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Comparison of aerated marsh-pond-marsh and continuous marsh constructed wetlands for treating swine wastewater

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Increased swine production in North Carolina has resulted in greater waste generation and is demanding some emerging new innovative technologies to effectively treat swine wastewater. One of the cost-effective and passive methods to treat swine wastewater is using constructed wetlands. The objective of this study was to evaluate the N removal under two N loads in 3 different wetland systems: aerated marsh-pond-marsh (M-P-M), aerated marsh-covered pond-marsh (M-FB-M), and continuous marsh (CM) with two days drain and five days flood cycle. Swine wastewater from an anaerobic lagoon was applied to the constructed wetland cells (11 m wide x 40 m length) at two N loading rates of 7 and 12 kg N ha^{-1} $day^{-1}$from June to July and August to September 2005, respectively. Weekly inflow and outflow samples were collected for N, P, TS, and COD analysis. Total N reductions (%) at low and high N loading rates were 85.8 and 51.8; 86.3 and 63.3; and 86.2 and 61.8 for M-P-M, M-FB-M, and CM, respectively. Aeration had no significant ($P > 0.05$) impact on N removal. However, significant ($P < 0.05$) differences were observed for wetland systems between low and high N loading rates. No difference ($P > 0.05$) in N reduction was found among wetland systems. Vegetation uptake of N was negligible, ranging from 1.2 to 1.8 %. No significant ($P > 0.05$) differences in TS and COD removal were observed between the wetland systems.

Keywords: Constructed wetlands, marsh-pond-marsh design, continuous marsh, N removal.

Introduction

Over the past decade constructed wetlands have been investigated as means for removing pollutants from wastewater,[1–10] especially from animal wastewater.[5–11] While this approach has been demonstrated to be effective, much remains to be learned about system processes and maximizing the treatment of animal wastewater. Wetland systems used in conjunction with land application can enhance the effectiveness for treating animal wastewater, especially nitrogen (N). Wetland operations can reduce land requirements for wastewater application. Previous studies[9,11] showed that N removal ranged from 40–60% when varied loads of N were applied through anaerobic lagoon swine wastewater to constructed marsh-pond-marsh wetlands. It was observed that oxygen might be a limiting factor for N removal from swine wastewater in constructed wetlands,[12,13] which could be introduced actively (mechanical aeration) or passively (intermittent wetland drainage by alternating the fill and drain cycle) to improve N removal efficiency. Intermittent drainage of wetland mesocosms and constructed wetlands were proved to be effective in treating dairy wastewater[14] and swine wastewater[11] respectively, as compared to continuous wastewater applications. Modification of constructed wetland design may enhance the oxygen input in wastewater and wetland soil substrate.

Assumptions were made in the construction of marsh-pond-marsh (m-p-m) wetland system to increase nitrification (oxidation status) in the pond area by promoting additional surficial oxygen transfer from the air through the open water surface and denitrification in the outflow marsh area.[9,15] However, such wetland system did not increase the N removal in swine wastewater;[16] suggesting that either the N loading was high or the surficial oxygen transfer was not enough to support nitrification. Furthermore, not only removal of N in m-p-m system occurs through a traditional nitrification-denitrification process but also by volatilization of ammonia in the pond section.[16,17] In order to study the impact of reducing ammonia volatilization in the pond section, two of the four m-p-m systems were covered with floating wetlands.[18]

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The objectives of this research were to (i) compare the effectiveness of covered and uncovered aerated ponds and continuous marsh systems for treating anaerobic lagoon swine wastewater and (ii) evaluate the effect of two N loading rates on N and P removal in covered and uncovered aerated ponds of m-p-m system and a continuous marsh system.

Materials and methods

Site description and constructed wetland design

Experiments were conducted using four m-p-m and two continuous marsh wetlands (11m × 40m) at the Swine Research Facility (250 sow farrow-feeder operation) located at the North Carolina Agricultural and Technical State University Farm in Greensboro, North Carolina, USA. The m-p-m wetlands were constructed in 1995 consisting of an 11m W × 10 m L marsh at both the influent and effluent ends with a central pond section 11 m × 20 m (Fig. 1). Both the influent and effluent marshes had a 15 cm water depth, whereas pond section had 75 cm water depth. The marsh sections were planted with broadleaf cattail (*Typha latifolia* L.) and American bulrush [*Schoenoplectus americanus (Pers.) Volkart ex Schinz & R. Keller*] in March 1996. Modifications were done in 2003 on cells 5 and 6 to achieve a continuous marsh system with a slope of 0.33%, from inlet to outlet end and planted with giant bulrushes (*Scirpus californicus*) (Fig. 2).

In 2004, an aeration system was installed in cells 1 through 4 and a floating cover in cells 2 and 3. Floating covers (geosynthetic material recycled closed-cell foam compressed into planks and covered with non-woven fabric) were constructed on-site by Environmental Fabrics Inc., Gaston, SC. The floating cover consisted of 13 strips (1.2 m W × 12 m L) of material sewn together lengthwise. The strips were connected in an alternating series of foam pads facing up and down. The finished material was located over the pond section and anchored on each side of the wetland system (WS) by using steel rods ~ 0.91 m long. A 0.15 m layer of peat soil was placed along each section of the cover where the foam pads face up. Cuttings of giant bulrush obtained from the marsh area of the wetland were planted on this area constituting the covered pond/floating bed (FB).

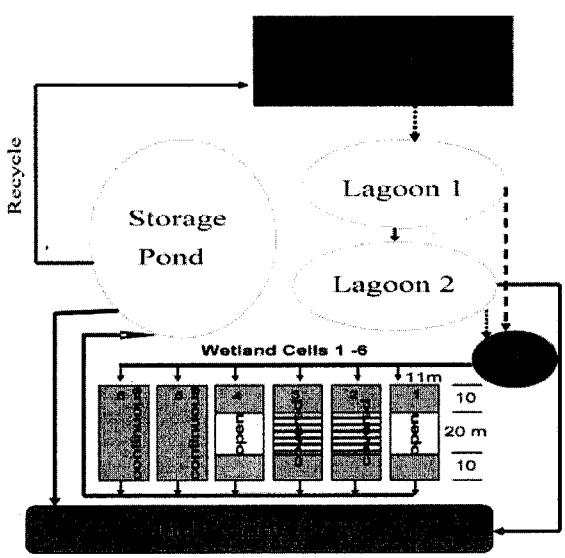

Fig. 1. Schematic (not to scale) of the wetland systems showing the wastewater source and distribution systems.

Experimental design

The study was conducted in two time periods during the summer of 2005. The first period was during June and July with a loading rate of 7 kg N ha^{-1} d^{-1} and 28 days residence time; and the second period was during August and September with a loading rate of 12 kg N ha^{-1} d^{-1} with 14 days residence time. In general, six wetland cells were used to include three wetland systems replicated two times. The three systems were (1) WS-1: cells 1 & 4 (uncovered aerated ponds); (2) WS-2: cells 2 & 3 (covered aerated ponds); and (3) WS-3: cells 5 & 6 with a five-day flood and two-day drain cycle (Fig. 2).

Wastewater holding capacity for WS-1 and WS-2 was ~130 m^3 while that for WS-3 was ~116 m^3. Wastewater from the Swine Research Facility was collected into a two-stage anaerobic lagoon system consisting of a primary lagoon (L1) with overflow into a secondary lagoon (L2) (Fig. 1). Wastewater from lagoon 1 was pumped using a Teel Heavy Duty Shallow Well Jet Pump (Model 2P897B, 1 HP, 1¼″ suction inlet and 1″ discharge outlet) to an 8000 L storage tank and then discharged by gravity to each WS. All discharge from the cells was collected in a storage pond where it was recycled for flushing of the swine production facility and land application.

The cells were equipped with tipping buckets wired with magnetic contacts (MPS-45 WG) and electronic totalizers/volume counters (Veeder-Root brand Series 7999- Mite Totalizer) at the influent and effluent ends to monitor flows. Mechanical flow measurements were verified periodically by manual checks. Cells WS-1 and WS-2 were aerated using two air compressors (GAST Model 1023, 3/4 HP) using four EDI Flex Air™T-Series membrane diffuser calibrated to produce an airflow rate of 70 liters per minute (LPM). The air was delivered to the pond through "H" manifold having four diffusers that rested on concrete blocks at the bottom of the pond.

Weekly wastewater samples were collected from auto samplers (ISCO 2700, Lincoln, NE) located at the inlet source (storage tank) and from the outlet of each cell. The auto samplers were programmed to grab two 70 ml samples per day. Each sampling bottle was acidified with concentrated sulfuric acid to lower the pH below 2.5. Samples were transferred weekly to the laboratory and stored at 4°C until analysis.

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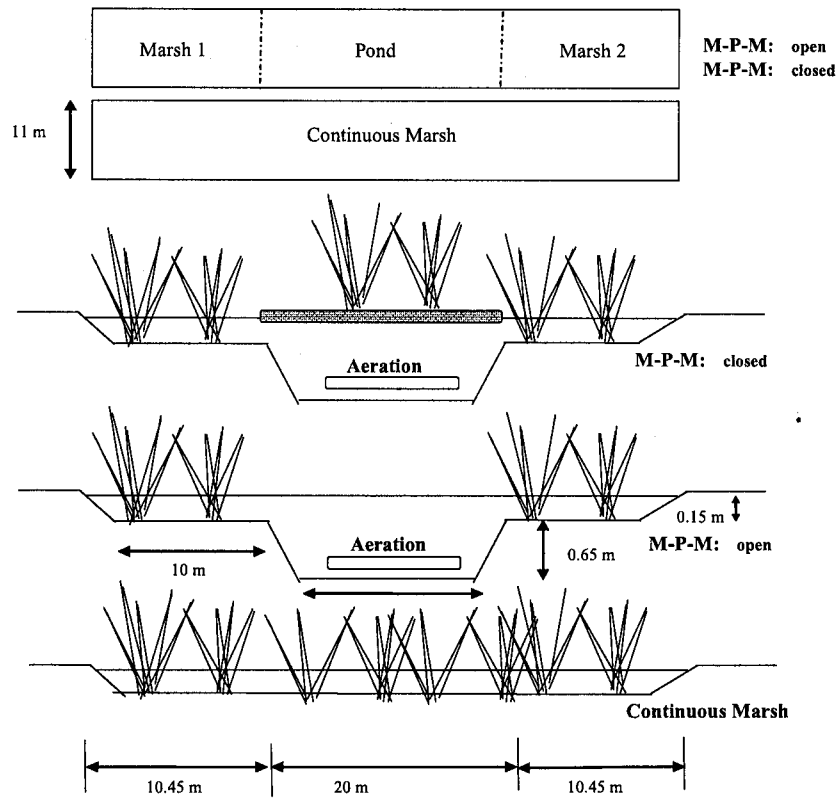

Fig. 2. Schematic (not to scale) of the wetland systems showing marsh-pond-marsh; open and closed ponds and continuous marsh.

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Analysis

Acidified wastewater samples were analyzed for total suspended solids (TSS), ammonium (NH_4-N), nitrate (NO_3-N), total Kjeldahl-N (TKN) and total phosphorus (TP). These samples were analyzed with the USEPA procedures as described by Kopp and McKee.[19] Chemical oxygen demand (COD) was analyzed using the Hach method.[20] Parameters measured on site included cell wastewater temperature, pH, dissolved oxygen, and redox potential.

Weekly wastewater flows and nitrogen concentrations were used to determine weekly rates of constituent mass load and discharge for each cell with the following equation:

$$R_{L/D} = [(C/10^6)^*Q]/A_w \quad (1)$$

where $R_{L/D}$ = nutrient mass load/discharge (kg ha^{-1} day^{-1})

C = Inlet/outlet constituent concentration of WS (mg L^{-1})

Q = average daily wastewater inflow/outflow (L day^{-1})

Aw = wetland area (ha)

Weekly treatment efficiencies (% mass removal) for each cell were determined with the following equation:

$$Eff = [(R_L - R_D)/R_L]^*100 \quad (2)$$

where Eff = treatment efficiency (%)

R_L = average rate of constituent mass load (kg $ha^{-1}day^{-1}$)

R_d = average rate of constituent mass discharge (kg $ha^{-1}day^{-1}$)

Wastewater residence time (wdt) was calculated as:

$$wdt = Vt/Ti \quad (3)$$

where, Ti = total wastewater inflow rate and Vt = total wastewater volume in cell. Ammonia volatilization test were carried out in the second experimental period using a wind tunnel. The measurement methods and calculations were made as described by Poach et al.[17]

Vegetation sampling

Plant samples were collected during the second experimental period. A 1 m^2 marsh area in each cell was sampled for bulrushes and cattails and a 0.04 m^2 pond area in each section was sampled for microphytes (duckweed). After recording their fresh weight, these samples were oven dried at 65°C for 72 hours and the dry weight was determined. The samples were ground for total N and P analysis. Total-P analysis was done using the perchloric acid digestion method and using a TRAACS 8000 Auto Analyzer. Total N was determined using a C-N-H-S analyzer (Perkin-Elmer model 2400).

N uptake by plant = Plant N concentration X Total dry weight of the plant / m^2.

Statistical analysis

The experimental design followed in the statistical design was a split-plot with the wetland system as the main plot factor and loading rate as the sub-plot factor. Repeated measures (samples) were taken weekly over time during the periods when each level of loading was in place. The analysis of variance was carried out with cells as random effect and an autocorrelation error structure for weeks (sampling time) within cells. Statistical analysis was conducted using SAS Mixed Procedures.[21]

Results

Water component

The Mean pH values in the wetland system ranged from 6.8 to 7.4 with an average water temperature ranging from 15.8 to 21.7°C. The mean oxygen content in the pond area was 3.3 mg L^{-1}, which was adequate to promote nitrification. Dissolved oxygen (DO) was lower in the inflow marsh than in the outflow marsh (Table 1). However, both marshes showed oxidation-reduction (-Eh) conditions at soil surface ranging from 27.6 to 63.0 mV. The total rainfall during the experimental period, June to September, was 208.3 mm and of this 61.6% occurred during June and July.

N-Removal

There was no difference in nitrogen removal between the wetland systems at each N loading (Fig. 3). Each wetland system showed a significant difference ($P < 0.05$) in N removal between the low and high N loading rates. Unlike WS-2 and WS-3, the removal in WS-1 also included loss of ammonia by volatilization. A 40% less reduction of N occurred at 12 kg ha^{-1} day^{-1} loading rate from that of 7 kg ha^{-1} day^{-1} loading rate in uncovered (WS-1) cells (Fig. 3). Aerating the WS-1 and WS-2 systems did not improve N reduction rate.

Total N mass balance in these wetland systems is presented in Table 2. For the uncovered WS-1 ammonia volatilization accounted for 12% of ammonia loss at lower loading rate, which is similar to the results reported by Poach et al.[16] Ammonia volatilization accounted for < 4% of removal in WS-2 and WS-3 cells (Table 2). Ammonia volatilization was four to five times higher in the open pond m-p-m system than the other two wetland systems. Higher NH_3volatilization in the WS-1 could be due to active aeration in the open pond area. Therefore, 52% reduction of N in WS-1 at 12 kg ha^{-1} day^{-1} includes 12% volatilization losses and hence if the ammonia loss was not accounted then all three systems had similar reduction. The total N removal in these systems ranged from 6.2 to 7.5 kg N $ha^{-1}day^{-1}$. Denitrification assumptions were made (total % reduction - % loss by plants and ammonia volatilization). Denitrification contributed to 4.5, 6.6 and 7.0 kg N $ha^{-1}day^{-1}$ of the total N removal in WS-1, WS-3 and WS-2, respectively. Vegetation uptake of N was

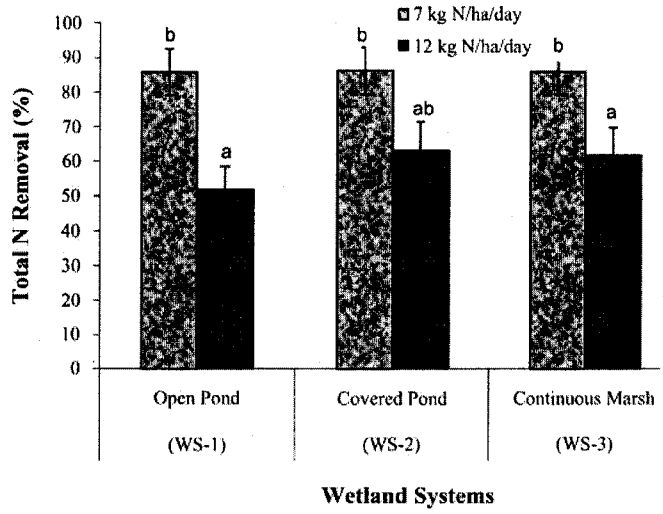

Fig. 3. Percent removal of total nitrogen (Total-N) at N loading rates 7 and 12 kg ha^{-1} d^{-1} during two time periods, June and July, and August and September, respectively. Vertical bars indicate standard error of the mean (n = 2) with the same letters representing no significant differences at $P > 0.05$.

Table 1. Means and standard deviations for pH, oxidation-reduction potential (Eh) and dissolved oxygen (mg L^{-1}) of the swine wastewater in the three wetland systems.

		pH		*Oxidation – reduction Eh(mV)*		*Dissolved oxygen mgL^{-1}*	
Wetland Systems		*Mean*	*SD*	*Mean*	*SD*	*Mean*	*SD*
Open Pond (WS-1)	Marsh Inflow	7.0	0.2	27.6	27.1	0.3	0.2
	Pond					3.2	1.4
	Marsh Outflow	7.1	0.2	66.9	50.0	0.8	0.5
Covered Pond (WS-2)	Marsh Inflow	7.2	0.2	43.5	33.1	0.6	0.4
	Pond					3.3	1.1
	Marsh Outflow	7.1	0.3	74.0	27.3	1.0	1.0
Continuous Marsh (WS-3)	Marsh Inflow	7.2	0.3	52.1	33.9	0.3	0.4
	Marsh Outflow	6.9	2.0	63.0	27.0	0.4	0.4

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Table 2. Nitrogen mass balance (12 kg ha^{-1} day^{-1}) for the three constructed wetland systems.

Wetland Systems	Mass In	Mass Out	Macrophyte uptake** Cattail/Bulrush	Macrophyte uptake** Duckweed	Ammonia volatilization	Denitrification	Total removal	N loss %
				kg ha^{-1}day^{-1}				
Open Pond (WS-1)	12	5.8	0.22	0.05	1.4	4.5	6.2	52
Covered Pond (WS-2)	11.9	4.4	0.18	*	0.3	7	7.5	63
Continuous Marsh (WS-3)	11.3	4.3	0.13	*	0.3	6.6	7	62

*No duckweed in wetland system.
**Macrophytes uptake of N for 120 days (Experimental Period).

in negligible amounts, ranging from 1.2 to 1.8%. Earlier research[16] showed that < 8 Kg N/ha exhibited insignificant ammonia volatilization. There was no difference in N uptake by plants between treatments with N loads of 7 and 12 Kg N ha^{-1}day^{-1}. Hence the total N mass balance has been calculated for the 12 Kg N ha^{-1}day^{-1}loading.

P-removal

There were no differences in total P (TP) removal between the wetland systems at each N loading rate. TP removal was determined on a mass basis (Fig. 4). System versus loading showed no significant difference ($P > 0.05$) in TP reduction. The mean value of TP mass entering the wetlands at low and high N loading rates were 6.3 ± 2.4 and 9.9 ± 3.1 kg ha^{-1} d^{-1}, respectively; whereas, the mean mass discharge of TP at low and high N loading rates were 3.0 ± 1.4 and 8.1 ± 3.0 kg ha^{-1} d^{-1}, respectively. Plant uptake of total P ranged from 0.01 to 0.04 kg ha^{-1}day^{-1} of total P removal. Phosphorus removal was significantly higher ($P < 0.05$) at the low N loading rate than at the high N loading rate in all wetland systems, which is in agreement with Reinhardt et al.[22] However, WS-1 showed a -10% removal at the high N loading rate. These conclusions are solely based on TP mass balance in the system. It appears that reflux of P from soil-sediment into water might have occurred in these wetlands which is in agreement with Kadlec and Knight.[23]

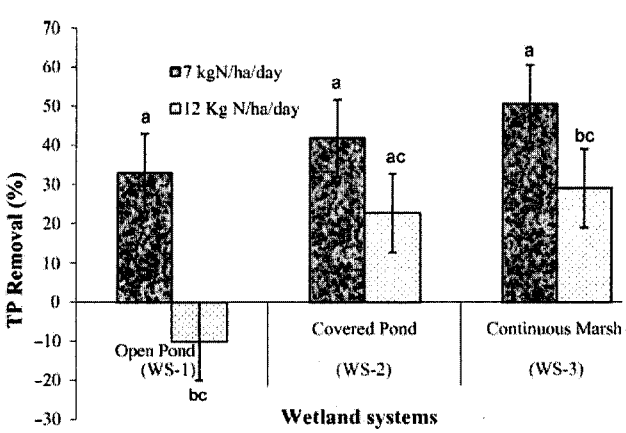

Fig. 4. Percent removal of total phosphorous (TP) at N loading rates 7 and 12 kg ha^{-1} d^{-1} during two time periods, June and July, and August and September, respectively. Vertical bars indicate standard error of the mean (n = 2) with the different letters representing significant differences at $P < 0.05$.

TSS and COD — removal

At the high and low N loading rates WS-1, WS-2 and WS-3 removed wastewater TSS by an average of 57.0 and 79.9, 54.1 and 83.1, and 63.6 and 80.7%, respectively. COD removal from wastewater at the high and low N loading rates for WS-1, WS-2 and WS-3 were 43.7 and 49.6, 62.5 and 61.0, and 50.4 and 51.4%, respectively (Fig. 5). No statistical differences ($P > 0.05$) were observed in TSS and COD reduction among the three wetland systems. Also, no significant difference in TSS and COD reduction was noticed between the low and high N loading. Aeration in covered and uncovered m-p-m wetlands showed no significant difference in reducing TSS and COD.

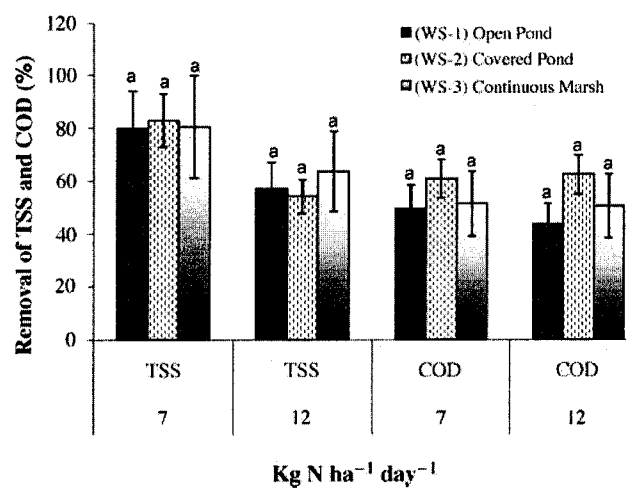

Fig. 5. Percent reduction of total suspended solid (TSS) and chemical oxygen demand (COD) at N loading rates 7 and 12 kg ha^{-1} d^{-1} during two time periods, June and July, and August and September, respectively. Vertical bars indicate standard error of the mean (n = 2) with the same letters representing no significant differences $P > 0.05$.

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Discussion

Nitrogen transformation occurs in wetlands by a number of physical, chemical and biological processes. Nitrogen removal is achieved by three major biological processes: mineralization, nitrification and denitrification. Nitrification-denitrification reactions are the dominant removal mechanisms in constructed wetlands. Denitrification cannot occur if NO_3^--N (nitrate-nitrogen) is not supplied adequately along with organic carbon. Normally, in many wetlands, the nitrification rate is much slower than the denitrification rate, so the first nitrification process controls the overall nitrification-denitrification rate. Nitrification has been found to be a limiting factor in constructed wetlands treated with animal wastewater for succeeding denitrification process.[10,12,13] The rate of nutrient removal by plants varies widely, depending on plant growth rate and concentration of the nutrient in plant tissue. At the lower loading rate with higher detention time treatment efficiency was improved which is in agreement with Reddy et al.[9] Also, these results are in agreement with Hunt et al.'s[10] work in North Carolina, where the wetlands showed a removal of 70 to 95% of the total N loads ranging from 3 to 36 kg N ha^{-1} day^{-1}. The introduction of diffused aeration to the wetland systems showed no improvement in N removal when compared to the results reported by Poach et al.[16] who studied m-p-m systems with intermittent drainage. The amounts of N removal occurring in the two aerated systems (WS-1 and WS-2) were similar to those reported by Poach et al.[16] However, aerating the open-pond (WS-1) system did increase the loss of nitrogen by volatilization. Interpretations of ammonia loss data from these systems suggest that the floating cover in WS-2 and the vegetative cover in WS-3 minimized volatilization (Table 2).

The modifications of the wetland operation made in this study to enhance the oxygen availability in the wetlands did not increase N treatment ability. The mechanical introduction of oxygen to WS-1 and WS-2 cells was designed to promote nitrification and increase nitrogen removal; instead it resulted in increased ammonia volatilization (which is an undesired pathway for nitrogen removal). The diffused membrane aeration of m-p-m was surprisingly not as effective as we hypothesized. These diffusers produced very low oxygen transfer efficiency of less than 3%, which were attributed to its low submergence depth of 0.8 m compared to typical depths of 4.5 m.[24] The pond section appeared not to be deep enough for the air being emitted from the diffusers to have sufficient contact time for oxygen to be transferred into water column instead diffused aeration only increased the ammonia volatilization due to increase in water column turbulence.[18] Not only low oxygen transfer efficiency, but also competition from heterotrophic organisms for oxygen in wetlands resulted in low nitrification and reduced N removal. The marsh areas in all systems functioned similarly in ammonia volatilization (< 4%); such lower ammonia volatilization was due to the vegetative cover in the marshes. Previous researchers investigating constructed wetlands treating swine wastewater found that the N conversion to gaseous forms was the major mechanism for wastewater N removal in pond area of m-p-m system.[10,16]

Phosphorus in swine wastewater is in the form of organic-P and inorganic-P. Reduction or removal of P in wetlands is achieved by four retention and uptake mechanisms: (i) chemical and physical adsorption; (ii) precipitation; (iii) plant and microbial uptake; and (iv) sedimentation.[23]The inorganic-P will be either adsorbed to the substrate matrix or become available to plants and microbes. Inorganic-P precipitates with calcium (Ca) and magnesium (Mg) in the wastewater and forms insoluble precipitates. The systems were not efficient in removing total P. The P removal efficiency in these wetlands might have decreased over the years, being that these systems were constructed in 1995. These wetlands were treated with swine wastewater for the past nine years and therefore the wetland soils may have become saturated resulting in less adsorption sites available on the surface of the substrate (soil). It appears that reflux of P from soil-sediment into water occurred in these wetlands which is in agreement with Kadlec and Knight.[23] Although phosphorus retention is an important attribute of wetlands, sediment attached phosphorus is subject to re-suspension and movement with water when sediments are disturbed. Adsorption, precipitation and plant uptake are generally considered finite, short-term retention mechanisms. Wetland soils can function as a source or sink for P[25] depending on the quality and quantity of native P. Phosphorus removal in constructed wetlands through previous studies have always posed a challenge; therefore, to accomplish more efficient removal in the wetland systems, pre and post-treatment of wastewater should be further investigated.

Treatment efficiencies for TSS and COD are in agreement with the results reported by Reddy et al.[9] in that there was no difference in removal based on N loading rate, which was as a result of the association of the organics with the solids. Poach et al.[11] indicated that alternate fill and drain cycles showed no significant effect on TSS and COD removal rates. Gearheart[26] also observed that an increase or decrease in retention time had no effect on TSS reduction due to the gravity settlement of suspended solids in wetlands.

Conclusion

The N mass reduction at 7 kg N $ha^{-1}day^{-1}$ was approximately 86% in all wetland systems. The systems showed no difference in treatment ability. At 12 kg N $ha^{-1}day^{-1}$ reduction of N was in the range of 51.8% to 63.3%. A significant difference ($P < 0.05$) was observed between the loading rates and treatment systems. At the higher loading rate the wetland systems showed no significant difference in treatment. However, at the lower loading rate, the higher removal was expected due to a higher detention time (28 days) as compared to the higher loading rate with a lower detention time (14 days). Aeration in the pond

section showed no improvement in nitrogen removal efficiency as nitrification was limited. The aerated ponds contributed to higher ammonia volatilization (>12% of total N load). The three WS showed no significant difference ($P > 0.05$) in treatment ability. However, Hunt et al.[12] indicated that a continuous marsh system with lower water depths than the m-p-m system was found to increase denitrification rates. Therefore, an increase in denitrification should result in increased treatment efficiency. These results represent summer months and may differ from early spring and late fall seasons. This is supported by our earlier study[11] that showed a significant difference in N removal between winter and summer months.

The constructed wetlands showed no significant difference in P reduction with an average mass reduction ranging from -10 to 53.1%, for the high and low N loading rates, respectively for all systems studied. COD and TSS reduction was noticeable throughout systems; however, mass reduction is not associated with mass load.

The inclusion of aeration in the pond section of the m-p-m constructed wetlands should be eliminated due to the fact that it increases ammonia volatilization while not effective in oxygen transfer into water column. It only increases the operation cost. However, past studies on m-p-m systems by Poach et al.[27] revealed that partial nitrification of wastewater before being applied to constructed wetlands can improve N removal efficiency. Similar results were also observed by Poach et al.[11] where different drain cycles were implemented. To avoid changing the natural biogeochemical function, it is important that the hydrology of the wetland, the inflow, outflow and residence time of the water, remain relatively undisturbed.

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