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Bioremediation Journal

Publication details, including instructions for authors and subscription information:

<http://www.tandfonline.com/loi/bbrm20>

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Matthew T. Moore^a, Robert Kröger^b, Martin A. Locke^a, Heather L. Tyler^a & Charles M. Cooper^a

^a US Department of Agriculture Agricultural Research Service National Sedimentation Laboratory, Water Quality and Ecology Unit, Oxford, Mississippi, USA

^b Department of Wildlife, Fisheries, and Aquaculture, Mississippi State University, Mississippi State, Mississippi, USA

Published online: 24 Jul 2013.

To cite this article: Matthew T. Moore, Robert Krger, Martin A. Locke, Heather L. Tyler & Charles M. Cooper (2013) Seasonal and Interspecific Nutrient Mitigation Comparisons of Three Emergent Aquatic Macrophytes, *Bioremediation Journal*, 17:3, 148-158, DOI: [10.1080/10889868.2013.807771](https://doi.org/10.1080/10889868.2013.807771)

To link to this article: <http://dx.doi.org/10.1080/10889868.2013.807771>

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Seasonal and Interspecific Nutrient Mitigation Comparisons of Three Emergent Aquatic Macrophytes

Matthew T. Moore,¹ Robert Kröger,² Martin A. Locke,¹ Heather L. Tyler,¹ and Charles M. Cooper¹

¹US Department of Agriculture
Agricultural Research Service
National Sedimentation
Laboratory, Water Quality and
Ecology Unit, Oxford,
Mississippi, USA

²Department of Wildlife,
Fisheries, and Aquaculture,
Mississippi State University,
Mississippi State, Mississippi,
USA

ABSTRACT The purpose of this study was to measure both summer and winter nutrient mitigation efficiencies of three aquatic plants found in agricultural drainage ditches in the lower Mississippi River Basin. Mesocosms (1.25 × 0.6 × 0.8 m) were filled with sediment and planted with monocultures of one of three obligate wetland plant species, *Typha latifolia*, *Thalia dealbata*, and *Sagittaria latifolia*, or left nonvegetated to serve as controls. Mesocosms were amended with nitrate, ammonium, and phosphate over a 4-h hydraulic retention time, followed by an 8-h flushing with nonamended water to assess residual nutrient leaching in both summer and winter exposures. Significant interactions between vegetation type and season were noted for both nitrate and total inorganic phosphorus concentrations and loads. Future research will focus on altering hydraulic retention time for improved efficiency, as well as the specific contribution of microbial activity to nutrient mitigation.

KEYWORDS best management practice, nitrogen, phosphorus, vegetation

INTRODUCTION

During the past 35 years, the United States has doubled its agricultural food production, leading to a nearly 3.5- and 7-fold increase in phosphorus (P) and nitrogen (N) fertilizer use, respectively (Tilman 1999). In 2007, more than 97.5 million ha of cropland were treated with fertilizer, which is approximately an 8% increase from applications in 2002 (US Department of Agriculture National Agricultural Statistics Service [USDA-NASS] 2009). Globally, the United States consumes 13% of the inorganic N fertilizers used, which when averaged over the entire land area of the country, equates to a 2.2-fold greater usage in the United States than the global average (Howarth et al. 2002). Likewise, the United States ranks third globally in annual N fertilizer consumption (10.7 Mt), behind India (10.85 Mt) and China (22.6 Mt) (Fixen and West 2002). Problems begin to arise not only because of the volume of fertilizer used, but also because where it may ultimately reside. Tilman (1999) indicated that less than 50% of applied N stays in the field or is harvested with the crop, leaving the remainder to impact both terrestrial and aquatic ecosystems. Fertilizer, according to Howarth et al. (2002),

Address correspondence to Matthew T. Moore, USDA-ARS National Sedimentation Laboratory, Water Quality and Ecology Unit, PO Box 1157, Oxford, MS 38655, USA. E-mail: matt.moore@ars.usda.gov

is the single largest source of reactive N, both in the United States and globally. Eutrophication (excessive nutrients in water), hypoxia, biodiversity loss, and habitat degradation in coastal ecosystems are consequences of the amount of reactive N in the environment (Galloway et al. 2003).

Snyder (2008) reported 22% of nitrate (NO_3^-) in the Mississippi-Atchafalaya River Basin originated in the lower Mississippi subbasin, which is also responsible for 16% of the total phosphorus (P) discharge. From 1999 to 2005, approximately 54% of the non-point source N in the Mississippi-Atchafalaya River Basin was from fertilizer (Snyder 2008). Currently delivering some $1.82 \text{ Tg N year}^{-1}$ to the northern Gulf of Mexico, the Mississippi River's dissolved N is 53% nitrate (NO_3^-), 43% organic N, and 4% ammonium (NH_4^+) (Dagg and Breed 2003). Of the total river N flux into the Atlantic Ocean, the Mississippi River is responsible for 31% (Dagg and Breed 2003).

According to the US Environmental Protection Agency (USEPA), nutrients are the third leading cause (10%) of water body impairment in the United States (USEPA 2011). By factoring in other listed impairments that may be directly related to nutrients (e.g., organic enrichment, algal growth, and ammonia), these causes combine to account for 20% of US impairments, surpassing both pathogens and metals (USEPA 2011). It is estimated that freshwater eutrophication, at the very least, costs an estimated \$2.2 billion annually (Dodds et al. 2009). Additionally, there is a global climate impact of excessive N in the environment, since denitrification of NO_3^- will produce the potent greenhouse gas, nitrous oxide (Tilman 1999). With the mixture of politics, policy, and economics, nutrient criteria development is an intensely debated subject. By basing nutrient criteria on the best available science, it is anticipated that conflict would be minimized and potential benefits to improve water quality maximized (Dodds and Welch 2000).

Several best management practices promote varied uses of vegetation to reduce nutrient runoff, whether through grass waterways, stiff grass hedges, constructed wetlands, or vegetated drainage ditches. Although few would question the ability of plants to accumulate nutrients through uptake during the growing season, only limited research has been conducted regarding the potential of nutrient uptake during the dormant season. Winter research has instead focused on nutrient leaching back into the water column. The objective of this re-

search was to compare nutrient mitigation capacities of specific aquatic plants during exposures occurring both in the summer (June, July, and August) and the dormant winter seasons (December, January, and February).

MATERIALS AND METHODS

In early April, individual mesocosms (1.25 m [length] \times 0.6 m [width] \times 0.8 m [height]) were filled with 25 cm of sand substrate topped with a 10-cm layer of sediment from wetlands at the University of Mississippi Field Station (UMFS), Abbeville, Mississippi. Sediment particle size distribution was analyzed on a Horiba (Horiba Instruments, Irvine, CA, USA) model LA-910 laser scattering particle size analyzer and calculated based on Mie scattering theory, using a relative refractive index of 1.32-000. Mean (SE) percent sand, silt, and clay fractions across mesocosms were 22.2 (1.77), 77.3 (1.80), and 0.48 (0.06), respectively.

Plant stocks were collected from the US Department of Agriculture (USDA) Natural Resource Conservation Service (NRCS), Plant Materials Center, Coffeerville, Mississippi, and from the UMFS, Abbeville, Mississippi. These two locations were chosen because of their lack of agrochemical exposure. Selected plant species were broadleaf cattail (*Typha latifolia* L.), powdery alligator flag (*Thalia dealbata* Fraser ex Roscoe), and broadleaf arrowhead (*Sagittaria latifolia* Willd.). All three species have been identified as common species within surface vegetated agricultural ditches in the Mississippi Delta, USA. Plant species have been classified as obligate wetland plants for the southeast region. Mesocosms were maintained under natural climatic conditions at the USDA Agricultural Research Service (ARS) National Sedimentation Laboratory (NSL), Oxford, Mississippi.

Three treatments were planted with a monoculture of a selected plant species and one treatment was nonvegetated, serving as the control. There were four treatments with three replicates per treatment for a total of 12 mesocosms. Mesocosms were randomly arranged.

Three months after transplanting to the mesocosms, the first series of nutrient mitigation studies examined the potential of each plant species to reduce NO_3^- , ammonium (NH_4^+), total inorganic phosphorus (TIP), and dissolved inorganic phosphorus (DIP) concentrations and loads in simulated water runoff. Two experiments were conducted in the summer, exactly 1 week apart. The second series of experiments took place during the

winter (January) examining the same constituency concentrations and loads and were again exactly 1 week apart.

An aqueous mixture was prepared in reservoirs and pumped into individual mesocosms at the water surface. Nonchlorinated well water (city of Oxford, Mississippi) was mixed with stock concentrations of laboratory grade sodium nitrate (NaNO_3), ammonium sulfate $[(\text{NH}_4)_2\text{SO}_4]$, and dipotassium hydrogen phosphate (K_2HPO_4) to prepare nutrient-enriched water for the simulated runoff exposure. Target concentrations for all N and P constituents were 5 mg/L for the first summer experiment, then they were reduced to 2.5 mg/L for the final summer and both winter experiments, due to concerns of overloading the experimental systems. Outflow from each mesocosm exited at the surface through a 0.95 cm (outer diameter [o.d.]) \times 0.64 cm inner diameter [i.d.] vinyl discharge hose at the opposite end of the mesocosm. Each mesocosm was hydrologically regulated for a 4-h water residence time. The 4-h targeted nutrient runoff exposure was delivered using Fluid Metering Inc. (FMI; Syosset, NY, USA) piston pumps, models QD-1 (flow range: 0–552 mL min^{-1}) and QD-2 (flow range: 0–1242 mL min^{-1}), at a constant, calculated rate (based on a 4-h residence time) for each individual mesocosm. After 4 h, new delivery hosing provided “clean” (unamended) municipal well water runoff for an additional 8 h to simulate hydraulic conditions possibly seen in the field which may include potential flushing of N and P from wetland systems.

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 for the duration of the experiment. Water samples were immediately decanted into respective volumes for nutrient analyses, partially filtered, and stored at 4°C until analysis. Nutrient analyses occurred within 1 week of the experiment.

Nitrate was analyzed with the cadmium reduction method, whereas NH_4^+ was analyzed by the standard phenate method (American Public Health Association [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 (Rochester, NY,

USA) Genesys 10 ultraviolet (UV) 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.

One-way analysis of variance (ANOVA) followed by Tukey-Kramer honestly significant difference (HSD) test was used to determine significant differences between particle size in mesocosms. Two-way ANOVA followed by an effects test and a Tukey-Kramer HSD test were performed to identify significant differences between vegetation treatments and season in JMP version 8.0.1 (SAS Institute, Cary, NC). Analyses were conducted with an alpha of .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 (NO_3^- , NH_4^+ , TIP, and DIP) concentrations were normalized according to initial unamended water N and P concentrations and subtracted from influent concentrations to determine a percentage reduction of concentrations through time for each mesocosm.

RESULTS

Nitrogen Concentrations

Percentage decreases in NH_4^+ during the summer ranged from $74 \pm 3\%$ (nonvegetated control) to $80 \pm 2\%$ (*S. latifolia*) (Figure 1a, b). In the winter, NH_4^+ decreases ranged from $68 \pm 8\%$ (nonvegetated control) to $79 \pm 3\%$ (*T. latifolia*). No significant interactions were noted between mesocosm vegetation and season ($p = .5385$). Likewise, no significant differences were noted between vegetation ($p = .3164$) or season ($p = .2571$). Plant species listed from greatest to least for removal efficiency in the summer were *S. latifolia* > *T. latifolia* = *T. dealbata* > nonvegetated control. Relative efficiency in the winter was *T. latifolia* > *S. latifolia* > *T. dealbata* > nonvegetated control (Figure 1a, b).

Nitrate concentration decreases ranged from $66 \pm 2\%$ (*S. latifolia*) to $73 \pm 1\%$ (*T. latifolia*) in summer exposures, whereas winter exposure decreases ranged from $64 \pm 1\%$ (*T. dealbata*) to $69 \pm 1\%$ (*S. latifolia*). A significant interaction was observed between vegetation and season ($p = .0023$), with *T. dealbata* lowering NO_3^- concentrations significantly more in summer than in winter. No other significant differences between mesocosm

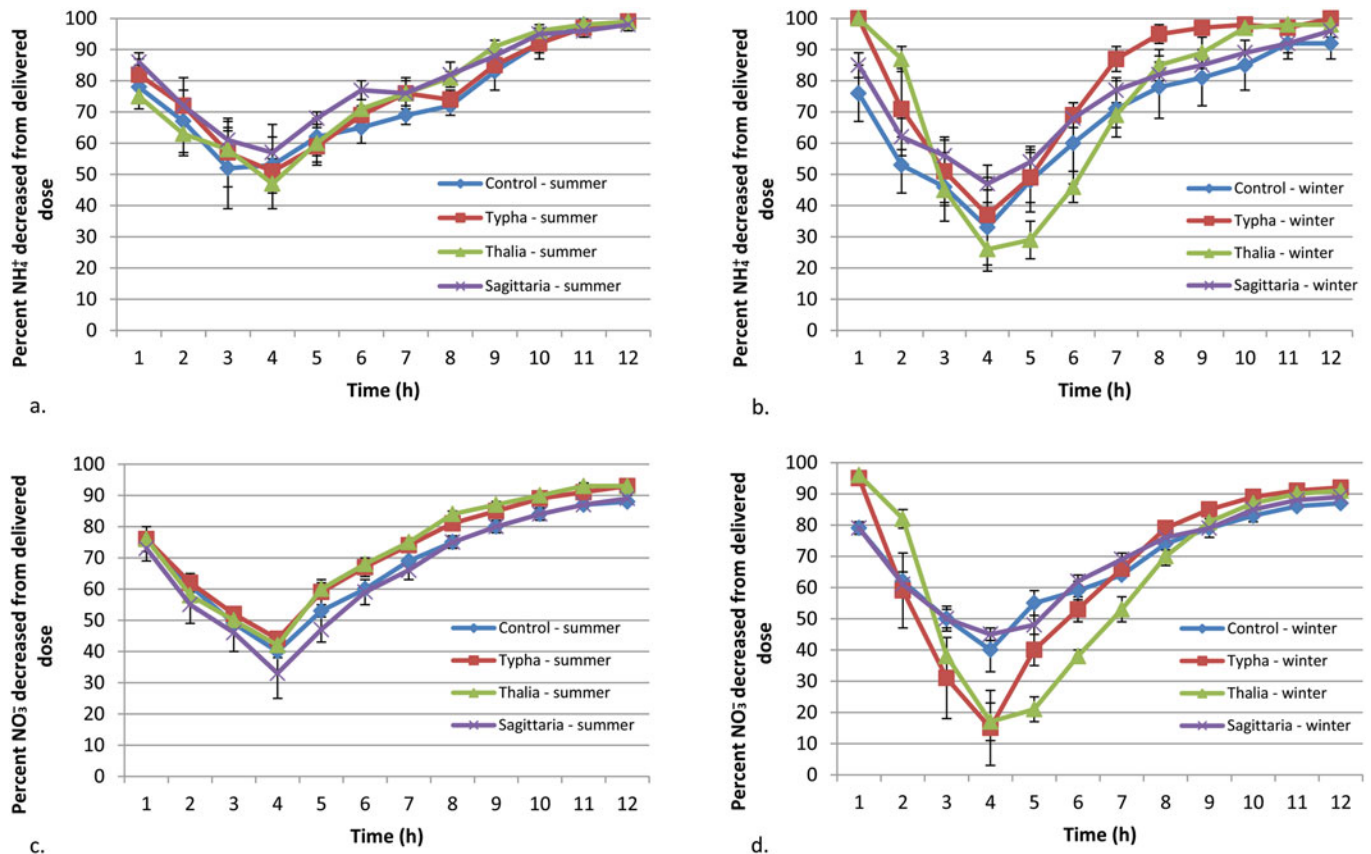


FIGURE 1 Mean percent load decreases (\pm SE) in NH_4^+ (a, b) and NO_3^- (c, d) during summer and winter exposures, respectively. (Figure available in color online.)

treatments in summer or winter were noted. Summer plant efficiencies were $T. latifolia = T. dealbata >$ nonvegetated control $> S. latifolia$. Winter plant efficiencies essentially reversed to $S. latifolia >$ nonvegetated control $> T. latifolia > T. dealbata$ (Figure 1c, d).

Nitrogen Loads

Decreases in both NH_4^+ and NO_3^- loads followed concentration decreases and were variable among plant species (Tables 1 and 2). No interactions between vegetation and season were observed in percent load decreases of NH_4^+ , either at the exposure peak (4 h) ($p = .5935$) or overall experiment (12 h) ($p = .5704$). Overall summer load reductions ranged from 24% in the nonvegetated control to 39% in $S. latifolia$. Overall summer plant efficiencies for NH_4^+ load decreases at 12 h were $S. latifolia > T. latifolia = T. dealbata >$ nonvegetated control. Winter overall load reductions ranged from 4% in the nonvegetated control to nearly 38% in $T. latifolia$. Winter plant efficiencies after the 12-h exposure were $T. latifolia > S. latifolia > T. dealbata$

$>$ nonvegetated control. Although statistically significant differences in inflow loads between species were observed ($p < .0001$), differences were not biologically significant to the overall function and efficiency of the mesocosm systems. In addition, these differences did not translate into significant differences in percent load reductions of NH_4^+ .

No significant interaction between vegetation and season was observed in the percent decrease of NO_3^- load during the peak 4-h exposure ($p = .3373$). Nor were there any significant differences noted due to vegetation ($p = .9197$). However, a significant interaction between vegetation and season was observed after 12 h ($p = .0021$), with both $T. latifolia$ and $T. dealbata$ decreasing NO_3^- loads significantly more in summer than in winter (Table 2). No other significant differences between mesocosm vegetation were noted. Overall summer plant efficiencies after the 12-h exposure for NO_3^- load decreases were $T. latifolia = T. dealbata >$ nonvegetated control $> S. latifolia$. Winter overall NO_3^- load reductions ranged from -5% ($T. dealbata$) to 12% ($S. latifolia$). Winter plant efficiencies after the 12-h exposure

TABLE 1 Inflow and Outflow Load Decreases \pm Standard Deviation (mg) for Ammonium (NH_4^+) in Summer and Winter Exposures via the Monospecific Obligate Wetland Vegetation and Nonvegetated Control

	Control	<i>T. latifolia</i>	<i>T. dealbata</i>	<i>S. latifolia</i>
Summer				
Inflow	32 \pm 9.5	24 \pm 8.8	21 \pm 7.1	30 \pm 6.5
Total outflow	23 \pm 1.1	17 \pm 5.9	14 \pm 3.2	18 \pm 5.0
Outflow 0–4 h	11 \pm 2.3	7.5 \pm 2.4	7.6 \pm 2.0	8.6 \pm 2.6
Outflow 4–8 h	9.9 \pm 0.94	7.1 \pm 3.0	5.3 \pm 1.4	7.6 \pm 2.2
Outflow 8–12 h	2.5 \pm 0.98	1.9 \pm 1.4	1.0 \pm 0.68	1.8 \pm 1.9
% Decrease at peak (4 h)	63 \pm 18	65 \pm 16	59 \pm 14	69 \pm 12
Total % decrease (12 h)	24 \pm 21	28 \pm 20	28 \pm 18	39 \pm 17
Winter				
Inflow	35 \pm 6.0	27 \pm 4.1	21 \pm 4.1	37 \pm 2.6
Total outflow	32 \pm 15	16 \pm 6.0	17 \pm 4.8	27 \pm 7.9
Outflow 0–4 h	16 \pm 5.7	8.9 \pm 4.3	7.0 \pm 1.1	13 \pm 3.0
Outflow 4–8 h	12 \pm 6.0	6.8 \pm 3.5	8.6 \pm 2.8	10 \pm 2.6
Outflow 8–12 h	3.7 \pm 3.8	0.64 \pm 1.0	0.96 \pm 1.0	3.2 \pm 2.4
% Decrease at peak (4 h)	52 \pm 20	65 \pm 21	66 \pm 8.1	63 \pm 11
Total % decrease (12 h)	4.3 \pm 52	38 \pm 26	20 \pm 20	25 \pm 25

were *S. latifolia* > nonvegetated control > *T. latifolia* > *T. dealbata*.

Phosphorus Concentrations

Average percent decreases in DIP during the summer ranged from 56 \pm 2% (nonvegetated control) to 66 \pm 2% (*T. dealbata*). In the winter, DIP decreases ranged from 51 \pm 10% (*T. dealbata*) to 63 \pm 2% (*S. latifolia*). No significant interactions were observed between vegetation and season ($p = .0935$). Nor did vegetation

($p = .7018$) or season ($p = .5875$) influence DIP concentration independently of each other. Plant species listed from greatest to least DIP removal efficiency in the summer were *T. dealbata* > *T. latifolia* > *S. latifolia* > nonvegetated control. Efficiency in the winter was *S. latifolia* > *T. latifolia* = nonvegetated control > *T. dealbata* (Figure 2a, b).

TIP concentration decreases ranged from 44 \pm 4% (*T. latifolia*) to 50 \pm 2% (*T. dealbata*) in summer exposures, whereas winter exposure decreases ranged from 48 \pm 3% (nonvegetated control) to 62 \pm 1% (*T. latifolia*).

TABLE 2 Inflow and Outflow Load Decreases \pm Standard Deviation (mg) for Nitrate (NO_3^-) in Summer and Winter Exposures via the Monospecific Obligate Wetland Vegetation and Nonvegetated Control

	Control	<i>T. latifolia</i>	<i>T. dealbata</i>	<i>S. latifolia</i>
Summer				
Inflow	1398 \pm 123.3	1114 \pm 317.8	942.2 \pm 187.4	1262 \pm 129.8
Total outflow	1198 \pm 112.0	792.9 \pm 211.0	675.6 \pm 134.1	1113 \pm 64.33
Outflow 0–4 h	552.0 \pm 63.67	379.0 \pm 97.57	365.7 \pm 86.46	531.3 \pm 125.8
Outflow 4–8 h	452.3 \pm 31.46	288.8 \pm 96.47	229.0 \pm 54.24	419.9 \pm 64.98
Outflow 8–12 h	193.6 \pm 25.85	108.6 \pm 43.46	75.06 \pm 15.25	161.7 \pm 33.42
% Decrease at peak (4 h)	56.25 \pm 1.258	59.00 \pm 5.692	56.33 \pm 4.885	51.67 \pm 15.28
Total % decrease (12 h)	14.25 \pm 2.986	28.33 \pm 6.919	28.00 \pm 4.427	10.83 \pm 12.73
Winter				
Inflow	1003 \pm 898.0	722.6 \pm 640.4	552.5 \pm 469.3	955.3 \pm 803.5
Total outflow	935.0 \pm 889.0	650.3 \pm 540.8	562.2 \pm 457.8	811.3 \pm 666.0
Outflow 0–4 h	452.4 \pm 450.5	340.8 \pm 311.8	246.5 \pm 221.9	392.9 \pm 343.1
Outflow 4–8 h	353.4 \pm 329.7	234.9 \pm 196.0	262.8 \pm 201.9	294.0 \pm 226.2
Outflow 8–12 h	129.1 \pm 109.0	56.17 \pm 46.16	52.96 \pm 35.91	124.5 \pm 99.59
% Decrease at peak (4 h)	57.50 \pm 7.767	49.83 \pm 21.87	58.33 \pm 8.664	59.17 \pm 4.167
Total % decrease (12 h)	7.750 \pm 11.00	4.000 \pm 20.01	–4.833 \pm 8.035	12.00 \pm 8.922

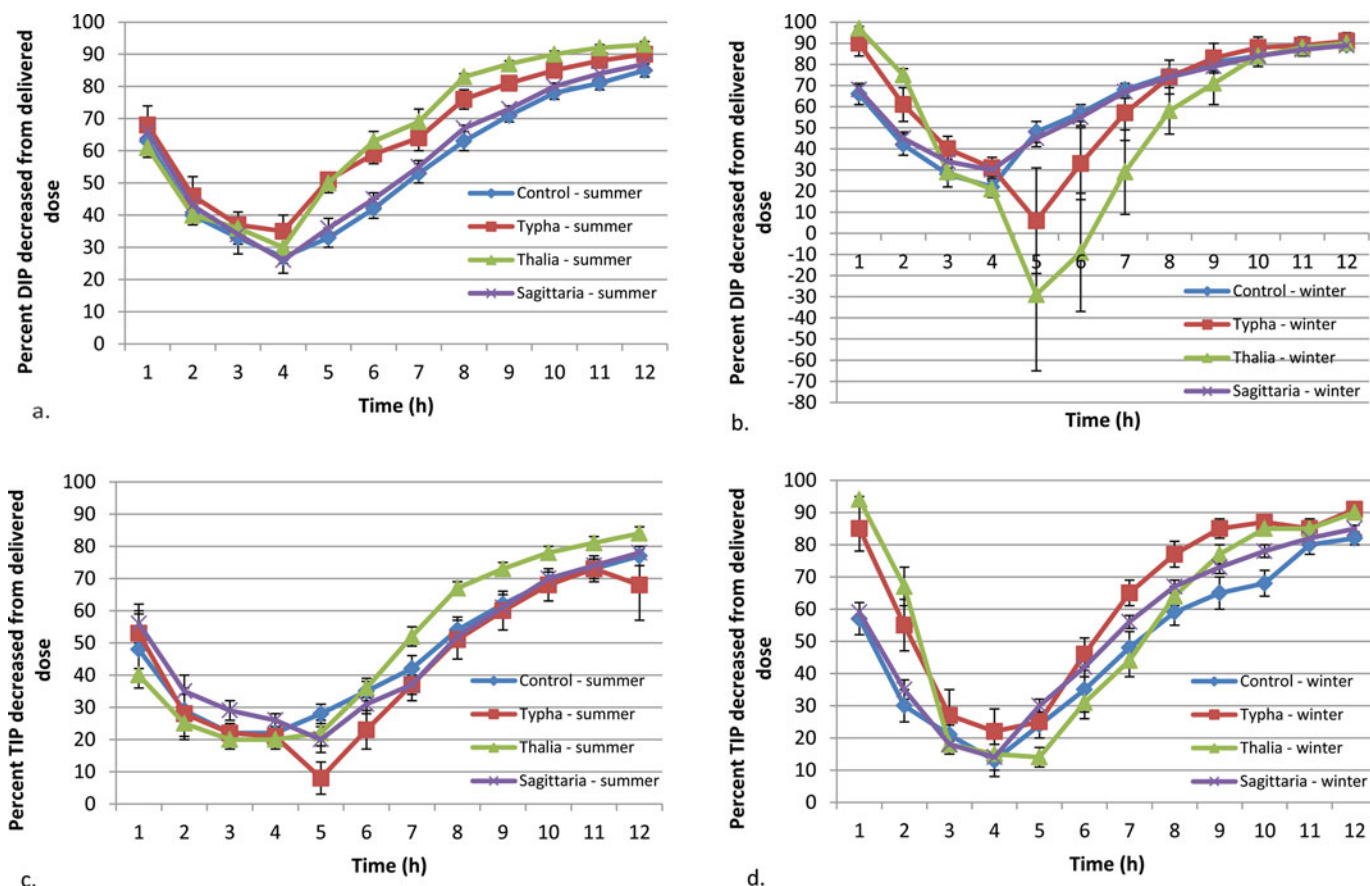


FIGURE 2 Mean percent load decreases (\pm SE) in DIP (a, b) and TIP (c, d) during summer and winter exposures, respectively. (Figure available in color online.)

A significant interaction was observed between vegetation and season ($p = .0104$), with *T. latifolia* decreasing TIP concentration significantly more in winter than in summer. In winter, *T. latifolia* decreased TIP concentration significantly more than nonvegetated controls ($p = .0120$). No differences were observed between mesocosms in summer exposures ($p \geq .6190$). Summer plant efficiencies for TIP decreases were *T. dealbata* > *S. latifolia* = nonvegetated control > *T. latifolia*. Winter plant efficiencies were *T. latifolia* > *T. dealbata* > *S. latifolia* > nonvegetated control (Figure 2c, d).

Phosphorus Loads

Significant differences between vegetation were noted in both inflow and outflow loads for both DIP and TIP ($p \leq .0002$). Analysis of load decreases centered on the percent decrease in loads relative to the inflow amount of each treatment. The possibility still remains that these differences in inflow may impact significant differences observed between mesocosm ef-

ficiencies. However, these differences are less than 1 order of magnitude and likely not of any biological significance to the efficiency of the mesocosm systems.

Decreases in both DIP and TIP loads, like NH_3 and NO_3^- loads, followed concentration decreases and were variable among plant species (Tables 3 and 4). Overall summer load reductions ranged from -18% in nonvegetated control to 13% in *T. dealbata*. Overall summer plant efficiencies after 12 h for load decreases in DIP were *T. dealbata* > *T. latifolia* > *S. latifolia* > nonvegetated control. Winter overall load reductions ranged from -47% in *T. dealbata* to -10% in *S. latifolia*. Winter plant efficiencies after the 12-h exposure were *S. latifolia* > nonvegetated control > *T. latifolia* > *T. dealbata*. No interactions between vegetation and plant species were observed in the percent decrease in DIP loads after the exposure peak (4 h) ($p = .1995$), nor the overall experiment (12 h) ($p = .0953$). When comparing plant species independently of season, *T. latifolia* lowered DIP concentrations significantly more than nonvegetated controls ($p = .0380$) after the 4-h

TABLE 3 Inflow and Outflow Load Decreases \pm Standard Deviation (mg) for Dissolved Inorganic Phosphorus (DIP) in Summer and Winter Exposures via the Monospecific Obligate Wetland Vegetation and Nonvegetated Control

	Control	<i>T. latifolia</i>	<i>T. dealbata</i>	<i>S. latifolia</i>
Summer				
Inflow	532 \pm 70.5	423 \pm 136	347 \pm 76.3	504 \pm 54.7
Total outflow	623 \pm 78.0	381 \pm 136	297 \pm 53.4	546 \pm 36.3
Outflow 0–4 h	276 \pm 21.7	188 \pm 56.1	170 \pm 26.4	250 \pm 31.3
Outflow 4–8 h	244 \pm 31.2	139 \pm 56.7	99.9 \pm 24.3	213 \pm 7.89
Outflow 8–12 h	102 \pm 26.1	51.3 \pm 28.2	27.2 \pm 6.85	83.3 \pm 33.42
% Decrease at peak (4 h)	40.8 \pm 6.13	46.5 \pm 11.9	41.8 \pm 7.57	42.3 \pm 8.59
Total % decrease (12 h)	–17.8 \pm 12.8	10.3 \pm 14.4	12.7 \pm 13.0	–10.0 \pm 11.7
Winter				
Inflow	463 \pm 68.4	359 \pm 46.4	278 \pm 29.1	444 \pm 34.3
Total outflow	518 \pm 107	405 \pm 154	402 \pm 175	486 \pm 60.6
Outflow 0–4 h	277 \pm 52.9	157 \pm 24.9	123 \pm 5.23	245 \pm 16.1
Outflow 4–8 h	178 \pm 56.0	203 \pm 128	235 \pm 147	174 \pm 34.5
Outflow 8–12 h	63.4 \pm 19.1	44.2 \pm 42.4	43.3 \pm 30.5	66.8 \pm 17.6
% Decrease at peak (4 h)	39.5 \pm 10.3	55.0 \pm 10.1	55.3 \pm 4.37	44.5 \pm 3.51
Total % decrease (12 h)	–11.8 \pm 12.3	–14.0 \pm 44.6	–47.0 \pm 69.9	–9.50 \pm 12.2

peak exposure period. However, by the end of the 12-h exposure, no differences between mesocosms were observed ($p = .6897$).

A significant interaction between vegetation and season was noted in TIP percent load decreases after both peak exposure (4 h) ($p = .0027$) and duration of the experiment (12 h) ($p = .0042$). The percent decrease in TIP observed in *T. dealbata* mesocosms after the 4-h peak exposure was significantly higher in winter than in summer ($p = .0040$). In contrast, total load decreases associated with *T. dealbata* after the 12-h exposure did

not differ significantly across season ($p = .9963$). An opposite trend was observed in *T. latifolia* mesocosms, where no significant differences were observed across seasons during the 4-h peak exposure. However, total percent decreases in TIP load after 12 h were higher in *T. latifolia* mesocosms in winter than in summer. In relation to individual species, *T. dealbata* decreased TIP load significantly more than *S. latifolia* after 4 h in the winter ($p = .0456$) (Table 4). At the end of the 12-h exposure, nonvegetated mesocosms decreased TIP load significantly more than *T. latifolia* mesocosms in winter.

TABLE 4 Inflow and Outflow Load Decreases \pm Standard Deviation (mg) for Total Inorganic Phosphorus (TIP) in Summer and Winter Exposures via the Monospecific Obligate Wetland Vegetation and Nonvegetated Control

	Control	<i>T. latifolia</i>	<i>T. dealbata</i>	<i>S. latifolia</i>
Summer				
Inflow	633 \pm 73.2	450 \pm 125	382 \pm 78.1	584 \pm 17.4
Total outflow	790 \pm 92.5	633 \pm 238	456 \pm 91.4	748 \pm 90.6
Outflow 0–4 h	345 \pm 34.3	251 \pm 67.6	223 \pm 39.0	301 \pm 40.6
Outflow 4–8 h	298 \pm 38.6	263 \pm 106	170 \pm 38.3	307 \pm 31.7
Outflow 8–12 h	145 \pm 25.7	118 \pm 67.2	63.5 \pm 17.1	133 \pm 26.1
% Decrease at peak (4 h)	30.3 \pm 10.7	30.5 \pm 11.0	26.7 \pm 5.72	36.2 \pm 8.57
Total % decrease (12 h)	–25.5 \pm 15.2	–39.0 \pm 27.3	–20.2 \pm 15.7	–28.5 \pm 18.9
Winter				
Inflow	501 \pm 65.7	399 \pm 46.6	303 \pm 28.8	487 \pm 41.7
Total outflow	753 \pm 130	430 \pm 55.7	381 \pm 18.6	652 \pm 56.3
Outflow 0–4 h	344 \pm 46.7	198 \pm 33.4	150 \pm 20.9	318 \pm 20.1
Outflow 4–8 h	290 \pm 59.2	181 \pm 47.2	186 \pm 11.2	238 \pm 25.8
Outflow 8–12 h	120 \pm 34.4	106 \pm 130	46.1 \pm 7.91	95.9 \pm 16.8
% Decrease at peak (4 h)	30.0 \pm 5.94	47.2 \pm 12.9	48.5 \pm 9.05	31.5 \pm 5.13
Total % decrease (12 h)	–50.3 \pm 19.2	–7.50 \pm 3.94	–27.0 \pm 14.1	–34.2 \pm 10.8

No significant differences between vegetation were observed in summer experiments after 4 or 12 h ($p \geq .5334$). Summer overall TIP load reductions ranged from -39% (*T. latifolia*) to -20% (*T. dealbata*). Overall summer plant efficiencies for TIP load decreases at 12 h were $T. dealbata > \text{nonvegetated control} > S. latifolia > T. latifolia$. Winter overall TIP load reductions ranged from -50% (nonvegetated control) to -8% (*T. latifolia*). Winter plant efficiencies after the 12-h exposure were $T. latifolia > T. dealbata > S. latifolia > \text{nonvegetated control}$.

DISCUSSION

Scientific literature acknowledges the ability of aquatic macrophytes to assimilate nutrients either through uptake or microbial decomposition on biofilms (Brix and Schlerup 1989; Bachand and Horne 2000; Hoagland et al. 2001; García-Lledó et al. 2011). According to Silvan, Vasander, and Laine (2004), colonizing and growing vegetation in wetlands is the main factor in N retention, while serving as a significant factor for P retention. Questions arise, however, as to specific plant uptake ability, whether or not that ability changes with changing seasons, and the influence of water residence time.

Studies vary as to the exact role plants play in nitrogen speciation removal in wetland systems. Reddy and DeBusk (1985) reported that plant uptake was responsible for 16–75% of total N removed in retention ponds. Wetland N removal of 47% was noted by Hoagland et al. (2001), whereas Fink and Mitsch (2004) saw a decrease of 41% of total N by mass in wetlands. Gottschall et al. (2007) reported that 9% of total Kjeldahl N uptake was attributed to plants. Data from the current study suggest that plant uptake is at least partially responsible for decreases in NO_3^- and TIP concentrations and loads, although a complete mass-balance determination was not possible because of sampling limitations of sediment and plants in mesocosms.

In earlier studies using some of the same types of vegetation, Moore and Kröger (2011) reported NH_4^+ load decreases at the experiment peak (4 h) of $56 \pm 3\%$, $67 \pm 0.2\%$, and $65 \pm 3\%$ for *T. dealbata*, *T. latifolia*, and nonvegetated controls, respectively, during summer exposures. Ammonium peak load reductions for the same vegetation in the current study were comparable for summer exposures (Table 1). Deaver et al. (2005) determined that *Ludwigia peploides* (yellow primrose) was

capable of reducing NH_4^+ concentrations by 83% from inflow to outflow in mesocosms similar to those used in the current study, again during summer exposures. A comparison between unplanted mesocosms with zero fluctuation and mesocosms with plants and zero fluctuation revealed a 44% increase in mean NH_4^+ -N removal in the presence of wetland plants (Tanner et al. 1999).

Research is much more abundant on NO_3^- removal by wetlands or associated plant uptake. Moore and Kröger (2011) found NO_3^- load decreases of $56 \pm 3\%$, $50 \pm 4\%$, and $60 \pm 2\%$, for *T. dealbata*, *T. latifolia*, and nonvegetated controls, respectively, at the peak (4 h) of a summer experimental exposure. These decreases correspond closely with those found in the current study during the summer (Table 2). However, overall percent decreases were slightly better in the current study for both *T. dealbata* and *T. latifolia* when compared with results from Moore and Kröger (2011). A 74% decrease in NO_3^- mass after wetland processing was discovered by Fink and Mitsch (2007), although an earlier study demonstrated only a 41% mass retention of NO_3^- - NO_2^- in wetlands studied over 2 years (Fink and Mitsch 2004). Kadlec (2010) reported an average wetland NO_3^- mass removal of 67% following 4 years of monitoring 28 events, even though individual-event removals ranged from 17% to 100%. The current study's NO_3^- load decreases at 4 h ranged from 52% to 59%, whereas those with 4-h residence times in Kadlec's (2010) study ranged from 74% to 86%.

Contributions of ammonification and denitrification to N removal have also been examined. García-Lledó et al. (2011) reported ammonification rates were responsible for approximately 10% of NO_3^- reduction in wetlands, and this was particularly evident in areas with vegetation. Bachand and Horne (2000) acknowledged that little is known about the denitrification ability and role of specific plants; however, they reported that NO_3^- removal could be achieved utilizing mixtures of grasses in addition to emergent, floating, and submerged macrophytes. Floating and submerged aquatic macrophytes also aid in the denitrification process by providing increased organic carbon availability. Additionally, N removal should increase over time as vegetation establishes and carbon stores build up (Bachand and Horne 2000). Although denitrification rates were not calculated in the current study, results concur with those from Bachand and Horne (2000) regarding the need for plant mixtures in order to

optimize nutrient removal, since no single plant was most effective.

According to studies from field ditches in the lower Mississippi River Valley, vegetated drainage ditches possessed 1.3 times the denitrification potential of un-vegetated ditches (Ullah and Faulkner 2006). Kröger et al. (2007) saw a decrease of 57% of dissolved inorganic N (DIN) ($\text{NH}_4^+ + \text{NO}_3^- + \text{NO}_2^-$) transport from fields after traveling through vegetated drainage ditches. Although the current study did not evaluate NO_2^- loads or concentrations, the other combined components of DIN yielded similar results with *T. dealbata*, *T. latifolia*, and *S. latifolia*. Ahiablame et al. (2011) determined that ditch uptake lengths for concentrations of NO_3^- -N were relatively long (sometimes up to 2300 m); therefore, the particular ditch drainage systems were not efficient at assimilating nutrients. Ditch descriptions for Ahiablame et al. (2011) indicated that riparian ditch vegetation was present, but no in-ditch vegetation was described. García-Lledó et al. (2011) noted that *T. latifolia*-dominated wetlands had greater N removal efficiencies than other wetlands dominated by different plant species receiving comparable hydraulic loads. Based on the current literature, it is evident that both drainage ditch length and vegetation are important factors in nutrient mitigation.

Because of the concerns with senescence and leaching of nutrients back into receiving water systems, the current study examined nutrient removal in systems during both summer and winter exposures. Two separate studies, Reddy and DeBusk (1985) and Pei, Yang, and Tian (2010), both noted that N and NO_3^- removals, respectively, were more effective in the summer than winter months. In both cases, the current study affirms those results (Tables 1 and 2).

Several studies have reported on the success of wetlands at decreasing various forms of P in effluent waters. Hoagland et al. (2001) reported P removal of 29%, whereas Fink and Mitsch (2004) saw 28% of total P and 74% of soluble reactive P (SRP or DIP) mass retained. The current study revealed much lower percent DIP mass retained in mesocosms (Table 3). Fink and Mitsch (2007) later found less mass retention of TP and DIP by wetlands (31% and 46%, respectively), although these values were still greater than those observed in the current study. Seasonal wetland P removal was examined by Maynard, O'Geen, and Dahlgren (2009) in two wetlands (2.3 and 7.3 ha) with hydraulic retention times of 11 and 31 h. Results of that study demonstrated wet-

land removal of 55–65% of particulate P (PP), 61–63% of dissolved reactive P (DRP or DIP), and 57–62% of bioavailable P (BAP). DIP reductions from the current study (Table 3) were much less effective than those reported by Maynard, O'Geen, and Dahlgren (2009). Moore and Kröger (2011) found percent decreases of DIP loads at the experiment peak (4 h) ranging from $40 \pm 1\%$ (*T. latifolia*) to $56 \pm 3\%$ (*T. dealbata*), with non-vegetated controls having a decrease of $41 \pm 7\%$ during summer exposures. In the current study, summer DIP load decreases at experiment peak (4 h) were $46.5 \pm 11.9\%$, $41.8 \pm 7.57\%$, and $40.8 \pm 6.13\%$ for *T. latifolia*, *T. dealbata*, and the nonvegetated controls, respectively. For TIP percent load reductions at experiment peak (4 h), Moore and Kröger (2011) reported values of $19 \pm 15\%$, $28 \pm 8\%$, and $20 \pm 8\%$ for *T. latifolia*, *T. dealbata*, and the nonvegetated controls, respectively. In the current study, summer TIP percent load reductions at experimental peak (4 h) were generally greater at $30.5 \pm 11.0\%$, $26.7 \pm 5.72\%$, and $30.3 \pm 10.7\%$ for *T. latifolia*, *T. dealbata*, and the nonvegetated controls, respectively; however, current values fell within the standard deviation from Moore and Kröger (2011).

Removal of P by vegetation from wastewaters can vary widely, from 5% (Gottschall et al. 2007) up to 73% (Reddy and DeBusk 1985; Cooke 1992; Greenway and Woolley 2000). Ultimate success of plant P uptake depends on several factors, including plant species, overall nutrient loading rates, and the particular composition of the wastewater (Gottschall et al. 2007). Kröger et al. (2008) and Smith (2009) both examined P removal through agricultural drainage ditches. After monitoring three vegetated agricultural drainage ditches in Mississippi, Kröger et al. (2008) reported a 44% decrease in inorganic P exiting drainage ditches. Ditch uptake lengths for P ranged from 40 to 1900 m and were correlated with ditch width and the sediment properties of percent organic matter, equilibrium P concentration, and readily exchangeable P (Smith 2009).

Current study results demonstrate no single aquatic emergent plant is most effective at N and P mitigation. For this reason, a mixture of vegetation in ditches would be the optimal management strategy. *Thalia dealbata* mesocosms decreased summer NO_3^- concentrations significantly more than in winter ($p = .0023$). Additionally, both *T. dealbata* and *T. latifolia* NO_3^- loads in summer were decreased significantly more than in winter ($p = .0021$). Conversely, *T. latifolia* mesocosms significantly decreased TIP concentrations

in winter, more than in summer ($p = .0104$). Although the current study utilized a 4-h water residence time, Kröger et al. (2011) demonstrated the use of low-grade weirs in ditches to increase residence time resulted in significantly higher NO_3 reductions.

Effective nutrient mitigation from intensive agricultural areas will require more than simply a silver bullet solution. Because year-round nutrient mitigation is complicated in vegetated ditches and wetlands due to plant senescence and re-release of nutrients into the water, using ditches to remove nutrients is only part of the overall suggested management plan. Truly effective mitigation will require proper management of nutrient inputs, installation and upkeep of multiple, successful edge-of-field management practices (e.g., vegetated drainage ditches, denitrification beds, etc.), balance of optimal runoff water residence time utilizing low-grade weirs when available, and routine monitoring for effectiveness of the system.

Due to lag time for water quality improvement (Meals, Dressing, and Davenport 2010) and the magnitude of the eutrophication issue, nutrient mitigation will take time. Turner and Rabalais (2003) concluded that the significant water quality changes occurring in the Mississippi River Basin over the last 200 years will in turn take decades of collaboration for improvement to be successful. Patience and perseverance must be exercised in order to face the water quality challenges of the 21st century.

ACKNOWLEDGMENTS

The authors thank especially Lisa Brooks, Toyal Flemons, and Landon Lee for analytical analyses and sample collection assistance. Mention of trade names or commercial products in this article is solely for the purpose of providing specific information and does not imply recommendation or endorsement by the US Department of Agriculture (USDA). The USDA is an equal opportunity employer and provider.

REFERENCES

- Ahiablame, L. M., I. Chaubey, D. R. Smith, and B. A. Engel. 2011. Effect of tile effluent on nutrient concentration and retention efficiency in agricultural drainage ditches. *Agric. Water Manage.* 98:1271–1279.
- American Public Health Association (APHA). 1998. *Standard methods for the examination of water and wastewater*, 20th ed. Washington, DC: APHA.
- Bachand, P. A. M., and A. J. Horne. 2000. Denitrification in constructed free-water surface wetlands: II. Effects of vegetation and temperature. *Ecol. Eng.* 14:17–32.
- Brix, H., and H. H. Scherlup. 1989. The use of aquatic macrophytes in water-pollution control. *Ambio* 18:100–107.
- Cooke, J. G. 1992. Phosphorus removal process in a wetland after a decade of receiving a sewage effluent. *J. Environ. Qual.* 21:733–739.
- Dagg, M. J., and G. A. Breed. 2003. Biological effects of Mississippi River nitrogen on the Northern Gulf of Mexico—A review and synthesis. *J. Marine Syst.* 43:133–152.
- Deaver, E., M. T. Moore, C. M. Cooper, and S. S. Knight. 2005. Efficiency of three aquatic macrophytes in mitigating nutrient runoff. *Int. J. Ecol. Environ. Sci.* 31:1–7.
- Dodds, W. K., W. W. Bouska, J. L. Eitzmann, T. J. Pilger, K. L. Pitts, A. J. Riley, J. T. Schloesser, and D. J. Thornbrugh. 2009. Eutrophication of U.S. freshwaters: Analysis of potential economic damages. *Environ. Sci. Technol.* 43:12–19.
- Dodds, W. K., and E. B. Welch. 2000. Establishing nutrient criteria in streams. *J. North Am. Benthol. Soc.* 19:186–196.
- Fink, D. F., and W. J. Mitsch. 2004. Season and storm event nutrient removal by a created wetland in an agricultural watershed. *Ecol. Eng.* 23:313–325.
- Fink, D. F., and W. J. Mitsch. 2007. Hydrology and nutrient biogeochemistry in a created river diversion oxbow wetland. *Ecol. Eng.* 30:93–102.
- Fixen, P. E., and F. B. West. 2002. Nitrogen fertilizers: Meeting contemporary challenges. *Ambio* 31:169–176.
- Galloway, J. N., J. D. Aber, J. W. Erisman, S. P. Seitzinger, R. W. Howarth, E. B. Cowling, and B. J. Cosby. 2003. The nitrogen cascade. *BioScience* 53:341–356.
- García-Lledó, A., O. Ruiz-Rueda, A. Vilar-Sanz, L. Sala, and L. Bañeras. 2011. Nitrogen removal efficiencies in a free water surface constructed wetland in relation to plant coverage. *Ecol. Eng.* 37:678–684.
- Gottschall, N., C. Coutin, A. Crolla, C. Kinsley, and P. Champagne. 2007. The role of plants in the removal of nutrients at a constructed wetland treating agricultural (dairy) wastewater, Ontario, Canada. *Ecol. Eng.* 29:154–163.
- Greenway, M., and A. Woolley. 2000. Changes in plant biomass and nutrient removal over 3 years in a constructed free water surface flow wetland in Cairns Australia. In *Proceedings of the Seventh International Conference on Wetland Systems for Water Pollution Control, International Water Association, Lake Buena Vista, Florida*, November 11–16, 2000, eds. K. R. Reddy and R. H. Kadlec, 707–718. Gainesville, Florida: Institute of Food and Agricultural Sciences, University of Florida.
- Hoagland, C. R., L. E. Gentry, M. B. David, and D. A. Kovacic. 2001. Plant nutrient uptake and biomass accumulation in a constructed wetland. *J. Freshwater Ecol.* 16:527–540.
- Howarth, R. W., E. W. Boyer, W. J. Pabich, and J. N. Galloway. 2002. Nitrogen use in the United States from 1961–2000 and potential future trends. *Ambio* 31:88–96.
- Kadlec, R. H., 2010. Nitrate dynamics in event-driven wetlands. *Ecol. Eng.* 36:503–516.
- Kröger, R., M. M. Holland, M. T. Moore, and C. M. Cooper. 2007. Hydrological variability and agricultural drainage ditch inorganic nitrogen reduction capacity. *J. Environ. Qual.* 36:1646–1652.
- Kröger, R., M. M. Holland, M. T. Moore, and C. M. Cooper. 2008. Agricultural drainage ditches mitigate phosphorus loads as a function of hydrological variability. *J. Environ. Qual.* 37:107–113.
- Kröger, R., S. C. Pierce, K. A. Littlejohn, M. T. Moore, and J. L. Farris. 2011. Decreasing nitrate-N loads to coastal ecosystems with innovative drainage management strategies in agricultural landscapes: An experimental approach. *Agric. Water Manage.* 103: 162–166.
- Maynard, J. J., A. T. O'Geen, and R. A. Dahlgren. 2009. Bioavailability and fate of phosphorus in constructed wetlands receiving

- agricultural runoff in the San Joaquin Valley, California. *J. Environ. Qual.* 38:360–372.
- Meals, D. W., S. A. Dressing, and T. E. Davenport. 2010. Lag time in water quality response to best management practices: A review. *J. Environ. Qual.* 39:85–96.
- Moore, M. T., and R. Kröger. 2011. Evaluating plant species-specific contributions to nutrient mitigation in drainage ditch mesocosms. *Water Air Soil Pollut.* 217:445–454.
- Murphy, R., and J. P. Riley. 1962. A modified single solution method for the determination of phosphate in natural waters. *Anal. Chim. Acta* 27:31–36.
- Pei, Y., Z. Yang, and B. Tian. 2010. Nitrate removal by microbial enhancement in a riparian wetland. *Bioresour. Technol.* 101:5712–5718.
- Reddy, K. R., and W. F. DeBusk. 1985. Nutrient removal potential of selected aquatic macrophytes. *J. Environ. Qual.* 14:459–462.
- Silvan, N., H. Vasander, and J. Laine. 2004. Vegetation is the main factor in nutrient retention in a constructed wetland buffer. *Plant Soil* 258:179–187.
- Smith, D. R. 2009. Assessment of in-stream phosphorus dynamics in agricultural drainage ditches. *Sci. Total Environ.* 407:3883–3889.
- Snyder, C. S. 2008. Nutrients and hypoxia in the Gulf of Mexico—An update on progress, 2008. *Better Crops With Plant Food* 92(2):16–22.
- Tanner, C. C., J. D'Eugenio, G. B. McBride, J. P. S. Sukias, and K. Thompson. 1999. Effect of water level fluctuation on nitrogen removal from constructed wetland mesocosms. *Ecol. Eng.* 12:67–92.
- Tilman, D. 1999. Global environmental impacts of agricultural expansion: The need for sustainable and efficient practices. *Proc. Natl. Acad. Sci. U. S. A.* 96:5995–6000.
- Turner, R. E., and N. N. Rabalais. 2003. Linking landscape and water quality in the Mississippi River Basin for 200 years. *BioScience* 53:563–572.
- Ullah, S., and S. P. Faulkner. 2006. Denitrification potential of different land-use types in an agricultural watershed, lower Mississippi valley. *Ecol. Eng.* 28:131–140.
- US Department of Agriculture National Agricultural Statistics Service (USDA-NASS). 2009. Census of Agriculture. United States summary and state data. www.agcensus.usda.gov/Publications/2007/Full_Report/usv1.pdf (accessed July 8, 2013).
- US Environmental Protection Agency. 2011. National summary of impaired waters and TMDL information. http://iaspub.epa.gov/waters10/attains_nation.cy.control?p_report.type=T#causes_303d (accessed December 2, 2011).