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Role of Rainfall Intensity and Hydrology in Nutrient Transport via Surface Runoff

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ABSTRACT

Loss of soil nutrients in runoff accelerates eutrophication of surface waters. This study evaluated P and N in surface runoff in relation to rainfall intensity and hydrology for two soils along a single hillslope. Experiments were initiated on 1- by 2-m plots at foot-slope (6%) and mid-slope (30%) positions within an alfalfa (*Medicago sativa* L.)–orchardgrass (*Dactylis glomerata* L.) field. Rain simulations (2.9 and 7.0 cm h^{-1}) were conducted under wet (spring) and dry (late-summer) conditions. Elevated, antecedent soil moisture at the foot-slope during the spring resulted in less rain required to generate runoff and greater runoff volumes, compared with runoff from the well-drained mid-slope in spring and at both landscape positions in late summer. Phosphorus in runoff was primarily in dissolved reactive form (DRP averaged 71% of total P), with DRP concentrations from the two soils corresponding with soil test P levels. Nitrogen in runoff was mainly nitrate (NO_3–N averaged 77% of total N). Site hydrology, not chemistry, was primarily responsible for variations in mass N and P losses with landscape position. Larger runoff volumes from the foot-slope produced higher losses of total P (0.08 kg ha^{-1}) and N (1.35 kg ha^{-1}) than did runoff from the mid-slope (0.05 total P kg ha^{-1}; 0.48 kg N ha^{-1}), particularly under wet, springtime conditions. Nutrient losses were significantly greater under the high intensity rainfall due to larger runoff volumes. Results affirm the critical source area concept for both N and P: both nutrient availability and hydrology in combination control nutrient loss.

Phosphorus and N are essential to crop and animal production, and are also the major nutrients controlling eutrophication of surface waters (Diaz and Rosenberg, 1995; Carpenter et al., 1998). Accelerated eutrophication has been identified as the most common water quality impairment in the USA (USEPA, 1996), with agriculture a major source of N and P in U.S. surface waters (USGS, 1999). Today, a growing strategy for reducing P losses from agricultural lands is to target "critical source areas" of P transport, where high concentrations of P are found in areas that are prone to surface runoff (Sharpley et al., 1994). Similar strategies have been proposed to address losses of N from agriculture, albeit with an emphasis on subsurface losses, as leaching of N has traditionally been the major pathway of concern to water quality (Heathwaite et al., 2000).

The hydrologic controls of critical source areas depend on interactions between climate, soils, field management, and geomorphology, all of which contribute to "variable source area hydrology" in which limited areas of a landscape contribute to watershed runoff (Gburek and Sharpley, 1998). Surface runoff may be generated by two, nonexclusive mechanisms: "infiltration excess" and "saturation excess." Infiltration excess runoff occurs when rainfall intensity exceeds the infiltration capacity of a soil. Saturation excess runoff occurs as a water table rises to the soil surface so that the soil's water storage capacity is exceeded. Saturation excess runoff includes both rain and soil water, while infiltration excess runoff is comprised predominantly of rain water (Nash et al., 2002).

Although both saturation and infiltration excess runoff generation mechanisms can occur simultaneously during a single storm, they are favored by certain climatic and geomorphic conditions. For instance, Srinivasan (2000), in a study of runoff generation from a grassed, colluvial soil in Pennsylvania, showed that saturation excess runoff was typically produced by frequent, low intensity spring time storms whereas infiltration excess runoff tended to be generated by sporadic, high-intensity summer storms. Related research by Needelman (2002) showed that saturation excess runoff from cultivated soils was promoted by the presence of subsurface features that temporarily perched water, such as a fragipan or pronounced argillic horizon. In contrast, Needelman (2002) found that runoff from soils with minimal subsurface discontinuities was exclusively by infiltration excess, if at all.

Landscape position clearly influences surface runoff generation processes. Saturation excess runoff is commonly observed in near-stream areas because of the proximity to the water table (Gburek and Sharpley, 1998). In some regions, lower positions in the landscape (colluvial foot-slopes, as defined by Conacher and Dalrymple, 1977) are associated with fragipans that have formed in colluvial soils and create seasonally perched water tables (Needelman, 2002). Conversely, upslope areas are often removed from perched and regional water tables. For instance, soils on steep, transportational mid-slopes (Conacher and Dalrymple, 1977) tend to be well-drained such that runoff from these soils is more likely to arise from intense rainstorms that exceed soil infiltration capacity.

Rainfall intensity affects surface runoff generation as well as concentrations of nutrients in runoff. Infiltration excess runoff requires sufficient rainfall intensity and duration for soil infiltration capacity to be overwhelmed, whereas saturation excess runoff may occur at extremely low rainfall intensities (Srinivasan et al., 2001). Sharpley (1985) found that the effective depth of interaction (EDI) between soil and runoff was positively related to rainfall intensity and erosion. Neal (1938) determined that rainfall intensity significantly affected runoff volume from trays packed with soils, and had an even more pro-

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Abbreviations: DRP, dissolved reactive phosphorus; EDI, effective depth of interaction; NH_4–N, ammonium-nitrogen; NO_3–N, nitrate-nitrogen; PP, particulate phosphorus; SS, suspended solids; TKN, total Kjeldahl nitrogen; TP, total phosphorus; WEP, water extractable phosphorus.

nounced effect on erosion. These conclusions were borne out by Fraser et al. (1999) monitoring surface runoff from fields planted with winter cover crops on soils that appear to have been prone to infiltration excess runoff. They observed increases in erosion and sediment-bound P concentrations in runoff with increased rainfall intensity. In contrast, Edwards and Daniel (1993) found that the intensity of rainfall was negatively related to concentrations (mg L^{-1}) of P and N in runoff, but was positively related to mass losses (kg ha^{-1}) of P and N in runoff, from grassed soils that had been broadcast with poultry litter.

To date, there have been limited efforts to link landscape and climatic variables controlling runoff generation processes (transport factors) with soil and management variables controlling nutrient availability to runoff (source factors). Even so, there has been widespread acceptance of site assessment indices, such as the P Index, that target critical source areas within agricultural landscapes for remedial action (Sharpley et al., 2003). A large and growing body of research uses small plots subjected to simulated rainfall to assess the influence of source factors on nutrients in surface runoff (e.g., Pote et al., 1999; Daverede et al., 2004). These studies provide quantitative insight into the role of individual source variables (soil P, applied manure, and mineral fertilizer P) in nutrient runoff. However, by controlling variables such as antecedent moisture and rainfall intensity, most rain simulations studies offer little to no insight into how source factors interact with transport factors. In one of the few studies examining the interaction of source and transport factors on nutrient runoff from soils, Zheng et al. (2004) conducted experiments with packed soil boxes equipped to regulate antecedent moisture and to simulate exfiltration (seeping or upwelling of groundwater). Their findings pointed to potentially profound differences in nutrient runoff by saturation excess vs. infiltration excess processes, particularly when exfiltration was introduced. Interestingly, the relative effects of a source factor, applied fertilizer, remained constant even as differences in transport potential altered nitrate–N (NO_3–N) and DRP concentrations by an order of magnitude.

Clearly, there is a need to better relate transport and source factors in the study of nutrient runoff and to better understand the relevance of common methods used to study nutrient runoff (e.g., rain simulation on small plots) with landscape and climatic processes influencing nutrient losses from agricultural fields. This study seeks to evaluate the influences of rainfall intensity and hydrology, as modified by soil properties, landscape position, and seasonal trends in antecedent soil moisture, on nutrient losses in surface runoff. Rainfall–runoff experiments were conducted using simulated rainfall to control rainfall intensity and evaluate seasonal trends in nutrient release from soils of a single catena.

MATERIALS AND METHODS

Study Area

The study was conducted within the Susquehanna River Basin, part of the Appalachian Valley and Ridge Physiographic Province of the northeastern USA, on a hillslope that is concave along the contour and concave downslope (Fig. 1). The hillslope contains Albrights soils (fine-loamy, mixed, semiactive, mesic Aquic Fragiudalfs) in the colluvial foot-slope position and Berks soils (loamy-skeletal, mixed, active, mesic Typic Dystrudepts) in the transportational, mid-slope position (Conacher and Dalrymple, 1977). Needelman (2002) observed that runoff from an Albrights soil was predominantly by saturation excess, whereas runoff from soils similar to the Berks soil in the current study, was by infiltration excess only. The hillslope is contour-cropped, divided into conventionally tilled fields that are rotated between corn (*Zea mays* L.), soybean (*Glycine max* L.), wheat (*Triticum aestivum* L.), and alfalfa. Approximately 5 Mg ha^{-1} yr^{-1} poultry (*Gallus gallus domesticus*) manure, corresponding to 205 kg TN ha^{-1} yr^{-1} and 85 kg TP ha^{-1} yr^{-1}, had been applied to the study area before 2001. The lowest field, which was planted with alfalfa and orchardgrass in 2001, served as the locus of this study. Immediately before planting the alfalfa–orchardgrass mix, 48 kg total N ha^{-1} and 7 kg TP ha^{-1} were broadcast as mineral fertilizer (17–6–30, 2% S) and then incorporated by chisel plow and disk. In 2002, when rain simulation experiments were performed, neither P nor N was applied to the established alfalfa–orchardgrass stand.

Rain Simulation Experiment

Runoff plots were established at two locations, on Albrights and Berks soils, within the alfalfa field. These locations were selected to contrast soil, slope, and hydrologic characteristics, while ensuring that management was consistent. At each location, four pairs of 1-m wide by 2-m long runoff plots were installed along a single elevation contour (Fig. 1). Slope gradients were 6 and 30% for the Albrights foot-slope and Berks mid-slope, respectively. Plots were isolated on the upper three sides by steel frames driven 5 cm into the soil and extending 5 cm above the soil. At the lower end of each plot, a gutter was inserted 5 cm into the soil with the upper edge level with the soil surface. The gutter was equipped with a canopy to exclude direct input of rainfall.

Rain simulations were conducted using a modified protocol of Sharpley et al. (2001). Portable rain simulators (Humphry et al., 2002) were equipped with either TeeJet 1/2 HH SS 50 WSQ (7.0 cm rain h^{-1}) or TeeJet 3/8 HH SS 24 WSQ (2.9 cm rain h^{-1}) nozzles (Spraying Systems Co., Wheaton, IL) approximately 305 cm above the soil surface. Rainfall from both nozzles had a coefficient of uniformity >0.83 within the 2 by 2 m area directly below the nozzle.

Two sets of rain simulations were conducted, one in May and one in September 2002, to assess runoff response under wet, spring and dry, summer site conditions. In the 14 d before the May rainfall simulations, the site received 5.3 cm natural rainfall, whereas only 2.0 cm natural rainfall fell in the 14 d before the September simulations. For every rain simulation event, the duration of the rain simulation was controlled by the amount of time necessary to generate 30 min of runoff, with a maximum event length of 150 min if no runoff occurred. In this area, a rainfall event of 150 min at 2.9 cm h^{-1} corresponds to a 10-yr rainfall return period, whereas an event of 150 min at 7.0 cm h^{-1} exceeds a 100-yr rainfall return period. Runoff volume was measured and a sample of runoff water collected on a 5 min interval for the full 30 min of the runoff event. In addition, a composite sample was collected at the end of the event.

Two weeks were required to complete a single set of rain simulations. Care was taken to minimize possible interactions between time, plot location, and rainfall intensity. For instance, plots were covered with waterproof tarpaulins between May rainfall simulation events to prevent 4.0 cm of natural rainfall from impacting the plots and were covered once during the

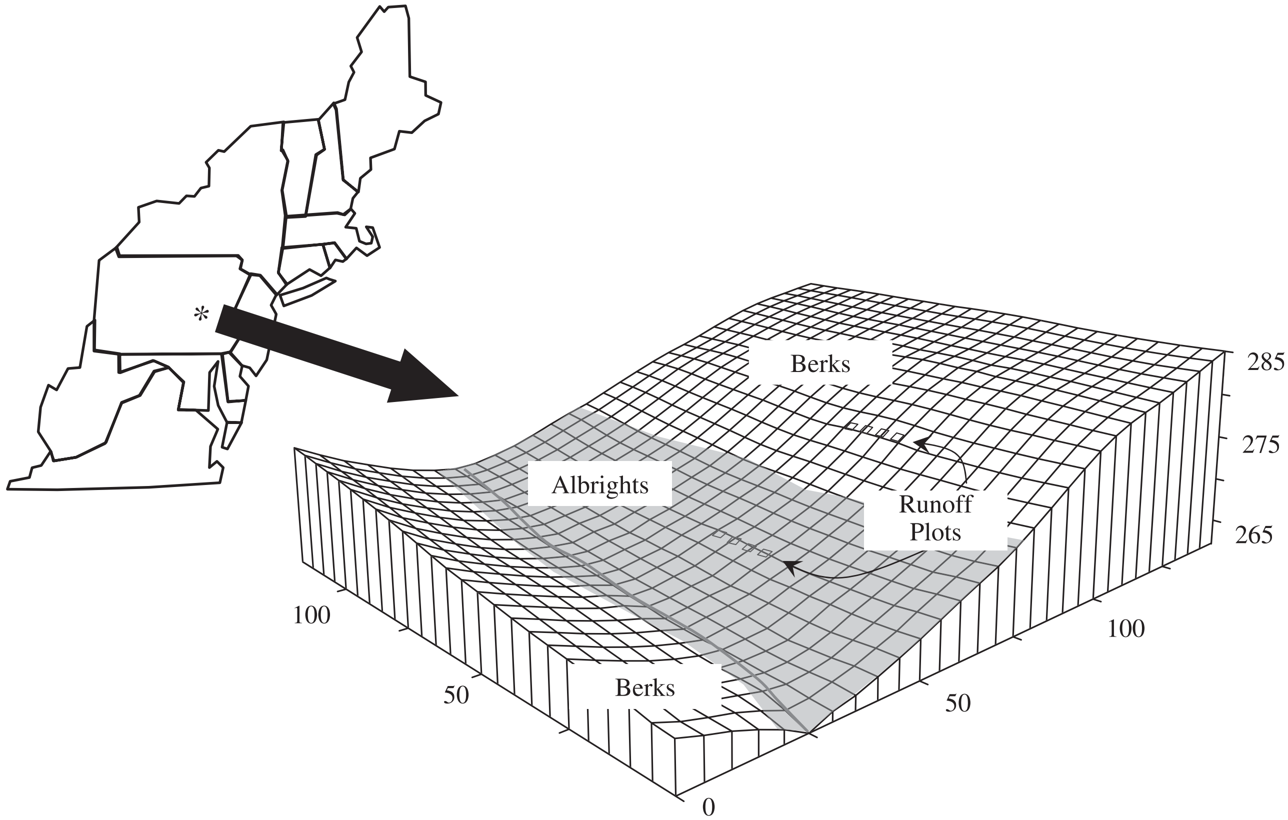

Fig. 1. Location of study site in eastern USA and 1-m contour map of hillslope showing location of runoff plots in Albrights (shaded) and Berks (unshaded) soils. Each set of plots includes four abutting pairs of 1 by 2 m runoff plots (eight plots per set).

September simulations to intercept a 2.2-cm natural downburst. For each rain simulation period (May vs. September), simulations were conducted on all plots under both rainfall intensities on two consecutive days. In the first week, half of the plots were subjected to rainfall at 7.0 cm h^{-1} while the other half received rainfall at 2.9 cm h^{-1}. After allowing plots to drain for 7 d to return to roughly the moisture conditions before the simulation period, a second round of simulations was conducted, with plots that had previously received one rainfall intensity now receiving the other rainfall intensity. For instance, if a plot had received 7.0 cm h^{-1} rainfall on the first week, it received 2.9 cm h^{-1} on the second week, and vice versa.

Approximately 3 d before each set of rainfall simulations, vegetation within the plots was mowed to approximately 4 to 5 cm height to ensure a uniform cover, simulating post-haying conditions. Residue was raked from the plots to minimize the contribution of nutrients in the organic residue to P and N runoff. Volumetric moisture (θ) of the surface soil (upper 4 cm) was measured with a capacitance sensor (ThetaProbe, Dynamax, Houston, TX) at six predetermined locations within each plot (Fig. 2) immediately before each rain simulation, at the start of runoff, and immediately after rainfall was terminated. Before each set of runoff experiments, 10 surface soil samples (2-cm diam., 5 cm deep) were collected from an area adjacent to each plot (Fig. 2).

Laboratory Analysis

Soil samples were combined and thoroughly mixed to obtain a single, composite sample for each plot. Samples were then air dried and sieved (2 mm). Total soil N and total soil C were determined by elemental analyzer (EA 1110, CE Elantech, Lakewood, NJ). Inorganic soil N (NH_4^+ and NO_2^- + NO_3^-) was extracted with 2 *M* KCl (solution/soil = 5:1, 1 h extraction) and determined colorimetrically (Mulvaney, 1996). Deionized water extractable P (WEP,solution/soil = 10:1, 1 h extraction) and Mehlich-3 P (Mehlich, 1984) were determined on air-dried samples. Filtrate P was determined colorimetrically, following a modified method of Murphy and Riley (1962), with spectrophotometer λ = 712 nm. Particle size analysis was conducted by the hydrometer method (Day, 1965). Soil pH was determined by mixing air dry soil with distilled water (solution/soil = 1:1).

Runoff samples were stored at 4°C. Total Kjeldahl N (TKN) and TP were measured on unfiltered runoff water by modified semimicro-Kjeldahl procedure following Bremner (1996). Inorganic N (NH_4^+ and NO_2^- + NO_3^-) and DRP were determined colorimetrically on filtered runoff samples (0.45 μm). Total N in runoff was calculated by summing the NO_3–N and TKN fractions. Suspended solids (SS) were determined by gravimetric analysis, after evaporating 200 mL of runoff water at 80°C.

Statistical Analysis

Data were evaluated using the Kolmogorov D test statistic to determine whether they were lognormally or normally distributed. Dissolved reactive P and TP in runoff were transformed logarithmically (natural) to comply with the assumption of Gaussian distribution. These P data were back-transformed for discussion in text following the method of Schmidt et al. (2002). Treatment effects were evaluated by paired *t* test. Relationships between individual variables were analyzed by least squares regression. Treatment differences discussed in the text are significant at $\alpha \leq 0.05$. Analyses were conducted with SAS, Version 8 (SAS Institute, 1999).

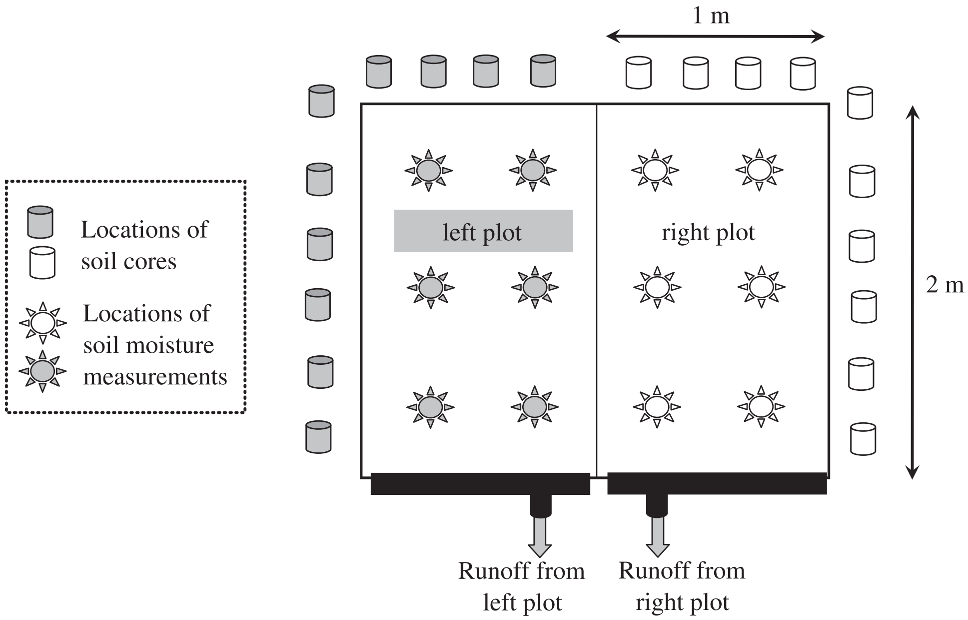

Fig. 2. Layout of paired runoff plots illustrating dimensions of abutting plots, separate runoff collection drains for each plot, location of soil moisture measurements within each plot, and location of 5-cm cores used to evaluate surface soil properties of each plot.

RESULTS AND DISCUSSION

Soils

The Albrights and Berks soils differed in several properties expected to influence hydrology and nutrient losses in runoff. A key difference was that the Albrights soil possessed a fragipan, beginning at approximately 60-cm depth, whereas the Berks soil did not. Lateral flow of water through these soils and perching of water above the fragipan produces seasonal differences in surface moisture content, with the somewhat poorly drained Albrights soil becoming substantially wetter than the well-drained Berks soil in the spring and fall. Differences in slope gradients (6% for Albrights vs. 30% for Berks) also contribute to observed differences in drainage and runoff generation. In a runoff monitoring study conducted on a nearby hillslope, Needelman (2002) found that a somewhat poorly drained, Albrights soil with fragipans was more prone to saturation excess runoff than were well-drained soils lacking fragipans. Furthermore, runoff occurred much more frequently from the somewhat poorly drained Albrights soil than from the well-drained soils. An additional property potentially affecting runoff generation was particle size distribution. The colluvial Albrights soil was finer textured than the Berks soil (Table 1). Estimated permeability of horizons within an Albrights pedon ranged from 0.5 to 5.0 cm h^{-1}, whereas the permeability throughout a typical Berks pedon ranged from 1.5 to 15 cm h^{-1} (Eckenrode, 1985).

Despite their occurrence in the same field, Berks and Albrights soils possessed significantly different Mehlich-3 P and WEP concentrations (Table 1), reflecting varying histories of manure application. Mehlich-3 P values of the Berks plots exceeded the environmental threshold of 202 mg kg^{-1} identified by Sharpley et al. (2001). Due to the seasonally perched water table, the Albrights soil is often inaccessible to manure spreading equipment in the spring when manure is typically applied. The well-drained Berks soil, however, is substantially drier at that time (Table 2), allowing regular access of equipment. In addition, until 2 yr before the study, the field in which both soils were located had been treated as two management units, largely due to the differential drainage of the Albrights and Berks soils. Thus, the Albrights soil had historically received less manure than had the Berks soil.

Table 1. Mean properties of Albrights and Berks surface soil at time of May and September rain simulations.

Soil	N	Mehlich-3 P	WEP (soil/soln, 1:10)	KCl extractable NO_2^- + NO_3^-	KCl extractable NH_4–N	Total N	Total C	pH	Particle-size distribution Sand	Silt	Clay
		mg kg^{-1}				%			%		
May											
Albrights	16	70 (9)†	6.0 (1.2)	7.1 (0.4)	6.2 (1.8)	0.19 (0.01)	2.3 (0.2)	6.6 (0.1)	25‡	48	27
Berks	16	256 (39)	22.8 (2.0)	13.3 (1.8)	13.5 (4.2)	0.23 (0.03)	2.5 (0.5)	7.0 (0.2)	43	36	21
September											
Albrights	16	69 (12)	8.6 (0.9)	10.8 (3.7)	16.3 (2.7)	0.26 (0.03)	3.0 (0.4)	6.4 (0.4)	–	–	–
Berks	16	216 (15)	20.9 (1.7)	10.3 (3.0)	22.0 (2.5)	0.23 (0.02)	2.3 (0.3)	6.4 (0.4)	–	–	–

† Standard deviations are presented in parentheses.
‡ Particle-size analysis conducted on a single sample of each soil composited from May and September samples.

Table 2. Mean hydrologic properties of runoff plots established in Albrights and Berks soil during May and September rain simulations.

Soil	Sequence of event	Rainfall intensity	Fraction of plots† generating runoff	Rainfall before runoff	Runoff	Volumetric moisture content, θ: Before rain	Start of runoff	End of event
	d	cm h^{-1}	%	cm	cm	m^3 m^{-3}		
				May				
Albrights	1	2.9	8/8	0.51 (0.11)‡	1.35 (0.48)	0.40 (0.02)§	0.44 (0.01)	0.44 (0.01)
Albrights	1	7.0	8/8	0.67 (0.36)	3.10 (0.91)	0.38 (0.05)	0.45 (0.01)	0.42 (0.05)
Berks	1	2.9	0/8	no runoff	no runoff	0.23 (0.03)	no runoff	0.36 (0.02)
Berks	1	7.0	3/8	4.28 (5.99)	0.21 (0.1)	0.23 (0.01)	0.39 (0.03)	0.36 (0.03)
Albrights	2	2.9	8/8	0.65 (0.16)	1.18 (0.47)	0.40 (0.01)	0.44 (0.02)	0.44 (0.01)
Albrights	2	7.0	8/8	0.69 (0.31)	3.03 (0.93)	0.40 (0.02)	0.44 (0.02)	0.43 (0.02)
Berks	2	2.9	6/8	2.39 (0.33)	0.17 (0.09)	0.24 (0.03)	0.38 (0.02)	0.38 (0.02)
Berks	2	7.0	8/8	1.68 (0.92)	0.64 (0.43)	0.26 (0.02)	0.41 (0.02)	0.41 (0.02)
				September				
Albrights	1	2.9	3/8	2.98 (0.32)	0.44 (0.34)	0.20 (0.07)	0.39 (0.01)	0.42 (0.03)
Albrights	1	7.0	8/8	5.67 (3.12)	1.07 (0.55)	0.22 (0.05)	0.43 (0.03)	0.43 (0.04)
Berks	1	2.9	0/8	no runoff	no runoff	0.16 (0.05)	no runoff	0.39 (0.03)
Berks	1	7.0	6/8	5.15 (3.99)	0.11 (0.05)	0.16 (0.07)	0.37 (0.08)	0.38 (0.03)
Albrights	2	2.9	5/8	2.13 (0.69)	0.53 (0.45)	0.34 (0.01)	0.43 (0.04)	0.44 (0.04)
Albrights	2	7.0	8/8	1.95 (3.99)	1.98 (0.05)	0.32 (0.07)	0.47 (0.08)	0.47 (0.03)
Berks	2	2.9	1/8	3.29 (0)	0.14 (0)	0.27 (0.03)	0.459 (0)	0.39 (0.05)
Berks	2	7.0	8/8	2.24 (1.35)	0.44 (0.46)	0.26 (0.04)	0.44 (0.03)	0.40 (0.01)

† Total of eight runoff plots per soil.
‡ Standard deviations of rainfall and runoff depth are presented in parentheses for plots generating runoff.
§ Soil moisture data are provided for all plots, even those that did not generate runoff.

Significantly greater concentrations of soil WEP were observed in September than in May for the Albrights soil, but not for the Berks soil (Table 1). Similarly, total soil C increased significantly from May to September sampling dates for the Albrights soil, but not for the Berks soil.

Inorganic N extracted by KCl from the upper 5-cm of soil accounted for <2% of total soil N (Table 1). Although total N did not differ significantly between soils, KCl-extractable inorganic N pools (NO_2^- + NO_3^- and NH_4–N) were greater in the Berks soil than in the Albrights soil. The seasonally saturated surface horizon of the Albrights soil is more likely to undergo prolonged periods of denitrification than the surface horizon of the Berks soil (Clement et al., 2002). In this region, denitrification can serve as a significant loss pathway for NO_3–N (Schnabel et al., 1996; Flite et al., 2001). Denitrification may also have contributed to the lower NO_2–N + NO_3–N extracted with KCl from the Albrights soils in May than in September, due to an elevated water table in May (Table 1). In addition, N_2 fixation associated with the leguminous alfalfa crop, along with mineralization of organic N during the growing season, likely augmented the KCl-extractable NH_4–N pools of both soils from May to September (Carpenter-Boggs et al., 2000).

Runoff Hydrology

Runoff hydrology was greatly influenced by interactions between seasonal soil moisture conditions (May vs. September), day of simulation (Day 1 vs. Day 2), soil/landscape location (Albrights/colluvial foot-slope vs. Berks/transportational mid-slope) and rainfall intensity (2.9 vs. 7.0 cm h^{-1}). Results suggest that runoff from the Albrights plots was generated by saturation excess in May, whereas runoff from the Albrights plots in September, and runoff from the Berks plots in both May and September, resulted from infiltration excess (Table 2 and Fig. 3). Volumetric soil moisture (θ) before the initiation of rain simulation events was greatest in May, particularly in the Albrights plots which were at or near surface saturation for the entire duration of both May events. Notably, shallow pits excavated adjacent to the Albrights plots revealed a water table within 3 cm of the soil surface in May, and several of the Albrights plots continued to yield runoff for at least 30 min after simulated rainfall ceased following May simulations. With the exception of the Albrights plots in May, θ was always significantly higher at the start of the second day of simulations than at the start of the first day of simulations. Regardless of moisture conditions at the start of the rainfall event, θ at time of runoff initiation did not differ significantly from θ at the end of the runoff event. This is because θ of the surface soil was generally at or near saturation when runoff was generated (approximately 0.45 m^3 m^{-3} for Albrights and 0.40 m^3 m^{-3} for Berks). No significant differences in any of the θ measurements (before rainfall, start of runoff, end of runoff) were observed on the basis of rainfall intensity (Table 2).

Runoff generation, as reflected by the number of plots in a particular soil/landscape location that actually produced runoff ("Fraction of plots" column, Table 2), was clearly controlled by soil moisture at the start of a rainfall event, inherent soil infiltration properties, and rainfall intensity. Due to initial moisture conditions in May, all Albrights plots generated runoff on both days under both rainfall intensities (Fig. 3). Despite relatively wet, initial conditions in the Berks soil in May (average θ = 0.23), very few of these steeply sloped plots produced runoff on the first day of simulations, and those that did produce runoff all received 7.0 cm h^{-1} rainfall. By the second day of the May rainfall simulations, all Berks plots yielded runoff under the 7.0 cm h^{-1} rain, and 75% of the plots receiving 2.9 cm h^{-1} rain produced runoff

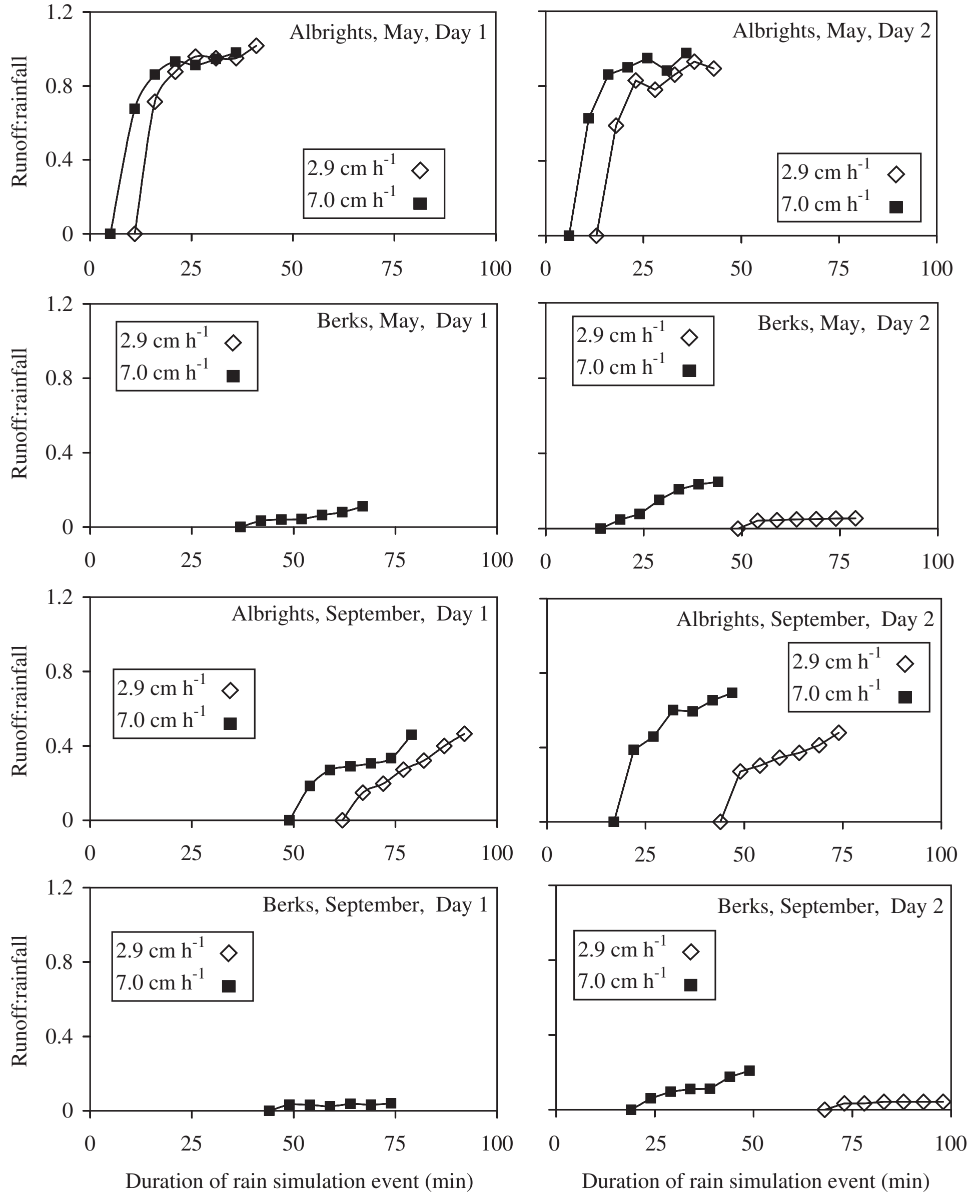

Fig. 3. Conversion of rainfall into runoff, represented by runoff to rainfall ratio (runoff/rainfall) for 5 min intervals over the duration of rain simulation event.

as well. In September, when initial soil moisture of the Berks plots was comparatively low (average $\theta = 0.16$), trends in runoff generation tracked rainfall intensity on both days, with roughly the same proportion of 7.0 cm h^{-1} plots producing runoff as in May, and considerably fewer 2.9 cm h^{-1} plots yielding runoff than in May (Table 2). Indeed, a 150-min event of 2.9 cm h^{-1} rainfall in September was insufficient to generate infiltration excess runoff from nearly all Albrights and Berks plots on Day 1 and from the well-drained Berks plots on Day 2.

Trends in rainfall infiltration before runoff were weakly related to soil moisture content at the start of the event by logarithmic model [Infiltration $= -4.1 \ln(\theta) - 2.9$, $r^2 = 0.33$)] with the relationship improving substantially when observations from a pair of abutting Berks plots sampled on Day 1 of rainfall simulations in September were excluded from the analysis [Infiltration $= -6.5 \ln(\theta) - 5.6$, $r^2 = 0.60$]. Runoff from the two Berks plots appeared to have been the result of dry, hydrophobic surface soil conditions. Runoff from these plots began 3 to 4 min after rainfall initiation in September, whereas it did not start until 40 to 45 min after rainfall initiation in May. Observations from these two plots in September point to an additional mechanism for runoff from dry soils, accentuated in the Berks plots due to their steep gradient. Hydrophobicity has been reported for other mineral soils in the region (Steenhuis et al., 2001). While hydrophobic runoff cannot be discounted, the nature of this runoff, confined to one area of the Berks soil under one set of conditions, suggests that runoff produced by hydrophobicity would likely infiltrate once it encountered nonhydrophobic soils.

Across all plots, average infiltration of rainfall before runoff was significantly greater in September, when soils were initially drier, than in May (Table 2). Similarly, for all but the Albrights plots in May, infiltration before runoff was lower at the start of Day 2 of simulation than on Day 1 due to elevated antecedent soil moisture conditions. In May, infiltration in the Albrights plots did not differ significantly between the Days 1 and 2 of simulation because of high levels of antecedent moisture. Rainfall intensity modified the effects of soil moisture and inherent infiltration properties (e.g., saturated hydraulic conductivity) on rainfall infiltration. Infiltration of rainfall before runoff was significantly greater under the 7.0 cm h^{-1} rainfall intensity than under the 2.9 cm h^{-1} rainfall intensity (Table 2).

Figure 3 illustrates the conversion of rainfall into runoff for those plots that produced runoff. Average runoff to rainfall ratio (runoff/rainfall) for each 5 min increment is presented for each group of events. Saturation–excess runoff from Albrights plots was characterized by nearly immediate runoff production in which runoff/rainfall rapidly approached 1.0. In contrast, infiltration–excess runoff from Albrights plots in September required more time for runoff to occur, and resulted in significantly less runoff than in May, with runoff/rainfall never obtaining a plateau over the duration of the September events. The delay in runoff generation and the lower runoff/rainfall indicate are clear indicators of infiltration excess runoff. Berks plots always produced significantly less runoff than did the Albrights plots, with less rainfall converted to runoff. For all events, the effect of lower rainfall intensity was to delay initiation of runoff relative to the 7.0 cm h^{-1} rain, and generate significantly less flow, confirming the findings of Neal (1938). However, once runoff was initiated, the fraction of rainfall converted to runoff (runoff/rainfall) was similar for both intensities (Fig. 3).

Runoff Water Quality

Soil (Albrights vs. Berks)

Significant differences in runoff nutrient content were observed between the two soils. Runoff DRP concentrations (Table 3) were significantly greater from the Berks soil (average = 1.04 mg L^{-1}) than from the Albrights soil (average = 0.34 mg L^{-1}). Differences in DRP (Berks was 3.1 times greater than Albrights) corresponded with relative differences in both Mehlich-3 P (Berks was 3.4 times greater than Albrights) and WEP (Berks was 3.0 times greater than Albrights). Despite greater DRP concentrations in runoff from the Berks soil, runoff DRP losses (Table 4) were significantly greater from the Albrights soil (average = 0.05 kg ha^{-1}) than from the Berks soils (average = 0.04 kg ha^{-1}), reflecting the larger amounts of runoff generated from the Albrights soil. Consequently, hydrologic differences between the two soils counteracted the effect of soil P release characteristics on mass P loss in runoff.

Factors controlling runoff DRP likely influenced differences in runoff TP between soils, as DRP was the dominant form of TP in runoff from both Albrights (DRP averaged 0.68% of TP) and Berks (DRP averaged 0.73% of TP) soils. In addition, greater concentrations of SS in runoff from the Berks soil (Table 3), representing higher erosion due to steeper slope gradient, likely increased the contribution of particulate P to TP in Berks runoff. Indeed, the difference between TP and DRP concentrations, which should primarily account for sediment-bound P with a minor contribution of dissolved organic P, was significantly greater in Berks runoff (averaging 0.39 mg L^{-1}) than in Albrights runoff (averaging 0.16 mg L^{-1}). As with DRP losses (Table 4), the greater volumes of runoff from the Albrights counteracted the greater TP concentrations in runoff from Berks, so that significantly greater total P losses were observed in runoff from the Albrights soil than from the Berks soil.

Both DRP and TP results clearly affirm the central precept of P site assessment indices; that is, it is the coincidence of high P source potential (availability of P to runoff) and high P transport potential (represented here by runoff depth) that controls P loss from soil, not simply source potential or transport potential alone (Lemunyon and Gilbert, 1993). Here, the greatest losses (kg ha^{-1}) of P in runoff came from the soil with the lowest soil P content (source potential), and the highest runoff (transport potential).

Table 3. Mean nutrient concentration of runoff plots established in Albrights and Berks soil during May and September rain simulations.

Treatment	DRP†	TP	NO_3–N	TKN	SS
	mg L^{-1}				g L^{-1}
Soil					
Albrights	0.34*	0.50*	6.9*	1.4*	0.22*
Berks	1.04	1.43	8.8	4.9	0.43
Month					
May	0.30*	0.51*	9.3*	1.6*	0.29
September	0.90	1.27	5.6	3.9	0.30
Day					
1	0.64	1.10	6.6*	3.7	0.28
2	0.51	0.67	8.3	2.0	0.31
Intensity					
2.9 cm h^{-1}	0.35*	0.55*	8.9*	1.7	0.26
7.0 cm h^{-1}	0.70	0.99	6.8	3.2	0.32

* Indicates significant difference ($P < 0.05$) within a treatment category.
† DRP = dissolved reactive phosphorus, TP = total phosphorus, NO_3–N = nitrate-nitrogen, TKN = total Kheldahl nitrogen, SS = suspended solids.

Table 4. Mean nutrient loss of plots that produced runoff from Albrights and Berks soils during May and September rain simulations.

Treatment	DRP†	TP	NO_3–N	TKN	SS
	kg ha^{-1}				
Soil					
Albrights	0.05*	0.08*	1.14*	0.21*	39.99*
Berks	0.04	0.05	0.36	0.12	12.36
Month					
May	0.04	0.06	1.18*	0.19	31.59
September	0.07	0.09	0.44	0.16	27.55
Day					
1	0.003	0.08	0.91	0.21	21.68
2	0.05	0.07	0.79	0.16	35.57
Intensity					
2.9 cm h^{-1}	0.03*	0.04*	0.71	0.12*	13.24*
7.0 cm h^{-1}	0.06	0.08	0.92	0.21	38.56

* Indicates significant difference ($P < 0.05$) within a treatment category.
† DRP = dissolved reactive phosphorus, TP = total phosphorus, NO_3–N = nitrate-nitrogen, TKN = total Kheldahl nitrogen, SS = suspended solids.

Soil-related trends in N runoff were similar to those of P, with significantly greater concentrations of NO_3–N and total N (NO_3–N + TKN) in runoff from the Berks soil (average NO_3–N = 8.8 mg L^{-1}; average total N = 13.6 mg L^{-1}) than from the Albrights soil (average NO_3–N = 6.9 mg L^{-1}; average total N = 8.3 mg L^{-1}). The higher NO_3–N concentrations in runoff from the Berks soil correspond with greater KCl-extractable NO_3–N in that soil (Table 1). Because NO_3 accounted for 83 and 65% of total N in runoff from Albrights and Berks soils, respectively, trends in total N content of runoff tracked those of NO_3–N. In addition, greater SS concentrations in runoff (Table 3) and greater KCl-extractable NH_4–N associated with the Berks soil (Table 1) also elevated total N in runoff. Losses (kg ha^{-1}) of NO_3–N and total N in runoff (Table 4) were consistent with P losses and were significantly greater from the Albrights soil (average NO_3–N = 1.14 kg ha^{-1}; average total N = 1.35 kg ha^{-1}) than from the Berks soil (average NO_3–N = 0.36 kg ha^{-1}; average total N = 0.48 kg ha^{-1}). Here then, from the standpoint of managing nutrient losses in surface runoff, conclusions regarding N and P are consistent: avoid application of nutrients to the Albrights soil where saturation–excess runoff produces greater P and N losses.

Seasonal Trends (May vs. September)

Differences in nutrient runoff between Albrights and Berks soils were modified significantly by seasonal timing of runoff events (late spring vs. late summer). Whereas soil-related trends in N transport were consistent with trends in P transport, seasonal trends in P and N transport were discordant. Specifically, DRP and TP concentrations in runoff were significantly greater in September runoff than in May, as were associated variances, whereas NO_3–N and total N concentrations were greater in May than in September runoff (Table 3).

Differences in the WEP of the Albrights soil between May and September suggest one possible explanation for the observed seasonal differences in runoff P concentrations. Specifically, greater availability of water-soluble P derived from plant biomass later in the growing season may have contributed to the increased DRP concentrations in runoff in September. Indeed, Gburek and Broyan (1974), comparing sequential, laboratory leachings of orchardgrass with seasonal trends in water quality for the larger watershed in which the present study was located, concluded that contributions of soluble P from vegetation could account for elevated summertime concentrations of P in runoff. Elsewhere, Sharpley (1981) found that an increase in the age of cotton (*Gossypium hirsutum* L.), sorghum (*Sorghum sudanense* Stapf.), and soybean from 42 to 82 d resulted in substantially greater contributions of soluble P from plant leaves to runoff, accounting for increases in runoff P by 20 to 60%. Although significant increases in WEP were observed from May to September in the Albrights soil, consistent with this hypothesis, none was observed in the Berks soil (Table 1). Notably, the hypothesis of depleted DRP concentrations due to increased infiltration of rain water and translocation of dissolved P out of the EDI was not supported by trends observed in this study. Thus, this study contradicts the findings of Zheng et al. (2004), derived from a highly controlled packed soil box study in which seasonal variations in nutrient sources did not exist.

In contrast with the greater concentrations of DRP and TP in September runoff, no significant differences in runoff DRP and TP losses (kg ha^{-1}) were observed between May and September events (Table 4), suggesting that hydrologic differences overwhelmed the seasonal effects of differential soil/vegetation P release. For the Albrights soil, large variability in runoff amounts related to different runoff generation mechanisms (saturation excess in May vs. infiltration excess in September) masked seasonal differences in runoff P concentrations. Although runoff generation mechanisms did not differ between May and September in the Berks soil, the lack of a significant difference in runoff P losses may be attributed to greater depth of runoff in May than in September, and fewer Berks plots generating runoff, hence lower degrees of freedom (Table 2).

Mean NO_3–N concentrations in runoff were significantly lower in September (5.6 mg L^{-1}) than in May (9.3 mg L^{-1}), as were losses (September = 0.44 kg ha^{-1}, May = 1.18 kg ha^{-1}). Trends in total N in runoff were consistent with those of NO_3–N, as NO_3–N accounted for the majority of total N in runoff (Tables 3 and 4). Significantly greater runoff volumes in May than in September undoubtedly contributed to the greater losses of N in May (Table 4). The lack of correspondence in KCl-extractable inorganic N (Table 1) and runoff N concentrations (Table 3) over time may reflect seasonal differences in hydrology. Greater infiltration of rainfall in September than in May could have translocated runoff-available N fractions from the EDI into the subsoil. Such a hypothesis was also offered by Pote et al. (2001), who reported negative correlations between TKN and NH_4–N concentrations in runoff and rainfall infiltration rate in grassed soils broadcast with swine (*Sus scrofa*) slurry. Elsewhere, Zheng et al. (2004) observed that NO_3–N concentrations in surface runoff from packed soil boxes exposed to simulated rainfall was greater under saturated conditions (average = 1.8 mg L^{-1}) than under freely draining conditions (average = 0.04 mg L^{-1}). In that study, exfiltration (seep) processes were also simulated. Runoff produced by exfiltration and rainfall contained even higher average NO_3–N concentrations (average = 8.2 mg L^{-1}) than did runoff produced from the saturated soils, but not as high as runoff produced from exfiltration alone (average = 75.4 mg L^{-1}). Zheng et al. (2004) attributed the different NO_3–N concentrations to the direction of flow through the soils, with exfiltrating water bringing with it NO_3–N from deeper within the soil profile. Again, similarities in the relative magnitude of runoff losses of N and P between May and September, despite seasonally different trends in soil N and P fractions, point to the importance of hydrologic factors in controlling mass losses of nutrients in runoff.

Daily Trends (Day 1 vs. Day 2)

Daily trends in nutrient runoff, both between and within individual events, varied by season (May vs. September) as well as by event. As illustrated in Fig. 4, two distinct sets of trends were observed in P dynamics during the 30-min runoff periods. In May, trends in runoff DRP and TP concentrations over the runoff periods were relatively static between Day 1 and Day 2. In September, pronounced declines in runoff DRP and TP concentration were observed from the first (5 min) runoff sample to the final (30 min) sample, with an average decline of 57% for DRP and 63% for TP on Day 1 and 36% for both DRP and TP on Day 2. Significantly greater concentrations of P were observed on Day 1 than on Day 2 in September.

In May, the relatively flat chemographs and absence of significant differences in P concentrations between Day 1 and Day 2 coincide with the greater saturation of the plots, which produced relatively uniform equilibrium flow conditions earlier in the rainfall period in comparison with the September events. The slightly higher concentrations of P in runoff on Day 2 of the May simulations (Fig. 4b and 4d) compared with Day 1 (Fig. 4a and 4c) reflect the contribution of additional plots on Day 2, particularly the well-drained Berks plots with high soil P, that did not run off on Day 1 (Table 1). Small declines in P concentrations were observed over the Day 1 runoff period in May (31 and 48% for DRP and TP, respectively), and even smaller declines (5 and 27% for DRP and TP, respectively) were evident over the Day 2 runoff period in May. Declines in P concentrations over a runoff period have been reported elsewhere, with concentrations ultimately trending to equilibrium levels (e.g., White et al., 2003). It is noteworthy that the relative magnitude of the decline in runoff P concentrations from beginning to end of the runoff period corresponds with mean infiltration for these events with September Day 1 (average infiltration = 6.9 cm) > September Day 2 (average infiltration = 4.1 cm) > May Day 1 (average infiltration = 3.1 cm) > May Day 2 (average infiltration = 2.6 cm). Pote et al. (2001) observed that greater rainfall infiltration rates were associated with lower mean event concentrations of DRP in soils receiving surface application of swine manure, positing that translocation of manure P with infiltrating rainwater decreased P available to runoff water. Zheng et al. (2004) observed that trends in runoff DRP concentrations during the runoff period were related to hydraulic gradient of packed soil boxes, with free-draining soils exhibiting declines in DRP over the course of an event and saturated soils producing relatively static concentrations of DRP in runoff.

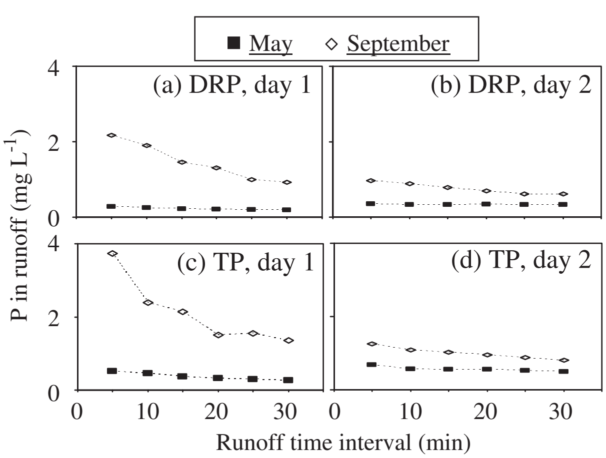

Fig. 4. Mean concentrations of dissolved reactive phosphorus (DRP) and total phosphorus (TP) in runoff during Day 1 and 2 runoff periods for May and September rainfall events.

Although soils of the current study had not recently received manure or other P fertilizer, it is possible that rainfall infiltration and translocation of soluble P out of the EDI contributed to the relative declines observed in DRP concentrations within individual events. Such a hypothesis presumes the depletion or exhaustion of a soluble P source at the soil surface by leachate and runoff (e.g., Vadas et al., 2004a). Indeed, the large declines in P concentrations observed over the course of the September runoff periods coincide with elevated concentrations of DRP in runoff that are possibly derived from plant sources. Trends in SS concentrations (Fig. 5), particularly for the September Day 1 runoff period when the greatest declines in DRP and TP concentrations were observed, suggest that physical processes were also of importance. For instance, the preferential erosion of low density organic materials, such as senescent plant litter, may have contributed substantially to both DRP and TP concentrations.

Depletion of available P sources is not the only explanation for the declining P concentrations observed during the individual runoff periods. A competing, nonexclusive hypothesis is that of dilution. Recall that flow increased over the runoff period with each 5 min sampling increment (Fig. 3). Declining P concentrations coincide with increasing flow during the runoff period, consistent with increasing dilution. Indeed, dilution has been widely cited as a cause of declining solute concentrations in runoff during flow events (e.g., Vadas et al., 2004b). It is likely that dilution and source depletion mechanisms operate simultaneously.

Unlike P, N concentrations in runoff (NO_3–N and TKN) were relatively static over time (e.g., Fig. 6). Concentrations of N in runoff did not decline over the

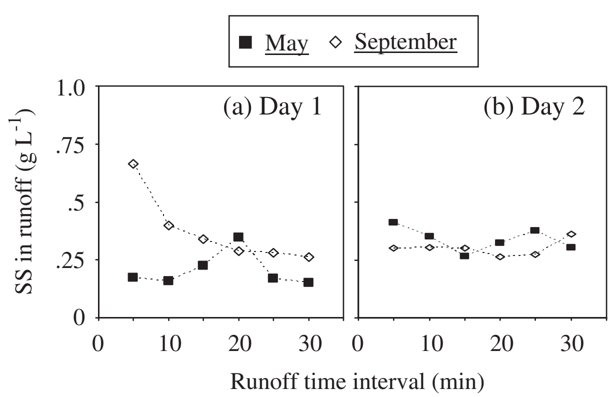

Fig. 5. Mean concentrations of SS in runoff over time on Days 1 and 2 of May and September rainfall simulations.

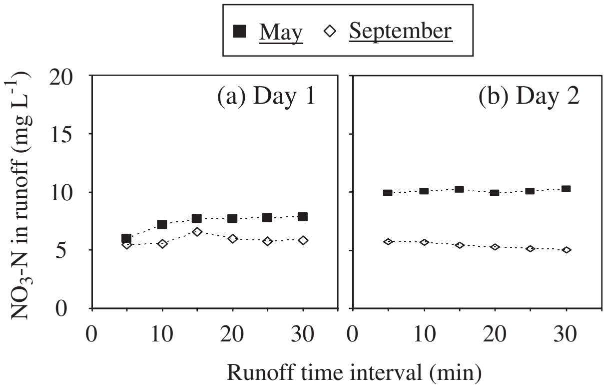

Fig. 6. Mean concentrations of NO_3–N in runoff over time on Days 1 and 2 of May and September rainfall simulations.

course of the runoff period, nor did they differ significantly between Day 1 or Day 2 events in May or September (Table 3). In fact, average NO_3–N concentrations increased slightly over the Day 1 May runoff period and from Day 1 to Day 2 in May, although the differences were not statistically significant. Elsewhere, Zheng et al. (2004) measured increasing NO_3–N concentrations in runoff over the course of a runoff period from packed soil boxes with exfiltrating flow. Their results highlight the importance of the zone of interaction (i.e., the source of runoff nutrients in soil) on N and P concentrations in runoff and downplay the role of dilution on nutrient concentrations in runoff. Specifically, runoff volumes and runoff nutrient concentrations produced by a combination of rainfall and exfiltration processes were significantly greater than those produced by rainfall on freely draining soils.

Rainfall Intensity

Rainfall intensity (7.0 vs. 2.9 cm h^{-1}) significantly affected DRP, TP, and NO_3–N concentrations in runoff, with rainfall intensity positively related to DRP and TP concentrations and negatively related to NO_3–N concentrations (Table 3). Previous studies examining nutrient concentrations in runoff related to rainfall intensity have pointed to erosion and dilution as processes controlling N and P concentrations in runoff (Fraser et al., 1999; Edwards and Daniel, 1993). Phosphorus trends in the current study are consistent with the findings of Sharpley (1985) who found rainfall intensity to be positively related to EDI, a key determinant of soil P desorption. More often, however, studies have found rainfall intensity to be negatively related to P concentration in runoff. For instance, Edwards and Daniel (1993) simulated two rainfall intensities (5 and 10 cm h^{-1}) following application of poultry litter to grassed plots, citing increased runoff volumes and related dilution of solutes as the cause of lower concentrations of P concentrations in runoff at higher rainfall intensity. In that study, N concentrations also declined with increasing rainfall intensity, pointing to dilution as a key control of N concentration in runoff. Thus, the trends observed in the current study related to rainfall intensity are consistent with the NO_3–N dilution and greater P desorption owing to increased EDI.

Rainfall intensity was positively related with mass losses of nutrients and sediment in runoff, with the exception of NO_3–N (Table 4). As described above, rainfall intensity clearly affected plot hydrology, particularly with regard to runoff generation under infiltration excess conditions and the volume of runoff produced under both infiltration excess and saturation excess conditions (Table 2). Differences in runoff nutrient losses related to rainfall intensity reflect these hydrologic differences as well as the different concentrations reported in Table 3. In the case of NO_3–N, the negative relationship between NO_3–N concentration in runoff and rainfall intensity (Table 3) and the positive relationship between runoff volume and rainfall intensity (Table 2) appear to counteract each other, resulting in no significant difference in NO_3–N loss between rainfall intensities.

CONCLUSIONS

Understanding the interaction of source and transport factors in nutrient runoff is key to the improved management of water quality. Many surface runoff studies evaluate source factors while controlling transport factors. By conducting rain simulation experiments under a variety of hydrologic conditions at two distinctly different landscape positions, this study provides insight into the relevance of traditional rain simulation experiments in describing processes controlling nutrient transport.

In the current study, runoff generation mechanisms differed between soils/landscape position as well as over time. Most rain simulation studies simulate infiltration excess runoff, which is not the dominant mechanism of runoff for poorly drained soils. Significant differences in infiltration, timing of runoff, and runoff volume were observed between saturation excess and infiltration excess events on the Albrights soil, which has a seasonally perched water table. As a result, mass losses of nutrients in runoff were significantly greater under saturation excess runoff than under infiltration excess runoff. These findings indicate the limitation of extrapolating traditional rain simulation findings to predict the export of nutrients from agricultural fields.

Despite profound differences in runoff generation processes, concentrations of DRP in runoff related well to concentrations of P in soil, and were in close agreement with runoff DRP concentrations predicted by Vadas et al. (2005) based on a single extraction coefficient derived from a large variety of traditional (infiltration excess only) rain simulation experiments. Furthermore, DRP concentration in runoff was positively related with rainfall intensity. Nitrogen concentrations in runoff appeared to be more susceptible to transport process influences than did P. In the case of N, no apparent link between soil N and N in runoff was found. However, increasing runoff depths with rainfall intensity were associated with diminished NO_3–N concentrations in runoff, presumably due to dilution. While consideration of runoff N is necessary to the development of nutrient management strategies, it is important to note that surface runoff is generally not seen as a dominant pathway for N transport.

Both P and N results clearly affirm the critical source area concept: it is the coincidence of high nutrient availability and high transport potential that control nutrient loss from soil, not simply source potential or transport potential alone. The soils examined in the current study contrasted substantially in properties related to nutrient source and transport potential. Here, the greatest export of nutrients was associated with the soil with the lowest nutrient content (lowest source potential), but the highest runoff potential.

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