Critical source area management of agricultural phosphorus: experiences, challenges and opportunities

Andrew N. Sharpley, Peter J. A. Kleinman, Don N. Flaten and Anthony R. Buda

ABSTRACT

The concept of critical source areas of phosphorus (P) loss produced by coinciding source and transport factors has been studied since the mid 1990s. It is widely recognized that identification of such areas has led to targeting of management strategies and conservation practices that more effectively mitigate P transfers from agricultural landscapes to surface waters. Such was the purpose of P Indices and more complex nonpoint source models. Despite their widespread adoption across the U.S., a lack of water quality improvement in certain areas (e.g. Chesapeake Bay Watershed and some of its tributaries) has challenged critical source area management to be more restrictive. While the role of soil and applied P has been easy to define and quantify, representation of transport processes still remains more elusive. Even so, the release of P from land management and in-stream buffering contribute to a legacy effect that can overwhelm the benefits of critical source area management, particularly as scale increases (e.g. the Chesapeake Bay). Also, conservation tillage that reduces erosion can lead to vertical stratification of soil P and ultimately increased dissolved P loss. Clearly, complexities imparted by spatially variable landscapes, climate, and system response will require iterative monitoring and adaptation, to develop locally relevant solutions. To overcome the challenges we have outlined, critical source area management must involve development of a 'toolbox' that contains several approaches to address the underlying problem of localized excesses of P and provide both spatial and temporal management options. To a large extent, this may be facilitated with the use of GIS and digital elevation models. Irrespective of the tool used, however, there must be a two-way dialogue between science and policy to limit the softening of technically rigorous and politically difficult approaches to truly reducing P losses.

Key words | agricultural landscapes, animal manure, fertilizer phosphorus, leaching, phosphorus indices, surface runoff, water quality

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INTRODUCTION

The accelerated eutrophication of freshwaters and, to a lesser extent, coastal waters is primarily driven by phosphorus (P), which has led to the P-based management of point and nonpoint sources (Schindler *et al.* 2008; Dale *et al.* 2010). While efforts to identify and limit point source inputs of P to surface waters have seen significant progress, nonpoint sources have remained more elusive and more difficult to identify, target, and remediate (Dubrovsky *et al.* 2010; U.S. Environmental Protection Agency 2010). As improvements in wastewater treatment technology to further lower discharge concentrations become prohibitively costly, attention has shifted to nonpoint sources, with an emphasis on developing strategies to curb agriculture's contributions to surface water P loadings (Hilton *et al.* 2006; Duriancik *et al.* 2008).

The attention now afforded to agricultural P management has heightened over the last 10 to 20 years, owing, in part, to highly visible cases of accelerated eutrophication, including the Chesapeake Bay, Florida Everglades, Great Lakes, Neuse River, and Gulf of Mexico (National Research Council 2008). Compounding concerns derived from these cases is the more recent admission that eutrophication

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mitigation efforts have not achieved the improvements predicted by watershed models and expected with widespread implementation of conservation strategies (Executive Order 13508 2009; Kovzelove *et al.* 2010).

Thus, there has been a paradigm shift in agricultural P management strategies from unilateral recommendation of conservation measures to address P loss across a watershed toward specific targeting of particular management practices on critical source areas within a watershed. This targeting includes both spatial and temporal dimensions. Spatially, the justification for critical source area management derives from findings that the major proportion of P loss from a watershed (~80%) derives from a small area (~20%) of the watershed (Pionke *et al.* 1997, 2000). Critical source areas within a watershed are essentially P hotspots with active hydrological connectivity to the stream channel (Walter *et al.* 2000; Gburek *et al.* 2007). Temporally, the justification for critical source area management derives from observations that most of the P exported from watersheds occurs during only a few storm events and that there are critical periods when certain management practices (e.g. broadcast P application and tillage) disproportionately exacerbate the risk of P loss.

Critical source area targeting has become the dominant paradigm for agricultural P management, as reflected by the widespread development and adoption of P site assessment indices (Sharpley *et al.* 2003). However, critical source area identification and management is not and cannot be the only tool in mitigating eutrophication related to agricultural sources. There must also be a parallel emphasis on long-term factors, such as farm and watershed scale P imbalances, legacy P sources and vertical stratification of P in soils, to realize fundamental, real, and lasting changes in agricultural P losses. This paper reviews the integration of critical source area management into strategies to curb agricultural P losses, highlights a variety of issues challenging critical source area management, and reveals opportunities for improved P management.

THE EVOLVING PHOSPHORUS INDEX

In the U.S., a site assessment tool, or P Index, was proposed in 1993 and eventually adopted into the U.S. Department of Agriculture's Natural Resource Conservation Service (NRCS) Conservation Practice Standard for nutrient management (i.e. the NRCS 590 Standard). The P Index was designed to identify and rank critical source areas of P loss based on site-specific source factors (soil P, rate, method, timing, and type of P applied) and transport factors (runoff, erosion, and proximity to streams) (Lemunyon & Gilbert 1993). The fundamental advantage of the P Index is to enable targeting of remedial management to critical source areas where high P source and transport potential coincide. This approach differs profoundly from others that are based solely on soil P concentrations. Although they require more information on site source and transport conditions, P Indices more reliably identify nonpoint sources of agricultural P and provide greater flexibility in remedial options and more cost-effective management recommendations.

Currently, 47 U.S. states have adopted the P Index as a site assessment tool to identify critical source areas and target remedial practices (Sharpley *et al.* 2003). In addition, versions of the P Index have been proposed for several Canadian provinces and Scandinavian countries (e.g. Finland, Norway, and Sweden). As different versions of the P Index have emerged, ostensibly to account for local topographic, hydrologic, soil, land use, and policy conditions, so too have differences in the P management recommendations that are made using the P Index. A survey of 12 P Indices from states in the southern U.S. revealed major differences in the way that Indices, even those from neighbouring states, rated site vulnerability to P loss (Osmond *et al.* 2006). Differences in management inferences derived from those P Index ratings for the same fields ranged from recommending no restrictions on field management (continue status quo or N-based management) to recommending the most restrictive remedial actions (no further P additions allowed). In addition to an obvious absence of cross-border coordination in Index development, some of this disparity may be attributed to the paucity of validation efforts by individual states to fully justify their version of the P Index. Some states have pursued rigorous validation of the P Index, or at least quantitative calibration of P Index components using tools such as rainfall simulators and unit source watersheds (e.g. Delaune *et al.* 2004; Harmel *et al.* 2005; Butler *et al.* 2010). However, many states have not had the resources, ability, or motivation to test the alternative versions of P Indices they have promulgated. Differences in state P Index performance also point to the complex nature of critical source areas and the inherent difficulty in their identification.

The lesson of Osmond *et al.* (2006), coupled with a poor public understanding of the P Index and public impatience over the slow rate of water quality improvements following P Index implementation, have culminated in a review and revision of the U.S. standard for nutrient management; the NRCS 590 Standard. In regions where P management has been highly politicized (e.g. Chesapeake Bay Watershed), there have been proposals to supplant the P Index with single, soil-based management guidelines that are easier for the public to understand. These proposals force more restrictive outcomes of site assessment, essentially using site assessment to drive local export of manure to other regions, but have had little to do with the substance of the Index itself, which includes soil P as a principal 'source' factor.

Many U.S. state P Indices are currently being revised to address some of the limitations described above. In addition, there has been a movement toward developing versions of the P Index that estimate runoff P loads. A growing number of states (e.g. Iowa, Oklahoma, Wisconsin, and Texas) have unveiled tools that estimate edge-of-field or watershed level P load changes with alternative management scenarios. Such load prediction tools directly report the potential water quality outcome of management changes (e.g. kg P $ha^{-1}\,yr^{-1}$) and are in particular demand by agencies and end users focused on enumerating watershed management outcomes. However, critics argue that the precision of the load predictions belies the uncertainty in the estimations, and that, at a minimum, they are not scalable between field and watershed.

Major advances have been made toward representing P source availability in the P Index, even unearthing failings in established P routines used by most fate-and-transport models (e.g. Vadas *et al.* 2007). However, representation of transport processes has been more elusive. Quantifying flow, a requirement of P load estimation, requires robust models that can reconcile field, landscape and, depending upon the inference scale, watershed hydrologic processes. Thus, debate remains over the appropriateness of using P Indices to predict edge-of-field P loss.

As P Indices evolve to load estimation tools, it is inevitable that they will be applied to the task of reconciling water quality thresholds with watershed management options. One of the first steps in this process is to determine whether the tool should estimate P loads (e.g. kg P $ha^{-1}\,yr^{-1}$) or concentrations (mg L^{-1}). This determination requires input by scientists and policy-makers alike. In the U.S., Clean Water Act regulations for impaired watersheds are based upon load allocations to particular land uses. Therefore, estimating P loads makes sense from the standpoint of regulatory convention. However, trophic response thresholds are generally tied to concentrations in a water body. For instance, in the cold, dry prairie region of Canada, Salvano & Flaten (2006) reported ranges in average P loads of only 0.02 to 0.16 kg P $ha^{-1}\,yr^{-1}$ for 14 regional watersheds. These loads were very low, even though P concentrations in surface waters were well above eutrophication thresholds (from 0.05 to 0.38 mg L^{-1}). Secondly, the interpretive thresholds for determining the degree of change in management required for a critical source area should consider the overall target for P loading or concentrations in the watershed.

CHALLENGES

Challenges facing modern P management have not changed since critical source area targeting became the dominant paradigm. Although the benefits of critical source area management can be readily documented at smaller watershed scales, in-stream processes serve to buffer downstream water quality improvements. In addition, vertical stratification of P in no-till soils and legacy sources of P may overwhelm the short-term benefits of remedial practices applied to critical source areas.

Locally restricted benefits

The local water quality benefit of targeting critical source areas for conservation management within a watershed is shown by work in sub-watersheds of the Little Washita River Watershed (54,000 ha) in central Oklahoma (Sharpley & Smith 1994). Nutrient export from two sub-watersheds (2 and 5 ha) was measured from 1980 to 1994, while conservation practices were installed on approximately 50% of the main watershed. Practices included construction of flood control impoundments, eroding gully treatment, and conservation tillage. Following conversion of conventional-till (moldboard and chisel plow) to no-till wheat (*Triticum aestivum* L.) in 1983, P export was reduced by 2.9 kg $ha^{-1}\,yr^{-1}$ (10-fold; Sharpley & Smith 1994). A year later, shaping eroding gullies decreased P loss five-fold and construction of an impoundment decreased P loss from the sub-watersheds by 13-fold (Sharpley *et al.* 1996). While the benefits of conservation management were observed at a sub-watershed scale (2–5 ha), there was no consistent decrease in P concentration at the outlet of the main Little Washita River Watershed (54,000 ha). The lack of remedial success at a larger scale is most likely a result of in-stream processes and the continued release of 'legacy P' already stored within the watershed system (Meals 1996; McDowell *et al.* 2002; Meals *et al.* 2009).

Vertical stratification of P with reduced tillage

Within our BMP strategies for controlling P loss, well-intentioned BMPs for reducing one form of P loss may, in fact, increase losses of another form, resulting in little net reduction in P loss or water quality improvements. For example, in response to deteriorating water quality in Lake Erie, U.S. in the 1960s and early 1970s, a coordinated, voluntary program was set in place to reduce P loads to the Lake (Baker & Richards 2002). Phosphorus loads have been monitored since 1975 to determine the effect of adopting BMPs such as conservation tillage (~50% no-till) and nutrient management planning (25% less P applied) in predominantly row crop agriculture (mainly corn, soybean, and wheat) in two Ohio watersheds with major tributaries to Lake Erie (Richards *et al.* 2002). As a result, mean annual flow-weighted dissolved P concentrations decreased 86% and total P 44% between 1975 and 1995 (Richards & Baker 2002). Subsequent to 1995, however, annual flow-weighted concentrations of dissolved P increased, while total P continued to decline. The trend of increasing dissolved P and decreasing total P may be attributed to a combination of several factors; an accumulation of P at the soil surface with conversion to no-till cropping and an increase in the proportion of fertilizer and manure broadcast, without incorporation, in the fall and winter (Baker & Richards 2009; Krieger *et al.* 2010).

One of the most important challenges of using a critical source area approach to manage P is to ensure that the tool correctly accounts for the processes and BMPs that control excess losses in a particular situation. For example, similarly to their counterparts in Ohio, researchers in Manitoba, Canada have measured increased losses of P from conservation tillage systems. In this region of the Northern Great Plains, the relatively flat landscapes and cold, dry climate result in a high proportion of runoff occurring during snowmelt, over frozen soils, with the majority of P loss in dissolved forms and relatively little loss of particulate P due to erosion (Tiessen *et al.* 2010). As a result, Tiessen *et al.* (2010) measured P losses that were 12% greater from conservation than conventional tillage in a twin watershed study near Miami, Manitoba, even though conservation tillage significantly decreased sediment and N loss. Furthermore, in Manitoba, P losses are not predicted accurately using existing critical source area tools that have been designed for land and climates where rainfall-induced erosion of particulate P from sloping landscapes is the main process of P transport (Salvano *et al.* 2009).

From these two case examples, it is clear that, whatever strategies are implemented, they should be done in an adaptive manner. The complexities imparted by spatially variable landscapes, climate, and system response will require iterative monitoring and adaptation, to develop locally relevant solutions. For example, system response can vary from a year to several decades and this time generally increases as spatial scale increases.

Legacy sources of P – interaction of historical additions and hydrology

Few management practices are suited to reversing the effects of historical additions of P to sites that serve as critical source areas. Obvious examples of legacy sources are soils that are highly saturated with regard to P. In the Chesapeake Bay Watershed, the University of Maryland Eastern Shore's research farm occupies the site of a former commercial poultry operation, with roughly 30 years of poultry litter application to farm soils in excess of crop requirement. These coastal plain soils are heavily ditched due to shallow regional water tables and possess soil test P concentrations nearly one order of magnitude above the threshold for crop requirement (Kleinman *et al.* 2007). Because of the ditches, nearly all fields may be considered hydrologically active and connected to local surface waters. Annual P loads from field ditches readily exceed 20 kg ha^{-1} yr^{-1} under normal precipitation regimes. Experiments to draw down soil test P by curtailing P application to the soils showed no significant change in soil test P over nearly one decade (Kleinman *et al.* 2011). Therefore, non-traditional practices may be required to address the overwhelming contributions of P from these critical source areas (e.g. Penn *et al.* 2007).

Less obvious legacy sources may be found in hydrologically active areas that possess soil test P at or near the agronomic optimum. Buda *et al.* (2009) monitored contour-cropped fields on a Pennsylvania hillslope, in which the bottom field possessed the lowest relative soil test P (roughly two-fold lower than the other fields). While this bottom field was the only one that did not receive P amendments during the study period, it yielded runoff volumes roughly 50-fold greater than the other fields included in the study. Runoff from the upper fields was largely disconnected from the loads observed from the bottom field. Annual loads of P from this hydrologically active field were >8 kg ha^{-1}, in comparison to 1 kg ha^{-1} or less from the other fields. This study highlights the ability of site hydrology to overwhelm source factors in determining P loss. More importantly, it points to the ability of hydrology to convert a modest source of P into a major P load. In such cases, careful adherence to critical source area management and non-traditional P runoff remediation practices may be required.

OPPORTUNITIES

Developing alternatives

Although P Indices were developed as strategic tools for P-based management, they were never intended to provide an easy or complete solution to the problem of increased P runoff brought about by localized P surpluses. Even so, like many simple tools and models, P Indices can produce black-and white numbers and colour-coded maps, providing a certain level of comfort that change was forthcoming and P loss would subside. The fact that this has not occurred has renewed the call for stricter guidelines for P application to agricultural land. Given the system buffering (on land and in water bodies) and legacy P effects, how long does it take after conservation measures are initiated before meaningful improvements in water quality can be expected or observed? Part of the current concern with critical source area assessment and the P Indexing approach is due to unrealistic expectations of how quickly impaired waters will take to respond following implementation of conservation measures at a larger watershed scale.

Clearly, there is an opportunity for a science-based approach to meet this challenge. If the aim is to significantly decrease nonpoint source loading of P to sensitive waters and we acknowledge system buffering and the slow release of P, it will be difficult to justify the widespread and continued application of P at levels that exceed crop removal. In effect, this might initiate measures to help farmers address the underlying challenge of localized excesses of manure P and provide cost-effective and practical options that encourage the transport of manure from areas with surplus P to areas with a deficit. This may also encourage the development of alternative uses for manure, such as burning for electricity generation and digestion for methane production, to become more economically viable. However, with bioenergy production, the P-rich biochar (remaining after burning) or sludge (remaining after digestion) must still be managed appropriately.

Improved identification of critical source areas using LiDAR digital elevation models

Digital elevation models (DEMs) represent a useful geospatial tool to automatically identify potential critical source areas on the landscape, particularly in regions where topography exerts an important control on surface runoff generation. The resolution of DEMs produced from topographic maps (10–30 m) has often been insufficient to identify critical source areas that occupied portions of small fields or hillslopes. However, there have been recent improvements in DEM resolution (<1 m) through the use of Light Detection and Ranging (LiDAR) technology. These improvements offer new opportunities to identify and map critical source areas and determine if these features are connected to surface waters. This is important because some critical source areas will rarely deliver P to surface waters, while others will frequently deliver P because of their position on the landscape (e.g. near-stream zone) or through other direct connections such as via rills and gullies.

Researchers have demonstrated a number of useful approaches for identifying critical source areas using high-resolution LiDAR DEMs. One of the more common techniques is to calculate the topographic index (Beven & Kirkby 1979), which is defined as the natural logarithm of the upslope contributing area (a) divided by the local slope gradient ($\tan \beta$). High values of the topographic index represent the potential for saturated areas to develop on the landscape and act as source areas for surface runoff. When information on the topographic index is combined with maps of soil P status, critical source areas can be readily identified (Page *et al.* 2005). The network index, a recent modification of the topographic index (Lane *et al.* 2004, 2009), can be used with LiDAR DEMs to identify saturated areas and evaluate their potential to connect to surface waters.

In addition to topographic index approaches, other researchers have mapped potential critical source areas using terrain analysis techniques. For example, work by Heathwaite *et al.* (2005) summarized the use of multiple flow accumulation algorithms (Quinn *et al.* 1991) to map surface and subsurface flow pathways for P transport and to evaluate their connectivity to surface waters. In the flat landscapes of the Netherlands, Sonneveld *et al.* (2006) used LiDAR DEMs to estimate internal versus external drainage at the field scale, and determine the potential for externally drained fields to contribute P runoff to surface waters. Finally, efforts to identify and map rill and gully networks using terrain analysis techniques with LiDAR DEMs (James *et al.* 2007) could eventually aid in determining whether upslope critical source areas directly connect to surface waters.

Clearly, there are a number of promising approaches to identify and map critical source areas on the landscape and assess their connectivity to surface waters using LiDAR DEMs. With an increasing number of states collecting statewide LiDAR datasets, the potential exists to quickly and effectively evaluate critical source areas at relevant management scales.

Establishing threshold criteria

The act of establishing thresholds for management of critical source areas must balance a variety of considerations. Strategically, there are arguments for varying these criteria by watershed, much as management within a watershed varies by site P loss potential. For instance, watersheds that are severely impaired by excess P may require substantial overall reductions in P loading to reach an ecologically appropriate long term target (e.g. P load reductions of 10% or more). Those substantial reductions will, therefore, require substantial changes to management practices. Conversely, due to a variety of social and economic reasons, targets and management changes in the short term might be much more modest, for example, to moderate further increases in P losses (e.g. 0% increase in P load).

RECOMMENDATIONS

At a field and farm level, research has demonstrated that edge-of-field reduction in nutrient and sediment loss can occur within months of changing P management. However, the spatial complexity of watershed systems increases this response time for P as a function of slow release of legacy P stored in soils and fluvial sediments to surface flow pathways. Critical source area tools are fundamentally sound, particularly when used over the short-term (e.g. a one-year planning cycle), but linkages between the implementation of long-term critical source area management and water quality benefits/improvements are still relatively unknown. Hence, there is a need for long-term validation of critical source area tools, especially in light of potential legacy P effects.

A range of voluntary and regulatory measures can be used to encourage implementation of nutrient management strategies as part of conservation programs to protect soil and water resources. In general, the success of these measures relates to how well farmers can afford to implement new management strategies and the concomitant level of support or incentives for their adoption. Clearly, continuing educational efforts with both the public and farmers regarding the importance and impact of BMPs on water quality are needed to achieve environmental goals and criteria. In some instances, local or regional governmental controls may be necessary to enhance prompt adoption of practices that will have a positive influence on environmental outcomes.

As we have moved from nutrient management that improves crop production to the environmental quality arena, measures have become more costly to farmers, which raises the old dilemma 'who benefits and who pays?' It is important to recognize that market prices do not always motivate farmers to manage nutrients in an environmentally sustainable way. Consumers can be given a choice about which products they buy, with premiums paid to farmers who provide more environmentally friendly products. Relatedly, another important question that should be asked of the public and agricultural communities is what price are we willing to pay for cheap clean water and low-cost agricultural grains, protein, and milk?

In some areas, we cannot and should not expect that pristine waters are achievable with ever increasing population densities and more intensive agricultural production systems to meet market demands. The bottom line is that this may require either a reassessment of water-use designations and/or far-reaching societal commitment and support of agricultural system changes.

Finally, to promote a proactive role for agriculture in this process, we need to make sure that we develop and use a collection of reliable and appropriate tools for critical source area identification and management. All these tools must consider the role of BMPs in mitigating P loss from land to water, as well as water quality targets, to effect lasting and meaningful changes. To be successful, the development and implementation of these tools will require an honest and forthright two-way dialogue between those developing the foundational science of these tools and those making and implementing nutrient management policies. Such dialogue will be essential to limit the softening of technically rigorous and politically difficult approaches to truly reducing excess P loading.

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