

## Field-Scale Evaluation of Phosphorus Leaching in Acid Sandy Soils Receiving Swine Waste

Nathan O. Nelson,\* John E. Parsons, and Robert L. Mikkelsen

### ABSTRACT

Accurate descriptions of P leaching are important because excess P applied to soils can enter surface water via leaching and subsurface transport, thereby negatively impacting water quality. The objectives of this study were to monitor P leaching in soils with a long-term history of waste application, relate soil solution P concentrations to soil P status, and quantify P leaching losses. Soil solution was monitored for 20 mo with samplers installed at 45-, 90-, and 135-cm depths in two pits (1 × 3 × 1.5 m) in Autryville (loamy, siliceous, thermic Arenic Paleudults) and Blanton (loamy, siliceous, semiactive, thermic Grossarenic Paleudults) soils located in a grazed pasture in Sampson County, NC, which had received swine waste for >20 yr. Maximum soil solution P concentrations at 45 cm exceeded 18 mg L<sup>-1</sup> in both soils. Soil solution P concentrations at 90 cm in the Blanton soil were similar to that at 45 cm indicating low P sorption. Soil solution P concentrations at 90 cm in the Autryville soil averaged 0.05 mg L<sup>-1</sup> compared to 10 mg L<sup>-1</sup> at 45 cm. A split-line model related soil solution P concentration to the degree of phosphorus saturation (DPS), identifying a change point at 45% DPS. Phosphorus movement past 45 cm equaled or exceeded surplus P additions for both soils. Long-term waste applications resulted in DPS > 90%, high soil solution P concentrations, and substantial vertical P movement. Phosphorus leaching should be considered when assessing long-term risk of P loss from waste-amended soils.

UNDER TYPICAL AGRICULTURAL practices soil P concentrations remain less than that required for environmentally significant P leaching (Sims et al., 1998; Lemunyon and Daniel, 2002; Hansen et al., 2002). However, long-term animal waste applications can increase soil P concentrations to the point where P leaching is a potential environmental threat. For example, Ham (1999) documented soil P concentrations in subsoils of waste application fields in the North Carolina Coastal Plain that were 10 times greater than P concentrations in fields not receiving waste. Elevated subsoil P concentrations from repeated waste applications were also documented by Sharpley et al. (1984), King et al. (1990), Mozaffari and Sims (1994), Maguire et al. (2000), and Novak et al. (2002).

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Because artificial drainage systems accelerate drainage water transport to surface water bodies and decrease the interaction of drainage water with subsoils, the majority of research concerning P loss through leaching has focused on artificially drained soils. Sims et al. (1998) summarized results of over 20 studies that reported P leaching losses through agricultural drainage systems. Although many of the studies reviewed found low P losses in drainage waters, several studies found P concentrations in excess of 1 mg P L<sup>-1</sup>. More recent studies also found elevated P concentrations in drainage water from artificially drained soils (Gardner et al., 2002; Beauchemin et al., 2003).

Although less of an immediate environmental concern, P leaching losses from naturally drained soils can still pose an environmental threat, particularly in sandy soils with low P sorption capacities that receive high P additions from animal waste (Breeuwsma and Silva, 1992; Sims et al., 1998). Early assessment of P leaching losses is advantageous because once a soil profile accumulates high amounts of P, downward movement can continue for many years even when P additions are reduced or eliminated (Breeuwsma and Silva, 1992; Breeuwsma et al., 1995; Schoumans and Groenendijk, 2000).

Recent studies using large intact soil columns or field lysimeters found elevated P concentrations in leachate resulting from inorganic or organic P additions, with mean and maximum P concentrations ranging from 0.01 to 0.8 mg L<sup>-1</sup> and 0.3 to 1.8 mg L<sup>-1</sup>, respectively (Leinweber et al., 1999; Turner and Haygarth, 2000; Djodjic et al., 2004; Godlinski et al., 2004). Leinweber et al. (1999) found positive correlations between surface (0–30 cm) soil P status and P concentrations in leachate while other authors did not. Djodjic et al. (2004) concluded that while surface soil P concentrations may be indicative of potential P release, subsoil properties, including P status, have a strong influence on P leaching.

Teflon suction cups or tension lysimeters have been used to monitor P leaching through unsaturated zones in the soil profile at depths ranging from 15 to 180 cm (Andersen, 1994; Tischner et al., 1998; Andersen and Xia, 2001; Brye et al., 2002). These authors found P concentrations ranging from 0.01 to 0.2 mg P L<sup>-1</sup> in the leachate, with the majority of the samples having less than 0.05 mg P L<sup>-1</sup>. Some of the monitored soils had elevated soil P concentrations in the surface, but the P status of the soil in the immediate vicinity of the samplers was relatively low (Andersen and Xia, 2001).

Although the risk for P loss through leaching is in-

**Abbreviations:** Al<sub>ox</sub>, oxalate-extractable aluminum; DPS, degree of phosphorus saturation; DRP, dissolved reactive phosphorus; Fe<sub>ox</sub>, oxalate-extractable iron; M3-P, Mehlich 3-extractable phosphorus; P<sub>ox</sub>, oxalate-extractable phosphorus; P<sub>w</sub>, water-extractable phosphorus; TDR, time-domain reflectometry; TP, total phosphorus.

creased by excess P applications to soils with low P sorption capacities, relatively few field-scale research studies have been conducted in such an environment. Novak et al. (2000) monitored P concentrations in shallow ground water beneath sandy soils receiving swine lagoon effluent that had P accumulations > 100 mg Mehlich 3-extractable phosphorus (M3-P) kg<sup>-1</sup> soil down to 110 cm. However, M3-P decreased to near background levels (1–5 mg kg<sup>-1</sup>) by a depth of 180 cm and shallow ground water at a depth of 3 m had correspondingly low average P concentrations (<0.05 mg P L<sup>-1</sup>).

Research shows that P movement is greatly increased with increasing soil P status (Maguire and Sims, 2002; Sims et al., 2002). However, reported research on P leaching in naturally drained soils has focused on P concentrations in leachate collected at depths where soil P status is well below the P sorption capacity, thus leading to relatively low estimates of P leaching losses (Turner and Haygarth, 2000; Brye et al., 2002; Djodjic et al., 2004; Godlinski et al., 2004). The objectives of this research study were to (i) monitor P leaching throughout the soil profile of soils with a long-term history of waste application and various degrees of P saturation, (ii) relate P leaching losses to soil P status, and (iii) quantify P leaching losses with respect to P additions.

## MATERIALS AND METHODS

### Site Description

The study was conducted on a 21-yr-old farrow-to-wean swine facility in Sampson County, NC, that was managed typical of farms in the Coastal Plain region. Swine waste generated on the farm is treated in an anaerobic lagoon, after which excess lagoon liquid is applied to approximately 19 ha of 'Coastal' Bermuda grass [*Cynodon dactylon* (L.) Pers.] pasture overseeded with triticale (*X Triticosecale* Wittmack). Pastures are alternately grazed and cut for hay. Lagoon liquid, ranging from 50 to 100 mg P L<sup>-1</sup>, is applied at a rate of 300 kg available N ha<sup>-1</sup> yr<sup>-1</sup> throughout the growing season (February through December), which overapplies P by 35 to 50 kg ha<sup>-1</sup> yr<sup>-1</sup> based on average P removal rates of 10 and 50 kg ha<sup>-1</sup> yr<sup>-1</sup> from pasture and hay fields, respectively, receiving swine lagoon effluent (Nelson and Mikkelsen, 2001). It is very likely, however, that pre-regulation waste applications (first 10–15 yr of farm operation) exceeded current application rates.

In situ suction soil solution samplers, constructed from mod-

ified polysulfone filter holders equipped with 0.45- $\mu$ m polyethersulfone membrane filters (Nelson and Mikkelsen, 2005), were installed in Autryville and Blanton soils (Table 1) located in the swine farm's waste application fields (Fig. 1a). Two pits (1 × 3 × 1.5 m) were excavated in each soil and samplers were installed 40 cm into the sidewalls of the pits so that samplers were beneath undisturbed soil (Fig. 1b). Two samplers were installed at each of three depths (45, 90, and 135 cm) in each pit, for a total of 24 samplers (two soils, two pits, two reps, three depths). Slope in the immediate area of the pits was 0 to 3%, however, the overall slope of the field was 4% with some areas as high as 9%.

Soil moisture was monitored on a subhourly basis with Campbell Scientific (Logan, UT) CS616 automated time-domain reflectometry (TDR) probes inserted horizontally into one pit wall of each soil type at depths of 10, 45, 68, 90, 114, and 135 cm. Time-domain reflectometry probes were field calibrated during the first 3 mo of operation. Precipitation and irrigation were continuously measured with a tipping-bucket rain gauge (TE525; Campbell Scientific) mounted 50 cm from the ground surface. Vegetation around the rain gauge was kept trimmed to <40 cm to avoid interference with rain and irrigation measurement. On-farm irrigation records were used to separate irrigation events from precipitation. After installation of soil solution samplers and TDR probes, the pits were filled in and the area surrounding the pits fenced off from the cattle. Each fall, triticale was seeded inside the fenced area near the same time it was overseeded in the pasture. Vegetation over the samplers was cut and removed throughout the study to maintain the plots in the same condition as the surrounding pasture.

A continuous vacuum of -12 to -10 kPa was maintained in each sampler by means of a hanging water column as described by Riekerk and Morris (1983), thereby allowing for continuous sample collection when soil water potential was greater than -12 kPa. Soil solution samples were removed from the samplers once every 12 to 20 d from September 2002 through April 2004, for a total of 42 sampling intervals. Samples were stored at <4°C and either frozen or analyzed within 2 wk of collection. Because soil solution samplers were equipped with 0.45- $\mu$ m polyethersulfone membrane filters, all samples were filtered during collection. Initial sampler testing showed that refiltering samples did not affect P concentration; therefore, samples were not refiltered. Dissolved reactive phosphorus (DRP) concentration in all samples was determined colorimetrically using a Lachat (Milwaukee, WI) QuickChem 8000 autoanalyzer instrument.

### Soil Analysis

Before pit excavation, three soil cores (9 cm in diameter by 90 cm deep) were removed from each pit for soil chemical

**Table 1.** Soil descriptions for Autryville and Blanton soils in which soil solution P concentrations were monitored.

Horizon	Depth cm	Description
<b>Autryville soil</b>		
Ap	0 to 20	very dark grayish brown (10YR 3/2) sand (2% clay)†; weak fine granular structure
E	20 to 58	brownish yellow (10YR 6/6) sand (2 to 4% clay); weak medium granular structure
Bt	58 to 80	yellowish brown (10YR 5/8) sandy loam (6 to 14% clay); weak medium subangular blocky structure
Bw	80 to 100	yellowish brown (10YR 5/8) loamy sand (8 to 9% clay); weak fine subangular blocky structure
E'	100 to 130	brownish yellow (10YR 6/8) sand (4 to 7% clay); single grained loose
B't	130 to 150+	yellowish brown (10YR 5/8) sandy loam; common medium distinct light gray (10YR 7/1) iron depletions; weak fine subangular blocky structure
<b>Blanton soil</b>		
Ap	0 to 20	dark brown (10YR 3/3) sand (1 to 2% clay); weak fine granular structure
E	20 to 125	brownish yellow (10YR 6/6) sand (1 to 3% clay); single grained, loose
Bt1	125 to 150	yellowish brown (10YR 5/8) sandy loam (5 to 8% clay); weak fine subangular blocky structure
Bt2	150 to 160+	yellowish brown (10YR 5/8) sandy loam; few coarse distinct yellowish red (5YR 5/8) and light grey (10YR 7/1) mottles; weak fine subangular blocky structure

† Clay is defined as soil particles < 2  $\mu$ m effective diameter as determined by the pipette method (Gee and Bauder, 1986).

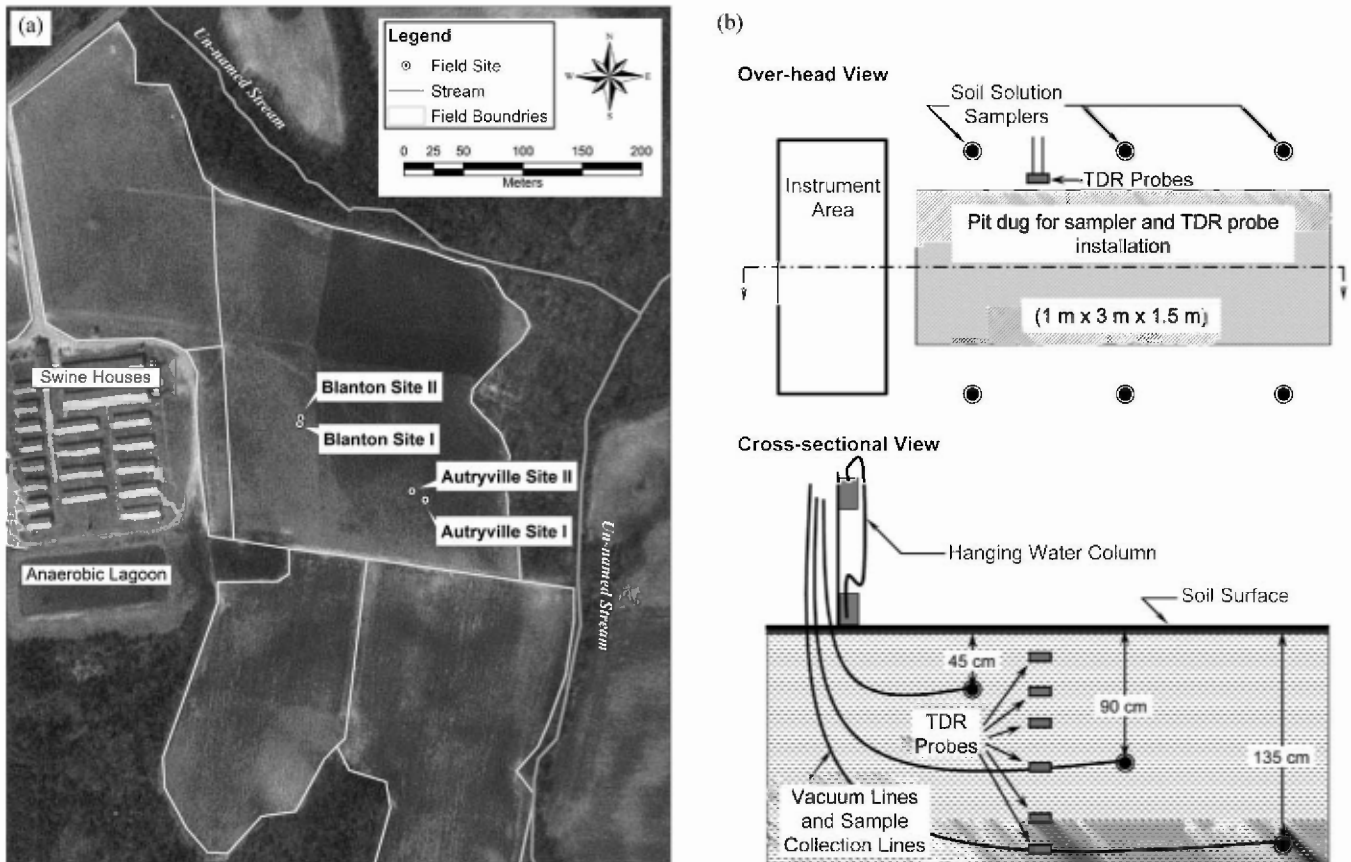


Fig. 1. (a) Location of pits where soil solution samplers were installed in Autryville and Blanton soils on a 20-yr-old swine farm in Sampson County, NC, and (b) schematic of soil solution samplers as installed at each site in Fig. 1a (time-domain reflectometry [TDR] probes were only installed at Blanton Site I and Autryville Site II).

analyses. Soil cores were dissected into 5-, 10-, or 15-cm segments based on soil morphology. Additional soil samples were taken from the side walls of the excavated pits from 90 to 140 cm deep in 10-cm increments. Soil samples were air-dried, ground, and passed through a 2-mm sieve before analyses. Samples were extracted for Mehlich 3–extractable phosphorus (M3-P) (Sims, 2000), water-extractable phosphorus ( $P_w$ ) (Self-Davis et al., 2000), and oxalate-extractable phosphorus, iron, and aluminum ( $P_{ox}$ ,  $Fe_{ox}$ ,  $Al_{ox}$ ) (Schoumans, 2000). The degree of phosphorus saturation (DPS) was calculated as:

$$DPS = P_{ox}(100)/[\alpha(Al_{ox} + Fe_{ox})] \quad [1]$$

where  $P_{ox}$ ,  $Al_{ox}$ , and  $Fe_{ox}$  are oxalate-extractable elements in  $mmol\ kg^{-1}$  soil and  $\alpha$  is 0.5 for noncalcareous sandy soils (van der Zee and van Riemsdijk, 1988; Schoumans and Groenendijk, 2000). Total phosphorus (TP) was determined by a modified sulfuric acid–hydrogen peroxide digestion procedure (Gasparatos and Haidouti, 2001). Particle size analysis of all samples used for chemical analysis was determined by the pipette method (Gee and Bauder, 1986). Phosphorus concentrations in M3-P,  $P_w$ , and TP solutions were determined colorimetrically using the Lachat QuickChem 8000. Phosphorus, Al, and Fe concentration in the oxalate extracts were determined by inductively coupled plasma spectrometry (ICP). A split-line model identifying two slopes ( $b_0$  and  $b_1$ ) and a change point ( $d_0$ ) was fit to the average soil solution DRP measured from each soil  $\times$  pit  $\times$  depth combination and the corresponding DPS (Eq. [2] and [3]):

$$DRP = a_0 + b_0 DPS, \text{ for } DPS \leq d_0$$

$$DRP = a_1 + b_1 DPS, \text{ for } DPS > d_0 \quad [2]$$

$$b_0 = [(a_1 - a_0) + b_1 d_0]/d_0 \quad [3]$$

Intact soil cores 6 cm tall by 5.5 cm in diameter were taken in triplicate from each horizon in each pit for determination of water retention, hydraulic conductivity, and bulk density. Water retention functions were determined with laboratory methods described by Klute (1986) and fit to moisture release curves with a modified form of the van Genuchten equation (Fayer and Simmons, 1995). Saturated hydraulic conductivity was determined using the constant head method as described by Klute and Dirksen (1986). Unsaturated hydraulic conductivity functions were determined using saturated hydraulic conductivity and the van Genuchten method (van Genuchten, 1980). Analysis of variance and contrasts for pair-wise comparisons of soil analysis and soil solution DRP concentrations were performed with SAS proc glm (SAS Institute, 1998). The split-line model and moisture release curves were fit with SAS proc nlin (SAS Institute, 1998).

#### Determination of Percolate Volume

Percolation volume for each sampling interval was determined using a water balance method, given as:

$$\Delta S = (P + I) - ET + D \quad [4]$$

where  $\Delta S$  is the change in storage (cm),  $P$  is precipitation (cm),  $I$  is irrigation (cm),  $ET$  is evapotranspiration, and  $D$  is the drainage (cm). A negative value for  $D$  indicates water movement

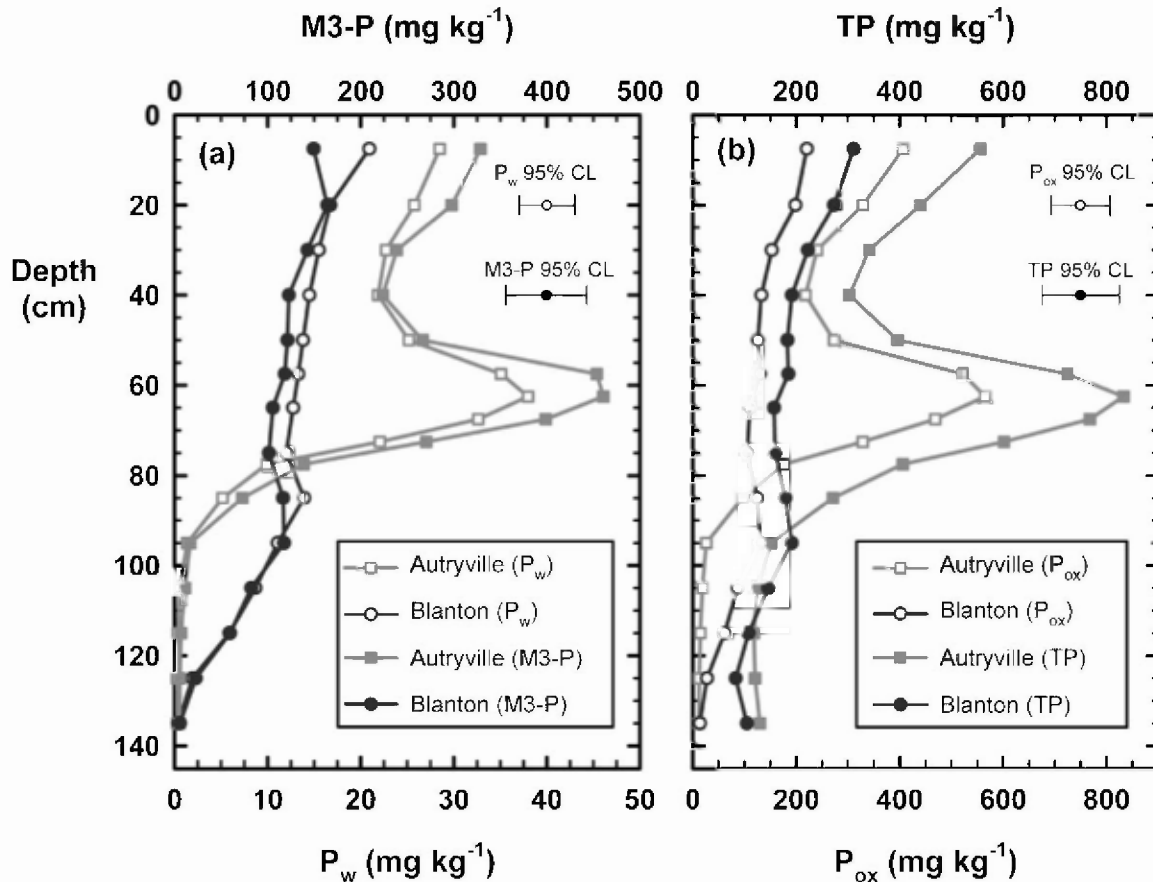


Fig. 2. (a) Mehlich 3-extractable phosphorus (M3-P) and water-extractable phosphorus (P<sub>w</sub>) concentrations with depth in the Autryville and Blanton soils (95% confidence limits [CL] represent ±43 and ±3 mg kg<sup>-1</sup> M3-P and P<sub>w</sub>, respectively) and (b) total phosphorus (TP) and oxalate-extractable phosphorus (P<sub>ox</sub>) concentrations with depth in the Autryville and Blanton soils (95% confidence limits [CL] represent ±74 and ±56 mg kg<sup>-1</sup> TP and P<sub>ox</sub>, respectively).

is downward through the soil profile and a positive value for *D* indicates upward flux of the soil water. Runoff was assumed to be negligible because the field was planted in pasture with good ground cover and the surface soil had very high saturated hydraulic conductivities (>35 cm h<sup>-1</sup>). Furthermore, there were no visual indications of runoff in the vicinity of the field plots.

Computationally, a water balance was determined separately for each soil layer, given as:

$$BAL_1 = \Delta S_1 - (P + I) + PET_1 \quad [5]$$

for the surface soil layer and:

$$BAL_i = \Delta S_i + D_{i-1} + PET_i \quad [6]$$

for subsurface soil layers, where BAL<sub>*i*</sub> is the water balance for layer *i*, PET<sub>*i*</sub> is the potential evapotranspiration removed from layer *i* (cm), and other terms are as previously defined. Potential evapotranspiration (PET) was determined by multiplying the reference ET by a crop coefficient. The reference ET was determined with the FAO56 Penman-Monteith combination method on an hourly time step (Allen et al., 1998). Crop growth was divided into three growth stages with three transition periods, where each growth stage had a different crop coefficient (Chamblee et al., 1995; Allen et al., 1998). The fraction of PET removed from each soil layer (PET<sub>*i*</sub>) was determined as:

$$PET_i = \lambda_i PET \quad [7]$$

where λ<sub>*i*</sub> are distribution coefficients 0.6, 0.32, 0.05, 0.02, 0.01, and 0 for soil layers 1 through 6, respectively, which were determined with the root distribution function:

$$y = 1 - 0.5^{z/m} \quad [8]$$

where *y* is the fraction of roots above depth *z* and *m* is the half-depth or depth at which *y* = 0.5 (Federer et al., 2003). The parameter *m* was set at 15 cm based on research from Beyrouty et al. (1990), Hays et al. (1991), and Bonachela (1996).

Using Eq. [5] and [6], drainage from each layer was computed as:

$$\begin{aligned} D_i &= BAL_i, \quad BAL_i \leq U_{\max(i)}, \\ D_i &= U_{\max(i)}, \quad BAL_i > U_{\max(i)} \end{aligned} \quad [9]$$

where U<sub>max(*i*)</sub> is the maximum upward flow of soil water that could have occurred during the sampling interval (cm).

If the water table was below the monitoring depth (135 cm), then the maximum upward flux of water into a soil layer [U<sub>max(*i*)</sub>] was determined by estimating average daily flux between the midpoints of consecutive soil layers using Darcy's law:

$$q = -K(h)_i [(h_{(i+1)} + z_{(i+1)}) - (h_i + z_i)] / (z_{(i+1)} - z_i) \quad [10]$$

where *q* is soil water flux (cm d<sup>-1</sup>), *K*(*h*) is the unsaturated hydraulic conductivity as a function of *h* (cm d<sup>-1</sup>), *h* is the soil water potential at the center of layer *i* (cm), and *z* is the distance from the surface to the center of layer *i* (cm). Moisture release curves were used to determine *h* from average daily

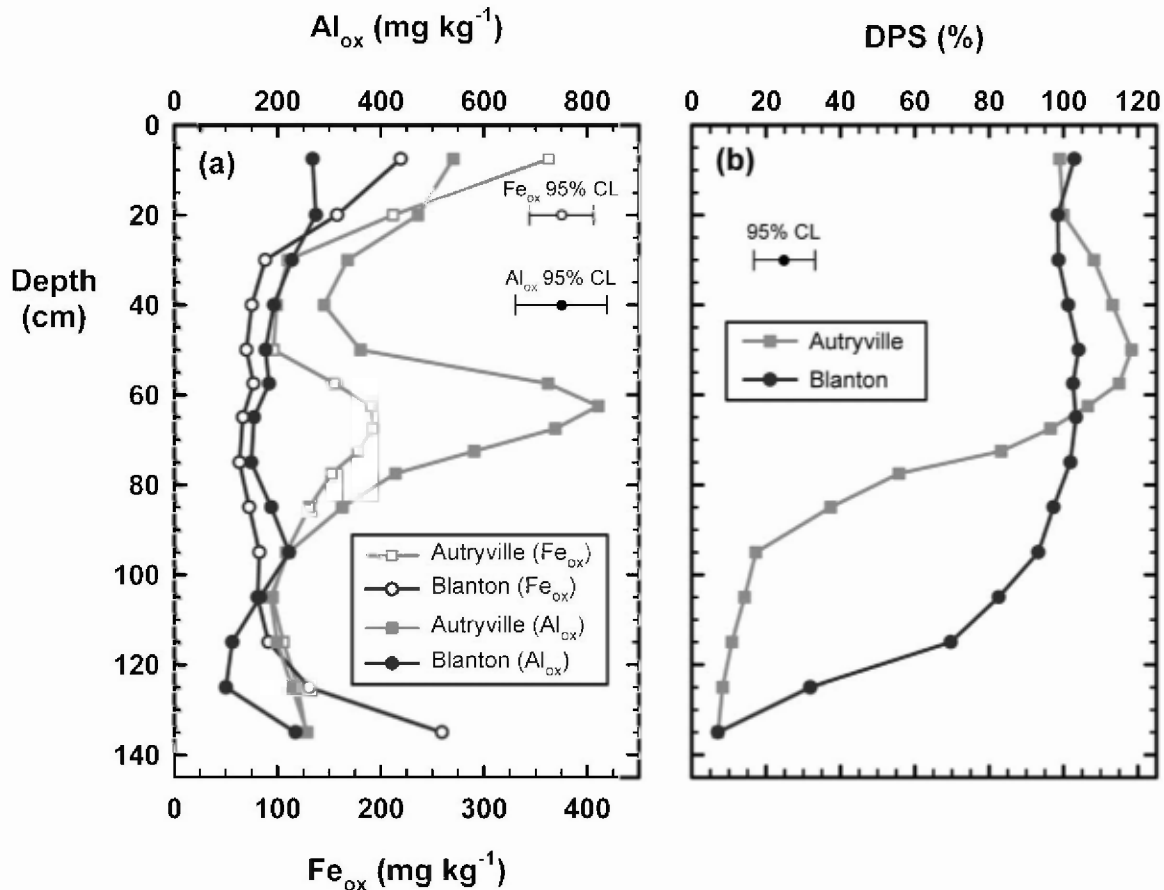


Fig. 3. (a) Oxalate-extractable aluminum ( $Al_{ox}$ ) and iron ( $Fe_{ox}$ ) and (b) degree of phosphorus saturation (DPS) with depth in the Autryville and Blanton soils (95% confidence limits [CL] represent  $\pm 88$  and  $\pm 30\ mg\ kg^{-1}$   $Al_{ox}$  and  $Fe_{ox}$ , respectively, and  $\pm 8\%$  DPS).

volumetric soil moisture content as measured by TDR. Hydraulic conductivity ( $K$ ) at each point was determined with the corresponding  $K(h)$  relationship. Because  $K(h)$  and moisture release relationships were different for each layer, the geometric mean interlayer conductivity ( $K_m$ ) replaced  $K(h)$  in Eq. [10] (Eq. [11], Haverkamp and Vauclin, 1979):

$$K_m = (K_i K_{i+1})^{1/2} \quad [11]$$

When the average daily flux calculated by Eq. [10] was negative the maximum upward flux was set to 0.

If the water table was above 135 cm (as determined by volumetric water content equal to saturation),  $U_{max(i)}$  was estimated using a steady state relationship between maximum upward water flux and water table depth (Skaggs, 1981). Daily estimates of  $U_{max(i)}$  were summed for each sampling interval.

Actual evapotranspiration removed from each layer (AET<sub>*i*</sub>) was determined as:

$$\begin{aligned} AET_i &= -\Delta S_i + (P + I) + D_{ij} = 1, \\ AET_i &= -\Delta S_i - D_{i-1} + D_{ij} > 1 \end{aligned} \quad [12]$$

where  $D_i$  is as computed in Eq. [9]. Notice that for  $D_i \leq U_{max(i)}$ ,  $AET_i = PET_i$ .

Phosphorus losses at each sampling depth were calculated by multiplying the P concentration in soil solution by the calculated drainage for each sampling interval. Annual P losses were computed by summing the P losses for each sampling interval through the entire year.

## RESULTS AND DISCUSSION

### Soil Phosphorus Distribution

Both Autryville and Blanton soils had significant P accumulations in the upper 75 cm of the soil profile (Fig. 2), with M3-P concentrations 40 to 90 times background concentrations for similar soils (Novak et al., 2000). There was a significant soil  $\times$  depth interaction for  $P_w$ , M3-P,  $P_{ox}$ , and TP ( $p < 0.001$ ). Concentrations of  $P_w$  and M3-P followed the same general trend, where the Autryville soil had higher concentrations than the Blanton in the upper 75 cm ( $p < 0.05$ ), the Blanton soil had higher concentrations from 85 to 115 cm ( $p < 0.05$ ), and both soils had similar P concentrations at 125 cm and below ( $p > 0.05$ ). Concentrations of  $P_{ox}$  and TP are higher in the upper 75 cm of the Autryville soil compared to the Blanton ( $p < 0.05$ ), but were similar below 85 cm ( $p > 0.05$ ) (Fig. 2b). Although patterns of P accumulation in the two soils are very different, the high soil P concentrations below 50 cm suggest that past excess P applications have saturated the phosphorus sorption capacity of the upper horizons in both soils.

The sum of  $Al_{ox}$  and  $Fe_{ox}$ , a useful indicator of the P sorption capacity of soils (Breeuwsma and Silva, 1992), ranged from 5 to 50  $mmol\ kg^{-1}$  in the Autryville soil and from 2 to 24  $mmol\ kg^{-1}$  in the Blanton soil, which would suggest that these soils have comparably lower

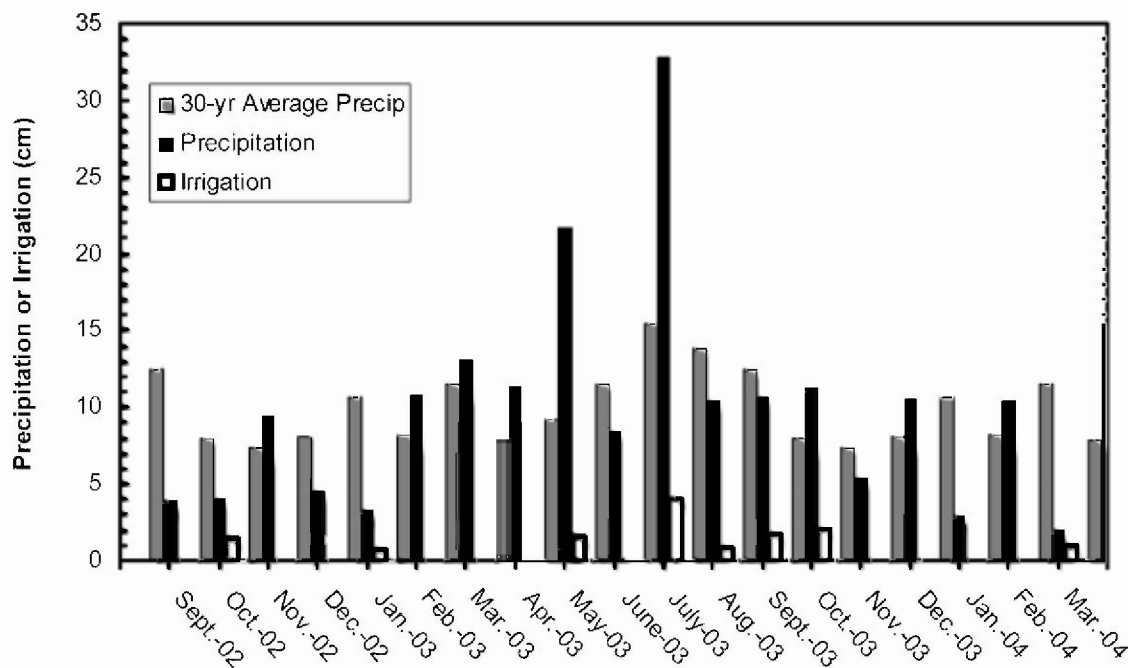


Fig. 4. Monthly precipitation and irrigation during the study compared to the 30-yr average precipitation.

P adsorption capacities than other soils of the Mid-Atlantic Coastal Plain (Sims et al., 2002). The P accumulation at 60 cm in the Autryville soil corresponds to the depth of the Bt horizon, where  $Al_{ox}$  and  $Fe_{ox}$  concentrations also increase (Fig. 3a). The DPS was greater than 90% in the upper 70 cm of the Autryville soil and the upper 100 cm of the Blanton soil (Fig. 3b). Despite the differences in P accumulation in the upper horizons of the soils, the DPS was relatively constant with depth down to 70 cm in the Autryville soil and 100 cm in the Blanton soil. The high DPS and lack of change in DPS with depth are further indicators that these soils are saturated with respect to P sorption capacity in the upper horizons and therefore potentially susceptible to P leaching. However, the DPS for both soils decreases to <10% at 135 cm.

### Drainage

The study contained both wet and dry periods relative to 30-yr average precipitation (Fig. 4). Precipitation was less than average during a few of the first and last months of the study. Year 2003 was a wet year with 149 cm of precipitation, compared to average annual precipitation of 124 cm. Despite the high rainfall in 2003, irrigation continued because the producer had to dispose excess lagoon liquid. The relative AET (AET/PET) for individual sampling intervals was as low as 0.5 and 0.2 in the Autryville and Blanton soils, respectively, during the drier months, but relative AET for the entire study period was greater than 0.9 for both soils.

In accordance with the higher precipitation, there was also more drainage in 2003 than in similar time periods of 2002 or 2004 (Table 2). Generally, drainage volumes were similar between the Blanton and Autryville soils with minor differences occurring in the summer of 2003 (data not shown).

Although the seasonally high water table is 130 and 150 cm in the Autryville and Blanton soils (Table 1), respectively, above average rainfall raised the water table above these depths during the summer of 2003 (Table 2). In the Autryville soil, the water table was between 114 and 135 cm deep for part of July and August 2003. In the Blanton soil, the water table was less than 135 cm deep for approximately 50% of the time samples were collected. When the water table was within the monitoring zone (135 cm), the assumption of no lateral water movement would not be valid for the saturated soil horizons. The possible errors introduced into the drainage calculations by the presence of the water table within the depth of measurements were minimal for the Autryville soil and the upper layers of the Blanton soil. However, drainages out of the 114- and 135-cm layers of the Blanton are considered best approximations given the circumstances.

### Phosphorus Concentrations in Soil Solution

The average soil solution DRP concentration at 45 cm in the Autryville soil was over two orders of magnitude greater than DRP concentrations at 90 and 135 cm (Table 3). By contrast, soil solution DRP concentrations were elevated in the Blanton soil down to 90 cm, but were below the detection limit ( $0.03 \text{ mg L}^{-1}$ ) at 135 cm. Although the DPS in the Autryville soil is greater than or equal to the DPS in the Blanton soil above 70 cm, soil solution DRP at 90 cm is two orders of magnitude less than the Blanton soil, confirming the importance of considering subsoil properties when estimating the fate and transport of P, as was also noted by Djodjic et al. (2004).

The soil solution DRP concentrations at 45 cm followed the same general trend for both soils throughout the course of the study (Fig. 5a). In the Blanton soil, soil

**Table 2. Precipitation, irrigation, evapotranspiration (ET), and drainage by depth in the Autryville and Blanton soils.**

	September– December 2002	January– April 2003	May– August 2003	September– December 2003	January– April 2004	Total
	cm					
Precipitation	22	38	73	38	31	202
Irrigation	1	1	6	4	2	15
	<u>Autryville soil</u>					
ET	18	20	44	20	23	125
Drainage						
10-cm depth	-17	-27	-55	-26	-21	-147
45-cm depth	-11	-22	-41	-19	-16	-109
68-cm depth	-10	-21	-39	-18	-15	-103
90-cm depth	-9	-21	-37	-18	-15	-100
114-cm depth	-9	-21	(-33)†	-21	-15	-99
135-cm depth	-9	-21	(-31)	-23	-15	-98
	<u>Blanton soil</u>					
ET	16	21	46	21	24	127
Drainage						
10-cm depth	-17	-27	-54	-27	-20	-144
45-cm depth	-13	-21	-38	-21	-14	-107
68-cm depth	-11	-21	(-35)	-20	-13	-101
90-cm depth	-11	-20	(-34)	-21	-13	-98
114-cm depth	-10	(-15)	(-35)	-21	-12	-92
135-cm depth	-9	(-13)	(-35)	(-21)	(-12)	-90

† Parentheses indicate intervals in which the soil layer was saturated for greater than 7 d.

solution DRP at 90 cm generally remained slightly less than the DRP at 45 cm and changes in DRP concentration tended to lag behind the changes observed at 45 cm. Dissolved reactive P concentrations at 90 cm in the Autryville soil remained relatively low for most of the study period, with the exception of April 2004, where DRP concentrations peaked at 0.2 mg P L<sup>-1</sup>. Similarly, soil solution DRP concentrations at 135 cm remained below the detection limit for most of the study except in spring 2004 (Fig. 5b). The high variability of P concentrations throughout the sampling period is consistent with results from other studies (Heckrath et al., 1995; Brye et al., 2002).

Increased drainage from overlying soils could potentially transport excess P from animal waste applications to the lower horizons. However, changes in soil solution DRP concentrations did not seem to follow a consistent pattern with either drainage or waste applications (Fig. 6). Furthermore, correlations between soil solution DRP and drainage were not significant ( $p > 0.2$ ). This is in contrast to Stamm et al. (1997) and Hodgkinson et al. (2002) who found increased P concentrations in drainage water following dairy and swine waste applications. These authors concluded that P concentrations increased after waste application because of preferential flow through the soil. It is unlikely that preferential flow is a signifi-

cant transport pathway in the poorly structured sandy soils of the present study, thus explaining the lack of correlation between waste applications and changes in DRP concentrations of soil solution.

The DRP concentrations at 45 cm in the Autryville soil and 45 and 90 cm in the Blanton soil all decreased dramatically during the summer of 2003, which could be a result of climate or crop management factors. Precipitation in May and July of 2003 was over double the 30-yr average (Fig. 4). The high drainage rates at the end of May 2003 would reduce the effective soil to solution ratio, which decreases P concentrations in quantity–intensity relationships (Hesketh and Brookes, 2000; Koopmans et al., 2002). Although this offers a possible explanation for the decreased soil solution DRP concentrations, the correlation between soil solution DRP concentration and rainfall rates was not significant over the course of the study ( $p > 0.20$ ). Further complicating the interpretation, the producer applied lime to the pasture in early April 2003, as the surface soil pH values were 4.8 and 4.6 for Autryville and Blanton soils, respectively. Lime addition could have displaced Al from cation exchange sites and neutralized hydroxyl-Al species, thereby increasing P sorption sites and decreasing DRP concentrations in the soil solution (Robarge and Corey, 1979).

Split-line models describing two linear relationships on either side of a threshold are used to relate DPS, or other measures of soil P status, to P<sub>w</sub> and/or soil solution DRP for use in defining environmental thresholds for P leaching (Hesketh and Brookes, 2000; Maguire and Sims, 2002; Koopmans et al., 2002; Nair et al., 2004). Similarly, a split-line model was fit to the average soil solution DRP measured from each soil × pit × depth combination and the corresponding DPS for the soils in this study. The split-line model identified a change point in the relationship between soil solution DRP and the soil DPS at 45% DPS, after which the DRP concentration increased rapidly with increasing DPS (Fig. 7a). This

**Table 3. Mean dissolved reactive phosphorus (DRP) concentrations in soil solution collected in Autryville and Blanton Soils from 1 Sept. 2002 to 4 Apr. 2004.**

Depth	Autryville		Blanton	
	Mean	CL† ±	Mean	CL ±
cm	mg L <sup>-1</sup>			
45	9.95	2.04	8.35	3.52
90	0.05	0.03	6.37	2.54
135	0.02	0.01	0.02	0.01

† 95% Confidence limit of the mean.

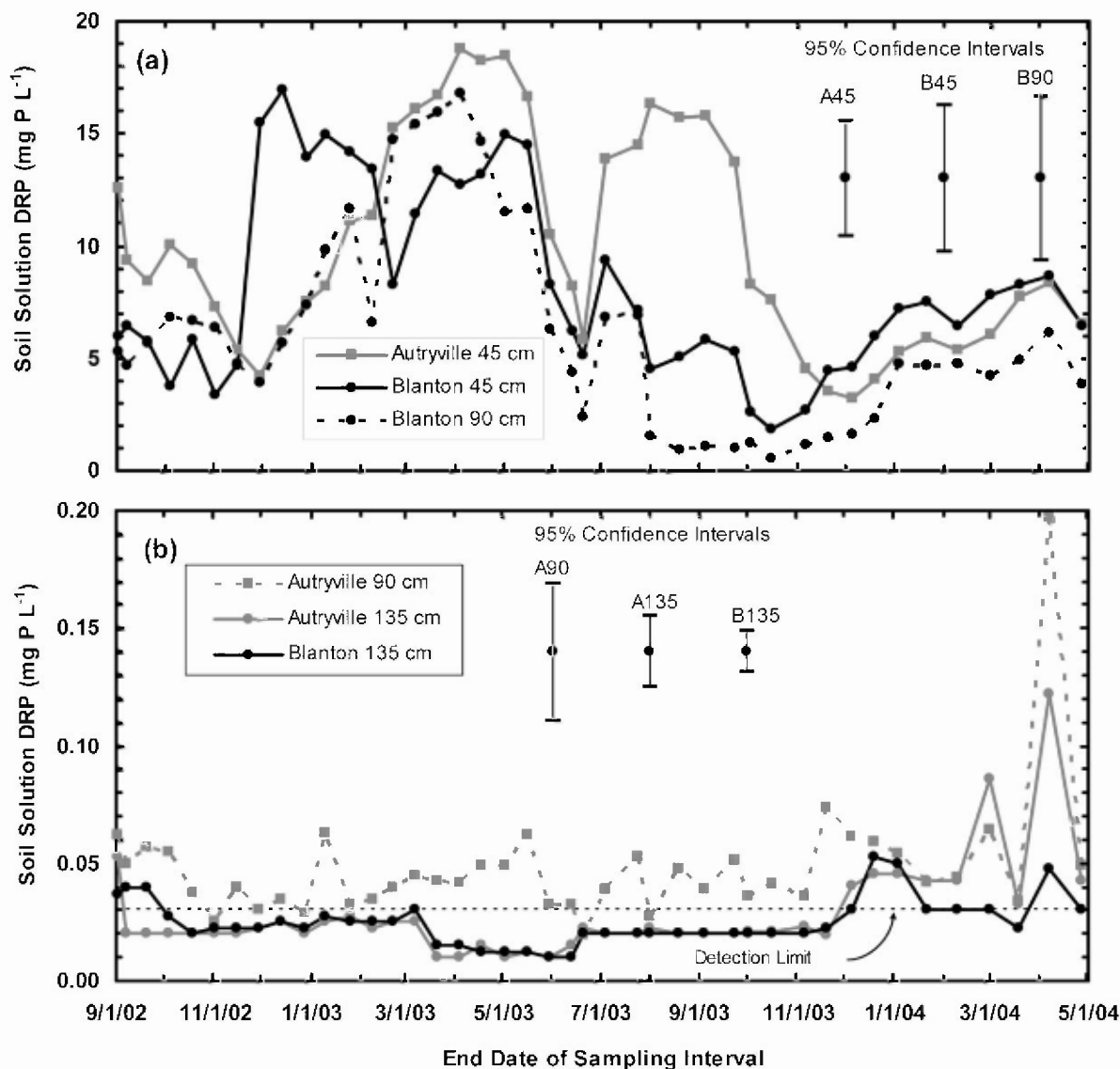


Fig. 5. Dissolved reactive phosphorus (DRP) concentrations in soil solution collected at (a) 45 cm in the Autryville soil and at 45 and 90 cm in the Blanton soil and (b) 90 and 135 cm in the Autryville soil and at 135 cm in the Blanton soil.

change point is slightly lower than the change point of 56% identified by Maguire and Sims (2002), which may be due to our lack of data points between 43 and 95% DPS. The relationship between  $P_w$  and DPS was also fit with the split-line model, identifying a change point at 59% DPS (Fig. 7b). The increase in the slope of the  $P_w$  vs. DPS relationship after the change point was much less than that in the DRP vs. DPS relationship, as indicated by a smaller  $b_1$  to  $b_0$  ratio (Table 4). The smaller  $b_1$  to  $b_0$  ratio is a result of the smaller soil to solution ratio for the  $P_w$  extraction. Koopmans et al. (2002) calculated that soil to solution ratios of 1:0.3 and 1:10 should have  $b_1$  to  $b_0$  ratios of 110 and 7.1, respectively, which are in general agreement with soil to solution and  $b_1$  to  $b_0$  ratios in Table 4. Koopmans et al. (2002) also calculated that the change point should decrease with decreasing soil to solution ratio, which is contrary to what was found in this study. However, this discrepancy could

also be due to a lack of data points between 43 and 95% DPS in the DRP vs. DPS relationship, thus suggesting that the change point would have been greater had there been more data points between the extremes. Although change points identified in this study agree with results of Maguire and Sims (2002), they are greater than the change point of 20% DPS found by Nair et al. (2004) and the change point of 25% DPS suggested by research in the Netherlands (Breeuwsma et al., 1995; Schoumans and Groenendijk, 2000).

#### Mass Losses of Dissolved Phosphorus

Phosphorus movement past 45 cm in both soils was not significantly different from the mass of P applied in waste over the course of the study (Table 5). Average crop P removal for Bermuda grass hay/pasture is approximately 30 kg P ha<sup>-1</sup> yr<sup>-1</sup>, assuming half the forage is



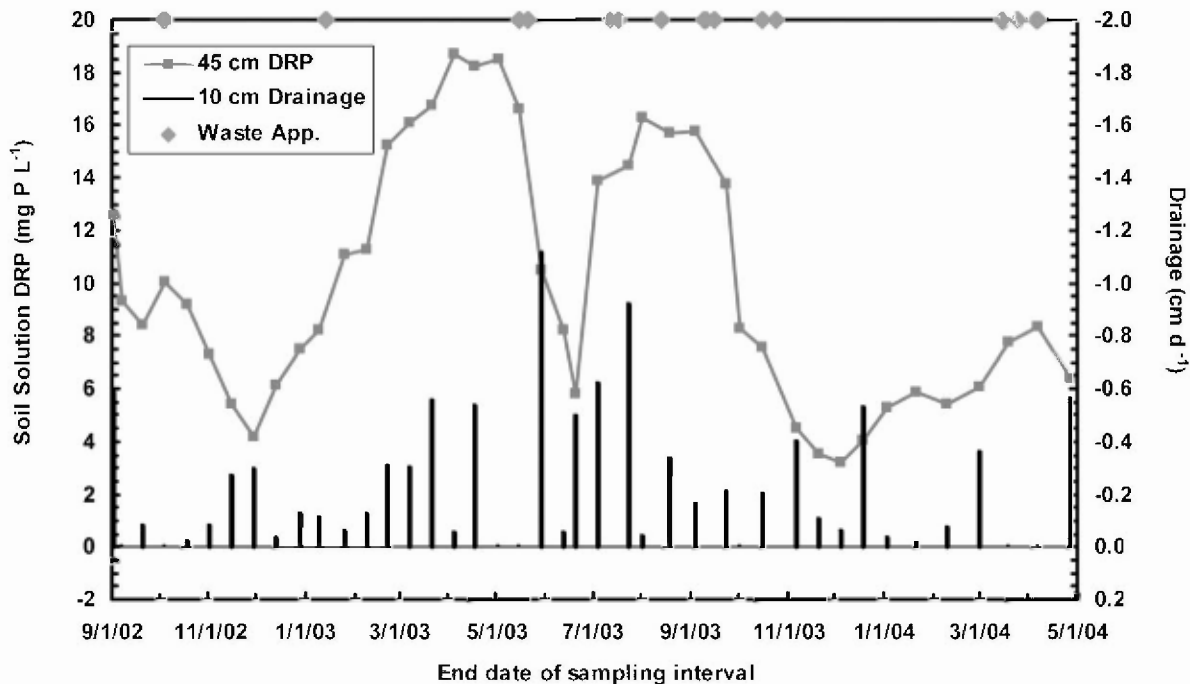


Fig. 6. Dissolved reactive phosphorus (DRP) concentrations in soil solution at 45 cm in the Autryville soil with corresponding drainage rates from the overlying soil layer (0–20 cm; bars) and waste applications.

grazed and half removed as hay (Nelson and Mikkelsen, 2001). Therefore, P movement past 45 cm in both soils was greater than estimated excess P applications for the 20-mo monitoring period. Although P movement past 45 cm was not greater than P applications in fall 2002 or spring 2004, the combination of high DPS and greater than average rainfall caused P movement past 45 cm to exceed P applications for the Autryville soil during 2003. Phosphorus leaching in excess of P application indicates P desorption from the upper 45 cm of the Autryville soil contributed to P leaching in 2003. Phosphorus desorption, resulting in P leaching in excess of P additions, has also been observed in laboratory scale studies (Beauchemin et al., 1996). Although 108 kg P were leached from the upper 45 cm of the Autryville soil only 0.4 kg were leached past 90 cm because of P adsorption between 45 and 90 cm, equating to an estimated 5% increase in the mass of adsorbed P ( $P_{ox}$ ) at that depth. Because the DPS is greater than 95% above 70 cm, it is possible that the P adsorption would occur between 70 and 80 cm, where the soil P status is below the P sorption capacity. Addition of 108 kg of P to the soil at 70 to 80 cm would increase  $P_{ox}$  by 27% and raise the DPS from 70 to 98%.

Phosphorus quantities leached from the P saturated soil horizons in this study are much greater than the P losses determined by other studies, which range from  $<0.1$  to  $3.2 \text{ kg P ha}^{-1} \text{ yr}^{-1}$  (Sims et al., 1998; Leinweber et al., 1999; Brye et al., 2002; Djodjic et al., 2004; Godlinski et al., 2004), with the exception of a tile-drained organic muck soil where losses were as high as  $31 \text{ kg ha}^{-1} \text{ yr}^{-1}$  (Duxbury and Peverly, 1978). However, P leached from soil horizons with  $\text{DPS} < 45\%$  are well within the range reported by other studies. This demonstrates that substantial vertical P movement can occur in the upper

portions of a soil profile even when soil solution P concentrations in the deeper depths are near detection limits. Excessive P leaching in the P saturated soils of the present study are due to the overapplication of P, the low P sorption capacity of the soils, and average annual rainfall in excess of evapotranspiration. Because P desorption from P saturated soil horizons will continue to maintain high soil solution P concentrations and result in P leaching even in the absence of P applications (Breeuwsma and Silva, 1992; Breeuwsma et al., 1995; Schoumans and Groenendijk, 2000), due attention should be given to evaluation of P leaching losses on soils with low P sorption capacities. While sandy soils with low P sorption capacities indicate potential for P leaching losses, site hydrology and subsoil characteristics also affect eventual fate and transport of P leached out of the root zone.

## CONCLUSIONS

Annual P applications of at least  $35 \text{ kg excess P ha}^{-1} \text{ yr}^{-1}$  for 20 yr on sandy soils resulted in elevated P concentrations in soil and soil solution below the root zone. Substantial P movement occurred in the upper horizons of P saturated soils while soil solution P concentrations remained very low in the lower horizons ( $\text{DPS} < 45\%$ ). Therefore, accurate assessments of P leaching in sandy soils must incorporate P status and P sorption capacity of the entire soil profile, not just the surface horizons.

A split-line model adequately described the relationship between average DRP concentration in soil solution and the DPS and identified a change point at 45% DPS. Slope above the change point was 140 times greater than the slope below the change point; therefore, increases in adsorbed P above 45% DPS have 140 times

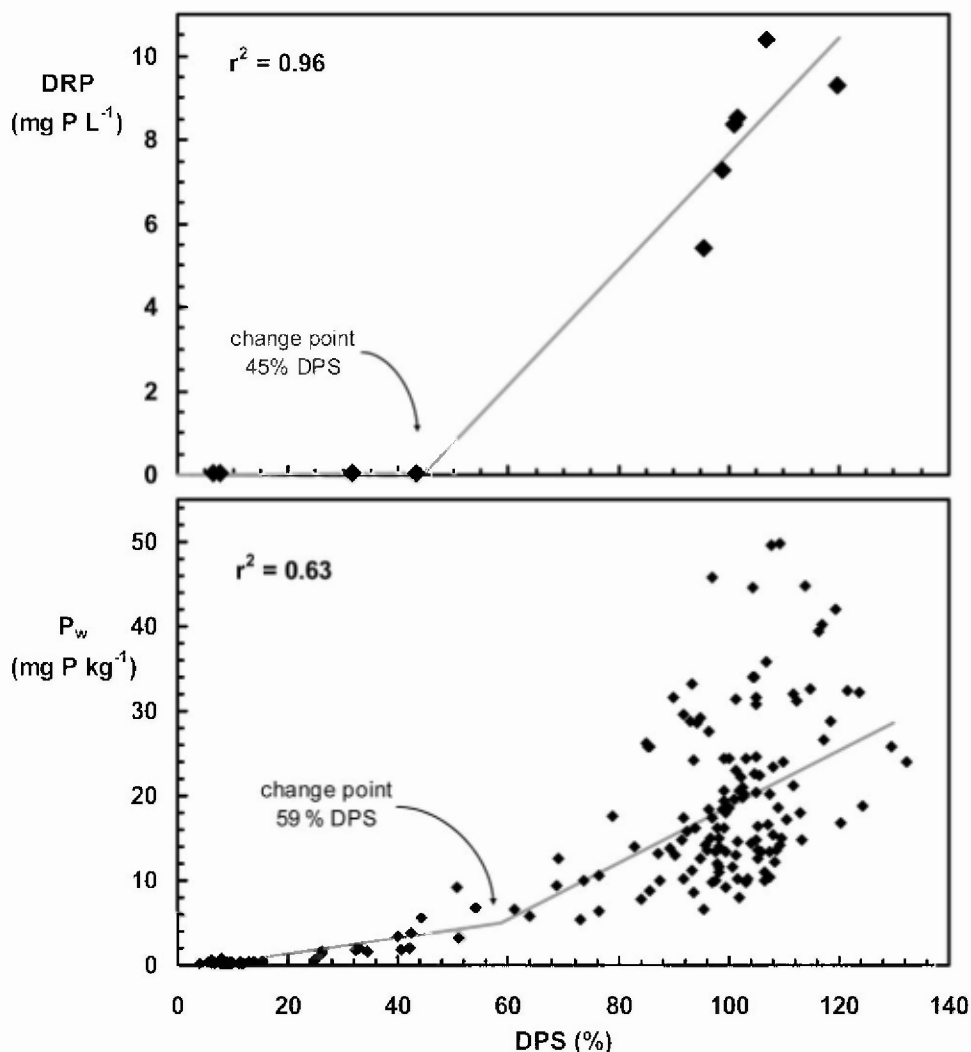


Fig. 7. Relationship between the (a) average soil solution dissolved reactive phosphorus (DRP) concentration and soil degree of phosphorus saturation (DPS) and (b) water-extractable phosphorus ( $P_w$ ) concentration and DPS for Autryville and Blanton soils fit with corresponding split-line models.

greater impact on soil solution P concentrations compared to soils with less than 45% DPS. A change point of 59% DPS was determined for the  $P_w$  vs. DPS relationship, suggesting that more data points for the soil solution DRP vs. DPS relationship may be needed to more precisely identify the change point.

Vertical P movement in excess of surplus P additions was observed in P saturated soil horizons when precipitation was above average, indicating that P desorption contributed up to 50% of P leached past 45 cm in the Autryville soil. Thus, P leaching could continue even in absence of P additions, demonstrating the need to consider long-term leaching losses when calculating potential environmental impacts of P loss from agricultural soils with low P sorption capacity.

Environmental consequences of the observed P leaching losses are determined by eventual P transport to surface water through subsurface lateral flow, which is in turn dependent on future P applications, subsurface flow rates and pathways, and soil characteristics through which lateral flow occurs. Although this analysis is be-

yond the scope of the present study, the results of the present study show that substantial quantities of P can be leached through soils with low P sorption capacities and further study on the fate and transport of P leached past the root zone is warranted. Future studies should include field evaluations of subsurface P transport and processed-based modeling efforts to predict P sorption, desorption, and transport in the root zone and through subsurface pathways. Modeling efforts would also be useful in identifying the future impacts of current P man-

Table 4. Parameter estimates for the split-line models fit to the relationships between soil solution dissolved reactive phosphorus (DRP) and degree of phosphorus saturation (DPS) and water-extractable phosphorus ( $P_w$ ) and DPS (Eq. [2]).

Dependent variable	$d_0$	$a_0$	$b_0$	$a_1$	$b_1$	Soil to solution ratio	$b_1$ to $b_0$ ratio
DRP	45.0	0.014	0.001	-6.18	0.14	1:0.1†	140
$P_w$	58.6	-0.60	0.097	-14.16	0.33	1:10	3.4

† Estimated based on average water content of  $0.16 \text{ cm}^3 \text{ cm}^{-3}$  for the study period and bulk density of  $1.6 \text{ g cm}^{-3}$ .

**Table 5. Phosphorus additions from waste application and P losses through leaching in Autryville and Blanton soils.**

	September– December 2002	January– December 2003	January– April 2004	Total
<b>kg P ha<sup>-1</sup></b>				
<b>Phosphorus balance</b>				
Waste-applied P	18	68	15	102
Estimated crop P removal†	10	30	10	50
Excess P applied	8	38	5	52
<b>P leaching losses</b>				
<b>Autryville</b>				
45 cm	7 (±0.8)‡	91 (±13)	10 (±6)	108 (±13)
90 cm	0.0 (±0.03)	0.3 (±0.2)	0.1 (±0.06)	0.4 (±0.3)
135 cm	0.0 (±0.0)	0.1 (±0.04)	0.0 (±0.05)	0.1 (±0.08)
<b>Blanton</b>				
45 cm	12 (±6)	61 (±19)	10 (±6)	83 (±31)
90 cm	5 (±3)	46 (±8)	4 (±8)	55 (±18)
135 cm	0.0 (±0.0)	0.1 (±0.02)	0.0 (±0.0)	0.1 (±0.02)

† Average P removal based on Bermuda grass hay/pasture with half of the forage grazed and half harvested for hay, removing 5 and 25 kg P ha<sup>-1</sup> yr<sup>-1</sup>, respectively (Nelson and Mikkelsen, 2001).

‡ 95% Confidence interval.

agement strategies compared to alternative best management practices.

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