Nutr Cycl Agroecosyst (2016) 104:373–392
DOI 10.1007/s10705-015-9748-8

PERSPECTIVE

Conservation practice effectiveness and adoption: unintended consequences and implications for sustainable phosphorus management

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Received: 6 April 2015 / Accepted: 12 October 2015 / Published online: 15 October 2015

Abstract Phosphorus (P) runoff from agricultural land continues to receive attention due to a widespread lack of reduction in losses combined with a series of high profile P-induced harmful algal blooms. Many widely adopted conservation practices (CPs), aimed at reducing P loss, target particulate P (PP) through reductions in erosion or entrapment of P within the terrestrial landscape. However, there is increasing evidence that in time, these CPs may in fact increase dissolved P (DP) losses. We reviewed the effectiveness of current CPs promoted in the U.S., the results from long-term in-stream monitoring following implementation of conservation schemes and field studies investigating P loss from buffer zones designed to trap PP. These studies showed that different CPs are required to target different forms of P loss and the tendency for farmers to implement strategies targeting PP over DP resulted in an increase in dissolved reactive P export post-implementation of 37–250 % in three of the five catchment monitoring studies. Buffer zones, such as grass and vegetative filter strips, managed riparian zones and wetlands were found to accumulate labile forms of soil P over time and, in some studies, became significant sources of both inorganic and organic DP. Furthermore, often overlooked microbial processes appear to play a key role in P release. Consequently, to improve the effectiveness of future conservation schemes, practices need to specifically target DP losses in addition to PP and recognize that CPs trapping P within the landscape are at risk of becoming legacy P sources.

Keywords Agricultural runoff · Conservation practices · Nutrient management · Surface runoff · Buffer zones · Water quality

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Introduction

Phosphorus (P) is an essential element for plant and animal growth and, thus, an important component of soil fertility in productive agriculture. However, diffuse losses of P from agricultural sources, while small in agronomic terms, can be environmentally significant and have been linked to the accelerated eutrophication of many streams, rivers and lakes (Carpenter et al. 1998; Smith et al. 2015b). Running alongside water quality issues, there are increasing concerns over the future supply of mineral P fertilizers (Cordell et al. 2009; Gilbert 2009), highlighting the need for more sustainable P use.

Efforts to reduce P loss from agriculture are driven by society's desire for clean, ecologically healthy waters and the large economic cost of dealing with the impacts of eutrophication. Dodds et al. (2008) estimated that on an annual basis, eutrophication costs the U.S. economy U.S.$2.2 billion due to decreased recreational use of impaired waters, decreased value of waterfront properties, restoration efforts for threatened and endangered species, and providing alternative drinking water. Consequently, a suite of conservation practices (CPs) aimed at decreasing P loss from agriculture have been developed and are promoted by the U.S. Department of Agriculture, Natural Resources Conservation Service (USDA-NCRS) (Table 1).

Despite significant economic investment and large scale implementation of such CPs at the field scale; however, there has been limited measured improvement in water quality at the catchment scale in both Europe and U.S. (Kronvang et al. 2005; Jarvie et al. 2013). For example, over 20 years of intensive schemes and large scale investment aimed at decreasing nitrogen (N) and P loads to the Chesapeake Bay have failed to result in major ecological improvements (Chesapeake Bay Foundation 2014; Sharpley et al. 2009b). The disconnect between reductions in P loss at the field scale and improvements in water quality at the catchment scale reflects the complexity of interactions between terrestrial soil processes, hydrological controls, in-stream P release and retention mechanisms, contributions from other non-agricultural sources, such as rural septic tanks and wastewater treatment plants, and the complex food webs in freshwater systems. Furthermore, there is growing recognition of the chronic and ubiquitous release of P from soils enriched in P as a result of past fertilization management, which buffer mitigation efforts and result in long lag times between implementation and measured improvements in water quality (Hamilton 2012; Meals et al. 2010).

The ability of soils or sediments to act as a sink for P depends on a complex mix of physical, chemical and biological processes. Changes in land management, as occurs following the implementation of CPs, alter these processes and it is unclear how this may impact soil P retention or release. For instance, many CPs focus on reduction in sediment loss and associated P via reductions in erosion or physical entrapment of nutrient-rich particulates present in surface runoff before they reach the watercourse but, do little to address dissolved P (DP) loss (Table 1). In fact, adoption of some CPs, such as no-till, have been shown to increase DP loss in some cases (e.g., Gaynor and Findlay 1995; Sharpley and Smith 1994). Similarly, accumulation of high P soil particles in critical areas adjacent to the stream, e.g., riparian buffers zones, can decrease their effectiveness over time and even transform buffer zones from P sinks to P sources (Hoffmann et al. 2009). Characterization of the sinks and stores of P, often termed legacy P, within a catchment, and assessment of the effectiveness of CPs in dealing with legacy P sources is essential to predict the expected outcome of restoration schemes.

This paper considers the emerging challenges of reducing P loss to surface waters through agricultural management. In particular, we review current literature to assess the effectiveness of typical U.S. conservation practices investigating the following hypothesis: The lack of success of large scale conservation schemes in improving P status in sensitive catchments is, in part, due to; (1) A focus on reducing particulate (PP) losses with limited emphasis on DP loss, and (2) the widespread adoption of conservation practices which trap P within sensitive areas of the landscape which over time transition from P sinks to P sources.

Conservation practices in the US: effectiveness for PP and DP loss at the field scale

Table 1 lists the CPs promoted by USDA-NRCS aimed at decreasing P loss from agriculture. They can be broadly split into three categories: farm inputs, source management and transport management. Source and transport measures are further categorized by USDA-NRCS as avoid, control and trap measures. Avoid measures aim to limit the loss of nutrients to runoff through reductions in the amounts of P applied and its availability in the soil, e.g., nutrient management and soil P testing, thus can generally be considered source management. Control measures target transport pathways and involve the adoption land management practices which increase infiltration, reduce runoff and erosion, e.g., adoption of no-till practices and the introduction of cover crops. The final line of defense are trap measures which physically retain nutrients and particulates in the terrestrial landscape before they reach the receiving waterbody, e.g., buffer strips, riparian corridors, wetlands.

A large volume of research has been conducted to evaluate these strategies at the field scale. Table 1 summarizes the effectiveness of the promoted CPs based on results from field trials and modelling studies

Table 1 USDA-NCRS promoted conservation practices to minimize the loss of P from agriculture and the range of effectiveness for dissolved P (as DRP) and particulate P found in field experiments across the U.S. and Canada

Practice	Description	Effectiveness (% reduction)		References
		Dissolved P	Particulate P	
Farm inputs				
Dietary P	Match animals nutritional requirements to feed	13–89	<1	Ebeling et al. (2002)[b], Ghebremichael et al. (2007)[a], Hamrahan et al. (2009)[b] and Jokela et al. (2012)[b]
Corn hybrids	Use of low phytic-acid corn in feed to reduce manure P	45–48	Negligible	Leytem et al. (2008)[c], Penn et al. (2004)[c] and Smith et al. (2004a)[b]
Feed additives	Addition of phytase enzyme to increase P utilization	n.s.—52	Negligible	Penn et al. (2004)[c] and Smith et al. (2004a, b)[b]
Source management/Avoid				
Nutrient management	Rate—based on soil testing. P inputs are based on crop requirements	10 % reduction for every 10 % reduction in STP	n.d.	Vadas et al. (2005)[d]
		>80 % reduction moving from N based to STP based litter application	Negligible	Sharpley et al. (2009a)[e]
		60–88 % reduction moving from N based to P based litter application	Negligible—67	Eghball and Gilley (1999)[b], Miller et al. (2011)[b] and Sweeney et al. (2012)[b]
	Application timing—apply during seasons with low runoff potential	41–42	Negligible	Schroeder et al. (2004)[b] and Sharpley (1997)[b]
	Application method—incorporate, band or inject P into conservation tilled soil	20–98[i]	n.s.—60 % increase	Eghball and Gilley (1999)[b], Kibet et al. (2011)[b], Kimmell et al. (2001)[e], Little et al. (2005)[b], Rotz et al. (2011)[a], Tarkalson and Mikkelsen (2004)[b] and Sweeney et al. (2012)[b]
Conservation crop rotation	Sequence different rooting depth and plant acquisition mechanisms to optimize soil P uptake	85	Negligible	Smith et al. (2015a)[e]
Soil inversion	Reduce P enrichment in topsoil	n.s.—92	Negligible—59 % increase	Quincke et al. (2007)[b], Sharpley (2003)[b] and Smith et al. (2007)[b]
Transport management/Trap				
Conservation cover	Permanent vegetative cover to increase soil infiltration and remove sediment bound P	n.s.—63	65–90	Sharpley and Smith (1991)[d] and Zhu et al. (1989)[e]

Table 1 continued

Practice	Description	Effectiveness (% reduction)		References
		Dissolved P	Particulate P	
Buffer strips/riparian zones	Slows flow, increases infiltration and removes sediment bound P	−258 to 88	35–96	Abu-Zreig et al. (2003)[b,k], Blanco-Canqui et al. (2004)[b], Chaubey et al. (1995)[b], Daniels and Gilliam (1996)[e], Dillaha et al. (1989)[b], Schmitt et al. (1999)[b], Lee et al. (2000)[b], Lowrance and Sheridan (2005)[e,k]
Constructed wetlands	Removes sediment bound P	−72[j] to 94	47–70	Beutel et al. (2014)[e,k], Jordan et al. (2003)[e,k], Kovacic et al. (2000)[e,k], Maynard et al. (2009a, 2009b)[e,k] and Pietro and Ivanoff (2015)[d,k]
Transport management/Control				
Strip cropping/contour tillage/terraces	Reduces erosion and transport of sediment bound P	Negligible—53	32–91	Alberts et al. (1978)[g], Gassman et al. (2006)[f] and Langdale et al. (1985)[g]
Conservation tillage	Reduces erosion and increases infiltration	−308 to −40	−33 to 96	Gaynor and Findlay (1995)[e], Schreiber and Cullum (1998)[e], Sharpley and Smith (1994)[e], Shipitalo et al. (2013)[e] and Smith et al. (2015a)[e]
Grass waterways	Slows flow, increases infiltration and removes sediment bound P	−83 to 81	45–89	Gassman et al. (2006)[f] and Smith et al. (2015a)[e]
Drainage water management	Controls drainage to reduce outflow volumes	Negligible—68	15–31	Evans et al. (1995)[d], Littlejohn et al. (2014)[e,k], Tan and Zhang (2011)[e] and Williams et al. (2015)[h]

[a] Model simulation of P loss following implementation of a 25 % reduction in dietary P intake

[b] Rainfall simulation experiments

[c] Reduction in WSP content of manure as a proxy for DP loss

[d] Review article

[e] Edge of field monitoring

[f] Modelling of 30 years reduction cf. baseline simulation using APEX (Edge of field) and SWAT (catchment)

[g] Paired catchment monitoring (5 years)

[h] Monitored tile drain outflow (subsurface pathway)

[i] Depending on method of incorporation

[j] Muskrat colonization in 2010 decreased effectiveness to 4 % for TP and led to a 72 % increase in DRP Release of DRP

[k] % effectiveness measured as difference between inflow and outflow loads, *n.s.* not significant, *n.d.* not determined

across the US and Canada. Many of these studies only considered total P loss, however where P loss was separated into dissolved and particulate forms there is a clear distinction between practices which are effective at reducing DP and those more effective for PP (Table 1).

Reductions in farm inputs through careful manipulation of animal diets can reduce DP loss by up to 91 % (Ebeling et al. 2002) and following the "4 R" approach to nutrient management, which is adding P at the Right rate to match crop needs, in the Right source to ensure correct N:P balance, at the Right time to avoid application within a few days of expected rainfall, and in the Right place through incorporation into the soil profile, was highly effective at reducing DP loss (Sharpley et al. 2009b; Tomer et al. 2014). However, while in some cases reducing the rate of application reduced PP loss (Eghball and Gilley 1999) and some evidence of increased PP loss was seen when manures were incorporated with disc tillage (Eghball and Gilley 1999), the effect of reducing farm inputs and source management on PP transfer was negligible. One key guiding principle around nutrient management is to avoid the accumulation of P within the soil in excess of crop requirements. While there is clear evidence that DP losses are highly correlated with soil test P (STP) concentrations (e.g., Heckrath et al. 1995; McDowell and Sharpley 2001), a study by Withers et al. (2009) suggested the reduction of STP in P enriched soils will have limited impact on PP loss because STP represents only a small proportion of the total P present in soils.

Transport measures, and strategies aimed at minimizing erosion (e.g., conservation tillage, cover crops) and trapping sediment (e.g., buffer strips) were the most effective at reducing PP loss (Table 1). However, the impact on DP was highly variable and implementation of these CPs actually increased the loss of DP in many studies and by more than 200 % for both vegetative buffers and conservation tillage.

Implementation at the catchment scale

The effectiveness of USDA-NCRS recommended CPs has been demonstrated at the field scale but does this relate to improvements in instream water quality? Table 2 summarizes the results from six long-term monitoring studies designed to assess the impact of implementing a range of different conservation practices on P transport to sensitive waterbodies. In the majority of the studies the implemented CPs targeted PP loss, although nutrient management was common to all but one. Significant reductions in TP loading were observed at four of the six sites and during the first 20 years of monitoring in the Lake Erie Basin. However, this appears to be mostly due to a reduction in PP loss and, where measured, the adoption of cover crops, vegetated buffer strips and no till reduced PP loads by 29–70 %. Conversely, DP loading was only reduced in the Cannonsville Reservoir Basin (Bishop et al. 2005) and during the first 20 years of monitoring in the Lake Erie catchment (Richards et al. 2002). In the other studies dissolved reactive P (DRP) loading actually increased over the monitoring period.

Reductions in DP loading in the Cannonsville Reservoir Basin were attributed mainly to improved nutrient management (Bishop et al. 2005). Infrastructure improvements including construction of a manure storage lagoon and improvements in roadways allowed more strategic application of manure in terms of timing, rate and more even distribution to fields further from the watercourse. Similarly the 85 and 88 % reduction in DRP within the Sandusky and Maumee Rivers between 1975 and 1995 was attributed to assumed reductions in P fertilizer inputs indicated by 25–40 % reduction in sales between 1980 and 1995. These reductions in DRP back up the results from field trials reviewed in Table 1 which indicate that nutrient management is the most effective strategy to decrease DP loss.

While some form of nutrient management was involved in all of the conservation schemes implemented in Table 2 this did not result in decreases in DRP loading in four of the six catchments and DRP loads were increased in three of the four catchments, where it was determined. Clearly, the type of agriculture appeared to have an effect on the effectiveness of strategies. Brannan et al. (2000), found that the implementation of CPs which mainly targeted PP, through erosion control and particulate trapping, reduced PP by 70 % (2.09–0.63 kg P ha^{-1}) in the arable dominated catchment but only by 35 % (4.59–3.00 kg P ha^{-1}) in the dairy dominated catchments, while DRP loading was increased to a greater extent in the arable catchment with a 52 % increase for dairy from 1.03 to 1.57 kg P ha^{-1} and a 250 % increase for arable from 0.10 to 0.35 kg P ha^{-1}. While

Table 2 Reduction in total, dissolved and particulate P loads found in U.S. catchment monitoring schemes following the implementation of a range conservation practices

Study	Location	Receiving catchment	Major land use	CPs implemented (main target)[a]	Study type (duration)		% Reduction		
							TP	PP	DP
Bishop et al. (2005)	Cannonsville Reservoir Basin (NY)	Cannonsville Reservoir	Dairy pasture and crop	Manure management (DP) Rotational grazing (PP) Drainage management (DP) Crop management (PP)	Paired catchment study. Pre- versus post-implementation monitoring (2 years pre-, 5 years post-)		n.d.	29	TDP: 43
Brannan et al. (2000)	Owl Run Watershed (VI)	Chesapeake Bay	Dairy pasture and crop	Manure management (DP) Stream fencing (DP + PP) Cover crops (PP) Field strip cropping (PP) Grass waterways (PP)	Pre- versus post-implementation monitoring (3 years pre-, 7 years post-)	Catchment outlet	54	66	TDP: 23 (DRP: −37, DUP: 66)
						Dairy dominated sub-catchment	25	35	TDP: 4 (DRP: −52, DUP: 54)
						Crop dominated sub-catchment	36	70	TDP: −117 (DRP: −250, DUP: −78)
Inandar et al. (2001)	Nomini Creek (VI)	Chesapeake Bay	Crop	Nutrient management (DP) No-till (PP) Filter strips (PP) Stabilization structures (PP)	Pre- versus post-implementation monitoring (3 years pre-, 7 years post-)		4–21	30–41	TDP: −61–86 (DRP: −92, DUP: −83 to −55)
Lemke et al. (2011)	Mackinaw River (IL)	Gulf of Mexico	Crop	Grass waterways (PP) Field buffers (PP) Strip -tillage (PP)	Paired catchment (7 years)		n.s.	n.d.	DRP: n.s.

Table 2 continued

Study	Location	Receiving catchment	Major land use	CPs implemented (main target)[a]	Study type (duration)		% Reduction		
							TP	PP	DP
Locke et al. (2008)	Beasley Later Watershed (MS)	Beasley Lake	Crop	Controlled drainage (DP) Field buffers (PP) Riparian forest planting (PP)	Lake monitoring (12 years)		4	n.d. (SS: 70)	n.d.
Richards et al. (2002)	Sandusky River (OH)	Lake Erie	Crop	Nutrient management (DP)	River monitoring 1975–1995 (20 years)	Sandusky River	46	n.d.	DRP: 88
	Maumee River (OH)			Conservation Tillage (PP)		Maumee River	42	n.d.	DRP: 85
Daloğlu et al. (2012)	Sandusky River (OH)				River monitoring 1995–2008 (13 years)	Sandusky River	n.d.	n.d.	DRP: −70
Michalak et al. (2013)	Maumee River (OH)					Maumee River	−58	−31	DRP: −218

[a] Based on expected effectiveness for each P form taken from Table 1, *n.s.* not significant, *n.d.* not determined, *TP* total P, *PP* particulate P, *TDP* total dissolved P, *DRP* dissolved reactive P, *DUP* dissolved unreactive P, *SS* suspended sediment

these loadings are relatively low, DRP, which is more bioavailable than PP, made up a much greater proportion of TP post CP implementation in both the arable catchment, increasing from only 4 to 21 % and the receiving river, increasing from 15 to 46 %. Although nutrient management was implemented, manure application rates in the arable fields were based on nitrogen requirements and were in excess of crop P requirements which may in part explain the large increase in DRP export. Interestingly export of TDP was not increased to the same extent as DRP. As part of nutrient management manure storage lagoons were constructed to allow for strategic timing of application. Conversion of organic P within manure to orthophosphate during storage was suggested as a potential reason for the reduction in organic P loads and increase in DRP. Thus, the role of biotic processes in influencing P availability are rarely considered, but add to the complexities of P source management and system response.

Information from the long-term monitoring of the Sandusky and Maumee Rivers in Ohio, major tributaries of Lake Erie is particularly enlightening. Implementation of CPs within these catchments dramatically reduced DRP and TP loads by more than 80 and 40 % respectively between 1975 and 1995, and in stream flow weighted mean concentrations of DRP decreased from 0.06 and 0.175 mg P L^{-1} in 1995, to 0.015 and 0.03 mg P L^{-1} in 2008 (Richards et al. 2002). These reductions corresponded with a reduction in P fertilizer inputs, as discussed earlier, and an increased adoption of no-till cultivation from virtually zero in 1975–50 % of cropland in 1995. However, despite these initial improvements, P loading, especially DRP, increased from 1995 onwards; and was 70 and 218 % larger in 2008 compared to 1995 in the Sandusky and Maumee Rivers, respectively. Furthermore annual P loads in the Sandusky River were between 350 and nearly 500 kg P $year^{-1}$ in 2006–2010, which is greater than prior to restoration efforts in 1995, when the annual P load was around 200 kg P $year^{-1}$ (Daloglu et al. 2012). Record breaking algal blooms in Lake Erie have occurred in recent years (Wynne et al. 2013). The most recent of which, in the summer of 2014 resulted in a 2 day shutdown of the City of Toledo's water supply to almost half a million people (Henry 2013; New York Times, August 2014). Modelling analysis of the long-term trends in water quality and changes in land management have implicated the widespread adoption of conservation tillage and broadcast application of fertilizer and manure during fall and winter, without incorporation, as contributors to the increased P loadings (Daloglu et al. 2012; Michalak et al. 2013; Smith et al. 2015a).

Conservation tillage imparts many benefits to the soil, including improving soil health and quality (Karlen et al. 2003, 2014; Sims et al. 1997). However, long-term surface application of P fertilizers and manures in the absence of conventional tillage can lead to accumulation of P in the soil surface (e.g., Cade-Menun et al. 2010; Mathers and Nash 2009; Vu et al. 2010) and encourage the formation of preferential flow paths (Shipitalo et al. 2000). Consequently conservation tillage has been shown to increase DP losses over time (e.g., Gaynor and Findlay 1995; Sharpley and Smith 1994) and may in fact increase PP loss, despite a reduction in erosion and sediment loss due to P enrichment of the particles transported (Gaynor and Findlay 1995). Therefore, management of P application must be adapted to minimize the potential for surface soil accumulation of P (Joosse and Baker 2011; Sharpley et al. 2012).

Adoption of conservation practices

Results from field trials and long-term catchment monitoring clearly show that different CPs are more effective against different forms of P loss and reductions in DP appear to be especially challenging. The main strategy effective at reducing off-farm DP loss is nutrient management (see Table 1), however, it appeared to have limited impact at the catchment scale (Table 2). In addition to the confounding impact of other conservation practices adoption rates play an important part in the success of conservation schemes. Insights into CP adoption and farmer CP decision making have recently been gained through reviewing the National Institute of Food and Agriculture (NIFA) Conservation Effects Assessment Project (CEAP) setup to assess the effectiveness of CPs on water quality across 13 impaired catchments throughout the US. Nutrient management plans were generally disliked by farmers mainly due to distrust around recommendations and a tendency to favor "insurance fertilization" (Osmond et al. 2015). As a result nutrient management plans were generally not fully implemented and in some cases, even after soil testing, corresponding fertilizer recommendations were not followed.

In contrast, practices which targeted sediment and therefore PP loss, namely conservation tillage, grassed waterways and terraces, were most widely adopted, regardless of whether dissolved or particulate pollutants were of most concern (USDA-NCRS 2012). Discussions with farmers have indicated that they are more likely to address water quality issues for pollutants they can see, i.e., sediments than invisible dissolved pollutants (Reimer et al. 2012) explaining the preference for erosion control measures.

In one catchment successful and widespread adoption of nutrient management was achieved but only following the hiring of a dedicated extension officer, highlighting the importance of farmer education and outreach (Osmond et al. 2012, 2015).

In summary, long-term monitoring has highlighted the limited impact and even negative effects of catchment scale implementation of CP on DP loss. Limited implementation of nutrient management plans and an adoption preference for strategies targeting PP over DP along with confounding effects of some of the practices actually increasing DP loss supports our hypothesis that a focus on PP over DP losses is in part responsible for the lack of success of many catchment scale conservation schemes.

Conservation practices increasing P loss: transitions from P sink to P source

Our second hypothesis relates to the unintended consequences of implementing CPs leading to increases in P loss. In addition to the potential negative impacts of adopting conservation tillage discussed above, strategies which focus on the trapping sediment (and associated PP) prior to it reaching the watercourse will gradually accumulate P within the landscape. We hypothesize that over time, vegetative filter strips and riparian buffers, wetlands and grassed waterways will become hotspots of legacy P and have the potential to transition from P sinks to P sources.

The speciation of P accumulating within these areas will depend in part on that of the upslope soils. The more labile soil P forms have been shown to accumulate in the clay size fraction of the soil. For example, following long-term fertilization, Leinweber et al. (1997) found that resin P concentrations were highest in the clay fraction, while H_2SO_4-P, associated with Ca-P forms and residual P accumulated in the silt and sand fractions respectively. Similarly for organic P (P_o), diester and labile monoester phosphates dominated the clay fraction while the more strongly retained inositol phosphates (including phytate-P) mainly accumulated in the sand fraction (McDowell and Stewart 2006). Smaller particles, mainly clays and silts, are selectively eroded during surface runoff and are often transported further than larger, heavier particles, like sand (Sharpley 1985). This selective transport suggests that there would be preferential accumulation of more labile PP within the transition zones. However, finer particles are less prone to deposition and may, thus, remain suspended in runoff and pass straight through these zones with little retention (Owens et al. 2007).

Further complication arises from the potential transformations occurring due to different physiochemical, biological and environmental conditions, such as increased carbon addition from vegetation and leaf litter in riparian zones, wetlands and grassed waterways and changes in redox conditions as a result of periods of inundation, add layers of complexity when considering the long-term impact of CPs which trap P in the landscape.

Buffers, such as grass (GFS) or vegetated filter strips (VFS), managed riparian zones and constructed wetlands, provide a disconnect between the edge-of-field and the watercourse. These areas function by slowing the flow of surface runoff, promoting sedimentation and infiltration, to act as a filter trapping sediment and removing DP through plant uptake (Hoffmann et al. 2009).

Experiments in the U.S. have shown that buffers and wetlands can be highly effective at reducing PP loss at the field scale but, the impact on DP appears highly variable and can in fact increase DP loss (Table 1).

Table 3 provides a summary of field studies assessing the impact of buffers on P loss. All but one of the 16 buffer filter strips and all but two of the five wetlands studies showed a significant reduction in TP loss in surface runoff. The difference in TP load between sites with buffers and sites without ranged from 21 to 96 % and the difference between inflow and outflow TP loads from wetlands ranged from 17 to 80 %. For DP, however, the impact was much more

Table 3 Impact of buffers, wetlands and grass waterways on total P (TP), particulate P (PP) and dissolved reactive P (DRP) losses from field experiments

Study design (location)	Design details	Time since implementation (years)	Additional comments	% Reduction							Reference
				Runoff	TP		PP		DRP		
				Volume	Conc	Load	Conc	Load	Conc	Load	
Grass, vegetated and riparian buffers											
Runoff plot experiment under natural rainfall (U.S. TN)	GFS	2		39–64	n.s.	68–76	n.s.	66–82	n.s.	66–73	Al-wadaey et al. (2012)
Runoff plot experiment under simulated rainfall (U.S. MO)	VFS	1		11–15	n.g.	n.g	n.g.	36–53	n.g.	37–54	Blanco-Canqui et al. (2004)
Edge of field monitoring (IT)	VFS	4		78	22	81	n.d.	n.d.	n.s.	83	Borin et al. (2005)
Runoff plot experiment under simulated rainfall (U.S. AR)	GFS	1		n.g.	67–97	43–90	n.d.	n.d.	68–96	40–88	Chaubey et al. (1995)
Runoff plot experiment under simulated rainfalll (U.S. VI)	GFS	6 simulated rainfall runs	Run 1–3	20–84	n.g.	67–95	n.g.	68–96	n.g.	−31 to 79	Dillaha et al. (1989)
			Run 4–6	−19 to 71	n.g.	34–91	n.g.	36–93	n.g.	−258 to 60	
Runoff and drainage plot experiment under natural rainfall (Ca)	GFS and VFS	1	Surface runoff	35–40	n.g.	85–86	n.d.	n.d.	n.g.	50–67	Duchemin and Hogue (2009)
			Drainage	−18 to 8		−418 to −345	n.d.	n.d.	n.g.	−33	
			Total	15		76	n.d.	n.d.	n.g.	22–33	
Runoff plot experiment under simulated rainfall. (U.S. IA)	GFS and VFS	4		7–21	n.g.	46–93	n.d.	n.d.	n.g.	28–85	Lee et al. (2000)
Edge of field monitoring. Outflow compared to inflow. (U.S. GA)	3 zone buffer	4	Zone 1 (grass)	68	n.s.	67	n.s.	56	n.s.	62	Lowrance and Sheridan (2005)
			Zone 2 (pine)	66	49	67	71	56	44	64	
			Zone 3 (hardwood)	44	n.s.	80	43	63	8	79	
Runoff plot experiments with natural rainfall. (FI and S)	GFS and VFS	7	Grass harvested annually	15	n.g.	38	n.d.	n.d.	n.g.	14	Uusi-Kämppä et al. (2000)
		7	Vegetation remains	n.s.	n.g.	27	n.d.	n.d.	n.g.	−64	
		3	Grass remains	−71	n.g.	−37	n.d.	n.d.	n.g.	−33	
Edge of field monitoring. (F)	GFS–grass harvested	9 CT	Conventional tillage (CT)	20	n.g.	36	n.g.	44	n.g.	7	Uusi-Kämppa and Jauhiainen (2010)
		9 CT+3 G	Grazed (G)	10	n.g.	13	n.g.	10	n.g.	18	
		9 CT+3 G+2 DD	Direct Drill (DD)	1	n.g.	14	n.g.	23	n.g.	8	
	VFS–vegetation remains	9 CT	Conventional tillage (CT)	10	n.g.	28	n.g.	48	n.g.	−60	
		9 CT+3 G	Grazed (G)	16	n.g.	21	n.g.	−7	n.g.	36	
		9 CT+3 G+2 DD	Direct Drill (DD)	30	n.g.	23	n.g.	28	n.g.	23	

Table 3 continued

Study design (location)	Design details	Time since implementation (years)	Additional comments	% Reduction							Reference
				Runoff	TP		PP		DRP		
				Volume	Conc	Load	Conc	Load	Conc	Load	
Edge of field monitoring. (NL)	GFS–grass harvested	3–4	Shallow sub-surface flow	n.g.	n.g.	61	n.g.	62	n.g.	62	Noiji et al. (2013)
			Deep subsurface flow	n.g.	n.g.	n.s.	n.g.	n.s.	n.g.	n.s.	
			Tile drained	n.g.	n.g.	n.s.	n.g.	n.s	n.g.	n.s.	
Constructed wetlands											
Wetland treats agricultural irrigation water, measured difference between inflow and outflow (U.S. WA)	Sediment basin connected to 2 surface flow wetlands (Total area: 1.6 ha)	7	Sediment basin (year 1/year 7)	n.d.	18/16	n.d.	n.d.	n.d.	13/29	n.d.	Beutel et al. (2014)
			Wetland 1 (year 4/year 7)	n.d.	46/30	n.d.	n.d.	n.d.	70/36	n.d.	
			Wetland 2 (year 4/year 7)	n.d.	36/29	n.d.	n.d.	n.d.	70/24	n.d.	
Wetland treats drainage from an agriculturally dominated catchment, difference between inflow and outflow (U.S. MD)	1.3 ha restored catchment scale wetland	11		n.g.	n.g.	−11 to 59	n.d.	n.d.	n.g.	−18 to 53	Jordan et al. (2003)
Wetland treats drainage from farm tile drains, difference between inflow and outflow (U.S. IL)	3 Small farm scale wetlands	3	Wetland 1 (year 1/year 3)	n.g.	n.g.	17/35	n.d.	n.d.	n.g.	17/47	Kovacic et al. (2000)
			Wetland 2 (year 1/year 3)	n.g.	n.g.	80/38	n.d.	n.d.	n.g.	90/38	
			Wetland 3 (year 1/year 3)	n.g.	n.g.	60/−54	n.d.	n.d.	n.g.	15/−27	
Grass waterways											
Paired catchment study—2 catchments with grass waterway, 2 without (GE)	Unmanaged waterway and annually cut waterway	5	Unmanaged	90	n.g.	n.g.	n.g.	76	n.s.	31	Fiener and Auerswald (2009)
			Annually cut	10	n.g.	n.g.	n.g.	88	n.s.	n.s.	
Runoff plot experiment. Inflow concentrations compared to outflow (U.S. OH)	Simulated grass waterways	2		n.g.	n.g.	n.s.	n.d.	n.d.	15	n.s.	Shipitalo et al. (2010)
Edge of field monitoring. Before and after implementation compared (U.S. IN)	Established on empherial gullies	6		n.g.	n.s.	n.s.	n.d.	n.d.	−61	−478	Smith et al. (2015a)

n.d. not determined, *n.g.* not given, *n.s.* not significant variable. Four of the 16 buffer strips appeared to be sources of DP and load reductions ranged from −258 % (Dillaha et al. 1989) to 88 % (Chaubey et al. 1995). Similarly to TP, two of the five wetlands appeared to release DP and reductions in load ranged from −27 to 90 % (Kovacic et al. 2000). While DP load reductions were observed in many studies, much of this impact was attributed to reductions in runoff volumes and where provided, the impact on DP concentration was often not significant (Al-wadaey et al. 2012; Borin et al. 2005; Lowrance and Sheridan 2005).

Evidence for reduction in buffer effectiveness over time was demonstrated by Dillaha et al. (1989) in a series of rainfall simulation experiments to experimental buffer plots. Increased TP loads were found between the first set of three simulated runoff events and the second set of three runoff events for all buffer plots and an increase in DRP load for three of the six plots. Hence, reductions in TP and DRP compared to the no buffer plots were much less during the second rainfall simulation and release of DRP from the buffer plots was increased (Table 3). Investigation of the effectiveness of field scale buffer zones has also shown a reduction in effectiveness over time (Table 3). The most effective buffers were between 1 and 4 years old. With the exception of one site where runoff, TP, DRP and PP losses were increased compared to a non-buffered control (Uusi-Kämppä et al. 2000), reductions in TP and DRP load in surface runoff ranged from 43 to 93 % and 22–88 % respectively compared to reductions of 13–38 % for TP and −64–36 % for DRP in buffers over 7 years old. Beutel et al. (2014) found a small decrease in TP load reductions from a constructed wetland treating irrigation runoff between years four and seven of operation and a large decrease in DRP reductions over this time from 70 % in year four to only 24–36 % in year seven. Similarly, Kovacic et al. (2007) found reductions in effectiveness of two out of three wetlands receiving tile drainage from cropland over the first 3 years of operation, over which time one wetland became a source of DRP.

While only a limited number of studies have been conducted on grassed waterways (Table 3), unmanaged grassed waterways established for 5 years were found to decrease DRP loading by 30 % in a paired catchment study in Germany as a result of reduction in runoff volume, but no significant effect on DRP concentration was found (Fiener and Auerswald 2009). However, Smith et al. (2015a) found that grassed waterways acted as a large source of DRP in the growing season after establishment in arable fields within the Lake Erie catchment. Grassed waterways act as a direct conduit for runoff to reach the watercourse so release of P from these zones is likely to have a large impact on water quality.

Processes controlling P release

Release of DP from buffer and wetland soils in many studies has been attributed to the accumulation of labile soil P pools. Studies comparing soil P concentrations in 10 buffer soils and one wetland soil to adjacent field soils showed a varying response in soil test P (STP) (Table 4) from no significant difference to increased and decreased concentrations. However, the bioavailable fraction, estimated through extraction with water or weak salt solutions, was dramatically increased by 39–146 % in all but two of the sites. While we were unable to find studies documenting the soil P concentrations in grassed waterways the large retention of PP (76–88 %) found by Fiener and Auerswald (2009) (Table 3) indicates the likely accumulation of P within these structures. Furthermore, many studies have documented the accumulation of bioavailable forms of P within agricultural drainage ditches (Nguyen and Sukias 2002; Sallade and Sims 1997; Vaughan et al. 2007).

The main mechanisms for P release from these transition zone soils is through (i) abiotic desorption and dissolution of P and a reduction in P retention due to saturation of sorption sites and (ii) biotic release following nutrient cycling through the microbial and plant pools.

Stutter et al. (2009) carried out incubation and leaching experiments to determine the importance of the different mechanisms in the remobilization of P from a range of buffer strip soils with varying STP concentrations. The addition of high P sediments to these soils, mimicking the accumulation of PP, did not decrease P sorption capacity and did not increase DRP leaching. This has been attributed to the increase in sorption sites following sediment addition along with added P. Further experiments found that manipulating conditions which promoted increased microbial activity promoted DRP release and increasing the soil temperature from 5 to 20 °C along with addition of P

Table 4 Percent difference in soil P fractions in buffer and wetland soils compared adjacent agricultural fields

Source (location)	Age	STP (method)	% increase versus field soils			
			Bioavailable P (method)	DOP (method)	MBP	Reference
Riparian buffer/UK	3	n.s (Olsen)	39 % (1 mM NaCl)	23 % (1 mM NaCl)	34 %	Stutter et al. (2009)
Riparian buffer/UK	8	n.s.	146 % (1 mM NaCl)	742 % (1 mM NaCl)	227 %	
Riparian buffer/UK	n.g.	56 % (Olsen)	125 % (WSP)	155 % (WSP_o)	212 %	Roberts et al. (2013)
Riparian buffer/U.S.	13	−0.43 % (Bray-1)	52 % (EPC_0)	n.d.	n.d.	Schroeder and Kovar (2008)
Harvested grass buffer/FI	17	n.s.	n.d.	n.d.	n.d.	Uusi-Kämppa and Jauhiainen (2010)
Vegetated buffer/FI	15	50 % (P_{AC})	n.d.	n.d.	n.d.	
Riparian buffer/NZ	4–5	208 % (Olsen)	65 % (WSP)	n.d.	n.d.	Cooper et al. (1995)
Riparian buffer/NZ	4–5	−25 % (Olsen)	36 % ($CaCl_2$-P)	n.d.	n.d.	Aye et al. (2006)
Riparian buffer/NZ	4–5	21 % (Olsen)	58 % ($CaCl_2$-P)	n.d.	n.d.	
Riparian buffer/NZ	4–5	−32 % (Olsen)	−14 % ($CaCl_2$-P)	n.d.	n.d.	
Wetland/NZ	4–5	−63 % (Olsen)	−50 % ($CaCl_2$-P)	n.d.	n.d.	

rich sediments increased DRP leaching from 0.05 to 150 mg P L^{-1}. Their findings suggest that saturation of P sorption sites had not occurred, even in the soils with the highest STP and that cycling through the microbial biomass plays an important role in remobilization of P.

This is supported by the buildup of the microbial P pool over time following establishment of riparian buffer zones and the large increase in the microbial P pool in riparian soils compared to adjacent field soils found by Stutter et al. (2009) and Roberts et al. (2013) (Table 4). The potential for microbial accumulation in drainage ditch sediments had also been demonstrated with 10–40 % of DRP reductions in fluvarium experiments attributed to microbial uptake (Sharpley et al. 2007). Microbial processes play a large role in the soil organic P cycle and Stutter et al. (2009) and Roberts et al. (2013) also showed a large increase in the labile organic P fraction extracted by either 1 mM NaCl or water (Table 4). The P_o fraction made up 90 % of the total P extracted with 1 mM NaCl in 8 year old VFS in the U.K. (Stutter et al. 2009) and 34 % of water extractable P in the 12 VSF soils investigated by Roberts et al. (2013), highlighting the role of P_o cycling within these zones.

Of the field studies on buffer effectiveness presented in Table 3 only two studies reported total dissolved P (TDP) concentrations (Dillaha et al. 1989; Noiji et al. 2013). In these studies dissolved unreactive P (DUP), generally considered to be mainly P_o, made up 20–50 % of TDP. Similarly, in the plot study by Dillaha et al. (1989), where buffer soils increased DRP concentrations relative to no buffer plots, there was a corresponding increase in DUP concentration of a similar magnitude of up to 200 %. At the catchment scale the impact of CPs on DUP appears mixed. Dissolved unreactive P was determined for two of the six locations presented in Table 2, Owl Run (Brannan et al. 2000) and Nomini Creek (Inandar et al. 2001). In the Owl Run catchment there was a decrease in DUP export from the dairy dominated catchment which resulted in an overall, small decrease in TDP loading, but an increase in DRP loads, attributed to changes in manure speciation during storage. However, in the arable dominated catchment where transport focused CPs were implemented, including grass waterways and strip cropping, there was a significant increase in both DUP and DRP loading resulting in a 117 % increase in TDP export. The Nomini Creek catchment is dominated by row crop agriculture and implementation of transport CPs, including buffers and no-till, increased DUP loads by 55–83 %, and DRP loads by 92 %. In both catchments, DUP made up a significant proportion of TDP, ranging from 25 and 78 % in the arable and dairy dominated catchments of Owl Run respectively and >80 % in Nomini Creek. The dominance of DUP and the bioavailability of some P_o forms to algae (Whitton et al. 1991; Whitton and Neal 2011), indicates that the impact of CPs on dissolved P_o export needs to be considered and that the potential of buffer zones to act as sources of P may be underestimated by monitoring programs solely measuring DRP.

In addition to DP, the bioavailability of PP export to waterbodies can be modified by buffer zones and wetlands. Preferential trapping of sand sized particles in buffers indicates that finer particles, which tend to be enriched with P, may remain in runoff passing through the buffer. This may be especially relevant if the physical retention properties of the buffer have been reduced due to the accumulation of sediment over time (Owens et al. 2007). Similarly spatial assessment of particulates stored within wetland soils have shown that clay particles travel further into the wetland and can accumulate near the outflow while less P rich sand particles prudentially accumulate nearer the inflow position (Maynard et al. 2009a). Transformation of PP speciation has also been documented within wetland soils with an increase in the proportion bioavailable P_o fraction found in the outflow of a constructed wetland in California compared to the irrigation water entering the wetland (Maynard et al. 2009b). This fraction was considered to be microbial in origin reinforcing the view that microbial processes play a large role in the bioavailability of both DP and PP reaching the watercourse.

Management and environmental factors influencing P mobilization

Management of buffer zones and the adjoining agricultural fields influences the potential for DP retention or mobilization (Table 3). One of two VFS in Sweden bordering an arable field, was found to be a source of DRP following 9 years of traditional cultivation practices (Uusi-Kämppä et al. 2000), likely due to large accumulation of PP. Implementing 3 years of grazing in the buffer to increase plant uptake, increased DRP retention resulting in the buffer returning to a modest DRP sink in the following 2 years under direct drill management. However, during grazing PP export increased and the buffer became a modest source of PP due to cattle grazing and treading damage on the soil. This highlights the different responses of differing P forms to management and the complexity of developing effective CPs. As this study suggests, management of vegetation within buffer strips influences performance. Buffers where vegetation (grass) was regularly cut and removed showed a reduction in DRP load, while comparable buffers where vegetation was retained were found to be significant sources of DRP (Table 3; Uusi-Kämppa and Jauhiainen 2010). In addition to the removal of soil P through plant harvest and removal the contribution of plant residues at the soil surface in unmanaged riparian zones can make a large contribution to DRP loss especially in cold climates where much of the annual P loss occurs during snowmelt (Lui et al. 2013; Uusi-Kämppa and Jauhiainen 2010).

Transport pathway also influences the effectiveness of buffer zones. Most research has focused on the impact on surface runoff but the effect on subsurface losses remain uncertain (Table 3). Duchemin and Hogue (2009) investigated the impact of VFS on P loss in surface runoff and subsurface drainage from runoff plots under natural rainfall to Canadian manure amended pastures, 1 year after implementation. The VFS significantly reduced TP and DRP load to surface runoff compared to non-buffered controls but there was a large increase in TP and DRP load in drainage waters from 40 mg TP and 4.41 mg DRP per plot per year to 149–183 mg TP and 7.08–7.93 mg DRP per plot per year. At this site, the increase in subsurface P loss did not negate the large reduction in P loss to surface runoff due to the dominance of this pathway in transporting the majority of annual P loads and total P loss was reduced by 7 kg P Ha^{-1}. However, where subsurface drainage is the dominant pathway buffer zones may not be effective CPs for P control. Noiji et al. (2013), compared P loss from pastures and fodder crop fields in the Netherlands bordered by buffer strips to unbuffered controls across a range of hydrological regimes. They found a significant reduction in TP and DRP load of 60 % for fields dominated by shallow subsurface flow, but no reduction in either parameter in fields dominated by deep subsurface flow or tile drains.

Environmental factors can exert control on mobilization of soil P stores. Wetlands, grassed waterways and riparian buffers prone to occasional flooding are all susceptible to changes in redox state. Redox conditions can have a large influence on P mobilization through the reductive dissolution of Fe–P under anaerobic conditions and subsequent release of P (e.g., Surridge et al. 2007; Scalenghe et al. 2010). Additionally large pulses of P can be released from the microbial biomass following wet-drying cycles and freeze–thaw events (Blackwell et al. 2010). The increased accumulation of microbial biomass P in buffer soils compared to field soils (Table 4) will increase this risk.

Adoption considerations

There has been widespread adoption of riparian buffer zones, including grass/vegetative buffer strips and wetlands across Europe and the U.S. (Collentine et al. 2015; Osmond et al. 2012). While it should be remembered that riparian zones provide a wide range of ecosystem benefits, in terms of P mitigation, their impact is unclear (Table 3) and possibly overstated.

It is important to consider the appropriateness of CPs on a site by site basis, implementing a "right strategy, right place" principle. In the U.S., the Farm Service Agency (FSA) administers a voluntary conservation scheme called the Conservation Reserve Program (CRP) where farmers receive an annual payment to set-aside land from agricultural production, and convert it to VFS, grass waterways and riparian buffers (FSA-USDA 2015 http://www.fsa.usda.gov/programs-and-services/conservation-programs/conservation-reserve-program/index). Sprague and Gronberg (2012) evaluated the relationship between total N and P export on 133 agricultural catchments across the U.S. and the area of land under CRP, which was roughly 8 % of total U.S. cropland in 2002. Modeling results showed that increased export of TP was significantly associated with an increase in area of land under CRP but that the association was stronger as the erodibility of soils, hence the proportion of PP transported, decreased.

Hydrology of the site can have a large influence on effectiveness of buffers. To effectively decrease P transport, run-off needs to flow evenly across the length of the buffer, i.e., via sheet flow (Hoffmann et al. 2009). However, during intense storms, runoff flow often converges into narrow flow paths, defined by site specific conditions, such as topography, soil type and physical characteristics, which may be influenced by land management (e.g., soil compaction). Surface runoff along narrow flow paths will concentrate sediment and nutrient transport through a small area of the buffer zone (Owens et al. 2007) and may rapidly saturate the physical retention capacity for both sediment and DP. Field observations by Dillaha et al. (1989) indicated that farmers implemented VFS following federal cost-share programs with little consideration of site hydrology. Hence, the majority of filter strips implemented on 18 farms in Virginia for water quality improvement purposes were ineffective due to generation of concentrated flow paths. Consequently, in some cases careful design and maintenance of buffers at specific locations may be more effective than widespread implementation of a set buffer zone.

In summary, enrichment of buffer zone soils with labile forms of P, compared to the contributing agricultural fields, has been demonstrated, suggesting that soils in buffer zones can be a significant source of both inorganic and organic DP. Field studies have demonstrated that the effectiveness of buffer strips and wetlands decreases over time and that the risk of P release increases under certain management (Table 3). Such release of DP occurs for example, when protecting fields under conventional tillage and where plant material remains on the soil surface, and under certain environmental conditions, for example episodic flooding. This supports our second hypothesis that over time CPs that trap P within the landscape can become P sources instead of sinks. From the catchment monitoring trials shown in Table 2, no decrease in DRP loads were found for schemes where buffer strips or grassed wetlands were implemented and in two of the four studies there was a 37–92 % increase in DRP loads post implementation.

Conclusions

Current conservation schemes are often not sufficient to meet water quality targets. A review of field studies assessing the effectiveness of recommended CPs in the U.S. indicated that different CPs need to be adopted depending on the pollutant form and that many CPs designed to reduce sediment and PP transport can increase the export of DP. Furthermore CPs which act by trapping P within the landscape, such as buffers, wetlands and grassed waterways can accumulate P over time and become enriched with labile forms of P compared to adjacent field soils. This may result in these P sinks transitioning into P sources. Consequently, in-stream water quality following CP implementation in catchment monitoring studies often shows reductions in PP export but no change, or even increases in DRP loading.

Of the current USDA-NCRS recommended CPs, nutrient management practices are the only strategies that consistently decrease DP losses (Table 1). However, while included in most catchment scale conservation schemes implementation of nutrient management plans meets farmer resistance. Furthermore, the fact that farmers are more reluctant to recognize the issue of dissolved nutrient losses compared to the easily visualized loss of particulates and studies, suggest that extensive education and outreach is required to improve nutrient management plan adoption (Osmond et al. 2015). Clearly, reductions in DP losses with current CPs are proving difficult. This has also been exacerbated by widespread adoption of CPs targeting erosion and PP which have been shown in some cases, to increase DP loss. For example conservation tillage and the widespread implementation of buffers and wetlands which accumulate P within the landscape.

Furthermore, due to the large timescales involved in decreasing soil P concentration through reduction or cessation of P inputs (Dodd et al. 2012; Meals et al. 2010), additional strategies to reduce DP losses from P enriched soils are urgently required. Possible strategies include vertical tillage and destratification of no-till soils to redistribute P accumulated at the soil surface (Kleinman et al. 2015), phytoextraction of P to accelerate the decline in soil P pools (Koopmans et al. 2004; van der Salm et al. 2009), especially in buffer zones and the use of Al or Fe industrial wastes as soil amendments to convert VFS soils acting as a DRP source into P sinks (Habibiandehkodi et al. 2015).

To improve the effectiveness, future conservation schemes require: (1) the selection of CPs based on the dominant form of P in runoff (DP or PP) and an understanding of possible tradeoffs between the two forms; (2) recognition that CPs designed to trap P within the landscape are at risk of becoming legacy P sources; (3) extensive education and outreach to ensure the widespread adoption of CPs targeting DP losses; and (4) inclusion of TDP in monitoring programs in addition to DRP as dissolved P_o losses can be significant, especially from riparian zones.

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