosphorus losses observed could result in deterioration of receiving waters. Schepers and Francis (1982) concluded that increased amounts of phosphorus were exported when grazing livestock were present at the time of runoff compared to when they were absent.

Wintering Sites

As discussed previously, snowmelt runoff can account for a large percentage of the total runoff in a watershed. On the Canadian prairies, greater than 80% of the total runoff occurs in the spring (Nicholaichuk 1967). Conditions are favorable for large amounts of surface runoff as the soil is frozen and infiltration is restricted (Granger et al. 1984). Areas used for winter feeding are of higher risk as they may have greater amounts of soil and plant disturbance resulting in an increase in surface runoff compared with grassland areas not used as over-wintering sites (Chichester et al. 1979).

As the majority of surface runoff can occur during the spring melt period, a large percentage of annual exported phosphorus may be removed at this time (Timmons and Holt 1977). Export values of TP reported by Chichester et al. (1979) on grazed areas range from 0.3 to 1.1 kg ha^{-1}. In the Haynes Creek study in Alberta, TP export from cattle wintering sites was in the range of

0.04 to 0.37 kg ha^{-1}. Export values for DP in Haynes Creek were in the order of 0.04 to 0.26 kg ha^{-1}, showing that the dominant form of phosphorus in the runoff at these sites was DP (Anderson et al. 1998).

The form of phosphorus was not always related to the degree of sediment removed from the area, indicating that erosion is not the major mechanism mobilizing phosphorus (Nash et al. 2000). Some studies observed high sediment loads with a greater percentage of the phosphorus in dissolved form. A study by Van Keuren et al. (1979) recorded no measurable soil loss during the study period from the summer-grazed grass watersheds and minimal TP losses (<0.1 to 0.1 kg ha^{-1}) while the winter-use area had considerably greater soil loss as well as greater phosphorus loss (0.3 to 1.2 kg ha^{-1}).

Fertilizer Impacts

Application of commercial fertilizers and manure for fertilizer can impact the amount of phosphorus entering watercourses. The common method of applying fertilizers to grassland is by broadcasting it on the surface. This practice leaves the phosphorus vulnerable to transport by surface runoff. The timing of application, the physical properties of the fertilizer, and the risk of erosion are factors that affect the amount of phosphorus in runoff (Gillingham and Thorrold 2000).

Nash et al. (2000) found significant amounts of phosphorus in dissolved form exported during rainfall runoff. There was a weak relationship between sediment losses and percent of phosphorus in dissolved form, indicating low erosion potential at this site. There was, however, a significant amount of phosphorus exported from the site, mainly due to rainfall runoff. As the main fraction of phosphorus exported was <0.45μm, standard methods of reducing nutrient transfer, such as buffer strips, may not be as effective. Alternative methods such as minimizing the amount of phosphorus susceptible to runoff, may be a more effective approach. Conversely, Heathwaite (1995) found that phosphorus transfer was related to the magnitude of the sediment load. Buffer strips were effective in trapping particulate matter and reducing phosphorus concentrations from that fraction. Some of the PP was thought to be related to the removal of undissolved fertilizer granules in the runoff samples.

Preedy et al. (2001) added phosphorus fertilizer, dairy slurry, and fertilizer and slurry to lysimeter plots. The greatest amount of PP was released from slurry. At lower runoff volumes, the proportion of DP increased, with the greatest contribution to DRP coming from phosphorus fertilizer and fertilizer and slurry treatments.

Several factors may affect the degree of phosphorus export from grassland under grazing. Nash et al. (2000) examined the relationships between TP and days since grazing (DG), total storm flow (TF), as well as season. When compared to fertilizers, the impact of manure was small, indicating that cattle did not mobilize large amounts of DP. They calculated a phosphorus addition of approximately 6 kg^{-1} ha^{-1} yr^{-1} from manure compared with fertilizer additions of 35 kg^{-1} ha^{-1} yr^{-1}. As well, the larger surface area of the fertilizer granules would make its potential mobility greater than that of the phosphorus contained in manure. Dissolved forms of phosphorus made up the majority of the runoff, with DRP making up 93% and DP 96% of the TP exported.

The presence of cow manure was listed as a contributing factor to the amount of DP exported in a grassland catchment in Australia (Nelson et al 1996) and in Nebraska (Schepers et al. 1982).

Smith et al. (1992) observed that exported amounts of soluble phosphorus (SP) from grasslands in the southern United States plains were generally small and similar to levels observed in other studies. They concluded that better cover resulted in lower sediment and nutrient discharge. The extent of the vegetative cover appeared to be more important than the quality of the cover.

Seasonally, studies have shown that phosphorus losses from plots were greatest during the initial runoff events (Jawson et al. 1982; Edwards and Daniel 1994). The amount of time since fertilization also impacts the amount of phosphorus exported. Nash et al. (2000) found that just more than half of the variability in TP was accounted for by the number of days since fertilization (DF). The inverse relationship noted between TP and DF was significant. Jawson et al. (1982) observed that the quality of runoff can begin to approach background levels after a relatively small number of runoff events.

IRRIGATED LANDS

Introduction

Irrigation water is applied to approximately 5.4% or 600 000 ha of Alberta's cultivated lands; the largest irrigated area in Canada. A large distribution and return flow infrastructure, consisting of more than 7500 km of irrigation canals, has been constructed to support this development. Irrigation increases the diversity of crops that can be grown in this semi-arid climate, and provides water for confined feeding operations, rural residents, and small communities. However, irrigated areas are often associated with increased agricultural inputs, such as fertilizers and pesticides (Bevans et al. 1998). Irrigation return flow streams, composed of excess supply water, surface and subsurface runoff, tile drainage, and groundwater, can adversely affect water quality in receiving bodies due to excess nutrients, sediment, salts, pesticides, and pathogenic bacteria (Bevans et al. 1998; Cuffney et al. 2000; Cessna et al. 2001). In the past, most studies examining the impact of irrigation return flow streams have focused primarily on salts (Branson et al.1975).

Methods

Most studies of irrigation return flow water quality are carried out at the watershed scale. Phosphorus concentrations or loads in supply waters are often compared with those in irrigation return flow streams (Miller et al. 1977). These loads are often converted to export coefficients to determine phosphorus flux per unit area from various land uses (Greenlee et al. 2000). Other studies have attempted to determine potential impact of irrigation return flows on receiving bodies using relative loads of phosphorus in the return flows and in the receiving bodies (Cessna et al. 2001). Flows from irrigation return flows are usually relatively low compared with those in the receiving bodies; therefore, few studies have identified irrigation return flows as having significant impact on water quality, despite having significantly higher phosphorus concentrations. However, in both of these cases, it is difficult to determine the proportion of the load that comes from runoff from the land and the amount that is generated or attenuated within the stream itself.

Phosphorus Flux from Irrigated Lands

Phosphorus losses from irrigated lands may be greater than from dryland due to increased soil-water contact, thereby increasing the amount of phosphorus in solution (Sharpley et al. 1999). Losses of PP can also be a concern with some methods of irrigation and crop management. In semi-arid regions, irrigation vastly increases the probability of overland flow due to increased antecedent soil moisture. All of these factors suggest that losses of phosphorus from irrigated lands may be greater than from corresponding dryland plots; however, there have been very few studies that directly compare phosphorus flux from dryland and irrigated watersheds.

Irrigation methods. The amounts and fractions of phosphorus lost greatly depend on the irrigation method employed. Sediment loads, which are associated with PP, tend to be much greater in flood-irrigated (Ebbert and Kim 1998) and furrow-irrigated fields (Carter et al. 1971; Lentz et al. 1998). Bondurant (1971) mathematically demonstrated that the transfer of phosphorus from soil to solution in irrigation furrows was minimal and consequently, many authors have suggested that by controlling sediment losses from irrigation, phosphorus losses will be avoided (Carter and Bondurant 1977). However, other studies from flood-irrigated regions seem to contradict these conclusions. In a flood-irrigated district of Saskatchewan, TP concentrations increased between two and five times more than inflow water while dissolved ortho-phosphorus concentrations increased up to 14 times (Cessna et al. 2001). Similarly, in a study conducted in the Eastern Irrigation District near Rolling Hills, Alberta, where approximately 40% of the fields are flood-irrigated, TP concentration doubled between irrigation source waters and return flows, whereas DP increased by a factor of 10 (Chambers and Ferguson 1991). The lower losses of DP predicted by Bondurant (1971) may be due to the minimal contact between soil and water in furrow irrigation methods and the high concentrations of DP in the irrigation supply water (Carter et al. 1974). Westermann et al. (2001) also cited furrow length, and hence, soil-water contact time, as a key factor influencing phosphorus losses.

In properly managed sprinkler-irrigated fields, runoff and sediment loads are minimized; however, DP loads may still be significant. In the Lethbridge Northern Irrigation District, Greenlee et al. (2000) reported increases in SRP concentrations between irrigation supply and return flow waters. Results were limited due to the high detection limits for SRP. Concentrations of DP and TP were also elevated in irrigation return flows compared with the Lower Little Bow River in Alberta (J. Little, AAFRD, unpublished data).

Phosphorus sources. Phosphorus in irrigation return flow streams is usually enriched compared with supply waters due to runoff from adjacent irrigated lands. However, it is often difficult to determine how much of the increase in phosphorus is due to inputs from the adjacent lands and the proportion that is due to erosion and other processes of production and attenuation within the stream channel itself. Phosphorus concentrations are often rapidly attenuated within return flow channels (Oosterveld and McMullen 1979).

Joseph and Ongley (1986) attempted to determine changes in water quality in irrigation return flows and to identify the sources of phosphorus within the irrigation return flows. They found that DP was enriched by a factor of four in irrigation return flows compared to supply waters. Total phosphorus loads were also higher in return flows and were related to the amount of suspended sediment in the water. Detailed fractionation of sediment phosphorus from the channel beds indicated that phosphorus in return flows had similar proportions of phosphorus fractions to those found in spring sediment loads in the receiving waters of the Bow River and the Oldman River. The authors, therefore, suggested that sediment-associated phosphorus was due to natural erosion and not anthropogenic sources. The impact of irrigation on hydrology must be noted as patterns of sediment-associated phosphorus were different in the receiving rivers in the summer and fall seasons, but remained similar in the irrigation return flows. While sediment may be due to naturally occurring erosion, it is likely accelerated in irrigation canals.

Phosphorus flux from irrigated watersheds. Olness et al. (1975) found that losses of total phosphorus were much greater from watersheds containing irrigated cotton (11.15 kg $ha^{-1}yr^{-1}$) than from dryland cotton (5.01 kg $ha^{-1}yr^{-1}$), alfalfa (2.48 kg $ha^{-1}yr^{-1}$), and wheat (2.94 kg $ha^{-1}yr^{-1}$). Dissolved phosphorus flux was also higher from irrigated watersheds (1.93 kg $ha^{-1}yr^{-1}$) than from dryland watersheds (0.71 to 1.23 kg $ha^{-1}yr^{-1}$). The irrigated watersheds were fertilized, but it was estimated that inorganic fertilizers accounted for only 1.5 kg ha^{-1} or about 3% of applied phosphorus. Estimated phosphorus losses were much greater than from other watersheds, which the authors attributed to a combination of above average precipitation and timing of tillage operations.

In Alberta irrigation districts, export coefficients for SRP had a wide range, from a low of 0.001 kg ha^{-1} yr^{-1} to a high of 0.19 kg ha^{-1} yr^{-1} (Greenlee et al. 2000). This estimate must be interpreted with caution as the majority of samples were below the relatively high detection limit for SRP (0.05 mg L^{-1}). During a storm event in 1999, export coefficients for irrigated land in the Battersea Drain, near Lethbridge, Alberta, were estimated at 0.002 to 0.048 kg ha^{-1} d^{-1} for TP and 0.001 to 0.029 kg ha^{-1} d^{-1} for DP (J. Little, AFFRD, unpublished data). Similarly, predominantly irrigated watersheds in the Lower Little Bow basin had export coefficients ranging from 0.004 to 0.019 kg ha^{-1} d^{-1} for TP, while dryland sub-basins had export coefficients ranging from 0.002 to 0.006 kg ha^{-1} d^{-1} during the same storm event. Export coefficients were highly dependent on watershed area.

As discussed previously, estimates of phosphorus flux from furrow-irrigated watersheds are generally higher than sprinkler-irrigated systems. Although differences were not significant, Oosterveld and McMullin (1979) reported slightly lower losses of 0.2 kg ha^{-1} yr^{-1} for a flood-irrigated watershed and 0.3 kg ha^{-1} yr^{-1} for a sprinkler-irrigated watershed in Alberta. These amounted to approximately 1.0% of the total applied phosphorus in the watersheds. In a flood-irrigated watershed in Saskatchewan, phosphorus concentrations were elevated in irrigation return flow streams. Although phosphorus losses amounted to 2.2% of the fertilizer phosphorus applied during one crop year, there was no impact on the receiving body due to the relatively low flows of the irrigation return flows (Cessna et al. 2001).

Irrigation can sometimes result in net inflows of sediment and phosphorus, especially if supply water quality is high in these parameters. Results from irrigated pasture in Nevada were also variable, indicating a net influx of SRP in irrigated watersheds using water high in total dissolved solids (TDS), and net export of phosphorus in watersheds with lower TDS in supply waters (Miller et al. 1977). In several studies of irrigation of the Snake River, Idaho, Carter et al. (1974) reported that net inflow of phosphorus exceeded the outflow in two large furrow-irrigated tracts of land. However, TP concentrations in river water were quite high and the furrows provided an opportunity for adsorption by sediments. Export coefficients suggest that a large quantity of phosphorus was still exported from the irrigation tracts (0.15 to 0.63 kg SRP ha^{-1} yr^{-1}; 0.61 to 2.31 kg TP ha^{-1} yr^{-1}). Furthermore, differences in phosphorus losses are likely attributable to reduced flows in irrigation return flows compared with supply waters.

FORESTED LANDS

Introduction

Nutrients in forest soils are influenced by soil composition, stand age, tree species, intrinsic properties of the site and environmental conditions (Huang and Schoneau 1996). Soil nutrients are also influenced by fire, windthrow, and insect outbreaks, which in turn, are strongly affected by climate and weather (Schindler 1998). Until a fire outbreak, biomass and nutrients appear to accumulate in forest systems. A few to several years following a fire, increases in the chemical and hydrological outputs from streams draining the burned areas are evident, depending on the severity and season of fire, climate, and weather (Schindler 1998).

Studies have been conducted in the Canadian boreal forest to examine the impacts of logging on surface water quality (Devito et al. 2000). Logging may impact runoff volumes, increasing the mass of nutrient loss to surface water. Kachanoski and de Jong (1982) studied the water cycle in forested and clearcut sites in northern Saskatchewan. They reported 63% of water loss was in surface flow from forest plots during snowmelt in spring. This percentage likely reflected frozen topsoil that prevented infiltration. About 25% of summer rainfall was lost out of the rooting zone as drainage. With such a large percentage of precipitation exported from forested sites, it is surprising that nutrient exports are quite low compared to agricultural land.

Methods

Most studies examining loss of phosphorus from forested land compare upstream and downstream loadings instream to different types of land use.

Phosphorus Export

Several authors have reported that forested lands discharge less phosphorus than agricultural or urbanized land (Keup 1968; Taylor et al. 1971; Cooke and Prepas 1998). Taylor et al. (1971) reported larger, average concentrations of TDP lost from farmland (0.022 mg L^{-1}) than from woodland (0.015 mg L^{-1}) in a 3-yr period. When reported as DP carried annually by streams, again, more phosphorus was lost from farmland, 0.2233 kg ha^{-1} yr^{-1}, than from woodland, 0.1424 kg ha^{-1} yr^{-1}.

Runoff plots in an undisturbed aspen-birch forest were studied by Timmons et al. (1977). Annual TP loss of 0.13 kg ha^{-1} yr^{-1} was reported, with more soluble inorganic phosphorus (ortho phosphorus) than organic phosphorus transported in surface runoff during two of the three years. The overall average showed ortho phosphorus and organic phosphorus were nearly equal. Average annual TP loss in the interflow was only 18% of annual TP loss in surface runoff.

Studies of phosphorus input to lakes have mainly concentrated on stream and overland runoff, and precipitation. McCullough (1998) investigated the importance of litterfall from forests as a direct source of phosphorus to lakes. The author found that PP made up 72% of all phosphorus measured, with a greater proportion of PP in May and June, corresponding to jack pine pollen dispersal. Phosphorus concentrations were highest in insects (9 mg g^{-1}) and lowest in broad leaves and needles (McCullough 1998). The jack pine withdrew approximately two-thirds of phosphorus before the needles fell from the tree. In summary, litterfall phosphorus concentration in the littoral zone was dependent on wind, rain, and biological productivity and activity.

Dillon et al. (1991) studied 32 forested headwater streams and reported TP exports were greater than atmospheric inputs, but the additions were likely from peat deposits or beaver ponds, which were found in most catchments where export exceeded deposition.

URBAN AND INDUSTRIAL POINT SOURCES

Introduction

Phosphorus is integral to our daily lives, not only as fertilizer for food production, but also as a product required in manufacturing or as an additive used in food products and in biochemical reactions. Waste from human activities, such as municipal sewage effluent and industrial wastewater, have been discharged into receiving waters such as lakes and rivers for dilution and purification. Although they are highly concentrated, they are generally easier to contain and manage than non-point source pollution.

Methods

Municipal wastewater treatment plant (MWTP) effluent, stormwater, and industrial effluent are considered point sources of pollution. The methods employed in researching these point source contributions to phosphorus loads of receiving bodies consist of comparing concentrations in the discharge stream and in the receiving body (lake or river) prior to and after the point source input. This can be tested temporally as in the study at Field Lake, Alberta, where water quality sampling was conducted before and after effluent discharge began (Mitchell 1998). However, in most other studies, the contributions are measured spatially, such as in urban areas where stormwater or MWTP effluent is monitored and compared to river water quality upstream and downstream from the effluent discharge (Alexander and Stevens 1976; Dixon 1994).

Municipal Wastewater Effluents

Processes. Discharges from MWTPs, combined sewer and stormwater systems are the largest pollution source by volume to water in Canada (Jefferson 2001). However, between 1983 and 1996, phosphorus loading from MWTPs has decreased by 37% (Chambers et al. 2001). Canada-wide, human waste is the largest contributor to municipal phosphorus loading, while dishwasher and laundry detergents and general cleaning products contributed ≥7% of the municipal phosphorus load (Environment Canada 1996).

In the early 1940s, sewer systems discharged household or sanitary wastewater and stormwater directly into receiving waters. As the deleterious effects of raw sewage on aquatic systems became apparent, most sewers were diverted to wastewater treatment plants. When rain or snowmelt exceeded the design capacity of the MWTP or sewer system, raw sewage and stormwater were released into receiving waters. In the event that volumes exceeded capacity, sewage water and stormwater were mixed in combined sewer overflows (CSO). These types of systems still exist in older parts of urban centers, while new systems built are separate system designs with separate pathways for sewage and stormwater.

Combined sewer overflow and stormwater are not routinely measured in Canada. Stormwater releases are confined to wet weather periods and concentrations are more dilute than raw sewage, treated sewage or CSO. Combined sewer overflow represents about 5% of MWTP annual discharge in Canada and is more dilute than raw sewage, but has higher nutrient concentrations than stormwater or treated sewage (Chambers et al. 2001). A rule of thumb is that stormwater for a given urban population will contribute about one tenth of the phosphorus load of a properly run secondary sewage treatment plant (USEPA 1983). Additionally, bioavailable phosphorus forms account for 65 to 100 % of sewage phosphorus, while stormwater is largely particulate (Chambers et al. 2001).

Sewage sludge refers to the organic and inorganic solids resulting from decomposition and settling of wastewater as it undergoes treatment (Warman 1997). Sludge is produced in every step of wastewater treatment, and can be incinerated, deposited in a landfill, or applied on land as an agricultural amendment. The quantity of sludge increases as sewage flows through primary, secondary, and tertiary treatment (Black et al. 1984). Biosolids are the portion of sludge that has been stabilized through digestion and meets public health regulations for land application.

The process of dealing with municipal wastewater has been an ongoing challenge for urban centers around the world. The technology to deal with this type of wastewater is increasingly complex and has decreased the amount of phosphorus released into receiving bodies. However, each individual MWTP is different in the manner it receives effluent, the population it serves, and the treatment process it employs. All these variables contribute to the complexity of determining the overall contribution of MWTP to the phosphorus load of a receiving water body.

Sources. In Canada, household TP released to MWTPs is estimated at 3.38 g capita^{-1} d^{-1}, or 35 588 t yr^{-1}. Depending on the sewage treatment, 63 to 93% of the phosphorus is removed (OMEE 1993). In the prairie provinces of Canada, secondary and tertiary treatments serve most of the population, and all major cities have at least secondary treatment. Within Alberta, Calgary is implementing advanced phosphorus removal, while Edmonton is separating some of its combined sewer systems that lead to CSOs (Chambers et al. 2001).

Alexander and Stevens (1976) examined a housing estate that contributed MWTP effluent to Lough Neagh in Northern Ireland. The housing estate was free of industrial effluent and had separate sanitary and storm drainage. Phosphorus contributions from domestic sewage were determined to be 1.8 g of TP capita^{-1} d^{-1} (Alexander and Stevens 1976). This number is comparable to the estimated value in Ontario, if the lowest percentage (63%) of phosphorus is removed through treatment (2.03 g capita^{-1} d^{-1}) (OMEE 1993).

There have also been studies done to characterize other components of urban sewer contributions. The City of Calgary, Alberta, uses an impoundment on the Elbow River, the Glenmore Reservoir, to supply half of its drinking water (Dixon 1994). Stormwater sewer outfalls were monitored from 1989 to 1993 to characterize the quality of the stormwater discharging into the Glenmore Reservoir and to estimate annual loading of pollutants under baseflow and storm events (Dixon 1994). According to literature reviewed by Dixon (1994) from 1974 to 1989, mean TP from residential land use was 0.04 to 0.98 mg L^{-1} and from city center land use was 0.66 to 0.90 mg L^{-1}. Dixon (1994) found the mean TP in the base flow was 0.059 mg L^{-1}, and in the storm runoff the average flow-weighted TP was 0.418 mg L^{-1}. The total annual TP exports per unit area were 0.201 kg ha^{-1} yr^{-1} at one outfall and 0.156 kg ha^{-1} yr^{-1} for the other, with an average of 0.179 kg ha^{-1} yr^{-1} (Dixon 1994). Dixon (1994) suggests that differences between TP exports between the two urban catchments may be due to the greater amounts of arterial and interchange area, and therefore, greater automobile traffic in one of the catchments.

In the 1980s, there was recognition that wetlands could be used to treat MWTP and stormwater effluents (Cooke 1992). In 1983, the town of Lac La Biche, located 220 km northeast of Edmonton, began using a new sewage treatment plant that discharged treated effluent into nearby Field Lake (Mitchell 1998). The sewage treatment plant is a three-cell aerated lagoon with a continuous discharge of approximately 2000 m^3 day^{-1} (0.023 m^3 s^{-1}) (Mitchell 1998). Annual TP loading after sewage effluent discharge was 3161 kg yr^{-1}, based on the approximate flow volume in 1997 and measured concentration of phosphorus in the effluent in 1993 and 1994 (Mitchell 1998). The lake was eutrophic prior to the discharge and has deteriorated to a hypereutrophic state (Mitchell 1998). After 4 yr, the phosphorus concentration was 14 times higher than prior to effluent discharge, and by 1997 the average phosphorus concentration had doubled 1986 levels (28 times higher than prior to effluent discharge) (Mitchell 1998).

Industrial Discharges

Most light industries discharge their wastes into nearby MWTP, landfills or the atmosphere, whereas large industries (pulp and paper mills, mining operations and large manufacturing plants) independently manage their wastes. Permits, which require adherence to strict guidelines, must be obtained prior to effluent discharge.

One large industry prevalent in Alberta is pulp and paper. Previously, environmental concerns with pulp mills were the effects of high biochemical oxygen demand (BOD), organochlorines, and colour (Chambers et al. 2000). Upgrades and design improvements to pulp mills have reduced these concerns; however, the effect of nutrient loading on receiving bodies has now come to light (Chambers et al. 2000). Bothwell (1992) concluded that the role of pulp mill effluent in river eutrophication is not appreciated due to the misconception that nutrient concentrations must be elevated substantially to have an effect. Mining operations, agro-industry, and electrical power generation are other point sources of phosphorus pollution in Alberta, but contribute much less to the phosphorus load of rivers and lakes.

Sources. In the 1970s, pulp mills discharged effluent into the northern rivers of Alberta. A bleached kraft pulp mill was opened in 1973 on the Wapiti River south of the Grande Prairie, Alberta (Noton et al. 1989). Impacts on the river became apparent in 1979 when the mill's processes changed and low winter flows caused concern with residents downstream in the town of Peace River (Noton et al. 1989). In general, background levels of phosphorus in the upper reaches of the Wapiti and Smokey Rivers averaged 0.12 mg L^{-1}, with higher values observed during spring flows (Noton et al. 1989). Total phosphorus from the effluent discharges varied, with the final effluent concentrations averaging 1.23 mg L^{-1}, while the storm sewer and cooling water effluent contained 0.47 mg L^{-1} (Noton et al. 1989). When calculated as average load, the mill only accounted for 3% of the estimated load above the Wapiti confluence with the Smoky River, reflecting high flow and naturally high TP concentrations in the Wapiti River. Overall, Noton et al. (1989) found that the pulp mill had a greater impact on the Wapiti River during low flow periods.

Coal mining is another important industry in Alberta. In 1984, an investigation was conducted to determine water quality up and down stream from the Coal Valley Mine, owned by Luscar Ltd. and operating since 1915 (Trew et al. 1990). Forms of phosphorus increased significantly downstream from the lease area (Trew et al. 1990). The Lovett River receives the point and non-point sources from the mine and then discharges into the Pembina River. Effects measured 13.5 km downstream from the outfall showed statistically significant increases in SRP. In the Lovett River, the median TP concentrations ranged from 0.012 mg L^{-1} to 0.029 mg L^{-1}, with the only significant change occurring at sites upstream and downstream effluent additions (Trew et al. 1990).

Combination of Industrial and Municipal

Nutrient studies in Alberta have tended to examine the point sources of municipal waste and industrial contributions together. These lead to larger watershed or basins studies such as the Northern River Basins Study (Chambers 1996) or the Frank Lake study in southern Alberta (White and Bayley 2001). These large regional basin studies have occurred in other provinces as well, including British Columbia where the Thompson River near Kamloops has been intensively studied (Bothwell and Culp 1993).

Sources. In southern Alberta, Frank Lake, a 1246-ha bulrush marsh, was restored with additions of municipal and agro-industrial wastewater (White and Bayley 2001). Secondary wastewater from a beef slaughterhouse and local municipality are discharged continuously into the lake. With a mean annual inflow of wastewater greater than 5000 m^3 d^{-1}, the wastewater contributes 11 mg L^{-1} of SRP and a total of 23 000 kg of phosphorus annually (White and Bayley 2001).

Although it produced lower volumes than the MWTP, the slaughterhouse contributed more than 80% of the TP load, with the remaining 6 to19% contributed by the MWTP (White and Bayley 2001). The average TP loading from Cargill meat packing plant and the MWTP is 4.70 g m^{-2} yr^{-1} (White and Bayley 2001). However, based on calculations from their numbers on phosphorus loading by source, the average TP loading from the Cargill Slaughterhouse was 5.00 g m^{-2} yr^{-1} and from the MWTP was 0.77 g m^{-2} yr^{-1} (White and Bayley 2001).

In northern Alberta, a large study was conducted to address nutrient additions from effluent to the Athabasca, Peace, and Slave River basins (Chambers 1996). These basins receive numerous contributions from municipal and industrial effluents. The study focused on continuous sources and not on contributions that are discharged on an annual or biannual basis.

The Athabasca River receives continuous point source effluent from nine MWTPs, while the Peace River receives effluent from six MWTPs (Chambers 1996). These two rivers converge into the Slave River, which then receives effluent from one additional MWTP (Chambers 1996). The contributions of all of the MWTP ranged from 0.2 to 4.6 g $capita^{-1}$ d^{-1} of TP (Chambers 1996). These per capita values for effluent TP are consistent with other studies that put the contributions of TP from MWTP around 2 g $capita^{-1}$ d^{-1} (Alexander and Stevens 1976; OMEE 1993).

In addition to the MWTP contributions, there are seven pulp mills, and three non-pulp mill industries that contribute continuous point source effluent to the Athabasca, Peace, and Slave River systems (Chambers 1996). The pulp mills contribute anywhere between 34 and 101 kg d^{-1} of TP (Chambers 1996). The three non-pulp mill industries consist of an oilsands project discharging utility wastewater with 6.86 kg d^{-1} of TP, a thermoelectric power station discharging process wastewater with 0.18 kg d^{-1} of $TP_{,}$ and an abandoned oil well discharging water with 0.03 kg d^{-1} of TP. All of these activities were noted as having an adverse effect on water quality (Chambers 1996).

The lowest TP concentrations in the Athabasca River were found above Jasper and increased downstream beyond Fort McMurray during the spring, fall, and winter (Chambers 1996). High flows in the summer may mask any impacts of nutrient loading (Chambers 1996). Anthropogenic contributions on an annual basis contribute 6 to 16% of the TP load in the Athabasca River and 22% of the TP load discharged from the Wapiti River to the Smoky River (Chambers et al. 2000). Total phosphorus, TDP, and SRP concentrations were 2 to 3 fold greater downstream from each effluent discharge on the Athabasca and Wapiti Rivers in the fall of 1994 (Chambers et al. 2000).

British Columbia. In 1972, Weyerhauser Canada started discharging kraft pulp mill effluent (KME) into the Thompson River in south-central British Columbia. By the fall of that year, there were numerous complaints of degraded water quality downstream (Bothwell and Culp 1993). The Thompson River is a nutrient-poor river, very low in phosphorus with background levels of 1 to 2 µg L^{-1} SRP (Bothwell 1992). A Federal-Provincial Task Force identified phosphorus from the pulp mill and Kamloops MWTP effluent as the agent responsible for the degraded water quality in 1976. After the release of the Federal-Provincial Task Force Report, the city of Kamloops began a phosphorus removal program and a winter hold back program (Bothwell and

Culp 1993). These procedures would essentially eliminate the discharge of DP between December and March, and markedly decrease phosphorus discharge throughout the year (Bothwell and Culp 1993).

As a result, Bothwell and Culp (1993) found that in the Thompson River SRP concentrations were highest in the winter (~5 to 7 µg L^{-1}) and lowest in the fall (~1 to 2 µg L^{-1}). SRP concentrations declined downstream from the pulp mill and this trend was most evident during late winter.

Discussion

Rivers having high nutrient concentrations prior to discharge, or already receiving large quantities of discharge, have a smaller potential to show primary production effects as a result of effluent discharge, than if the receiving body is nutrient poor (Chambers et al. 2000). The impacts of effluent discharge may be more prominent in low flow situations, where less dilution ability can lead to greater impact from low phosphorus concentrations (Chambers 1996). However, Bothwell (1992) found that even with KME discharge concentrations below 0.5 mg L^{-1} of DP (1 mg L^{-1} TP) and river dilutions of 100 fold, the continuous elevation of soluble phosphorus was still high enough to trigger algal production in rivers that were phosphorus limited.

The ability of water bodies to sustain constant loading of MWTP effluent has been questioned (Cooke 1992). Although significant responses have been seen when phosphorus removal systems are implemented at the point source and nonpoint sources of phosphorus are curtailed, internal mechanisms, such as phosphorus stored in the system by the vegetation, detritus, and sediments may make high levels of phosphorus available for years or decades (Larsen et al. 1979). For example, Frank Lake in southern Alberta is still a net nutrient retention marsh; however, it was suggested that continued high nutrient loading to the marsh could result in phosphorus export (White and Bayley 2001).

In addition, point sources are generally much easier to control. As noted by Fraser (1987), the point source and tributary phosphorus load to Lake Erie, Canada, had decreased by 50% between 1967 to 1987; however, the real challenge was to control the non-point sources coming from different land uses.

Summary

The contributions of phosphorus from urban sources come in many different forms, such as MWTP effluent, stormwater effluent, and industrial effluents, and these are all point source phosphorus loads. The phosphorus contributions of these point sources have decreased in recent times with the advent and implementation of new, more efficient, phosphorus removal technology.

Based on our findings, MWTP effluent, in a worst-case scenario, can be estimated at 2 g $capita^{-1}$ d^{-1}. Stormwater runoff is more dilute than the MWTP effluent, and although CSO effluent has higher concentration of nutrients, it only accounts for 5% of total effluent discharge annually. However, with municipal and industrial effluent, the contribution of phosphorus to the load in the receiving body depends on the degree of effluent treatment, the receiving waters ability to dilute, and the morphological characteristics of the receiving water. Therefore, each system is unique and must be addressed individually.

SEPTIC SYSTEMS

Introduction

According to the 2001 Statistics Canada census, 19.1% or 569,647 Albertans live in rural areas. These areas are defined as areas lying outside urban areas and are likely serviced by private septic systems. Septic systems are household wastewater disposal systems composed of a septic tank and a septic field. Solids settle out in the septic tank, while less dense materials float to the surface. The remaining effluent is distributed to a septic field (also known as a tile bed or leaching field) via perforated pipes. The effluent then slowly drains through a gravel bed to the soil below (Chambers et al. 2001).

Methods

Phosphorus flux from septic fields is usually via groundwater; therefore, many tools employed in groundwater investigations are used to track septic tank effluent movement to surface waters. Installation of piezometers along groundwater flow paths, phosphorus budgets, and Darcy flux are methods that will be discussed in detail in Part 3: Groundwater. In addition to these methods, researchers have tracked the path of bromide tracers and dyes through household wastewater treatment systems (Robertson et al. 1998). Sosiak and Trew (1996) used a septic leachate detector in their study of Pine Lake, Alberta. This device measured conductivity and fluorescence from organic compounds associated with byproducts of laundry detergents, bleaches, and sewage degradation. Water samples were collected to verify these results.

Phosphorus Movement from Septic Fields

In theory, most phosphorus from septic tanks should be retained within the septic field. However, septic systems must be properly sited and maintained to work efficiently. In a survey of central Ontario cottages, Dillon et al. (1986) reported an average of 61% of septic systems were not properly designed, constructed, or maintained. Furthermore, when phosphorus adsorption capacities in septic fields are exceeded, phosphorus can be leached to groundwater or surface waters. Adsorption capacity in the vadose (unsaturated) zone can be exceeded relatively rapidly, as phosphorus reached the water table after only 2 mo of operation in a medium-textured calcareous sand (Robertson et al. 1998).

Within groundwater, phosphorus moves at a much slower rate than the groundwater itself. Robertson et al. (1998) estimated that the movement of phosphorus in groundwater was retarded by factors of <20 to 100 in 10 central Ontario septic beds compared with groundwater velocity. Phosphorus can be adsorbed onto the surface of several minerals, combining with aluminum and iron hydroxides in acid soils and with calcium compounds in calcareous soils. Despite the observed retardation, phosphorus velocity was still high enough in some cases to be a threat to surface waters (approx. 1 m yr^{-1}).

Concentrations of phosphorus in groundwater are usually about an order of magnitude less than concentrations in septic tank effluent (Reneau and Pettry 1976). Phosphorus concentrations are controlled by soil type, pH, and redox conditions. Under reduced conditions, phosphate concentrations are often decreased due to the precipitation of iron-phosphorus complexes. Phosphate concentrations are also lower under oxidized conditions with low pH compared with neutral or alkaline conditions. Aluminum and iron minerals often have lower solubilities in these conditions and are often precipitated out of solution. The highest concentrations of phosphate in septic plumes were found in oxidizing, calcareous sands at circumneutral pH (Robertson et al. 1998). Calcium-phosphorus complexes are highly soluble under these conditions.

Based on an estimated per capita volume of municipal wastewater of 160 L d^{-1}, and a total phosphorus influent concentration of 15 mg L^{-1}, Chambers et al. (2001) estimated per capita production of phosphorus of 2.4 g d^{-1} or 0.876 kg yr^{-1}. Of this, 25 to 40% is retained within the septic tank, while additional amounts are retained within the septic field. Robertson et al. (1998) calculated average phosphorus retention coefficients of 72% from 10 mature septic systems in Ontario. Using these retention values, the amount of phosphorus released from septic fields in Alberta was approximately 140 000 kg yr^{-1}. However, a much smaller portion of phosphorus enters surface waters. Shannon and Brezonik (1972) estimated that only 10% of phosphorus from septic fields in the immediate vicinity of the lake, and 1% of phosphorus from septic fields in remote areas of the watershed, contributed to Florida lakes.

Theoretical estimates of annual loading from septic fields and sewage lagoons near Pine Lake, Alberta were 79 kg and 39 kg, respectively (Sosiak and Trew 1996). These estimates represent 5.9% of the total annual load to the lake. Mitchell (1997) also reported similar relative contributions to the overall phosphorus budget from Lac Ste. Anne and Lake Isle. However, an extremely high proportion of the phosphorus budget (67%) of Jackfish Lake near Edmonton was attributed to septic systems (Chambers et al. 2001). This lake has a very small watershed area and is surrounded by cottages.

ATMOSPHERIC CONTRIBUTIONS

Introduction

Atmospheric deposition has also been identified as a source of phosphorus to surface waters. However, this source has not been given much consideration until more recently. Atmospheric phosphorus is associated with organic (e.g., pollen) and inorganic dust particles and will contain varying concentrations of phosphorus depending on its source (Brunner and Bachofen1998). Phosphorus can also be released into the atmosphere by microbial reduction methods that generate volatile phosphorus compounds, such as PH_3 or P_2H_4, in sewage sludge, faeces, landfill, and compost (Brunner and Bachofen1998). These gases move into the atmosphere where they are oxidized and released in aerosol droplets.

Methods

Quantifying phosphorus content of air is difficult because of the small concentrations present and because phosphorus is generally in an aerosol form or associated with small particles (less than 0.1 µm). Rainfall concentrations may also be below detection limits, necessitating concentration before analysis (Brunner and Bachofen1998). Error may also result during the filtering process from silicate interference (i.e., in dusts) during the molybdenum blue method, or from phosphate adsorption or release by laboratory glassware (Brunner and Bachofen 1998). Further difficulty arises as phosphorus deposition measures wet and dry deposition. Dry deposition rates will be higher on a wet surface and the phosphorus content of rain may vary. For example, fog water may contain up to 10 times greater concentrations of phosphorus than most rain (Brunner and Bachofen1998). Trew et al. (1978) sampled TP in snowfall and dustfall and white ice during winter and in rainfall and dustfall during summer. Estimates of phosphorus deposition from the atmosphere may be inaccurate due to the difficulty in measurement or experimental method (Brunner and Bachofen 1998).

Atmospheric Losses

Vapourized and uncontaminated condensed water (rainfall) should not contain phosphorus. Most phosphorus in rainfall results from washout of atmospheric particulate material whose composition and quantity governs the concentration in rainfall (Keup 1968). Reported phosphorus concentrations have ranged from trace to as high as 80 µg L^{-1} in a Cincinnati suburb (in Keup 1968). In general, phosphorus deposition ranges between 0.04 and 2 kg ha^{-1} yr^{-1} (Brunner and Bachofen 1998) . A study conducted in the Western Lake Michigan Drainages reported 0.38 kg ha^{-1} of phosphorus deposited from the atmosphere near Chicago and almost a magnitude less in a remote forested area in northern Wisconsin (0.05 kg ha^{-1}) (Robertson 1996).

Sosiak and Trew (1996) reported rainfall, snow, and dry deposition contributed approximately 61.6 kg of TP directly to the surface of Pine Lake (4000 ha) between May 12 and October 26. This represented about 3.1% of the TP load to Pine Lake. This is comparable to Bennett et al. (1999) who reported wet and dry deposition accounted for 4.5% of the total watershed phosphorus budget of Lake Mendota, Wisconsin. Trew et al. (1978) reported 259.1 kg of TP deposited directly onto Baptiste Lake, 33.1 kg were deposited as snowfall in winter, and 225.9 kg as rainfall and dustfall from the period of April 20 to September 5. These values are much greater than those reported by Sosiak and Trew (1996). The annual deposition rate of 33.2 mg m^{-2} yr^{-1} reported by Trew et al. (1978) is also greater than the 20.3 mg m^{-2} yr^{-1} reported by Shaw et al. (1990) at Narrow Lake, Alberta.

Phosphorus deposition from the atmosphere is expected to differ among regions due to factors such as wind, precipitation, vegetation, and land use. In areas having high soil disturbance (e.g., agricultural regions), dry fall out or dust fall may contribute a greater amount of phosphorus per unit surface area of water than atmospheric deposition from forest zones having minimal land disturbance (Trew et al. 1978). Brunner and Bachofen (1998) identified combustion of coal and forest fires as the main anthropogenic sources of atmospheric phosphorus.

PART 3: PHOSPHORUS FLUX WITHIN THE AQUATIC ENVIRONMENT

GROUNDWATER

Introduction

Groundwater can be an important contributor of phosphorus to surface water bodies (Hayashi and Rosenberry 2001; McDowell and Sharpley 2001); however, studies assessing phosphorus contributions from groundwater are limited. There are two major reasons for this lack of data: (1) phosphorus was considered to be immobile within the soil profile due to the high adsorption capacity for phosphate ions by calcite, other clay minerals, and iron and aluminum hydroxides, and (2) groundwater characterization is costly to initiate at the watershed scale. It is now widely recognized that phosphorus, especially organic forms found in manure, can be leached to the groundwater via preferential flow through macropores (Whalen and Chang 2001). Phosphorus can also move downwards when the phosphorus sorption capacity of surface soil is exceeded, as in heavily manured soils or mature or malfunctioning septic systems (Robertson et al. 1998).

Methods

Calculation of groundwater flux to surface water is a two-step process. First, the total flux of groundwater to surface water is estimated by one of several methods described below. This estimated volume is then multiplied by the phosphorus concentration determined from porewater and/or groundwater samples. Groundwater chemistry tends to change significantly in the last few metres before it discharges to surface water, due to reactions with organic-rich surface-water sediments, interaction with riparian vegetation, and mixing with surface water. Groundwater tends to become more reducing in the hyporheic zone, and this can result in denitrification of nitrate (Mengis et al. 2000; Devito et al. 2000;). Two studies in Alberta and a third study in Wisconsin found that phosphorus concentrations tend to be lower in groundwater than in adjacent lake pore water (Brock et al. 1982; Shaw et al. 1990) and seepage-meter water (Zilkey 2001). Increased phosphorus concentrations in the hyporheic zone are attributed to an increase in reducing conditions (Moore and Reddy 1994) and to adsorption on or desorption from particulate matter in lake-bottom sediments (Shaw et al. 1990). Thus, in addition to being a source of phosphorus to surface water, groundwater also enhances phosphorus recycling from lake or stream-bottom sediments to surface water.

There are several different methods employed to measure groundwater contributions to surface waters. The simplest of these is the water balance method, which is a mass balance of all water entering and leaving the water body. Inflows generally include precipitation and surface water inflow, while surface water outflows, evaporation, and change in volume are subtracted. The residual volume remaining after this calculation is assumed to be the contribution from groundwater (Sosiak and Trew 1996). Therefore, groundwater is not measured directly and there can be large errors associated with this method since many of the parameters are difficult to accurately measure (i.e., evaporation, overland runoff directly to water body, sediment release of phosphorus) (Shaw et al. 1990). Often groundwater contribution is overestimated as the residual accounts for all unquantified parameters (Sosiak and Trew 1996).

Groundwater flux can also be measured by calculating Darcy's flux from wells adjacent to the water body or from mini-piezometers within the stream or lake bed itself. This method is generally more reliable than water balance methods; however, there is potential for the mini-piezometers installed within sediments to become plugged with sediment (Cartwright et al. 1979).

Seepage meters can also be used to measure *in situ* seepage flux. Seepage meters are inexpensive, relatively easy to use, and provide direct measurements of seepage flux that can be extrapolated over the entire lake area to give reasonable estimates of total contributions from groundwater. There is some evidence to suggest that seepage meters may overestimate seepage rates (Shaw and Prepas 1989) and that anoxic conditions within the seepage meters may result in overestimations of phosphorus concentrations due to release from sediments (Belanger and Mikutel 1985).

Groundwater Contributions of Phosphorus to Surface Water

Although investigations of phosphorus transport to surface waters from groundwater are relatively rare, two comprehensive studies have been conducted in Alberta. Shaw et al. (1990) quantified phosphorus inputs into Narrow Lake in central Alberta using several methods, including a water balance, Darcy flux from piezometers adjacent to the lake, Darcy flux from mini-piezometers installed in the lake sediments, and flux from seepage meters. Results from the four methods varied widely with the greatest flux of groundwater phosphorus inputs estimated from the water balance method. Although seepage meters may overestimate phosphorus flux due to the creation of anoxic zones within them, this was assumed to be negligible as pore-water phosphorus concentrations in the lakebed sediments were similar to phosphorus concentrations of seepage water. Mini-piezometers installed on the lakebed gave the lowest estimates for groundwater inputs; however, these estimates were deemed unreliable due to obstruction of the intake sites with sediment. Using pore-water phosphorus concentrations with flux data from water balance, piezometers, and seepage meters provided estimates of phosphorus loading of 58, 35, and 23 mg m^{-2} yr^{-1}, respectively, and a mean value of 0.39 kg ha^{-1} yr^{-1}. Overall, groundwater was estimated to have contributed 30% of the water to the lake and was identified as the single most important phosphorus source to the epilimnetic zone of the lake. The annual input of phosphorus by groundwater was nearly five times that from surface runoff.

The second study was conducted in the Battersea Drain, which is part of the Lethbridge Northern Irrigation District. Flux between a shallow aquifer and an irrigation return flow channel was quantified using seepage meters. Piezometers and water table wells were installed to characterize groundwater flow patterns and water chemistry. Overall, annual seepage rates into the Battersea Drain were estimated at 0.02 m^3 m^{-2} d^{-1} or 7.3 m^3 m^{-2} yr^{-1} (Zilkey 2001), with lower fluxes observed during the irrigation season (May to mid-October), and higher values in the late fall and early spring. The flux of TP contributed from groundwater for a 16-ha catchment to the drain was 0.05 kg ha^{-1} yr^{-1} (1.1 g m^{-2} yr^{-1}). Although this phosphorus flux represented only 0.16% of the annual phosphorus mass applied to the land, the flux rate was 20 times higher than the phosphorus flux to Narrow Lake. Dissolved phosphorus flux increased along the groundwater flow path, likely due to reducing conditions. The total amount of DP lost was 0.21 kg yr^{-1} or 0.01 kg ha^{-1} yr^{-1}.

WETLANDS AND RIPARIAN AREAS

Introduction

Natural wetlands have been described as transition zones between uplands and deepwater aquatic systems (Mitsch 1995). Wetlands perform a variety of functions in the watershed, including water storage, groundwater recharge, water purification, sediment retention, flood protection, nutrient recycling, provision of wildlife habitat, and biodiversity (Mitsch and Gosselink 1993; Young 1994). Wetlands are classified according to the area they inhabit. Fringe wetlands are situated around lakes or other deepwater systems, while instream wetlands are part of the river or stream. Riparian wetlands receive only seasonal flooding but are otherwise separated from the waterway except for return flows and lateral flows from uplands (Mitsch 1995). Constructed wetlands, which are constructed by humans in nonwetland areas, have received a lot of attention for their ability to retain nutrients (Kadlec and Knight 1996).

Phosphorus retention in wetlands results from deposition and uptake into storage compartments, namely the soils, vegetation, and plant litter within the wetland ecosystem. If the annual input is greater than output, the wetland acts as a sink, removing phosphorus from circulation (Johnston 1991; Mitsch and Gosselink 1993; Mitsch 1995). Conversely, if the annual input is less than the output, the wetland acts as a source of phosphorus for receiving waters. A single wetland can be a sink for an inorganic form of nutrient and a source for an organic form of the same nutrient (Mitsch 1995). Peverly (1982) reported an instream wetland as a source of phosphorus in the first year of study and a sink for phosphorus in the second year. The most important influence on phosphorus retention is the hydrologic balance, namely retention time of surface water and timing of precipitation events. To a lesser extent, phosphorus retention is influenced by internal processes such as litter fall patterns, plant uptake and production, and the rate of microbial processes (Neely and Baker 1989). The literature contains many studies of standing stocks, but few have documented flux between wetland vegetation, litter, surface water, groundwater, and soil (Johnston 1991).

Methods

The methods used to establish nutrient budgets in wetland ecosystems range from simple “black-box”, inflow-outflow sampling to more intensive investigations of internal processes (e.g., soils, litter, vegetation). Although the former method is useful for net retention, it does not account for internal processes that may be contributing to phosphorus export from the system.

Measurement of standing stock gives an estimate of the total amount of nutrients in wetland storage compartments at a particular time. Vegetation is a dynamic storage compartment that retains and releases phosphorus depending on the time of year. The soil compartment, on the other hand, is a standing stock that changes little with time (Johnston 1991). It is important to appropriately time sampling of dynamic storage compartments so that results can be compared among years of study and with other literature. For example, aboveground biomass sampled in the summer may contain high concentrations of phosphorus, while samples collected after senescence may show reduced nutrient concentrations due to translocation to the root zone. In this example, sampling in summer or in late fall will yield different results.

The capacity of soils to remove phosphorus from surface water is often estimated in the laboratory by equilibrating soil samples with solutions containing inorganic phosphate (Nichols 1983; White et al. 2000). McDowell et al. (2001b) assessed phosphorus sorption capacity of sediment by shaking samples in deionized water containing known concentrations of phosphorus. Curves of phosphorus sorbed against phosphorus in solution were fitted to the Langmuir equation and a sorption maximum was calculated. Phosphorus desorption studies were also conducted by McDowell et al. (2001b) by shaking replicate samples of varying soil to water ratios. Laboratory studies used to determine sediment phosphorus sorption capacity may; however, not be representative of actual storage capacity. Studies investigating long-term applications of phosphorus from wastewater or fertilizer frequently show that soils can fix more phosphorus than estimated by short-term laboratory experiments. Soils thought to be saturated with adsorbed SRP regained their adsorption capacity after 2 to 3 mo. This was likely because of occlusion and precipitation of adsorbed SRP (Nichols 1983).

Phosphorus exchange between the solid and solution phase is understood to be dependent on phosphorus equilibrium between the two phases. The equilibrium theory has been used to determine phosphorus release from soil or sediment to the ambient water (Koski-Vähälä and Hartikainen 2001). Phosphorus uptake from sorption experiments is expressed by one of two indices: the equilibrium phosphorus concentration (EPC), or the sorption index. Simply stated, the EPC is the SRP concentration where no net adsorption or desorption of phosphorus by the sediments takes place (Taylor and Kunishi 1971; Munn and Meyer 1990). The EPC is interpolated from regressions of final SRP concentrations in incubation flasks and the amount of phosphorus adsorbed or desorbed from the sediments (Munn and Meyer 1990). A low EPC value indicates a large capacity of the sediments to adsorb phosphorus. Conversely, a high EPC value indicates a low phosphorus sorption capacity in the sediment (Munn and Meyer 1990; Koski-Vähälä and Hartikainen 2001). The EPC value dictates the critical phosphorus concentration below or above which solids will release or retain phosphorus and thus describes phosphorus loading risk (Koski-Vähälä and Hartikainen 2001). Koski-Vähälä and Hartikainen (2001) reported that EPC increased in order of pH 7 < pH 7 anoxic conditions < pH 9.

The sorption index is calculated as the amount of phosphorus (μg) absorbed by sediment (g^{-1}) from an initial phosphorus concentration of 2 mg L^{-1} divided by the SRP concentration in the solution after a 1 h incubation period. The sorption index is a useful indicator of the phosphorus buffering capacity of sediments, where a large index value indicates that a large amount of phosphorus can be sorbed by the sediment without increasing the concentration in the water at equilibrium (Munn and Meyer 1990). Comparing the two indices, Munn and Meyer (1990) reported that EPC was a better indicator of the sorption capacity of sediments in two headwater streams. The EPC represented sorption at normal stream water concentrations, while the sorption index is calculated from incubations using much greater concentrations than those found in the streams under study.

The best predictor of phosphorus sorption on wetland soil is the amount of oxalate-extractable iron and aluminum in the sediment (Richardson 1985). Simard et al. (1993) found there was a significant relationship between phosphorus retention and the amount of ammonium-oxalate extractable aluminum in the A, B, and C soil horizons. In this study, the rapid phosphorus sorption capacity was measured after 48 h of contact with solutions of 0.005 M $CaCl_2$ containing graded amounts of phosphorus. A composite sample from a large number of samples (> 30) was necessary to give an accurate estimation of the phosphorus retention capacity at a given site (Simard et al. 1993).

In addition to estimating sorption capacity, sedimentation rates have been used to estimate long-term storage of phosphorus in the soil. Richardson and Craft (1993) used stable cesium-137 isotopes to determine short-term (about 25 yr) sedimentation rates. Johnston et al. (1984) also used the depth of cesium-137 to estimate net sedimentation rates in a seasonally flooded lakeside wetland in northeastern Wisconsin.

Processes

Physical processes. The ability of wetland sediments (hydric soils) to store phosphorus is governed by sorption, precipitation, and incorporation (biological immobilization) (Syers et al. 1973; Neely and Baker 1989; Mitsch and Gosselink 1993; Richardson and Craft 1993; White and Bayley 2001). Adsorption of phosphorus in freshwater sediments is a function of the interaction of redox potential (oxygen status), pH, iron, aluminum, and calcium minerals, and the concentration of native soil phosphorus (Syers et al. 1973; Mitsch and Gosselink 1993; Richardson and Craft 1993). Under aerobic conditions and in acid soils, inorganic phosphorus is adsorbed on hydrous oxides of iron (Fe^{3+}) and aluminum (Al^{3+}), and may precipitate as insoluble iron-phosphorus and aluminum-phosphorus (Keup 1968; Mitsch and Gosselink 1993; Richardson and Craft 1993). Acidic to slightly acidic conditions also promote greater phosphorus sorption and precipitation with clay particles (Mitsch and Gosselink 1993). At pH values greater than 7, precipitation as insoluble calcium-phosphorus is the dominant transformation (Richardson and Craft 1993). In anaerobic sediment, described as having redox potential below +250 mV, Fe^{3+} is reduced to the more soluble ferrous iron (Fe^{2+}) and releases associated phosphorus (Mitsch and Gosselink 1993; Richardson and Craft 1993).

Adsorption and deposition of sediment can result in large fluxes of phosphorus from surface water to wetland soils, and is thought to be responsible for most of the nutrient trapping in wetlands (Johnston et al. 1984). Authors have reported that more than 80% of phosphorus entering a wetland is stored in the sediments (Hammer 1989; Richardson and Craft 1993). Although this is a large proportion of incoming phosphorus, removal rates differ according to soil type, land use, and anthropogenic influence. Phosphorus flux by sediment deposition is generally greater in mineral soils compared to organic soils that have almost no capacity to fix phosphorus (Nichols 1983; Johnston 1991). In mineral soils, phosphorus retention averaged 1.5 g m^{-2} yr^{-1} in comparison to 0.26 g m^{-2} yr^{-1} in organic soils (Johnston 1991). Johnston et al. (1984) estimated the accumulation of phosphorus by sedimentation in a seasonally flooded lakeside wetland was 2.6 g m^{-2} yr^{-1}, of which 46% was inorganic phosphorus and 54% was organic phosphorus. Neilson and Mackenzie (1977) found that only 14% of the phosphorus lost from an organic watershed was associated with sediment, suggesting that sediment phosphorus loss was not as important as soluble phosphorus loss in organic areas.

In the Prairie Pothole region of South Dakota, annual accumulation of phosphorus in wetland sediments was 0.57 g m^{-2} yr^{-1} in cultivated watersheds compared to 0.30 g m^{-2} yr^{-1} in uncultivated watersheds (Johnston 1991). Eagle Lake Marsh, Iowa, a prairie marsh receiving agricultural runoff, was able to store 0.62 g m^{-2} yr^{-1} of phosphorus (Davis and van der Valk 1978). Richardson and Craft (1993) reported similar results, stating permanent storage of phosphorus in natural prairie marshes was below 1.0 g m^{-2} yr^{-1} and averaged 0.5 g m^{-2} yr^{-1}. Wetlands receiving no anthropogenic inputs were reported to retain 9 to 80 % of phosphorus loads (0.17 to 0.73 g m^{-2} yr^{-1}) (Johnston 1991).

Cooper and Gilliam (1987) measured phosphorus deposited with sediment in a wooded wetland. The total deposition of phosphorus in the riparian wetland during a 25-yr period was estimated to be half of that entering from agricultural areas. Cooper and Gilliam (1987) found that the EPC of riparian sediments was higher than the DP concentration in surface runoff water entering the riparian areas in their watershed; thus, the sediments would not be expected to remove phosphorus in solution from the runoff water. Lowrance et al. (1984) estimated retention of TP entering the riparian areas of their watershed in Georgia to be 30%. Peterjohn and Correll (1984) found that the DP concentrations in surface water entering and leaving their riparian areas were unchanged. The authors concluded that riparian buffers can remove phosphorus attached to sediment reasonably well, but are relatively ineffective in removing soluble phosphorus.

Phosphorus accretion. Adsorption-precipitation by soils is not necessarily permanent and is at least partially reversible. Laboratory experiments conducted by Johnson et al. (1976) showed sediments release phosphorus when agitated during storm periods. Compared to soils with high phosphorus adsorption capacity, soils with low adsorption capacity can be expected to readily release phosphorus (Nichols 1983). Long-term storage associated with burial in the soil and litter layers is more permanent. Even when other phosphorus storage compartments are saturated, sediment burial can continue to effectively remove phosphorus at a rate similar to the sedimentation rate (Howard-Williams 1985; White et al. 2000).

Frank Lake, Alberta, is a restored, natural wetland that receives secondarily treated wastewater from a beef processing facility and from the Town of High River (White 1997). Slaughterhouse wastewater and municipal wastewater accounted for at least 80% and 6 to 19% of the TP load to the wetland, respectively. Mean phosphorus loading to the wetland was 4.7 g m^{-2} yr^{-1} (White and Bayley 2001). Inflow and outflow sampling showed that SRP was reduced by 57% and TP reduced by 64% between July 1994 and June 1995. White et al. (2000) reported phosphorus burial rates of 38.5 g m^{-2} yr^{-1} near the wastewater outflow compared to other sites across the same marsh (24 g m^{-2} yr^{-1}). Sedimentation rates were reported to be 30 mm yr^{-1} at the point of wastewater discharge. Approximately 60% of phosphorus inputs into Frank Lake since 1990 have been retained in the sediments (79 662 kg out of 141 760 kg applied) (White 1997; White et al. 2000). Compared to reference wetlands, Frank Lake sediments had nearly 50% less phosphorus sorption capacity (White et al. 2000).

Another wetland example in Alberta is the Blue Quill's School wetland wastewater treatment project in St. Paul (Kent 1987). The wetland has been receiving pretreated wastewater for more than 40 yr. High TP and SRP concentrations were found immediately downstream of the discharge point, ranging from 7.6 to 10.8 mg L^{-1} and 6.0 to 7.5 mg L^{-1}, respectively, in surface water (Kent 1987). The author attributed the limited reduction of phosphorus in this zone to saturation of the sediment's adsorption capacity from many years of wastewater discharge. Few submerged macrophytes and lower invertebrate diversity were also found in this zone. However, the author reported spreading of fresh manure on a field adjacent to the south side of the outfall zone, but does not report the impact this practice may have on wetland water quality. Downstream 250 m from the outfall, TP and SRP were reduced by 90% compared to influent concentrations (Kent 1987).

Hydrological processes. In addition to adsorption, sedimentation, and burial processes, phosphorus retention in wetlands is affected by local hydrology. In natural systems, hydrology often results in the seasonal export of phosphorus, while constructed wetlands are usually designed to treat the water that flows through them. Hydrology, retention time, and seasonal variability have all been incorporated into the design of constructed wetlands to maximize nutrient retention. Phosphorus retention tends to be greater in constructed or managed systems than in unmanaged, natural wetlands.

Residence time is defined as the amount of time a fluid spends in a reactor (i.e., a wetland) (Kadlec and Knight 1996). In general, increasing the residence time will result in greater treatment opportunity within a wetland (Raisin and Mitchell 1995). The Des Plaines River Wetlands Demonstration Project is a constructed riparian wetland receiving up to 40% of the river's average flow before being returned to the river through control structures (Kadlec and Hey 1994; Mitsch 1995). Mitsch (1992) reported greater phosphorus retention when flows were maintained at 50 mm wk^{-1} than at 300 mm wk^{-1}. In four of the Des Plaines wetland cells, the proportion of phosphorus retained was 83 to 96% during low flows and 63 to 68% during high flows. Although the constructed cells were able to retain a greater percent of the phosphorus load during low flows, the actual load retained was greater during high flows.

In natural wetlands, phosphorus retention ranges from 5 to 20% of nutrient loads (Neely and Baker 1989; Mitsch and Gosselink 1993), but can vary according to runoff characteristics of the local watershed and water level influences on fringe or instream wetlands. Old Woman Creek, a fringe wetland situated on the coast of Lake Erie, Ohio, received phosphorus loadings of 8.0 g m^{-2} yr^{-1}, but only retained 0.8 g m^{-2} yr^{-1} (approximately 10%) (Mitsch and Gosselink 1993). The effectiveness of this wetland to ameliorate nutrient loading from the agricultural watershed was dependent on the amount of annual runoff and the water level of the lake (Mitsch 1995). Other natural wetlands also retained smaller percentages of phosphorus due to fluctuating flow. Heron Pond, a forested wetland in southern Illinois, retained approximately 3.6 g m^{-2} yr^{-1} or 4.5% of the total phosphorus load (Mitsch and Gosselink 1993). Davis et al. (1981) in Neely and Baker (1989) reported Eagle Lake Marsh retained 20% of the 3.1 kg ha^{-1} SRP load it received. Clausen and Johnson (1990) assessed the role of lake level on TP retention within a lakeside wetland and found export from the wetland decreased as the lake level increased.

Biological processes. Macrophytes, phytoplankton, and microorganisms are all important wetland components capable of assimilating large amounts of phosphorus. However, studies show that the latter two components are more efficient than macrophytes at removing phosphorus from the water column (Howard-Williams 1985; Richardson and Craft 1993). Emergent macrophytes take up phosphorus from the sediment and release much of it back to surface water upon senescence. Few studies have attempted to quantify net annual retention of nutrients in plant biomass (Johnston 1991), but as much as 35 to 75% of assimilated phosphorus can be returned to the water column and litter compartments (Johnston et al. 1991; Richardson and Craft 1993). Estimates of phosphorus retained by woody wetland vegetation were variable (between 0 and 56%); however, the highest retention estimate included storage in root biomass that most researchers ignored (Johnston 1991). Although litter losses have averaged 20 to 45% of plant uptake, the annual deposition of fresh plant litter will aid in long-term incorporation of phosphorus in organic sediments (Davis and van der Valk 1983).

The short-term net effect of rooted emergents is to transfer phosphorus from the sediment to the water column, while root and residual decomposition products result in long-term phosphorus storage via peat accumulation. Furthermore, Davis and van der Valk (1983) suggested nutrient assimilation by roots of emergent vegetation may free exchange sites on sediment particles by stimulating movement of ions from surface water into sediment interstitial water. Kadlec (1986) compared SRP and DP concentrations in surface water and interstitial water in the Delta Marsh, Manitoba. Water 15 cm below the sediment-water interface contained nearly five times the SRP and three times the DP compared to surface water. Although uptake of phosphorus by vegetation may not directly benefit water quality (Johnston 1991; Richardson and Craft 1993), macrophytes may be important mechanisms for increasing the sorption capacity of wetland soils (Nichols 1983)

Seasonality. Many authors have reported the influence of season on a wetland’s ability to retain phosphorus. Season often influences hydrology, as water level and flow rate tend to increase in spring due to snowmelt and precipitation. As discussed previously, increased flow rate and water levels often impact a wetland’s ability to retain phosphorus. Thus, it is not surprising that retention in natural systems is reduced in winter and spring and is greatest in summer and fall when flows are lowest. Kadlec and Hey (1994) reported removal efficiencies from 60 to 100% in summer and 27 to 100% in the winter. Despite higher incoming concentrations during summer, there are higher removals of phosphorus (Kadlec and Hey 1994), likely due to incorporation into living biomass of bacteria, algae, and vascular macrophytes that are more productive at this time (Mitsch and Gosselink 1993).

Seasonal differences were also apparent at Frank Lake, Alberta, as average TP reduction decreased from 71% in May to October to 26% in February and 19% in April. Soluble reactive phosphorus retention showed similar trends. Although the author reported that seasonality of export was not shown in Frank Lake as the marsh retained phosphorus even during February (White 1997), retention was clearly reduced. Richardson and Craft (1993) surmise that sedimentation processes are unaffected by low temperature or reduced microbial activity as for some other processes; thus, some wetlands continue to show net retention of phosphorus even in winter.

FATE OF INSTREAM PHOSPHORUS

Introduction

The flux of phosphorus among storage compartments in flowing water is complex. Similar to wetland environments, retention or export of phosphorus in streams depends on the physical, chemical, and biological composition of individual aquatic systems. The rate at which phosphorus is removed will vary from river to river, within sections of a river, and with seasonal changes (Keup 1968). Instream storage depends on many factors, including macrophyte biomass and the uptake of DP by aquatic biota, the extent of the riparian zone, and the concentration of nutrients in the overlying water (Kronvang et al. 1999). Phosphorus flux is further influenced by transformations between PP and DP caused by changes in the equilibrium stream DP concentration, and re-suspension of streambed or streambank PP. Many factors influence the direction of the phosphorus transformations during transport, including the time of year, the relative amount of phosphorus entering from different sources, and water management.

Methods

Sediment sorption-desorption studies in lotic systems are similar to those reported for wetlands, and include ammonium-oxalate extractable aluminum as an index for phosphorus retention capacity in sediment and the determination of EPC (see Wetlands and Riparian Areas - Methods). Laboratory experiments were also used to determine abiotic versus biotic mechanisms of sorption, the effect of sediment size fraction on sediment phosphorus sorption characteristics, and the sorption capacities of phosphorus of sediment from different habitats (Munn and Meyer 1990).

The main mechanism for determining the fate of phosphorus in flowing water has involved tracer studies using stable phosphorus-32 isotopes (Keup 1968; Essington and Carpenter 2000; Haggard et al. 2001). Spiraling, which also requires stable isotopes to calculate, is a more recent term used to describe the downstream distance a nutrient molecule travels while cycling from dissolved to particulate form and back to dissolved form (Essington and Carpenter 2000; Haggard et al. 2001). The length required to complete one spiral is composed of uptake length (Sw), the average distance a nutrient molecule travels in the water column before it is removed, and turnover length (Sp), the distance required for a nutrient molecule to be regenerated or released from the particulate form. Sw is used to assess nutrient retention in streams and is described as an index of stream utilization of nutrients supplied by the terrestrial environment (Haggard et al. 2001).

Munn and Meyer (1990) used rhodamine dye and chloride as tracers to observe mixing and to quantify travel time in a 20-m reach of stream. Solutes of known concentrations were dripped into the stream and samples taken every 10 min at 5-m intervals within the reach.

Processes

Phosphorus spiraling. Many authors have tried to determine the stream length required for phosphorus removal in flowing water, but results have been variable. Ball and Hooper (1963) reported the average distance an atom traveled downstream before it was removed from water ranged between 412 and 10 273 m. Although the phosphorus-32 tracer was rapidly taken up and released by vegetation, equilibrium was not reached for about 15 to 20 d, at which time release and uptake by vegetation were in steady state. Different phosphorus uptake rates were thought to be a function of phosphorus-32 traveling downstream as either a particulate or in a more readily available soluble state. Others found 75% of a phosphorus-32 spike was assimilated within 100 m in one reach of a Tennessee stream (Keup 1968). The majority was assimilated by periphyton, with only minor amounts sorbed by other organisms, sediments, and detritus (Keup 1968). Haggard et al. (2001) reported significant summer SRP retention in streams in areas having high CFO densities and pasture. Soluble reactive phosphorus uptake lengths ranged from 200 to 900 m.

Munn and Meyer (1990) studied the net impact of different physical, chemical, and biotic attributes of habitat on phosphorus retention. They reported uptake rates of 22 and 3 $\mu g\ m^{-1}\ min^{-1}$ and uptake lengths of 32 m and 666 m for two cobble-bed streams in North Carolina and Oregon, respectively. Biotic sorption of phosphorus by sediments was more important than abiotic sorption in both streams as determined by low EPC values for untreated sediments when compared with EPC values for sterilized sediments. However, in the Oregon stream, the EPC value of sterilized sediments was less than the augmented stream water concentration during the additions, and a portion of uptake was likely due to abiotic processes.

Hydrological processes. Hydrologic factors relating to stream flow may be the single-most dominant factor controlling phosphorus concentrations in flowing water. Many studies have determined that rivers and streams act as nutrient sinks during low flow and are sources of nutrients during periods of high flow (Keup 1968; Johnson et al. 1976; Hill 1982; House and Denison 1998; Haggard et al. 2001). Haggard et al. (2001) reported the uptake length for SRP was shortest when discharge was lowest at all study sites and increased at higher discharge rates. The authors attributed greater phosphorus retention during low flow to reduced velocities that allowed for greater interaction between the water column and benthic sediments and biota, increasing the opportunity for phosphorus uptake. In addition, Keup (1968) stated that reduced velocity and turbulence during low flows limits the capacity of a river or stream to transport suspended material, facilitating phosphorus storage in the sediment.

While low flows promote instream phosphorus retention, increasing flow rates cause scouring and resuspension of previously deposited materials (Keup 1968). Kronvang et al. (1999) reported 80% of phosphorus retained during low flow periods in summer was resuspended in the first storm events in autumn and winter. The same authors reported that instream bed sediments and marginal bank zones amounted to 23.4% of the gross TP export from the river basin. Johnson et al. (1976) simulated storm flow conditions in the laboratory and found that sediments released phosphorus when agitated. They attributed increases in SRP concentration with increasing discharge rate during storm runoff to input from surface runoff and release from streambed sediments. Castillo et al. (2000) were one of few who reported elevated SRP concentrations in summer and attributed this to lower flows that resulted in less dilution of point-source, wastewater treatment plant effluent.

The impact of increased channel flow not only scours and resuspends bottom sediments, but increases the potential for streambank erosion. Streambank erosion, estimated at 30 mm yr^{-1} in a Danish stream, accounted for 10 to 50% of suspended material instream (Vought et al. 1994). Miller et al. (1982) estimated 10% of TP export in 10 Great Lakes Region streams was due to streambank erosion. However, increased channel flow that is great enough to overflow streambanks may remove phosphorus by trapping it on the flood plain. Kronvang et al. (1999) measured sedimentation of nutrients into riparian areas and reported that entrapment of sediment on the floodplain represented 2.7 to 5.4% of TP export from Gjern River Basin.

Stream order. The influence of flow on phosphorus retention is also apparent at different stream orders. Net retention of PP in stream sediments averaged between 3.7 and 8.3 g m^{-2} yr^{-1} in first to fourth order streams from March to mid-September (Kronvang et al. 1999). The same authors reported an increase in net retention with stream order up to the third order, with a decrease from third to fourth. This decrease in retention was attributed to mean slope, which decreased from first order to third order streams, but increased from third order to fourth order. The increase in slope was thought to increase flow velocity thereby decreasing phosphorus retention.

McDowell et al. (2001b) found that water samples taken furthest upstream contained, on average, the lowest concentrations of DRP and TP in a 39.5-ha watershed in central Pennsylvania, United States. Conversely, storm flow produced the opposite effect, with greater DRP and TP concentrations found upstream rather than downstream. This paralleled a decrease in the percentage of near-stream soils (<60 m from the channel) in excess of 200 mg kg^{-1} STP. Part of the reduced phosphorus concentrations downstream during stormflow was attributed to dilution.

Physical-chemical processes. In mineral soils, phosphorus desorption-sorption has been found to control phosphorus exchange. The sorption capacity of suspended sediment is greater than that found in surface sediment particles because erosion selectively transports finer particles with higher phosphorus sorption capacity (Koski-Vähälä and Hartikainen 2001). Koski-Vähälä and Hartikainen (2001) estimated the potential for labile phosphorus in resuspended material to be desorbed under experimental conditions. Equilibrium phosphorus concentrations increased with decreasing solid concentration. High pH or anoxic conditions increased the amount of labile phosphorus by two to three times. Equilibrium phosphorus concentration values increased under anoxic conditions but not as drastically as at high pH. The maximum phosphorus sorption capacity decreased considerably at pH 9 concomitantly with an increase in EPC. The effect of pH on mobilization was dependent on the concentration of suspended solids in the system (Koski-Vähälä and Hartikainen 2001). At pH <5 and >7, phosphorus release was much greater with lower suspended solids concentration (< 170 mg L^{-1}) with no effect seen at solids of 1700 mg L^{-1}. Ionic strength (salt concentration) is higher in interstitial waters than in the upper water layers, and this promotes phosphorus sorption to and decreases phosphorus desorption from solid surfaces. A slight increase in ionic strength can reduce EPC in the system. Elevated pH favours the role of resuspended solids as a phosphorus source more than a low redox potential does. In conjunction with rising pH, intensive primary production decreases the DP concentration in the water column for internal loading. An increase in EPC at high pH can be explained by the fact that at high pH, iron- and aluminum-bound phosphorus will be released.

Taylor and Kunishi (1971) studied the availability of phosphate adsorbed on streambed sediment and adjacent soils in a watershed draining agricultural land. They concluded that in the Mahantango watershed, Pennsylvania, sediments, stream banks, and cultivated field soils contributed little to the SRP load of the stream as the meadow soils, banks, and sediments all tended to adsorb SRP that may leave the more highly fertilized areas by runoff or erosion. The authors do not specify the distance the runoff traveled from fertilized fields to surface water. Kunishi et al. (1972) studied the ability of suspended material to adsorb phosphate in the same Mhantango watershed. The authors reported that solid material had a large capacity to adsorb phosphorus from solutions containing greater than 10 ug L^{-1} of phosphorus. The ability of suspended material to adsorb phosphorus depended on the character of the material as well as the solid-solution ratio and the initial phosphate concentration.

Flow rates can also affect phosphorus sorption capacity of the sediments. Hill (1982) reported the mean value of the sorption index for 10 Duffin Creek sediment samples was 142.3 $\pm$ 37.8, whereas for the Nottawasaga River samples it was 85.6 $\pm$ 26.2. The two means were significantly different, the Duffin Creek pool sediments having a higher buffering capacity. Coarse sandy sediments had a lower phosphorus sorption capacity than silty sediments. Low EPC and considerable phosphorus buffering capacity of sediments in the two rivers suggest that sediment sorption is a major mechanism for phosphorus retention during low summer flows.

Simard et al. (1993) reported the amount of DP in water and bioavailable phosphorus in sediments of the Beaurivage Watershed in Quebec were significantly greater in tributaries where animal unit densities were greater than elsewhere in the watershed. There was no significant difference in the amounts of PP among zones, suggesting that the phosphorus enrichment of the Beaurivage river water originated from leaching rather than from runoff additions.

Biological processes. In addition to sediment sorption, DP is retained instream through uptake by benthic algae and macrophytes (Hill 1982). However, this may account for only a small proportion of annual phosphorus flux through an individual river reach, with uptake of 1 to 4% or between 5 to 19 mg m^{-2} of phosphorus (Pelton et al. 1998). Ball and Hooper (1963) showed that periphyton (bottom covering film of microscopic plants) was the primary source of rapid biological uptake of phosphorus (rates not reported).

The ability of instream organisms or sediments to assimilate stream nutrients may be affected by nutrient loading. McColl (1974) reported that in nutrient-poor waters, added phosphate-phosphorus was removed rapidly by filamentous algae and trapped sediment. In another stream where nutrients were abundant, phosphate-phosphorus was not significantly different up to 100 m downstream (McColl 1974). Hill (1982) used a mass balance approach to study transport of SRP and major cations in reaches of the Nottawasaga River and Duffin Creek, southwestern Ontario. The study was conducted during low flow in summer. These two rivers receive phosphorus inputs from cropland and sewage treatment plants. Root uptake by riparian vegetation and aquatic insect emergence may account for only a small percentage of the phosphorus removed in a river reach (Meyer and Likens 1979).

LAKES AND RESERVOIRS

Introduction

Lakes differ from other aquatic systems in that they act as receiving bodies for inflowing waters. They are often recognized for their ability to have a purifying effect, with water exiting lakes often having better quality than inflowing tributaries. Water quality in lakes often receives more attention than lotic or wetland systems because of their recreational value. Eutrophication, which can be caused by excess phosphorus, can reduce the aesthetic and recreational value of lakes due to increased primary production.

Methods

Methods used to develop phosphorus budgets in lakes are similar to those reported previously for wetlands and lotic systems, including adsorption-desorption studies of lake sediment. In addition, sediment traps are used to assess nutrient removal via sedimentation. These traps combine all processes affecting particle formation and deposition from the time the trap is deployed. They also provide information regarding seasonal trends in deposition that is often not apparent in sediment cores (Poister et al. 1994). Coring also assists in the determination of sedimentation rates (McCullough 1998).

Primary production plays an important role in nutrient recycling in lakes. Phosphorus fixation by primary producers can be obtained by measuring inorganic carbon incorporation during photosynthesis using the stable carbon-14 isotope in laboratory incubations of water-column samples in combination with average molar carbon:nitrogen:phosphorus ratios of standing particulate matter. Adjustments must be made for light intensity to coincide with natural water-column light profiles (Poister et al. 1994). Assumptions relating to this method are (1) that the incorporation corresponds to the amount of carbon converted to photosynthetic biomass in the natural environment, and (2) that the ratios of carbon:nitrogen:phosphorus used to calculate phosphorus incorporation stay constant throughout the ice-free season (Poister et al. 1994).

Processes

Sedimentation. Whereas hydrology plays a large role in phosphorus retention in flowing water and shallow wetlands, sedimentation of particulate and settling plankton is probably the most important removal mechanism of phosphorus from the water column in lakes. Poister et al. (1994) studied sedimentation and recycling in three Wisconsin lakes, two of which were recharged by groundwater and precipitation, the third recharged by surface water and groundwater. The authors reported that seasonal trends in particulate matter flux to sediment varied among lakes but was consistent between years within each lake. Kronvang et al. (1999) reported similar inlake TP retention of 0.30 g m^{-2} yr^{-1}.

Internal recycling. Nutrient sedimentation is often counterbalanced by recycling, where biologically occluded forms of phosphorus are converted to biologically available forms and reused in primary production (Poister et al. 1994). Poister et al. (1994) reported that 85 to 90% of phosphorus demand from primary production is met by internal recycling. Thus, lakes receiving little external loading can maintain substantial levels of primary production through internal cycling. Welch and Jacoby (2001) reported that internal loading was the dominant source of phosphorus in summer and could be as high as 50% of the TP measured in the water column. Sosiak and Trew (1996) also found that approximately 61% of the TP loading to Pine Lake was from sediment release and other internal sources, compared to about 36% from surface runoff. Inlake DP and SRP concentrations were greatest in spring, with the DP ranging from 82 $\mu g\ L^{-1}$ in spring to 24 $\mu g\ L^{-1}$ in late summer. Reduced DP concentration in the surface water was attributed to uptake by aquatic plants (Sosiak and Trew 1996), but was more likely due to uptake by periphyton (Ball and Hooper (1963). The same authors also reported a vertical gradient in TP and DP during the summer, reflecting the internal release of phosphorus from the bottom sediments. Most of the phosphorus accumulating at the bottom was dissolved and in a form readily available to algae. Internal loading rate for Pine Lake was reported as 1.2 mg $m^{-2}\ d^{-1}$.

Hydrological processes. As in wetlands and flowing systems, water management may play a key role in lake phosphorus dynamics. Klotz and Linn (2001) collected lake sediment samples and dried them for different lengths of time to simulate the effect of water-level drawdown on phosphorus release from sediment. They reported the amount of phosphorus released when dried was greater than continuously wet controls, indicating potential for phosphorus release from sediment when water levels are drawn down. This was supported in previous work conducted by Fabre (1988), De Groot and Van Wijk (1993), Qui and McComb (1994, 1995), Olila et al. (1997) and Watts (2000). Further, the amount of drying needed to promote phosphorus release was only 4 d (Klotz and Linn 2001). Qiu and McComb (1995) in Klotz and Linn (2001) reported that drying killed approximately 75% of microbial biomass in wetland sediments, and that most phosphorus flux in dried sediment came from expired bacteria cells. Watts (2000) found the biotic release of phosphorus resulting from drying was greatest for reservoir sediments of high organic phosphorus that tend to have higher microbial biomass (Sparling et al. 1985). Klotz and Linn (2001) compared phosphorus release from organic sediment with that from relatively sandy sediment and concluded organic content was more important in phosphorus release than phosphorus concentration in the sediments. The freezing of sediments had the same effect on phosphorus release as drying (Klotz and Linn 2001). Although drawdown is important in lakes, the impact that drying and freezing have on intermittent streams or irrigation return-flow channels is unknown. These processes may be important in southern Alberta, where water regulation is highly prevalent throughout the growing season.

Biological processes. There are many biotic communities in lakes that influence phosphorus retention and cycling in lakes, including primary producers, bacteria, zooplankton, and fish. Bacteria and zooplankton are important to the transformation of phosphorus to dissolved bioavailable forms used by primary producers as a source of nutrients. Fish also excrete DP.

In temperate lakes, the production-decomposition cycle of macrophytes, and the associated biogeochemical fluxes, follow an annual cycle (Carpenter and Lodge 1986). Most phosphorus uptake in macrophytes is retained through the roots and then translocated to shoots, where it may eventually enter the lake water by release from living or decaying shoots (Carpenter and Lodge 1986). Phosphorus leaching from macrophytes is strongly dependent on shoot phosphorus content, with 75% of leached phosphorus occurring in a soluble reactive form that is rapidly assimilated by phytoplankton (Carpenter 1980). Phosphorus release rates may be minor with reports of less than 5% of uptake rates throughout the year (Smith 1978).

Macrophyte stands are sinks for particulate matter and sources of DP and organic carbon (Carpenter and Lodge 1986), depending on the season. During periods of active growth, macrophytes and epiphytes are a net sink for phosphorus (Howard-Williams and Allanson 1981), and a net source during periods of senescence (Landers 1982). Release and uptake of nutrients by fallen litter vary from species to species, from site to site for the same species, and seasonally in prairie glacial marshes (Davis and van der Valk 1978).

At Eagle Lake, Iowa, the main nutrient sinks were the litter layer and the organic soil. Microbes associated with fallen litter extract nutrients from marsh water. Nutrients in microbial biomass were added to the nutrient pool in the dead *Typha* shoot tissue. As decomposition progresses, some of the nutrients are released slowly and may be flushed from the marsh in surface water, reabsorbed by macrophytes, algae, or microorganisms, or transformed chemically or microbiologically. The wetland floor may be anaerobic, thus decomposition is incomplete. Partially decomposed litter accumulates with time as organic sediment. A certain amount of the nutrients in the decomposing *Typha* litter and its associated microbial tissues will not be released and will become immobilized semi-permanently in the organic sediment. The litter serves as a nutrient sink via microbial uptake and physical-chemical precipitation, but is the first stage in the development of the organic sediment, which is the ultimate, long-term site of nutrient immobilization in the marsh (Davis and van der Valk 1983). Davis et al. (1983) reported that macrophytes accumulated more than 10 times the amount of phosphorus that entered the marsh in precipitation, drainage, and runoff combined. However, the plants extract nitrogen and phosphorus primarily from substrate interstitial water; therefore, the uptake by emergent macrophytes would not appreciably reduce surface water nitrogen and phosphorus concentrations (Davis et al. 1983). Thus, it is not so much the action of plants that act as a sink, but rather the contribution of litter that makes macrophytes important.

PART 4: WATERSHED-SCALE CASE STUDIES

INTRODUCTION

This chapter reviews watershed studies that have been conducted in various parts of the world. There are few watershed-scale studies completed in Alberta that document phosphorus sources and sinks. The second part of this chapter focuses on the watershed-scale studies that have been conducted in the province, including the Pine Lake and Baptiste Lake studies, Crowfoot Creek Watershed, and the Alberta Environmentally Sustainable Agriculture (AESA) Stream Survey. Where possible, an attempt was made to quantify and partition phosphorus sources according to land use.

METHODS

Watershed-scale studies of phosphorus fluxes are less common than small-plot studies, but several were conducted in the late 1960s and early 1970s (Kunishi et al. 1972; Schuman et al. 1973; Burwell et al. 1974), and they began to re-emerge in the 1990s (Chichester and Richardson 1992; Anderson et al. 1998). Generally, these studies involve instrumenting catchments for hydrometric measurements and taking flow-weighted water samples. The results are then related back to management practices or land cover types. At the larger scale, there is greater variability in management practices, climate, and hydrology, and this makes it more difficult to quantify phosphorus losses and to determine phosphorus sources (Menzel et al. 1978; Anderson et al. 1998).

The most successful projects used small watersheds (<10 ha) to study a single management type (Schuman et al. 1973; Burwell et al. 1974; Chichester and Richardson 1992). These studies are generally restricted to identifying trends as statistics are rarely applied, and if they are, results are often not significant due to large coefficients of variation. Paired watershed studies may allow comparisons to be made among different management practices, provided differences in watershed hydrology are minimal (Johnson et al. 1979; Chichester and Richardson 1992).

The introduction and growth of Geographic Information Systems (GIS) technology and spatial analysis has allowed for more detailed studies of larger watersheds comparing multiple land uses. The simplest models relate the proportion of land in a certain land use type to instream water quality or loads, while more sophisticated models take into account differences in landscape form, distance to water (Soranno et al. 1996; Wente 2000), and/or critical source areas (Gburek et al. 2000).

At larger scales, the relationship between land use and instream phosphorus concentrations is often unclear, as phosphorus dynamics are much more dependent on soil characteristics than other parameters (Byron and Goldman 1989; Johnson et al. 1997; Bouraoui et al.1999; Calhoun et al. 2002). However, several studies have found relationships between human and livestock densities and/or wastes (i.e., point sources) with instream phosphorus levels (Muir et al.1973; Johnes 1996; McFarland and Hauck 1999; Castillo et al. 2000). Despite the relationships that have been found, Steegen et al. (2001) contend that measuring water quality at the outlet of the watershed and relating it back to land use is insufficient for determining phosphorus fluxes. As in the Burwell et al. (1974) study, there are often confounding phosphorus sources and instream processes that make it difficult to quantify phosphorus losses at the watershed scale. Models that account for multiple sources, transport mechanisms, and hydrological factors are essential for further quantification of phosphorus fluxes (Haygarth and Jarvis 1999).

GENERAL WATERSHED-SCALE STUDIES

Although relationships between land use and water quality have been studied at the larger scale, very few studies have been able to attribute phosphorus losses to specific land use activities. In the Great Lakes region of Canada, Miller et al. (1982) examined water quality at the outflows of 10 agricultural basins. They developed predictive relationships for TP export based on regression equations and independent estimates of specific sources. They estimated that 70% of the agricultural contributions to surface waters were from runoff from cultivated land, while 20% were due to livestock, and 10% were attributed to streambank erosion. Other agricultural sources (subsurface drains, unimproved land) were only minor contributors. The major assumption was that all phosphorus from cropland was associated with sediment. Although the authors reported that SRP was often a significant proportion of TP, regression models for SRP were generally not successful.

Johnson et al. (1976) reported much lower proportions of DP contributed from cultivated lands and barnlots (around 20%) in New York State. However, background levels of DP, estimated from groundwater and inputs from groundwater, were quite high due to geologic factors, contributing 45% of the DP load. Point sources, including municipal discharge and septic systems, accounted for 35% of the DP load. Overall, it was estimated that less than 1% of phosphorus applied in manure reached the stream.

Although phosphorus losses from surface runoff are usually less than 1% of applied phosphorus (Dunigan et al. 1976; Douglas et al. 1998), even small amounts can have negative impacts on aquatic ecosystems. Additionally, several watershed-scale studies have reported much larger phosphorus losses. Chichester and Richardson (1992) reported phosphorus losses of 6.3% and 11.8% of applied fertilizer from conservation tillage and conventional tillage, respectively. Similarly, mean annual losses of phosphorus of 7.5% were reported for a South Devon, United Kingdom, watershed (McDowell and Trudgill 2000). However, these values were based on the assumption that fertilizer phosphorus was the only source of phosphorus in the watershed and did not account for other sources or instream processes.

Robertson (1996) studied the Western Lake Michigan Drainages to identify, describe, and explain, where possible, the major factors that affect observed water quality conditions and trends. In the study, the basin was divided into subunits based on dominant land use. The inputs of phosphorus to the Western Lake Michigan Drainages came from fertilizers (54.8%), manure (41.7%), atmosphere (2.2%), and point sources (1.2%). This amounted to 31 000 000 kg or 6.09 kg ha^{-1} yr^{-1}. The amount of phosphorus exported by each fraction, assuming that 100% of point sources were exported, amounted to 72.5% from nonpoint sources, 14.8% by point sources directly to Lake Michigan and Green Bay and 12.7% from point sources in the basin. Total export from the Western Lake Michigan Drainages amounted to 1 421 000 kg or 0.28 kg ha^{-1} yr^{-1}, nearly 300 times less than input. The sub-basins with the highest inputs of phosphorus (highest use of fertilizer and manure) also had the greatest outputs. The forested sub-basin, having greater than 85% forested land area, had inputs of 0.87 kg ha^{-1} yr^{-1} and exported 0.08 kg ha^{-1} yr^{-1}. Land area having greater than 50% agriculture had phosphorus inputs of 9.48 kg ha^{-1} yr^{-1} to 11.33 kg ha^{-1} yr^{-1} and corresponding exports of 0.32 kg ha^{-1} yr^{-1} to 0.64 kg ha^{-1} yr^{-1}.

Similarly, Cooke and Prepas (1998) compared forested and agricultural watersheds and also found greater export from regions dominated by agriculture. They reported export values of 0.05 kg ha^{-1} yr^{-1} to 0.22 kg ha^{-1} yr^{-1} in the forested watersheds and export values of 0.12 kg ha^{-1} yr^{-1} to 0.82 kg ha^{-1} yr^{-1} in agricultural watersheds during their 2-yr study near Baptiste Lake, Alberta. The authors found that 60 to 91% of the phosphorus export was in the dissolved form from the agricultural watersheds.

ALBERTA CASE STUDIES

Pine Lake Diagnostic Study

The Pine Lake Diagnostic Study is the most comprehensive study documenting watershed-scale processes in Alberta to date (Sosiak and Trew 1996). Pine Lake is a small, intermittently stratified, eutrophic freshwater lake with a surface area of nearly 4 km^2 and a mean depth of 5.3 m. A combination of cow-calf wintering sites and human sewage systems have caused accumulation of phosphorus in lake sediments. The study was initiated in response to water quality deterioration, including algae and macrophyte growth, and occasional fish kills. One of the main project objectives was to provide a detailed phosphorus budget for Pine Lake, including improved estimates of external and internal phosphorus loading. It further attempted to identify the most important sources of nutrients in the various stream sub-basins to establish critical or high priority areas for restoration.

Daily mean flow estimates were calculated for all significant inflows and outflows between February and November 1992. Separate stage-discharge curves were developed for the period when channels contained ice, and for the ice-free period. Weekly grab and/or daily composite samples were collected on all known surface inflows until flows ceased. Automated ISCO samplers were used to collect TP samples between March and August 1992. Annual phosphorus loadings to the lake were measured at the mouth of each stream and short-term loading was measured during spring runoff.

Sampling was also conducted upstream and downstream at some locations to help pinpoint four small areas containing winter livestock feeding and calving areas, and resort septic leachate. These were known to contribute significant amounts of the TP and DP loadings.

The three main aquifers identified in the vicinity had higher water levels than the lake level, indicating the lake was recharged by groundwater. Six nests of wells were installed at both ends of the lake and at central locations on the east and west sides. Mini-piezometers were installed at three shoreline areas to allow preliminary sampling of shallow groundwater in or near the lake. A septic leachate detector was used to identify areas of potential septic leachate. Laboratory experiments were conducted on intact sediment cores to verify estimated sediment release rates.

In 1992, it was determined that approximately 61% of the TP loading was from sediment release and other internal sources, compared to about 36% from surface runoff (Table 1; Sosiak and Trew 1996). Inlake DP and SRP were greatest in spring, the former ranging from 82 $\mu g\ L^{-1}$ in spring to 24 $\mu g\ L^{-1}$ in late summer. The difference was attributed to uptake by aquatic plants (Sosiak and Trew 1996). The same authors also reported a vertical gradient in TP and DP during the summer, reflecting the internal release of phosphorus from the bottom sediments. Most of the phosphorus that accumulated at the bottom was dissolved and in a form readily available to algae. Internal loading rate for Pine Lake was 1.2 $mg\ m^{-2}\ d^{-1}$.

Table 1. Percent contributions of phosphorus to Pine Lake, Alberta by various sources.

Source	%
Internal loading	61.0
Surface runoff	27.0
Theoretical sewage load	6.0
Atmospheric deposition	3.0
Diffuse runoff	3.0

The Baptiste Lake Study

The Baptiste Lake Study was initiated in 1976 to establish the trophic status of the lake and to assess the impact of past and future shoreline and watershed development. In their assessment, Trew et al. (1978) attempted to quantify nutrient sources. Baptiste Lake is situated north of Edmonton, Alberta, in the interior plains region. The Baptiste Lake watershed drains an area of approximately 30 900 ha through 12 tributary streams flowing into the lake and one outlet stream. Land area in the watershed is approximately 58% forested, 25% standing water, 16% in agricultural production, and less than 1% is urban land. Buried bedrock channels and surficial deposits also contribute groundwater to the lake. The lake’s surface area is about 917 ha, with a mean depth of 9.3 m and maximum depth of 27.5 m.

Water samples and stream discharge measurements were taken throughout 1976, with more intensive sampling taken at peak flow. Tributary stream contributions were assumed to be representative of surface runoff contributing to the stream. Two deep test-holes were drilled to verify the existence of buried pre-glacial bedrock channels. Although the channels were present, they were thought to pass below the lake basin and were not included as potential inputs used in the calculations of hydraulic residence time or nutrient loads in the lake. Three observation wells were placed in the sand and gravel unit found at the base of each channel and were used to monitor groundwater pressure and chemistry of the aquifer. Twenty-three additional wells were installed to monitor the fluctuations and chemistry of shallow groundwater.

To predict nutrient export from diffuse runoff areas, Trew et al. (1978) extrapolated from coefficients (or average of two coefficients) of the stream basin(s) in which the relative portion of land use types were most similar to the diffuse runoff area in question. Theoretical loadings were also calculated using nutrient export coefficients from the literature. The authors noted there was no validity in constructing budgets using coefficients from geographic locations other than from the watershed under investigation since the quantity and quality of nutrient runoff is site-specific, and heavily influenced by factors such as vegetation, soil structure, topography, and precipitation. Atmospheric contributions were quantified by analyzing the total mass of phosphorus in snow, ice, and rainfall (which includes dustfall).

Selected coefficients were applied to different land use areas that were determined through aerial photograph interpretation. Rural and agricultural land was assigned an export coefficient of 0.5 kg ha^{-1} yr^{-1} and 0.1 kg ha^{-1} yr^{-1} was assigned to forested land. The authors assumed that the net contribution of phosphorus from wetlands to the lake was zero. The export coefficient determined by Trew et al. (1978) for forest land use was similar to that used in the literature, but was slightly less for agricultural land (0.3 kg ha^{-1} yr^{-1} TP compared to 0.5 kg ha^{-1} yr^{-1} TP).

Theoretical loading estimates, excluding urban (cottage) land, for agricultural and forested land were 4 226.6 kg yr^{-1}. Total phosphorus concentrations in bedrock channels were 0.50 mg L^{-1}, although reported for reference only, with a range in load from 0.061 to 0.568 kg yr^{-1}. Compared to surface runoff, groundwater contributions were insignificant. Atmospheric inputs (snowfall, rainfall, and dustfall) contributed a total of 258.2 kg TP to the lake in 1978, approximately 88% of which was deposited as rainfall and dustfall. An annual rate of TP loading based on average precipitation was estimated at 308 kg yr^{-1} (33.2 mg m^{-2} yr^{-1} TP). Trew et al. (1978) compared atmospheric deposition rates with coefficients similar to those used to calculate surface runoff and found that this method overestimated TP deposition by a factor of about three (100 mg m^{-2} yr^{-1}). Table 2 summarizes the percent contribution of each source of phosphorus to Baptiste Lake.

Table 2. Percent contributions of phosphorus to Baptiste Lake, Alberta by various sources.

Source	1976	1977	1978
Agriculture	-	-	45.0
Forests	-	-	33.0
Tributaries	69.0	66.0	-
Diffuse runoff	7.0	13.0	6.0
Precipitation	12.0	12.0	6.0
Urban runoff	4.0	3.0	3.0
Domestic effluent	8.0	6.0	7.0

No attempt was made to quantify phosphorus release from sediment. However, sampling the water column immediately above the sediment showed differences in SRP release between basins within Baptiste Lake. Soluble reactive phosphorus in the south basin increased throughout the summer but did not change concentration following winter stratification. The north basin exhibited a peak in SRP during the winter. The authors attribute the difference in release to the higher sediment load contributed to the south basin by 78.9% of the total discharge to the lake. Total phosphorus loads contributed by runoff from urban land, including septic systems, was estimated at 102.3 kg yr^{-1}. Phosphorus contributions from wildlife and decomposition were not quantified.

Haynes Creek Watershed

Anderson et al. (1998) investigated field runoff from four cattle wintering sites, three cultivated field sites, and one aspen forested control catchment within the Haynes Creek watershed. Runoff from cultivated field and control sites was measured before entering the stream and additional sites were monitored within Haynes Creek. For the wintering sites, monitoring sites were established upstream and downstream and contributions were deduced from changes in nutrient loads. This study was conducted for a period of 2 yr (1995 to 1996) with very different climatic conditions.

Runoff from the cultivated sites was restricted to the period of spring snowmelt and was quite variable between the 2 yr of the study. In 1995, runoff from the three cultivated sites averaged losses of 0.037 and 0.047 kg ha^{-1} yr^{-1} for DP and TP, respectively. In 1996, snowmelt runoff losses were 17 times higher for DP (0.634 kg $ha^{-1}yr^{-1}$) and more than 60 times higher for TP (2.871 kg ha^{-1} yr^{-1}). Losses of TP, PP, and DP, tended to be higher from a catchment with a grassed waterway seeded to an alfalfa-brome mixture. The higher values of PP were attributed to a break in the grassed waterway. However, this site had been treated with inorganic fertilizer and liquid hog manure, while the other two sites only had applications of inorganic fertilizer. Of the remaining two sites, losses of phosphorus tended to be greater from the site without a grassed waterway; however, the proportion of DP was higher from the site with a grassed waterway. Runoff was not analyzed prior to entering the grassed waterway so it is not possible to determine the effect it had on TP and DP levels.

Phosphorus concentrations in runoff from the control site were much higher than expected, and often greater than for the cultivated fields. The high background values were attributed to wildlife, atmospheric deposition, and leaching of phosphorus from poplar leaves.

Anderson et al. (1998) also measured concentrations and loads within Haynes Creek to determine the effect of various sources. Runoff losses from non-point sources (i.e., cultivated fields) were obscured by the high flows, which caused the destruction of several beaver dams and the release of large amounts of organic matter into the stream. Additionally, wetlands may have confounded the relationship between land use and water quality.

The impact of cattle wintering sites was measured by monitoring upstream and downstream of the cattle wintering sites. Therefore, no direct samples of runoff were taken and results were often obscured by dilution. However, nutrient concentrations loads were often much higher downstream of wintering sites. Estimated export values for TP ranged from 0.04 to 0.37 kg ha^{-1} yr^{-1}, with most of the phosphorus in dissolved forms. Loads were lower from a wintering ground that was separated from the stream by a grassed strip.

Lower Little Bow River Watershed

Water quality and flow in the Lower Little Bow (LLB) watershed has been monitored since 1998. This watershed is composed of five sub-basins draining directly to the mainstem of the river and two tributaries, the Sorgaard drain and the Pitchfork drain, that are major return flows for the Lethbride Northern Irrigation District (LNID). The LLB is a regulated river with releases from Travers Reservoir of 0.85 m^3 s^{-1} in winter and mean monthly flows of 0.77 to 2.27 m^3 s^{-1} from May to September, depending on irrigation demand. Therefore, sampling was conducted on a weekly or biweekly basis and during runoff events instead of flow-weighted sampling.

Land use was classified for each of the basins in the first year of the study. This agricultural watershed shows a wide range of land uses and agricultural intensity. Native prairie dominates the uppermost sub-basins and is used as pasture land. There is still a large proportion of native prairie in the wide river valley along the lower reaches, but dryland and irrigated cropland are also present. In the irrigation return flow sub-basins, irrigated land and confined feeding operations (CFOs) are relatively high, especially in the Sorgaard drain. Although cattle are grazed along the length of the river, no reliable estimate of livestock numbers was obtained.

Water quality in the return flow streams had higher concentrations of TP and DP compared with mainstem sites. Concentrations of phosphorus significantly increased in mainstem sites downstream of the confluence with the irrigation return flow streams, during low flow years. Export coefficients for TP calculated for the irrigation season (mid-April to mid-October), tended to be higher from irrigated watersheds; however, results were extremely variable (0.007 to 0.18 kg ha^{-1}) and dependent on sub-basin area. Therefore, these were deemed not to be reliable indicators of land use impacts.

Correlations among water quality and land use, soil type, and landscape variables showed some significant relationships with instream phosphorus concentrations. Median TP concentrations were correlated with the proportion of cereals and irrigated land, while inverse correlations were observed between median TP and the proportion of native prairie. Stronger relationships were found between maximum TP concentrations, which were observed following an intense rainfall event. This finding emphasizes the importance of a transport mechanism for linking land use and water quality. Maximum TP concentration correlated with the proportion of cereals, potatoes, irrigated land, and the density of CFOs. Soil type was also a factor with medium-textured glacial-lacustrine soils showing significant associations with maximum TP concentrations. This suggests that land use and soil type can play a role in determining phosphorus export from this watershed.

Higher median DP values were associated with greater proportions of irrigated land and cereals and inversely correlated with native prairie. Unlike TP values, maximum DP values showed only one significant relationship with soil type (Orthic Dark Brown medium-coarse textured soils of glacio-fluvial origin). Multiple-regression analysis suggested that most of the variation in maximum TP and median DP concentrations could be explained by the proportion of cereals in the watershed.

Although relationships were apparent at the watershed scale, localized influences cannot be ignored. One of the irrigation return flows had high loads of TP, but only a small proportion of this was DP. In this case, erosion of the return flow channel may have been an important contributor to instream phosphorus concentrations. Total phosphorus concentrations in the other irrigation flow relatively high proportions of DP. A reservoir just upstream of the confluence with the mainstem allowed many solids to settle out.

These localized influences illustrate the difficulty in describing meaningful land use relationships, as variables controlling phosphorus export are difficult to measure and operate at different scales. In spite of this, useful information can be derived from broad-scale, rudimentary analyses.

Crowfoot Creek Watershed

The Crowfoot Creek Watershed study was conducted to determine whether agricultural practices were contributing to deterioration of surface water quality in the watershed and to identify specific land uses having an impact. The study was initiated in response to concerns from area residents and the findings of previous studies that identified high concentrations of nutrients and coliform bacteria in the watershed. Agricultural inputs of herbicides and fertilizers are in the top 25th percentile in Alberta.

Crowfoot Creek is a small tributary of the Bow River, in an area of intensive agricultural activity in southern Alberta. The watershed is located approximately 85 km east of the City of Calgary and is approximately 160 000 ha in size. The area is predominantly agricultural in nature, with the Village of Standard as the only urban area within the watershed boundary. Water quality was monitored at 28 sites during a 4-yr period, with water quantity and quality data collected. A map of the land cover in the watershed, including native and improved pasture, annual crops, and summerfallow, was prepared using LANDSAT imagery.

Hydrology of the watershed was dominated by return flows into the creek from the Western Irrigation District (WID) infrastructure. Flows were returned to the creek from the WID from early May until late September in all years. Natural flows consisted of snowmelt and rainfall runoff.

Levels of phosphorus were greatest during the spring and early summer as the phosphorus in several wetlands along the creek was flushed out by snowmelt runoff and WID return flow water. The WID water tended to be of better quality than the water in the creek and improved the overall quality by dilution. Levels of TP exceeded the Alberta Water Quality Guidelines of 0.05 mg L^{-1} (Alberta Environment 1999) most of the time and were greatest during spring snowmelt and rainfall events. Total dissolved phosphorus made up the majority of the TP in this watershed.

The impact of landcover on phosphorus in the watershed was variable. The percentage of TP made up by TDP varied with the type of landcover adjacent to the watercourse. Sub-basins with greater amounts of grassland adjacent to the watercourse appeared to have greater percentages of TDP, while in sub-basins with greater amounts of cultivation adjacent to the watercourse the reverse was true. During rainfall events, the percentage of TP as TDP stayed fairly constant in reaches where grassland dominated the area adjacent to the creek, while in areas with greater amounts of cultivated land the percentage of TDP decreased.

While some trends were apparent in this study, many variables may have impacted the final outcome. The large size of this watershed resulted in great variation in rainfall. The WID return flows varied in response to consumer demand; therefore, the amount of flushing that occurred varied from year to year. Critical source areas were not determined, making it difficult to determine the impact of land cover on water quality. The contribution of wetlands during various stages of the year was variable, so the level of contribution of phosphorus from these sources is unknown.

AESA Watershed Monitoring Program

The Alberta Environmentally Sustainable Agriculture (AESA) Stream Survey was initiated in 1997. This program monitors water quality in 23 watersheds that are all less than 150 000 ha and contain active stream gauging stations. The watersheds represent regional and provincial characteristics in agricultural production and runoff processes, and may be used to represent similar watersheds throughout Alberta.

Each year, the selected watersheds are given a water quality index score for nutrients, bacteria, and pesticide concentrations. The index is calculated based on the number of parameters that exceed flow-weighted median concentration of all streams, the frequency with which they exceed this level and by how much they exceed the median value. Table 3 summarizes the results obtained from the data collected in 1999 through 2001. Those watersheds with the greatest intensity of agriculture complied the least with TP guidelines for the protection of aquatic life (AESA Stream Survey 1999; AESA Stream Survey 2000; AESA Stream Survey 2001). In general, high nutrient concentrations found within all watersheds corresponded with high runoff events.

Table 3. Percent total phosphorus compliance with water quality guidelines for the protection of aquatic life in watersheds having variable agriculture intensity.

Watershed type	% Compliance		
	1999[z]	2000[y]	2001[y]
Low agricultural intensity	48	53	57
Moderate agricultural intensity	19	27	33
High agricultural intensity	4	9	5
Irrigation	39	58	45

[z]A relatively wet year compared to normal.
[y]A relatively dry year compared to normal.

PART 5: SUMMARY AND CONCLUSIONS

PHOSPHORUS SOURCES AND SINKS

Many studies have discussed the influence of local factors on the export of phosphorus from terrestrial and aquatic systems. Research has generally focused on quantifying one contributing source or sink at a time (e.g., cultivated fields, native grassland, septic field losses, and wetlands as covered in Parts 2 and 3). Although this increases our understanding of how individual processes influence phosphorus flux, watershed-scale studies require understanding of the interactions between aquatic and terrestrial environments.

Although phosphorus losses from non-point sources are difficult to quantify, researchers have, with limited success, been able to quantify phosphorus losses from various land uses. A common theme throughout all studies reviewed was the inherent variability of phosphorus flux and its dependency on local factors. In cropping systems, the main factors influencing phosphorus flux are tillage practices, timing of application, properties of the fertilizer, and risk of erosion. From grasslands, phosphorus loss is influenced by grazing intensity, number of livestock, time of year (frozen or unfrozen ground), slope, and proximity to the watercourse. The contribution of phosphorus from municipal and industrial effluent depends on the degree of effluent treatment, the receiving water's ability to dilute and the characteristics of the receiving water. In total, the impact of phosphorus loss from various land uses will depend on the likelihood of the nutrient reaching surface water.

Once in the receiving body, the phosphorus cycle becomes more complex. Phosphorus flux in lentic and lotic environments is influenced by the physical, chemical, and biological composition of the system. There are short-term and long-term storage compartments that retain and release phosphorus at variable rates according to pH and redox conditions of overlying water and sediment. The biota (macrophytes, phytoplankton, zooplankton, and bacteria) represent short-term storage compartments that, upon senescence and death, release much of the stored phosphorus back to the water. Remaining phosphorus will be buried in the litter layer and upon decay a portion will be released to the water column. As litter is buried by sediments, the remaining phosphorus is placed in long-term storage. Thus, aquatic systems act as a source of nutrients and as a sink for nutrients.

Considering all the factors, it is not surprising that each watershed seems to have a unique phosphorus budget. The quantity and quality of phosphorus in runoff is site-specific, and this results in a large range of export coefficients for each source (Table 4). Combining the many variables with varying land use and management practices, a watershed-scale study will only produce phosphorus budgets that are suitable for the watershed that is being evaluated. Those studies that have attempted to budget phosphorus at the watershed scale have not incorporated essential watershed components or have made assumptions to compensate for gaps in knowledge (Trew et al. 1978).

To increase knowledge with limited expense, plot studies or small watershed studies have been conducted. However, very few of these studies have attempted to link these two different scales, even though watershed size will influence phosphorus export. Where it has been attempted, several studies found that a large proportion of SRP from the land is rapidly attenuated instream through adsorption (Schuman et al. 1973) or dilution (Kunishi et al. 1972; Anderson et al. 1998). However, Kronvang et al. (1999) reported that about 80% of adsorbed phosphorus was resuspended in the first large storm event. Many studies that have tried to link the two different scales have not accounted for important sources and sinks, such as wetlands (Anderson et al. 1998) and livestock operations (Burwell et al. 1974). Studies examining larger, multiple land use watersheds, have shown relationships between instream phosphorus concentrations and land use, but few have been able to account for phosphorus loads from specific sources. GIS-based models that take into account land use and hydrology may be better able to link these scales and processes.

Table 4. Summary of literature values of phosphorus export coefficients from various watershed sources.

Source	TP ($kg\ ha^{-1}\ yr^{-1}$)	DP ($kg\ ha^{-1}\ yr^{-1}$)
Atmospheric deposition	0.0231 - 1.487	-
Groundwater	0.01 - 0.390	-
Cropped land	0.000 - 38.000	0.023 - 1.230
Grassland	0.020 - 9.400	0.130 - 3.290
Irrigated land	0.002 - 11.150	0.001 - 1.930
Urban stormwater	0.156 - 0.201	-
Forested land	0.046 - 0.350	0.142
Source	**TP source ($g\ m^{-2}\ yr^{-1}$)**	**TP sink ($g\ m^{-2}\ yr^{-1}$)**
Streambank erosion	0.003 - 0.145	-
Wetlands	0.000 - 0.700	0.005 - 6.5
Lakes	0.730 - 4.380	0.300

There is some evidence that internal loading may contribute more to phosphorus concentrations in surface water than surrounding land use (Sosiak and Trew 1996; Haggard et al. 2001). However, there are very few studies that have been conducted in flowing water that try to quantify this contribution in Alberta. Based on findings of others, phosphorus retention in Alberta’s lotic systems can be expected to be greatest in summer when flows are lowest and biological activity is highest. Long-term storage is more likely in wetlands and riparian zones than in rivers and streams as vegetation aids burial of phosphorus (Davis and van der Valk 1978). Furthermore, where natural hydrology is intact, greater nutrient retention in aquatic systems, particularly rivers and streams, can be expected.

IMPLICATIONS FOR THE SOIL PHOSPHORUS LIMITS PROJECT

The findings of this review have some important implications for the Soil Phosphorus Limits Project, which is being carried out in Alberta. Many studies have documented the inherent variability of phosphorus export from different watersheds and the variability in STP:DP ratios. Therefore, the data collected in the Soil Phosphorus Limits Project microwatershed study should be carefully considered.

The sites selected for edge-of-field analysis of runoff are all situated on cultivated land, with the exception of the Stavely site, which is located on ungrazed, native grass. Numerous authors have documented a broad range of phosphorus concentrations in runoff from cultivated fields (Table 4 and Appendix 1). The results were heavily influenced by land management (tillage, fertilization, irrigation) and local site characteristics (topography, vegetation, climate). Therefore, the results from the microwatershed study may also be highly variable. Since cultivated fields are the focus of the study, all of these variables will have to be accounted for to properly interpret the results. The study may, however, be neglecting important sources and placing too much emphasis on STP.

There are many contradicting studies that show inconsistent results for the relationship between STP and DP, particularly at the watershed scale where the variability in land management is high. Sims (2000) suggests that STP measurements, in addition to assessing soil phosphorus fertility, provide information on regions that may be at greater risk of phosphorus loss to water. This is particularly important in subwatersheds with high animal densities as soils with higher STP values will be enriched with SP and PP. The critical soil phosphorus value for optimum crop yield and limited phosphorus loss in surface runoff will define a critical phosphorus limit. In the United States, researchers have identified threshold STP values for agriculture and for the environment using a risk-based approach. Agronomic thresholds range from 12 mg kg^{-1} in Idaho to 50 mg kg^{-1} in Arkansas, while environmental thresholds range from 50 mg kg^{-1} in Idaho to 200 in Texas (Sharpley et al. 1999). However, nearly all states in the United States have opted for a phosphorus index approach that incorporates source and transport factors, rather than environmental soil phosphorus limits. The determination of STP and the relationship between STP and DP, alone, will not provide the appropriate foundation for generating phosphorus limits for agricultural land in Alberta. Particulate phosphorus also needs to be included in analyses as sediment, especially smaller fractions such as clay, may be highly enriched in phosphorus. Therefore, smaller events must be taken into account since they may contribute more phosphorus than rare, large events during the long term (Quinton et al. 2001). Soil erosion estimates cannot be relied upon to calculate phosphorus losses.

McDowell et al. (2001a) reported that 90% of surface runoff came from the area immediately adjacent to the stream. Therefore, targeting management practices within critical source areas (CSAs) may be the most effective way to address phosphorus transport to surface waters. However, control measures that can reduce PP do not always have the same effect on DP. For example, conservation tillage reduces PP, but can increase dissolved fractions due to leaching from vegetation residues and surface applied fertilizers and manure. Losses from fertilized fields, especially of DP forms, can be minimized by incorporation. Losses of inorganic fertilizer are often higher than manure immediately following application (Bundy et al. 2001), but inorganic fertilizer is more readily available to plants; therefore, the risk of runoff is rapidly reduced. In general, less than 1% of applied phosphorus is lost in surface runoff (Dunigan et al. 1976; Douglas et al. 1998), with the exception of phosphorus loss from frozen soils (Nicholaichuk and Read 1978; Anderson et al. 1998). While this small percentage may be agronomically insignificant, these amounts can have a large impact on receiving water.

When ranked according to greatest TP export, cultivated land ranked highest, while grasslands ranked highest for greatest DP export (Table 4). If this is true for Alberta, grasslands should be included in the microwatershed study since the majority of land found in the critical source areas is grassland.

Phosphorus limits for cultivated lands may not provide adequate protection for surface waters, given that most of the CSA is grassland. Grassland in Alberta is considered representative of baseline conditions and is used to determine the impact of more intensive agricultural development. Although unimproved grassland areas tend to have low export values, with the majority of losses occurring during snowmelt runoff, much of the grassland region is now used for grazing or wintering livestock. The addition of manure and decreased vegetative cover from grazing results in a relatively large export of phosphorus. If more than 90% of surface runoff is generated in the area immediately adjacent to the stream (McDowell et al. 2001a), then a risk-based approach that considers grassland management may be more important than focusing on cultivated land that mainly occurs in the uplands. Djodjic et al. (2002) reported that only 10.4% in 1995 and 5.2% in 1996 of the total watershed area was classed as having a high potential for phosphorus movement in a watershed in Sweden.

Research in Alberta should focus on developing monitoring programs that can establish background concentrations of phosphorus in soils and sediments in the province. This will provide a better understanding of background phosphorus concentrations as well as insight into the phosphorus sorption capacity of these storage areas. Much of this research has already been initiated through the Phosphorus Mobility Study.

RECOMMENDATIONS AND CONCLUSIONS

In southern Alberta, many of the inherent traits of rivers have been lost and are now regulated to meet irrigation demand rather than the needs of the ecosystem. Annual drying cycles in irrigation return-flow drains may, for example, increase phosphorus release from sediments in spring, resulting in greater export of phosphorus from the watershed. The impacts of drought may have similar effects on lakes and reservoirs. More research should be conducted to determine how present water management, regulation, and drought are impacting phosphorus release in Alberta’s lakes, reservoirs, rivers, streams, and irrigation return-flow drains. A greater understanding of instream processes will help identify the contributions of phosphorus from terrestrial or aquatic environments.

Little emphasis has been placed on the interaction between groundwater and surface water or on groundwater as a source of phosphorus to surface waters. This omission is likely due to the expense associated with outfitting a watershed with groundwater monitoring equipment. Many studies have shown that groundwater in some areas significantly contributes to lake recharge.

This may prove to be of greater significance in irrigated areas of Alberta, particularly as phosphorus leaches into groundwater and is transported to surface water. Research in Alberta suggests that while phosphorus concentration in groundwater is low, it can rapidly increase within the last few metres of its flow path. Therefore, groundwater may serve as a mechanism for recycling phosphorus from lakebed or riparian sediments, in addition to directly contributing phosphorus to surface water. Although considered a minor contributor, groundwater contributions have not been properly evaluated. With drought conditions and water quantity issues becoming more of a concern, understanding groundwater-surface water interactions will be invaluable.

In addition to instream processes and groundwater and surface water interactions, atmospheric deposition may be an important source of phosphorus to Alberta's surface water, particularly in the southern region. High winds combined with exposed soil increase the risk of phosphorus transport. Studies should be developed to consider dust fall and precipitation in phosphorus budget calculations.

Phosphorus is a valuable, agricultural nutrient that is essentially lost from terrestrial systems when improperly managed. Studies are currently being conducted to develop science-based phosphorus limits for agricultural land in Alberta. From the many studies that were reviewed in this document, it is clear that there are many mechanisms and factors that influence phosphorus export from the land as well as within aquatic systems. Caution must be used when interpreting export coefficients derived from various land uses, as the values may only be valid for the region of origin. Data collected in the Soil Phosphorus Limits Study should not be extrapolated to regions that differ significantly from the collection site. While agriculture may contribute significantly to instream DP concentrations in one region, the percent contribution may be smaller in another, depending on land use and management. Attention should be given to identifying critical source areas in Alberta and developing risk-based phosphorus models.

PART 6: ABBREVIATIONS AND GLOSSARY

BAP Biologically available phosphorus

BMP Beneficial management practices

BOD Biochemical oxygen demand

CFO Confined feeding operation

Constructed wetlands Those that have been purposely constructed by humans in an area where a wetland did not previously exist

CSA Critical source area

CSO Combined sewer overflow

DIP Dissolved inorganic phosphorus

DP Dissolved phosphorus the equivalent of TDP

DRP Dissolved reactive phosphorus

EPC Equilibrium phosphate concentration: the concentration where no net adsorption or desorption of phosphorus takes place

Fringe wetlands Wetlands that fringe larger bodies of water, i.e., lakes (Mitsch 1995)

GIS Global information system

Instream wetlands Part of the river or stream they are protecting (Mitsch 1995)

KME Kraft Mill Effluent

Lentic Pertaining to a lake or other nonflowing water body (Kadlec and Knight 1996)

Littoral The shoreward zone of a lake or wetland. The areas where water is shallow enough to allow the dominance of emergent vegetation (Kadlec and Knight 1996)

LLB Lower Little Bow River

Lotic Pertaining to flowing water bodies such as streams and rivers (Kadlec and Knight 1996)

MWTP Municipal wastewater treatment plant

N Nitrogen

P Phosphorus

PP Particulate phosphorus used to describe sediment associated phosphorus

Redox potential The potential of a soil to oxidize or reduce chemical substances (Kadlec and Knight 1996)

Riparian wetlands Receive only seasonal flooding, but otherwise are separated from the river except for return flows and lateral flows from uplands (Mitsch 1995)

SP Soluble phosphorus

SRP Soluble reactive phosphorus used as the equivalent of DRP, orthophosphorus and DIP

STP Soil-test phosphorus

TDS Total dissolved solids

TP Total phosphorus

WID Western Irrigation District

PART 7: REFERENCES

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PART 8: APPENDICES

Appendix 1. Contribution of phosphorus to aquatic systems via surface runoff from cropping systems.

		Phosphorus fraction exported (kg ha^{-1} yr^{-1})					
Source[z]	Location	TP	DP	SRP	PP	BAP	Author
Cultivated land	Haynes Creek, AB	0.028-0.262	0.023-0.183	-	-	-	Anderson et al. 1998
Conventional tillage	Maryland	0.099-0.160	0.005-0.024	0.009-0.024	-	-	Angle et al. 1984
No till	Maryland	0.008-0.029	0.004-0.029	0.003-0.025	-	-	Angle et al. 1984
Row crops	-	2.000	-	-	-	-	Beaulac & Reckhow 1982
Conventional tilled corn	Kentucky	-	-	0.137	-	-	Blevins et al. 1990
Chisel-plow corn (across slope)	Kentucky	-	-	0.072	-	-	Blevins et al. 1990
No-till corn	Kentucky	-	-	0.036	-	-	Blevins et al. 1990
Level-terraced watershed	Iowa	-	-	0.193	0.211	-	Burwell et al. 1974
Contour-cropped corn	Iowa	-	-	0.122	0.793	-	Burwell et al. 1974
Fallow	Minnesota	33.340	0.190	-	-	-	Burwell et al. 1975
Continuous corn	Minnesota	18.190	0.410	-	-	-	Burwell et al. 1975
Corn in rotation	Minnesota	8.660	0.230	-	-	-	Burwell et al. 1975
Oats in rotation	Minnesota	5.250	0.240	-	-	-	Burwell et al. 1975
Hay in rotation	Minnesota	0.680	0.660	-	-	-	Burwell et al. 1975
Mixed land use	Raisin River, MI	-	-	0.120	-	-	Castillo et al. 2000
Conventional tillage	Texas	1.500	-	0.700	-	-	Chichester & Richardson 1992
Conservation tillage	Texas	0.8	-	0.700	-	-	Chichester & Richardson 1992
Winter-applied manure	Wisconsin	0.550-8.090	-	-	-	-	Converse et al. 1976
A1	Baptiste Lake, AB	0.14, 0.12	-	-	0.03, 0.04	-	Cooke and Prepas 1998
A2	Baptiste Lake, AB	0.82, 0.57	-	-	0.07, 0.06	-	Cooke and Prepas 1998
Cropland	Lake Erie	0.000-1.650	-	-	-	-	Coote et al. 1982
Surface runoff & SBR drainage	Essex County, ON	-	0.090-0.160	-	0.170-0.290	-	Culley et al. 1983
Surface runoff & SBR drainage	Essex County, ON	-	-	-	0.210-0.420	-	Culley et al. 1983
Mixed land use	Illinois	0.700-1.100	-	-	-	-	David & Gentry 2000
Winter wheat	Oregon	11.900	0.100	-	-	-	Douglas et al. 1998
Spring pea	Oregon	13.300	0.100	-	-	-	Douglas et al. 1998
Fallow	Oregon	31.300	0.200	-	-	-	Douglas et al. 1998
Zero-till	Essex County, ON	-	-	0.940-1.920	-	-	Gaynor & Findlay 1995
Ridge-tillage	Essex County, ON	-	-	0.750-1.180	-	-	Gaynor & Findlay 1995
Surface runoff, no-till	Nebraska	1.440	0.350	-	-	0.460	Eghball & Gilley 1999
Surface runoff, disked	Nebraska	1.140	0.040	-	-	0.150	Eghball & Gilley 1999

[z] F. is fertilizer; SBR is sub-surface runoff; P. is plots; MP is moldboard plow.

Appendix 1. Continued.

Source[z]	Location	Phosphorus fraction exported (kg ha^{-1} yr^{-1})					Author
		TP	DP	SRP	PP	BAP	
F.	Nebraska	1.730	0.640	-	-	0.660	Eghball & Gilley 1999
Compost (N)	Nebraska	2.210	0.200	-	-	0.420	Eghball & Gilley 1999
Compost (P)	Nebraska	1.040	0.080	-	-	0.220	Eghball & Gilley 1999
Manure (N)	Nebraska	1.560	0.210	-	-	0.350	Eghball & Gilley 1999
Manure (P)	Nebraska	0.770	0.090	-	-	0.180	Eghball & Gilley 1999
Control	Nebraska	0.470	0.010	-	-	0.050	Eghball & Gilley 1999
Ridge tillage, manure	Minnesota	-	-	0.060	-	-	Ginting et al. 1998
Ridge tillage, no manure	Minnesota	-	-	0.040	-	-	Ginting et al. 1998
Ridge tillage, manure	Minnesota	0.310	-		-	-	Ginting et al. 1998
Ridge tillage, no manure	Minnesota	1.540	-		-	-	Ginting et al. 1998
MP with & w/out manure	Minnesota	0.010-0.060	-	0.070	-	-	Ginting et al. 1998
MP, chisel plow & ridge tillage	Minnesota	0.480-1.200	0.050-0.330	-	-	-	Hansen et al. 2000
MP with broadcast F.	Minnesota	0.800-3.000	-	-	-	-	Hansen et al. 2000
MP, banded F.	Minnesota	0.300-0.550	-	-	-	-	Hansen et al. 2000
Chisel plow, broadcast F.	Minnesota	0.200-0.380	-	-	-	-	Hansen et al. 2000
Chisel plow, banded F.	Minnesota	0.200-0.830	-	-	-	-	Hansen et al. 2000
Ridge till, broadcast F.	Minnesota	0.025-0.200	-	-	-	-	Hansen et al. 2000
Ridge till, banded F.	Minnesota	0.010-0.380	-	-	-	-	Hansen et al. 2000
Mixed agricultural use	Saline River, MI	-	-	0.084	-	-	Johengen 1991
Cereals, vegetables and canola	UK	0.650	-	-	-	-	Johnes 1996
Conventional tillage	Iowa	38.000	0.010-0.100	-	-	-	Johnson et al. 1979
Tillage plant	Iowa	17.000	0.100-0.280	-	-	-	Johnson et al. 1979
Ridge tillage	Iowa	3.000	0.120-0.300	-	-	-	Johnson et al. 1979
Chisel-till broadcast, sorghum P.	Kansas	0.400-0.800	0.013-0.019	-	-	0.032-0.067	Kimmell et al. 2001
Chisel-till knife, sorghum P.	Kansas	0.255-0.453	0.010-0.015	-	-	0.027-0.039	Kimmell et al. 2001
Ridge-till broadcast, sorghum P.	Kansas	0.925-1.320	0.292-0.349	-	-	0.342-0.510	Kimmell et al. 2001
Ridge-till knife, sorghum P.	Kansas	0.504-0.847	0.057-0.098	-	-	0.077-0.166	Kimmell et al. 2001
No-till broadcast, sorghum P.	Kansas	0.752-9.130	0.300-0.358	-	-	0.389-0.408	Kimmell et al. 2001
No-till knife, sorghum P.	Kansas	0.284-0.675	0.031-0.116	-	-	0.044-0.203	Kimmell et al. 2001
Chisel-till broadcast, soybean P.	Kansas	0.077-0.614	0.002-0.018	-	-	0.005-0.061	Kimmell et al. 2001
Chisel-till knife, soybean P.	Kansas	0.073-0.770	0.002-0.012	-	-	0.005-0.075	Kimmell et al. 2001
Ridge-till broadcast, soybean P.	Kansas	0.082-0.606	0.004-0.061	-	-	0.011-0.132	Kimmell et al. 2001
Ridge-till knife, soybean P.	Kansas	0.066-0.342	0.002-0.031	-	-	0.006-0.039	Kimmell et al. 2001

[z] F. is fertilizer; SBR is sub-surface runoff; P. is plots; MP is moldboard plow.

Appendix 1. Continued.

		Phosphorus fraction exported (kg ha^{-1} yr^{-1})					
Source[z]	Location	TP	DP	SRP	PP	BAP	Author
No-till knife, soybean P.	Kansas	0.003-0.470	0.000-0.028	-	-	0.000-0.069	Kimmell et al. 2001
Corn, high fertilizer	New York	-	-	0.130-0.490	-	-	Klausner et al. 1974
Corn , med. F.	New York	-	-	0.050-0.160	-	-	Klausner et al. 1974
Wheat/bean rotation high & med. F.	New York	-	-	0.180-0.370	-	-	Klausner et al. 1974
Wheat/bean rotation, med. F.	New York	-	-	0.040-0.090	-	-	Klausner et al. 1974
Wheat, medium F. rate, good mgmt	New York	-	-	0.170-0.320	-	-	Klausner et al. 1974
Wheat, medium F. rate, poor mgmt	New York	-	-	0.080-0.210	-	-	Klausner et al. 1974
Fall-fertilized summerfallow	Swift Current, SK	2.900	-	1.200	-	-	Nicholaichuk & Read 1978
Stubble	Swift Current, SK	0.400	-	0.100	-	-	Nicholaichuk & Read 1978
Summerfallow	Swift Current, SK	1.400	-	0.200	-	-	Nicholaichuk & Read 1978
Dryland cotton	Oklahoma	5.010	0.710	0.680	-	-	Olness et al. 1975
Dryland wheat	Oklahoma	2.940	0.540	0.440	-	-	Olness et al. 1975
Dryland alfalfa	Oklahoma	2.480	1.230	1.130	-	-	Olness et al. 1975
Contour-cropped corn, high F.	Iowa	-	-	0.171	1.050	-	Schuman et al. 1973
Contour-cropped corn, F.	Iowa	-	-	0.110	0.581	-	Schuman et al. 1973
Level-terraced watershed	Iowa	-	-	0.049	0.085	-	Schuman et al. 1973
Cultivated land	Belgium	1.100-4.900	-	-	-	-	Steegen et al. 2001
Raised beds, no-till	Texas	0.070-0.350	-	0.140	-	-	Torbert et al. 1996
Raised beds, chisel plow	Texas	0.750-1.100	-	0.030	-	-	Torbert et al. 1996
MP, manure	Minnesota	0.508	-	0.112	-	-	Zhao et al. 2001
Ridge tillage, manure	Minnesota	0.674	-	0.433	-	-	Zhao et al. 2001
SBR MP, manure	Minnesota	0.006	-	0.005	-	-	Zhao et al. 2001
SBR ridge tillage, manure	Minnesota	0.177	-	0.140	-	-	Zhao et al. 2001
Moldboard plow, urea	Minnesota	0.635	-	0.055	-	-	Zhao et al. 2001
Ridge tillage, urea	Minnesota	0.289	-	0.080	-	-	Zhao et al. 2001
SBR, moldboard plow, urea	Minnesota	0.002	-	0.0001	-	-	Zhao et al. 2001
SBR, ridge tillage, urea	Minnesota	0.038	-	0.012	-	-	Zhao et al. 2001

[z] F. is fertilizer; SBR is sub-surface runoff; P. is plots; MP is moldboard plow.

Appendix 2. Contribution of phosphorus to aquatic systems via surface runoff from grasslands.

		Phosphorus fraction exported (kg ha^{-1} yr^{-1})					
Source[z]	Location	TP	DP	SRP	PP	BAP	Author
Pasture		1.000	-	-	-	-	Beaulac & Reckhow 1982
	Ohio	-	-	-	-	-	Chichester et al. 1979
Unimproved land	Lake Erie	0.080	-	-	-	-	Coote et al. 1982
Surface runoff/subsurface drainage	Essex County, ON	-	0.150	-	0.090	-	Culley et al. 1983
Surface runoff/subsurface drainage	Essex County, ON	-	3.290	-	0.210	-	Culley et al. 1983
Grassland, rough fescue	Alberta	?	-	-	-	-	Dormarr & Willms 1998
Grassland, poultry litter	Arkansas	1.980	-	-	-	-	Edwards & Daniel 1994
Grassland, inorganic fertilizer	Arkansas	2.680	-	-	-	-	Edwards & Daniel 1994
Grassland, control	Arkansas	1.210	-	-	-	-	Edwards & Daniel 1994
Grassland, super-phosphate F.	New Zealand	-	-	-	-	-	Gilligham & Thorrold 2000
Unimproved pasture,	SW England	-	-	-	-	-	Hawkins & Schofield 1996
	UK	-	-		-	-	Haygarth & Jarvis 1999
	UK	-	-	-	-	-	Heathwaite, L. 1995
Grazed grassland	Idaho	-	-	-	-	-	Jawson et al. 1982
Grassland, grazed	Lake Wabumum, AB	0.200-1.420	-	0.100-1.160	-	-	Mitchell & Hamilton 1982
Grassland, manure & F.	S. Australia	1.900-9.400	-	1.700-9.200	-	-	Nash et al. 2000
Grassland, super-phosphate F.	Australia	1.040	0.140	-	-	-	Nelson et al. 1996
Grassland, super-phosphate F.	Australia	1.110	0.840	-	-	-	Nelson et al. 1996
Rotational grazing	Oklahoma	1.270	0.130	0.026	-	-	Olness et al. 1975
Continuous grazing	Oklahoma	4.600	0.140	0.025	-	-	Olness et al. 1975
Paddocks, fescue, bermuda grass	Georgia	-	-	3.000-9.700	-	-	Pierson et al. 2001
Grassland, not grazed during events	Nebraska	0.0920	-	-	-	-	Schepers & Francis 1982
Rotationally grazed, F.	Iowa	-	-	0.216	0.067	-	Schuman et al. 1973
Grassland, native	OK and Texas	0.195-0.218	-	-	-	-	Sharpley 1995
Grassland, various management	OK and Texas	0.020-4.390	-	-	-	-	Smith et al. 1992
Grassland	UK	-	-	0.010	-	-	Smith et al. 1992
Native prairie	Lake Mendota, WI	0.300	-	-	-	-	Soranno et al. 1996
Native grassland	Minnesota	0.110	-		-	-	Timmons & Holt 1977

[z] F. is fertilizer; SBR is sub-surface runoff; P. is plots; MP is moldboard plow.

Appendix 3. Contribution of phosphorus to aquatic systems via surface runoff from irrigated land.

Source	Location	Phosphorus fraction exported (kg ha^{-1} yr^{-1})					Author
		TP	DP	SRP	PP	BAP	
Low slope	Snake River, Idaho	0.610	-	0.150	-	-	Carter et al. 1974
High slopes	Snake River, Idaho	2.310	-	0.630	-	-	Carter et al. 1974
Irrigated land	Alberta irrigation districts		-	0.001-0.190	-	-	Greenlee et al. 2000
Irrigated land	Battersea Drain	0.002-0.048	0.001-0.029	-	-	-	J. Little Unpublished
Irrigated cotton	Oklahoma	11.150	1.930	1.650	-	-	Olness et al. 1975
Flood irrigated	S. Alberta (Hayes)	0.200	-	-	-	-	Oosterveld & McMullin 1979
Sprinkler-irrigated	S. Alberta (Hayes)	0.300	-	-	-	-	Oosterveld & McMullin 1979

Appendix 4. Contribution of phosphorus to aquatic systems via surface runoff from other sources.

		Phosphorus fraction exported (kg ha^{-1} yr^{-1})				
Source	Location	TP	DP	SRP	PP	Author
		Forested Land				
-	-	0.200	-	-	-	Beaulac & Reckhow 1982
F1	Baptiste Lake, AB	0.09, 0.13			0.06, 0.065	Cooke and Prepas 1998
F2	Baptiste Lake, AB	0.05, 0.22			0.11, 0.03	Cooke and Prepas 1998
Two Creek	Athabasca River, AB	0.13	-	0.032	-	Mitchell Unpublished
Sakwatamau River	Athabasca River, AB	0.07	-	0.019	-	Mitchell Unpublished
Stream D	Baptiste Lake, AB	0.05	-	0.030	-	Mitchell Unpublished
Stream E.	Baptiste Lake, AB	0.12	-	0.030	-	Mitchell Unpublished
Stream F	Baptiste Lake, AB	0.12	-	0.030	-	Mitchell Unpublished
Stream K	Baptiste Lake, AB	0.15	-	0.08	-	Mitchell Unpublished
Stream L	Baptiste Lake, AB	0.03	-	0.02	-	Mitchell Unpublished
Stream D	Baptiste Lake, AB	0.09	-	0.04	-	Mitchell Unpublished
Stream E	Baptiste Lake, AB	0.07	-	0.02	-	Mitchell Unpublished
Stream F	Baptiste Lake, AB	0.15	-	0.03	-	Mitchell Unpublished
Stream K	Baptiste Lake, AB	0.15	-	0.09	-	Mitchell Unpublished
Stream L	Baptiste Lake, AB	0.07	-	0.03	-	Mitchell Unpublished
Stream D	Baptiste Lake, AB	0.16	-	0.09	-	Mitchell Unpublished
Stream E	Baptiste Lake, AB	0.25	-	0.09	-	Mitchell Unpublished
Stream F	Baptiste Lake, AB	0.25	-	0.08	-	Mitchell Unpublished
Stream K	Baptiste Lake, AB	0.02	-	0.01	-	Mitchell Unpublished
Stream L	Baptiste Lake, AB	0.22	-	0.10	-	Mitchell Unpublished
Woodland	-	-	0.142	-	-	Taylor et al. 1971
Aspen-Birch Forest	-	0.130	-			Timmons et al. 1977
Marchell Experimental Forest - Spring	Minnesota, U.S.A.	0.272	-	0.130	-	Verry & Timmons 1982
Marchell Experimental Forest - Summer	Minnesota, U.S.A.	0.046	-	0.010	-	Verry & Timmons 1982
Marchell Experimental Forest - Fall	Minnesota, U.S.A.	0.032	-	0.008	-	Verry & Timmons 1982
Marchell Experimental Forest - Annual	Minnesota, U.S.A.	0.350	-	0.148	-	Verry & Timmons 1982

Appendix 4. Continued.

		Phosphorus fraction exported (kg ha^{-1} yr^{-1})				
Source	Location	TP	DP	SRP	PP	Author
		Urban Industrial				
Stormwater Outfall 1	Calgary, AB	0.201	-	-	-	Dixon 1994
Stormwater Outfall 2	Calgary, AB	0.156	-	-	-	Dixon 1994
Cargill Slaughterhouse	Frank Lake, AB	50.000	-	-	-	White & Bayley 2001
MWTP	Frank Lake, AB	7.700	-	-	-	White & Bayley 2001
		Groundwater				
	Narrow Lake, AB	0.390	-	-	-	Shaw et al. 1990
	Battersea Drain	0.010	-	-	-	Zilkey 2001
		Septic Systems				
kg $tank^{-1}$ yr^{-1} immediate area	Florida	0.138	-	-	-	Shannon & Brezonik 1972
kg $tank^{-1}$ yr^{-1} remote area	Florida	0.014	-	-	-	Shannon & Brezonik 1972
kg $capita^{-1}$ yr^{-1}	Canada	0.21	-	-	-	Chambers et al. 2001
kg $capita^{-1}$ yr^{-1}	Dorset, ON	0.80	-	-	-	Dillon et al. 1994
		Atmosphere				
Wet deposition	Lake Mendota, WI	0.262	-	-	-	Bennett et al. 1999
Dry deposition	Lake Mendota, WI	0.262	-	-	-	Bennett et al. 1999
Rainfall and snowfall	Rawson Lake, ON	0.24-0.53				Schindler et al. 1976
	Narrow Lake, AB	0.203	-	-	-	Shaw et al. 1989
Extrapolated from 8 mths data	Baptiste Lake, AB	0.0231	-	-	-	Trew et al. 1978
Forest Net precipitation-Spring	Minnesota, U.S.A.	0.187	-	0.054	-	Verry & Timmons 1982
Forest Net precipitation-Summer	Minnesota, U.S.A.	0.965	-	0.494	-	Very & Timmons 1982
Forest Net precipitation-Fall	Minnesota, U.S.A.	0.335	-	0.172	-	Verry & Timmons 1982
Forest Net precipitation-Annual	Minnesota, U.S.A.	1.487	-	0.720	-	Verry & Timmons 1982

Appendix 5. Retention or export of phosphorus in lentic and lotic systems.

Table A5.1. Phosphorus (P) source and sink values.

Author	Location	Description	Flux	P fraction measured	Source ($g\ m^{-2}\ yr^{-1}$)	Sink ($g\ m^{-2}\ yr^{-1}$)	±%
		Wetlands and riparian areas					
Clausen & Johnson 1990	Vermont	-	-	TP	-	1.1	-
Davis & van der Valk 1978	Eagle Lake Marsh	Receiving ag. runoff	Inflow-outflow	-	-	0.62	-
Davis et al. 1981	Eagle Lake Marsh	Prairie marsh	-	SRP	-	3.1 kg/ha	-
Johnston 1991	South Dakota	Mineral soil	Water to Soil	TP	-	1.46	-
Johnston 1991	South Dakota	Organic soil	Water to Soil	TP	-	0.26	-
Johnston 1991	South Dakota	Cultivated w.	Water to Soil	TP	-	0.57	-
Johnston 1991	South Dakota	Uncultivated w.	Water to Soil	TP	-	0.30	-
Johnston 1991	South Dakota	-	Inflow-outflow	TP	-	0.17 to 0.73	9.0-80.0
Johnston 1991	-	-	Water/vegetation	-	0.70	0.30	-
Johnston 1991	-	% of plant uptake	Litter loss	-	-	-	20.0-45.0
Johnston et al. 1984	-	Fringe wetland	-	TP	-	2.60	-
Kadlec & Hey 1994	-	Summer	-	-	-	-	60.0-100.0
Kadlec & Hey 1994	-	Winter	-	-	-	-	27.0-100.0
Kent 1987	St. Paul, AB	Receiving MWWTP	-	-	-	-	90.0
Kronvang et al. 1999	Gjern River Basin, Denmark	Riparian Wetland	Flood event	TP	-	0.16-6.50	2.7-5.4
Mitsch 1992	Des Plains River	Constructed riparian	Low flow	TP	-	0.45-0.69	83.0-96.0
Mitsch 1992	Des Plains River	Constructed riparian	High flow	TP	-	1.49-1.54	63.0-0.68
Mitsch 1995	Ohio	Instream	-	TP	-	2.900	-
Mitsch & Gosselink 1993	Old Woman Creek	-	-	-	-	0.8	10.0
Mitsch et al. 1979	S. Illinois	Riparian	-	TP	-	3.60	-
Nichols 1983	-	Mod.-cold climate	-	TP	-	0.005-0.220	-
Nichols 1983	-	Undisturbed organic soil	-	TP	-	0.100-0.200	-
Richardson & Craft 1993	-	Everglades	Inflow-outflow	-	-	0.40	-
Richardson & Qian 1999	-	-	-	-	-	1.00	-
Verry & Timmons 1982	Minnesota, U.S.A.	Peatland	Spring	TP	-	0.459	74.0
Verry & Timmons 1982	Minnesota, U.S.A.	Peatland	Summer	TP	-	0.183	46.0

Table A5.1. Continued.

Study	Location	Description	Flux	P fraction measured	Source ($g\ m^{-2}\ yr^{-1}$)	Sink ($g\ m^{-2}\ yr^{-1}$)	$\pm$%
			Wetlands and riparian areas				
Verry & Timmons 1982	Minnesota, U.S.A.	Peatland	Fall	TP	-	0.065	43.0
Verry & Timmons 1982	Minnesota, U.S.A.	Peatland	Annual	TP	-	0.707	61.0
Verry & Timmons 1982	Minnesota, U.S.A.	Peatland	Spring	SRP	-	0.217	82.0
Verry & Timmons 1982	Minnesota, U.S.A.	Peatland	Summer	SRP	-	0.007	8.0
Verry & Timmons 1982	Minnesota, U.S.A.	Peatland	Fall	SRP	-	0.005	15.0
Verry & Timmons 1982	Minnesota, U.S.A.	Peatland	Annual	SRP	-	0.229	60.0
White 1997	Frank Lake, AB	Receiving MWWTP	-	TP	-	3.2	64.0
White 1997	Frank Lake, AB	Receiving MWWTP	-	SRP	-	-	57.0
			Rivers and streams				
Haggard et al. 2001	Oklahoma	-	-	SRP	-	200-900m[z]	-
Kronvang et al. 1999	Gjern River Basin, Denmark	mid-March to mid-September	-	TP	-	3.7-8.3	-80.0[y]
Nielson & Mackenzie 1977	-	-	-	-	-	-	14.0
			Streambank erosion				
Coote et al. 1982	Lake Erie	-	-	TP	-	0.003-0.11	-
Culley & Bolton 1983	Essex County, ON	-	-	TP	-	0.145	-
			Lakes and reservoirs				
Kronvang et al. 1999	-	-	-	TP	-	0.30	-
Sosiak & Trew 1996	Pine Lake, AB		Internal release	TP	4.38	-	-61.0
Welch & Cooke 1995	AB	13 eutrophic lakes	Internal release	TP	1.168$\pm$ 1.168	-	-
Welch & Cooke 1995	Typical	-	Internal release	TP	0.73-1.825	-	-
Welch & Jacoby 2001	Western Washington	14 lakes	Internal release	TP	-	-	-68.0 $\pm$ 21.0

[z] Sink defined as uptake length.
[y] % of retained P resuspended in storm event.

Table A5.2. Retention of phosphorus in wetlands (g m^{-2} yr^{-1}).

		Phosphorus fraction retained					
Source	Location	TP	DP	SRP	PP	BAP	Author
No anthropogenic inputs		0.170-0.730	-	-	-	-	Johnston 1991
Mineral soils		1.460	-	-	-	-	Johnston 1991
Organic soils		0.260	-	-	-	-	Johnston 1991
Cultivated watersheds		0.570	-	-	-	-	Johnston 1991
Uncultivated watersheds		0.300	-	-	-	-	Johnston 1991
Prairie pothole, uncultivated watershed	South Dakota	0.570	-	-	-	-	Martin and Hartman 1987
Prairie pothole, uncultivated watershed	South Dakota	0.300[z]	-	-	-	-	
Fringe wetland		2.600	Inorg. 1.2, Org. 1.4	-	-	-	Johnston et al. 1984
Instream wetland	Ohio	2.900	-	-	-	-	Mitsch 1995
Riparian wetland	S. Illinois	3.600	-	-	-	-	Mitsch et al. 1979
Fringe wetland	Vermont	11 kg ha^{-1} yr^{-1}	mean/day x 365	-	-	-	Clausen and Johnson 1990
Moderate to cold climate wetlands		0.005-0.220	-	-	-	-	Nichols 1983
Undisturbed organic wetland soils		0.100-0.200	-	-	-	-	Nichols 1983
Prairie wetland		0.500	-	-	-	-	Richardson and Craft 1993
Prairie wetland		< 1.00 (mean 0.500)					Richardson and Craft 1993
Prairie wetland receiving ag. runoff		0.620	-	-	-	-	Davis and van der Valk 1978
Riparian wetland	Denmark	0.160-6.500[y]					Kronvang et al. 1999
Constructed riparian wetland, low flow	Des Plains River	0.45-0.69					Mitsch 1992
Constructed riparian wetland, high flow	Des Plains River	1.49-1.54					Mitsch 1992
Instream wetland?	Old Woman Creek	0.800					Mitsch and Gosselink 1993
		1.000					Richardson and Qian 1999

[z] Water to vegetation. Note vegetation can release 0.70 g m^{-2} yr^{-1} back to water (Johnston et al. 1991).
[y] Overbank flood event.

Table A5.3. Phosphorus (P) source and sink values.

Study	Location	Description	Flux	P fraction measured	Source g m^{-2} yr^{-1}	Sink g m^{-2} yr^{-1}	±%
		Wetlands and riparian areas					
Davis and van der Valk 1978	Eagle Lake Marsh	Prairie marsh receiving ag runoff	Inflow-outflow			0.62	
Davis et al. 1981	Eagle Lake Marsh			SRP		3.1 kg/ha	
Johnston 1991		Mineral soil	Water to Soil	TP		1.46	
Johnston 1991		Organic soil	Water to Soil	TP		0.26	
Johnston 1991		Cultivated watershed	Water to Soil	TP		0.57	
Johnston 1991		Uncultivated watershed	Water to Soil	TP		0.30	
Johnston 1991			Inflow-outflow	TP		0.17 to 0.73	
Johnston 1991			Water to vegetation			0.30	
Johnston 1991			Vegetation to water		0.70		
Johnston et al. 1984			Fringe wetland			2.60	
Kadlec and Hey 1994		Summer					60.0-100.0
Kadlec and Hey 1994		Winter					27.0-100.0
Kent 1987	St. Paul, Alberta	Receiving MWWTP					90.0
Kronvang et al. 1999	Gjern River Basin, Denmark	Riparian Wetland	Overbank flood event	TP		0.16 to 6.50	2.7-5.4
Mitsch 1992	Des Plains River	Constructed riparian	Low flow			0.45	83.0
Mitsch 1992	Des Plains River	Constructed riparian	Low flow			0.69	96.0
Mitsch 1992	Des Plains River	Constructed riparian	High flow			1.49	63.0
Mitsch 1992	Des Plains River	Constructed riparian	High flow			1.54	68.0
Mitsch and Gosselink 1993	Old Woman Creek					0.8	10.0
Richardson and Craft 1993		Prairie marsh	Inflow-outflow			< 1.00 av. 0.50	
Richardson and Qian 1999						1.00	
White 1997				TP			64
White 1997				SRP			57
		Lakes and reservoirs					
Sosiak and Trew 1996		Internal release		TP	4.38		-61.0