

Rainfall intensity and phosphorus source effects on phosphorus transport in surface runoff from soil trays

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Abstract

Phosphorus runoff from agricultural fields amended with mineral fertilizers and manures has been linked to freshwater eutrophication. A rainfall simulation study was conducted to evaluate the effects of different rainfall intensities and P sources differing in water soluble P (WSP) concentration on P transport in runoff from soil trays packed with a Berks loam and grassed with annual ryegrass (*Lolium multiflorum* Lam.). Triple superphosphate (TSP; 79% WSP), low-grade super single phosphate (LGSSP; 50% WSP), North Carolina rock phosphate (NCRP; 0.5% WSP) and swine manure (SM; 70% WSP), were broadcast (100 kg total P ha⁻¹) and rainfall applied at 25, 50 and 75 mm h⁻¹ 1, 7, 21, and 56 days after P source application. The concentration of dissolved reactive (DRP), particulate (PP), and total P (TP) was significantly ($P < 0.01$) greater in runoff with a rainfall intensity of 75 than 25 mm h⁻¹ for all P sources. Further, runoff DRP increased as P source WSP increased, with runoff from a 50 mm h⁻¹ rain 1 day after source application having a DRP concentration of 0.25 mg L⁻¹ for NCRP and 28.21 mg L⁻¹ for TSP. In contrast, the proportion of runoff TP as PP was greater with low (39% PP for NCRP) than high WSP sources (4% PP for TSP) averaged for all rainfall intensities. The increased PP transport is attributed to the detachment and transport of undissolved P source particles during runoff. These results show that P source water solubility and rainfall intensity can influence P transport in runoff, which is important in evaluating the long-term risks of P source application on P transport in surface runoff.

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1. Introduction

Accelerated eutrophication, the biological enrichment of surface waters stemming from anthropogenic inputs of nutrients, is the most common surface water impairment in the United States (U.S. Environmental Protection Agency, 1996; U.S. Geological Survey, 1999); limiting water use for drinking, recreation, and industry (Howarth et al., 2000; Sharpley, 2000). As P most often limits freshwater eutrophication, the decrease

Abbreviations: DCP, dicalcium P; DM, dry-matter; DRP, dissolved reactive P; LGSSP, low-grade single superphosphate; NCRP, North Carolina rock phosphate; PP, particulate P; SSP, single superphosphate; SM, swine manure; TP, total P; TSP, triple superphosphate; WSP, water soluble P.

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of P loss in runoff has become a major target to minimize surface water degradation (Carpenter et al., 1998; Boesch et al., 2001). According to U.S. Environmental Protection Agency (2004), agriculture is the primary source of non-point source pollution degrading the quality of streams and lakes. The potential for P loss can increase with the continual application of P as mineral fertilizers and manures. Thus, controlling accelerated eutrophication of fresh waters mainly requires decreasing P inputs to surface waters (Shigaki et al., 2006a).

Two major factors influencing P movement in runoff are transport and source (Gburek et al., 2000; McDowell et al., 2004; Sharpley et al., 2001). Transport factors are those mechanisms causing P movement within the landscape, such as erosion and runoff. Source factors contribute P to runoff and include soil P concentration and the rate, timing, method, and form of P applied (as fertilizer and/or manure). Related to “source factors,” there are many types of P fertilizers available such as organic (bone meal, cottonseed meal, fish emulsions, manures and processed sewage sludge) and inorganic (reactive rock phosphates, acidulated phosphates and thermophosphates) materials (Leikam and Achorn, 2005). The most common P fertilizer used is acidulated triple superphosphate (TSP) due its high P solubility and thus, immediate availability of P for crop uptake. Other fertilizers with lower water P solubility, such as highly reactive North Carolina rock phosphate (NCRP) and low-grade single superphosphate (LGSSP) can be used to maintain a longer-term source of P due to their residual availability and lower cost (Chien and Menon, 1995; Leon et al., 1986; Prochnow et al., 2004). Recent studies have shown a close relationship between the water soluble P (WSP) of land-applied fertilizers and manure on P runoff. Kleinman et al. (2002) found the WSP concentration of dairy, poultry, and swine manure (SM) applied to the surface of three soils to be highly correlated with dissolved reactive P (DRP) in surface runoff ($R^2=0.86$). Shigaki et al. (2006b) evaluated P loss in surface runoff after applying TSP (79% WSP), LGSSP (50% WSP), NCRP (0.5% WSP), or SM (30% WSP) at a rate of 100 kg P ha⁻¹. They found that cumulative losses for DRP and TP followed the increase in source WSP (TSP>LGSSP>SM>NCRP).

The influence of transport factors runoff and erosion on P movement in runoff is well known (Hart et al., 2004; Quinton et al., 2001). An increase in P transport (mainly as particulate P, PP) has been observed with increasing runoff and erosion induced by greater rainfall intensities (Quinton et al., 2001). Further, Sporre-Money et al. (2004) found that high intensity rainfall

causes a greater proportion of total P in runoff to be particulate P (PP) than low-intensity rainfall. Fraser et al. (1999) monitoring surface runoff from fields planted with winter cover crops observed that with an increase in rainfall intensity from 1 to 10 mm h⁻¹, there was an increase in erosion (14 to 114 kg ha⁻¹ h⁻¹), DRP (15 to 218 g ha⁻¹ h⁻¹), and PP loss (16 to 319 g ha⁻¹ h⁻¹) in runoff. Also, Sharpley (1985a) reported that the effective depth of interaction between surface soil and runoff P for five Oklahoma and Texas soils, increased from 2.7 to 14.2 mm with an increase in rainfall intensity from 50 to 160 mm h⁻¹ at a 2% soil slope and from 6.9 to 27.3 mm at a 20% slope. However, there are few studies controlling variables such as rainfall intensity and P source WSP. This information is needed to provide a better understanding of P transport process in order to develop site-specific P management strategies that minimize P loss and protect water quality.

This paper describes the effects of rainfall intensity on P transport in surface runoff from packed soil trays amended with P sources varying in WSP. Simulated rainfall (25, 50, and 75 mm h⁻¹) was used following the protocol of the National Phosphorus Research Project (2006). In the present study, mineral P sources and swine manure (usually defined as an organic source) were used. However, we will be referring both to mineral and organic P additions as “sources of P” to facilitate clarity of our discussion.

2. Material and methods

2.1. Soil and P sources

2.1.1. Soil collection

The surface 7.5 cm of a Berks loam (loamy-skeletal, mixed, active, mesic Typic Hapludults) was collected from the agricultural watershed FD-36, in April 2005. FD-36 is a sub-watershed (39.5 ha) in the Mahantango Creek watershed, which drains into the Susquehanna River and ultimately the Chesapeake Bay. The sampled soil depth equates to the depth of soil that is used in the runoff trays (5 cm). Soil was air-dried and sifted through a 19-mm sieve, and thoroughly mixed prior to use in runoff trays. Pre-treatment of soil was minimal and a coarse sieve used to remove stones and plant residues. Ten subsamples of soil were collected, combined, thoroughly mixed by end-over-end shaking, and sieved (2 mm) prior to physical and chemical characterization.

2.1.2. Soil analyses

Soil was analyzed for pH with a glass electrode on a mixture of 5 g air-dried soil with 5 mL distilled water.

Soil cation exchange capacity (CEC) was determined utilizing 1 M NH_4OAc adjusted to pH 7.0 (Hendershot et al., 1993). Extractable Fe and Al were determined by shaking 0.5 g of soil in 20 mL of 1 M NH_4OAc (adjusted to pH 4.8) for 2 h (Olson, 1965; McLean, 1965). Particle size analysis was conducted by the hydrometer method after dispersion with sodium hexametaphosphate (Day, 1965). Mehlich-3 extractable soil P was determined by shaking 1 g of soil with 10 mL of 0.2 M CH_3COOH , 0.25 M NH_4NO_3 , 0.015 M NH_4F , 0.013 M HNO_3 , and 0.001 M EDTA for 5 min (Mehlich, 1984). Total soil P was analyzed by modified semi-micro Kjeldahl procedure (Bremner, 1996). In all cases, P concentrations were determined on filtered (Whatman No. 1 paper¹) and neutralized extracts and digests by colorimetric method of Murphy and Riley (1962).

2.1.3. P sources

Three mineral P sources varying in water soluble P and swine manure were selected for this study (Table 2). Mineral P sources used were LGSSP (obtained by the acidulation of the apatite concentrated Araxa rock phosphate), TSP and NCRP. Water solubility of mineral sources was determined by Association of Official Analytical Chemists Official Method 977.01; and ammonium citrate — EDTA solubility by Method 993.31 (Association of Official Analytical Chemists, 1995).

Dry-matter (DM) concentration of SM was determined gravimetrically after drying at 70 °C. The WSP concentration of swine manure was determined by shaking 1 g (dry weight) of field moist manure with 200 mL distilled water for 1 h (Sharpley and Moyer, 2000). The total P concentration of swine manure was determined by the modified semi-micro Kjeldahl procedure (Bremner, 1996). Water extracts and digests were filtered (Whatman No. 1 paper) and the P determined on water extracts and neutralized digests by the colorimetric method of Murphy and Riley (1962).

2.2. Runoff trays and rainfall simulation

Stainless steel runoff trays packed with air dried and sieved (19 mm) soil, were used in accordance with the National P Research Project Protocol (National P Research Project, 2006). The trays were 1-m long, 20-cm wide and 5-cm deep with side and back walls 2.5 cm higher than the soil surface, and 5-mm drainage holes in the base (Fig. 1). Cheese cloth was placed in the bottom

of the tray, followed by sufficient soil, which was slightly tamped to achieve a uniform bulk density of 1.4 g cm^{-3} . This bulk density approximates that of the sampled Berks soil. After the trays were packed with soil, annual ryegrass (*Lolium multiflorum* Lam.) was planted in to the trays to guarantee a vegetative cover for this study. Runoff trays were watered regularly to promote grass growth but did not induce surface runoff. The grass was cut 7 days after planting to allow root establishment. After incubation and growth for 60 days, the grass was cut to a 5-cm height, 2 days before P source application to ensure a uniform cover among the trays. To ensure that initial soil moisture was consistent among treatments, soil trays were irrigated to approximately field capacity (soil moisture 30%). Each P source was surface applied 1 day before the first rainfall event at a rate of $100 \text{ kg TP ha}^{-1}$. Each treatment was replicated four times.

The P application rate of $100 \text{ kg TP ha}^{-1}$ is equivalent to an N-based manure application to grass in this region (Beegle, 2002). Although this rate is higher than normal fertilizer rates (50 to 75 kg TP ha^{-1} ; Beegle, 2002), it was used to compare the relative effects of P source application on P transport in runoff. Lower rates of P source were not used due to the high variability on uniformity of application found for the extremely small amount of NCRP and TSP needed ($<1 \text{ g}/0.2 \text{ m}^2$ tray).

The grass was cut to a 5-cm height after the last rainfall (56 days after P source application), grass DM determined gravimetrically after drying at 70 °C and total P concentration measured on ground ($<0.05 \text{ mm}$) duplicate 0.2 g subsamples by the modified semi-micro Kjeldahl digestion procedure (Bremner, 1996). Digests were filtered (Whatman No. 1 paper) and the P determined on water extracts and neutralized digests by the colorimetric method of Murphy and Riley (1962).

Runoff was generated by applying artificial rainfall on inclined (3%) soil runoff trays with a TeeJet™ 2HH-SS24WSQ², TeeJet™ 2HH-SS30WSQ² and TeeJet™ 2HH-SS50WSQ² nozzles approximately 2.5 m above the soil to achieve terminal velocity (Bowyer-Bower and Burt, 1989; Humphry et al., 2002; Shelton et al., 1985). Rainfall was delivered at 25, 50 and 75 mm h^{-1} with coefficient uniformity >0.83 within the 4 m^2 area directly below the nozzle (coefficient of uniformity = standard deviation \times mean of rainfall intensity as determined on a 20-cm grid spacing). These rates represent return periods of 2, 5, and 10 years, respectively, in south central Pennsylvania (Flippo, 1977). For each intensity, rainfall duration was controlled to give the same total amount of rainwater applied to each tray. For instance, rainfall lasted

¹ Mention of trade names does not imply endorsement by the U.S. Department of Agriculture.

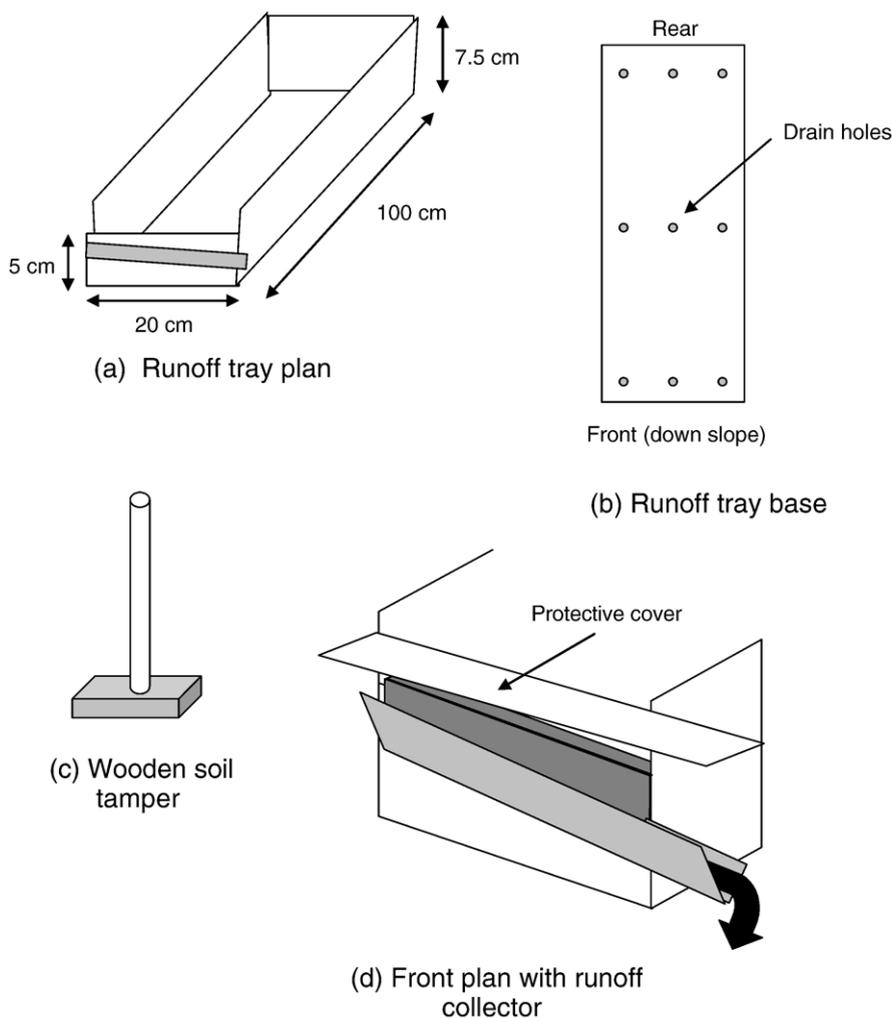


Fig. 1. Schematic diagram of the runoff tray used in this study, including (a) overall dimensions, (b) drain holes in the base of the tray, (c) wooden soil tamper to fill tray with soil, and (d) front plan with runoff collector.

60, 30 and 20 min at 25, 50, and 75 mm h⁻¹ intensities, respectively, giving 5 L of rain water. Rainfall was applied 1, 7, 21, and 56 days after P source application. The total volume of runoff generated from each tray was collected via a gutter, equipped with a canopy to exclude direct input of rainfall and inserted at the lowest edge of the runoff tray (Fig. 1).

2.2.1. Water analysis

A 500 mL sample of runoff was taken from each container after thorough mixing and agitation, and a sub-sample of 50 mL was immediately filtered through a 0.45- μ m pore diameter membrane filter and stored at 4 °C. Filtered samples were analyzed within 24 h of collection and unfiltered samples no more than 7 days after the completion of the rainfall simulation. The concentration of DRP in filtered runoff samples was determined on a

0.45 μ m filtered sample. The concentrations of both total dissolved P (TDP) and total P (TP) were determined on filtered and unfiltered runoff samples, respectively, following acid persulfate digestion and filtration (Whatman No. 40 filter paper²) (Patton and Kryskalla, 2003). Each of these P measurements was conducted in duplicate. Phosphorus in all stream water filtrates and neutralized digests was determined by the colorimetric molybdenum-blue method of Murphy and Riley (1962). Solutions were neutralized using *p*-nitrophenol indicator (color change at pH 7.0) and drop-wise addition of either 0.5 M H₂SO₄ or 1.0 M NaOH. As DRP was consistently >95% of TDP, colloidal and dissolved organic P that passed through the 0.45 μ m filter was assumed to be

² Mention of trade names does not imply endorsement by the USDA.

Table 1
Properties of the P sources used in this study

P source	Total P	Water soluble P		Citrate-EDTA+H ₂ O soluble P		Moisture content
	g kg ⁻¹	g kg ⁻¹	% total P	g kg ⁻¹	% total P	%
North Carolina rock P	133	0.6	0.5	16.4	12	0
Low-grade single super P	79	39.9	50.5	54.9	70	0
Triple super P	201	159.0	79.1	185.3	92	0
Swine manure	1.0	0.7	70.0	ND [†]	ND [†]	90

[†] ND Data not determined.

negligible and not presented in this study. Further, PP was calculated as the difference between TP and DRP.

It should be emphasized that use of trays packed with soil and simulated rainfall are intended to evaluate treatment effects in P transport under controlled conditions and do not replace field plots. Thus, they are to be used to evaluate the specific effects of certain variables in P transport and not to quantify losses that may occur under a field setting.

2.3. Statistical analysis

Statistical analyses were performed using the Statistical Analysis System (SAS Institute, 2001). Data were analyzed for normality using Proc Univariate (SAS Institute, 2001), and was normally distributed, so it was not necessary to transform the data. Analysis of variance (ANOVA) techniques were used to determine treatment effects. Comparisons of means were performed using Duncan's studentized range test at $P < 0.05$ (SAS Institute, 2001).

3. Results and discussion

3.1. Soil and phosphorus source properties

Physical and chemical soil properties of Berks loam soil selected for this study are presented in Table 2. Mehlich-3 extractable soil P concentration (69 mg kg⁻¹)

was slightly greater than the optimum value for cultivated soils in Pennsylvania (60 mg kg⁻¹; Beegle, 2002) and less than a third the environmental soil P threshold for Pennsylvania (200 mg kg⁻¹; Weld et al., 2003). The site from which the Berks soils sample was collected, had not received any fertilizer or manure P in the previous 10 years.

For the P sources used, large differences were observed in WSP concentration, which increased in the order NCRP, SM, LGSSP, and TSP. However, NCRP had a higher TP concentration (133 g kg⁻¹) than LGSSP (79 g kg⁻¹) and SM (1 g kg⁻¹). As a result, the percent water solubility of TP increased in the order NCRP (0.5%), LGSSP (50.5%), SM (70.0%), and TSP (79.1%) (Table 1).

There was no significant difference ($P > 0.05$) in dry-matter production of ryegrass among the P source amendments and control. Ryegrass yield averaged 25.8 g (± 3.7) DM tray⁻¹ during 60 days between P source application and final rainfall and grass harvest. This grass production is equivalent to 7850 kg DM ha⁻¹ year⁻¹ (or 5.5 tons acre⁻¹) and is consistent with area yields (Beegle, 2002). Additionally P uptake by ryegrass was not affected ($P > 0.05$) by applied P; averaging 0.07 g (± 0.02) TP tray⁻¹ or 13.6 kg P ha⁻¹ year⁻¹. The lack of P source response by ryegrass results from the fact that Mehlich-3 extractable P of source soil used (69 mg kg⁻¹; Table 2), was greater than optimum crop response values (60 mg kg⁻¹). As ryegrass production and cover were similar across all treatments and control, vegetative cover did not influence the transport of P in runoff from the soil trays.

3.2. Runoff phosphorus and phosphorus source

Single relationships were used to describe the dependence of WSP on DRP and PP concentrations in runoff 1 day after P source application (Figs. 2 and 3). At each rainfall intensity, there was a significant ($P < 0.01$) exponential increase in runoff DRP and PP with P source WSP 1 day after application (R^2 ranged from 0.95 to 0.98). Similar relationships between P source WSP and runoff DRP and PP were observed 7,

Table 2
Properties of Berks soil used in this study

Soil	Mehlich-3 P	Total P	CEC [‡]	pH	C	Al	Fe	Sand	Silt	Clay
	mg kg ⁻¹		meq 100 g ⁻¹		g kg ⁻¹	mg kg ⁻¹		%		
Berks	69	457	10.7	6.3	17.1	23.7	22.5	48	34	18

[‡]CEC, cation exchange capacity.

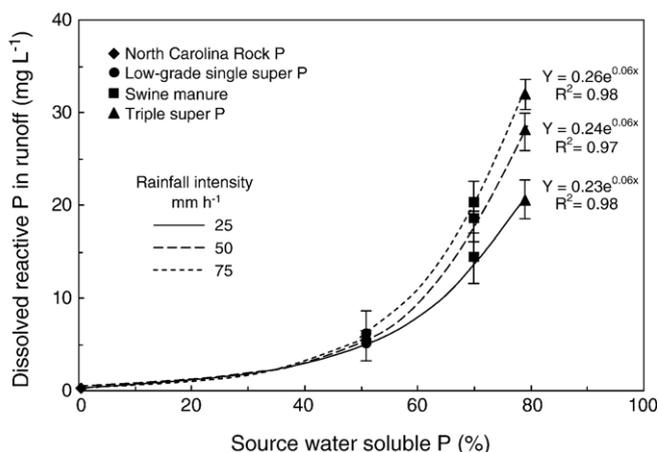


Fig. 2. Variation in the concentration of dissolved reactive P in runoff with source water soluble P, 1 day after source application.

21, and 56 days after P source application, although the magnitude and significance of concentration change was less. Most of the TP was as DRP (89%) averaged for all sources, rainfall intensity, and events (Table 3).

The concentration of DRP in surface runoff decreased rapidly with time after the application of LGSSP, TSP, and SM, at each rainfall intensity (Table 3). For instance, the concentration of DRP 1 day after applying TSP was 28.21 mg L^{-1} and after 56 days was 0.74 mg L^{-1} , with a 50 mm h^{-1} rainfall intensity. The decline in DRP in runoff from 1 and 56 days after P source application averaged for the three rainfall intensities, increased as a function of P source WSP (5.03 mg L^{-1} for LGSSP, 15.54 mg L^{-1} for SM, and 26.22 mg L^{-1} for TSP). In fact, DRP decline was closely correlated with P source WSP ($R^2 > 0.95$).

For NCRP, having a low WSP (0.5% of TP as WSP), the decline in DRP concentration between 1 and 56 days rain was 0.09 mg L^{-1} averaged for the three rainfall intensities (Table 3). There was a lower decrease in PP with time after P source application compared with DRP, although at the highest rainfall intensity (75 mm h^{-1}), TSP decreased 1.94 mg L^{-1} between 1 and 56 days rains, LGSSP 1.53 mg L^{-1} , and SM 1.00 mg L^{-1} . For NCRP there was no consistent effect of rainfall–runoff timing on PP in runoff, such that the slow dissolution of NCRP maintained a large amount of undissolved fertilizer particles at the soil surface that could be transported during runoff (Table 3). Sharpley (1997) also observed declining concentrations in runoff DRP over successive runoff events from 10 different soils mixed with poultry manure. The decline is due to dilution of source P on the surface and sorption of P translocated into the soil by infiltration (Vadas et al., 2004).

3.3. Runoff phosphorus and rainfall intensity

There was a significant ($P < 0.01$) increase in DRP and PP concentration in runoff as rainfall intensity increased from 25 to 75 mm h^{-1} 1 day after P source application (R^2 ranging from 0.95 to 0.98; Figs. 2 and 3). For all P sources 1 day after application, the concentration of DRP, PP, and TP was significantly greater ($P < 0.05$) in runoff with a 75 than 25 mm h^{-1} rainfall intensity, except for runoff DRP for NCRP. Although there was a trend of increasing P concentration as rainfall increased, values for the 50 mm h^{-1} intensity were not consistently greater than for 25 mm h^{-1} or lower than 75 mm h^{-1} intensities. Rainfall intensity has no effect ($P > 0.05$) on DRP concentration in runoff from the controls (Table 3), for PP, however, runoff concentrations 7, 21, and 56 days after P source application were greater for 75 than

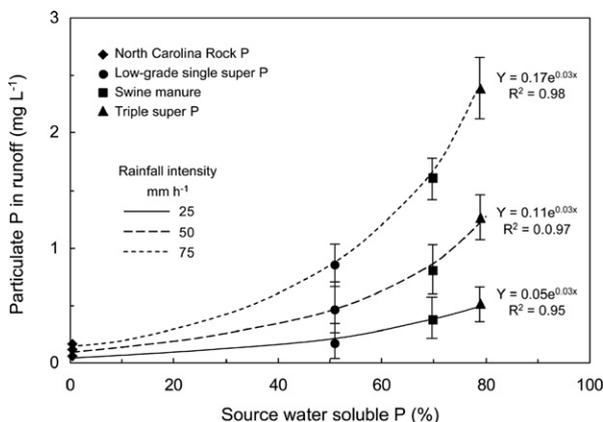


Fig. 3. Variation in the concentration of particulate P in runoff with source water soluble P, 1 day after source application.

Table 3

Concentration of dissolved reactive, particulate and total P in surface runoff from soil trays as a function of P source WSP, days after application and rainfall intensity [†]

Rain mm h ⁻¹	Control				NCRP				LGSSP				TSP				SM			
	1 d	7 d	21 d	56 d	1 d	7 d	21 d	56 d	1 d	7 d	21 d	56 d	1 d	7 d	21 d	56 d	1 d	7 d	21 d	56 d
<i>Dissolved reactive, P mg L⁻¹</i>																				
25	0.20a	0.16a	0.13a	0.10a	0.23a	0.23a	0.14a	0.12a	5.08a	1.27a	0.58a	0.43a	20.75a	2.48a	0.91a	0.68a	14.41a	3.62a	2.75a	1.92a
50	0.22a	0.23a	0.17a	0.14a	0.25a	0.24a	0.19a	0.16a	5.43ab	1.31ab	0.58a	0.46a	28.21b	2.69a	1.18a	0.74a	18.52b	4.29b	2.93a	2.04a
75	0.27a	0.26a	0.20a	0.18a	0.27a	0.26a	0.21a	0.20a	6.11b	1.54b	0.73b	0.62b	32.07b	3.46a	1.83b	0.94a	20.18b	4.46b	2.98a	2.23a
<i>Particulate P, mg L⁻¹</i>																				
25	0.08a	0.11a	0.05a	0.06a	0.10a	0.13a	0.06a	0.06a	0.62a	0.83a	0.08a	0.16a	0.62a	0.43a	0.32a	0.13a	0.21a	0.54a	0.36a	0.42a
50	0.09ab	0.14a	0.11ab	0.10ab	0.11a	0.19ab	0.13ab	0.11ab	0.87a	1.15ab	0.25b	0.19a	1.17b	0.37a	0.35a	0.15a	1.06b	0.65a	0.33a	0.46a
75	0.19b	0.15a	0.13b	0.14b	0.27b	0.25b	0.22b	0.20b	1.69b	1.43b	0.27b	0.16a	2.32c	0.84a	0.77b	0.38b	1.68c	1.15b	0.69b	0.68a
<i>Total P, mg L⁻¹</i>																				
25	0.28a	0.27a	0.18a	0.16a	0.33a	0.36a	0.20a	0.18a	5.70a	2.10a	0.66a	0.59a	21.37a	2.91a	1.23a	0.81a	14.62a	4.16a	3.11a	2.34a
50	0.31a	0.37ab	0.28ab	0.24ab	0.36ab	0.43ab	0.32ab	0.27ab	6.30a	2.46ab	0.83a	0.65a	29.38b	3.06a	1.53a	0.89a	19.31b	4.94a	3.26a	2.50a
75	0.47b	0.41a	0.33b	0.32b	0.54b	0.51b	0.43b	0.40b	7.80b	2.97b	1.00a	0.78a	34.39c	4.30b	2.60b	1.32b	21.86b	5.61b	3.67a	2.91a

[†] Concentrations of each P form as a function of rainfall intensity for each source overtime followed by different letters are significantly different ($P < 0.05$) as determined by analysis of variance.

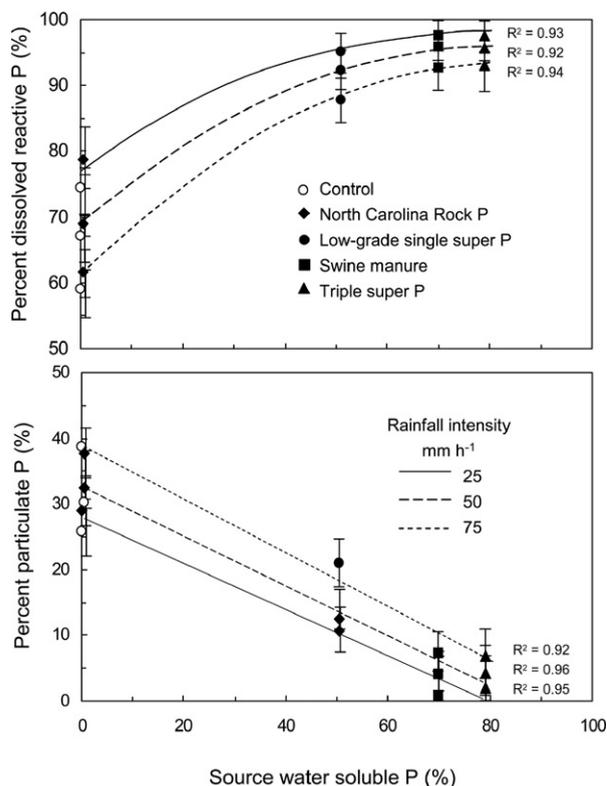


Fig. 4. Variation in the percent of total P as dissolved reactive and particulate P in runoff with source water soluble P, 1 day after source application, as a function of rainfall intensity.

25 mm h⁻¹ intensity. Total P concentrations in runoff for the 75 mm h⁻¹ intensity were consistently greater than the 25 mm h⁻¹ intensity for all P sources and rainfall timing, except for LGSSP and SM after 21 days (Table 3).

The increase in the proportion of runoff TP as DRP with increasing P source WSP was significant ($P < 0.01$) for each rainfall intensity 1 day after P application (Fig. 4). Although similar trends were observed for runoff 7, 21, and 56 days after P source application, the magnitude of change and significance was much less than 1 day after P source application. With an increase in rainfall intensity, there is a greater degree and depth of interaction between runoff and surface soil (Ahuja et al., 1981; Sharpley, 1985a). Kleinman et al. (2006) also observed that when rainfall intensity increased from 29 to 70 mm h⁻¹, the resulting runoff had a greater DRP (0.35 and 0.70 mg L⁻¹, respectively) and TP concentration (0.55 and 0.99 mg L⁻¹, respectively). There is thus, a greater mass of soil and thereby, larger pool of P that can be released to runoff water at high compared to low rainfall intensities; hence high runoff DRP concentrations.

There was a concomitant increase in the proportion of runoff TP as PP 1 day after application of P sources

(Fig. 4). Clearly, as rainfall intensity increased from 25 to 75 mm h⁻¹, the proportion of TP in runoff as PP increased for all sources. The increase in proportion of TP as PP with increasing rainfall intensity was greater with less soluble P sources (Fig. 4). For example, 1 day after P application, percent PP increased from 30 to 50% for NCRP (0.5% WSP), 11 to 22% for LGSSP (50% WSP), 1 to 8 for SM (70% WSP), and 3 to 7 for TSP (79% WSP) as rainfall intensity increased from 25 to 75 mm h⁻¹. These results suggest a difference in the fate in soil and runoff water of the different P sources used in this study.

It is likely that the slower rate of dissolution of the NCRP resulted in an increased movement or erosion of undissolved fertilizer particles in runoff, with highest percentage of PP for the highest intensity. For example, an increase in rainfall intensity (60 to 120 mm h⁻¹) was found to increase the erosive energy of resulting runoff (Sharpley, 1985b). In addition, several studies have shown that as erosion increases, the proportion of P exported in runoff as PP also increases (Sharpley and Smith, 1990). For instance, Sharpley et al. (1978) observed appreciably greater PP loss in surface runoff (4.92 kg P ha⁻¹), following the application of dicalcium P (DCP), a slow release fertilizer compared with the application of single superphosphate (SSP; 2.63 kg ha⁻¹) to permanent pasture in New Zealand. However, the loss of DRP was lower with DCP (2.17 kg ha⁻¹) than SSP (2.80 kg P ha⁻¹). Also, Sharpley and Syers (1982) evaluated the effects of SSP (50 kg P ha⁻¹) applied in a solid or liquid form, on the transport of DRP and PP in surface runoff from established pasture using field plots and natural rainfall. They observed that the application of solid SSP resulted in a much greater PP concentration of surface runoff (4.02 mg L⁻¹) than with liquid SSP (1.62 mg L⁻¹). It is likely that undissolved fertilizer particles may be transported directly in surface runoff following the solid fertilizer application in addition to sediment-bound PP, and that this effect is greater at higher rainfall intensities (Figs. 3 and 4).

4. Conclusions

Results from this research clearly show the influence of water P solubility of P sources and rainfall intensities on P concentration in surface runoff. For each rainfall intensity used in this study, there was a significant and exponential increase in runoff DRP and PP with P source WSP (TSP > SM > LGSSP > NCRP). Phosphorus concentrations in surface runoff were higher in the first rainfall event 1 day after P source application and decreased with time.

With the increase in rainfall intensity from 25 to 75 mm h⁻¹ the proportion of TP as PP was greater for NCRP (the less soluble P source), which is attributed to the slow rate of dissolution of this P source resulting in an increased movement of undissolved fertilizer particles in runoff. Although TSP provides a ready and large amount of P readily available for plant uptake, it is also an immediate source of P to runoff. With time after application, P enrichment of runoff rapidly declines due to depletion of this P source. Following application of a lower WSP P source, such as LGSSP or NCRP, runoff P enrichment is not as great as with TSP.

Whilst the proportion of runoff TP as PP after NCRP application was greater than with other sources, the loss as DRP was lower. In the short term, therefore, NCRP will have less effect on runoff P enrichment and biological response of receiving waters, as DRP is immediately available for algal uptake (Sharpley, 1993). However, PP can be a long-term source of P to algal growth in surface waters.

Using the information gained from this research, it may be possible to make fertilizer management decisions on which P source could be applied if environmental impacts were a major concern and driving variable. Given that the pH of the Berks soil used in this study was 6.3, it is likely that NCRP dissolution may be greater in more acidic soils (pH < 6.0). Additional research is needed on a wider range of soil types to extend this important information.

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References

Ahuja LR, Sharpley AN, Yamamoto M, Menzel RG. The depth of rainfall–runoff–soil interaction as determined by ³²P. *Water Resour Res* 1981;17:969–74.

Association of Official Analytical Chemists. Fertilizers — phosphorus. In: Cunniff P, editor. Official methods of analysis of AOAC

International. Agricultural Chemicals; Contaminants; Drugs, 16th edition. Arlington, VA: AOAC International; 1995. p. 1–12.

Beegle DB. Soil fertility management. In: Martz E, editor. *Agronomy guide 2002*. Publications distributions center, College of Agricultural Sciences. University Park, PA: The Pennsylvania State University; 2002. p. 18–42.

Boesch DF, Brinsfield RB, Magnien RE. Chesapeake Bay eutrophication: scientific understanding, ecosystem restoration, and challenges for agriculture. *J Environ Qual* 2001;30:303–20.

Bowyer-Bower TAS, Burt TP. Rainfall simulators for investigating soil response to rainfall. *Soil Technol* 1989;2:1–16.

Bremner JM. Nitrogen — total. In: Sparks DL, editor. *Methods of soil analysis*. Part 3. Soil Sci Soc Am Book Series, Madison, WI: Soil Sci Soc Am; 1996. p. 1085–121.

Carpenter SR, Caraco NF, Correll DL, Howarth RW, Sharpley AN, Smith VH. Nonpoint pollution of surface waters with phosphorus and nitrogen. *Ecol Appl* 1998;8:559–68.

Chien SH, Menon RG. Factors affecting the agronomic effectiveness of phosphate rock for direct application. *Fertil Res* 1995;41:227–34.

Day PR. Particle fractionation and particle size analysis. In: Black CA, et al, editor. *Methods of soil analysis*. Part 1. Agronomy Monograph Madison, WI: Am Soc Agron; 1965. p. 545–67.

Flippo Jr HN. *Floods in Pennsylvania*. Bulletin, vol. 13. Harrisburg, PA: Pennsylvania Department of Environmental Resources; 1977.

Fraser AI, Harrod TR, Haygarth PM. The effect of rainfall intensity on soil erosion and particulate phosphorus transfer from arable soils. *Water Sci Technol* 1999;39(12):41–5.

Gburek JW, Sharpley AN, Forlmar GJ. Critical areas of phosphorus export from agricultural watersheds. In: Sharpley AN, editor. *Agriculture and phosphorus management: the Chesapeake Bay*. Boca Raton, FL: Lewis Publishers; 2000. p. 83–104.

Hart MR, Quinn BF, Nguyen ML. Phosphorus runoff from agricultural land and direct fertilizer effects: a review. *J Environ Qual* 2004;33:1954–72.

Hendershot WH, Lalande H, Duquete M. Ion exchange and exchangeable cations. In: Carter MR, editor. *Soil sampling and methods of analysis*. Boca Raton, FL: Lewis Publishers; 1993. p. 167–76.

Howarth RW, Anderson DA, Church TM, Greening H, Hopkinson CS, Huber W, et al. *Clean Coastal Waters: Understanding and Reducing the Effects of Nutrient Pollution*. National Research Council. National Academy Press, Washington, D. C. 2000; 405 pages.

Humphry JB, Daniel TC, Edwards DR, Sharpley AN. A portable rainfall simulator for plot-scale runoff studies. *Appl Eng Agric* 2002;18:199–204.

Kleinman PJA, Sharpley AN, Moyer BG, Elwinger GF. Effect of mineral and manure phosphorus sources on runoff phosphorus. *J Environ Qual* 2002;31:2026–33.

Kleinman PJA, Srinivasan MS, Dell CJ, Schmidt JP, Sharpley AN, Bryant RB. Role of rainfall intensity and hydrology in nutrient transport via surface runoff. *J Environ Qual* 2006;35:1248–59.

Leikam DF, Achorn FP. Phosphate fertilizers: production, characteristics, and technologies. In: Sims JT, Sharpley AN, editors. *Phosphorus: agriculture and the environment*. Agron Monograph-Madison, WI: Am Soc Agron; 2005. p. 23–50.

Leon LA, Fenster WE, Hammond LL. Agronomic potential of eleven phosphate rocks from Brazil, Colombia, Perú, and Venezuela. *Soil Sci Soc Am J* 1986;50:798–802.

McDowell RW, Biggs BJB, Sharpley AN, Nguyen L. Connecting phosphorus loss from agricultural landscapes to surface water quality. *Chem Ecol* 2004;20:1–40.

- McLean EO. Aluminum. In: Black CA, et al, editor. Methods of soil analysis. Part 2. Agronomy Monograph Madison, WI: Am Soc Agron; 1965. p. 963–73.
- Mehlich A. Mehlich-3 soil test extractant: a modification of Mehlich-2 extractant. *Commun Soil Sci Plant Anal* 1984;15:1409–16.
- Murphy J, Riley JP. A modified single solution method for the determination of phosphate in natural water. *Anal Chim Acta* 1962;27:31–6.
- National Phosphorus Research Project. National phosphorus research project for simulated rainfall — surface runoff studies. Southern Extension-Research Activity- Information Exchange SERA-IAG; 2006. Available at http://www.sera17.ext.vt.edu/Documents/National_P_protocol.pdf. last verified October 12, 2006.
- Olson RV. Iron. In: Black CA, et al, editor. Methods of soil analysis, Part 2. Agronomy Monograph Madison, WI: Am Soc Agron; 1965. p. 978–98.
- Patton CJ, Kryskalla JR. Methods of analysis by the U.S. Geological Survey National Water Quality Laboratory: evaluation of acid persulfate digestion as an alternative to Kjeldahl digestion for the determination of total and dissolved nitrogen and phosphorus in water. U.S. Geological Survey, Water Resources Investigations Report 03-4174. Denver, CO: U.S. Geological Survey, Branch of Information Services, Federal Center; 2003. 33 pp.
- Prochnow LI, Chien SH, Carmona G, Henao J. Greenhouse evaluation of phosphorus sources produced from a low-reactive Brazilian phosphate rock. *Agron J* 2004;96:761–8.
- Quinton JN, Catt JA, Hess TM. The selective removal of phosphorus from soil: is event size important? *J Environ Qual* 2001;30: 538–45.
- SAS Institute. SAS OnlineDoc. Version 8.02. Cary, NC: SAS Inst.; 2001.
- Sharpley AN. Depth of surface soil–runoff interaction as affected by rainfall, soil slope and management. *Soil Sci Soc Am J* 1985a;49:1010–5.
- Sharpley AN. The selective erosion of plant nutrients in runoff. *Soil Sci Soc Am J* 1985b;49:1527–34.
- Sharpley AN. Assessing phosphorus bioavailability in soils and runoff. *Fertil Res* 1993;36:259–72.
- Sharpley AN. Rainfall frequency and nitrogen and phosphorus in runoff from soil amended with poultry litter. *J Environ Qual* 1997;26:1127–32.
- Sharpley AN, editor. Agriculture and phosphorus management: the Chesapeake Bay. Boca Raton, FL: CRC Press; 2000. 229 pages.
- Sharpley AN, Syers JK. Transport of phosphorus in surface runoff as influenced by liquid and solid fertilizer phosphate addition. *Water Air Soil Pollut* 1982;19:321–6.
- Sharpley AN, Smith SJ. Phosphorus transport in agricultural runoff: the role of soil erosion. In: Boardman J, Foster IDL, Dearing JA, editors. Soil erosion on agricultural land. Chichester, England: J. Wiley and Sons; 1990. p. 351–66.
- Sharpley AN, Moyer B. Phosphorus forms in manure and compost and their release during simulated rainfall. *J Environ Qual* 2000;29: 1462–9.
- Sharpley AN, Syers JK, Gregg PEH. Transport in surface runoff of phosphorus derived from dicalcium phosphate and superphosphate. *NZ J Sci* 1978;21:307–10.
- Sharpley AN, McDowell RW, Kleinman PJA. Phosphorus loss from land and water: integrating agricultural and environmental management. *Plant Soil* 2001;237:287–307.
- Shelton CH, von Bernuth RD, Rajbhandari SP. A continuous-application rainfall simulator. *Trans Am Soc Agric Eng* 1985;28 (4):1115–9.
- Shigaki F, Sharpley AN, Prochnow LI. Animal-based agriculture, phosphorus and management and water quality in Brazil: options for the future. *Sci Agric* 2006a;63(2):194–209.
- Shigaki F, Sharpley AN, Prochnow LI. Phosphorus source and the transport of phosphorus in surface runoff. *J Environ Qual* 2006b;35:2229–35.
- Sporre-Money JL, Lanyon LE, Sharpley AN. Low-intensity sprinkler for evaluating phosphorus transport from different landscape positions. *Appl Eng Agric* 2004;20(5):599–604.
- U.S. Environmental Protection Agency. Environmental indicators of water quality in the United States. EPA 841-R-96-002. Washington, D.C: U.S. Government Printing Office; 1996. 34 pages.
- U.S. Environmental Protection Agency. Managing manure nutrients at concentrated animal feeding operations. EPA-821-B-04-006. Washington, D.C.: U.S. Environmental Protection Agency, Office of Water (4303T); 2004. Available on-line at <http://www.epa.gov/guide/cafo/> (last verified October 12, 2006); 230 pages.
- U.S. Geological Survey. The quality of our nation's waters: nutrients and pesticides. U.S. Geological Survey Circular, vol. 1225. Denver, CO: USGS Information Services; 1999. <http://www.usgs.gov> (last verified October 12, 2006); 82 pages.
- Vadas PA, Kleinman PJA, Sharpley AN. A simple method to predict dissolved phosphorus in runoff from surface-applied manures. *J Environ Qual* 2004;33:749–56.
- Weld JL, Beegle DB, Gburek WJ, Kleinman PJA, Sharpley AN. The Pennsylvania phosphorus index: version 1. Extension publications, CAT US 180 5m3/03ps, 4591; 2003.