

Phosphorus Legacy: Overcoming the Effects of Past Management Practices to Mitigate Future Water Quality Impairment

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The water quality response to implementation of conservation measures across watersheds has been slower and smaller than expected. This has led many to question the efficacy of these measures and to call for stricter land and nutrient management strategies. In many cases, this limited response has been due to the legacies of past management activities, where sinks and stores of P along the land–freshwater continuum mask the effects of reductions in edge-of-field losses of P. Accounting for legacy P along this continuum is important to correctly apportion sources and to develop successful watershed remediation. In this study, we examined the drivers of legacy P at the watershed scale, specifically in relation to the physical cascades and biogeochemical spirals of P along the continuum from soils to rivers and lakes and via surface and subsurface flow pathways. Terrestrial P legacies encompass prior nutrient and land management activities that have built up soil P to levels that exceed crop requirements and modified the connectivity between terrestrial P sources and fluvial transport. River and lake P legacies encompass a range of processes that control retention and remobilization of P, and these are linked to water and sediment residence times. We provide case studies that highlight the major processes and varying timescales across which legacy P continues to contribute P to receiving waters and undermine restoration efforts, and we discuss how these P legacies could be managed in future conservation programs.

DURING THE LAST DECADE, it has become apparent that many watershed-based conservation programs have failed to deliver improvements in water quality within timescales predicted by watershed managers and scientists (Jarvie et al., 2013; Meals et al., 2010; Mulla et al., 2008). Examples include the Chesapeake Bay watershed (USEPA, 2010; Reckhow et al., 2011), Mississippi River basin (Dale et al., 2010), Florida's inland and coastal waters (USEPA, 2011), and the Lake Erie basin (Sharpley et al., 2012), where conservation practices to decrease nutrient losses (particularly P) from agriculture were put in place 20 to 30 yr ago to minimize water quality degradation due to eutrophication. In trying to understand the apparent lack of water quality response, questions have been asked about the effectiveness of the conservation measures introduced and whether they are being correctly located or implemented at a sufficient scale and intensity across watersheds (Sharpley et al., 2009). Increasing scrutiny is now being directed toward the legacy of past land management practices, especially in terms of sinks and stores of P in the watershed. It has been suggested that conservation strategies have not adequately taken account of legacy P issues arising from past land management practices (Kleinman et al., 2011a; Sharpley et al., 2011).

A portion of P inputs from nonpoint and point sources within a watershed can accumulate at various locations along transport pathways within the land–freshwater continuum. These include soils, in downslope areas, and in ditch, stream, river, lake, wetland, riparian, and estuarine sediments and biomass (Fig. 1). Accumulated P can be remobilized or recycled, acting as a continuing source to downstream water bodies for years, decades, or even centuries (McDowell et al., 2002). This has been referred to as *legacy P* (Kleinman et al., 2011a). Legacy P is particularly problematic because it is characterized by intermediate storage and remobilization along slow or tortuous flow paths between the original source (agricultural fields or point-source discharges) and the watershed outlet.

When external P loads to rivers, lakes, and bays are reduced, legacy P may effectively mask or buffer the impacts of contemporary conservation measures (Hamilton, 2012; Meals et

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Abbreviations: BMP, best management practice.

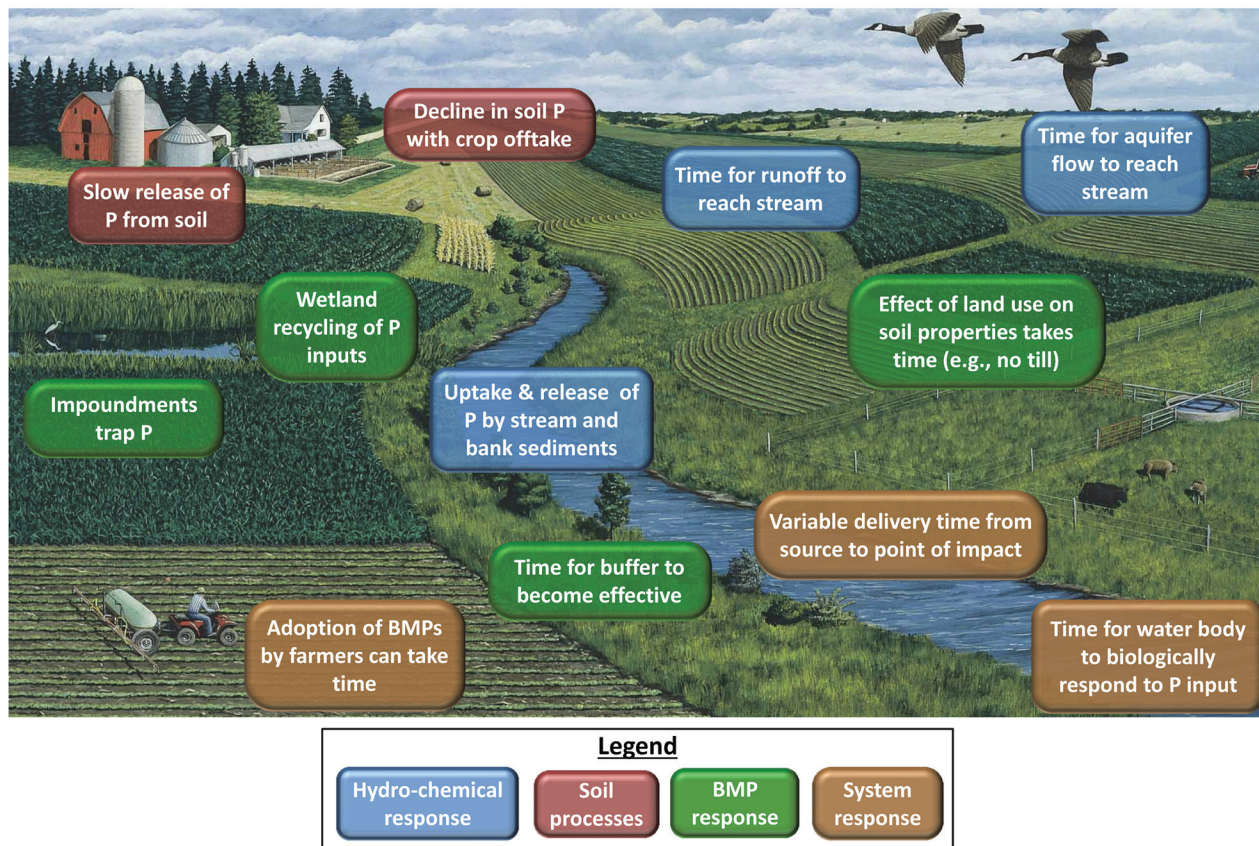


Fig. 1. Conceptual representation of natural, managed, soil, and best management practice (BMP) processes influencing the lag time for system response.

al., 2010; Spears et al., 2012). This makes it difficult to distinguish the effects of current conservation measures from historical land management. In many cases, legacy P has increased pressure on watershed organizations to explain and, more importantly, to respond to an apparent failure of water quality initiatives following significant economic investment. For instance, US\$24 billion has been invested in conservation measures through the 2008 U.S. Farm Bill (P.L. 110–246; Monke and Johnson, 2010) between 2005 and 2010, while the desired outcomes of conservation programs in such basins as the Chesapeake Bay, Lake Erie, and Mississippi River have not yet been attained. Not achieving water quality targets has prompted regulatory agencies to reevaluate nutrient management guidelines and standards (e.g., Natural Resources Conservation Service, 2011, 2012), Concentrated Animal Feeding Operations regulations (e.g., USDA and USEPA, 1999; USEPA, 2008), nutrient criteria for flowing and impounded waters (e.g., USEPA, 2010, 2011), and the Clean Water Act itself.

In this study, we take a watershed-scale view of legacy P by examining common drivers that control the physical transport (hereafter referred to as a *cascade*) and biogeochemical spiraling (i.e., absorption–adsorption–desorption) of P along the continuum from soils to rivers, lakes, and reservoirs via surface and subsurface flow pathways. We emphasize that the challenges faced in managing P cycling in watersheds cannot be regarded solely as phenomena of today’s management. Instead, we demonstrate that the abundance of recalcitrant sources of P in watersheds is a symptom of the wider decoupling of local and regional macronutrient cycles. We examine the spatial distribution of legacy P hotspots within watersheds and the

timescales for “drawdown” of legacy P stores along the land–freshwater continuum. We provide individual case studies that illustrate the controls and timescales associated with legacy P issues in soils, rivers, and lakes. We explore the unintended consequences of best management practices (BMPs), which were designed to retain P within the landscape and protect surface waters from agricultural P losses but the legacies of which are now contributing to P exported from watersheds. We conclude by examining how the legacies of past management practices have both positive and negative impacts on water quality and what strategies and technologies are available to address legacy P.

Legacy Phosphorus: A Symptom of Decoupled Macronutrient Cycles

Phosphorus, along with N and C, are essential elements for biological processes, so biogeochemical cycles of P, N, and C are closely interrelated and support productivity and diversity across terrestrial and aquatic ecosystems (Manzoni and Porporato, 2011). In pristine and relatively undisturbed ecosystems, the interdependence of P, N, and C cycling results from the following chemical and biological processes: uptake into and decomposition of biomass; interactions with other elements (e.g., Fe, Mn, Mo, and S) via microbial metabolism; and hydrologic fluxes that physically connect terrestrial and aquatic ecosystems and mediate macronutrient exchanges (Burgin et al., 2011; Jarvir et al., 2012a; Manzoni and Porporato, 2011).

Modern agriculture, with its heavy dependence on fertilizer applications and its low nutrient-use efficiency, fundamentally

disrupts P and N cycles (Elser and Bennett, 2011). Meeting the energy and food demands of a growing and increasingly urbanized population has dramatically altered P, N, and C cycling and increased ecosystem fluxes (Townsend and Porder, 2012). While N is harnessed from the atmosphere using the Haber–Bosch process, mining of geologic P deposits has exploited finite global reserves that have a residence time of about 100,000,000 yr, thus short-circuiting the sedimentary P cycle (Childers et al., 2011). Large mass transfers of P and N into relatively small and concentrated areas of agricultural production (Cordell et al., 2009; Elser, 2011) have resulted in widespread regional decoupling of P and N cycling rates from C in intensively managed agricultural systems. In pristine and low-intensity, extensively managed environments, P and N transfers in soils and water are largely biologically mediated. In intensively managed agricultural systems, however, where feed and fertilizer inputs of P and N are significantly greater than outputs in produce, biological controls can become overwhelmed (Sharpley et al., 2006, 2007). This leads to an increasing dominance of inorganic forms and critical shifts in the stoichiometry of nutrients (ratios of P, N, and C) delivered to downstream ecosystems (Dodds et al., 2010; Evans-White et al., 2009; Singer and Battin, 2007). Disruptions to the P, N, and C cycles and increased fluxes of inorganic and, to a certain degree, organic P and N fractions result in elemental ratios that can be very different from the requirements of organisms responsible for primary production.

Phosphorus is unique among the major nutrient elements in that its reactivity as an anion, key to its role as the “backbone element” of biochemical systems (Sharpley, 2011), results in a disproportionately large concentration in solid phases relative to solution phases. Indeed, the bulk of the literature on P availability in soils, sediments, and the environment documents and quantifies the sorption processes (rapid adsorption to the surfaces of cationic minerals, precipitation with electrolytes, and physical occlusion during long-term weathering) that convert dissolved and soluble forms of P into increasingly recalcitrant forms in the solid phase (Jones et al., 1984). For instance, the sorption potential of P by most soils and sediments is at least several orders of magnitude greater than the corresponding concentration of P in solution (Frossard et al., 1995, 2000). Consequently, sinks for the dissolved forms of P are ubiquitous, and sorption has helped to dampen solution signatures of a P surplus. As the degree of P sorption saturation increases, however, so does the release of sorbed P to solution, in this case to runoff or infiltrating water (Hooda et al., 2000; Leinweber et al., 1997; Nair et al., 2004; Sharpley, 1995; Sims et al., 2002). Considerable research has been devoted to quantifying this phenomenon, from parameterized models describing P sorption isotherms to models describing P sorption saturation (Bolster et al., 2012). Consequently, even as inputs of P are curtailed, solid-phase P, which is a legacy of previous activities, can be mobilized directly as particulate P or indirectly as dissolved P and thereby constitute the major source of P in the land–freshwater continuum (McDowell and Sharpley, 2002; Sharpley, 1996; Sharpley et al., 2004). In certain cases where peaty soils predominate, organic sources of P can be important (Condrón et al., 2005; Frossard et al., 2000).

The capacity of soils and sediments to adsorb P led early land managers to incorrectly conclude that its transport in dissolved

form in runoff and in subsurface flow was absent or unimportant (Sharpley et al., 1996; Sims et al., 1998). What has become apparent is that transport of P along pathways of concentrated or preferential flow is responsible for the downgradient transfer of environmentally significant loads of P along the land–freshwater continuum. Features such as rills, gullies, soil macropores, artificial drainage channels and tile lines, and bedrock fractures all serve as potential conduits for particulate and dissolved P, bypassing the many sinks that exist. Phosphorus can accumulate along pathways of preferential flow (Simard et al., 2000; Sims et al., 1998), saturating the sorption capacity of the exposed surfaces and eventually converting these finite stores of legacy P into potential sources of P sometime in the future.

The land–freshwater continuum is not simply a passive transport system. Surface and subsurface pathways of P transfer are influenced by cycling and exchange between different inorganic and organic pools and hydrologic processes that result in varying residence times for P (Gburek and Sharpley, 1998; McDowell and Sharpley, 2001; Meals et al., 2010). This is conceptualized in Fig. 1 and gives rise to distinct spatial and temporal discontinuities in P, N, and C storage and release. Furthermore, local changes in the behavior of one element can have major downstream impacts on the responses of others. For example, increased inputs of labile C to rivers from terrestrial sources as a result of the disruption of soil C cycles (Dawson et al., 2012) or from septic tank discharges (Palmer-Felgate et al., 2010) fuel higher rates of microbial respiration in riverbed sediments. As a result, microbial degradation of organic matter releases P and increases O₂ depletion, resulting in reductive dissolution of Fe oxyhydroxides, which release sorbed P. Higher rates of organic C release from soils have also been linked to steric stabilization of colloidal particles, promoting enhanced export and downstream transport of colloids and, potentially, their associated P load (Neal et al., 2008; King and Jarvie, 2012). Both of these “knock-on” effects of increased export of organic C have the potential to increase the mobilization of legacy P stores and serve to demonstrate that the storage and remobilization of legacy P in watersheds can be closely linked to changes in other macronutrient cycles.

Phosphorus Cascades and Spirals: Physical and Biogeochemical Drivers of Legacy Phosphorus

There is a suite of processes common to agricultural landscapes that controls the mobilization and transfer of legacy P from the soil via surface and subsurface hydrologic pathways to rivers and lakes (Fig. 1). These can be described in relation to the physical processes that control the “cascade” of P through the watershed and the biogeochemical processes that mediate the “spiraling” of P among water, sediment, and biota. The term *cascade* is commonly used to describe the transfer of nutrients, sediment, and water from areas of runoff and sediment erosion within the landscape, via transport pathways and intermediate stores, to the watershed outlet or coastal zone (Burt and Allison, 2010). Here, we examine the cascade of P from the soil and through the watershed as conceptualized in Fig. 1.

The cascade of water and sediment through the landscape is initiated by runoff generation. In contrast, the potential accumulation of legacy P is mediated by surface and subsurface

flow pathways of dissolved, colloidal, and particulate P as it cascades through the watershed and by the residence times in different environmental compartments. The conveyance capacity of particulate P through fluvial systems is limited by the entrainment and transport capacity of the flow, with areas of flow separation and divergence, and consequent reductions in flow velocities, resulting in net deposition and accumulation of particulate P. The episodic nature of these cascades can give rise to waves of sediment that can accumulate rapidly (e.g., vertical accretion on floodplains) but which require much higher energy events or longer time frames to remove (e.g., through channel or bank erosion). This process of material accumulation is referred to as “fast in–slow out” (Trimble, 2010).

Within the cascade of P through the watershed from source to outlet, the physical retention of P is not limited to P accumulation in solid phases alone (e.g., soils and sediment in transit). Dissolved constituents (be they conservative, e.g., Cl^- , or nonconservative, e.g., P) are also subject to retention through the storage of water along slower flow pathways. These pathways include groundwater, wetlands, riparian and hyporheic zones, and lakes, which contribute to a legacy of dissolved P with varying residence times. In fact, water and solute travel times in watersheds have been shown to follow an approximate power-law function, thus retaining a long chemical memory of past inputs. This is true even for conservative analytes such as Cl^- , where there are no additional biogeochemical retention or release processes involved (Kirchner et al., 2000, 2011).

Superimposed on this physical cascade of water and sediment is the biogeochemical cycling of P as it moves downslope and downstream along the land–freshwater continuum (referred to as *nutrient spiraling*; Newbold et al., 1983). Nutrient spiraling can also be applied to the cycling of P at the soil or hillslope scale, where spiraling lengths are shorter and have more intermittent downstream transport of nutrients than rivers (Manzoni and Porporato, 2011). The mechanisms of P exchange and spiraling have been reviewed extensively elsewhere for soils and hillslopes (Beven et al., 2005; Haygarth et al., 2006; Withers et al., 2002), streams and rivers (Haggard and Sharpley, 2007; House, 2003; Reddy et al., 1999; Withers and Jarvie, 2008), lakes (Boström et al., 1988; Søndergaard et al., 2005), and wetlands (Reddy et al., 1999). Common biogeochemical processes that control P spiraling along the land–freshwater continuum include: mineral precipitation and dissolution (House, 2003; Pierzynski et al., 2005); sorption and desorption (Froelich, 1988; House et al., 1998; Reddy et al., 1999; Sharpley et al., 2012); organic P mineralization (Condrón et al., 2005; Jakobsen et al., 2005), uptake, or process regulation by primary producers (Dodds, 2003; Jarvie et al., 2002; Spears et al., 2008a; Søndergaard et al., 2007) and microorganisms (Gächter and Meyer, 1993); and molecular diffusion (Frossard et al., 1995, 2000; Moore and Reddy, 1994).

Clearly, hydraulics, hydrology, geomorphology, and land management exert interacting controls on P fluxes along the land–freshwater continuum and the physical cascades and biogeochemical spiraling of P among pools. The variable residence and turnover times of these different pools of P across the landscape give rise to distinct spatial and temporal patterns of P retention, storage, and release that often reflect legacy P but also natural P contributions from across the watershed. For example,

most (>80%) of land-based (agricultural) P originates from only a small proportion (<20%) of the watershed area (Pionke et al., 1997, 2000). These “critical source areas” are essentially P hotspots of legacy P accumulation activated by overland flow (Gburek et al., 2007; Walter et al., 2000). Many of the legacy P hotspots also occur where there is a dramatic change in water and sediment retention times, promoting the sedimentation and accumulation of particulate P or uptake of P by biota. For example, locations where slope or channel geometry serve to lower water velocity (e.g., at the foot of hillslopes) and promote sediment deposition (e.g., where fast-flowing water enters a standing water body) are particular hotspots for P accumulation. This also applies to areas of slower flow and higher water retention time in rivers, as these promote the deposition of fine sediment and organic matter. In large standing water ecosystems, the positioning of hotspots can be dependent on a mosaic of physical disturbance and biogeochemical processing mechanisms, for example the focusing of legacy P to deep sediment zones in lakes (Hilton et al., 1986; Spears et al., 2007). Transitional “ecotones” between terrestrial and fluvial environments (e.g., riparian wetlands, floodplains) and between groundwater and fluvial environments (hyporheic zones) also provide important areas of P accumulation.

In the following sections, we examine the spatial patterns and timescales of P retention and recycling that give rise to distinct hotspots of legacy P within the landscape and in surface waters. We review the timescales of P recovery after remediation in terrestrial, river, and standing water environments (see Table 1). We describe P legacies as:

- *Terrestrial P legacies*—These encompass the effect of land and nutrient management, which in many cases was based on sound agronomic and soil productivity decisions, on the buildup of soil P to levels in excess of crop needs, as well as the modification of the connectivity of terrestrial sources to fluvial transport.
- *River P legacies*—These encompass the retention and remobilization of P through physical cascades and biogeochemical spirals along channels draining the edge of fields and via streams and rivers to receiving standing waters.
- *Standing water P legacies*—These encompass the deposition and remobilization of abiotic inorganic and biotic organic P as a function of ecosystem drivers, such as water residence times, water depth, ecological interferences, and the occurrence of extreme climate events.

We also provide five specific case studies for soils, rivers, and standing water that exemplify the locations of legacy P accumulation and the timescales during which the P legacy continues to contribute to receiving waters.

Terrestrial Phosphorus Legacies: From Soil to Hillslopes and Groundwater

The accumulation of agricultural P in soils represents the most pervasive legacy source of P to the environment. Phosphorus surpluses are locally evident across the world (MacDonald et al., 2011), reflecting soil fertility programs, local intensification of livestock production and the ensuing land application of manure, and, to a lesser extent, local geologic sources. Phosphorus legacy in soils arises because the buildup of P in soil occurs much

more rapidly than the decline due to crop uptake, such that the decline can be on the order of decades, depending on how much P has accumulated in the soil (Cox et al., 1981; Sharpley et al., 2009). In studies comparing soils under forest and agricultural land use, concentrations of total P under agriculture can range from two- to 10-fold greater than background concentrations (geologic or native P) under forests (by $\sim 300 \text{ mg kg}^{-1}$), a legacy of historical P fertilization (Nash and Hannah, 2011; Sattari et al., 2012; Sharpley et al., 1985; Syers et al., 2008; Vitousek et

al., 2009). For instance, Brazil's national soil fertility program, launched in the 1970s, targeted the Cerrado region of central Brazil for continuous additions of phosphate fertilizer (Goedert, 1983), which profoundly increased soil P reserves, often above agronomic optimums, and transformed the region's agricultural lands into some of the most productive in the country. In China, the drive for national food security encouraged the use of manure and fertilizer resources, with rates of P addition to agricultural

Table 1. Example ranges of time lags for recovery generated by legacy P contributions for soils, rivers, and lakes.

System	Location	Parameter	Decline†	Time lag yr	Decline $\text{mg kg}^{-1} \text{ yr}^{-1}$	Reference
Soils‡						
Thurlow loam under small grains	Montana	Olsen P	60–6	9	6.0	Campbell (1965)
Portsmouth fine sandy loam under small grains	North Carolina	Mehlich-1 P	54–26	9	3.1	Cox et al. (1981)
Haverhill clay under wheat	North Carolina	Olsen P	74–33	14	2.9	Cox et al. (1981)
Haverhill clay under wheat	North Carolina	Olsen P	135–70	14	4.6	Cox et al. (1981)
Portsmouth fine sandy loam under corn	North Carolina	Mehlich-3 P	100–20	16–18	4.7	McCollum (1991)
Ruston fine sandy loam under bermudagrass hay	Oklahoma	Mehlich-3 P	258–192	6	11.0	Sharpley et al. (2007)
Othello silt loam under corn	Maryland	Mehlich-3 P	488–465	5	4.6	Sharpley et al. (2009)
Range of soils under arable and pasture	Ireland	Morgan's P	>8 to 5–8§	7–15	0.4	Schulte et al. (2010)
Carroll clay loam under wheat	Manitoba	Olsen P	71–10	8	7.6	Spratt et al. (1980)
Carroll clay loam under wheat	Manitoba	Olsen P	135–23	8	14.0	Spratt et al. (1980)
Carroll clay loam under wheat	Manitoba	Olsen P	222–50	8	21.5	Spratt et al. (1980)
Waskada loam under wheat	Manitoba	Olsen P	88–23	8	8.1	Spratt et al. (1980)
Waskada loam under wheat	Manitoba	Olsen P	200–50	8	18.9	Spratt et al. (1980)
Waskada clay loam under wheat	Manitoba	Bray-1 P	140–50	8	11.3	Wagar et al. (1986)
Waskada clay loam under wheat	Manitoba	Bray-1 P	320–80	8	30.0	Wagar et al. (1986)
Ste. Rosalie clay under corn	Montreal	Mehlich-3 P	125–109	4	4.0	Zhang et al. (2004)
Nicollet–Webster loam under corn–soybean rotation	Iowa	Bray-1 P	60–5	27	2.0	Dodd and Mallarino (2005)
Nicollet–Webster loam under corn–soybean rotation	Iowa	Bray-1 P	95–8	27	3.2	Dodd and Mallarino (2005)
Webster–Canisteo loam under corn–soybean rotation	Iowa	Bray-1 P	42–8	27	1.3	Dodd and Mallarino (2005)
Webster–Canisteo loam under corn–soybean rotation	Iowa	Bray-1 P	85–9	27	2.8	Dodd and Mallarino (2005)
Rivers						
In-channel sediment and P storage and remobilization	lowland permeable rivers (Tern, Lambourn, Frome, Piddle), UK	fine sediment and particulate P	NA¶	<1	NA	Collins and Walling (2007a, 2007b), Ballantine et al. (2009)
	River Lambourn, UK	Dissolved reactive P	30	0.5	NA	Jarvie et al., 2006
	Yorkshire rivers, UK	Fine sediment and Total P	NA	<1	NA	Owens et al., 2001
Watershed-scale fluvial sediment storage and remobilization	Coon Creek, WI; Powder Creek, MT; Amazon basin	bulk sediment	NA	decades–centuries	NA	Trimble (2010)
Floodplain sediment and P storage and remobilization	Murder Creek, GA	bulk sediment	NA	centuries–millennia	NA	Jackson et al. (2005)
	River Swale, UK	fine sediment, total P, and heavy metals	NA	decades–centuries	NA	Walling et al. (2003)
Reactivation of legacy P by floodplain restoration	River Yare, UK	molybdate-reactive P	NA	decades–centuries	NA	SurrIDGE et al. (2012)
	River Lobau, Germany	total P	NA	decades–centuries	NA	Schönbrunner et al. (2012)
Riparian restoration	Vermont	total P	20% decrease	2	NA	Meals and Hopkins (2002)
		dissolved P	20–50% decrease	2	NA	Meals and Hopkins (2002)

Table 1. Continued.

System	Location	Parameter	Decline†	Time lag yr	Decline mg kg ⁻¹ yr ⁻¹	Reference
			Lakes#			
Loch Leven; WWTP upgrades, industrial source controls, farm BMPs	UK	total P	95(i)–55(i)	30	NA	Case Study 5
Lake Müggelsee; WWTP upgrades, farm BMPs, industrial source controls, P-free detergents	Germany	total P	230–185(i)	6	NA	Kohler et al. (2005)
Barton Broad; WWTP upgrades, diversion of P-rich inflow	UK	total P	308(i)–95(i)	20	NA	Phillips et al. (2005)
Little Mere; WWTP effluent diversion	UK	total P	2350(ii)–167(ii)	>11	NA	Moss et al. (2005)
Shagawa Lake; WWTP upgrades	Minnesota	total P	51–30	5	4.2	Larsen et al. (1979)
Shagawa Lake; WWTP upgrades	Minnesota	dissolved P	21–4.5	5	3.3	Larsen et al. (1979)
Lake Champlain; WWTP upgrades and farm BMPs	New York, Quebec, Vermont	total P	no significant change	18		Lake Champlain Basin Program (2008)

† Soil units are mg P kg⁻¹ soil; river units are µg P L⁻¹; and lake units are (i) summer to autumn or (ii) annual mean values as µg P L⁻¹.

‡ Soil samples collected from the surface 0–15- or 0–20-cm depth and received no P during the stated period.

§ Morgan's P > 8 mg kg⁻¹ is ranked as excess soil P status and 5–8 mg kg⁻¹ is ranked as target soil P status.

¶ NA, not applicable.

WWTP, waste water treatment plant; BMP, best management practice.

lands in many parts of the country greatly exceeding those in the United States and northern Europe (Vitousek et al., 2009).

Legacy P reserves in soils have also developed in regions where specialization and intensification of crop and livestock production result in localized manure nutrient imbalances, such as those found in the intensive poultry-producing region of the Chesapeake Bay's Delmarva Peninsula and northwest Arkansas (Kleinman et al., 2012; Slaton et al., 2004). Soil P concentrations on the Delmarva Peninsula have increased to levels that are nearly one order of magnitude greater than what is required for crop growth due to continuous application of poultry litter (Buda et al., 2010). The legacy of historic P management of these soils

and their likely response to a changing P management is outlined in Case Study 1. Reducing these high soil P concentrations to levels that have a lower potential to enrich runoff by ceasing P applications may take decades or more, depending on the degree of soil P accumulation (Dodd et al., 2012). Similar response times (7–15 yr) were estimated by Schulte et al. (2010) for soil test P to decline from 14.3 and 8.3 mg kg⁻¹ Morgan's soil P (i.e., excessive, >8 mg kg⁻¹, Index 4) to 7.5 and 4.7 mg kg⁻¹, respectively (i.e., optimum, 5.1–8 mg kg⁻¹, Index 3).

The physical cascade of P in most landscapes arises because a large fraction of total P loss from soils is in particulate form, pointing to the importance of erosion as a mechanism

Case Study 1

The Legacy of Poultry Litter Management: Princess Anne, Maryland

In areas of intensive livestock production, the inefficient transfer of fertilizer nutrients from manures to crops has been and continues to be the primary concern for building up legacy P in soils because manure P invariably exceeds crop P requirements when manure is applied to meet crop N requirements. The effect of transitioning from N- to P-based application rates has been evaluated in a number of settings, including on an Othello silt loam (a fine-silty, mixed, active, mesic Typic Endoaquult) at the University of Maryland Eastern Shore's Research Farm, Princess Anne, MD. The farm is situated on the Atlantic Coastal Plain, bordering the Chesapeake Bay to which it drains, and has high soil P concentrations (Mehlich-3 P range 400–500 mg kg⁻¹ and total P generally >1000 mg kg⁻¹) as a result of >20 yr of poultry litter additions (for additional details, see Kleinman et al., 2007, 2011a). From 2000 to 2010, poultry litter was applied at three P-based rates: 0 kg P ha⁻¹ yr⁻¹ (corresponding to no P); 20 to 58 kg P ha⁻¹ yr⁻¹ (crop P requirement); and 40 to 116 kg P ha⁻¹ yr⁻¹ (crop N requirement), with anhydrous NH₃ fertilizer added to meet crop N requirements where the litter rate was insufficient.

Surface runoff was monitored from 2000 to 2005 while soils were sampled from 2000 to 2010. During the runoff monitoring period, no differences were detected in P losses, reflecting high variability in runoff generation and the control of soil P on surface runoff P losses. During the 10-yr soil monitoring period, no significant differences in Mehlich-3 extractable P or weak salt extractable P were detected among the litter application treatments, despite the annual removal of approximately 20 to 40 kg P ha⁻¹ with crop (corn [*Zea mays* L.] and soybean) harvest.

While it is anticipated that decreasing poultry litter application rates to these soils will eventually result in declines in soil P and associated reductions in soil P availability to runoff waters, this study clearly demonstrates that standard strategies to draw down soil P using crop harvest will require multiple decades to produce a significant effect. Therefore, additional measures are required to address such extreme reserves of soil P.

for mobilizing all forms of P, including legacy P. Erosional processes can result in immediate transfer of particulate P to stream systems but, in many cases, result in progressive cycles of downslope sedimentation and remobilization, giving rise to the accumulation of legacy P in downslope areas. Eroded sediments are typically enriched with P relative to the bulk soil (up to a factor of five times) due to the preferential removal of fine soil particles with greater P content during the process of erosion (Sharpley, 1980; Sharpley et al., 2002). Soil conservation measures such as no-till, cover crops, buffers, grass waterways, and diversion drainage are critical to controlling these legacy P transfers from landscapes to stream networks (Kleinman et al., 2011b; Sims and Kleinman, 2005). Conversely, changes in land use that exacerbate erosion can dramatically increase legacy P transfers. For instance, Carpenter et al. (1999) cited studies in which suburban development increased erosion to $>20 \text{ Mg ha}^{-1}$. Elsewhere, Duan et al. (2012) observed significantly higher particulate P loads in watershed effluent following residential development of agricultural land.

Once P sources are mobilized, sorption and desorption of P by soil and sediment dominate P transfers via surface and subsurface pathways (Buda et al., 2009b; Kleinman et al., 2011b; Sharpley et al., 2009). Therefore, strong correlations exist between P concentrations in soils and dissolved P concentrations in runoff (Vadas et al., 2005), with the elevated P concentrations of most soils reflecting historical fertilization activities, hence legacy P sources. In general, large loads of dissolved P are attributed to critical source areas of high soil P concentration (Kleinman et al., 2007). Even where soil P concentrations are lower, however, sources of legacy P can contribute significantly to runoff, especially in critical source areas of hydrologically active landscapes or hotspots (Gburek and Sharpley, 1998; Gburek et al., 2007). The potential for hydrology to overwhelm P loss drivers, in translating a modest source of soil P to a high field-scale loss of P, is highlighted in Case Study 2 for the Mattern Watershed in Pennsylvania.

The combined effects of modifications to land management and hydrology on P export can also be seen in how P inputs to

Lake Erie have responded to changes in farming practices during the last 30 yr, as described in Case Study 3. This case study clearly demonstrates the spatial and temporal legacy that changes in nutrient, land use, and drainage management in the source watersheds of Lake Erie can have on nutrient inputs to the lake and on recent algal responses. Importantly, the Lake Erie Basin study of land management and water quality response exemplifies several terrestrial P legacy factors that relate to the creation and management of hotspots on the landscape, scaling from field to watershed, and the temporal scale of terrestrial system response. Here, the widespread adoption of conservation tillage dramatically reduced P loads during a period of about 20 yr. With the change in tillage, however, there was a slow but continued buildup of soil test P with long-term broadcast applications of P, which eventually translated into increased dissolved P discharge to Lake Erie. Also, an increase in hydrologic connectivity, via tile drainage of land to rivers, eventually increased the potential of that land to contribute P to Lake Erie.

Legacy P transmission from soils is not restricted to surface runoff alone. An increasing number of studies have highlighted the contribution of *shallow* groundwater to P loss (Domagalski and Johnson, 2011; Holman et al., 2008). While P transfers to groundwater are generally assumed to be negligible due to the high potential for soils to retain P by adsorption (Addiscott and Thomas, 2000), they can be particularly problematic in regions with high-P soils, as demonstrated by Vadas et al. (2007). In that study, the researchers monitored dissolved P concentrations in the shallow ($<3\text{-m}$) groundwater on an intensely ditch-drained poultry farm located on Maryland's Lower Eastern Shore. Soil P concentrations on the farm were severely vertically stratified, ranging from 400 mg kg^{-1} Mehlich-3 P at the surface (0–20 cm) to 40 mg kg^{-1} Mehlich-3 P deeper in the subsoil (40–60 cm). In general, mean dissolved P concentrations in the shallow groundwater closely mirrored the P concentration trends in the soil, with the highest dissolved P concentrations near the soil surface (0.65 mg L^{-1}) and progressively lower concentrations with increasing soil depth.

Case Study 2

Hydrologic Mobilization of Legacy Phosphorus on Hillslopes: The Mattern Watershed, Pennsylvania

While we can measure the effect of past land use on soil test P legacy buildup, these legacy sources can be overwhelmed by the site hydrology as it translates to P loss (Fig. 2). This is important for mitigation of legacy impacts, in that transport management needs to be considered in terms of activating and mobilizing P sources.

Buda et al. (2009a, 2009b) monitored strip-cropped fields along a hillslope in which the agronomic soil test P ranged from near optimum in the riparian area (72 mg kg^{-1} Mehlich-3 P) to roughly three times the concentration required to support crop production upslope (177 mg kg^{-1} Mehlich-3 P; Fig. 2c). Runoff volumes from the riparian area were 27 times greater than runoff volumes from the upslope area (1376 to 18 L, respectively; Fig. 2c) due to the presence of a fragipan that supported seasonal saturation of riparian soils (Buda et al. (2009a, 2009b). As a result, the P load was 3.2-fold greater, despite the fact that runoff P concentrations from the riparian area were 5.8-fold less than those from upslope areas (Fig. 2d). Extrapolation of the hillslope study conducted by Buda et al. (2009a, 2009b) across the watershed in which the hillslope was located suggests that a small, hydrologically active riparian buffer zone representing only 6% of the total watershed area and with Mehlich-3 soil P of roughly half the concentration of upslope soils, is capable of producing a legacy yield of P in the runoff equivalent to 73% of the watershed P loss. This highlights the ability of site hydrology to overwhelm source factors in determining the activation and loss of legacy P. More important, 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.

Where deeper groundwater is concerned, much longer transmission pathways and lag times can occur before legacy P is ultimately delivered to receiving waters (Phillips and Lindsey, 1999; Sprague et al., 2011). In studying wells and springs, Lindsey et al. (2003) suggested that groundwater residence times can range from 0 to >50 yr, with a median age of 10 yr. In the Pocomoke River watershed (Coastal Plain uplands), the apparent age of the water ranged from 0 to 50 yr, with the age in the vicinity of streams from 0 to 23 yr. In Polecat Creek watershed (Piedmont crystalline region), ages ranged from 2 to 30 yr, in Muddy Creek watershed (Valley and Ridge carbonate) from 10 to 20 yr, and in the East Mahantango Creek watershed (Valley and Ridge siliciclastic) from 0 to >50 yr. Clearly, where BMPs are proposed, the lag between implementation and measurable improvements in stream water quality will be highly variable where P passes mainly through subsurface pathways.

River Phosphorus Legacies: Spiraling Sinks and Sources in Streams and Rivers

In river channels, legacy P stores accumulate as a result of the deposition of particulate P as fluvial bed sediments (Ballantine et al., 2009; Rawlins, 2011; Svendsen and Kronvang, 1993), sorption of dissolved P onto riverbed sediments (Haggard et al., 2001; 2005; Jarvie et al., 2005; Stutter et al., 2010) or onto suspended sediments that are subsequently deposited on the riverbed (Owens and Walling, 2002), or by incorporation of water-column P into plant or microbial biomass (Drake et al., 2012; Aldridge et al., 2010; Schade et al., 2011) (Fig. 1). The storage of P within fluvial channels is temporary and subject to remobilization (Dorioz et al., 1998; Spears et al., 2008b) but can account for between about 10 and 80% of annual P fluxes (Ballantine et al., 2006, 2009; Jarvie et al., 2012b).

In-stream P retention and release show distinct spatial patterns, influencing legacy effects at the river reach scale (Weigelhofer et al., 2012) and along the river continuum (Kerr et al., 2011; Powers et al., 2012; Vannote et al., 1980). These spatial patterns are linked to changes in the balance of physicochemical and biological processes driving P storage and remobilization (Withers and Jarvie, 2008). For example, higher rates of biological P retention have been reported in headwater environments, where shallow water, light penetration to the benthic interface, and lower water volume to the benthic surface area provide greater potential for biological P exchange processes (Bukaveckas, 2007). The balance between biotic and abiotic P retention

tends to shift along the river continuum, with abiotic P retention often increasing in significance downstream, linked to higher ambient dissolved P concentrations (Aldridge et al., 2010). At the river reach scale, hotspots of P retention and cycling occur in

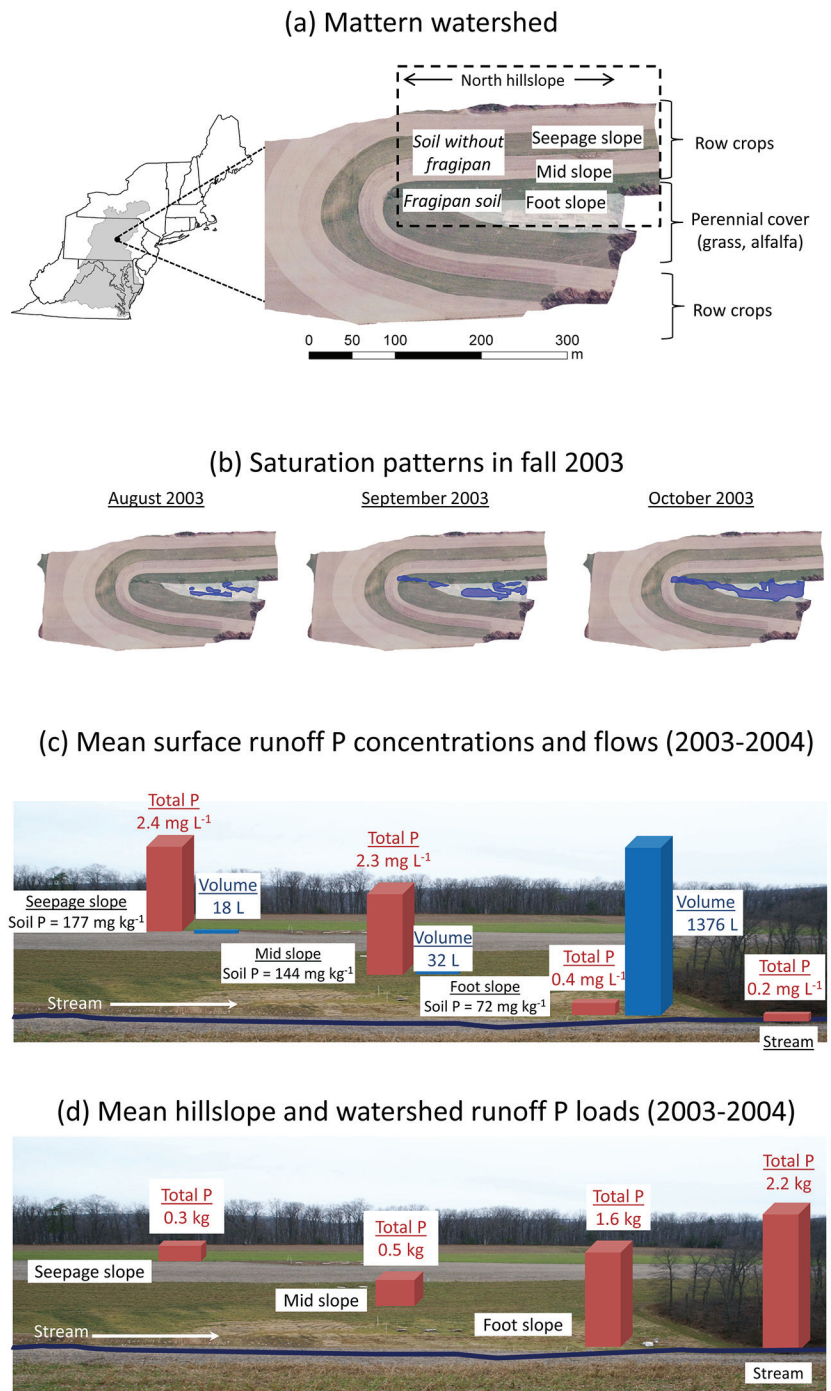


Fig. 2. Results from a 2-yr (2003 and 2004) study of P losses in surface runoff within the Mattern experimental watershed (Buda et al., 2009a, 2009b) showing (a) the location of the watershed within the Chesapeake Bay basin, as well as the location and land-use characteristics of the north hillslope where runoff monitoring was conducted at three different landscape positions, (b) changes in surface saturation patterns during the late summer and early fall of 2003, (c) mean total P concentrations in soils and runoff waters, as well as total runoff volumes for the 2003 to 2004 sampling period, and (d) mean hillslope and watershed total P loads. Mean hillslope total P loads were calculated using runoff total P data that were averaged across four closed runoff plots (plot area = 2 m²) at each landscape position. Assuming that the area of the plot represented the maximum contributing area for runoff and P loss, we then scaled these total P load estimates to the area of the hillslope above each set of plots.

zones of transient storage: areas of slower flows and higher water retention times, such as pools, eddies, dead zones, and channel margins where fine sediment and organic matter accumulate (Meals et al., 2010). These are recognized as zones of intense microbial activity (driven by an abundant C source), where P is released as a result of enhanced rates of organic matter breakdown, anoxia in the bed sediments, and the release of polyphosphate when cells become anaerobic or decompose (Ensign and Doyle, 2005; Hupfer et al., 2007; Withers and Jarvie, 2008).

In-stream legacy P may be physically remobilized by gradual winnowing and downstream transport of fine particulates under low-flow conditions (Jarvie et al., 2006) or by more turbulent mobilization of particulate P, scouring of algal biomass, and release of dissolved P from pore waters during high-flow events (Doriot et al., 1989; Drake et al., 2012). The release of dissolved P from bed sediments into the overlying water column may occur as a result of a reversal in diffusion gradients, often linked to sudden decreases in dissolved P in the river water, following point-source remediation, for example (Haggard et al., 2005; House and Denison, 2000; Jarvie et al., 2005, 2006). Biogeochemical recycling of legacy P in bed sediments is also driven by microbial processing of organic matter in the bed sediment, which releases dissolved P and, as the bed sediment becomes more anoxic, results in the reductive dissolution of Fe(III) oxyhydroxides releasing bound phosphate into the pore waters (House, 2003). The upward migration of the redox boundary to, or above, the sediment–water interface can result in the release of these pools of recycled legacy P into the overlying river water (House, 2003; Jarvie et al., 2008; Palmer-Felgate et al., 2010).

These processes are exemplified in Case Study 4, which monitored the response of bed sediment P in the River Lambourn, UK, following a sudden and dramatic reduction in point-source P inputs (Jarvie et al., 2006). While the response time for the

drawdown of in-channel legacy P from the River Lambourn bed sediments was rapid (~ 6 mo), the time lags associated with physical storage and remobilization of in-channel sediment were highly variable (Table 1). River P legacy response times are dependent on a range of geomorphological and anthropogenic factors, including the nature and particle size of the sediment, the river flow regime, the channel hydromorphology, and factors such as channel modifications and the presence of impoundments such as ponds, lakes, reservoirs, and canals (Hamilton, 2012; Palmer-Felgate et al., 2009).

Typical lag times of decades or more have been reported for the retention of bulk sediment in river channels (Trimble, 2010), especially where the upstream sediment supply exceeds the limited conveyance capacity of the river to transport sediment downstream (Hamilton, 2012). The portion of sediment of the greatest significance for P transport and P legacy, however, is the finer fraction (silt and clay), which tends to be more mobile and, owing to a higher specific surface area, carries a greater P load per unit mass than the bulk sediment (Naden, 2010). Indeed, in-stream storage of fine surface bed sediment can be relatively short term, i.e., until the next high-flow event remobilizes the deposited sediment, with residence times of <1 yr in many river systems (Collins and Walling, 2007a, 2007b; Owens et al., 2001). Longer term retention of riverine legacy P, on timescales of centuries, may occur from storage of sediment and particulate P in seasonally inundated floodplains, with subsequent remobilization by channel migration and bank erosion (Jackson et al., 2005; Walling et al., 2003). The restoration of floodplains and riparian wetlands may also inadvertently activate long-term legacy P stores by rewetting floodplains and reestablishing a hydrologic connection between the river and its floodplain (Schönbrunner et al., 2012; Surridge et al., 2012).

Case Study 3

The Legacy of Land Management on Phosphorus Export: Lake Erie Basin

The importance of long-term monitoring for assessing watershed response to conservation management and documenting the legacies of management change is demonstrated by studies conducted by Richards et al. (2002b, 2009) in the Lake Erie Basin. Phosphorus loads in two Ohio watersheds (Maumee and Sandusky rivers watersheds), which are major tributaries to Lake Erie, have been monitored since 1975 to determine the effect of adopting best management practices such as conservation tillage and nutrient management planning in predominantly row-crop agriculture (corn, soybean, and wheat). Averaged across both watersheds, mean annual flow-weighted dissolved and total P concentrations decreased 86 and 44%, respectively, between 1975 and 1995 (Richards et al., 2002a). Since 1995, annual flow-weighted concentrations of dissolved P have increased, while particulate (and total P) has continued to decline (Richards et al., 2009). The trend of increasing dissolved P and decreasing total P inputs to the lake can be attributed to a combination of several factors: a change in rainfall distribution pattern; a legacy of chronic excess of P applied above that removed in crops; a buildup of P at the soil surface with conversion to no-till cropping; and increased applications of fertilizer and manure without incorporation in the fall and winter (Joosse and Baker, 2011; Sharpley et al., 2012).

In 2006 and 2007, widespread fall application of fertilizer, and a warmer and wetter than normal winter (175% of average annual rainfall occurred October–January), coupled with a threefold greater rainfall than normal in August 2007 (25 cm), exacerbated dissolved P runoff. The magnitude of dissolved P loss could not be solely explained by greater precipitation and runoff but also reflects fertilizer management and the steady increase in surface soil test P concentrations, which have characterized the past decade. Dissolved P loads in 2010 were the highest in 35 yr of monitoring and coincided with a time when Lake Erie was more biologically responsive to inputs (Richards et al., 2010). Also, while P loss from tile-drained soils is generally low, there has been a dramatic increase in the extent of tile drainage since 2005 in the Lake Erie Basin. More fields with tile drainage that connects to ditches and streams has increased the contributing sources areas of legacy P to Lake Erie.

Standing Waters: Sinks and Sources of Legacy Phosphorus in Impoundments and Lakes

Standing waters (impoundments, lakes, and reservoirs) are widely recognized for their ability to trap sediments and associated nutrients such as P (Vörösmarty et al., 2003). While the presence of such structures on rivers can have a short-term benefit by reducing downstream P fluxes, over the long term their sediment storage capacity can be diminished to such an extent that they are no longer a sink for fluvial P (Fig. 1). As a result, the legacy of P built up behind river impoundments or dams can become an important source of P and potential risk for downstream water quality.

In a study on the lower Susquehanna River in Maryland, Hirsch (2012) documented recent increases in P loss from the Conowingo Dam, a reservoir that is approximately 85% full with sediment. As the Conowingo Dam has approached capacity, fluxes of P entering the reservoir have come into relative balance with those leaving the reservoir and entering Chesapeake Bay. Furthermore, the relationship between P concentration and flow has changed markedly during several decades, such that P concentrations in reservoir discharges $>2832 \text{ m}^3 \text{ s}^{-1}$ (100,000 cfs) are currently two- to threefold higher than those that occurred in the past at similar flow rates. As a result, the highest P concentration during the 34-yr period of record (2.3 mg L^{-1}) occurred during Tropical Storm Lee in 2011, the third largest flood in history on the Lower Susquehanna River. The event, which accounted for 61% of the annual P flux in 2011 and 22% of the P flux during the past decade, demonstrates the potential for large storms to mobilize and transport substantial loads of the legacy P stored behind river impoundments.

Lakes also slow the speed of P transport through the watershed. The residency of P within lakes is driven by a mosaic of biogeochemical processes. Dissolved P can be rapidly sequestered into the food web and returned to bed sediments in organic and particulate forms or be relinquished from the lake through groundwater or surface water outflows. Particulate P can settle quickly to the lake bed, especially under calm conditions, and is subject to diagenetic processes as described above for rivers. Phosphorus that has accumulated in the sediments with time is a major source of legacy P.

The release of legacy P from bed sediments to the overlying water column in lakes is termed *internal P loading* and can exhibit strong seasonality, as shown in Case Study 5. A wide range of physicochemical processes regulate this internal loading (Boström et al., 1988). Bed sediments with a low equilibrium P concentration and a high P sorption capacity (e.g., as indicated by high Fe contents; Jensen et al., 1992) have been shown to support high levels of internal P loading in lakes. In deeper lakes, where prolonged periods of thermal and dissolved O_2 stratification may occur, internal loading of legacy P to the water column is mainly contained within anaerobic bottom waters and, therefore, may not be flushed out of the system very quickly through surface water outflows. This containment results in relatively low rates of export of legacy P from deeper lakes once P inputs from the watershed have been reduced.

In shallow lakes, where prolonged periods of stratification are rare, legacy P is more likely to be flushed out of the system after release from the bed sediments. In these systems, biological structure and function can strongly influence the magnitude of internal P release, and threshold changes (i.e., regime shifts; Scheffer, 1998) can occur, resulting in alternate “stable

Case Study 4

Short-Term In-Channel Fluvial Phosphorus Legacies: River Lambourn, UK

Declines in P concentrations following a reduction in point-source P inputs were investigated in a rural chalk stream, the River Lambourn, UK (Jarvie et al., 2006). The rapid response in water-column P concentrations following point-source P reductions provided a “controlled” experiment to quantify the legacy P contribution and its duration (Fig. 3). This would be difficult to achieve for nonpoint-source controls, where part of the measured P legacy would probably result from lags in the drawdown of P inputs from the land surface, as well as the in-stream P legacy. River-reach mass balance (Fig. 3a) and sediment equilibrium P concentrations (EPC_0 ; House and Denison, 2000; Fig. 3b) were measured to evaluate the changes in in-channel P storage and release along a 2.5-km river reach before and after the point-source reductions under baseflow conditions. Further details of the River Lambourn and methodologies can be found in Jarvie et al. (2006). The river reach mass balance studies allowed direct quantification of net gains and losses in dissolved reactive P (DRP) loads along the reach, and the EPC_0 measurements were used to examine whether riverbed sediments were acting as net sinks or sources of DRP during this time.

The mass balance measurements along the river reach immediately below the effluent input showed a sudden switch from net uptake of DRP before the point-source reductions to net release of DRP after the point-source P reductions of 70 to 90% (Fig. 3a). The EPC_0 measurements provided corresponding independent verification of a shift of bed sediments from net sinks of DRP to net sources of legacy DRP after point-source P reductions (Fig. 3b).

The DRP net release from bed sediments after reductions in point-source P inputs occurred as a result of a reversal in DRP diffusion gradients out of bed sediments following the sudden reductions in DRP in the overlying water column. There was a lag time of around 6 mo before the bed sediment EPC_0 values had declined to values that were close to equilibrium with the overlying river water DRP concentrations and for a drawdown in legacy P release from the bed sediments. This was during spring and summer, however, under stable baseflow conditions in a relatively low-energy chalk stream environment (Collins and Walling 2007b). Under higher flows and more flashy or responsive hydrologic regimes, with more active sediment erosion and deposition, a more rapid removal of higher EPC_0 sediment and shorter legacy P recovery lag time would be expected (Jarvie et al., 2006).

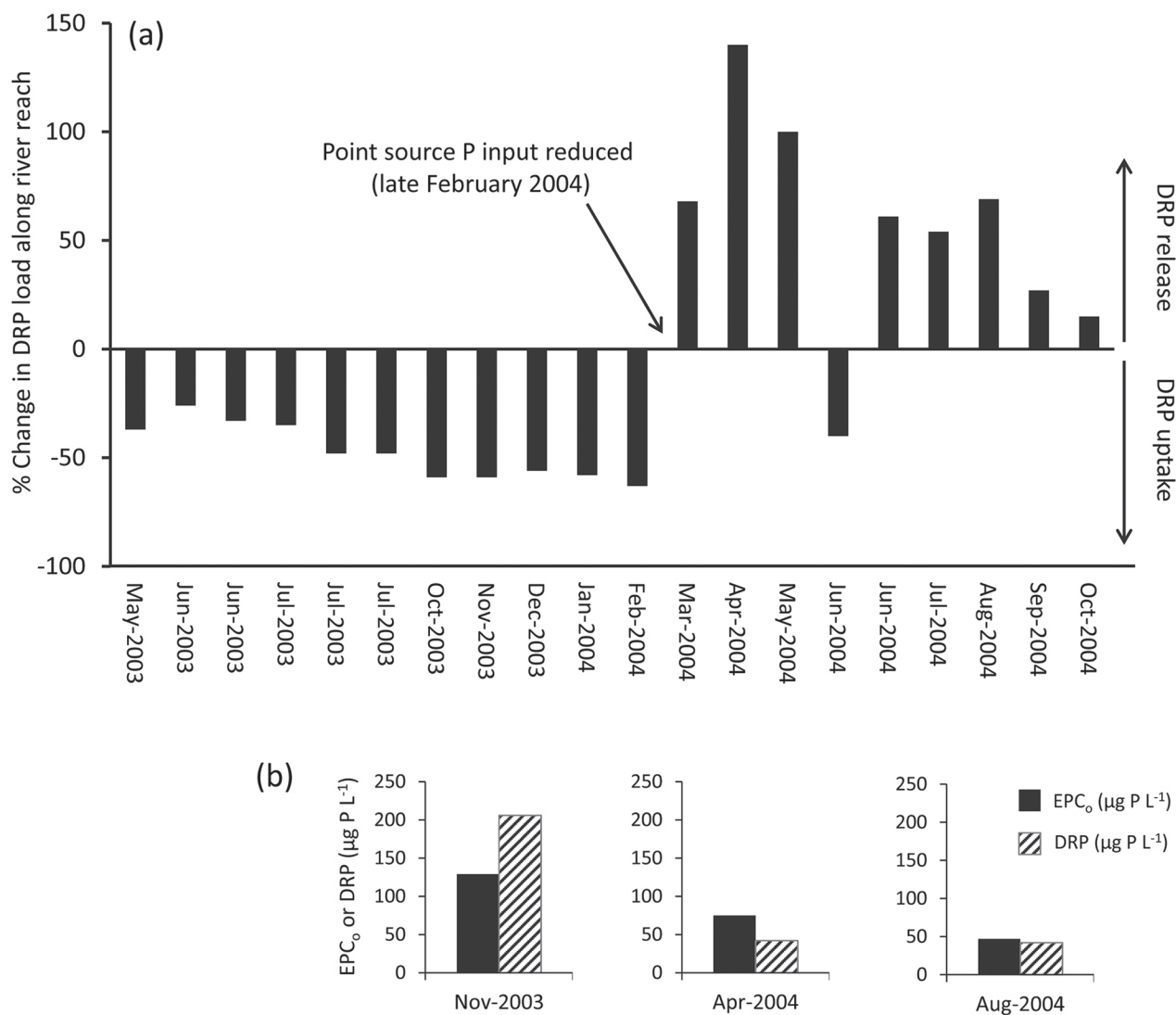


Fig. 3. (a) Mass balance results showing the changes in dissolved reactive P (DRP) load (by comparing river and effluent inputs and river outputs) along a 2.5-km reach of the River Lambourn, UK (data from Jarvie et al., 2006)—before reductions in point-source P inputs, negative values showed net losses of dissolved P along the reach linked to uptake of dissolved P by the riverbed sediments, while after reductions, there was a net gain of dissolved P as a result of the release of dissolved P from the riverbed sediments; and (b) equilibrium P concentrations (EPC₀) for the riverbed sediments and corresponding water-column dissolved P concentrations collected within the same 2.5-km reach downstream of the point-source input in November 2003, April 2004, and August 2004. These data provide independent verification of bed sediments switching from net sinks to net sources after reductions in point-source P inputs. Where EPC₀ < DRP (November 2003), sediments acted as a net sink for river water dissolved P; where EPC₀ > DRP (April 2004), sediments acted as a net source of dissolved P to the overlying river water; where EPC₀ ≈ DRP (August 2004), sediments were approaching equilibrium with the river water.

states.” These can include a turbid-water, phytoplankton-dominated state, which occurs when watershed P loading is high. Under these conditions, P cycling between the water column and sediment is perpetuated through high turnover of the phytoplankton community and results in rapid coupling between lake-bed and water-column P stores. The second is a clear-water, macrophyte-dominated state, which occurs when watershed P loadings are low. Under these conditions, P cycling between the water column and the lake bed is slower, passing mainly through the macrophytes rather than the phytoplankton. As a result, phytoplankton growth is nutrient limited in the clear-water state and macrophyte growth is light limited in turbid water (Scheffer, 1998). The importance of alternate stable states in regulating the cycling of legacy P in lakes is becoming more apparent, however, and the role of microorganisms, both heterotrophic and autotrophic, in driving these switches requires

clarification (Genkai-Kato and Carpenter, 2005; Genkai-Kato et al., 2012; Spears et al., 2009; Vadeboncouer et al., 2002). Affecting a switch from a turbid-water to a clear-water state by altering the magnitude of, and the balance between, internal (legacy) and watershed P loading is usually the main aim of changes in watershed management practices that are focused on lake restoration, especially in shallow lakes where sudden regime shifts are more likely (Carpenter et al., 1999).

In-lake biogeochemical processes regulate the release of legacy P from lake bed sediments following a reduction in external P inputs to lakes. In fact, these processes are important for controlling legacy P both before and after management and should be considered continuous. Jeppesen et al. (2005), reviewing the recovery of 35 lakes after external nutrient load reduction, found that internal P loading from legacy sources had delayed ecological recovery of these lakes by 10 to 15 yr. Similarly, Søndergaard et al. (2007)

found that internal P loading had delayed the recovery of Lake Søbygård, Denmark, for about 20 yr. Although driven, primarily, by the rate of release of internal stores of legacy P, the length of this recovery process is also affected by lake-specific factors such as water retention time, biological structure, pollution history, sediment P composition and concentrations, and lake morphology (Sas, 1989). A wider range of environmental factors, such as extreme weather events, fish stocking practices, and other types of pollution associated with industrial practices, are also important in regulating response times in lakes following P management (Spears et al., 2011). Seasonal changes in P balances and concentrations also occur during the recovery period, as reflected in observed changes in water-column total P concentrations (Søndergaard et al., 2005, 2007). In general, during the early years of recovery, winter to spring and autumn in-lake total P concentrations tend to decrease, and sediment P release during the summer to early autumn tends to increase. As the magnitude of legacy P release from the sediments decreases with time, summer in-lake total P concentrations also decrease.

This process and the importance of internal P loading are illustrated for Loch Leven, a shallow (average depth 4 m) lake in Scotland, UK, in Case Study 5. The trends in in-lake P concentrations following watershed management reflected a change in dominance from external to internal P loading, and these trends altered the timing of P delivery to downstream ecosystems (May et al., 2012; Spears et al., 2012).

Best Management Practices: The Good, the Bad, and the Legacy

Best management practices, which promote the retention of eroded soil, sediment, water, and nutrients within the landscape (such as buffer zones and wetlands) have been successful in storing P and thus smoothing out peaks in P delivery to downstream receiving waters (Reddy et al., 1999; Uusi-Kamppa et al., 2000). These BMPs protect downstream aquatic environments from high P loads and concentrations; without them, the impacts of agricultural P losses on downstream water quality would have been more acute. By concentrating P storage at specific locations in the landscape, however, these practices can create longer term and continuing sources of legacy P to receiving waters. The significance of these BMP sources of legacy P for downstream water bodies and aquatic ecosystems depends on: (i) their hydrologic connectivity to the receiving waters (Powers et al., 2012); (ii) changes in the timing and processes of legacy P remobilization, which influence the form of P exported (dissolved and particulate) (Edwards and Withers, 2007); and (iii) fluxes from legacy P stores relative to other contemporaneous P source fluxes (point and nonpoint) within a watershed.

Best management practices can become inadvertent causes of legacy P transfers within watersheds. For instance, Sheppard et al. (2006) found that several vegetated buffer strips in Manitoba, Canada, contained surface soil P levels 33% greater than source fields, and runoff P from the strips was 18% greater than that entering. No-till management decreased erosion and particulate P runoff from wheat (*Triticum aestivum* L.) in Oklahoma

Case Study 5

Legacy Phosphorus and Lake Recovery after Remediation: Loch Leven, UK

Loch Leven is a shallow (mean depth 3.9 m) lake that is relatively large by UK standards (13.3-km² surface area). The lake has a long and well-documented history of eutrophication problems. Efforts were made to restore the lake to good water quality during the late 1970s to early 1990s by reducing the external P inputs by about 60% (Bailey-Watts and Kirika, 1999; D'Arcy et al., 2006; May and Spears, 2012; May et al., 2012). This was mainly achieved by reducing industrial inputs from a woolen mill in 1976 to 1986 (Phase 2, Fig. 4a and 4c) and by upgrading sewage treatment works in 1987 to 1999 (Phase 3, Fig. 4a and 4c). Farmers were also encouraged to change their management practices and use less fertilizer within the watershed. In-lake winter and spring total P concentrations responded rapidly after Phase 2, with concentrations falling by about 75% in winter and 60% in spring compared with pre-management (Phase 1) levels. Further decreases in spring concentrations to about 50% of pre-management levels were observed after Phase 2. Nevertheless, summer concentrations remained high, with concentrations actually increasing. This switch from high baseline total P concentrations throughout the year to low winter, spring, and autumn total P concentrations, with a pronounced summer total P peak, suggests a switch in dominance from external to internal P loading in the lake during this period. The magnitude of the summer total P peak decreased in the 8 yr after Phase 3, as did concentrations in general across all seasons; however, summer total P concentrations of around 80 $\mu\text{g L}^{-1}$, and winter and spring baseline total P concentrations of around 30 $\mu\text{g L}^{-1}$, were still common.

Because the lake delivers water to a river downstream, the quality of this river is also affected by changes in outflow water quality that are driven by the release of legacy P within the lake during the recovery process. These changes affect the stoichiometry of nutrients delivered to downstream ecosystems, which affects their structure and function and alters their biogeochemical processes. Nutrient delivery from recovering lakes to downstream ecosystems is also important in terms of mass transfer because this affects loads to sensitive water bodies farther downstream, such as standing waters and estuaries. These loads are affected by variations in the rate at which water flows through the lake (i.e., the flushing rate). For example, seasonal changes in downstream total P loads from Loch Leven between the pre-management (1968–1975) and post-management and recovery (2000–2005) periods are compared in Fig. 4b. A strong decrease in the downstream delivery of total P was observed during the spring, and a moderate increase was observed during winter, summer, and autumn.

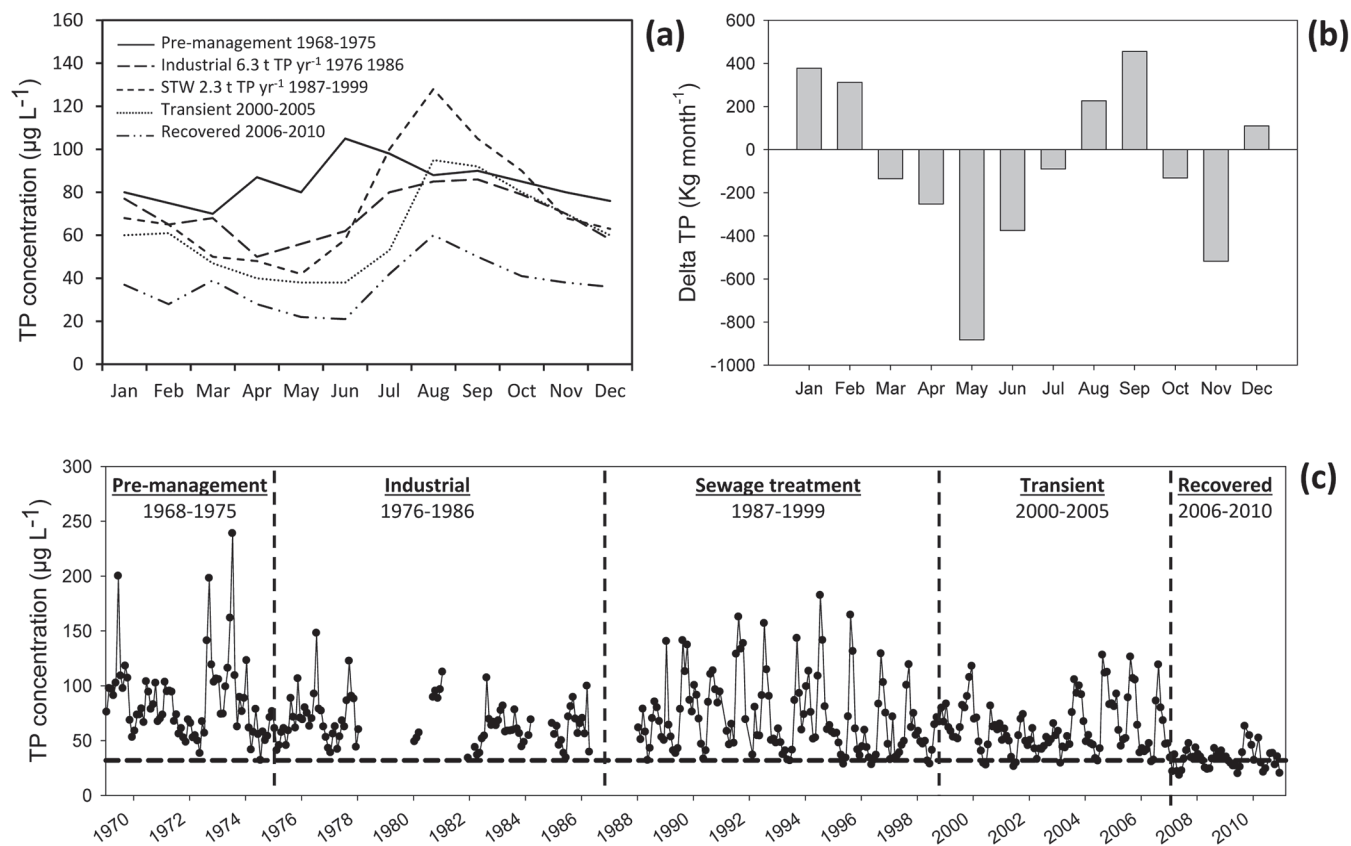


Fig. 4. Variation in monthly total P (TP) concentrations throughout key management periods, illustrating the impact of legacy P on the recovery of Loch Leven: (a) changes in monthly average in-lake TP concentrations during the recovery period; (b) change (delta) in cumulative monthly downstream delivery of TP between the post-management transient recovery period (2000–2005) and the pre-management period (1968–1975), where positive values represent increased losses from the lake; and (c) long-term variation in monthly TP concentration, 1970 to 2010. The horizontal dashed line in (c) represents the total P target of $32 \mu\text{g L}^{-1}$ for a high-alkalinity, shallow lake, as set under the EC Water Framework Directive. Water quality data were taken from the Loch Leven long-term monitoring data set (<https://gateway.ceh.ac.uk/home>).

(Sharpley and Smith, 1994) and soybean [*Glycine max* (L.) Merr.] in Ohio (Richards et al., 2009; Sharpley et al., 2012; Case Study 3); however, P that accumulated at the soil surface became a source of dissolved P in runoff.

Similar to their counterparts in Oklahoma and Ohio, researchers in Manitoba, Canada, 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, MB, even though conservation tillage significantly decreased sediment and N loss. At the landscape scale, construction in the late 1920s of Conowingo Dam on the Susquehanna River in Pennsylvania controlled P-rich sediment deposition in the Chesapeake Bay and minimized flooding (Hirsch, 2012). Some 85 yr later, however, it has become a potentially problematic pool of P stored in the river system bed sediments.

While wetlands and buffer strips may provide physical and biogeochemical hotspots for trapping and processing P, their importance as sources of legacy P is governed by the interactions between nutrient processing rates, water residence times, and, critically, the hydrologic connectivity between the legacy P source and the stream network (Palmer-Felgate et al.,

2011b; Reddy et al., 2011). Riparian buffers, wetlands, and constructed wetlands that discharge directly to surface waters or groundwater throughout the year are a potential source of legacy P to river ecosystems. Such wetlands are effective at trapping and converting less bioavailable particulate-bound P from nonpoint sources, delivered via the physical P cascade, into a sustained source of dissolved and more bioavailable P (Ardon et al., 2010) via biogeochemical processing (Palmer-Felgate et al., 2011a; Pant and Reddy, 2003). If these wetlands discharge directly to headwater stream networks, or are hydrologically connected with the stream via shallow groundwater, riparian, or hyporheic pathways (Lapworth et al., 2011), the resulting legacy P contributions can provide a source of bioavailable P to baseflow dissolved P concentrations and help maintain headwater streams in a sustained eutrophic state. Spring and summer baseflows are times of eutrophication risk, when legacy P leaking from these BMP wetlands can interact with stream biogeochemical spirals and contribute to primary production, resulting in potentially greater impairment to stream water quality and ecology (Edwards and Withers, 2007).

The inherent spatial and temporal variability in terrestrial and fluvial system response to management change begs the question of how long monitoring programs are needed to reliably demonstrate the success or failure of implemented remedial measures. For instance, if additional funds had not been secured in 1995 to continue water quality monitoring in

the Maumee and Sandusky watersheds of the Lake Erie basin, conclusions regarding the long-term land management impacts of conservation (Richards et al., 2002a, 2002b) would have been misleading. The laboratory conducting the monitoring fought to continue sample collection and analysis and eventually was able to demonstrate the long-term legacy of management change on water quality, as described in Case Study 3.

In a recent systematic review of chemical and ecological recovery in aquatic ecosystems, Verdonschot et al. (2013) indicated that in most reported case studies it was unlikely that the post-management monitoring was sufficient to capture the full recovery period to its endpoint. The majority of studies considered in this review reported post-management monitoring periods of <5 to 10 yr across rivers, lakes, estuaries, and coastal waters. This is clearly insufficient given the timelines described in the case studies presented here, especially when recovery in fluvial systems with low hydraulic residence times is considered. In Loch Leven, the lack of apparent response in P concentration immediately following costly catchment management measures led stakeholders to question the success of the implemented BMPs and to propose alternatives. Only through comprehensive continuous monitoring and experimentation, for >40 yr, was it possible for researchers and water quality managers to identify the role of legacy P sources in prolonging recovery in this lake and reducing the implementation of potentially unnecessary additional catchment management measures, where lake bed sediments were the dominant P source.

Living with the Past

The major goal for managing legacy P stores is to minimize their impacts on the quality and ecology of downstream receiving waters. A key factor for ecological impairment is the timing and form of legacy P mobilization and delivery to rivers. As described above, legacy P contributions in bioavailable (dissolved, inorganic) form, which are delivered at times of greatest biological demand (low flows in spring and summer), pose the greatest ecological risks to rivers. Thus, armed with our better understanding of system legacies, can smarter ecological engineering of BMPs lead to more effective BMPs across spatial and temporal scales?

Watershed management activities that disturb or decouple the interrelated P, C, and N cycles are of particular concern for biogeochemical mobilization of P during these ecologically critical times. For example, cycling of P stores along the land–freshwater continuum is closely related to organic C turnover, and it is increasingly recognized that nutrient management for water body restoration needs to incorporate management of organic C (Stanley et al., 2012). Organic C plays a major role as an energy source fueling microbial activity, which in turn controls organic matter mineralization, the release of P from organic matter, the rate of microbial P processing, and redox conditions that control P sorption, precipitation, and dissolution (House, 2003). For example, failing to adequately manage organic C loading to rivers from wastewater and septic tank effluents is a common cause of biogeochemical activation and release of in-channel legacy P stores under baseflow conditions (Jarvie et al., 2005; Palmer-Felgate et al., 2010).

Under low flows during the summer, when water retention times and temperatures are optimal for microbial activity,

breakdown of organic solids results in high rates of microbial respiration. This results in elevated biochemical O₂ demand, which, in turn, generates anaerobic conditions at the sediment–water interface and in the overlying water column, promoting reductive dissolution of ferric phosphate minerals and Fe oxyhydroxides. This microbially mediated mineral dissolution releases bound legacy P, stored in the stream sediments, into the water column in the form of highly bioavailable inorganic P (phosphate). Thus, failing to manage organic C loading from wastewater can activate streambed legacy P stores in an algal-available form at times of high ecological risk, even where the P has originated from other nonpoint (agricultural) sources (Palmer-Felgate et al., 2010).

As easily mined mineral P sources become scarcer and more costly to extract and process, it may become cost effective to recycle P locally from legacy P hotspots. Clearly, there are management techniques that allow us to divert and concentrate excess legacy P into exploitable and recyclable forms and allow us, even at a local scale, to start to close the disrupted P cycle. For example, many farm ponds and lagoons contain large amounts of P collected in runoff from agricultural facilities (Sharpley et al., 1989). While these ponds are efficient at trapping P on farms, they can ultimately become a source of dissolved P. Thus, recycling of deposited sediment enriched with P on agricultural land may become a cost-effective reuse of a finite resource (Cordell et al., 2009; Sharpley et al., 2012; Syers et al., 2008).

There are also increasing numbers of chemical and biological byproduct materials, originating as waste streams from various processes, which can be used as amendments to sequester P from either terrestrial or fluvial systems (Buda et al., 2012; Callahan et al., 2002; Penn et al., 2012; Stout et al., 1999, 2000). These materials include but are not limited to fluidized gas desulfurization gypsum, fly ash, Fe gels, red muds, and struvite, and their generation and reuse has been reviewed by Chardon et al. (2012), Egemose et al. (2012), and Weng et al. (2012). As pressures to minimize the size and extent of hotspots increases, so will the costs willing to be incurred to trap, remove, and recycle P.

In lakes, the use of P-sequestering materials to improve water quality and reduce downstream legacy P issues is increasing (Kennedy and Cooke, 1982; Mehner et al., 2008; Meis et al., 2012; Reitzel et al., 2005). These materials can be carefully selected to amend the physicochemical properties of bed sediments and reduce the likelihood of legacy P release to the water column (Hickey and Gibbs, 2009; Hupfer and Hilt, 2008; Leader et al., 2008; Spears et al., 2013). Their use as a means of sequestering P needs to be explored further. Similarly, sediment removal has been conducted to reduce the impacts of legacy P in lakes, although the destructive nature of this technique may limit its applicability in lakes of high conservation value.

Conclusions

The legacies of previous management and the remobilization of P stored in land, river, and lake systems are clearly important issues. Because of the lag time between BMP implementation and water quality improvements, remedial strategies should consider the time necessary for re-equilibration of watersheds and water bodies, where nutrient sinks may become sources of P with only slight changes in watershed management and hydrologic

response. Clearly, hotspots of P accumulation and legacy across watersheds are subject to complex interactions of hydraulics, hydrology, geomorphology, and land management. It will take time to address the P legacy at the field or headwater scale to “cascade” and “spiral” down to the larger watershed scale. The lags in water-quality response to BMP implementation at the downstream watershed scale will be determined by the key rate-limiting (slowest responding) parts of the system. Furthermore, environmental change (e.g., impacts of climate change on C cycles and hydrology) will have important implications for recovery, because of shifting baselines in hydrologic response, for instance.

A better understanding of the spatial and temporal aspects of watershed response to nutrient load reductions in both flowing and standing water bodies is needed, as well as an understanding of the scale at which responses may occur in a more timely fashion. It is likely that local water quality and quantity benefits may become evident more quickly at a smaller subwatershed scale. This is an important outcome in itself to help demonstrate change and foster accountability and ultimately wider adoption of conservation practices. As shown in the Coastal Plain poultry litter experiments, however, even at smaller scales, improvements at the field level may not immediately translate to the subwatershed. It is also important to accept in any watershed-P loss reduction strategy that it is essential to address the overall physical and social complexity of individual systems and the mitigation of non-agricultural sources of P. Only this will bring about lasting improvements in water quality as evidenced under all hydrologic (storm and non-storm) conditions.

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