Modeling phosphorus transport in agricultural watersheds: Processes and possibilities

A.N. Sharpley, P.J.A. Kleinman, R.W. McDowell, M. Gitau, and R.B. Bryant

ABSTRACT: Modeling phosphorus (P) loss from agricultural watersheds is key to quantifying the long term water quality benefits of alternative best management practices. Scientists engaged in this endeavor struggle to represent processes controlling P transport at scales and time frames that are meaningful to farmers, resource managers, and policy makers. To help overcome these challenges, we reviewed salient issues facing scientists that model P transport, providing a conceptual framework from which process-based P transport models might be evaluated. Recent advances in quantifying the release of soil P to overland and subsurface flow show that extraction coefficients relating soil and flow P are variable but can be represented as a function of land cover or erosion. Existing information on best management effects on P export should be linked to watershed models to better represent changes in P transport. The main needs of P transport models are inclusion of flexible coefficients relating soil and overland flow P, fertilizer and manure management and P loss, stream channel effects on edge-of-field P losses prior to water body input, and linkage of watershed and water-body response models. However, it is essential that the most appropriate model be carefully selected, according to a user's needs in terms of available input data, level of predictive accuracy, and scale of simulation being considered.

Keywords: Agricultural runoff prediction, erosion, eutrophication, manure management, nonpoint source models, overland flow, subsurface flow, Total Maximum Daily Loads, water quality modeling, watersheds

Phosphorus (P), an essential nutrient for crop and animal production, can accelerate freshwater eutrophication (Carpenter et al. 1998; Sharpley 2000). Recently, the U.S. Environmental Protection Agency (USEPA 1996) identified eutrophication as the most ubiquitous water quality impairment in the United States, with agriculture being a major contributor of P (USGS 1999). Eutrophication restricts water use for fisheries, recreation, industry, and municipalities through the increased growth of undesirable algae and aquatic weeds and the oxygen depletion caused by their death and decomposition. An increasing number of surface waters have experienced periodic and massive harmful algal blooms (e.g., *Cyanobacteria* and *Pfiesteria*), which contribute to summer fish

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kills, unpalatability of drinking water, and the formation of carcinogens during water chlorination, and that are linked to neurological impairment in humans (Burkholder and Glasgow 1997; Kotak et al. 1993).

In response to these concerns, the USEPA has begun to enforce section 303(d) of the 1972 Clean Water Act, requiring states to develop lists of impaired waters that do not meet designated water quality standards. As part of this process, Total Maximum Daily Loads (TMDLs) must be assigned to impaired waters. A TMDL specifies the maximum amount of a pollutant that a water body can receive and still meet water quality standards. Pollutant loadings are estimated for point and nonpoint sources of pollution plus some margin of safety (i.e., consideration of seasonal variations and allowance for reasonably foreseeable increases in pollutant load), and used to assign responsibility for reducing pollutant loads within a watershed. For nutrients such as P, implementation of the TMDL process is based upon first combining watershed data with process-based models to allocate nutrient loads to specific land uses, then forecasting the effect of alternative management practices on watershed P loadings, and finally requiring sufficient changes in management to meet established TMDLs in an impaired watershed (USEPA 2000).

Given general environmental concerns and regulatory pressure to reduce P loadings to surface waters, much research is now focused on better understanding factors controlling P loss from agricultural watersheds. Because of the time and expense involved in the field assessment of management impacts on P loss, models often represent a more efficient and feasible means of evaluating management alternatives. Numerous process-based models have been developed to simulate the fate of P in soil and its transport to surface waters. In their most comprehensive form, such models integrate information over a large scale, helping to define watershed scale processes relevant to P transport, highlighting appropriate best management practices (BMPs), and identifying critical source areas where BMPs are most likely to affect watershed-scale P losses.

The Agricultural Nonpoint Pollution Source (AGNPS; Young et al. 1989, 1995) model was originally developed to provide estimates of runoff water quality from watersheds of up to 20,000 ha (50,000 ac) and to quantify the effects of BMPs targeted to specific areas. To make model output more meaningful to decision makers such as conservationists and farmers, AGNPS, which ran on a storm or flow event basis, was recently superseded by an annualized version, AnnAGNPS (Croshley and Theurer 1998). The model operates on a cell basis that makes it possible to analyze spatially discrete management units (fields) within a watershed, thereby enabling identification of individual fields that may serve as critical source areas of nutrient export.

The Soil and Water Assessment Tool (SWAT) model was developed to assess the impact of land management on water quality in watersheds and large river basins (Arnold et al. 1998). The model runs on a continuous time step and is currently being used in a variety of large-scale studies to estimate the off-site impacts of climate and management on water use and nonpoint source loadings.

Other process-based nutrient transport models include but are not limited to Areal Nonpoint Source Watershed Environment Response Simulation—2000 (ANSWERS-2000; Beasley et al. 1985; Bouraoui and Dillaha 1996); Guelph Model for Evaluating the Effects of Agricultural Management Systems on Erosion and Sedimentation (GAMES; Cook et al. 1985); Hydrologic Simulation Program - Fortran (HSPF; Johanson et al. 1984); Agricultural Runoff Model (ARM; Donigian et al. 1977); Erosion-Productivity Impact Calculator (EPIC; Sharpley and Williams 1990); and the lumped parameter model Generalized Watersheds Loading Functions (GWLF; Haith and Shoemaker 1987). For more detailed information on these models and their approaches, reviews by Hook (1997), Leavesley et al. (1990), National Research Council (2000a), and Rose et al. (1990) are available.

Export coefficient models have also been widely used to predict P loading of receiving water bodies (Beaulac and Reckhow 1982; Hanrahan et al. 2001; Johnes et al. 1996). Export coefficients define P loss from a particular source or land use in a watershed, and are usually derived from actual field measured losses of P (Johnes 1996; Johnes and Heathwaite 1997). The models calculate watershed export of P as the sum of individual loads from each source in the watershed. This approach accounts for the complexity of land-use systems and the spatial distribution of data from various sources (point and nonpoint). As export coefficients are empirical, these types of models are as accurate as input data (as are process-based models) (Hanrahan et al. 2001). Coefficients derived from short term or infrequent (i.e., monthly) monitoring of small drainage areas, however, can contribute to predictive variability (Lathrop et al. 1998).

A common limitation of model application is the lack of detailed parameterization data on soil physical, chemical, and biological properties as well as on crop and tillage information. To compensate for this limitation, existing databases are increasingly being linked to nonpoint source models, often via Geographical Information Systems (GIS) (Arnold et al. 1998; National Research Council 2000a). Key input data for nutrient transport models generally involve land use, soil texture, topography, and management practices. Once these data are in digital form, GIS techniques can be used to combine them with experimental or model results to extrapolate other properties needed for model application. In addition to regional assessments, this approach can be used to make comparative studies of the effectiveness of different remedial measures. Using mathematical models to calculate typical P transport values over a wide range of soil textures, slopes, and crops can serve as a quick and inexpensive way of making these assessments.

In order to identify critical sources of P export within larger areas, however, data obtained from experimental plots, field and hill slope monitoring, and models must often be extrapolated to broader scales. The accuracy of such "scaled-up" estimates depends greatly on how processes occurring at finer scales of spatial resolution relate to processes governing P transport at watershed and even more general scales. Although scaled models may enable process-based prediction of P transfers from readily available data, and, thus be used to assess to field and farm scale management practices, one must remember that system function may be scale- and time-dependent. For example, soil particle detachment, deposition, and resuspension, along with relative amounts of overland and subsurface flow, will differ between plot and hillslope scales and may contribute to scale dependent P losses.

In this paper we will discuss:

a. existing process-based formulations for soil P release and transport in overland and subsurface flow;
b. the effect of mineral fertilizer and manure management on P transport in

overland and subsurface flow;

c. phosphorus loss relative to landscape position of P source and transport pathways;
d. use of existing information and databases via GIS to facilitate parameter capture and scale up from plot to field to watershed;
e. the possibilities for nonpoint source modeling in defining future BMPs that minimize P export; and
f. how channel processes affect watershed export of P and the resulting impact on the receiving water body.

Hydrologic processes affecting P transport are examined in detail by Gburek et al. (this issue) and will not be addressed in detail here.

Soil Phosphorus Release and Transport

Dissolved P. Most nonpoint source models simulate dissolved P transport in overland flow as a function of the extractability of P in the surface 5 cm (2 in) of soil (e.g., AnnAGNPS, EPIC, SWAT). This can be represented by:

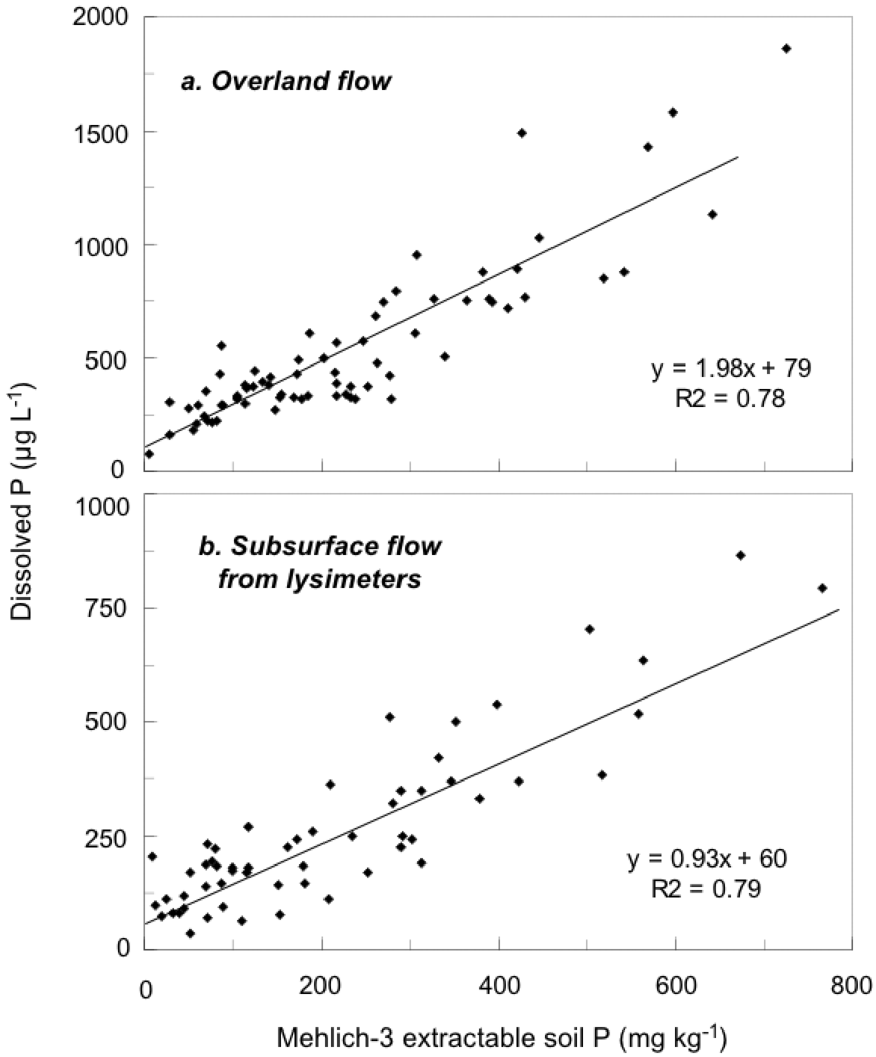
Dissolved P = extraction coefficient * available soil P * overland flow volume (1)

where dissolved P is orthophosphate loss in overland flow (kg ha^{-1}), available soil P is the amount of P in a unit depth of surface soil (usually 5 cm) (2 in) as estimated by recommended soil test P (STP) methods (kg ha^{-1} 5 cm^{-1}) (lb ac^{-1} 2 in^{-1}), and extraction coefficient is the fraction of STP that can be released to overland flow for a given flow event volume (cm).

For modeling purposes, extraction coefficients can be determined as the slope of the linear regression of STP and overland flow dissolved P (Figure 1a). Although some research has described this relationship as either curvilinear (Lory et al. 2001) or consisting of two linear regressions intersecting at a soil P threshold above which P release is greater (Kleinman et al. 2000; McDowell and Condron 1999; McDowell and Sharpley 2001a), these options are only sometimes statistically better (i.e., have higher regression coefficients) than the single linear relationship reported by Pote et al. (1999) and Sharpley et al. (1996).

A similar relationship holds for subsurface flow P and surface STP, although the slope of the relationship (0.93) is almost half that for overland flow (slope of 1.98) (Figure 1b).

Figure 1
Relationship between the concentration of dissolved P in overland (a) and subsurface flow (b) from 30 cm deep lysimeters and the Mehlich-3 extractable soil P concentration of surface soil (0-5 cm) from a central PA watershed (adapted from McDowell and Sharpley, 2001a and Sharpley et al., 2001).



The dependence of dissolved P transport on subsurface and overland flow suggests the importance of preferential flow pathways, such as earthworm burrows and old root channels, in the downward movement of P through the soil profile (Kleinman et al. 2003; McDowell and Sharpley 2001a; Sims et al. 1998). Other studies have found a similar relationship between surface soil P and P loss in subsurface flow. For example, Heckrath et al. (1995) found that STP (Olsen P) >60 mg kg^{-1} (> 60 ppm) in the plow layer of a silt loam caused the dissolved P concentration in tile drainage water to increase dramatically (150 to 275 µg L^{-1}) (150 to 275 ppb). They postulated that above this concentration, which exceeds that needed by major crops for optimum yield (about 20 mg kg^{-1} (20 ppm); Ministry of Agriculture, Food and Fisheries 1994), the potential for subsurface P movement in drained lands greatly increases.

Most models use a constant extraction coefficient value, assuming that STP extractability is similar among soils. A re-analysis of P transport in relation to STP (as Mehlich-3 P) for several Oklahoma watersheds revealed a range of extraction coefficient values (Figure 2). These watersheds had a dominant soil type of Kirkland silt loam but were managed differently, either as

Figure 2
Relationship between Mehlich-3 extractable soil P and the concentration of dissolved P in overland flow from cropped and grassed fields in Oklahoma (data adapted from Sharpley et al., 1991 and Smith et al., 1991).

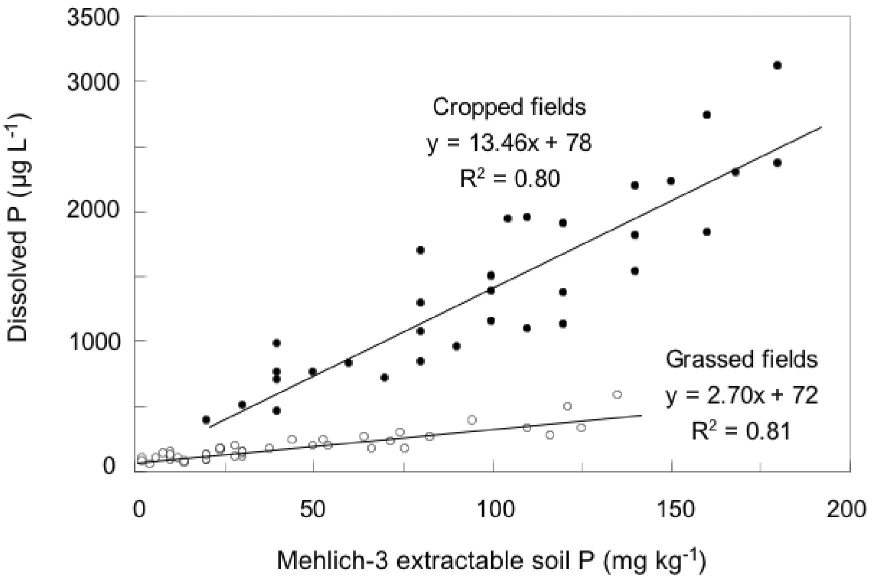
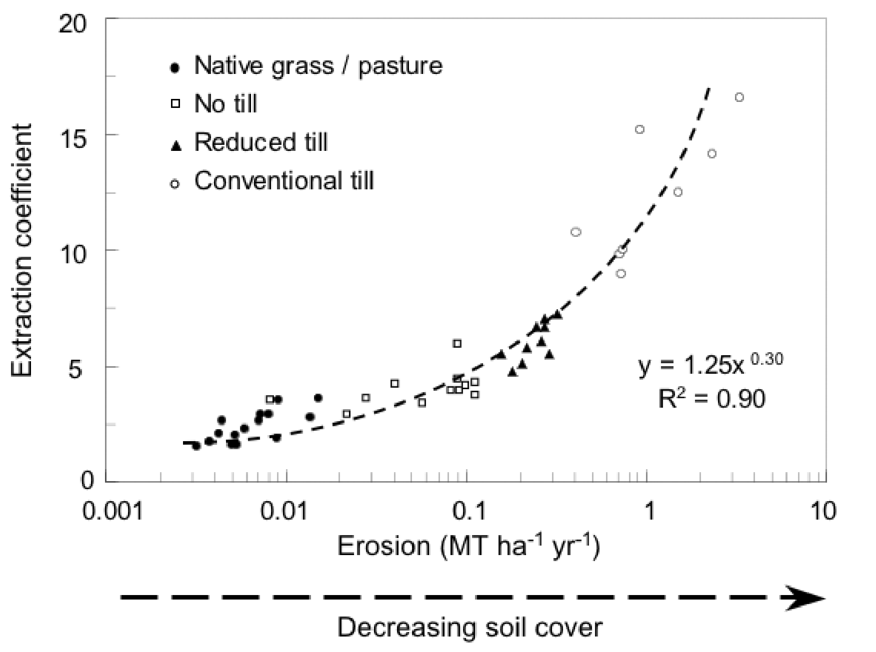


Figure 3
Extraction coefficient (slope of the relationship between soil test P and dissolved P in overland flow) as a function of erosion to represent soil vegetative cover for sites in Arkansas, Oklahoma, New York, and Pennsylvania (data adapted from Pote et al., 1999; McDowell and Sharpley, 2001a;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



native grass or cultivated (moldboard plow) wheat. Extraction coefficients were much greater for cropped (13.46) than grassed watersheds (2.70). This difference can be attributed to a lower degree of interaction between surface soil and overland flow with a protective grass cover than for a cropped situation, where the soil is more exposed to overland flow. This finding is supported by several studies by Sharpley et al. (1996), which found that specific regression equations between STP and overland flow P varied with soil type and management. For instance, regression slopes were flatter for grass (4.1 to 7.0, mean 6.0) than for cultivated land (8.3 to 12.5, mean 10.5). However, regression slopes were too variable to allow the use of a single or average relationship between STP and overland flow P for all soils under the same management, due to the inherent variability between soils and the soil-specific nature of soil P release to overland flow. Factors that influence P release among soils include the dominant forms of P in soil, texture, aggregate diffusion, degree of interaction between soil and water, organic matter content, vegetative soil cover, and P sorption capacities (Sharpley 1983; 1999).

We undertook a more detailed evaluation of land use affects on the relationship between STP and overland flow dissolved P by re-analyzing data published by McDowell and Sharpley (2001a), Pote et al. (1999), and Sharpley and Smith (1994). Using erosion as a factor approximating land cover, the extraction coefficient (slope of the linear relationship between STP and dissolved P in overland flow) increased with greater erosion or reduced soil cover (Figure 3). A larger soil P extraction coefficient represents a greater release of P as dissolved P in overland flow per unit of STP increase. With more erosion or decreased soil cover there is a greater interaction between soil and overland flow (Sharpley 1985b), and thereby soil P release (Figure 3). Erosion is represented on a logarithmic scale in Figure 3 to more clearly show the range in values. On a normal (nonlogarithmic) scale, however, extraction coefficient values plateau between 15 and 20 (not shown). Although most models currently use a fixed soil P extraction coefficient, this assumption is clearly not the case. The effect of land management on soil P release and overland flow dissolved P may be more accurately represented or approximated as a function of erosion (Figure 3).

Particulate P. As the sources of particulate P in overland flow and stream include eroding surface soil, stream banks, and channel beds, processes determining erosion also control particulate P transport. In general, eroded particulate material is enriched with P compared to source surface soil, due to the preferential transport of finer (i.e., clay-sized), more sorptive soil and organic particles or greater P content than coarser inorganic particles (i.e., sand-sized). Sharpley (1985a) found that the plant available P content of sediment in overland flow was on average three times greater (or more enriched) than that of source soil and 1.5 times greater for total, inorganic, and organic P. The degree of P enrichment is expressed as a P enrichment ratio (PER): the P concentration of discharged sediment divided by that of source soil. After assembling enrichment ratio information for nonpoint source modeling, Menzel (1980) concluded that for particulate P, a logarithmic relationship as in Equation 2 seemed to hold for a wide range of soil vegetative conditions:

$$\text{Ln PER} = 2.00 - 0.16 \text{ Ln sediment discharge} \quad (2)$$

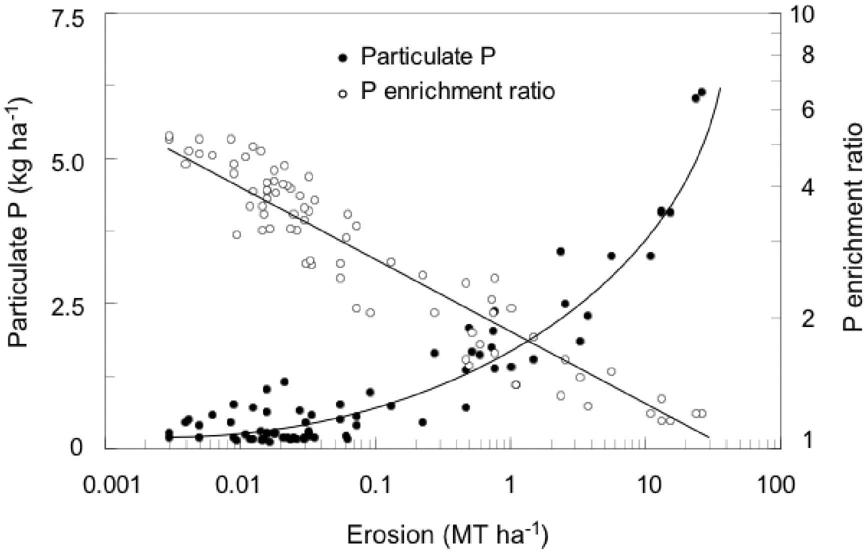
where the units of sediment discharge are kg ha^{-1} (lb ac^{-1}). Most nonpoint source models adopted this approach to predict particulate P transport in overland flow. This relationship is based on the well-documented increase in particulate P loss as erosion increases (Figure 4). Based on the total P concentrations of source soils for each of the watersheds represented in Figure 4, PER decreases with an increase in erosion. As erosion increases, there is less particle-size sorting by overland flow, less clay-sized particles are transported in proportion to total soil loss, and P enrichment, thus, decreases (Figure 4).

Once an appropriate PER is obtained from sediment discharge, particulate P loss can be calculated as:

$$\text{Particulate P} = \text{total soil P} * \text{sediment concentration} * \text{PER} * \text{overland flow volume} \quad (3)$$

where particulate P is loss in overland flow (kg ha^{-1}), total soil P is the amount in a unit depth of surface soil (usually kg ha^{-1} 5 cm^{-1}) (lb ac^{-1} 2 in^{-1}), sediment concentration is g sediment L^{-1} overland flow, and PER is calculated from Equation 2, for a given flow event volume (cm).

Figure 4
Particulate P loss and enrichment ratio of eroded sediment as a function of erosion in overland flow from watersheds at El Reno, OK (adapted from Sharpley et al., 1991 and Smith et al., 1991).



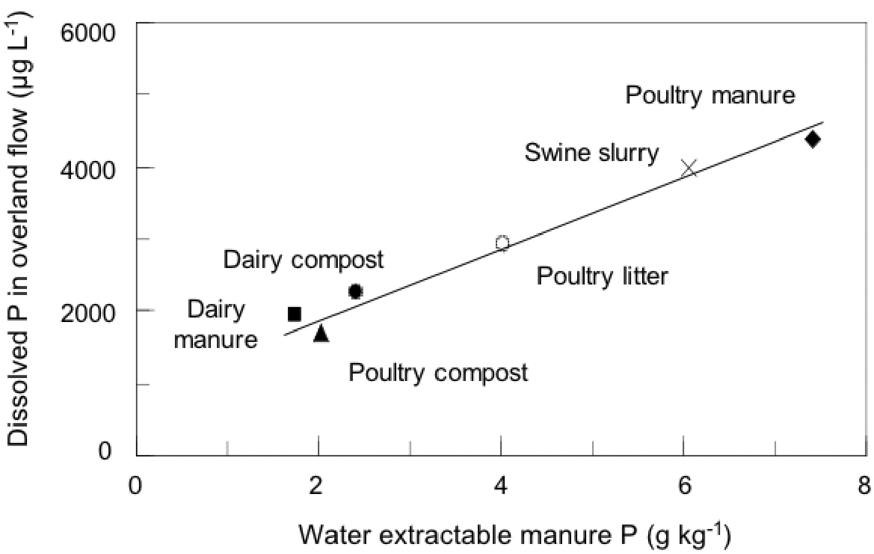
The remainder of this discussion describes how land management, landscape, and stream channel processes affect this predictable loss of P as dissolved, particulate, and total P, and in turn how this may be addressed in simulation models.

Fertilizer and Manure Management

Management of fertilizer and manure, as it affects P availability to overland flow over the short term, can profoundly affect prediction of P transport in overland flow. While soil P represents a source of P enrichment in overland flow, the application of fertilizer and manure to soil, including form, method, timing, and rate of P application, can temporarily overwhelm relationships derived between STP and P in overland flow (Sharpley and Tunney 2000). As such, accounting for fertilizer and manure management in P transport models is essential to their accuracy. However, most models do not directly address the effect of applied P (particularly manure) on P transport in overland flow. Instead, they incorporate added P into the soil P pool and adjust or recalibrate the extraction coefficient. Thus, P transport in overland flow as affected by the amount, form, method, and time after applying P is, in general, poorly represented and underpredicted. The following discussion shows how fertilizer and manure management effects on overland flow P are predictable.

Form of P applied. Mineral fertilizer and manure represent concentrated sources of soluble P that can greatly increase dissolved P losses in overland flow. Consequently, the concentration of soluble P in these sources may provide effective predictions, depending upon the solubility of the P source, method of application, rate of application, and timing of application relative to the overland flow event. Figure 5, adapted from Kleinman et al. (2002), illustrates the relationship between dissolved P concentration in overland flow and water extractable P concentration of several manures, composts, and diammonium phosphate, all broadcast on a Hagerstown soil (fine, mixed, semiactive, mesic Typic Hapludalf) at a rate of 100 kg total P ha^{-1} (90 lb total P ac^{-1}). Application of P sources increased dissolved P concentrations in overland flow 4 - 26 times that observed from unamended soil and shifted the partitioning of P in overland flow from primarily particulate P (particulate P was greater than 90% of total P for the unamended soil) to primarily dissolved P (dissolved P was greater than 60% of total P for the amended soils). Notably, the correlation between water extractable P and overland flow dissolved P concentration is linear for manures, but appears to plateau when mineral fertilizer is used. The cause of this plateau likely reflects differences in the physical properties of the manures and the mineral fertilizer. As the manures protected the surface soil from erosion to a certain

Figure 5
Relationship between water extractable manure P and the dissolved P in overland flow one week after manure or mineral fertilizer was broadcast (100 kg total P ha^{-1}) on a Hagerstown silt loam soil (7 cm hr^{-1} rainfall for 30 minutes) (adapted from Kleinman et al., 2002).



extent (see also McDowell and Sharpley 2001b), soil loss was greater from the fertilizer (1.2 g L^{-1})(1200 ppm) than the three manure treatments (average = 0.6 g L^{-1}) (600 ppm) (Kleinman et al., 2002). This would increase the extent to which soluble P from the fertilizer was sorbed by eroded sediments (Sharpley et al., 1981).

Method of P application. Equally important as the source of P for P availability to overland flow is the method by which that source is applied to soil (Mueller et al. 1984; Romkens et al. 1973; Zhao et al. 2001). Surface application of manure and mineral fertilizer concentrates P at the extreme soil surface, where it is vulnerable to removal by overland flow (Eghball and Gilley 1999; Sharpley et al. 1984). Sharpley (1985b) reported an effective depth of interaction (EDI) between overland flow and soil P of 1.3 to 37.4 mm (0.05 to 1.47 in), depending upon rainfall intensity and slope gradient. As a result, surface placement may greatly increase dissolved P losses. Injection, knifing, and immediate incorporation by cultivation remove manure and fertilizer from the EDI zone, but, in the case of cultivation, may also result in increased vulnerability to particulate P losses due to increased erosion potential (Andraski et al. 1985; Romkens et al. 1973).

Figure 6, adapted from Kleinman et al. (2002), illustrates the relationship between total P concentration in overland flow and suspended solids for soils receiving broadcast or incorporated manure and mineral fertilizer. Clearly, erosion as measured by suspended solids provides effective prediction of total P under incorporated (cultivated) conditions. This relationship is obscured by broadcasting, as water soluble P from manure and mineral fertilizer overwhelm soil P contributions to runoff (Sharpley and Tunney 2000). Notably, total P concentrations were significantly higher when P was surface applied rather than incorporated.

Timing of P application. The timing of P application relative to the occurrence of an overland flow event modifies the effect of P source and application method on P concentrations in the overland flow (Sharpley 1997; Westerman and Overcash 1980). Immediately following application of a P source, the potential for P loss peaks and then declines over time, as applied P increasingly interacts with the soil and is converted from soluble to increasingly recalcitrant forms (Edwards and Daniel 1993). Sharpley and Syers (1979) reported declining dissolved P (from > 250 to < 100 μg L^{-1}) (> 250 to < 100 ppb) and total P concentrations (from > 700 to 100 μg L^{-1}) (> 700 to 100 ppb) in tile drainage over one month following temporary, intensive grazing of paddocks by dairy cattle. Similarly, Gascho et al. (1998) observed exponential declines in dissolved P concentrations in overland flow (from > 5000 to < 1000 μg L^{-1}) (> 5000 to < 1000 ppb) roughly one month after mineral fertilizer application.

Figure 7, adapted from Kleinman and Sharpley (2003), shows the interaction between P application rate and timing of application relative to the overland flow event and the dissolved P concentrations. Two manures (dairy and poultry) with varying concentrations of water soluble P (819 mg kg^{-1} and 3129 mg kg^{-1}, respectively) (819 and 3129 ppb) were broadcast at 10, 25, 50, 75, 100, and 150 kg total P ha^{-1} (or 9, 22, 45, 67, 90, and 134 lb total P ac^{-1}) to a Lewbeach soil (coarse-loamy, mixed, semiactive, frigid Typic Fragiudept). Overland flow dissolved P concentration is linearly correlated with the rate of total P addition for both manures, with the magnitude modified by the concentration of water soluble P in the manure. Timing of manure application and overland flow dissolved P concentration is consistently described by exponential decay.

In addition to affecting the availability of P to overland flow, manure and mineral fertilizer P sources may directly impact soil physical properties that control runoff and erosion. Over the short term, surface application of manure, particularly at high loading rates, may increase soil cover, protecting it from raindrop impact and aggregate dispersion (Barthès et al. 1999; McDowell and Sharpley 2002). Over the long term, addition of manure may increase soil organic matter levels, which in turn affect porosity, aggregate stability, and infiltration, all of which are factors that affect runoff and erosion potential (Gilley and Risse 2000; Oades and Waters 1991; Rousseva 1989).

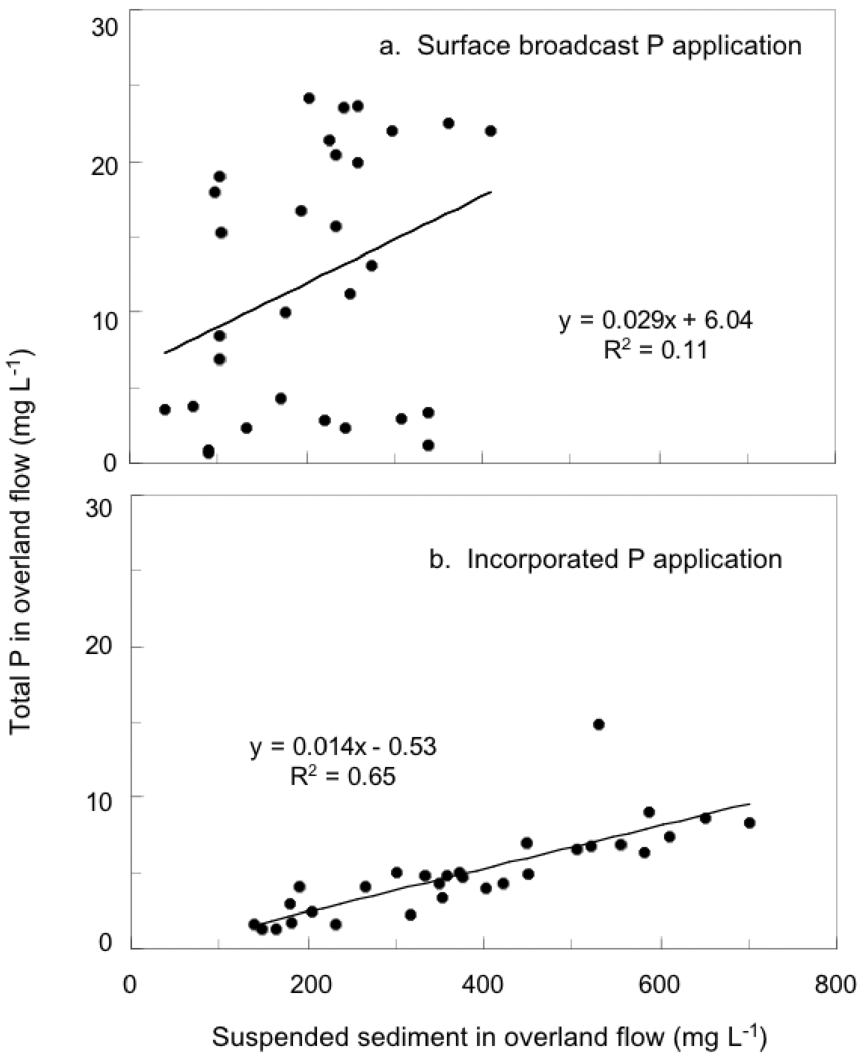
Landscape Position

The potential for P loss within the landscape is a function of erosion, soil P concentration (e.g., the addition of manure), and management (e.g., cropping). The nature of the erosion process means that finer-sized particles, which also contain much more P than coarse-sized particles, are preferentially eroded. 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 with coarse particles (Maguire et al. 1998; McDowell and Sharpley 2002). Once P is in solution, the transition between dissolved and particulate forms during overland flow can change, mediated by the sorption and desorption properties of the sediments. As P has a strong affinity to react with sediments, the sorption and desorption of P to and from sediments are two of the main processes that regulate the behavior and concentration of P in fresh waters. Much work has shown that sediments can remove or release P to the overlying water, depending on factors such as the affinity and saturation of sediment P sorbing sites and the kinetics of exchange (House et al. 1995; Sharpley et al. 1981).

To investigate the influence of varying concentrated P sources along a simulated hillslope length, McDowell and Sharpley (2002) studied P loss from soils treated with manure at 75 kg P ha^{-1} (or 67 lb P ac^{-1}) at various positions upslope. Dissolved reactive P concentration was more closely related to the proportion of clay in sediment of overland flow before (r = 0.98) than after (r = 0.56) manure application. This was attributed to the transport of larger, low-density particles as flocculates after applying manure. Surprisingly, the concentration of dissolved and particulate P 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. Supplementary evidence by McDowell et al. (2001b), using ^{33}P suggests that the 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, total P loss (mainly as particulate P 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 total P loads and should not be discounted in studying areas of potential P loss within a watershed. Furthermore, when simulating watershed export of P, landscape position must be considered in linking source and transport processes controlling P loss.

Figure 6

Relationship between the concentration of suspended sediment and total P in overland flow for surface broadcast (a) and incorporated (b) manure or fertilizer applications (100 ka total P ha^{-1}) to a Hagerstown silt loam soil (7 cm hr^{-1}) rainfall for 30 minutes) (adapted from Kleinman et al., 2002).



Spatial Data Requirements for Modeling

Models that assess nonpoint sources of P loss from agricultural lands rely on spatial data as input. Land use, soil properties, and topographic data that include stream locations and watershed boundaries are commonly required inputs. The relationship between model characteristics and spatial data requirements can be visualized in the context of the scale diagram (Figure 8) as originally proposed by Hoosbeek and Bryant (1993). The watershed scale rests high upon the axis of "Organizational Hierarchy," covering a relatively large geographical area, whereas much of the research on soil P interactions and processes is at the soil horizon (i-1) or pedon/plot scale (i-0) level. For soil and most other data layers, model resolution (related to the geographical scale at which watershed processes are to be represented) greatly affects the size of associated spatial databases. For example, the number of soil delineations on a map at a scale of 1:250,000 (State Soil Geographic Database (STATSGO) data) increase when the same landscape area is mapped at a scale of 1:25,000 (Soil Survey Geographic Database (SSURGO) data), because of the greater detail depicted. This change of scale also dramatically increases the size of soil property databases, which must now include information on all individual soils that were previously grouped into associations. However, at some point, the relationship between scale and database size is of lesser concern. In a scale range from 1:24,000 to 1:12,000, the amount of detail is not greatly affected because the size of naturally occurring landforms does not change

Figure 7
Interaction of poultry manure application rate and timing of overland flow event on dissolved P in overland flow on a Lewbeach silt loam soil (7 cm hr-1 rainfall for 30 minutes) (adapted from Kleinman and Sharpley, 2002).

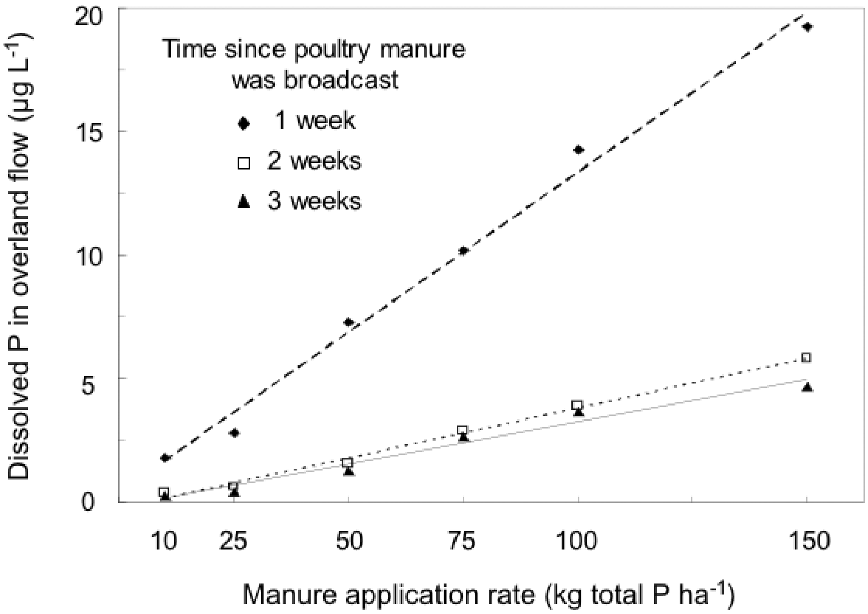
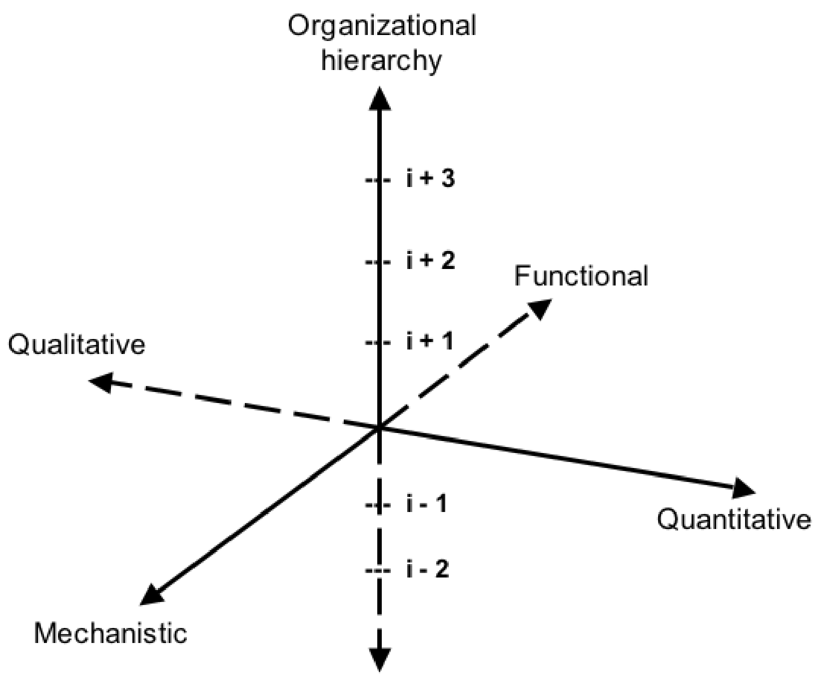


Figure 8
The relationship between model characteristics and spatial data requirements in the context of a scale diagram (adapted from Hoosbeek and Bryant, 1993).



(Hudson and Culver 1994). Similarly, if boundaries of individual fields can be mapped at a scale of 1:24,000, a database at a scale of 1:12,000 will show the same number of fields.

Data and parameter requirements also increase rapidly as models become more mechanistic to better represent the physical and chemical processes involved in P loss. Every new variable introduced for the purpose of better representing a process requires a populated field in the database. The net result of this move to process-based representation of management effects on P loss is that, as watershed size, level of mechanistic detail, and spatial data resolution increase, then databases, model interactions, and run times become unmanageable.

The complexity of managing these large databases in support of a watershed model, therefore, limits the degree of spatial resolution of existing models, even in cases where spatial databases exist at higher resolutions. Spatial parameters are frequently "lumped" so that units having similar soil, land use, and topographic characteristics respond the same to driving variables, such as those used to simulate runoff generation. Spatially lumped parameters can pose a problem when responses from lumped units cannot distinguish between relative spatial locations of individual units. As we showed earlier, the spatial location of P sources and transport pathways in relation to each other and to the stream channel is a critical factor in determining P export from a watershed to a water body.

Whether or not a model is directly linked to GIS, these systems are generally used to acquire, summarize, and provide spatial data input. The development of GIS databases has seen rapid advances in the last decade, but data requirements for process-based, watershed-scale modeling can stretch the limits of spatial data availability. Digital elevation models (DEMs) are generally available at resolutions that provide adequate topographic information for modeling large watersheds of moderate to high relief. But the resolution of commonly available DEMs may not provide adequate topographic data for watersheds of low relief or for detailed field scale modeling.

The STATSGO digital soil survey database (1:250,000) is universally available for the United States, but map units are comprised of soil associations that effectively lump soils that may have very different runoff generation and soil P characteristics. The more detailed SSURGO database (1:25,000) has limited coverage, and its survey areas correspond to political boundaries. Watersheds frequently cross political boundaries, thus decreasing the probability that SSURGO data are available for complete coverage of a particular watershed of interest. There is also limited availability of land use data, which can be time sensitive in accordance with the rate of land use changes. Recent coverage of remotely sensed data are readily available for most areas of interest, but land use interpretations derived from remotely sensed data are time consuming and expensive. If future models are to be useful and effective, modelers should consider the availability of spatial data when choosing data requirements.

To overcome the spatial data limitations thus far identified, a nested modeling approach is recommended. Field and farm scale models that incorporate the knowledge of P source and transport processes involved in P loss can be supported with highly detailed spatial databases that are already available in some areas or could be easily developed in others. Results and generalizations from these models could be aggregated to represent sub-basins in a simpler, less mechanistic model that requires lower spatial resolution, but draws on relationships derived from the results of more process-based modeling at the larger, more-detailed scale. Similarly, results from sub-basin models could be further aggregated to represent whole watersheds, several hundreds of square kilometers in size. Then, with enough data on the processes operating in individual component sub-watersheds, we could use the principles of mapping to derive generalizations about large watersheds that span multiple physiographic regions, such as the Chesapeake Bay Watershed and Mississippi River Basin. Map units of the Major Land Resource Areas (MLRAs) of the United States are defined on the basis of topography, soils, and land use, and therefore are ideally suited for extrapolating detailed studies of whole watersheds to the broader area of the MLRA map unit.

Defining Future Best Management Practices

The implementation of P control measures has often been carried out with insufficient knowledge as to how well-suited the practices are to controlling P loss in a given situation. A large number of BMPs exist; their suitability varies according to the particular situation. Given that BMP impacts are largely site-specific (Baker and Johnson 1983; Deere and Company 1995; USEPA 1993), defining future BMPs for P control depends a great deal on being able to establish the effectiveness of these BMPs under the variety of field conditions that are constantly encountered.

Watershed monitoring studies have been used to assess the effectiveness of BMPs in decreasing P export. However, there has been a recent increase in model-based BMP assessments, such as those by Hamlett and Epp (1994), Mostaghimi et al. (1997), Osei et al. (2000), Phillips et al. (1993), and Walter et al. (2001). Various process-based models exist that can be used to describe the response of land areas and pollutant movement to various BMPs. These include the AnnAGNPS, ANSWERS-2000, and SWAT models described earlier. Another model that can be configured for BMP assessment is GWLF (Haith and Shoemaker 1987; as modified by Schneiderman et al. 1998).

There are several factors that complicate BMP assessment in a watershed situation, including:

- large range of topographic and soil conditions,
- the limited numbers of BMPs or BMP combinations that may be studied at once without confounding the study,
- issues of replication, especially with BMPs that require large areas and are costly to implement,
- the lengthy time periods that may be required to adequately describe the functioning of BMPs (Baker and Johnson 1983), and
- the need to rely on natural rainfall (where rain simulator studies may not be applicable).

In turning to models, we try to eliminate some of these complications. While it is recognized that models greatly simplify the natural system, they do provide a means of carrying out complex BMP evaluations. Model outputs should be adequate when a comparative analysis rather than an absolute value is desired (Novotny and Olem 1994).

The large amounts of data that have accumulated over the years can be extremely useful in working on a modeling approach to BMP evaluation (Gitau et al. 2001). Integration of BMP literature studies with modeling is schematized in Figure 9. An initial step in modeling BMPs would be the characterization of the BMPs of concern with regard to mechanisms of operation, such as source (soil P and form, rate, and method of P applied) and transport (runoff and erosion) factors controlling P loss. This characterization would enable the identification of source and transport mechanisms impacted by particular BMPs, and thus the determination of model changes that would be necessary to fully represent the BMPs (Gitau et al. 2001).

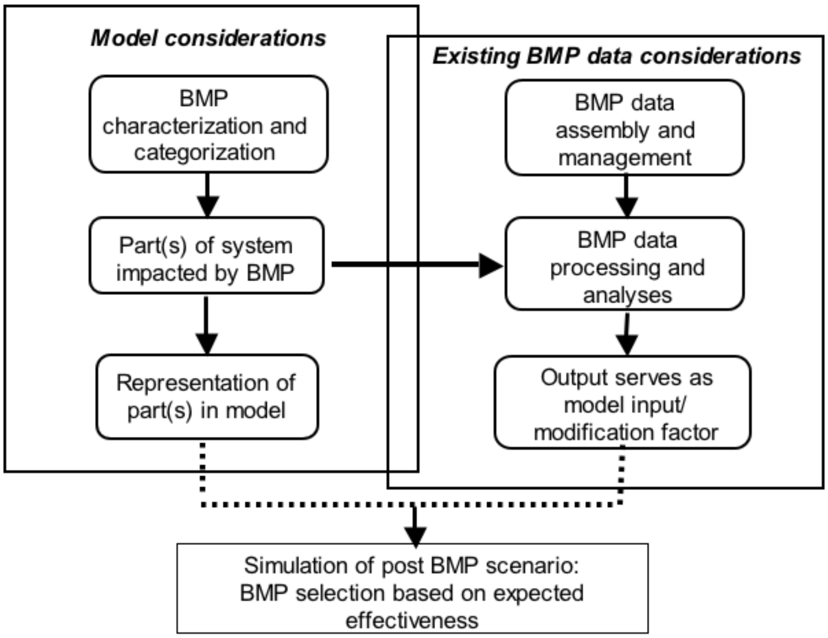
In integrating system characterization with BMP studies, there is the need to identify, assemble, and analyze all available data on BMP effectiveness, such as in the ongoing effort described by Gitau et al. (2001). This effort involves the development of an interactive BMP database, from which analyzed data can be extracted. An example of the type of information on BMP effectiveness that can be gleaned from such a database is shown in Figure 10. Clearly, there is a wide range in dissolved P reduction after BMP implementation that is influenced by site-specific factors and weather (Figure 10). Still, general trends in efficiency are apparent, with total P losses being much reduced, although dissolved P loss usually increases after adoption of conservation tillage, due to a build up of surface soil P (Figure 10 and McDowell and McGregor 1984; Sharpley and Smith 1994). Similar reduction efficiencies are also developed for particulate P losses with BMP implementation (Gitau et al. 2001). Outputs from data analyses provide values that can be used as model inputs or modification factors, which enables simulation of post-BMP scenarios, thereby providing a basis for BMP selection. This is especially useful when the model is not configured to accept direct input changes in describing particular BMPs.

In general, modeling offers a rapid, efficient, and low cost alternative to field assessment of BMP effectiveness, thus facilitating BMP selection and the move towards identifying suitable BMPs for P loss mitigation.

Transport and Impact of P in Surface Waters

In-channel processes modify the potential for agriculture to impact a downstream freshwater body. Surface water impacts drive activities such as TMDL development, so understanding in-channel P transport processes and the impact of transported P on downstream water bodies is necessary to link upstream changes in agricultural management with downstream

Figure 9
Integrating modeling with existing information on Best Management Practices to better predict their effectiveness in minimizing P export from agricultural watersheds.



water quality impacts.

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 and the dilution and hydraulics of flowing water. In-channel processes include those that control the availability of P to flowing waters and those that define where that availability is expressed.

In fluvial systems with good hydraulic mixing (such as shallow flowing streams), the availability of P in stream sediments can be estimated by the equilibrium P concentration (EPC_0) at zero net sorption or desorption. 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 a given site. Here, the EPC_0 will have a major influence on the concentration of P in solution: P will desorb from sediments if its concentration 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).

Variability in sediment EPC_0 along a stream or river channel is characteristically high, reflecting physical hydraulic processes, management of land adjacent to the stream, form of P occurring in the sediment, and biotic processes. Biologically mediated P release during the decomposition of organic matter in sediments can be an important source of dissolved P at times of high temperature and low flow at upstream sites with organic-rich sediments (Klotz 1991). However, under most other conditions (non-base flow, mineral sediments), when P loads are expected to be high, biotic processes have little effect on EPC_0 and stream flow P, and abiotic processes dominate. 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. 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.

Under conditions of high flow (i.e., storm flow), P in stream flow does not equilibrate with sediment EPC_0 and is a function of the kinetics of P release. 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) (McDowell and Sharpley 2002; Stone and Murdoch 1989). Thus, hydrologic processes that control sediment particle size distribution have important implications for P delivery and fate in river systems. In a recent review of land use and sediment yield, Walling (1999) indicated that fluvial systems have a 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 channel processes have on P transport in relatively small watersheds, McDowell et al. (2001a) 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 (100 ac) second-order agricultural watershed. Base flow dissolved P concentrations were greater at the watershed outflow (42 $\mu g\ L^{-1}$ at flume 1) (42 ppb) than at the upper-most flume (28 $\mu g\ L^{-1}$ at flume 4) (28 ppb), while the inverse occurred during storm flow (304 $\mu g\ L^{-1}$ at flume 4 and 128 $\mu g\ L^{-1}$ at flume 1) (304 and 128 ppb, respectively) (Figure 11). Similar trends in total P concentration were observed.

During storm flow, in-channel decreases in P concentration were indicative of the dilution of P from a critical source area above the uppermost flume (flume 4), where an area of high soil P intersected an area of high overland flow potential (Figure 11). During base flow, the increase in P concentrations downstream was clearly controlled by channel sediments (Figure 11), such that the P sorption maximum of the uppermost flume (flume 4) sediment (532 mg kg^{-1}) (532 ppm) was far greater than the outlet flume (flume 1) sediment (227 mg kg^{-1}) (227 ppm). Paralleling these trends, the EPC_0 of sediment at flume 1 was greater than at flume 4 (34 to 0.4 $\mu g\ L^{-1}$) (34 to 0.4 ppb). Sediment EPC_0 trends were highly correlated to base flow dissolved P concentrations (28 $\mu g\ L^{-1}$ at flume 4 and 42 $\mu g\ L^{-1}$ at flume 1) (28 and 42 ppb, respectively) (Figure 11).

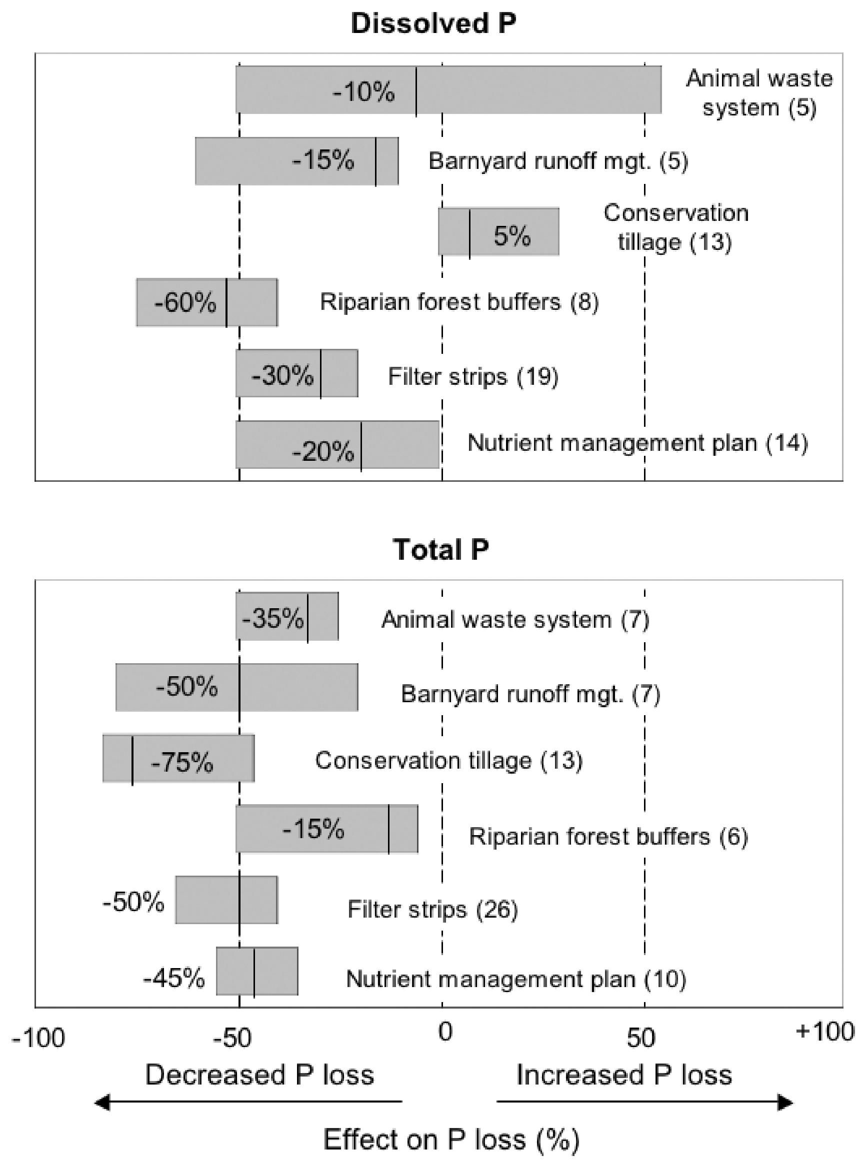
In a much larger watershed, McDowell et al. (2002) examined the processes controlling sediment P release to the Winooski River in Vermont, the largest tributary to Lake Champlain. The amount of iron-oxide strip P (algal-available P) in the river sediments adjacent to agricultural land (3.6 mg kg^{-1}) (3.6

ppm) was significantly greater ($P<0.05$) than that in sediments adjacent to forested land (2.4 mg kg^{-1}) (2.4 ppm). Notably, impoundment (731 mg kg^{-1}) (731 ppm) and reservoir sediments (803 mg kg^{-1}) (803 ppm) had greater total P concentrations than did river sediments (462 mg kg^{-1}) (462ppm). This result was attributed to more fine sediments (< 63 μm) in impoundments and reservoirs (64%) than in rivers (33%). Consequently, impoundment and reservoir sediments had lower abilities to release P to solution in the short term, thereby acting as P sinks. Internal recycling of P from reservoir sediments (e.g., release of P from iron-P compounds or biotic mediated release by sulphur-reducing bacteria in an anaerobic hypolimnion) can be an important source of P with regard to total watershed management and TMDL implementation, especially during reservoir stratification (Baldwin et al. 2000; Golterman 2001; Gray and Kirkland 1986; Knuuttila et al. 1994).

The results of this research clearly 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 store of P with the potential to release a large amount of P to overlying waters over the long term. In the short term, however, river flow and physical properties of the sediments will influence the amount of sediment P leaving the watershed in the Winooski River. Thus, modeling of channel processes must account for variability in flow, local sources of P, sediment properties, and P resuspension in streams, particularly near the point of impact (Hanrahan et al. 2001). In the past, however, the complexity of these processes and fluvial systems has limited the ability of watershed models to simulate in-channel transport and transformations of P.

Figure 10

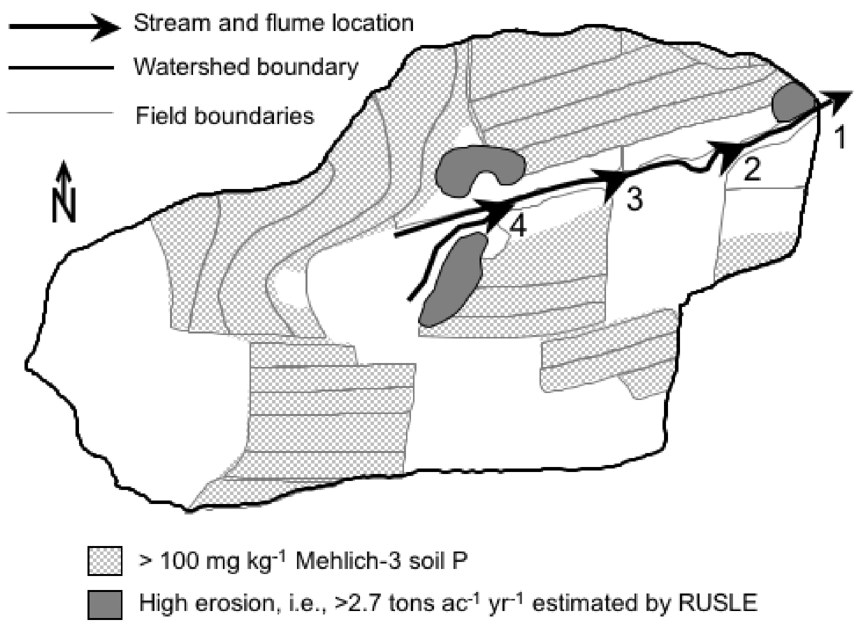
Range in effectiveness of various BMPs in reducing dissolved and total P loss in overland flow based on published information (adapted from Gitau et al., 2001). Number of studies is in parenthesis. The solid line through each box represents the median value of effectiveness.



The impact of P transport to downstream water bodies. Biological responses differ drastically among water bodies, according to variations in geographic location, climate, water residence times, and surface area and depth of water body. The Cannonsville Reservoir (part of the New York City water supply system) flushes in a matter of months, while Cayuga Lake (the longest Fingerlake in New York State) has a mean water residence time of about 12 years (National Research Council 2000b). The Chesapeake Bay has a completely different set of critical biological indicators in comparison with the Gulf of Mexico (National Research Council 2000a). In fact, the ratio of watershed drainage area and Bay water volume (2410 km^2 km^{-3}) (1500 mi^2 mi^{-3}) is nearly an order of magnitude greater than any other lake or bay in the world (next is Gulf of Finland at 380 km^2 km^{-3}) (240 mi^2 mi^{-3}). As a result, simulation as well as management of the biological response within the Chesapeake Bay presents unique challenges because of the relatively large area for nutrient source inputs that must be considered.

Models of specific water bodies of concern have been developed and can also be characterized in terms of scale, level of mechanistic detail, and degree of computation. Most of these models operate on similar time scales as watershed models and are driven in part by nutrient loading. A detailed description of these models is beyond the scope of this review and readers are directed to National Research Council (2000a) for a more detailed description of several aquatic models.

Figure 11
The distribution of high Mehlich-3 soil P (>100 mg kg^{-1}) and erosion (>6 Mg ha^{-1} yr^{-1}) [>2.7 tons acre^{-1} yr^{-1}]) and dissolved P concentration in stream and baseflow (mean of 1997-2000 data) in relation to P sorption properties of channel sediment at four flumes in FD-36 (data adapted from McDowell et al., 2001a).



Flume	Dissolved P		Stream sediment	
	Stormflow	Baseflow	P sorption max	EPC_0
	µg L^{-1}		mg kg^{-1}	µg L^{-1}
1	128	42	227	34
2	174	36	295	13
3	202	37	330	4
4	304	28	532	4

Although we can simulate P loss in overland flow, the related effects of agricultural management, and how nutrients cycle within a water body, it is still difficult to relate P loss as a function of watershed management to the biological response of a receiving water body. Because of the scales involved, the connectivity, and the dominant processes in terrestrial and aquatic systems, watershed and water body response models have tended to develop independently. Summer et al. (1990) attempted to link watershed (AGNPS) and lake process (FARMPND) models. However, a lack of adequate water monitoring data (chemistry and flow rate) limits rigorous testing of the models' ability to simulate a lake's response to changes in agricultural management and climate.

Suggestions for Future Model Development

This discussion has presented background information on processes controlling P transport in overland and subsurface flow from agricultural landscapes and how nonpoint source models have attempted to simulate P losses. New information on soil and site dependency of extraction coefficients relating STP and overland flow dissolved P and the use of enrichment ratios to estimate particulate P transport should be incorporated into these models. Also, incorporation of new formulations describing the release and transport of inorganic and organic P from manure in overland and subsurface flow will improve model predictions of P loss following land application of manures. However, it is clear that there is a great deal of information already available on the fate and transport of P in agricultural landscapes and the effectiveness of various BMPs at minimizing this loss through source or transport controls. To better use existing data rather than reinventing the wheel, researchers are developing ways to apply this information through innovative database management and integration with existing models.

Many complex models are available and are gaining greater acceptance as computers become cheaper and more powerful and as managers and planners become more comfortable using them. Models have a strong appeal to policymakers and managers because they yield clear numerical results that make it possible to gauge progress. However, they can sometimes bring false confidence and misconceptions (Boesch et al. 2001). It has been said that while all models are wrong, some are useful. Modelers must clearly define what their model is and is not designed to do. Likewise, users must decide what they want to accomplish with a model. They must consider the scale (field, watershed, or basin), time (flow event, annual, or multi-year), and level of accuracy (0.1 or 10 kg ha^{-1} yr^{-1}) (0.1 or 10 lb ac^{-1}) that needs to be simulated, as well as the amount of parameterization data available. A key to useful simulation of P loss is selection of the appropriate model and the data to run it. If, for instance, one needs to identify areas in a watershed at greater risk for P loss to target remedial BMPs, then site vulnerability tools such as the P Index are available (Lemunyon and Gilbert 1993; Gburek et al., 2000). However, P indices are not designed to quantify P loss, as are many nonpoint source models described earlier in this discussion.

It is clear that there can be a great deal of uncertainty in model computations. Uncertainty arises with an imperfect representation of the physics, chemistry, and biology of the real world, caused by numerical approximations, inaccurate parameter estimates and data input, and errors in measurements of the variables being computed. Whenever possible, this uncertainty should be represented in the model output (e.g., as a mean plus standard deviation) or as confidence limits on the output of a time series of concentrations or flows. The tendency for decision makers to "believe" models because of their presumed deterministic nature and "exact" form of output must be tempered by responsible use of the models by engineers and scientists, so that model computations or predictions are not oversold or given more weight than they deserve. Above all, model users should determine that the model computations are reasonable in the sense of providing output that is physically realistic and based on input parameters that are within accepted ranges.

Based on our current knowledge of nonpoint source modeling, we have identified the following gaps and research needs:

- Flexible extraction coefficients relating soil test P and overland flow P that are a function of soil type and/or land use, rather than current fixed default values, need to be developed. One possibility would be to have three options: (a) a coefficient value related to some soil type or land use characteristic, (b) a user-defined coefficient value if available, and (3) an existing default value.
- The effects of manure management on overland flow P via direct release of P from manure are poorly addressed. Formulations should be developed that simulate P loss as a function of manure type (dry or liquid), method of application (surface, injection, or incorporation), and impact of soil physical properties influencing overland flow (soil aggregation, infiltration, flow volumes, and soil-water holding capacity).
- The importance of field or landscape position relative to the stream channel in determining watershed export of P via variable source area hydrology and channel chemical transport pathways should be considered.
- Stream channel effects in terms of dilution, channel sedimentation and erosion, sediment P resuspension, and sediment sorption and release of P are not simulated, even though these processes can dramatically alter predicted edge-of-field losses prior to watershed export. Some models include an in-stream decay rate for P, which in reality can be fluvial-system specific, and most models do not account for P resuspension. This modeling need will be of critical importance as watershed models become an integral part of the TMDL decision-making process and BMP evaluation.
- Nested modeling efforts that develop parameters for generalized models of large watersheds, which are based on results and knowledge gained through mechanistic modeling of field scale processes and smaller sub-watersheds are needed. There is a lot of information on processes controlling P export and the relative effectiveness of various BMPs, but a greater emphasis should be placed on integrating these databases into modeling efforts.
- Greater linkage between watershed and water body response models is needed. This lack is an important shortcoming of most predictions of nonpoint source impacts on the chemical and biological response of receiving water bodies. Linkage and interfacing of these models will enable the translation of agricultural management effects on the export of P from watersheds to the point of impact, in terms of the chemical or biological responses of receiving water bodies.
- Finally, there is a lack of stream flow monitoring of P concentrations and loads for a wide range (both geographically and across land use) of watersheds. This lack of data severely limits the representation, calibration, and validation of nonpoint source models. Typically, monitoring strategies are fixed interval sampling schemes, ranging from weekly to monthly (National Research Council 2000a). As the sampling interval lengthens, the accuracy of P load estimates generally decreases (Pionke et al. 1999). To accurately characterize P loads, sampling strategies need to represent the range in flows and P concentrations over a period of time. This is also a key weakness in evaluating the effectiveness of BMP implementation at decreasing the watershed export of P. In fact, future TMDL implementation may require absolute measures of P loss reductions. As a result, model scenarios and comparisons may identify potential BMPs, but watershed monitoring will still be needed to verify model output and improve model predictions.

Conclusion

The role of modeling will be more and more important over the next decade in making management and policy decisions related to conservation programs and water quality enhancement and enforcement. The availability of water monitoring data is increasing in response to water quality concerns in the United States and other parts of the world and providing new opportunities to develop, calibrate, and test watershed models. As we progress, we need to take an interdisciplinary approach that involves hydrologists, soil scientists, engineers, economists, animal scientists, and possibly rural sociologists.

With the knowledge that many and varied working models exist, our efforts should be directed to the improvement or adaptation of existing models, rather than reinventing or developing new models, except where major limitations have been clearly defined. Finally and most importantly, it is essential that the user carefully select the most appropriate model that will provide the right level of predictive accuracy, make use the available input data, and provide the appropriate scale of simulation for both time and space.

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