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REGULAR ARTICLE

Managing agricultural phosphorus for water quality protection: principles for progress

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Abstract
Background The eutrophication of aquatic systems due to diffuse pollution of agricultural phosphorus (P) is a local, even regional, water quality problem that can be found world-wide.
Scope Sustainable management of P requires prudent tempering of agronomic practices, recognizing that additional steps are often required to reduce the downstream impacts of most production systems.
Conclusions Strategies to mitigate diffuse losses of P must consider chronic (edaphic) and acute, temporary (fertilizer, manure, vegetation) sources. Even then, hydrology can readily convert modest sources into significant loads, including via subsurface pathways. Systemic drivers, particularly P surpluses that result in long-term over-application of P to soils, are the most recalcitrant causes of diffuse P loss. Even in systems where P application is in balance with withdrawal, diffuse pollution can be exacerbated by management systems that promote accumulation of P within the effective layer of effective interaction between soils and runoff water. Indeed, conventional conservation practices aimed at controlling soil erosion must be evaluated in light of their ability to exacerbate dissolved P pollution. Understanding the opportunities and limitations of P management strategies is essential to ensure that water quality expectations are realistic and that our beneficial management practices are both efficient and effective.

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Introduction

Anthropogenic eutrophication, the ecologicaltransformation of water bodies induced by nutrient pollution, is a global phenomenon readily witnessed in developing and developed worlds alike. Anthropogenic eutrophication transcendsscales,from ponds to local streams and reservoirs to regional watersheds andinternational estuaries. The detrimental impacts of eutrophication range from the decline of aquatic resources (wild and cultured) that support coastal, riverine and lacustrine communities, to the degradation of water for human consumption and recreation, to the expansion of acutely toxic algal blooms that can directly impact human health. Despite limited information on the economic costs of eutrophication, the many local, provincial, domestic and international mitigation programs that have mounted to combat causes of eutrophication, suggest a magnitude of cost equivalent to a significant fraction of national economies.

The role of phosphorus (P) in eutrophication has expanded over the past half century as anthropogenic sources of P have grown and as our understanding of eutrophication processes has advanced. Once regarded as mainly a concern to freshwater eutrophication, P has since been implicated as a primary or contributing nutrient in coastal eutrophication (Correll 1998; Howarth and Paerl 2008), notably Northern Europe's Baltic Sea, China's Changjiang Estuary and North America's Gulf of Mexico, among others. From the perspective of societal development, the phenomenon of P-induced eutrophication can be attributed to the pervasive use of phosphates. Along with point sources of P (e.g., sewage treatment plants, factories) and non-point sources associated with urbanization (e.g., stormwater runoff, septic systems), agriculture is consistently identified as one of the largest contributors of P to surface waters(Duriancik et al. 2008; Kronvang et al. 2009; Torrent et al. 2007). Indeed, in developed countries where significant reductions have been made in point-source P pollution, agricultural non-point sources are often the greatest source of P to eutrophic water bodies (Dubrovsky et al. 2010; U.S. Environmental Protection Agency 2010).

Managing agricultural sources of P to curb eutrophication is an inherently difficult task, hindered by the complexity of watershed processes, variability in appropriate management practices and vagaries of climate. At one level, the basic cause of continued P loss from agriculture represents the discrepancy between what is economically optimum for agricultural production and what is required to keep P transfers to surface waters below trophic response thresholds. Grappling with this discrepancy represents one of the great challenges of modern agriculture, perhaps even modern society. At another level, significant non-point source P pollution can occur even when environmental sources of P are not obvious; e.g., vegetation and dung can combine to contribute significantly to dissolved P to runoff from pastures where P is in agronomic deficit (McDowell et al. 2007). It is therefore important to understand that although agronomic optimization of P is essential to tackling agricultural causes of eutrophication, tackling crop P use efficiency alone may be insufficient to reverse eutrophication in many cases. Additional steps are often required (Sharpley et al. 2006; Sims and Kleinman 2005).

Considerable research and experience now exists that provide insight into practices and strategies for minimizing P transfers from agricultural lands to surface waters (Sharpley et al. 2006; Sims and Kleinman 2005). This review elucidates processes underlying non-point P pollution and illuminates practices and strategies required to control that pollution. Recognizing the complexity and site-specific nature of P losses from agriculture, we highlight key opportunities and challenges for managing agricultural P to enhance water quality.

Comprehensive management includes acute and chronic sources of P

At the most elemental level, tackling diffuse P pollution from agriculture begins with identifying and managing P sources at the field scale. Major sources include recently applied P (i.e. fertilizer, manure, dung) and "legacy" P in soils from previous P applications.

Applied P – an acute, temporary source

The high concentrations of P in recently applied sources can elevate dissolved P in surface runoff and leachate to concentrations many fold greater than background, although the effect is transient and is associated with surface application methods. Referred to as "wash-off" or "incidental transfer", P transfer associated with surface applied sources is at maximum potential in the first runoff events after application (Austin et al. 1996), generally returning to near background levels over several months (McDowell and Catto 2005; Preedy et al. 2001). High rates of P applied to soils prone to runoff during periods of high rainfall or shortly before flood irrigation events produce the greatest potential for acute transfers of applied P to water bodies (Kleinman and Sharpley 2003; Nash et al. 2000; Withers et al. 2003).

The initial magnitude of P concentrations in runoff from applied P wash-off is generally a function of the rate of manure or commercial fertilizer application (Fig. 1) and the solubility of applied P (Kleinman et al. 2007a). Mobilization of dissolved P losses from wash-off can be reduced with amendments that lower solubility of P in the applied source (Moore and Miller 1994; Moore et al. 2000; Stout et al. 2000). Notably, the P in mineral fertilizers tends to be so soluble that a disproportionate fraction of applied fertilizer P is translocated into soil by infiltrating rainfall where it is rapidly sorbed. Consequently, wash-off of surface-applied mineral fertilizer P is comparable to what is observed with P sources of lesser solubility (Kleinman et al. 2002).

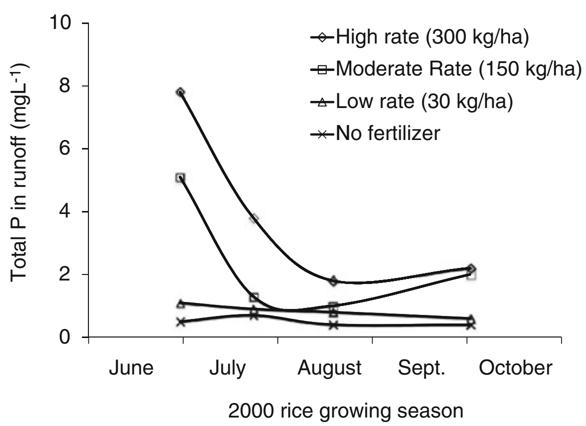

Fig. 1 Runoff (surface and subsurface combined) P concentrations from rice paddy soils in China's Taihu Lake watershed, China. Wash off of applied P is greatest in initial events after application. With the highest application rates, runoff P concentrations remain above background through to the end of the rice growing season. Adapted from Zhang et al. (2005)

Both the magnitude and the duration of P transfers to runoff associated with applied P wash-off isreduced by application methods that promote rapid or immediate incorporation of P sources into soil. Removing applied P from the effective depth of interaction between surface runoff and soil (Sharpley 1985) and promoting contact of that P with soil to promote sorption is achieved by techniques as commonplace assubsurface banding ofmineral fertilizers. With manures, incorporation has traditionally been by tillage, which involves trade-offs due to greater erosion potentional, but a growing number of low-disturbance incorporation technologies are now available:injectors for liquid manures;subsurface applicators for dry manures; aerators that improve infiltration of liquid manure and rain water (Maguire et al. 2011).

Soil P – a chronic source

While the contribution of surface applied P sources to runoff diminishes during the weeks and months after application, the contribution of soil P erosion, dissolution and desorption to runoff persists over time. First and foremost, soil erosion presents the greatest concern to most P mitigation programs. The concentration of P attached to soil particles is generally several orders of magnitude greater than that in the soil solution. In addition, erosional processes preferentially remove the finest particles of soil, resulting in enrichment in sediment P concentrations that can be up to five times greater than those found in bulk soil from which the sediment erodes (Sharpley et al. 2002). When left unchecked, erosion-related losses of soil P can match P removed by crop harvest, threatening crop production in areas where soil P is low. Consequently, soil conservation has been and will continue to be a high priority in eutrophication remediation strategies.

In recent years, there has been a recognition that release and mobilization of dissolved P from agricultural soils is a much more pervasive contributor to eutrophication than historically acknowledged. Arguments have been made that, due to its immediate biological availability, dissolved inorganic P pollution has a disproportionately large impact on eutrophication, compared with sediment-bound P derived from erosion. Phosphorus desorption is correlated with the saturation of a soil's P sorption capacity by past applications of P in excess of crop removal (Sharpley and Rekolainen

1997). Although a variety of measures of soil P have been promoted as indicators of dissolved P release to runoff, comparably strong relationships have been obtained between dissolved P concentration in surface and subsurface flow and either agronomic soil tests or so-called environmental soil tests (Fig. 2).

Adjust management to account for the overwhelming role of hydrology on P transfers

Although much is known of the chemical controls of P release from soils and applied sources to runoff water, the hydrologic controls of P transfer (mobilization, transport and delivery from field to water body) change with scale (from field to watershed), are highly location specific and are dynamic in nature. It is clear that site and watershed hydrology exert an overwhelming effect on whether agricultural P will become a downstream water quality concern (Gburek et al. 2002; Pionke et al. 2000). Within watersheds, variable source area hydrology (Hewlett and Hibbert 1967) can produce severe spatial heterogeneity in the potential for P to be transferred from field to water body (Sen et al. 2008; Srinivasan and McDowell 2009; Walter et al. 2000). Watersheds prone to variable source area hydrology possess zones that contribute disproportionately to runoff (e.g., Pionke et al. 1997), with the size and location of runoff-generating areas determined by the interaction of soil moisture, topography and geomorphology (Buda 2011; Needleman et al. 2004). As a result, the inherent potential to mobilize agricultural P differs greatly from one field, even one area of a field, to the next (Gburek et al. 2007). Furthermore, hydrologic processes are often discontinuous within agricultural landscapes, such that runoff from some areas contributes P directly to downstream water bodies while runoff from other areas does not (Sharpley et al. 2008).

The dynamic nature of hydrologic processes produces profound temporal variability in P transport. Variability in hydrologic flows is generally the overarching determinant of watershed P loadings (e.g., Puustinen et al. 2007). Storm characteristics (intensity, duration) and antecedent conditions (especially soil moisture) all contribute to this variability, sometimes with sufficient regularity to allow meaningful management inferences. For instance, in the province of Manitoba, Canada, 80 to 90% of runoff occurs during spring-time snow-melt over frozen soils (Tiessen et al. 2010). Adjusting annual P applications in this region to avoid application to frozen soils minimizes the potential for P wash-off. Such generalizations are fine for strategic planning (e.g., Sharpley et al. 2003), but do not help in tactical or operational planning requiring daily or weekly decisions. More recently, there has been growing interest in the potential to tie short-term weather forecasts to hydrologic routines that estimate site potential for runoff (Buda et al. 2010; Dahlke et al. 2008; Wisconsin Manure Advisory System 2010). At this point, these daily models have not advanced beyond the proof-of-concept phase; however, they do

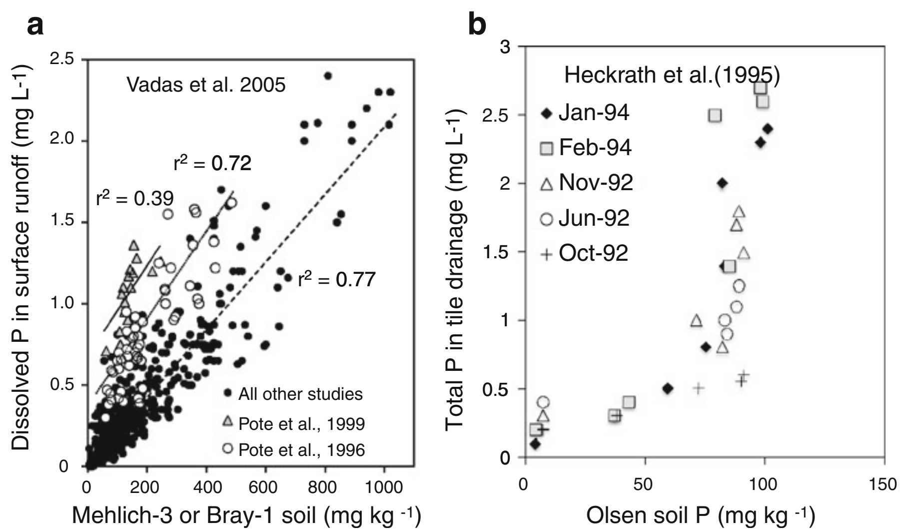

Fig. 2 Concentrations of P in surface runoff and tile drainage waters as a function of soil test P concentration. Surface runoff data (**a**) are adapted from a summary of studies in the U.S. A. by Vadas et al. (2005). Drainage data (**b**) are adapted from long-term experiments on the Broadbalk plots in the U.K. described by Heckrath et al. (1995)

provide insight to a future where farmers can easily obtain reliable weather based information that helps them make environmentally sound management decisions that are also cost beneficial.

Consider the potential for subsurface P transport

Old generalizations discounting environmentally significant losses of P via subsurface pathways are no more acceptable today than are past generalizations discounting the transfer of dissolved forms of P in overland flow. Subsurface transport of P from agricultural fields does occur, and large off-site loads associated with acute (applied) and chronic (edaphic) sources have been documented (Kleinman et al. 2007b; van Es et al. 2004). To impact surface waters, P must both leach vertically and be transported laterally to a stream. Vertical leaching potential is largely determined by soil physical characteristics, as P leaching occurs primarily via macropores that are continuous with sources within the solum (Djodjic et al. 1999; Simard et al. 2000; Sims et al. 1998). Leaching of dissolved P has been tied to surface soil P desorption (Heckrath et al. 1995; McDowell and Sharpley 2001) and applied sources (Chardon et al. 2007; Geohring et al. 2001; Weaver et al. 1988), with particulate P transfers frequently documented (Heathwaite and Dils 2000), particularly following tillage (e.g., Schelde et al. 2006). Lateral subsurface transport of P in subsurface flow can be induced by subsurface drainage, with which it is most commonly documented (Sims et al. 1998), but also occurs in association with sandy strata and bedrock fractures (Kleinman et al. 2007b; Kleinman et al. 2009).

Considerable evidence suggests that long-term accumulation of P in surface soils can produce chronic losses of P to drainage waters. The concept of a soil P threshold to protect drainage water quality is supported by an array of studies that have reported statistically and environmentally significant increases in subsurface P losses above a particular value (Heckrath et al. 1995; Maguire and Sims 2002; McDowell and Sharpley 2001). In the Netherlands, soil P sorption saturation (also the “degree of P saturation” or “P saturation ratio”) has been used to determine when unacceptably high concentrations of P can leach from soils (Breeuwsma and Silva 1992).

However, soil P alone does not determine the risk of P loss, as soil physical properties as well as drainage and tillage management can overwhelm the role of surface soil P saturation in subsurface P transport (Fig. 3).

Growing interest exists in the potential to curtail subsurface P transfers of applied P sources using tillage. Due to the importance of soil macropores as a conduit for P transport (Heathwaite and Dils 2000), practices that encourage macropore connectivity, particularly with the soil surface, can also encourage P leaching. Shipitalo and Gibbs (2000) pressurized smoke into tile drains to identify macropores (primarily earth worm burrows) connecting the drains with the soil surface. Notably, most of the pores were located within 2 m of the drains. A variety of studies have tested the role of tillage before or after manure application on P leaching. While some studies have found that tillage exacerbates P losses by enhancing particulate P transfers to drainage waters (Schelde et al. 2006), others have found lower subsurface losses due to lesser solute transfer with tillage (Djodjic et al. 2002; Kleinman et al. 2009; Shipitalo et al. 2000). Indeed, new manure injection technologies are being advanced that mix injected manure with soil to plug macropores and prevent bypass flow.

Confront legacy sources of P

Building soil P reserves is a long-establishedsoil fertility practice, as is the maintenance of soil P at levels sufficient for crop growth (e.g., Tisdale and

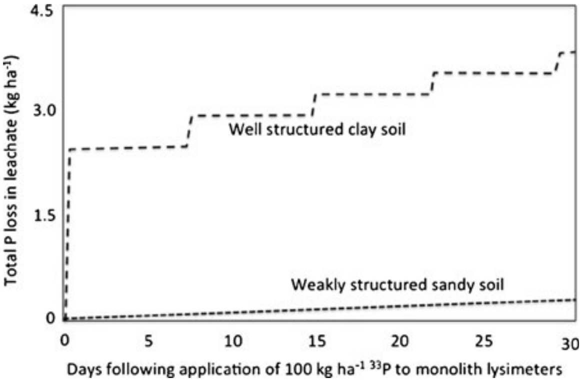

Fig. 3 Calculated losses of ^{33}P in leachate from monolith lysimeters. The well structured clay soil possess a significantly greater potential to transmit applied P than the weakly structured sandy soil, illustrating the importance of continuous macropores from the surface to the subsurface to vertical leaching of P. Adapted from Djodjic et al. (1999)

Nelson 1956). However, the application of P to agricultural soil in excess of crop requirement and the related saturation of a soil's P sorption capacity will promote chronicrelease of P to runoff water that is not addressed by most agronomic and conservation practices. Soil P concentrations may rise to such a level that their return to optimal levels for crop production may require decades to reverse (Halvorson and Black 1985; McCollum 1991).

In many cases, system level imbalances in P, such as national fertilizer policies or regional export of P from areas of crop production and import of P into areas of intensive livestock production, account for local accumulations of excess P in soils (Maguire et al. 2009; Sims et al. 2000). Globally, national efforts to address food security through land reclamation have produced profound accumulations of P in soils, such as today in China or historically in Brazil's Cerrado region (Fig. 4). In the USA, specialization and intensification of crop and livestock production has yielded gross discrepancies in the local P balances (Lanyon 2005; Sims et al. 2005). As illustrated by county level P balances in the USA (Fig. 4), significant P deficits (>15 kg ha^{-1}) exist in mid-western counties that produce most of the country's grain while significant surpluses (>30 kg ha^{-1}) are found in association with livestock production. Because most of the P fed to livestock is excreted in manure, which tends to be applied locally, the counties with P surpluses also possess the highest soil P levels.

Several case studies from the Chesapeake Bay region of the eastern USA illustrate the conse-

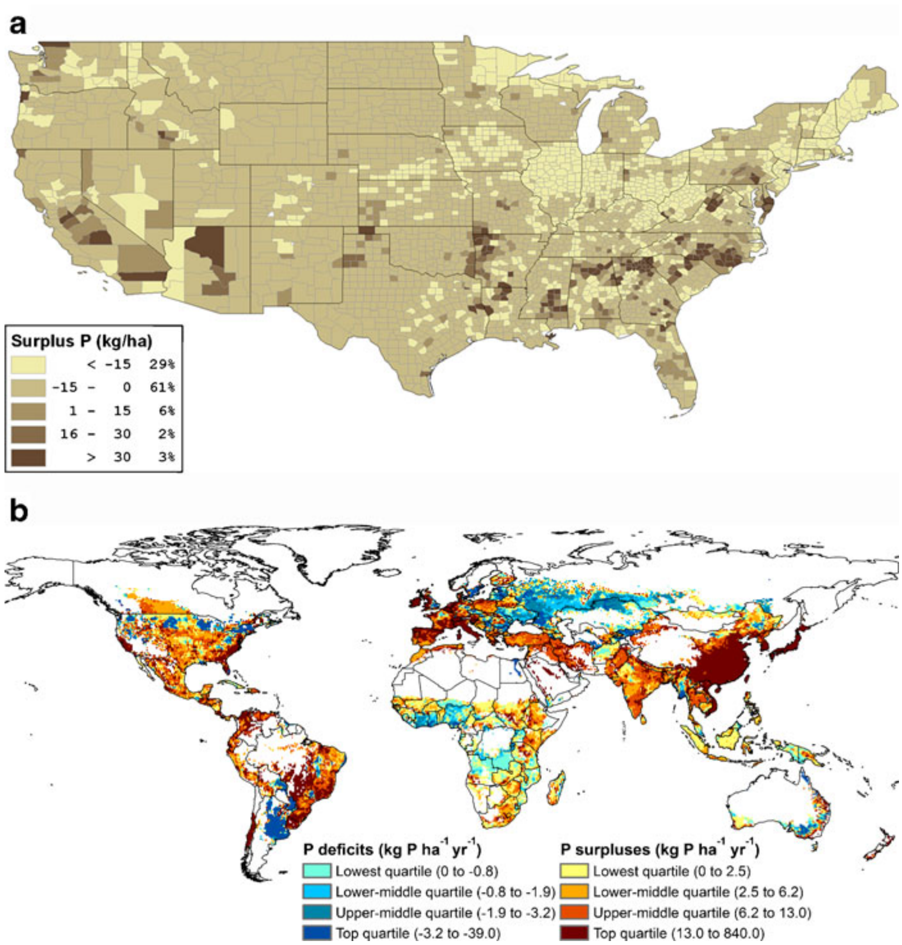

Fig. 4 County level balances of manure P in the U.S.A. (**a**, adapted from Maguire et al. 2009) and regional agronomic balances across the globe, (**b**, adapted from MacDonald et al. 2011). The accumulation of P in agricultural soils is a necessary part of soil fertility management, but accumulations well in excess of crop requirement can be found world-wide as a function of livestock concentration or food-security derived policies

quence of legacy P sources on water quality. For example, on the Delmarva Peninsula, a 14,000 km^2 spit of coastal plain underlain by marine sands, P concentrations in agricultural soils are often much higher than what is required for crop production (Sims and Kleinman 2005). Soil P concentrations have risen over 40 years in some areas of the Delmarva Peninsula due to the continued N-based application of poultry litter from broiler (meat poultry) operations that produce more than 600 million birds annually. Field drainage ditches are pervasive due to the flat landscape and shallow water table and >90% of P loadings to ditches are derived from sub-surface flow. At a site in which soil test P was nearly one order of magnitude above crop production requirements, Kleinman et al. (2007b) observed little impact of short-term fertilizer and manure wash-off processes when compared with overwhelming contribution of legacy sources of P in soils and ditch sediments. Indeed, annual P loadings were as high as 26 kg ha^{-1} $year^{-1}$ from some ditches, with nothing available in the way of conventional mitigation processes to curb this loss.

Lower livestock densities are observed in many of the upland reaches of the Chesapeake Bay Watershed, but even there local sources of legacy P impact water quality. Buda et al. (2009) monitored runoff from three fields located at different landscape positions on a hillslope in Pennsylvania, part of the Chesapeake Bay Watershed. The lowest field occupied a wet area of the hillslope that accounted for 97% of the overland flow. Although the lowest field had significantly lower soil P (Mehlich-3 P=78 mg kg^{-1}) than the upslope fields (Mehlich-3 P=144-177 mg kg^{-1}), soil P concentrations of the lower field exceeded the amount required for crop production due to historical applications of manures (50 mg kg^{-1}). Chronic dissolved P concentrations in overland flow, which accounted for 70% of total P, were strongly tied to soil test P. However, losses of P from the lowest field (8 kg ha^{-1} $year^{-1}$) were much greater than the upper fields (≤1 kg ha^{-1} $year^{-1}$) owing to the greater hydrologic activity. These extremely large losses indicate that, under the right conditions, even a modest source of soil P can contribute to very large legacy loads.

To address legacy P sources, novel practices may be required. If mobilization of dissolved P from an agricultural soil is the primary concern and erosion is kept in check, then some forms of deep tillage may be used to promote mixing of surface soils that are highly P saturated, with subsoils that have a high P sorption capacity. Sharpley (2003) reported reductions of 47% total P concentration in runoff following plowing of soils with severe vertical stratification of P and establishment of an erosion-protecting cover. There is growing interest in intercepting runoff waters (surface and subsurface) and filtering them to remove P (Cox et al. 2005; Penn et al. 2007). Opportunities exist at sites of concentrated flow (especially drainage lines and ditches), adjacent to areas of acute P mobilization potential (e.g., barnyards) and even in subsoils.

Balance soil conservation and water quality priorities to avoid vertical P stratification

Managing agricultural P to protect water quality may require adjustments to other conservation strategies. This is most apparent with tillage management. While no-till and other forms of reduced tillage are key to controlling erosion and associated particulate P losses, these tillage systems can exacerbate dissolved P losses with time. Specifically, the absence of tillage aggravates the stratification of soil and residual fertilizer P in the soil profile (Holanda et al. 1998; Selles et al. 1999; Sharpley et al. 1993; Vu et al. 2009), thus concentrating applied P at the soil surface within the critical zone that readily contributes dissolved P to runoff water (Sharpley 1985). Both acute wash-off of applied P and chronicreleaseof soil P to runoff may be exacerbated (Krieger et al. 2010; Tiessen et al. 2010; Verbree et al. 2010). Prudent conservation strategies must therefore consider the unintended consequences of severe vertical P stratification to prevent inadvertent dissolved P pollution.

A growing number of case studies illustrate the trade-offs in managing fertilizer and manure P in no-till systems (e.g. Kleinman et al. 2009; Sharpley and Smith 1994; Uusitalo et al. 2007). In the nearly level prairie region of Manitoba, Canada, Tiessen et al. (2010) reported the results of a 14-yr paired watershed study comparing conventional tillage and no till management. Following a 4-year calibration period in which the two, 4–5 ha watersheds were managed in conventional tillage (15–25% residue cover after planting), one of the watersheds was converted to no-till (45–73% residue cover after planting). The conservation tillage system was allowed to "stabilize" for several years prior to the 4-year measurement period during which the two watersheds were compared. After accounting for the inherent differences in runoff behavior between the two watersheds, annual losses of total P in runoff from the no-till watershed increased 12%, even though particulate P losses decreased 37% with no-till due to 65% less erosion (Fig. 5). The greater total P losses in runoff reflect an increase in dissolved P in runoff from the no-till watershed, a function of severe P stratification at the soil surface and potentially the release of dissolved P from plant residue on the soil surface. In colder climates, the release of dissolved P from plant residue is exacerbated by repeated freeze thaw cycles that lyse plant cells (e.g., Bechmann et al. 2005). Thus, Tiessen et al. (2010) argue that strategies to control diffuse P pollution in areas where dissolved P is the primary concern should include practices that lower the accumulation of soil P and plant residue at the soil surface.

Control a diversity of sources of P in pasture-based systems

Even low-input, pasture-based systems can yield environmentally significant loads of P. Runoff P losses from dung can account for 20–40% of the total P losses from pastures. The risk of loss of dung-P depends on stock type (usually cattle>deer>sheep) and stocking rate (i.e., number of dung deposits) and decreases with time since deposition due to the gradual formation of a crust that prevents the interaction of runoff with the bulk of the dung beneath (McDowell 2006). A similar decrease with time occurs for mineral fertilizers (Austin et al. 1996; Nash et al. 2000), depending on solubility (as discussed above). Losses of fertilizer-P can account for 50% of losses from the paddock, but the average is about 10% in New Zealand pastures (McDowell et al. 2007), and can be decreased to <5% (on a watershed scale) if low water solubility fertilizers (e.g., reactive phosphate rock) are used (McDowell et al. 2010). A third source, accounting for about 20% of P losses, arises from grazing, whereby runoff extracts P from the vacuoles of freshly grazed plants, or plants that have been trampled upon or are in a state of natural decay. The final source, the soil accounts for the remainder of P losses. As mentioned above, the magnitude of losses depends on soil P concentration, but variation also occurs due to trampling and the physical disturbance of surface soil resulting in particulate P losses. This is usually only a factor for cattle and deer (Curran-Cournane et al. 2011).

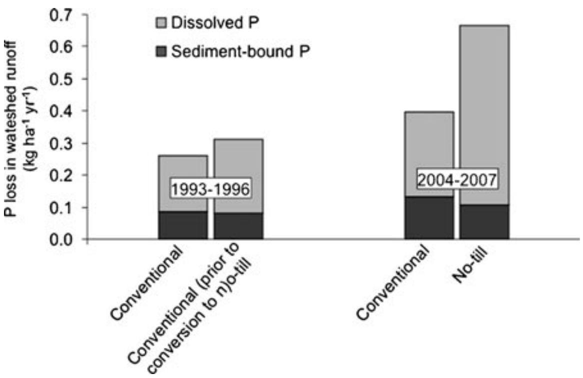

Fig. 5 Runoff P losses from paired watersheds in Manitoba, Canada. Following a 4 year calibration period (1993–1996), no till was implemented in one watershed. 7 years after conversion to no-till, runoff total P loss increased by 12%, after accounting for inherent differences between the watersheds. Although sediment-bound P losses declined with no-till, dissolved P losses increased substantially. Data adapted from Tiessen et al. (2010)

Where paddocks are intersected by waterways, fencing is necessary to prevent direct deposition of dung-P and additional P inputs associated with disturbance of waterway beds or banks. Many studies have documented a decrease in P loading of streams or rivers after the installation of fencing. McDowell (2008) noted a 90% decrease in P losses of a headwater watershed in southern New Zealand after red deer (*Cervus elaphus*) were fenced-out of an area used by the deer for wallowing (Fig. 6).

Livestock that defecate and urinate in and near streams can potentially contribute significant amounts of N and P over time. By observing four pastures where cattle had access to streams in New York, U.S.A., James et al. (2007) were able to estimate fecal P contributions to streams. On average, roughly 30% of all dungpatsexpected from a herd fell within 40 mof a stream, and 7% were deposited directly into streams. Extrapolated to all grazed pastures in the area, cattle excreta deposited in streams were equivalent to 12% of surface water P loads attributed to all forms of agriculture (point and non-point). While programs

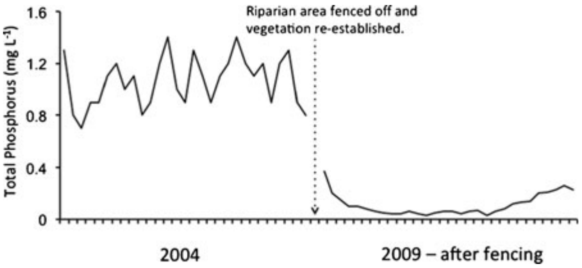

Fig. 6 Concentrations of total P in a stream draining a New Zealand red deer paddock. Captive deer create large wallows in near-stream areas. Access to the stream was restricted and the riparian area re-vegetated, resulting in a 90% reduction in total P concentrations. Adapted from McDowell (2008)

may subsidize streambank fencing, farmer participation is often mixed. Because riparian areas serve as watering sources for cattle as well as shade, alternative amenities away from the stream or controlled access must be considered.

Target critical source areas for cost effective remedial management

Critical source area management has become the dominant approach to targeting agricultural P practices for water quality protection (Sharpley et al. 2003), justified by studies that identified minor areas within watersheds (<20%) contribute to the majority of P loadsin watershed effluent (>80%) (e.g., Pionke et al. 1997, 2000). Critical source area management seeks to identify critical source areas of P export then to target remedial practices to those areas. Critical source area management is considerably more cost effective than approaches that blanket a watershed with remedial practices (Gitau et al. 2006). However, identifying critical source areas where P mobilization is likely to occur is a difficult task, even in areas with a surfeit of agronomic and physiographic data.

Critical source areas are identified with site assessment tools, the P Index being the best known (Sharpley et al. 2003), that apply information on the sources of P (source factors) and on the mechanisms for transferring that P to surface waters (transport factors). Experience and advances in critical source area identification and management have yielded several generations of site assessment tools, some rooted in watershed models and capable of estimating off site P loads (White et al. 2009), others geared toward educating farmers on how best to curb those P loads (Osmond et al. 2006; Sharpley et al. 2011). Ultimately, critical source area management shifts the application of P away from soils that are prone to losses to water and toward soils where losses are less likely to occur. Critical source area management can only remain effective over the long-term if P is in balance at the watershed scale, otherwise this approach simply delays the inevitable, ultimately turning sinks of P into watershed sources.

Experience in the Little Washita River watershed (54,000 ha) of Oklahoma, U.S.A. offers insight into the consequence of ignoring critical source areas (Sharpley and Smith 1994). Conservation practices, including flood control impoundments, treatment of eroded gullies and conservation tillage were installed on about 50% of the watershed. Monitoring of several small sub-watershed (2–5 ha) confirmed that these practices reduced P exportfrom the smaller watersheds in which they were implemented by more than an order of magnitude (Sharpley and Smith 1994). However, despite improvements at the sub-watershed scale, no consistent decline in P concentration was observed at the outlet of the main Little Washita River watershed in which these sub-watersheds were located. The lack of remedial success at the large watershed scale was thought to reflect the unfocused nature of practice implementation, which missed most critical source areas of P within the watershed. In addition, it is likely that the continued release of P already stored within the watershed system contributed to a lack of response at a watershed level.

Establish clear, realistic and relevant management objectives

The establishment of management guidelines for agricultural P in relation to nonpoint sources and water quality impairment can generate considerable controversy, especially when these guidelines extend beyond normal agronomic recommendations. Ideally, remedial strategies to curb agricultural P losses should be clearly tied to specific water quality objectives or use designations. However, causal links between implemented conservation practice and water quality benefit are not easily established. Indeed, the data or models used to describe these links are often a first line of contention, such as in the eastern U.S.A.'s Chesapeake Bay Watershed, where metrics of remedial progress and new rules derived from the Chesapeake Bay Model are seen as arbitrary, scientifically unsupportable and imposing an unfair burden to agriculture (Stallman 2011). While litigation may be inevitable in some corners of the globe, the likelihood for conflict is obviously influenced by public understanding of, and sympathy toward, the basis of the management restrictions. Education, social capital and public participation in the decision-making process help to address some, but not all, sources of conflict (Mullen and Allison 1999).

An initial challenge of any watershed program combating eutrophication is to distinguish between acceptable and unacceptable losses of P. A surprisingly profound step in this process is to determine whether the appropriate metric for evaluation should be loads (kg ha^{-1}) or concentrations (mg L^{-1}) of P in watershed discharge. For example, in the semi-arid prairie province of Manitoba, Canada, P loads to water are very low (0.02 to 0.16 kg P ha^{-1} $year^{-1}$), but concentrations of P in water (0.05 to 0.38 mg L^{-1}) are well above eutrophication thresholds (Salvano et al. 2009). As a result, while P loads do not appear to present a water quality problem in the Manitoba region, high P concentrations at times of greatest biological sensitivity can result in impairment.

Strategies to reduce P concentrations (mg L^{-1}) in water can be quite different from those designed to lower P loads (kg ha^{-1}). Watersheds that are severely impaired by excess P loads may require major changes to management practices, including hydrologic management to dampen high flows (e.g., drainage water management). In comparison, P source management alone may be sufficient to address intermittently high concentrations of P. Regardless of which measure of P loss is selected (P load or P concentration), these thresholds and their interpretation require careful consideration by scientists and policy makers.

Once the appropriate metric for P loss to surface water is determined (i.e., P load or concentration), targets for aquatic ecosystem restoration must consider social, economic and political realities. In the Chesapeake Bay Watershed, a 1987 compact established a target of 40% reduction in P (and N) inputs to the Bay by 2000 to restore ecosystem health (Simpson and Weammert 2007; U.S. Environmental Protection Agency 1987). Initial load reductions were encouraging (~25%) as watershed programs tackled "low hanging fruit," but these reductions were insufficient to meet the ambitious targets. Consequently, new agreements have been promulgated that institute stepped, or phased in, load reductions to improve water quality in the Bay by 2025 (Chesapeake Bay Program 2009). In a case of diminishing returns, it is now recognized that incrementally greater load reductions will be more difficult and costly to achieve. It is also likely that expectations about desired endpoints in coastal ecosystem restoration efforts will be frustrated by shifting baselines in ecosystems like the Chesapeake Bay (Duarte et al. 2009). Therefore, the development of appropriate water quality targets must involve an open and honest dialogue so that technically difficult and politically unpopular decisions are not moderated for expediency.

To retain support over the long-term, eutrophication mitigation programs must convey realistic expectations and provide accurate representation of the uncertainty in watershed management. Given the many sources of P and the complex processes of P transfer within a watershed, anticipating changes in water quality can be perilous, particularly at large, regional scales. Legacy sources and slow nutrient delivery have the potential to severely delay water quality response to remedial strategies or at least to limit short-term water quality benefits of the strategies (Meals et al. 2010). Reversing the legacy of long-term nutrient loadings on an aquatic ecosystem can lag well behind nutrient load reductions. Articulating these factors to the public and to decision makers is essential to setting appropriate expectations. Since our understanding of legacy sources and delivery lags is still developing, most watershed models do not adequately represent these processes. Indeed, many watershed models (EPIC, GLEAMS, ANSWERS, SWAT) have lacked an ability to accurately represent wash-off (incidental transfer) processes for P (Vadas et al. 2007) and in-stream stores and sinks of P (Haggard and Sharpley 2007), even though they have been widely used to project changes in watershed P discharge. Therefore, accurately conveying uncertainty and the state of scientific understanding are key to establishing appropriate expectations for watershed programs, while not using them as excuses for management inaction or softening of land management restrictions.

Conclusions

There is now a solid awareness of the ties between agriculture and anthropogenic eutrophication, refocusing interest on improved agronomic P management. Experience over the past 30 to 40 years has strengthened our understanding of agricultural P sources and the many mechanisms of transfer to water bodies. Likewise, an ever-widening range of efforts to curtail diffuse losses of P from agricultural lands offers insight into successful and unsuccessful strategies to manage P for water quality protection in different situations. While on-farm actions are always required, agro ecosystems with long term challenges such as severe P surpluses require efforts beyond the local farm gate. Indeed, tackling eutrophication is a societal concern that should engage agricultural and non-agricultural communities alike. Throughout this process, many legitimate socio-economic concerns will be raised (e.g., farm profitability, regional development) and a variety of socio-economic tools (e.g., financial incentives, appropriate adaptation periods) will be required to address those concerns. However, we must also remember and respect the fundamental biophysical principles that ultimately determine whether or not agricultural phosphorus losses will be reduced.

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