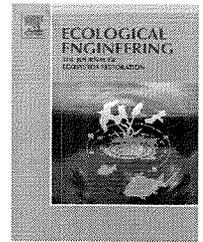


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Changes in sediment–water column phosphorus interactions following sediment disturbance

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ABSTRACT

Stream and ditch sediments are an important part of the ecosystem that can play a pivotal role in buffering nutrient and contaminant concentrations in surface water. Removal of existing sediments and deposition of new sediments will change the physiochemical parameters controlling P transport in drainage ditches; however, these phenomena are not well documented. This study was conducted to determine the impact of severe storms on sediment P interactions with water column P in tile-fed drainage ditches within agricultural watersheds in northeast Indiana. Easily exchangeable P (MgCl_2 extractable) in ditch sediments ranged from 0.5 to 1.32 mg kg^{-1} . The equilibrium P concentration (EPC_0) between the aqueous phase and sediments ranged from 0.02 to 0.11 mg L^{-1} . When comparing sediment EPC_0 with dissolved P concentrations in ditch water, sediments acted as a P sink in five of the seven data collection sites. The ability of the sediment to buffer water column P, as measured by the slope of the linear relation used to calculate sediment EPC_0 , ranged from 2.7 to 13.8 $\text{mg P sorbed kg}^{-1}$ sediment per unit in mg PL^{-1} in the aqueous solution. In contrast to previous studies, chemical amendments did not alter easily exchangeable P in sediments nor sediment EPC_0 and the slopes observed in the linear P sorption isotherms. While chemical amendment can be used to increase P sorption by sediments and reduce exchangeable P in sediments, this showed that this practice should be targeted to times of the year when they would be the most effective.

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1. Introduction

Agriculture has been identified as one of the primary contributors to non-point source nutrient losses in the USA (USEPA, 1990). Excessive phosphorus (P) has evolved to become the main concern in many freshwater systems where anthropogenic eutrophication is a water quality problem (Schindler, 1977). Despite efforts to minimize nutrient loss from agricultural sources, nutrients (particularly P) often reach sur-

face waters in sufficient amounts to accelerate or maintain eutrophication in agriculturally dominated watersheds (Carpenter et al., 1998; Daniel et al., 1998). Nutrients are transported from the landscape or discrete sources downstream to primary surface waters of concern and are subject to interactions with many abiotic and biotic processes in the stream and drainage ditch network. Some information is available on how fluvial processes affect nutrient transport in streams (Klotz, 1988; Haggard et al., 1999; McDowell and Sharpley, 2003), but

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little information is available on sediment P interactions and transport through drainage ditches (except see Smith et al., 2005).

Stream and ditch sediments interact with P in the water column and as such play a pivotal ecological role in buffering water column P concentrations, and thus the P delivered downstream. Specifically, sediments often adsorb P until an equilibrium P concentration (EPC_o) is obtained (Taylor and Kunishi, 1971; Klotz, 1988; Haggard et al., 1999). However, these interactions are not limited to abiotic processes because microbial processes account for up to 43% of P uptake (Khoshmenesh et al., 1999). Sediment EPC_o has often been used to determine whether sediments are a P source or P sink or are in equilibrium with respect to P exchange in the water column. When sediment EPC_o is greater than P concentrations in the water column, the sediments will be a source to the water column, whereas when the P concentrations in the water column are greater than sediment EPC_o , the sediments will act as a sink to P in the water column. Recently, Haggard et al. (2004) have suggested that point sources of P to the water column, such as waste water treatment plants, control sediment EPC_o , especially in P-enriched systems.

Historically, chemical additions have been used to alleviate P release at the sediment–water interface in lakes and reservoirs (Welch et al., 1982; Welch and Schriever, 1994). More recently, the use of chemical amendments to reduce stream and ditch sediment EPC_o and P availability has been studied (Smith et al., 2005; Haggard et al., 2004). Physical disturbance, resulting from flood events or mechanical dredging of drainage ditches may also alter the interaction between sediments and P in the water column. Mechanical dredging could remove P-laden sediments and possibly increase P retention in drainage ditches. Storm events may transport ditch sediments downstream and deposit sediments from the landscape that may serve as a P source or sink to the ditch water after the storm runoff subsides. Little information has been gathered with regard to how physical sediment disturbance alters the ecological role of sediments, specifically with respect to P transport in streams and ditches. The objectives of this research were to characterize the impact of sediment disturbance from a severe storm event on sediment–water column P interactions in three tile-fed drainage ditches in northeast Indiana, USA, and to determine if chemical amendment of these sediments could be used to reduce downstream delivery of P from anthropogenic inputs.

2. Materials and methods

Seven sites on three agricultural drainage ditches were sampled and studied to evaluate sediment–P interactions before and after a severe storm event. These sites have also been intensively studied with daily collection of water quality samples and continuous monitoring of discharge and precipitation. A brief description of these seven sites including watershed size and predominant land use are listed in Table 1 and the watersheds are presented in Fig. 1. A detailed description of the watersheds has been previously described (Smith et al.,

Table 1 – Identification of sampling sites on the three watersheds, contributing area, and land use for seven sites in three watershed used for sediment and water collection

ID	Area (ha)	Land use/cropping
A-Small	299	79% Agriculture 15% Grass/pasture 4% Forest
A-Large	2181	77% Agriculture 16% Grass/pasture 6% Forest
A-X-large	4307	78% Agriculture 14% Grass/pasture 6% Forest
B-Small	311	85% Agriculture 8% Grass/pasture 6% Forest
B-Large	1421	83% Agriculture 12% Grass/pasture 3% Forest
C-Small	373	83% Agriculture 10% Grass/pasture 4% Forest
C-Large	1340	73% Agriculture 17% Grass/pasture 5% Forest

2005); this study also presents results and discussion of the sediment–P interactions observed before any storm events.

Sediments from the seven sites were collected on June 19, 2003, hereafter referred to as pre-disturbance sediments. On July 14, 2003, approximately 1 week after a series of severe storms passed through northeast Indiana, sediment and water samples were collected from each of the seven sites. Ten liters of water from each site were collected in Nalgene bottles. Two subsamples of water were also collected at this time: one 60-mL subsample was filtered and acidified for determination of soluble nutrients, and another 60-mL aliquot was taken for total nutrient analysis. Four liters of sediment were also collected by removing the upper 2 cm of sediments with a trowel and placing them into Nalgene bottles. Ditch water was poured into the Nalgene containers of sediment, so as not to expose the sediments to air, and reduce the potential for physiochemical changes that could occur due to drying and/or oxidation.

Upon arrival in the laboratory, sediment samples were sieved to 2 mm, and bulk water samples were filtered through a 0.45 μm membrane filter. The water samples were then split into five aliquots of 1 L each. Four of these 1 L aliquots were spiked with additional P at 0.1, 0.25, 0.5, or 2.0 mg PL^{-1} using KH_2PO_4 , while the remaining aliquot remained unspiked at the ambient P concentration. For example, if the original P concentration at a site was 0.06 mg PL^{-1} , then the five 1-L solutions would contain 0.06, 0.16, 0.31, 0.56, and 2.06 mg PL^{-1} .

Water samples filtered in the field were analyzed for dissolved P using ICP-OES, and $\text{NH}_4\text{-N}$ and $\text{NO}_3\text{-N}$ were analyzed using a Lachat flow injection analyzer.

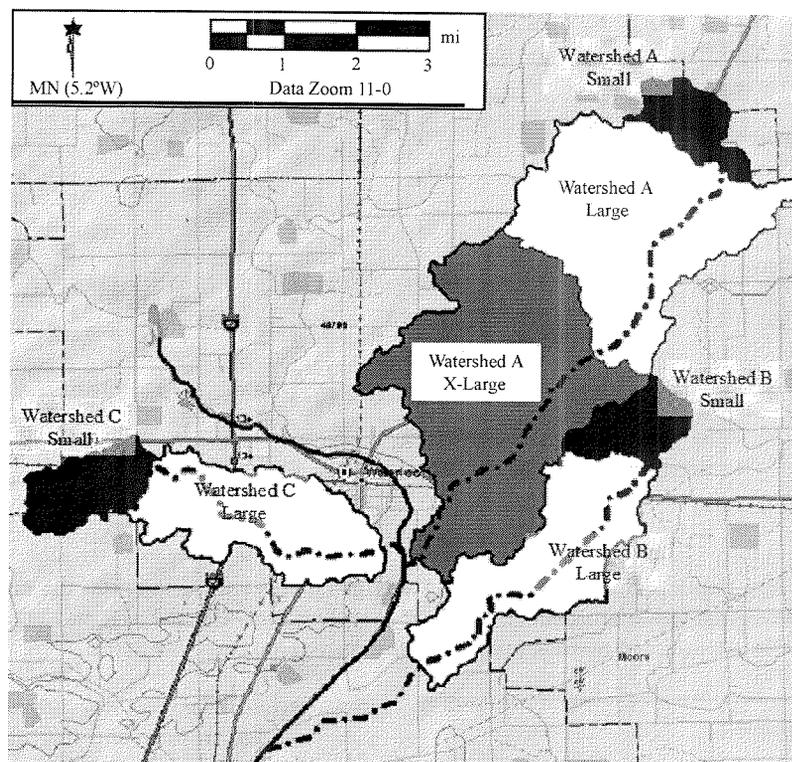


Fig. 1 – Map of sites used for sediment and water collection on tile-fed drainage ditches from a total of seven sites along three ditches.

Twenty grams of wet sediment from each site were placed in 18 centrifuge tubes with a 200-mL capacity. A 100 mL volume of each of the six solutions was then placed into each of three centrifuge tubes, and shaken on an oscillating shaker at $180 \text{ oscillations min}^{-1}$ for 1 h, and finally centrifuged at $5800 \times g$ for five min. The resulting supernatant was filtered through a $0.45 \mu\text{m}$ membrane filter using vacuum filtration, acidified to pH 2 with concentrated HCl, and then analyzed for dissolved P in the supernatant using ICP-OES. Remaining sediments were then dried at 60°C to determine the dry mass of sediments in each centrifuge tube.

A linear sorption isotherm was used to evaluate P uptake by the ditch sediments where the amount of P sorbed by sediments from the aqueous phase (mg P kg^{-1} sediment) was regressed against the initial P concentration in the aqueous phase. Sediment EPC_0 is determined as the x-intercept of this relation and represents the point of no P adsorption or desorption by the ditch sediments from the aqueous solution. Sediment EPC_0 can be compared to the water column P concentration in the ditches to determine if the sediments are acting as a source (sediment $\text{EPC}_0 > \text{water column P}$), sink (sediment $\text{EPC}_0 < \text{water column P}$), or in equilibrium (sediment $\text{EPC}_0 = \text{water column P}$). The slope of linear relation used to estimate sediment EPC_0 represents a measure of the ability of sediments to adsorb per unit increase in solution P concentrations, i.e. a measure of the P buffering capacity.

Readily exchangeable P (ExP) was determined as the P content extracted from ditch sediments using a 1 M MgCl_2 solution prepared with a nanopure water source. Approximately

20 g wet sediment were put into 200 mL capacity centrifuge tubes and then 100 mL of the 1 M MgCl_2 solution was added. The sediment slurry was shaken and centrifuged as described earlier, and the supernatant was filtered, acidified, and analyzed. The remaining sediments were dried to determine dry mass, and ExP calculated as $\text{mg P extracted kg}^{-1}$ sediment. A partitioning index (PI) was estimated by dividing the ExP in 1 g sediment by the amount of P in 1 mL of ditch water (after Triska et al., 1994); the PI indicates relative availability of P with respect to the water column and sediments where values greater than 1 indicate more sediment P compared to the water column.

To evaluate the efficacy of chemical amendments, 250 g of wet sediment from each site were treated with 0.5 g aluminum sulfate ($(\text{Al}_2(\text{SO}_4)_3 \cdot 14\text{H}_2\text{O})$, commonly referred to as alum) and 0.5 g calcium carbonate (CaCO_3). This rate of chemical treatment has been used previously to study the impacts of chemical treatment on removal of P from the water column in natural streams and agricultural drainage ditches (Haggard et al., 2004; Smith et al., 2005). The alum-treated sediments were used with the six P solutions described above and extracted as described above.

Exchangeable P and the partitioning index were analyzed statistically using general linear model procedures (PROC GLM) in SAS Version 8.0 (SAS Institute, Cary, NC). Means were separated using Fisher's protected LSD. Equilibrium P concentrations and buffering capacity were analyzed using regression techniques based on the sorption isotherm data. Correlation coefficients (R^2) were all above 0.85, and were significant at $P < 0.05$.

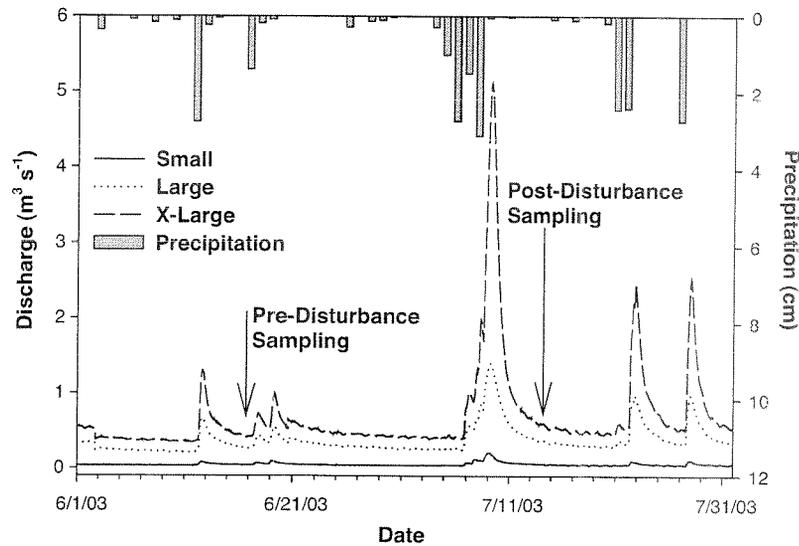


Fig. 2 – Precipitation and flow depth at the small, large, and X-large sites in watershed A for the months of June and July, 2003. Arrows indicate the original (pre-disturbance) sediment sampling that occurred in June 2003 and is cited by Smith et al. (2005) and the post-disturbance sediment sampling for the current study in July 2003.

3. Results

Discharge of the ditches at the three sites on ditch A and daily precipitation for the months of June and July in 2003 are shown in Fig. 2. Peak discharge occurred on July 9, 2003, after several days of intense storms. Hydrograph responses in watersheds B and C were similar to those observed in watershed A (Fig. 3). Ditches in these watersheds were not stagnant during this study, as the 2003 hydrologic year was very wet, so field tiles were draining into the ditches throughout the year.

Soluble nutrient concentrations in ditch water are given in Table 2. Post-disturbance water samples from sites located within watershed A had dissolved P concentrations of approximately two times greater than the dissolved P in the samples collected prior to disturbance in June 2003. Dissolved P

in watershed B increased by 30% when comparing the samples collected in June and July of 2003 (Table 2), whereas the changes in dissolved P were relatively minor in watershed C for samples collected in June and July 2003. There were 36.9, 28.2, and 14.2 kg P discharged from large watersheds A, B, and C, respectively, from June 19 to July 14, 2003. Ammonium-N concentrations in ditch water were relatively low in watersheds A and B for post-disturbance samples. In watershed C, water at the small site was relatively high in $\text{NH}_4\text{-N}$; however, with increasing distance downstream, this concentration decreased by approximately 75% by the time it reached the large site. Nitrate-N concentrations in watershed A increased by more than 250% with increasing distance downstream, while in watersheds B and C, the $\text{NO}_3\text{-N}$ concentrations increased by more than 500% between small and large sites.

A reduction of 51% ($P < 0.001$) in exchangeable P (ExP) occurred between the small and large sites in watershed A (Table 3). As in watershed A, ExP was reduced by 52% ($P < 0.001$) as water flowed from the small to large site in watershed B. However, in watershed C, there was very little change in ExP between the small and large sites. Following treatment of sediments with alum, there was not a significant change in ExP in most of the samples between the alum treated sediments and the untreated sediments. In sediment samples taken from the small and X-large sites on ditch A however, there were significant reductions in ExP following alum treatment (38 and 11% reductions, respectively; $P < 0.05$).

The highest observed partitioning index was at the small site in watershed B, which contained low P in water and fairly high ExP in sediments (Table 3). In both watershed A and B, partitioning index values were reduced from the small to large sites; however, this reduction was only significant in watershed A ($P < 0.001$). Partitioning index values in watershed C were 10.5 at the small to large sites. As with the ExP, partitioning index values were only significantly reduced following alum additions in the small and X-large sites on ditch A.

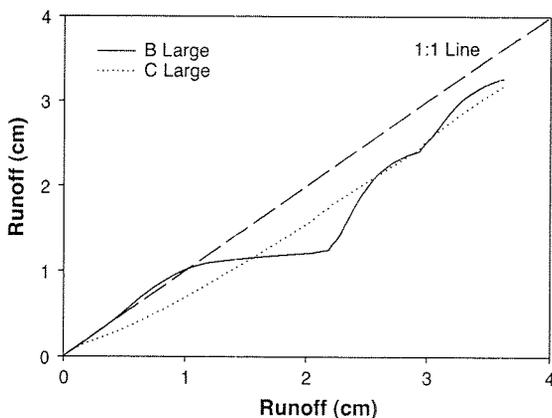


Fig. 3 – Cumulative runoff from watersheds B large (solid line) and C large (dotted line) regressed against the cumulative runoff from watershed A large. The dashed line represents the one to one line, or the line that would represent runoff from two watersheds being exactly the same.

Table 2 – Dissolved phosphorus and nitrogen concentrations in drainage ditches from water samples collected before and after sediment disturbance

Watershed	Site	Dissolved P		NH ₄ -N		NO ₃ -N	
		Pre-disturbance (mg L ⁻¹)	Post-disturbance (mg L ⁻¹)	Pre-disturbance (mg L ⁻¹)	Post-disturbance (mg L ⁻¹)	Pre-disturbance (mg L ⁻¹)	Post-disturbance (mg L ⁻¹)
A	Small	0.02	0.06	0.09	0.03	7.14	1.63
	Large	0.03	0.07	0.11	0.01	11.24	3.04
	X-large	0.05	0.06	0.17	0.02	11.99	4.35
B	Small	0.08	0.12	0.07	0.01	13.42	0.44
	Large	0.06	0.08	0.08	0.04	6.77	2.69
C	Small	0.06	0.06	0.12	0.22	2.25	0.16
	Large	0.09	0.06	0.12	0.06	5.66	2.93

Data from pre-disturbance are adapted from Smith et al. (2005).

Table 3 – Exchangeable P and partitioning index for untreated and alum treated ditch sediments from sites along tile-fed drainage ditches from three watersheds before and after sediment disturbance

Watershed	Site	Exchangeable P						Partitioning index					
		Pre-disturbance		Post-disturbance		Pre-disturbance		Post-disturbance		Pre-disturbance		Post-disturbance	
		Untreated (mg P kg ⁻¹)	Alum-treated (mg P kg ⁻¹)	Untreated (mg P kg ⁻¹)	Alum-treated (mg P kg ⁻¹)	Untreated (dimensionless)	Alum-treated (dimensionless)	Untreated (dimensionless)	Alum-treated (dimensionless)	Untreated (dimensionless)	Alum-treated (dimensionless)		
A	Small	4.79	2.50	1.01	0.63	240.0	125.0	16.2	10.1	125.0	16.2	10.1	
	Large	1.01	0.43	0.49	0.46	33.7	14.4	7.6	7.0	14.4	7.6	7.0	
	X-large	1.57	0.17	0.70	0.36	31.5	3.4	12.0	6.2	3.4	12.0	6.2	
B	Small	9.35	1.73	1.32	1.34	117.0	21.6	10.8	11.0	21.6	10.8	11.0	
	Large	0.49	0.59	0.63	0.56	8.1	9.9	7.7	6.9	9.9	7.7	6.9	
C	Small	1.72	1.08	0.61	0.71	28.7	18.0	10.5	12.5	18.0	10.5	12.5	
	Large	1.79	0.72	0.60	0.71	19.9	8.0	10.5	12.5	8.0	10.5	12.5	

Table 4 - Sediment equilibrium P concentration (sediment EPC_0) and the slope of the linear relation used to estimate sediment EPC_0 for untreated and alum-treated ditch sediments from three watersheds before and after sediment disturbance

Watershed	Site	Sediment EPC_0						Slope					
		Pre-disturbance			Post-disturbance			Pre-disturbance			Post-disturbance		
		Untreated ($mg P L^{-1}$)	Alum-treated ($mg P L^{-1}$)	Alum-treated (mg $P L^{-1}$)	Untreated ($mg P L^{-1}$)	Alum-treated ($mg P L^{-1}$)	Alum-treated ($mg P L^{-1}$)	Untreated ($L kg^{-1}$)	Alum-treated ($L kg^{-1}$)	Alum-treated ($L kg^{-1}$)	Untreated ($L kg^{-1}$)	Alum-treated ($L kg^{-1}$)	Alum-treated ($L kg^{-1}$)
A	Small	0.078	0.017	0.035	0.033	0.016	1.34	1.37	1.37	8.57	7.35	7.35	
	Large	0.055	0.008	0.016	0.020	0.016	0.53	0.57	0.57	3.25	5.08	5.08	
	X-large	0.067	-0.002	0.027	0.107	0.027	0.41	0.52	0.52	2.72	5.14	5.14	
B	Small	0.050	-0.023	0.034	0.080	0.034	1.07	1.07	1.07	13.80	26.10	26.10	
	Large	-0.020	-0.001	0.071	0.023	0.071	0.60	0.59	0.59	7.38	7.59	7.59	
C	Small	0.051	0.004	0.052	0.035	0.052	1.85	4.18	4.18	11.60	13.20	13.20	
	Large	0.110	-0.011	0.043	0.068	0.043	0.52	0.50	0.50	4.92	4.89	4.89	

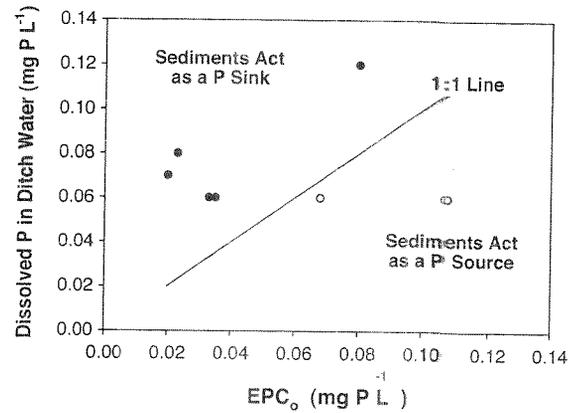


Fig. 4 - Dissolved P in the ditch water plotted as a function of sediment equilibrium P concentration for sediments that act as a potential P source (○) or P sink (●).

Equilibrium P concentrations (EPC_0) in watershed A ranged from 0.02 to 0.11 $mg L^{-1}$ (Table 4). There were small differences in watershed A EPC_0 values between the small and large sites; however, EPC_0 at the X-large site was more than five times higher than at the large site. In watershed B, a 75% reduction in EPC_0 values occurred between the small and large sites. The EPC_0 nearly doubled in watershed C between the small and large sites. Inconclusive results were obtained following application of alum to sediments. At the small and large sites in watershed A there was very little change in EPC_0 when alum was applied to sediments. Alum treatment decreased EPC_0 in the X-large site in watershed A (75%), the small site in watershed B (58%), and the large site in watershed C (37%), while there was an increase due to alum application in the large site on watershed B (209%) and the small site in watershed C (49%).

The P buffering capacity of sediments ranged from 2.7 to 13.8 (Table 4). In all three watersheds, the P buffering capacity decreased with increased distance downstream. In watershed A, the P buffering capacity decreased by 68% between the small site and the X-large site. In watershed B, the P buffering capacity decreased by 6.4 $L kg^{-1}$ between the small and large sites, while there was a 6.7 $L kg^{-1}$ decrease between small and large sites in watershed C. Alum treatment increased the P buffering capacity of sediments in four of the seven sites, decreased the P buffering capacity at the small site on watershed A, and had very little effect at the large sites on watersheds B and C.

Soluble P in ditch water was plotted against the EPC_0 of sediments (Fig. 4). Using a 1:1 line as a guide, sediments with EPC_0 values that plot above the line will theoretically act as a sink for water column P, while values that plot below this line indicate that sediments may be a potential P source to the water column. At five of the seven sites, the P concentrations in ditch waters were greater than sediment EPC_0 at those sites, indicating that P transport downstream should be retarded as the sediments adsorb P from the water column. Sediment EPC_0 was greater than P levels in ditch waters at the other two sites (X-large site on watershed A and large site on watershed C), suggesting that P could be released from the sediments thereby increasing the P concentration in ditch water (see McDowell and Sharpley, 2003). Sediment EPC_0 and water

column P concentrations had not reached equilibrium at any of the sites following this series of storm events.

4. Discussion

4.1. Water chemistry

Nitrogen and dissolved P concentrations were different following this series of storm events compared to what was measured a month before in June 2003. Nitrogen ($\text{NH}_4\text{-N}$ and $\text{NO}_3\text{-N}$) was generally less in this study than that reported in Smith et al. (2005), whereas dissolved P concentrations were greater in the current study. Nitrate-N and $\text{NH}_4\text{-N}$ were likely different because stream discharge in the current study was 1.3 times that measured in June 2003, where dilution likely was responsible for some of the decrease. The current study collected water samples on the falling side of a large storm event, and dissolved N stored in the tile drained upland soils may be transported during a first flush phenomenon which may result in decreased concentrations once the stream returns to baseflow. $\text{NH}_4\text{-N}$ concentrations tended to decrease as water flowed downstream, whereas $\text{NO}_3\text{-N}$ concentrations tended to increase, suggesting nitrification could be playing an important role in the N transport of these ditches.

Phosphorus is not as mobile as $\text{NO}_3\text{-N}$, so transport of P in drainage ditches may be delayed compared to N, especially $\text{NO}_3\text{-N}$. Furthermore, this study collected water samples at a point in the hydrograph with elevated flow, when water column P may not interact with bottom sediments as readily. Thus, the elevated P concentration in ditch water was likely due to the drainage from the field tiles. This hypothesis is supported by the fact that P in ditch water was greater than bottom sediment EPC_o in July 2003, indicating that the sediments probably are not contributing to dissolved P transport. This is in contrast to the results obtained in June 2003 (Smith et al., 2005), where sediment EPC_o tended to be greater than P concentrations in the ditch water. Under these conditions, sediments were a likely source of dissolved P to the water column (see Smith et al., 2005 for more details).

4.2. Ditch bottom sediment phosphorus

Bottom sediment ExP was generally less at all sites in July 2003 compared to that observed in June 2003 by Smith et al. (2005). The combination of greater water column P and less sediment ExP resulted in partitioning indices as much as an order of magnitude lower compared to values reported in June 2003 (see Smith et al., 2005). These data support the hypothesis that fresh sediments were deposited during the episodic runoff from the storms, and that the sources of P in water shifted from internal sources in June 2003 to external sources from the tiled fields in July 2003. It may be expected that the sediment ExP would be high following chronic exposure of sediments to dissolved P from the tiles fields, whereas sediment ExP should be reduced with fresh sediment deposition after P-enriched bottom sediments are resuspended and transported downstream. Sediment disturbance is a source of total suspended solids and turbidity in the water column, and as such can be detrimental to species dependant on the pen-

etration of light into the water column. The data presented here are evidence that periodic events that disturb sediments can also be ecologically beneficial, as the sediments present following the disturbance can act as a sink for contaminants, such as P, thereby buffering the contaminant concentrations in the water column.

Differences between sediment EPC_o in the current study and that observed in June 2003 before the series of storm events were somewhat variable. Overall, sediment EPC_o in the current study was within the same range (-0.02 to 0.11 mg L^{-1}) reported in June by Smith et al. (2005). It may be expected that deposition of fresh sediment would result in reduced sediment EPC_o ; however, this was not the case in these ditch sediments or at least this was not a consistent finding across the drainage ditches. The general trends in sediment EPC_o remained the same before and after the series of storms.

While sediment EPC_o may have been in the same range following the series of storms, the ability of the sediments to adsorb P changed dramatically following deposition of fresh sediments. The slopes of the linear sorption isotherms used to calculate sediment EPC_o increased an order of magnitude following the deposition of fresh sediments, indicating the P buffering capacity dramatically increased in July 2003 compared to that observed in June 2003 (see Smith et al., 2005).

4.3. Alum treatment of sediments

Addition of alum to sediment in the current study did not have a consistent effect on sediment EPC_o and ExP, whereas before sediment disturbance, alum treatment of sediment resulted in decreased sediment EPC_o and ExP (see Smith et al., 2005). The treatment of sediments with alum would be expected to produce results similar to the previous study by Smith et al. (2005). Sediments that have been exposed to prolonged elevated P concentrations, e.g. from municipal effluent discharges, have shown substantial decreases in sediment EPC_o and ExP (Haggard et al., 2004). Similar results have been reported in studies treating reservoir bottom sediments with alum to reduce P flux, especially under anaerobic conditions (Haggard et al., 2005; Welch et al., 1982; Welch and Schriever, 1994). Thus, alum treatment should be an effective means to reduce internal P sources in streams and reservoirs.

Since the current study did not show consistent effects of alum, it appears that fresh sediments were likely deposited, whereas the previous sediments that had been exposed to P sources were resuspended and transported further downstream. It appears that the natural process of sediment resuspension and deposition provides an important ecological benefit by temporarily reducing the downstream delivery of P through sequestration by sediments. Furthermore, the current study suggests that in-stream best management practices increasing P sequestration by sediments should be temporarily targeted and should focus on periods of sustained baseflow and smaller storm events. Chemical treatment of sediments immediately prior to a disturbance event would result in the removal of the alum-amended sediments and diminished benefit from this practice. The effects of such a situations should be explored because while the benefit with regard to in-stream dissolved P retention would not be evident, the transport of alum amended sediment may benefit

downstream aquatic systems. These results also indicate that physical treatment of sediments for the removal of P from the water column should also be explored. Since ditch dredging is a common practice in the tile-fed drainage ditches of northeast Indiana, and the impact of this physical sediment treatment on P transport within these drainage networks should be studied.

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