

REVIEWS AND ANALYSES

Estimating dissolved phosphorus losses from legacy sources in pastures: The limits of soil tests and small-scale rainfall simulators

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Abstract

A legacy of using P fertilizers on grazed pastures has been enhanced soil fertility and an associated increased risk of P loss in runoff. Rainfall simulation has been extensively used to develop relationships between soil test P (STP) and dissolved P (DP) in runoff as part of modeling efforts scrutinizing the impact of legacy P. This review examines the applicability of rainfall simulation to draw inferences related to legacy P. Using available literature, we propose a mixing layer model with chemical transfer to describe DP mobilization from pasture soils where readily available P in the mixing layer is rapidly exhausted and contact time controls DP concentrations responsible for subsequent DP mobilization. That conceptual model was shown to be consistent with field monitoring data and then used to assess the likely effect of rainfall simulation protocols on DP mobilization, highlighting the influence of soil preparation, scale and measurement duration, and, most important, hydrology that can facilitate the physical transport of P into and out of surface flow. We conclude that rainfall simulation experimental protocols can have severe limitations for developing relationships between DP in runoff and STP that are subsequently used to estimate legacy P contributions to downstream water resources.

1 | INTRODUCTION

Excessive P concentrations in streams and water impoundments are a feature of many agricultural catchments where P amendments are applied to intensively grazed pastures (Grayson et al., 2001; McDowell et al., 2019; Pierson et al., 2001; Sharpley et al., 2007, 2012). In these systems P is lost in runoff as part of a solid, attached to a solid, or as a solute (i.e., dissolved in water). The terms “particulate” and “dissolved”

are commonly used to define P materials that pass through or are retained by a 0.45- μm filter. Particulate P (i.e., $>0.45\ \mu\text{m}$) comprises crystalline P, adsorbed P, and P in organic matter. On the other hand, dissolved P (DP; i.e., $<0.45\ \mu\text{m}$) is generally considered to be dominated by inorganic orthophosphate ($\text{H}_2\text{PO}_4^-/\text{HPO}_4^{2-}$), which is the form of P utilized by both terrestrial and aquatic organisms (Robinson et al., 1994; Walton & Lee, 1972); P attached to or contained in colloidal materials that are $<0.45\ \mu\text{m}$ (Beckett & Hart, 1993; Haygarth & Sharpley, 2000); and some organic P species, with mono- and diphosphate esters most common (Kolowith et al., 2001). In well-managed pastures, unless there is a

Abbreviations: DP, dissolved phosphorus; DRP, dissolved reactive phosphorus; STP, soil test phosphorus; TP, total phosphorus.

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predisposition to erosion (i.e., high slope, gully formation, dispersive soils, excessive animal treading damage), P in runoff is generally dominated by DP that is reactive in an acid-molybdate solution (Murphy & Riley, 1962); this DP is known as dissolved reactive P (DRP) and is considered to be dominated by orthophosphate (Fleming & Cox, 1998; Nash & Murdoch, 1997; Schuman et al., 1973; Sharpley et al., 1994).

Phosphorus exports in runoff from pasture-based grazing systems can be conveniently divided into “systematic” (i.e., base or background) and “incidental” (i.e., management) components (Haygarth & Jarvis, 1999). Phosphorus amendments can contribute directly to P exports soon after their application and by increasing soil fertility and P cycling. Although incidental P mobilized directly from amendments can account for a substantial portion (30–80%) of total farm P exports, when appropriately managed, that figure can be <10% (Nash & Hannah, 2011; Nash et al., 2019). Of more concern appear to be P surpluses and the increasing P concentrations in soil (Condon, 2004; Gourley et al., 2012; Neville et al., 2005; Stutter et al., 2015; Tian et al., 2017), the so-called “legacy P” (Kleinman et al., 2011) that represents a recalcitrant baseline underpinning systematic exports (Haygarth et al., 2014; Jarvie et al., 2013; Sharpley et al., 2013).

Various models have been used to investigate and regulate P exports from pasture-based grazing systems (Borah & Bera, 2004; Cichota & Snow, 2009; Kumaran et al., 2011). Process-based models should be ideal for this purpose, but there are often insufficient data to either adequately parameterize process models or provide confidence in their predictions at a field scale (Vaze et al., 2011). In the absence of better alternatives, various index systems (Melland et al., 2004; Sharpley et al., 2017) and hydrological modeling paired with an estimate of P export potential (Davies et al., 2006; Donigian et al., 1984; Novotny et al., 1978) have been used. Such tools are often informed by empirical relationships between soil test P (STP) and P concentrations developed from rainfall simulation studies (Vadas et al., 2005).

At the field scale, relationships between STP and DP concentrations in runoff can be tenuous. For example, in a study of 61 events (>2,500 L or ~0.15 mm) from Darnum in southeastern Australia (1994–2000), Nash et al. (2005) used an additive model to explain 61% of the variation in log-transformed total P (TP) concentrations (of which DRP comprised >93% of TP). Systematic (i.e., background) and incidental components of TP were all highly significant (Figure 1). However, despite STP at the Darnum site increasing from 17 to 57 mg Olsen P kg⁻¹ (0-to-100-mm deep soil sample) between 1994 and 2000, there appeared to be no relationship between STP and the systematic P concentrations in runoff water (Figure 1d). Such findings are consistent with other modeling of that area (Government of Victoria, Department of Primary Industries, unpublished data, 2011) and with field-scale studies elsewhere (Owens & Shipitalo, 2006).

Core Ideas

- Increased soil P is a legacy of fertilizer applications to pastures.
- Soil test P is often used to assess the risk of legacy P to down-stream water resources.
- We review P mobilization processes and rainfall simulation methodology.
- Rainfall simulation is shown to have major limitations for assessing the risks of legacy P.

The effectiveness of using STP to estimate P loads in some of the more complex modeling systems has been problematic (Cichota & Snow, 2009; Hollaway et al., 2018; Kumaran et al., 2011; Radcliffe et al., 2009). In most fate-and-transport models, changes in STP account for a small proportion of the total variability in estimated loads compared with contributions from hydrologic variability or incidental source contributions (e.g., fertilizer application, manure spreading). For example, Harmel et al. (2006) reported that “Significant linear relationships existed between STP and DP, particulate P, and TP loads, although the variability was quite large with adjusted $R^2 < 0.19$...” In New Zealand, load data from 46 sites with varying land uses and scales (e.g., <1 to >1,000 ha) were estimated “reasonably well” ($R^2 > .80$; $P = .001$) by the OVERSEER Nutrient Budgets model (Gray et al., 2016). However, that result could largely be attributed to a small number (e.g., $n = 5$) of farm-scale (e.g., 100–1,000 ha) dairy and/or mixed farming systems, where incidental factors appeared important, rather than to the skilled prediction of STP-derived DP losses.

The use of simulated rainfall and small plots (i.e., <100 m²) to study and estimate the effects of legacy P on P exports has been justified on the basis that those systems mimic the processes responsible for DP mobilization and saturated excess overland flow rather than quantify DP exports per se (Burkitt et al., 2011; Dougherty, Nash, et al., 2008; McDowell, Drewry, et al., 2003; McDowell, Monaghan, & Morton, 2003; Sharpley & Kleinman, 2003). Given the potential importance of legacy P, we investigate the veracity of using rainfall simulation to study processes controlling DP in runoff from pastures and to develop relationships (e.g., STP vs. P concentration) commonly used to estimate the contribution of legacy P to those losses. To do this, we briefly review conceptual models used to describe DP mobilization in pasture-based grazing systems in the absence of incidental factors. A mixing layer model with chemical transfer is described for DP mobilization from the grazing soils (i.e., pasture) and used to evaluate the influence of experimental protocols in small-scale rainfall simulation studies on inferences related to P source.

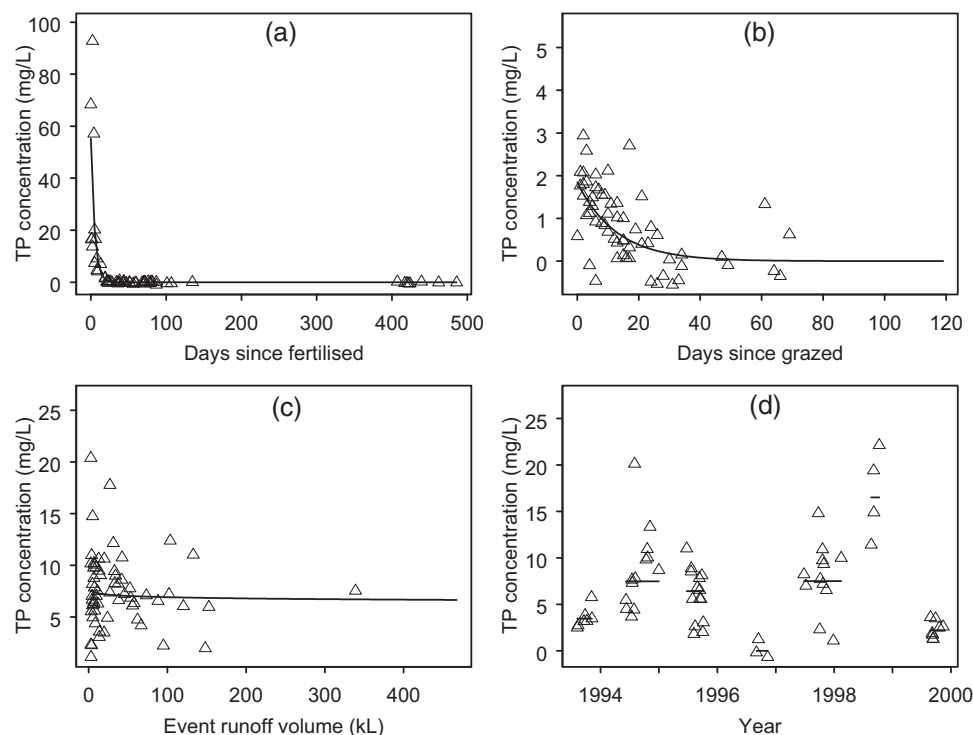


FIGURE 1 Relationships between adjusted (partial residual) total P (TP) concentrations in overland flow and (a) days since fertilizer was applied, (b) days since last grazed, (c) overland flow volume for the event, and (d) the effect of year for a site at Darnum in the Gippsland region of southeastern Australia. Points are back-transformed partial residuals (i.e., data with other model term estimates subtracted out). Lines are the relevant terms from the fitted model. Adapted from Nash et al. (2005)

2 | UNDERSTANDING DP MOBILIZATION FROM PASTURE SOILS

2.1 | Processes of dissolved P mobilization

Studies of P runoff from pasture soils under natural rainfall are consistent in describing an early spike in DP mobilization, then a decline as a storm stabilizes at a slightly lower rate and runoff DP concentrations are diluted with growing flow rate. Figure 2 is an example of such a relationship derived from field-scale (~2.5 ha) monitoring of a storm at Darnum in the Gippsland region of southeastern Australia. When data from that site were adjusted for flow, a characteristic peak in P concentrations at the start of a storm was generally followed by a slow decay (Nash, 2002). Similar processes in DP mobilization are revealed by irrigation studies. In border-check (also called border-dyke or flood) irrigation, water is applied at a constant rate from the inlet at the top of laser-leveled bays whose dimensions vary depending on water application rates, soil infiltration characteristics, and farm geometry (i.e., grade, 0.17–0.33%; length, 100–800 m; width, ~30–100 m). As water moves down the bay, the highest P concentrations are found in samples recovered from just behind the true wetting front, and increase with distance from the water inlet, presumably reflecting initial P mobilization (Process 1) from

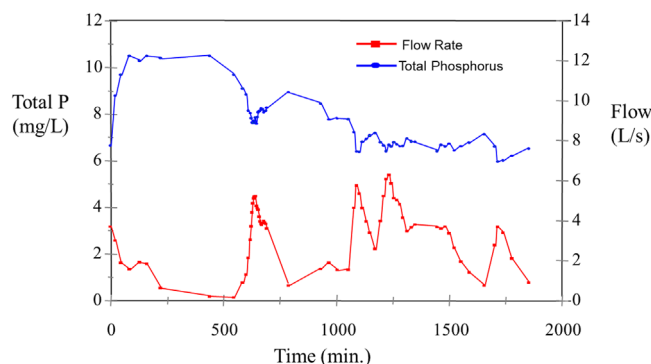


FIGURE 2 Within-storm variation in total P concentration and flow from a field in the Darnum region of West Gippsland, Australia, on 6 Nov. 1994. Adapted from (Nash & Murdoch, 1997)

the mixing zone (Nash, Webb, et al., 2007). Behind the wetting front, infiltration rates rapidly decline, and overland flow increases. Dissolved P concentrations in overland flow decline back up the bay toward the inlet but continue to increase as water moves down the bay, with the rate of increase tending to decline with distance from the wetting front (Nash & Barlow, 2009). Flume studies conducted in the absence of infiltration show similar slow P mobilization into overland flow (i.e., Process 2) (Barlow, 2003; Doody et al., 2006).

2.2 | Dissolved P mobilization

A range of conceptual models has been used to simulate solute mobilization into overland flow at the soil surface, many applicable to understanding DP loss in surface runoff from pasture soils. Such models include mixing layer models, interfacial-diffusion controlled models (Shi et al., 2012), interfacial diffusion and raindrop dispersion models, and empirical models (Shi et al., 2011). In pasture systems, evidence supporting a mixing layer model with chemical transfer includes mathematical modeling (Deng et al., 2006), rainfall simulation studies where water was applied on successive days (Sharpley, 1995), and field studies where strong relationships between overland flow volume and flow-weighted DP concentrations between storms have not been found (Nash et al., 2000, 2005).

Historically, DP mobilization has been conceptualized as a “desorption” reaction that occurs in a finite volume of surface soil where mixing is facilitated by raindrop impact and flowing water, and there is no chemical transfer into the mixing zone (Sharpley, 1985; Sharpley et al., 1981). Essentially this is a single mixing layer model, not dissimilar to laboratory experiments where a small sample of soil is shaken in a set volume of water (Sharpley et al., 1981). Dissolved P concentrations can be estimated from the product of “desorb-able” (i.e., mobilizable) P per gram of soil, the water/soil ratio, and the total mass of soil in the mixing zone (i.e., interacting zone) (e.g., Equation 1; Sharpley et al., 1981).

$$C_{ro} = (K P_o S 30^\alpha \cdot W^\beta) / R \quad (1)$$

where C_{ro} is the mean DP concentration for a 30-mm rainfall event starting from time zero; P_o is the desorbable P content of the soil; S is the mass of soil in the interacting zone; W is the water/soil ratio; R is the total rainfall during the event; and K , α , and β are constants.

In a single-layer mixing model, the depth of soil and water mixing determines the quantity of P available for mobilization, with a depth of ~3 mm expected in many well-managed pastures. High-intensity, simulated rainfall (65 mm h⁻¹) on bare soil packed into boxes amended with ³²P has been used to demonstrate that the depth of interaction between surface runoff and soil occurs within 3 mm of the surface (Ahuja et al., 1981). Natural rainfall is often less intense than simulated rainfall, and, in pasture-based grazing systems, the plant canopy and surface detritus would intercept raindrops. Both would be expected to lower the kinetic energy available for aggregate disruption and mixing (Sharpley, 1985), lowering the thickness of the mixing layer. On the other hand, in the field, poaching (i.e., surface perturbations) caused by animal traffic may well increase the “apparent” (i.e., effective) depth of interaction between surface runoff and soil P by increas-

ing the soil surface area and decreasing soil stability (Drewry, 2006; Drewry et al., 2008; Greenwood & McKenzie, 2001).

It is highly unlikely that a discrete layer of fixed depth exists in heterogeneous pasture soils or even that true overland flow is the only P transport pathway. The layer of detritus (e.g., decomposing organic matter including thatch, fragments of dead organisms and fecal material, and associated biological communities) near the soil surface is probably a reasonable approximation of a mixing layer that transitions to the soil below. However, although overland flow may occur, in many systems interflow in the surface detritus is common and functionally indistinguishable from true overland flow (Nash et al., 2002). Compaction of soil below the mixing layer by animal traffic would help suppress infiltration (Alderfer & Robinson, 1947; Drewry, 2006; Houlbrooke et al., 2009), contributing to interflow that manifests itself in the form of spongy, saturated surface soil that exudes water on compression (e.g., grazing animals). In these systems P is often exported offsite in a few large storms each year, and exports continue after rain when true overland flow ceases (Nash, 2002).

Given the inherent limitations of single mixing layer model, it is not surprising that more recent models of solute mobilization have similarities with interfacial-diffusion controlled models (Parr et al., 1987; Richardson & Parr, 1988; Wallach & van Genuchten, 1990) that provide for chemical transfer into or within the mixing layer (Wallach et al., 1988; Zhang et al., 1997). A mixing layer model with chemical transfer can be considered to have two phases. Initially (Process 1), contaminants (e.g., P) are rapidly mobilized from a layer or zone near the surface due to the physical interaction (i.e., mixing) of soil and water, as in a single mixing layer model. A second (Process 2), slower mobilization dominated by diffusion and hydraulic dispersion (i.e., mechanical mixing by velocity variations at the microscopic level during advective transport) transfers contaminants into the mixing layer from below and facilitates solute egress from aggregates, plants, and detritus both within and to the mixing layer.

Convective transport of dissolved pollutants such as P early in a storm (Process 1) is well documented (Wallach, 1991). Initially high rates of pollutant mobilization decline as labile supplies are exhausted (i.e., P mobilization is supply limited). On the other hand, the second slower process (Process 2) is rate limited. Although slow mobilization (i.e., Process 2) declines exponentially with diffusion path length, the residual pool of accessible P is greater due to an overall slower mobilization rate than Process 1 and access to a larger pool of P, especially from below. It follows that, although P mobilization tends to peak early in a storm, it often quickly stabilizes and marginally declines thereafter (Shi et al., 2011; Wallach & Shabtai, 1992).

In pastures, P mobilization rates are likely to be influenced by infiltrating water carrying P into the soil and away

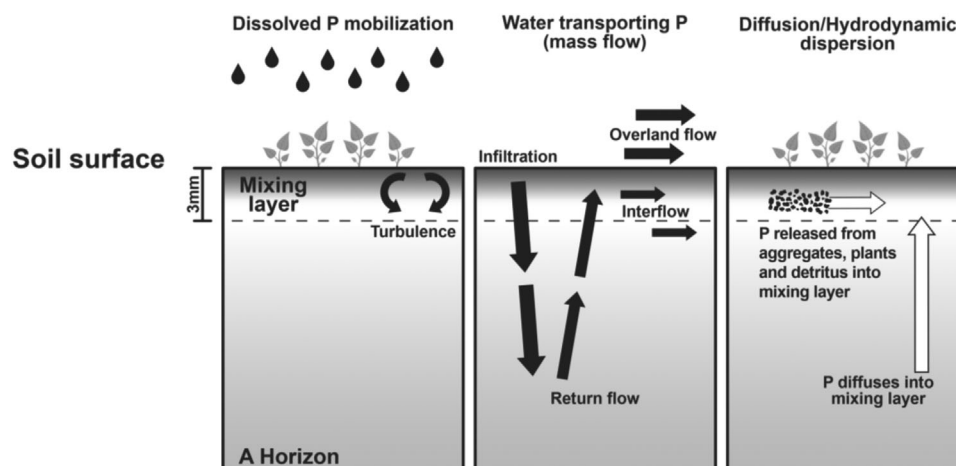


FIGURE 3 Field-scale processes contributing to dissolved P mobilization

TABLE 1 Initial soil moisture content (pre-rainfall), depth simulated rainfall applied to generate 30 min of runoff, depth of runoff generated in 30 min, and dissolved and total P concentration in runoff from unmanured Onteora and Marlow soils in New York and New Hampshire, respectively

Day of rainfall	Initial soil moisture $\text{m}^3 \text{m}^{-3}$	Rainfall applied mm	Runoff volume	Dissolved P mg L^{-1}	Total P
Onteora silt loam (coarse-loamy, mixed, semiactive, frigid Aquic Fragiudept)					
Day 1	0.13 (0.05)	64.0 (31.7)	1.7 (0.7)	0.22 (0.21)	0.92 (0.42)
Day 2	0.31 (0.03)	38.0 (2.1)	3.4 (1.8)	0.22 (0.13)	0.78 (0.31)
Day 3	0.33 (0.02)	38.0 (3.9)	5.2 (3.6)	0.18 (0.09)	0.97 (0.31)
Marlow sandy loam (coarse-loamy, isotic, frigid Oxyaquic Haplorthod)					
Day 1	0.20 (0.06)	95.6 (25.7)	3.9 (2.6)	0.38 (0.24)	0.55 (0.30)
Day 2	0.33 (0.03)	58.6 (14.5)	7.9 (3.4)	0.35 (0.25)	0.52 (0.27)

Note. Data adapted from Srinivasan et al. (2007). Number in parentheses indicates 1 SD.

from the mixing layer (Wallach et al., 2001) or, conversely, by return flow, especially from subsurface interflow (Snyder & Woolhiser, 1985), transporting additional P into the mixing layer (Figure 3) (Govindaraju, 1996). As a result, soil water content (i.e., antecedent soil moisture) and infiltration characteristics have been shown to influence P concentrations in runoff (Ahuja, 1990; Buda et al., 2009; Kleinman et al., 2006; Needelman, 2002; Snyder & Woolhiser, 1985; Vadas et al., 2008). For example, Srinivasan et al. (2007) found that DP and TP concentrations of runoff induced by simulated rainfall (60 mm h^{-1} to collect 30-min runoff) were a function of antecedent soil moisture and soil infiltration characteristics (Table 1). With successive daily rainfalls, consistent with the soil textures, soil moisture increased 154 and 65% for Onteora silt loam and Marlow sandy loam soils, respectively. Over these rainfalls, the amount of rainfall to induce runoff decreased and volume collected 30 min after initiation increased for both soils (Table 1).

2.3 | Sources of DP mobilized from pasture soils

Rather confusingly, the term “desorption” has often been used to describe DP mobilization into overland flow. Although “desorption” may have a broader context in relation to DP mobilization that can include physical processes (e.g., diffusion-controlled desorption) (Koopmans et al., 2004; Sharpley et al., 1981), it is questionable if the term “desorption” is appropriate for pasture-based grazing systems. Moreover, using the term “desorption” implies that reactions between orthophosphate and adsorption surfaces primarily control DP mobilization.

Dissolved P mobilization commences when rainfall or irrigation water washes over pasture plants and onto the soil surface. Although some irrigation waters may contain appreciable concentrations of P (Barlow et al., 2005), it is generally accepted that most P exports are sourced from

the soil–plant system (Greenhill et al., 1983). Numerous studies have shown that water can extract significant concentrations of P from the plant canopy (McDowell et al., 2011; Sharpley, 1981; Timmons et al., 1970), especially after grazing (Mundy et al., 2003). Plant exudates can contribute to a surface coating of soluble P mixed with dust that can be removed by rainfall. Phosphorus inside standing detritus and intracellular P are more difficult to mobilize (Dougherty et al., 2004).

On arrival at the soil surface, water is exposed to both inorganic and organic sources of P in the mixing layer. In pasture-based grazing systems, organic P often comprises >50% of total soil P (Perrott & Sarathchandra, 1989; Perrott et al., 1990; Richardson, 1994; Stutter et al., 2012, 2015). For example, a study of 15 pasture sites in southeastern Australia found that, whereas total P concentrations decreased with depth, organic P concentrations were generally three times higher than inorganic P concentrations, especially in the top 20 mm of soil (McLaughlin et al., 2011). Although relationships between organic carbon and P in overland flow have been demonstrated (Halliwell et al., 2000), estimating the proportion of P exports derived from inorganic (e.g., mineral fertilizer) versus organic P sources (e.g., animal manures) remains problematic.

In the absence of recently applied fertilizers or other P amendments (e.g., feces), the concentration of mobilizable P (e.g., orthophosphate) derived from inorganic sources would be controlled by sorption/desorption reactions and to a lesser extent by dissolution of precipitates (Norrish & Rosser, 1983; Sample et al., 1980; Sanyal & DeDatta, 1991; Schulthess & Sparks, 1991). Although in field soils chemical equilibria are unlikely to be uniform (White & Beckett, 1964), it could be expected that P concentrations in runoff attributable to desorption and dissolution of precipitates would only vary marginally over a year.

Phosphorus mobilized from organic sources, on the other hand, is likely to fluctuate with substrate availability and biological activity (Bromfield & Jones, 1970; Harrison 1987; Jones & Bromfield, 1969; Kaila, 1949). In Australian studies of “hayed-off” *Phalaris tuberosa* L.–*Trifolium subterraneum* L. pasture, 60–83% of total plant P was water soluble (i.e., orthophosphate), and >62% of that P was “leached” by simulated rainfall (Bromfield & Jones, 1972; Jones & Bromfield, 1969). In those studies, plants grown in more fertile conditions contained more TP, contained a higher proportion of total P in a water-soluble form, and lost a similar percentage of water-soluble P to leaching, implying proportionally higher P mobilization potential. Modeling based on these studies suggests that plants could contribute up to 56 kg ha^{−1} of water-soluble P annually to pastures (Nash & Halliwell, 1999). Phosphorus exports from rainfed pasture systems are commonly 10% of that figure (Dougherty et al., 2004; Harmel et al., 2006;

Vadas et al., 2015). More extensive reviews of soil organic P cycling related to pasture systems can be found elsewhere (Dalai, 1977; Nash et al., 2014).

The physical location of P released from organic sources in relation to other soil constituents is also likely to affect P mobilization. Although P in the form of orthophosphate (e.g., from plant exudates) on the surface of detritus may participate in adsorption processes with inorganic soil components, a considerable portion of P derived from organic sources would likely be encapsulated by the organic materials themselves. Moreover, at microsites where organically derived P may be sequestered, anionic organic matter may compete with P for adsorption sites in ligand exchange (Holford, 1989; Schulthess & Sparks, 1991), and, in acidic soils, organic matter may lower the activity of iron and aluminum (Vijayachandran & Harter, 1975), which would otherwise precipitate P from solution (Bloom et al., 1979; Lobartini et al., 1998; Thomas, 1975). There is little quantitative information on these mechanisms.

Clearly, the orthophosphate component of DP exported from pasture-based grazing systems can originate from inorganic or organic sources as a result of dissolution or desorption processes. The relative importance of a particular P source or process is likely to vary between seasons (Magid & Nielson, 1992; Pote et al., 1999a; Saunders & Metson, 1971; Shand et al., 1994). Understanding that variation and underlying causality would appear to be useful for developing robust mitigation strategies.

3 | THE ROLE OF EXPERIMENTAL PROTOCOLS IN RAINFALL SIMULATION OUTCOMES

Numerous studies have investigated relationships between STP and P exports in simulated overland flow. A selection of those studies and their experimental protocols are presented in Table 2. Rainfall simulation experiments can be divided into three classes based on soil or site preparation. For repacked trays, soil is often collected from the field, air dried, processed (e.g., crushed and sieved), and repacked into trays at near field soil bulk density (e.g., Kleinman et al., 2004). Some of the soil layering may be reconstituted (e.g., 0–20 mm) and/or plant cover may be established. For Intact turfs, wedges of soil and the associated plant cover are recovered from the field and placed in trays. For field plots, borders of various types are used to hydrologically isolate plots and collect overland flow in situ. The plots are often located in commercial fields or experimental plots. Specific aspects of rainfall simulation experiments that may affect their ability to accurately estimate the effects of legacy P on P exports are considered below.

TABLE 2 Selected rainfall simulation experiments and associated experimental protocols

Reference	Land use/pasture species	Simulation conditions (L × W × H)	Runoff analytes ^a	Pre-wetting	Application rate mm h ⁻¹	Duration	Slope %	Country
Pote et al. (1996)	fescue (<i>Festuca arundinacea</i> Schreb.)	field plots (6 m by 1.5 m)	DRP, BAP	saturated 2 d prior	100	30 min runoff	5	U.S.
Pote et al. (1999a)	fescue (<i>Festuca arundinacea</i> Schreb.)	field plots (6 m by 1.5 m)	DRP	saturated 2 d prior	75	30 min runoff	5	U.S.
Pote et al. (1999b)	tall fescue pasture	field plots (3 m by 1.5 m) maintained (0.1–0.2 m) by mowing	DRP	saturated 2 d prior	75	30 min runoff	7	U.S.
McDowell & Sharpley (2001)	cultivated fields/some grassland	Impermeable trays (1 m by 0.15 m by 0.15 m) packed with air-dried, sieved soil (<2 mm) soil from 0 to 0.075 m. U.K. soils were field plots (1 m by 0.15 m) with barriers to 0.075 mm.	DRP	unspecified	50	30 min	5	U.S., U.K.
Weld et al. (2001)	variable	Soil blocks in boxes (1 m by 0.2 m by 0.3 m) were subjected to three simulated rainfall events at 1-d intervals.	DRP	unspecified	65	30 min	–	U.S.
Fang et al. (2002)	sheep pasture, corn, soybean	Trays (0.61 m by 0.15 m by 0.1 m) were packed with soils from 0 to 0.2 m with vegetation and crop debris removed, trays were free draining, and duplicate simulations were performed.	DRP, BAP, TP	saturated	60	30 min	4	U.S.
McDowell & Sharpley (2002)	resown orchardgrass (<i>Dactylis glomerata</i> L.)	Field plots (1, 2, 4, 6, 8, and 10 m by 1 m) were mown to 0.02 mm.	DRP, TDP, TP	variable	70	60 min runoff	5	U.S.
Torbert et al. (2002)	bermudagrass (<i>Cynodon dactylon</i> L.)	Field plots (3 m by 2 m) with grass maintained by mowing were subjected to three simulated rainfall events.	DRP	unspecified	50	30 min runoff	~5	U.S.
McDowell, Drewry, et al. (2003a)	sheep and cattle pastures and cultivated grassland	Intact turfs (top 0.1 m) placed in trays (0.1 m ² ; depth, 0.125 m) with three holes for drainage were subjected to treading damage prior to rainfall simulation.	DRP, TDP, TP	maintained ~55% v/v	15	60 min runoff	5	New Zealand

(Continues)

TABLE 2 (Continued)

Reference	Land use/pasture species	Simulation conditions (L × W × H)	Runoff analytes ^a	Pre-wetting	Application rate	Duration	Slope	Country
McDowell, Monaghan, & Carey (2003)	pasture	Intact turfs (top 0.1 m) placed in boxes (1.05 m by 0.2 m by 0.125 m) with 6 mm by 2 mm holes for drainage received simulated rainfall before and after fertilizer treatments.	DRP, TDP, TP	soils stored outside	15	60 min runoff	5	New Zealand
McDowell Monaghan, & Morton (2003)	ryegrass (<i>Lolium perenne</i> sp.)	Trays (1.05 m by 0.2 m by 0.075 m) with 6 mm by 3 mm holes for drainage were packed with air-dried and sieved (6 mm) soil from 50 to 100 mm. Pasture was planted 8 mo prior to simulation and maintained by clipping (~0.05–0.08 m), with clippings returned to soil.	DRP, TDP, TP	soils stored outside and irrigated	15	60 min runoff	5	New Zealand
Sharpley & Kleinman (2003)	orchardgrass (<i>Dactylis glomerata</i> L.)	National Phosphorus Research Project Rainfall Simulator Protocol: Field plots (2 m by 1 m) received simulated rainfall on successive days. Water Erosion Prediction Project rainfall Protocol: Field plots (3.05 m by 10.7 m) received simulated rainfall on successive days.	DRP, TDP, TP	unspecified	75	30 min runoff	~5	U.S.
McDowell & Condron (2004)	grassland	Intact turfs (1 m by 0.2 m by 0.075 m) with pasture trimmed to 50 mm with 6 mm by 2 mm drainage holes	DRP	unspecified	15	60 min runoff	5	New Zealand
Schroeder et al. (2004)	hay or pasture production	Paired field plots (1 m by 1 m) were mown 1 wk prior to three rainfall simulations at 48-h intervals.	DRP, TP	presoaked 1 d prior	75	30 min runoff	–	U.S.
SERA-17 (2004)	variable	Field plots (2 m by 0.75–2 m) mown to ~0.1 m 1 wk prior to should receive simulated rainfall on consecutive days.	DRP, BAP, TDP, TP	saturated 2 d prior	~70 or 10-yr storm for location	30 min runoff	>2	U.S.

(Continues)

TABLE 2 (Continued)

Reference	Land use/pasture species	Simulation conditions (L × W × H)	Runoff analytes ^a	Pre-wetting	Application rate	Duration	Slope	Country
Dougherty et al. (2006b)	dairy pasture, ryegrass (<i>Lolium perenne</i> L.)	Trays (1 m by 0.2 m by 0.125 m) with 6 mm by 3 mm holes for drainage were repacked with sieved soil (<6-mm, 0-to-10-mm, and 10-to-100-mm depths packed as per field). Pasture was planted and the trays placed outside and irrigated with pasture maintained by clipping (clippings returned to soil) for 4 mo prior to simulation.	DRP, TDP	unspecified	30	~30 min application time with 20 min runoff	5	Australia
Shigaki et al. (2007)	annual ryegrass (<i>Lolium multiflorum</i> Lam.)	Trays (1 m by 0.2 m by 0.05 m) with 9 mm by 5 mm drainage holes were repacked with air-dried sieved (<19 mm) soil. Ryegrass with ryegrass sown 60 d before rainfall simulations and maintained by mowing.	DRP, TDP, TP	irrigated to field capacity 1 d prior	varying intensities of 25, 50, 75	varying durations of 60, 30, 20 min rainfall	3	U.S.
Dougherty, Nash, et al. (2008)	dairy pasture, ryegrass/clover pastures	Large field plots (50 m by 25 m) and small field plots (2 m by 0.75 m) were used in a nested design and had simulated rainfall applied on two occasions.	DRP, TDP, TP	unspecified	large plot 8; small plot 80	10 h application; 20 min runoff	5, 5	Australia
Dougherty, Nicholls, et al. (2008)	dairy pasture, ryegrass (<i>Lolium perenne</i> L.) and clover (<i>Trifolium repens</i> L.)	Large field plots (50 m by 25 m) were subjected to natural rainfall or sprinkler irrigation.	DRP, TDP, TP	unspecified	Irrigation ~8	Irrigation ~10 h	5	Australia
Robertson & Nash (2008)	sheep pasture, ryegrass (<i>Lolium perenne</i> L.), phalaris (<i>Phalaris aquatica</i> L.), and subterranean clover (<i>Trifolium subterranean</i> L.)	Field plots (2 m by 0.75 m) were subjected to simulated rainfall.	DRP, TP	unspecified	70	30 min runoff or terminated if no runoff occurred (e.g., >105 or 60 min)	1–4	Australia

(Continues)

TABLE 2 (Continued)

Reference	Land use/pasture species	Simulation conditions (L × W × H)	Runoff analytes ^a	Pre-wetting	Application rate	Duration	Slope	Country
Burkitt et al. (2010)	dairy pasture, ryegrass (<i>Lolium perenne</i> sp.) and clover (<i>Trifolium repens</i> L.)	Trays (0.8 m by 0.2 m by 0.1 m) with 20 mm by 2 mm holes for drainage were packed with soil from 0 to 20 mm and from 20 to 100 mm (100 mm total depth). Packed trays were resown with perennial ryegrass and placed in a glasshouse (day, 20 ± 5 °C; night, 10 ± 2 °C) >8 mo. before simulations with pasture maintained <50 mm.	DRP, TP	saturated 1 d prior	40	20 min runoff	5	Australia
Burkitt et al. (2011)	ryegrass (<i>Lolium perenne</i> sp.)	Trays (0.8 m by 0.2 m by 0.1 m) with 6 mm by 10 mm holes for drainage were packed with soil (<10 mm) from 0 to 20 mm and from 20 to 100 mm (100 mm total depth). Packed trays were resown with perennial ryegrass and placed in a glasshouse (day, 20 ± 5 °C; night, 10 ± 2 °C) >4 mo before simulations with pasture maintained <50 mm. Fly wire mesh was suspended over the trays, and drainage holes were plugged during simulations.	DRP, TDP, TP	Irrigated 1 d prior	50	20 min runoff	5	Australia
Dougherty et al. (2011)	long-term pasture sites	Trays (1 m by 0.2 m by 0.1 m) with 6 mm by 5 mm holes for drainage were packed with soil, sown with ryegrass (<i>Lolium perenne</i> sp.) and grown in a shade house for >9 mo.	TDP, TP	unspecified	45	30 min runoff	5	Australia

^aBAP, biologically available P; DRP, dissolved reactive P; TDP, total dissolved P; TP, total P.

3.1 | Soil preparation

Soil preparation prior to the application of simulated rainfall is likely to affect relationships developed using repacked trays. Increased P availability following wetting and drying cycles is well documented (Blackwell et al., 2010; Frossard et al., 2000), and, even if the soil was sampled in layers, it is highly unlikely that natural soil P stratification (Dougherty, Nash, et al., 2006) (i.e., the mixing layer and transition zone below it) could be re-established. The “apparent” mixing layer would contain only a fraction of the total soil P that would exist in the field, much like field soils that have been cultivated as part of pasture renovation (Dougherty, Davies, et al., 2006; Watkins et al., 2012).

Drying, crushing, and repacking trays is also highly likely to alter soil physical properties, including structure and aggregate stability, that affect the diffusion pathways that are important for rate-dependent P mobilization (Process 2) and soil infiltration characteristics that affect mass transfer of P into and out of the mixing zone (see Hydrology below). The anomalies created by reconstructing the soil profile do not occur to the same extent with intact turfs or field plots; this in part explains why small-scale, high-intensity rainfall simulation may underestimate natural runoff P concentrations from pastures (Dougherty, Nash, et al., 2008).

A lack of significant differences in extraction coefficients (i.e., gradient of a linear plot of STP against DRP concentration) between repacked trays and field plots (Vadas et al., 2005) has been used to argue that repacked trays with re-established pasture soils adequately represent, and compare well with, P mobilization under field conditions. However, a lack of significant differences is not of itself a measure of similarity. Moreover, it is unlikely that the chemical and physical attributes of a field soil could be reproduced by regrowing pasture in a glasshouse ($20 \pm 5^\circ\text{C}$) for <1 yr prior to experimentation, more so in the absence of “normal soil biology” and where harvested vegetation is removed. In field-scale studies, pastures that have been cultivated prior to being resown do not exhibit precultivation P concentrations in soil water (Watkins et al., 2012) and runoff (Nash, Webb, et al., 2007) until long after pasture is re-established (i.e., $>>1$ yr).

3.2 | Scale and measurement duration

Effects of flow path length and velocity are two interrelated parameters that affect soil–water contact time and P mobilization. The “time of concentration” (i.e., time for the entire watershed to contribute to measured flow and therefore the time available for soil–water interaction) (Grimaldi et al., 2012) is a concept used in hydrology to estimate the effect of a rain event and is related to the time to peak flow and rainfall

intensity (Michailidi et al., 2018). In rainfall simulation studies where P exports are often measured for <60 min after overland flow has commenced, the time of concentration is relatively short when compared to field-scale studies where overland flow persists for hours (Figure 2). Consequently, rapid, supply-limited P mobilization (i.e., Process 1) would be expected to dominate P export concentrations in simulated rainfall experiments. However, at the field scale, most P exports occur in a few major runoff events each year (Owens & Shipitalo, 2006) that often last longer than 1 h (Nash, 2002). It follows that in the field, P exports are more likely to reflect rate-dependent P mobilization (i.e., Process 2) rather than P mobilized directly from the mixing layer.

It is possible to simulate rainfall over larger areas to address the issues associated with scale (Sharpley & Kleinman, 2003). However, the additional water required for larger experiments tends to place an upper limit on the scale at which experiments can be conducted and the associated replication. In addition, it can be difficult to find larger plots that have consistent longitudinal and cross-sectional grades that do not encourage concentrated flow along the borders and thereby generate atypical field hydrology (Verbree et al., 2010).

3.3 | Hydrology

In most rainfall simulation studies, the hydrology is referred to using terms such as runoff, surface runoff, overland flow, or saturated overland flow. Although the latter description is true for the outlet, it is unlikely to be the case for other areas of the tray or plot. In small-scale simulation experiments, hydrology often varies between replicates within soil types, as demonstrated by good relationships between STP and P concentration and poor relationships (i.e., $R^2 < .02$) between STP and P loads (Pote et al., 1996). Because P concentrations in runoff reflect net P mobilization, understanding the hydrology of simulation experiments is important for interpreting their outcomes (Kleinman et al., 2006).

A schematic diagram describing the possible hydrology of experimental trays is presented in Figure 4a. It is likely that infiltration excess overland flow, saturation excess overland flow, and interflow are occurring simultaneously in different locations (Nash et al., 2002). The predominant hydrology will vary with soil infiltration rate and hydraulic conductivity (i.e., water transmission rate) (Nash, 1984) and with the degree of anisotropic behavior (i.e., variation in soil behavioral characteristics, such as hydraulic conductivity, when the direction of measurement is changed, for example from the vertical to horizontal plane).

Hydrological variability associated with small-scale rainfall simulation experiments has been used to justify prewetting soils shortly (i.e., $\sim 1\text{--}2$ d.) before both tray-based

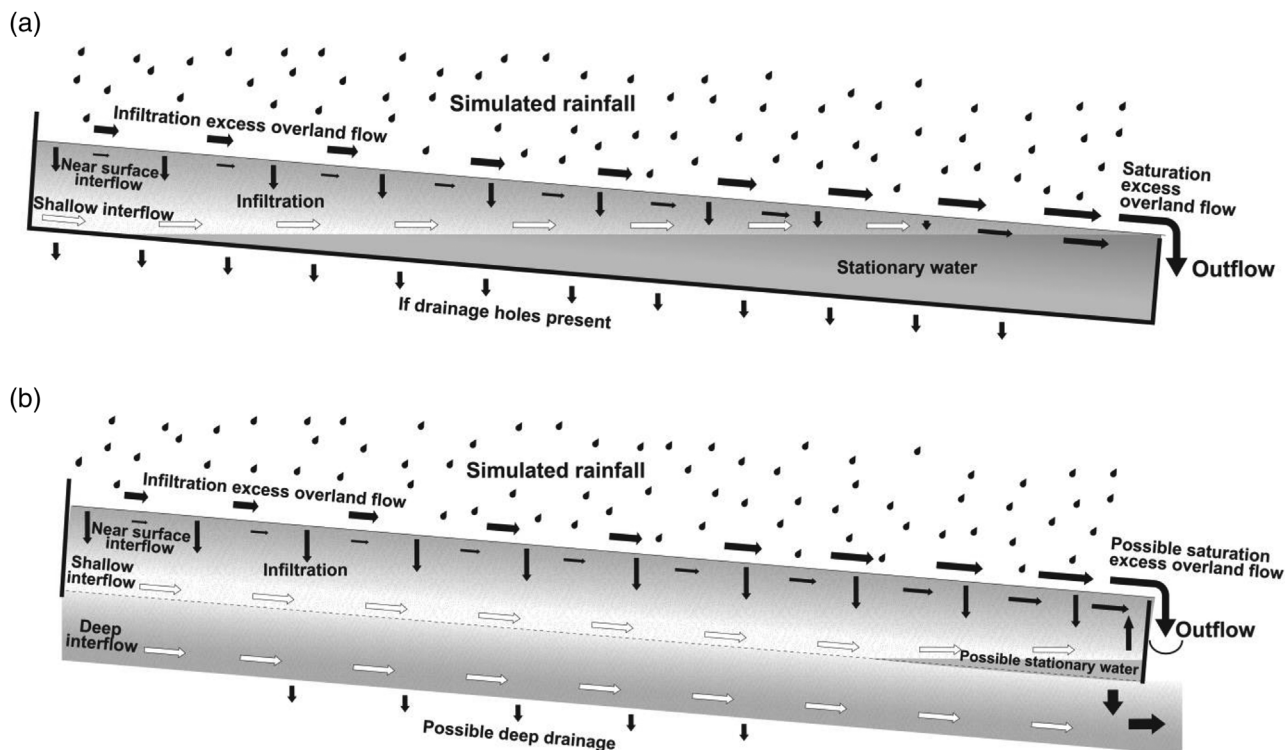


FIGURE 4 Cross-sectional profiles of possible hydrology in (a) trays and (b) plots used in small-scale rainfall simulation experiments

(Burkitt et al., 2011) and plot-based experiments commence (Pote et al., 1996) (Table 2). Prewetting is expected to move P from the mixing zone into subsoil storage. In that case, the simulation is assessing the short-term (i.e., <2 d) ability of Process 2 to resupply P into the mixing layer for a soil at near field capacity. Again, soil P buffering is expected to affect that resupply. It should be noted that any macropores (e.g., cracks) created during the recovery of intact soils are likely to increase both infiltration rates and hydraulic conductivity, thereby facilitating P storage that may be atypical of field soils.

Where field plots are used, the characteristics of the site (e.g., topography, initial soil water content) determine which is most prevalent, saturation excess- or infiltration excess-overland flow or interflow (Figure 4b) (Srinivasan et al., 2007). Pedological features are also critically important in gaining an understanding of the response of plot runoff to applied rainfall (as simulated or natural rainfall) (Buda et al., 2009; Needelman 2002). For instance, Needelman (2002) demonstrated using simulated rainfall that the presence of subsurface features that temporarily perched water, such as a fragipan or argillic horizon, led to saturation excess overland flow from alfalfa. However, on soils with minimal subsurface discontinuities, runoff was generated exclusively by infiltration excess overland flow (Needelman, 2002). In a watershed adjacent to that in the Needelman (2002) study (i.e., FD-36 in east-central Pennsylvania), Buda et al. (2009) showed that landscape position greatly influenced runoff

under natural rainfall conditions over 3 yr (i.e., 2002–2004). At the foot-slope site, saturation excess overland flow was nearly 20 times greater than infiltration excess overland flow; conversely, runoff from upslope sites was dominated by infiltration excess overland flow (Buda et al., 2009).

As an approximation, early in experiments, infiltrating water would be expected to transport P from the mixing layer into the soil. Conversely, later in an event, especially where subsoil conductivity is impaired, for example by grazing animals, or subsoils have low saturated hydraulic conductivity compared with topsoils, shallow (i.e., <100 mm) interflow may re-emerge at the surface, diluting the overland flow and near-surface interflow. Both overland flow and interflow increase with scale (e.g., run-on from areas higher in the catchment increases with scale) (Nash et al., 2002). It follows that different site preparation (e.g., pre-watering) and experimental protocols (e.g., scale) can emphasize different hydrological processes and thereby experimental outcomes, albeit with similar numerical outcomes in some cases (Cornish et al., 2002; Sharpley et al., 1982). Because of the effects of scale (McDowell & Sharpley, 2002), simply extending the duration of experiments is unlikely to replicate field-scale hydrology and mobilization processes.

Hydrological variability between sites and “replicates” has been investigated in several rainfall simulation studies (1 m wide by 2 m long; SERA-17, 2004). Overland flow was measured from side-by-side (i.e., A and B) 2-m² plots in New York (Honeoye loam) subjected to 70 mm h⁻¹ simulated

TABLE 3 Variability in overland flow volume and P transport from 1-m-wide and 2-m-long runoff plots under simulated rainfall following National Phosphorus Runoff Project protocols (SERA-17, 2004) in New York and Arkansas

Site Plot	1		2		3		4		5		6	
	A	B	A	B	A	B	A	B	A	B	A	B
Honeoye loam (fine-loamy, mixed, semiactive, mesic Glossic Hapludalf), New York (Kleinman et al., 2001)												
Mehlich-3 extractable soil P, mg kg ⁻¹												
	15.4	15.0	36.3	31.2	55	57.2	33	34.1	115.2	126.8	77.8	75.1
Initial soil moisture, m ³ m ⁻³												
Day 1	0.26	0.26	0.28	0.28	0.27	0.27	0.29	0.29	0.25	0.25	0.23	0.23
Day 2	0.33	0.33	0.31	0.31	0.27	0.27	0.27	0.27	0.33	0.33	0.31	0.31
Day 3	0.33	0.33	0.31	0.31	0.36	0.36	0.36	0.36	0.35	0.35	0.33	0.33
Volume of runoff, L ^a												
Day 1	45.55	38.75	1.77	3.69	26.50	8.45	19.55	10.45	38.50	36.00	29.00	24.80
Day 2	33.85	36.55	16.75	12.50	33.50	18.70	33.20	28.10	52.20	49.00	27.80	40.55
Day 3	39.78	46.75	37.35	27.75	30.60	30.50	32.50	33.10	52.15	46.88	43.80	45.70
Total P concentration, mg L ⁻¹												
Day 1	0.269	0.263	0.497	0.538	0.855	0.622	0.269	0.228	0.395	0.461	0.544	0.425
Day 2	0.228	0.180	0.281	0.257	0.419	0.347	0.156	0.204	0.568	0.353	0.317	0.323
Day 3	0.275	0.240	0.443	0.299	0.491	0.467	0.156	0.240	0.347	0.389	0.252	0.365
Total P loss, mg												
Day 1	12.27	10.21	0.88	1.99	22.67	5.26	5.27	2.38	15.21	16.59	15.79	10.54
Day 2	7.70	6.57	4.71	3.22	14.03	6.49	5.17	5.72	29.67	17.30	8.82	13.11
Day 3	10.96	11.20	16.54	8.31	15.01	14.24	5.07	7.93	18.10	18.24	11.02	16.69
Captina silt loam (fine-silty, siliceous, active, mesic Typic Fragiudult), Arkansas (Brennan et al., 2019)												
Mehlich-3 extractable soil P, mg kg ⁻¹												
	145.0	139.5	125.8	132.5	146.2	158.4	150.0	162.8	140.8	152.1		
Initial soil moisture, m ³ m ⁻³												
	0.22	0.30	0.26	0.31	0.32	0.33	0.29	0.32	0.28	0.30		
Volume of runoff, L ^a												
	16.80	26.19	9.85	31.30	8.00	24.50	7.60	17.05	6.70	34.45		
Total P concentration, mg L ⁻¹												
	1.882	2.212	1.052	1.404	1.450	1.306	8.350	2.940	1.664	1.472		
Total P loss, mg												
	31.62	57.92	10.36	43.95	11.60	32.00	63.46	50.13	11.15	50.71		

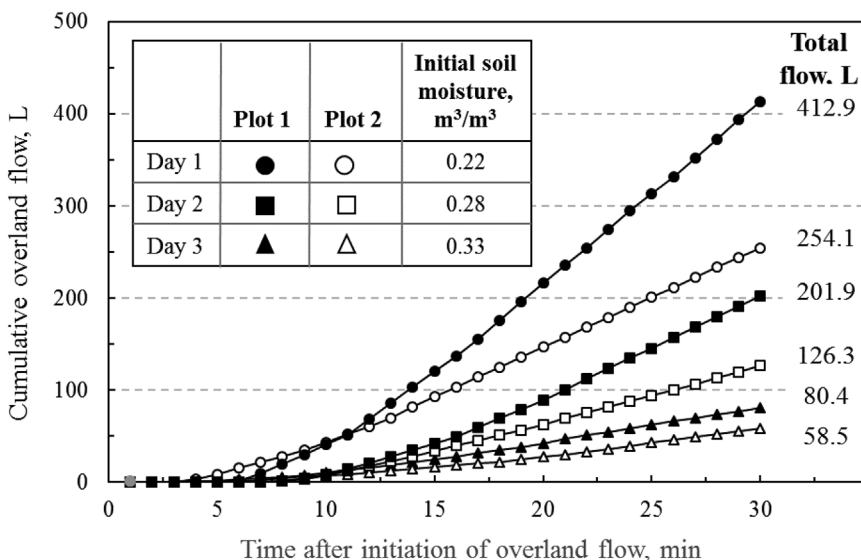
^aVolume of overland flow generated in 30 min after runoff initiation with simulated rainfall at 70 mm h⁻¹ and coefficient of uniformity of >85% uniformity.

rainfalls on three consecutive days (Table 3). Despite having similar moisture contents initially, runoff volumes between duplicates varied up to a factor of 3, especially on Days 1 and 2. Even though STP (as indicated by Mehlich-3 P) was similar between duplicate plots at a given site, TP concentrations and TP losses were widely divergent (e.g., 4.4-fold greater TP from Plot A than Plot B at Site 3; Table 3). The same trends were apparent for a Captina silt loam in Arkansas subject to 70 mm h⁻¹ simulated rainfall. Here, runoff from Plot B was consistently greater than from replicate Plot A (1.6- to 5.1-fold; Table 3) for most of the five sites, which contributed up to a 4.5-fold greater load from Plot B than from Plot A (i.e., site 5; Table 3).

For larger runoff plots (3.05 m wide by 10.7 m long) (Gilley & Flanagan, 2007; Laflen et al., 1991), simulated rainfall (75 mm h⁻¹ and coefficient of uniformity of >90% uniformity) was applied to replicate side-by-side plots in the mixed land-use watershed FD-36, a 39.5-ha subwatershed of Mahantango Creek Watershed, Pennsylvania (Sharpley & Kleinman, 2003). Even at this larger plot scale, overland flow volumes varied between plots for each of three consecutive daily rainfalls, with runoff from Plot 1 consistently greater than that from Plot 2 (Figure 5).

Variation in hydrological processes (e.g., overland flow initiation, duration, and volume) across scales (e.g., plot areas of 2–32.6 m²) and between plots can be advantageous in

FIGURE 5 Overland flow from paired 32.6-m² plots generated in 30 min after runoff initiation with 75 mm h⁻¹ simulated rainfall on three consecutive days



studying P mobilization processes. Findings from such plot studies demonstrate the variability in factors driving P exports from agricultural landscapes, especially where initial P mobilization is independent of hydrology and largely supply limited (e.g., for incidental factors such as effluent spreading and fertilizer application). In those instances, careful examination of between-plot hydrology could also facilitate interpretation of P transport and mobilization/demobilization processes and therefore provide valuable information on relationships between soil, fertility, and land management and the concentration of P in runoff (Kibet et al., 2014; Sharpley & Kleinman, 2003; Srinivasan et al., 2007; Wainwright et al., 2000). However, where P mobilization is rate limited and heavily reliant on contact time, flow path, and path length, as is most likely the case for legacy P (i.e., Process 2), hydrological variation across and between replicates in most rainfall simulation experiments is likely to preclude careful examination of P export processes. In either case, quantitative extrapolation from simulation experiments to field scale is inappropriate.

3.4 | Soil test P methodology

Numerous soil tests have been used to infer DP export potential (Maguire & Sims, 2002; Nash, Hannah, et al., 2007; Sims, 1993, 2000). Agronomic soil tests (i.e., 0–10 cm) were designed to relate soil fertility to plant growth/yield (i.e., predict response to additional fertilizer). They generally measure “quantity” (the concentration of soil P that is potentially available for uptake during the crop cycle) or “intensity” (the immediately available P; i.e., P activity or orthophosphate P concentration in soil solution (Moody & Bolland, 1999). In Australasia, Olsen P and Colwell P are the two most common agronomic soil tests. Both use bicarbonate solutions (0.5 M NaHCO₃ at pH 8.5) to extract P. The shorter-

duration (30 min) Olsen P test measures both P quantity (i.e., supply) and intensity (i.e., concentration), whereas the Colwell P (16 h) predominantly measures P quantity (Moody & Bolland, 1999).

Because P mobilization is both a function of the P concentration in solution and the ability to resupply that P, neither test would be expected to reliably predict short-term (i.e., <48 h) mobilization of DP by water or changes in P availability over the year, especially those attributable to organic sources and organically mediated P availability (Fox et al., 1990). The so-called environmental tests include water soluble P and dilute (0.01 M) calcium chloride extractable P, which both measure P intensity, and P sorption saturation, which estimates the extent of soil binding sites occupied by P and, therefore, P desorption potential (Kleinman, 2017). Given that most small-scale rainfall simulation studies appear to focus on short-term P mobilization from the mixing layer (Process 1), there should be strong relationships (e.g., $R^2 > .7$) between STP and DP in runoff and even stronger relationships for the environmental tests, much as a single-layer mixing model would suggest (Sharpley et al., 1981). None of the common soil tests consider organic sources of P and their possible indirect (i.e., effects on P sorption) contributions to P exports.

4 | CONCLUDING DISCUSSION

An ability to estimate the systematic (i.e., background) component of P exports is necessary for evaluating the effects of legacy P and optimizing mitigation measures. Although it is widely acknowledged that empirical relationships between STP and P concentrations derived from small-scale simulation studies do not quantitatively translate well to the field scale, by necessity, STP is currently the tool of choice for assessing systematic P export potential.

Despite the regular differences in experimental protocols (Table 2), most small-scale rainfall simulation studies find good relationships between STP and P concentrations in runoff. Being of short duration, such studies replicate the rapid mobilization of P early in a storm (i.e., Process 1) in a similar way to laboratory adsorption experiments. Their time scale evens out some hydrological variation, albeit atypical of most field-scale hydrology. Furthermore, intensive soil sampling around the time of rainfall simulation minimizes the temporal and spatial variation that would be expected in the field (Department of Natural Resources & Environment, 2002; Morton et al., 2000; Mountier & During, 1967; Simpson et al., 2015).

The limited setup conditions imposed by rainfall simulation protocols constrain the evaluation of factors and mechanisms that have produced STP vs. DP relationships. Consequently, extrapolation to larger scale is precarious (Shepherd et al., 2013), especially given that the simulation data from which such equations are derived do not represent the time-dependent, rate-limited process (i.e., Process 2) accounting for most P exports or field scale hydrology (i.e., transport mechanisms). It is therefore not surprising that most field-scale studies yield poor quantitative relationships (i.e., $R^2 < .2$) between STP and systematic P concentrations.

All else being equal, one would expect DP concentrations in runoff to increase with an increase in STP. However, in the field all things are rarely equal. Farming systems are complex, with interrelated capital (e.g., soil type, topography), management (i.e., history, grazing strategy), and environmental (e.g., climate) components that are conditionally dependent on one another. Is it possible to encapsulate that complexity in a single metric (i.e., extraction co-efficient or soil test) derived from small-scale rainfall simulation experiments that can be used to estimate systematic P concentrations and loads, even within a soil type, without reference to hydrology?

There are modeling platforms (e.g., Bayesian Belief Networks) that use cause-and-effect relationships and probability theory that can be used to iteratively develop conceptually sound mechanistic (i.e., quasi process) models (Lucci et al., 2014; McDowell et al., 2009; Nash & Hannah, 2011; Nash et al., 2010; Nash, Waters, et al. 2013). Importantly, these systems can interface with conventional modeling (Barton et al., 2012; Bicking et al., 2019; Nash, Riffkin, et al., 2013; Phan et al., 2016; Pollino & Henderson, 2010). Small-scale rainfall simulation studies have potentially an important role to play in providing input to such modeling efforts. However, more detailed analyses of the hydrology and other scale-dependent processes (i.e., the system components), rather than simply reporting apparent relationships, is needed. The emphasis needs to be more on “how and why,” rather than on “what” happened and working iteratively toward a more process-oriented approach.

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David M. Nash: Conceptualization; Writing-original draft. A. J. Weatherley: Conceptualization; Writing-review & editing. Peter J.A. Kleinman: Conceptualization; Writing-review & editing. Andrew N. Sharpley: Conceptualization; Writing-review & editing.

CONFLICT OF INTEREST

The authors declare no conflict of interest.

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