Attachment 37:

Phosphorus loss from land to water: integrating agricultural and environmental management
Sharpley et al. 2001

Plant and Soil **237:** 287–307, 2001.

Phosphorus loss from land to water: integrating agricultural and environmental management

Andrew N. Sharpley*, Richard W. McDowell & Peter J. A. Kleinman
*USDA-ARS, Pasture Systems and Watershed Management Research Unit Building 3702, Curtin Road, University Park, PA 16802-3702, USA *Corresponding author: Tel: +1-814-863-0948; Fax: +1-814-865-2058; E-mail: ans3@psu.edu*

Key words: eutrophication, fertilizers, manures, nonpoint source pollution, runoff, water quality

Abstract

Phosphorus (P), an essential nutrient for crop and animal production, can accelerate freshwater eutrophication, now one of the most ubiquitous forms of water quality impairment in the developed world. Repeated outbreaks of harmful algal blooms (e.g., *Cyanobacteria* and *Pfiesteria*) have increased society's awareness of eutrophication, and the need for solutions. Agriculture is regarded as an important source of P in the environment. Specifically, the concentration of specialized farming systems has led to a transfer of P from areas of grain production to animal production. This has created regional surpluses in P inputs (mineral fertilizer and feed) over outputs (crop and animal produce), built up soil P in excess of crop needs, and increased the loss of P from land to water. Recent research has shown that this loss of P in both surface runoff and subsurface flow originates primarily from small areas within watersheds during a few storms. These areas occur where high soil P, or P application in mineral fertilizer or manure, coincide with high runoff or erosion potential. We argue that the overall goal of efforts to reduce P loss to water should involve balancing P inputs and outputs at farm and watershed levels by optimizing animal feed rations and land application of P as mineral fertilizer and manure. Also, conservation practices should be targeted to relatively small but critical watershed areas for P export.

Introduction

Phosphorus (P), an essential nutrient for crop and animal production, can accelerate freshwater eutrophication (Carpenter et al., 1998; Sharpley, 2000). Recently, the US Environmental Protection Agency (1996) and US Geological Survey (1999) identified eutrophication as the most ubiquitous water quality impairment in the US. Eutrophication restricts water use for fisheries, recreation, and industry due to the increased growth of undesirable algae and aquatic weeds and oxygen shortages caused by their death and decomposition. Also, an increasing number of surface waters have experienced periodic and harmful algal blooms (e.g., *Cyanobacteria* and *Pfiesteria*), which contribute to summer fish kills, unpalatability of drinking water, formation of carcinogens during water chlorination and links to neurological impairment in humans (Burkholder and Glasgow, 1997; Kotak et al., 1993).

Although concern over eutrophication is not new, there has been a profound shift in our understanding of, and focus on, sources of P in water bodies. Since the late 1960s, the relative contributions of P to water bodies from point and non-point sources has changed dramatically. On one hand, great strides have been made in the control of point source discharges of P, such as the reduction of P in sewage treatment plant effluent. These improvements have been due, in part, to the ease in identifying point sources. On the other hand, less attention has been directed to controlling non-point sources of P, due mainly to the difficulty in their identification and control (Sharpley and Rekolainen, 1997). Thus, control of non-point sources of P is a major hurdle to protecting fresh surface waters from eutrophication (Sharpley and Tunney, 2000; Sharpley et al., 1999a; Withers et al., 2000).

While a variety of non-point sources, ranging from suburban lawns to construction sites to golf courses, contribute P to water bodies, agriculture, particularly intensive livestock agriculture, is receiving more and more attention (Lander et al., 1998; Sharpley, 2000).

This may be attributed to the evolution of agricultural systems from net sinks of P (i.e., deficits of P limit crop production) to net sources of P (i.e., P inputs in feed and mineral fertilizer can exceed outputs in farm produce). Before World War II, for example, farming communities tended to be self-sufficient in that they produced enough feed locally to meet animal requirements and could recycle the manure nutrients effectively to meet crop needs. As a result, nutrients were generally recycled in relatively localized areas. After World War II, farming systems became more specialized in the USA, with crop and livestock operations in different regions of the country. Today, less than a third of the grain is produced on farms where it is grown (Lanyon, 2000). This has resulted in a major one-way transfer of P from grain-producing areas to animal-producing areas (Sharpley et al., 1998b; Sims, 1997).

As animals inefficiently utilize P in feed (only 30% is retained), most of the P entering livestock operations ends up in manure, which is usually land applied locally. Animal manure can be a valuable resource for improving soil structure and increasing vegetative cover, thereby reducing surface runoff and erosion potential. However, in many areas of concentrated animal production, manures are normally applied at rates designed to meet crop nitrogen (N) requirements and to avoid groundwater quality problems created by leaching of excess N. This often results in a build up of soil test P above amounts sufficient for optimal crop yields, which can increase the potential for P loss in runoff as well as in leachate (Haygarth et al., 1998; Heckrath et al., 1995; Sharpley et al., 1996).

The ultimate goal of agricultural and environmental P management is to balance P inputs to the farm with outputs in primary produce such that no excess P is applied and soil P concentrations are kept at an optimum level for agronomic performance and minimal environmental impact. However, because of the potential for major changes in agricultural management and negative economic impacts, it is necessary to explore short-term or temporary fixes. In the USA, this has led the Environmental Protection Agency (EPA) and the Department of Agriculture (USDA) to devise a joint strategy for sustainable nutrient management for animal feedings operations (AFOs; USDA-USEPA, 1999). This strategy proposes a variety of voluntary and regulatory approaches, whereby all AFOs develop and implement comprehensive nutrient management plans by the year 2008. An important part of this strategy outlines how acceptable application rates of P as mineral fertilizer or manure will be determined.

In the USA, agencies charged with developing these strategies (i.e., EPA and USDA) have challenged the scientific community to provide technical leadership in developing sound criteria that identifies the risk of P loss from agricultural land to water (Sharpley et al., 1999b). The aim of this paper is to present research on P loss from land to water and show how this information is being used to define and support P management strategies that maintain agricultural production and protect water quality. We will discuss those factors controlling P loss in the context of developing practical tools for agricultural and environmental P management.

Assessing the Risk for Phosphorus Loss

Water quality concerns have forced many states in the USA to consider developing recommendations for land application of P and watershed management based on the potential for P loss in agricultural runoff (Sharpley et al., 1996; USDA-USEPA, 1999). Currently, these recommendations center on the identification of a threshold soil test P level above which the enrichment of P in surface runoff is considered unacceptable (Table 1). Existing agronomic guidelines may not be appropriate for water quality protection, and agronomic soil testing data may need to be reinterpreted to address environmental objectives (Sims and Sharpley, 1998). Specifically, agronomic soil test interpretations (i.e., low, medium, optimum, high) are based on the expected response of a crop to P, and cannot be directly translated to estimates of environmental risk, such as runoff P enrichment potential.

Even when soil testing data are properly reinterpreted for runoff enrichment potential, they provide an incomplete assessment of the potential for P loss from a site, as such data do not account for processes controlling the transport of P in surface runoff and subsurface flow (Kleinman et al., 2000). For example, adjacent fields having similar soil test P levels, but differing susceptibilities to surface runoff and erosion due to contrasting topography and management, may have substantially different P loss potentials (Sharpley and Tunney, 2000).

Generally, most P exported from agricultural watersheds comes from only a small part of the landscape during a few relatively large storms, where hydrologically active areas of a watershed contributing surface

Table 1. Threshold soil test P values and P management recommendations (adapted from Lory and Scharf, 2000; Sharpley et al., 1996)

State	Environmental soil P threshold mg kg^{-1}	Soil test P method	Management recommendations for water quality protection
Arkansas	150	Mehlich-3	*At or > 150 Mg P kg^{-1}:* apply no P, provide buffers next to streams, overseed pastures with legumes to aid P removal, and provide constant soil cover to minimize erosion.
Colorado	100	Olsen	*> 100 Mg P kg^{-1}:* hog producers with >36,000 lbs capacity, no P applied unless runoff is minimal.
Delaware	50	Mehlich-1	*> 50 Mg P kg^{-1}:* apply no more P until soil is significantly decreased.
Idaho	50 & 100	Olsen	*Sandy soils > 50 Mg P kg^{-1}:* *Silt loam soils > 100 Mg P kg^{-1}:* apply no more P until soil P is significantly decreased.
Kansas	100–200	Bray-1	Regions of the state coincide with high (eastern) to low (western) runoff. Swine producers must eliminate manure applications above the threshold.
Ohio	150	Bray-1	*> 150 Mg P kg^{-1}:* decrease erosion and/or eliminate P additions.
Oklahoma	130	Mehlich-3	*30 B 130 Mg P kg^{-1}:* half P rate on slopes > 8%. *130 B 200 Mg P kg^{-1}:* half P rate and adopt measures to decrease surface runoff and erosion. *> 200 Mg P kg^{-1}:* P rate not to exceed crop removal.
Maine	40–100	Morgan	Apply no P in sensitive (40 Mg P kg^{-1}) and non-sensitive watershed (100 Mg P kg^{-1}).
Maryland	75	Mehlich-1	*Use P index >75 Mg P kg^{-1}:* soils with high index must reduce or eliminate P additions.
Michigan	75	Bray-1	*75 B 150 Mg P kg^{-1}:* P application should equal crop removal. *> 150 Mg P kg^{-1}:* apply no P from any source.
Mississippi	70	Lancaster	*> 70 Mg P kg^{-1}:* no P added
Texas	200	Texas A&M	*> 200 Mg P kg^{-1}:* P addition not to exceed crop removal
Wisconsin	75	Bray-1	*< 75 Mg P kg^{-1}:* rotate to P demanding crops and decrease P additions. *> 75 Mg P kg^{-1}:* discontinue P additions.

runoff to streamflow are coincident with areas of high soil P (Gburek and Sharpley, 1998; Pionke et al., 1997). To be most effective, risk assessment must consider 'critical source-areas'; areas within a watershed that are most vulnerable to P loss in surface runoff (Gburek and Sharpley, 1998). Critical source areas are dependent on the coincidence of transport (surface runoff, erosion, and subsurface flow) and site management factors (functions of soil, crop, and management) (Table 2). Transport factors mobilize P sources, creating pathways of P loss from a field or watershed. Site management factors are typically well defined and reflect land use patterns related to soil P status, mineral fertilizer and manure P inputs, and tillage (Table 2).

Even in regions where subsurface flow pathways dominate P transport, areas contributing P to drainage waters appear to be localized to soils with high soil P saturation and hydrological connectivity to the drainage network (Schoumans and Breeuwsma, 1997). Therefore, soil P levels alone have little meaning vis a vis P loss potential unless they are used in conjunction with estimates of potential surface runoff and subsurface flow.

Table 2. Factors influencing P loss from agricultural watersheds and its impact on surface water quality

Factors	Description
Transport	
Erosion	Total P loss strongly related to erosion.
Surface runoff	Water has to move off or through a soil for P to move.
Subsurface flow	In sandy, organic, or P-saturated soils, P can leach through the soil.
Soil texture	Influences relative amounts of surface and subsurface flow occurring.
Irrigation runoff	Improper irrigation management can induce surface runoff and erosion of P.
Connectivity to stream	The closer the field to the stream, the greater the chance of P reaching it
Channel effects	Eroded material and associated P can be deposited or resuspended with a change in stream flow. Dissolved P can be sorbed or desorbed by stream channel sediments and bank material.
Proximity of P-sensitive water	Some watersheds are closer to P-sensitive waters than others (i.e., point of impact).
Sensitivity P input	Shallow lakes with large surface area tend to be more vulnerable to eutrophication.
Site management	
Soil P	As soil P increases, P loss in surface runoff and subsurface flow increases.
Applied P	The more P (mineral fertilizer or manure), the greater the risk of P loss.
Application method	P loss increases in the order: subsurface injection; plowed under; and surface broadcast with no incorporation.
Application timing	The sooner it rains after P is applied, the greater the risk for P loss

Development of the phosphorus index

To overcome the limitations of using a soil P threshold as the sole measure of site P loss potential, the US Natural Resource Conservation Service (NRCS), in cooperation with research scientists, developed a site assessment tool for P loss potential (i.e., the P index, Table 3). The P index was designed as a screening tool for use by field staff, watershed planners, and farmers to rank the vulnerability of sites to P loss in surface runoff (Lemunyon and Gilbert, 1993).

Calculating site vulnerability to phosphorus loss

The P index accounts for and ranks transport and site management factors controlling P loss in surface runoff and sites where the risk of P movement is expected to be higher than that of others (Tables 3 and 4). Site vulnerability to P loss in surface runoff is assessed by selecting rating values for a variety of transport (Table 3) and site management factors (Table 4).

To calculate transport potential for each site, erosion, surface runoff, leaching potential, and connectivity values were first summed (Table 3). Dividing this summed value by 23, the value corresponding to 'high' transport potential (erosion is 7, surface runoff is 8, leaching potential is 0, and connectivity is 8), a relative transport potential was determined. This normalization process assumes that when a site's full transport potential is realized, 100% transport potential is realized. Thus, transport factors <1 represent a fraction of the maximum potential (Table 3).

Calculation of site management factors of the P index are based on the Mehlich-3 P concentration of surface soil samples collected at each site and P application as mineral fertilizer or manure as determined from annual farmer surveys (Table 4). The correction factor of 0.2 for soil test P is based on field data which

Table 3. Phosphorus loss potential due to transport characteristics in the P index

Characteristics	Relative Ranking					Field Value
Soil Erosion	Soil loss (Tonnes/ha/year)					
Soil Runoff Class	Very Low	Low	Medium	High	Very High	
	0	**1**	**2**	**4**	**8**	
Subsurface Drainage	Very Low	Low	Medium	High	Very High	
	0	**1**	**2**	**4**	**8**	
Leaching Potential	Low		Medium		High	
	0		**2**		**4**	
Connectivity	Not connected†		Partially connected‡		Connected§	
	0	**1**	**2**	**4**	**8**	

Total Site Value (sum of erosion, surface runoff, leaching, and connectivity values):

Transport Potential for the Site (total value / 23) ¶:

†Field is far away from water body. Surface runoff from field does not enter water body.
‡Field is near but not next to water body. Surface runoff sometimes enters water body, e.g., during large intense storms.
§Field is next to a body of water. Surface runoff from field always enters water body.
¶The total site value is divided by a high value (23).

showed a 5-fold greater concentration of dissolved P in surface runoff with an increase in mineral fertilizer or manure addition compared to an equivalent increase in Mehlich-3 P (Sharpley and Tunney, 2000).

A P index value, representing cumulative site vulnerability to P loss, is obtained by multiplying summed transport and site management factors (Table 5). The P index values are normalized so that the break between high and very high categories is 100. This is done by calculating a site P index value, assuming all transport and source factors are high. Erosion is set at 7 tonnes ha^{-1} considered a high value for Pennsylvania and soil test P is set at 200 mg kg^{-1} Mehlich-3 P, which is proposed as a non-site specific threshold for Pennsylvania (Beegle, 2000). The break between medium and high and low and medium is calculated using the same method and soil test P concentrations of 50 and 30 mg Mehlich-3 P kg^{-1}, respectively. These Mehlich-3 P levels correspond to crop response and fertilizer recommendations for Pennsylvania, with 50 mg kg^{-1} sufficient for production and no response to added P and 30 mg kg^{-1} the low value (Beegle, 2000).

Management interpretations of the phosphorus index

Since its inception, two major changes have been introduced to the P index. First, source and transport factors were related in a multiplicative rather than additive fashion, in order to better represent actual site vulnerability to P loss. For example, if surface runoff does not occur at a particular site, its vulnerability should be low regardless of the soil P content. In the original P index, a site could be ranked as very highly vulnerable based on site management factors alone, even though no surface runoff or erosion occurred. On the other hand, a site with a high potential for surface runoff, erosion or subsurface flow but with low soil P is not at risk for P loss, unless P as mineral fertilizer or manure is applied. Second, an additional transport factor reflecting distance from the stream was incorporated into the P index. The contributing distance categories in the revised P index are based on hydrological analysis. This analysis considers the probability (or risk) of occurrence of a rainfall event of a given magnitude which will result in surface runoff to the stream (Gburek et al., 2000).

In addition to its function as a practical screening tool, the P index can also be used to identify agricultural areas or management practices that have the greatest potential to accelerate eutrophication. As such, the P index will identify alternative management options available to land users, providing flexibility in developing remedial strategies. Some general recommendations are given in Table 6. In considering these recommendations, one should keep in mind that P management is very site-specific and requires a well-planned, coordinated effort between farmers, extension agronomists, and soil conservation specialists.

In its current form, the P index is not a quantitative predictor of P loss in surface runoff or subsurface flow from a watershed. Rather it is a qualitative assessment tool to rank site vulnerability to P loss. Ultimately, the

Table 4. Phosphorus loss potential due to site management characteristics in the P index

Site Characteristics	Relative Ranking					Field Value
	Very Low	Low	Medium	High	Very High	
Soil Test P			Soil test P (mg/kg)			
Loss Rating Value			Soil Test P ∗ 0.2			
Fertilizer P Rate			Fertilizer Rate (kg P/ha)			
Fertilizer Application Method and Timing	Placed with planter or injected more than 2" deep	Incorporated <1 week after application	Incorporated >1 week or not incorporated >1 following application in spring–summer	Incorporated >1 week or not incorporated following application in autumn–winter	Surface applied on frozen or snow covered soil	
	0.2	**0.4**	**0.6**	**0.8**	**1.0**	
Loss Rating Value			Fertilizer P Application Rate ∗ Loss Rating for Fertilizer P Application Method and Timing			
Manure P Rate			Manure application (kg P/ha)			
Manure Application Method and Timing	Placed with planter or injected more than 2" deep	Incorporated <1 week after application	Incorporated >1 week or not incorporated >1 following application in spring–summer	Incorporated >1 week or not incorporated following application in autumn–winter	Surface applied on frozen or snow covered soil	
	0.2	**0.4**	**0.6**	**0.8**	**1.0**	
Loss Rating Value			Manure P Application Rate ∗ Loss Rating for Manure P Application Method and Timing			
					Total Site Management Value (sum of soil, fertilizer, and manure P loss rating values):	

Table 5. Worksheet and generalized interpretation of the P index

P Index	Generalized interpretation of the P index
Low < 30	LOW potential for P loss. If current farming practices are maintained, there is a low probability of adverse impacts on surface waters.
Medium 30–70	MEDIUM potential for P loss. The chance for adverse impacts on surface waters exists, and some remediation should be taken to minimize the probability of P loss.
High 70–100	HIGH potential for P loss and adverse impacts on surface waters. Soil and water conservation measures and a P management plan are needed to minimize the probability of P loss.
Very high > 100	VERY HIGH potential for P loss and adverse impacts on surface waters. All necessary soil and water conservation measures and a P management plan must be implemented to minimize the P loss.

P index rating for a site = Transport potential value × Site management value/45 †. 145 is the value to normalize the break between high and very high to 100. The following is used:
Transport value (23/23; i.e., 1.0)
Erosion is 7 tonnes/ha per year, 7
Surface runoff class is very high, 8
Field is connected, 8
Site management (145)
Soil test P is 200, 40
Fertilizer P application is 30 kg P/ha, 30
Manure P application is 75 kg P/ha, 75

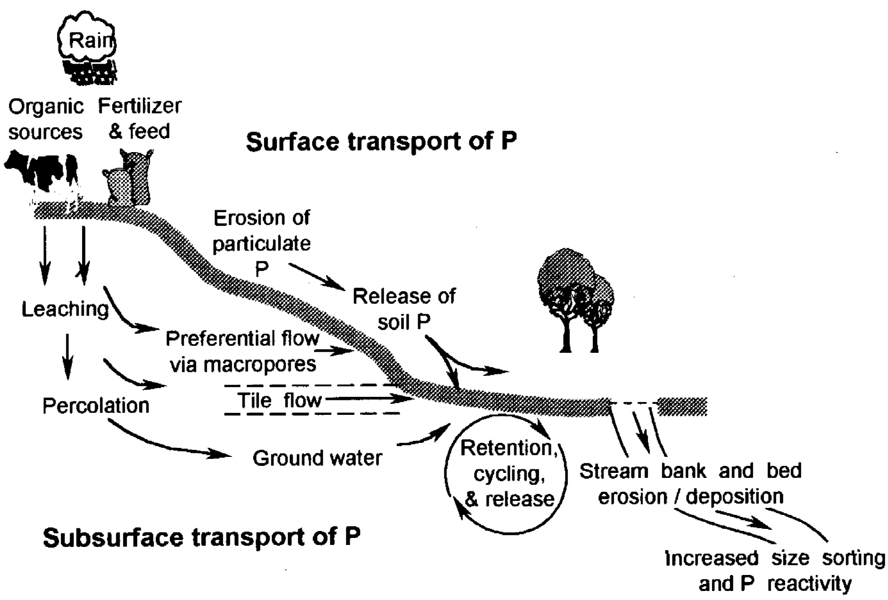


Figure 1. Transport and site management factors influencing the potential for P loss from agricultural land to surface waters.

Table 6. Management options to minimize nonpoint source pollution of surface waters by soil P

Phosphorus index	Management options to minimize nonpoint source pollution of surface waters by soil P
(LOW) < 30	*Soil testing*: have soils tested for P at least every 3 years to monitor build-up or decline in soil P. *Soil conservation*: follow good soil conservation practices. Consider effects of changes in tillage practices or land use on potential for increased transport of P from site.
(MEDIUM) 30–70	*Nutrient management*: consider effects of any major changes in agricultural practices on P losses *before* implementing them on the farm. Examples include increasing the number of animal units on a farm or changing to crops with a high demand for fertilizer P. *Soil testing*: have soils tested for P at least every 3 years to monitor build-up or decline in soil P. Conduct a more comprehensive soil testing program in areas that have been identified by the P Index as being most sensitive to P loss by surface runoff, subsurface flow, and erosion. *Soil conservation*: implement practices to reduce P losses by surface runoff, subsurface flow, and erosion in the most sensitive fields (i.e., reduced tillage, field borders, grassed waterways, and improved irrigation and drainage management). *Nutrient management*: any changes in agricultural practices may affect P loss; carefully consider the sensitivity of fields to P loss before implementing any activity that will increase soil P. Avoid broadcast applications of P fertilizers and apply manures only to fields with lower P Index values.
(HIGH) 70–100	*Soil testing*: a comprehensive soil testing program should be conducted on the entire farm to determine fields that are most suitable for further additions of P. *Soil conservation*: implement practices to reduce P losses by surface runoff, subsurface flow, and erosion in the most sensitive fields (i.e., reduced tillage, field borders, grassed waterways, and improved irrigation and drainage management). Consider using crops with high P removal capacities in fields with high P Index values. *Nutrient management*: in most situations fertilizer P, other than a small amount used in starter fertilizers, will not be needed. Manure may be in excess on the farm and should only be applied to fields with lower P Index values. A long-term P management plan should be considered.
(VERY HIGH) > 100	*Soil testing*: a comprehensive soil testing program must be conducted on the entire farm to determine fields that are most suitable for further additions of P. *Soil conservation*: implement practices to reduce P losses by surface runoff, subsurface flow, and erosion in the most sensitive fields (i.e., reduced tillage, field borders, grassed waterways, and improved irrigation and drainage management). Consider using crops with high P removal capacities in fields with high P Index values. *Nutrient management*: fertilizer and manure P should not be applied for at least 3 years and perhaps longer. A comprehensive, long-term P management plan must be developed and implemented.

P index is an educational tool that brings interaction between the planner and farmer in assessing environmental management decisions required to improve the farming system on a watershed rather than political basis.

Transport factors

Transport factors are critical to site assessment as they translate potential P sources into actual loss from a field or watershed. Factors controlling the transport of P within agricultural watersheds are conceptualized in Fig. 1. The main controlling factors and those considered in the P index are erosion, surface runoff, subsurface flow, and distance or connectivity of the site to the stream channel. The justification for inclusion of each of these factors is given below.

Erosion

Erosion is a mechanism of P transport that preferentially removes finer-sized soil particles (Haygarth and Sharpley, 2000). As a result, the P content and reactivity of eroded material is usually greater than source

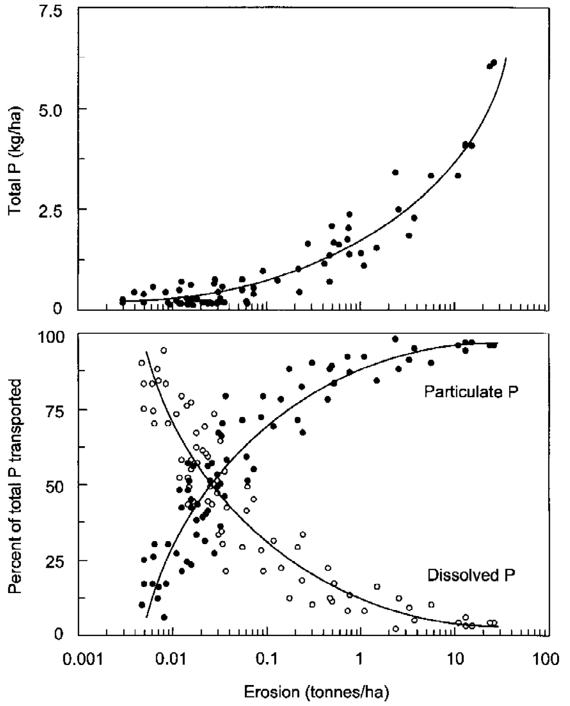


Figure 2. Total P loss and percentage of total P in dissolved and particulate forms a function of erosion in surface runoff from watersheds at E1 Reno, OK (adapted from Sharply et al., 1991; Smith et al., 1991).

soil. For example, Sharpley (1985b) found that under simulated rainfall, the enrichment of soil test P (Bray-1 P) and total P content of sediment in surface runoff from several soils compared to the whole soil, ranged from 1.2 to 6.0 and 1.2 to 2.5, respectively. These P enrichment ratios increased as erosion decreased, favoring the relative movement of fine-particles (<2 μm) with greater P content over coarse particles (> 5 μm) with lower P content.

The effect of erosion on P movement is illustrated by a 15-year study of runoff from several grassed and cropped watersheds in the Southern Plains (Fig. 2; Sharpley et al., 1991; Smith et al., 1991). Increasing erosion from native grass, no-till and conventional-till wheat (*Triticum aestivum* L.) resulted in an increase in total P loss, of which a greater proportion was transported as particulate P. Accompanying the increase in particulate P movement, was a relative decrease in dissolved P movement (Fig. 2).

Surface Runoff

The potential for P loss in surface runoff from a given site can be extremely high. The transport of dissolved

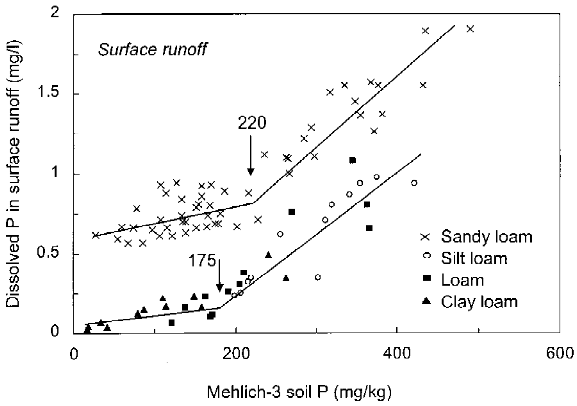


Figure 3. Relationship between the concentration of dissolved P in surface runoff and Mehlich-3 extractable soil P concentration of surface soil (0–5 cm) from the FD-36 watershed, Northumberland Co., PA (adapted form McDowell and Sharply, 2001).

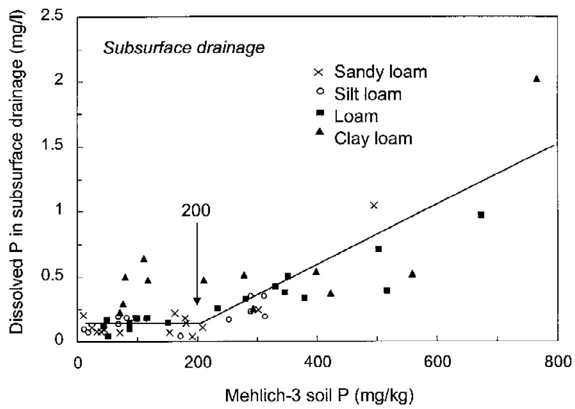


Figure 4. Relationship between the concentration of dissolved P in subsurface drainage from 30-cm deep lysimeters and the Mehlich-3 extractable soil P concentration of surface soil (0–5 cm) from the FD-36 watershed, Northumberland Cp., PA (adapted from McDowell and Sharply, 2001).

P in runoff is initiated by the release of P from soil, plant material, and suspended sediments (Fig. 1). This process occurs when rainfall interacts with a thin layer of surface soil (1–5 cm) before leaving the field as surface runoff (Sharpley, 1985a). The proportion of rainfall and depth of soil involved are highly dynamic due to variations in rainfall intensity, soil tilth, and vegetative cover, making them difficult to quantify in the field.

Subsurface Flow

Generally the P concentration in water percolating through the soil profile by leaching is small due to sorption of P by P-deficient subsoils. Exceptions occur in organic soils, where the adsorption affinity and capacity for P sorption are low due to the predominance of negatively charged surfaces (Duxbury and Peverly, 1978; Miller, 1979; White and Thomas, 1981). Other soils that are susceptible to movement include sandy soils with low P sorption capacities, waterlogged soils where Fe(III) has been reduced to Fe(II), and well structured soils prone to preferential flow through macropores and earthworm burrows (Bengston et al., 1992; Sharpley and Syers, 1979; Sims et al., 1998).

Because of the variable paths and time of water flow through a soil with subsurface drainage, factors controlling P loss in subsurface flow are more complex than for surface runoff. Subsurface flow includes artificial and natural drainage, where artificial drainage includes percolating water intercepted by installed drainage systems, such as mole and tile drains (Fig. 1). In general, the greater contact time between subsoil and natural subsurface flow than artificial drainage, results in lower losses of dissolved P in natural subsurface flow (Sharpley and Rekolainen, 1997; Sims et al., 1998).

Distance or connectivity to the stream channel

In order to translate the potential for P transport in surface runoff and subsurface flow from a given site to the potential for P loss in stream flow, it is necessary to account for whether water leaving a site actually reaches the stream channel. For instance surface runoff and subsurface flow may occur at various locations in a watershed and not reach the stream channel (Gburek et al., 2000). Thus, the location of a field in relation to the stream channel may determine whether runoff from the field reaches the channel and actually leaves the watershed. For a simple assessment of this factor, a site can be categorized as either not connected to the stream channel or connected to the channel by direct runoff, drainage ditch, or similar topographic feature.

Site Management Factors

A number of site management factors control P loss from agricultural lands. These include soil test P concentration, as well as rate, type (mineral fertilizer or manure), and method of P application (Fig. 1). These factors reflect day-to-day farm operations, while the transport factors discussed earlier tend to represent inherent soil, topographic and climatic properties.

Soil phosphorus

The loss of dissolved P in surface runoff is highly dependent on the P content of surface soil, as illustrated in Fig. 3. These data were obtained from several locations within a 40 ha watershed (FD-36) in south-central Pennsylvania (Northumberland Co.) using a portable rainfall simulator (Miller, 1987), following a protocol developed for the National P Project (Sharpley et al., 1999b). Briefly, either field plots (1-m wide and 2-m long) or packed boxes of soil (15-cm wide and 1-m long) were subjected to a rainfall intensity of 7 cm/h to produce 30-min of surface runoff, and a P concentration was determined for the entire 30-min event. This intensity for 30 min has an approximate 5-year return frequency in south-central Pennsylvania. To assess the role of soil test P on surface runoff P concentrations, field soils were selected to give a wide range in Mehlich-3 P concentrations (from 15 to 500 mg/kg).

A change point in the relationship between soil and surface runoff P, representing the interception of significantly different regression slopes ($P < 0.05$), is clearly visible at Mehlich-3 P values of 220 mg/kg (sandy loam soil) and 175 mg/kg (silt loam, loam and clay loam soils) (Fig. 3). Notably, for each of the soils, the potential for soil P release above this change point is greater than below it (McDowell and Sharpley, 2001; McDowell et al., 2001).

In a review of earlier studies, Sharpley et al. (1996) found that the specific regression equations between soil P and surface runoff P vary with soil type and management. For instance, regression slopes were flatter for grass (4.1–7.0, mean 6.0) than for cultivated land (8.3–12.5, mean 10.5). However, regression slopes were too variable to allow the use of a single or average relationship between soil test P and runoff P for all soils under the same management, probably due to inherent variability between soils. This variability is supported by the findings of Pote et al. (1999), who reported significantly different regression equations for three Ultisols of differing texture ($p < 0.05$). Also, the variation in the relationships presented in Fig. 3, as well as the corresponding change points, illustrates the soil specific nature of soil P release to surface runoff. Factors which 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 sorption capacities (Sharpley, 1983, 1999).

The concentration of P in subsurface flow is also related to surface soil P. In an experiment examining leachate from 30-cm deep lysimeters taken from the FD-36 watershed and subjected to simulated rainfall as described above for the National P Project, McDowell and Sharpley (2001) found the concentration of dissolved P in subsurface flow from the lysimeter increased (0.07–2.02 mg/l) as the Mehlich-3 P concentration of surface soil increased (15 to 775 mg/kg; Fig. 4). These data manifest a change point that was similar to the change point identified for surface runoff. They concluded that the dependence of subsurface P transport on surface soil P is evidence of the importance of P in preferential flow pathways such as earthworm burrows and old root channels.

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 soil test P (Olsen P) >60 mg/kg in the plow layer of a silt loam, caused the dissolved P concentration in tile drainage water to increase dramatically (0.15–2.75 mg/l). They postulated that this level, which is well above that needed by major crops for optimum yield (about 20 mg/kg; Ministry of Agriculture, Food and Fisheries, 1994), is a critical point above which the potential for P movement in land drains greatly increases. Similar studies suggest that soil P thresholds can vary threefold as a function of site hydrology, relative drainage volumes, and soil P release (desorption) characteristics (McDowell et al., 2001; Sharpley and Syers, 1979).

Application of phosphorus as mineral fertilizer or manure

The application of mineral fertilizer and manure to soil may dramatically increase P loss in surface runoff and subsurface flow. For example, 14 days after applying either 0, 50, or 100 kg P/ha in dairy manure to a Berks silt loam (Typic Dystrochrept) with a Mehlich-3 P content of 75 mg/kg, we applied artificial rainfall following the National P Project protocol (Sharpley et al., 1999b) and observed dissolved P concentrations in surface runoff (7 cm/h rainfall for 30 min) of 0.25, 1.35, and 2.42 mg/l, respectively (Fig. 5).

Table 7 summarizes findings from a variety of studies examining the effect of mineral fertilizer and manure management on runoff P concentration. From this and earlier reviews (Sharpley and Rekolainen, 1997), it is clear that the loss of P is influenced by the rate, time, and method of application; form of P

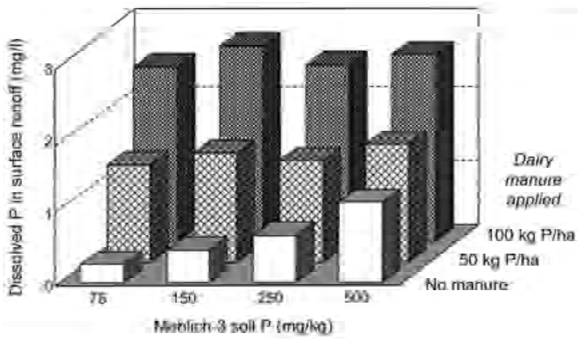


Figure 5. The concentration of P in surface runoff from a grassed Berks silt loam, as a function of Mehlich-3 soil P concentration and amount of dairy manure applied 2 weeks before the rainfall.

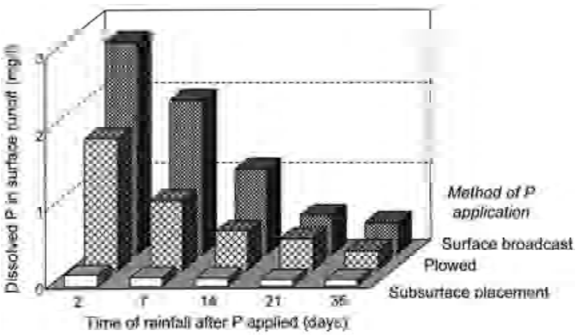


Figure 6. The effect of application method and timing of rainfall after application of dairy manure (100 kg p?ha) on the concentration of P in surface runoff from a grassed Berks silt loam.

added, amount and time of rainfall after application; and vegetative cover. In addition, the portion of applied P transported in runoff appears to be greater from conventional- than conservation-tilled watersheds. In one instance, McDowell and McGregor (1984) found mineral fertilizer P application to no-till corn actually reduced P transport, probably due to increased vegetative cover afforded by fertilization. Similarly, others found manure applications can reduce P loss in runoff via improved soil structure, aeration and water holding capacity afforded by added organic matter, as well as reducing erosion via increased vegetative over (Pote et al., 1996; Sharpley et al., 1998c).

Table 7 also illustrates that the loss of applied P in subsurface artificial drainage is appreciably lower than in surface runoff. Although it is difficult to distinguish between losses of mineral fertilizer, manure, or native soil P, without the use of expensive and hazardous radioactive tracers, total losses of applied P in runoff are generally less than 10% of that applied, unless rainfall immediately follows application or where surface runoff has occurred on steeply sloping, poorly drained,

Table 7. Effect of mineral fertilizer and manure application on P loss in surface runoff and fertilizer application on P loss in tile drainage

Land use	P added ($kg\ ha^{-1}\ yr^{-1}$)	Phosphorus loss ($kg\ ha^{-1}\ year^{-1}$) Dissolved	Total	Percent applied [a]	Reference and location
Surface runoff					
Mineral Fertilizer					
Grass	0	0.02	0.22		McColl et al., 1977;
	75	0.04	0.33	0.1	New Zealand
No-till corn	0	0.70	2.00		McDowell and McGregor, 1984;
	30	0.80	1.80		Mississippi
Conventional corn	0	0.10	13.89		
	30	0.20	17.70	12.7	
Wheat	0	0.20	1.60		Nicolaichuk and Read, 1978;
	54	1.20	4.10	4.6	Saskatchewan, Canada
Grass	0	0.50	1.17		Sharpley and Syers, 1976;
	50	2.80	5.54	8.7	New Zealand
Grass	0	0.17	0.23		Uhlen, 1988;
	24	0.25	0.31	1.2	Norway
	48	0.42	0.49	1.0	
Dairy Manure [b]					
Alfalfa	0	0.10	0.10		Young and Mutchler, 1976;
-spring	21	1.90	3.70	17.1	Minnesota
-autumn	55	4.80	7.40	13.3	
Corn	0	0.20	0.10		
-spring	21	0.20	0.60	2.4	
-autumn	55	1.00	1.60	4.7	
Poultry Manure					
Grass	0	0.00	0.10		Edwards and Daniel, 1992;
	76	1.10	2.10	2.6	Arkansas
Grass	0	0.10	0.40		Westerman et al., 1983;
	95	1.40	12.4	12.6	North Carolina
Swine Manure					
Fescue	0	0.10	0.10		Edwards and Daniel, 1993a;
	19	1.50	1.50	7.4	Arkansas
	38	4.80	3.30	8.4	
Aritifical Drainage					
Corn	0	0.13	0.42		Culley et al., 1983;
	30	0.20	0.62	0.7	Ontario, Canada
Oats	0	0.10	0.29		
	30	0.20	0.50	0.7	
Potatoes + Wheat + Barley					Catt et al., 1997;
Minimal till	102	0.26	8.97	8.8	Woburn, England
Conventional till	102	0.35	14.38	14.1	
Alfalfa	0	0.12	0.32		
	30	0.20	0.51	0.6	
Grass - 0–30 cm	32	0.12	0.38	1.1	Heathwaite et al., 1997;
- 30–80 cm	32	0.76	1.77	5.5	Devon, UK
Grass	0	0.08	0.17		Sharpley and Syers, 1979;
	50	0.44	0.81	1.3	New Zealand

[a] Percent P applied lost in runoff.
[b] Manure applied in either spring or autumn.

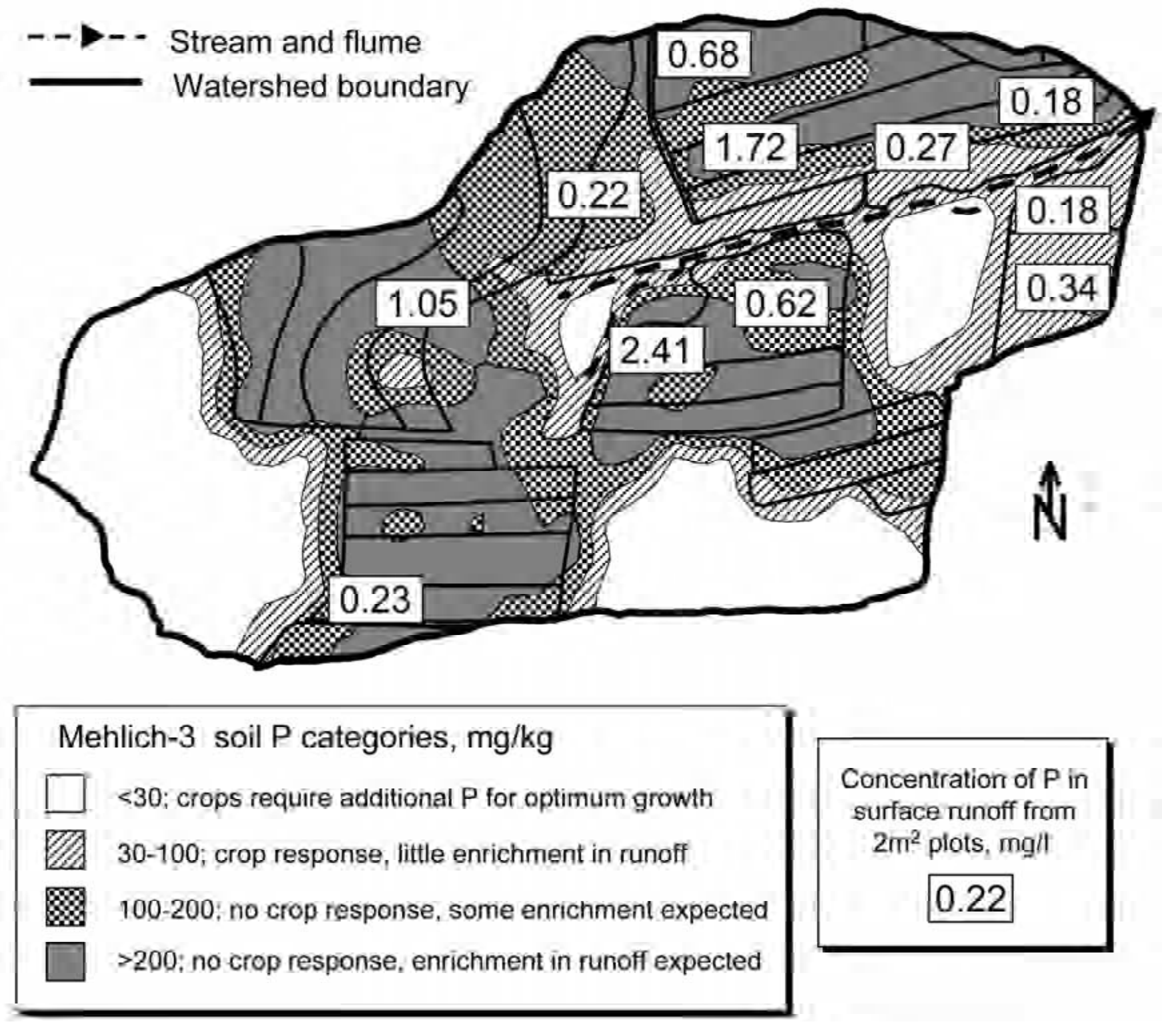


Figure 7. Distribution of Mehlich-3 soil P (0–5 cm soil depth) and concentration of dissolved P in surface runoff from 2-m^2 plots within the FD-36 watershed, Northumberland Co., PA.

and/or frozen soils. The high proportion of manurial P in runoff reported by the studies summarized in Table 7 may result from high manure application and generally less flexibility in application timing than for mineral fertilizer. Such inflexibility in scheduling of manure application results from the continuous production of manure throughout the year and a frequent lack of manure storage facilities.

Although we have shown soil P is important in determining P loss in surface runoff, applying P to soil can override soil P in determining P loss. For example, in our simulated rainfall study in the FD-36 watershed, the dissolved P concentration of surface runoff increased with Mehlich-3 P concentration in the upper 5 cm of soil (Fig. 5). When dairy manure was broadcast on these grassed soils, the dissolved P concentration of surface runoff 14 days later, was greater than with no manure (Fig. 5). Furthermore, the application of increasing quantities of manure P to these soils masked the effect of soil P concentration on surface runoff P.

Phosphorus application method and timing relative to rainfall also influences the concentration of P removed in runoff. For example, several studies have shown a decrease in P loss with an increase in the length of time between P application and surface runoff (Edwards and Daniel, 1993b; Sharpley, 1997; Westerman et al., 1983). This decrease can be attributed to the reaction of added P with soil and dilution of applied P by infiltrating water from rainfall that did not cause surface runoff. For instance, in our rainfall simulation studies in the FD-36 watershed, the dissolved P concentration of surface runoff from the Berks silt loam decreased from 2.75 to 0.40 mg/l when rainfall occurred 35 days rather than 2 days after a surface broadcast application of 100 kg P/ha as dairy manure (Fig. 6).

Although the concentration of P at the soil surface serves as the primary source of P to runoff, incorporation of manure into the soil profile either by tillage or subsurface placement, decreases the potential for P loss in surface runoff (Fig. 6). For example, the dissolved P concentration of surface runoff from a Berks silt loam 2 days after the surface application of 100 kg P/ha dairy manure was 2.75 mg/l. When the same amount of manure was incorporated by plowing to a depth of 10 cm, dissolved P in surface runoff was 1.70 mg/l, and when the manure was placed 5 cm below the soil surface, dissolved P in surface runoff fell to 0.15 mg/l (Fig. 6).

In an earlier field study, Mueller et al. (1984) found that incorporation of dairy manure by chisel plowing reduced total P loss in runoff from corn 20-fold compared to no-till areas receiving surface applications. However, the concentration of P in surface runoff did not decrease as dramatically as the mass of P lost. This was due to an increase in infiltration rate with manure incorporation and consequent decrease in surface runoff volume. In fact, surface runoff volume from no-till corn was greater than from conventional-till corn. Thus, P loss in runoff decreased by a dilution of P at the soil surface and reduction in runoff with incorporation of manure.

Testing the P index

Although there is a great deal of research documenting the justification of the transport and source factors included in the P index, there has been little site evaluation of index ratings. The original and modified versions of the P index have been used to assess the potential for P loss in several regions including the Delmarva Peninsula (Leytem et al., 1999; Sims, 1996), Oklahoma (Sharpley, 1995), Texas (McFarland et al., 1998), Vermont (Jokela et al., 1997), and Canada (Bolinder et al., 1998). However, few comparisons of P index ratings and measured P loss have been made. In Nebraska, Eghball and Gilley (1999) found correlation coefficients (r) as high as 0.84, between total P loss from simulated rainfall-runoff plots and P index ratings, when erosion losses were strongly weighted in the P index.

Using the National P Project rainfall simulator (Sharpley et al., 1999b), we measured the dissolved P concentration in surface runoff from 48, 1×2-m plots within the FD-36 watershed to evaluate the ability of the P index to rank site vulnerability to P loss on a plot scale. A selection of dissolved P concentrations of surface runoff within the watershed is given in Fig. 7, along with surface soil (0–5 cm depth) Mehlich-3 P illustrating the large variation in surface runoff P concentrations found between plots. At some sites, rain simulation was conducted approximately 2 weeks after manure application. At other sites, no manure had been applied for at least 9 months prior to rain simulation. Thus, the range in dissolved P concentration was a function of soil P concentration and manure application.

The P index was applied to each plot within the FD-36 watershed. Using soil survey, land management, and topographic information, erosion was calculated by the Revised Universal Soil Loss Equation (RUSLE) and surface runoff by the curve number approach (Sharpley et al., 1998a). Site management factors of the P index were calculated from Mehlich-3 P concentration of surface soil (0–5 cm depth) and P application rate, method, and timing as shown in Table 4. The final P index rating for each plot was calculated as the product of transport and site management factors as described in Table 5. Due to our use of plot data, the evaluation of the P index did not account for landscape factors such as site position or connectivity, which precludes the interpretation of P index rating values by the management categories given in Tables 5 and 6.

Figure 8 illustrates the relationship between plot P index ratings and dissolved P in surface runoff. The two variables were strongly associated (r^2 = 0.78; P = 0.001). This strong association indicates the P index can accurately account for and describe a site's potential for P loss if surface runoff were to occur (Fig. 8).

In addition to this plot-scale assessment of the P index, a watershed-scale validation is required of the index, leaving a number of questions that must be addressed. For instance, are the areas identified to be at greatest risk for P loss, actually sources of most of the P exported? In the same vein, will remediation of high risk areas identified by the index decrease P export in stream flow from a watershed? Conversely, can low

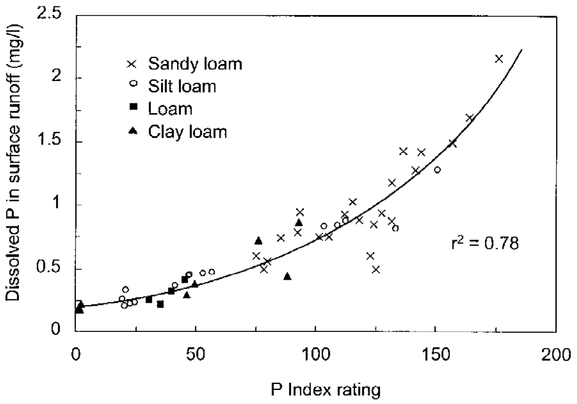


Figure 8. Relationship between the concentration of dissolved P in surface runoff and P index rating 2-m^2 plots within the FD-36 watershed, Northumberland Co., PA.

vulnerability areas receive more liberal P management without increasing P export?

Finally, and perhaps most critical to the use of the P index to as a guide to P management practices, will be the development of overall risk assessment classifications (see Tables 5 and 6). These classifications and interpretations must be developed with careful consideration of local management options, industry infrastructures, and State and Federal policy programs. With further development and testing, the P index will be a valuable tool to identify critical areas of P export, so that alternative management options and remedial measures can be identified. Limited resources and assistance can then be better used to target remedial measures to areas where they will have the most benefit.

Remedial measures

Remedial measures must begin with the long-term objective of increasing P use-efficiency, by attempting to balance P inputs within a watershed with P outputs, while simultaneously improving management of soil, manure and mineral fertilizer P. Reducing P loss in agricultural runoff may be brought about by source and transport control strategies, such as those listed in Table 8. In the past, much attention has been focused on erosion control as a means of controlling P loss from agricultural land. Increasingly, however, attention is being directed toward source management and the control of dissolved P losses in surface runoff.

Source management

Source management attempts to minimize the buildup of P in the soil above levels sufficient for optimum crop growth, by limiting the quantity of P in manure that must be applied to land, and controlling the amount of P that is applied in a localized area. Techniques for source management include:

- Manipulation of dietary P intake by animals may help reduce P inputs in feed; often the major cause of P surplus. Phosphorus intake in excess of minimum dietary requirements do not appear to confer any growth or health advantages and actually decreases profitability through increased feed costs (Knowlton and Kohn, 1999). Carefully matching dietary P inputs to livestock requirements can reduce the amounts of P excreted by animals.
- Increasing the efficiency of P uptake by livestock from feed. A significant amount of the P in grain is in phytate (phytic acid), a form of P that cannot be digested by monogastric animals such as pigs and chickens. As a result, it is common to supplement feed with mineral forms of P, which contribute to P enrichment of manures and litters. Enzymes such as phytase, which break down phytate into forms available to monogastric animals, can be added to feed to increase the efficiency of grain P absorption by pigs and poultry. Such enzymes reduce the need for P supplements in feed and potentially reduce the P content of manure. Also, corn hybrids are available which contain low amounts of indigestible phytate P. Pigs and chickens fed ‘low-phytic acid’ corn grain excreted less P in manure than those fed conventional corn varieties (Ertl et al., 1998).
- Use of manure and soil testing data to improve nutrient management. Farm advisors and resource planners are now recommending that the P content of both manure and soil be determined by soil test laboratories before land application of manure.
- Use of amendments to decrease P solubility in soil and manure. Commercially available manure amendments, such as slaked lime ($CaOH_2$) or alum ($Al_2(SO_4)_3$), are used to reduce ammonia (NH_3) volatilization, leading to improved animal health and weight gains. Coincidentally, these amendments can also greatly decrease the water solubility of P in poultry litter, thereby decreasing dissolved P concentrations in surface runoff (Moore et al., 2000; Shreve et al., 1995). Perhaps the most important benefit of manure amendments

Table 8. Best Management Practices for control of nonpoint sources of agricultural P and N

Practice	Description	Impact on loss[a] P	Impact on loss[a] N
Source Measures			
Feed additives	Enzymes increase nutrient utilization by animals	Decrease	Decrease
Crop hybrids	Low phytic-acid corn reduces P in manure	Decrease	Neutral
Manure management	Compost, lagoons, pond storage; barnyard runoff control; transport excess out of watershed	Decrease	Decrease
Rate added	Match crop needs	Decrease	Decrease
Timing of application	Avoid autumn and winter application	Decrease	Decrease
Method of application	Incorporated, banded, or injected in soil	Decrease	Decrease
Crop rotation	Sequence different rooting depths	Neutral	Decrease
Manure amendment	Alum reduces NH_3 loss and P solubility	Decrease	Decrease
Soil amendment	Flyash, Fe oxides, gypsum reduce P solubility	Decrease	Neutral
Cover crops/residues	If harvested can reduce residual soil nutrients	Decrease TP	Increase DP
Plowing stratified soils	Redistribution of surface P through profile	Decrease	Neutral
Transport Measures			
Cultivation timing	Not having soil bare during winter	Decrease	Decrease
Conservation tillage	Reduced and no-till increases infiltration and reduces soil erosion	Decrease TP Increase DP	Decrease Increase NO_3
Grazing management	Stream exclusion, avoid overstocking	Decrease	Decrease
Buffer, riparian, wetland areas, grassed waterways	Removes sediment-bound nutrients, enhances denitrification	Decrease TP neutral DP	Decrease
Soil drainage	Tiles and ditches enhance water removal and reduce erosion	Decrease TP Increase DP	Decrease TN Increase NO_3
Strip cropping, contour plowing, terraces	Reduces transport of sediment-bound nutrients	Decrease Neutral DP	Decrease Neutral NO_3
Sediment delivery structures	Stream bank protection and stabilization, sedimentation pond	Decrease	Decrease
Critical source area treatment	Target sources of nutrients in a watershed for remediation	Decrease	Decrease

[a] TN is total N, NO_3 is nitrate, TP is total P, and DP is dissolved P.

for both air and water quality would be an increase in the N:P ratio of manure (via reduced N loss because of NH_3 volatilization) that would more closely match crop N and P requirements.

- Transporting manure P from areas of P excess to areas of P deficiency. At present, manure is rarely transported more than 15 km from where it is produced, restricting application options. However, it must be shown that the recipient farms are more suitable for manure application than manure-rich farms and that measures are managed on recipient farms to avoid soil P build up.

- Composting may also be considered as a management tool to improve manure distribution. Although composting tends to increase the P concentration of manure, the volume is reduced and thus, transportation costs are reduced.

- Separating solids from liquids may increase the number of management options available for some types of manure such as dairy and swine. This process results in some separation of the nutrients as well, leaving a large proportion of the available N in the liquid fraction and a large proportion of the P will be in the solid fraction. While this does not change the total amount of nutrients that must be handled, it may enable better targeting of the individual nutrients to locations where they will do the most good and/or have less potential for causing environmental problems. Also, because the solid fraction is more concentrated it may be feasible to transport it to more remote fields.

- Using manure as a source of 'bioenergy'. For example, dried poultry litter can be burned directly or converted by pyrolytic methods into oils suitable for use to generate electric power. Liquid manures can be digested anaerobically to produce methane which can be used for heat and energy.
- Improving management of P application rate, method, and timing to minimize the potential for P loss in runoff. As we have shown, P loss in runoff increases with greater applications of P as mineral fertilizer or manure (Table 7 and Figs. 5 and 6). Incorporation of manure into the soil profile either by tillage or subsurface placement, decreases the potential for P loss in runoff by lowering the concentration of P at the soil surface and a reducing runoff volume (Mueller et al., 1984; Pote et al., 1996).

Transport management

Transport management refers to efforts to control the movement of P from soils to sensitive locations such as bodies of fresh water. Phosphorus loss via surface runoff and erosion may be reduced by conservation tillage and crop residue management, buffer strips, riparian zones, terracing, contour tillage, cover crops and impoundments (e.g., settling basins). Basically, these practices reduce rainfall impact on the soil surface, reduce surface runoff volume and velocity, and increase soil resistance to erosion. Conversion from furrow irrigation to sprinkler to drip irrigation significantly reduces irrigation erosion and runoff. Furrow treatments such as straw mulching and use of polyacrylamides will also reduce in-furrow soil movement (Lentz et al., 1998).

Despite these advantages, any one of these measures should not be relied upon as the sole or primary means of reducing P losses. These practices are generally more efficient at reducing sediment P than dissolved P. Also, P stored in stream and lake sediments can provide a long-term source of P in waters long after inputs from agriculture have been reduced. Several researchers have indicated little decrease in lake productivity with reduced P inputs following implementation of conservation measures (Gray and Kirkland, 1986; Young and DePinto, 1982). Thus, the effect of remedial measures in the contributing watershed will be slow for many cases of poor water quality. Therefore, immediate action may be needed to reduce future problems.

Integrating P and N management

Farm N inputs are usually more easily balanced with plant uptake than are P inputs, particularly where confined livestock operations exist. In the past, separate strategies for either N or P have been developed and implemented at farm or watershed scales. Because of different critical sources, pathways, and sinks controlling P and N export from watersheds, remedial efforts directed at either P or N control can negatively impact the other nutrient (Table 8). For example, basing manure application on crop N requirements to minimize nitrate leaching to ground water can increase soil P and enhance potential P losses (Sharpley et al., 1998b; Sims, 1997). In contrast, reducing surface runoff losses of total P via conservation tillage can enhance N leaching and even increase algal available P transport (Sharpley and Smith, 1994).

These positive and negative impacts of conservation practices on N and P loss potential should be considered in the development of sound remedial measures. Clearly, a technically sound framework must be developed that recognizes critical sources of P and N export from agricultural watersheds so that optimal strategies at farm and watersheds scales can be implemented to best manage both P and N. One approach, explored by Heathwaite et al. (2000) and Sharpley et al. (1998a), is to employ the P index to target P management on critical source areas of P and assume N-based management on all other areas. With such an approach, however, careful consideration must be given to the potential long-term consequences of N management on P loss and vice versa.

Bridging agricultural and environmental management

In order to initiate real and lasting changes in agricultural production, emphasis must be placed on consumer-based programs and education rather than assuming that farmers will absorb the burden. Acceptance of best management practices (BMPs) will not be easy. Because farmers' decisions are generally shaped by regional and often global economic pressures and constraints, which they have little or no control over, there is often reluctance to adopt management practices that do not address these concerns. Clearly, new ways of using incentives to help farmers implement BMPs are needed. The challenge is to recognize how social policy and economic factors influence the nutrient-management agenda.

Equally important is that everyone is affected by and can contribute to a resolution of nutrient-related concerns. Rather than assume that inappropriate farm management is responsible for today's water quality problems, the underlying causes of the symptoms must be addressed. As shown above, much of today's problems relate to marketplace pressures, the breakdown and imbalances in global P cycling, and economic survival of farms. Research is, thus, needed to develop programs that encourage farmer performance and stewardship to achieve previously agreed upon environmental goals. These programs should focus on public participation to resolve conflicts between economic production efficiency and water quality. In the US, there are numerous sources of technical assistance and financial cost-share and loan programs to help defray the costs of constructing or implementing practices that safeguard soil and water resources (US Environmental Protection Agency, 1998). Watershed-based programs, such as the New York City Watershed Agriculture Program, have been established to provide technical assistance and financial support to farmers participating in water quality protection programs (National Research Council, 2000).

Stakeholder alliances encourage collaborative relationships among concerned parties. Such alliances have been formed in response to recent public health issues related to the nutrient enrichment of waters in the eastern US. In the Chesapeake Bay, stakeholder alliances have developed among state, federal, and local groups and the public to work together to identify critical problems, focus resources, include watershed goals in planning, and implement effective strategies to safeguard soil and water resources (Chesapeake Bay Program, 1995, 1998).

In Australia, it was found that an awareness of agricultural or environmental problems and potential solutions did not necessarily cause people to change their behavior to correct such problems (Wilkinson and Cary, 1993). Solutions have to be adapted in practical ways to individual circumstances. The Australian National Land and Water Resources Audit has recognised this by investigating the capacity of rural communities to implement changes to help protect soil and water (National Land and Water Resources Audit, 1998).

One barrier to the design and implementation of BMPs is that the assessment and monitoring implemented by government is often perceived as a top-down process. Walker et al. (1996) recommend a bottom-up process, whereby policy maker and user can select soil and water quality indicators at a local level. Such a process would seem equally if not more fundamental to the successful identification and adoption of new management systems. A concerted attempt has been made in Australia to take this approach by devolving primary responsibility for local monitoring and resource management to land managers themselves through the provision of government funds for the national Landcare and Waterwatch programs (SCARM-ARMCANZ, 1997).

Finally, P applications at recommended rates can reduce P loss in agricultural runoff via increased crop uptake and cover. It is of vital importance that we implement management practices that minimize soil P buildup in excess of crop requirements, reduce surface runoff and erosion, and improve our capability to identify fields that are major sources of P loss to surface waters.

Summary

A growing focus on nutrient transfers from agricultural lands to water has served to accelerate our understanding of the environmental consequences of P management in agriculture. Phosphorus imbalances at farm and watershed scales, often related to concentrated animal feeding operations, aggravate diffuse P losses through the gradual accumulation of P in soils, and the application of P at times of high transport potential. While erosion of particulate P from agricultural soils remains a dominant concern, the transport of dissolved, or soluble P in surface runoff and subsurface flow is also important.

Research at plot, field and watershed scales emphasizes the importance of critical source areas, where high P availability and high transport potential overlap, as major contributors to P losses from agricultural lands to water. The development of tools such as the P index represents a major advance in identifying critical source areas, as highlighted by our research relating P index ratings to plot-scale P losses.

Management strategies to minimize P loss to water may be brought about by optimizing P use-efficiency, refining animal feed rations, using feed additives to increase P absorption by the animal, moving manure from surplus to deficit areas, and targeting conservation practices, such as reduced tillage, buffer strips and cover crops, to critical source areas within a watershed. However, because farmers' decisions are influenced by regional and even global economics over which they have little control, we should explore the use of incentives to aid in implementation of innovative measures that minimize on-farm surpluses of P and reduce P losses.

References

Beegle D 2000 Soil fertility management. *In* Agronomy Guide 1999–2000. Eds. N Seroktin and S Tibbetts. pp 19–46. University Park, PA: Publications Distributions Center, Pennsylvania State University.

Bengston L, Seuna P, Lepisto A and Saxena R K 1992 Particle movement of meltwater in a subdrained agricultural basin. J. Hydrol. 135, 383–398.

Bolinder M A, Simard R R, Beauchemin S and MacDonald K B 1998 Indicator of risk of water contamination: methodology for the phosphorus component. pp 11–21, Agri-environmental Indicator Project Report No. 24, Agriculture and Agri-food Canada.

Burkholder J A and Glasgow H B Jr 1997 *Pfiesteria piscicidia* and other Pfiesteria-dinoflagellates behaviors, impacts, and environmental controls. Limnol. Oceanogr. 42, 1052–1075.

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. Ecol. Appl. 8, 559–568.

Catt J A, Johnston A E and Quinton J N 1997 Phosphate losses in the Woburn erosion reference experiment. *In* Phosphorus Loss from Soil to Water. Eds. H Tunney, O T Carton, P C Brookes and A E Johnston. pp 374–377. CAB International, Wallingford, UK.

Chesapeake Bay Program 1995 *In* The State of the Chesapeake Bay 1995. Eds. R Magnien, D Boward, and S Bieber. U.S. Government Printing Office, Washington, DC. 45 p.

Chesapeake Bay Program 1998 Federal Agencies Chesapeake Ecosystem Unified Plan. Chesapeake Bay Program, Annapolis, MD.

Culley J L B, Bolton E F and Bernyk V 1983 Suspended solids and phosphorus loads from a clay soil: I. Plot studies. J. Environ. Qual. 12, 493–498.

Duxbury J M and Peverly J H 1978 Nitrogen and phosphorus losses from organic soils. J. Environ. Qual. 7, 566–570.

Edwards D R and Daniel T C 1992 Potential runoff quality effects of poultry manure slurry applied to fescue plots. Trans. ASAE 35, 1827–1832.

Edwards D R and Daniel T C 1993a Runoff quality impacts of swine manure applied to fescue plots. Trans Am. Soc. Agric. Eng. 36, 81–80.

Edwards D R and Daniel T C 1993b Drying interval effects on runoff from fescue plots receiving swine manure. Trans. Am. Soc. Agric. Eng. 36, 1673–1678.

Eghball B and Gilley J E 1999 Phosphorus risk assessment index and comparison with results of runoff studies. p 30. *In* Agronomy Abstracts 1999. Am. Soc. Agron., Madison, WI.

Ertl D S, Young K A and Raboy V 1998 Plant genetic approaches to phosphorus management in agricultural production. J. Environ. Qual. 27, 299–304.

Gburek W J and Sharpley A N 1998 Hydrologic controls on phosphorus loss from upland agricultural watersheds. J. Environ. Qual. 27, 267–277.

Gburek W J, Sharpley A N, Heathwaite A L, and Folmar G J 2000 Phosphorus management at the watershed scale: a modification of the phosphorus index. J. Environ. Qual. 29, 130–144.

Gray C B J and Kirkland R A 1986 Suspended sediment phosphorus composition in tributaries of the Okanagan Lakes, BC. Water Res. 20, 1193–1196.

Haygarth P M and Sharpley A N 2000 Terminology for phosphorus transfer. J. Environ. Qual. 29, 10–15.

Haygarth P M, Hepworth L and Jarvis S C 1998 Form of phosphorus transfer and hydrological pathways from soil under grazed grassland. Eur. J. Soil Sci. 49, 65–72.

Heathwaite A L, Griffiths P, Haygarth P M, Jarvis S C and Parkinson R J 1997 Phosphorus loss from grassland soils: implications of land management for the quality of receiving waters. pp 177-186. *In* Freshwater Contamination, Proc. Rabat Symp., April–May 1997, IHAS Publ. No 243.

Heathwaite A L, Sharpley A N and Gburek W J 2000 Integrating phosphorus and nitrogen management at catchment scales. J. Environ. Qual. 29, 158–166.

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. J. Environ. Qual. 24, 904–910.

Jokela W, Magdoff F and Durieux R 1997 Soil testing for improved phosphorus management. Vermont Cooperative Extension Service, Burlington, VT.

Kleinman P J A, Bryant R B, Reid W S, Sharpley A N and Pimental D 2000. Using soil phosphorus behavior to identify environmental thresholds. Soil Sci.

Knowlton K F and Kohn R 1999 We've got to stop overfeeding phosphorus. Hoard's Dairyman 144, 437.

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 Res. 27, 495–506.

Lander C H, Moffitt D, and Alt K 1998 Nutrients Available from Livestock Manure Relative to Crop Growth Requirements. Resource Assessment and Strategic Planning Working Paper 98–1. US Department of Agriculture, Natural Resources Conservation Service, Washington, DC. (http://www.nhq.nrcs.usda.gov/land/pub s /nlweb.html).

Lanyon L E 2000 Nutrient management: Regional issues affecting the Chesapeake Bay. *In* Agriculture and Phosphorus Management: The Chesapeake Bay. Ed. A N Sharpley. CRC Press, Boca Raton, FL.

Lemunyon J L and Gilbert R G 1993 The concept and need for a phosphorus assessment tool. J. Prod. Agric. 6, 483–496.

Lentz R D, Sojka R E, and Robbins C W 1998 Reducing phosphorus losses from surface-irrigated fields: emerging polyacrylamide technology. J. Environ. Qual. 27, 305–312..

Leytem A B, Sims J T, Coale F J Sharpley A N and Gburek W J 1999 Implementing a phosphorus site index for the Delmarva Peninsula: challenges and research needs. pp. 336. *In*: Agronomy Abstracts 1999. Am. Soc. Agron., Madison, WI.

Lory J A and Scharf P C 2000 Threshold phosphorus survey. SERA-17 Minimizing phosphorus losses from agriculture. (http://ces.soil.ncsu.edu/sera17/).

McColl R H S, White E and Gibson A R 1977 Phosphorus and nitrate runoff in hill pasture and forest catchments, Taita, New Zealand. N.Z. J. Mar. Freshwater Res. 11, 729–744.

McDowell L L and McGregor K C 1984 Plant nutrient losses in runoff from conservation tillage corn. Soil Tillage Res. 4, 79–91.

McDowell R W and Sharpley A N 2001 Approximating phosphorus release to surface runoff and subsurface drainage. J. Environ. Qual. 30, 508–601.

McDowell R W, Sharpley A N, Brookes P A and Poulton P 2001 Relationships between soil test phosphorus and phosphorus release to solution. Soil Sci. (in press).

McFarland A, Hauck L, White J, Donham W, Lemunyon J and Jones S 1998 Nutrient management using a phosphorus risk index for manure application fields. pp 241–244. Proceedings of Manure Management in Harmony with the Environment and Society, Soil and Water Conservation Society, Feb. 10–12, 1998, Ames, IA.

Miller M H 1979 Contribution of nitrogen and phosphorus to subsurface drainage water from intensively cropped mineral and organic soils in Ontario. J. Environ. Qual. 8, 42–48.

Miller W P 1987 A solenoid-operated, variable intensity rainfall simulator. Soil Sci. Soc. Am. J. 51, 832–834.

Ministry of Agriculture, Food and Fisheries 1994 Fertilizer recommendations for agricultural and horticultural crops. Ministry of Agriculture, Fisheries, and Food, Reference Book 209. HMSO, London, England. 48 p.

Moore P A Jr, Daniel T C and Edwards D R 2000 Reducing phosphorus runoff and inhibiting ammonia loss from poultry manure with aluminum sulfate. J. Environ. Qual. 29, In press.

Mueller D H, Wendt R C and Daniel T C 1984 Phosphorus losses as affected by tillage and manure application. Soil Sci. Soc. Am. J. 48, 901–905.

National Land and Water Resources Audit 1998 National Land and Water Resources Audit: Draft Strategic Plan — 1998–2001. National Land and Water Resources Audit, Canberra, Australia.

National Research Council (NRC) 2000 Watershed Management for Potable Water Supply. National Academy Press, Washington, DC.

Nicholaichuk W and Read D W L 1978 Nutrient runoff from fertilized and unfertilized fields in western Canada. J. Environ. Qual. 7, 542–544.

Pionke H B, Gburek W J, Sharpley A N, and Zollweg J A 1997 Hydrologic and chemical controls on phosphorus losses from catchments. *In* Phosphorus Loss to Water from Agriculture. Eds. H Tunney, O Carton and P Brookes. pp 225–242. CAB International, Cambridge, England.

Pote D H, Daniel T C, Sharpley A N, Moore P A Jr, Edwards D R and Nichols D J 1996 Relating extractable phosphorus to phosphorus losses in runoff. Soil Sci. Soc. Am. J. 60, 855–859.

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. J. Environ. Qual. 28, 170–175.

SCARM-ARMCANZ 1997 Evaluation Report on the Decade of Landcare Plan — National Overview, Australian Government Publishing Service, Canberra, Australia.

Schoumans O F and Breeuwsma A 1997 The relation between accumulation and leaching of phosphorus: laboratory, field and modelling results. *In* Phosphorus loss from soil to water. Eds. H Tunney, O T Carton, P C Brookes and A E Johnston. pp 361–363. CAB International Press, Cambridge, UK.

Sharpley A N 1983 Effect of soil properties on the kinetics of phosphorus desorption. Soil Sci. Soc. Am. J. 47: 462–467.

Sharpley A N 1985a Depth of surface soil-runoff interaction as affected by rainfall, soil slope, and management. Soil Sci. Soc. Am. J. 49, 1010–1015.

Sharpley A N 1985b The selective erosion of plant nutrients in runoff. Soil Sci. Soc. Am. J. 49, 1527–1534.

Sharpley A N 1995 Identifying sites vulnerable to phosphorus loss in agricultural runoff. J. Environ. Qual. 24, 947–951.

Sharpley A N 1997 Rainfall frequency and nitrogen and phosphorus in runoff from soil amended with poultry litter. J. Environ. Qual. 26, 1127–1132.

Sharpley A N 1999 Phosphorus availability. Section D, pages 18–38. *In* M. Sumner, Handbook of Soil Science. CRC Press, Boca Raton, FL.

Sharpley A N 2000 Editor Agriculture and Phosphorus Management: The Chesapeake Bay. CRC Press, Boca Raton, FL.

Sharpley A N and Rekolainen S 1997 Phosphorus in agriculture and its environmental implications. *In* Phosphorus Loss from Soil to Water. Eds. H Tunney, O T Carton, P C Brookes and A E Johnston. pp 1–54. CAB International Press, Cambridge, UK.

Sharpley A N and Smith S J 1994 Wheat tillage and water quality in the Southern Plains. Soil Tillage Res. 30: 33–48.

Sharpley A N and Syers J K 1976 Phosphorus transport in surface runoff as influenced by fertilizer and grazing cattle. N.Z. J. Sci. 19, 277–282.

Sharpley A N and Syers J K 1979 Loss of nitrogen and phosphorus in tile drainage as influenced by urea application and grazing animals. N.Z. J. Agric. Res. 22, 127–131.

Sharpley A N and Tunney H 2000 Phosphorus research strategies to meet agricultural and environmental challenges of the 21st century. J. Environ. Qual. 29, 176–181.

Sharpley A N, Daniel T C, Sims J T and Pote D H 1996 Determining environmentally sound soil phosphorus levels. J. Soil Water Conserv. 51, 160–166.

Sharpley A N, Daniel T C , Sims J T, Lemunyon J, Steven R A and Parry R 1999a Agricultural phosphorus and eutrophication. US Department of Agriculture — Agricultural Research Service, ARS-149 U.S. Govt. Printing Office, Washington, DC 34 p.

Sharpley A N, Gburek W J and Folmar G 1998a Integrated phosphorus and nitrogen management in animal feeding operations for water quality protection. *In* Animal Feeding Operations and Ground Water: Issues, Impacts, and Solutions. Eds. R W Masters and D Goldman. pp. 72–95. National Ground Water Association, Westerville, OH.

Sharpley A N, Gburek W J and Heathwaite A L 1998b Agricultural phosphorus and water quality: Sources, transport and management. J. Agric. Food Chem. Finland 7, 297–314.

Sharpley A N, Smith S J, Williams J R, Jones O R and Coleman G A 1991 Water quality impacts associated with sorghum culture in the Southern Plains. J. Environ. Qual. 20, 239–244.

Sharpley A, Daniel T, Wright B, Kleinman P, Sobecki T, Parry R and Joern B 1999b National research project to identify sources of agricultural phosphorus loss. Better Crops 83, 12–15.

Sharpley A N, Meisinger J J, Breeuwsma A., Sims T, Daniel T C and Schepers J S 1999c Impacts of animal manure management on ground and surface water quality. p. 173–242. *In* J. Hatfield, Effective Management of Animal Waste as a Soil Resource. Ann Arbor Press, Chelsea, MI.

Shreve B R, Moore P A Jr, Daniel T C, Edwards D R and Miller D M 1995 Reduction of phosphorus in runoff from field-applied poultry litter using chemical amendments. J. Environ. Qual. 24, 106–111.

Sims J T 1996 The Phosphorus Index: a phosphorus management strategy for Delaware's agricultural soils. Fact Sheet ST-05, Delaware Cooperative Extension Service, Newark. DE.

Sims J T 1997 Agricultural and environmental issues in the management of poultry wastes: recent innovations and long-term challenges. *In* Uses of By-Products and Wastes in Agriculture. Eds. J Rechcigl and H C MacKinnon. pp 72–90. Am. Chem. Soc., Washington, DC.

Sims J T and Sharpley A N 1998 Managing agricultural phosphorus for water quality protection: future challenges. *In* Soil Testing for Phosphorus: Environmental Uses and Implications. Ed. J T Sims. pp. 41–43. Southern Cooperative Series Bulletin No. 389, SERA-IEG 17, University of Delaware, Newark, DE.

Sims J T, Simard R R and Joern B C 1998 Phosphorus losses on agricultural drainage: historical perspectives and current research. J. Environ. Qual. 27, 277–293.

Smith S J, Sharpley A N, Naney J W, Berg W A and Jones O R 1991 Water quality impacts associated with wheat culture in the Southern Plains. J. Environ. Qual. 20, 244–249.

Uhlen G 1988 Surface runoff losses of phosphorus and other nutrient elements from fertilized grassland. Norwegian J. Agric. Sci. 3, 47–55.

US Department of Agriculture and US Environmental Protection Agency 1999 Unified national strategy for Animal Feeding Operations. March 9, 1999. (http://www.epa.gov/owm/finafost.htm).

US Environmental Protection Agency 1996 Environmental indicators of water quality in the United States. EPA 841-R-96-002. US EPA, Office of Water (4503F), U.S. Govt. Printing Office, Washington, DC.

US Environmental Protection Agency 1998 Water quality criteria and standards plan: priorities for the future. EPA-823-F-98-011. US EPA, Office of Water (4304F), US Govt. Printing Office, Washington, DC.

US Geological Survey 1999 The quality of our nation's waters: Nutrients and pesticides. US Geological Survey Circular 1225, 82 p. USGS Information Services, Denver, CO. http://www.usgs.gov.

Walker J, Alexander D, Irons C, Jones B, Penridge H and Rapport D 1996 Catchment health indicators: an overview. *In* Indicators of Catchment Health: a Technical Perspective. Eds. J Walker and D J Reuters. pp 3–18. CSIRO, Collingwood, Victoria, Australia.

Westerman P W, Donnely T L and Overcash M R 1983 Erosion of soil and poultry manure — a laboratory study. Trans. Am. Soc. Agric. Eng. 26, 1070–1078, 1084.

White R E and Thomas G W 1981 Hydrolysis of aluminum on weakly acidic organic exchangers: implications for phosphorus adsorption. Fertil. Res. 2, 159–167.

Wilkinson R L and Cary J W 1993 Monitoring Soilcare in North East Victoria. Report of the School of Agriculture and Forestry, University of Melbourne, Melbourne, Australia.

Withers P J A, Davidson I A and Foy R H 2000. Prospects for controlling diffuse phosphorus loss to water. Environ Qual. 29, 167–175.

Young T C and DePinto J V 1982 Algal–availability of particulate phosphorus from diffuse and point sources in the lower Great Lakes basin. Hydrobiologia 91: 111–119.

Young R A and Mutchler C K 1976 Pollution potential of manure spread on frozen ground. J. Environ. Qual. 5, 174–179.

Attachment 38:

Assessment of Chlorophyll-*a* as a Criterion for Establishing Nutrient Standards in the Streams and Rivers of Illinois
Royer et al. 2008

Assessment of Chlorophyll-*a* as a Criterion for Establishing Nutrient Standards in the Streams and Rivers of Illinois

Todd V. Royer* Indiana University
Mark B. David, Lowell E. Gentry, Corey A. Mitchell, and Karen M. Starks University of Illinois at Urbana-Champaign
Thomas Heatherly II and Matt R. Whiles Southern Illinois University

Nutrient enrichment is a frequently cited cause for biotic impairment of streams and rivers in the USA. Efforts are underway to develop nutrient standards in many states, but defensible nutrient standards require an empirical relationship between nitrogen (N) or phosphorus (P) concentrations and some criterion that relates nutrient levels to the attainment of designated uses. Algal biomass, measured as chlorophyll-*a* (chl-*a*), is a commonly proposed criterion, yet nutrient–chl-*a* relationships have not been well documented in Illinois at a state-wide scale. We used state-wide surveys of >100 stream and river sites to assess the applicability of chl-*a* as a criterion for establishing nutrient standards for Illinois. Among all sites, the median total P and total N concentrations were 0.185 and 5.6 mg L^{-1}, respectively, during high-discharge conditions. During low-discharge conditions, median total P concentration was 0.168 mg L^{-1}, with 25% of sites having a total P of $\geq$0.326 mg L^{-1}. Across the state, 90% of the sites had sestonic chl-*a* values of $\leq$35 μg L^{-1}, and watershed area was the best predictor of sestonic chl-*a*. During low discharge there was a significant correlation between sestonic chl-*a* and total P for those sites that had canopy cover $\leq$25% and total P of $\leq$0.2 mg L^{-1}. Results suggest sestonic chl-*a* may be an appropriate criterion for the larger rivers in Illinois but is inappropriate for small rivers and streams. Coarse substrate to support benthic chl-*a* occurred in <50% of the sites we examined; a study using artificial substrates did not reveal a relationship between chl-*a* accrual and N or P concentrations. For many streams and rivers in Illinois, nutrients may not be the limiting factor for algal biomass due to the generally high nutrient concentrations and the effects of other factors, such as substrate conditions and turbidity.

The Federal Clean Water Act requires states to identify impaired water bodies and develop plans to reduce impairment. Nutrient enrichment, mainly with nitrogen (N) and phosphorus (P), is a frequently cited cause of impairment for streams and rivers (USEPA, 2000a). Nutrient loading can degrade the ecological integrity of streams and create human health concerns. For example, the U.S. Environmental Protection Agency (USEPA) has set a drinking water standard of 10 mg NO_3–N L^{-1} to prevent methemoglobenemia. No drinking water standard exists for P; however, P enrichment can affect drinking water supplies by stimulating blooms of toxin-producing organisms, such as cyanobacteria. Nutrient enrichment in streams stimulates algal growth with resulting impacts on habitat quality, trophic relations, community structure, dissolved O_2 concentrations, pH, and aesthetic qualities (e.g., Miltner and Rankin, 1998).

The goal in developing nutrient standards for streams and rivers is to prevent a particular ecological condition (e.g., excessive algal biomass) by controlling the presumably limiting factor for algal growth. The rationale behind nutrient standards is that ecological impairment in nutrient-enriched streams is due, at least in part, to excess algal biomass and the effect of the excess biomass on dissolved O_2 deficits via respiration and decomposition. However, the cause-and-effect relationship among nutrients, algal biomass, and O_2 deficits is complicated by other environmental factors that can maintain low algal biomass despite abundant nutrients (Dodds and Welch, 2000). For example, scouring, shading, grazing, and temperature can affect algal biomass independently of nutrient enrichment. Further complicating the development of nutrient standards is the fact that algae in streams and rivers occur in multiple forms, such as sestonic cells, epilithic biofilms, and filamentous mats. These various forms may differ in their response to nutrient enrichment and the degree to which they are affected by other environmental factors.

Chlorophyll-*a* (chl-*a*) is a commonly used proxy for algal biomass and has been proposed as a criterion for identifying streams that fail to

Published in J. Environ. Qual. 37:437–447 (2008).
doi:10.2134/jeq2007.0344
Received 28 June 2007.
*Corresponding author (troyer@indiana.edu).
© ASA, CSSA, SSSA
677 S. Segoe Rd., Madison, WI 53711 USA

Todd V. Royer, School of Public & Environmental Affairs, Indiana Univ., Bloomington, IN 47405. Mark B. David, Lowell E. Gentry, Corey A. Mitchell, and Karen M. Starks, Dep. of Natural Resources and Environmental Sciences, Univ. of Illinois at Urbana-Champaign, Urbana, IL 61801. Lowell E. Gentry, current address: Dep. of Crop and Soil Sciences, Michigan State Univ., East Lansing, MI 48824. Thomas Heatherly II and Matt R. Whiles, Dep. of Zoology and Center for Ecology, Southern Illinois Univ., Carbondale, IL 62901.

Abbreviations: chl-*a*, chlorophyll-*a*; DRP, dissolved reactive phosphorus; Q, discharge.

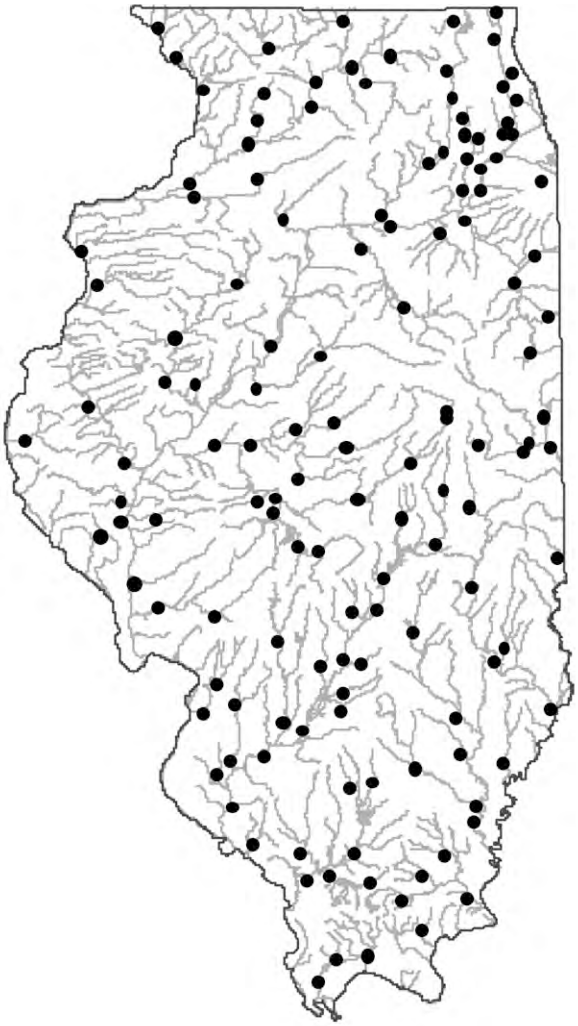


Fig. 1. Map of Illinois showing the major river networks and the distribution of the 138 sites used for the study.

attain their designated use(s) due to nutrient loading (e.g., USEPA, 2000b). Because of the large number of streams and rivers that will likely require management intervention to reduce nutrient loading, it is critical that the selected criterion be strongly associated with numerical water quality standards and attainment of designated uses (Reckhow et al., 2005). In Illinois, the strength of the relationship between nutrients and chl-*a* has not been assessed at a state-wide scale, and the appropriateness of sestonic or benthic chl-*a* as a state-wide criterion for establishing nutrient standards is unknown. Previous work in Illinois suggested that in wadeable agricultural streams sestonic chl-*a* was a poor indicator of eutrophic conditions (Figueroa-Nieves et al., 2006; Morgan et al., 2006), but its applicability in larger rivers has not been examined.

Across large geographic regions, such as states or level III ecoregions (e.g., Woods et al., 2006), it is often difficult to statistically relate nutrient concentrations to algal biomass (as chl-*a*) due to spatial and temporal variations in hydrology, light, temperature, and land use factors that influence algal abundance (Dodds et al., 2002). Nevertheless, the development of nutrient standards for streams will likely occur independently within each state, potentially with the goal of producing single, state-wide standards for N and P. To assist the state of Illinois with development of nutrient standards, we examined nutrient–chlorophyll relationships throughout the streams and rivers of the state. Because the state anticipates single, state-wide standards, our approach was at the state-wide scale. Our goals included examination of state-wide relationships between nutrients and chl-*a* as well as more mechanistically focused studies aimed at identifying environmental factors other than nutrients that may affect algal biomass in the streams and rivers of Illinois.

Materials and Methods

State-wide Surveys

Quantifying the ecological response to nutrient enrichment across Illinois presents a challenge due to the diversity of stream types and land uses within Illinois. Illinois covers slightly more than 150,000 km^2 and has a latitudinal gradient of 627 km, from 36°58′ N at the south to 42°30′ N at the northern boundary. Much of Illinois is rural and dominated by intensive row-crop agriculture with large inputs of N and P fertilizer (David and Gentry, 2000). The southern region of Illinois has more extensive tracks of hardwood forests and includes the Shawnee National Forest. Streams in the northeastern portion of the state are influenced strongly by urbanization, with the Illinois River system receiving the wastewater effluent from the approximately 8 million people living in the greater Chicago region. Statewide, land cover is 76% agricultural, 12% forest, 6% urban, 4% wetland, and 2% other uses (Illinois Department of Agriculture, 2001).

We conducted two state-wide surveys in 2004 designed to document conditions during distinct seasonal and hydrological conditions. A smaller state-wide survey was conducted in 2005 in conjunction with the artificial substrate study described below. Sites selected for the survey ranged in size from small, wadeable streams to the large rivers of the state but did not include the Mississippi, Ohio, or Wabash Rivers because these rivers were not wadeable even at low discharge and could not be safely sampled from bridges. Most sites corresponded to locations used by Illinois EPA in their ambient water quality monitoring network. The goal in site selection was to identify a representative group of sites that would allow for generalization to the streams and rivers of the state as a whole. A complete listing of all sites is presented in Appendix A.

The first survey examined 138 sites distributed across the state (Fig. 1) and was conducted from May to early July when most of the streams were at higher than baseflow discharge (Q) but not flooded. An analysis of 103 of the sites that were gauged by the US Geological Survey indicated that, at the time of sampling, average discharge across the sites was 81% of the long-term mean discharge for the month of May. For the second survey we revisited 109 of the sites during September when the streams were at baseflow. In this paper we refer to the first survey as the high-Q survey and the second survey as the low-Q survey. During all surveys, we sampled sestonic chl-*a* and benthic chl-*a* (if present), estimated canopy cover, and collected water samples for the determination of total P, dissolved reactive P (DRP), organic P, total N, NO_3–N, NH_4–N, organic N, and dissolved silica. We used portable probes and meters to make on-site measurements of water temperature, pH, specific conductivity, and turbidity. Turbidity was measured at three locations across the width of each stream and averaged. Samples for dissolved constituents were filtered through a 0.45-μm membrane. Sample processing and preservation followed standard procedures (APHA, 1998).

At each site we established three cross-sectional transects separated by approximately 50 m. At each transect that had gravel or cobble substrate, we collected a representative rock for determination of benthic chl-*a* density. We did not attempt to sample benthic chl-*a* from soft sediments or sand. At the most up-stream transect, we collected three 500-mL samples from the left, right, and center of the channel for determination of sestonic chl-*a* concentration. Samples for chl-*a* were stored in the dark on ice until they were processed at the end of each day. For each sestonic chl-*a* sample, a known volume of water was filtered through a Whatman GF/F filter (0.7 μm), and the filters were immediately placed in individual plastic Petri dishes, wrapped in aluminum foil, and placed on ice. Rocks were individually wrapped and placed on ice for transport to the laboratory where they were processed within 30 d of collection (see below).

Nutrient and Chlorophyll Analyses

Nitrate concentrations were measured using ion chromatography (DX-120; Dionex, Sunnyvale, CA) with a detection limit of 0.1 mg L^{-1} of NO_3–N. Ammonium, DRP, and silica concentrations were analyzed colorimetrically by flow injection analysis with a QuikChem 8000 (Lachat, Loveland, CO) using the automated sodium salicylate, the automated ascorbic acid, and the automated heteropoly blue methods, respectively. Method detection limits were 0.01 mg NH_4–N L^{-1}, 0.005 mg P L^{-1}, and 0.2 mg SiO_2 L^{-1}. Water samples for total P were digested with sulfuric acid (11.2 N) and ammonium persulfate (0.4 g per 50 mL of sample), which converted all forms of P into DRP, and then analyzed as described previously. Samples for total N were digested with sulfuric acid, copper sulfate, and potassium sulfate in an aluminum block digester (BD-46; Lachat) that converted organic N compounds to ammonia, which was then analyzed as described previously. Organic N and P were determined as the difference between the total and the dissolved inorganic forms of each nutrient.

In general, we followed the procedures for chl-*a* analysis as described in detail by Morgan et al. (2006), with some modifications. Samples for sestonic chl-*a* were extracted in the dark with 90% acetone for 24 h at 4°C. For benthic chl-*a*, rocks were thoroughly scraped of material using a wire brush, and the dislodged material was collected onto a Whatman GF/F filter. Chl-*a* was extracted in the dark at 4°C for 24 h with 90% ethanol. For sestonic and benthic samples, 30 s of sonication was used to promote extraction. Chlorophyll-*a* was determined using a UV-Vis spectrophotometer (Aquamate; ThermoElectron, Waltham, MA). To correct for pheophytin, absorbance was determined before and after acidification as described in Morgan et al. (2006). The areal surface on the rocks from which the benthic chl-*a* was collected was determined using the aluminum foil method (Steinman and Lamberti, 1996).

Embarras River and Kaskaskia River Surveys

The 2004 surveys revealed unexpectedly high concentrations of sestonic chl-*a* in some of the larger rivers of the state. We examined this phenomenon further in 2005 by conducting synoptic surveys on the Embarras and Kaskaskia rivers. The surveys included sampling sestonic chl-*a* at multiple sites on the mainstem rivers and in several tributaries of each river. An 82-km study reach was used on the Embarras River, and a 36-km reach was used on the Kaskaskia. The goal was to determine if the high mainstem concentrations were the result of tributary loading or in-channel production. All methods were as described previously.

Artificial Substrate Study

Substrate condition is a critical factor for the development of periphyton in streams and can vary substantially among sites. We attempted to reduce the effects of this confounding variable by placing unglazed ceramic tiles at 35 sites across the state and measuring benthic chl-*a* accrual. The goal was to examine nutrient–chl-*a* relationships when provided a common and suitable benthic substrate. At nine sites, the water was too deep to place tiles, but sestonic chl-*a* and nutrient samples were collected. At the remaining 26 sites, a 20 cm × 20 cm tile was placed near the center of the channel at each of three transects and anchored in place with reinforcing bars. This method may not have accounted for the accrual of floating filamentous algal mats, but floating mats of algae were not observed at the time of placement or retrieval of the tiles at any of the sites. Tiles were placed in July 2005 and retrieved after 5 wk of incubation. Sestonic chl-*a* and nutrient samples were collected in conjunction with placement and retrieval of the tiles and analyzed as described previously. Benthic chl-*a* on the tiles was determined by thoroughly scraping a known area and processing the sample as described previously.

Data Analysis

Relationships between chl-*a* and environmental variables (including nutrients) were examined with Pearson correlation analysis or simple linear regression. Differences between the high-Q and low-Q surveys were examined with a two-sample *t* test if the data were normally distributed or with a nonparametric test if data could not be normalized. Normality of all data sets was examined with the Kolmogorov-Smirnov test ($\alpha = 0.05$). Water temperature was normally distributed, and the benthic chl-*a* data were normalized with a $\log_{10}(X + 1)$ transformation (Zar, 1999). Sestonic chl-*a*, nutrient, and turbidity data could not be normalized and therefore were examined with the Kruskal–Wallis nonparametric test. All statistical analyses were conducted with MINITAB release 14.2.

Results

Water Chemistry

Water temperatures were not different between the high-Q and low-Q surveys ($p = 0.143$) and averaged 21°C during both time periods. There was a large range in specific conductivity among the streams, from approximately 100 to >2000 μS cm^{-1}, but there was little difference between the high-Q and low-Q pe-

Table 1. Distribution of water chemistry values from the 2004 state-wide surveys.

	Minimum	25th Percentile	Median	75th Percentile	Maximum
High-Q† survey (May–July, n = 138)					
DRP‡ (mg L^{-1})	<0.005	0.038	0.069	0.156	1.9
Total P (mg L^{-1})	0.013	0.123	0.185	0.326	2.0
NH_4–N (mg L^{-1})	0.008	0.040	0.058	0.089	0.387
NO_3–N (mg L^{-1})	0.10	1.0	4.3	10.2	20.2
Total N (mg L^{-1})	0.37	2.2	5.6	11.0	20.9
Silica (mg L^{-1})	1.5	6.7	9.6	11.8	16.6
pH	7.0	7.7	7.9	8.1	8.7
Specific conductivity (µS cm^{-1} @ 25°C)	106	586	658	751	2240
Turbidity (NTU§)	<1	21	36	61	614
Low-Q survey (Sept., n = 109)					
DRP (mg L^{-1})	0.001	0.029	0.081	0.345	2.8
Total P (mg L^{-1})	0.007	0.112	0.168	0.456	2.8
NH_4–N (mg L^{-1})	0.002	0.011	0.022	0.042	0.696
NO_3–N (mg L^{-1})	<0.05	0.18	1.5	3.9	18.0
Total N (mg L^{-1})	0.21	1.0	2.5	5.0	18.7
Silica (mg L^{-1})	1.3	6.4	8.6	11.2	29.2
pH	6.8	7.6	7.9	8.2	8.9
Specific conductivity (µS cm^{-1} @ 25°C)	132	556	664	814	3246
Turbidity (NTU)	<1	10	18	29	159

† Low-Q, low discharge; high-Q, high discharge.

‡ Dissolved reactive phosphorus.

§ Nephelometric turbidity units

riods (Table 1). Turbidity declined significantly between the two surveys (p < 0.001) from a median of 36 nephelometric turbidity units during the high-Q period to a median of 18 nephelometric turbidity units during the low-Q period. The distribution of nutrient concentrations during the high-Q and low-Q periods is shown in Table 1. Across the state, the median total P concentration was 0.185 mg L^{-1} during the high-Q survey and 0.168 mg L^{-1} during the low-Q survey. There was no statistical difference between the two surveys in total P or DRP. The maximum DRP and total P values of ≥2 mg L^{-1} were recorded from streams in which the discharge was dominated by wastewater effluent.

Total N, nitrate, and ammonium concentrations were significantly lower during the low-Q survey than during the high-Q survey, but even during the low-Q period 75% of the streams had a total N concentration of 1.0 mg L^{-1} or greater (Table 1). The high nitrate concentrations reflect the heavily fertilized, agricultural landscape that typifies much of Illinois. As total N concentrations increased, nitrate comprised a greater fraction of the total N, particularly during the high-Q survey. Ammonium concentrations were generally 1 to 2 orders of magnitude lower than nitrate concentrations, and only 25% of the sites had ammonium N concentrations >0.089 mg L^{-1} during the high-Q survey or >0.042 mg L^{-1} during the low-Q survey.

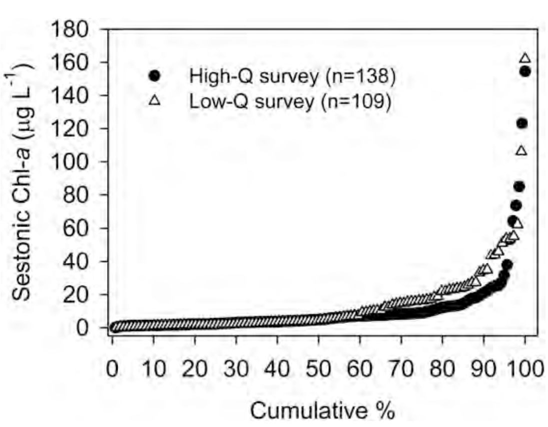


Fig. 2. Distribution of sestonic chlorophyll-*a* (chl-*a*) concentrations across the state of Illinois during the 2004 high-discharge (high-Q) and low-Q surveys (see text for explanation). See Appendix A for the list of sites used in each survey.

Sestonic chl-*a*

The median sestonic chl-*a* value was 5 µg L^{-1} during the high-Q and low-Q surveys, and statistically there was no difference in sestonic chl-*a* concentrations during the two time periods (p = 0.642) (Fig. 2). Across the state, 90% of the sites had sestonic chl-*a* values of ≤35 µg L^{-1}. There was no correlation between benthic chl-*a* (see below) and sestonic chl-*a*, suggesting that sloughing of periphyton was not the major source of algal cells to the water column. During all surveys, watershed area was the best predictor of sestonic chl-*a* (Fig. 3).

Large streams and rivers are capable of supporting planktonic algal communities and can accumulate sestonic cells from tributary inputs. The synoptic surveys on the Embarras and Kaskaskia Rivers suggested that the direct relationship between watershed area and sestonic chl-*a* was due mainly to in-channel production rather than to tributary inputs. Across the 82-km study reach on the Embarras River, mainstem sestonic chl-*a* values increased from 52 to 97 µg L^{-1}. In the Kaskaskia, mainstem sestonic chl-*a* increased from 30 to 86 µg L^{-1} along the 36-km study reach. The mainstem Embarras had a mean sestonic chl-*a* concentration of 69 µg L^{-1} (SD = 19; n = 5), whereas the tributaries had a mean concentration of 11 µg L^{-1} (SD = 12; n = 5). In the Kaskaskia, the mainstem had a mean of 63 µg L^{-1} (SD = 28; n = 4), whereas the tributaries had a mean of 8 µg L^{-1} (SD = 7; n = 9). We were unable to calculate sestonic chl-*a* loads because discharge data were not available for the tributaries. However, the tributaries were significantly smaller than the mainstem rivers, indicating that tributary loading could not account for the downstream increase in sestonic chl-*a* observed in both mainstem rivers.

There was no relationship between sestonic chl-*a* and any nutrient measure or other environmental factor during the high-Q survey. During the low-Q survey, there was no correlation between sestonic chl-*a* and total P for the data set as a whole. However, for those sites that had both canopy cover ≤25% and total P of ≤0.2 mg L^{-1} there was a correlation

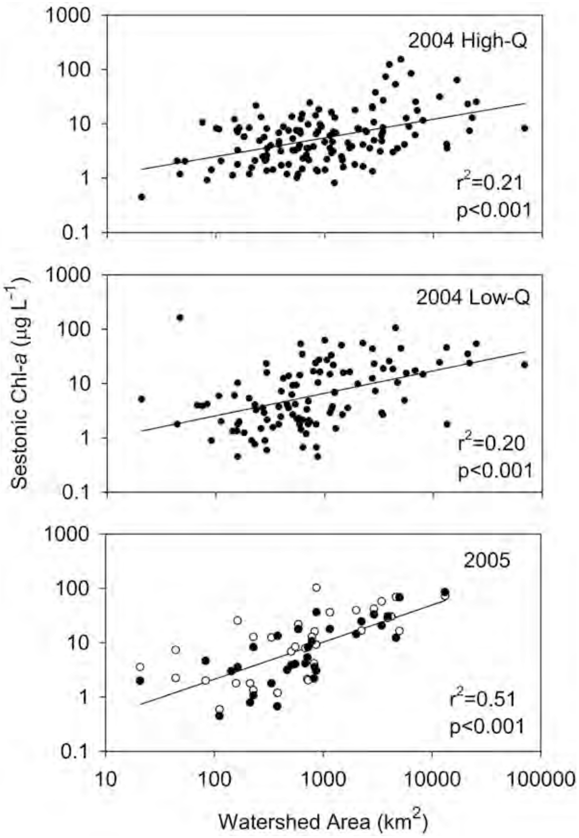


Fig. 3. Relationship between watershed area and sestonic chlorophyll-*a* (chl-*a*) concentrations across the state of Illinois during the 2004 surveys (see text for explanation) and during 2005. Samples for 2005 were collected in July (open symbols, n = 35) and 5 wk later in August or September (filled symbols, n = 35). See Appendix A for the list of sites used in each survey.

between total P and sestonic chl-*a* (Pearson correlation = 0.62; $p < 0.001$) (Fig. 4). Based on the 38 sites that met these criteria, there appeared to be a threshold value for total P of about 0.07 mg L^{-1}. Below that threshold, sestonic chl-*a* was $\leq$5 µg L^{-1}, whereas sestonic chl-*a* ranged from 1 to 55 µg L^{-1} among sites with $\geq$0.07 mg L^{-1} of total P and an open canopy.

Benthic chl-*a*

Less than 50% of the sites sampled during both the high-Q and low-Q surveys contained gravel or cobble substrate for analysis of benthic chl-*a*. Among the sites that contained suitable substrate, the median benthic chl-*a* value was 3 mg m^{-2} during the high-Q survey (n = 31) and 14 mg m^{-2} during the low-Q survey (n = 46). There was a significant increase in benthic chl-*a* between the two time periods (t = 7.04; $p < 0.001$; df = 74) (Fig. 5). During the low-Q survey, there was no relationship between benthic chl-*a* and any nutrient measure among the 46 sites that contained coarse substrate. During the high-Q survey, however, there was a weak correlation between total N and log-transformed benthic chl-*a* (Pearson correlation = 0.33; p = 0.07).

Of the 26 sites at which tiles were placed, 20 accrued benthic chl-*a* during the 5-wk incubation. At the other six sites, the tiles were buried by shifting sand and fine sediments. No significant flooding occurred at the sites during the incubation period (late July to September), and discharge declined in 14 of the 15 sites that were gauged (Table 2). Among the 20 sites that accrued benthic chl-*a*, the density of chl-*a* on the tiles ranged from 3 to 67 mg m^{-2}, with a median value of 11 mg m^{-2}. Although the sites spanned a range in N and P concentrations (Table 2), there was no relationship between chl-*a* accrual on the tiles and any nutrient measure or other environmental factor.

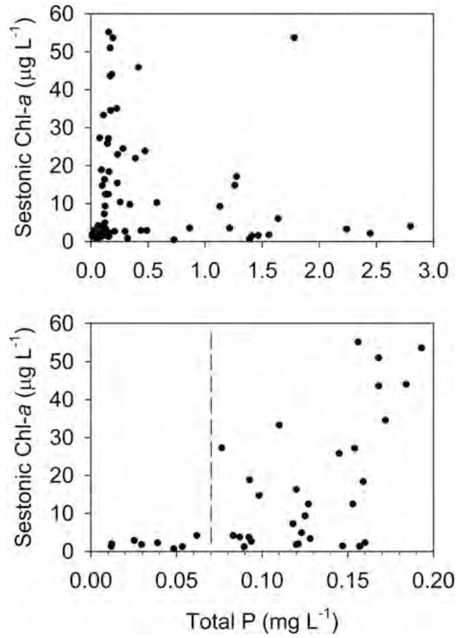


Fig. 4. Relationship between total P and sestonic chlorophyll-*a* (chl-*a*) concentrations during the 2004 low-discharge survey using all sites (upper panel), and only sites with an open canopy (<25%) and total P concentrations of <0.2 mg L^{-1} (lower panel; n = 38). The dashed vertical line indicates an apparent threshold value of 0.07 mg L^{-1} total P.

Discussion

The establishment of defensible nutrient standards for streams and rivers requires a strong linkage between attainment of designated uses and the criterion used to measure the effect of nutrient enrichment. A successful criterion should allow resource managers to accurately predict attainment status based on the measured value of the criterion (Reckhow et al., 2005). Additionally, it is desirable from a management standpoint to have a single standard that can be applied to a large geographic region, such as a state or ecoregion, meaning the criterion must be broadly applicable to a potentially large range of stream types. Our goal was to examine patterns and relationships between algal biomass (as chl-*a*) and nutrient concentrations or other environmental factors and to do so at a state-wide scale. We focused on algal biomass as a potential criterion for nutrient standards because algae often respond directly to nutrient loading and because excess algal biomass can negatively affect O_2 concentrations, habitat quality, biotic community structure, and the aesthetic value of streams—all of which can affect attainment of designated uses.

Sestonic chl-*a* can occur in streams as a result of sloughing of periphyton or in-channel production if conditions are favorable (Swanson and Bachmann, 1976; Lohman and Jones, 1999).

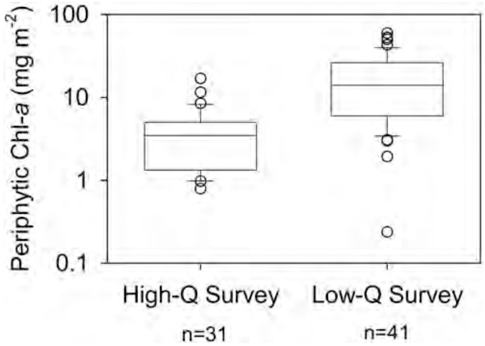


Fig. 5. Box-and-whisker plots of benthic chlorophyll-*a* (chl-*a*) density during the 2004 high- discharge (high-Q) and low-Q surveys (see text for explanation). Horizontal lines indicate the 25th, 50th, and 75th percentiles; whiskers indicate the 10th and 90th percentiles. Circles represent sites outside the 10th or 90th percentiles.

Similar to Lohman and Jones (1999), we found no correlation between benthic chl-*a* density and sestonic chl-*a* concentration, suggesting that in-channel production was the main source of sestonic chl-*a*. This conclusion is supported by the synoptic surveys on the Embarras and Kaskaskia Rivers in which we documented downstream increases in mainstem sestonic chl-*a* that were not attributable to tributary inputs. Across Illinois, sestonic chl-*a* was directly related to watershed area, and this pattern has been observed in other geographic regions (Van Nieuwenhuyse and Jones, 1996; Lohman and Jones, 1999). We did not find the expected relationship between watershed area and total P because several relatively small streams (drainage area <800 km^2) had total P concentrations >1.0 mg L^{-1} during low discharge conditions. Point-source discharges can elevate P concentrations in small streams, but the high rate of algal washout may keep sestonic chl-*a* concentrations lower than the P concentration would predict. In a study of agricultural streams in central Illinois, sestonic chl-*a* was correlated with total P only if a site that received wastewater effluent was excluded from the analysis (Morgan et al., 2006). In the present study, the positive relationship between sestonic chl-*a* and watershed area indicated that rivers and large streams supported conditions that favored the development of sestonic algal communities. As channel size increases, rivers tend to be less influenced by riparian shading and have lower flushing rates than do smaller streams. Van Nieuwenhuyse and Jones (1996) suggested that physical conditions and nutrients co-regulate sestonic chl-*a* concentrations in rivers, and our results support this notion.

We observed a correlation between total P and sestonic chl-*a* but only by limiting the analysis to sites with an open canopy and <0.2 mg L^{-1} total P (Fig. 4). There seemed to be a threshold value of approximately 0.07 mg L^{-1} total P that would be protective of excessive water column chl-*a*, but only eight sites out of 109 had total P concentrations below the apparent threshold, which limits our ability to generalize to the state as a whole. This apparent threshold for Illinois agrees closely with the concentration of 0.075 mg L^{-1} total P suggested by Dodds et al. (1998) as a boundary between mesotrophic and eutrophic conditions in streams of the temperate zone. We suggest that sestonic chl-*a* is not a useful criterion for streams and small rivers (drainage areas <2000 km^2), but it may have some applicability for larger rivers. Among sites with drainage areas >2000 km^2, there was a trend for increasing sestonic chl-*a* with increasing total P concentration (both on a $\log_{10}$ scale), but there was considerable variability, and the relationship was not statistically significant.

Table 2. Periphyton accrual on artificial substrate and the change in environmental variables from the start to the end of the 5-wk incubation. Description of the sites is given in Appendix A.

Site	Chl-*a*‡	Discharge		Total P		Total N		NO_3–N		Turbidity (NTU†)	
		Start	End	Start	End	Start	End	Start	End	Start	End
	mg m^{-2}	m^3 s^{-1}		mg L^{-1}							
1	67	0.04	0.03	1.92	2.50	9.8	14.4	9.4	14.2	1	2
2	43	4.05	3.42	0.22	0.13	0.6	0.7	0.4	0.3	6	7
3	33	0.17	0.05	0.20	0.18	1.2	2.1	0.2	1.6	7	7
4	30	13.41	9.91	2.94	4.25	1.4	4.8	0.7	3.5	8	16
5	27			0.62	0.60	1.2	1.4	0.1	0.4	14	16
6	19			0.74	0.92	4.5	9.1	3.8	8.7	5	19
7	18	0.15	1.56	0.11	0.08	3.5	3.5	3.2	3.2	6	2
8	17	3.45	1.53	0.03	0.04	0.6	0.4	0.1	0.2	2	12
9	14			0.10	0.10	2.8	1.4	2.0	0.9	17	17
10	12	1.08	0.65	0.07	0.05	5.8	1.7	5.5	0.1	7	5
11	11			0.11	0.09	0.7	1.3	0.2	0.6	11	15
12	10	1.73	1.25	0.24	0.19	1.5	0.5	0.1	0.2	10	15
13	9	0.34	0.05	0.15	0.15	0.9	0.6	0.1	0.1	9	14
14	9	0.31	0.17	0.23	0.20	1.1	0.6	0.5	0.3	25	9
15	7	0.42	0.18	0.07	0.07	0.5	0.6	0.1	0.1	4	6
16	7	0.57	0.37	0.09	0.07	0.5	0.8	0.1	0.4	10	12
17	6	0.37	0.27	1.43	1.26	12.7	11.8	11.3	11.6	42	14
18	5			0.09	0.05	0.6	0.2	0.2	0.1	3	3
19	4	0.09	0.05	0.50	0.22	0.9	1.7	0.7	1.3	28	69
20	3			0.17	0.12	0.7	0.4	0.1	0.1	4	11

† Nephelometric turbidity units.

‡ Chlorophyll-*a*.

The relationship between nutrients and the density of benthic chl-*a* is often confounded by factors such as flooding, grazing, and shading, which can make it difficult to separate the effects of human disturbance from natural variation (Dodds et al., 2002). After scouring of periphyton during high discharge, algae begin to accrue in relation to factors such as nutrient availability, light, and grazing, which makes the time since last disturbance a critical co-factor in explaining nutrient–periphyton relationships (Biggs, 2000). This presents practicable problems for state-wide monitoring programs, particularly for sites that are not continuously gauged for discharge. It often is filamentous macro-algae, rather than epilithic biofilms, that reach nuisance levels in streams (Welch et al., 1988). Filamentous mats can have a very patchy distribution within a stream reach, which makes representative sampling difficult. In the current study, we encountered significant mats of filamentous algae only during the low-Q survey. Although nutrient loading is necessary for nuisance algal blooms, factors such as scouring, turbidity, and riparian shading can create situations of high nutrient concentrations and low algal biomass, as occurred during the high-Q survey.

We could not establish a nutrient–benthic chl-*a* relationship, even when a common and suitable substrate for algal growth was used, suggesting among-site variation in other (unmeasured) environmental factors influenced chl-*a* accrual more strongly than did nutrient availability. During both the high-Q and low-Q survey, the 25th percentiles for total N and total P were greater than the breakpoints reported by Dodds et al. (2006) for the nutrient–benthic chl-*a* relationships identified for temperate zone streams. This suggests that, for the state as a whole, nutrients occurred in excess of algal demands and that a factor(s) other than nutrients limited algal biomass. Additionally, this implies that for many streams and rivers in Illinois nutrients would have to be lowered below some threshold concentration before a response in algal biomass would occur (Dodds et al., 2002).

An ecoregion approach to using benthic chl-*a* as an indicator of nutrient enrichment has shown promise in streams of the Mid-Atlantic USA (Pan et al., 1999) and to some extent for streams in North America and New Zealand (Dodds et al., 2002). Whether an ecoregion approach would improve the nutrient–benthic chl-*a* relationship in Illinois streams is unknown, and the paucity of sites with measurable benthic chl-*a* in the current study precludes an ecoregion analysis. However, the Central Corn Belt Plains Ecoregion represents approximately 50% of the state of Illinois (Woods et al., 2006), and estimates have been made for baseline nutrient concentrations for various corn belt ecoregions. The total N and total P concentrations reported here and elsewhere (e.g., Royer et al., 2004; Gentry et al., 2007) for Illinois streams are generally 1 to 2 orders of magnitude greater than the estimated background total N and total P concentrations for corn belt ecoregions (Smith et al., 2003; Dodds and Oakes, 2004). Although historic data on nutrient concentrations for Illinois streams are sparse, it is clear that present-day nutrient concentrations are likely greatly elevated from background conditions, and this may obscure differences between ecoregions in algal–nutrient relationships.

Fewer than 50% of the sites we examined had coarse substrate from which to sample benthic chl-*a*. Because we did not attempt to sample benthic chl-*a* from sand and soft sediments, we cannot assess the role of these substrates in supporting benthic chl-*a*. However, our observations indicate that streams and rivers in Illinois that contain soft sediments are consistently turbid even during periods of low discharge (see Table 1) due to the recurring suspension of fine sediments. A previous study of 14 agricultural streams in central Illinois found that water column turbidity was a strong predictor of benthic chl-*a* (Figueroa-Nieves et al., 2006). Morgan et al. (2006) examined benthic chl-*a* in two open canopy agricultural streams in Illinois and found a significant inverse relationship between chl-*a* and water depth, suggesting that water column turbidity was limiting light penetration to the streambed. Based on the accumulated evidence, we suggest that benthic chl-*a* in Illinois streams and rivers is strongly influenced by substrate conditions and water column turbidity (which are themselves related) and that this may preclude isolating the direct effect of nutrients on benthic chl-*a* at a state-wide scale.

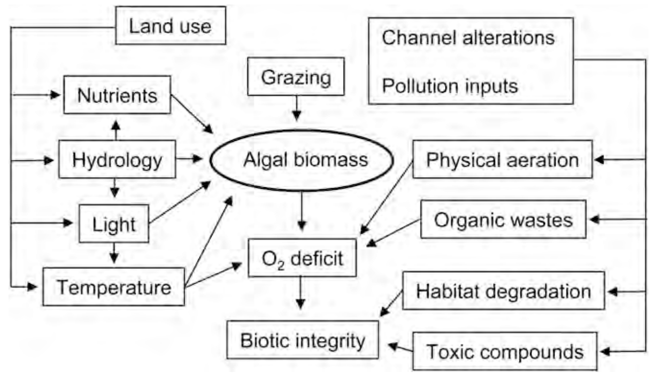


Fig. 6. Factors that influence the biotic integrity of Illinois streams. Land use, channel alterations, and pollution inputs can directly and indirectly decouple the expected cause-and-effect relationship between nutrient loading, algal biomass, O_2 deficit, and biotic integrity. Not all factors and interactions are shown.

Conclusions

Several challenges exist in the process of developing nutrient standards for streams and rivers. For example, unlike drinking water standards developed via toxicity testing to prevent a physiological outcome in individuals, nutrient standards are designed to prevent an ecological outcome (i.e., biotic impairment) across large geographic regions in the face of multiple stressors that influence biotic integrity. Furthermore, the ultimate goal of improving and protecting the biotic integrity of streams may not be accomplished solely with the implementation of nutrient standards. Biotic integrity is an outcome of many interacting factors (Fig. 6), and it may often be the case that nutrient concentration plays a relatively minor role in causing biological impairment of a particular site. Physical habitat quality played a strong role in controlling stream macroinvertebrate communities across Illinois, and habitat quality and nutrient concentrations were related, confounding efforts to isolate the influence of each (Heatherly et al., 2007).

Although nutrients are not the sole determinant of stream health, factors such as habitat quality, hydrology, light, temperature, and grazing are less amenable to management practices, leaving nutrients as the focus for attempts to control algal biomass and protect biotic integrity (Dodds and Welch, 2000). Currently in Illinois, it seems that for many streams and rivers nutrients may not be the limiting factor for algal biomass, due to the generally high nutrient concentrations and the effects of other factors, such as substrate conditions and turbidity. Nevertheless, management of nutrient concentrations in Illinois is important because ecological processes other than algal growth, including heterotrophic respiration, are influenced by nutrient loading (Dodds, 2006). Illinois is also a major contributor of N and P to the Mississippi River and Gulf of Mexico (Goolsby et al., 1999; David and Gentry, 2000), and efforts within the state to reduce nutrient loading may have far-reaching benefits.

Acknowledgments

We thank Josh Burke, Heather Clodfelter, Emily Keane, Erin Lewis, Tessa Marusarz, Marshall McDaniel, Allyson Morgan, Paul Terrio, Tom Tucker, and Kate Von Holle for assistance with collecting and processing samples. Personnel at Illinois EPA assisted with site selection. Funding was provided by the State of Illinois through the Illinois Council on Food and Agricultural Research (C-FAR), Water Quality SRI.

Appendix A. The 138 sites used in the study, watershed area for each site, and the corresponding Ambient Water Quality Monitoring Network code used by the Illinois Environmental Protection Agency (IEPA). Sites are organized by major drainage basins, and all location references are to towns or cities in Illinois. All of the listed sites were used in the 2004 high-Q survey, those sites used in other aspects of the study also are indicated.

IEPA code	Site name and location	Watershed area	Sites used in 2004 low-Q survey	Sites used in 2005 survey	Site number for artificial substrate study
		km^2			
	Ohio River Basin				
AD-02	Cache River at Forman	632	x		
AK-02	Lusk CR near Eddyville	111	x	x	8
AT-06	Saline River near Gibsonia,	2751	x		
ATF-04	North Fork Saline River near Texas City	448	x		
ATGC-01	Bankston Creek near Harrisburg	202			
ATH-05	South Fork Saline River near Carrier Mills	381	x	x	
	Little Wabash & Wabash River Basin				
BC-02	Bonpas Creek at Browns	591	x	x	
BE-01	Embarras River near Billet	6224			
BE-07	Embarras River at Ste. Marie	3926	x	x	13
BE-09	Embarras River near Diona	2380			
BE-14	Embarras River at Camargo	466	x	x	14
BEF-05	North Fork Embarras River near Oblong	824	x	x	20
BM-02	Sugar Creek near Elbridge	158	x		
BP-01	Vermilion River near Danville	3341	x		
BPG-09	North Fork Vermilion River near Bismarck	679	x		
BPJ-03	Salt Fork Vermilion River near Oakwood	1267	x		
BPJC-06	Saline Branch near Mayview	212	x	x	1
BPK-07	Middle Fork Vermilion River near Oakwood	1119	x		
C-21	Little Wabash River near Effingham	622	x		
C-22	Little Wabash River near Clay City	2929	x	x	
C-23	Little Wabash River at Carmi	7998	x		
CA-03	Skillet Fork near Carmi	2740	x		
CA-05	Skillet Fork at Wayne City	1202	x		
CD-01	Elm River near Toms Prairie	686	x	x	12
	Illinois River Basin				
D-23	Illinois River at Marseilles	21,391	x		
D-32	Illinois River at Valley City	68,801	x		
DA-04	Macoupin Creek near Macoupin	787	x	x	
DA-06	Macoupin Creek near Kane	2248	x	x	
DB-01	Apple Creek near Eldred	1046	x		
DD-04	Mauvaise Terre Creek near Merritt	378	x	x	
DE-01	McKee Creek at Chambersburg	883	x		
DG-01	La Moine River at Ripley	3349			
DG-04	La Moine River at Colmar	1696	x		
DJ-08	Spoon River near Seville	4237	x	x	
DJ-09	Spoon River near London Mills	2751			
DJB-18	Big Creek near Bryant	106	x		
DJL-01	Indian Creek near Wyoming	163	x	x	2
DK-12	Mackinaw River near Green Valley	2828			
DK-13	Mackinaw River near Congerville	2010	x	x	16
DQ-03	Big Bureau Creek at Princeton	508	x	x	4
DS-06	Vermilion River at McDowell	1427	x		
DS-07	Vermilion River near Leonore	3240			
DT-06	Fox River at Algonquin	3634			
DT-38	Fox River at Montgomery	4486	x		
DT-46	Fox River at Dayton	6843			
DTD-02	Blackberry Creek near Yorkville	181	x		
DTG-02	Poplar Creek at Elgin	91	x		
DTK-04	Nippersink Creek near Spring Grove	497	x		
DV-04	Mazon River near Coal City	1178	x		
DZZP-03	Farm Creek at East Peoria	158	x		
	Sangamon River Basin				

(continued)

Appendix A. Cont'd.

IEPA code	Site name and location	Watershed area	Sites used in 2004 low-Q survey	Sites used in 2005 survey	Site number for artificial substrate study
		km^2			
E-09	Sangamon River at Decatur	2429			
E-25	Sangamon River near Oakford	13,191	x	x	5
E-26	Sangamon River at Riverton	6781	x		
E-28	Sangamon River near Monticello	1484	x		
E-29	Sangamon River at Fisher	622	x		
EI-02	Salt Creek near Greenview	4672	x	x	3
EI-06	Salt Creek near Rowell	868	x	x	
EID-04	Sugar Creek near Hartsburg	862	x	x	6
EIE-04	Kickapoo Creek at Waynesville	588	x		
EIG-01	Lake Fork near Cornland	554	x	x	15
EL-01	Spring Creek at Springfield	282	x		
EO-01	South Fork Sangamon River near Rochester	2253			
EO-02	South Fork Sangamon River at Kincaid	1456	x		
EOH-01	Flat Branch near Taylorville	715	x	x	19
EZU-01a	Big Ditch near Dewey	142	x	x	10
	Kankakee River Basin				
F-01	Kankakee River near Wilmington	13,339	x		
F-02	Kankakee River at Momence	5941			
FL-02	Iroquois River near Chebanse	5416	x		
FL-04	Iroquois River at Iroquois	1777			
FLI-02	Sugar Creek at Milford	1155	x		
	Des Plaines River & Lake Michigan Basins				
G-07	Des Plaines River near Gurnee	601	x		
G-08	Des Plaines River at Russell	319			
G-22	Des Plaines River near Des Plaines	932			
G-39	Des Plaines River near Riverside	1632	x		
GB-10	Du Page River near Naperville	570			
GB-11	Du Page River at Shorewood	839	x		
GBK-05	West Branch Du Page River near Warrenville	233	x		
GBK-09	West Branch Du Page River near West Chicago	75	x		
GBL-10	East Branch Du Page River at Lisle	148	x		
GG-02	Hickory Creek at Joliet	277	x		
GI-01	Sanitary & Ship Canal at Romeoville				
GL-09	Salt Creek at Western Springs	295	x		
GLA-02	Addison Creek at Bellwood	47	x		
H-01	Calumet Sag Channel at Sag Bridge	1008			
HBD-04	Thorn Creek at Thornton	269	x		
HCC-07	North Branch Chicago River at Niles	259	x		
HCCC-02	North Branch Chicago River at Deerfield	52			
	Mississippi River Tributaries				
II-03	Marys River at Welge	293	x		
IX-04	Cache River at Sandusky	606	x		
IXJ-02	Big Creek near Balcom	21	x	x	
JN-02	Cahokia Canal near Collinsville	155	x		
JNA-01	Canteen Creek near Collinsville	67	x		
JQ-05	Cahokia Creek at Edwardsville	549	x		
KCA-01	Bay Creek at Nebo	417	x		
KI-02	Bear Creek near Marcelline	904	x		
LD-02	Henderson Creek near Oquawkao	1119	x		
LF-01	Edwards River near New Boston	1153	x	x	11
MJ-01	Plum River at Savanna	707	x		
MN-03	Apple River near Elizabeth	536	x		
MQ-01	Galena River at Galena	508	x		
	Big Muddy River Basin				
N-11	Big Muddy River at Plumfield	2056			
N-12	Big Muddy River at Murphysboro	5618	x		
NC-07	Beaucoup Creek near Vergennes	1238	x		
ND-01	Crab Orchard Creek near Carbondale	704	x		
ND-04	Crab Orchard Creek near Marion	83	x	x	
NJ-07	Casey Fork near Mount Vernon	228	x	x	
NK-01	Rayse Creek near Waltonville	228	x	x	

(continued)

Appendix A. Cont'd.

IEPA code	Site name and location	Watershed area	Sites used in 2004 low-Q survey	Sites used in 2005 survey	Site number for artificial substrate study
		km^2			
	Kaskaskia River Basin				
O-02	Kaskaskia River at Cooks Mills	1225	x		
O-07	Kaskaskia River near Carlyle	7081			
O-08	Kaskaskia River at Vandalia	5025	x	x	
O-10	Kaskaskia River near Cowden	3445	x	x	9
O-20	Kaskaskia River near Venedy Station	11,378	x		
O-31	Kaskaskia River near Tuscola	293			
OC-04	Richland Creek near Hecker	334	x	x	17
OD-06	Silver Creek near Troy	399	x		
OD-07	Silver Creek near Freeburg	1202			
OI-08	Shoal Creek near Breese	1904			
OI-09	Shoal Creek near Walshville	728	x	x	
OK-01	East Fork Kaskaskia River near Sandoval	293	x		
OKA-01	North Fork Kaskaskia River near Patoka	101			
OL-02	Hurricane Creek near Mulberry Grove	394	x		
ON-01	Hickory Creek near Bluff City	202	x		
OQ-01	Beck Creek at Herrick	251			
OT-02	West Okaw River near Lovington	290	x		
OZC-01	Plum Creek near Baldwin	158	x	x	
	Rock River Basin				
P-04	Rock River near Joslin	24,732	x		
P-06	Rock River at Como	22,670			
P-14	Rock River at Byron	20,694	x		
P-15	Rock River at Rockton	16,480			
PB-02	Green River near Deer Grove	834	x	x	18
PB-04	Green River near Geneseo	2598			
PH-16	Elkhorn Creek near Penrose	378	x		
PHI-01	Five Mile Creek near Brookville	44	x	x	7
PL-03	Kyte River at Daysville	464	x		
PQ-10	Kishwaukee River at Garden Prairie	575	x		
PQ-12	Kishwaukee River near Perryville	2846	x		
PQC-06	South Branch Kishwaukee River near Fairdale	1002	x		
PW-08	Pecatonica River at Freeport	3434	x		

References

American Public Health Administration. 1998. Standard methods for the examination of water and wastewater. 18th ed. APHA, Washington, DC.

Biggs, B.J.F. 2000. Eutrophication of streams and rivers: Dissolved nutrient-chlorophyll relationships for benthic algae. J. N. Am. Benthol. Soc. 19:17–31.

David, M.B., and L.E. Gentry. 2000. Anthropogenic inputs of nitrogen and phosphorus and riverine export for Illinois, USA. J. Environ. Qual. 29:494–508.

Dodds, W.K. 2006. Eutrophication and trophic state in rivers and streams. Limnol. Oceangr. 51:671–680.

Dodds, W.K., J.R. Jones, and E.B. Welch. 1998. Suggested classification of stream trophic state: Distributions of temperate stream types by chlorophyll, total nitrogen, and phosphorus. Water Res. 32:1455–1462.

Dodds, W.K., and R.M. Oakes. 2004. A technique for establishing reference nutrient concentrations across watersheds affected by humans. Limnol. Oceanogr. Methods 2:333–341.

Dodds, W.K., V.H. Smith, and K. Lohman. 2002. Nitrogen and phosphorus relationships to benthic algal biomass in temperate streams. Can. J. Fish. Aquat. Sci. 59:865–874.

Dodds, W.K., V.H. Smith, and K. Lohman. 2006. Erratum: Nitrogen and phosphorus relationships to benthic algal biomass in temperate streams. Can. J. Fish. Aquat. Sci. 63:1190–1191.

Dodds, W.K., and E.B. Welch. 2000. Establishing nutrient criteria in streams. J. N. Am. Benthol. Soc. 19:186–196.

Figueroa-Nieves, D., T.V. Royer, and M.B. David. 2006. Controls on chlorophyll-*a* in nutrient-rich agricultural streams in Illinois, USA. Hydrobiologia 568:287–298.

Gentry, L.E., M.B. David, T.V. Royer, C.A. Mitchell, and K.M. Starks. 2007. Phosphorus transport pathways to streams in tile-drained agricultural watersheds. J. Environ. Qual. 36:408–415.

Goolsby, D.A., W.A. Battaglin, G.B. Lawrence, R.S. Artz, B.T. Aulenbach, R.P. Hooper, D.R. Keeney, and G.J. Sensland. 1999. Flux and sources of nutrients in the Mississippi-Atchafalaya River Basin. Topic 3. Report of the integrated assessment on hypoxia in the Gulf of Mexico. NOAA Costal Ocean Program Decision Analysis Series No. 17. NOAA Costal Ocean Program, Silver Spring, MD.

Heatherly, T., M.R. Whiles, T.V. Royer, and M.B. David. 2007. Relationships between habitat quality, water quality, and macroinvertebrate assemblages in Illinois streams. J. Environ. Qual. 36:1653–1660.

Illinois Department of Agriculture. 2001. Land cover of Illinois statistical summary 1999–2000, On-line report available at http://www.agr.state.il.us/gis/stats/landcover/mainpages/stats_statewide.htm (verified 7 Dec. 2007).

Lohman, K., and J.R. Jones. 1999. Nutrient-sestonic chlorophyll relationships in northern Ozark streams. Can. J. Fish. Aquat. Sci. 56:124–130.

Miltner, R.J., and E.T. Rankin. 1998. Primary nutrients and the biotic integrity of rivers and streams. Freshwater Biol. 40:145–158.

Morgan, A.M., T.V. Royer, M.B. David, and L.E. Gentry. 2006. Relationships among nutrients, chlorophyll-*a*, and dissolved oxygen in agricultural streams in Illinois. J. Environ. Qual. 35:1110–1117.

Pan, Y., R.J. Stevenson, B.H. Hill, P.R. Kaufmann, and A.T. Herlihy. 1999. Spatial patterns and ecological determinants of benthic algal assemblages in Mid-Atlantic streams, USA. J. Phycol. 35:460–468.

Reckhow, K.H., G.B. Arhonditsis, M.A. Kenney, L. Hauser, J. Tribo, C. Wu, K.J. Elcock, L.J. Steinberg, C.A. Stow, and S.J. McBride. 2005. A predictive approach to nutrient criteria. Environ. Sci. Technol. 39:2913–2918.

Royer, T.V., J.L. Tank, and M.B. David. 2004. Transport and fate of nitrate in headwater agricultural streams in Illinois. J. Environ. Qual. 33:1296–1304.

Smith, R.A., R.B. Alexander, and G.E. Schwarz. 2003. Natural background concentrations of nutrients in streams and rivers of the conterminous United States. Environ. Sci. Technol. 37:3039–3047.

Steinman, A.D., and G.A. Lamberti. 1996. Biomass and pigments of benthic algae. p. 295–313. *In* F.R. Hauer and G.A. Lamberti (ed.) Methods in stream ecology. Academic Press, San Diego, CA.

Swanson, C.D., and R.W. Bachmann. 1976. A model of algal exports in some Iowa streams. Ecology 57:1076–1080.

USEPA. 2000a. Atlas of America's Polluted Waters, EPA 840-B00-002. Office of Water, USEPA, Washington, DC.

USEPA. 2000b. Nutrient criteria technical guidance manual, EPA 822-B-00-002. Office of Water, USEPA, Washington, DC.

Van Nieuwenhuyse, E.E., and J.R. Jones. 1996. Phosphorus-chlorophyll relationship in temperate streams and its variation with stream catchment area. Can. J. Fish. Aquat. Sci. 53:99–105.

Welch, E.B., J.M. Jacoby, R.R. Horner, and M.R. Seeley. 1988. Nuisance biomass levels of periphytic algae in streams. Hydrobiologia 157:162–168.

Woods, A.J., J.M. Omernik, C.L. Pederson, and B.C. Moran. 2006. Level III and IV ecoregions of Illinois. EPA/600/R-06/104. USEPA National Health and Environmental Effects Research Lab., Corvallis, OR.

Zar, J.H. 1999. Biostatistical Analysis, 4th ed. Prentice Hall, Upper Saddle River, NJ.

Attachment 39:

Establishing nutrient criteria in streams
Dodds and Welch 2000

J. N. Am. Benthol. Soc., 2000, 19(1):186–196

BRIDGES

BRIDGES is a recurring feature of J-NABS intended to provide a forum for the interchange of ideas and information between basic and applied researchers in benthic science. Articles in this series will focus on topical research areas and linkages between basic and applied aspects of research, monitoring policy, and education. Readers with ideas for topics should contact Associate Editors, Nick Aumen and Marty Gurtz.

Criteria for setting nutrient levels in lotic ecosystems are relevant to US states and other countries in the process of setting water-quality regulations. There are few articles in the peer-reviewed literature on this topic, and policy makers have had little information from which to base their decisions for streams. This lack of information is particularly troublesome because of the large number of streams and rivers that have impaired water quality, and the ever-increasing pace of urban and agricultural development. In addition to the effects of high nutrient concentrations on stream ecosystem structure and function, high nutrient concentrations, particularly nitrate, may have adverse effects on human health.

Nick Aumen, naumen@bellsouth.net
Marty Gurtz, megurtz@usgs.gov
Co-editors

Establishing nutrient criteria in streams

WALTER K. DODDS[1]

Division of Biology, Kansas State University, Manhattan, Kansas USA 66506

EUGENE B. WELCH

Department of Civil and Environmental Engineering, University of Washington, Seattle, Washington USA 98195

"Every child deserves to grow up with water that is pure to drink, lakes that are safe for swimming, rivers that are teeming with fish. We have to act now to combat these pollution challenges with new protections to give our children the gift of clean, safe water in the 21st century."

—President Clinton, 23 February 1999, Baltimore

The US Environmental Protection Agency (USEPA) and the US Department of Agriculture (USDA) recently have been directed to set criteria for nutrients in rivers, lakes, and estuaries. State reports compiled by the USEPA (National Water Quality Inventory: 1996 Report to Congress) claim that 40% of streams or rivers surveyed were impaired because of the nutrients N and P, but no well-defined standards have been proposed to determine if nutrients impair flowing waters (USEPA 1998). A rational framework for determining criteria is necessary because the USEPA has been charged with establishing maximum acceptable levels of nutrients in streams and rivers by 2001 as part of the Clean Water Action Plan. State and tribal governments will use these criteria to set total maximum daily loads (TMDLs) for nutrients and adopt their own standards by 2003. The USEPA, the USDA, and other national governmental agencies (e.g., US Geological Survey [USGS], US Army Corps of Engineers), state and tribal officials, and private parties will set these criteria.

Data analyses are needed to explain the relationships between stream algae and nutrients, which previously have received attention from researchers. Given the potential economic impacts of nutrient control, the process by which nutrient levels are set likely will engender controversy. Basing the criteria on the best scientific data available will minimize conflict and maximize the potential benefits related to controlling nutrients in streams.

Although the question of how to set nutrient criteria is framed above in terms of US politics and policies, other countries also are interested in nutrient criteria for streams, particularly developed countries where industrialization, urbanization, and modern agriculture have resulted in extensive nutrient discharge into water courses. The following discussion will be based

[1] E-mail address: wkdodds@ksu.edu

primarily on examples from the US, but the general principles could apply to any watershed.

We 1st will address why nutrient criteria are needed. Next, we will discuss the scientific basis behind possible criteria. Last, we discuss ways to account for variability in streams (i.e., factors that may decouple nutrient concentrations from biomass). The main thrust of this paper is to outline what scientific methods currently are available to managers for setting nutrient criteria, given specific reasons for setting the criteria.

Why do we need nutrient criteria?

Reasons for nutrient criteria include: 1) adverse effects on humans and domestic animals, 2) aesthetic impairment, 3) interference with human use, 4) negative impacts on aquatic life, and 5) excessive nutrient input into downstream systems. Each of these will be discussed in sequence; they share several characteristics, but they also have facets that may result in criteria being set at different levels.

High levels of some nutrients may have adverse effects on human health. Control of NO_3^- levels particularly is important to avoid metheamoglobemia. Furthermore, NO_3^- consumption has been correlated with stomach cancer (Hartman 1983). Although correlation does not guarantee causation, NO_3^- could be regulated as a carcinogen in the future.

Eutrophication from N and P causes proliferation of algal masses, some of which may be toxic. In one of the worst cases, eutrophication caused Cyanobacteria to bloom in the stagnant Murray-Darling River system (Australia) during a drought, leading to livestock deaths and concerns about impacts on humans (Bowling and Baker 1996). Such toxic blooms are most likely to occur in very enriched, slow-moving, and nonturbid rivers.

Eutrophication causes taste and odor problems in lakes (Arruda and Fromm 1989, Wnorowski 1992), but these negative effects have not been linked directly to trophic state of streams and rivers. Such problems often can be traced to production of odorous metabolites by Cyanobacteria (e.g., geosmin) and other algae and their subsequent leakage into surrounding waters. Algae that cause taste and odor problems can reach high biomass in eutrophic streams and rivers, both in the phytoplankton of slow-moving rivers and the periphyton of shallow streams.

Aesthetic impairment is more difficult to quantify, but usually is associated with filamentous algal forms. Nuisance levels may be reached somewhere between 100 and 200 mg/m^2 chlorophyll (Horner et al. 1983, Nordin 1985, Welch et al. 1988, Quinn 1991). Enriched waters often have benthic chlorophyll concentrations >150 mg/m^2, and many stream users find high levels of algal growth objectionable (Welch et al. 1989, V. Watson, University of Montana, personal communication). A link also may exist between property values and trophic state in lotic waters, as has been documented for lakes (Michael et al. 1996). However, to our knowledge, such analyses have not been conducted for rivers and streams.

Excessive growth of algae and macrophytes can interfere with human uses of flowing waters. Such interference is exemplified by problems caused by the filamentous green alga *Cladophora*. Exorbitant amounts of this alga can slow water flow in canals (decreasing delivery rates and increasing water losses), interfere with swimming, and snag fishing lures (Dodds and Gudder 1992). Furthermore, excessive algal growth may clog screens on water intakes for water treatment plants and industries.

High concentrations of NH_3 in the water column clearly are toxic to aquatic animals (Russo 1985). For example, levels of >1 mg/L NH_3-N in Ohio streams have negative impacts on the fish communities (Miltner and Rankin 1998). Negative impacts on aquatic life related to stimulation of algal biomass by increased nutrients are subtler. As a system becomes more productive, different species of algae may become more competitive and species composition can shift (Kelly and Whitton 1995, Pan et. al. 1996, Kelly 1998). However, unless such species shifts cause specific water-quality symptoms (e.g., toxic algae) or aesthetic problems (e.g., very long streamers of filamentous algae), the public is unlikely to be concerned.

Nutrient enrichment may adversely affect stream animal communities. Enriched streams have increased invertebrate biomass and altered invertebrate communities (Bourassa and Cattaneo 1998). Community structure has been correlated directly with P concentration (Miltner and Rankin 1998). Excessive levels of algae were damaging to invertebrates (Nordin 1985).

Changes in community structure may be viewed as problematic, particularly if game fish are affected. In extreme cases, levels of primary production can be stimulated by nutrients; organic C will build up in the system and cause a subsequent low dissolved O_2 (DO) and high pH event. Fish and invertebrates will grow poorly and even die if the O_2 depletion and pH increases are severe (Welch 1992).

Because streams drain into lakes and oceans, eutrophication caused by influx of nutrients from flowing waters is a concern for downstream lake and coastal areas. Examples of marine eutrophication are the zone of hypoxia that develops in the Gulf of Mexico (Rabalais et al. 1998) and the production of toxic estuarine dinoflagellate blooms (Burkholder and Glasgow 1997). Eutrophication problems in lakes are well documented, and the control of external and internal nutrient loading necessary to minimize eutrophication can be calculated (Cooke et al. 1993). Requirements for control of nutrient loading to lakes and coastal marine systems may lead to more stringent nutrient criteria in rivers than those required for controlling instream eutrophication, especially in localities where stream algae are limited by factors other than nutrients.

What are the scientific bases for criteria?

In this section we discuss what nutrients and what forms of those nutrients should be used to set criteria. We also describe some models and approaches that can be used to set criteria. Last, we offer some discussion on how criteria may vary depending upon the reason for the criteria.

What nutrients and forms should be used to set criteria?

The traditional view is that P limits primary production in fresh waters (e.g., Correll 1998), and N limits it in the ocean. However, nutrient bioassays and correlation analyses do not substantiate this point of view. Data were compiled from 158 bioassays reported in the literature in which the response of stream periphyton to nutrient fertilization was measured. Of the studies, 13% showed stimulation by N alone, 18% by P alone, 44% by simultaneous N and P additions, and 25% by neither nutrient (W. K. Dodds, unpublished data). The absolute proportions associated with each type of limitation should not be viewed as a general guide to nutrient limitation in streams. However, bioassay results do suggest that both N and P can limit primary producers in streams.

Correlation analyses also do not support the idea of P as the sole limiting nutrient in rivers and streams. Mean and maximum benthic chlorophyll correlated better with total N (TN) than total P (TP) in the water column in several hundred streams. Nitrogen and P occur in several forms in rivers and streams, including dissolved organic and inorganic forms and in particulate material. All of these forms together are referred to as TN and TP. Total N does not include dissolved N_2 gas. A regression model using both nutrients explained the highest proportion of the variance in biomass (Dodds et al. 1997). Thus, both N and P can control primary production in at least some streams and rivers.

Control of P alone may cause P to limit and lower algal biomass, as has occurred in many lakes (Sas 1989, Cooke et al. 1993). However, if pulses of P occur, they can be taken up in excess of requirements and stored inside algal cells in a process called *luxury consumption*. This stored P can allow algae to grow even if P concentrations are low in the water column. If controlling such P pulses is impossible (e.g., pulses associated with high runoff events in spring), control of N could become necessary. For example, Lohman and Priscu (1992) demonstrated that intracellular P concentrations in *Cladophora* increased, while P availability in the surrounding water decreased. Thus, *Cladophora* was N limited, even though analyses of available N and P in the river water column suggested P limitation. Given the bioassay and correlation data, and that periphyton can consume P in excess of immediate needs, setting nutrient criteria for both N and P makes sense.

Unless clear limitation by other nutrients has been demonstrated in a particular system, N and P should be assumed to be the dominant nutrients controlling the trophic states of streams and rivers. Fortunately, nuisance and some toxic heterocystous Cyanobacteria that can use N_2 gas as a N source generally are not part of eutrophic stream periphyton, but may occur in the plankton of slowly flowing rivers (Bowling and Baker 1996). The decreased dominance of heterocystous Cyanobacteria in streams leads to some situations where N control alone may lead to decreases in algal biomass. Controlling N and not P inputs in lakes can encourage blooms of nuisance Cyanobacteria (Stockner and Shortread 1988). However, the strategy of controlling N alone should be viewed with caution, especially in plankton-dominated rivers. More data on cyanobacterial problems in eutrophic streams are necessary before we can be certain that N control will not lead to cyanobacterial dominance.

Control based on measured levels of dissolved inorganic N and P may not be effective because these pools are replenished rapidly by remineralization in surface waters (Dodds 1993). Correlation of algal biomass with dissolved inorganic nutrients was poor in some studies (Dodds et al. 1997), but not all (Biggs and Close 1989). Also, lake managers are aware of problems with using dissolved inorganic nutrient concentrations to set nutrient criteria. Last, most of the data linking land-use practices to N and P loading have been reported in TN and TP (Loehr 1974), so basing criteria on total nutrients for calculating TMDLs is more practical than using dissolved inorganic nutrients.

Two caveats are necessary to the generalization that TN and TP should be emphasized. First, if nutrients are released directly into streams in dissolved inorganic form, their influence may be more intense and localized near the point source of release. Dissolved inorganic nutrients will be taken up rapidly, which can lead to a very high, localized concentration of biomass (Hynes 1969). Second, some models using seasonal means of dissolved inorganic nutrients to predict algal biomass have been very successful (Biggs 1995, 2000), and some sites have considerably more data on dissolved than total nutrients on which to base decisions.

At what concentrations should criteria be set?

One difficulty in setting criteria involves assessment of the trophic state of a stream or river. Stated another way, how can we declare that a river or stream is in an unacceptable trophic state if there is no basis for scaling the trophic state relative to other rivers? A generally accepted system for classifying the trophic states of streams and rivers is lacking (Dodds et al. 1998). In general, trophic state is classified by nutrients and algal biomass. System metabolism may be more relevant to ecosystem function, but difficulties with methods and limited data have precluded use of production and respiration to classify trophic states of lakes and streams. One classification system proposed for streams relies upon the cumulative frequency distributions of chlorophyll and nutrients. The lower 1/3 of the distribution sets the range for oligotrophic streams, and the upper 1/3 for eutrophic streams. This approach is consistent with the convention of classifying trophic state into 3 categories, while basing classification on the actual distribution of biomass and nutrient levels found in streams (Dodds et al. 1998). The published classification was based on only 286 temperate streams. More data are necessary to determine how well this classification scheme applies to rivers from different ecoregions, how distributions of nutrients correlate to algal biomass, and how well such classification represents pristine conditions. Analyses of existing databases may provide a valuable tool in extending this approach to trophic classification.

A few models directly link TN and TP to benthic algal biomass in streams (e.g., Lohman et al. 1992, Dodds et al. 1997, Bourassa and Cattaneo 1998, Chételat et al. 1999). Such models can be applied to estimate algal biomass as a function of water column nutrients. A similar correlation approach has been very successful in managing eutrophication in lakes and reservoirs. Extension of these models to link in-stream nutrient concentrations to known sources of nutrient loading also has been described (Dodds et al. 1997).

Models describing the correlation between nutrients and chlorophyll in lakes differ from those for streams because benthic chlorophyll may be much more variable in streams as a result of the effects of floods, turbidity, and grazing. This difference is exemplified by the ratios of maximum to mean chlorophyll. This ratio describes the variance in level of chlorophyll, with high numbers denoting a high variance. The maximum/mean chlorophyll ratio is 4.5 for stream benthos compared to 1.7–2.6 for lake phytoplankton (Dodds et al. 1998). Furthermore, total water column nutrients usually are correlated strongly with chlorophyll because phytoplankton contain chlorophyll, N, and P. This linkage leads to high correlation coefficients between total nutrients and algal biomass in lakes. This relationship is not as highly coupled in streams when *benthic* chlorophyll and *water col-*

umn TN and TP are considered. Thus, the correlation models developed for stream benthic algae contain a much greater degree of uncertainty than those for lakes.

Biggs (2000) proposes a correlation method that considers hydrodynamic disturbance and inorganic nutrients in New Zealand streams, that is pertinent for predicting benthic algal biomass. Such an approach may prove useful within an ecoregion, and could be used to provide a sliding scale of nutrient criteria, with higher nutrient content allowed in more hydrodynamically unstable rivers (i.e., criteria may be more lenient because of regular scouring of algal biomass in rivers that flood frequently).

An alternative approach to correlation models also has been developed. This method consists of sampling nutrients in reference stream reaches where chlorophyll levels are deemed acceptable. Gary Ingman (Montana Department of Environmental Quality) and Vicki Watson (University of Montana) proposed this technique for use in the Clark Fork River in Montana (Dodds et al. 1997). General regional criteria have yet to be established using this method. In systems where the entire stream receives nutrient loading, or regions where all watersheds are enriched, locating suitable reference reaches may be impossible. Data from other similar streams should be used to identify the obtainable baseline nutrient concentrations in those cases.

A regression model linking TP to river phytoplankton is available (Van Niewenhuyse and Jones 1996). This model can be used to set TP criteria. The TP levels can be used to calculate corresponding TN concentrations with the Redfield ratio (Harris 1986). This model captures additional variance when watershed area is considered.

Setting nutrient criteria is difficult based on subjective impressions of what constitutes excessive levels of benthic algae. However, >200 mg/L of benthic chlorophyll generally produces a very green stream bottom (Welch et al. 1988). To further complicate matters, filamentous green algae have a less desirable appearance than brown-colored diatoms, even when the biomass of the 2 is similar. Moreover, a large amount of the variance in benthic chlorophyll levels in streams is not related to nutrient levels. We simply do not have the data in the US to predict when benthic algal community structure will shift to more nuisance forms with changes in nutrients. Preliminary data from Canada indicate that rhodophytes make up a large portion of the algal community when biomass is low, and *Cladophora* and *Melosira* prefer high nutrient water (Chételat et al. 1999). More research clearly is needed in this area, both original research and analysis of existing data. Thus, criteria based on current data will need to be set based on what amount of chlorophyll is acceptable, not on how nutrient amounts and ratios will influence algal communities.

Dissolved O_2 deficit and high pH are perhaps the most severe algal-related problems affecting the aquatic life-support characteristics of a river or stream. Deficits of DO can occur when respiration of organic C produced by photosynthetic processes in the stream exceeds the ability of reaeration to supply DO. Depletion of DO in streams was described years ago (Odum 1956). However, the severity of the deficit is difficult to predict in specific situations. Deficits of DO are most likely to occur in rivers with laminar flow (slow, non-turbulent flow), when a large algal biomass is present, with high water temperature, and during times of low light (early morning or after protracted cloudy periods). Given that such events rarely are recorded (though they may occur frequently), and that so many factors are related to DO depletion rates, existing data for most streams are insufficient to develop nutrient criteria for avoiding DO deficits. Such models probably will be developed in the future and development will be facilitated by recent improvements in tools for measuring and storing temporal data on instream DO concentrations. As more data become available, it will be possible to directly link frequency and severity of low DO events with nutrient loading.

Similar problems exist for predicting pH excursions. High pH is promoted by laminar flow and sunny conditions that, respectively, minimize atmosphere-to-water transport of CO_2 and maximize photosynthetic uptake of CO_2. Again, limited data for most streams hamper prediction of the degree of pH excursions as a function of TMDLs of N and P.

Nutrient criteria also could be set relative to other streams on a regional or national basis. Dodds et al. (1998) combined data from the EPA eutrophication survey (Omernik 1977) and several hundred streams and rivers in the US and analyzed the resulting cumulative frequency distributions. Half of the systems had TP >0.04

mg/L, and ½ had TN >0.9 mg/L. If the target is to bring streams and rivers to nutrient levels at or below current means, then using frequency distributions would be a viable approach to setting nutrient criteria. Problems with using such frequency distributions are discussed below.

Nutrient criteria may be more stringent when potential eutrophication of systems fed by rivers is a factor driving adoption of criteria. A common classification system suggests that 35 μg/L TP and a mean of 8 μg/L chlorophyll constitutes the dividing line between eutrophic and mesotrophic lakes (Organization for Economic Cooperation and Development [OECD] as cited in Rast et al. 1989). In contrast, data from Dodds et al. (1997) suggest that maximum benthic chlorophyll values are likely to exceed 200 mg/m^2 at 90 μg/L TP, and mean values of chlorophyll of 50 mg/m^2 are likely with 55 μg/L TP. Thus, unacceptable levels of chlorophyll may occur at much lower nutrient concentrations in lakes than streams.

Streams and rivers are less likely to accumulate as much algal biomass as lakes, given the same TP, because the lentic planktonic habitat is considerably more benign. Thus, there is fairly low chlorophyll yield per unit nutrient in streams. Comparing streams that flood at moderate frequency to more hydrodynamically stable artificial and spring-fed streams substantiates this view. Much higher benthic chlorophyll yield per unit TP than predicted by Dodds et al. (1997) is possible in controlled laboratory streams, outdoor artificial streams, or spring-fed rivers (Welch et al. 1992, Walton et al. 1995, Anderson et al. 1999, Welch et al., in press).

Likewise, planktonic chlorophyll yield is less in flowing waters than in lakes. A river with 8 μg/L chlorophyll would have ~48 μg/L TP, using the relationship proposed by Van Niewenhuyse and Jones (1996) for suspended chlorophyll in rivers as a function of TP. This value is ~1.4 times greater than the proposed mesotrophic/eutrophic boundary value for lakes and reservoirs (OECD as cited in Rast et al. 1989).

Last, a missing link in the above discussion is how to relate instream TN and TP concentrations to nonpoint and point sources of nutrients (i.e., to set TMDLs). Models predicting nutrient loading in streams need to be developed if mitigation strategies based on water column nutrients are to be successful. A method for determining instream TN and TP concentrations based on loading from point sources has been developed for use in the Clark Fork River (Dodds et al. 1997). Simple correlation techniques using data available in various regions may yield a relationship that can be used to predict what management strategies are necessary to bring nutrients from point sources, and consequently algal biomass, to target levels.

What factors may alter responses to nutrient control?

Variation of benthic algal biomass occurs among areas with different geology, land-use practices, and as a function of other biotic and abiotic factors. In this section, we discuss how regional differences (ecoregions) may play a role in setting nutrient criteria. In general, the relationships described above that can be used to set criteria based on algal biomass response, represent average responses.

Nutrient criteria should be set after considering the natural state of streams and rivers in an ecoregion. For example, in watersheds with high-PO_4^{3-} rock that is weathering at significant rates, low P concentrations may never occur. Large rivers will have higher TP, and yield of suspended algae will be different than in smaller streams (Van Niewenhuyse and Jones 1996). Furthermore, some watersheds have very high natural NO_3^- weathering rates (Halloway et al. 1998). Such areas naturally high in nutrients occur in several places in the US (Omernik 1977). Clearly, if nutrient levels naturally are high in a watershed, restrictive nutrient criteria cannot be met. Furthermore, when pristine systems are absent, determining natural baselines could be impossible.

Considerably greater levels of accuracy for prediction of benthic algal biomass with regression models are possible if region-specific data are available. For example, the general data sets used in regression models relating water column nutrients to benthic algae developed by Dodds et al. (1997) have a maximum r^2 of 0.43. Data from Missouri streams alone have r^2 values ranging from 0.47–0.60, depending upon year and whether TN or TP is used to predict algal biomass (Lohman et al. 1992). Biggs (1995) was able to construct a model for algal biomass with an r^2 of 0.89 in a region of New Zealand by normalizing for the effect of floods and using conductivity as a surrogate for nutrients. Fur-

TABLE 1. Various potential nutrient criteria set using different outcomes of concern related to instream nutrient concentrations. TN = total N, DIN = dissolved inorganic N.

Outcome	N (mg/L)	Total P (mg/L)	Comments
Toxicity, human	10 NO_3		US national standard
Toxicity, aquatic life, acute	0.03–5 NH_3		Fish and invertebrate data (Russo 1985)
Toxicity, aquatic life, chronic	0.005–1 NH_3		Fish data (Russo 1985, Miltner and Rankin 1998)
Oxygen deficit, pH excursion	?	?	Probably greater than levels presented below
Mean benthic chlorophyll <50 mg/m^2	0.47 TN	0.055	Large data set (Dodds et al. 1997)
Mean benthic chlorophyll <50 mg/m^2	0.25 TN	0.021	Lohman et al. (1992)
Maximum benthic chlorophyll <200 mg/m^2	3.0 TN	0.415	Calculated form Dodds et al. (1997)
Significant effect on biotic integrity index using invertebrates and fish	1.37 inorganic N	0.17	Headwater streams, Ohio (Miltner and Rankin 1998); effects less apparent in larger rivers
Systems with nutrient concentrations in upper ½	0.9 TN	0.04	Dodds et al. (1998)
Planktonic stream chlorophyll <8 μg/L	0.29 TN	0.042	Calculated from Van Nieuwenhuyse and Jones (1996); chlorophyll level from Organization for Economic Cooperation and Development (OECD, as cited in Rast et al. 1989); TN set by Redfield ratio (Harris 1986)
Lake mesotrophic/eutrophic boundary (planktonic chlorophyll <8 μg/L)	0.25 TN	0.035	OECD (as cited in Rast et al. 1989); TN set by Redfield ratio
Values set by State of Montana and cooperators	0.30 TN	0.020	Tri-State Implementation Council, Clark Fork Voluntary Nutrient Reduction Program
Levels leading to periphyton and macrophyte control	1.0 DIN	<0.020 (total dissolved)	Bow River, Alberta (A. Sosiak, Alberta Environmental Protection, personal communication)
Levels set to control summer phytoplankton		0.07	Tualatin River, Oregon (R. Burkhart, Oregon Department of Environmental Quality, personal communication).
Levels recommended to control maximum periphyton below 200 mg/m^2 for 50 d accrual	0.019 DIN	0.002 (soluble reactive)	(Biggs 2000)

thermore, all relationships that have been developed to date are from temperate regions, with most data from North America and New Zealand. Subtropical or polar regions could have quite different relationships. Thus, if data are available for an ecoregion, they should be used to set criteria for that region. Extant data such as state and tribal water quality records, USGS National Water Quality Assessment Program data, and Environmental Monitoring and Assessment Program results may serve as sources for such analyses.

Streams in a local region also may exhibit different relationships between TN or TP and benthic chlorophyll than those observed with larger-scale data sets. Thus, large, generalized data sets should not be the 1st choice for setting criteria, if local data are available. For example, the TN and TP values that yield a mean benthic chlorophyll of 50 mg/m^2, were lower for the detailed data set from Missouri than those from a larger data set (Table 1).

Nevertheless, one should not expect that the nutrient concentration yielding a given periphytic biomass will be markedly different among regions if other factors (i.e., light, grazing, etc.) are similar. For example, most regression relationships for chlorophyll-TP in lakes show slopes or chlorophyll:TP ranging from 0.5–1.0 (Ahlgren et al. 1988). Invertebrate grazing may result in low chlorophyll yield per unit nutrient in streams regardless of ecoregion (Bourassa and Cattaneo 1998), as is the case in lakes.

One potential problem with the ecoregion approach is that variation over time and space within a small area may be as great as the variation among ecoregions. The nutrient bioassays of Wold and Hershey (1999) demonstrate high variation of responses to N or P additions in 6 watersheds within 100 km of each other. The responses also were variable across season. Similar seasonal responses have been documented in New Zealand streams (Francoeur et al. 1999).

All the data sets that have been published linking algal biomass to water column nutrients in rivers and streams have a potential statistical problem (Lohman et al. 1992, Biggs 1995, Van Niewenhuyse and Jones 1996, Dodds et al. 1997, Chételat et al. 1999). Investigators may have introduced bias in site selection because sites were not selected randomly. In many cases, study sites are selected specifically to represent the broadest possible range of site types. Thus, extremely eutrophic and oligotrophic systems may be overrepresented. Such models may work well for the streams used to construct the models, but their application should be viewed with caution. For example, Dodds et al. (1997) reported relationships among nutrients and chlorophyll derived from literature values. The investigators who conducted this literature analysis had no way of knowing why investigators choose to investigate particular sites or if all data were reported. Streams with low amounts of periphyton may have been excluded, or researchers may have preferred to work in pristine systems. Last, much ecological investigation has concentrated on temperate, forested streams, which may have low levels of nutrients and where canopy cover may have restricted algal growth. Temperate forested streams may not be globally representative of all streams because they provide $<1/3$ of the runoff from the earth's continents (Dodds 1997). Thus, future sampling strategies to generate data that will be used to link stream eutrophication with nutrients should attempt to avoid investigator-specific biases. The models for setting criteria should be based on representative streams with data taken from the full population of streams and with each type of stream sampled in proportion to its relative occurrence. Such an approach has been taken in lakes (Peterson et al. 1999). Large data sets such as those collected by the USGS water quality monitoring network of the National Water Quality Assessment Program may be useful because sites could be selected from the databases to provide data specific to individual ecoregions.

If streams and rivers are turbid as a result of suspended particles, nutrient enrichment will have less influence on trophic status of the entire system. Sediments attenuate light, which becomes the factor limiting ecosystem production. However, even in turbid systems, enrichment may increase periphyton and macrophyte production in shallow portions of the river. Similarly, extensive shading by a riparian canopy will inhibit algal growth. Both conditions reduce chlorophyll yield per unit nutrient.

If macrophyte production predominates in streams and rivers, setting nutrient criteria will be difficult. We are not aware of any general published relationships between water column nutrients and macrophyte biomass. Such relationships may be very difficult to establish for macrophytes that are able to acquire nutrients from sediments through their root systems. However, nutrient control resulted in lowered macrophyte biomass in the Bow River, Alberta (A. Sosiak, Alberta Environmental Protection, personal communication), so future work on macrophyte-nutrient relationships could yield useful predictive models.

Conclusions

Many factors can regulate primary producers in streams, including nutrient availability, hydrodynamics, grazing, turbidity, riparian shading, and human impacts (e.g., addition of toxic compounds, global change, introduced species, watershed development). However, nutrient inputs are usually the most effectively managed factor. Factors in addition to nutrients need to be considered mainly because they can lead to cases of low algal biomass with high nutrients. Although these additional factors may decouple nutrient enrichment from algal biomass, most of these (e.g., flooding, grazing, turbidity) are not easily controlled at most sites. Thus, we are left with setting nutrient criteria as the primary way to mitigate problems of excessive algae.

Developing a single value that can be used for nutrient criteria in streams and rivers will be difficult, given the variety of reasons for setting the criteria (Table 1). To protect human health, no more than 10 mg/L NO_3^--N should be present. To avoid chronic toxicity by NH_3, no more than 0.02 mg/L NH_3^--N should be present. If the concern is eutrophication, then setting criteria for TN and TP is most reasonable.

If streams are not turbid, preventing maximum benthic chlorophyll levels from exceeding 200 mg/m^2 is reasonable because streams with higher levels are not aesthetically pleasing, and their recreational uses may be compromised. For benthic chlorophyll to remain below 200 mg/m^2 at the very least, TN should remain below 3 mg/L, and TP below 0.4 mg/L. Based on cumulative frequency distributions of nutrients, and assuming that ~½ the systems in the US have been impaired by excessive nutrients, levels of TN and TP would be set at 0.9 and 0.4 mg/L, respectively. If a mean of 50 mg/m^2 chlorophyll is the target (thus ensuring chlorophyll is <100 mg/m^2 most of the time), TN should be 0.47 and TP 0.06 mg/L. Lower levels for nutrient criteria should be considered for regions with more pristine systems (e.g., TN and TP levels of 0.3 and 0.02 mg/L, respectively, were chosen for the Clark Fork River in Montana, Table 1). If systems downstream are to be protected, even lower stream nutrient concentrations will be necessary in some situations.

A significant amount of monitoring data are necessary to refine recommendations for nutrient criteria. Some regions and agencies have data that can be used for this purpose. Data that would be useful to collect or glean from existing sources for many more systems include seasonal means and maxima for benthic and planktonic chlorophyll, associated water column nutrients, and diurnal DO concentrations for a variety of stream types. Such data should be collected in a way that avoids sampling bias. Data on macrophyte abundance related to nutrients, reference streams with acceptable algal and macrophyte biomass, and factors related to dominance by nuisance algal and macrophyte species also are sorely lacking for many regions.

Establishing rational criteria will require bridging the gap between managers and scientists. The managers will provide the realistic assessment of what needs to be accomplished, whereas the scientists can suggest the best available means to reach the management goals. Continued interplay between applied and basic approaches will be necessary if eutrophication in streams is to be controlled in an efficient manner.

Acknowledgements

W. D. thanks Kirk Lohman for early conversations and subsequent collaborations that led to much of this work. Jack Jones, Vicki Watson, Val Smith, and Jan Stevenson also provided useful input into developing these ideas. Barry Biggs, Debra Hart, Marty Gurtz, Amanda Parker, and 2 anonymous reviewers commented on the manuscript and Dolly Gudder provided technical assistance. This research was supported in part by the Konza LTER grant from the National Science Foundation. This paper is contribution no. 99-375-J from the Kansas Agricultural Experiment Station.

Literature Cited

AHLGREN, I., T. FRISK, AND L. KAMP-NIELSEN. 1988. Empirical and theoretical models of phosphorus loading, retention and concentration vs. lake trophic state. Hydrobiologia 170:285–303.

ANDERSON, E. L., G. M. SCHIMEK, E. B. WELCH, J. M. JACOBY, AND R. R. HORNER. 1999. Periphyton removal related to phosphorus and grazer biomass level. Freshwater Biology 41:633–651.

ARRUDA, J. A., AND C. H. FROMM. 1989. The relationship between taste and odor problems and lake enrichment from Kansas lakes in agricultural watersheds. Lake and Reservoir Management 5:45–52.

BIGGS, B. J. F. 1995. Periphyton biomass dynamics in gravel bed rivers: the relative effects of flows and nutrients. Freshwater Biology 22:209–231.

BIGGS, B. J. F. 2000. 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., AND M. E. CLOSE. 1989. The contribution of disturbance, catchment geology and land use to the habitat template of periphyton in stream ecosystems. Freshwater Biology 33:419–438.

BOWLING, L. C., AND P. D. BAKER. 1996. Major cyanobacterial bloom in the Barwon-Darling River, Australia, in 1991, and underlying limnological conditions. Marine and Freshwater Research 47: 643–657.

BOURASSA, N., AND A. CATTANEO. 1998. Control of periphyton biomass in Laurentian streams (Québec). Journal of the North American Benthological Society 17:420–429.

BURKHOLDER, J. M., AND H. B. GLASGOW. 1997. *Pfiesteria piscicida* and other *Pfiesteria*-like dinoflagellates: behavior, impacts, and environmental controls. Limnology and Oceanography 42:1052–1075.

CHÉTELAT, J., F. R. PICK, AND A. MORIN. 1999. Periphyton biomass and community composition in rivers of different nutrient status. Canadian Journal of Fisheries and Aquatic Sciences 56:560–569.

COOKE, G. D., E. B. WELCH, S. A. PETERSON, AND P. R. NEWROTH. 1993. Restoration and management of lakes and reservoirs. 2nd edition. Lewis Publishers, Boca Raton, Florida.

CORRELL, D. 1998. The role of phosphorus in the eutrophication of receiving waters: a review. Journal of Environmental Quality 27:261–266.

DODDS, W. K. 1993. What controls levels of dissolved phosphate and ammonium in surface waters? Aquatic Sciences 55:132–142.

DODDS, W. K. 1997. Distribution of runoff and rivers related to vegetative characteristics, latitude, and slope: a global perspective. Journal of the North American Benthological Society 16:162–168.

DODDS, W. K., AND D. A. GUDDER. 1992. The ecology of *Cladophora*. Journal of Phycology 28:415–427.

DODDS, W. K., J. R. JONES, AND E. B. WELCH. 1998. Suggested criteria for stream trophic state: distributions of temperate stream types by chlorophyll, total nitrogen and phosphorus. Water Research 32:1455–1462.

DODDS, W. K., V. H. SMITH, AND B. ZANDER. 1997. Developing nutrient targets to control benthic chlorophyll levels in streams: a case study of the Clark Fork River. Water Research 31:1738–1750.

FRANCOEUR, S. N., B. J. F. BIGGS, R. A. SMITH, AND R. L. LOWE. 1999. Nutrient limitation of algal biomass accrual in streams: seasonal patterns and a comparison of methods. Journal of the North American Benthological Society 18:242–260.

HALLOWAY, J. M., R. A. DAHLGREN, B. HANSEN, AND W. H. CASEY. 1998. Contribution of bedrock nitrogen to high nitrate concentrations in stream water. Nature 395:785–788.

HARRIS, G. P. 1986. Phytoplankton ecology: structure, function and fluctuation. Chapman and Hall, Cambridge, UK.

HARTMAN, P. E. 1983. Nitrate/nitrite ingestion and gastric cancer mortality. Environmental Mutagenesis 5:111–121.

HORNER, R. R., E. B. WELCH, AND R. B. VEENSTRA. 1983. Development of nuisance periphytic algae in laboratory streams in relation to enrichment and velocity. Pages 121–134 *in* R. G. Wetzel (editor). Periphyton of freshwater ecosystems. Dr. W. Junk Publishers, The Hague, The Netherlands.

HYNES, H. B. N. 1969. The biology of polluted waters. Liverpool University Press, Liverpool, UK.

KELLY, M. G. 1998. Use of community-based indices to monitor eutrophication in rivers. Environmental Conservation 25:22–29.

KELLY, M. G., AND B. A. WHITTON. 1995. The trophic diatom index: a new index for monitoring eutrophication in rivers. Journal of Applied Phycology 7:433–444.

LOEHR, R. C. 1974. Characteristics and comparative magnitude of non-point sources. Journal of the Water Pollution Control Federation 46:1849–1872.

LOHMAN, K., J. R. JONES, AND B. D. PERKINS. 1992. Effects of nutrient enrichment and flood frequency on periphyton biomass in northern Ozark streams. Canadian Journal of Fisheries and Aquatic Sciences 49:1198–1205.

LOHMAN, K., AND J. C. PRISCU. 1992. Physiological indicators of nutrient deficiency in *Cladophora* (Chlorophyta) in the Clark Fork of the Columbia River, Montana. Journal of Phycology 28:443–448.

MICHAEL, H. J., K. J. BOYLE, AND R. BOUCHARD. 1996. Water quality affects property prices: a case study of selected Maine lakes. Miscellaneous Report 398. Maine Agricultural and Forest Experiment Station, Orono, Maine. (Available from: http://www.umaine.edu/mafes/ or Maine Agricultural and Forest Experiment Station, 5782 Winslow Hall, University of Maine, Orono, Maine 04469–5782.)

MILTNER, R. J., AND E. T. RANKIN. 1998. Primary nutrients and the biotic integrity of rivers and streams. Freshwater Biology 40:145–158.

NORDIN, R. N. 1985. Water quality criteria for nutrients and algae (technical appendix). Water Quality Unit, Resources Quality Section, Water Management Branch, British Columbia Ministry of the Environment, Victoria, BC. (Available from: http://www.env.gov.bc.ca or Water Quality Section, BC Environment, PO Box 9340, Station Provincial Government, Victoria, BC V8W 9M1.)

ODUM, H. T. 1956. Primary production in flowing waters. Limnology and Oceanography 1:102–117.

OMERNIK, J. M. 1977. Nonpoint source-stream nutrient level relationships: a nationwide study. EPA-600/3-77-105. Special Studies Branch, Corvallis Environmental Research Laboratory, Office of Research and Development, US Environmental Protection Agency, Corvallis, Oregon.

PAN, Y., R. J. STEVENSON, B. H. HILL, A. T. HERLIHY, AND G. B. COLLINS. 1996. Using diatoms as indicators of ecological conditions in lotic systems: a regional assessment. Journal of the North American Benthological Society 15:481–495.

PETERSON, S. A., N. S. URQUHART, AND E. B. WELCH. 1999. Sample representativeness: a must for reliable regional lake condition estimates. Environmental Science and Technology 33:1559–1565.

QUINN, J. M. 1991. Guidelines for the control of undesirable biological growths in water. Consultancy Report No. 6213/2. National Institute of Water and Atmospheric Research, Hamilton, New Zealand.

RABALAIS, N. N., R. E. TURNER, W. J. WISEMAN, AND Q. DORTCH. 1998. Consequences of the 1993 Mississippi River flood in the Gulf of Mexico. Regulated Rivers: Research and Management 14:161–177.

RAST, W., V. H. SMITH, AND J. A. THORNTON. 1989. Characteristics of eutrophication. Pages 37–63 *in* S. Ryding and W. Rast (editors). The control of eutrophication in lakes and reservoirs. UNESCO and Parthenon, Paris.

RUSSO, R. C. 1985. Ammonia, nitrite and nitrate. Pages 455–471 *in* G. M. Rand and S. R. Petrocelli (editors). Fundamentals of aquatic toxicology. McGraw-Hill, Washington, DC.

SAS, H. 1989. Lake restoration by reduction in nutrient loading: expectations, experiences, extrapolations. Academia Verlag Richarz, St. Augustin, Germany.

STOCKNER, J. G., AND K. S. SHORTREAD. 1988. Response of *Anabaena* and *Synechococcus* to manipulation of nitrogen:phosphorus ratios in a lake fertilization experiment. Limnology and Oceanography 33: 1348–1361.

USEPA (US ENVIRONMENTAL PROTECTION AGENCY). 1998. Notice of availability of clean water action plan. Federal Register 63:14109–14112.

VAN NIEWENHUYSE, E. E., AND J. R. JONES. 1996. Phosphorus-chlorophyll relationship in temperate streams and its variation with stream catchment area. Canadian Journal of Fisheries and Aquatic Sciences 53:99–105.

WALTON, S. P., E. B. WELCH, AND R. R. HORNER. 1995. Stream periphyton response to grazing and changes in phosphorus concentration. Hydrobiologia. 302:31–46.

WELCH, E. B. 1992. Ecological effects of wastewater. 2nd edition. Chapman and Hall, London, UK.

WELCH, E. B., E. L. ANDERSON, J. M. JACOBY, B. F. BIGGS, AND J. M. QUINN. Invertebrate grazing of filamentous green algae in outdoor channels. Verhandlungen der Internationalen Vereinigung fur theoretische und angewandte Limnologie 27 (in press).

WELCH, E. B., R. R. HORNER, AND C. R. PATMONT. 1989. Prediction of nuisance periphytic biomass: a management approach. Water Research 23:401–405.

WELCH, E. B., J. M. JACOBY, R. R. HORNER, AND M. R. SEELEY. 1988. Nuisance levels of periphytic algae in streams. Hydrobiologia 157:161–168.

WELCH, E. B., J. M. QUINN, AND C. W. HICKEY. 1992. Periphyton biomass related to point-source enrichment in seven New Zealand streams. Water Research 26:669–675.

WNOROWSKI, A. 1992. Tastes and odours in the aquatic environment: a review. Water SA (Pretoria) 18: 203–214.

WOLD, A. P., AND A. E. HERSHEY. 1999. Spatial temporal variability of nutrient limitation in 6 north shore tributaries to Lake Superior. Journal of the North American Benthological Society 18:2–14.

About the Authors

Walter Dodds is an Associate Professor at Kansas State University. He specializes in algal ecology with emphasis on nutrients and algae in streams. Gene Welch is a Professor at the University of Washington. He has spent many years working on eutrophication control in lakes and streams. Both authors are involved in developing a manual with the USEPA that can be used by state, tribal, and regional officials to guide establishment of nutrient criteria.

Attachment 40:

Nitrogen and Phosphorus Losses from Winter Disposal of Dairy Manure1
Klausner et al. 1976

Nitrogen and Phosphorus Losses from Winter Disposal of Dairy Manure[1]

S. D. Klausner, P. J. Zwerman, and D. F. Ellis[2]

ABSTRACT

An evaluation of surface runoff losses of inorganic nitrogen and total soluble phosphorus from fields receiving winter applications of dairy manure was conducted. Runoff losses, as derived from natural precipitation, were accumulated for the time period 1 Jan. to 31 Mar. for 3 consecutive years. The influence of rate of manure application (35, 100, and 200 metric tons/ha) and climatological variability within and between years was of primary importance. Average runoff values of inorganic nitrogen for the three rates of application were 16, 1, and 0.2 kg/ha for 1972, 1973, and 1974, respectively. Phosphorus values averaged 3.5, 0.7, and 0.01 kg/ha for the 3 respective years. Adverse weather conditions during the winter application in 1972 were largely responsible for increased nutrient discharges in runoff. Results indicated that manure disposal during active thaw periods can result in increased nutrient losses. Losses were minimized when manure was applied and then covered with snow, melting at a later date. The 35 metric tons/ha rate of application, applied on frozen soil and then covered with snow, resulted in nutrient losses that differed little from areas that received no manure.

Additional Index Words: surface runoff, fertilization, water quality, animal waste.

The timing of manure applications to the soil is an important consideration in farm management. Concern has been expressed about applications during winter months and the effects of land runoff on receiving waters.

From the standpoint of decreased nutrient movement, the optimum time of manure application would be when stream flow decreases and evapotranspiration begins to maximize (11). In New York State this period coincides with the growing season. Because of time, labor, storage, and land limitations it is difficult to apply annual manure accumulations prior to and during the growing season, therefore, fall and winter applications are practiced.

Without manure, losses of nitrogen and phosphorus in winter runoff from a watershed can be expected to be primarily from residual soil fertility (9), leaching of organic material on the soil surface (3, 15) and from precipitation (4, 5, 10). Additional nutrient losses during the winter can be expected from manure disposal when a combination of poor manure management techniques are practiced in combination with inclement weather (9, 12).

Winter runoff losses can be difficult to predict because of the wide variation in climatological sequences within a year and between years. Characteristically, the Northeast and North Central states have 2.5 cm or more of snow on the ground for 100-140 days each winter (1). The presence of snow in these areas is very much a matter of chance (7) and the soil beneath the snow may previously have been subject to *concrete freezing* or *honeycomb freezing*. In the former case, water infiltration may be difficult or impossible. In the latter case, infiltration may be good (14).

The following presentation is concerned with the losses of inorganic nitrogen (NH_4-N + NO_3-N) and total soluble phosphorus in surface runoff associated with various rates of winter-applied dairy manure for 3 consecutive years.

METHODS AND MATERIALS

The experimental area at the Cornell Research Farm, Aurora, New York is composed of approximately 12 ha of a Glossoboric Hapludalf fine loamy, mixed, mesic. The soil texture is silt loam and is moderately to somewhat poorly drained. A total of 48 plots (each 0.32 ha; 61 by 53.0 m) were installed in 1956. The study was maintained as a drainage experiment until 1965. From 1969–1972, these plots were used to evaluate water quality from farming systems using varying rates of mineral fertilizers. In the winter of 1972, free stall dairy cow manure was substituted for mineral fertilizer.

The manure treatments for a continuous corn (*Zea mays* L.) cropping system were 35, 100, and 200 wet metric tons/ha. Three different manure rate treatments were applied as a single application for 3 consecutive years. The 100 and 200 metric ton/ha rates were replicated twice; the 35 metric ton/ha rate was replicated four times for a total of eight experimental plots per year. The eight plots designed for a winter application represent one-third the number of plots in a much larger experiment. The remaining 16 plots during 1972 were slated for a manure application either in the spring or summer months. These plots, instrumented identically to the eight under discussion, can serve as a series of check plots for the winter of 1972 only.

Manure without bedding was supplied by a local dairy farmer from his free stall operation and was stored for varying lengths of time in an exposed pile. Each load of manure was weighed and applied with conventional manure handling equipment. Upon application, manure samples were taken from each spreader load. Representative samples for analysis were composite samples of three loads. Dry matter content was determined on one subsample. A second subsample was homogenized with an equal weight of water in a blender for 1 minute. The homogenate was analyzed for NH_4-N (2), total Kjeldahl-N (2), total soluble P (16) and total-P (16).

Surface water was controlled by a series of small interception cross ditches and broad shallow runoff ditches up and down slope. Individual plots had surface slopes ranging from 2 to 4%. Runoff water from each plot was diverted into a 30.5-cm H-flume which recorded the flow variation with time. After measurement, a subsample of approximately 1% was collected by an electrically driven Coshocton wheel. The sample was further divided by a splitter arrangement which collected either 10% or 20% of the subsample. The integrated water sample taken over the entire period of flow was collected in an underground storage tank. After each runoff event, a 250-ml subsample of the suspension was taken for analysis. The remaining suspension was pumped into an above ground storage tank for sediment analysis. Sediment losses are not reported here. Each installation was insulated and contained a heating lamp to avoid freezing during the winter.

Runoff samples of 250 ml were centrifuged at 37,000 RCF (relative centrifugal force) for 30 minutes. The supernatant was analyzed for NH_4-N (13), NO_3-N (8), inorganic-P (6), and total soluble P (16).

All data presented are for the liquid fraction of surface runoff as derived from natural precipitation during the time period from 1 Jan. to 31 Mar. for 1972, 1973, and 1974.

[1] Contribution from the Dep. of Agronomy, Cornell University, Ithaca, NY 14853. Supported in part by the Environmental Protection Agency (EPA Grant No. S800767). Agronomy Paper No. 1125. Received 20 Mar. 1975.

[2] Research Associate, Professor of Soil Conservation, and Extension Specialist, respectively.

Table 1—Quantity of nitrogen, phosphorus, and dry matter applied in dry manure

Rate	Year	NH_4-N	Total-N	Total soluble-P	Total-P	Dry matter
metric tons/ha		kg/ha				
35	1972	23	149	8	69	8,596
100		187	454	18	108	21,851
200		214	772	60	178	49,237
35	1973	29	162	19	46	8,321
100		57	416	38	113	29,477
200		136	958	111	210	44,330
35	1974	64	262	19	43	7,893
100		189	568	47	132	23,980
200		375	1,143	120	254	59,158

Table 2—Comparison of several weather parameters for a 3-year period. Sum of Jan., Feb., and Mar.

	1972	1973	1974
Total precipitation, cm	17.5 (−0.8)*	15.9 (−1.8)	14.4 (−3.0)
Snowfall, cm	190 (−16)	88 (−83)	89 (−38)
Frozen soil, days†	62	12	0
Snow cover, days	75	39	64
Runoff, cm‡	2.9	2.8	0.1

* Deviations from 14 and 24 year means for precipitation and snowfall, respectively.
† Average daily soil temperature ≤ 0C at the 10 cm depth under sod.
‡ Averaged over all winter applied plots.

RESULTS AND DISCUSSION

Dairy manure was applied in mid-Feb. 1972 and in mid-Jan. during 1973 and 1974 on frozen soil. The designated rates of application were made on a wet weight basis. Varying lengths of manure stored in the open prior to application explains the variability of N and P inputs from year to year (Table 1).

A comparison of several weather parameters, as summed over the months of Jan., Feb., and Mar., for the 3 year period are given in Table 2. These data indicate that the winter of 1972 was fairly typical, while 1973 and 1974 had below average snowfall. There were striking differences in the number of days the soil was frozen and the number of days with a snow cover over the soil surface. During the 3-month period, the soil was frozen for 62 days in 1972, 12 days in 1973, and 0 days in 1974. Soil temperatures were taken at the weather station on the farm at a 10 cm depth under sod. A dense sod cover would tend to insulate the soil to a greater degree than the corn stubble on the experimental plots. The number of days when the soil was actually frozen may be an underestimate. The soil might be frozen at the surface, but not at 10 cm. Still, the data may be meaningful for relative comparisons of the 3 years.

The cumulative losses of N and P in surface runoff are given in Fig. 1. There were significantly greater nutrient losses in 1972 than in 1973 and 1974. In 1972 there was a negligible difference in nutrient runoff between the control plots and the 35 metric tons/ha treatment. A larger, but nonsignificant difference occurred at the 200 metric tons/ha rate of application. The losses from the 100 metric tons/ha rate were significantly higher than for other treatments. During 1973 and 1974, N and P losses did not show meaningful differences with respect to the year or to the rate of manure application.

The greater losses measured in the winter of 1972 in comparison to 1973 and 1974 were due to a complex series of circumstances at the time of disposal. Figure 1 shows the tremendous increased losses of nitrogen and phosphorus associated with the 100 metric tons/ha rate. The 35 and 200 metric tons/ha rates were applied on bare frozen soil. Almost immediately, 40 cm of snowfall covered the manure, thus delaying the disposal of the 100 metric tons/ha rate for 10 days. When the snow had partially thawed, the 100 metric tons/ha rate was applied on 15-20 cm of dense melting snow overlying frozen soil, a disposal condition exemplifying the worst possible manure management practice. Figure 2 illustrates the weekly distribution of snowfall for the three winter periods. The arrows denote the time of manure disposal.

The first snowmelt event occurred 10 days after the disposal of the 35 and 200 metric tons/ha and during the disposal of the 100 metric tons/ha rate. This event accounted for a very large portion of the total 1972 winter runoff losses (Table 3). The disposal of manure on melting snow is undoubtedly disadvantageous in comparison to manure spread earlier and then covered with snow, melting at a later date. Spreading on top of melting snow not only eliminates snow cover protection, but also causes rutting of the snowpack and soil beneath which aids in the channelization of runoff water. Channels formed in the snow or soil may have an influence on runoff at a later date.

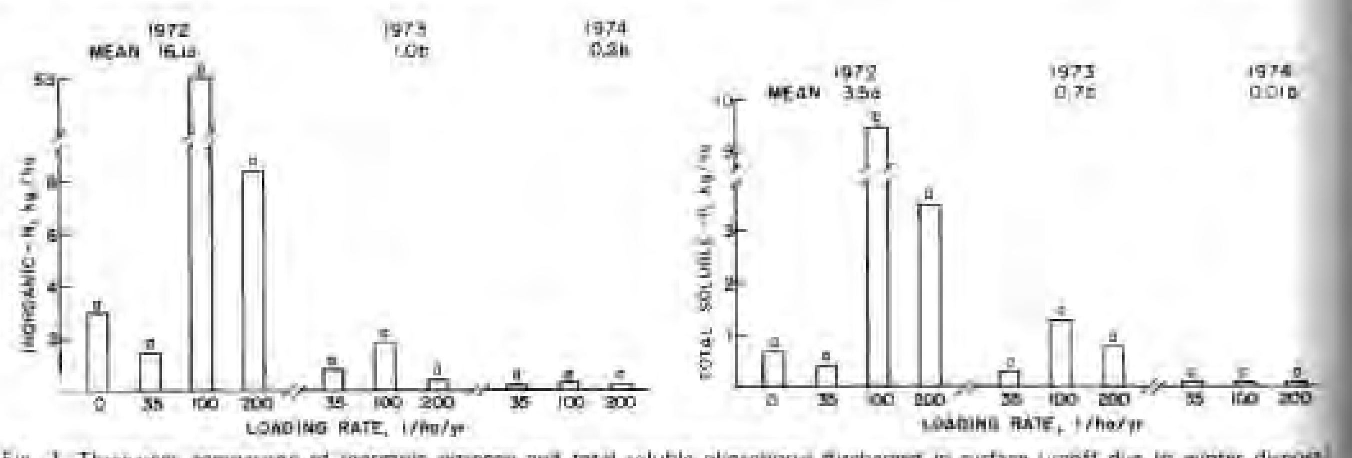


Fig. 1—Three-year comparison of inorganic nitrogen and total soluble phosphorus discharged in surface runoff due to winter disposal. Cumulative losses are from 1 Jan. to 21 Mar.

The excessive nutrient losses associated with the 100 metric tons/ha treatments in 1972 was not a reflection of the rate of application, but of the direct influence of the weather and soil surface conditions at the time of application. Had those conditions been less severe, nutrient losses from the 100 metric tons/ha rate would probably have taken an intermediate position between the losses resulting from the 35 and 200 metric tons/ha applications.

The rate of application effect varied from year to year. Ignoring the 100 metric tons/ha rate for the winter of 1972, because it was applied under very dissimilar conditions, the 200 metric tons/ha rate resulted in approximately four times the nitrogen and phosphorus losses in runoff as did the 35 metric tons/ha rate during 1972. Nutrient losses for all three rates of application were essentially identical in 1973 and 1974. Moreover, the 35 and 200 metric tons/ha rates applied across an array of weather patterns showed no significant nutrient loss differences compared to the control area in 1972, or between years which varied considerably in snowfall, days of snow cover, and frozen soil.

SUMMARY AND CONCLUSIONS

Accumulative winter losses (1 Jan.–31 Mar.) of soluble N and P in the liquid fraction of runoff as a function of three different rates of manure application (35, 100, and 200 metric tons/ha) are presented. Emphasis was placed on the rate of application and the variation in nutrient losses that may be expected from year to year.

Nutrient losses of nitrogen and phosphorus, averaged over all rates of application, were considerably greater in 1972 than in 1973 or 1974. The winter of 1972 had typical precipitation, while the 1973 and 1974 winter periods were below average. Average runoff losses of nitrogen for the three rates of application were 16, 1, and 0.2 kg/ha for 1972, 1973, and 1974, respectively. Phosphorus values averaged 3.5, 0.7, and 0.01 kg/ha for the 3 respective years.

Adverse weather conditions during and after the winter disposal in 1972 were largely responsible for increased nutrient discharges in runoff. This was especially evident at the 100 metric tons/ha rate which was applied on top of

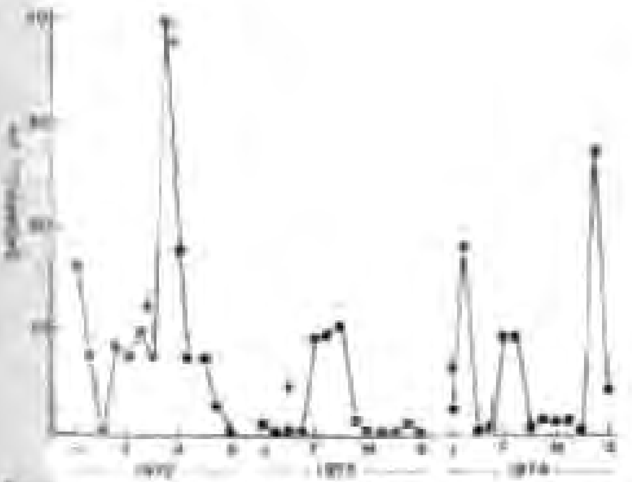


Fig. 2—Weekly distribution of snowfall for three consecutive winter periods. Arrows indicate the period of manure application.

Table 3—Nutrient losses and runoff from a single snowmelt event (29 Feb.) in comparison to total winter runoff losses in 1972

Rate	Runoff		Inorganic N		Total soluble P	
	29 Feb.	Jan.–Mar.	29 Feb.	Jan.–Mar.	29 Feb.	Jan.–Mar.
metric tons/ha	cm		kg/ha			
0	2.7 a	3.3 a	2.1 a	3.1 a	0.6 a	0.1 a
35	0.9 a	1.5 a	1.0 a	1.4 a	0.3 a	0.3 a
100	4.1 a	7.0 a	43.6 b	54.2 b	6.8 b	9.3 b
200	1.8 a	1.6 a	4.1 a	8.3 a	1.1 a	3.3 a

* Means followed by the same letter are not statistically significant at the 5% level.

melting snow. This single snowmelt event illustrated the necessity for avoiding spreading of manure during active thaw periods.

The data clearly indicated that manure disposal during active thaw periods can result in excessive nutrient losses, while nutrient losses were minimized when manure is applied and then covered with snow, melting at a later date. The 35 metric tons/ha rate of application, applied on frozen soil and then covered with snow before a thaw period, resulted in the lowest nutrient losses when compared to areas that received no manure at all.

LITERATURE CITED

1. Anonymous. 1941. Climate of the United States. Yearbook of Agric. System of Documents, U. S. Government Printing Office, Washington, D.C. p.728.
2. Bremner, J. M. 1965. Analysis of total nitrogen and inorganic forms of nitrogen. p. 1149–1232. In Methods of soil analysis. Am. Soc. of Agron., Madison, Wis.
3. Bryant, J. C., and C. S. Slater. 1948. Runoff water as an agent in the loss of soluble materials from certain soils. Iowa State Coll. J. Sci. 22:269–312.
4. Daniel, H. A., H. M. Elwell, and H. J. Harper. 1938. Nitrate nitrogen content of runoff water from plots under different cropping systems on soil classified as Vernon fine sandy loam. Soil Sci. Soc. Am. Proc. 3:230–233.
5. Feth, J. H. 1966. Nitrogen compounds in natural water—a review. U. S. Geol. Surv., Menlo Park, Calif. 1st Quarter. p. 41–58.
6. Fiske, C. H., and Y. Subbarow. 1925. The colorimetric determination of phosphorus. J. Biol. Chem. 66:375.
7. Garstka, W. U. 1964. Snow and snow survey. Handbook of applied hydrology. Sec. 10. McGraw-Hill Publishing Co., New York.
8. Jacobs, M. B., and S. Hochheiser. 1958. Continuous sampling and ultramicro determination of nitrogen dioxide in air. Anal. Chem. 30:426–428.
9. Klausner, S. D., P. J. Zwerman, and D. F. Ellis. 1974. Surface runoff losses of soluble nitrogen and phosphorus under two systems of soil management. J. Environ. Qual. 3:42–46.
10. Likens, G. E. 1972. The chemistry of precipitation in the Central Finger Lakes Region. Cornell Univ. Water Res. Marine Sci. Center, Ithaca, N. Y. Tech. Rep. 50. p. 1–61.
11. Loehr, R. C., S. D. Klausner, and T. W. Scott. 1973. Disposal of agricultural wastes on land. Fourth Geneva Conf., Montana State Univ. p. 101–178.
12. Midgley, A. R., and D. E. Dunklee. 1945. Fertility runoff losses from manure spread during the winter. Univ. of Vermont Agric. Exp. Sta. Bull. 523. p. 1–19.
13. Russell, J. A. 1944. The colorimetric estimation of small amounts of ammonia by the phenol-hypochlorite reaction. J. Biol. Chem. 156:457–461.
14. Storey, H. C. 1955. Frozen soil and spring and winter floods. The Yearbook of Agric. U. S. Government Printing Office, Washington, D.C. p. 179–184.
15. Timmons, D. R., R. F. Holt, and J. J. Latterell. 1970. Leaching of crop residues as a source of nutrients in surface runoff waters. Water Resour. Res. 6:1367–1375.
16. U. S. Environmental Protection Agency. 1971. Methods for chemical analysis of water and wastes. EPA, Anal. Qual. Contr. Lab., Cincinnati, Ohio.

Attachment 41:

Pollution Potential and Corn Yields from Selected Rates and Timing of Liquid Manure Applications
Phillips et al. 1981

Pollution Potential and Corn Yields from Selected Rates and Timing of Liquid Manure Applications

P. A. Phillips, J. L. B. Culley, F. R. Hore, N. K. Patni

ABSTRACT

A 6-year study was conducted to determine the effects of rate and time of liquid manure application, chemical fertilizer application, and no fertilizer, on the chemical composition of surface and subsurface water and on crop yield. Liquid manure was applied at three rates of 224, 560 and 897 kg/(ha•yr) of N in accordance with four application schedules (i.e. spring, fall, split rates in spring and fall, and winter). In all cases except winter application, manure was incorporated by plowing at time of application.

During spring snow-melt, surface runoff concentrations of inorganic N, P, and K from winter-applied manure increased approximately in proportion to increased application rate. Also, they were much higher than concentrations from spring, fall, spring-fall, and chemical fertilizer treatments.

In contrast to spring snowmelt surface runoff, tile drain effluent NO_3-N concentrations from the plots receiving manure at nearly 900 kg/(ha•yr) of N appeared to be little different from the plot chemically fertilized with 134 kg/(ha•yr) of N. However, at and above the 560 kg/(ha•yr) of N (140 kg/(ha•yr) of P) rates of manure the drain effluent PO_4-P concentrations tended to be higher than the concentration resulting from chemical fertilizer applications.

Most of the nitrogen and phosphorus in surface runoff during June storms was associated with suspended sediment that resulted from erosion. Neither the amounts of sediment nor their total N and total P contents were affected by manure or fertilizer applications. Although the concentrations of inorganic N and PO_4-P in the water portion of June storm runoff were small (<3 percent) compared to those in the sediment, plots with higher rate spring-applied manure tended to have higher concentrations of inorganic N, PO_4-P and K.

No significant differences in silage corn yields were observed amongst any of the manure and the chemical fertilizer treatments.

Based on trends in the water quality results, it is concluded that winter application of manure at any rate on areas that contribute runoff directly to bodies of surface water is not recommended. Non-winter applications of manure at and above rates of 560 kg/(ha•yr) of N may also lead to water quality impairment.

Article was submitted for publication in November 1979; reviewed and approved for publication by the Soil and Water Division of ASAE in March 1980. Presented as ASAE Paper No. 79-2117.

Contribution No. 1-114 from the Engineering and Statistical Research Institute, No. 51 from the Land Resource Research Institute, and No. 891 from the Animal Research Institute.

The authors are: P.A. PHILLIPS, Engineering and Statistical Research Institute, J.L.B. CULLEY, Land Resource Research Institute, F.R. HORE, Engineering and Statistical Research Institute, and N.K. PATNI, Animal Research Institute, Research Branch, Agriculture Canada, Ottawa, ON.

Acknowledgements: The authors wish to acknowledge the assistance of D. J. Gall, Engineering & Statistical Research Institute, with the runoff measuring installations and their operation, C. O'Meara, Land Resource Research Institute, with the laboratory analyses of water samples, and W. B. Woods, B. Dow and R. Boyd with field cropping operations.

INTRODUCTION

Changes in water quality indicators such as bacterial counts, pesticide residues, sediment loads, heavy metals, BOD, COD, and dissolved chemical concentrations in groundwater and surface water, have been linked to normal agricultural practices such as tillage, continuous cropping, use of pesticides, and spreading of manure and chemical fertilizers. In 1973, a long-term field plot experiment was initiated to study the effects of time and rate of liquid manure application on land cropped continuously in corn. The objective was to determine the nutrient losses to water supplies and accumulations in soil over a wide range of manure management alternatives. This paper is concerned with the effects of manure on nitrogen (N), phosphorus (P) and potassium (K) in subsurface and surface runoff, and on crop response.

EXPERIMENTAL DESIGN AND PROCEDURES

A gradually sloping (0.8 percent) field of an imperfectly drained Aquic Eutrochrept on the Central Experimental Farm, Ottawa, Ontario, was divided into 14 plots, each 75.6 m by 11.6 m, that has been cropped continuously in silage corn since 1973 (Fig. 1). The soil (Mountain series) is a sandy clay loam to a depth of about 80 cm covering clay loam. Selected manure and fertilizer treatments (Table 1) were randomized and applied each year to these plots. Manure rates were based on the mass of N and were chosen to be 224, 560, and 897 kg/(ha•yr) of N in accordance with four alternate application schedules: spring (usually the first week of May), fall (usually the last week of September), one-half of rate in spring - one-half in fall, and winter (usually the first week of December when ground is frozen and/or snow covered). One of the remaining two plots received no amendment while other received chemical fertilizer at the annual rate of 134 kg/(ha•yr) of N, 49 kg/(ha•yr) of P, and 93 kg/(ha•yr) of K, broadcast prior to seeding.

Each plot contained a subsurface plastic drain, positioned longitudinally down the center line of the plot at a depth of 0.7 m, which discharged into a continuous recording flow meter housed in an instrument pit (Fig. 1). Exterior runoff and cross flow between plots was controlled by a system of ditches and perimeter dikes and the plot surface runoff was measured by a continuous-recording flume mounted in the instrument pit (Fig. 1).

  0001-2351/81/2401-0139$02.00139
This article is reprinted from the TRANSACTIONS of the ASAE (Vol. 24, No. 1, pp. 139 - 144, 1981)

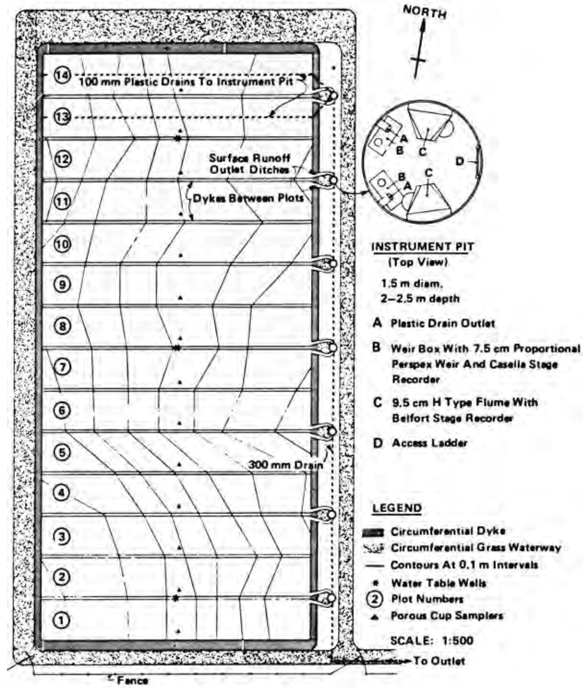


FIG. 1 Experimental plot area at the Central Experimental Farm, Ottawa.

Attempts were made to maintain flow measuring equipment during freezing months but results from December to April proved unreliable, so the equipment was removed at this time.

Manured plots received dairy cattle liquid manure (average dry matter content of 8.8 percent) spread directly from tanker trucks since treatments began in the fall of 1973. Manure volumes required to match the chosen application rates of N were estimated from an initial analysis of the manure N, one day prior to spreading. At the time of application, manure samples were taken for analysis of total N, P and K to determine the actual application rates. Except for the winter treatments, the manured plots were plowed down immediately to minimize odor problems. All plots were fall plowed and 6 of the plots (spring, spring-fall schedules) received two plowings per year. All plots were disked three times prior to seeding. Corn (United H-7) was seeded in mid-May at 46,200 plants/ha. During the first week of September, four 2.5 m length rows from the east and west halves of each treatment were harvested and weighed. Subsamples were dried at 70 °C and weighed to determine yields on a dry-weight basis.

TABLE 1. Planned Manure Treatments And Average Rates (Standard Deviations In Brackets) Of Nitrogen, Phosphorous And Potassium Applied To Plots Over Six Years Ending Spring, 1979.

Plot no.	Treatment K		Nutrient applied, kg/(ha•yr)		
	Timing schedule	Rate of N kg/(ha•yr)	N	P*	K†
12	Spring	224	230 (27)	58 (7)	201 (69)
7	Fall	224	228 (19)	54 (6)	239 (92)
11	Spr.-Fall	112/112	212 (50)	57 (21)	191 (63)
3	Winter	224	223 (33)	48 (10)	174 (28)
2	Spring	560	570 (69)	150 (32)	505 (198)
10	Fall	560	568 (67)	134 (19)	600 (254)
6	Spr.-Fall	280/280	572 (51)	141 (16)	551 (204)
5	Winter	560	526 (85)	124 (29)	413 (77)
4	Spring	897	891 (125)	234 (42)	792 (290)
8	Fall	897	923 (96)	217 (28)	973 (405)
9	Spr.-Fall	448/448	897 (67)	222 (22)	870 (328)
13	Winter	897	861 (115)	203 (42)	678 (109)
1	Spring	Chem. Fert.	134	49	93
14	Control	0	0	0	0

* P x 2.29 = P_2O_5
† K x 1.2 = K_2O

Flow events induced by snow-melt or rainfall, which involved at least 11 of the 14 plots, were sampled manually and subjected to nutrient analysis to determine the effects of treatments. Recognizing that water quality during a given event can change with time, samples were taken at intervals during each event to give a representative average value. For long events such as spring snowmelt, which extend over several weeks, grab samples were taken twice daily; sampling for events of relatively short duration, such as storm runoff, was done at the beginning, peak and end of the flow event. Therefore average nutrient concentrations for snow-melt and rainfall events were expected to be based on at least 2 to 3 samples per event-day. All samples were stored in glass bottles at 1°C prior to analysis.

Snow-melt surface runoff samples were analyzed for nitrate plus nitrite (NO, + NO_2) nitrogen, (Keng and Menage, 1970)ammonium-nitrogen (NH_4 -N)(Quin *et.*

al., 1974) and orthophosphorus (PO_4-P)(Sowden 1972) using an autoanalyser. Potassium (K) concentrations were determined using atomic absorption spectroscopy. Storm surface runoff samples that contained suspended sediment were centrifuged at 15,000 rpm for 10 min; the sediments were analyzed for total N and total P (after digestion in concentrated HNO_3- $HClO_4$ acid). The supernatant received the same analyses as snow-melt runoff samples. Subsurface drainage water was analyzed for (NO_3 + NO_2)-N, NH_4-N, PO_4-P and K in a manner similar to snowmelt runoff samples. As NO_2 concentrations in water and waste water are generally very low, and as NO_2 is readily converted to NO_3 by bacteria (Sawyer and McCarty, 1967), (NO_3 + NO_2)-N is denoted as NO_3-N below.

Total nitrogen of the manure and a few runoff samples was determined by the standard Kjeldahl method. Prior to determining phosphorus and potassium manure was digested in nitric-perchloric acid. The P in solution was determined by colorimetry while the K was determined by atomic absorption spectroscopy.

RESULTS AND DISCUSSIONS

Manure Application

Six-year averages of N, P and K applied to the experimental plots are shown in Table 1. Mean N rates actually applied are all within 6 percent of the chosen rates. The standard deviations of the N applied are mostly within about 13 percent of the chosen rates indicating a consistent application program. The applications of P and K show more variability but met or exceeded the yearly crop requirements.

Nutrient Content of Runoff Water

During the six-year study period there were occasions when rainfall induced flow on several plots. However, there were only a total of 12 surface and subsurface "Flow Events" used to determine the effect of treatments as was explained in the procedures. Spring snow-melt constituted the major portion of the annual surface and subsurface runoff (eight events) with June storms and late fall rains contributing small amounts of runoff some years (four events). Snow-melt surface runoff was estimated to average about 15 cm per year and contained little suspended sediment. June storm runoff volumes were relatively small (< 1 cm per event) compared to spring snow-melt, but carried substantial amounts of sediment from the plot surface. The potential for storm runoff was negligible during July, August, and September due to the protective effects of the corn plant. Late in the fall there i s also a potential for subsurface flow in this region. However, with the exception of two very brief events, fall rainfall during the course of this study was insufficient to cause much subsurface runoff at this site. Therefore snow-melt accounted for over 90 percent of the runoff from these plots.

The treatment effects are discussed in terms of nutrient concentrations in the runoff. Although treatment comparisons of amounts lost based on concentrations and flow volumes are important, slight non-uniformity in the very gradual plot slopes caused appreciable plot variation in flow volumes during events, and precluded these comparisons. Others (Zwerman *et al.*, 1974; Klausner *et al.*, 1976, Gast *et al.*, 1978) have recognized such variations in surface and subsurface runoff from plots where treatment comparisons were made.

Nutrient concentrations for each snow-melt event represent the average of 12 to 24 plot-samples, while 2 or 3 plot-samples were obtained per rainfall induced flow event.

Surface water: Winter applications of manure at all three rates consistently resulted in higher concentrations of (NO_3 + NH_4)-N*,PO_4-P and K (Table 2) in snowmelt runoff water compared to other schedules and spring applied chemical fertilizer. In addition, these data show that increasing the rate of winter applied manure increased the concentration of these constituents in runoff

____

* Total Kjeldahl nitrogen (TKN) was not done routinely but results from selected spring runoff samples from winter-spread plots suggested the concentration of TKN was approximately twice the concentration of (NO_3 + NH_4)-N.

TABLE 2. Average N, P, And K Concentrations In Surface Runoff During Five Springtime Flow Events From Plots Receiving Three Rates Of Manure At Four Scheduled Times Of Application.

Nutrient	Source	Application rate, kg/(ha•yr) of N	Time of application: Fall	Winter	Spring	Spr.-Fall
(NO_3 + NH_4) -N	Manure	224	ND*	6.7	3.5	2.2
mg/L	Manure	560	3.9‡	11.6	3.0	4.8 ‡
	Manure	897	2.0§	23.3	4.7	4.8 ‡
	None	0			2.4†	
	Fertilizer	134			2.1†	
PO_4 -P	Manure	224	ND	2.70	0.54	0.10
mg/L	Manure	560	0.78‡	4.11	0.32	1.13 ‡
	Manure	897	0.36§	10.68	0.80	0.77 ‡
	None	0			0.09†	
	Fertilizer	134			0.08†	
K	Manure	224	ND	7.2	2.6	1.4
mg/L	Manure	560	3.5‡	14.3	2.7	3.4 ‡
	Manure	897	3.5§	30.2	6.5	5.7 ‡
	None	0			1.1†	
	Fertilizer	134			1.3†	

* no data † Four-year average ‡ Three-year average § Two-year average

TABLE 3. Average N, P, And K Concentrations In Water Portions Of Surface Runoff From Plots Receiving Three Rates Of Manure At Four Scheduled Times Of Application During Two June Storms.

Nutrient	Source	Application rate, kg/(ha•yr) of N	Time of application			
			Fall	Winter	Spring	Spr.-Fall
$(NO_3 + NH_4)$ -N mg/L	Manure	224	1.1	1.9	2.0	1.8
	Manure	560	1.0	1.1	5.1	2.3
	Manure	897	1.3	1.4*	6.5	4.1
	None	0			0.9	
	Fertilizer	134			2.2	
PO_4-P, mg/L	Manure	224	0.18	0.24	0.26	0.33
	Manure	560	0.35	0.51	0.70	0.61
	Manure	897	0.52	1.20*	1.48	1.95
	None	0			0.12	
	Fertilizer	134			0.39	
K, mg/L	Manure	224	2.6	3.5	2.7	3.1
	Manure	560	3.8	3.2	8.7	4.3
	Manure	897	7.0	4.2*	23.4	16.3
	None	0			1.7	
	Fertilizer	134			6.1	

* One storm

approximately in direct proportion to the rate applied. Schulte *et al.* (1979) in a 3-yr comparative study of winter, spring and fall-applied swine manure, also observed higher concentrations of inorganic N and orthophosphate from non-incorporated winter applications as well as increased concentration with increased rate. The results of Klausner et al. (1976) on runoff losses from winter-applied dairy cattle manure, when converted from unit area losses to concentration units, showed similar increased values of N and P with increased manure application rates except for N at the highest application rate of 200 t/ha.

The random nature of the N, P and K concentrations in the spring surface runoff resulting from the other non-winter treatments suggests that the effects of such treatments did not differ although the chemical fertilizer treatment tended to yield the lowest nutrient concentrations, particularly the P concentration. Schulte *et al.* (1979) obtained a similar trend but higher concentrations for P, possibly because manure was incorporated by disking only, in contrast to this study in which the manure and fertilizer were incorporated in the soil by plowing and disking.

In June, 1975 and 1976, storms caused surface runoff but the amount, typically 4 mm in 1976, was small compared to spring runoff. Treatments did not affect the runoff sediment concentrations which averaged about 10 g/L, nor did they affect total N and total P contents of the sediments, which were typically 0.8 and 0.3 percent respectively. Therefore, sediment-associated N and P concentrations in the runoff were about 80 mg/L and about 30 mg/L respectively, regardless of treatment. The fact that manure for all treatments was well incorporated into the soil prior to runoff could explain this lack of treatment differences. The presence of sediments in this runoff can be attributed to lack of protective crop vegetation and the experimental facilities which reflect conditions comparable to those where zero distance exists between the cropped area and the body of water receiving the surface runoff.

Analyses of the water portion of June-storm surface runoff showed concentrations of $(NO_3 + NH_4)$-N, PO_4-P and K increased from two to six times (Table 3) with increased application rates of 560 and 897 kg/(ha•yr) of N for spring and spring-fall applied manure. It does not seem surprising that runoff from these particular plots was enriched in N, P and K, as high amounts of manure were applied just weeks previously, as compared to more than 5 months previously for the other plots. However, this difference apparently had no influence on the erosion nutrient losses. The effect of rate on concentrations of PO_4-P was evident also on the fall and winter plots, but was not apparent for $(NO_3 + NH_4)$-N or K. The chemical fertilizer treatment concentrations were comparable to the overall mean concentrations of 2.3 mg/L for $(NO_3 + NH_4)$-N, 0.63 mg/L for PO_4-P, and 6.5 mg/L for K. It should be noted that these dissolved inorganic N and P concentrations in June storm runoff were small (<3 percent) compared to the sediment associated N and P.

In summarizing the results so far, winter spreading affected the quality of snowmelt runoff and spring manure application influenced runoff quality from June storms. In both types of runoff the level of dissolved nutrients were roughly in proportion to the rates applied.

Subsurface water: No NH_4-N was detected in drain effluents except for low concentrations (<0.5 mg/L) that resulted from winter applications at the two highest rates.

Treatment effects on concentrations of NO_3-N, PO_4-P and K (Table 4) for the three springtime tile drain flow events were slightly different from those for the surface flows.

Although time of application had little influence on the average NO_3-N concentrations, they did increase with increased application rates for all four application schedules. This was not unexpected; similar increased concentrations have occurred with increased nitrogen applications from chemical fertilizer (Gast *et al.*, 1978). However, even at the 897 kg/(ha•yr) of N rate of application, concentrations of NO_3-N were comparable to those under the chemical fertilizer treatment.

A comparison between the N concentrations of springtime subsurface drainage and surface runoff resulting from the high application rates, shows that the drain NO_3-N concentrations were 4 to 5 times higher than the surface $(NO_3 + NH_4)$-N concentrations except for the winter schedule treatment; however, the N in surface runoff for the winter treatment consisted largely of NH_4-

TABLE 4. Average N, P, And K Concentrations In Tile Drain Effluent During Three Springtime Flow Events From Plots Receiving Three Rates Of Manure At Four Scheduled Times Of Application.

Nutrient	Source	Application rate, kg/(ha•yr) of N	Time of application			
			Fall	Winter	Spring	Spr.-Fall
NO_3-N mg/L	Manure	224	11.0	12.3*	6.9	5.1
	Manure	560	12.1	14.0	12.8	19.4
	Manure	897	25.3	16.8	20.2	20.7
	None	0			8.4	
	Fertilizer	134			18.4	
PO_4-P, mg/L	Manure	224	0.03	0.03*	0.02	0.02
	Manure	560	0.04	0.13	0.17	0.06
	Manure	897	0.05	0.11	0.04	0.04
	None	0			0.01	
	Fertilizer	134			0.01	
K, mg/L	Manure	224	0.8	0.3*	0.7	0.8
	Manure	560	0.9	0.7	0.8	0.8
	Manure	897	1.0	1.2	0.8	1.0
	None	0			0.4	
	Fertilizer	134			0.3	

* Two-year average

N. Results from 50 measurements summarized by Baker and Johnson (1977) indicate that, in general, concentrations of NO_3-N in subsurface flow from tile drains are higher than concentrations in surface runoff.

The effect of high manure applications on PO_4-P concentrations in drain effluents is more pronounced than the NO_3-N results when compared to the control plot values. At the 560 and 897 kg/(ha•yr) of N rates of manure application (i.e. about 140 and 220 kg/(ha•yr) of P respectively), concentrations tend to be higher than those for the low manure application rate and the chemical fertilizer treatments, particularly for the winter application schedule. Both the no-fertilizer and the chemical fertilizer treatments yielded a similar concentration of PO_4-P which was not greatly lower than the concentrations resulting from the spring and spring-fall plots manured at the rate of 224 kg/(ha•yr) of N. The PO_4-P concentrations in all drain effluents were about one order of magnitude lower than those from surface runoff for comparable treatments.

Tofflemire and Chen (1976) measured 5-day PO_4-P absorption rates and capacities of A horizons ranging from 220 kg/ha of P for sand and gravel outwash materials to about 980 kg/ha of P for medium to coarse textured surface layers. Absorption rates and capacities of B horizons were generally higher than those of A horizons, and long term absorption rates were estimated to be at least twice the 5-day rates. It is reasonable to assume that PO_4-P saturation of the soil above the tile drains did not occur over the first 6 years of this study, if one considers the application rates given in Table 1. Bielby *et al.* (1973) observed considerable leaching of poultry liquid manure when applied to frozen, snow covered sandy loam. They speculated that thawing of the soil may have caused channeling thus resulting in higher PO_4-P in subsurface runoff from winter spread plots as compared to other application times.

Drainage water concentrations of K from manure applied plot were generally about twice the control plot values and were little affected by either rate or time of manure application.

During the study period, there were two subsurface runoff events induced by late fall rains. Runoff volumes averaged about 1 cm for each event. The average concentrations of PO_4-P and K (Table 5) were similar to those in snow-melt induced subsurface runoff (Table 4). However, the NO,-N concentrations were noticeably higher for most plots. Also the NO,-N concentrations,

TABLE 5. Average N, P, And K Concentrations In Subsurface Runoff From Plots Receiving Three Rates Of Manure At Four Scheduled Times Of Application During Two Late Fall Rains.

Nutrient	Source	Application rate, kg/(ha•yr) of N	Time of application			
			Fall	Winter	Spring	Spr.-Fall
NO_3-N mg/L	Manure	224	23.2	23.9*	17.3	15.8
	Manure	560	25.0	24.5	9.8*	30.0
	Manure	897	33.8	38.2	32.7*	44.4
	None	0			11.4	
	Fertilizer	134			30.7	
PO_4-P, mg/L	Manure	224	0.02	0.01*	0.01	0.02
	Manure	560	0.03	0.11	0.23*	0.03
	Manure	897	0.06	0.03	0.02*	0.02
	None	0			0.01	
	Fertilizer	134			0.01	
K, mg/L	Manure	224	0.8	0.7*	0.7	0.7
	Manure	560	0.9	0.7	0.4*	0.8
	Manure	897	1.2	1.0	0.7*	1.1
	None	0			0.6	
	Fertilizer	134			0.4	

* One event

TABLE 6. Six-year Average (1974-1979) Silage Corn Yields From Plots Receiving Three Rates Of Manure At Four Scheduled Times Of Application.

Source	Application rate, kg/(ha•yr) of N	Time of application			
		Fall	Winter	Spring	Spr.-Fall
		- - - - - - t/ha - - - - - - -			
Manure	224	11.68	11.13	11.82	11.16
Manure	560	10.59	11.65	11.92	11.69
Manure	897	12.72	11.70	10.87	12.00
None	0			8.60	
Fertilizer	134			11.39	

Standard error of the mean = 0.32

although generally higher than in the snow-melt induced drainage, show little difference between manure and chemical fertilizer treatments.

Silage Corn Yields

Since treatments were not replicated within years, plot uniformity was assessed by measuring corn yields in 1973 prior to the first application of manure or fertilizer. Yields averaged 10.01 t/ha from plots 1 to 5 with a coefficient of variability (c.v.) of 4 percent, 9.05 t/ha from plots 6 to 9 with a c.v. of 3 percent, and 8.62 t/ha from plots 10 to 14 with a c.v. of 6 percent. Thus, a significantly (Tukey's W test at 5 percent) decreasing productivity from south to north was apparent.

Corn yields, averaged over the 6 yr of treatment, were not significantly increased with increased manure application rate, nor were they affected by time of application (Table 6). Similarly, average annual yields from all manured plots (11.58 t/ha) and those from the chemically fertilized plot (11.39 t/ha) were not significantly different. However, the yield from the no-fertilizer treatment was significantly lower than those from all other treatments. These results, which indicate no improvement in corn yields at the higher rates of 560 and 897 kg/(ha•yr) of N, are consistent with those results from a 2-yr study in Indiana (Sutton *et al.,* 1978) where no yield response was observed from swine liquid manure applied in the spring at N rates above 250 kg/(ha•yr).

With spring application scheduling, consistently lower values for yield have been observed each year at the highest manure rate compared with the 560 kg/(ha•yr) rate of N. Elsewhere, high springtime application rates of manure have resulted in decreased silage corn yields possibly as a result of increased soil salinity during the growing season (Haghiri *et al.*, 1978, Shortall and Liebhardt 1975, Mathers and Stewart 1974).

CONCLUSIONS

On the basis of six years of study, and under these soil conditions, snow-melt induced runoff contributes the bulk of the total annual runoff. Winter-applied manure resulted in considerably higher concentrations of nitrogen, phosphorus and potassium in runoff than spring, fall and spring-fall applications. The higher the rate of winter application, the higher the concentrations

but this relationship was not as apparent for the non-winter application schedules. Therefore, winter applications of manure on areas that contribute snow-melt runoff directly to bodies of surface water should not be recommended.

Limited data from rain storm-induced surface runoff indicate that spring plow-down of manure at rates of 560 and 897 kg/(ha•yr) of N leads to greater concentrations of inorganic N and K in the water portion (excluding sediment) of the runoff compared to spring plow-down of manure at 224 kg/(ha•yr) of N or chemical fertilizer application at the rate of 134 kg/(ha•yr) of N.

Non-winter applications of manure at rates above 224 kg/(ha•yr) of N have not affected corn yields, but the tendency toward increased soluble orthophosphorus concentrations in tile drain effluent for manure rates at and above 560 kg/(ha•yr) of N (about 140 kg/(ha•yr) of P) compared to the chemical fertilizer application of 49 kg/(ha•yr) of P suggest that these rates of manure P could lead to water quality impairment.

References

1 Baker, J. L. and H. P. Johnson. 1977. Impact of subsurface drainage on water quality. p. 91-98. In: Proceedings of the Third National Drainage Symposium, ASAE Publ. 1-77.

2 Bielby, D. G., D. A. Tel, and L. R. Webber. 1973. Phosphorus in percolates from manured lysimeters. Can. J. Soil Sci. 53:343-346.

3 Gast, R. G., W. W. Nelson, and G. W. Randall. 1978. Nitrate accumulation in soils and loss in tile drainage following nitrogen application to continuous corn. J. Environ. Qual. 7:258-261.

4 Haghiri, R., R. H. Miller, and T. J. Logan. 1978. Crop response and quality of leachate as affected by land application of beef cattle waste. J. Environ. Qual. 7:406-412.

5 Keng, J. and P. M. A. Menage. 1970. Automated determination of ammonium and nitrate in soil extracts by distillation. Analyst 95:379-382.

6 Klausner, S. D., P. J. Zwerman, and D. F. Ellis. 1976. Nitrogen and phosphorus losses from winter disposal of dairy manure. J. Environ. Qual. 5:47-49.

7 Mathers, A. C. and B. A. Stewart. 1974. Corn silage yield and soil chemical properties as affected by cattle feedlot manure. J. Environ. Qual. 3:143-147.

8 Quin, J. R. J. G. Boisvert, and I. Wood. 1974. Semi-automated ninhydrin assay of Kjeldahl nitrogen. Anal. Biochem. 58:609-614.

9 Sawyer, C. N. and P. L. McCarty. 1967. Chemistry for sanitary engineers. McGraw Hill, New York. p. 429.

10 Schulte, D. D., M. B. Tokarz, and G. E. Laliberte. 1979. Nitrogen and phosphorus transport from swine-manured fields during snowmelt runoff events in the Red River Valley. ASAE Paper No. 79-2116 ASAE, St. Joseph, MI 49085.

11 Shortall, J. G. and W. C. Liebhardt. 1975. Yield and growth of corn as affected by poultry manure. J. Environ. Qual. 4:186-191.

12 Sowden, F. R. 1972. Effects of silicon on automated methods for the determination of phosphate in water. Can. J. Soil Sci. 52:237-243.

13 Sutton, A. L., D. W. Nelson, V. B. Mayrose and J. C. Nye. 1978. Effects of liquid swine waste applications on corn yield and soil chemical composition. J. Environ. Qual. 7:325-333.

14 Tofflemire, T. J. and M. Chen. 1976. Phosphate removal by sands and soils. p. 151-170. In: Land as a waste management alternative. Proceedings of the 1976 Cornell Agricultural Waste Management Conference.

15 Zwerman, P. J., S. D. Klausner, and D. Ellis. 1974. Land disposal parameters for dairy manure. p. 211-221. In: Processing and management of agricultural wastes. Proceedings of the 1974 Cornell Agricultural Waste Management Conference.

Attachment 42:

Manure management and nutrient loss under winter conditions: A literature review
Srinivasan et al. 2006

R2012-023
S James Electronic Filing - Received, Clerk's Office, 10/16/2012

Manure management and nutrient loss under winter conditions: A literature review

M.S. Srinivasan, R.B. Bryant, M.P. Callahan, and J.L. Weld

ABSTRACT: Excessive losses of nitrogen (N) and phosphorus (P) from agricultural fields have detrimental impacts on environmental quality. Nutrient management guidelines, such as the P index, are designed to minimize the risk of nutrient loss with minimal disruption to the whole farm operation. Restricting winter spreading of manure, which is common to most management guidelines developed for cold climates, is a contentious issue in the northern-tier states of the United States and almost all provinces of Canada. Producers have strong opinions with regard to the merits of winter spreading and arguments against the alternative practice of manure storage. The purpose of this paper is to review the results of scientific studies relevant to the issue of winter spreading of manure, and identify needs for additional research in this area. Collectively, these studies illustrate the complexity of N and P dynamics in response to a wide spectrum of winter conditions. They do shed some light on the potential for nutrient loss following manure application during winter with respect to cropping system effects on runoff, manure mulching effects, manure properties, and differences due to manure placement relative to a snow pack and timing of application. However, process-level understanding of nutrient loss following manure application during winter is still lacking, and critical variables that control hydrologic and transport processes under winter conditions are not fully identified or understood. Extensive watershed-scale observations in combination with plot and field scale experiments that focus on specific processes should yield sufficient knowledge and data to develop empirical models, a useful first step in developing more detailed understanding of nutrient losses associated with manure spreading under winter conditions.

Keywords: Frozen soil, manure, nitrogen, nutrient loss, phosphorus, snow, winter

Manure management associated with livestock operations has received much attention because land application of excess manure is viewed as a method of disposal (Shuyler and Meek, 1989). Khaled et al. (1980) indicated that manure application rates for crop utilization are usually low compared to those for disposal purposes. Although they are essential for plant and animal production, phosphorus (P) in fresh waters and nitrogen (N) in saline coastal waters lead to eutrophication, which has become one of the most ubiquitous water quality impairments in the United States and other parts of the world. Losses of P and N from agricultural fields have been identified as a major source of these elements in water bodies. Many states have adopted nutrient management guidelines, such as the phosphorus index, that are based on the best available knowledge of soil-nutrient-hydrology interactions that affect nutrient losses in runoff to surface water bodies or movement to ground waters.

Restrictions placed on winter spreading of manure, which are common to most management guidelines developed for cold climates, potentially affect farm operations in many of the northern-tier states of the United States and almost all provinces of Canada. In the northeast and north-central United States, approximately 2.5 cm (1 in) of snow is recorded on the ground for 100 to 140 days during the winter months (USDA, 1941), and approximately 45 percent of the United States experiences freezing weather conditions during winter (Formanek et al., 1990). Restricting winter spreading is a contentious issue in these areas, because producers have strong opinions with regard to the merits of winter spreading and arguments against the alternative practice of manure storage. Restrictions on winter spreading are based more on commonly held perceptions than on research, because studies of soil and manure interactions and the hydrological processes that affect nutrient transport under winter conditions are limited and the results of observational studies are mixed. The purpose of this paper is to review the results of scientific studies relevant to the issue of winter spreading of manure, such as the effects of winter conditions on infiltration and nutrient transport via runoff and erosion.

Rationale for winter manure spreading. Despite the perceived soil and water quality concerns and unfavorable weather conditions for operating equipment and working outdoors, winter manure spreading is widely practiced. Literature indicates the following reasons for this practice (Fleming and Fraser, 2000):

1. *No need for manure storage structures.* Manure storage structures are not popular among producers despite many public cost-sharing programs. Manure storage structures need periodic maintenance. Improperly or poorly maintained manure storage structures can become point sources of pollution.

2. *More time available for manure spreading.* Fewer on-field activities occur during winter months than the growing season, allowing producers more time to apply manure. Crop production activities during the growing season may allow very little time for producers to spread both stored and fresh manure.

3. *Reduced soil compaction.* Manure application on frozen ground during winter season results in less soil compaction.

Thus, for economical and practical reasons, many producers still practice winter manure spreading. The practice can be optimized through judicious decisions about timing and rate of manure spreading, and where the application occurs on the landscape (Kongoli and Bland, 2002a; Kongoli and Bland, 200

M.S. Srinivasan is associated with AgResearch Limited in Mosgiel, New Zealand. **Ray Bryant** is affiliated with the U.S. Department of Agriculture (USDA) Agricultural Research Service in the Pasture Systems and Watershed Management Research Unit in University Park, Pennsylvania. **Michael Callahan** is affiliated with the DelVal Soil and Environmental Consultants Incorporated in Doylestown, Pennsylvania. **Jennifer L. Weld** is associated with the USDA Natural Resources Conservation Service in Pottsville, Pennsylvania.

R2012-023
S James Electronic Filing - Received, Clerk's Office, 10/16/2012

Manure spreading can be combined with recommended best management practices (BMPs) to reduce the risk of nutrient loss. However, the remaining level of risk in comparison to winter storage and alternative spreading options is still the subject of debate. Manure stored in winter is generally applied in early spring. Bubenzer and Converse (1975) observed that the relative effect of such early spring application on soil and surface water quality compared to intermittent winter spreading is not known.

Current guidelines for winter manure management. For lack of sufficient scientific data, current winter manure spreading guidelines largely rely on the common sense of the applicator (Fleming and Fraser, 2000). Very early on, field studies recognized that nutrient export from winter-applied manure can be a severe nonpoint source problem to water bodies (Milne, 1976). In a field study in Wisconsin, Hensler et al. (1971) observed lower crop yields when manure was applied throughout the winter than in the spring. The US Department of Agriculture Natural Resources Conservation Service's Nutrient Management Standard 590 requires conservation measures when manure is applied to frozen soils with slopes greater than nine percent (Madison et al., 2003). Many states have formulated additional standards for winter manure management.

Table 1 presents the current winter manure management guidelines for many states in the United States. Whereas almost all states treat manure as a nutrient source, Maine treats manure as a nutrient source as well as a waste (refer to Table 1). It is unclear how manure management varies with this perspective. However, whether it is treated as a nutrient source or as a waste, manure spreading is restricted under winter conditions in that state. Also, many of the guidelines do not distinguish between manure form—solid or liquid. In addition to Nutrient Management Standard 590, the majority of states have drawn upon state regulations for manure management. These state regulations may be reflective of local management activities. Winter manure management guidelines listed in Table 1 can be summarized as follows:

- Avoid manure spreading on areas that have "high risk" for runoff,
- Avoid manure spreading on steep slopes, and
- Avoid manure spreading on fields adjacent to water bodies.

It is evident that the manure management guidelines listed in Table 1 are very similar to management practices recommended for non-winter periods. Implementation of these guidelines in the field largely rely on the common sense of the applicator, as there may not be tools or maps available to locate areas suitable for spreading. Thus, these guidelines are very good starting points, but additional research-based tools are needed for effective implementation.

In Canada, nutrient management guidelines advise application of manure on watershed areas that have a lower probability of generating runoff, and these areas are termed as "safe" areas (Fleming and Fraser, 2000). Another form of restriction is to prohibit manure spreading anywhere on the landscape during specific periods of winter as is practiced in some Canadian provinces. In Manitoba, large-scale (greater than 400 animal units) livestock operations cannot spread manure from November 10 to April 15 (Fleming and Fraser, 2000). There is no restriction on small-scale units. In Quebec, no manure spreading is allowed between October 1 and March 31 (Fleming and Fraser, 2000). More severe winter conditions in Canada may have dictated these time-restricted guidelines. Such time-restricted guidelines exist only in Maine and Vermont in the United States (refer to Table 1).

Hydrologic processes under winter conditions. *Infiltration.* Winter periods in the northern United States are characterized by alternate freezing and thawing, which are known to affect soil structure and infiltration. Freeze-thaw cycles disrupt soil aggregates (Bullock et al., 1988) and accelerate soil crusting, resulting in decreased infiltration and erosion resistance (Zuzel and Pikul, 1990). The potential for nutrient transport, either in soluble form in runoff, adsorbed to soil particles, or as a component of manure particles, will vary in accordance with these processes.

Depending on the presence of soil organic matter and soil moisture content at the time of freezing, the Yearbook of Agriculture for 1955 identifies four types of frozen soil structure: concrete, honeycomb, stalactite, and granular (Storey, 1955). Presence of even a small layer of concrete structure can drastically decrease soil infiltration rate, whereas the presence of other frozen soil structures, such as honeycomb, will have little or no effect on infiltration, even across large areas (Storey, 1955). Many factors can affect the existence and extent of formation of frozen soils. Since honeycomb and stalactite frost are most prevalent in meadows and pastures, and granular is indicative of forest soils, any alteration of cultivated soils that increases their similarity to the conditions present in either of these two types of land use would decrease the occurrence of concrete frost. This includes increased organic matter (humus), crop stubble, and manure applications (Storey, 1955).

Subsequent studies confirm this early characterization of frozen soil and the effects on infiltration. Willis et al. (1961) observed that infiltration rate of frozen soil decreases with increasing soil water content at the time of freezing. Studies by Lee and Molnau (1982) supported this observation. They concluded that there is a strong inverse relationship between the soil moisture content at the time of freezing and the final infiltration rate. Steenhuis et al. (1981) noted that not all frozen soils are impermeable, and the permeability of frozen soils varies with temperature and extent of pores blocked by ice. Zuzel and Pikul (1987) indicated that soils frozen under low moisture conditions may become granulated and provide little impediment to infiltration, whereas soils frozen under high moisture contents often freeze into massive, dense, concrete-like structures that are nearly impermeable to water.

Infiltration of snowmelt is also dependent on soil conditions at the time of melt. While frozen soil may lead to runoff of snowmelt, unfrozen soil may allow infiltration depending on soil moisture conditions. Gutting et al. (1998a) reported the existence of saturated soil conditions under thawing conditions, resulting in very little infiltration of snowmelt. Studies have shown that on recently thawed fine-textured soils, very little of the snowmelt infiltrates (e.g., Baker, 1972).

Studies have also recognized the mulching effect of manure under winter conditions and its effect on moderating soil temperatures. The timing of manure application can affect the soil freezing process itself. For instance, manure applied during the fall season and left on the plowed surface moderated the soil temperature extremes over early winter (Young and Mutchler, 1976). On fall-plowed fields, manure may work effectively as a mulch to control or reduce soil erosion and runoff from spring snowmelt, thereby contributing to the conservation of soil, enhancement of infiltration, and replenishment of soil moisture for crop use (Young and Mutchler,

Table 1. Current winter manure management guidelines in the United States.

Region	Source	Nutrient application practice	Reference
National standard	590 standard[*]	When a risk of runoff exists, no nutrient application to snow covered, saturated, or frozen land is permitted.	USDA-NRCS, 2003a
		Manure application on frozen soils greater than 9 percent slope must include conservation practices.	Madison et al. (2003)
Alabama	590 standard[*]	No nutrient application to land that is at field capacity, snow covered, frozen, or does not have a growing crop in the fall and winter months.	USDA-NRCS, 2002a
Alaska	590 standard[*]	When a risk of runoff exists, no nutrient application to snow covered, saturated, or frozen land is permitted.	USDA-NRCS, 2001a
Colorado	590 standard[*]	When a risk of runoff exists, no nutrient application to snow covered, saturated, or frozen land is permitted.	USDA-NRCS, 2004a
Connecticut	590 standard[*]	No nutrient application when the following conditions may occur: saturated, or frozen, snow-covered land or flooding as determined by weather, field conditions or when application occurs between November and March.	USDA-NRCS 2002b
Delaware	590 standard[*]	No nutrient application during snow-covered, frozen, or saturated conditions.	USDA-NRCS 2002c
Idaho	590 standard[*]	Winter application of solid manure is permitted on land with 0 to 2% slopes and with no potential for runoff.	USDA-NRCS, 2004b
Illinois	590 standard[*]	When a risk of runoff exists, no nutrient application to snow covered, saturated, or frozen land.	USDA-NRCS, 2002d
	State regulations	State regulations allow for application on frozen or snow covered land with less than a 5% slope and controlled erosion.	Illinois Dept. of Agriculture, 2001
Indiana	590 standard[*]	When a risk of runoff or loss from inlet tile flow exists, no nutrient application to snow covered, saturated, or frozen land is permitted.	USDA-NRCS, 2001b
Iowa	590 standard[*]	No nutrient application to snow covered, saturated, or frozen land with greater than 5% slope and greater than tolerable soil loss.	USDA-NRCS, 2001c
Kansas	590 standard[*]	When a risk of runoff exists, no nutrient application to snow covered, saturated, or frozen land is permitted.	USDA-NRCS, 2004c
Kentucky	590 standard[*]	No nutrient application to snow covered, saturated, or frozen land with the following exceptions: mineral fertilizer within 30 days of crop and solid manure application with 75 ft. setbacks from streams, sinkholes, and sensitive areas.	USDA-NRCS, 2001d
Louisiana	590 standard[*]	When a risk of runoff exists, no nutrient application to snow covered, saturated, or frozen land is permitted.	USDA-NRCS, 2003b
Maine	590 standard 633 standard[†] State regulations	No nutrient application on snow covered, ice covered, or frozen ground: between December 1 and March 15, and on saturated ground when potential for runoff exists.	USDA-NRCS, 2001e USDA-NRCS, 2002e Dept. of Agriculture, Food, and Rural Resources, 2001
Maryland	590 standard[*]	When a risk of runoff exists, no nutrient application to snow covered, saturated, or frozen land is permitted.	USDA-NRCS, 2001f
	State regulations	State regulations allow frozen or snow covered ground application of manure when no other management option exists with the following restrictions: minimized application rates, ground must have vegetative cover, slope is 7% or less, erosion loss is tolerable, and 100 ft. setback from all surface waters.	Maryland Dept. of Agriculture, 1999
Massachusetts	590 standard[*]	When a risk of runoff exists, no nutrient application to snow covered, saturated, or frozen land is permitted. The Massachusetts Nutrient Transport Index determines runoff risk and fields rated Medium or High will be restricted.	USDA-NRCS, 2002f

1976). Kongoli and Bland (2002a) reported that when applied to frozen soil or early in spring before a snowfall, manure tends to increase the soil temperature, thereby promoting earlier thawing, increased infiltration, and reduced runoff and soil loss.

Kongoli and Bland (2002a) reported that, due to differences in surface albedo, manure applied on top of snow actually retarded the melting rate in proportion to the manure application rate. Depending on soil conditions, the slower melting rate is likely to increase the infiltration rate of the melt water (Young and Mutchler, 1976; Kongoli and Bland, 2002a; Kongoli and Bland, 2005). Also, manure applied on snow, because of its dark color, absorbed more sunlight and resulted in more moderate soil temperature

Table 1. Continued.

Region	Source	Nutrient application practice	Reference
Michigan	590 standard	When a risk of runoff exists, no nutrient application of phosphorus to snow covered, saturated, or frozen land or when restricted by Generally Accepted Agricultural and Management Practices for Manure Management and Utilization or recommendation of Michigan State University Extension is permitted. A Manure Application Risk Index evaluation must be conducted on all fields receiving winter applications of manure.	USDA-NRCS, 2002g
	State regulations	State regulations allow applications to frozen or snow covered ground when runoff and erosion are controlled with the following restrictions: solid manure must be applied to ground with less than 6% slope and liquid manure must be applied to ground with less than 3% slope.	Michigan Dept. of Agriculture, 2004
Minnesota	590 standard	Manure may be applied to frozen or snow covered land with the following restrictions: 300 ft. setback from all surface waters, solid manure may only be applied to land with 4 T/A or less soil loss, and liquid manure may only be applied to land with 2 T/A or less soil loss.	USDA-NRCS, 2001g
Missouri	590 standard	When a risk of runoff or flooding exists, no nutrient application to snow covered, saturated, or frozen land is permitted.	USDA-NRCS, 2001h
Montana	590 standard	When a risk of runoff exists, no nutrient application to snow covered, saturated, or frozen land is permitted. Runoff risk is determined by RUSLE (soil loss calculations). When slope is less than 2% or RUSLE determined soil loss is less than 5 T/A, runoff risk is considered negligible.	USDA-NRCS, 2004d
Nebraska	590 standard	When a risk of runoff exists, no nutrient application to snow covered, saturated, or frozen land is permitted.	USDA-NRCS, 2000
Nevada	590 standard	Application to snow covered, saturated, or frozen land is not permitted unless: risk of runoff is low or very low, application rates meet specified threshold or crop utilization levels, liquid applications do not exceed 20% of the available water holding capacity and appropriate set backs are followed.	USDA-NRCS, 2002h
New Hampshire	590 standard	When a risk of runoff exists, no nutrient application to snow covered, saturated, or frozen land is permitted.	USDA-NRCS, 2001i
New Jersey	590 standard	When a risk of runoff to sensitive areas (such as wellheads, surface waters, or sinkholes) exists, no nutrient application to snow covered, saturated, or frozen land is permitted.	USDA-NRCS, 2004e
New York	590 standard	When a risk of runoff exists (as determined by the Phosphorus index), no nutrient application to snow covered, saturated, or frozen land is permitted.	USDA-NRCS, 2003c
North Dakota	590 standard	When a risk of runoff exists, no nutrient application to snow covered, saturated, or frozen land is permitted.	USDA-NRCS, 2002i
Ohio	590 standard	Inorganic fertilizer applications are permitted on ground with at least 50% residue cover or a growing and established crop.	USDA-NRCS, 2003d
	633 standard	Manure applications are permitted with the following restrictions: the ground must have 90% residue cover, 200 ft. minimum setback from grassed waterways, surface drainage ditches, streams, surface inlets, and other water bodies, maximum liquid manure rate is 5000 gal/A, maximum solid manure rate is 10 T/A (manure < 50% moisture) and 5 T/A (manure > 50% moisture), and additional contour strip requirements for fields with a slope greater than 6%.	USDA-NRCS, 2003e
Oklahoma	590 standard	No nutrient application of nutrients to snow covered, saturated, or frozen land is permitted.	USDA-NRCS, 2004f
Oregon	590 standard	When a risk of runoff exists, no nutrient application to snow covered, saturated, or frozen land is permitted.	USDA-NRCS, 2001j

in spring and early thawing (Young and Mutchler, 1976).

Surface management practices such as plowing can also influence infiltration processes under winter conditions. Pikul et al. (1996) reported that forming tillage slots at the time of planting in late summer or early fall for winter wheat does not improve infiltration in winter as loose dry surface soil sloughs into the tillage channel. Under such conditions, Schillinger and Wilkins (1997) suggested that tillage after the soil freezes

Table 1. Continued.

Region	Source	Nutrient application practice	Reference
Pennsylvania	590 standard*	Fall and winter manure application are permitted when the ground has permanent vegetation, a cover crop, or 25 % residue cover.	USDA-NRCS, 2001k
	State regulations	Application of manure is permitted in frozen, snow covered and saturated conditions with the following restrictions: 100 ft. setback (slope less than 8%) and 200 ft. setback (slopes greater than 8%) from streams, springs, lakes, ponds intakes to agricultural drainage systems, and other surface conveyances; and no nutrient application is permitted in vegetated concentrated flow areas.	Pennsylvania Code, 1997
Rhode Island	590 standard*	When a Medium or High transport potential exists (as determined by the Phosphorus Index), no nutrient application is permitted to snow covered, saturated, or frozen land is permitted.	USDA-NRCS, 2001l
South Carolina	590 standard*	When a risk of runoff exists, no nutrient application to snow covered, saturated, or frozen land is permitted.	USDA-NRCS, 2001m
South Dakota	590 standard*	When a risk of runoff exists, no nutrient application to snow covered, saturated, or frozen land is permitted.	USDA-NRCS, 2004g
Tennessee	590 standard*	When a risk of runoff exists, no nutrient application to snow covered, saturated, or frozen land is permitted.	USDA-NRCS, 2003f
Utah	590 standard*	Application to snow covered, saturated, or frozen land is not permitted unless: risk of runoff is low or very low (as determined by the Utah Manure Application Risk Index), application rates meet specified threshold or crop utilization levels, liquid applications do not exceed 20% of the available water holding capacity and appropriate set backs are followed.	USDA-NRCS, 2004h
Vermont	590 standard* Accepted agricultural practices	No manure application is permitted on frozen or snow-covered ground and from December 1 to April 1.	USDA-NRCS, 2002j Vermont Agency of Agriculture, Food and Markets, 1995
Virginia	590 standard*	No manure application is permitted on frozen, snow or ice covered, or saturated ground with one exception. Solid manure may be applied when needed to land with less than 6% slope and 60% cover.	USDA-NRCS, 2001n
Washington	590 standard*	When a risk of runoff exists (as determined by the Phosphorus Index), no nutrient application to snow covered, saturated, or frozen land is permitted.	USDA-NRCS, 2003g
West Virginia	590 standard*	No nutrient application is permitted on frozen, snow covered or saturated ground is permitted.	USDA-NRCS, 2004i
Wisconsin	590 standard*	Manure application is permitted to frozen and snow covered ground with the following restrictions: no nutrient application within Surface Water Quality Management Area and 200 ft. setback from sinkholes, wells, fractured bedrock, and gravel pits.	USDA-NRCS, 2002k
Wyoming	590 standard†	When a risk of runoff exists, no nutrient application to snow covered, saturated, or frozen land is permitted.	USDA-NRCS, 2002l

* **USDA-NRCS 590 nutrient management practice standard.**

† **USDA-NRCS 633 waste utilization practice standard.**

might improve infiltration. However, energy requirements for tilling frozen soil may be extremely high, depending on the vertical extent of frost and type of freezing. However, Ulén (2003) in a study in Sweden concluded that plowed and unplowed fields did not show any difference in snowmelt times and rates.

Snowmelt-runoff and erosion. Our understanding of seasonal runoff generation processes is very limited. Runoff during winter periods largely comes from snowmelt, which could be very different from rainfall-runoff generation processes, such as saturation excess and infiltration excess. Winter runoff generation may fall in the category of infiltration excess when the ground is frozen and snowmelt cannot enter the soil. On the other hand, snowfall prior to soil freezing may prevent soil from freezing, leading to saturation-excess runoff conditions, wherein the snowmelt in excess of soil storage will be routed downslope. The third type of runoff generation that may occur during winter is by rainfall on snowpack. Often, warm air and incoming rainfall can quickly melt snowpacks and lead to large runoff events. Very little literature is available to assist in identifying the runoff-risk areas during winter. Though there is a perception that it is safe to apply manure on flat areas under winter conditions, according to Midgley and Dunklee (194 when the soil is frozen, runoff, and hence nutrient transport, may occur on any slope. The average losses may be greater on steep slopes (Midgley and Dunklee, 1945). McCool (1990) indicated that even the BMPs are often not sufficient to control runoff and soil erosion (and associated nutrient transport) during events of rain and/or melting snow on frozen ground.

Tigerman and Rosa (1951) observed that

frozen soils are strongly associated with increased runoff, soil erosion, and sediment production. From a basic process perspective, Storey (1955) described four characteristics of frozen soils that are important to snowmelt, runoff and the potential for flooding:

1. Structure of the frozen soil,
2. Depth of penetration of the frost,
3. Persistence of soil frost, and
4. Areal extent of frozen soil.

The structure of frozen soil determines the rate of movement of water into the soil. The depth and persistence of frozen soil determines how quickly it will thaw, and the areal extent can increase the size of the contributing runoff producing area (Storey, 1955). With respect to persistence of soil frost, Baker (1983) wrote that the rate at which snow melts (and frozen soil persists) is dependent on influx energy as affected by weather, vegetation, and topography.

How these basic processes combine to determine the risk of runoff, erosion, and nutrient transport is a function of the prevailing soil and climate conditions. Consequently, the results of observational studies and general conclusions in relation to snowmelt, runoff, and erosion based on these studies are sometimes conflicting and must be interpreted within the context of local soil and climate conditions. In cold climates, Young and Mutchler (1976) and Hansen et al. (2000) observed that snowmelt-runoff may exceed rainfall-runoff. Where this is the case, the risk of nutrient loss and transport to surface water bodies is especially sensitive to manure applications during the winter.

Tillage operations that increase surface roughness and thus surface storage, decrease runoff but not necessarily sediment loss (Schillinger and Wilkins, 1997; Ginting et al., 1998b; Hansen et al., 2000). Ginting et al. (1998b) observed that snowmelt runoff may not be as erosive as runoff caused by rain. Hansen et al. (2000) concluded the same and suggested that snowmelt runoff is not expected to cause substantial interrill erosion because snow normally melts gradually and because soil detachment is limited when the soil is frozen. Conversely, Schillinger and Wilkins (1997) state that soil loss in runoff can be high when snowmelt or rain occurs on thawed soil overlying a subsurface frozen layer. For example, in north-central Oregon, Zuzel et al. (1982) reported that 86 percent of soil erosion on winter wheat was caused by rapid snowmelt or rainfall on thawing soil.

Wischmeier and Smith (1978) indicated that as much as 90 percent of soil loss from the dry-farmed croplands of the Pacific Northwest is caused by surface thaws and snowmelt.

Klausner et al. (1971) suggested that if manure has to be applied during winter, it should be limited to the early periods before the first heavy snowfall, so that the snow/ice sheet covering the manure may reduce nutrient transport in runoff. They also suggested that if good manure spreading could not be achieved during winter because of snow accumulation or frozen ground, the manure should be stored under cover.

Nutrient transport from winter-applied manure. Studies that have looked at winter manure application and nutrient losses during snowmelt and thawing conditions differ in methods and scales (spatial and temporal), but the majority of these studies observed substantial nutrient losses (Converse et al., 1976; Klausner et al., 1976; Young and Mutchler, 1976; Philips et al., 1981; Lorimor and Melvin, 1996). Klausner et al. (1976) spread dairy manure at three different rates for three consecutive winters and, although wide variations in climatic sequences within a year or between years significantly influenced study results, they observed excessive nutrient losses when manure was spread during active thaw periods. Philips et al. (1981) made a similar observation with respect to manure application during snowmelt. Hensler et al. (1970) applied fresh dairy manure on frozen soils for two consecutive winters and found that the runoff losses were variable. During the first year of observation, more N and P losses were noted in runoff than occurred during the second year. They observed that the dry winter conditions during the second year resulted in minimal nutrient losses. On the other hand, Young and Mutchler (1976) found very little differences in thawing rates from manured versus unmanured alfalfa plots, and in nutrient losses from manured and unmanured corn plots.

Christensen and Tiedje (1990), Ryan et al. (2000), and Jacinthe et al. (2002) reported the presence of significant quantities of inorganic N in runoff during early spring, even when no manure was applied. Higher levels of N could be due to the death and lysis of organisms which release N during the thawing process, and snowmelt runoff during early spring may potentially carry the inorganic N to water bodies (Ryan et al., 2000). Also, increased infiltration during thawing periods may leach excess N to the ground water (Jacinthe et al., 2002).

Phosphorus transport losses during snowmelt periods are poorly understood and have not been investigated for most types of soils (Ulen, 2003). Winter application of manure is as much of a balancing act as manure application in any other season; the costs and benefits must be weighed. Although it is likely that dissolved P concentrations will increase in runoff waters, the amount of runoff volume will likely decrease because of the mulching effects of the manure (Converse et al., 1976; Young and Mutchler, 1976; Young and Holt, 1977). If no manure is applied, then dissolved P concentrations will be lowered, but the site will be more prone to sediment loss, therefore particulate P loss will increase (Young and Holt, 1977; Ginting et al., 1998a). However, sediment loss may be minimal due to the management conditions present at the site, such as no-till, a good alfalfa cover, or just a relatively flat field that has soil properties that do not favor sediment detachment and transport. Management practices, such as no-till, that leave the surface smooth and favor the accumulation of nutrients at the surface in the form of manure or decomposing crop residues pose greater threats of nutrient losses on frozen soils (Young and Mutchler, 1976; Kekolainen, 1989; Hansen et al., 2000). Sharpley et al. (1991) reported increases in soluble nutrient losses in rainfall-runoff under conservation tillage conditions, and the same appears to be true for winter conditions. Conservation tillage that involves reduced or no soil inversion may lead to accumulation of P on the surface (Ismail et al., 1994), resulting in large soluble P losses in snowmelt runoff (Hansen et al., 2000).

Ulen (2003), in a plot-scale study in Sweden, observed high concentrations of dissolved and particulate P during snowmelt, and the majority of particulate P loss was associated with colloidal clay particles. As grassed buffers may be ineffective in trapping the colloidal size particles, Ulen (2003) suggested that the loss of these particles may be controlled by improving the soil structure through practices such as liming. McDowell et al. (2001) found the proportion of coarse sediment was greater at the start of rainfall-runoff and decreased as the event proceeded. According to McDowell et al. (2001), as the erosion caused by the kinetic energy of rainfall and overland flow equilibrates over the rainfall-runoff period, the proportion of coarse sediment in runoff drops. It is unclear whether similar exhaustion occurs during gentler snowmelt (Ulen, 2003). Under extremely cold conditions where the soils remain frozen for long periods, snowmelt may occur when the bulk of the soil remains frozen. Yli-Halla et al. (1995) suggested that under such frozen soil conditions in Finland, dissolved as well as sediment P losses are minimal, but as the soil thaws in the spring, large concentrations of dissolved P were observed. Dissolved P losses peaked during wet winters (Yli-Halla et al., 1995).

Studies do not necessarily concur on the influence of manure on snowmelt and nutrient loss during winter. Blais and West (1999) applied liquid manure in the late fall on frozen ground and in the spring on unfrozen ground, and found no significant differences in N contamination between seasons both in surface runoff and subsurface drainage water. On the other hand, liquid manure can significantly accelerate melting of the snow pack and increase P loss in runoff (Braun, 1990, cited in Kongoli and Bland, 2002a). According to Kongoli and Bland (2002a), the lower the water content of the manure, the lower the risk of nutrient loss from winter application. Liquid manure presumably has a higher thermal conductivity than solid-bedded manure because of its higher water content and more finely divided solids, which prevents it from functioning as insulation (Kongoli and Bland, 2002a).

Relative placement of manure with respect to snow and timing of application relative to snowmelt may be important factors affecting the risk of nutrient loss, but studies are contradictory and not supportive of the development of BMPs. Young and Mutchler (1976) suggested that the manure applied in fall might stay below the melting snow layer and nutrients from this manure might travel along the soil surface, resulting in high concentrations in runoff. Conversely, when manure is on top of the snow, runoff water passes over bare soil beneath the snow, resulting in lower concentrations of nutrients in runoff. They therefore concluded that when manure is on top of snow there is less chance for nutrients to move to surface water than when snow is on top of manure. However, they also indicated that under heavy spring rainfall, snowmelt conditions resulted in both large runoff volumes and high nutrient concentrations.

Kongoli and Bland (2002a) proposed that the presence of manure on the soil surface checks runoff velocities and allows more time for infiltration, thereby decreasing the total volume of runoff and nutrient transport. Presumably, this effect would occur under a snow pack, if the underlying soil is not frozen. Similarly, surface applied manure tends to dry out, and when thawing begins, the dried manure retains some snowmelt (Kongoli and Bland, 2002a; Kongoli and Bland, 2002b).

Klausner et al. (1976) observed that when manure was applied on top of melting snow, nutrient losses were great, but manure applied at higher rates prior to snowmelt did not transport as much nutrients. Also, when manure was covered with snow and melting occurred at a later date, nutrient losses were not as great. Studies have shown that the first runoff event that follows manure application tends to transport more nutrients than subsequent events (Hensler et al., 1970; Qiu et al., 1996). The effect of time between manure application and snowmelt/runoff is similar to conclusions reached by Edwards and Daniel (1994), based on rainfall on manure. Van Vliet et al. (2002), from a field study in British Columbia, Canada, reported that total nutrient losses from fall-applied manure were always greater than the allowed water quality standards during the first three runoff events, irrespective of management practices adopted. Slower decay rates of manure during winter periods than other seasons because of reduced microbial activity, might result in larger availability of nutrients for longer periods than any other season.

Few studies are available that directly link winter manure management and water quality of water bodies, and even these studies present contradicting conclusions. Greathouse et al. (1971), based on a one-year study in Michigan, reported no differences in N and P concentrations as a result of winter pasturing of animals along water bodies. Milne (1976) studied N and P concentrations in a stream flowing through a ranch in Montana. Sheep, hogs, and cattle were reared along the stream and the stream recorded increases in N concentrations by three fold and ortho-P concentrations by four fold within the ranch. These nutrient contributions represented direct deposit of animal feces in and around the stream segment. However, Witzel et al. (1969) observed that nutrient losses from spring runoff from four small watersheds were essentially the same, though some of the watersheds received winter-spread manure while others did not. Meals (1996), from a watershed-scale monitoring study in St. Albans Bay Watershed, Vermont, reported that 13 percent of P and 17 percent of N from winter-applied manure were lost in runoff. Clausen (1987) observed five percent loss in P from winter-applied manure on cornfields in the LaPlatte River Watershed, Vermont. Variations in climatic conditions could have resulted in these varying observations in Vermont.

Summary and Conclusion

As Kongoli and Bland (2002a) rightly pointed out, most of the field studies on manure management under winter conditions predate 1980. Donue (1983) attributed the paucity of empirical studies on snowmelt to the difficult and uncomfortable weather conditions that prevail during winter. Based on a review of the literature, some key research and management issues that need to be addressed are:

1. Characterization of the changes in the physical and chemical properties of manure under winter conditions as they affect N and P release rates.

2. Development of strategies and methods that relate the findings of small-scale experiments to large-scale soil, landscape, and climate patterns.

3. Collection of sufficient data to establish the linkages among watershed-scale water quality, winter manure spreading practices, and winter conditions that affect hydrologic and erosion processes.

4. Development of empirical models of snowmelt and nutrient transport for use in evaluating current winter spreading practices and developing BMPs.

5. Development of alternate methods of manure application. For example, Klausner et al. (1971) suggested manure injection in shallow snow or on frozen soil to a depth of a few centimeters might be possible method of manure application under winter conditions.

Data collected at various scales would allow identification of critical variables that control hydrologic and transport processes under winter conditions. Extensive watershed-scale observations in combination with plot- and field-scale experiments that are focused on specific processes should yield sufficient knowledge and data to develop empirical models, which are useful first steps in developing more detailed understanding of manure management and associated nutrient [illegible] under winter conditions. The best [illegible] for progress may be to develop [illegible] specific models for application to [illegible] geographical areas having similar soil [illegible] landscape characteristics and winter [illegible] conditions that result in most proba- [illegible] scenarios for soil freezing, snow cover, and [illegible] conditions.

Acknowledgements

The authors would like to acknowledge the assistance provided by Dr. Peter Kleinman located with the U.S. Department of Agriculture Agricultural Research Service's Pasture Systems and Watershed Management Research Unit, in University Park, Pennsylvania and Dr. D.B. Beegle at The Pennsylvania State University in University Park, Pennsylvania for help in compiling and organizing the materials for this manuscript.

References Cited

[illegible] 1973. Prediction of spring runoff. Water Resources Research [illegible].
[illegible] and C. Wall. 1999. Nitrogen losses in surface and [illegible] water from spring and late fall liquid manure application on a [illegible] clay soil. Abstract of Technical Paper in Canadian Journal of Soil Science [illegible].
[illegible] C.D. and J.C. Converse. 1978. Impact of freezing and thawing soil conditions on the movement of [illegible] from agricultural lands. Technical report. Water Resources Center, University of Wisconsin, Madison. 44 pp.
[illegible] M.S., W.D. Kemper, and S.D. Nelson. 1988. Soil cohesion as affected by freezing, water content, time and tillage. Soil Science Society of America Journal 52:[illegible].
[illegible] S. and J.M. Tiedje. 1987. [illegible] and vigorous N2O production by soil at spring thaw. Journal of Soil Science [illegible].
[illegible] J.C. 1987. [illegible] field study. Pp. [illegible]. In LaPlatte River watershed water quality monitoring and analysis program – Year 8. D.W. Meals (ed.) Program Report No. 9. Vermont Water Resources Research Center, University of Vermont, Burlington, Vermont.
[illegible] J.C., G.D. Bubenzer, and W.H. Paulson. 1976. Nutrient losses in surface runoff from winter spread manure. Transactions of the ASAE 19:517-519.
[illegible] of Agriculture, Food and Rural Resources. [illegible]. Chapter 565 Nutrient Management Rules. Maine Nutrient Management Program, Augusta, Maine.
[illegible] T. 1983. Relations of field studies and modeling in the prediction of snowmelt runoff. Journal of Hydrology [illegible].
[illegible] D.R. and T.C. Daniel. 1994. Quality of runoff from fescue grass plots treated with poultry litter and inorganic fertilizer. Journal of Environmental Quality 23:579-584.
[illegible] R. and H. Fraser. 2000. Impacts of winter spreading of manure on water quality. Literature review. Ridgetown College, University of Guelph, Ridgetown, Ontario.
Fonstad[illegible] G.E., G.R. Mischel[illegible] and W.R. Fran[illegible]. 1990. Conservation applications impacted by soil freeze-thaw. Pp. 108-112. In Frozen soil impacts on agricultural, range, and forest lands. K.R. Cooley (ed.) Proceedings of International Symposium, Spokane, Washington. CRREL Special Report No. 90-1. Cold Regions Research and Engineering Laboratory, U.S. Army Corps of Engineers, Hanover, New Hampshire.
Ginting, D., J.F. Moncrief, S.C. Gupta, and S.D. Evans. 1998a. Corn yield, runoff, and sediment losses from manure and tillage systems. Journal of Environmental Quality 27:1396-1402.
Ginting, D., J.F. Moncrief, S.C. Gupta, and S.D. Evans. 1998b. Interaction between manure and tillage system on phosphorus uptake and runoff losses. Journal of Environmental Quality 27:1403-1410.
Grossbauer[illegible] T.R., W.T. Magee, N.R. Kevern, and B.C. Ball. 1971. Effect of wintering and pasturing cattle along rivers and streams on water quality and bank erosion. Report No. 141. Michigan Agricultural Experimentation Station, East Lansing, Michigan.
Hansen, N.C., S.C. Gupta, and J.F. Moncrief. 2000. Snowmelt runoff, sediment, and phosphorus losses under three different tillage systems. Soil and Tillage Research 57:93-100.
Hensler, R.F., R.J. Olsen, S.A. Witzel, O.J. Attoe, W.H. Paulson, and R.F. Johannes. 1970. Effect of method of manure handling on crop yields, nutrient recovery and runoff losses. Transactions of the ASAE 13:726-731.
Hensler, R.F., W.H. Erhardt, and L.M. Walsh. 1971. Effect of manure handling systems on plant nutrient cycling. Pp. 254-257. In Livestock waste management and pollution abatement. Proceedings of International Symposium on Livestock Wastes. The Ohio State University, Columbus, Ohio.
Illinois Department of Agriculture. 2005. Livestock management regulations. Livestock Management Facilities Program. Springfield, Illinois.
Ismail, I., R.L. Blevins, and W.W. Frye. 1994. Long-term no-tillage effects on soil properties and continuous corn yields. Soil Science Society of America Journal 58:193-198.
Jacinthe, P.A., W.A. Dick, and L.B. Owens. 2002. Overwinter soil denitrification activity and mineral nitrogen pools as affected by management practices. Biology and Fertility of Soils 36:1-9.
Khaleel, R., K.R. Reddy, and M.R. Overcash. 1980. Transport of potential pollutants in runoff water from land areas receiving animal wastes: A review. Water Research 14:421-436.
Klausner, S.D., P.J. Zwerman, and T.W. Scott. 1971. Land disposal of manure in relation to water quality. Pp. 36-46. In Agricultural wastes: Principles and guidelines for practical solutions. Cornell University Conference on Agricultural Waste Management, Syracuse, New York.
Klausner, S.D., P.J. Zwerman, and D.F. Ellis. 1976. Nitrogen and phosphorus losses from winter disposal of dairy manure. Journal of Environmental Quality 5:47-49.
Kongoli, C.E. and W.L. Bland. 2002a. Influence of manure application on surface energy and snow cover: Field experiments. Journal of Environmental Quality 31:1166-1173.
Kongoli, C.E. and W.L. Bland. 2002b. Influence of manure application on surface energy and snow cover: Model development and sensitivities. Journal of Environmental Quality 31:1174-1183.
Lee, R.W. and M.D. Marston[illegible]. 1982. Infiltration into frozen soils using simulated rainfall. Paper No. 82-2048. American Society of Agricultural Engineering, St. Joseph, Michigan.
Lorimor, J.C. and J.C. Melvin. 1996. Nitrogen losses in surface runoff from winter applied manure. Final Report. University of Iowa, Ames, Iowa.
Madison, F., K. Kelling, L. Massie, and L.W. Good. 2003. Guidelines for applying manure to cropland and pasture in Wisconsin. University of Wisconsin Cooperative Extension Publication No. A3392. University of Wisconsin, Madison, Wisconsin.
Maryland Department of Agriculture. 1999. Maryland manure management manual. Section 1. Nutrient recommendations. Maryland Department of Agriculture, Annapolis, Maryland.
Michigan Department of Agriculture. 2004. Generally accepted agricultural and management practices for manure management and utilization. Michigan Commission of Agriculture, Lansing, Michigan.
McCool, D.K. 1990. Crop management effects on runoff and soil loss from thawing soil. Pp. 171-176. In Frozen soil impacts on agricultural, range, and forest lands. K.R. Cooley (ed.) Proceedings of International Symposium, Spokane, Washington. CRREL Special Report 90-1. Cold Regions Research and Engineering Laboratory U.S. Army Corps of Engineers, Hanover, New Hampshire.
McDowell, R., A.N. Sharpley, and G. Folmar. 2001. Phosphorus export from an agricultural watershed: Linking source and transport mechanisms. Journal of Environmental Quality 30:1587-1595.
Meals, D.W. 1996. Watershed-scale response to agricultural diffuse pollution control programs in Vermont, USA. Water Science and Technology 33(4-5):197-204.
Midgley, A.R. and D.E. Dunklee. 1945. Fertility runoff losses from manure spread during the winter. University of Vermont, Agricultural Experiment Station, Bulletin 523:1-19.
Milne, C.M. 1976. Effect of a livestock wintering operation on a western mountain stream. Transactions of the ASAE 19:749-752.
Pennsylvania Code. 1997. Title 25, Pennsylvania Code, Section 83.294. State Conservation Commission, Harrisburg, Pennsylvania.
Phillips, P.A., J.L.B. Culley, F.R. Hore, and N.K. Patni. 1981. Pollution potential and corn yields from selected rates and timing of liquid manure applications. Transactions of the ASAE 24:139-144.
Pikul, J.L., Jr., D.E. Wilkins, J.K. Aase, and J.F. Zuzel. 1996. Contour ripping: A tillage strategy to improve water infiltration into frozen soil. Journal of Soil and Water Conservation 51:76-83.
Qiu, G., J. Leonard, J. Feddes, and W. McGill. 1996. Effects of fresh and composted manure on runoff water quality when applied on snow. Paper No. 96-205. Canadian Society of Agricultural Engineering, Winnipeg, Manitoba.
Rekolainen, S. 1989. Effect of snow and soil frost melting on the concentrations of suspended solids and phosphorus in two rural watersheds in western Finland. Aquatic Sciences 51:211-224.
Ryan, M.C., R.G. Kachanoski, and R.W. Gillham. 2000. Overwinter soil nitrogen dynamics in seasonally frozen soils. Canadian Journal of Soil Science 80:541-550.
Schillinger, W.F. and D.E. Wilkins. 1997. Deep ripping fall-planted wheat after fallow to improve infiltration and reduce erosion. Journal of Soil and Water Conservation 52:198-202.

Sharpley, A.N., R.J. Smith, J.R. Williams, O.R. Jones and G.A. Coleman. 1991. Water quality impacts associated with sorghum culture in the southern plains. Journal of Environmental Quality 20:239-244.

Shirley, C. and J.W. Meek. 1999. EPA guidelines concerning agricultural manure management practices. Pp. 23-28. In Dairy manure management. Northeast Regional Agricultural Engineering Service (ed.). Proceedings from the Dairy Manure Management Symposium. Cornell University, Ithaca, New York.

Steenhuis, T.S., G.D. Bubenzer, J.C. Converse, and M.F. Walter. 1981. Winter-spread manure nitrogen loss. Transactions of the ASAE 24:436-449.

Storey, H.C. 1955. Frozen soil and spring and winter floods. Pp. 179-184. In The Yearbook of Agriculture. U.S. Government Printing Office, Washington, D.C.

Trimble, M.Y. and J.M. Rosa. 1951. Erosion from melting snow on frozen ground. Journal of Forestry 47:807-809.

Uhlen, B. 1988. Concentrations and transport of different forms of phosphorus during snowmelt runoff from an illite clay soil. Hydrological Processes 17:747-758.

U.S. Department of Agriculture (USDA). 1941. Climate and Man: Year Book of Agriculture. U.S. Government Printing Press, Washington D.C.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). 2000. Code 590 Nutrient management practice standard. USDA-NRCS Lincoln, Nebraska.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). 2001a. Code 590 Nutrient management practice standard. USDA-NRCS Palmer, Alaska.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). 2001b. Code 590 Nutrient management practice standard. USDA-NRCS Indianapolis, Indiana.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). Code 590 Nutrient management practice standard. USDA-NRCS Des Moines, Iowa.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). Code 590 Nutrient management practice standard. USDA-NRCS Lexington, Kentucky.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). Code 590 Nutrient management practice standard. USDA-NRCS Bangor, Maine.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). Code 590 Nutrient management practice standard. USDA-NRCS Amherst, Massachusetts.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). Code 590 Nutrient management practice standard. USDA-NRCS Saint Paul, Minnesota.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). Code 590 Nutrient management practice standard. USDA-NRCS Columbia, Missouri.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). Code 590 Nutrient management practice standard. USDA-NRCS Durham, New Hampshire.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). Code 590 Nutrient management practice standard. USDA-NRCS Portland, Oregon.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). Code 590 Nutrient management practice standard. USDA-NRCS Harrisburg, Pennsylvania.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). Code 590 Nutrient management practice standard. USDA-NRCS Warwick, Rhode Island.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). 2001m. Code 590 Nutrient management practice standard. USDA-NRCS Columbia, South Carolina.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). 2001n. Code 590 Nutrient management practice standard. USDA-NRCS Richmond, Virginia.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). 2001o. Code 590 Nutrient management practice standard. USDA-NRCS Spokane, Washington.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). 2002a. Code 590 Nutrient management practice standard. USDA-NRCS Auburn, Alabama.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). 2002b. Code 590 Nutrient management practice standard. USDA-NRCS Tolland, Connecticut.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). 2002c. Code 590 Nutrient management practice standard. USDA-NRCS Dover, Delaware.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). 2002d. Code 590 Nutrient management practice standard. USDA-NRCS Champaign, Illinois.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). 2002e. Code 633 Waste utilization practice standard. USDA-NRCS Bangor, Maine.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). 2002f. Code 590 Nutrient management practice standard. USDA-NRCS Champaign, Illinois.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). 2002g. Code 590 Nutrient management practice standard. USDA-NRCS East Lansing, Michigan.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). 2002h. Code 590 Nutrient management practice standard. USDA-NRCS Reno, Nevada.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). 2002i. Code 590 Nutrient management practice standard. USDA-NRCS Bismarck, North Dakota.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). 2002j. Code 590 Nutrient management practice standard. USDA-NRCS Colchester, Vermont.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). 2002k. Code 590 Nutrient management practice standard. USDA-NRCS Madison, Wisconsin.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). 2002l. Code 590 Nutrient management practice standard. USDA-NRCS Casper, Wyoming.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). 2003a. Code 590 Nutrient management practice standard. USDA-NRCS Washington, D.C.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). 2003b. Code 590 Nutrient management practice standard. USDA-NRCS Alexandria, Louisiana.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). 2003c. Code 590 Nutrient management practice standard. USDA-NRCS Syracuse, New York.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). 2003d. Code 590 Nutrient management practice standard. USDA-NRCS Columbus, Ohio.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). 2003e. Code 633 Waste utilization practice standard. USDA-NRCS Columbus, Ohio.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). 2003f. Code 633 Waste utilization practice standard. USDA-NRCS Nashville, Tennessee.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). 2004a. Code 590 Nutrient management practice standard. USDA-NRCS Lakewood, Colorado.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). 2004b. Code 590 Nutrient management practice standard. USDA-NRCS Boise, Idaho.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). 2004c. Code 590 Nutrient management practice standard. USDA-NRCS Salina, Kansas.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). 2004d. Code 590 Nutrient management practice standard. USDA-NRCS Bozeman, Montana.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). 2004e. Code 590 Nutrient management practice standard. USDA-NRCS Somerset, New Jersey.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). 2004f. Code 590 Nutrient management practice standard. USDA-NRCS Stillwater, Oklahoma.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). 2004g. Code 590 Nutrient management practice standard. USDA-NRCS Huron, South Dakota.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). 2004h. Code 590 Nutrient management practice standard. USDA-NRCS Salt Lake City, Utah.

U.S. Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). 2004i. Code 590 Nutrient management practice standard. USDA-NRCS Morgantown, West Virginia.

Van Vliet, L.J.P., B.J. Zebarth, and G. Derksen. 2002. Effect of fall-applied manure practices on runoff, sediment, and nutrient surface transport from silage corn in south coastal British Columbia. Canadian Journal of Soil Science 82:445-456.

Vermont Agency of Agriculture, Food and Markets. Accepted agricultural practices. Vermont Agency of Agriculture, Food and Markets, Montpelier, Vermont.

Attachment 43:

Nutrients and sediment in frozen-ground
runoff from no-till fields receiving liquid-dairy
and solid-beef manures
Komiskey et al. 2011

doi:10.2489/jswc.66.5.303

Nutrients and sediment in frozen-ground runoff from no-till fields receiving liquid-dairy and solid-beef manures

M.J. Komiskey, T.D. Stuntebeck, D.R. Frame, and F.W. Madison

Abstract: Nutrients and sediment in surface runoff from frozen agricultural fields were monitored within three small (16.0 ha [39.5 ac] or less), adjacent basins at a no-till farm in southwest Wisconsin during four winters from 2003 to 2004 through 2006 to 2007. Runoff depths and flow-weighted constituent concentrations were compared to determine the impacts of surface-applied liquid-dairy or solid-beef manure to frozen and/or snow-covered ground. Despite varying the manure type and the rate and timing of applications, runoff depths were not significantly different among basins within each winter period. Sediment losses were low (generally less than 22 kg ha^{-1} [20 lb ac^{-1}] in any year) and any statistical differences in sediment concentrations among basins were not related to the presence or absence of manure or the amount of runoff. Concentrations and losses of total nitrogen and total phosphorus were significantly increased in basins that had either manure type applied less than one week preceding runoff. These increases occurred despite relatively low application rates. Lower concentrations and losses were measured in basins that had manure applied in fall and early winter and an extended period of time (months) had elapsed before the first runoff event. The highest mean, flow-weighted concentrations of total nitrogen (31.8 mg L^{-1}) and total phosphorus (10.9 mg L^{-1}) occurred in winter 2003 to 2004, when liquid-dairy manure was applied less than one week before runoff. On average, dissolved phosphorus accounted for over 80% of all phosphorus measured in runoff during frozen-ground periods. The data collected as part of this study add to the limited information on the quantity and quality of frozen-ground runoff at field edges, and the results highlight the importance of manure management decisions during frozen-ground periods to minimize nutrients lost in surface runoff.

Key words: edge-of-field—manure—nutrients—sediment—snowmelt

The surface application of manure to frozen and/or snow-covered cropland soils is a common practice on livestock farms located in midlatitude, continental-climate regions. The frozen-ground period (FGP) provides an opportunity to haul manure with minimal soil compaction, reduces the volume of manure storage required, and can increase the time available for field preparation and planting in the spring. One alternative to spreading manure during the FGP is to have adequate manure storage; however, storage compresses the time available for spreading, increases capital investment, and can pose potential environmental problems when leaks or failures occur (Kongoli and Bland 2002).

Limited studies (most predating 1980) have evaluated both the quality and quantity of surface runoff from areas receiving surface-applied manure during FGPs. A review of some of these studies showed substantial nutrient losses due to wintertime manure application (Srinivasan et al. 2006). However, some studies suggest that the application of manure to frozen and/or snow-covered soils can have no or minimal effects (Ginting et al. 1998; Young and Holt 1977) or can potentially reduce nutrient losses by reducing the volume of runoff (Kongoli and Bland 2002). It is difficult to determine exactly why varying conclusions could be made; however, there is a variety of differences in the way the studies were conducted. Some of these differences included, but were not limited to the means by which runoff was generated (snowmelt only versus rain on snowmelt) and the management style and on-farm conditions (manure amount, type of manure, application timing, tillage, field residue, soil type and texture, and slope) during the monitoring periods.

Studies have also suggested that the timing of manure applications during FGPs can influence nutrients exported in runoff. However, these studies also show conflicting results. Klausner et al. (1976) suggested that applying manure prior to snow reduces nutrients lost compared to applying manure during snowmelt, whereas Young and Mutchler (1976) showed that applying manure on top of snow keeps manure and associated nutrients out of snowmelt flow paths. Despite limited and conflicting study results, the application of manure during the FGP has been shown to have adverse effects on aquatic resources if the applied manure enters water bodies (Milne 1976; Young and Mutchler 1976; Madison et al. 1998). Most states have developed guidelines for manure management during FGPs (Srinivasan et al. 2006). These guidelines vary from state to state and have been revised and updated as more information is available.

The agricultural management practice of no-till has been promoted as a way to reduce nutrients and sediments lost in surface runoff. Most studies have shown reduced runoff volume and sediment loss from no-till fields during unfrozen conditions, compared to conventionally-tilled fields (Mickelson et al. 2001; Cox and Hendricks 2000; Andraski et al. 1985). However, limited roughness in no-till fields can potentially lead to increased runoff and nutrient losses during FGPs. Young and Mutchler (1976) showed that no-till practices posed more of a threat to nutrient losses on frozen soils than conventional tillage because the field surface is relatively smooth and nutrients accumulate at the soil surface. Hansen et al. (2000) found that this surface accumulation resulted in greater soluble phosphorus (P) losses in snowmelt runoff.

Matthew J. Komiskey and **Todd D. Stuntebeck** are physical scientists with the US Geological Survey at the Water Science Center in Middleton, Wisconsin. **Dennis R. Frame** and **Fred W. Madison** are codirectors of the Discovery Farms Program, a University of Wisconsin-Extension program, in Pigeon Falls, Wisconsin.

In response to limited data and conflicting literature results available regarding wintertime runoff from no-till fields and the effects of manure applied during FGPs, research was conducted at a privately owned livestock farm. Surface runoff quantity and quality data from runoff events were collected over a four-year period to quantify runoff losses, the timing of these losses, and the conditions in which losses typically occurred. The runoff results, in addition to the manure-application data, were then used to evaluate the impacts of applying liquid-dairy or solid-beef manure to frozen and/or snow-covered ground.

Materials and Methods

Site Description. The US Geological Survey, in cooperation with the University of Wisconsin Discovery Farms program, installed three surface monitoring stations in grassed waterways within three small (6.8, 7.0, and 16.0 ha [16.9, 17.2, and 39.5 ac]), adjacent basins at a privately owned farm in southwest Wisconsin (figure 1). Each basin contained portions of two separate farm fields with 4% to 6% average slope. Field soils consisted of primarily Tama silt loam (Fine-silty, mixed, superactive, mesic Typic Argiudoll) overlying fractured limestone bedrock. The soils had not been tilled for approximately twenty years. Soil properties—pH, organic matter, and soil test P values (Bray-1 method)—were similar among basins (table 1). Average soil test P values were approximately 80 ppm (160 lb ac^{-1}) in each basin. Fields were planted on the contour in a rotation of two years of corn (*Zea mays* L.) followed by one year of soybeans (*Glycine max* L.). Terraces and grassed waterways were used as conservation practices to reduce soil erosion and move runoff water from the field. Corn was harvested as silage or grain, with about two-thirds of corn-grain residue removed for livestock bedding for a beef-finishing enterprise. Solid-beef manure (SBM) from the farm and liquid-dairy manure (LDM) from a neighboring confined-dairy farm were periodically surface applied to the cropped fields to meet nutrient requirements.

Monitoring stations were installed, maintained, and operated according to the methods described in Stuntebeck et al. (2008). H-flumes were used to quantify the volume of runoff, while refrigerated samplers collected discrete, time-based samples during natural rainfall- and/or snowmelt-induced runoff events. Samples were generally

Figure 1

Monitoring locations, basin sizes, and field locations at a no-till farm in southwest Wisconsin.

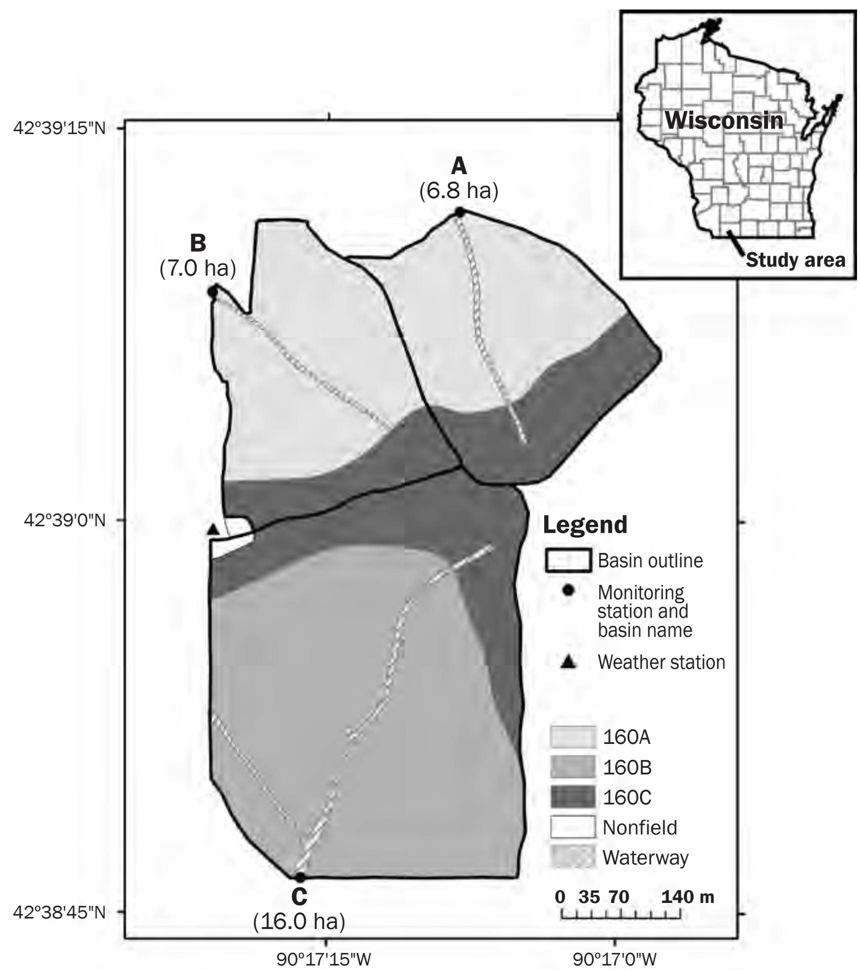


Table 1

Soil properties of Tama silt loam for three monitored basins located at a no-till farm in southwest Wisconsin, sampled October 2006.

Basin name	Sample depth (cm)	Soil pH	Soil organic matter (%)	Soil phosphorus (ppm)
A				
	0 to 2.5	7.1	4.8	142
	2.5 to 15.2	7.3	3.4	83
	0 to 15.2	7.2	3.6	97
	15.2 to 30.5	7.2	2.7	25
B				
	0 to 2.5	7.0	4.5	143
	2.5 to 15.2	7.0	3.4	86
	0 to 15.2	7.0	3.4	82
	15.2 to 30.5	7.1	2.8	56
C				
	0 to 2.5	7.0	4.5	102
	2.5 to 15.2	7.2	3.4	48
	0 to 15.2	7.2	3.8	82
	15.2 to 30.5	7.3	2.8	20

retrieved within 24 hours of the end of a runoff event and were subsequently iced and transported to the Water and Environmental Analysis Laboratory at the University of Wisconsin–Stevens Point.

A weather station on the farm collected weather and soil temperature data (figure 1). Measurements included rainfall amount and intensity, soil temperature at 2, 5, 10, 20, 40, and 80 cm (0.8, 2.0, 3.9, 7.9, 15.7, and 31.5 in), and air temperature. Frozen precipitation (snow, ice pellets, and hail) was estimated using data from the National Weather Service station in Platteville, Wisconsin, approximately 18.2 km (11.3 mi) to the northeast (National Weather Service 2007). Annual rainfall (1971 to 2000) averages 917 mm (36.1 in), and snowfall averages 1,041 mm (41.0 in) at this weather station (Midwestern Regional Climate Center 2007a; Midwestern Regional Climate Center 2007b).

Volumes were measured for all runoff events, and water samples were collected and analyzed for most runoff events for the entire period from November 2003 through September 2007; however, only runoff events that occurred when the ground was frozen, at any measured depth at the weather station, were included in this analysis. Frozen-ground periods typically extended from mid-December through mid-March. Runoff events included those derived from snowmelt only, a combination of rain and snowmelt, and rainfall only. Runoff volume was divided by the basin area and multiplied by a numerical factor to determine runoff depth (mm [in]), which normalized the runoff volume for differences in basin size.

A flow-weighted composite sample was produced for each runoff event at each station by calculating the percentage of the total event volume that each discrete sample represented, collecting appropriate aliquots from each discrete sample using a churn splitter, and combining aliquots. The composite sample was analyzed for concentrations of total dissolved solids, suspended sediment, chloride, nitrate plus nitrite–nitrogen, ammonium nitrogen, total Kjehldahl nitrogen, dissolved reactive P, and total P. The total nitrogen (TN), organic N, and particulate P fraction of each sample were calculated from the reported results. The resultant constituent concentrations for each flow-weighted sample were multiplied by the event volume and a numerical factor to determine constituent losses (kg ha^{-1} [lb ac^{-1}]) for each event in each basin.

Each monitored basin included portions of two separately managed fields; therefore, manure application rates were adjusted to reflect rates for a basin scale, rather than a field scale (table 2). These adjusted rates, deemed "effective application rates," are field-application rates that account for the percent of each basin that received manure. For example, the effective application rate of a 20 ha (49.4 ac) field (within a 40 ha [98.8 ac] basin) that received manure on the entire field would be reported as half that of the actual field application rate.

Manure was surface applied to the fields by the producer at a time, rate, location, and method typical of his operation. Following is a brief synopsis of manure applications and some of the field conditions at the time of application during each study year. Additional details can be found in table 2.

- Some LDM was applied to basins A and B in September of 2003. Additional LDM was applied on frozen and snow-covered ground in February of 2004, less than one week before the start of snowmelt. Some LDM was applied to basin C in November of 2003, on top of frozen, but not snow-covered, ground.
- Some SBM was applied to all three basins in September and October of 2004. Some LDM was also applied to basin C in October of 2004. Basin C then received four separate, SBM applications on frozen and snow-covered ground in January and February of 2005. Some applications were made on top of melting snow.
- Some SBM was applied to basins A and B on frozen and snow-covered ground in early December of 2005 and early January of 2006. Basin C received only SBM in September and October of 2005.
- Some SBM was applied to all three basins in the fall and before the ground froze in mid-December of 2006. Basins A and B then received three spreader loads of SBM mixed with snow three days before runoff occurred in March of 2007.

Runoff depths and flow-weighted constituent concentrations (and thus constituent losses) were compared using nonparametric tests of data ranks among sets of data for each of the three basins within each year. Differences in depths and concentrations among the monitored basins were considered to be significant at $p < 0.05$. Only runoff events that were mutually common among all three basins were used in the statistical analyses. Typically, runoff events were defined from the time when runoff started to the time runoff ended, unless the event contained multiple peaks or lasted several days. In some of these cases, the event was subdivided to provide information regarding runoff volume and constituent-concentration changes throughout the event. Some small runoff events were not sampled, but sample volumes were computed; consequently, the number of values statistically compared for runoff volumes was greater than that for constituent concentrations. Unsampled events typically represented 6% or less of the total runoff volume measured in any FGP.

Results and Discussion

Precipitation and Surface Runoff. During winter 2003 to 2004, approximately 744 mm (29.3 in) of frozen precipitation fell on the basins. Using an averaged ratio of 14 in (355.6 mm) of snowfall to 1 in (25.4 mm) of liquid-water equivalent determined by Baxter et al. (2005), the liquid-water equivalent (WE) of this frozen precipitation was approximately 53 mm (2.1 in). An additional 127 mm (5.0 in) of rain fell on the basins when the ground was frozen. The approximate freeze/thaw dates extended from November 25, 2003, to March 25, 2004. This time frame represents the period in which the ground was persistently frozen at any measured depth. Five runoff events were induced by both snowmelt only and rain on snow during the FGPs between February 18 and 28, 2004 (figure 2a). Runoff depths of 32, 28, and 19 mm (1.26, 1.10, and 0.76 in) were recorded during this period for basins A, B, and C, respectively (table 3). This frozen period of runoff comprised nearly 60% of the total runoff volume monitored for the entire year (October 1, 2003, to September 30, 2004). Approximately 15% of the total precipitation that fell during the FGP was measured as runoff. Runoff depths were not significantly different among the three basins, even though basins A and B received LDM about five days before the start of runoff (tables 2 and 4).

During the winter of 2004 to 2005, approximately 925 mm (36.4 in) of frozen precipitation fell (WE = 66 mm [2.6 in]). Additional rainfall during this FGP totaled 66 mm (2.6 in). The approximate freeze/thaw dates extended from December 13, 2004,

Table 2
Manure-application history for each of three monitored basins at a no-till farm in southwest Wisconsin, 2003 to 2007. All manure applications occurred between fall harvest and the end of each period of frozen-ground runoff.

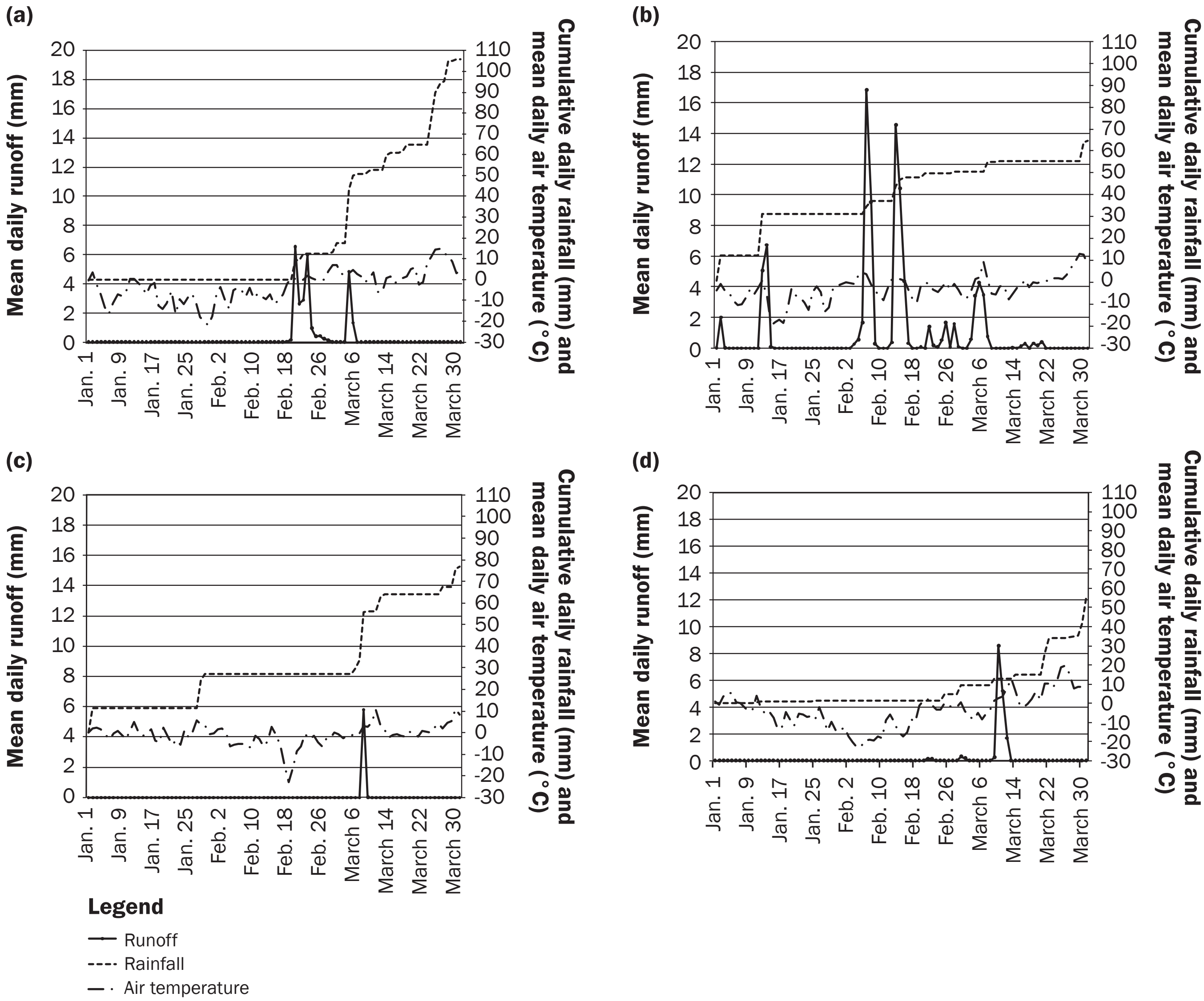
Basin name	Application date	Field name	Manure type	Analysis ($g\ kg^{-1}$ or $g\ L^{-1}$) N	P	K	Dry matter (%)	Application rate ($kL\ ha^{-1}$ or $Mg\ ha^{-1}$)	Area applied in basin (%)	Effective application rate ($kL\ ha^{-1}$ or $Mg\ ha^{-1}$)*
Fall/winter 2003 to 2004										
A	9/19/2003	160A	LDM	2.3	0.3	1.5	5	40.2	66	26.7
A	2/14/2004	160A	LDM	2.3	0.3	1.5	5	40.2	46	18.3
B	9/19/2003	160A	LDM	2.3	0.3	1.5	5	40.2	77	31.1
B	2/14/2004	160A	LDM	2.3	0.3	1.5	5	40.2	22	8.9
C	Nov. 2003	160B	LDM	2.3	0.3	1.5	5	65.5	76	50.1
Fall/winter 2004 to 2005										
A	9/18/2004	160C	SBM	10.6	3.3	7.3	25	5.4	34	1.8
A	10/6/2004	160A	SBM	10.6	3.3	7.3	25	8.3	66	5.5
B	9/18/2004	160C	SBM	10.6	3.3	7.3	25	5.4	22	1.2
B	10/6/2004	160A	SBM	10.6	3.3	7.3	25	8.3	77	6.4
C	9/18/2004	160C	SBM	10.6	3.3	7.3	25	5.4	22	1.2
C	10/9/2004	160B	LDM	2.2	0.4	1.7	5	38.4	76	29.3
C	10/29/2004	160B	SBM	10.6	3.3	7.3	25	16.0	51	8.1
C	1/1/2005	160B	SBM	8.7	2.9	6.1	22	14.6	20	3.0
C	1/28/2005	160B	SBM	8.7	2.9	6.1	22	14.6	15	2.2
C	2/12/2005	160B	SBM	8.7	2.9	6.1	22	7.3	51	3.7
C	2/19/2005	160B	SBM	8.7	2.9	6.1	22	11.2	39	4.4
Fall/winter 2005 to 2006										
A	12/1/2005	160A	SBM	8.7	2.9	6.1	22	5.6	33	1.9
A	1/1/2006	160A	SBM	8.7	2.9	6.1	22	5.6	33	1.9
B	12/1/2005	160A	SBM	8.7	2.9	6.1	22	5.6	39	2.2
B	1/7/2006	160A	SBM	8.7	2.9	6.1	22	5.6	39	2.2
C	Sep./Oct. 2005	160B	SBM	8.7	2.9	6.1	22	20.2	76	15.4
Fall/winter 2006 to 2007										
A	10/11/2006	160A	SBM	8.8	3.4	5.4	30	8.3	33	2.7
A	10/28/2006	160A	SBM	8.8	3.4	5.4	30	8.3	33	2.7
A	12/20/2006	160C	SBM	8.8	3.4	5.4	30	8.3	34	2.8
A	3/6/2007	160A	SBM	—	—	—	—	29.1	4	1.3
B	10/11/2006	160A	SBM	8.8	3.4	5.4	30	8.3	39	3.2
B	10/28/2006	160A	SBM	8.8	3.4	5.4	30	8.3	39	3.2
B	12/20/2006	160C	SBM	8.8	3.4	5.4	30	8.3	22	1.8
B	3/6/2007	160A	SBM	—	—	—	—	29.1	13	3.8
C	10/28/2006	160B	SBM	8.8	3.4	5.4	30	9.6	76	7.4
C	12/27/2006	160C	SBM	8.8	3.4	5.4	30	16.6	22	3.7

Notes: N = nitrogen. P = phosphorus. K = potassium. LDM = liquid-dairy manure. SBM = solid-beef manure with bedding corn stalks. — = manure analysis and percent dry matter not determined.

* Effective application rate is the application rate multiplied by the percentage of area applied within the basin.

to April 3, 2005. Twenty-two runoff events were recorded, totaling 111, 96, and 70 mm (4.38, 3.79, and 2.75 in) for basins A, B, and C, respectively. A majority of runoff occurred during rain on snow events on February 5 to 7 and February 13 to 15, 2005 (figure 2b). Approximately 70% of total precipitation that fell during the FGP was measured as runoff and comprised nearly 100% of the total monitored runoff for the entire year. This percentage was much greater than during the 2003 to 2004 FGP, possibly because a

Figure 2
Mean daily runoff, cumulative daily rainfall, and mean daily air temperatures from January 1 to March 31 for each of the monitored winter periods: (a) 2004, (b) 2005, (c) 2006, and (d) 2007. Date for frozen-ground periods preceding January 1 are not shown because no runoff occurred at any station. Fields were not necessarily frozen for the entire period shown for each year.



particularly widespread ice layer had formed on the soil surface and limited infiltration. Runoff depths did not differ significantly among the three basins, even though basin C received multiple applications of SBM throughout the FGP, some on melting snow.

During the winter of 2005 to 2006, approximately 945 mm (37.2 in) of frozen precipitation fell (WE = 68 mm [2.7 in]). Rainfall during this FGP totaled 64 mm (2.5 in), and the approximate freeze/thaw dates extended from December 2, 2005, to March 26, 2006. Although precipitation amounts were similar to those in previous years, only one FGP runoff event occurred during a rain on snow event on March 8 to 9, 2006 (figure 2c). This event was the only runoff recorded for the entire year. An early snow pack formed in November and December, limiting soil frost development and possibly allowing increased infiltration. Approximately 5% of total precipitation that fell during the FGP was measured as runoff. No statistics were computed on the data for this FGP.

During the winter of 2006 to 2007, approximately 1,168 mm (46.0 in) of frozen precipitation fell (WE = 84 mm [3.3 in]). Rainfall during this FGP totaled 58 mm (2.3 in), and the approximate freeze/thaw dates extended from December 1, 2006, to March 24, 2007. Six runoff events were recorded, totaling 7, 15, and 27 mm (0.28, 0.60, and 1.06 in) for basins A, B, and C, respectively. A majority of runoff occurred on March 10 to 12, 2007, (figure 2d) which was almost exclusively from melting snow due to warm air temperatures. This runoff comprised about 80% of the total monitored runoff for the entire year. Approximately 12% of total precipitation that fell during the FGP was

Table 3
Total runoff depth and mean flow-weighted concentrations of sediment, total nitrogen (TN), and total phosphorus (TP) in runoff during frozen-ground periods (FGP) for winters from 2003 to 2004 through 2006 to 2007.

Basin name	Residue type*	Runoff (mm)	Sediment (mg L^{-1})	TN (mg L^{-1})	TP (mg L^{-1})
FGP winter 2003 to 2004 (*n* = 6 [runoff], *n* = 5 [constituents])					
A	Corn/corn	32.0a	38.7a	31.8c	10.9c
B	Corn/corn	27.9a	80.7b	21.2b	8.4b
C	Corn/soybean-corn†	19.3a	17.0a	3.8a	2.3a
FGP winter 2004 to 2005 (*n* = 22 [runoff], *n* = 9 [constituents])					
A	Corn/soybean	111.2a	14.1a	3.1a	1.8a
B	Corn/soybean	96.2a	17.9a	3.1a	1.9a
C	Soybean-corn†/corn-soybean†	69.8a	45.1a	11.5b	5.8b
FGP winter 2005 to 2006‡					
A	Soybean/corn	13.70	61.0	8.0	7.7
B	Soybean/corn	<0.01	<0.01	<0.01	<0.01
C	Corn-soybean†/corn	3.70	205.0	11.0	5.6
FGP winter 2006 to 2007 (*n* = 6 [runoff], *n* = 3 [constituents])					
A	Corn/corn	7.1a	24.7a	4.0a	3.1a
B	Corn/corn	17.0a	32.4a	7.8a	6.7a
C	Corn/soybean-corn†	27.0a	248.1b	5.7a	3.6a

Notes: Values within a column and year followed by the same letter are not significantly different at $p < 0.050$; letters correspond to nonparametric analysis based on data ranks rather than means as shown.

* Both the residue and the upcoming crop are listed. For example, corn/corn shows that the basin had corn removed in the previous fall (thus the residue) and is going back into corn the next spring.

† This basin contains two fields. The first residue type listed comprises 80% of the basin area, while the second listed comprises 20%.

‡ Statistics not computed.

measured as runoff. Surface runoff depths did not differ statistically among the three basins.

Runoff depths compared among each of the three basins were not different in any given year, despite each basin receiving different types of manure at varying rates and times. Other studies have shown decreased runoff due to manure applications (Ginting et al. 1998; Young and Holt 1977; Kongoli and Bland 2002). It is possible that the relatively low application rates used in this study were not great enough to affect runoff volumes.

Despite relatively similar precipitation amounts during each FGP, both the number and volume of runoff events varied greatly from year-to-year. From site observations, it appeared that runoff amounts were more related to the timing and type (snow/sleet/rain) of precipitation, intensity of precipitation (rainfall), air temperatures, and snow-pack properties such as depth, water equivalent, ice layers, and temperature, rather than soil temperatures or frost depth alone.

Sediment. Mean flow-weighted concentrations of suspended sediment in runoff during FGPs were low, ranging between 7 and 248 mg L^{-1} (table 3). Sediment losses during FGPs were also low, with each site typically having less than 22 kg ha^{-1} (20 lb ac^{-1}) measured during any FGP (table 4). Sediment concentrations in basin B were significantly greater than in the other two basins during the 2003 to 2004 FGP, while concentrations in basin C were significantly greater than in the other two basins during the 2006 to 2007 FGP.

These differences were not associated with the presence or absence of LDM or SBM, nor were they influenced by the amount of runoff. Rather, basin-specific conditions were likely contributing factors. Greater concentrations in basin B during the 2003 to 2004 FGP were likely the result of terrace maintenance, and, to a lesser degree, to the reshaping of the waterway during the installation of the monitoring site in the fall of 2003. The greater concentrations in basin C during the 2006 to 2007 FGP were likely the result of soil erosion from end rows due to disturbance by farm equipment during harvest, which was observed at a location near the monitoring station. Regardless, the relatively low suspended sediment concentrations and losses observed are consistent with Hansen et al. (2000), who showed that wintertime runoff is not expected to cause significant soil erosion because of the slower rate of snowmelt compared to rainfall-induced runoff. Soil particles may also be less likely to become entrained in runoff when frozen together.

Nitrogen. During the FGP of winter 2003 to 2004, mean flow-weighted concentrations and losses of TN in runoff from basin A were significantly greater than in basin B, and TN in both basins A and B was greater than in basin C (table 3, 4). Individual event-mean concentrations of TN ranged from 9.3 to 69.7 mg L^{-1} in basins A and B and from 2.8 to 5.1 mg L^{-1} in basin C. Total nitrogen losses ranged from 0.7 to 10.1 kg ha^{-1} (0.6 to 9.0 lb ac^{-1}) among basins for the same period. These FGP losses comprised about 80%, 60%, and 40% of annual TN losses in basins A, B, C, respectively.

The greatest TN concentrations for the 2003 to 2004 FGP were observed during the onset of runoff in basins A and B and generally decreased over time. Total nitrogen concentrations were relatively constant throughout the runoff period in basin C. Although the amount of nitrogen applied in manure was relatively similar among the basins (table 4), basins A and B (which received LDM on top of snow, less than one week preceding snowmelt) exhibited significantly greater TN concentrations and losses. The greater concentrations observed in basin A compared to B are likely due to the greater effective application rate during the February 14, 2004, LDM application. These observations suggest that both the timing and amount of applied LDM were important factors for concentrations and losses of TN in runoff during FGPs.

Assuming that only nitrogen (N) from manure contributed to TN in runoff, TN losses from basins A, B, and C comprised 10%, 6%, and 1% of the TN that was applied after crop harvest (table 4). If only the TN added during the February LDM application to basins A and B was a significant TN component in runoff, 24% and 29% of this TN applied was measured in runoff, respectively. These percentages should be considered to be conservatively high since it was assumed

Journal of Soil and Water Conservation 66(5):303-312 www.swcs.org

Table 4

Suspended sediment, total nitrogen (TN), and total phosphorus (TP) losses in runoff; nutrients applied in manure and the percentage of those nutrients measured in runoff; and days between manure application and runoff during frozen-ground periods (FGP) for winters from 2003 to 2004 through 2006 to 2007. Calculated percentages assume that only nutrients applied in manure contributed to runoff.

	Runoff losses (kg ha^{-1})			Total nutrients applied in manure (kg ha^{-1})					Nutrients in manure applied on snow only (kg ha^{-1})					
Basin name	Suspended sediment	TN	TP	Manure type	N	Applied N in runoff (%)	P	Applied P in runoff (%)	Manure type	N	Applied N in runoff (%)	P	Applied P in runoff (%)	Days between last manure application and first runoff event
FGP winter 2003 to 2004														
A	12.3	10.1	3.5	LDM*	103	10	15	24	LDM*	42	24	6	62	4
B	22.7	5.9	2.4	LDM*	92	6	13	18	LDM*	20	29	3	70	6
C	3.3	0.7	0.4	LDM*	114	1	17	2	na	na	na	na	na	88
FGP winter 2004 to 2005														
A	15.7	3.4	2.0	SBM†	77	4	24	9	na	na	na	na	na	88
B	17.2	3.0	1.8	SBM†	81	4	25	7	na	na	na	na	na	87
C	31.5	8.0	4.0	Both	279	3	80	5	SBM†	116	7	38	11	0
FGP winter 2005 to 2006														
A	8.4	1.1	1.1	SBM†	33	3	11	9	SBM†	33	3	11	9	67
B	<0.01	<0.01	<0.01	SBM†	38	<1	12	<1	SBM†	38	<1	12	<1	67
C	7.5	0.4	0.2	SBM†	135	<1	45	<1	na	na	na	na	na	159
FGP winter 2006 to 2007														
A	1.8	0.3	0.2	SBM	73	<1	28	<1	SBM	‡	‡	‡	‡	3
B	5.0	1.3	1.1	SBM	72	2	28	4	SBM	‡	‡	‡	‡	4
C	67.0	1.5	1.0	SBM†	96	2	38	3	na	na	na	na	na	56

Notes: na = not applicable. N = nitrogen. P = phosphorus.

* Liquid-dairy manure (LDM), average analysis 2.3 g L^{-1} nitrogen, 0.8 g phosphorus oxide L^{-1}, 1.9 g potassium oxide L^{-1}, and 5% dry matter.

† Solid-beef manure (SBM) with bedding corn stalks, average analysis 9.3 g N kg^{-1}, 7.2 g phosphorus oxide kg^{-1}, 7.4 g potassium oxide kg^{-1}, and 30% dry matter.

‡ The amount of total nutrients applied on snow is unknown but is assumed to be relatively small.

that no N sources other than manure were significant contributors.

During the FGP of winter 2004 to 2005, concentrations and losses of TN in runoff from basin C were significantly greater than those in basins A and B. Individual event-mean concentrations of TN ranged from 3.3 to 23.8 mg L^{-1} in basin C and from 1.7 and 7.4 mg L^{-1} in basins A and B. The greatest TN concentrations in basin C occurred during runoff periods in which SBM was recently applied on top of melting snow. This suggests that the timing of applied SBM was also an important factor for TN losses in runoff. The TN losses recorded during this FGP ranged from 3.0 to 8.0 kg ha^{-1} (2.7 and 7.1 lb ac^{-1}) and comprised nearly 100% of the TN measured for the entire year in each basin. Assuming that only N from manure contributed to TN in runoff, TN losses from basins A, B, and C comprised 4%, 4%, and 3% of the TN that was applied in manure after crop harvest, respectively. If only the N added to basin C during the January and February SBM applications was a significant TN component in runoff, 7% of this TN applied was measured in runoff. This percentage was lower than the one for the LDM applied shortly preceding snowmelt to basins A and B during the FGP of winter 2003 to 2004, even though approximately four times more TN was applied and runoff was over twice as great. While the application of LDM or SBM to frozen and snow-covered ground shortly preceding snowmelt increased TN concentrations and losses in runoff, these observations suggest that contributions to runoff from the application LDM may be greater than those for SBM applied at similar rates, times, and field conditions.

Despite the fact that SBM was applied to basins A and B shortly preceding runoff during the 2006 to 2007 winter, TN concentrations and losses were not significantly different among basins. This is possibly due to the fact that only a small amount of TN in SBM (actual amount applied was unknown, but observations indicated that the application was a mixture of manure and snow) was applied to basins A and B, which suggests that the rate of SBM applied can also impact TN concentrations and losses. Individual event-mean concentrations in runoff ranged from 3.0 to 8.9 mg L^{-1} among the basins, with measured TN losses comprising less than 2% of that applied in manure. Total nitrogen losses during this FGP comprised about 50%, 100%, and 70% of annual TN losses in basins A, B, C, respectively.

The forms of N can be used to help evaluate the potential impacts of manure applications to the environment that are not necessarily described by TN alone. Nitrogen inputs—particularly nitrate-nitrogen—to water bodies limited by N are a well-documented concern (Cerco 1995; Rabalais et al. 2001, 2002). Another concern is N toxicity to fish and other aquatic organisms. Toxic forms of N to aquatic species include unionized ammonia and the ammonium ion (USEPA 1999). Ammonia is generally considered to be more toxic, with the degree of toxicity primarily dependent upon pH and temperature. Ammonium nitrogen (ammonium N) reported in this article is the total concentration reported for the sum of both unionized ammonia and ammonium ion. Actual amounts of unionized ammonia were not calculated because pH and temperature were not measured.

The highest mean flow-weighted concentrations of ammonium N occurred during the FGP of winter 2003 to 2004 (table 5). Individual event-mean concentrations of ammonium N during this period ranged from 1.6 to 43.6 mg L^{-1} at basin A and from 1.2 to 18.6 mg L^{-1} at basin B. Each basin received LDM on frozen and snow-covered ground shortly preceding runoff. In basin A, event-mean concentrations exceeded 20 mg L^{-1} for several events on consecutive days when manure was observed in the runoff water. The highest ammonium N concentrations during these events were often associated with high organic-N concentrations but relatively low nitrate plus nitrite N concentrations. Ammonium N concentrations in runoff ranged between 0.1 and 1.4 mg L^{-1} at basin C, which received only LDM in November 2003 on top of frozen—but not snow-covered—ground. At the onset of runoff in this basin, the LDM had been applied to the basin for nearly three months.

During the FGP of winter 2004 to 2005, mean flow-weighted concentrations of ammonium N in basin C were not as high as those from basins A and B during the previous winter. Individual event-mean ammonium N concentrations were also lower in basin C, ranging from 0.4 to 9.5 mg L^{-1}. These lower concentrations occurred despite the fact that basin C received SBM on top of frozen and snow-covered ground shortly preceding runoff and that approximately four times more N was applied in the SBM application than the LDM application the previous year. The lower ammonium N concentrations could have been the result of dilution by greater runoff volumes, different runoff conditions, and/or by inherently lower ammonium N concentrations in the SBM compared to the LDM.

Phosphorus. The total phosphorus (TP) concentrations and losses in runoff during FGPs and the relations between these losses and manure applications were similar to those for TN. During the FGP of winter 2003 to 2004, mean flow-weighted concentrations and losses of TP in runoff from basin A were significantly greater than in basin B, and TP in both basins A and B was greater than in basin C (tables 3 and 4). Individual event-mean TP concentrations ranged from 2.7 to 28.3 mg L^{-1} in basins A and B and from 1.6 to 3.2 mg L^{-1} in basin C. As was the case for TN, the timing of the LDM application was likely an important factor for greater concentrations and losses of TP in runoff during FGPs. Also, the greater effective application rate of LDM in basin A on February 14, 2004, likely increased TP concentrations and losses compared to those in basin B.

Table 5

Mean flow-weighted concentrations of nitrogen and phosphorus species in runoff during frozen-ground periods (FGP) for winters from 2003 to 2004 through 2006 to 2007.

	Nitrogen			Phosphorus	
Basin name	Nitrate plus nitrite N (mg L^{-1})	Ammonium N (mg L^{-1})	Organic N (mg L^{-1})	Dissolved reactive P (mg L^{-1})	Particulate P (mg L^{-1})
FGP winter 2003 to 2004					
A	2.0	12.2	17.5	8.2	2.7
B	0.8	9.9	10.5	6.7	1.8
C	1.6	0.4	1.9	2.0	0.3
FGP winter 2004 to 2005					
A	0.9	0.7	1.5	1.8	0.0
B	1.0	0.8	1.3	1.9	0.0
C	0.6	1.9	9.0	5.2	0.5
FGP winter 2005 to 2006					
A	2.1	1.7	4.2	6.6	1.1
B	<0.01	<0.01	<0.01	<0.01	<0.01
C	1.3	0.8	8.9	3.2	2.4
FGP winter 2006 to 2007					
A	0.6	1.0	2.5	1.5	1.6
B	0.3	3.2	4.3	6.5	0.2
C	0.9	1.9	2.9	3.1	0.5

Assuming that only P from manure contributed to TP in runoff, TP losses from basins A, B, and C measured 24%, 18%, and 2% of the P that was applied in manure after crop harvest. If only the P added during the February LDM application to basins A and B was a significant TP component in runoff during the FGP of the winter of 2003 to 2004, 62% and 70% of TP applied was measured in runoff, respectively. These percentages should be considered to be conservatively high since it was assumed that no P sources other than manure were significant contributors. Total phosphorus measured in runoff during this FGP comprised about 80%, 66%, and 54% of the annual TP losses in basins A, B, and C, respectively.

During the FGP of the winter of 2004 to 2005, concentrations and losses of TP in basin C were significantly greater than in basins A and B. Individual event-mean TP concentrations ranged from 1.0 to 4.8 mg L^{-1} in basins A and B and from 2.3 to 11.2 mg L^{-1} in basin C. The timing of SBM application in relation to runoff was likely an important factor for these greater concentrations and losses. As with TN, the losses during this FGP comprised nearly 100% of the TP losses measured for the entire year. Assuming that only P from manure contributed to P in runoff, TP losses from basins A, B, and C measured 9%, 7%, and 5% of the TP that was applied in manure after crop harvest, respectively. If only the TP added during the January and February SBM applications to basin C was a significant TP component in runoff during this FGP, 11% was measured in runoff. This percentage was lower than for the LDM applied shortly preceding snowmelt to basins A and B during the FGP of winter 2003 to 2004, even though approximately eight times more TP was applied and runoff was over twice as great. Similar to the findings for TN, these observations suggest that the TP contributions to runoff from the application of LDM may be greater than those for SBM applied at similar rates, times, and field conditions.

Total phosphorus concentrations and losses were not significantly different among basins for the monitored runoff events during the FGP of winter 2006 to 2007, which is most likely a result of only a relatively small amount of TP being applied in basins A and B shortly preceding runoff. This result suggests that SBM applications made prior to the FGP can result in lower TP concentrations and losses in runoff compared to applications made to frozen and snow-covered ground shortly preceding runoff. Concentrations in runoff ranged from 0.6 to 7.3 mg L^{-1} among the basins, with measured TP losses in runoff being 4% or less of that applied in manure. The TP losses during this FGP comprised 76%, 100%, and 78% of the annual TP losses in basins A, B, and C, respectively.

Differentiating between P forms in runoff can help managers to determine the potential impacts on aquatic resources and the best ways to mitigate these impacts. Approximately 75% of all TP measured in runoff during the FGP of the winter of 2003 to 2004 was dissolved reactive phosphorus, and greater than 90% of dissolved reactive phosphorus was measured for the FGP of winter 2004 to 2005 (table 5). On average, dissolved reactive phosphorus accounted for over 80% of all P measured in runoff during FGPs throughout the study period. Variations in P (and dissolved reactive phosphorus) concentrations and losses measured in runoff were likely related to the timing, rate, and form of manure applications, characteristics of runoff, and field condition at the time of runoff—not just soil-test P values alone. These results are supported by other studies that have shown soil-test P to be a good indicator of the potential for dissolved P to be lost in runoff, except when manures are applied (Vadas et al. 2005a, 2005b; McDowell and Sharpley 2002). In this study, soil-test P values were relatively similar among the three basins, yet dissolved reactive phosphorus in runoff was wide-ranging.

Summary and Conclusions

Nutrients and sediment in surface runoff from frozen agricultural fields were monitored within three small (16.0 ha [39.5 ac] or less), adjacent basins at a no-till farm in southwest Wisconsin during four winters from 2003 to 2004 through 2006 to 2007. Runoff depths and flow-weighted constituent concentrations were compared to determine the impacts of surface-applied LDM or SBM to frozen and/or snow-covered ground.

Average runoff volumes were highly variable among years, but runoff depths were not significantly different among basins within each FGP. Neither the type of manure nor the rates of application significantly affected runoff volumes. Runoff was more likely related to the form, timing, and intensity of precipitation (rainfall), air and soil temperatures, and snow-pack properties, such as depth, water equivalent, and temperature. Runoff during FGPs comprised from 60% to 100% of the total runoff measured within any given year. Most FGP runoff occurred in February and/or March.

Sediment concentrations and losses in runoff during the FGPs were low (generally less than 22 kg ha^{-1} [20 lb ac^{-1}] in any year). Any nutrients associated with sediment particles were therefore also low. Although statistical comparisons showed that suspended sediment concentrations and losses occasionally differed among basins within a given year, these differences were not related to the presence or absence of LDM or SBM, nor were they influenced by the amount of runoff. Rather, localized in-field conditions were likely contributing factors.

Concentrations and losses of nitrogen and phosphorus were significantly greater in basins that had either LDM or SBM applied to frozen and snow-covered ground less than one week preceding runoff. These increases occurred despite relatively low manure-application rates. Lower concentrations and losses were measured in basins that had manure applied in fall and early winter and an extended period of time (months) had elapsed before runoff. Nutrient losses measured in runoff during the FGPs were substantial, accounting for 40% to 100% of the annual total nitrogen and TP losses in any given year. Greater than 80% of all P measured in runoff during FGPs was dissolved.

The application of manure to cropped fields during FGPs is part of the management strategy for many livestock producers in continental-climate regions. The results of this study indicate that both LDM and SBM applied to frozen and snow-covered fields less than one week preceding runoff can significantly contribute to nitrogen and phosphorus losses in runoff. Future research targeted at defining the relationships between the timing of manure applications and the amount of nutrients in runoff would allow for determining suitable application periods or field conditions that would potentially minimize nutrient contributions to runoff from agricultural fields. Additional research examining the differences between applications rates and methods, cropping types and/or tillage types, and more detailed analysis of the impacts of different manure types would enable manure-management guidance among different producer types and management styles.

Acknowledgements

The authors would like to thank the farm family for access and for generous offerings of their time and equipment. Discovery Farms staff—Kevan Klingberg, Susan Frame, and Tim Popple—were instrumental in providing on-farm information. Funding was also provided by the Wisconsin Milk Marketing Board, Wisconsin Dairy Business Association, Wisconsin Farm Bureau, Professional Dairy Producers of Wisconsin, Wisconsin Pork Producers Association, and the Natural Resources Conservation Service. Further thanks are extended to Dave Owens (US Geological Survey) for technical assistance.

References

Andraski, B.J., T.C. Daniel, B. Lowery, and D.H. Mueller. 1985. Runoff results from natural and simulated rainfall for four tillage systems. Transactions of the American Society of Agricultural Engineers 28:1219-1225.

Baxter, M.A., C.E. Graves, and J.T. Moore. 2005. A climatology of snow-to-liquid ratio for the contiguous United States. Weather and Forecasting 20:729–744.

Cerco, C.F. 1995. Response of Chesapeake Bay to nutrient load reductions. Journal of Environmental Engineering 121:549-557.

Cox, F.R., and S.E. Hendricks. 2000. Soil test phosphorus and clay content effects on runoff water quality. Journal of Environmental Quality 29:1582-1586.

Ginting, D., J.F. Moncrief, S.C. Gupta, and S.D. Evans. 1998. Corn yield, runoff, and sediment losses from manure and tillage systems. Journal of Environmental Quality 27:1396-1402.

Hansen, N.C., S.C. Gupta, and J.F. Moncrief. 2000. Snowmelt runoff, sediment, and phosphorus losses under three different tillage systems. Soil & Tillage Research 57:93-100.

Klausner, S.D., P.J. Zwerman, and D.F. Ellis. 1976. Nitrogen and phosphorus losses from winter disposal of dairy manure. Journal of Environmental Quality 5:47-49.

Kongoli, C.E., and W.L. Bland. 2002. Influence of manure application on surface energy and snow cover: Field experiments. Journal of Environmental Quality 31:1166-1173.

Madison, F.W., K.A. Kelling, L. Massie, and L. Ward-Good. 1998. Guidelines for applying manure to cropland and pasture in Wisconsin. University of Wisconsin Extension Publications. Rep. A3392.

McDowell, R., and A. Sharpley. 2002. Phosphorus transport in overland flow in response to position of manure application. Journal of Environmental Quality 31:217-227.

Mickelson, S.K., P. Boyd, J.L. Baker, and S.I. Ahmed. 2001. Tillage and herbicide incorporation effects on residue cover, runoff, erosion, and herbicide loss. Soil & Tillage Research 60:55-66.

Midwestern Regional Climate Center. 2007a. Historical Climate Data. Precipitation Summary. Plattesville, Wisconsin. http://mrcc.sws.uiuc.edu/climate_midwest/historical/precip/wi/476646_psum.html.

Midwestern Regional Climate Center. 2007b. Historical Climate Data. Snowfall Summary. Plattesville, Wisconsin. http://mrcc.sws.uiuc.edu/climate_midwest/historical/snow/wi/476646_ssum.html.

Milne, C.M. 1976. Effect of a livestock wintering operation on a western mountain stream. Transactions of the American Society of Agricultural Engineers 19:749-752.

National Weather Service. 2007. Weather Data from Platteville, Wisconsin. http://www.ncdc.noaa.gov/oa/ncdc.html.

Rabalais, N.N., R.E. Turner, and W.J. Wiseman, Jr. 2001. Hypoxia in the Gulf of Mexico. Journal of Environmental Quality 30:320-329.

Rabalais, N.N., R.E. Turner, Q. Dortch, D. Justic, V.J. Bierman, Jr., and W.J. Wiseman, Jr. 2002. Nutrient-enhanced productivity in the northern gulf of Mexico: Past, present and future. Hydrobiologia 475-476:39-63.

Srinivasan, M.S., R.B. Bryant, M.P. Callahan, and J.L. Weld. 2006. Manure management and nutrient loss under winter conditions: A literature review. Journal of Soil Water Conservation 61(4):200-209.

Stuntebeck, T.D., M.J. Komiskey, D.W. Owens, and D.W. Hall. 2008. Methods of data collection, sample processing, and data analysis for edge-of-field, streamgaging, subsurface-tile, and meteorological stations at Discovery Farms and Pioneer Farm in Wisconsin, 2001-7: U.S. Geological Survey Open-File Report 2008-1015.

USEPA (US Environmental Protection Agency). 1999. 1999 Update of ambient water quality criteria for ammonia. Rep. EPA-822-R-99-014.

Vadas, P.A., B.E. Haggard, and W.J. Gburek. 2005a. Predicting dissolved phosphorus in runoff from manured field plots. Journal of Environmental Quality 34:1347-1353.

Vadas, P.A., P.J.A. Kleinman, A.N. Sharpley, and B.L. Turner. 2005b. Relating soil phosphorus to dissolved phosphorus in runoff: A single extraction coefficient for water quality modeling. Journal of Environmental Quality 34:572-580.

Young, R.A., and R.F. Holt. 1977. Winter applied manure: Effects of annual runoff, erosion, and nutrient movement. Journal of Soil and Water Conservation 32(5):219-222.

Young, R.A., and C.K. Mutchler. 1976. Pollution potential of manure spread on frozen ground. Journal of Environmental Quality 5:174-179.

Attachment 44:

Phosphorus and sediment loading to surface waters from liquid swine manure application under different drainage and tillage practices
Coelho et al. 2012

R2012-023
S James Electronic Filing - Received, Clerk's Office, 10/16/2012
Agricultural Water Management 104 (2012) 51–61

Contents lists available at SciVerse ScienceDirect

Agricultural Water Management

journal homepage: www.elsevier.com/locate/agwat

Phosphorus and sediment loading to surface waters from liquid swine manure application under different drainage and tillage practices

B. Ball Coelho[a,*], R. Murray[b], D. Lapen[c], E. Topp[b], A. Bruin[b]

[a] Ontario Ministry of Agriculture, Food and Rural Affairs, Agriculture Development Branch, 581 Huron St., Stratford, Ontario, Canada N5A 5T8
[b] Agriculture & Agri-Food Canada, Southern Crop Protection and Food Research Centre, 1391 Sandford St., London, Ontario, Canada N5V 4T3
[c] Agriculture & Agri-Food Canada, Eastern Cereal Oilseed Research Centre, 960 Carling Ave., Ottawa, Ontario, Canada K1A 0C6

ARTICLE INFO

Article history:
Received 10 March 2011
Accepted 25 October 2011
Available online 17 November 2011

Keywords:
Blind inlet
Hickenbottom
Preferential flow
Runoff
Surface inlet
Tile drainage
Tillage

ABSTRACT

Phosphorus (P) and sediment can move from agricultural land to surface waters, deteriorating its quality. This study was undertaken to improve understanding of partitioning of P and sediment to surface water via overland runoff and underground drainage pathways, and identify control measures. Over two full years, and including important winter events, P and sediment load overland and through tile were quantified from micro-catchments with relevant drainage and management practices imposed. Crop nutrients were supplied by liquid swine manure, either injected under minimum tillage management or surface-applied and incorporated under conventional till. Winters were temporally important for loadings from both runoff and drainage tile, particularly during rain on snow. A single event of 50 mm rain on snow over 2 days contributed more than 80% of the P_{dop} (dissolved organic + particulate P) and sediment that moved overland, and contributed 28% of P_{dop} and 20% of the sediment that moved through drainage tile during that season. Loads of P and sediment in both overland runoff and tile drainage were greater in non-growing seasons (NGSs) than growing seasons (GSs). For example, loading overland averaged 0.14 kg ha^{-1} dissolved reactive phosphate (DRP) and 1551 kg ha^{-1} sediment in NGSs, and 0.04 kg ha^{-1} DRP and 42 kg ha^{-1} sediment in GSs (four catchments, two seasons, runoff DRP first GS only). Through drainage tile, DRP load averaged 0.08 kg ha^{-1} in NGSs and 0.01 kg ha^{-1} in GSs from one field, A, and 0.02 kg ha^{-1} in NGSs and 0.003 kg ha^{-1} in GSs from another field, B; P_{dop} load was 0.07 kg ha^{-1} in NGSs and 0.02 kg ha^{-1} in GSs, similar from both Fields A and B; and sediment load was 23 kg ha^{-1} in NGSs from Field A, 8 kg ha^{-1} in NGSs from Field B, and 2 kg ha^{-1} in GSs from both fields. It is therefore important to manage movement during NGSs, particularly when runoff occurs over frozen soil. Movement through drainage tile comprised 31, 24 and 16% of the overland + subsurface DRP, total P (P_t) and sediment loads, respectively. Presence or type (blind inlet or hickenbottom) of surface inlet had little impact on P and sediment loading. Artificial drainage reduced overland + subsurface load to surface water to one-third for P_t and one-tenth for sediment, and is therefore a suitable strategy for controlling both P and turbidity in surface water. Overland + subsurface DRP load was unchanged by artificial drainage. Preferential flow of liquid swine manure to drainage tile only occurred with injection, in the year the drains were installed, in one of two fields. Along with being infrequent, the incidental DRP load through tile drains comprised only 2% of the annual P_t load from the catchment. The associated minimum tillage system reduced overland P_t and sediment runoff load 3- and 6-fold, respectively, relative to conventional till with broadcast incorporated manure.

1. Introduction

Excess phosphorus (P) and sediment in surface waters can threaten the survival of aquatic species (Staton et al., 2003). The major pathways of P and sediment movement to surface water from agricultural land having artificial drainage are overland (runoff and erosion) and through subsurface tile drains. Particulate-P (P bound to sediment) generally dominates P loss from row-cropped agricultural systems (Panuska and Karthikeyan, 2010; Richards et al., 2008; Verbree et al., 2010), and so controlling sediment movement

Abbreviations: Bi, blind inlet; CP, common practice; DRP, dissolved reactive phosphorus; GS, growing season; GS1, growing season 1; GS2, growing season 2; Hb, hickenbottom inlet; LSM, liquid swine manure; MT, minimum till; ND, natural drainage; NGS, non-growing season; NGS1, non-growing season 1; NGS2, non-growing season 2; Ni, no surface inlet to drainage tile; P, phosphorus; P_{dop}, dissolved organic + particulate P; P_t, total P; PT, pre-treatment time period.

* Corresponding author. Tel.: +1 519 271 9269; fax: +1 519 273 5278.
E-mail address: bonnie.ball@ontario.ca (B. Ball Coelho).

0378-3774/$ – see front matter. Published by Elsevier B.V.
doi:10.1016/j.agwat.2011.10.020

should be effective to reduce P loading. Speciation of P can vary widely with soil texture, however. Eastman et al. (2010) found that particulate P comprised 80% of total movement from clay loam but only 20% from sandy loam. Speciation can also vary with event type. Through drainage systems with surface inlets, particulate pollutants move with major storms, while most dissolved pollutants move in snowmelt waters (Ginting et al., 2000).

Tile drainage may be an important pathway for P and sediment movement to surface waters in some agricultural watersheds (Gentry et al., 2007; Jamieson et al., 2003; Stone and Krishnapan, 2002). Relative importance of the drainage tile pathway is inconsistent (Ball Coelho et al., 2010; Culley and Bolton, 1983; Simard et al., 2000; Watson et al., 2007; Zhao et al., 2001), likely in part due to effects of differing edaphic conditions (e.g. soil texture, slope) on the frequency of runoff occurrences. The proportion of total loading to surface water that moves through drainage tile also varies with management, and with temporal and spatial scales of measure. Certain management practices (e.g., no-till) are very effective in controlling P and sediment movement overland, while practices to control movement through tile drains are not well characterized. For example, tillage effects on sediment load through drains, and partitioning between drains and runoff are inconsistent (Gaynor and Findlay, 1995; Oygarden et al., 1997; Schelde et al., 2006; Zhao et al., 2001). With manure or biosolids as a nutrient source, P can move to drainage tile depending on the degree of preferential flow (Ball Coelho et al., 2007; Lapen et al., 2008). Overland runoff of P was not quantified in these studies, however, so the relative importance of these incidental loads to tile drains is not known. Incidental losses associated with manure application were less than annual P losses from soil, in a Swiss watershed with intensive livestock production (Lazzarotto et al., 2007).

Drainage system characteristics may also affect how much P and sediment move through tile drains. Surface inlets, which are commonly used in combination with subsurface drains in areas of low elevation, may alter the relative contributions from drainage tile to total movement (Ball Coelho et al., 2010). Schilling and Helmers (2008) surmised that up to 80% of sediment movement through drainage tile in a humid temperate watershed (IN, USA) was associated with surface runoff from rainwater flowing through inlets, based on assumptions about observed changes in concentrations during storms. They noted the need for monitoring more drainage tile networks to determine how different drainage system designs impact pollutant loading. Setbacks from surface inlets are recommended for waste application in some jurisdictions. In humid temperate USA, for example (Ohio), a setback of 30 m is required for surface application where there is no vegetation, 11 m for 50% vegetative or residue cover, and 61 m for winter application on frozen or snow covered soil (NRCS-Ohio, 2003).

Much of the overland runoff in temperate regions occurs over winter. In Norway, 90% of annual P discharge in runoff from agricultural fields occurred in winter (Syversen, 2002). Snowmelt is the predominant source of overland P and sediment runoff in temperate semi-arid USA (Minnesota, Wisconsin) (Ginting et al., 2000; Panuska and Karthikeyan, 2010). Measurement is difficult in winter, and so data are lacking. Few studies have quantified the net year-round effects of artificial drainage or other management practices on total overland plus subsurface load from agricultural lands. Due to the predominance of winter movement, recommendations based on results from short-term studies conducted only during the growing season may not be effective. This could be one of the reasons why results from implementation of recommendations at the watershed scale often fall short of expectations (Forster and Rausch, 2002; Inamdar et al., 2001; Richards et al., 2005).

Knowledge about the relative P and sediment contributions from subsurface and overland pathways to total movement from agricultural land to surface water as varied with drainage system, crop management and time will help to identify target pathways, and control practices for allocating resources. Our objectives were to identify management options which would result in the greatest overall improvement to water quality, and to inform recommendations for setbacks from surface inlets. To accomplish these objectives, we quantified the net year-round effect of subsurface drainage, based on surface inlet type and management practice (tillage and manure application method), on P and sediment loading to surface water through both drainage tile and overland runoff.

2. Materials and methods

Management treatments of either minimum till (MT) with injected manure or common practice (CP) consisting of conventional till (moldboard plough and secondary tillage) and broadcast-incorporated manure, were imposed in micro-catchments created with blind (Bi) or hickenbottom (Hb) surface inlets (described in Ball Coelho et al., under revision) or no inlet (Ni) at all, in two adjacent fields, referred to as Field A and Field B, located at 43°01/N, 81°12/W in Southern Ontario, Canada (Fig. 1). Relevant physical and chemical soil properties are listed in Ball Coelho et al. (under revision).

At Field A, which was 96 m × 104 m, three catchments were created during January–April 2007, one with Hb and two with Bi inlets (Fig. 1a). Drainage tile runs, 15 m long from inlet to catchment edge, were installed in mid-January 2007 centered in each 30 m × 16 m catchment with the surface inlet at the upslope end of the tile drain. A fourth catchment of equal area and slope but with no artificial drainage, was created in August 2007, and seeded with a rye cover crop. This catchment is hereafter referred to as natural drainage (ND). Surfaces were graded to 1% slope, and berms were constructed between catchments for equal area contribution to surface inlets and overland flow (Fig. 1a). Gutters were installed to route runoff from each catchment into a separate weir box (illustrated in Ball Coelho et al., under revision).

In Field B, which was 147 m × 85 m and had no overall slope, a berm was constructed to create two areas of equal size within the field, one for CP management and the other for MT (Fig. 1b). Drainage tile runs, 6.1 m apart and 85 m long, were installed on 19 January 2007, and a Hb was installed at the lowest elevation within each of the two management areas. In each management area, two catch basins were installed at the field edge where tile drains flowed into a header. One tile line had a Hb and the other had no surface inlet. A fall in elevation was created between the tile drain and the header to allow collection of samples and determination of flow from a weir box fitted below the outlet inside each basin.

At Field A, 2007 was year 1 for both tillage regimes beginning 23–24 April when CP catchments were disked, as the site was not cultivated prior to 2007 (mowed grass). At Field B, minimum till commenced fall 2006 when the CP (but not MT) catchment was fall-ploughed. Prior to that (>10 year) Field B was managed under conventional tillage (alfalfa and vegetables in 2006), so CP had long term history. Subsequent field operations were performed in common across fields.

At both fields, liquid swine manure (LSM) was applied in May (Table 1) using a plot-scale applicator with in-tank mixing (Nuhn Industries, Sebringville, Ontario, Canada), electromagnetic flow meter (Krohne Inc., Peabody, MA, USA) and console (Raven Industries, Sioux Falls, ND, USA). For CP, LSM was surface-applied using a drop hose with an inter-hose spacing of 0.3 m and was incorporated the same day as application. An S-tine cultivator with rolling harrows was used for incorporation in GS1, with one pass at Field A and two passes at Field B (for leveling purposes). Incorporation was by disking in GS2. For MT, the LSM was injected using vibroshank (Kongskilde Ltd., Strathroy, Ontario, Canada) with 0.11-m sweeps

Table 1
Dry matter (DM), and P concentration and amount supplied by manure either injected in a minimum till (MT) system, or surface applied in a conventional till system (common practice, CP), at a rate of 37.4 m^3 ha^{-1} (except 38.3 m^3 ha^{-1} on Field A in 2007) on two fields.

Field and catchment	Application date	DM		Phosphorus			
		Concentration (g kg^{-1})[b]	SD[a] (g kg^{-1})[b]	Concentration (g kg^{-1})[b]	SD (g kg^{-1})[b]	Amount (kg ha^{-1})	SD (kg ha^{-1})
A MT & CP	22 May 2007	58	0.6	2.5	0.60	94	23.1
A MT & CP	14 May 2008	100	7.5	2.5	0.31	97	11.7
B MT	23 May 2007	52	0.5	1.9	0.13	72	4.9
B CP	23 May 2007	53	2.4	2.2	0.42	80	15.6
B MT	15 May 2008	79	7.0	1.9	0.28	71	10.6
B CP	16 May 2008	55	7.0	1.3	0.31	47	11.4

[a] Standard deviation.
[b] As applied.

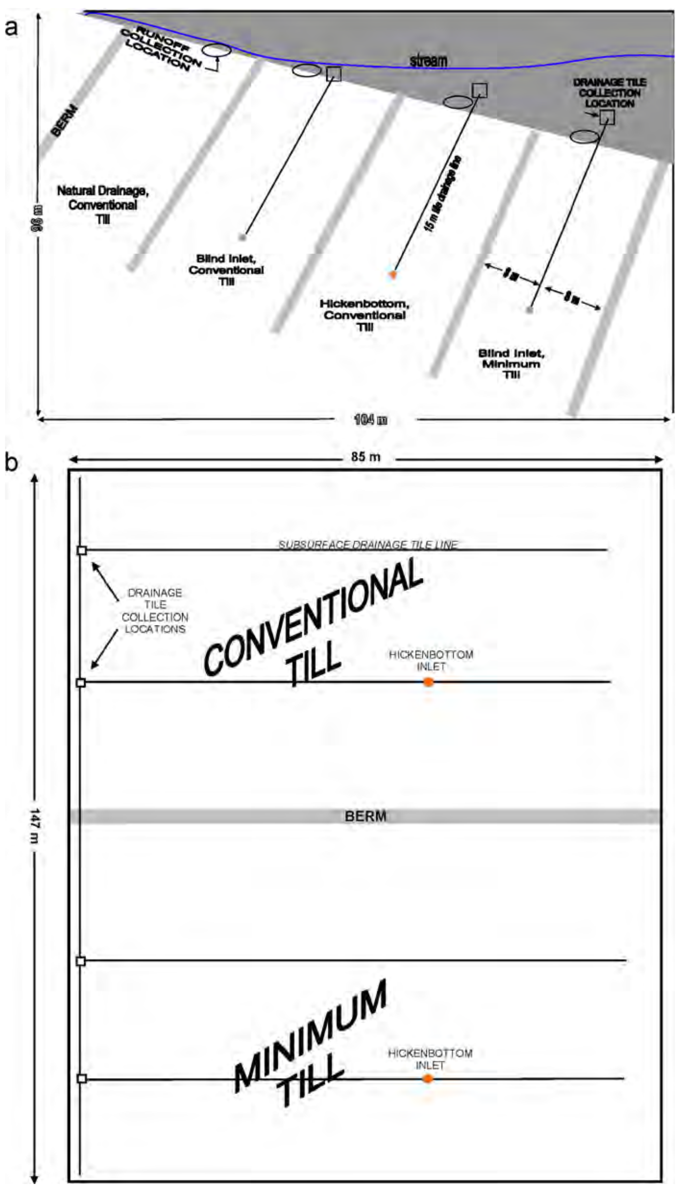


Fig. 1. Design of catchments within (a) Field A, and (b) Field B, including subsurface drainage tile lines, surface inlets, and tile drainage and runoff collection locations.

positioned between a coulter in front and adjustable fluted coulters on 0.75-m centers. There were no setbacks from surface inlets for tillage or manure application in either field. Both fields were planted with corn in 0.75-m rows, on 25 May 2007 and 23 May 2008. In MT catchments, corn rows were aligned with manure injection zones. The CP catchments were spring- (2009) rather than fall- (2008) ploughed due to inclement weather (rain and snow) in fall 2008 which prevented combining.

Drainage tile (Fields A and B) and runoff (Field A only) flows were measured using automated float/weir box systems. Every 5 min a UL16 data logger recorded the height of a float hanging from a pulley (FS 15, Lakewood Systems Ltd., Edmonton, Alberta, Canada). The float rested within a stilling well inside a wooden box that contained a V-notch weir anchored below the tile drain or gutter outlets. From the height of water flowing over the 90° V-notch weir, volume of flow in m^3 s^{-1} (Q) was calculated as follows:

$$Q = 2.5 \times h^x \quad (1)$$

where h equals head or height (m) above the bottom of notch provided by the float, and x is a constant. For each weir box, x was obtained by measuring heights of known flows. Weir box assemblies were housed in plywood structures in the field to protect the transducer from rain and reduce wind noise on the cable hanging from the pulley.

Water, usually 800 mL, was collected using programmable automated samplers (Isco Inc., Lincoln, NE, USA) with hoses positioned inside the weir boxes. When drainage tile flow was continuous, samplers were usually programmed for collection every 8 h (sometimes 12 h). If rain was imminently forecast, samplers were programmed for more frequent collection (e.g., every 3 h). If tile drains were not flowing, liquid level actuators (Isco Inc., Lincoln, NE, USA) positioned inside the boxes triggered sampling to commence with flow. For start-up drainage tile flows, samples were collected with decreasing frequency over time because P and sediment movement through tile drains is episodic, with most of the load occurring during the first few hours following intense rainfall (Ball Coelho et al., 2010). The automated collection schedule was every 15 min for 6 samples (i.e., sampled every 15 min during a minimum time span of 1.5 h if flow was sufficient for continuous operation according to the liquid actuator), followed by every 30 min for 2 samples, every 1 h for 7 samples, every 3 h for 1 sample, and every 6 h thereafter to fill the 24 bottles held in each sampler. Automated collection of samples from summer runoff water was more frequent (15 min intervals) than from drainage tile since runoff hydrographs are usually of shorter duration and higher intensity than those from drainage tile, particularly in summer. Over the winter, slow continuous runoff flows occurred with snowmelt and surface drainage over frozen ground. During these times runoff sample collection was less frequent (e.g., 6 h). Runoff flows were often recorded manually in winter (timed collection or height over the V, usually daily) because boxes were freezing (especially overnight) which caused disabling of the float recorder in ice. Grab samples were collected as necessary, for example when sampler lines were frozen.

Suspended solids concentrations in drainage tile and runoff water were determined from oven-dry (90 °C) weights obtained by filtering (0.45 μm) known volumes of water (usually 200 mL). When manure moved to tile drains (occurred in GS1), the organic fraction of total suspended solids was determined to distinguish between manure and sediment. The affected water samples were dried at 60 °C, transferred to crucibles, ashed at 500 °C for 4 h in a muffle furnace and re-weighed (based on Hirota and Szyper, 1975). Water filtrates and separate aliquots of unfiltered water were usually frozen prior to analyses. Using flow injection (Lachat Instruments, Milwaukee, WI, USA) colorimetry, concentration of dissolved molybdate-reactive phosphate (DRP) (Diamond, 2000) was determined in filtered water, and total P (P_t) in unfiltered water by in-line digestion (Liao, 2001) if sediment concentration was less than 0.03 g L^{-1}. More turbid samples were heated in a block digester with persulfate and sulfuric acid for 3 h at 130 °C, then molybdate-reactive phosphate was determined by flow injection to provide the P_t measure. P_t is comprised of dissolved reactive + dissolved unreactive (or organic) + particulate organic + particulate inorganic P. Dissolved organic (unreactive) + particulate P was calculated by subtracting DRP from P_t, and is referred to as P_{dop}. This fraction was comprised mainly of particulate P in the selected water samples for which P_t was determined in filtrates.

Drainage tile monitoring began in mid-March, and runoff mid-May 2007 in all catchments except ND. Runoff monitoring began at ND in fall 2007 after the CP catchments were fall ploughed and ground cover became comparable to the other catchments. Sediment and nutrient loads were calculated for each increment of time between automated water sampling events at each outlet, using the product of the volume of flow since the previous sample and the measured concentration. Missing flows (e.g. float failure, ice) were estimated based on manual measurements when available; or using flow rates at adjacent outlets or interpolation over time. Loads were converted to a hectare basis using contributing areas of 480 m^2 (catchment size) for runoff, and 529, 548 and 542 m^2 for drainage tile in CP Bi, CP Hb and MT Bi catchments, respectively (due to varied length of tile drains from catchment edge to outfall) at Field A; and 519 m^2 for drainage tile at Field B (6.1 m systematic drainage tile spacing × 85 m tile drain run), ignoring possible additional area contributing to the lines with surface inlets. Growing- (GS) and non-growing season (NGS) flows and loads were calculated separately because of differing temporal trends in management effects and flow proportions to various pathways. Flows and loads were summed over the following time periods:

PT, 16 March–15 May 2007;
GS1, 16 May–24 October 2007;
NGS1, 25 October 2007–31 May 2008;
GS2, 1 June–15 November 2008;
NGS2, 16 November 2008–31 March 2009.

The pre-treatment (PT) sum provided background data, being prior to the differing manure applications and the manifest of tillage management effects. Several drainage tile flow events and one runoff event were captured during the PT time period. The start of NGS1 coincided with initiation of monitoring the ND catchment. Concentrations and loads of P in runoff water during GS2 are not presented because the samples were contaminated by glyphosate [N-(phosphonomethyl) glycine, $C_3H_8NO_5P$] herbicide applied to weeds growing around the gutters. Rains washed herbicide into the gutters, and runoff P concentrations increased markedly in samples collected following the herbicide application. Since runoff was minimal throughout GS2 (flows were 85, 8, 10 and 0.1 mm from ND, CP Bi, CP Hb and MT Bi catchments, respectively), the exclusion of GS2 runoff P load did not impact the outcome of the study. When manure moved to tile drains (GS1), organic suspended solids were subtracted from total solids to calculate inorganic suspended solids subsurface flow-weighted concentrations and loads, so that sediment movement would not be confounded with contributions from manure.

3. Results

3.1. P and sediment to surface water from runoff (Field A)

Although concentrations were often greater in GSs than NGSs, DRP and sediment loads in runoff were greater during NGSs than GSs (Figs. 2 and 3), as were runoff flow volumes (NGS, 145 ± 60 mm > GS, 7 ± 5 mm, average of three catchments containing drainage tile and two seasons, see Ball Coelho et al., under revision for detailed flow data). Runoff DRP load averaged 0.10 ± 0.07 kg ha^{-1} in NGS1 and 0.15 ± 0.11 kg ha^{-1} in NGS2 vs. 0.04 ± 0.04 kg ha^{-1} in GS1, average of the same three catchments. Sediment runoff load from these three catchments averaged 534 ± 788 kg ha^{-1} in NGS1 and 143 ± 225 kg ha^{-1} in NGS2 vs. 69 ± 48 kg ha^{-1} in GS1 and 5 ± 5 kg ha^{-1} in GS2. Runoff P_{dop} loads on the other hand did not follow this temporal trend, averaging 0.7 ± 1 kg ha^{-1} in NGS1, 0.4 ± 0.3 kg ha^{-1} in GS1, but only 0.08 ± 0.09 kg ha^{-1} in NGS2 (Fig. 2). The preceding runoff P_{dop} load averages from catchments with artificial drainage were less than from the catchment with natural drainage, where P_{dop} loads were 2.8 kg ha^{-1} in NGS1 and 0.13 kg ha^{-1} in NGS2.

Similarly to P_{dop}, sediment concentrations and loads in runoff were generally less from areas with drainage tile than from the ND catchment (Fig. 3). Runoff sediment load from ND in NGS1 was 19-fold that of areas where subsurface drainage was installed. Most movement occurred during the largest sediment load event on 8–9 January 2008 with 50 mm rain on top of about 0.3 m of melting snow. Estimated load from this event might have been associated with errors in measurement due to flooding, so NGS1 loads were summed excluding this rain-on-snow event for comparison. Runoff sediment load from ND (570 kg ha^{-1}) was greater than from areas with tile drains (83 kg ha^{-1}, average of the other 3 catchments) with the uncertain data excluded. Erosion rates as estimated from ND with the 8–9 January event included (10 Mg ha^{-1}) are not uncommon in Southern Ontario, Canada (Lobb and Kachanoski, 1999; Wall et al., 1991), and with the 8–9 January 2008 load excluded, erosion was within the range of estimates from flat Brookston clay in Ontario of 400–900 kg ha^{-1} (Culley and Bolton, 1983; Gaynor and Bissonnette, 1992). Runoff P loads over the 2 day snowmelt averaged (four catchments) 1 kg P_{dop} ha^{-1} (83% of the NGS1 load) and 0.02 kg DRP ha^{-1} (21% of the NGS1 load).

An important runoff event during the GS occurred on 19 June 2007 with 20 mm rain 4 weeks after LSM application. On this day runoff loads from CP (0.47 ± 0.096 kg P_{dop} ha^{-1} and 0.016 ± 0.005 kg DRP ha^{-1}) were greater than that from MT (0.09 kg P_{dop} ha^{-1} and 0.004 kg DRP ha^{-1}). Runoff P_{dop} concentrations from CP Hb (12.5 mg L^{-1}) and CP Bi (11.4 mg L^{-1}) were at their maximum on 19 June 2007, and were probably associated with the broadcast manure application method. Between May 2007 and March 2009, runoff P_{dop} and sediment loads, respectively, averaged greater from CP (1.6 ± 1.5 and 1036 ± 1249 kg ha^{-1}) than from MT (0.2 and 183 kg ha^{-1}); albeit with large variability between the two CP catchments (Figs. 2 and 3). Management effects on runoff DRP load were different, averaging less from CP catchments containing drainage tile (0.2 ± 0.04 kg ha^{-1}) than from both the MT catchment (with drainage tile) and the CP ND catchment (both 0.4 kg ha^{-1}, $\sum$(GS1 + NGS1 + NGS2)) (Fig. 2).

3.2. P and sediment to surface water from tile drains (Fields A and B)

As with runoff, the dominant temporal trend from drainage tile was that both P and sediment loading were greater in NGSs than

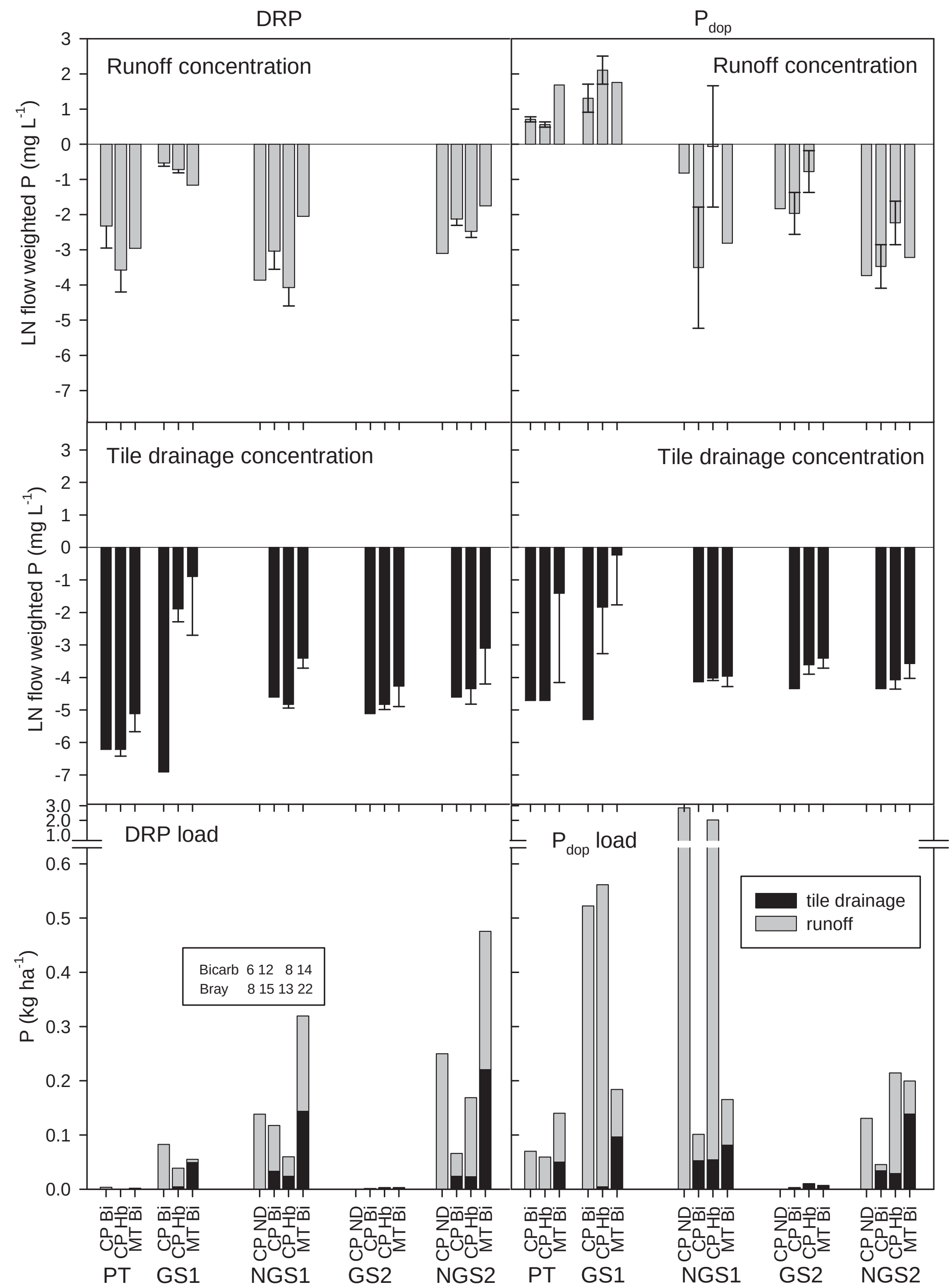


Fig. 2. Natural logarithm (LN) of flow-weighted concentrations, and loads of DRP (left column) and P_{dop} (dissolved organic P + particulate) (right column) from Field A summed over pre-treatment (PT) and two growing season (GS) and non growing season (NGS) time periods in runoff and tile drainage water with blind (Bi) or hickenbottom (Hb) surface inlets or natural drainage (ND), under common practice (CP) or minimum tillage (MT). Excluded: runoff P in GS2 due to contamination by glyphosate. Error bars for runoff are from CP Bi and CP Hb, and for tile drains are from the comparable management and drainage system at Field B. Values of soil sodium bicarbonate- (Bicarb) and Bray-extractable P (mg kg^{-1}) (Olsen and Sommers, 1982) in the top 0.2 m of soil determined in April 2007 are aligned with their respective catchments.

in GSs (Figs. 2–5). Similarly, flow volumes were greater in NGSs (292 ± 28 at Field A, 182 ± 10 mm at Field B) than in GSs (15 ± 13 at Field A, 19 ± 20 mm at Field B, averages of two seasons). Contributions from the 2-day snowmelt event of 8–9 January 2008 at Field A were important for P_{dop} (0.02 kg ha^{-1}, 28% of the NGS1 drainage tile load) and sediment (3.5 kg ha^{-1}, 20% of the NGS1 drainage tile load), but not DRP (0.003 kg ha^{-1}, 4%). At Field B, losses were 0.006 kg P_{dop} ha^{-1}, 1.1 kg sediment ha^{-1} and 0.0011 kg DRP ha^{-1}, representing 12, 13 and 5% of NGS1 drainage tile P_{dop}, sediment and DRP loads, respectively.

During NGSs, presence or type of surface inlets did not have much effect on P and sediment load to tile drains (Figs. 2–5). Greatest P and sediment load in NGS1 at Field B for example, was through a drainage tile line with no surface inlet (MT Ni, Figs. 4 and 5). During GS1, there was an interaction between the presence of a surface inlet and manure management. Greatest P and sediment load at

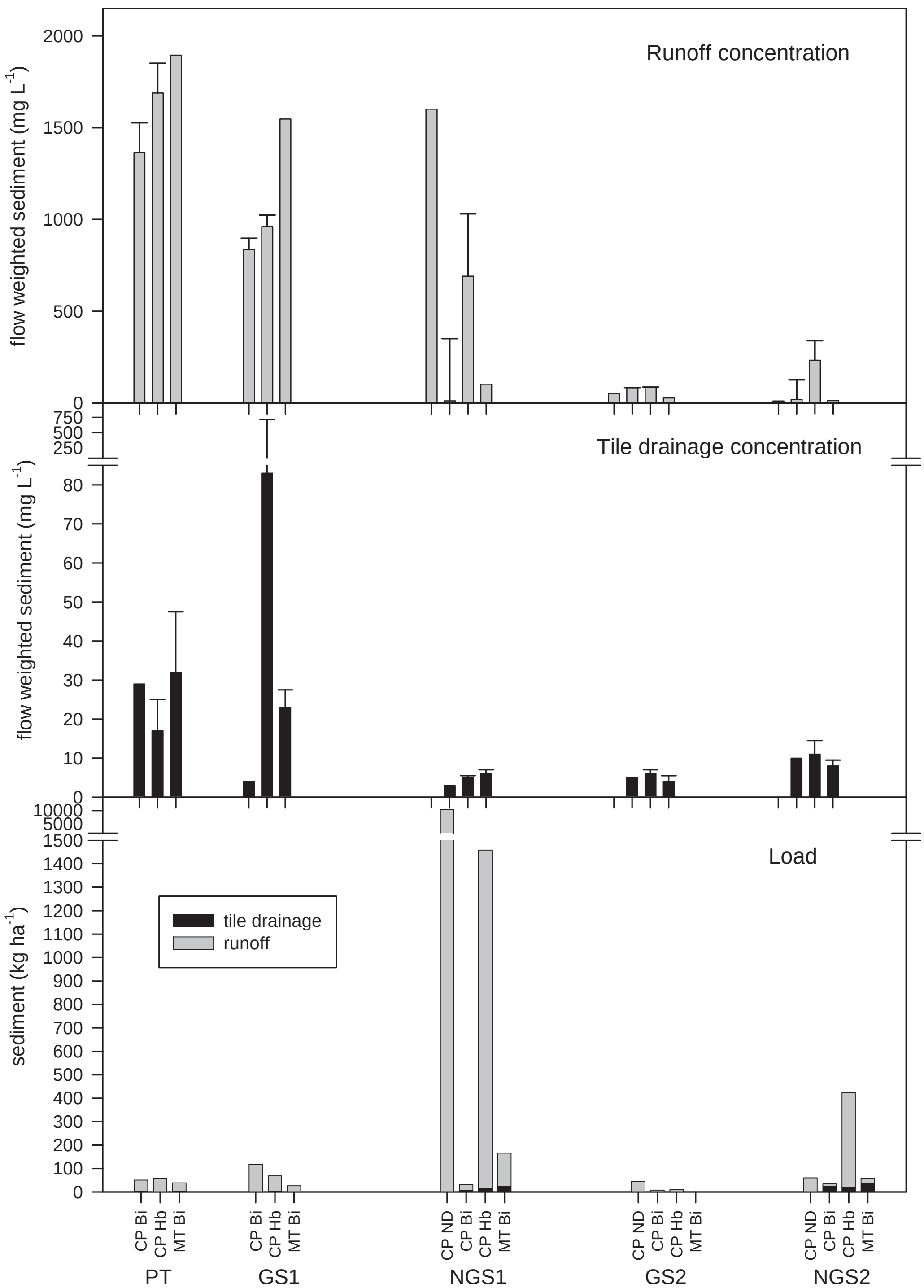


Fig. 3. Flow-weighted concentrations, and loads of sediment from Field A summed over pre-treatment (PT) and two growing season (GS) and non growing season (NGS) time periods in runoff and tile drainage water with blind (Bi) or hickenbottom (Hb) surface inlets or natural drainage (ND) under common practice (CP) or minimum tillage (MT). Excluded: suspended organic solids in GS1 when manure moved into drainage tile. Error bars for runoff are from CP Bi and CP Hb and for tile drains are from the comparable management and drainage system at Field B.

Field B occurred where there was a surface inlet and where LSM was broadcast incorporated (CP Hb). The rain-induced movement 4 weeks after manure application contributed 0.0029 kg DRP ha^{-1}, 0.024 kg P_{dop} ha^{-1}, 11.7 kg ha^{-1} inorganic solids and 2.3 kg ha^{-1} organic suspended solids on 19 June 2007, and comprised most of the load from CP Hb outlet at Field B during GS1. Loads from the other Field B outlets were insubstantial in GS1 (Figs. 4 and 5). From the comparable outlet (CP Hb) at Field A, loads during this rain event were 0.0048 kg DRP ha^{-1}, 0.0035 kg P_{dop} ha^{-1} and 1.6 kg suspended solids ha^{-1}. The suspended solids from the CP Hb outlet at Field B were comprised of 16% organic matter, and concentrations increased to 0.42 mg DRP L^{-1}, 3.9 mg P_{dop} L^{-1} and 1831 mg sediment L^{-1}. From the CP Hb outlet at Field A concentrations increased to 1.2 mg DRP L^{-1}, 1.4 mg P_{dop} L^{-1}, and 548 mg sediment L^{-1}.

At the time of application, LSM did not move to drainage tile at Field B. At Field A however, application-induced movement to tile drains occurred in GS1 where LSM was injected (MT Bi

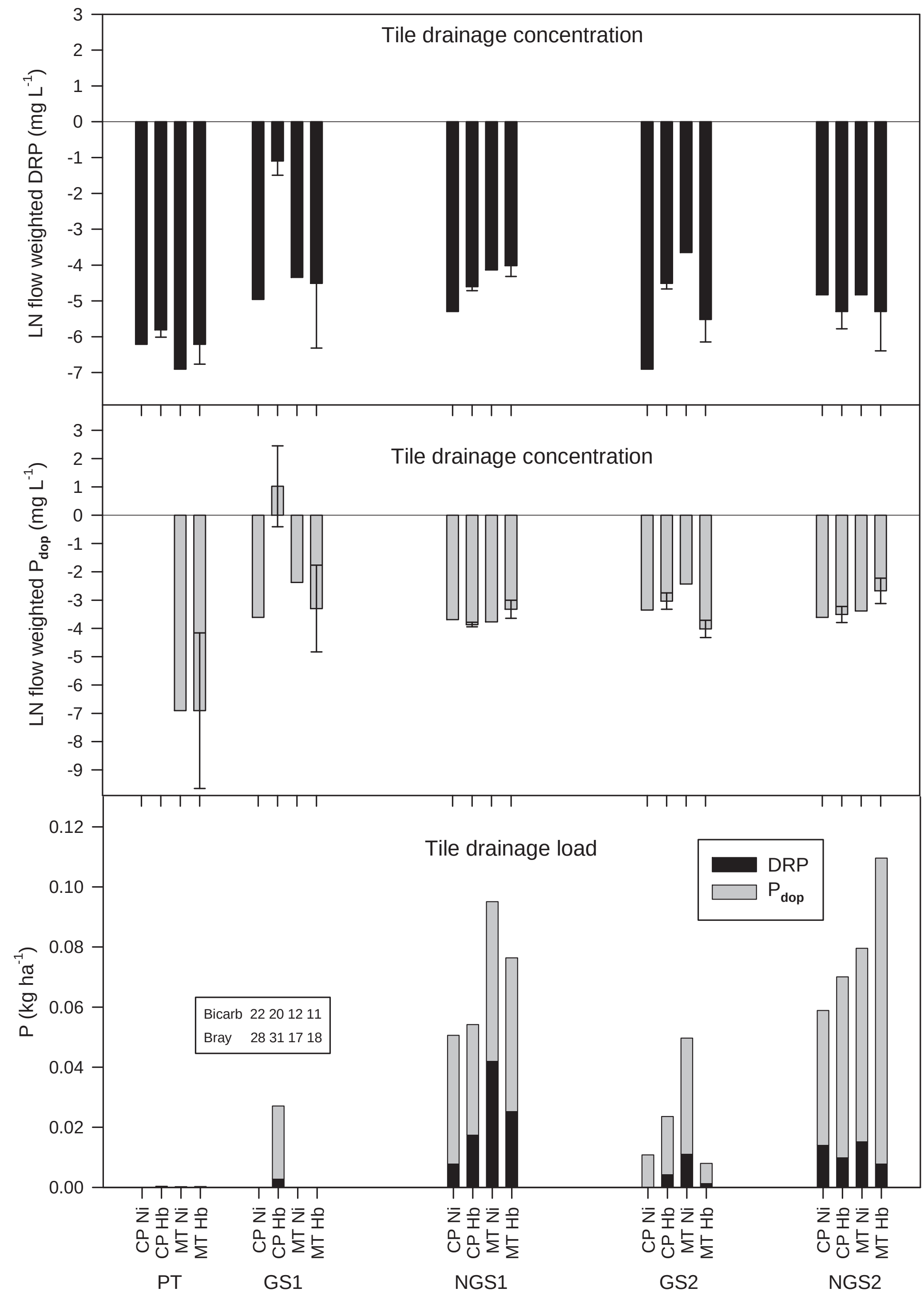


Fig. 4. Natural logarithm (LN) of flow-weighted concentrations, and loads of DRP and P_{dop} from Field B summed over pre-treatment (PT) and two growing season (GS) and non growing season (NGS) time periods in tile drainage water with hickenbottom (Hb) or no surface inlet (Ni) under common practice (CP) or minimum tillage (MT). Error bars are from the comparable management and drainage system at Field A. Values of soil bicarbonate- (Bicarb) and Bray-extractable P (mg kg^{-1}) in the top 0.2 m of soil determined in April 2007 are aligned with their respective catchments.

catchment). Spikes in DRP and P_{dop} concentrations contributed to greater flow-weighted P concentrations from MT Bi than from other outlets in GS1 (Fig. 2). The drainage water from the affected tile also had elevated suspended solids, which were comprised of 40–50% organic matter on the days of and immediately following application. The tile drainage water composition was similar to that of the applied LSM, which contained 64% organic solids. Injected manure that moved to drainage tile contributed loads of 0.033 kg DRP ha^{-1}, 0.087 kg P_{dop} ha^{-1}, 1.0 kg ha^{-1} inorganic solids and 0.1 kg ha^{-1} organic suspended solids during 22–29 May 2007. In GS2, there was no application-induced movement of LSM to drainage tile, and no rain-induced movement to surface inlets in the days and weeks following application (rainfall ≤ 15 mm d^{-1}) at either field. Topsoil water content was near saturation at the time

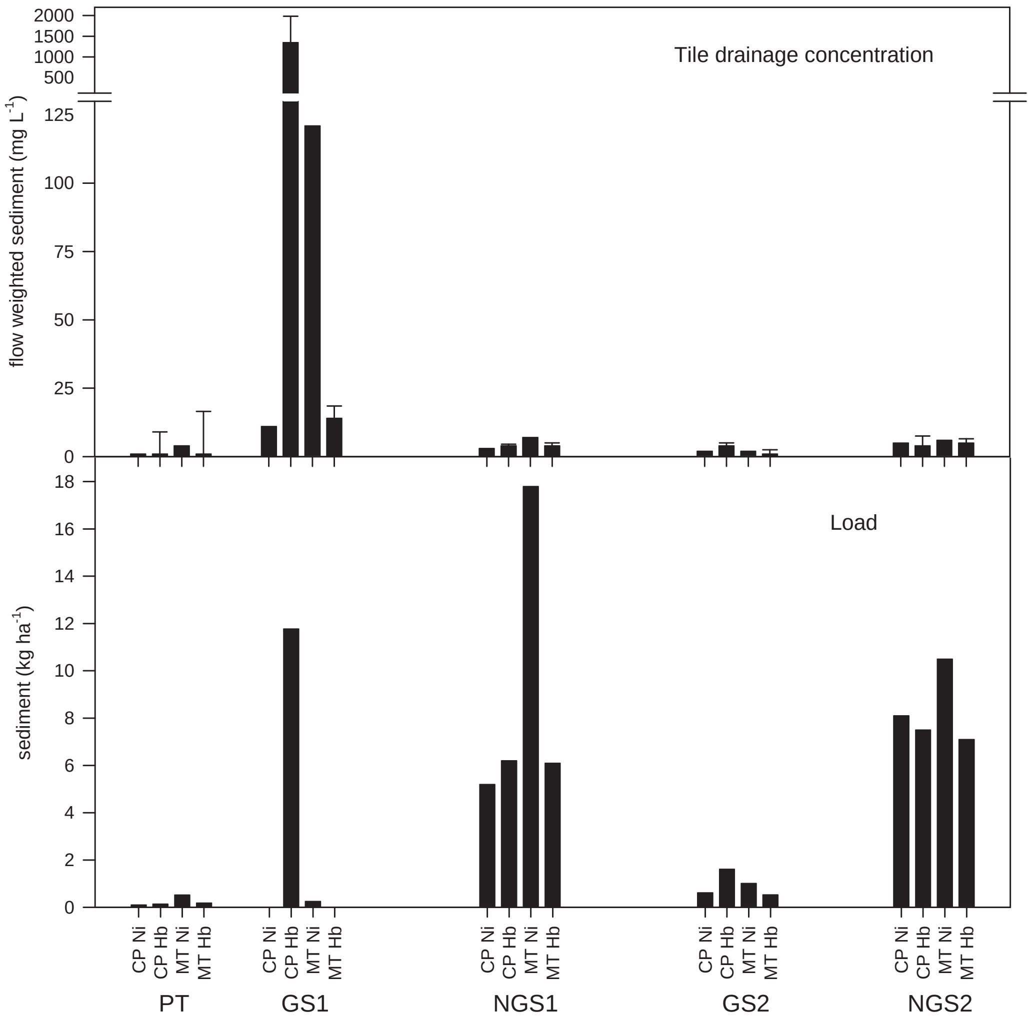


Fig. 5. Flow-weighted concentrations, and loads of sediment from Field B summed over pre-treatment (PT) and two growing season (GS) and non growing season (NGS) time periods in tile drainage water with hickenbottom (Hb) or no surface inlet (Ni) under common practice (CP) or minimum tillage (MT). Error bars are from the comparable management and drainage system at Field A.

of LSM application in both years (Ball Coelho et al., under revision), and the tile drains were flowing at the time of application at Field A, but not at Field B in both years.

While greater DRP load from MT than from other outlets at Field A during GS1 was associated with preferential movement of manure to the tile, the trend also occurred during NGS1 and NGS2 (Fig. 2). It was not likely tillage-related because during PT greater DRP load to the tile drain from MT (0.0012 kg ha^{-1}) than from CP (0.0002 ± 0.000014 kg ha^{-1}) also occurred at Field A, and greater DRP load to tile with MT than CP did not occur at Field B. The trend at Field A may have been caused by greater flow from the MT than other outlets (Ball Coelho et al., under revision), greater atmospheric deposition, or greater available soil P in the MT than other catchments at Field A (Fig. 2). When measured from a 0.3 m pack in January 2009, snow had greater P concentration in MT (0.059 ± 0.014 mg DRP L^{-1}) than in the other catchments at Field A (0.011 ± 0.014 mg L^{-1}) and Field B (0.010 ± 0.0005 mg L^{-1}). This P of unknown origin would add about 0.06 kg P ha^{-1} cumulatively over three equivalent depth snow packs.

Variation in Bray-extractable P within Field B did not correlate with drainage tile DRP load (Fig. 4). Soils were not P-saturated with respect to environmental or agronomic indicators. Soil P values were less than the proposed threshold 'change point' of 60 mg kg^{-1} above which P moves to drainage tile and below which tile water concentrations remain less than 0.3 mg L^{-1} (Heckrath et al., 1995). Soil P values were also less than the value of sodium bicarbonate extractable P (30 mg kg^{-1}) associated with rare probability of a profitable response to P application by field grains such as corn and soybeans (OMAFRA, 2009).

3.3. *Partitioning of load to surface water (Field A)*

The proportion of the total load (overland + subsurface) that moved through tile drains over the 2-year study was 31 ± 16% of DRP (21 ± 1% in CP, 49% in MT); 24 ± 25% of P_t (10 ± 7% in CP, 52% in MT); and 16 ± 13% of sediment (average of three catchments with artificial drainage, GS2 excluded for P). Sediment load averaged 796 ± 1021 kg ha^{-1} overland vs. 54 ± 18 kg ha^{-1} through drainage tile over the 2 years. Only CP Bi in NGS2 had greater sediment load through tile drains than overland, and runoff flow was small from this catchment. Runoff flows were notably sensitive to minor variations in catchment slopes.

Net effects of artificial drainage were reduced loading of: P_t (1.3 ± 1.1 kg ha^{-1}, overland + subsurface < 3.4 kg ha^{-1} ND, overland); P_{dop} (1.2 ± 1.5 kg ha^{-1} CP, 0.4 kg ha^{-1} MT, overland + subsurface < 3.0 kg ha^{-1} ND, overland); and sediment (973 ± 1285 kg ha^{-1} CP, 224 kg ha^{-1} MT, overland + subsurface < 10,376 kg ha^{-1} ND, overland). Total DRP load was similar with or without tile drains, on average (0.4 kg ha^{-1}), although greater under MT (0.8 kg ha^{-1}) and less under CP (0.2 ± 0.03 kg ha^{-1}, overland + subsurface) than under ND for $\sum$(NGS1 + NGS2). Total sediment movement was reduced by the presence of drainage tile to 1/19th in NGS1 and 1/7th in GS2. Sediment movement was not reduced by artificial drainage in NGS2, when the CP catchments were not fall-ploughed (in 2008) and total sediment load averaged only 172 ± 218 kg ha^{-1} from the three catchments containing drainage tile and 59 kg ha^{-1} without drainage tile.

4. Discussion

During rainfall events, erosion is the result of both detachment by raindrops and transport by runoff, whereas during snowmelt, flowing water leads to erosion (Ginting et al., 2000). These processes explain the observed trends of elevated P and sediment concentrations following summer storms and greater flow-weighted concentrations in summer than in winter runoff in most cases (Figs. 2 and 3). Summer rains likewise increased concentrations of P and sediment in drainage tile water. Particulate P transport in the first flush of drainage water following dry weather has been attributed to the washing into drainage tile (with no surface inlets) of fine material from the sides of cracks, fissures and earthworm burrows (Simard et al., 2000).

While sediment and P concentrations were greatest during the GS, loading was greater over winter (Figs. 2–5) due to more flow. These temporal trends have also been observed: on the Canadian prairies for overland transport (Tiessen et al., 2010); in lacustrine landscapes in MN, USA for movement through inlets, with 40% attributed to snowmelt and the rest due to rainfall (Ginting et al., 2000); and in both forested and grain-sown watersheds in Spain for runoff dissolved phosphate and sediment loading, despite most erosive rainfalls occurring during summer (Casalí et al., 2010). In Spain, the effect was attributed to less forest and cereal canopy cover in winter to reduce raindrop impact. With respect to our study, the effect can also be attributed to reduced infiltration in frozen soil, resulting in rapid overland flows. Ollesch et al. (2006) predicted that erosion with partly frozen soil would be 50-fold more than with unfrozen soil, from a German catchment with winter cereals and winter rape. In WI, USA, organic matter enrichment was greater in snowmelt than rainfall runoff, and was attributed to low flow rates over frozen surfaces (Panuska and Karthikeyan, 2010). From the prairie system, dissolved losses of P exceeded particulate runoff losses (Tiessen et al., 2010), which was not always the case in our study. Differing processes in more humid regions such as Ontario, Canada, are likely due to higher flows over frozen surfaces, and greater frequency of snowmelt in combination with rain events than in semi-arid Canada or US Midwest, where most runoff is from 'lower flow' snowmelt. Given that loading was dominated by a few events, long term data collection is important to accrue information about critical weather conditions, such as rain on frozen ground, and snowmelts.

Based on trends in sediment loading in OH, USA rivers, Richards et al. (2008) suggested that while agricultural best management practices have been successful in reducing erosion overall, better management in winter and spring is an area for future improvement. Poor control of loading over winter and early spring may be due in part to the ineffectiveness of saturated (Liu et al., 2008) or frozen buffers. To control movement over frozen ground, Tiessen et al. (2010) recommended minimizing nutrients at the surface. Use of tine applicators places less manure nutrients at the surface as compared with broadcast application, while reducing the risk of preferential flow to tile drains (Lapen et al., 2008). Reducing nutrients at the surface might also be accomplished by injection as opposed to broadcasting manure. To avoid possible preferential flow to drainage tile in susceptible systems (macro-porous soils), the practice of injection may be accompanied by precautionary measures such as careful attention to rate and style of equipment, or increased viscosity of applied material. The avoidance of application over newly installed tile drains was indicated by the present study.

Application- (or rain-) induced movement did not occur after soil over the tile drains had settled for 1 year. Incidental movement to drainage tile at Field A (not previously cultivated), but not at Field B (having had long-term cropping history), may have resulted from differences in soil structure between the two fields. Contaminated flow from injected manure in a Perth County, Ontario study which included above optimal application rates, contributed a maximum of 0.05 kg DRP ha^{-1} into tile drains that were well-settled and had no surface inlets (Ball Coelho et al., 2007). This amount is similar to the incidental load of 0.03 kg DRP ha^{-1} with injected manure in the present study.

While injected manure moved to drainage tile in the year of installation at Field A (at application), the contribution to total load was minor in proportion to annual (GS1 + NGS1) total (overland + subsurface) load of suspended solids (0.6%), and was 9% of the DRP and 25% of the P_{dop} from the MT Bi catchment. The preferential movement to the drainage tile amount was unimportant relative to the total loading from the CP system. Relative to CP, the combination of injected manure with minimum tillage, over 2 years, reduced runoff load of sediment by 6-fold and of P_{dop} by 8-fold, and increased DRP by 2-fold, resulting in a net P_t reduction. No-till usually reduces overland dissolved- and total-P movement (Andraski et al., 2003). However, dissolved P can sometimes be greater from no-till, as a result of high near-surface nutrient content (Gilley et al., 2008), for example from surface-applied fertilizer or freezing of plant tissue on the surface (Tiessen et al., 2010). As with P transport overland, minimum tillage usually reduces erosion. Shipitalo et al. (2010) reported that a no-till watershed had an average sediment concentration of approximately 1/5th that of a tilled watershed. Tiessen et al. (2010) also found less overland sediment transport with conservation than conventional tillage, from both rainfall and snowmelt events.

Preferential movement to tile drains at time of application did not occur where manure was surface applied. Surface application did however, result in rainfall-induced movement to drainage tile with a surface inlet 4 weeks after application in GS1, contributing 0.02 kg P_{dop} ha^{-1} and 0.003 kg DRP ha^{-1}. This amount was small as compared with the amount that moved in runoff with rain on the recently broadcasted manure, which was 0.38 kg P_{dop} ha^{-1} and 0.012 kg DRP ha^{-1} (calculated as the P runoff from CP catchments less that from the MT catchment with injected manure on 19 June 2007). A rainfall simulation study in soil of similar texture and steeper slope also demonstrated greater overland movement from surface-applied than injected amendment following application, even with immediate incorporation of the broadcast material (Topp et al., 2008). Combined, our two studies support consideration of injection as a separate category when recommending minimum separation distances from surface water, with injection requiring less setback than surface-applied incorporated material. The data also support a requirement of a specified setback distance from surface inlets when manure is surface-applied, even when incorporated within 24 h.

Other than the rain-induced movement of broadcast manure, surface inlets did not have much effect on loading. The minimal inlet effect we observed was similar to findings from MN, USA where subsurface drainage water quality was similar from fields (flat landscapes) with or without surface inlets (Ginting et al., 2000). Ponding at the inlet allows entrained particles to settle, thus limiting sediment and sorbed P transport to surface waters (Thoma et al., 2005). Movement of solids through drains with surface inlets varied as much as 1–125 $kg\,ha^{-1}\,yr^{-1}$, the average being 46 $kg\,ha^{-1}\,yr^{-1}$ during their 3-year study (Ginting et al., 2000). Wide variability across seasons and catchments was similarly observed in our study. Loads from both drainage tile and runoff varied more with flow volume (i.e., landscape variability) than with type of drainage or management. Others have noted the over-riding effect of flow on loading. Flow volume had a larger influence on P loads than management (grazing intensity, P rate) in Australian pasture (Melland et al., 2008). Kleinman et al. (2006) noted that site hydrology rather than chemistry was primarily responsible for variations in mass N and P losses by runoff in a NE, USA watershed. Given our painstaking efforts to equalize catchment characteristics, the sensitivity of P losses to landscape (e.g., greater runoff P load from catchments having larger runoff volumes) along with the randomness of flow complicates the interpretation of management effects on loading.

The recommended limit of 0.035 mg $P_t\,L^{-1}$ for preventing eutrophication (CCME, 2004) was exceeded in runoff seasonal flow-weighted concentrations in most cases for DRP and all cases for P_t except MT Bi in GS2. Runoff flow-weighted DRP exceeded about 10-fold that of tile drainage water (Fig. 2). Watson et al. (2007) also found DRP and P_t concentrations greater in overland flow than in drainage water from grazed grassland. In tile drainage water, flow-weighted concentrations exceeded the limit for eutrophication in a few cases for DRP (CP Hb at Fields A and B in GS1; MT Bi at Field A in GS1 and NGS2, Figs. 2 and 4) and more frequently for P_t. Runoff seasonal flow-weighted sediment concentrations exceeded the aquatic limit of 46 $mg\,L^{-1}$ (Vondracek et al., 2003) as well as that in tile drainage water in most cases (Fig. 3). Tile drainage water flow-weighted sediment concentrations were less than the aquatic maximum allowable limit with three exceptions in GS1 (Figs. 3 and 5).

Fifteen-fold more sediment moved overland than through tile drains over 2 years (catchments where drainage was installed). From systems having greater slope, the proportion of total loading from overland runoff would be greater than what we observed from a minor slope of 1%. In watersheds that are extensively tile drained, the contributing area from drainage tile in proportion to runoff would be greater than in our study, which was approximately 1:1. This would increase the relative importance of partitioning through tile drains on scaling up. In a short-term study in Quebec, Canada, sediment partitioning was also greater overland than through drains (75 vs. 31 $kg\,ha^{-1}\,yr^{-1}$, Jamieson et al., 2003). Watson et al. (2007) measured similar P_t loads in drainage water (0.1–1.5 $kg\,ha^{-1}\,yr^{-1}$) and overland from grazed systems in Ireland, but their soil had unusually high organic matter (0.12 $g\,g^{-1}$). The relative importance of the drainage tile pathway with respect to the overall P budget also depends on whether DRP or P_t is the criterion. Drainage tile contributed more important proportions of DRP total overland + subsurface load than of P_t or sediment over the monitored time scale. While DRP is more available to aquatic organisms in the short term, P_t cycles into algal-available forms over time (Sharpley et al., 2000), and therefore partitioning between DRP and P_{dop} at field exit may be a mute point.

The net effects of artificial drainage (investigated by comparing loads from catchments with and without drainage tile) were decreased loads of P_t and P_{dop} (2.5-fold) and sediment (10-fold), no change to DRP, and increased flow and nitrate transport (Ball Coelho et al., under revision) (Field A). Eastman et al. (2010) reported that subsurface drainage controlled P transport in sandy loam but not in a clay loam site. Our results from intermediate soil texture (loam) are in agreement with the following studies: tile drainage reduced P transport by 30% as compared to naturally drained grassland in England (Simard et al., 2000); tile drained areas generated more total flow and less sediment load than naturally drained areas (Richards et al., 2008); losses of P_t (only 6% of which was via drainage tile) and erosion were reduced with tile drainage (5 and 3482 $kg\,ha^{-1}$, respectively) vs. natural drainage (8 and 4986 $kg\,ha^{-1}$) from clay loam in LA, USA (Sims et al., 1998).

5. Conclusions

- Significant transport of sediment and P in both overland runoff and tile drainage occurred during snowmelt events.
- Since much of the annual P and sediment load occurs during winter, it is necessary to address how to manage for snowmelt events when buffers do not work.
- Runoff was an important pathway for P loading.
- Tile drains contributed a substantial proportion of the DRP, but less of the P_t and sediment total load for $\sum$(overland + subsurface).
- Surface inlets did not have a major effect on loading.
- Artificial drainage as compared with natural drainage reduced P_{dop} and sediment, but not DRP loading.
- Minimum tillage system with injected manure as compared with conventional tillage and broadcast manure reduced overland runoff P_{dop} and sediment but not DRP load.

Acknowledgements

Financial support was provided by the Ontario Ministry of Agriculture Food and Rural Affairs Nutrient Management Research Program, and the Interdepartmental Recovery Fund for Species at Risk. Sponsors had no role in experimental design, report writing or decision to submit this paper.

References

Andraski, T.W., Bundy, L.G., Kilian, K.C., 2003. Manure history and long-term tillage effects on soil properties and P losses in runoff. J. Environ. Qual. 32, 1782–1789.
Ball Coelho, B., Bruin, A.J., Staton, S., Hayman, D., 2010. Sediment and nutrient contributions from subsurface drains and point sources to an agricultural watershed. Air Soil Water 3, 1–21.
Ball Coelho, B., Lapen, D.R., Murray, R., Topp, E., Bruin, A.J., Khan, B. Nitrogen loading of offsite waters from liquid swine manure application under different drainage and tillage practices. Agric. Water Manage, under revision.
Ball Coelho, B.R., Roy, R.C., Topp, E., Lapen, D.R., 2007. Tile water quality following liquid swine manure application into standing corn. J. Environ. Qual. 36, 580–587.
CCME (Canadian Council of Ministers of the Environment), 2004. Canadian water quality guidelines for the protection of aquatic life. Phosphorus, in: Canadian environmental quality guidelines, Winnipeg. pp. 8. Available from: http://ceqg-rcqe.ccme.ca (accessed July 2009).
Casalí, J., Giménez, R., Díez, J., Álvarez-Mozos, J., Del Valle de Lersundi, J., Goñi, M., Campo, M.A., Chahor, Y., Gastesi, R., López, J., 2010. Sediment production and water quality of watersheds with contrasting land use in Navarre (Spain). Agric. Water Manage. 97, 1683–1694.
Culley, J.L.B., Bolton, E.F., 1983. Suspended solids and phosphorus loads from a clay soil: II. Watershed study. J. Environ. Qual. 12, 498–503.
Diamond, D., 2000. QuikChem Method 10-115-01-1-A. Determination of Ortho-Phosphate in Waters by Flow Injection Analysis Colorimetry. Lachat Instruments, Milwaukee, WI.
Eastman, M., Gollamudi, A., Stämpfli, N., Madramootoo, C.A., Sarangi, A., 2010. Comparative evaluation of phosphorus losses from subsurface and naturally drained agricultural fields in the Pike River watershed of Quebec, Canada. Agric. Water Manage. 97, 596–604.
Forster, D.L., Rausch, J.N., 2002. Evaluating agricultural nonpoint-source pollution programs in two Lake Erie tributaries. J. Environ. Qual. 31, 24–31.
Gaynor, J., Bissonnette D., 1992. The effect of conservation tillage practices on the losses of phosphorus and herbicides in surface and subsurface drainage waters. SWEEP Report No. 60.
Gaynor, J.D., Findlay, W.I., 1995. Soil and phosphorus loss from conservation and conventional tillage in corn production. J. Environ. Qual. 24, 734–741.
Gentry, L.E., David, M.B., Royer, T.V., Mitchell, C.A., Starks, K.M., 2007. Phosphorus transport pathways to streams in tile-drained agricultural watersheds. J. Environ. Qual. 36, 408–415.
Gilley, J.E., Sabatka, W.F., Eghball, B., Marx, D.B., 2008. Nutrient transport as affected by rate of overland flow. Trans. ASABE 51, 1287–1293.
Ginting, D., Moncrief, J.F., Gupta, S.C., 2000. Runoff, solids, and contaminant losses into surface tile inlets draining lacustrine depressions. J. Environ. Qual. 29, 551–560.

Heckrath, G., Brookes, P., Poulton, P., Goulding, K., 1995. Phosphorus leaching from soils containing different phosphorus concentrations in the Broadbalk experiment. J. Environ. Qual. 24, 904–910.

Hirota, J., Szyper, J.P., 1975. Separation of total particulate carbon into inorganic and organic components. Limnol. Oceanogr. 20 (5), 896–900.

Inamdar, S.P., Mostaghimi, S., McClellan, P.W., Brannan, K.M., 2001. BMP impacts on sediment and nutrient yields from an agricultural watershed in the coastal plain region. Trans. ASAE 44 (5), 1191–1200.

Jamieson, A., Madramootoo, C.A., Enright, P., 2003. Phosphorus losses in surface and subsurface runoff from a snowmelt event on an agricultural field in Quebec. Can. Biosyst. Eng. 45, 1.1–1.7.

Kleinman, P.J.A., Srinivasan, M.S., Dell, C.J., Schmidt, J.P., Sharpley, A.N., Bryant, R.B., 2006. Role of rainfall intensity and hydrology in nutrient transport via surface runoff. J. Environ. Qual. 35, 1248–1259.

Lapen, D.R., Topp, E., Edwards, M., Sabourin, L., Curnoe, W.E., Gottschall, N., Bolton, P., Rahman, S., Ball Coelho, B.R., Payne, M., Kleywegt, S., McLaughlin, N.B., 2008. Effect of liquid municipal biosolid application method on tile and ground water quality. J. Environ. Qual. 37, 925–936.

Lazzarotto, P., Prasuhn, V., Stamm, C., 2007. Spatial predictions of P losses from soil and manure and monitoring data in a small agricultural catchment point to soil P as the main source. In: Proceedings of the 5th International Phosphorus Workshop (IPW5). Diffuse Phosphorus Loss: Risk Assessment, Mitigation Options and Ecological Effects in River Basins, Silkeborg, Denmark, 3–7 September, pp. 247–250.

Liao, N., 2001. QuikChem Method 10-115-01-3-A. Determination of Phosphorus by FIA Colorimetry (In-line Persulfate Digestion Method). Lachat Instruments, Milwaukee, WI, pp. 18.

Liu, X., Zhang, X., Zhang, M., 2008. Major factors influencing the efficacy of vegetated buffers on sediment trapping: a review and analysis. J. Environ. Qual. 37, 1667–1674.

Lobb, D.A., Kachanoski, R.G., 1999. Modelling tillage erosion in the topographically complex landscapes of southwestern Ontario, Canada. Soil Till. Res. 51, 261–277.

Melland, A.R., McCaskill, M.R., White, R.E., Chapman, D.F., 2008. Loss of phosphorus and nitrogen in runoff and subsurface drainage from high and low input pastures grazed by sheep in southern Australia. Aust. J. Soil Res. 46, 161–172.

NRCS-Ohio (Natural Resources Conservation Service-Ohio), 2003. Conservation Practice Standard. Waste Utilization. Code 633. June. Section IV, FOTG, pp. 1–14.

Ollesch, G., Kistner, I., Meissner, R., Lindenschmidt, K.E., 2006. Modelling of snowmelt erosion and sediment yield in a small low-mountain catchment in Germany. Catena 68, 161–176.

Olsen, S.R., Sommers, L.E., 1982. Phosphorus. In: Page, A.L., Miller, R.H., Keeney, E.R. (Eds.), Methods of Soil Analysis Part 2, vol. 9. Am. Soc. Agron. Soil Sci. Soc. Am., Madison, WI, pp. 403–430.

OMAFRA (Ontario Ministry of Agriculture, Food and Rural Affairs), 2009. Agronomy Guide for Field Crops–Publication 811. Queen's Printer for Ontario, Toronto.

Oygarden, L., Kvaerner, J.K., Jenssen, P.D., 1997. Soil erosion via preferential flow to drainage systems in clay soils. Geoderma 76, 65–86.

Panuska, J.C., Karthikeyan, K.G., 2010. Phosphorus and organic matter enrichment in snowmelt and rainfall – runoff from three corn management systems. Geoderma 154, 253–260.

Richards, R.P., Baker, D.B., Crumrine, J.P., Kramer, J.W., Ewing, D.E., Merryfield, B.J., 2008. Thirty-year trends in suspended sediment in seven Lake Erie tributaries. J. Environ. Qual. 37, 1894–1908.

Richards, R.P., Loftus, T.T., Czajkowski, K.P., 2005. Ohio rock creek land use and water quality trends in rock creek watershed (Seneca Co., OH) 1982–2005: impact of core-4 conservation practices. Project Report, Heidelberg College. USDA Research, Education and Economics Information System. Available from: http://www.reeis.usda.gov/web/crisprojectpages/201638.html (accessed 19.11.09.).

Schelde, K., de Jonge, L.W., Kjaergaard, C., Laegdsmand, M., Rubaek, G.H., 2006. Effects of manure application and plowing on transport of colloids and phosphorus to tile drains. Vadose Zone J. 5 (1), 445–458.

Schilling, K.E., Helmers, M., 2008. Tile drainage as karst: conduit flow and diffuse flow in a tile-drained watershed. J. Hydrol. 349 (3-4), 291–301.

Sharpley, A.N., Foy, B., Withers, P., 2000. Practical and innovative measures for the control of agricultural phosphorus losses to water: an overview. J. Environ. Qual. 29, 1–9.

Shipitalo, M.J., Bonta, J.V., Dayton, E.A., Owens, L.B., 2010. Impact of grassed waterways and compost filter socks on the quality of surface runoff from corn fields. J. Environ. Qual. 39, 1009–1018.

Simard, R.R., Beauchemin, S., Haygarth, P.M., 2000. Potential for preferential pathways of phosphorus transport. J. Environ. Qual. 29, 97–105.

Sims, J.T., Simard, R.R., Joern, B.C., 1998. Phosphorus loss in agricultural drainage: historical perspective and current research. J. Environ. Qual. 27, 277–293.

Staton, S.K., Dextrase, A., Metcalfe-Smith, J.L., Di Majo, J., Nelson, M., Parish, J., Kilgour, B., Holm, E., 2003. Status and trends of Ontario's Sydenham River ecosystem in relation to aquatic species at risk. Environ. Monit. Assess. 88, 283–310.

Stone, M., Krishnapan, B.G., 2002. The effect of irrigation on tile sediment transport in headwater stream. Water Resour. 36, 3439–3448.

Syversen, N., 2002. Effect of a cold-climate buffer zone on minimizing diffuse pollution from agriculture. Water Sci. Technol. 45 (9), 69–76.

Tiessen, K.H.D., Elliott, J.A., Yarotski, J., Lobb, D.A., Flaten, D.N., Glozier, N.E., 2010. Conventional and conservation tillage: influence on seasonal runoff, sediment, and nutrient losses in the Canadian Prairies. J. Environ. Qual. 39, 964–980.

Thoma, D.P., Gupta, S.C., Strock, J.S., Moncrief, J.F., 2005. Tillage and nutrient source effects on water quality and corn grain yield from a flat landscape. J. Environ. Qual. 34, 1102–1111.

Topp, E., Monteiro, S.C., Beck, A., Ball Coelho, B., Boxall, A., Duenk, P., Kleywegt, S., Lapen, D.R., Payne, M., Sabourin, L., Li, H., Metcalfe, C., 2008. Runoff of pharmaceuticals and personal care products following application of biosolids to an agricultural field. Sci. Total Environ. 396, 52–59.

Verbree, D.A., Duiker, S.W., Kleinman, P.J.A., 2010. Runoff losses of sediment and phosphorus from no-till and cultivated soils receiving dairy manure. J. Environ. Qual. 39, 1762–1770.

Vondracek, B., Zimmerman, J.K.H., Westra, J.V., 2003. Setting an effective TMDL: sediment loading and effects of suspended sediment on fish. J. Am. Water Resour. Assoc. 39, 1005–1015.

Wall, G.J., Pringle, E.A., Sheard, R.W., 1991. Intercropping red clover with silage corn for soil erosion control. Can. J. Soil Sci. 71, 137–145.

Watson, C.J., Smith, R.V., Matthews, D.I., 2007. Increase in phosphorus losses from grassland in response to Olsen-P accumulation. J. Environ. Qual. 36, 1452–1460.

Zhao, S.L., Gupta, S.C., Huggins, D.R., Moncrief, J.F., 2001. Tillage and nutrient source effects on surface and subsurface water quality at corn planting. J. Environ. Qual. 30, 998–1008.

Attachment 45:

A BARREL FULL OF HOLES:
A CASE STUDY OF
PENNSYLVANIA REGULATIONS ON
HIGH DENSITY LIVESTOCK FARM
POLLUTION
July 2004

A BARREL FULL OF HOLES: A CASE STUDY OF PENNSYLVANIA REGULATIONS ON HIGH DENSITY LIVESTOCK FARM POLLUTION

Citizens for Pennsylvania's Future (PennFuture)
July, 2004

Foreword

The fate of the efforts to clean up the Chesapeake Bay will be determined in large part on achieving significant reductions in the pollution that comes from agriculture. Pennsylvania has committed to ambitious, but necessary, reductions in the amount of nitrogen and phosphorus it contributes to the Chesapeake Bay – the Commonwealth must find some way to cut nitrogen pollution delivered to the Bay by at least 15.5 million pounds per year and phosphorus pollution must be cut by at least 150,000 pounds.

The lion’s share of this pollution comes from agriculture, primarily from the nutrients – nitrogen and phosphorus - contained in animal manure disposed of on farm fields. A significant amount of the nutrient content of animal manure eventually runs off of farm fields and makes its way into local waterways and the Chesapeake Bay. The watersheds of the heavily agricultural lower Susquehanna River basin, including the Octoraro/Pequea/Conowingo Watershed (Watershed 7-K of the State Water Plan), are an intense source of nutrient pollution to the Bay.

The primary regulatory programs that attempt to control the amount of nutrients that end up in the water are the Nutrient Management Program and the Concentrated Animal Feeding Operation (CAFO) permitting system. Both of these programs fail to grapple with the fundamental problem of regulating nutrient pollution in heavily agricultural watersheds like Watershed 7-K – the sheer amount of animals waste that is being generated there. As the report will show, more than 2.8 million pounds of nitrogen is being generated by livestock in this watershed, and much of it is disposed of on fields in the watershed.

There is no provision in either the Nutrient Management Program or the CAFO permitting system that will limit the expansion of the livestock industry even in watersheds, like 7-K, that are already substantially overloaded with manure. As the livestock industry continues to put more and more animals in these areas, the failure to meet this challenge head on will ensure failure of efforts to reduce nutrient pollution of local waterways and the Chesapeake Bay.

As the report shows, the implementation of both of these programs only adds to the likelihood that they will be unable to bring about reductions in agricultural nutrient pollution from watersheds that are home to intensive livestock operations.

However, two new developments will help improve the situation:

- First, the Environmental Hearing Board recently ruled that nutrient management plans must now account for phosphorus. This ruling will help protect local water quality, but will also result in many acres of farm fields being ruled out for manure disposal exacerbating the manure disposal problem.

- Second, Governor Rendell signed a law that requires livestock operations that export manure to create nutrient balance sheets for fields where the manure will be spread, requires the records to be turned into the county conservation districts, and requires commercial manure haulers and brokers to be certified. If this law is fully implemented and strictly enforced, it will finally provide a complete picture of where, when, and how manure is disposed of. That information may finally expose the full extent of the manure disposal crisis in watersheds that host high animal densities. The response to that crisis will seal the fate of efforts to restore local water quality and the Chesapeake Bay clean up effort.

Pennsylvania's primary regulatory program to control pollution from agriculture, the Nutrient Management Program, is not achieving its goal of protecting and improving water quality, in the Octoraro/Conowingo/Pequea watersheds (Watershed 7-K of the State Water Plan). Nutrient Management Plans cannot achieve adequate control of agricultural pollution for three reasons: 1) the watershed is overloaded with manure; 2) the plans lack enforceable provisions that strictly control the disposal of animal waste, and; 3) there is a lack of adequate enforcement of the implementation of the plans. As a result, the program contains and controls nutrient pollution as well as a barrel full of holes contains water.

Citizens for Pennsylvania's Future (PennFuture) conducted a case study in Watershed 7-K (Pequea, Conowingo, Octoraro and Big Elk creek watersheds). This case study illustrates the impact of the failure of Pennsylvania's laws and regulations on the health, environment and economy of the region and beyond.

KEY FINDINGS

Pennsylvania's primary regulatory program to control pollution from agriculture, the Nutrient Management Program fails to protect water quality in the Octoraro/Conowingo/Pequea watersheds, Watershed 7-K of the State Water Plan. The key findings are listed below.

Implementation of Nutrient Management Plans

- Despite an intense effort by the Lancaster County Conservation District to bring livestock operators into compliance with their nutrient management plans, 59 percent of the operations are in violation of their plans.

The watershed is overloaded with manure and nutrients

- There are a total of 64 livestock operations that have nutrient management plans.
- 76,972,254 gallons of liquid manure are generated each year (enough to fill more than 3,800 railroad tanker cars).
- 58,624 tons of dry manure are generated each year (equal in weight to more than 27,000 Ford Explorers).
- Liquid manure storage capacity in the watersheds is 39,197,393 gallons.
- A total of 2,815,115 pounds of nitrogen from animal manure is generated in the watersheds each year.

Unaccounted-for manure and nitrogen

- 24,673,329 gallons of liquid manure (32 percent of the total liquid manure generated) and 14,060 tons of dry manure (23 percent of the total) is sent off the farm to manure haulers or other farmers. This exported manure is not covered by any approved nutrient management plan.

- Almost 50 percent of the nitrogen, 1,403,326 pounds, is exported. The manure with the higher nitrogen content is far more likely to be exported than manure with lower nitrogen content.
- 89 percent of the livestock operations send some of their manure to manure brokers or other farmers.
- 45 percent of the livestock operators who export manure fail to keep records of the manure transfers.

Watersheds, stream segments and drinking water supplies are heavily impacted by nutrients.

- The Chester Water Authority's Octoraro Reservoir serves more than 200,000 people in the City of Chester and surrounding communities in Delaware County. Treatment plant operators frequently measure nitrate levels in the reservoir and in tributaries in excess of 10 mg/L, the drinking water standard for nitrates, and experience episodic incidents of manure runoff from frozen ground. Even after treatment, nitrates are still detectable in the finished water at levels ranging from 0.2 to 7.6 parts per million (ppm).
- 27 operations generating a total of 50,230,210 gallons of liquid manure and 21,190 tons of dry manure are located in high quality watersheds. 23 percent of the liquid manure and 21 percent of the dry manure is exported and not accounted for in any approved nutrient management plan.
- Half of the high quality watersheds in 7-K contain stream segments that are impaired by agricultural pollution.
- A total of 35 operations are sited in watersheds containing stream segments impaired by agricultural pollution. These operations generate 1,413,227 pounds of nitrogen; 42 percent or 601,269 pounds of nitrogen are exported and not accounted for in any approved nutrient management plans.

Introduction

Pennsylvania's primary program to control pollution from agriculture, the Nutrient Management Program, is not achieving its goal of protecting and improving water quality in Watershed 7-K. The plans cannot achieve adequate control of agricultural pollution for three reasons: 1) the watershed is overloaded with nutrients; 2) the plans lack enforceable provisions that strictly control the disposal of animal waste, and; 3) there is a lack of adequate enforcement of the implementation of the plans.

Agricultural pollution is one of the two primary causes of water quality degradation in Pennsylvania. Erosion of sediment into waterways and over-application of fertilizer to fields severely damages almost 3,000 miles of Pennsylvania streams.

Pennsylvania relies heavily on nutrient management plans to control agricultural non-point nitrogen pollution of lakes and waterways and regional water bodies like the Chesapeake Bay. In 1993, the Pennsylvania General Assembly passed the Nutrient Management Act. The goal of the Act was to reduce nutrient pollution to state streams and lakes and the Chesapeake Bay. It targets the storage and disposal of manure generated at livestock facilities, termed Concentrated Animal Operations (CAOs), which have an animal-to-land ratio of two animal equivalency units (AEUs) or more per acre. Later legislation required stricter permits for Concentrated Animal Feeding Operations (CAFOs), which are larger facilities with both the CAO-defined AEU density and more than 300 AEUs in total.

Manure contains the plant fertilizers nitrogen and phosphorus. When too much of either fertilizer enters waterways, it promotes the growth of algae and other aquatic plants. Algae blooms block sunlight from reaching beneficial aquatic plants, and when the algae die, their decomposition uses up available oxygen in the water making it unfit for aquatic life like fish.

The Nutrient Management Act requires regulated livestock operators to write and implement nutrient management plans that detail how manure will be stored and when and where it will be disposed of. The plans are supposed to ensure that the manure is spread in a time and manner that optimizes its fertilizer value for crops, prevent excess fertilizer from being applied to the land, and minimize the opportunity for the fertilizer to run off into waterways or leach into groundwater. However, the program does not necessarily prevent water pollution. Even if a facility pollutes the water, it cannot be held accountable if the pollution resulted from an activity conducted in accordance with the plan.

To evaluate the effectiveness of the nutrient management program, PennFuture conducted a review of the implementation of the nutrient management program in the Octoraro/Conowingo/Pequea watersheds in Lancaster and Chester counties - Watershed 7-K of the State Water Plan. The review consisted of obtaining all of the nutrient management plans and CAFO permits approved in the watersheds and analyzing the information contained in them. The review uncovered a system wholly inadequate to ensure protection of water quality in watersheds overloaded with animals, manure and nutrients and completely unable to bring about reductions in pollution.

Our review of these plans has revealed that the effectiveness of the nutrient management program is severely compromised by four inherent flaws:

- There is an overload of animal waste and nutrients in the watershed;
- A significant portion of the manure and nitrogen generated in the watersheds is exported to fields not covered by approved nutrient management plans;
- Some of the provisions of the nutrient management plans are not enforceable;
- Even with a vigorous outreach and oversight effort, the plans' elements are not being fully implemented and a majority of operators are in violation of their plans.

Profile of Watershed 7-K

Watershed 7-K of the State Water Plan comprises the Pequea, Conowingo and Little Conowingo, Octoraro, and Big Elk creek watersheds. Agriculture is by far the predominant land use in these watersheds. About 85 percent of the land is in some kind of agricultural use. Watershed 7-K straddles the Chester-Lancaster county line and continues over the state border into Maryland. There are high quality streams in all of these watersheds, most notably the Conowingo, but many of the stream miles are degraded by agricultural pollution and officially listed as "impaired" by the Department of Environmental Protection (DEP)[1]. There are also widespread high levels of nitrate/nitrite concentrations in groundwater[2].

Watershed 7-K also contains a public water supply reservoir, the Octoraro Reservoir, which serves 200,000 people in the City of Chester and surrounding communities in Delaware County. Water intake from the reservoir into the treatment plant is occasionally interrupted because of high nitrate levels coming from the tributaries to the reservoir, especially in the winter months when manure runs off frozen or snow-covered fields. At these times, the Chester Water Authority pulls water from the Susquehanna River to dilute the reservoir water in

[1] 303 (d) list of impaired waters
[2] Watersheds, An Integrated Water Resource Plan for Chester County, Pennsylvania and Its Watersheds, Chester County Comprehensive Plan, Sept. 17, 2002

order to meet drinking water standards for nitrates[3]. Treatment plant operators frequently measure nitrate levels in the reservoir and in tributaries in excess of 10 milligrams per liter, the drinking water standard for nitrates. Even after treatment, nitrates are still detectable in the finished water at levels ranging from 0.2 to 7.6 parts per million.[4]

The Nutrient Management Program in Watershed 7-K

Because the watershed straddles two counties, both the Chester and Lancaster County Conservation Districts administer the nutrient management program for livestock operations in Watershed 7-K. There are a total of 64 livestock operations that have nutrient management plans in the watershed – nine in Chester County and 55 in Lancaster County. Of these, 19 are large enough to be classified as CAFOs[5] – four in Chester County and 15 in Lancaster County.

Manure and Nitrogen in the Watershed

The amount of manure and nitrogen generated in the watershed is so large that it poses significant disposal challenges. According to the nutrient management plans, 76,972,254 gallons of liquid swine and cow manure are generated in the watershed each year. In addition, chickens and dairy operations generate 58,624 tons of dry manure yearly. Combined, the liquid and dry manure contains 2,815,115 pounds of nitrogen.

Since nutrient management plans currently do not require balancing for phosphorus, there is no information in the nutrient management plans about the total amount of phosphorus generated or applied to land in the watershed. However, a very rough estimate obtained by adding up the animals reported in the nutrient management plans and calculating the amount of phosphorus contained in the manure using standard numbers contained in the Penn State Agronomy Guide suggest that about 1.6 million pounds of phosphorus are being generated in the watershed each year.

The aim of the nutrient management planning process is to ensure that the nutrients contained in the manure that is spread on crops is balanced against the needs of the crops that will grow there. This is supposed to ensure that the nutrients are taken up by the crops and will not find their way into the nearest stream or the groundwater. However, when a significant number of the livestock facilities in a particular area operate under contracts that require them to import feed rather than use crops grown locally, the connection between numbers of

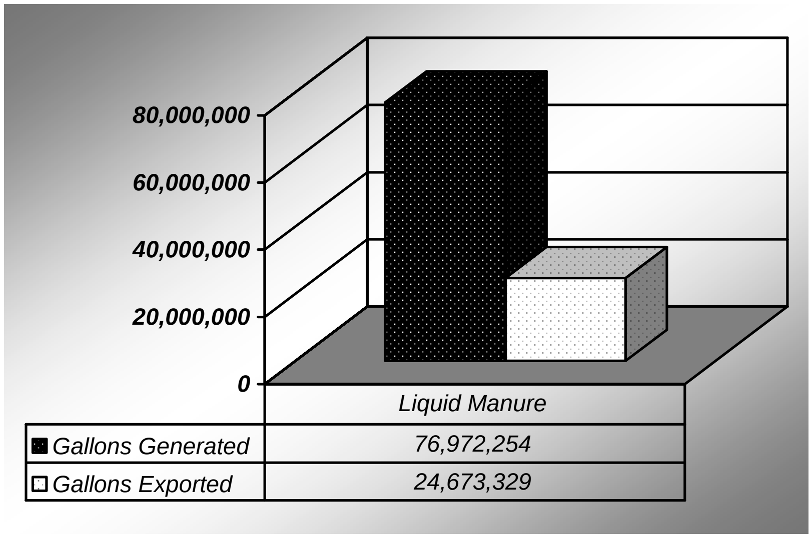
[3] Letter to DEP from the Chester Water Authority in reference to CAFO permit application for McMichael CAFO, August 25, 2000
[4] Watersheds, An Integrated Water Resource Plan for Chester County, Pennsylvania and Its Watersheds, Chester County Comprehensive Plan, Sept. 17, 2002.
[5] A CAFO is a livestock operation that meets the density threshold for Pennsylvania's Nutrient Management Program – 2 AEUs per acre – and also has at least 300 AEUs total or any operation with more than 1000 AEUs regardless of density or with a discharge to surface waters during a storm event at less than a 25 yr. 24 hr storm. Once an operation is that large, it is required to obtain a National Pollution Discharge Elimination System permit for CAFOs from DEP.

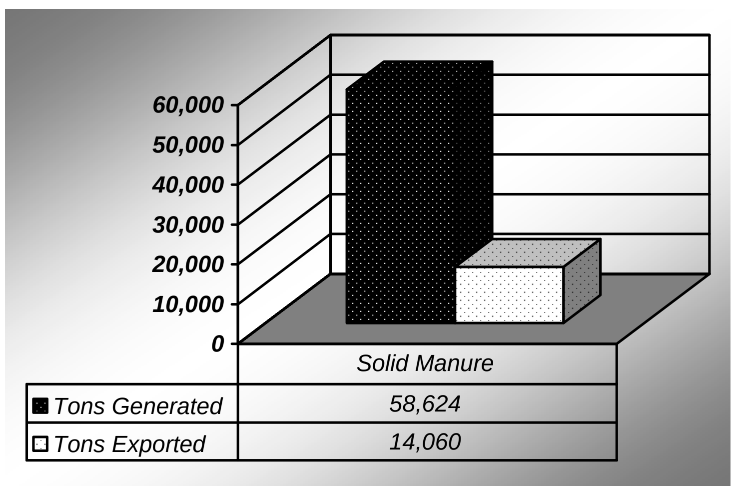
animals and cropland is broken[6]. In this circumstance, which exists in Watershed 7-K, more nutrients are imported into the watershed in the form of feed and ultimately processed by the animals into manure than can be used by the crops grown there.

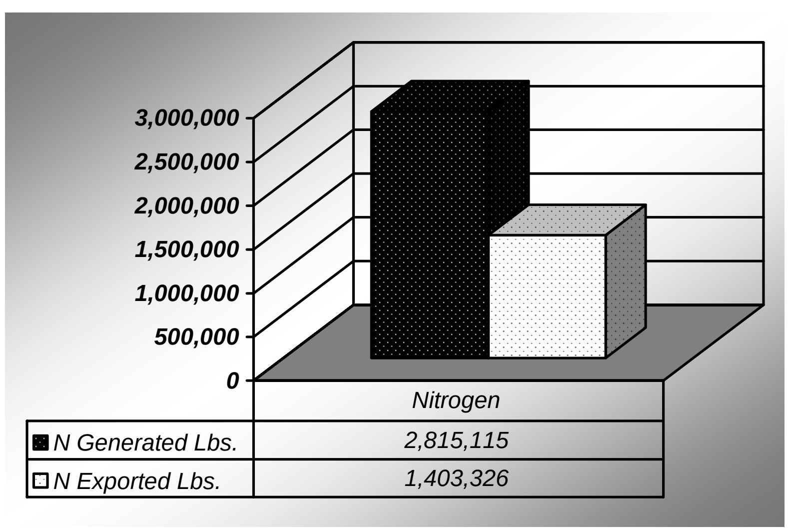
To make matters worse, the regulations contain a major loophole that allows livestock operators to “export” the manure to other farmers – shipping it off the operation that generated it to fields not covered by an approved nutrient management plan. Fully 89 percent of the livestock operators in the watershed export some manure. Our review of the nutrient management plans shows that 35 percent of the liquid manure and 23 percent of the solid manure is being exported to fields not covered by approved nutrient management plans. In addition, the manure with the highest concentrations of nitrogen, swine and chicken manure, is more likely to be exported. As a result 50 percent of the nitrogen generated in the watershed is exported.

There is some transfer of this manure between watersheds, most of it bound for mushroom operations. However, only a few farms list the brokers that supply the mushroom operations in their nutrient management plans. A rough estimate based on that reporting suggests that about 20 percent of the nitrogen is transferred to mushroom growers. The pattern of manure transfers among neighbors and family members in these watersheds would indicate that most of the manure is transferred to nearby fields and remains in its home watershed.

[6] A. E. Nord and L. E. Lanyon, “Managing Material Transfer and Nutrient Flow in an Agricultural Watershed,” Journal of Environmental Quality, March-April 2003.


80,000,000
60,000,000
40,000,000
20,000,000
0
Liquid Manure
Gallons Generated
76,972,254
Gallons Exported
24,673,329


60,000
50,000
40,000
30,000
20,000
10,000
0
Solid Manure
Tons Generated
58,624
Tons Exported
14,060


3,000,000
2,500,000
2,000,000
1,500,000
1,000,000
500,000
0
Nitrogen
N Generated Lbs.
2,815,115
N Exported Lbs.
1,403,326

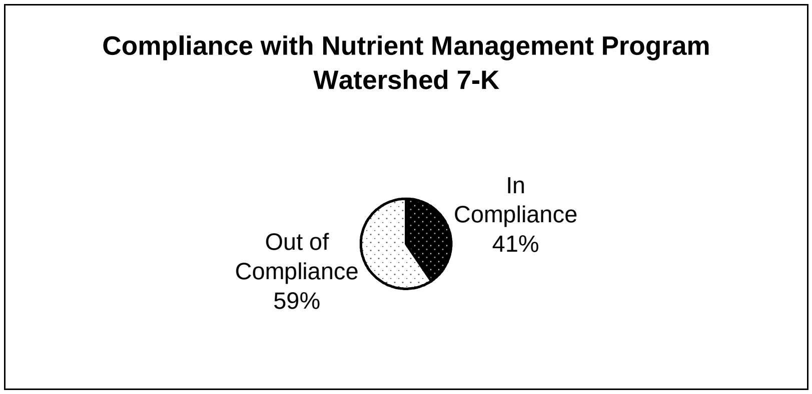
Implementation of the Nutrient Management Plans

County Conservation District staffs have no authority to enforce the Nutrient Management Program or cite the operators for being in violation of nutrient management plans. Operators in chronic violation of their plans must be referred to the State Conservation Commission, which has only three employees to handle enforcement for the entire state and has taken only ten enforcement actions over the history of administering the program. In addition, the Commission is under the purview of both the Department of Agriculture and the Department of Environmental Protection, with the Chair switching between the two. Consistent enforcement of regulations is nearly impossible under this bifurcated system.

The role of the conservation districts is to provide assistance to livestock operators in an effort to bring them into full compliance with the provisions of the nutrient management plans. Indeed, our review showed that there has been a significant effort on the part of the Lancaster County Conservation District to gain full implementation of the nutrient management plans. For instance, the conservation districts are charged with reviewing nutrient management plans every three years to evaluate the implementation of the plan and to inspect the operation to see if it has changed enough to warrant revisions to the plan. The Lancaster County Conservation District conducted timely reviews on all 46 of the nutrient management plans that reached the three-year deadline. In Chester County, conservation district staff had reviewed only three of nine operations, but all but two of them were due for their three year review.

However, despite the significant effort to ensure implementation of the nutrient management plans, 38 of the 64 operations reviewed — 59 percent — had not fully implemented their nutrient management plans, or had not kept important records that would allow the conservation district staff to determine that manure was being properly handled[7]. Of the operators that export manure, 46 percent of them were missing the manure transfer sheets that would at least identify the fields or manure brokers who had received their manure. Therefore, approximately 23 percent of the liquid manure and 32 percent of the dry manure was largely unaccounted for.

[7] Our compliance figure is based on the compliance status in most recently available conservation district review.



Very often, manure disposal in this watershed is handled informally among neighbors and extended family networks. Many of the operators are members of plain sects, and record-keeping is not their forte. When records are available, they indicate that manure is frequently sent to farms not listed in the nutrient management plans. The notes in the review files indicate that conservation district staff provides ongoing assistance to livestock operators in an attempt to bring them into full compliance with their plans. As a result, many operations fall in and out of compliance during the course of the review cycle.

While record-keeping violations may seem trivial, the lack of complete records completely undermines the goals of the Nutrient Management Program – to ensure no more nutrients are put on fields than crops can use and to reduce nutrient pollution of local streams and larger water bodies like the Chesapeake Bay. The heart of the program consists of planning and management, and record-keeping is the primary regulatory requirement.

The only way conservation district staff and water quality managers at DEP can determine the real loading rates in the watersheds for nitrogen and phosphorus would be to have complete records of manure transfers and tabulate the data. In the absence of such a system, any estimates of nutrient loadings or projections of reductions are based on information so incomplete as to make them meaningless for use by water quality managers.

Concentrated Animal Feeding Operations in the Watershed

Of the 64 intensive livestock operations in Watershed 7-K, there are 19 facilities large enough to qualify as Concentrated Animal Feeding Operations (CAFOs). These are generally operations that have an animal-to-land ratio of more than 2 animal equivalency units (AEUs) per acre and more than 300 AEUs in total. Once a livestock operation reaches that threshold, it is required to obtain a CAFO water pollution control permit from DEP and, sometimes, a permit for its manure storage structure.

Of the 19 facilities that are large enough to qualify as a CAFO, six do not currently have CAFO permits. Of the six, one has a permit application pending and another has applied for a permit for its manure storage structure, and DEP indicates that it is anticipating the permit application for that operation. Another was informed that it was required to apply for a permit, but DEP has not yet received the application. Two of the operations have a mix of poultry and dairy animals, and currently DEP does not require dairy operations with dairy herds under 300 AEUs to apply for permits even when the poultry operation puts the entire operation over the 300 AEU threshold.

The CAFOs generate almost 85 percent of the total manure and 60 percent of the nitrogen in the watershed — more than 65 million gallons of liquid manure and more than 34,000 tons of solid manure containing more than 1.6 million pounds of nitrogen. About 36 percent of the nitrogen is exported to fields not covered by an approved nutrient management plan. Five of these operations raise only hogs, four of them are dairies, and ten of them have a mix of hogs, poultry and cattle.

There were no records of any inspections by the conservation districts or DEP for three of these facilities. Of the 16 facilities inspected, 11 were in violation of either their CAFO permit or their nutrient management plans and five were in full compliance.

Impaired Watersheds

Half of the livestock facilities in this review are located in watersheds where the entire streams or significant stream segments do not meet water quality standards because of agricultural runoff and nutrient pollution. These 32 facilities generate a total of almost 43 million gallons of liquid manure and more than 20,000 tons of dry manure. This manure contains 1.25 million pounds of nitrogen. About a quarter of the manure is exported, but since the manure with the highest concentration of nitrogen is more likely to be exported, 44 percent of the nitrogen in the impaired watersheds is being exported to fields not covered by an approved nutrient management plan.

Currently, DEP does not consider the cumulative impact of the amount of manure and nitrogen being generated in the watershed when evaluating applications for new or expanding facilities large enough to be CAFOs. CAFO permits are “non-discharge” permits, so no discharge of manure from manure storage structures is allowed except during very large rain storms. The increased pollution from the inevitable polluted farm field runoff is not considered, nor is the potential for manure exported to fields not covered by an approved nutrient management plans taken into account. As a result, DEP does not attempt to limit manure generation or land application in impaired watersheds.

Special Protection Watersheds

There are 27 livestock facilities, or 42 percent of the operations, located in high quality watersheds. These facilities generate more than 50 million gallons of liquid manure and more than 21,000 tons of dry manure. This manure contains about 1.5 million pounds of nitrogen and about 34 percent of that is exported. Unfortunately 14 of the high quality streams also contain segments impaired by agricultural runoff. Right now, DEP does not take the potential for a CAFO to degrade water quality or to make it harder to clean up existing water quality problems into account when considering permit applications. Usually, facilities applying for water pollution control permits in high quality watersheds must demonstrate that their activities will not degrade the streams. DEP does not require this demonstration for CAFO applications, however, apparently based on the “no discharge” requirement. But as mentioned above, preventing discharges from the storage structures does not prevent nutrients from running off fields and into high quality streams after the manure is spread and causing serious water pollution problems.

Winter Spreading

The spreading of manure onto frozen or snow-covered fields is discouraged, but not prohibited in the nutrient management program. Manure spread onto frozen or snow-covered fields is merely being disposed of since there are no plants growing to take up the nutrients and it is highly likely to wash into the nearest stream with a quick melt or rainfall. The practice is routine and widespread and poses significant threats to water quality. During the winter, the Chester County Water Authority frequently must pump in water from the Susquehanna River to dilute the Octoraro reservoir water in order to reduce the nitrate levels sufficiently to meet drinking water standards. [8]

[8] Letter to DEP from the Chester Water Authority in reference to CAFO permit application for McMichael CAFO, August 25, 2000

Recommendations

IMPLEMENTATION

- A nutrient management program enforcement officer should be placed at each conservation district (or, in areas with few CAOs one enforcement officer could handle several conservation districts in a region) whose sole responsibility would be to inspect CAOs and CAFOs for compliance with their plans and permits.
- Chronic violators should be referred to the State Conservation Commission.
- Oversight and funding of the State Conservation Commission should be solely under the Department of Environmental Protection.
- Chronic violators should be barred from receiving federal or state funding for technical assistance or purchase of equipment.
- Nutrient management plans should contain enforceable provisions.
- The plans should also address the phosphorous content of manure spread on all fields.

TRACKING MANURE GENERATION AND DISPOSAL

- Exporters should be required to create balance sheets for both nitrogen *and* phosphorous for manure sent to other farmers' fields.
- Conservation districts should use the nutrient management review process to collect data about the amount of manure generated and its nutrient content and submit it to DEP.
- Livestock operators should be required to submit all manure transfer sheets to conservation districts each year. These should be considered public documents.
- Conservation districts should be required to tabulate all manure transfer data and submit it to DEP each year so that water quality managers can use the information for program implementation including the development and implementation of Total Maximum Daily Load (TMDL) and Chesapeake Bay Program Tributary Strategies.

REQUIREMENT TO OBTAIN A CAFO PERMIT

- Pennsylvania's should add the language of the U.S. Environmental Protection Agency (EPA) requiring all facilities that contain certain numbers of animals to obtain CAFO National Pollution Discharge Elimination System (NPDES) permits in addition to its density thresholds that currently trigger the requirement to get a permit.
- DEP should require all livestock facilities that meet the regulatory definition of a CAFO to obtain a permit.
- DEP should consider cumulative impact, impaired watersheds, TMDLs and limestone geology when determining that a livestock operation needs a CAFO NPDES permit.

- Any facility that has created a discharge should be required to obtain a CAFO NPDES permit.

REVISIONS TO CURRENT POLICY

- Require NPDES CAFO applicants in high quality watersheds to satisfy all antidegradation program requirements.
- Spreading on frozen or snow covered ground should be prohibited.
- In watersheds where a TMDL and a nutrient reduction program have been established, no additional manure should be allowed to be applied to land unless it is expressly accounted for in the TMDL. Thus, if a farming operation intends to expand (or establish itself) it would have to develop an alternative manure utilization plan. The Administration should place a temporary moratorium on CAFO expansions in watersheds that are impaired for nutrients and no TMDL has yet been developed.