Evaluating Plant Species-Specific Contributions to Nutrient Mitigation in Drainage Ditch Mesocosms

M. T. Moore · Robert Kröger

Abstract Eutrophication of surface water bodies is a worldwide concern. In the USA alone, excessive nutrients are blamed for nearly 5,700 impairments of surface water bodies. Innovative measures, such as maximizing drainage ditch nutrient retention, are being examined to decrease the amount of nitrogen (N) and phosphorus (P) running off agricultural lands and into aquatic receiving systems. The goal of this experiment was to measure the nutrient mitigation ability of six aquatic plants typically found in agricultural drainage ditches in the lower Mississippi River Basin. Experimental mesocosms ($1.25 \times 0.6 \times 0.8$ m) were filled with sediment and planted with monocultures of one of six obligate wetland plant species (*Typha latifolia* (broadleaf cattail), *Panicum hemitomon* (maidencane), *Thalia dealbata* (powdery alligator-flag), *Echinodorus cordifolia* (creeping burhead), *Myriophyllum spicatum* (Eurasian watermilfoil), and *Saururus cernuus* (lizard's tail)), while three replicates were left non-vegetated to serve as controls. Mesocosms were amended with 5 mg L^{-1} (each) of nitrate, ammonia, dissolved inorganic phosphorus, and total inorganic phosphorus, while nitrite amendments (1 mg L^{-1}) were also made over a 4-h hydraulic retention time. Following the 4-h exposure, "clean" (non-amended) water was flushed through mesocosms for an additional 8 h to assess residual leaching of nutrients. Outflow water concentrations and loads decreased for all examined forms of N and P. In certain cases, there were significant differences between plant species; however, for the majority, there was no statistical difference in percent decrease between plant species. While native aquatic vegetation shows promise for mitigation of nutrient runoff, further studies altering the hydraulic retention time for improved efficiency should be conducted.

Keywords Best management practice · Nitrogen · Phosphorus · Vegetation

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1 Introduction

As the world population increases, pressure on food and fiber production to meet people's basic needs increases as well. By 2050, the global population is expected to be nearly 12 billion, and the global demand for grain is expected to be twice that of 2002 demands (Tilman et al. 2002). Farmers in the USA,

one of the world's leading agricultural exporters, must continually work at minimizing costs while increasing crop yields. With increased production acreage there is a concomitant increase in fertilizer and chemical usage to efficiently and effectively raise successful crops. Seitzinger (2008) noted that anthropogenic souces of nitrogen (N) are being amended to soils at twice the natural rate. Over the last four decades, there has been a nearly 7-fold and 3.5-fold increases in global N and phosphorus (P) fertilization, respectively (Tilman 1999). Non-point source runoff, possibly carrying excess nutrients and pesticides from these agricultural fields, enters receiving water bodies and may ultimately enter a larger body of water, such as the Mississippi River or the Gulf of Mexico, causing potential downstream environmental damage.

The Mississippi River Basin (MRB) is home to some of the most productive agricultural land in the USA. At 3 million km^2, the MRB is responsible for approximately 90% of the freshwater inflow into the Gulf of Mexico and covers 40% of the lower 48 states (Day et al. 2003). Intensification of wheat, corn, and soybean production in this area has been implicated as a source of land alteration and water quality shifts observed since 1950 (Donner 2003). Goolsby et al. (2000) noted that historical data collected since the early 1900s indicated increasing nitrate (NO_3^-) concentrations (sometimes up to a factor of five) in the Mississippi River and some of its tributaries. Of the dissolved N found in the Mississippi River on an annual basis, 53% is NO_3^-, while 4% is ammonium (NH_4^+) (Turner and Rabalais 1991). Nitrate flux into the Gulf of Mexico has tripled in the last 30 years, with the greatest increase coming between 1970 and 1983 (Goolsby et al. 2000). According to Howarth et al. (1996), the northern Gulf of Mexico received 1.82 billion kg N year^{-1}. An estimated 565 kg N km^{-1} year^{-1} (25% of the MRB net anthropogenic input) travels through the Mississippi River system into the Gulf of Mexico (Howarth et al. 1996). Of the total river N flux entering the Atlantic Ocean, 31% is attributed to the Mississippi River alone (Dagg and Breed 2003). Similarly, P has increased in surface waters. Global mobilization of P has nearly tripled compared to its natural cycling (Smil 2000). Studies on P loads reported that, in one third of US rivers and streams, in excess of 90% of the P load may be due to non-point source pollution from agriculture and urban areas (Newman 1996).

Global increases of nutrient transport have resulted in significant concern over eutrophication and hypoxic conditions in receiving waters. Several on-field agricultural best management practices exist to try and reduce the amount of nutrient transport from the production landscape; however, additional practices are needed to couple with upland efforts at stemming nutrient movement into receiving systems. Kröger et al. (2007, 2008) reported vegetated agricultural drainage ditches were able to decrease 57% and 44% of a farm field's dissolved inorganic nitrogen (DIN) load and inorganic P effluent load, respectively. Using the vegetated agricultural drainage ditch as an innovative land management practice, a study was conducted to determine if differences in nitrogen and phosphorus mitigation existed among six aquatic plants species typically found in Mississippi River Basin drainage ditches surrounding agricultural fields.

2 Methods and Materials

Twenty-one Rubbermaid™ tub mesocosms (1.25 (L)×0.6 (W)×0.8 m (H)) were planted with six obligate wetland plant species: *Typha latifolia* L. (broadleaf cattail), *Panicum hemitomon* J.A. Schultes (maidencane), *Thalia dealbata* Fraser ex Roscoe (powdery alligator-flag), *Echinodorus cordifolius* (L.) Griseb (creeping burhead), *Myriophyllum spicatum* (L.) (Eurasian watermilfoil), and *Saururus cernuus* (L.) (lizard's tail) (Table 1). Plants were collected and transplanted from natural populations in control ponds at the University of Mississippi Field Station during the month of April. Each species had three replicates, with three non-vegetated mesocosms to serve as controls (total sample size=21). Mesocosms consisted of a 25 cm sand substrate with an overlying 10 cm layer of sediment (25±3% sand; 75± 3% silt) from wetlands at the University of Mississippi Field Station (UMFS). Mesocosms were maintained under natural climatic conditions at the National Sedimentation Laboratory, USDA-ARS, Oxford, MS. Three months after transplanting, a nutrient mitigation study examined the potential of each plant species to reduce N and P concentrations and loads.

Mesocosms were treated with a simulated high nutrient (5 mg L^{-1}) runoff concentration. This high concentration has been noted in stormwater runoff

Table 1 Obligate wetland species used in the experiment on nutrient decreases

Species	Wetland species type	Family	Density (stems/m^2)	Native/Introduced
Typha latifolia L.	Monocot emergent	Typhaceae	19.25	Native
Panicum hemitomon J.A. Schultes	Monocot emergent	Poaceae	571.8	Native
Thalia dealbata Fraser ex Roscoe	Monocot emergent	Marantaceae	54.1	Native
Echinodorus cordifolius (L.) Griseb.	Monocot emergent	Alismataceae	34.45	Native
Myriophyllum spicatum L.	Dicot rooted submerged	Haloragaceae	75.5	Introduced
Saururus cernuus L.	Dicot emergent	Saururaceae	24.7	Native

All species are common to agricultural drainage ditches in the Mid-South, depending on ditch hydrology

over the growing season post-fertilization (Kröger et al. 2007, 2008). All examined forms of N and P were delivered at 5 mg L^{-1}, while nitrite (NO$_2^-$) was delivered at 1 mg L^{-1}. Nutrient stocks were prepared from laboratory grade potassium phosphate dibasic (KH$_2$PO$_4$), sodium nitrate (NaNO$_3$), ammonium sulfate (NH$_3$SO$_4$), and sodium nitrite (NaNO$_2$). Nutrient concentrations were delivered via Fluid Metering Inc. (FMI™) piston pumps, models QD-1 (0–552 ml min^{-1}) and QD-2 (0–1242 ml min^{-1}) at a rate specific to each mesocosm. This rate was back-calculated by determining a 4-h specific retention volume for each mesocosm. After 4 h, clean water (no nutrient amendment) was delivered for a subsequent 8 h (two times the retention volumes) to determine residual leaching of N and P from each system. Water samples were taken pre-exposure and from delivered clean water to determine background N and P concentrations. Sampling occurred in duplicate every hour for 12 h for each mesocosm outflow over the duration of the experiment (total water samples=546). Water samples were immediately decanted into respective volumes for nutrient analyses, filtered if necessary and stored at 4°C until analysis. Nutrient analyses occurred within 1 week of the experiment.

2.1 Nutrient Analyses

All water samples were analyzed for NO$_3^-$, NO$_2^-$, ammonia (NH$_3$), dissolved inorganic P (DIP) and total inorganic P (TIP). Particulate P (PP) was determined by subtracting DIP from TIP. Nitrate and NO$_2^-$ were analyzed with the cadmium reduction method, while NH$_3$ was analyzed by the standard phenate method (APHA 1998). Total inorganic P was determined by the ammonia persulfate method

(Murphy and Riley 1962, APHA 1998). Dissolved inorganic P was similarly determined by the method of Murphy and Riley (1962), after 0.45 μm cellulose membrane filtration. Nutrient analysis absorbance detection was performed using a ThermoSpectronic Genesys 10UV spectrophotometer. Detection ranges for nutrient species were 0.001–10 mg L^{-1} at 880 nm for P, and 0.005–10 mg L^{-1} at 530 nm for N, in a 50 mm flow cell.

Statistical analyses included one-way analysis of variance (ANOVA; F test), post hoc Tukey's honestly significant differences tests, and t tests on normally distributed data (natural logarithm transformed). Analyses were conducted with an alpha of 0.05. Nutrient load of inflow and outflow water was determined by multiplying the known inflow/outflow concentration by the measured inflow rate. Outflow rate was assumed to equal the inflow rate, with constant water level maintained within each mesocosm throughout the duration of the experiment. Nutrient (TIP, DIP, NO$_3^-$, NO$_2^-$, and NH$_3^-$) concentrations were normalized according to initial un-amended water N and P concentrations and subtracted from influent concentrations to determine a percentage reduction of concentrations through time for each mesocosm. Mesocosm nutrient outflow concentration peaked at 4-h post-amendment; however, for certain treatments, peak concentrations were attained at the 3-h sampling period.

3 Results

3.1 Phosphorus Concentrations

Prior to the experiment, all mesocosms were filled to the required 4-h retention water levels. Initial filling of

mesocosms released sediment into suspension, increasing pre-exposure TIP concentrations (0.41–5.24 mg P L^{-1}) (Fig. 1a). At 0 h, TIP had a mean concentration of 2.53±0.29 mg P L^{-1}, while the mean concentration of DIP was 0.58±0.07 mg P L^{-1}. This resulted in a PP concentration of 1.95±0.27 mg P L^{-1}. After 1 h, DIP concentration increased to 2.6±0.18 mg P L^{-1}, and PP

load dropped to 0.57±0.08 mg P L^{-1}. Particulate P concentrations remained low (<0.5 mg P L^{-1}) for the duration of the experiment. Total inorganic P concentrations only returned to pre-exposure concentrations 12 h post-initiation of the experiment (Fig. 1a). Of the TIP components (PP and DIP), DIP contributed 80–90% for the duration of the experiment.
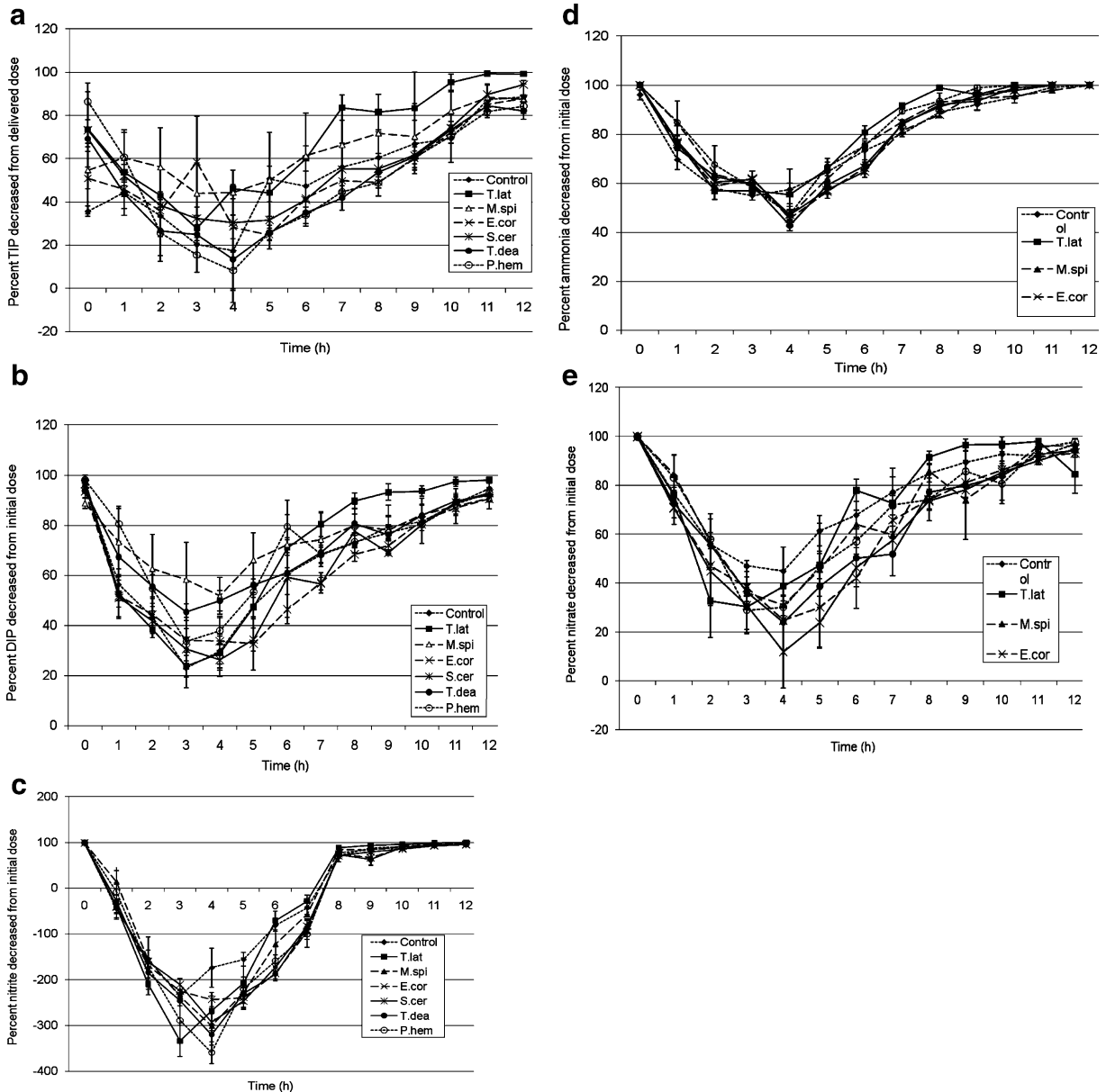
Fig. 1 Percentage of nutrients (a TIP, b DIP, c NO_2^-, d NH_3, e NO_3^-) removed from the initial delivered mixing chamber concentrations. Each mesocosm had a 4-h retention time, with a 4-h dose, and two clean water cycles of 4 h each. *T. lat* Typha *latifolia*, *M. spi Myriophyllum spicatum*, *E. cor Echinodorus cordifolia*, *S. cer Saururus cernuus*, *T. dea Thalia dealbata*, *P. hem Panicum hemitomon*

Combining the TIP 3 h and 4 h time series for each species treatment, *M. spicatum* and *P. hemitomon* were significantly different ($F_{6,35}$=2.63; p=0.03) from the other species. *M. spicatum* had the greatest concentration decrease between inflow and outflow for TIP (44%) while *P. hemitomon* had the least (11%).

Across all treatments there was a 23–52% decrease in DIP concentrations (Fig. 1b). The non-vegetated control had the least effect on the outflow concentration of DIP (23%). Combining 3- and 4-h time series, *M. spicatum* had significantly greater DIP concentration decrease than all other species ($F_{6,35}$=5.33; p=0.0005). *M. spicatum* had significantly higher decreases (55%) than *S. cernuus* (28.4%), the non-vegetated control (26.4%), and *T. latifolia* (26.3%).

3.2 Phosphorus Loads

Total load decrease for TIP and DIP was highly variable between species over the 12 h experiment as well as at the height of the exposure (4 h). At the height of the exposure, all treatments (vegetated and non-vegetated) removed TIP and DIP from influent runoff. The load decrease range for TIP at 4 h was 19%–42%, while the load decrease range for DIP at 4 h was 40%–59%. Low TIP decrease ranges were a

result of high levels of suspended sediment moving through the system in the initial startup. For the majority, the highest TIP and DIP loads moved through each mesocosm from 0 to 4 h with the associated storm runoff amendment (Table 2). For TIP decreases, there were significant differences between non-vegetated control and *T. latifolia*, and *M. spicatum* and *E. cordifolia* ($F_{6,14}$=3.65, p≤0.05). *M. spicatum* and *E. cordifolia* had significantly higher percentages of load decreases than *T. latifolia* and the non-vegetated control. There were no significant differences between DIP decreases; however, *M. spicatum* reduced the greatest load, while *T. latifolia, S. cernuus* and the non-vegetated control removed the least amount of DIP.

3.3 Nitrogen Concentrations

All treatments significantly elevated NO_2^- concentrations for the duration of the runoff amendment (Fig. 1c). Mesocosms were thought to be sufficiently aerobic to provide an ideal environment for nitrification. In some instances (*T. latifolia, T. dealbata*, and *P. hemitomon*), NO_2^- concentrations increased by over 300%: *T. latifolia* 2.6 $mg L^{-1}$ (0.61±0.09 mg L^{-1}), *T. dealbata* 3.04 $mg L^{-1}$ (0.71±0.10 $mg L^{-1}$), *P. hemitomon* 3.6 $mg L^{-1}$ (0.80±0.13 $mg L^{-1}$). Com-

Table 2 Inflow and outflow load decreases±standard deviation (mg) for total inorganic P (TIP) and dissolved inorganic phosphate (DIP) via the mono-specific stands of obligate wetland vegetation

	Control	T. latifolia	M. spicatum	E. cordifolia	S. cernuus	T. dealbata	P. hemitomon
TIP							
Inflow	545±30	436±57	613±53	566±38	650±19	581±60	433±33
Total outflow	774±132	457±91	694±196	805±84	883±18	897±85	669±55
Outflow 0–4 h	344±49	220±22	288±65	303±56	348±15	360±35	257±27
Outflow 4–8 h	271±45	170±37	258±81	330±22	354±3	361±24	274±32
Outflow 8–12 h	158±41	66±32	147±51	171±14	180±6	175±29	137±10
Total% decrease (12 h)	−42±22	−3±9	−13±30	−41±7	−36±4	−55±14	−55±8
% Decrease at peak (4 h)	20±8	19±15	42±11	39±8	35±3	28±8	31±8
DIP							
Inflow	555±141	343±58	578±65	566±94	557±21	767±54	594±107
Total outflow	622±134	345±74	519±193	684±49	676±18	738±87	559±69
Outflow 0–4 h	275±40	184±27	213±78	277±23	291±4	309±37	231±23
Outflow 4–8 h	234±54	127±33	193±69	282±12	267±22	288±23	215±46
Outflow 8–12 h	112±43	33±12	112±46	125±16	118±6	140±31	112±10
Total% decrease (12 h)	−14±5	0.8±5	3±17	−24±10	−61±34	4±6	3±7
% Decrease at peak (4 h)	41±7	40±1	59±5	43±5	41±4	56±3	55±6

bining the 3- and 4-h time series, there was a significant difference in percentage changes in NO_2^- concentrations between treatments (ANOVA, $F_{6,35}=$ 3.24; $p=0.009$). This result was attributed to the single difference between non-vegetated control (180%) and *P. hemitomon* (380%) (post hoc Tukey's test) (Fig. 1c). The major difference between these two treatments was in plant density, from absent to the highest. There were no significant differences between other treatments.

Percentage decreases in NH_3 ranged from 42.7% to 57.35% (Fig. 1d). There were no significant differences in percentage decrease between species and non-vegetated control ($F_{6,14}=1.45$; $p=0.26$). However, the non-vegetated control and *T. latifolia*, on average, had higher NH_3 concentration decreases (57% and 55%, respectively) than all other species (42–47%) (Fig. 1d).

Nitrate concentration decreases were between 11.8% and 45% (Fig. 1e), and there were no significant differences between any of the species and the non-vegetated control ($F_{6,14}=1.25$, $p=0.34$). *S. cernuus* removed the lowest percentage of NO_3^- (11.8%) at the height of the exposure, while the non-vegetated control removed the greatest percentage (44.9%). The non-vegetated control had a significantly larger effect on reducing NO_3^- concentration than

reducing DIP ($p \leq 0.05$). In some replicates (*S. cernuus* and *E. cordifolia*) NO_3^- concentration increased above the initial amended dose, again suggesting occurrence of aerobic nitrification. Nitrate concentrations returned to background levels at 8 h, which was 2 h earlier than NH_3.

3.4 Nitrogen Loads

Load decreases followed concentration decreases, where NO_3^- and NH_3 had variable load decrease efficiencies among plant species, while increasing loads in the outflow compared to inflow were observed over time in the mesocosms (Tables 3 and 4). Ammonia decrease was greater than 50% for all treatments. *M. spicatum* (70%) and *S. cernuus* (68%) removed the highest percentage of NH_3 loads while *T. latifolia* (56%) removed the least. There was a significant difference in NH_3 decrease percentage between *M. spicatum* and *T. latifolia* ($F_{6,14}=4.56$, $p \leq 0.01$) (Table 4). Similarly, *M. spicatum* removed a higher percentage of NO_3^- load from the mesocosm (59%). *Echinodorus cordifolia* and the non-vegetated control also removed high percentages of NO_3^- load (63% and 60%, respectively). There were no significant differences between NO_3^- removal percentages between species (Table 3).

Table 3 Inflow and outflow load decreases±standard deviation (mg) for nitrite (NO_2^-) and nitrate (NO_3^-) via the mono-specific stands of obligate wetland vegetation

	Control	T. latifolia	M. spicatum	E. cordifolia	S. cernuus	T. dealbata	P. hemitomon
NO_2^-							
Inflow	73±13	37±4	54±9	58±9	58±3	69±12	63±15
Total Outflow	293±66	165±18	235±34	278±35	280±12	344±58	306±54
Outflow 0–4 h	151±26	93±7	112±17	127±18	128±6	162±33	145±28
Outflow 4–8 h	132±37	70±11	116±14	140±16	143±6	169±26	153±31
Outflow 8–12 h	9.4±3	2±0.6	6±2	10±2	9±1	12±2	7±1
Total% decrease (12 h)	−298±15	−352±32	−344±24	−384±15	−384±11	−402±31	−402±43
% Decrease @ peak(4 h)	−199.7±6	−168±10	−121±13	−127±5	−62±70	−143±22	−150±41
NO_3^-							
Inflow	412±35	238±11	323±45	309±66	286±45	370±59	344±42
Total outflow	343±72	213±33	302±43	327±51	336±17	408±63	343±68
Outflow 0–4 h	155±23	112±9	123±9	130±16	132±5	162±32	141±26
Outflow 4–8 h	140±38	83±23	133±16	151±22	156±9	186±17	152±35
Outflow 8–12 h	47±12	18±2	46±18	45±10	47±5	59±17	50±19
Total% decrease(12 h)	10±9	10±13	3±17	−12±19	−22±16	−10±3	2±10
% Decrease @ peak(4 h)	60±2	50±4	59±5	63±18	49±9	56±3	58±8

Table 4 Inflow and outflow load decreases±standard deviation (mg) for ammonia (NH_3) via the mono-specific stands of obligate wetland vegetation

	Control	T. latifolia	M. spicatum	E. cordifolia	S. cernuus	T. dealbata	P. hemitomon
NH_3							
Inflow	26±2	19±2	25±2	26±1	28±1.1	28±2	23±2
Total Outflow	18±3	10±2	15±0.8	18±0.7	18±0.5	18±1	12±1
Outflow 0–4 h	9±1	6±0.8	8±1	9±0.5	9±0.3	9±0.6	7±0.9
Outflow 4–8 h	7±2	4±0.9	7±0.1	8±0.2	9±0.3	8.3±0.2	6±0.6
Outflow 8–12 h	1±0.4	0.2±0.1	1±0.3	0.7±0.03	0.7±0.2	0.84±0.35	0.25±0.1
% Decrease @ peak (4 h)	65±3	67±0.2	70±6	67±0.6	68±0.3	56±3	71±5
Total DIN decrease @ peak(4 h) ($NH_3+NO_3^-+NO_2^-$)	38%	28%	40%	32%	28%	29%	32%

M. spicatum, the only submerged aquatic plant, demonstrated higher percentages of decrease for DIP (59%), NH_3 (70%), and NO_3^- (59%). In some cases these percentages were significantly higher than any other treatment. However, the non-vegetated control also had high percentages of DIP (41%), NH_3 (65%), and NO_3^- (60%) decrease.

4 Discussion

Most scientific literature examining P removal from agricultural or urban runoff focuses on the overall ability of constructed wetlands, rather than comparing specific plant removal efficiencies. Literature on P removal using constructed wetlands varies from efficient (Tanner 1996) to highly variable (Kovacic et al. 2000). Many studies report less than 50% total phosphorus (TP) removal in constructed wetlands, including Braskerud (2002) (21%–44%); Fink and Mitsch (2004) (28%); Hoagland et al. (2001) (29%); and Fink and Mitsch (2007) (31%). Earlier studies detailed TP removal ranging from 71%–93% in constructed wetlands (Moustafa et al. 1996, Tanner 1996).

Lee et al. (2003) reported that a switchgrass buffer (*Panicum virgatum*) was able to remove 58% of PO_4-P and 78% TP. When used in combination with a woody buffer, PO_4-P and TP removal rates increased to 91% and 80%, respectively (Lee et al. 2003). Results from the current study demonstrated that a different species of *Panicum* (*hemitomon*) was less efficient for P removal, with only 11% TIP being removed. Reddy and De Busk (1985) examined TP

removal efficiencies of several aquatic plants, including *Eichhornia crassipes* (water hyacinth), *Pistia stratiotes* (water lettuce), *Hydrocotyle umbellata* (pennywort), *Lemna minor* and *Spirodela polyrhiza* (duckweeds), *Azolla caroliniana* (Carolina mosquito-fern), *Salvinia rotundifolia* (butterfly fern), and *Egeria densa* (Brazilian waterweed), which had removal efficiencies ranging from 12% to 73%.

Current results from all plant species indicated 23–52% decrease in DIP (synonymous with soluble reactive phosphorus (SRP)) concentrations, which agree with most of the published literature available on SRP retention. Fink and Mitsch (2007) determined SRP retention of 46% in a marsh system, although an earlier study (Fink and Mitsch 2004) reported SRP wetland retention of 74%. A decrease in DIP load of 44% was achieved when runoff traveled through a vegetated agricultural drainage ditch in north Mississippi (Kröger et al. 2008). Greenway and Woolley (1999) cited retention of SRP by <13% after passing through a constructed wetland.

Some studies question, however, the role plants play in P removal. Yang et al. (2001) reported the main removal mechanism for SRP was chemical adsorption in soil-bed systems which were not yet saturated. To the contrary, Silvan et al. (2004) determined that 25% of added P was retained in plants of constructed wetlands. Fink and Mitsch (2007) reported greater TP removal in an emergent marsh than open water, suggesting the importance of vegetation in the nutrient mitigation process.

In the current study, NO_2^- concentrations were elevated in all treatments. Obviously the environment was not conducive for complete nitrification/denitri-

fication processes to occur. The extent of nitrification (process of ammonia being converted to nitrite and then nitrate) is dependent on oxygen availability (Vymazal 2007). While NH_3 concentrations in the current study were reduced 42–57%, NO_3^- concentrations were only reduced 12–45%, providing ample conditions for increasing NO_2^- concentrations (an intermediate in both nitrification and denitrification processes) to exist within the systems. Denitrification rates are directly affected by factors such as NO_3^- concentrations, temperature, pH, sediment oxygen concentrations, and organic matter (Seitzinger et al. 2006, Lindau et al. 2008). The success or failure of N removal in wetland systems is controlled by the nitrification/denitrification processes, which are most often associated with microbial communities either in or on sediment or submerged macrophytic vegetation (Bastviken et al. 2003).

Many studies have reported on the TN removal capacities of constructed wetlands ranging from 18% to 92% (Tanner 1996, Greenway and Woolley 1999, Kovacic et al. 2000, Hoagland et al. 2001, Fink and Mitsch 2007). For those above-referenced studies looking specifically at NO_3^- mitigation, efficiencies were similar. Decreases in NO_3^- concentrations in the current study were variable (12–45%), but within range of some reported literature values. Kovacic et al. (2000) reported a 28% decrease in NO_3^- concentrations from inflow to outflow of a constructed wetland. An annual NO_3^- decrease of 74% was reported by Fink and Mitsch (2007) in a constructed wetland, which was slightly higher than an earlier study (Fink and Mitsch 2004) where the same authors reported 41% retention of NO_3^- and NO_2^-. Scott et al. (2008) noted >90% decrease in NO_3^- concentrations within the first 500 m downstream of inflow within a constructed wetland.

While no significant differences emerged between vegetated systems and the non-vegetated control in the current research with regard to NO_3^- or NH_3 mitigation, previous studies have reported the significant impact of vegetation on N species removal in constructed wetlands. *Typha* species in general have been reported as capable of removing N (either as total N or NO_3^-) (Bachand and Horne 2000, Martin et al. 2003). Likewise, several studies have examined the differences between vegetated and unvegetated systems and their impact on N removal. Tanner et al. (1999) examined NH_4^+ removal in systems planted

with *Schoenoplectus tabernaemontani* (C.C. Gmel) Palla (soft stem bulrush) versus those left unplanted. Results emphasized greater efficiency in planted systems (54–71% removal) as opposed to those left unplanted (10–28% removal) (Tanner et al. 1999). David et al. (1997) demonstrated spring NO_3^- removal rates were higher in planted versus unplanted mesocosms. More than 90% of N removal was associated with aquatic plants in a study done by Rogers et al. (1991), while gravel systems without vegetation were deemed inefficient and were quickly overloaded by nutrients.

The current research indicates the following plant species efficiency in removal of phosphorus: *M. spicatum*>*E. cordifolia*>*S. cernuus*>*P. hemitomon*>*T. dealbata*>non-vegetated control/*T. latifolia*. For nitrogen forms, plant removal efficiency was *M. spicatum*>non-vegetated control>*E. cordifolia*/*P. hemitomon*>*T. dealbata*>*T. latifolia*/*S. cernuus*. The progression toward natural plant assemblages and their ability to mitigate nutrients is encouraged; however, replication of diverse assemblages or plant specific species diversity is a common problem in ecological experiments. Future research lines will incorporate monocultures of plant species placed in series (in separate mesocosms) to increase the diversity of plant assemblages vying for opportunities to mitigate nutrients in runoff.

With increased food and fiber production necessary for a growing global population, the planet's natural resources are being pushed to their limits. Increasing fertilizer use, as a result of growing food needs, has resulted in a tumultuous relationship between the agricultural ecosystem and their aquatic receiving systems. Tilman (1999) suggested that 35 years ago, agriculture was a minor source of off-site degradation but today is rapidly becoming the major source of nutrient loading to terrestrial and aquatic ecosystems. The solution to nutrient eutrophication and hypoxia issues in the USA will not be a simple one, neither will there be a quick fix. Allowing agricultural runoff to flow through vegetated systems (wetlands or drainage ditches) is just one step toward decreasing nutrient runoff in aquatic receiving systems. As Turner and Rabalais (2003) pointed out, the present situation developed over decades, and it will take just as long if not longer for rehabilitation of water quality in nutrient-laden systems to be successful.

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