Dynamics of phosphorus transfers from heavily manured Coastal Plain soils to drainage ditches

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Abstract: Understanding the dynamics of phosphorus (P) transport in agricultural drainage ditches is essential to their improved management for water quality protection. Seven ditches draining soils with a 20+ year history of receiving poultry litter were monitored: two for five years and five for one year. Ditches receiving runoff from point sources (e.g., barns) exported 4.3 to 25.3 kg total P ha^{-1} (3.8 to 22.6 lb total P ac^{-1}) from 2005 to 2006, while ditches draining areas with only nonpoint source contributions exported 2.6 to 4.8 kg total P ha^{-1} (2.3 to 4.3 lb total P ac^{-1}) during that period. High concentrations of P in field soils (Mehlich-3 P averaged 441 mg kg^{-1}, or parts per million) and ditch soils (Mehlich-3 P averaged 171 mg kg^{-1}) suggest that desorption is the key nonpoint source process controlling P in ditch flow. Over five years, annual total P losses from two ditches with only nonpoint source P contributions were 1.4 to 26.2 kg ha^{-1} (1.3 to 23.4 lb ac^{-1}). Overland flow from the fields to these two ditches accounted for ≤ 8% of annual ditch P export, pointing to groundwater as a key pathway for P transport to ditches. Because P export from ditches was primarily in storm flow and groundwater sampling was primarily during base flow, this study does not provide compelling insight into the role of groundwater in ditch P transport. Only occasionally did dissolved P concentrations in groundwater and ditch flow correspond, and P export from the ditches occurred primarily in storm flow. Sampling of algal mats formed on the bottom of ditches suggests that floating algae may exacerbate sediment-related P transport. Results point to the need for new ditch management practices that can sequester dissolved forms of P and trap floating sources of P, in combination with traditional methods that primarily address sediment-bound P.

Key words: drainage ditches—groundwater—nonpoint source pollution—phosphorus—poultry litter—runoff

Drainage ditch networks are ubiquitous in the poultry-producing region of the Delmarva Peninsula due to the flat coastal plain landscape and predominance of poorly drained soils requiring artificial drainage to support building and transportation infrastructures, as well as crop production. The Peninsula's proximity to the eutrophic Chesapeake Bay and its highly concentrated poultry industry have long made the Peninsula the focus of nutrient management concern (Boynton 2000). Annually, the Delmarva Peninsula produces roughly 600 million broiler chickens (Delmarva Poultry 2006). Because poultry litter (manure + bedding material)

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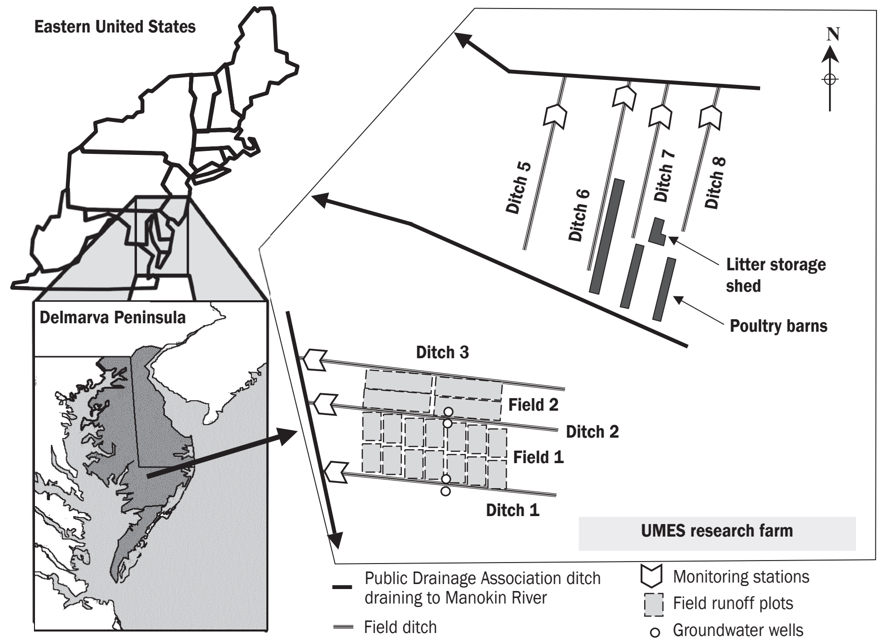
Figure 1
Map of research site, showing location of research farm on Delmarva Peninsula; layout of buildings, fields, and ditches; and distribution of ditch monitoring stations, field plots, and groundwater monitoring wells.



Note: Ditch 4 is not one of the seven ditches in this study.

has historically been applied to field soils on and around the farms where it is generated, nutrient application rates to agricultural soils of the Delmarva Peninsula are among the highest in the Chesapeake Bay watershed (Taylor and Pionke 2000). Consequently, the majority of agricultural soils in counties housing the Delmarva poultry industry have soil nutrient levels, particularly P, exceeding what is required for crop production (Sims 2000). The extensive network of ditches draining nutrient-rich soils that receive regular application of poultry litter creates a high potential for nutrient transfers from the Delmarva's agricultural land to surface waters (Mozaffari and Sims 1994).

Efforts to address agricultural nutrient losses have historically focused on field and barn management, while drainage ditch management has focused on minimizing water excesses on farm land. It is likely that ditches can serve a variety of functions related to nutrient transport. The most obvious role of ditches is as managed conduits of nutrients from field soils to downstream water bodies. In this role, ditches short-circuit overland and subsurface flow pathways, directly linking field edges with local streams and rivers. Undoubtedly, ditches also serve to store nutrients, buffering the impact of field runoff on downstream water quality though sedimentation, chemisorption, and biological uptake (Klotz 1988; Haggard and Sharpley 2006). However, work in fluvial systems suggests that such storages are mostly temporary due to the processes of desorption, mineralization, and resuspension (Kunishi et al. 1972; Sallade and Sims 1997). Some investigators have hypothesized that the long-term accumulation of nutrients in ditches may buffer the effect of recent improvements in nutrient management on water quality in the Chesapeake Bay, at least over the near term (e.g., McCoy et al. 1999).

Despite the prevalence of drainage ditches on the Delmarva Peninsula and their obvious potential to contribute to agricultural nutrient losses, limited research has been conducted in the area of P transport. A few studies have examined processes affecting the fate of P in Delmarva drainage ditches, focusing primarily on interactions between P and ditch soils (sometimes termed ditch "sediments"). Sallade and Sims (1997) examined properties of soils obtained from the bottom of 17 Delmarva drainage ditches. They observed that the equilibrium P concentration at zero sorption (EPC_0) of the ditch soils overestimated dissolved P concentrations in grab samples of ditch water collected during winter months. Conversely, EPC_0 of ditch soils underestimated the dissolved P concentrations in grab samples obtained during summer months. Sallade and Sims (1997) hypothesized that differences between ditch water quality and EPC_0 reflected seasonal cycles of oxidation/reduction and biological immobilization/mineralization that affected P solubility in the ditch soils. Elsewhere, Vaughan et al. (2007) documented large differences in P sorption and desorption properties of ditch surface soils, with roughly three-fold differences in P sorption saturation and EPC_0 across a variety of shallow and deep drainage ditches. While such studies highlight the potential for different chemical processes to affect the water quality of ditch effluent, they address only a subset of potential processes controlling ditch water quality and provide limited insight into the dynamics of ditch water quality.

Given the paucity of information on P transport to and from drainage ditches, the general objective of this study is to elucidate processes controlling P transport in field ditches draining heavily manured coastal plain soils. Specific objectives are to (1) assess variability in ditch P export between various ditches draining a former poultry farm, (2) examine causes of temporal variability in P export, and (3) evaluate sources and pathways of P exported from drainage ditches.

Materials and Methods

Study Site. The study site is on the University of Maryland Eastern Shore research farm in Princess Anne, Maryland (38°12'22" N and 75°40'35" W), part of the 24,000 ha (59,000 ac) Manokin River Watershed (figure 1). The Manokin River is a tributary of the Chesapeake Bay, which is less than 10 km (6 miles) downstream of the University of Maryland Eastern Shore research farm. Elevation at the farm averages 5.5 m (18 ft) above mean absolute sea level. Annual rainfall averages 1,110 mm (44 in) and temperature averages 13°C (55°F) (USDA Natural Resources Conservation Service 2006). A solar powered meteorological station records wind speed and direction, temperature, precipitation, relative humidity, and solar radiation on a five-minute basis.

Prior to its purchase by the University of Maryland Eastern Shore in 1997, the farm had been a commercial broiler operation for 20+ years. The farm includes three broiler houses and a litter storage shed. Broilers are occasionally raised on the farm at decidedly lower intensities than when the farm was an active commercial operation, and litter continues to be stored temporarily in the shed prior to its application to farm soils. As such, key point sources of P persist on the farm, albeit with lesser potential to contribute P than in the past.

Fields on the farm are maintained in a corn/wheat/soybean rotation. Field soils belong to the poorly drained Othello series (fine-silty, mixed, active, mesic Typic Endoaquult) which is derived from silty eolian sediments underlain by coarser marine sediments (Matthews and Hall 1966). Soils are extensively ditched (figure 1), with most fields bounded by at least one deep (>2 m [> 7 ft]) tax ditch maintained by the local Public Drainage Association and at least two shallow (< 1.5 m [< 5 ft]) ditches that are managed by the farm operator. Soils are also heavily enriched with P due to their long history of receiving poultry litter at rates often exceeding annual crop removal. During the period of this study (2001 to 2006), soils were fertilized to meet corn N demand (~9.4 Mg ha^{-1} or 150 bushel ac^{-1} yield goal), either using poultry litter generated from the farm (50 to 150 kg N ha^{-1} and 40 to 120 kg P ha^{-1} [45 to 134 lbs N ac^{-1} and 36 to 107 lbs P ac^{-1}]) or liquid ammonia (120 to 150 kg N ha^{-1} [107 to 134 lb N ac^{-1}]).

Ditch Monitoring. Seven ditches were equipped with flow monitoring stations—two in May 2001 (ditches 1 and 2) and the remainder in May 2005 (ditches 3, 5, 6, 7, and 8) (figure 1). Each ditch monitoring station consisted of an H-flume, ranging in size from 0.5 m (1.5 ft) to 0.8 m (2.5 ft), and an automatic sampler (Sigma 900max, Hach Corporation, Loveland, Colorado) powered by solar panel. The automatic sampler was controlled by a pressure transducer and programmed to collect flow proportional samples (samples 1 through 4 every 95 L [25 gal], samples 5 through 8 every 190 L [50 gal], and samples 9 to 96 every 380 L [100 gal]) that later combined to form a single, composite sample for the event. Samplers were removed annually between November 27 and December 15 due to concerns regarding freezing and returned between March 25 and April 15. However, stage was continuously recorded through the year by float pulley shaft encoder.

Storm event samples obtained by the automatic samplers were retrieved within 48 to 72 hours of each event, and both a filtered (0.45 μm) and unfiltered sub-samples were obtained. Samples were stored at 4°C (39°F) prior to analysis. In addition, grab samples representing low flow conditions were obtained every two to four weeks, including during the winter months when the automatic samplers were absent. Grab samples were processed identically to the storm samples.

Surface Flow from Fields. Two fields adjacent to ditches 1 and 2 (identified as fields 1 and 2) were monitored for overland flow from June to November 2003 as part of a separate experiment assessing poultry litter management on runoff water quality (figure 1). Eighteen 0.1 ha (0.25 ac) field plots were located on field 1, representing approximately 70% of the total field area. Four 0.3 ha (0.64 ac) field plots were located on field 2, representing 60% of the field area. Overland flow from each of the plots was channeled by 15 cm (6 in) earthen berms to a monitoring station consisting of a 0.2 m (9 in) H-flume and automatic sampler. Stage was monitored by pressure transducer which was used to trigger the automatic sampler. Flow was sampled along the hydrograph at intervals and flow volumes described above for the ditches. Water samples were collected and processed within the time frame and by the methods described above.

Subsurface Water Quality. Shallow groundwater wells were installed at various points within the site in 2003 and are described in detail by Vadas et al. (2007). For this study, only water quality data from wells adjacent to ditches 1 and 2 were evaluated (figure 1). Briefly, wells are constructed of a 3.75 cm (1.5 in) diameter inner pipe of solid polyvinylchloride (PVC) that is embedded within a 5 cm (2 in) diameter screened PVC pipe. A series of rubber K-packers glued to the solid pipe create separate cells along the well that are accessed by flexible plastic tubing, which allows water within each cell to be sampled independently. Groundwater was sampled from two discrete depth zones—45 to 75 cm (1.5 to 2.5 ft) and 90 to 133 cm (3.0 to 4.4 ft). Such samples represent background groundwater P concentrations that likely contribute to base flow from ditches when groundwater is in chemical equilibrium with soil. A 250 ml (8.5 oz) sample was collected for each depth, filtered promptly (0.45 μm), and stored at 4°C (39° F) until analysis.

Soil and Algae Sampling. Field soils and soils from areas around likely point sources (barns, litter storage shed) were sampled in 2006 to a depth of 5 cm (2 in). This shallow depth is considered the key zone of interaction between soil P and runoff (Sharpley 1985a) and is therefore preferable to a traditional agronomic sampling depth of 15 cm (6 in). Samples were obtained at roughly 30 m (100 ft) intervals for field soils and 5 m (17 ft) for point source soils and then combined to form a single composite sample. Field soils were sampled along transects paralleling the ditches, approximately half way between the ditch and the midpoint of a field. Sampling of ditch soils (0 to 5 cm [0 to 2 in]) was conducted at approximately 10 m intervals (Vaughan et al. 2007). Unlike field soils, ditch soil samples were analyzed individually. All soil samples were promptly transported to the laboratory, air dried, and sieved (2 mm) prior to analysis.

To assess the potential contribution of algal mats to ditch water quality, samples of algae were collected in September 2006. A 20 cm diameter sieve (2 mm mesh) was submerged under the algal mat and raised to the surface to isolate a section of the mat of known area, and all algae within that area was collected. Algal samples were placed in sealed plastic bottles and promptly refrigerated (4°C [39°F]) prior to analysis.

Laboratory Analysis. Dissolved reactive P (DRP) of filtered water samples was determined colorimetrically via phosphomolybdate reduction using a Lachat QuickChem Method 10-115-01-1-A (Diamond 1995). Total P was measured on unfiltered water samples by alkaline persulfate digestion (Patton and Kryskalla 2003), with digest P determined colorimetrically by a modified method of Murphy and Riley (1962), with spectrophotometer $\lambda = 712$ nm. Total solids were determined on 2005-2006 samples only by oven-drying samples at 70°C (158°F) for 48 hours.

Soils were analyzed for Mehlich-3 P (Mehlich 1984) by shaking 2.5 g of soil with 25 mL of Mehlich-3 solution (0.2 *M* CH_3COOH + 0.25 *M* NH_4NO_3 + 0.015 *M* NH_4F + 0.013 *M* HNO_3 + 0.001 *M* EDTA) for 5 minutes. The supernatant was filtered (Whatman #1) and P determined colorimetrically in the neutralized filtrate. Soil samples were also extracted with acid ammonium oxalate by shaking 0.5 g of air-dried soil with 20 mL of 0.1*M* $(NH_4)_2C_2O_4 \times H_2O$ + 0.1*M* $H_2C_2O_4 \times 2H_2O$ (pH adjusted to 3.0) in the dark for 4 hours. Extractable Al, Fe, and P (A_{lox}, F_{eox}, P_{ox}) were determined in the supernatant by inductively coupled plasma optical emission spectroscopy (Ross and Wang 1993).

Algae samples were analyzed in the condition they were obtained from the field to minimize the potential for transformations associated with drying. As such, samples were analyzed on a dry weight equivalent basis, by determining the moisture content of a subsample gravimetrically following oven drying at 70°C (158°F) for 48 hours. Water extractable P of the algae samples was analyzed by extracting 1 g of dry-weight equivalent algae with 100 mL of distilled water on an end-over-end shaker for 60 minutes. The mixture was centrifuged (about 2,900 g for 20 minutes to facilitate filtration) and filtered through a Whatman #1 filter paper prior to colorimetric determination of P. Total P was measured by modified semimicro-Kjeldahl procedure (Bremner 1996), with P determined by inductively coupled plasma optical emission spectroscopy.

Data Analysis. Phosphorus sorption saturation of ditch and field soil samples was estimated from oxalate data, expressed in mmol kg^{-1}, as follows:

$$\text{P sorption saturation (\%)} = [P_{ox}\,(A_{lox} + Fe_{ox})^{-1}] \times 100\ . \quad (1)$$

To facilitate comparisons among ditches, data were evaluated on annual (June 1 to April 30) and monthly bases. Concentration data (mg L^{-1} or g L^{-1}) were averaged over a given time period on a flow proportional basis. Relationships between soil and water properties were evaluated by least squares regression. To correct for the occasional missing sample (e.g., due to automatic sampler malfunction), ratings curves were developed between ditches with comparable flow and water quality characteristics (r^2 = 0.64 to 0.99).

For ditches 1 and 2 only, temporal trends in flow and water quality were evaluated using averaged properties of the two ditches. For these ditches, hydrograph separation was conducted by semilog method to partition flow into base and storm components (Hall 1968). This method assumes that where storm flow ends, the hydrograph is comprised solely of base flow and is the beginning of a line when the hydrograph is plotted in a semilog scale. The line is projected back to the time of the hydrograph peak, and another line is used to connect the peak with the time when flow begins to increase. The area below these lines is considered base flow and the area above considered storm flow.

The relationship and change point between precipitation and storm flow was determined using a split-line model following the method of McDowell and Sharpley (2001). This model describes two linear relationships whose slopes are significantly different from each other on either side of the change point.

Below the change point,

$$\text{Flow} = \text{m1(precipitation)} + c\ , \quad (2)$$

and above the change point,

$$\text{Flow} = \text{m1(precipitation)} + \text{m2(precipitation} - \text{change point)} + c\ , \quad (3)$$

where c is the intercept, m1 is the slope of the linear relationship below the change point, and m2 is the difference in slopes below and above the change point. The four parameters (m1, m2, change point, and c) were estimated by nonlinear regression using SAS's DUD protocol (SAS Institute 1999). Since nonlinear regression does not yield true coefficients of determination, these were determined by fitting a linear equation to the relationship of observed and predicted values.

All analyses were performed using SAS version 8 (SAS Institute 1999). Results discussed in the text were considered significant at $\alpha = 0.05$.

General Properties of Drainage Ditches and Their Watersheds

The seven field ditches monitored in the study included a variety of agricultural sources of nutrients, from nonpoint to point. Ditches 6 through 8 were all possibly influenced by local point sources, which included poultry barns and a litter storage facility (figure 1), whereas the remainder of the ditches appeared to be affected only by nonpoint sources, such as recently-applied fertilizers and manures, or field and ditch soils. The highest Mehlich-3 P concentrations were found in association with the litter storage shed at the upper end of ditches 7 and 8, reflecting extensive presence of spilled litter (table 1). Concentrations around the storage shed were approximately seven to eight times that of the field soils. Even so, Mehlich-3 P concentrations of all field soils were at least twice the environmental threshold of 150 mg kg^{-1} identified by the Maryland P Site Index (Coale 2005). The P content of field soils reflects the history of intensive poultry litter application during the period when the farm was a commercial broiler operation.

The Mehlich-3 P of field soils was up to five times that of ditch soils (table 1). Concentrations of P in the soils of ditches draining point sources were higher than those observed in other ditches, likely due to P translocated from the point sources to the ditch. For the other ditches, the discrepancy in P concentrations in the field and ditch soils suggests that the apparent accumulation of sediment and soil material at the bottom of the ditches is not the result of recent erosion from adjacent fields. If this were the case, enrichment of P in eroded sediments due to the preferential removal of finer particles would result in greater, rather than lesser ditch P concentrations (see Sharpley 1985b). Indeed, Vaughan (2005) reported only limited evidence of recent sediment deposition along the seven field ditches. A well-established vegetative cover within the ditches also served to stabilize ditch banks, minimizing bank erosion.

Ditches ranged in depth from 0.3 to 1.0 m (1 to 3 ft), but were all shallow in comparison with the Public Drainage Association ditches to which they drained (>2.0 m [>6 ft]). Ditches 1 through 3 were the deepest, and therefore expected to have a greater contribution of base flow than the remaining ditches. Ditches 1 through 3 were also somewhat longer than the other ditches and had the largest watersheds (table 1). Watershed areas ranged from 0.8 to 2.8 ha (2.0 to 6.9 ac) for the seven drainage ditches. Given the flat, coastal plain setting, the source of drainage water in ditches is probably not equivalent to the watersheds delineated via topographic survey, particularly for the deeper drainage ditches which had significant base flows (ditches 1 to 3). Undoubtedly sources of groundwater that contribute to ditch flow are not so much controlled by the subtle interfluves defining surface catchments as they are by water table depth, subsurface features (e.g., aquitards), and drainage ditch properties (e.g., depth).

Table 1
Properties of drainage ditches and contributing watersheds.

Ditch	Dimensions l × w × d (m)	Watershed area* (ha)	Potential sources of P to drainage ditches	Field soils (0 to 5 cm) Mehlich-3 P (mg kg^{-1})	Field soils (0 to 5 cm) P saturation†	Ditch soils (0 to 5 cm) Mehlich-3 P (mg kg^{-1})	Ditch soils (0 to 5 cm) P saturation	Point sources Mehlich-3 P (mg kg^{-1})
1	280 × 1.0 × 0.5	2.74	Nonpoint source	421	78%	83	20%	—
2	300 × 1.0 × 1.0	1.77	Nonpoint source	447	66%	96	20%	—
3	320 × 1.0 × 1.0	2.00	Nonpoint source	449	66%	99	19%	—
5	160 × 0.3 × 0.3	1.10	Nonpoint source	467	81%	138	24%	—
6	200 × 0.5 × 0.5	1.04	Point (1 barn) and nonpoint source	457	82%	141	23%	267
7	160 × 0.3 × 0.3	1.20	Point (2 barns) and nonpoint source	382	70%	205	24%	267; 3,113
8	180 × 0.6 × 0.6	0.82	Point (2 barns, litter shed) and nonpoint source	466	79%	438	36%	267; 3,113

* Based upon topographic survey to delineate watershed boundary.

† P sorption saturation = $P_{ox} (Fe_{ox} + Al_{ox})^{-1}$.

Differential Phosphorus Export from Seven Ditches, 2005 to 2006

Phosphorus Concentrations. Field drainage ditches exhibited a wide range of flow and P export characteristics (table 2). Flows were greatest from ditches 1 through 3, corresponding to their greater depths and lengths, as well as to their larger watersheds. Concentrations of total P in flow from these ditches and ditch 5, all of which drained nonpoint sources only, were the lowest of the seven ditches, averaging ≤1.02 mg L^{-1}. In contrast, average total P concentrations in effluent from ditches 6 through 8, which drained point sources, ranged from 3.01 to 6.17 mg L^{-1}. Regardless of relative standing, total P concentrations from all ditches were well in excess of the 0.02 mg L^{-1} threshold for eutrophication suggested for flowing waters of low velocity (Dodds et al. 1997, 1998).

For nearly all ditches, DRP accounted for the majority of total phosphorus (TP) concentrations in runoff, with DRP averaging 66% of TP across the seven ditches. It is notable that the proportion of TP that was dissolved reactive was no less from the ditches draining nonpoint source areas than those draining point sources. This high proportion of DRP points to readily soluble sources of P as key controls of P in ditch flow. Indeed, the high concentrations of P in field soil and even in ditch soils, as well as their high degree of P sorption saturation (table 1), suggest that desorption is the key nonpoint source process controlling P in ditch flow. High concentrations of P in soils around point sources provide evidence of the potential for these sources to contribute to effluent P from ditches 6 through 8.

Concentrations of DRP also appeared to be affected by sediment concentrations in ditch effluent. As depicted in figure 2A, annual average concentrations of DRP were negatively related to total solids, consistent with the findings of Sharpley et al. (1981) who documented the sorption of dissolved P by suspended particles in overland flow. When the relationship between DRP and total solids concentration is depicted with monthly (as opposed to annual) data (figure 2B), a weak relationship can still be discerned, highlighting the role of total solids as a factor controlling DRP concentrations between events from individual ditches. The

Table 2
Effluent and watershed characteristics of seven drainage ditches in 2005.

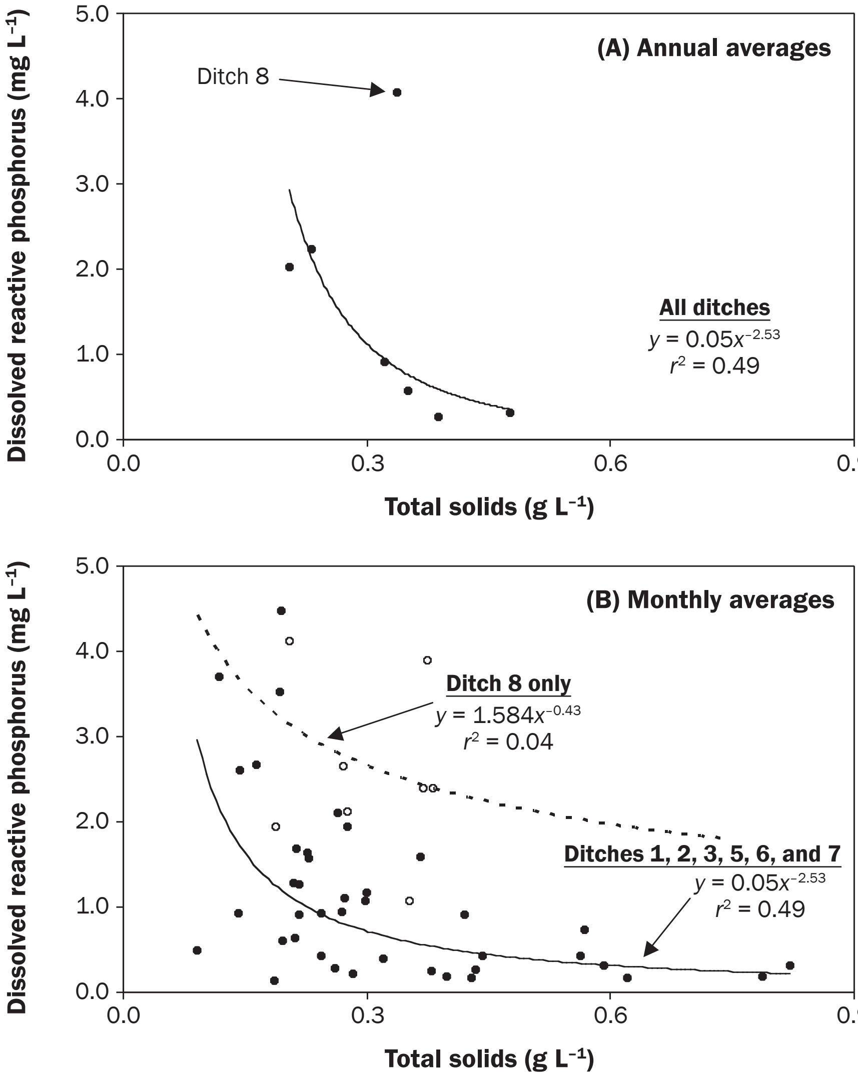
Ditch	Flow (m^3)	Concentrations* DRP (mg L^{-1})	Concentrations* TP (mg L^{-1})	Concentrations* Total solids (g L^{-1})	Loads DRP (kg)	Loads TP (kg)	Loads Total solids (Mg)	Losses† DRP (kg ha^{-1})	Losses† TP (kg ha^{-1})	Losses† Total solids (Mg ha^{-1})
1	6,163	0.25 (0.13)	1.02 (0.74)	0.39 (0.18)	1.6	7.3	2.4	0.6	2.7	0.9
2	11,556	0.31 (0.27)	0.51 (0.47)	0.47 (0.28)	4.4	5.9	5.5	2.5	3.3	3.1
3	12,827	0.56 (0.43)	0.75 (0.46)	0.35 (0.22)	7.9	9.7	4.5	3.6	4.8	2.3
5	3,174	0.90 (0.26)	0.90 (0.52)	0.31 (0.20)	2.9	2.9	1.0	2.6	2.6	0.9
6	1,414	2.02 (1.45)	3.20 (1.57)	0.20 (0.07)	3.3	4.5	0.3	3.2	4.3	0.3
7	3,450	2.22 (1.44)	3.01 (1.39)	0.10 (0.06)	7.7	10.4	0.8	6.4	8.6	0.7
8	3,382	4.05 (1.75)	6.17 (2.05)	0.34 (0.07)	13.7	20.9	1.1	16.6	25.3	1.4

Notes: DRP = dissolved reactive phosphorus; TP = total phosphorus.

* Flow weighted averages with standard deviations (of all measured concentrations) presented in parentheses.

† Based upon topographic survey to delineate watershed boundary.

Figure 2
Relationship of dissolved reactive P (DRP) and total solids concentration in effluent of seven drainage ditches showing (A) annual and (B) monthly averages.



obvious outlier in this relationship is the DRP in effluent from ditch 8, which appears to be primarily controlled by other factors. Of all the seven ditches monitored in this study, ditch 8 appeared to be most directly impacted by point sources. Ditch 8 terminated at the litter storage shed. Substantial quantities of litter were observed around the shed, reflected by the Mehlich-3 P and P sorption saturation of the soil around the shed (table 1). Vehicle traffic around the shed probably compacted the surrounding soil, hampering infiltration of rainfall and roof runoff. In addition, a discernable swale between the shed and the ditch was observed to channel runoff directly to ditch 8 during wet periods.

Given the shallow depth of the ditches, flat landscape, and high concentration of dissolved P sources, the substantial concentrations of total solids in effluent and large fraction of total P that was not DRP (34%) are also remarkable. Vaughan (2005) described these drainage ditches as decidedly stable systems from the standpoint of pedogenesis, with well-established vegetative cover and few areas where the processes of erosion, scouring, and deposition were evident. One ubiquitous source of total solids and sediment-bound P in the ditches is algae. Given the large amount of available nutrients, algal mats formed in virtually all areas of standing water during the periods between flow events. Standing water was observed for longer periods in the deepest ditches (2 and 3), which also had the greatest losses of total solids. Because of their low density, algal mats readily float in rising ditch water, and it is likely that they were removed during larger flows. Such mats can also serve to float other material, such as mineral sediments, that become trapped in their matrix. Algal mats sampled as part of this study contained high concentrations of total P (2,883 mg kg^{-1}, dry weight basis), only a small fraction of which was water extractable (443 mg kg^{-1}, dry weight basis). Therefore, most P in ditch effluent associated with algae would not be measured as DRP. Although a variety of factors would prevent the loss of all algal P within a ditch (especially entrapment by vegetation), algal mats represent a substantial reservoir of P with a high potential to contribute to P in ditch effluent.

Phosphorus Losses. The high concentrations of P in ditch effluent resulted in generally high loads (kg). When normalized by watershed area (kg ha^{-1}), losses of total P from ditches ranged from 2.6 to 25.3 kg ha^{-1}, equivalent to 2.3 to 22.6 lbs total P ac^{-1} (table 2). In comparison, Correll et al. (1992) reported annual total P losses of 4.2 kg ha^{-1} (3.8 lbs ac^{-1}) from coastal plain watersheds located on the western edge of the Chesapeake Bay. Given the relatively low precipitation from 2005 to 2006 (table 3), it is likely that the losses observed over this period were in fact lower than would be expected over the long term. Indeed, losses of total P from ditches 1 and 2, which had a five-year monitoring history, were at least five times lower than in the three previous years (table 3).

Losses from the ditches were both a product of the sources of P controlling P concentrations in effluent, as well as the volume of ditch effluent. Despite relatively low flow, ditches 6 through 8 had some of the highest losses of P, owing to high concentrations of P associated with point sources (table 2). However, larger flows from ditches 1 through 3, which only drained nonpoint sources, caused P losses from these ditches to be comparable to those from ditch 6, despite the lower concentration of P in their effluent (table 2).

Total solids losses were highest from ditches 2 and 3 (table 2), which were the deepest ditches in the study (table 1). It is possible that much of the total solids, as well as much of the effluent P that was not in

Table 3
Average characteristics of effluent from ditches 1 and 2 from 2001 to 2006.

Year	Flow (m^3)	Precip. (mm)	Concentrations* DRP ($mg\ L^{-1}$)	TP ($mg\ L^{-1}$)	Total solids ($g\ L^{-1}$)	Loads DRP (kg)	TP (kg)	Total solids (Mg)	Losses† DRP ($kg\ ha^{-1}$)	TP ($kg\ ha^{-1}$)	Total solids ($Mg\ ha^{-1}$)
2001 to 2002	3,308	762	0.07 (0.03)	0.95 (0.14)	NA	0.2	3.2	NA	0.1	1.4	NA
2002 to 2003	10,507	991	0.66 (1.02)	4.83 (2.38)	NA	7.0	51.1	NA	3.1	22.7	NA
2003 to 2004	16,805	1,191	1.23 (1.09)	3.51 (2.66)	NA	18.9	59.0	NA	8.4	26.2	NA
2004 to 2005	17,999	1,371	0.67 (0.55)	2.05 (1.02)	NA	12.1	37.0	NA	5.4	16.4	NA
2005 to 2006	7,579	922	0.30 (0.16)	0.70 (0.51)	0.43 (0.19)	3.0	6.6	3.3	1.4	2.9	1.44

Notes: DRP = dissolved reactive phosphorus; TP = total phosphorus; NA = not available.

* Flow weighted averages with standard deviations presented in parentheses.

† Based upon topographic survey to delineate watershed boundary.

dissolved reactive form, were associated with floating algae. Ditches 2 and 3 contained water for nearly all of the year, supporting the copious production of algae that formed floating mats. On average, algal mats contained 172 mg m^{-2} total P (0.0002 oz $yard^{-2}$). Assuming an average ditch length of 310 m (1,000 ft) for ditches 2 and 3 and an average ditch width of 1 m (3 ft), algae covering the base of one of these ditches could contain 0.05 kg (0.1 lbs) of total P. Following storm events, algae were observed to quickly recolonize the bottom of ditches within one to two weeks. Therefore, given that seven major events occurred during the seven months of the 2005-2006 flow year when primary productivity in the ditches was high, floating algae could contribute as much as 0.35 kg (0.77 lbs) of total P, most of which could be considered sediment bound. Although it is unlikely that algae are completely removed from a ditch during storm flow, the cumulative loss of P in floating algae may account for a substantial proportion of sediment-associated P exported from these ditches.

Total solids losses were also very high from ditch 8 (table 2), which had neither the depth nor the algal growth of ditches 2 and 3. Noticeable quantities of poultry litter were scattered around the storage shed at the upper end of this ditch. It is possible that the high total solids losses and large amount of TP that was not DRP (8.7 kg ha^{-1}) were associated with floating litter. As such, floating sources of P (algae, litter) may be important contributors to P loss from drainage ditches.

Phosphorus Export from Two Nonpoint Source Ditches, 2001 to 2006

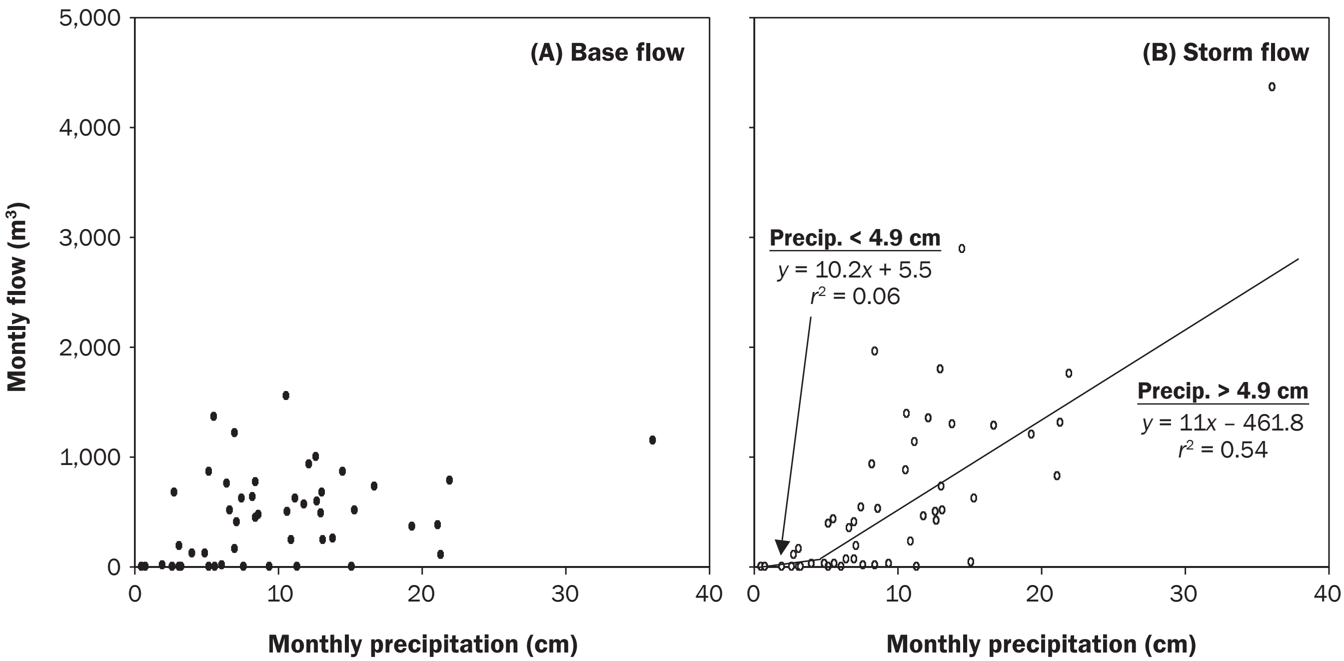
Effluent from ditches 1 and 2 exhibited considerable temporal variability in flow and water quality over the five years of monitoring (table 3). During that period, annual flow volumes from the two ditches had a coefficient of variation of 55%, driven largely by fluctuations in precipitation. Indeed, annual flow and precipitation were strongly related by linear regression (r^2 = 0.95). An average of 60% of the total flow from ditches 1 and 2 occurred as storm flow, with annual averages ranging from 47% to 75%. The shallow depths and small watersheds of the two ditches resulted in a limited base flow capacity. Total monthly base flow volumes varied from 0 to 1,558 m^3 (55,020 ft^3), regardless of the total monthly precipitation amount (figure 3A). Storm flow occurred during most months for which the total precipitation was greater than 4.9 cm (2.0 in), as identified by the change point in the relationship of monthly storm flow volume and precipitation (figure 3B). Antecedent moisture conditions and rainfall intensity undoubtedly caused variations in the storm flow volumes; however, in general, above the change point, storm flow volume exhibited a direct relationship with precipitation amount.

Given the 60% contribution of storm flow (including subsurface and overland flow components) to ditch effluent volumes, it is not surprising that an average of 58% of P losses occurred in association with storm flow. Annual contributions of storm flow to ditch effluent P loss ranged from 47% to 71% over the study period. A relatively minor fraction of total P in effluent from the two ditches was in dissolved reactive form, averaging only 27%. Even so, concentrations of DRP in ditch effluent over the five-year monitoring period averaged 0.59 mg L^{-1}, while total P averaged 2.4 mg L^{-1}. Dissolved reactive P concentration and annual flow volume were strongly, positively related by linear model (r^2 = 0.72), suggesting that a greater number of soluble P sources (soil, manure, point sources) are activated during wet years in which the highest flows occur. No significant relationship was observed between total P concentration and average annual flow volume. Notably, the relative contribution of DRP to total P in runoff increased consistently each year, from 7% for the 2001-2002 flow year, to 49% for the final year of the study. Over the first four flow years, the growing contribution of DRP to total P coincides with increasing flow. However, there was a decline in flow during the 2005-2006 flow year, yet that was the year in which DRP contributed the most to total P. Other explanations for observed trends in DRP are not readily apparent. Indeed, management of fields surrounding ditches 1 and 2 was consistent across the entire period of the study, ruling out changes in management as a cause of the variability in effluent P concentrations.

Temporal trends in P losses (kg ha^{-1}) largely followed those in flow, particularly for DRP (table 3). While annual losses of total P varied nearly 19 fold over the five years, there was up to an 80 fold difference in annual DRP losses. Although losses of P during the 2001-2002 and 2005-2006 flow years were relatively low and comparable to those reported for other agricultural watersheds (e.g., Udawatta et al. 2004), losses during the other years were alarmingly high, particularly given the fact that the ditch 1 and ditch 2 watersheds contained only nonpoint sources of P. During the 2003-2004 flow year alone, an average of 59.0 kg (130 lbs) total P was exported from each ditch, equivalent to 26.2 kg ha^{-1} (23.4 lbs ac^{-1}). Consequently, this study points to ditches as potentially major sources of P to surface waters, particularly for

Figure 3

Relationship of average monthly (A) base flow and (B) storm flow from ditches 1 and 2 to precipitation at the site.



agricultural systems that are highly saturated with P during periods of high flow.

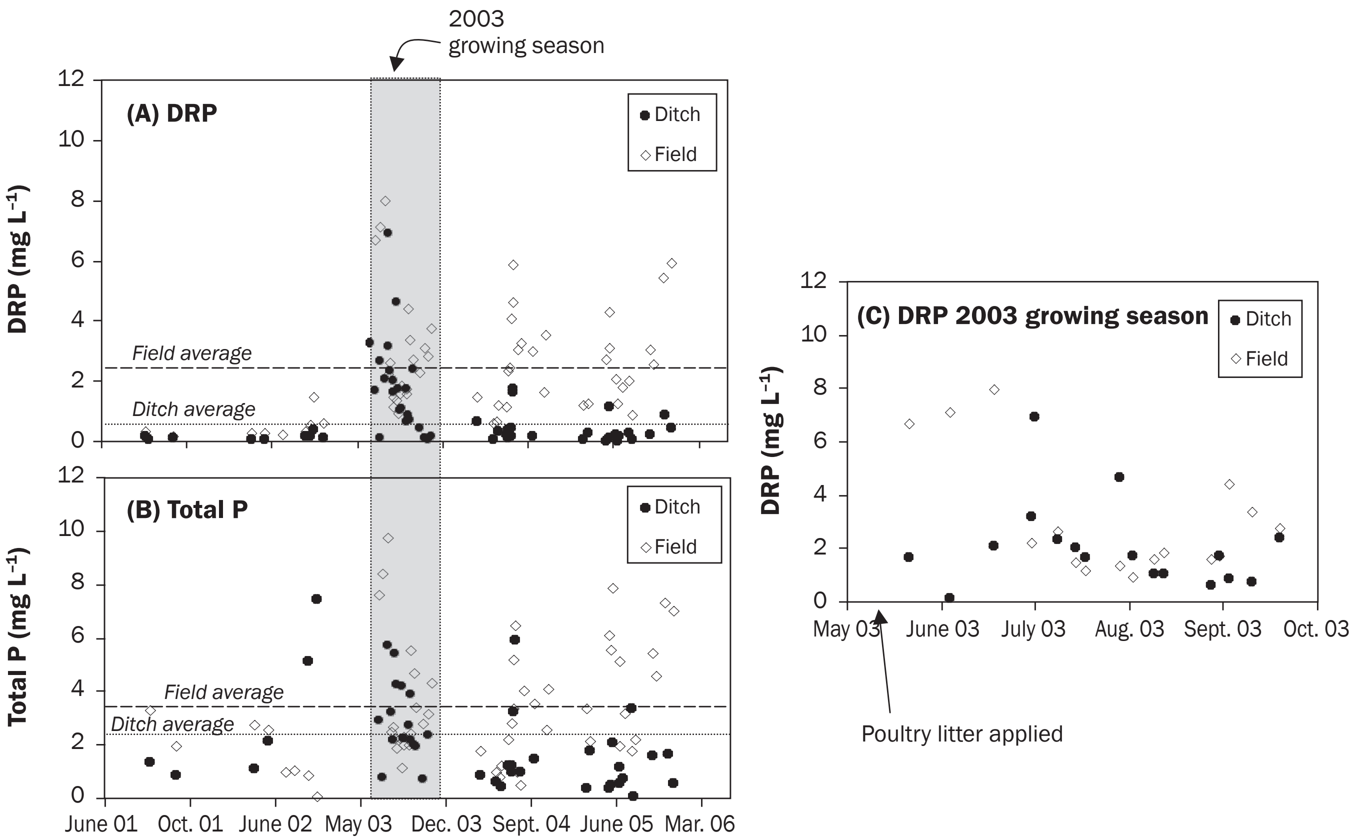
Sources of P Exported from Drainage Ditches. Overland flow from fields accounted for a minor fraction of the total effluent volume from ditches 1 and 2, averaging ≤8%, pointing to subsurface sources as the primary source of flow in the two ditches. Similarly, overland flow appeared to contribute relatively little P to ditch effluent, accounting for <6% of DRP in ditch effluent and <4% of total P. Concentrations of DRP and total P in overland flow from adjacent fields tended to be greater than those in ditch effluent, with DRP and total P concentrations in overland flow averaging 1.8 and 1.4 times that in ditch effluent, respectively (figures 4A and 4B). No significant relationships between P concentrations in overland flow and ditch effluent were discerned. Indeed, trends in P concentrations of overland flow and ditch effluent appeared to be somewhat disconnected. As depicted in figure 4C for the 2003 growing season, DRP concentrations in overland flow from the field plots peaked in the June events, which directly followed poultry litter application to the fields, and then dropped to concentrations comparable to those observed in ditch effluent. This trend is consistent with that observed elsewhere (e.g., Kleinman and Sharpley 2003), in which the contribution of soluble P in applied manure to dissolved P in runoff declines with subsequent events. However, no such trend was observed with DRP concentrations in the ditch effluent. Notably, the increase in DRP concentrations of overland flow in September, 2003, coincides with crop harvest (figure 4C), pointing to senescent crop residue as a possible source (e.g., Sharpley 1981). Again, DRP trends in ditch effluent did not track those in overland flow of the fields.

While the overland flow findings clearly implicate subsurface flow as a major contributor of P to ditches, groundwater wells adjacent to the two ditches provided inconclusive evidence of the role of subsurface P transport in ditch P losses. As illustrated in figure 5, concentrations of DRP in groundwater samples obtained on a monthly basis at depths of 45 to 75 cm (1.5 to 2.5 ft) and 90 to 133 cm (3.0 to 4.4 ft) were roughly 10 to 20 times lower than those observed in ditch flow. Average groundwater DRP concentrations were 0.05 mg L^{-1} for the 45 to 75 cm samples and 0.02 mg L^{-1} for the 90 to 133 cm samples, whereas DRP concentrations in ditch effluent averaged 0.49 mg L^{-1} over the same period. Because these concentrations do not represent storm flow conditions, when most P is transported in ditch flow, it is unclear whether the differences between groundwater and ditch DRP concentrations necessarily point to other sources of DRP in ditch effluent. Indeed, concentrations of DRP from the wells adjacent to ditches 1 and 2 were occasionally in the range of those observed in ditch flow, even though storm flow was not targeted (figure 5). Furthermore, high DRP concentrations in groundwater taken after storm events from wells in field centers and adjacent to other ditches and the Manokin River (Vadas et al. 2007) point to the potential for significant groundwater DRP transport. This may be particularly true when conditions are ripe for macropore (bypass) flow through the fine-textured loess mantle or for lateral flow to ditch across the top of subsoil clay layers of the Othello soils. Therefore, it remains likely that subsurface storm flow represents a key source of P observed in ditch effluent.

Another source of DRP in ditch effluent was the ditch soils themselves. As described above, several studies have made inferences regarding ditch water quality from the properties of ditch soils. Recently, Smith et al. (2006) used the EPC_0 of exposed ditch soils to explain changes in DRP of ditch water quality before and after dredging. While EPC_0 was not measured in the current study, Vaughan (2005) observed an average EPC_0 of 0.30 mg L^{-1} in three surface samples obtained from various ditches in the vicinity of the study area, all of which

Figure 4

Temporal trends in dissolved reactive P (DRP) and total P concentrations in effluent from ditches 1 and 2 and overland flow of adjacent field plots.

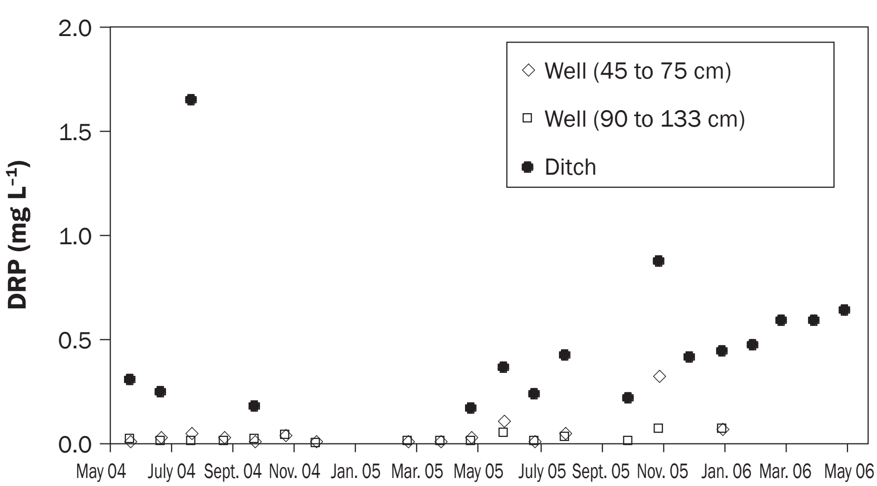


drained nonpoint sources only. Even though this concentration does not quite match the average flow-weighted DRP concentration observed in the effluent of ditches 1 and 2 (0.59 mg L^{-1}), it does indicate that ditch soils themselves can support relatively high concentrations of DRP in ditch effluent.

Much of the P in the ditch effluent was in forms other than DRP and was probably sediment bound. Given the presence of a stable vegetative cover within the ditches and few signs of bank erosion or scouring, the obvious sources of sediment-bound P to runoff are erosion from adjacent field soils and vegetation, especially floating algae. Over the five years, an average of ten storm events occurred during the seven months when primary production would be expected to be highest (April through October). Based upon the assumptions of algal P content described above and an average surface area of 290 m^2 (3,122 ft^2), approximately 0.50 kg total P (1.1 lb) could potentially be contributed to the annual effluent of each ditch by algae. Again, most of this would be associated with sediment-bound P. During years of low flow, such a contribution can be quite significant, but is clearly dwarfed by the losses observed during high-flow years when erosion from field soils would be expected to play a more important role (e.g., 2003 to 2004).

Figure 5

Temporal trends in dissolved reactive P (DRP) in effluent from ditches 1 and 2 and groundwater wells located next to the ditches.



Summary and Conclusions

This study highlights the potential for ditches to serve as key conduits of P from agricultural point and nonpoint sources to surface waters. Due to the dynamic nature of P transport, an integrated approach to managing P loss is required. Cost effective strategies for managing P losses from drainage ditches must involve the targeted application of remedial practices, either over space or over time. In the current study, drainage ditches that serviced areas with point sources had much higher overall P losses than those draining nonpoint sources only, even though the size of the point source ditches and their corresponding watersheds were smaller. Given the high loadings from the point sources, a first goal in remedial management should be to curtail P contributions from these sources to the ditches. Possible activities include removing spilled litter around the litter storage shed, routing of roof runoff away from areas adjacent to the shed, and reclaiming compacted areas to improve infiltration and curtail channelized overland flow.

Substantial variability in flow, exemplified by an 80-fold difference in annual DRP losses from two drainage ditches monitored for five years, resulted in losses from nonpoint source areas during high-flow years that were comparable to losses from point source areas during low-flow years. Given the importance of storm flow to P transport, practices that can lower the peak of the hydrograph, such as flow control structures and detention basins, have a great potential to decrease off-site loadings. These practices may also serve to promote sedimentation, hence the transport of sediment-bound P. Despite the flat landscape, a considerable portion of the total P load from ditches was in non-dissolved reactive forms (e.g., sediment-bound forms)

In agricultural systems that are highly saturated with respect to soil P sorption, the potential for desorption of soil P to runoff and groundwater is high. Practices to filter dissolved P are undoubtedly expensive and should either be targeted to ditches with greatest potential for dissolved P losses (e.g., those draining point sources) or to collection points in drainage systems where the largest volume of drainage effluent can be treated with a single filter. Given the potential importance of ditch soils to dissolved P in effluent, one strategy may be to treat the ditch soils with P-sorbing materials that minimize the potential for P desorption (e.g., Callahan et al. 2002). Alternatively, use of P-sorbing materials at critical control points within ditch drainage systems, such as in flow control structures and detention basins, may also be possible.

One potentially important source of P identified by this study is floating debris. The low density of algae and litter make them highly susceptible to removal by ditch flow. Treatment of ditches to minimize algal blooms, such as applying materials to render P insoluble, may help to minimize algal mats. In addition, some of the structures identified above (flow control, detention basins) may help to trap floating materials.

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