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Invited paper

Nutrient mitigation capacity in Mississippi Delta, USA drainage ditches

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Agricultural drainage ditches reduce N and P runoff concentrations from production landscapes.

A B S T R A C T

Eutrophication and hypoxia within aquatic systems are a serious international concern. Various management practices have been proposed to help alleviate nutrient loads transported to the Gulf of Mexico and other high-profile aquatic systems. The current study examined the nutrient mitigation capacity of a vegetated (V) and non-vegetated (NV) agricultural drainage ditch of similar size and landform in the Mississippi Delta. While no statistically significant differences in ammonium, nitrate, or dissolved inorganic phosphorus mitigation between the two ditches existed, there were significant differences in total inorganic phosphorus percent load reductions (V: 36% ± 4; NV: 71% ± 4). However, both agricultural drainage ditches were able to mitigate nutrients, thus reducing the load reaching downstream aquatic receiving systems. Further studies examining ecosystem dynamics within drainage ditches such as sediment and plant nutrient partitioning, as well as microbial processes involved, are needed to provide a better understanding of natural nutrient variability, seasonality and flux.

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1. Introduction

According to the US Census Bureau (2009), the global population increases by nearly three individuals each second. This translates into an additional 79.4 million individuals annually who need food and fiber for continued survival. From 1950 to 2000, the earth's population more than doubled (Carvalho, 2006). Over the past 35 years, there has been a doubling of agricultural food production resulting in not only water quality concerns, but also land and fertilizer use issues. Tilman (1999) noted that over the last 35 years there has been an almost 7-fold increase in nitrogen (N) fertilization, nearly 3.5-fold increase in phosphorus (P) fertilization, and a 1.1-fold increase in land in cultivation. The next doubling in food production, anticipated to occur during the next 50 years, will likely triple the annual rates of N and P released in the world, require 18% more arable land than is currently in production, and be a major driver in environmental change (Tilman, 1999; Tilman et al., 2001).

In 2000, agricultural N fertilizer consumption for the world was a reported 81.7 million tons (Fixen and West, 2002). Such fertilizer is the single largest source of reactive N (e.g. ammonia, ammonium, nitrate, nitrite, etc.) not only in the world, but also in the US (Howarth et al., 2002). On a per-capita basis, the US consumes 3-fold greater inorganic N fertilizer than the global average (Howarth et al., 2002). It is this reactive N which is partly responsible for eutrophication and hypoxia of waterways, as well as loss of biodiversity, habitat degradation, shifts in food chain structure, fisheries impairment, and outbreaks of nuisance aquatic species (Tilman, 1999; Galloway et al., 2003).

Within the fertile Mississippi River Alluvial Valley, one particular form of reactive N, nitrate (NO_3^-), has increased by factors of two to sometimes more than five in surface waters since the early 1900s (Goolsby et al., 2000). Turner and Rabalais (1991) reported that of the annual dissolved N in the Mississippi River, 53% is NO_3^-, 43% is organic N, and 4% is ammonium (NH_4^+). Approximately 1.82 Tg N yr^{-1} is delivered to the northern Gulf of Mexico by the Mississippi River (Howarth et al., 1996; Dagg and Breed, 2003). Although much attention is given to N inputs because of the Gulf of Mexico hypoxia issue, P transport from agricultural fields is also of concern. Based on a global budget approach, Bennett et al. (2001) estimated a 75% net P storage increase in terrestrial and freshwater ecosystems as compared to preindustrial storage levels. More than 90% of the P load in about one-third of US rivers and streams can be attributed to non-point source pollution, which includes agriculture (Newman, 1996). Model simulations conducted by Alexander et al. (2008) revealed that agricultural sources in watersheds contributed more than 70% of delivered N and P. Delivery of P vastly differs on local and regional scales, because of reservoir trapping

(Alexander et al., 2008). Accumulation of nutrients, such as N and P, may not initially result in the appearance of adverse freshwater ecosystem effects. Years may pass between soil accumulation and the appearance of actual ecosystem effects (Reed-Anderson et al., 2000). Likewise, an abrupt appearance of ecosystem effects may occur if a gradual increase in nutrient accumulation passes a threshold (Heckrath et al., 1995). Eutrophication of freshwater systems may be accelerated by an influx of P from agricultural catchments, therefore, specific management practices are needed at the edge-of-field to help remediate runoff (McDowell et al., 2003). To achieve efficient control of Gulf nutrient loads, a diversified management approach is needed which takes into account N and P sources (both terrestrial and atmospheric) and downstream reservoir P sources (Alexander et al., 2008).

One such management practice being closely examined is the use of vegetated agricultural drainage ditches for nutrient mitigation. While studies have demonstrated vegetated ditches' ability to mitigate pesticides (Moore et al., 2001; Cooper et al., 2002, 2004; Bennett et al., 2005), only recently have analyses been conducted on ditch nutrient mitigation capacity (Kröger et al., 2007, 2008). In fact, early reports on drainage ditches conducted by the USDA referred to vegetation as little more than "the most common form of obstruction" (Ramser, 1947). As time has passed, so has our understanding of the vegetated drainage ditch. Ramser (1947) commented that "the presence of vegetation in a drainage ditch indicates that the landowner is not deriving the full benefit…for which the ditch was intended." The focus on ditch ecosystems now includes environmental benefits, in addition to, the drainage advantages offered. The current study sought to compare the differences between a vegetated versus non-vegetated agricultural drainage ditch (of equal shape and size) in reducing concentrations and loads of ammonium, nitrate, nitrite (NO_2^-), total Kjeldahl nitrogen (TKN), dissolved inorganic orthophosphate (DIP), total inorganic phosphorus (TIP), total phosphorus (TP), total solids (TS), suspended solids (SS), and dissolved solids (DS).

2. Materials & methods

2.1. Study site location

Two agricultural drainage ditches (V: vegetated; NV: non-vegetated) within the Delta Conservation Demonstration Center (DCDC) (Metcalf, Mississippi, USA) were chosen for this nutrient mitigation comparison experiment. Each ditch was approximately 320 m in length, with a mean top width of 6 m (V: 6 ± 0.26 m; NV: 5.9 ± 0.31 m) and mean bottom width of 3.2 m (V: 3.2 ± 0.24 m; NV: 3.13 ± 0.21 m). Both drainage ditches were hydro-geomorphically similar in length, width and slope. The southernmost ditch lacked in-stream vegetation (non-vegetated), while the northernmost ditch was vegetated with a mixture of *Typha latifolia* L. (cattail), *Sparganium americanum* Nutt. (American bur-reed), and *Juncus effusus* L. (soft rush). The soils of the DCDC are classified Sharkey (very-fine, montmorillonitic, nonacid, thermic, Vertic Haplaquept) and Dundee series (fine-silty, mixed, thermic Typic Endoaqualfs).

2.2. Simulated storm event

A single simulated storm runoff (nutrients and sediment: duration = 7 h) was introduced to the non-vegetated ditch in mid-April 2006 and subsequently to the vegetated ditch in early May 2006. Sources of nitrogen (N) and phosphorus (P) in simulated runoff were Green Fields™ 34-0-0 (17% NH_4; 17% NO_3) and Triple Super Phosphate™ 0-46-0 (46% P_2O_5), respectively. Because of the large volume of nutrient amendment needed for each exposure, two 190 L chambers were used to mix identical doses of 26.4 kg of Green Fields™ fertilizer and 10 kg of Triple Super Phosphate™. Two Fluid Metering Inc. (FMI™) piston pumps, model QD-1, each calibrated at 450 mL min⁻¹ delivery were used to transfer nutrient slurry contents from mixing chambers to the exposed drainage ditch during the 7 h exposure (Fig. 1). Target ditch concentrations for NO_3^-, NH_4^+, and orthophosphate (PO_4) were 5 mg L⁻¹. To accurately simulate storm runoff components, a sediment slurry was also amended into each ditch during its respective exposure. Using a 3800 L mixing tank and an Atwood™ V500 (~1900 L h⁻¹) bilge pump for delivery, a staggered sediment slurry amendment was made over 7 h in both experiments (Table 1). In addition to the nutrient and sediment slurry, during the 8 h experiment,
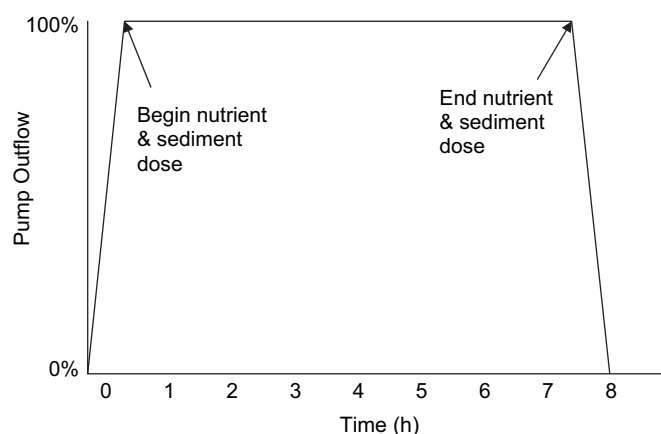

Fig. 1. Hypothesized chemograph for sediment and nutrient dose for the vegetated and non-vegetated drainage ditches at DC/DC.

groundwater was pumped into the ditch inflow at a rate of 1135 L min⁻¹ for the first 30 min (prior to nutrient-sediment addition), followed by a rate of 2270 L min⁻¹ for the next 7 h (nutrient-sediment addition), and concluded with a rate of 1135 L min⁻¹ for the final 30 min (no nutrient-sediment addition). This groundwater component simulated the storm water runoff typically generated off agricultural fields surrounding the respective drainage ditches.

2.3. Water, sediment, and plant sample collection

Grab samples of ditch water were collected in 230 mL polyethylene cups (Fisher Scientific, Pittsburgh, PA) at 30 min intervals for 8 h, then again at 10 h, 16 h, 24 h, and 48 h at each of the eight spatial sampling locations (0, 5, 10, 20, 40, 80, 160, and 320 m), along both vegetated and non-vegetated drainage ditches. Collected samples were stored on ice immediately and returned to the laboratory for nutrient analysis within 24 h of collection. Sediment samples (top 2.5 cm; approximately 200 g wet weight) were collected in both the non-vegetated and vegetated ditches at 1, 3, 6, 10, 16, 24, and 48 h post-event at the same locations of water samples. The sediment sampling protocol was similarly used to gather plants in the vegetated ditch. Approximately 250 g (wet weight) of plant material was collected at each of the eight spatial sampling locations. A representative collection of plants from each sampling location were collected at each specified timeframe. Only the portion of the plant exposed in the water column was saved for nutrient analysis. Both sediment and plant samples were wrapped in heavy duty aluminum foil, stored on ice, and transported to the laboratory within 24 h of collection. Upon arrival, samples were stored in a freezer (−20 °C) and subsequently air dried in a greenhouse to a constant weight for nutrient analyses. Water samples for solid determination were collected at all sampling locations in both the vegetated and non-vegetated ditches at 0 h, 1 h, 4 h, 8 h, 16 h, 24 h and 48 h. Groundwater samples were taken prior to the simulated amendment experiment to determine background N and P levels.

2.4. Nutrient analyses

Water samples were analyzed for NH_4^+, NO_3^-, NO_2^-, DIP, TIP, TKN, TP, TS, SS, and DS. Sediment and plant samples were analyzed for TKN and TP only. The cadmium reduction method was used to analyze NO_3^-, whereas NH_4^+, DIP and TIP were analyzed using the phenate and ascorbic acid methods, respectively, according to standard methods (Murphy and Riley, 1962; APHA, 1998). Solids analyses were likewise performed according to standard methods (APHA, 1998). Total solids were a summation of dissolved and suspended solids. All analyses were performed using a ThermoSpectronic Genesys 10 ultraviolet (UV) spectrophotometer. Sediment (10 mg), plant (20 mg) and water (20 mL) TP and TKN were determined by digesting

Table 1
Staggered sediment amendment dosing schedule for both vegetated and non-vegetated drainage ditches, 2006.

Time	Concentration
0.5–1.5 h	63 mg L⁻¹
1.5–2 h	130 mg L⁻¹
2–3 h	183 mg L⁻¹
3–5 h	250 mg L⁻¹
5–6 h	183 mg L⁻¹
6–6.5 h	130 mg L⁻¹
6.5–7.5 h	63 mg L⁻¹

the respective samples with sulfuric acid in a micro-Kjeldahl block digester. Mercuric oxide was used as a catalyst, and potassium sulfate was added to raise the boiling temperature of the digestion and speed up the conversion. The digest was diluted with 20 mL of deionized (D.I.) water, and run on a QuikChem Lachat FIA+ 8000 series. The Lachat was fitted with an autosampler (ASX 500 series), a reagent pump (RP-100 series) and a dual resolution diluter. Nitrite concentrations were determined on a Dionex Ion Chromatograph (IC) fitted with a IonPac ASH-HC anion exchange column. The IC system also contained an EC40 Eluent generator, GP50 gradient pumps and a CD25 conductivity detector. Chromeleon chromatography workstation interfaced with the IC for data analysis. All sediment, plant and water samples for both the IC and Lachat were bracketed by a multi-level calibration using five standards. Mid-level standards and D.I. blanks were sampled every 25th sample to check shifts in retention time.

2.5. Statistical analyses

Reductions in individual nutrient and sediment concentrations found in vegetated and non-vegetated drainage ditches following the simulated storm event were compared using standard student *t*-tests (two tail, unequal variance). All data were log-transformed to meet the assumptions of normality for the *t*-test. Linear regressions determined trends between percent nutrient (and sediment) concentrations and load reductions versus time along each respective drainage ditch. Loads were calculated by multiplying the influent and effluent concentrations (mg L^{-1}) by the volume (L) of water discharged over a set time period. Reductions were based on instantaneous values collected at ditch inflows and outflows. All values were significant at an alpha of 0.05.

3. Results and discussion

3.1. Sediment load reductions in drainage ditches

Sediment loads were amended to both vegetated and non-vegetated drainage ditches at a mean dose of 143.14 ± 35.7 mg L^{-1}, with a maximum targeted concentration of 250 mg L^{-1} between 3
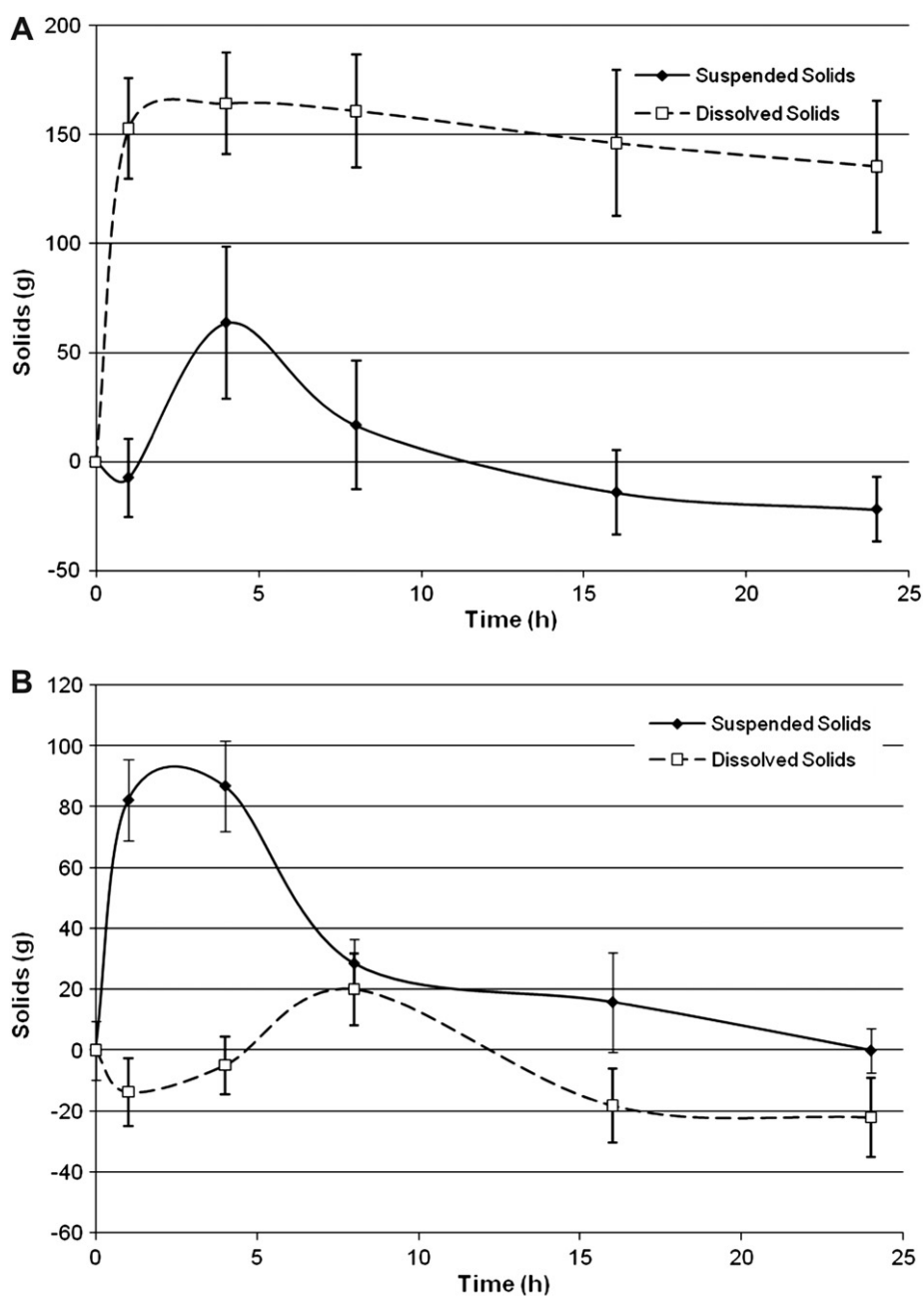

Fig. 2. Mean suspended and dissolved solids (g) for A) vegetated and B) non-vegetated ditches for all transects sampled per time interval (0–24 h). Total solids are the sum of suspended and dissolved solids. Mean values are based on the difference (+ve or −ve) between observed background levels and measured samples.

and 5 h (Table 1). Initial background total solid loads were significantly different between vegetated (431 $\pm$ 9.24 g) and non-vegetated (289 $\pm$ 22.5 g) ditches. As a result concentrations could not justifiably be compared; however, the percentage of suspended to dissolved solids would provide insight to sediment dynamics. Fig. 2 shows mean solid load across all transects per time interval sampled per ditch, based on an increase or decrease from the observed background levels. In the vegetated ditch (Fig. 2A) dissolved and suspended solid loads peaked at 4 h, with dissolved solid component remaining significantly higher than suspended solids for the duration of the amendment. The proportion of suspended solid to total solid component at the height of the sediment slurry amendment was less than 28% for the vegetated ditch. Similarly, in the non-vegetated ditch (Fig. 2B), suspended solids peaked at 4 h; however, in contrast, dissolved solids peaked at the end of the storm amendment (7 h). Unlike the vegetated ditch, the suspended sediment component comprised >95% of the total solid proportion for the duration of the amendment in the non-vegetated ditch. This proportional comparison highlights the lack of suspended sediments within the vegetated drainage ditch, suggesting an increase in sedimentation. It is well known that in-stream vegetation will increase friction and roughness of stream channels, thus decreasing water velocity, increasing sedimentation and decreasing suspended sediments from the overlying water column (Dieter, 1990; Abt et al., 1994; Braskerud, 2002; Schoonover et al., 2006).

3.2. Nitrogen species reductions in drainage ditches

Prior to amendment background concentrations of NH_4^+ (V: 0.409 $\pm$ 0.08 mg L^{-1}; NV: 0.08 $\pm$ 0.12 mg L^{-1}), NO_3^- (V: 6.2 $\pm$ 1.06 mg L^{-1}; NV: 0.57 $\pm$ 0.09 mg L^{-1}) and NO_2^- (V: 0.392 $\pm$ 0.06 mg L^{-1}; NV: 0.03 $\pm$ 0.003 mg L^{-1}) were significantly higher ($p \leq 0.0001$) in vegetated than non-vegetated ditches. These differences were attributed to a rainfall event ($\pm$April 5th) prior to the 2 experiments (NV: April 9th; V: May 8th). The non-vegetated ditch contained water at the start of the experiment, but the water was a result of a rainfall event four days prior to the experiment, thus containing relatively low concentrations of N species. The significantly higher concentrations in the vegetated ditch prior to the introduction of the simulated storm water amendment, were hypothesized as a result of the water being reduced in volume through time through evapotranspiration, thus concentrating in situ N species. Increased background concentrations for N in the vegetated ditch significantly skewed the predictions for reduction of N species for the first six sampling time intervals (0–2.5 h). Thus all figures, and load reductions referred to are based on measured values between 2.5 h and 8 h of the amendment. This was acceptable as F^- tracer evaluations confirmed retention times of between 2 and 2.5 h for the respective vegetated and non-vegetated ditches (Kröger et al., 2009). However, inflow nutrient loads and concentrations originally calculated to be similar between both vegetated and non-vegetated ditches were significantly higher in the non-vegetated ditch for NH_4^+ ($p \leq 0.0001$) and NO_3^- ($p \leq 0.01$). This difference was attributed to subjective delivery and human error. This limited statistical comparisons of the original values and concentrations between the two systems, but still allowed comparisons between overall percentage reduction capacities of the ditches.

Ammonium concentrations peaked within the vegetated ditch at 2.5 h, and within the non-vegetated ditch at 7.5 h. Fig. 3 describes the percentage reduction of NH_4^+ as the difference between inflow (0–20 m) and outflow (320 m) concentrations at each time interval sampled. Through time, both vegetated and non-vegetated ditches had variable outflow reductions of NH_4^+ from influent concentrations (Fig. 3). Such variability was also noted by Greenway and Wooley (1999) who reported 8–95% removal of NH_3^--N in nine wetlands located in Queensland, Australia, used to treat municipal wastewater. The percentage difference decreased between inflow and outflow in the vegetated ditch ($r^2 = 0.672$), while there was only a slight downward trend in NH_4^+ reduction for the non-vegetated ditch ($r^2 = 0.25$). The trend was similar (as shown by regressions with similar gradients), but with less variability in the vegetated pond. The sinuous NH_4^+ reduction distribution (Fig. 3) for the non-vegetated ditch was hypothesized to be associated with fluxes and pulses of suspended sediment. Suspended sediment has a strong cation exchange capacity and thus a high suspended sediment pulse would have a concomitant high NH_4^+ concentration.
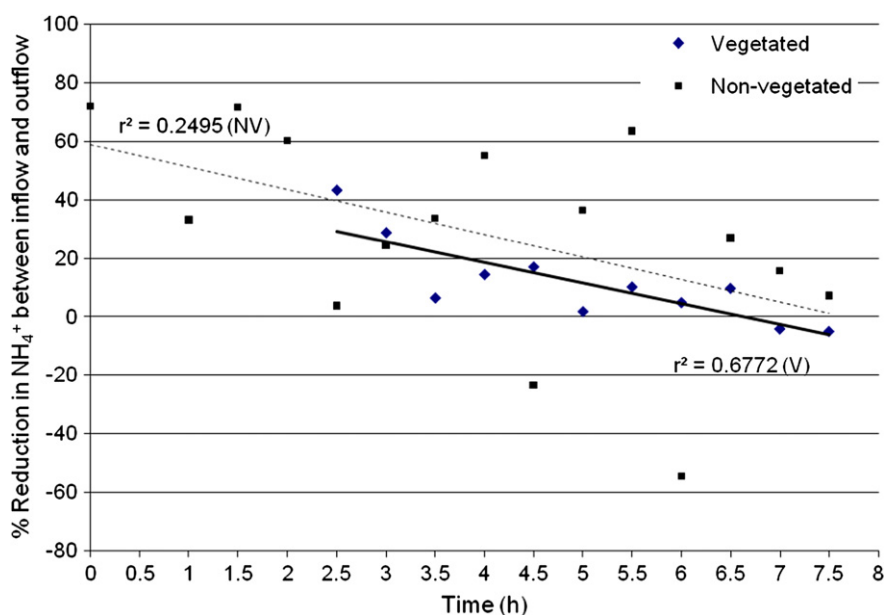

Fig. 3. Percentage reduction of NH_4^+ between mean inflow (0–20 m) and outflow (320 m) concentrations for respective time intervals for both vegetated (solid line) and non-vegetated (dashed line) drainage ditches. A +ve value indicates a lower outflow concentration than inflow; while a −ve value denotes a higher outflow concentration than inflow.

Averaging all time steps, the mean percentage (%) NH_4^+ reductions for the vegetated ditch (11.67 ± 4.28) and non-vegetated (19.05 ± 10.09) were not significantly different ($p = 0.512$). Few, if any, drainage ditch studies exist to allow further comparisons between vegetated systems and non-vegetated systems. Research primarily has been on constructed wetland efficiency of NH_4^+ removal. Jordan et al. (2003) reported 25% NH_4^+ removal in a two-year study of a 1.3 ha wetland receiving inflow from a 14 ha agricultural watershed. Tanner et al. (1999) examined NH_4^+ removal efficiencies of gravel-bed constructed wetlands either planted (with *Schoenoplectus tabernaemontani*) or unplanted when exposed to low to zero water level fluctuations. Results from that study indicated unplanted mesocosms removed only 10–28% of NH_4^+, while planted mesocosms removed 54–71% of NH_4^+ (Tanner et al., 1999). Removal of NH_4^+ in Tanner et al. (1999) was described by a sorption – plant uptake microbial model ($r^2 = 0.97$–0.99). It is difficult to compare the current study with that of Tanner et al. (1999) especially since microbial assessments were not part of the current study.

Nitrate concentrations peaked in the vegetated ditch at 3 h, with concentrations slowly decreasing with time over the course of the amendment ($r^2 = 0.1751$). The non-vegetated ditch had a double pulse of high NO_3^- concentrations (3 h; 4.32 ± 0.3 mg L^{-1} and 6.5 h; 4.93 ± 0.68 mg L^{-1}), with NO_3^- concentrations returning to background levels at 8 h. Nitrate reduction percentage distributions were different between vegetated and non-vegetated ditches (Fig. 4). The vegetated NO_3^- reduction distribution showed a steady decline through time ($r^2 = 0.6225$), a small variability in reduction percentage and a mean percentage reduction of 3.16% ± 3.86. Similarly, the non-vegetated NO_3^- reduction distribution showed variability between time intervals ($r^2 = 0.532$) and showed decreasing trend in NO_3^- percentage reductions through time. The average percent NO_3^- reduction for the non-vegetated ditch (3.37 ± 12.6) was not significantly different from the vegetated ditch ($p = 0.8$) but was significantly more variable in output. Wetland microcosms have demonstrated varied NO_3^- removal efficiencies (8–95%) depending on changing hydraulic loading and carbon addition rates (Ingersoll and Baker, 1998). Kovacic et al. (2000) reported a 28% decrease in NO_3^--N concentrations from inflow to outflow in a constructed wetland. Other studies have reported between 52 and 90% NO_3^- removal through the use of wetlands (Jordan et al., 2003; Fink and Mitsch, 2007; Scott et al., 2008). Earlier studies have reported success with removal of NO_3^- in vegetated buffers, albeit not vegetated drainage ditches. Switchgrass (*Panicum virgatum*) buffers have been reported to remove 62% of NO_3^--N, while the combination of switchgrass and woody buffers increased NO_3^--N removal to 85% (Lee et al., 2003).

Interestingly, sediment TKN concentrations increased significantly with time in the vegetated ditch ($r^2 = 0.9526$, $p \leq 0.0001$) (Fig. 5), while there was no change in TKN concentrations in the non-vegetated ditch sediments. Concomitantly, the relative percent mass contribution of water, sediment and plant highlights this sediment difference (Table 2A). In comparison, there were higher relative load percentages of TKN ($NO_3^- + NO_2^- + NH_4^+$) in the water column of the non-vegetated ditch as compared to the vegetated ditch as a result of the lack of N sorbed to non-vegetated sediments (Table 2A). Overall, wetland macrophytes contributed a small portion to TN load reduction (Table 2A). This differs from results reported by Rogers et al. (1991) who determined that plant uptake was responsible for more than 90% of N removal. Again, this study and the current study differed greatly in system substrates. Rogers et al. (1991) utilized a gravel bed system, while the current study was conducted within a sediment substrate system. Table 3 compares the relative percentage load reductions of the individual nitrogen species between non-vegetated and vegetated ditches. The vegetated ditch had a significantly higher relative reduction of TKN loads (92%) than the non-vegetated ditch (77%, $p = 0.04$). According to Silvan et al. (2004), wetland vegetation is a significant factor in nutrient retention, with 70% of added N retained in plant material. Increased N removal in vegetated versus similar but non-vegetated systems was also noted by several studies (Bowmer, 1987; Gersberg et al., 1989; Rogers et al., 1991; Farahbakhshazad et al., 1995; Kadlec, 1995; Soto et al., 1999; Yang et al., 2001; Martin et al., 2003).

Both systems significantly increased NO_2^- loads in the outflow. Nitrite is an intermediary by product of denitrification in reduced conditions as well as nitrification in oxidized conditions. The high N loadings from amendments and the shallow, oxidized water column environment, potentially significantly boosted nitrification and denitrification potentials in both vegetated and non-vegetated ditches. Nitrification and denitrification are considered responsible
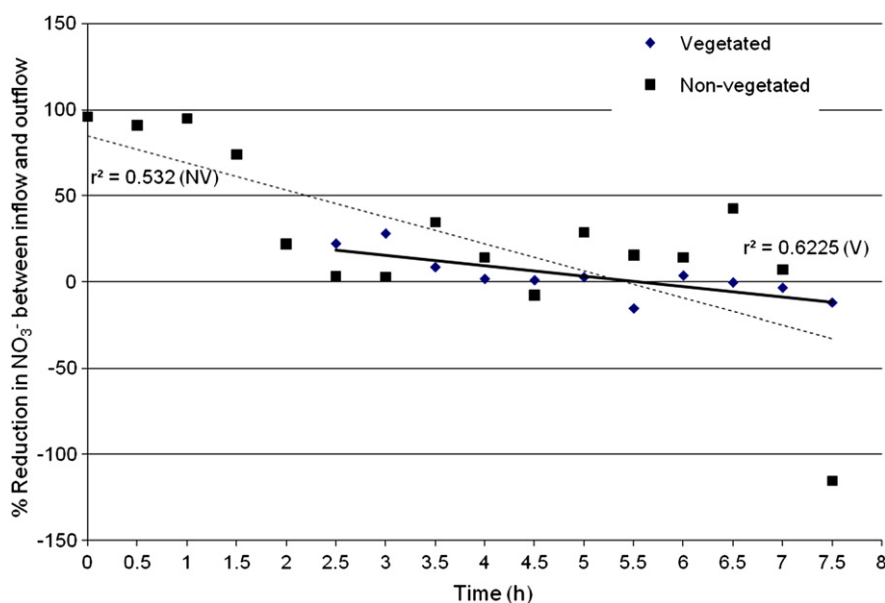

Fig. 4. Percentage reduction of NO_3^- between mean inflow (0–20 m) and outflow (320 m) concentrations for respective time intervals for both vegetated (solid line) and non-vegetated (dashed line) drainage ditches. A +ve value indicates a lower outflow concentration than inflow; while a −ve value denotes a higher outflow concentration than inflow.
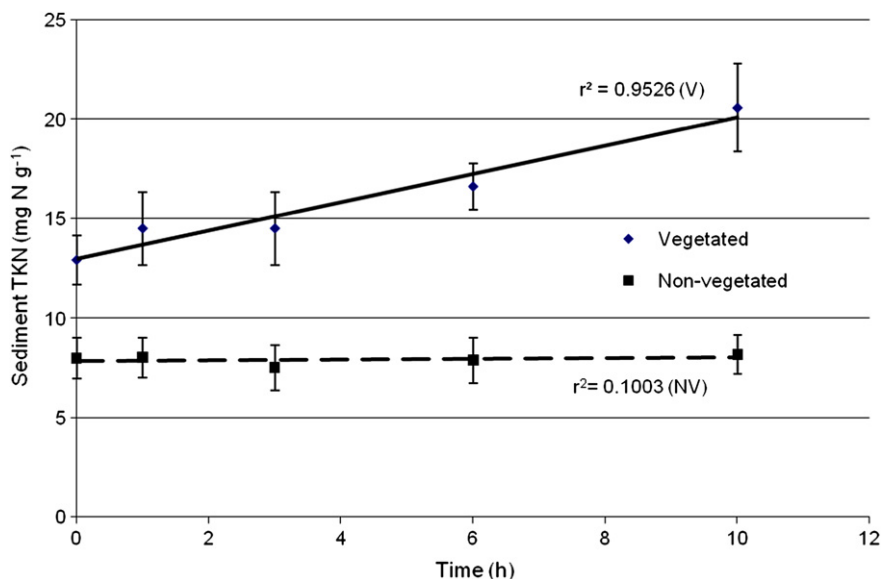
Fig. 5. Mean sediment TKN concentrations (mg N g^{-1}) for all sampling locations within the vegetated (solid line) and non-vegetated (dashed line) ditches through time.

for controlling N turnover in systems, and these processes are generally associated with sediment, macrophyte, or algal microbial communities (Bastviken et al., 2003). Microbial biofilms associated with vegetation provide additional surface area and may boost denitrification efforts in various aquatic systems (Eighmy and Bishop, 1989; Eriksson and Weisner, 1999; Körner, 1999). According to Seitzinger (2008), denitrifying bacteria present in anoxic sediments of aquatic systems play the most significant role in long term N removal. Ullah and Faulkner (2006) found denitrification potentials of vegetated ditches in the Mississippi Delta were 1.3 times higher than non-vegetated ditches.

3.3. Phosphorus species reductions in drainage ditches

Few studies have quantified effects of storm size on P loss. Sharpley et al. (2008) examined 248 storm flows over a 10-year period on a sub-watershed in Pennsylvania, determining that 93% of storms had a return period (i.e. chances of event occurring) of <1 year. Data from that study indicated that in those <1 year return storm events, 63% of the total flow was exported, while 54%, 45% and 47% of dissolved reactive P (DRP), particulate P (PP), and TP were exported, respectively (Sharpley et al., 2008). There were no significant differences ($p = 0.255$) in background TP concentrations between non-vegetated and vegetated systems, though the vegetated system had elevated DIP concentrations (V: 0.583 mg L^{-1}; NV: 0.488 mg L^{-1}). Again, through delivery error, inflow TP concentrations were significantly higher (NV: 5.34 ± 0.29 mg L^{-1}) in the non-vegetated ditches than the vegetated ditches (V: 3.23 ± 0.258 mg L^{-1}).

Several studies have examined the effectiveness of P removal in constructed wetlands, which share similar characteristics with vegetated drainage ditches (hydroperiod, hydric soils and hydrophytes). Although several studies have reported P removal percentages from 29 to 65% (Hoagland et al., 2001; Gu and Dreschel, 2008), Kovacic et al. (2000) saw only 2% P removal with highly variable results among treatment wetlands over the three year study. When examining TP removal, studies reveal consistently effective results ranging from 21% up to 93% depending upon location, hydraulic loadings, and other site-specific factors (Tanner, 1996; Braskerud, 2002; Fink and Mitsch, 2004, 2007).

As seen with nitrogen species, TP concentrations were significantly elevated between 0 and 2.5 h within the vegetated ditch due to evapotranspiration, concentrating the P within the water column. However, unlike N, high DIP background levels also occurred in the non-vegetated ditch. This was likely due to DIP desorbing from sediments with the recent rainfall, elevating background DIP concentrations in the overlying water column. Mean DIP concentrations peaked at 2.5 h within the vegetated ditch, and at 2.5 and 4.5 h in the non-vegetated ditch. This mirrors the double flush phenomenon seen in NO$_3^-$ concentrations. Fig. 6 describes the percent reduction through time between inflow and outflow DIP for vegetated and non-vegetated ditches. Both vegetated ($r^2 = 0.2958$) and non-vegetated ($r^2 = 0.1141$) ditches consistently reduced DIP concentrations, with DIP reductions more variable than NH$_4^+$ and NO$_3^-$ through time. There were no statistically significant differences ($p = 0.3$) between the mean percentage reduction of DIP for vegetated and non-vegetated ditches through

Table 2
Relative load contributions of water, sediment and plant compartments for A) TKN within vegetated and non-vegetated drainage ditches and B) TP within vegetated and non-vegetated ditches.

Site (m)	Water (%)		Sediment (%)		Plant (%)	
	V	NV	V	NV	V	NV
A)						
0	74	87	26	13	0	–
5	58	87	42	13	0	–
10	66	87	34	13	0	–
20	77	85	24	15	0	–
40	66	77	34	23	0.05	–
80	13	27	87	73	0.1	–
160	70	69	30	31	0.01	–
320	26	16	74	84	0.04	–
B)						
0	84	89	16	12	0	–
5	86	87	14	13	0	–
10	77	88	23	12	0	–
20	82	82	18	18	0	–
40	83	84	17	16	0.01	–
80	31	30	68	70	0.04	–
160	83	62	17	38	0.01	–
320	17	9	83	91	0.03	–

Table 3
Nitrogen species loads (NH_4^+, NO_3^-, NO_2^- and TKN) for inflow and outflow from a non-vegetated (NV) and vegetated (V) ditch respectively when loads are calculated from 2.5 h to 8 h amendment to remove the influence of the first flush from standing water.

	Ammonium		Nitrate		Nitrite		TKN	
	NV	V	NV	V	NV	V	NV	V
Average inflow (kg) ± S.E.	$0.03 \pm 2.0 \times 10^{-3}$	$0.02 \pm 1.0 \times 10^{-3}$	0.24 ± 0.02	0.17 ± 0.01	$2.0 \times 10^{-3} \pm 2.0 \times 10^{-4}$	$2.0 \times 10^{-3} \pm 1.0 \times 10^{-4}$	0.32 ± 0.03	0.19 ± 0.02
Average outflow (kg) ± S.E.	0.02 ± 0.001	$0.01 \pm 1.0 \times 10^{-3}$	0.21 ± 0.01	0.16 ± 0.01	$3.0 \times 10^{-3} \pm 2.0 \times 10^{-4}$	$3.0 \times 10^{-3} \pm 2.0 \times 10^{-4}$	0.22 ± 0.02	0.07 ± 0.01
Total inflow load	0.81 ± 0.02	0.42 ± 0.05	11 ± 1.0	6.8 ± 1.1	$0.01 \pm 2.0 \times 10^{-3}$	$0.02 \pm 3.0 \times 10^{-3}$	11 ± 0.75	9.5 ± 1.9
Total outflow load	0.23	0.16	2.3	1.78	0.04	0.04	2.5	0.78
% Reduction	71	61	78	74	−79	−57	77	92

Fig. 6. Percentage reduction of DIP between mean inflow (0–20 m) and outflow (320 m) concentrations for respective time intervals for both vegetated (solid line) and non-vegetated (dashed line) drainage ditches. A +ve value indicates a lower outflow concentration than inflow; while a −ve value denotes a higher outflow concentration than inflow.
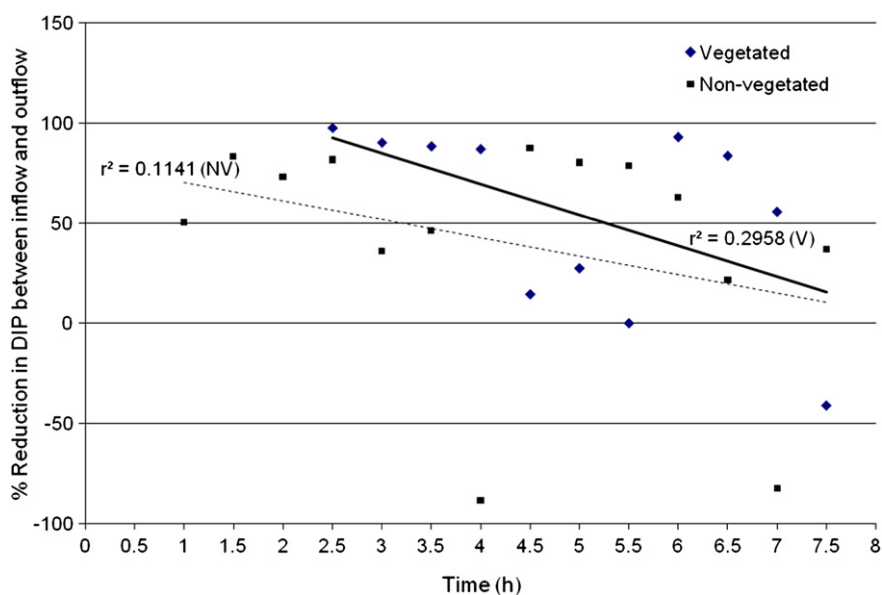

Fig. 7. Percentage reduction of TIP between mean inflow (0–20 m) and outflow (320 m) concentrations for respective time intervals for both vegetated (solid line) and non-vegetated (dashed line) drainage ditches. A +ve value indicates a lower outflow concentration than inflow; while a −ve value denotes a higher outflow concentration than inflow.
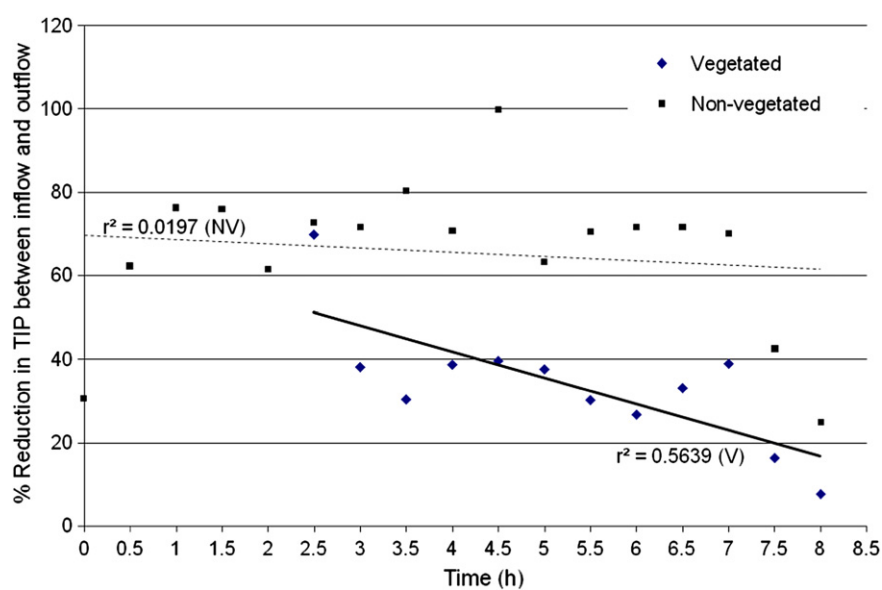
Table 4
Phosphorus species loads (DIP, TIP and TP) for inflow and outflow from a non-vegetated (NV) and vegetated (V) ditch respectively when loads are calculated from 2.5 h to 8 h amendment to remove the influence of the first flush from standing water.

	DIP		TIP		TP	
	NV	V	NV	V	NV	V
Average inflow (kg) ± S.E.	0.03 ± 0.01	$0.01 \pm 2.0 \times 10^{-3}$	0.29 ± 0.02	0.28 ± 0.02	0.19 ± 0.01	0.20 ± 0.02
Average outflow (kg) ± S.E.	$0.01 \pm 1.0 \times 10^{-3}$	$2.0 \times 10^{-3} \pm 1.0 \times 10^{-4}$	0.08 ± 0.01	0.19 ± 0.01	$0.03 \pm 2.0 \times 10^{-3}$	$0.04 \pm 3.0 \times 10^{-3}$
Total inflow load	5.1 ± 0.28	3.2 ± 0.48	6.3 ± 0.03	5.0 ± 0.65	5.5 ± 0.55	3.2 ± 0.65
Total outflow load	0.16	0.03	0.87	2.0	0.28	0.46
% Reduction	97	99	86	60	95	86

time (V: 54 ± 14.1; NV: 32.7 ± 18.7). Examining concentrations, the vegetated ditch never showed DIP outflow concentrations that exceeded inflow concentrations. However, the non-vegetated ditch at 4 h, and at 7 h, had DIP outflow concentrations that were significantly ($p < 0.05$) higher than the mean inflow DIP concentrations, suggesting desorption of labile P from non-vegetated ditch sediments. Kröger et al. (2008) reported a 44% reduction in DIP load over two years in vegetated ditches in north Mississippi, which was similar to that of Fink and Mitsch (2007). In California, Maynard et al. (2009) determined 61–63% of DIP was removed using a constructed wetland. An earlier study by Fink and Mitsch (2004) described removal of 74% of the DIP (i.e. soluble reactive phosphorus – SRP). In unsaturated soil bed systems, chemical adsorption was the main mechanism for DIP removal (Yang et al., 2001).

Interestingly, at no point during the amendment experiment was TIP export higher than inflow concentrations for both vegetated and non-vegetated ditches (Fig. 7). Though variable in percent reductions, the non-vegetated ditch reduced TIP levels consistently better than the vegetated ditch (Fig. 7). On average the vegetated ditch systems retained $36\% \pm 3.6$ of the influent TIP, while the non-vegetated ditch retained a significantly greater proportion ($71\% \pm 4.1$; $p \leq 0.001$). Kröger et al. (2008) showed TIP load reductions for vegetated ditches in the Loess Hills Region of Mississippi were closer to 50%. This difference could be explained by the volume of water within the DCDC ditch and the higher flow rate. Smaller primary intercept systems with less water volumes will more than likely produce better contaminant reduction potentials. There were no significant differences in sediment TP retention with time between vegetated and non-vegetated ditches. This is mirrored when the relative percent mass contribution for sediment, plant and water is evaluated (Table 2B). Bioavailability of sediment-P is a critical aspect of eutrophication (McDowell et al., 2003). Transport of soluble P concentrations to freshwater ecosystems is strongly affected by ditch sediments (Zhuan-xi et al., 2009). According to McDowell and Sharpley (2003), up to 35% of P sorbed during flow may be attributed to the sediment microbial community; however, fluvial sediment-P concentrations (either uptake or release) are rarely considered when determining edge-of-field P export management (McDowell and Sharpley, 2001; Sharpley et al., 2002).

Plants percentage contribution to TP load reduction was very low in the current study (Table 3B). These results differ from those of Reddy and DeBusk (1985) and Silvan et al. (2004) who attributed plant uptake of P responsible for 12–73% of removal. However, Wen and Recknagel (2006) determined dense vegetation inhibited P removal by periphyton and co-precipitation mechanisms. Total P concentrations were not significantly different between water ($p = 0.89$) or sediment ($p = 0.91$) between vegetated and non-vegetated ditch systems (Table 2B). Similarly the reduction of P species loads was similar between vegetated and non-vegetated ditches for DIP and TP (Table 4). However, the non-vegetated drainage ditch reduced a significantly ($p \leq 0.05$) greater relative percentage of TIP loads (86%) than the vegetated ditch (60%).

4. Conclusion

The current study sought to address the question of nutrient mitigation in two Mississippi Delta drainage ditches, one with vegetation and one without vegetation. Variability in certain N and P species outflow concentrations, and percentage distributions in both systems suggests a closer examination of intricate nutrient biogeochemical transformation mechanisms. The non-vegetated drainage ditch could be termed a dredged ditch, as it was maintained without vegetation. The difference between a dredged system and this non-vegetated system examined was the 10–30 cm layer of highly reactive, absorptive organic sludge that has immense interaction effects with the overlying water column. Often dredging removes vegetation and accumulated sediments, exposing the claypan of the drainage systems. Shigaki et al. (2008) quantified effects of ditch dredging on sediment-P dynamics. When overlying water was low in P concentration, undredged sediments released more than 10 times the P than did dredged sediments (Shigaki et al., 2008). When P-amended water (2.5 mg L^{-1}) was run over ditch sediments, undredged sediments removed 19% more than dredged sediments. Smith and Pappas (2007) evaluated the difference in dredged and non-dredged sediments in mitigating NO_3^-, NH_4^+ and soluble P. Nutrient uptake rates were significantly higher in pre-dredged sediments for all nutrient species, as dredging reduced the specific surface area in sediments for nutrient adsorption as well as removing the associated micro-biota responsible for transformations. Results from this study indicated both agricultural drainage ditches (vegetated or non-vegetatively maintained) were able to decrease nutrient loads being delivered to aquatic receiving systems, and warrant inclusion in management strategies in reducing non-point source pollution from agricultural landscapes. Increases in TIP reduction by the non-vegetated ditch could be linked to the sediment-P sorption characteristics of the two systems. Research is now underway to examine the P dynamics within Mississippi Delta drainage ditches to better address this question. Management decisions regarding TIP mitigation with vegetated or non-vegetated ditches need to be made with consideration of anticipated or historical P transport and concentrations in potential runoff. Additionally, drainage ditches could be designed with alternating sections of vegetation and open water to better address various nutrient mitigation requirements. Another concern with drainage ditch nutrient mitigation, as with constructed wetland contaminant mitigation, is the question of decreasing contaminant removal. Further research is needed to evaluate effective sizes (including lengths) of drainage ditches necessary to treat targeted runoff volumes with the goal of reaching an acceptable level of nutrient outflow. With this information, more informed decisions regarding BMP placement in the production landscape can be made.

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