

CONNECTING PHOSPHORUS LOSS FROM AGRICULTURAL LANDSCAPES TO SURFACE WATER QUALITY

R. W. MCDOWELL^{a,*}, B. J. F. BIGGS^b, A. N. SHARPLEY^c and L. NGUYEN^d

^aAgResearch Ltd, Invermay Agricultural Centre, Private Bag 50034, Mosgiel, New Zealand;

^bNational Institute of Water and Atmospheric Research, P.O. Box 8602, Riccarton, Christchurch, New Zealand; ^cUSDA-ARS, Pasture Systems and Watershed Management Research Unit, Curtin Road, University Park, PA 16802-3702, USA; ^dNational Institute of Water and Atmospheric Research, Gate 10 Silverdale Road, Box 11-115, Hamilton, New Zealand

(Received 23 May 2003; In final form 16 September 2003)

The loss of phosphorous (P) from the landscape is commonly viewed as deleterious for surface water quality. However, the quantities lost and the impact this can have on surface waters depends on numerous mechanisms that occur whilst en route. The aim of this review is to give an outline of these mechanisms and thus how sources of P in the agricultural landscape are connected to the impairment of surface water quality. Processes are dealt with by examining the potential for P loss from the landscape and its availability to aquatic plants during flow overland and subsurface flow and once in streamflow or a lake or reservoir. By examining the connectivity between P loss and the impact on surface water quality, potential mitigation and management of P losses are discussed for various aquatic systems.

Keywords: Connectivity; Phosphorus; Stream; Channel; Agriculture; Hydrology; Eutrophication

1 INTRODUCTION

Losses of phosphorus (P) from land, an essential nutrient for crop and animal production, have caused major problems world-wide in streams, rivers and lakes through the effects on aquatic ecosystem production (Carpenter *et al.*, 1998; Sharpley, 2000). Although these losses might be minor compared with the amount of fertilizer applied to land, aquatic primary producers can be extremely sensitive to even minor increases in P. For example, freshwater algae can sequester P at the picomolar level and thus even very minor increases in P concentrations can potentially stimulate algal production (Hudson *et al.*, 2000). Thus, in waterways where primary production is P-limited, increasing P supply to concentrations in the low parts per billion range has led to high growths of algae and associated eutrophication (e.g., Bothwell, 1985; Biggs, 2000a). Recently, the US Environmental Protection Agency (1996) identified eutrophication as the most ubiquitous water quality impairment in the US, with agriculture a major contributor of P (US Geological Survey, 1999). Similarly, extensive

* Corresponding author. E-mail: richard.mcdowell@agresearch.co.nz

problems with eutrophication have been reported in other areas such as New Zealand (Biggs, 1985), Australia (Harris, 1994), and Europe (Whitton, 2000). The outcome of such anthropogenic eutrophication is that water use for fisheries and recreation is restricted due to the increased growth of undesirable (and sometimes toxic) algae and aquatic weeds, and oxygen shortages caused by the decomposition of plant matter (*e.g.*, Cyanobacteria and *Pfiesteria*). Such blooms contribute to summer fish kills, unpalatability of drinking water, the formation of carcinogens during water chlorination, and have been tentatively linked to neurological impairment in humans (Kotak *et al.*, 1993; Burkholder and Glasgow, 1997). The economic impact of eutrophication on fishing and water treatment industries in the eastern US alone has amounted to over \$1 billion over the last decade (Greer, 1997).

Attention has centred on agriculture as a primary source of P loss to surface waters and thereby eutrophication. This is due to an extensive number of catchment-scale studies that have shown much higher concentrations of P in streams draining areas of agricultural land compared with undeveloped areas (*e.g.*, Smith *et al.*, 1993; Cooper and Thomsen, 1988). Also, it is easier to identify and mitigate sources of P than nitrogen (N, another limiter of eutrophication), although there are some uncontrollable P inputs via precipitation (*c.* 0.1–6.5 kg ha⁻¹ yr⁻¹; Newman, 1995). In addition, intensification and specialization of farming systems has led to regional surpluses of P imported in fertilizer and animal feed compared with P exported in farm produce (Carpenter *et al.*, 1998). Now many farms have soil P concentrations well in excess of plant needs with increased potential for P loss (Sharpley, 2000).

We have begun to realize that P loss does not occur from the entire catchment and small areas within the catchment can dominate P losses to streams. The dominance of these small areas, termed critical source areas (CSAs), is dependant upon many factors, including soil type, topography, management (*e.g.*, inputs of fertilizer and manure, and off-takes in crops or forage), and transport processes that are dependent upon environmental and hydrological conditions. The interaction between these factors is complex and varies spatially and temporally. However, in general, CSAs are defined by a high concentration of P available to flow and a high potential for flow, equating to a high potential for loss. Nevertheless, the interactions between P availability and flow regimes are intricate and complicated further by the influence of downstream effects which mediate the eventual impact of landscape P loss on surface water quality. Numerous reviews have dealt with P loss from land to water or the impact once in the stream (*e.g.*, Correll, 1998; Sharpley, 2000). Little attempt has been made to link the two systems together. Hence, our objectives in this review are to outline the major interactions and processes involved in connecting P loss from the landscape to the site of impact, whether that is a stream, river or lake. This review is split into three sections. The first deals with P loss from the landscape, whilst the second deals with processes that alter the behavior and impact of P once in the aquatic system. Finally, a third section outlines a secondary objective, which is to examine some management practices for P in relation to the connectivity between mitigating losses from the landscape and associated impacts on freshwater ecosystems.

2 TERMINOLOGY

Terminology needed to understand this review is divided into three categories: those that relate primarily to flow pathways and aquatic systems (limnological); those that relate to P impact in aquatic ecosystems (ecological); and those that relate to the behavior and bioavailability of P (chemical). A theoretical schematic of interactions is given in Figure 1.

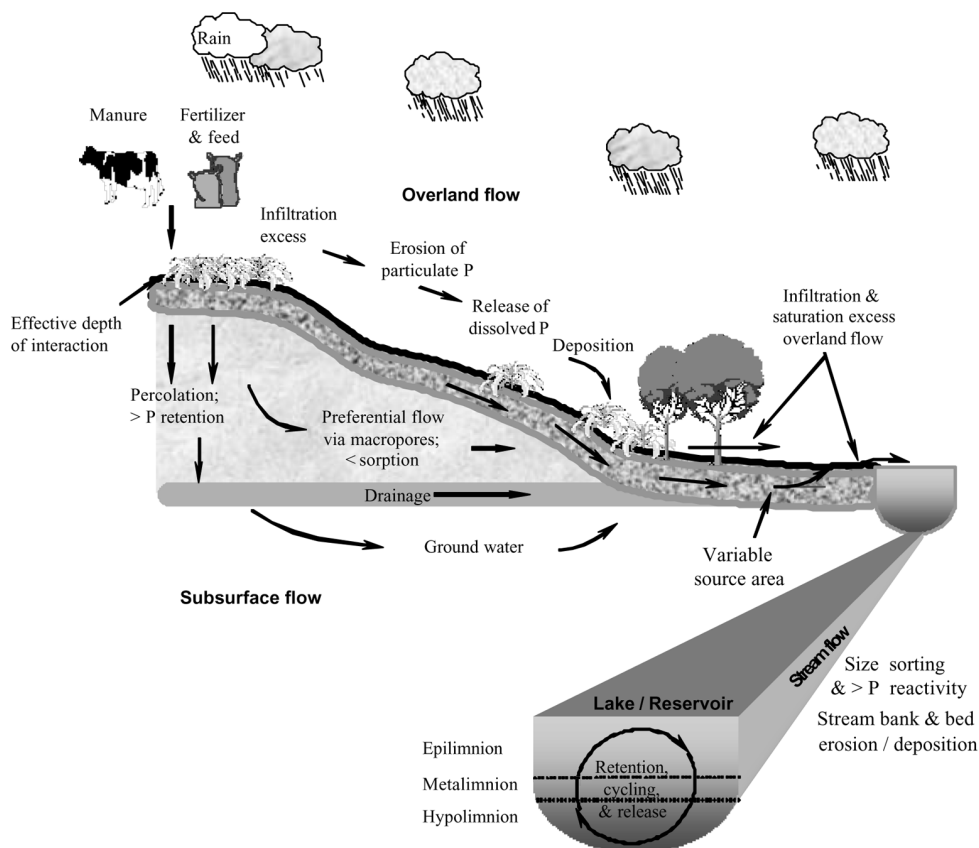


FIGURE 1 Conceptual diagram of processes that transport P from the landscape to surface water (adapted from McDowell *et al.*, 2003).

2.1 Limnological Terminology

Flow pathways and aquatic systems are referred to by:

Catchment (or watershed). Land surface delineating a drainage area from which water is discharged in stream flow.

Stream. Water that flows in a natural channel.

Runoff. That part of precipitation which ends up in streams or lakes (*i.e.* the combined flow of surface water, subsurface drainage and groundwater pathways).

Overland flow (or surface runoff). That part of rainfall or snowmelt which flows overland to streams or directly to lakes.

Subsurface flow (or drainage). That part of rainfall or snowmelt which infiltrates the soil and moves to streams or lakes as ephemeral, shallow, or perched flow above the water table. In agricultural landscapes, the downward movement (or *leaching*) of subsurface flow can be intercepted by artificial drainage systems such as tile drains.

In addition to overland and subsurface flow, other contributors of flow to streams and lakes include direct precipitation and *groundwater flow*, which is precipitation that has become part of the ground water *runoff*, via flow through the *hyporheic zone*, the saturated area beneath stream banks and beds where a mixture of surface and ground water occurs.

In deep reservoirs and lakes, stratification into layers of different water density may occur due to temperature differentials. The region of warm water near the surface of stratified water bodies is called the *epilimnion*, the cold deep water the *hypolimnion* and the intermediate layer the *metalimnion*. This stratification prevents water and P from mixing and is only broken down by turbulence or large changes (*e.g.*, changing seasons) in temperature. This is called *turnover*. This break-down and mixing can often lead to changes in the ecology and trophic status of the water body.

2.2 Ecological Terminology

For surface water bodies, many trophic states exist. In water bodies where secondary production (*i.e.* animal growth) and respiration exceeds primary production (nutrient related algae and plant growth), an external source of energy is required. Such water bodies are called *heterotrophic*; a forested catchment where leaf fall is the main source of energy is an example. In water bodies where primary production is equal to or greater than secondary production, four trophic states are possible:

Oligotrophic. Water bodies with low nutrient supply that typically have good water clarity (the exception being Bog lakes).

Mesotrophic. Intermediate in nutrient supply between oligotrophic and eutrophic states.

Eutrophic. Well nourished systems rich in nutrients.

Hypereutrophic. Water bodies with an excessive supply of nutrients.

The growth of aquatic plants will vary between three basic forms, each posing a different problem for management if an over-abundance occurs causing a decrease in water quality:

Phytoplankton. Microscopic algae suspended in the water column that take up P directly from water. They include cyanobacteria which have gas vacuoles allowing them to float to the surface for maximal light exposure, also causing a surface scum.

Periphyton. Dominated by algae, but also inclusive of some bacteria, fungi and protozoa. These attach to surfaces such as rocks, woody debris or vascular plants. The two main types of algae are the many cell long chain filamentous algae and single cell non-filamentous algae attached to the bed.

Macrophytes. Vascular plants with roots, stem and leaves that overall are larger than 0.5 mm. They may be rooted in sediment or free-floating. Those that are rooted obtain their P from bed sediment and foliar uptake and also help to minimize sediment disturbance.

Where each class of plant occurs depends on P supply and the physical environment. For instance, in deep, turbid water, periphyton does not grow due to light limitation, but would support phytoplankton, which could float near the surface. On the other hand, fast flowing rivers do not support phytoplankton, which are swept away before becoming established. Macrophytes can take up P from sediment and streamflow, and following grazing or dieback, P in macrophytes may become available for phytoplankton. Excessive phytoplankton growth can restrict macrophyte growth via light limitation. However, management to decrease phytoplankton may allow macrophytes to re-establish.

2.3 Chemical Terminology

In terms of P forms, total P (TP) in flow can be separated into dissolved P (DP) or particulate P (PP) by filtration through a 0.45 μm membrane filter, and in-turn, inorganic or organic fractions. Inorganic P or orthophosphate is commonly referred to as that detectable by

molybdate-based colorimetry of Watanabe and Olsen (1965). Some confusion can arise in the detection of inorganic P by colorimetry via the enhanced hydrolysis of loosely-bound inorganic and organic P forms (Stevens, 1979), or small colloidal material depending upon filtration (Sinaj *et al.*, 1998; McDowell and Sharpley, 2001a). Most confusion can be overcome by carefully defining P forms. Here, P is simply defined as dissolved reactive P (DRP) if previously filtered ($<0.45\ \mu\text{m}$) and detectable by colorimetry, otherwise samples without the filtration method specified (but still largely inorganic P) are termed reactive P (RP). A filtered and unfiltered sample subjected to a digestion for TP yields total dissolved P (TDP) and TP, with PP calculated as the difference between TP and TDP. A fraction commonly defined as dissolved organic P (DOP), but more accurately described as dissolved unreactive P (DURP) due to the aforementioned interference, is obtained as the difference between TDP and DRP.

In terms of availability to aquatic plants, and thus potential for eutrophication, RP or DRP is generally considered fully available while only a proportion of PP is available (Biggs, 2000a; Table I; Sharpley, 2000). A large proportion of DOP can become algal available in the aquatic environment if it is broken down to orthophosphate by extracellular enzymes such as alkaline phosphomonoesterase, phosphodiesterase, phospholipase and phytase (Whitton *et al.*, 1991; Wetzel, 2001). The use of DRP is advocated for assessing the nutrient status of rapidly flowing streams and rivers (especially where there are moderate to high concentrations of inorganic sediments which can be high in P; Biggs, 2000a). Conversely, TP is advocated for assessing the nutrient status of lakes, as much PP is present in algae where it can be solubilized and made available for further algal growth (Dodds and Welch, 2000). An additional measurement is Mat P or the percentage mass of periphyton mat that is composed of P (Table I). Mat P is determined following digestion of a small subsample of the mat. This parameter incorporates all P in the mat and does not distinguish between organic and inorganic fractions, nor that contained in different biotic components (*e.g.*, algae, protozoa, micro-invertebrates, etc.), but can be useful in determining growth rates and for use in modeling (Table I).

3 THE POTENTIAL FOR P LOSS FROM THE LANDSCAPE

3.1 Dissolved P

In acidic soils, P occurs largely as Al- and Fe-phosphates, whereas in neutral to alkaline soils P occurs largely as Ca- and Mg-phosphates sorbed onto the surface of Ca and Mg carbonates (Lindsay, 1979). Phosphorus is most available and mobile at pH 6–7. However, combinations of Al-, Fe-, Ca- and Mg-P can occur in most agricultural soils. Consequently, forested landscapes generally lose less P compared to agricultural landscapes where the pH is maintained nearer an optimum for plant growth (5.5–7.0). Phosphorus loss is supplemented by the inherent P content of the soils, weathered from parent rock material with different P concentrations as either fluoroapatite or apatite. Basic igneous rock contains little P as apatite, while limestone, andesite, unweathered loess, sandstone, gneiss, and diabase contain 1.3%, 0.16%, 0.07%, 0.04%, 0.03% and 0.03% P, respectively. Organic P also forms a significant part of soil P especially in acidic soils and soils that contain much organic matter and N. Landuse can have a significant impact on the quantity of P in organic forms with the ratio of organic P to inorganic P decreasing from forest to grassland to arable systems.

The solubility of soil P is controlled by three chemical characteristics: (i) concentration of P in solution; (ii) quantity of P in the soil that equilibrates with the solution; and (iii) buffering capacity of the soil controlled by sorption strength and the saturation of sorption sites with P, and influenced soil chemical conditions (*e.g.*, pH). Coupled with soil P solubility,

the release of P into flow varies with time. Kinetic exchange experiments using ^{33}P , have confirmed that rapidly exchangeable soil P (within 60 seconds) is closely related to P in overland or subsurface flow (McDowell *et al.*, 2001a). With time, P transport in overland flow becomes less related to this pool and more dependent upon the slow diffusion of P from the inside of the soil aggregate (Sharpley and Ahuja, 1983; McDowell and Sharpley, 2003a). This serves to illustrate that soil P release to overland flow is a function of the surface area available to the solution, as well as the quantity of P in soil.

During most high-flow events, the major release of P occurs at the start of the flow event when the labile pool of P is lost along with any mobile PP. At any one time, the load of DP in flow (overland or subsurface, kg) can be described by the product of soil test P (STP; such as Olsen P), flow volume (mm) and an extraction coefficient which describes the fraction of STP that can be released into flow. The extraction coefficient can be obtained from the slope of a plot of STP and overland flow DRP. For overland flow, this relationship is generally linear, while for subsurface flow the relationship is often described as either curvilinear (Lory *et al.*, 2001) or represented by two linear regressions intersecting at a soil P threshold above which P release is greater (Heckrath *et al.*, 1995; McDowell and Sharpley, 2001a, b). A recent paper by Koopmans *et al.* (2002) showed that in theory all relationships between DRP in flow and STP will be curvilinear, while the degree of curvature is dependent upon a number of factors, such as:

- The range of STP measured. The likelihood of detecting a curvilinear relationship decreases with the range of STP measured.
- Soil to solution ratio. Curvature increases as soil to solution ratio decreases (Fig. 2A).

In general, less soil is in contact with overland flow than with subsurface flow, and hence the relationship between STP and DRP in overland flow can appear linear. Field and laboratory data for UK and USA soils show the influence of different flow regimes and soil to solution ratios on the relationship (Fig. 2B and C).

Factors that enhance P extractability relative to STP include soil texture (P is more mobile in sandy soils), soils with a low P retention capacity, waterlogged soils where P is mobilized under reducing conditions, and the application of P above plant requirements.

Some organic P forms are more mobile than inorganic P forms. For example, Chardon *et al.* (1997) showed the enhanced downward movement through soil of organic P forms to depths of 70 cm or more following several years of application of swine manure to a sandy soil in the Netherlands. Studies using known organic P compounds such as deoxyribose nucleic acid, adenosine 5'-triphosphate and phospholipids have demonstrated their lesser affinity with soil compared to orthophosphate (Stewart and Tiessen, 1987; Leytem *et al.*, 2002). A notable exception are inositol hexaphosphates, which tend to be more strongly bound to the soil than orthophosphate (Turner *et al.*, 2002a). Some of these compounds are also available for use by algae (via exocellular enzymes) and thus can pose a problem in surface waters (Whitton *et al.*, 1991). At present, studies relating the loss of organic P species to specific soil characteristics are few. While most identify that the source of organic P is largely manurial in origin, and thus subject to manure management, the background contribution of plant and soil is less well understood and the subject of much ongoing research (Turner *et al.*, 2002b).

For inorganic P, the accuracy of relationships between P loss in flow, soil extractability and STP is subject to scale effects and variability. Preferential flow pathways, such as earthworm burrows and old root channels, are very important in the downward movement of P through the soil profile and could be missed during sampling (Sims *et al.*, 1998; McDowell and Sharpley, 2001b; Kleinman *et al.*, 2003). Jensen *et al.* (1998) showed that the concentration and saturation of P in the soil lining earthworm burrows and not the bulk soil was a major

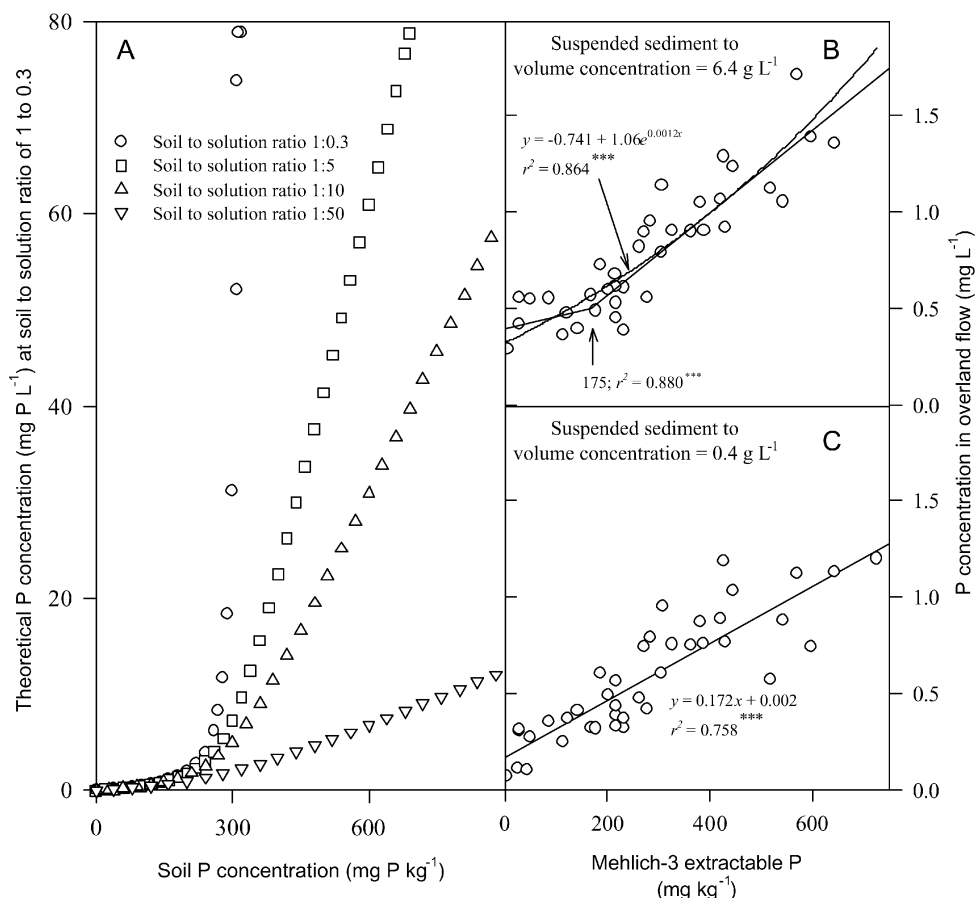


FIGURE 2 Plots showing the influence of soil to solution ratio on the theoretical expression of P in flow (A: theoretical soil of 300 mg kg^{-1} P sorption maximum and 1.5 L mg^{-1} Langmuir affinity constant, Koopmans *et al.*, 2002), and field and laboratory data for P in overland flow with two different (B and C: McDowell and Sharpley, 2001c) mean suspended sediment concentrations (*i.e.* eroded soil to solution ratios). Note the fit of a split-line and exponential model in B is a much better fit than a linear function, while the opposite is true in C.

determinant in the P lost in subsurface flow. As such, preferential flow pathways are often a conduit for the rapid transfer of surface applied P, either as manure or fertilizers, to drains and ditches (Withers *et al.*, 2001). On a catchment scale, this pathway is emphasized by the considerable quantity of P that can be lost during base flow in a stream when the flushing potential of a storm event is absent (*e.g.*, when liquid manure is spread on soils; Preedy *et al.*, 2001; Withers *et al.*, 2001). Such losses are made worse by the coincidence of manure spreading or deposition from grazing animals onto wet soils (Stamm *et al.*, 1998; Monaghan *et al.*, 2002). The accelerated transfer from this unfortunate combination is termed an 'incidental transfer' and can result in the loss of P in concentrations of 30 mg L^{-1} or more (Haygarth *et al.*, 2000; Withers *et al.*, 2001).

By ignoring incidental transfers, many researchers have used a constant extraction coefficient value (assuming STP extractability is similar among soils) to estimate the load of DP to surface water. This may be a reasonable enough assumption for fields not receiving any P inputs, or for background (*i.e.*, non-incidental) P losses. However, most fields will be subject to some proportion of incidental P loss subject to fertilizer and/or manure application rates.

Fortunately, with increasing scale these inputs can be buffered depending on land use. Catchments with a high sediment load also exhibit a strong buffering potential against incidental losses. This was examined by [Sharpley *et al.* \(2002\)](#), who reanalyzed data published by [Sharpley and Smith \(1994\)](#), [Pote *et al.* \(1999\)](#), [McDowell and Sharpley \(2001c\)](#). They found that the extraction coefficient (slope of the linear relationship between STP and overland flow P) increased with greater erosion or decreased soil cover (Fig. 3). It was assumed that erosion is a direct consequence of land cover and a surrogate estimate of the degree of surface soil–overland flow interaction ([Sharpley, 1985a](#)). A larger soil P extraction coefficient yields more DP into overland flow per unit STP increase, caused by a greater interaction between soil via erosion and overland flow (Fig. 3; [Sharpley *et al.*, 2002](#)). On a catchment scale, some evidence suggests that DP loss in overland flow is a function of erosion and land management, such as the proportion of cultivated fields present in catchments (*e.g.*, [Smith *et al.*, 1993](#); [Ekholm *et al.*, 2000](#)).

3.2 Erosion and Particulate P

Much TP loss can be in particulate form. Eroded particulate material is enriched with P compared to surface soil, due to the preferential transport of light and highly P-sorptive fines compared to coarse-sized particles. [Sharpley \(1985b\)](#) found that the STP content of suspended sediment was, on average, three times greater than bulk soil (0–5 cm depth) and 1.5 times greater in terms of TP. The degree of P enrichment (ER) is expressed as the quotient

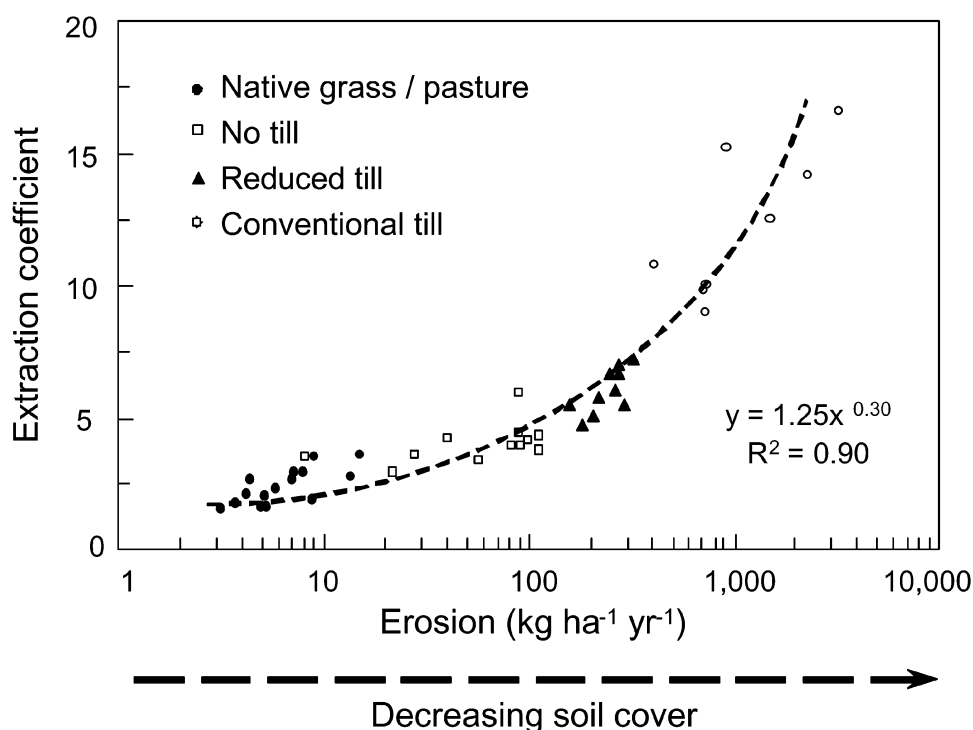


FIGURE 3 Extraction coefficient (slope of the relationship between soil test P and DP in overland flow) as a function of erosion to represent soil vegetative cover for sites in Arkansas, Oklahoma, New York, and Pennsylvania (data adapted from [Sharpley and Smith, 1994](#); [Pote *et al.*, 1999](#); [McDowell and Sharpley, 2001c](#); [Sharpley *et al.*, 2002](#)).

of P concentration of sediment in flow and in the contributing soil. Menzel (1980) concluded that, for PP, a logarithmic relationship ($\text{Ln ER} = 2.00 - 0.16 \text{ Ln sediment discharge}$) was appropriate for a wide range of vegetative conditions (*e.g.*, Fig. 4). Based on TP concentrations for several catchments (see Fig. 4), ER decreases with increasing erosion. As erosion increases, there is less particle-size sorting by overland flow, proportionately less clay-sized particles transported, and P ER decreases (Fig. 4). Once an appropriate particulate ER is obtained for sediment discharge, PP loss can be calculated as the product of soil TP (kg), sediment concentration (g L^{-1}), overland flow volume (L) and ER.

Quinton *et al.* (2001) emphasized that P lost from low intensity rainfall events was equally important as from high intensity rainfall events. McDowell and Sharpley (2002a) showed that the load of P as PP in overland flow increases with plot length. However, this can change if manure has been freshly applied. Here, the preferential loss of manure in overland flow overwhelms P loss from soil, and is enhanced by movement in low-density flocs associated with light particulate material (McDowell and Sharpley, 2002b). The transport of flocs plays a major role in the long-distance transport of suspended sediment in the fluvial system (Stone and Murdoch, 1989). Consequently, provided there is a connection between manure on a hillslope and the stream, then the potential for P loss is greater than has been traditionally thought (*i.e.* solely through the erosion of sediment and DP).

In unmanured soils, antecedent moisture conditions have a significant effect on the load and form of P lost. Soils that are wetter will have a greater potential for flow and thus P movement than dry soils. However, due to slaking and dispersion effects, dry soils can produce more PP in flow than wet soils when overland flow is suddenly produced, such as could occur during infiltration-excess overland flow. McDowell and Sharpley (2002a, b) showed that the potential for a soil to supply flow with P was much larger at the start of flow compared to the end of the flow event. The difference is greater in those soils with poor aggregate stability and the potential for slaking and dispersion (where the aggregates break apart due to osmotic pressures) is high.

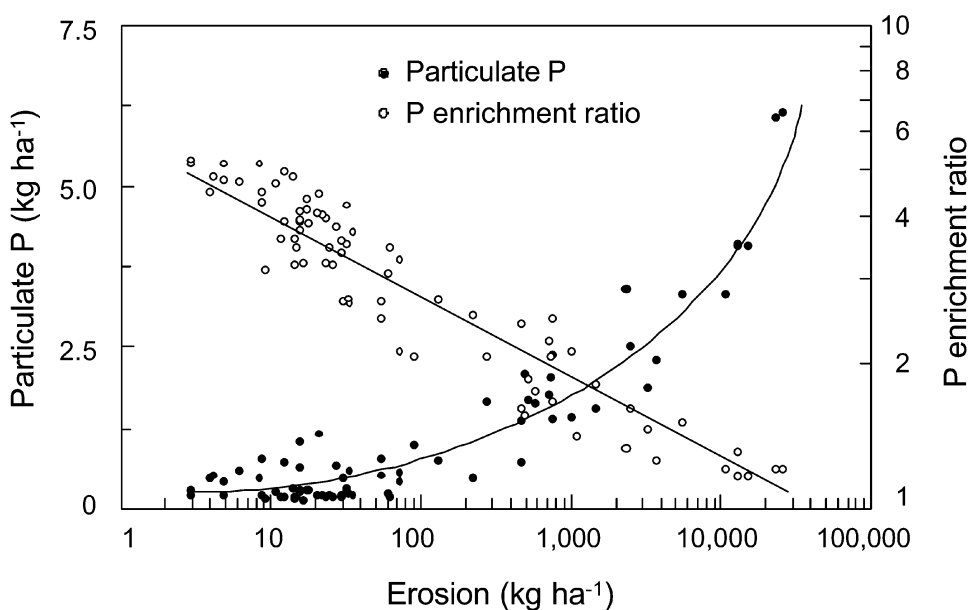


FIGURE 4 Particulate P loss and ER ratio of eroded sediment as a function of erosion in overland flow from catchments at El Reno, OK (adapted from Sharpley *et al.*, 1991; Smith *et al.*, 1991; Sharpley *et al.*, 2002).

3.3 The Impact of Land Management on P Loss

Fertilizer and manure management can profoundly affect P transfers in overland flow. While soil P represents a source of P in overland flow, the application of fertilizer and manure to soil, including factors such as the type, method, timing and rate of P application, can temporarily overwhelm relationships derived between STP and P in overland flow ([Sharpley and Tunney, 2000](#); [Sharpley *et al.*, 2001](#)). As such, accounting for fertilizer and manure management and minimizing the risk of incidental transfer is important for the mitigation of P loss.

Mineral fertilizer and manure can greatly increase DP losses in overland flow. [Kleinman *et al.* \(2002\)](#) illustrated the relationship between DRP concentration in overland flow and water extractable P (as an estimate of P in overland flow) concentration of several manures, composts, and diammonium phosphate surface applied to a Hagerstown soil (fine, mixed, semi-active, mesic Typic Hapludalf) at a rate of 100 kg TP ha⁻¹. Application of P increased DRP in overland flow by a factor of 4–26 times that lost from unamended soil, and shifted the majority of TP lost from particulate (90% in unamended soil) to dissolved (60% in amended soils) forms.

Equally important is the method of manure or fertilizer application ([Romkens *et al.*, 1973](#); [Mueller *et al.*, 1984](#); [Zhao *et al.*, 2001](#)). Surface application of manure and mineral fertilizer concentrates P at the soil surface where it is vulnerable to loss by overland flow ([Sharpley *et al.*, 1984](#); [Eghball and Gilley, 1999](#)). Depending upon rainfall intensity and slope gradient, the effective depth of interaction (EDI) between overland flow and P in topsoil ranges from approximately 1–40 mm ([Sharpley, 1985a](#)). Hence, injection, knifing or incorporation by cultivation removes P from the EDI.

Modifying the timing of application relative to likely flow events can significantly alter P concentrations in flow ([Westerman and Overcash, 1980](#); [Sharpley, 1997](#)). Following application, the potential for P loss is large and declines exponentially with time as P interacts with the soil and is converted to increasingly recalcitrant forms ([Edwards and Daniel, 1993](#)). [Sharpley and Syers \(1979\)](#) reported declining DRP (from >250 to <100 µg L⁻¹) and TP concentrations (from >700 to 100 µg L⁻¹) in tile drainage over one month following temporary, intensive grazing of paddocks by dairy cattle. Similarly, [Gascho *et al.* \(1998\)](#) observed exponential declines in DRP concentrations in overland flow (from >5000 to <1000 µg L⁻¹), roughly one month after fertilizer application.

In addition to affecting the availability of P to overland flow, manure and fertilizer P sources may directly impact soil physical properties that control runoff and erosion. In the short-term, surface application of manure, particularly at high loading rates, may increase soil cover, protecting it from rain drop impact and aggregate dispersion, especially if a crust has formed ([Bárthes *et al.*, 1999](#); [Smith *et al.*, 2001](#); [McDowell and Sharpley, 2003b](#)). Over the long term, addition of manure may increase soil organic matter levels, which, in-turn, affects porosity, aggregate stability, infiltration and P loss ([Rousseva, 1989](#); [Oades and Waters, 1991](#); [Gilley and Risse, 2000](#); [McDowell and Sharpley, 2003b](#)).

3.4 The Hydrologic Driver of P Loss from Landscapes

While we must consider sources of P such as soil and manures, the actual load of P loss from the landscape will be dictated by the volume of runoff. In other words, the transport of P in flow determines whether potential losses are translated into actual losses from a catchment.

Rainfall is the primary driving force behind P transfer, although some movement of P via wind erosion is also likely to occur in some regions. Rainfall events can be classified as those of:

- Low intensity and high frequency that tend to move P either in subsurface flow or overland flow in saturated areas.
- High intensity and low frequency that tend to move P by exceeding the infiltration capacity of soil and producing overland flow from a thin layer of P-rich topsoil (Fig. 1).

Due to the greater kinetic energy and erosive power of high frequency storms, more TP is lost during overland flow in particulate forms than in subsurface flow. For example, [Pionke *et al.* \(1996\)](#) showed that a few short, intense storms accounted for about 90% of the annual P export from an upland catchment. Similarly, Hortonian overland flow (limited by infiltration rate) will likely have a greater capacity to detach and move soil particles than overland flow caused by saturation-excess conditions (limited by soil water storage capacity).

In humid and temperate climates, saturation excess overland flow is described by variable source area (VSA) hydrology ([Ward, 1984](#); [Srinivasan *et al.*, 2002a](#)). Flow from these areas varies with time, expanding and contracting rapidly during a storm as a function of precipitation, temperature, soil-type, topography, ground water and moisture status over the catchment. The onset of flow from these areas is limited by soil water storage capacity and thus usually results from high water tables or soil moisture contents in near-stream areas. During a rainfall event, area boundaries will migrate upslope as soils saturate. In dry summer months, overland flow will come from areas closer to the stream than during wetter winter months, when the boundaries expand away from the stream channel. In catchments where infiltration-excess overland flow dominates, areas of the catchment can alternate between sources and sinks of overland flow, as a function of soil properties (largely infiltration rate), rainfall intensity, duration and antecedent moisture condition.

Transport and loss of P generally occurs from areas where overland flow contributes to stream flow, although some subsurface flow pathways may be important under certain hydrologic conditions. Loss of P in subsurface flow is generally less than that in overland flow, and will decrease as the degree of soil–water contact increases, due to sorption by P-deficient subsoils ([Haygarth *et al.*, 1998](#)). Exceptions occur where organic matter may accelerate P loss together with Al and Fe, the soil has a small P sorption capacity (*e.g.*, some sandy soils), or where subsurface flow travels from P-rich topsoil via macropores or is intercepted by artificial drainage (Fig. 1).

It has been thought that macropore flow only occurs when the soil is saturated. Later studies refined this to non-ponded conditions with saturated surface layers ([Adreini and Steenhuis, 1990](#)). Recent evidence using tracer dyes has shown that macropore flow may account for over half of total flow in unsaturated, undisturbed cores of well structured, dry clay soil ([Bootlink and Bourma, 1991](#)). Furthermore, on a larger, field scale, recent evidence has shown that macropore flow of effluent can occur almost immediately, once effluent has been applied ([Monaghan *et al.*, 2002](#)).

The importance of hydrology, and in-turn P losses, varies with scale. While laboratory studies using repacked or intact soil boxes have elucidated many mechanisms involving P losses in overland flow, the relevance of these to field losses *per se* is unclear (*e.g.*, [Sharpley, 1985a; 1995](#); [Srinivasan *et al.*, 2002b](#)). In plot studies, the hydrologic response time or the time taken for flow from the farthest point of the plot to reach the monitoring point is smaller compared to field scale studies. Furthermore, rainfall intensity declines logarithmically with time meaning that we can approximate peak flow from a plot as proportional to the contributing area ([Smith, 1992](#); [Nash *et al.*, 2002](#)). As such, peak flows per unit area from a plot are likely

to be greater than from a field. This is commonly found when sediment loads from small plots (20 m²) are upscaled and found to be much larger than that from much larger plots (500 m²) ([Le Bissonnais *et al.*, 1998](#)). However, when rainfall and overland flow are constant, longer flow paths will tend to have greater and faster flow volumes and erosion than smaller plots, but also more opportunity for the selective erosion of P-rich fines. Indeed, McDowell and Sharpley (2002b) showed that PP loss increased as flowpath length increased from 1 to 10 m. The relative proportion of dissolved P decreased as a result of the selective erosion of highly P sorptive clays (clay concentrations of 24% and 34% in 2- and 10-m long flowpaths, respectively).

Unfortunately, whilst the principles that apply to one scale can be applied to another, due to variability in the landscape and rainfall, the processes that cause P to be lost or gained in overland flow can occur at the same time in different positions in the landscape. Furthermore, antecedent soil moisture conditions within a catchment are not a direct indicator of the potential for P loss in overland flow. Very dry soils not expected to produce overland flow can do so and lose much P due to hydrophobicity, and slaking and dispersion effects that result in the loss of P-rich clay-sized material ([McDowell and Sharpley, 2002c](#)). This effect can also occur on a catchment scale where the selective erosion of fines only exacerbates the problem ([Rodda *et al.*, 1998](#)).

As discussed, the hydrologic and physiochemical factors controlling P loss vary in space and time. Increased net precipitation (determined as the difference between precipitation and evapotranspiration) to a catchment can increase the amount of discharge and the quantity of P lost by accelerating those transformations that occur before and after P reaches stream flow. However, we need to not only consider the quantity of P lost, but also the concentration and form of P loss. For example, whereas dissolved forms of P are immediately available to aquatic flora, particulate forms of P can represent a longer-term source of P via desorption. Depending upon how far down the water cycle the area of interest is (*e.g.*, first-order stream to lake), new flow and associated P can either remove and flush, or deposit and add, to the system.

3.5 Variation During Overland Flow

A potential for P loss does not necessarily translate to P loss from the landscape. During overland flow, soil and associated P is lost in order of increasing particle density and weight, while the opposite occurs for deposition during overland flow and in stream flow ([Slattery and Burt, 1997](#)). Thus, fine and/or light soil particles that contain many Al- and Fe-oxides and associated P or humic-associated P are transported before coarser and/or heavier sized particles. Eroded fine particles will be able to maintain an equilibrium stream P concentration for longer than coarser particles with less P in reserve. However, the concentration of P in water in equilibrium with fine particles can be much less (relative to the total concentration of P in the particle) than from coarse particles, which have a lower affinity for P and will release it faster initially ([Maguire *et al.*, 1998](#); [McDowell and Sharpley, 2002a](#)). It has been suggested that, once P is in solution, the transition between dissolved and particulate forms during overland flow can change, mediated by sorption/desorption properties of the sediments. However, recent evidence suggests that the majority of change during flow may simply be due to deposition and dilution ([McDowell and Sharpley, 2002b](#)), until flow reaches the stream where sorption/desorption from the underlying sediment then has a significant influence ([McDowell *et al.*, 2001b](#)). Indeed, much work has shown that sediments can remove or release P to the overlying water depending upon factors such as the affinity and saturation of sediment P sorbing sites and the kinetics of exchange ([Sharpley *et al.*, 1981](#); [House *et al.*, 1995](#)).

To investigate the influence of varying concentrated P sources along a simulated hillslope length, [McDowell and Sharpley \(2002a, b\)](#) studied P loss from soils treated with manure at

75 kg P ha⁻¹ at various positions upslope. Dissolved reactive P concentration was more closely related to the proportion of clay in sediment in overland flow before ($r = 0.98$) compared to after ($r = 0.56$) manure application. This was attributed to the transport of larger, low-density particles as flocs after applying manure. Surprisingly, the concentration of dissolved and PP fractions decreased with increasing flowpath length, due to dilution and deposition rather than sorption of P by surface soil during overland flow. This implied that sorption and/or desorption processes were either not occurring during the period of flow downslope, or they were complete by the time flow had reached the collection point. Using ³³P, supplementary evidence by McDowell *et al.* (2001a) suggests that P in overland flow is in a state of quasi-equilibrium and effectively complete, leaving dilution and deposition as the major processes affecting dissolved P loss. However, TP loss (mainly as PP derived from erosion) from one soil type was significantly more than from the other soil type studied, even with manure applied. Thus, while P loss in overland flow is affected by where manure is applied relative to flowpath length, initial soil P concentration has a major effect on TP loads and should not be discounted when looking at areas of potential P loss within a catchment.

4 CONNECTING P LOSS TO P FATE IN SURFACE WATERS

Once P from the landscape reaches the stream, in-channel processes can modify the potential for agriculture to impact a downstream freshwater body. As surface water impacts are the ultimate determinant for remedial efforts, understanding in-channel processes and how these vary the impact of transported P on downstream water bodies is necessary to link changes in agricultural management with downstream water quality impacts.

4.1 In-Channel Processes

The concentration of P in streams and rivers is defined by the inputs of P from contributing sources, such as overland flow from adjacent land, stream bank erosion, and by dilution and hydraulics of flowing water. These control the availability of P to flowing waters and define where availability is expressed. Two main processes mediate in-channel changes in P concentrations: (a) sediment geochemical processes, and (b) biotic physiological processes. In their review, Reddy *et al.* (1999) concluded that P retention by streams is dominated by physical processes such as flow velocity, discharge, and water depth. Abiotic processes controlling P retention in streams are dominated by sediment sorption reactions. However, biological uptake can account for a large proportion of DP in streams. Phosphorus assimilation rate in streams, measured as a function of time, is related to the length of stream and flow velocity (Reddy *et al.*, 1999).

4.1.1 Sediment Processes

Sediments within the fluvial system are either derived from overland flow or stream bank erosion. Sediments derived from stream banks will largely consist of subsoil that is relatively depleted of P. The proportion of sediment within the fluvial system will depend on the age of the channel network. For example, in areas with recent gully formation (channel rejuvenation), subsoil material will dominate (Olley *et al.*, 1993). Furthermore, P derived from subsoil materials in these systems will be less readily available to be released to water and the subsoils will likely represent a net sink for P (McDowell and Sharpley, 2001d). As a result of the erosion of subsoils, which are often dominated by silt-sized particles, the

predominant form of P transport in these fluvial systems is PP, whereas in sandy catchments most P is transported in dissolved form (Baldwin *et al.*, 2002).

In fluvial systems with good hydraulic mixing (such as shallow flowing streams), the availability of P in sediments can be estimated by the equilibrium P concentration at zero net sorption or desorption (EPC_0). Under conditions of low flow (*i.e.*, base flow) a state of quasi-equilibrium exists, whereby the kinetics of P release or uptake are practically complete by the time a volume of water flows by. For these situations, the EPC_0 will influence P concentration in solution, whereby P will desorb from sediments if the concentration of P in stream flow is less than the sediment's EPC_0 , or conversely, P in stream flow will adsorb to sediments if the concentration is greater than the EPC_0 of the sediment (Kunishi *et al.*, 1972). Significant input or depletion of P from the sediment can also alter the EPC_0 , thereby affecting stream P concentrations.

4.1.2 Biotic Processes

Strong downstream DRP gradients or losses with time have been recorded in artificial streams lined with inert substrates, but containing significant growths of periphytic algae (*e.g.*, Horner *et al.*, 1990). This indicates that algal uptake and growth can also have an important influence on stream DRP concentrations. Variability in sediment EPC_0 and uptake by periphyton along a stream or river channel is characteristically high, reflecting physical hydraulic processes, the management of land adjacent to the stream and the form of P occurring in the sediment. This leads to the concept of P-spiralling or the distance travelled downstream by one P molecule as it completes one cycle of uptake and transformations from dissolved to organic forms and back into flow (Newbold *et al.*, 1981; Ellwood *et al.*, 1983). The first definitive measurement of P-spiralling length was reported for a first-order woodland stream in Tennessee (USA) using $^{32}PO_4$ as the tracer. The P spiralling length was 190 m, 165 m of which was in the water while the remainder was in fine particulate organic matter. Other North American workers found that spiralling length ranged from 23 m in November to 99 m in August when the concentration of coarse particulate material was less and P was moving largely in dissolved form (Mulholland *et al.*, 1985). However, during storms the distance travelled by P in particulate material can increase by one or two orders of magnitude above typical distances of 1–3 m (Melack, 1995). Differences in geology can have a profound effect on P-spiralling length. Munn and Meyer (1990) found that a stream with granite bedrock had a spiralling length of 85 m, whilst in a stream with P-rich volcanic bedrock the spiralling length was 687 m.

Factors influencing the EPC_0 and P-spiralling are notoriously variable along a stream reach. For example, where in-stream geomorphic processes cause size sorting, or where sediments are enriched with P due to local contributions of P-rich overland flow, sediments can represent a significant source of P to stream flow, even when inputs from runoff have ceased. For instance, sediment EPC_0 that is greater than the concentration of P passing in stream flow will result in net P desorption until the P concentration in stream flow is the same as the sediment EPC_0 . The reservoir of P able to contribute to the EPC_0 can be approximated from measures of sorbed P in the sediment, and, in-turn, the effectiveness of sediment as a P sink can be estimated from its P sorption capacity or sorption maximum.

4.1.3 Other Processes Affecting P Concentrations in Stream Waters

A number of other processes may also induce changes in sediment P release to stream waters. These include a rise in stream water pH, extraction of P from dead periphyton cells and

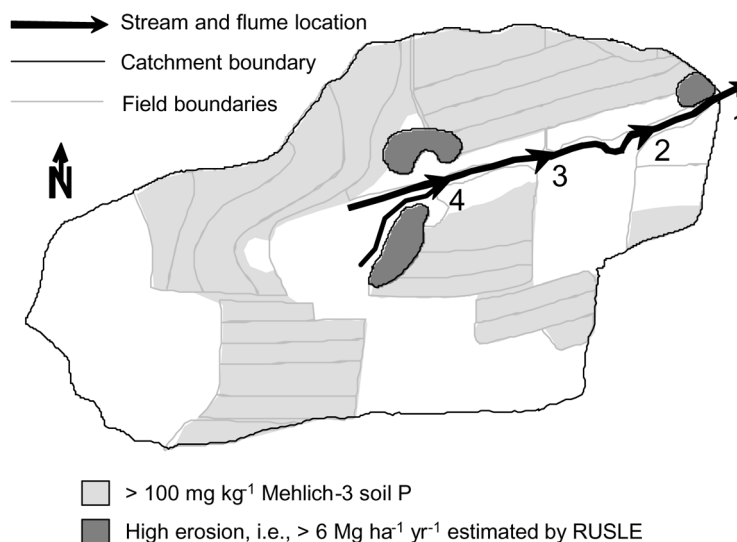
mineralization, the hydrolysis of organic P species, and changes in sediment crystallinity and subsequent P release under anaerobic conditions ([Stream Solute Workshop, 1990](#); [Fox, 1993](#); [Baldwin *et al.*, 2002](#)). For example, the potential of sediments exposed to wetting–drying cycles cause a change in Fe-oxide crystallinity making P more recalcitrant to release; the same is observed with periodically re-wetted stream bank sediments, compared to stream bed sediments ([Qiu and McComb, 1995](#); [Baldwin, 1996](#); [McDowell and Sharpley, 2001d](#)). Furthermore, facultative anaerobic bacteria may uptake P and store it as polyphosphates (up to 20% of their dry weight) and release it under anaerobic conditions ([Davelaar, 1993](#); [Gächter and Meyer, 1993](#)). Other studies have shown bacteria can account for a considerable proportion of total sediment P uptake and release (30–40%, [Khoshmanesh *et al.*, 1999](#); [McDowell and Sharpley, 2003c](#)). However, the potential for this P to interact with the overlying water column is dependant upon the mass transfer of P from the sediment water via physical disturbance during high flow events. Some molecular diffusion is likely to occur under low flow conditions. However, the contribution of this mechanism to P release is probably small ([Baldwin *et al.*, 2002](#)). Invariably, under conditions of rising flow a flush of P into stream flow may occur that is independent of the sediment EPC_0 but is subject to the kinetics of P release.

Following the initial flush of P from P-rich pore water, under most flow conditions the speed of P release is inversely related to particle size and P sorption capacity (*i.e.*, clay-sized particles sorb P more readily than coarser-sized particles [Stone and Murdoch, 1989](#)). Thus, hydrologic processes that control sediment particle size distribution have important implications to P delivery and fate in river systems. For instance, in a recent review of land use and sediment yield, [Walling \(1999\)](#) indicated that fluvial systems have considerable capacity to buffer changes in sediment delivery, whereby rivers with a low sediment delivery ratio will exhibit a large buffering capacity and *vice versa*.

Illustrating the effects of in-channel processes on P transport in relatively small catchments, [McDowell *et al.* \(2001b\)](#) described the mechanisms controlling P release from soil and stream sediments in relation to storm and base flow at four flumes along the channel of a 40 ha, second order agricultural catchment. Base flow DRP concentrations were greater at the catchment outflow ($42 \mu\text{g L}^{-1}$ at flume 1) than at the upper-most flume ($28 \mu\text{g L}^{-1}$ at flume 4), while the inverse occurred during storm flow ($304 \mu\text{g L}^{-1}$ at flume 4 and $128 \mu\text{g L}^{-1}$ at flume 1) (Fig. 5). Similar trends in TP concentration were observed. However, it is questionable whether short-term pulses in DRP have much ecological impact in streams ([Humphrey and Stevenson, 1992](#)).

During storm flow, in-channel decreases in P concentration were indicative of dilution of P originating from a CSA above the upper-most flume (flume 4), where an area of high soil P intersected an area of high erosion and overland flow potential (Fig. 5). During base flow, the increase in P concentrations downstream was clearly controlled by channel sediments, such that the P sorption maximum of the upper-most flume (flume 4) sediment (532 mg kg^{-1}) was far greater than the outlet flume (flume 1) sediment (227 mg kg^{-1}) (Fig. 5). Paralleling these trends, the EPC_0 of sediment at flume 1 was greater than at flume 4 (34 to $4 \mu\text{g L}^{-1}$). Sediment EPC_0 trends were highly correlated to base flow dissolved P concentrations ($28 \mu\text{g L}^{-1}$ at flume 4 and $42 \mu\text{g L}^{-1}$ at flume 1; Fig. 5).

In a much larger catchment, [McDowell *et al.* \(2002a\)](#) examined the processes controlling sediment P release to the Winooski River, VT, the largest tributary to Lake Champlain (Fig. 6). Iron-oxide strip P (algal-available P) of the river sediments adjacent to agricultural land (3.6 mg kg^{-1}) was significantly greater ($P < 0.05$) than that of sediments adjacent to forested land (2.4 mg kg^{-1}). This was especially the case in the tributary adjacent to a dairy farm where overland flow was directly contributing to the enrichment of fluvial sediment (Fig. 6). Notably, impoundment (731 mg kg^{-1}) and reservoir sediments (803 mg kg^{-1}) had greater TP



Flume	Dissolved P		Stream sediment	
	Stormflow	Baseflow	P sorption max	EPC ₀
	$\mu\text{g L}^{-1}$		mg kg^{-1}	$\mu\text{g L}^{-1}$
1	128	42	227	34
2	174	36	295	13
3	202	37	330	4
4	304	28	532	4

FIGURE 5 The distribution of Mehlich-3 soil P ($>100\text{ mg kg}^{-1}$), erosion ($>6\text{ Mg ha}^{-1}\text{ yr}^{-1}$) and DP concentration in stream and baseflow (mean of 1997–2000 data) and P sorption properties of channel sediment at four flumes in FD-36 (adapted from McDowell *et al.*, 2001b).

concentrations than river sediments (462 mg kg^{-1}). This was attributed to more fines ($<63\mu\text{m}$) in impoundments and reservoirs than in river sediments. Furthermore, turbulence at the confluence of two tributaries resulted in the shift of the particle size towards coarser particles. This also increased the release rate of these sediments and their sensitivity to incoming P sources (Fig. 6). Consequently, impoundment and reservoir sediments had lower abilities to release P to solution in the short-term, thereby acting as sinks for sediments rich in P, provided the system was maintained in an aerobic state. However, if sediments, like soils, become anaerobic through oxygen depletion and stratification, P release increases via iron and sulphate reducing bacteria (Boström *et al.*, 1988; Lovley *et al.*, 1991).

In general, the impoundment of rivers causes a decrease in the downstream transport of sediment and P by increasing retention within the impoundment. As of 1971, about 4000 km^3 of water was impounded world-wide by dams, with more rivers being modified as pressures increase on water resources for agricultural, urban and industrial uses (Petts, 1984). On average, sediment yield is decreased by three quarters by impoundments, and nutrient enrichment of these waters can commonly occur due to the retention and release

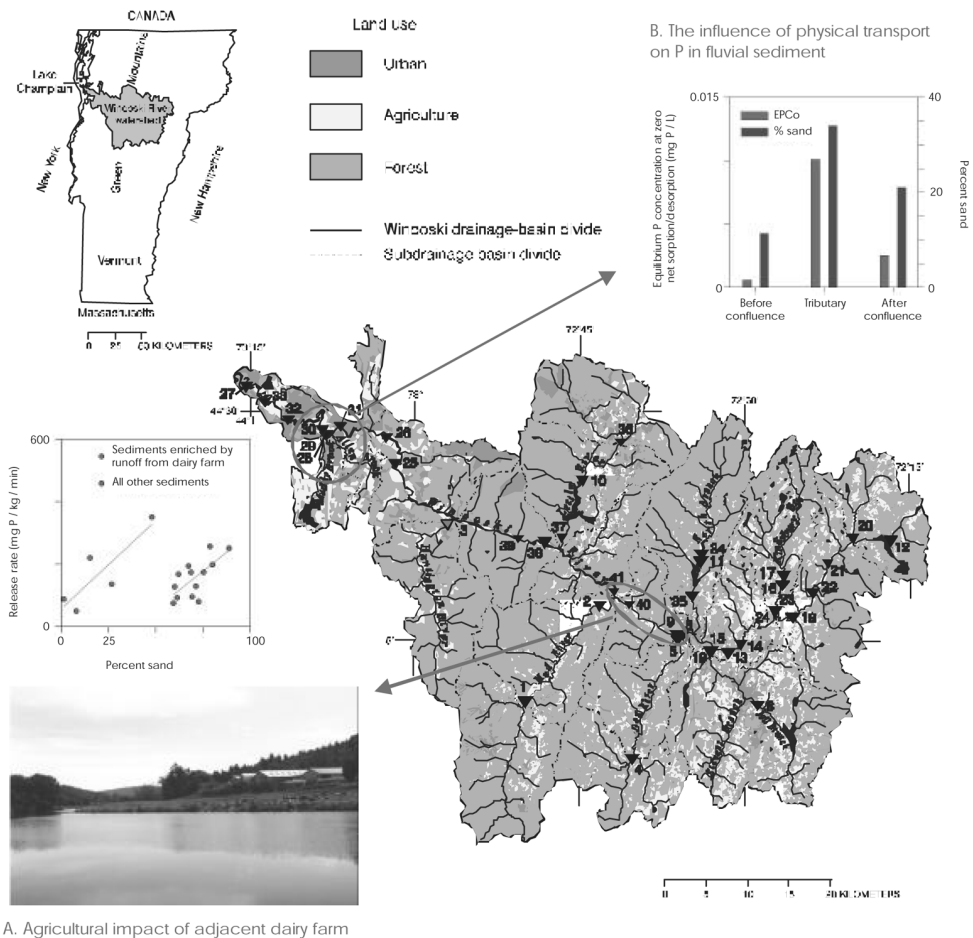


FIGURE 6 The location, distribution and impact of land use and physical transport processes on P in fluvial sediments within the Winooski River catchment, VT (adapted from McDowell *et al.*, 2002a).

of P from P-rich fines. Downstream, periodic discharge can increase the sediment and P load via stream-bank erosion and release light particulate material and P from biological production of organic matter. In contrast, the much greater flushing rates of natural ecosystems (rivers and lakes) enables a much greater potential for sediment and P distribution to equilibrate over the entire stream length (Higgins and Kim, 1981). The influence of flushing rates and P inputs to rivers and lakes will be discussed in the following section.

The results of the research on the Winooski River catchment demonstrate that there is a strong influence of fluvial hydraulics on the properties of sediment within river systems. The input and delivery of fine sediment enriched with P was influenced by adjacent land use. The fluvial sediment, particularly at the outflow of the river into Lake Champlain, represents a P storage pool, which has a long-term potential to release a large amount of P to overlying waters. In the short-term, however, river flow and the physical properties of the sediments will influence the amount of sediment P leaving the catchment. Thus, in connecting sediment P loss from the landscape to channel processes, variability in flow, local sources of P, and sediment properties must be taken into account, particularly near the point of impact. Because of these complexities, channel processes and changes in P forms and

loads are not currently simulated in many models that estimate P loss from catchments ([Hanrahan *et al.*, 2001](#)).

4.2 The Impact of P Transport to Streams and Rivers

The extent to which P transport to streams and rivers stimulates the productivity and growth of benthic algae (commonly called periphyton), phytoplankton, or macrophytes depends on whether productivity is limited by P (productivity in some streams is limited by N; *e.g.*, [Chessman *et al.*, 1992](#); [Francoeur *et al.*, 1999](#)), or other limiters such as light, temperature, flood disturbances and grazing invertebrates ([Biggs, 1996](#)). Periphyton responses to increased P load are usually evident as blooms of filamentous algae (*e.g.*, *Cladophora*, also known as ‘blanket weed’) in streams that have prolonged periods of stable flow, are unshaded, and have a gravel/cobble bed. In such situations, periphyton can respond to even minute increases in P concentrations because the laminar boundary layer surrounding the cells and within the mat is eroded by moving water. This means that boundary layer diffusion limitation is minimized and the rate limiting step is biochemically mediated uptake kinetics. Notwithstanding this, in natural streams broad positive correlations between P concentrations (as DRP and TP) and periphyton biomass are still observed (*e.g.*, [Biggs and Close, 1989](#); [Biggs, 1995](#); [Dodds *et al.*, 1997](#); [Biggs, 2000a](#)). This has also been supported by a number of investigations in experimental streams ([Bothwell, 1989](#)). Based on his experiments with variable DP loading, [Bothwell \(1989\)](#) suggested that this anomaly between cellular growth rate limitation at extremely low concentrations of P and high biomass only occurring at high P concentrations, was due to nutrient limitation at two different scales. Bothwell suggested that there was a shift from cellular controlled growth rate limitation of mats to community controlled growth rate as mats developed:

- Growth of cells at the surface of the mat are unlikely to be P-limited at sub-parts per billion DP concentrations, but as the thickness of the mat develops, diffusion of P molecules to the base of the mat is restricted (either through restricted water movement within the mat or due to uptake by cells nearer the upper surface). This results in P-starvation of the underlying layers and eventual decay leading to spontaneous sloughing of mats.
- With higher P concentrations in the overlying water, rates of molecular diffusion to the underlying layers increase, resulting in the accumulation of much higher biomass before sloughing occurs (see [Borchardt, 1996](#), for a more detailed explanation) and the maintenance of high biomass for much longer periods in the stream ([Biggs, 2000a](#)).

Changes in the type of periphyton in shallow streams and phytoplankton in deep, slow flowing rivers also occur as P loading increases. These changes are generally toward more undesirable types of algae. For example, communities in oligotrophic streams (*i.e.* streams with low nutrient loadings) tend to be dominated by diatoms and filamentous algae, whereas in eutrophic streams (*i.e.* streams with high nutrient loadings) communities tend to be dominated by filamentous green algae capable of forming very long filaments ([Chétalet *et al.*, 1999](#); [Biggs, 2000a](#)).

Shifts in the N:P ratio can also have important effects on the community composition of periphyton in shallow streams. For example, [Biggs and Smith \(2002\)](#) reported that communities with a N:P ratio of less than 10 (*i.e.* low in N relative to P) had a very high proportion of nitrogen fixing Cyanobacteria in the community, whereas these organisms were rare in streams with a high N:P ratio (*i.e.* low P relative to N). Also, the diversity of algae was much greater at low N:P ratios, where nitrogen-fixing taxa were more abundant ([Biggs and Smith, 2002](#)). In slower flowing oligotrophic rivers, phytoplankton tends to be dominated

by a low biomass of diatoms, whereas these communities are often replaced by a high biomass of Cyanobacteria in eutrophic rivers.

Macrophyte responses to increased P loads are more difficult to predict. In the first instance, macrophytes need stable flows and sunlight to proliferate (Riis and Biggs, in press). However, whether they respond to increases in P varies depending on free-stream water vs. sediment pore water P concentrations and the ability of the particular macrophyte species to utilize root vs. leaf uptake paths. For example, [Chambers *et al.* \(1989\)](#) found that biomass, shoot density and tissue nutrient concentrations of the dominant species (*Potamogeton crispus*) in a nutrient rich river were largely determined by sediment and not water P availability. Biggs and Malthus (1982) reported an extremely high biomass for *Lagarosiphon major* from a flowing water section of a hydro-power impoundment in New Zealand where water P concentrations were $<5 \mu\text{g L}^{-1}$, but the plants were rooted in fine schist sediments with a high natural P content. Indeed, [Barko *et al.* \(1991\)](#) have concluded that in most streams and rivers root uptake is the main pathway for P uptake. This indicates that controlling surface water P loads may only be effective in preventing excessive growth of macrophytes if there is no geological source of P in the sediment or the sediment pools are likely to become depleted following a persistent decrease in incoming soluble or PP.

Problems associated with excess periphyton and macrophyte biomass accumulation in streams become prominent during summer low flows and tend to be sporadic. Some common stream values that may be compromised by such growths include contact recreation, degradation of water quality (reductions in dissolved oxygen and increases in pH), clogging of water intake structures, creating taste and odour problems, fouling/suffocation of fish spawning and rearing sites and degradation of benthic biodiversity (Biggs, 2000b). The effects on water quality and ecosystem degradation are only moderately well quantified, and a number of cause-effect assumptions need careful testing. Shifts in benthic community structure are clearly apparent across a range of regimes with streams having low P-loads and associated oligotrophic conditions of low periphyton production (peak chlorophyll *a* biomass $<50 \text{ mg m}^{-2}$) usually being dominated by a diverse, healthy invertebrate fauna, whereas streams having high P-loads and eutrophic conditions (peak chlorophyll *a* biomass $<200 \text{ mg m}^{-2}$) are often dominated by snails, worms and midges which are indicative of an unhealthy ecosystem (Biggs, 2000b). These latter communities have only limited value as food for many fish species such as brown trout (*Salmo trutta*).

One of the critical issues with understanding and modelling the effects of increased P loadings on stream ecosystems is how we can best measure instream concentrations of P in an ecologically meaningful way. The traditional approach in lakes has been to use TP because of the rapid uptake and storage of P by phytoplankton followed by partial re-mineralization as phytoplankton decay ([Welch, 1992](#); [Chapra, 1997](#)). However, local deposition and decay of periphyton in streams may only occur in very slow flowing streams and pools and there can be high flow events to flush the deposited organic matter from the streams (Table I). The use of mat P concentrations (*i.e.* TP of the periphyton mat normalized to mat biomass; *e.g.*, Biggs, 1995) is appealing because of the potentially close relationship between concentrations and algal growth rates (*e.g.*, [Auer and Canale, 1982](#)). However, there are difficulties in relating these back to nutrient supply regimes due to the potential problem of bias caused by non-phototrophic sediments, and analyses are expensive (Table I).

The variance explained by models based on dissolved, particulate and mat nutrient concentrations varies widely. Models of dissolved and mat-P concentrations explain the most variance in periphyton biomass, particularly when combined with time available for communities to grow between bed cleansing floods ([Biggs, 2000a](#)). However, while a useful variable in ecological studies, there are still many questions over the use of mat nutrient concentrations for eutrophication management, as noted above, and much further research is

needed before the link between dissolved nutrient supply rates and mat nutrient concentrations can be quantified. It would appear for the present that the dissolved nutrient-biomass models for maximum biomass might be the most useful tools for managing the eutrophication of temperate gravel/cobble bed streams (Biggs, 2000a). In these models, the DRP status of the streams is determined following monthly monitoring for at least a year so that short-term variability associated with uptake processes and hydrological events can be averaged.

4.3 The Impact of P Transport to Lakes and Reservoirs

The usual response of a lake to a large sudden input of P is also a rapid increase in algal productivity (Vollenweider, 1968; Correl, 1998). However, unless P inputs are sustained in a continuous or pulsed manner, productivity will tail-off after a few weeks. In contrast, to restore an eutrophic lake requires a decrease of P inputs to levels less than P removal within the lake.

The rate at which algal productivity increases or decreases within a lake, differs greatly, with variations in geographic location, climate, water residence times, and surface area

TABLE I Summary of General Technical and Practical Advantages and Disadvantages of Different Forms of Nutrients for Use in Benthic Algal Biomass – Nutrient Regression Models in Streams/Rivers.

DRP

Advantages: A direct measure of the bio-available form of P and therefore mechanistically sound. Point source effluent effects can be assessed directly. Temporal variability moderate–low relative to other nutrients (*e.g.*, CV ~20–110% for DRP; Biggs and Close, 1989; Biggs, 1995). Analyses are relatively quick and cheap.

Disadvantages: Single measurements in time are a poor indicator of P supply regime because of effects of algal uptake and geochemical transformations in the stream. Contribution of subsurface springs/seeps is difficult to account for. About a year of monthly measurements is best to obtain a reliable estimate of mean P supply concentrations (fewer samples may be possible where flow regimes are relatively stable). Phosphorus bound to organic matter *might* become available somewhere downstream and therefore the projected DRP supply could underestimate the actual supply. Low levels of detection required.

Total P

Advantages: Incorporates all forms of P thus giving a measure of the overall, potential, nutrient supply. Supply from subsurface inflows/ groundwater is incorporated in measure. Extensive data is available on landuse – TP loadings. A widely used variable in lake eutrophication management might be useful for comparing lentic *vs.* lotic enrichment processes (*e.g.*, Dodds *et al.*, 1998).

Disadvantages: Correlated with chlorophyll in water column. Thus, a proportion of TP in streams is probably derived from suspended benthic algae, which means that there might be some circular reasoning in its application. Therefore the approach requires the following assumptions: the particulates/algae will eventually deposit in downstream reaches; a proportion of P in these particulates will become available to algae; and the proportion of bio-available P will be similar among streams, and over time, regardless of the type of particulates (*e.g.*, organic *vs.* inorganic). Analyses require a digestion step which makes processing more expensive. Frequent monitoring is required to get good estimates of mean TP concentrations (*e.g.*, weekly for a year) because of moderate–high temporal variability (*e.g.*, CV ~30–500% for TP; Biggs and Close, 1989).

Mat P

Advantages: A direct measure of the P status of the periphyton and can be related to specific growth rates through mechanistic models such as the Droop model (*e.g.*, Auer and Canale, 1982). Integrates the history of P supply, including mineralized nutrients from deposited organics and subsurface supply from seeps and groundwater.

Disadvantages: Difficult to relate back to supply concentrations of dissolved or TP (therefore, difficult to use as a basis for managing nutrient loads). Results likely to be biased to varying degrees by the amount and type of silt deposited in the mat. The influence of silts will increase as the biomass:silt ratio decreases. Analysis requires a digestion step and a measurement of organic biomass which increases costs. Moderate temporal variability moderate–high sampling frequency is required (CV of mat %P commonly ~90–200%; Biggs, 1995).

and depth of water body. In addition, P loading of water bodies is further influenced by the ratio of drainage basin area to lake area. In general, inputs of P increase with the proportion of the surrounding land to water ratio. For example, the ratio of catchment drainage area and Bay water volume in the Chesapeake Bay ($2410 \text{ km}^2 \text{ km}^{-3}$) is nearly an order of magnitude greater than any other lake or bay in the world (next is the Gulf of Finland at $380 \text{ km}^2 \text{ km}^{-3}$). Hence, the potential for deleterious effects via P inputs to the Chesapeake Bay is very high.

By decreasing P loads to levels that exist in lakes with good water quality, the recovery of lakes with poor water quality and mean size and water flushing time can take between 2 and 10 years (Wetzel, 2001). In general, the residence time, or the inverse, flow-through rate, of P is related to sediment removal and phytoplanktonic production (Vollenweider, 1990). Provided the duration of P loading is short, lakes that were oligotrophic will respond slowly to increases in P loads and decrease rapidly back to oligotrophic conditions if loadings are decreased. However, if sustained P loads have sediments enriched with P, and the lake has been eutrophic for some time, it will take much longer to recover oligotrophic status.

In lakes and reservoirs with anoxic hypolimnia, the magnitude of internal P loading can be high as P is solubilized under reducing conditions. The magnitude of P release can be estimated from the mean rate of P release from the sediments, the surface area of anoxic sediments and the period of anoxia. Shallow lakes with active sediment deposition and a high internal loading of P can be especially susceptible and take a long time to recover oligotrophic status. Coupled with this, recent evidence has shown that the potential for P input derived from internal loading can be greater from P-enriched stream bank sediments transported to the lake during a storm event compared to bed load typically with less P (McDowell and Sharpley, 2001d).

It is fortunate that in most lakes the internal contribution of P from sediments is a small proportion of the total inputs. Hence, restriction of P loads into a lake can be an effective way of decreasing aquatic biomass production. Indeed, recent evidence has shown that decreasing P fertilizer sales and more efficient soil management over the past 20 years has decreased P losses into the north-western Ohio River, which drains into Lake Erie (Calhoun *et al.*, 2002).

Anthropogenic inputs of P are significantly influenced by the alteration of flow regimes such as those found in artificial reservoirs compared to natural lakes. Straškraba (1996) noted that the P retention capacity (*i.e.* the proportion of P load not lost from the water body via its outflow) is correlated to water retention time, defined as the time (days) taken to fill the water body from empty (Fig. 7). Whereas, water retention times of lakes are typically between 1 and 7 years, reservoirs are highly variable, but commonly <100 days. Consequently, over long periods there is little difference between P retention capacities. However, a distinct difference is evident for times of less than 300 days and especially for retention times <100 days. This is related to the quicker sedimentation of PP and removal of outflow water from deep within the stratified water of reservoirs *vs.* the predominantly slower sedimentation of particles and the removal of surface outflows in lakes. Thus, surface water rich in P in lakes is less likely to be retained than in reservoirs (Fig. 7).

In addition to P inputs and other nutrients such as N, the interaction of other factors can influence the effect of P inputs. Models integrating these effects are numerous and for a discussion reviews such as those by Cloern (2001) should be consulted. Briefly, the factors influencing P impact the most, excluding P sources such as soil, sediment, fertilizers and manures already discussed, are (Newton and Jarrell, 1999):

- *Water residence time.* A short residence time (*e.g.*, <7 days) will not allow phytoplankton to increase before being swept away.

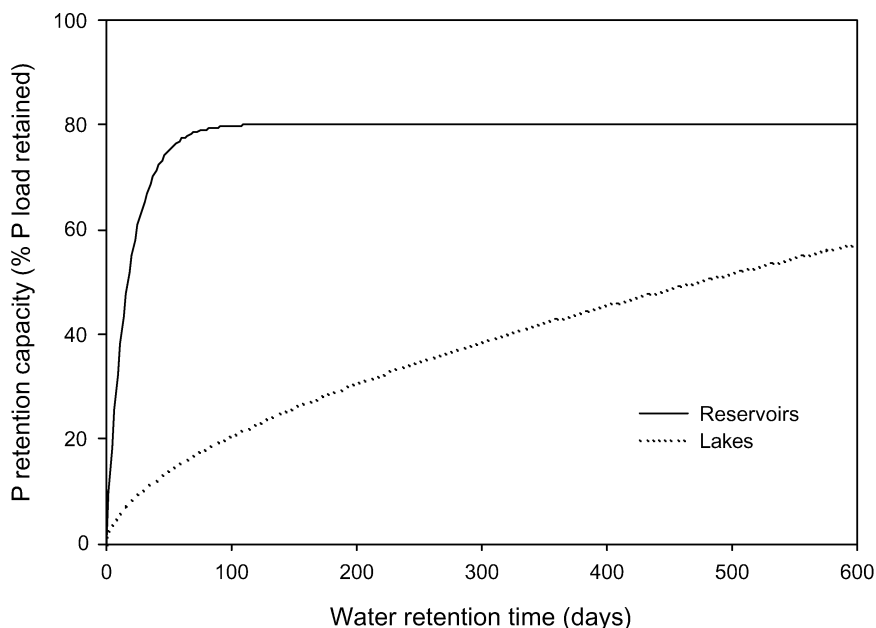


FIGURE 7 The relationship between the P retention capacity of reservoirs and lakes and their water retention times (adapted from Straškraba, 1996).

- *Temperature.* Algae grow faster at higher temperatures, hence the proliferation of aquatic plants in summer, especially blue-green algae which are tolerant of higher temperatures than other algae. Decreasing temperature also increases dissolved oxygen concentrations and fish survival. Increases in temperature arise from either direct sunlight or inputs of warm water (e.g., irrigation water), and measures such as riparian planting for shade can decrease temperature in narrow streams.
- *Light.* Obviously light is required for plant growth, provided sufficient nutrients and warm temperatures are available. Riparian vegetation, suspended sediment and dissolved (e.g., tannins) or particulate organics decrease light penetration and availability.
- *Depth.* This can affect sensitivity to nutrient inputs by increasing the distance required for light to travel to the bottom of the water body, thereby increasing the ability of particles in the water to scatter, reflect and absorb light. The point at which light is too low to support plant growth is termed the compensation depth. In shallow waters (e.g., <2 m depth), decreasing suspended sediment inputs will increase light transmission and, provided the bottom sediments are sufficiently enriched with P, then periphyton and macrophytes may grow even though overlying water is poor in P.
- *Stratification.* Caused by differences in density in response to temperature (water is densest at 4 °C) which results in layers of water within deep water bodies that do not mix. In summer, warm water layers (epilimnion) float on top of denser cold layers (hypolimnion), while the reverse may happen in winter whereby cold water near 4 °C is at the bottom but even colder water lies on top. This thermal stratification is broken down either by wind turbulence or by temperature changes causing surface layers to cool or warm compared to lower layers thus creating a turnover of water layers. Because the hypolimnion is isolated from the atmosphere, dissolved oxygen can be depleted by microbes acting on organic matter causing anoxic conditions and P to be released from the sediment. When turnover occurs, P becomes available to the surface water and if the right conditions occur then rapid algal growth will result. Conversely, reduced Fe can become oxidized and sequester P.

- *Ground water inputs.* Although inputs of P from ground water are low, generally $<0.02 \text{ mg L}^{-1}$ (Wetzel, 2001), inputs of 0.1 mg L^{-1} or above can occur due to geology or sometimes management (Breeuwsma and Silva, 1992; Newton and Jarrell, 1999). Catchment groundwater enters the stream system via the hyporheic zone, an area where hydrologic and biogeochemical processes are active and therefore are likely to impact on the quality of receiving waters. Solute transport through this zone has been found to be orders of magnitude slower than movement in stream channels (m day^{-1} vs. m min^{-1} ; Triska *et al.*, 1993), increasing residence times and biogeochemical P transformation.

5 CONNECTING MANAGEMENT OF P LOSS WITH SURFACE WATER QUALITY

5.1 Management to Decrease P Loss from the Landscape

Effective nutrient management ultimately aims to balance farm P inputs with off-takes in crop or animal products, while efficient mitigation of P loss from the landscape involves placing P away from CSAs likely to lose much P. There are numerous options available to mitigate the potential for P loss. Although most are orientated at decreasing the P available to flow, some can also mitigate P if in a hydrologically active area.

Cultivation immediately after application can decrease P losses if erosion is minimized. Periodic tillage of the soil may also decrease P loss by redistributing high-P topsoil throughout the root zone and disrupting subsurface P loss pathways (*i.e.* macropores; Djodjic *et al.*, 2002). Applications of manure or fertilizer during drier periods to avoid precipitation or snowmelt will further decrease the potential for P loss in overland and subsurface flow by increasing the contact time (and uptake) with soil and crop.

Phosphorus inputs to receiving waters from point sources such as dairy shed wastewaters are often controlled by treating wastewaters in a two-staged waste stabilization pond (WSP) system before applying to agricultural land or discharged into receiving waters. Such WSP systems, which consists of an anaerobic pond and then an aerobic pond, normally removes $<40\text{--}65\%$ of the TP in dairy wastewaters (Nguyen and Davies-Colley, 1998). To enhance P removal, wetlands (Cooke *et al.*, 1992; Tanner *et al.*, 1995; 1998) and some features of the advanced integrated wastewater pond systems (Green *et al.*, 1995) such as high rate algal ponds (HRAP), maturation ponds and algae settling ponds are often incorporated into the WSP system. The high pH (9–11) in HRAP, resulting from algal photosynthesis (Nurdogan and Oswald, 1995; Green *et al.*, 1996) enhances P removal via chemical precipitation with calcium in pond waters.

The presence of cover crops and residues can help decrease P loss by decreasing the kinetic energy of raindrop impact and thus erosion and P in overland flow. Equally, anything that keeps surface roughness high or intercepts overland flow, which encourages infiltration and sediment retention, can be effective. Such measures include riparian zones, buffer strips, terracing, cover crops, contour tillage and impoundments or small reservoirs. However, these measures are better at stopping particulate than DP transport. Gitau *et al.* (2001) recently summarized the range in efficiencies for best management practices to mitigate P loss (Fig. 8).

Other remedial measures include manure and soil treatment and amendment to decrease P solubility and potential release to overland flow; feeding animals no more P than they actually need; use of soil testing to guide future P application (particularly as manure); and redistribution of manure within and among farms (Sharpley, 2000). Further decreases can be achieved by identifying CSAs and targeting conservation measures such as adding high P sorbing materials like coal combustion by-products (Stout *et al.*, 2000).

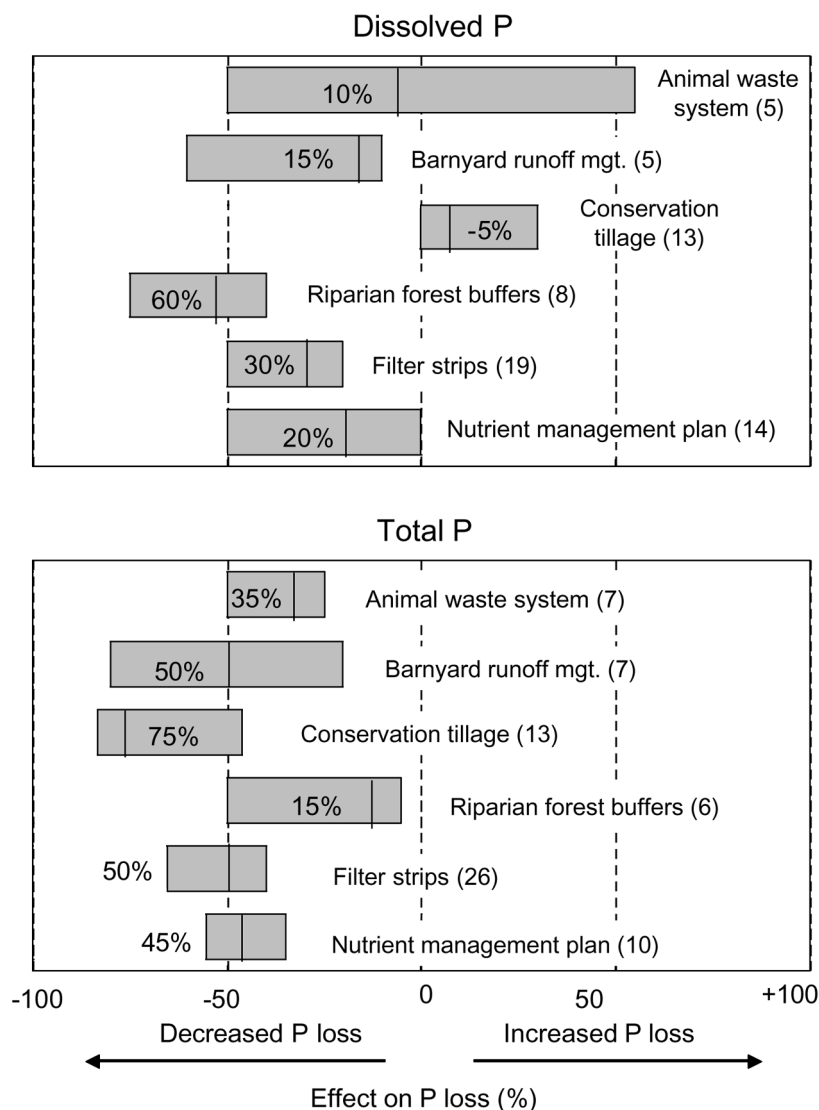


FIGURE 8 Range in effectiveness of various BMPs in reducing dissolved and TP loss in overland flow based on published information (adapted from Gitau *et al.*, 2001; Sharpley *et al.*, 2002). Number of studies is in parenthesis. The solid line through each box represents the median value of effectiveness.

Due to the time and expense involved in field assessment of management impacts on P loss, models often represent a more efficient and feasible means of evaluating management alternatives. Indeed, numerous process-based models have been developed to simulate the fate of P in soil and its transport to surface waters (*e.g.*, AGNPS, Young *et al.*, 1989; EPIC, Sharpley and Williams, 1990). In their most comprehensive form, such models integrate information over a large scale, helping to define catchment-scale processes relevant to P transport, highlighting appropriate best management practices (BMPs) and identifying CSAs where BMPs are most likely to affect catchment-scale P losses. Perhaps one of the simplest models is the ‘P index’, which utilizes data on a field by field basis to determine a degree of risk of P loss and an appropriate mitigation strategy. The P index has or is being used in many

counties for mitigating P at the source due to its ease of use, transparency and good uptake by the end-user. For a comprehensive review on modeling P loss, including the 'P index', the reader is directed to [Sharpley *et al.* \(2002\)](#).

5.2 Edge of Field Management

5.2.1 Wetlands

In many natural ecosystems much P is prevented from reaching downstream areas by wetlands. However, P removal by wetlands generally declines after a period of years or decades depending on loading rates, hydraulic retention time, wastewater characteristics, wetland substratum and wetland areas ([Reddy *et al.*, 1995](#); [Fennessy and Cronk, 1997](#)). The P removal capacity of wetlands depends on the physical–chemical–microbiological processes that influence organic and inorganic P uptake in wetland sediments. These include, sorption–precipitation of DRP by wetland substrate (*e.g.*, soil, gravel, minerals and peat), sedimentation–deposition of PP, and P assimilation by microbial and plant biomass ([Reddy *et al.*, 1999](#)). Microbial P represents a dynamic P pool of limited size while plant P, unless removed by harvesting, is released back to wetlands via decomposition of plant litter ([Johnston, 1991](#)). Thus, the most important P removal processes are sorption and sedimentation ([Cooke *et al.*, 1992](#); [Reddy *et al.*, 1995](#); [Nguyen, 2000a](#)). Wetland redox potential can influence P dynamics through interaction with Fe compounds; P is likely to be released under anaerobic conditions via the reduction of crystalline ferric (Fe^{3+}) P complexes. However, anaerobic conditions can also enhance P sorption since highly-dispersed gel-like Fe^{2+} oxides or hydroxides have a greater surface area for P sorption than crystalline Fe^{3+} forms ([Patrick and Khalid, 1974](#); [Nguyen *et al.*, 1997](#)).

In addition to natural wetlands, constructed wetlands can be built with soil, gravel, zeolite, limestone and slag ([Hylander and Siman, 2001](#)). A problem for constructed wetlands that treat highly organic wastewaters (*e.g.*, dairy wastewaters) is the clogging of pore space by accumulated refractory (humic acid, fulvic acid and humin) organic solids ([Nguyen, 2000b](#)). [Tanner *et al.* \(1998\)](#) reported a substantial decline in P removal efficiency in constructed wetlands after 3 years of treating dairy wastewater, caused by the accumulation of organic solids and subsequent wetland short-circuiting.

5.2.2 Buffer Strips – Riparian Areas

Phosphorus loss from the landscape can be decreased by establishing edge-of-field riparian buffer strips and riparian wetlands to intercept water runoff from agricultural lands ([Flenniken *et al.*, 2001](#)). In a study by [Smith \(1989\)](#), riparian pasture retirement effects on P in overland flow from two steep hillslopes were examined over 22 months by comparing DRP concentrations in overland flow at retired and grazed riparian sites. The event flow-weighted PP and DRP loads were 80% and 55% less, respectively, at the retired site, indicating the effectiveness of riparian buffer strips in decreasing P loss via overland flow to waterways in the short term. This effectiveness depends on the width of buffer strips, vegetation density, soil characteristics (*e.g.*, water infiltration rate and P sorption capacity), vegetation type, placement within the landscape and slope ([Fennessy and Cronk, 1997](#)). Efficiency of P retention increases as riparian buffers become wider ([Osborne and Kovacic, 1993](#); [Fennessy and Cronk, 1997](#)). [Quinn *et al.* \(1993\)](#) estimated a dense grass buffer strip of 10–20 m wide (5–10% hillslope length) was sufficient to remove 60–75% of P from overland flow in areas of medium ($15\text{--}20^\circ$) slope and low (4 mm hr^{-1}) drainage rate.

Primarily, riparian areas work by promoting sedimentation and improving soil structure and infiltration (Lyons *et al.*, 2000). Grassy riparian vegetation is better than woody riparian vegetation in decreasing P loss by decreasing the potential for flow in channels and enhancing entrapment in sheet flow (Lyons *et al.*, 2000). Sedimentation of PP depends on soil particle size. Sand or silt-sized particles are readily deposited within riparian zones while P bound to clay particles are not (Uusi-Kampala *et al.*, 1997). Retention of DP is less than PP and decreases with time as P sorption sites become occupied (Reddy *et al.*, 1995; Uusi-Kampala *et al.*, 1997). Cooper *et al.* (1995) noted a riparian pasture that had been set-aside for 12 years acted as a source rather than a sink of DRP to receiving waters. Studies conducted by others (Dillaha *et al.*, 1989; Osbourne and Kovacic, 1993) have also demonstrated that buffer strips may act as a source of DRP while still acting as a sink for TP.

Edge of field riparian management not only impacts on overland flow P removal but also has a strong influence on stream nutrient spiralling (Cooper *et al.*, 1987; Hearne and Howard-Williams, 1988). Fencing off pastures from grazing allows palatable aquatic macrophytes to flourish and decreases nutrient spiralling length, whereas riparian afforestation can shade out periphyton and macrophytes thus increasing spiral length. The placement of riparian afforestation within a catchment is therefore expected to have a major influence on downstream surface water quality. Quinn *et al.* (1993) reported that forested (*Pinus radiata*) riparian areas at the headwaters of grassland dominated catchments in New Zealand adversely influenced downstream P water quality (TP concentrations from grazed and riparian forested influenced catchments of 0.488 and 1.195 mg L⁻¹, respectively). This was attributed to a lack of ground cover within the riparian zone. Quinn *et al.* (1993) suggest that tree planting should be sufficiently sparse to allow development of vegetative ground cover. Fennessy and Cronk (1997) concluded that buffer strips should be located along headwater reaches where most catchment water originates. Storing water high in the catchment also decreased downstream erosion. In addition, streams and riparian areas in headwaters tend to be narrower than downstream channels, taking less land out of production while maximizing nutrient retention/removal compared to the targeting of larger channels downstream (Fennessy and Cronk, 1997).

5.2.3 Drainage Ditches and Networks

Drainage networks connecting open ditches and mole and tile drains act as a conduit of nutrients to streams bypassing riparian buffers. To minimize this, constructed wetlands could be located at the end of tile and mole drains and vegetation management implemented in open ditches (Raisin and Mitchell, 1995). Phosphorus in drainage ditch sediments is released to overlying water depending on the redox potential at the sediment–water interface, water depth, sediment EPC₀, sediment P status and overlying P concentration (Sallade and Sims, 1997; Young and Ross, 2001).

Periodic drain clearance (c. 5 years) removes vegetation and sediments to improve water flow. This practice should be targeted to drains where sediment is P saturated (if known). In conjunction, farm practices such as P fertilizer application, grazing intensity and land application of manures and effluent should be carefully managed to minimize P loss to these ditches (Sims *et al.*, 1998). To counter this P loss, studies in Maryland, Illinois and Iowa indicate that wetlands could remove up to 43% of P from drainage waters, depending on the water retention time within wetlands. This removal will be governed by the ratio of wetland size to the contributing drainage area (Woltemade, 2000). Tanner *et al.* (2002) showed that during the first 3 months, constructed wetlands installed at the end of subsurface drains (wetland surface area c. 1–2% of the drained catchment area) removed c. 80% of DRP

load. However, the long-term benefit of these systems is disputable as they become less efficient at DRP and PP retention due to siltation and P saturation.

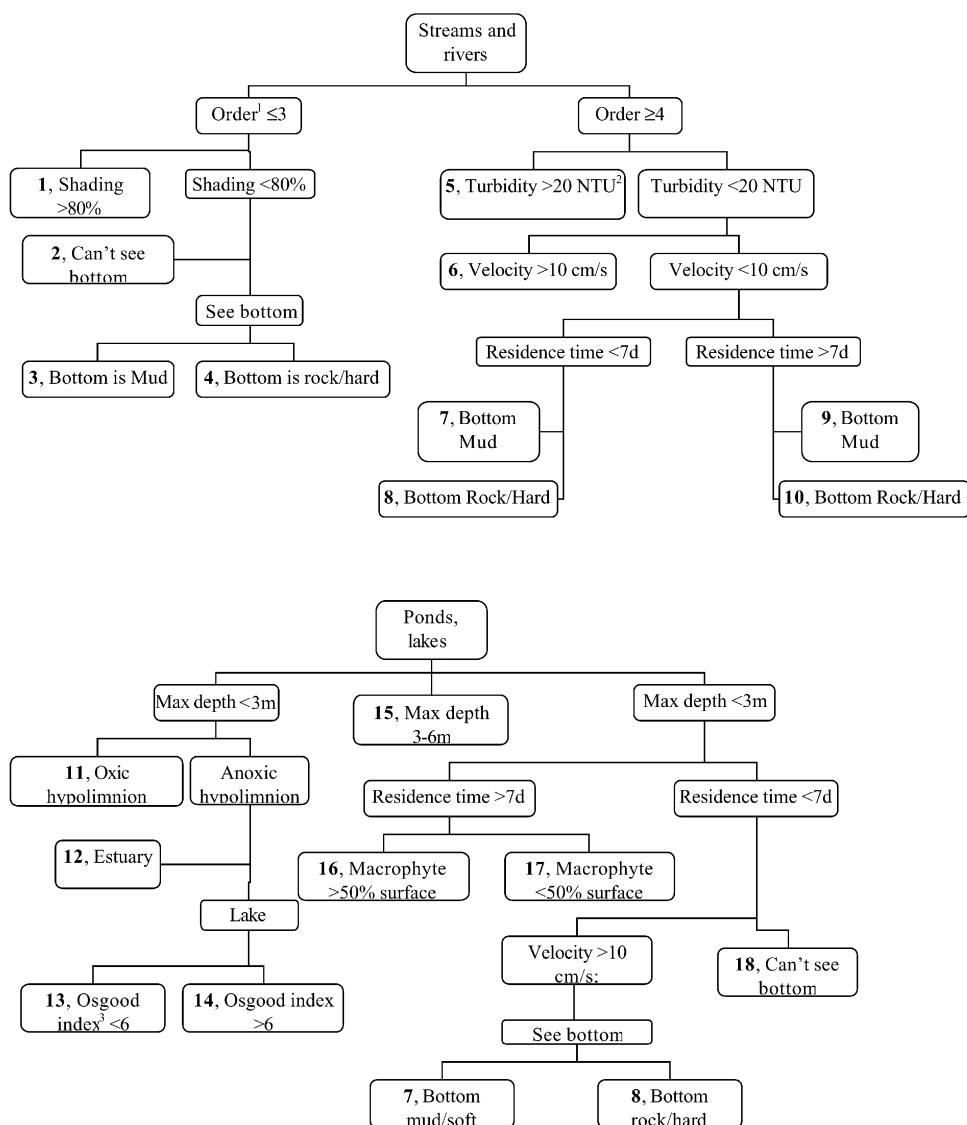
5.3 Integrating and Managing P Loss from the Landscape with Impacts in Rivers and Lakes

Successful management of surface water quality requires knowledge of how the system responds to increases or decreases in P loads. Some management strategies that decrease P loads may not be suitable for some water bodies. Indeed, decreasing P loads may have no effect at all and simply represent a waste of time, effort and resources. In many areas it may not be possible or practical to attain P loadings low enough to prevent periphyton blooms because of, for example, natural P from nutrient rich rocks (*e.g.*, Biggs, 1995). In such situations management may better focus on controlling the frequency or duration of such bloom events rather than just focusing on maximum biomass. For some in-stream environmental values such as aesthetics, it might be that the duration of blooms is as important, or even more important, than the specific levels of biomass (Biggs, 2000a). Critical P concentrations required to prevent biomass criteria from being exceeded need to link to the expected duration of low flow events. Streams and rivers which flood regularly, causing washout of the periphyton mats, can assimilate higher loads of P before developing problem blooms than waterways that have few flood disturbances and long periods available for growth (Biggs, 2000b).

To maximize environmental benefits, landscape P management needs to be connected to likely behavior and effect on surface water quality in various systems from small streams to large lakes. Newton and Jarrell (1999) devised a system of surface water classification and sensitivity in relation to increases and decreases in P loads. Figure 9 and Table II outline the classification and management of specific surface water bodies. Where mitigation of P loads is required, best management practices such as those outlined above and summarized in Figure 8 can be implemented. By targeting the right BMP to CSAs and surface water bodies within a catchment, much of the P loss can be mitigated thus maximizing the environmental aims whilst minimizing agronomic impact.

In addition to the classification system of Newton and Jarrell (1999), Biggs (2000a) developed an empirical model to predict the state of stream eutrophication as a function of both DRP concentrations and frequency of bed-cleansing flood events (Fig. 10). This model shows that the extent of periphyton biomass development is a result of the interaction between nutrient concentrations and time available for biomass accrual whereby it is possible to have high P concentrations without eutrophic conditions if flood events are frequent, resulting in short accrual periods. However, if there are long periods between flood events (*e.g.*, >100 days), then eutrophic conditions can develop with only low P concentrations (*e.g.*, <0.5 $\mu\text{g L}^{-1}$). Similar models (often called 'phosphorus loading' or 'Vollenweider' models) have been developed for phytoplankton biomass in lakes (see Chapra, 1997).

Due to variability in soil types and management, generalizations across scale are rarely applicable. As a consequence, the best scale for management should be that which effects the greatest mitigation of P loss with the best result. Due to the non-specific nature of farm boundaries in relation to catchment boundaries, management is best achieved at the scale where most benefit is evident, the catchment scale. This requires knowing where within the catchment most P loss is occurring (*i.e.* CSAs) and what the right suite of BMPs is to mitigate P loss. On the other hand, implementation may be best achieved at a farm scale, where there is more control over land use changes and more importantly, proper continued maintenance. Incentives can be used to encourage BMP adoption by farmers either passively via education, explaining the savings to be made by decreasing fertilizer inputs if soil P is



¹ Stream order is counted from headwaters to ocean and only increase when two stream of the same order meet.

² Nephelometric turbidity units (inorganic turbidity only).

³ Osgood index is defined as $z/A^{0.5}$ where z is the mean depth of the water body and A is surface area (km^2).

FIGURE 9 The classification of surface water bodies. Bold numbers refer to Table II. Adapted from Newton and Jarrell (1999).

above the agronomic optimum or actively via policy, legislation or financial aid (Rhodes *et al.*, 2002; McDowell *et al.*, 2003). One such tool that utilizes both scales to obtain best results is the P index (McDowell *et al.*, 2002b; Sharpley *et al.*, 2002). Already used in several countries, variations on the P index are being or under consideration for incorporation into strategies to address P loss. For example, in the USA the use of the P index is being promoted

TABLE II Water Body Description, Functioning, Sensitivity to Rising (↑) and Falling (↓) P Loads and Possible Management.

Description	Present functioning		Possible management scenarios	
	↑	↓	↑	↓
1. Heterotrophic small stream	Typically light limited. May be sensitive to increase in SS	Low	Low	Maintain riparian vegetation, shading and access to flood plain to prevent siltation of riparian areas
2. Phytoplankton-dominant small stream	Not light limited, but P limited	Medium	Medium	Decrease P inputs and monitor TP concentrations
3. Macrophyte-dominant small stream	Controlled by vascular plants and macrophytes, which can get sufficient P from sediment	Low	Low	Decreasing SS load most important
4. Periphyton-dominant small stream	Responds quickly to changes in P load as little sediment is present	High	High	As periphyton obtain P from water column, decrease essential
5. Heterotrophic large stream	Characterized by high SS, hence, light limited. Little potential for eutrophication	Low	Low	Groundwater P may be important
6. Periphyton-dominant large stream	High flow prevents macrophytes from establishing	Medium	Medium	Possibly decreasing SS load, but trophic state may shift to eutrophic once light limitation is removed
7. Macrophyte-dominant large stream or lake	Residence time insufficient for phytoplankton growth. Mud acts as good substrate for macrophytes	Low	Low	Decreasing P concentrations in the water column. Groundwater P input may be important
8. Periphyton-dominant large stream or lake	Residence time insufficient for phytoplankton growth but clear enough for periphyton	Medium	Medium	Efficient P uptake from water or mud means that only decrease of both will prevent growth
9. Large stream-shifts between phytoplankton and macrophyte dominance	Phytoplankton may shade out macrophytes, but temperature can be a critical factor	Medium	Low	Susceptible to sustained P inputs
10. Large stream-shifts between phytoplankton and periphyton dominance	Sufficient residence time for phytoplankton which could shade out periphyton	High	High	Sediment retention of P is low so should respond quickly to P decrease
11. Oligotrophic or mesotrophic deep lake	Stratification present, but DO still present in hypolimnion. Oxidic conditions bind P to sediment	Extreme	Medium-High	Macrophyte dominance may benefit fish

(continued)

TABLE II Continued.

<i>Description</i>	<i>Present functioning</i>	↑	↓	<i>Possible management scenarios</i>
12. Estuary	Sufficient P is supplied from marine water	Low	Low	Not important
13. Stratified deep lake – hypolimnetic P transfer	P build up in anoxic hypolimnion which is commonly transferred to epilimnion by wind turbulence	Medium–High	Low	Difficult to restore, requires chemical treatment (<i>e.g.</i> , Cu addition) or hypolimnetic aeration
14. Stratified deep lake	Less hypolimnetic P transfer than above. External loads important. P released to algae at turnover	High	Low	P loads need to decrease to prevent further eutrophication, but dependent upon loading history
15. Polymictic lake	Short-term stratification means many mixing events move P to epilimnion, but depends on trophic state	High–Low	High–Low	Sensitive if oligotrophic and insensitive if eutrophic determines effort that should go into P management
16. Shallow lake – macrophyte dominant	Macrophytes keep SS and P low in water column	High	Low	Macrophyte removal hard without introducing phytoplankton and increasing SS. Leave as is
17. Shallow lake – few macrophytes	SS and P higher than above, mean loadings irrelevant	Low	Low	Ascertain internal–external loads for management
18. Shallow lake with short residence time, high turbidity	Residence time insufficient to allow phytoplankton to grow. Light limited	Low	Low	Sediment and P removal may help but require identification of CSAs

Source: Adapted from Newton and Jarrell (1999).

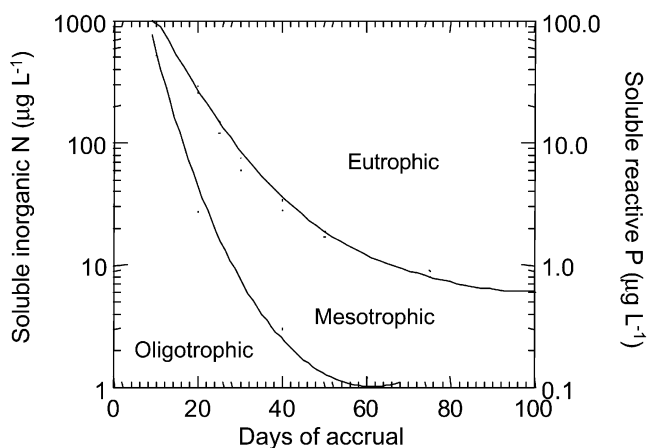


FIGURE 10 Nomograph of mean monthly dissolved nutrient concentrations in stream waters and mean days of accrual that are predicted to result in oligotrophic, mesotrophic and eutrophic periphyton biomass conditions in gravel/cobble bed streams. The oligotrophic–mesotrophic boundary is set at 60 mg m^{-2} maximum chlorophyll *a* and the mesotrophic–eutrophic boundary is a maximum of $200 \text{ mg chlorophyll a m}^{-2}$ (after [Dodds *et al.*, 1998](#)). These boundaries also equate to criteria for the protection of benthic biodiversity (oligo/mesotrophic), and aesthetics and trout fishery values (meso/eutrophic) in New Zealand streams ([Biggs, 2000b](#)). The lines delineating the trophic boundaries were calculated using a model with $R^2 = 0.74$, $N = 30$ streams (adapted from [Biggs, 2000a](#)).

on a state by state basis, aided by federal agencies ([Sharpley *et al.*, 2003](#)). In New Zealand, the P index is now being used commercially as part of a nutrient management package for farmers ([Hart *et al.*, 2002](#)). Such approaches bode well for increasing awareness of the P loss problem and towards development of methods for minimizing agricultural impacts.

Once the need for decreasing P loss on a catchment scale has been established, management then needs to focus on areas largely on the field scale to decrease P loss. Since the early 1980s, several studies have investigated the long-term (7–10 yr) effectiveness of BMPs to reduce P export from agricultural catchments. These studies quantified nutrient loss prior to, and after, BMP implementation or attempted to use untreated catchments as controls. Overall, these studies showed that BMPs decreased P export from a variety of catchments in several areas of the USA ([Goldstein and Ritter, 1993](#); [Richards and Baker, 1993](#); [Bottcher and Tremwell, 1995](#)). However, it is evident that several factors are critical to effective BMP implementation. These factors include targeting catchments that respond most effectively to BMPs, identifying CSAs, and accounting for both catchment and estuary response time and their capacity to buffer added P ([Sharpley and Rekolainen, 1997](#)).

The importance of targeting BMPs within a catchment or basin is shown by several studies across the US. In the Chesapeake Bay catchment, a coordinated and intensive program of BMP adoption was implemented in the mid-1990s to decrease nutrient inputs to the Bay through a combination of education, cost-share, and technical assistance ([Boesch *et al.*, 2001](#)). Those BMPs used included stream bank protection, manure storage, cover crops, and decreased tillage with an emphasis on decreasing sediment (and associated P) in overland flow. Nutrient management plans with field-by-field nutrient recommendations were developed for about one quarter of the agricultural land by 1997. Despite these efforts, nonpoint source P load estimates have been decreased by only 9% after 10 years, compared to 58% of P from point sources ([Boesch *et al.*, 2001](#)).

In the Little Washita River catchment (54,000 ha) in central Oklahoma, studies also showed little change in P export following BMP implementation ([Sharpley and Smith, 1994](#)) (Fig. 11). Nutrient export from two subcatchments (2 and 5 ha) were measured

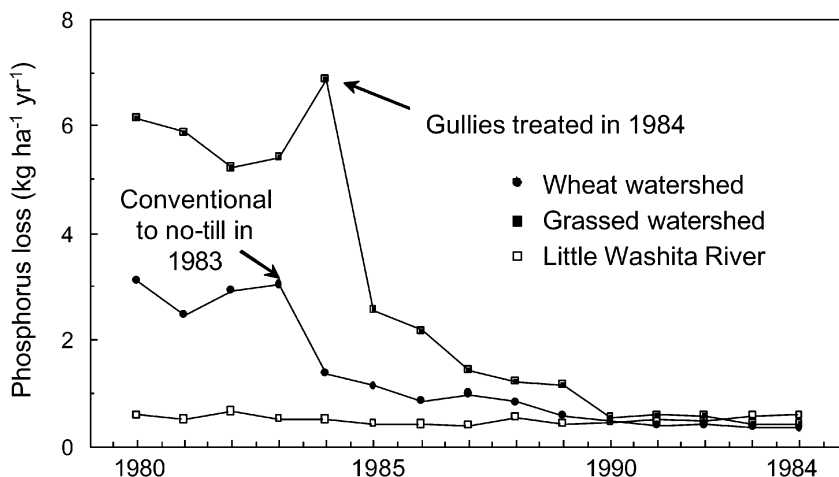


FIGURE 11 Annual P loss from the Little Washita River catchment and from two sub-catchments that were (i) in grass with eroding gullies treated in 1984, and (ii) in wheat, converted from conventional to no-till in 1983 (adapted from [Sharpley and Smith, 1994](#); [Sharpley *et al.*, 1996](#)).

from 1980 to 1994, while BMPs were installed on about 50% of the main catchment. Practices included construction of flood control impoundments, eroding gully treatment, and conservation tillage. Following conversion of conventional-till (moldboard and chisel plough) to no-till wheat in 1983, nitrogen export was decreased $14.5 \text{ kg ha}^{-1} \text{ yr}^{-1}$ (3-fold) and P loss $2.9 \text{ kg ha}^{-1} \text{ yr}^{-1}$ (10-fold; Fig. 9; [Sharpley and Smith, 1994](#)). One year later, eroding gullies were shaped and an impoundment constructed in the other subcatchment, and N and P loss decreased dramatically (5- and 13-fold, respectively; [Sharpley *et al.*, 1996](#)). However, there was no effect of BMP implementation on P concentration in flow from the main Little Washita River catchment (Fig. 11). Thus, a lack of effective targeting of BMPs and control of major sources of P export in the Little Washita River catchment did not contribute to any consistent decrease in catchment export of P.

These examples clearly demonstrate that careful targeting of BMPs at an appropriate level of intensity over sufficient time is required to effectively decrease P export from catchments. However, in order for such BMPs to be successfully implemented on-farm, they must also be economic. At present, there is a lack of knowledge on their economic effectiveness, not only on agronomic performance, but also on environmental effectiveness. To correct this imbalance, work should be directed at a life cycle analysis of whole ecosystems from plot to farm to catchment scale. In other words, we need to connect the environmental economics from the farm to the site of end impact (stream or lake).

6 CONCLUSIONS

Loss of P from agricultural landscapes occurs in dissolved and particulate form. While DP concentration represents a readily available source of P to aquatic plants, TP is preferred as an estimate of the long-term P loading and impact on surface water bodies (particularly lakes). The potential for P loss from the landscape is a function of the availability of P to flow and flow mechanisms that transport P to surface waters. While simple managements can be implemented to decrease potential P availability, such as fertilizer and manure

management and maintaining soil P concentrations within crop requirements, management of flow regimes is more difficult. Hydrological flow pathways within the landscape are notoriously fickle and vary widely depending upon environmental conditions. Coupled to this, changes that occur once P is in flow also alter the concentration and form of P lost. Recent work has outlined some of these changes, such as the dominance of DP forms lost in flow from manured soils, the enrichment of TP (largely as P-rich fines) as flow events and flow paths increase, and the changes that occur in the erosion of sediment and PP with soil moisture conditions. Better knowledge has enabled us to derive BMPs to mitigate P loss from the landscape and, coupled with an analysis of the landscape, allow us to target these to CSAs which correspond to an overlap of high potential P availability and flow. Once P has left the field, additional managements can be implemented to prevent P moving further downstream. These include the use of riparian buffer strips to mitigate P, largely as PP, and natural or constructed wetlands. However, these measures have a finite lifespan and can later serve as a P source.

Despite efforts to retain P within the landscape, inevitably P will make it to surface waters. Here the impact will vary not only depending on the type of water body (small stream to large lake), but also the variety of aquatic plants present. In some cases it may not be possible to prevent P concentrations rising to levels high enough to support excessive aquatic plant life due to the geology of substrata. For these situations, a BMP should focus on minimizing when an algal bloom is most unwanted. With agricultural systems naturally tending towards a state of eutrophication, it is imperative that the perceived problem of eutrophication be linked with P loss mechanisms. The connectivity between the two requires knowledge of water body function and targeting the correct BMPs for the desired job. This also requires us to know if this is feasible, not only economically but also culturally. In essence, life-cycle analysis of the whole ecosystem is required to determine if environmental and agricultural objectives can be maximized within the same catchment and where policy and lawmaking can assist in the process.

Acknowledgements

This review was partly funded by the New Zealand Foundation for Research, Science and Technology (contracts AGRX002 and CO1X0023).

References

- Adreini, M. S. and Steenhuis, T. S. (1990). Preferential flow paths under conventional and conservational tillage. *Geoderma*, **46**, 85–102.
- Auer, M. T. and Canale, R. P. (1982). Ecological studies and mathematical modelling of *Cladophora* in Lake Huron. 3. The dependence of growth rates on internal phosphorous pool size. *Journal of Great Lakes Research*, **8**, 93–99.
- Baldwin, D. S. (1996). Effects of exposure to air and subsequent drying on the phosphate sorption characteristics of sediments from a eutrophic reservoir. *Limnology and Oceanography*, **41**, 1725–1732.
- Baldwin, D. S., Mitchell, A. M. and Olley, J. M. (2002). Pollutant-sediment interactions: Sorption, reactivity and transport of phosphorus. in Haygarth, P. M. and Jarvis, S. C. (eds.), *Agriculture, hydrology and water quality*. CABI International, Oxford, UK, pp. 265–280.
- Barko, J. W., Gunnison, D. and Carpenter, S. R. (1991). Sediment interactions with submersed macrohyte growth and community dynamics. *Aquatic Botany*, **41**, 41–65.
- Barthès, B., Albrecht, A., Asseline, J., De Noni, G. and Roose, E. (1999). Relationship between soil erodibility and topsoil aggregate stability or carbon content in a cultivated Mediterranean highland (Aveyron, France). *Communications in Soil Science and Plant Analysis*, **30**, 1929–1938.
- Biggs, B. J. F. (1985). The use of periphyton in the monitoring of water quality. Biological Monitoring in Freshwaters: proceedings of a seminar. in Pridmore, R. D. and Cooper, A. B. (eds.), *Biological monitoring in freshwater*. Ministry of works and Development, Wellington, New Zealand, pp. 117–142.
- Biggs, B. J. F. (1995). The contribution of disturbance, catchment geology and landuse to the habitat template of periphyton in stream ecosystems. *Freshwater Biology*, **33**, 419–438.

- Biggs, B. J. F. (1996). Patterns in benthic algae of streams. in Stevenson, R. J., Bothwell, M. L. and Lowe, R. L. (eds.), *Algal ecology: freshwater benthic ecosystems*. Academic Press, San Diego, pp. 31–56.
- Biggs, B. J. F. (2000a). Eutrophication of streams and rivers: dissolved nutrient-chlorophyll relationships for benthic algae. *Journal of the North American Benthological Society*, **19**, 17–31.
- Biggs, B. J. F. (2000b). New Zealand Periphyton Guidelines: Detecting, monitoring and managing enrichment of streams. New Zealand Ministry for the Environment publication. 122 pp. Available from: <http://www.mfe.govt.nz/publications/water/nz-periphyton-guide-jun00.pdf>
- Biggs, B. J. F. and Malthus, T. J. (1982). Macroinvertebrates associated with various aquatic macrophytes in the backwaters and lakes of the upper Clutha Valley, New Zealand. *New Zealand Journal of Marine and Freshwater Research*, **16**, 81–88.
- Biggs, B. J. F. and Close, M. E. (1989). Periphyton biomass dynamics in gravel bed rivers: the relative effects of flows and nutrients. *Freshwater Biology*, **22**, 209–231.
- Biggs, B. J. F. and Smith, R. A. (2002). Taxonomic richness of stream benthic algae: effects of flood disturbance and nutrients. *Limnology and Oceanography*, **47**, 1175–1186.
- Boesch, D. F., Brinsfield, R. B. and Magnien, R. E. (2001). Chesapeake Bay eutrophication: scientific understanding, ecosystem restoration, and challenges for agriculture. *Journal of Environmental Quality*, **3**, 303–320.
- Booltink, H. W. G. and Bourma, J. (1991). Physical and morphological characterization of bypass flow in a well structured clay soil. *Soil Science Society of America Journal*, **55**, 1249–1254.
- Borchardt, M. A. (1996). Nutrients. in Stevenson, R. J., Bothwell, M. L. and Lowe, R. L. (eds.), *Algal ecology: freshwater benthic ecosystems*. Academic Press, San Diego, pp. 184–227.
- Boström, B., Anderson, J. M., Fleischer, S. and Jansson, M. (1988). Exchange of phosphorus across the sediment-water interface. *Hydrobiologia*, **170**, 229–244.
- Bothwell, M. L. (1985). Phosphorus limitation of lotic periphyton growth rates: an intersite comparison using continuous-flow troughs (Thompson River system: British Columbia). *Limnology and Oceanography*, **30**, 526–542.
- Bothwell, M. L. (1989). Phosphorus-limited growth dynamics of lotic periphytic diatom communities: areal biomass and cellular growth rate responses. *Canadian Journal of Fisheries and Aquatic Sciences*, **46**, 1293–1301.
- Botcher, A. B. and Tremwell, T. (1995). BMPs for water quality improvement. *Ecological Engineering*, **5**, 341–356.
- Breeuwsma, A. and Silva, S. (1992). Phosphorus fertilization and environmental effects in the Netherlands and the Po region (Italy). Report No. 57, The Winand Staring Centre for Integrated Land, Soil and Water Research, Wageningen, The Netherlands, 39 p.
- Burkholder, J. A. and Glasgow, H. B., Jr. (1997). *Pfiesteria piscicidia* and other *Pfiesteria*-dinoflagellates behaviors, impacts, and environmental controls. *Limnology and Oceanography*, **42**, 1052–1075.
- Calhoun, F. G., Bigham, J. M. and Slater, B. K. (2002). Relationships among plant available phosphorus, fertilizer sales, and water quality in Northwestern Ohio. *Journal of Environmental Quality*, **31**, 38–46.
- Carpenter, S. R., Caraco, N. F., Correll, D. L., Howarth, R. W., Sharpley, A. N. and Smith, V. H. (1998). Nonpoint pollution of surface waters with phosphorus and nitrogen. *Ecological Applications*, **8**, 559–568.
- Chambers, P. A., Prepas, E. E., Bothwell, M. L. and Hamilton, H. R. (1989). Roots versus shoots in nutrient uptake by aquatic macrophytes in flowing waters. *Canadian Journal of Fisheries and Aquatic Sciences*, **46**, 435–439.
- Chapra, S. C. (1997). *Surface water-quality modeling*. McGraw-Hill, New York.
- Chardon, W. J., Oenema, O., del Castilho, P., Vriesema, R., Japenga, J. and Blaauw, D. (1997). Organic phosphorus in solutions and leachates from soils treated with animal slurries. *Journal of Environmental Quality*, **26**, 372–378.
- Chessman, B. C., Hutton, P. E. and Burch, J. M. (1992). Limiting nutrients for periphyton growth in sub-alpine, forest, agricultural and urban streams. *Freshwater Biology*, **28**, 349–361.
- Chétaillet, J., Pick, E. R., Morin, A. and Hamilton, P. B. (1999). Periphyton biomass and community composition in rivers of different nutrient status. *Canadian Journal of Fisheries and Aquatic Sciences*, **56**, 560–569.
- Cloern, J. E. (2001). Our evolving conceptual model of the coastal eutrophication problem. *Marine Ecology Progress Series*, **210**, 223–253.
- Cooke, J. G., Stub, L. and Mora, N. (1992). Fractionation of phosphorus in the sediment of a wetland after a decade of receiving a sewage effluent. *Journal of Environmental Quality*, **21**, 726–733.
- Cooper, A. B. and Thomsen, C. E. (1988). Nitrogen and phosphorus in streamwaters from adjacent pasture, pine, and native forest catchments. *New Zealand Journal of Marine and Freshwater Research*, **22**, 279–291.
- Cooper, A. B., Smith, C. M. and Smith, M. J. (1995). Effects of riparian set-aside on soil characteristics in an agricultural landscape: Implications for nutrient transport and retention. *Agriculture, Ecosystems and Environment*, **55**, 61–67.
- Cooper, J. R., Gilliam, J. W., Daniels, R. B. and Robarge, W. P. (1987). Riparian areas as filters for agricultural sediment. *Soil Science Society of America Journal*, **51**, 416–420.
- Correll, D. L. (1998). The role of phosphorus in the eutrophication of receiving waters: A review. *Journal of Environmental Quality*, **27**, 261–266.
- Davelaar, D. (1993). Ecological significance of bacterial polyphosphate metabolism in sediments. *Hydrobiologia*, **207**, 179–192.
- Dillaha, T. A., Reneau, R. B., Mostaghimi, S. and Lee, D. (1989). Vegetative filter strips for agricultural nonpoint source pollution control. *Transactions of the American Society of Agricultural Engineers*, **32**, 513–519.
- Djordjic, F., Bergström, L. and Ulén, B. (2002). Phosphorus losses from a structured clay soil in relation to tillage practices. *Soil Use and Management*, **18**, 79–84.

- Dodds, W. K. and Welch, E. B. (2000). Establishing nutrient criteria in streams. *Journal of North American Benthological Society*, **19**, 186–196.
- Dodds, W. K., Smith, V. H. and Zander, B. (1997). Developing nutrient targets to control benthic chlorophyll levels in streams: A case study of the Clark Fork River. *Water Research*, **31**, 1738–1750.
- Dodds, W. K., Jones, J. R. and Welch, E. B. (1998). Suggested classification of stream trophic state: distributions of temperate streamthopes by chlorophyll, total nitrogen, and phosphorus. *Water Research*, **32**, 1455–1462.
- Edwards, D. R. and Daniel, T. C. (1993). Drying interval effects on runoff from fescue plots receiving swine manure. *Transactions of the American Society of Agricultural Engineers*, **36**, 1673–1678.
- Eghball, B. and Gilley, J. E. (1999). Phosphorus and nitrogen in runoff following beef cattle manure or compost application. *Journal of Environmental Quality*, **28**, 1201–1210.
- Ekholm, P., Kallio, K., Salo, S., Pietiläinen, O.-P., Rekolainen, S., Laine, Y. and Joukola, M. (2000). Relationship between catchment characteristics and nutrient concentrations in an agricultural system. *Water Research*, **34**, 3709–3716.
- Elwood, J. W., Newbold, J. D., O'Neil, R. V. and Van Winkle, W. (1983). Resource spiraling: An operational paradigm for analyzing lotic ecosystems. in Fontaine, T. D., III and Bartell, S. M. (eds.), *Dynamics of lotic ecosystems*. Ann Arbor Science, Ann Arbor, Michigan, pp. 3–27.
- Ennessy, M. S. and Cronk, J. K. (1997). The effectiveness and restoration potential of riparian ecotones for the management of non-point source pollution, particularly nitrate. *Critical Reviews in Environmental Science and Technology*, **27**, 285–317.
- Flenniken, M., Mceldowney, R. R., Leininger, W. C., Fraser, G. W. and Trlica, M. J. (2001). Hydrologic response of a montane riparian ecosystem following cattle use. *Journal of Range Management*, **54**, 567–574.
- Fox, L. E. (1993). The chemistry of aquatic phosphate: inorganic processes in rivers. *Hydrobiologia*, **253**, 1–16.
- Francoeur, S. N., Biggs, B. J. F., Smith, R. and Lowe, R. L. (1999). Nutrient limitation of algal biomass accrual: Seasonal patterns and comparison of methodology. *Journal of the North American Benthological Society*, **18**, 242–253.
- Gächter, R. and Meyer, J. S. (1993). The role of microorganisms in mobilization and fixation of phosphorus in sediments. *Hydrobiologia*, **253**, 103–121.
- Gascho, G. J., Wauchope, R. D., Davis, J. G., Truman, C. C., Dowler, C. C., Hook, J. E., Sumner, H. R. and Johnson, A. W. (1998). Nitrate–nitrogen, soluble, and bioavailable phosphorus runoff from simulated rainfall after fertilizer application. *Soil Science Society of America Journal*, **62**, 1711–1718.
- Gilley, J. E. and Risse, L. M. (2000). Runoff and soil loss as affected by the application of manure. *Transactions of the American Society of Agricultural Engineers*, **43**, 1583–1588.
- Gitau, M. W., Schneiderman, E., Gburek, W. J. and Jarrett, A. R. (2001). An evaluation of best management practices installed in the cannonsville reservoir watershed, New York, in *Proc. "9th National Non Point Source Monitoring Workshop"*, 27–30 August 2001, Indianapolis, IN.
- Goldstein, A. L. and Ritter, R. D. (1993). A performance-based regulatory program for phosphorus control to prevent the accelerated eutrophication of Lake Okeechobee, Florida. *Water Science and Technology*, **28**, 13–26.
- Green, F. B., Lundquist, T. J. and Oswald, W. J. (1995). Energetics of advanced integrated wastewater pond systems. *Water Science and Technology*, **31**, 9–20.
- Green, F. B., Bernstone, L. S., Lundquist, T. J. and Oswald, W. J. (1996). Advanced integrated wastewater pond systems for nitrogen removal. *Water Science and Technology*, **33**, 207–217.
- Greer, J. (1997). In harm's way? The threat of toxic algae. *Marine Notes*, July–August 1997, (<http://www.mdsg.umd.edu/MDSG/Communications/MarineNotes/index.html>), pp. 1–4.
- Hanrahan, G., Gledhill, M., House, W. A. and Worsfold, P. J. (2001). Phosphorus loading in the Frome catchment, UK: Seasonal refinement on the coefficient modeling approach. *Journal of Environmental Quality*, **30**, 1738–1746.
- Harris, G. P. (1994). Nutrient loadings and algal blooms in Australian waters—a discussion paper. Land and Water Resources, Research and Development Corporation, Department of Defence Occasional Paper No. 12/94, 18 p.
- Hart, M. R., Elliott, S., Petersen, J., Stroud, M. J., Cooper, A. B., Nguyen, M. L. and Quin, B. F. (2002). Assessing and managing the potential risk of phosphorus losses from agricultural land to surface waters, in L. D. Currie and P. Loganathan (eds.), *Proc. Dairy Farm Management* 13–14 February, Massey University, Palmerston North, New Zealand, Occasional Report No 15. Fertilizer and Lime Research Centre, Massey University, Palmerston North, New Zealand, pp. 155–166.
- Haygarth, P. M., Hepworth, L. and Jarvis, S. C. (1998). Forms of phosphorus transfer in hydrological pathways from soil under grazed grassland. *European Journal of Soil Science*, **49**, 65–72.
- Haygarth, P. M., Heathwaite, A. L., Jarvis, S. C. and Harrod, T. R. (2000). Hydrological factors for phosphorus transfer from agricultural soils. *Advances in Agronomy*, **69**, 154–178.
- Hearne, J. W. and Howard-Williams, C. (1988). Modelling nitrate removal by riparian vegetation in a spring-fed stream: the influence of land-use practices. *Ecological Modelling*, **42**, 179–198.
- Heckrath, G., Brookes, P. C., Poulton, P. R. and Goulding, K. W. T. (1995). Phosphorus leaching from soils containing different phosphorus concentrations in the Broadbalk experiment. *Journal of Environmental Quality*, **24**, 904–910.
- Higgins, J. M. and Kim, B. R. (1981). Phosphorus retention models for Tennessee Valley Authority reservoirs. *Water Resources Research*, **17**, 571–576.
- Horner, R. R., Welch, E. B., Seeley, M. R. and Jacoby, J. M. (1990). Responses periphyton to changes in current velocity, suspended sediment and phosphorus concentration. *Freshwater Biology*, **24**, 215–232.

- House, W. A., Denison, F. H., Smith, J. T. and Armitage, P. D. (1995). An investigation of the effects of water velocity on inorganic phosphorus influx to a sediment. *Environmental Pollution*, **89**, 263–271.
- Hudson, J. J., Taylor, W. D. and Schindler, D. W. (2000). Phosphate concentrations in lakes. *Nature*, **406**, 54–56.
- Humphrey, K. P. and Stevenson, R. J. (1992). Responses of benthic algae to pulses in current and nutrients during simulations of subscouring spates. *Journal of the North American Benthological Society*, **11**, 37–48.
- Hylander, L. D. and Siman, G. (2001). Plant availability of phosphorus sorbed to potential wastewater treatment materials. *Biology and Fertility of Soils*, **34**, 42–48.
- Jensen, M. B., Jørgensen, P. R., Hansen, H. C. B. and Lielsen, N. E. (1998). Biopore mediated subsurface transport of dissolved orthophosphate. *Journal of Environmental Quality*, **27**, 1130–1137.
- Johnston, C. A. (1991). Sediment and nutrient retention by freshwater wetlands: Effects on surface water quality. *Critical Reviews in Environmental Control*, **21**, 491–565.
- Khoshmanesh, A., Hart, B. T., Duncan, A. and Beckett, R. (1999). Biotic uptake and release of phosphorus by a wetland sediment. *Environmental Technology*, **29**, 85–91.
- Kleinman, P. J. A., Sharpley, A. N., Moyer, B. G. and Elwinger, G. F. (2002). Effect of mineral and manure phosphorus sources on runoff phosphorus. *Journal of Environmental Quality*, **31**, 2026–2033.
- Kleinman, P. J. A., Neeldeman, B. A., Sharpley, A. N. and McDowell, R. W. (2003). Using soil phosphorus profile data to assess phosphorus leaching potential from manured soils. *Soil Science Society of America Journal*, **67**, 215–224.
- Koopmans, G. F., McDowell, R. W., Chardon, W. J., Oenema, O. and Dolfing, J. (2002). Soil phosphorus quantity-intensity relationships to predict increased soil phosphorus loss to overland and subsurface flow. *Chemosphere*, **48**, 679–687.
- Kotak, B. G., Kenefick, S. L., Fritz, D. L., Rousseaux, C. G., Prepas, E. E. and Hrudey, S. E. (1993). Occurrence and toxicological evaluation of cyanobacterial toxins in Alberta lakes and farm dugouts. *Water Research*, **27**, 495–506.
- Kunishi, H. M., Taylor, A. W., Heald, W. R., Gburek, W. J. and Weaver, R. N. (1972). Phosphate movement from an agricultural watershed during two rainfall periods. *Journal of Agricultural and Food Chemistry*, **20**, 900–905.
- Le Bissonnais, Y., Benkhadra, H., Chaplot, V., Fox, D., King, D. and Daroussin, J. (1998). Crusting, runoff and sheet erosion on silty loamy soils at various scales and upscaling from m² to small catchments. *Soil and Tillage Research*, **46**, 69–80.
- Leytem, A. B., Mikkelsen, R. L. and Gillam, J. W. (2002). Sorption of organic phosphorus compounds in atlantic coastal plain soils. *Soil Science*, **167**, 652–658.
- Lindsay, W. L. (1979). *Chemical equilibria in soils*. Wiley-Interscience, New York, NY, 449 pp.
- Lory, J. A., Daniel, T. C., Burch, T., Kleinman, P. J. A. and Scharf, P. C. (2001). Nonlinear effects of soil test phosphorus extraction method and sampling depth on runoff phosphorus predictions. *Agronomy Abstracts*, p. 109. Am. Soc. Agron., Madison, WI.
- Lovley, D. R., Phillips, E. J. P. and Lonergan, D. J. (1991). Enzymatic versus non-enzymatic mechanisms for Fe(III) reduction in aquatic sediments. *Environmental Science and Technology*, **25**, 1062–1067.
- Lyons, J., Trimble, S. W. and Paine, L. K. (2000). Grass versus trees: Managing riparian areas to benefit streams of Central North America. *Journal of the American Water Resources Association*, **36**, 919–930.
- Maguire, R. O., Edwards, A. C. and Wilson, M. J. (1998). Influence of cultivation on the distribution of phosphorus in three soils from NE Scotland and their aggregate size fractions. *Soil Use and Management*, **14**, 147–153.
- McDowell, R. W. and Sharpley, A. N. (2001a). Soil phosphorus fractions in solution: influence of fertilizer and manure, filtration and method of determination. *Chemosphere*, **45**, 737–748.
- McDowell, R. W. and Sharpley, A. N. (2001b). Phosphorus losses in subsurface flow before and after manure application to intensively farmed land. *Science of the Total Environment*, **278**, 113–125.
- McDowell, R. W. and Sharpley, A. N. (2001c). Approximating phosphorus release from soils to surface runoff and subsurface drainage. *Journal of Environmental Quality*, **30**, 508–520.
- McDowell, R. W. and Sharpley, A. N. (2001d). A comparison of fluvial sediment phosphorus (P). Chemistry in relation to location and potential to influence stream P concentrations. *Aquatic Geochemistry*, **7**, 255–265.
- McDowell, R. W. and Sharpley, A. N. (2002a). Phosphorus transport in overland flow in response to position of manure application. *Journal of Environmental Quality*, **31**, 217–227.
- McDowell, R. W. and Sharpley, A. N. (2002b). Effect of plot scale and an upslope phosphorus source on phosphorus loss in overland flow. *Soil Use and Management*, **18**, 112–119.
- McDowell, R. W. and Sharpley, A. N. (2002c). The effect of antecedent moisture conditions on sediment and phosphorus loss during overland flow: Mahantango Creek catchment, Pennsylvania, USA. *Hydrological Processes*, **16**, 3037–3050.
- McDowell, R. W. and Sharpley, A. N. (2003a). Phosphorus solubility and release kinetics as a function of soil test P concentration. *Geoderma*, **112**, 143–154.
- McDowell, R. W. and Sharpley, A. N. (2003b). Phosphorus and sediment loss in overland flow in relation to soil carbon quality. *Journal of Environmental Quality*, **32**, 207–214.
- McDowell, R. W. and Sharpley, A. N. (2003c). Uptake and release of phosphorus from overland flow in a stream environment. *Journal of Environmental Quality*, **32**, 937–948.
- McDowell, R. W., Sinaj, S., Sharpley, A. N. and Frossard, E. (2001a). The use of isotopic exchange kinetics to assess phosphorus availability in overland flow and subsurface drainage waters. *Soil Science*, **166**, 365–373.
- McDowell, R., Sharpley, A. and Folmar, G. (2001b). Phosphorus export from an agricultural watershed: linking source and transport mechanisms. *Journal of Environmental Quality*, **30**, 1587–1595.

- McDowell, R. W., Sharpley, A. and Chalmers, A. T. (2002a). Chemical characterisation of fluvial sediment: The Winooski River, Vermont. *Ecological Engineering*, **18**, 477–487.
- McDowell, R. W., Sharpley, A. and Kleinman, P. J. A. (2002b). Integrating phosphorus and nitrogen decision management at watershed scales. *Journal of the American Water Resources Association*, **38**, 479–491.
- McDowell, R. W., Monaghan, R. M. and Morton, J. (2003). Soil phosphorus concentrations to minimize potential P loss to surface waters in Southland. *New Zealand Journal of Agricultural Research*, **46**, 239–253.
- Melack, J. M. (1995). Transport and transformations of P, fluvial and lacustrine ecosystems. in Tiessen, H. (ed.), *Phosphorus in the Global Environment: Transfers, Cycles and Management*. SCOPE 54 and John Wiley and Sons, Chichester, UK, pp. 245–254.
- Menzel, R. G. (1980). Enrichment ratios for water quality modeling. in Knisel, W. (ed.), *CREAMS—A Field Scale Model for Chemicals, Runoff and Erosion from Agricultural Management Systems*, Vol. III. Supporting Documentation, USDA, Cons. Res. Rep. 26. U.S. Govt. Printing Office, Washington, DC, pp. 486–492.
- Monaghan, R., Horne, D., Hedley, M. and Nguyen, L. (2002). Water and contaminant loads from artificial subsurface drainage systems in New Zealand. AgResearch Client Report for Fonterra Research Centre, AgResearch Ltd, Invermay Agricultural Centre, Mosgiel, New Zealand, 73 p.
- Mueller, D. H., Wendt, R. C. and Daniel, T. C. (1984). Phosphorus losses as affected by tillage and manure application. *Soil Science Society of America Journal*, **48**, 901–905.
- Mulholland, P. J., Newbold, J. D., Elwood, J. W., Ferren, L. A. and Webster, J. R. (1985). Phosphorus spiralling in a woodland stream: Seasonal variations. *Ecology*, **66**, 1012–1023.
- Munn, N. L. and Meyer, J. L. (1990). Habitat-specific solute retention in two small streams, An intersite comparison. *Ecology*, **71**, 2069–2082.
- Nash, D., Halliwell, D. and Cox, J. (2002). Hydrological mobilization of pollutants at the field/slope scale. in Haygarth, P. M. and Jarvis, S. C. (Eds.), *Agriculture, Hydrology and Water Quality*. CABI International, Oxford, UK, pp. 225–242.
- Newbold, J. D., Elwood, J. W., O'Neill, R. V. and Van Winkle, W. (1981). Measuring nutrient spiralling in streams. *Canadian Journal of Fisheries and Aquatic Science*, **38**, 860–863.
- Newman, E. I. (1995). Phosphorus inputs to terrestrial ecosystems. *Journal of Ecology*, **83**, 713–726.
- Newton, B. J. and Jarrell, W. M. (1999). A procedure to estimate the response of aquatic systems to changes in phosphorus and nitrogen inputs. Natural Resources Conservation Service, National Water and Climate Center, Portland, OR, USA. 37pp. Available from: www.wcc.nrcs.usda.gov/water/quality/common/aquasens.pdf
- Nguyen, M. L. (2000a). Phosphate incorporation and transformation in surface sediments of a sewage-impacted wetland as influenced by sediment sites, sediment pH and added phosphate concentration. *Ecological Engineering*, **14**, 139–155.
- Nguyen, M. L. (2000b). Organic matter composition, microbial biomass and microbial activity in gravel-bed constructed wetlands treating farm dairy wastewaters. *Ecological Engineering*, **16**, 199–221.
- Nguyen, M. L. and Davies-Colley, R. J. (1998). Nitrogen and phosphorus cycling in waste stabilisation ponds and their influence on the quality of discharged effluents: A review. *NIWA Technical Report*, **43**, 32.
- Nguyen, M. L., Cooke, J. G. and McBride, G. B. (1997). Phosphorus retention and release characteristics of sewage-impacted wetland sediments. *Water, Air, and Soil Pollution*, **100**, 163–179.
- Nurdogan, Y. and Oswald, W. J. (1995). Enhanced nutrient removal in high rate ponds. *Water Science and Technology*, **31**, 33–43.
- Oades, J. M. and Waters, A. G. (1991). Aggregate hierarchy in soils. *Australian Journal of Soil Research*, **29**, 815–828.
- Olley, J. M., Murray, A. S., Mackenzie, D. H. and Edwards, K. (1993). Identifying sediment sources in a gullied catchment using natural and anthropogenic radioactivity. *Water Resources Research*, **29**, 1037–1043.
- Osborne, L. L. and Kovacic, D. A. (1993). Riparian vegetated buffer strips in water quality restoration and stream management. *Freshwater Biology*, **29**, 243–258.
- Patrick, W. H., Jr. and Khalid, R. A. (1974). Phosphorus release and sorption by soils and sediments: Effects of aerobic and anaerobic conditions. *Science*, **186**, 53–55.
- Petts, G. E. (1984). *Impounded rivers*. John Wiley and Sons, New York, New York.
- Pionke, H. B., Gburek, W. J., Sharpley, A. N. and Schnabel, R. R. (1996). Flow and nutrient patterns for an agricultural hill-land watershed. *Water Resources Research*, **32**, 1795–1804.
- Pote, D. H., Daniel, T. C., Nichols, D. J., Sharpley, A. N., Moore, P. A., Jr., Miller, D. M. and Edwards, D. R. (1999). Relationship between phosphorus levels in three Ultisols and phosphorus concentrations in runoff. *Journal of Environmental Quality*, **28**, 170–175.
- Preedy, N., McTiernan, K., Mathews, R., Heathwaite, L. and Haygarth, P. (2001). Rapid incidental phosphorus transfers from grassland. *Journal of Environmental Quality*, **30**, 2105–2112.
- Qiu, S. and McComb, A. J. (1995). Planktonic and microbial contributions to phosphorus release from fresh and air-dried sediments. *Freshwater Research*, **46**, 1039–1045.
- Quinn, J. M., Cooper, A. B. and Williamson, R. B. (1993). Riparian zones as buffer strips: A New Zealand perspective. in Bunn, S. E., Pusey, B. J. and Price, P. (Eds.), *Ecology and Management of Riparian Zones in Australia*. LWRDC Occasional Series No. 05/93. LWRDC, Canberra and CCISR, Griffith University, Maroolia, Queensland, pp. 53–88.
- Quinton, J. N., Catt, J. A. and Hess, T. M. (2001). The selective removal of phosphorus from soil: Is size event important? *Journal of Environmental Quality*, **30**, 538–545.
- Raisin, G. W. and Mitchell, D. S. (1995). The use of wetlands for the control of non-point source pollution. *Water Science and Technology*, **32**, 177–186.

- Reddy, K. R., Diaz, O. A., Scinto, L. J. and Agami, M. (1995). Phosphorus dynamics in selected wetlands and streams of the lake Okeechobee Basin. *Ecological Engineering*, **5**, 183–207.
- Reddy, K. R., Kadlec, R. H., Flaig, E. and Gale, P. M. (1999). Phosphorus retention in streams and wetlands: A review. *Critical Reviews in Environmental Science and Technology*, **29**, 83–146.
- Rhodes, H. M., Leland, L. S. and Niven, B. E. (2002). Farmers, streams, information, and money: does informing farmers about riparian management have any effect? *Environmental Management*, **30**, 665–677.
- Richards, R. P. and Baker, D. B. (1993). Trends in nutrient and suspended sediment concentrations in Lake Erie Tributaries, 1975–1990. *Journal of Great Lakes Research*, **19**, 200–211.
- Riis, T. and Biggs, B. J. F. (in press). Hydrologic and hydraulic control of macrophyte establishment and performance in streams. *Limnology and Oceanography*.
- Rodda, H., Nguyen, L. and Elliott, S. (1998). The development of a model to predict dissolved phosphorus in surface runoff from grazed hill-country pastures in New Zealand. NIWA Client Report AGR80201 for AgResearch, National Institute of Water and Atmospheric Research Ltd, Hamilton, New Zealand, 26 p.
- Romkens, M. J. M., Nelson, D. W. and Mannering, J. V. (1973). Nitrogen and phosphorus composition of surface runoff as affected by tillage method. *Journal of Environmental Quality*, **2**, 292–295.
- Rousseva, S. (1989). A laboratory index for soil erodibility assessment. *Soil Technology*, **2**, 287–299.
- Sallade, Y. E. and Sims, J. T. (1997). Phosphorus transformations in the sediments of Delaware's Agricultural Drainageways: II. Effect of reducing conditions on phosphorus release. *Journal of Environmental Quality*, **26**, 1579–1588.
- Sharples, A. N. (1985a). Depth of surface soil-runoff interaction as affected by rainfall, soil slope, and management. *Soil Science Society of America Journal*, **49**, 1010–1015.
- Sharples, A. N. (1985b). The selective erosion of plant nutrients in runoff. *Soil Science Society of America Journal*, **49**, 1527–1534.
- Sharples, A. N. (1995). Dependence of runoff phosphorus on extractable soil phosphorus. *Journal of Environmental Quality*, **24**, 920–926.
- Sharples, A. N. (1997). Rainfall frequency and nitrogen and phosphorus in runoff from soil amended with poultry litter. *Journal of Environmental Quality*, **26**, 1127–1132.
- Sharples, A. N. (ed.) (2000). *Agriculture and phosphorus management: the Chesapeake Bay*. CRC Press, Boca Raton, FL.
- Sharples, A. N. and Syers, J. K. (1979). Loss of nitrogen and phosphorus in tile drainage as influenced by urea application and grazing animals. *New Zealand Journal of Agricultural Research*, **22**, 127–131.
- Sharples, A. N. and Ahuja, L. R. (1983). A diffusion interpretation of soil phosphorus desorption. *Soil Science*, **135**, 322–326.
- Sharples, A. N. and Williams, J. R. (eds.) (1990). *EPIC-erosion/productivity impact calculator: I. Model documentation*. U.S. Department of Agriculture Tech. Bull. 1768.
- Sharples, A. N. and Smith, S. J. (1994). Wheat tillage and water quality in the Southern Plains. *Soil and Tillage Research*, **30**, 33–38.
- Sharples, A. N. and Rekolainen, S. (1997). Phosphorus in agriculture and its environmental implications. in Tunney, H., Carton, O. T., Brookes, P. C. and Johnston, A. E. (eds.), *Phosphorus loss from soil to water*. CAB International Press, Wallingford, Oxon, UK, pp. 1–54.
- Sharples, A. N. and Tunney, H. (2000). Phosphorus research strategies to meet agricultural and environmental challenges of the 21st century. *Journal of Environmental Quality*, **29**, 176–181.
- Sharples, A. N., Menzel, R. G., Smith, S. J., Rhoades, E. D. and Olness, A. E. (1981). The sorption of soluble P by soil material during transport in runoff from cropped and grassed watersheds. *Journal of Environmental Quality*, **10**, 211–215.
- Sharples, A. N., Smith, S. J., Stewart, B. A. and Mathers, A. C. (1984). Forms of phosphorus in soil receiving cattle feedlot waste. *Journal of Environmental Quality*, **13**, 211–215.
- Sharples, A. N., Smith, S. J., Zollweg, J. A. and Coleman, G. A. (1996). Gully treatment and water quality in the Southern Plains. *Journal of Soil and Water Conservation*, **51**, 512–517.
- Sharples, A. N., Kleinman, P. J. A., McDowell, R. W. and Weld, J. L. (2001). Assessing site vulnerability to phosphorus loss in an agricultural watershed. *Journal of Environmental Quality*, **30**, 2026–2036.
- Sharples, A. N., Kleinman, P. J. A., McDowell, R. W., Gitau, M. and Bryant, R. B. (2002). Modelling phosphorus transport in agricultural watersheds: Processes and possibilities. *Journal of Soil and Water Conservation*, **57**, 425–439.
- Sharples, A. N., Weld, J. L., Beegle, D. B., Kleinman, P. J. A., Gburek, W. L., Moore, P. A. and Mullins, G. (2003). Development of phosphorus indices for nutrient management planning strategies in the U.S. *Journal of Soil and Water Conservation*, **58** (in press).
- Sims, J. T., Simard, R. R. and Joern, B. C. (1998). Phosphorus losses on agricultural drainage: Historical perspectives and current research. *Journal of Environmental Quality*, **27**, 277–293.
- Sinaj, S. J., Mächler, F., Frossard, E., Faisse, C., Oberson, A. and Morel, C. (1998). Interference of colloidal particles in the determination of orthophosphate concentrations in soil water extracts. *Communications in Soil Science and Plant Analysis*, **29**, 1091–1105.
- Slattery, M. C. and Burt, T. P. (1997). Particle size characteristics of suspended sediment in hillslope runoff and stream flow. *Earth Surface Processes and Landforms*, **22**, 705–719.
- Smith, C. M. (1989). Riparian pasture retirement effects on sediment, phosphorus, and nitrogen in channelised surface runoff from pastures. *New Zealand Journal of Marine and Freshwater Research*, **23**, 139–146.

- Smith, C. M., Wilcock, R. J., Vant, W. N., Smith, D. G. and Cooper, A. B. (1993). Towards sustainable agriculture: freshwater quality in New Zealand and the influence of agriculture. MAF Policy Technical Paper 93/10. Wellington, New Zealand, 208 pp.
- Smith, J. A. (1992). Precipitation. in Maidment, D. R. (ed.), *Handbook of hydrology*. McGraw-Hill, New York, NY, pp. 3.1–3.47.
- Smith, K. A., Jackson, D. R. and Withers, P. J. A. (2001). Nutrient losses by surface run-off following the application of organic manures to arable land. 2. Phosphorus. *Environmental Pollution*, **112**, 53–60.
- Smith, S. J., Sharpley, A. N., Williams, J. R., Berg, W. A. and Coleman, G. A. (1991). Sediment-nutrient transport during severe storms, in *Proceedings of the Fifth Federal Interagency Sedimentation Conference*, Las Vegas, Nevada (USA), 48–55.
- Srinivasan, M. S., Gburek, W. J. and J.M. Hamlett. (2002a). Dynamics of stormflow generation: A hillslope-scale field study in east-central Pennsylvania, USA. *Hydrologic Processes*, **16**, 649–665.
- Srinivasan, M. S., Kleinman, P. J. A. and Gburek, W. J. (2002b). National phosphorus runoff project: Hydrologic evaluation of runoff boxes. Internal Report, USDA-ARS, Pasture Systems and Watershed Management Research Unit, University Park, PA, 6p. Available at: <http://pswmru.arsup.psu.edu/phosphorus/ProgressReports/Hydrology.pdf>
- Stamm, C., Fluhler, H., Gächter, R., Luenberger, J. and Wunderli, H. (1998). Preferential transport of phosphorus in drained grassland soils. *Journal of Environmental Quality*, **27**, 515–522.
- Stewart, J. W. B. and Tiessen, H. (1987). Dynamics of soil organic phosphorus. *Biogeochemistry*, **4**, 41–60.
- Stevens, R. J. (1979). Evaluation of an enzymatic method for orthophosphate determination in freshwaters. *Water Research*, **13**, 767–770
- Stone, M. and Murdoch, A. (1989). The effect of particle size, chemistry and mineralogy of river sediments on phosphate adsorption. *Environmental Technology Letters*, **10**, 501–510.
- Stout, W. L., Sharpley, A. N. and Landa, J. (2000). Effectiveness of coal combustion by-products in controlling phosphorus export from soils. *Journal of Environmental Quality*, **29**, 1239–1244.
- Straškraba, M. (1996). Lake and reservoir management. *Verh. Internat. Verein. Limnol.*, **26**, 193–209.
- Stream Solute Workshop. (1990). Concepts and methods for assessing solute dynamics in stream ecosystems. *Journal of the North America Benthological Society*, **9**, 95–119.
- Tanner, C. C., Clayton, J. S. and Upsdell, M. P. (1995). Effect of loading rate and planting on treatment of dairy farm wastewaters in constructed wetlands-II. Removal of nitrogen and phosphorus. *Water Research*, **29**, 27–34.
- Tanner, C. C., Sukias, J. P. S. and Upsdell, M. P. (1998). Relationships between loading rates and pollutant removal during maturation of gravel-bed constructed wetlands. *Journal of Environmental Quality*, **27**, 448–458.
- Tanner, C. C., Nguyen, M. L. and Sukias, J. P. S. (2002). Last ditch effort to reduce nutrient export from drained dairy pastures using constructed wetlands, in Currie, L. D. and Loganathan, P. (eds.), *Proc. Dairy Farm Soil Management*, 13–14 February 2002, Fertilizer & Lime Research Centre, Massey University, Palmerston North, New Zealand, pp. 247–254.
- Triska, F. J., Duff, J. H. and Avanzino, R. J. (1993). Patterns of hydrological exchange and nutrient transformation in the hyporheic zone of a gravel-bottom stream: examining terrestrial-aquatic linkages. *Freshwater Biology*, **39**, 259.
- Turner, B. J., McKelvie, I. D. and Haygarth, P. M. (2002a). Characterisation of water-extractable soil organic phosphorus by phosphatase hydrolysis. *Soil Biology and Biochemistry*, **34**, 27–35.
- Turner, B. L., Paphazy, M. J., Haygarth, P. M. and McKelvie, I. D. (2002b). Inositol phosphates in the environment. *Philosophical Transactions of the Royal Society, London*, **357**, 449–469.
- U.S. Environmental Protection Agency. (1996). Environmental indicators of water quality in the United States. EPA 841-R-96-002. U.S. EPA, Office of Water (4503F), U.S. Govt. Printing Office, Washington, DC.
- U.S. Geological Survey. (1999). The quality of our nations' waters: Nutrients and pesticides. U.S. Geological Survey Circular 1225, 82 p. USGS Information Services, Denver, CO. Available at: <http://www.usgs.gov>
- Uusi-Kamppa, J., Turtola, E., Hartikainen, H. and Ylaranta, T. (1997). "The interactions of buffer zones and phosphorus runoff". in Haycock, N. E., Burt, T. P., Goulding, K. W. T. and Pinay, G. (eds.), *Buffer Zones: Their Processes and Potential in Water Protection*. Quest Environmental, Hertfordshire, UK, pp. 43–53.
- Vollenweider, R. A. (1968). Scientific fundamentals of the eutrophication of lakes and flowing waters, with particular reference to nitrogen and phosphorus as factors in eutrophication. Paris, Report for the Organisation for Economic Cooperation and Development. DAS/CSI/68.27, 192 pp.; Annex, 21 pp.; Bibliography, 61 pp.
- Vollenweider, R. A. (1990). Eutrophication: Conventional and non-conventional considerations and comments on selected topics. *Mem. Ist. Ital. Idrobiol.*, **47**, 77–134.
- Walling, D. E. (1999). Linking land use, erosion and sediment yields in river basins. *Hydrobiologica*, **410**, 223–240.
- Ward, R. C. (1984). On the response to precipitation of headwater streams in humid areas. *Journal of Hydrology*, **74**, 171–189.
- Watanabe, F. S. and Olsen, S. R. (1965). Test of an ascorbic acid method for determining phosphorus in water and NaHCO₃ extracts from soil. *Soil Science Society of America Proceedings*, **29**, 677–678.
- Welch, E. B. (1992). *Ecological effects of wastewater: applied limnology and pollutant effects*. Chapman and Hall, 425 p.
- Westerman, P. W. and Overcash, M. R. (1980). Short-term attenuation of runoff pollution potential for land-applied swine and poultry manure, in *Proc. "4th International Symposium on Livestock Wastes"*, April 1980, Amarillo, TX. in *Livestock waste – A renewable resource*. American Society of Agricultural Engineers, St. Joseph, MI, pp. 289–292

- Wetzel, R. G. (2001). *Limnology: lakes and river ecosystems*. Academic Press, San Deigo, CA, 1006 pp.
- Whithers, P. J. A., Clay, S. D. and Breeze, V. G. (2001). Phosphorus transfer in runoff following application of fertilizer, manure, and sewage sludge. *Journal of Environmental Quality*, **30**, 180–188.
- Whitton, B. A. (2000). Increases in nuisance macro-algae in rivers: a review. *Verh. Internat. Verein. Limnol.*, **27**, 1257–1259.
- Whitton, B. A., Grainger, S. J. L., Hawley, G. R. W. and Simon, J. W. (1991). Cell-bound and extracellular phosphatase activities of cyanobacterial isolates. *Microbial Ecology*, **21**, 85–98.
- Woltemade, C. J. (2000). Ability of restored wetlands to reduce nitrogen and phosphorus concentrations in agricultural drainage water. *Journal of Soil and Water Conservation*, **55**, 303–309.
- Young, E. O. and Ross, D. S. (2001). Phosphate release from seasonally flooded soils: A laboratory microcosm study. *Journal of Environmental Quality*, **30**, 91–101.
- Young, R. A., Onstad, C. A., Bosch, D. D. and Anderson, W. P. (1989). AGNPS: A nonpoint-source pollution model for evaluating agricultural watersheds. *Journal of Soil and Water Conservation*, **44**, 168–173.
- Zhao, S. L., Gupta, S. C., Huggins, D. R. and Moncrief, J. F. (2001). Tillage and nutrient source effects on surface and subsurface water quality at corn planting. *Journal of Environmental Quality*, **30**, 998–1008.