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Ecological Engineering 23 (2004) 165–175
ECOLOGICAL ENGINEERING
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Swine wastewater treatment by marsh–pond–marsh constructed wetlands under varying nitrogen loads

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Received 14 May 2004; received in revised form 12 August 2004; accepted 1 September 2004

Abstract

The research objective was to investigate the ability of marsh–pond–marsh (m–p–m) constructed wetlands to treat wastewater from a confined swine operation over varying nitrogen loads. Swine wastewater was applied to six, m–p–m wetlands in Greensboro, NC, USA, during two experimental periods, summer and winter. The efficiency of each system to remove the following wastewater constituents was determined: total suspended solids (TSS); chemical oxygen demand (COD); nitrogen (N); phosphorus (P). During the study, the wetlands removed an average 35–51% of TSS, 30–50% of COD, 37–51% of total N, and 13–26% of total P from swine wastewater. For wastewater COD and N, treatment efficiency was significantly lower during the winter experimental period compared to the summer. Treatment efficiency for all constituents tended to decrease with decreasing air temperatures and increasing rainfall amounts. While these m–p–m wetlands treated more N than an equal area of farm land they were not superior in their N treatment ability compared to previously studied continuous-marsh systems.
Published by Elsevier B.V.

Keywords: Constructed wetlands; Animal wastewater; Nitrogen removal; Marsh–pond–marsh design

1. Introduction

Traditionally, animal manure produced on the farm is applied to cropland and pastureland. Through crop utilization of the nutrients in animal manure, land application provides the dual benefits of independence from commercial fertilizers and management of manure. However, this form of manure management becomes inadequate where the assimilative capacity of the land and cropping system is outpaced by the production of manure nutrients. Under these conditions, excess nutrients may impact surface and groundwater thus excess nutrients must be accommodated in a different manner.

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0925-8574/$ – see front matter. Published by Elsevier B.V.
doi:10.1016/j.ecoleng.2004.09.001

Animal operations with limited land area can avoid the need for offsite treatment if they adopt management options that increase their onsite treatment capacity. On farms that produce liquid manure, one option to reduce manure nutrients prior to land application is to use a constructed wetland. A constructed wetland is an appealing option because it provides an operationally passive and cost effective form of nutrient removal (Hill et al., 1999, Kadlec and Knight, 1996). While constructed wetlands have been used for decades to treat municipal and industrial wastewater, only recently has the ability of this technology to treat liquid animal manure been examined by significant research efforts (Hunt et al., 2002; Hunt and Poach, 2001; Knight et al., 2000). This research has shown that constructed wetlands can effectively treat animal wastewater prior to land application and, thereby, reduce the nutrient load to crop and pastureland (Knight et al., 2000).

While previous research has examined the ability of constructed wetlands to treat animal wastewater, little effort has been made to analyze how wastewater treatment is affected by wetland design. The designs for constructed wetlands generally belong to one of two classes: subsurface flow or surface flow. In the USA, surface-flow systems have been preferred for animal wastewater treatment (Knight et al., 2000); generally, these systems have been configured either as a continuous-marsh or a marsh–pond–marsh (m–p–m).

As the name implies, the marsh–pond–marsh design is a continuous-marsh design bisected by a deeper, open-water or pond section. The pond section was added to the wetland design to promote the input of oxygen to the wastewater (Hammer, 1994; Reaves, 1996). The promotion of oxygen input was expected to improve the ability of constructed wetlands to treat animal wastewater. For example, both Hammer (1994) and Reaves (1996) stipulated that oxygen input facilitated by the pond section would enhance wastewater nitrification. Because nitrification limits the removal of N from animal wastewater, the enhancement of nitrification should in turn increase wastewater N removal.

A comparison of m–p–m and continuous-marsh systems treating dairy wastewater found that the two designs performed similarly (Moore et al., 1995), even though research has shown that the pond section can increase the oxygen concentration of wastewater (Cathcart et al., 1994). The results of Moore et al. (1995) should not be seen as indicative because of the limited and unique scope of the study. Moore et al. (1995) only examined treatment under one wastewater application rate. At their application rate the wetlands received high loads of chemical oxygen demand (550 kg ha^{-1} day^{-1}) and total Kjeldahl nitrogen (42 kg ha^{-1} day^{-1}). To achieve a fuller comprehension of the impact of the pond section, thorough studies need to be conducted on animal wastewater treatment by both systems. While intensive research has been conducted on the treatment of animal wastewater by continuous-marsh systems (Hunt et al., 2002, 2003; Poach et al., 2002, 2003), comparable research has not been conducted on m–p–m systems. The wastewater treatment ability of m–p–m systems needs to be critically examined to determine the optimal wetland design for animal wastewater treatment.

The objective of this research was to investigate the ability of m–p–m constructed wetlands to treat wastewater from a confined swine operation over varying N loads. To meet this objective, the research was conducted using six m–p–m wetland systems, which allowed the unique ability to simultaneously examine wetland treatment at six different N loading rates. Results were compared to literature results to investigate the hypothesis that m–p–m systems remove more N than continuous-marsh systems.

2. Materials and methods

2.1. Site description

The experiment was conducted using six m–p–m wetlands at the swine research facility of the North Carolina A&T State University farm in Greensboro, NC, USA. The wetlands (11 m × 40 m) were constructed in 1995. Each wetland system (WS) consisted of an 11 m × 10 m × 0.15 m marsh at both the influent and effluent ends and an 11 m × 20 m × 0.75 m pond section separating the marshes (Fig. 1). The marsh sections were planted with *Typha latifolia* L. (broadleaf cattail) and *Schoenoplectus americanus* (Pers.) Volkart ex Schinz and R. Keller (American bulrush) in March 1996.

2.2. Experimental design

The study was conducted during two experimental periods: the first or winter period was from November of 2000 to February of 2001; and the second or summer period was from May to August of 2001. During the study, two on-site sources of wastewater were used to provide each WS with a different N load, while targeting all systems to receive similar hydraulic loads. The first source was the primary lagoon (L_1) of a two-stage anaerobic lagoon that received manure flushed from the swine house (Fig. 1). The second source was a storage pond that had received the outflow from the constructed wetlands since their initial operation in 1997 (Reddy et al., 2001). Wastewater from L_1 was transferred by a submersible pump to an 8000 L storage tank and discharged by gravity to all but the last WS (WS-6). Wastewater from the storage pond was transferred by a shallow-well pump to all but the first WS (WS-1). Thus WS-1 received wastewater only from L_1; WS-6 received wastewater only from the storage pond.

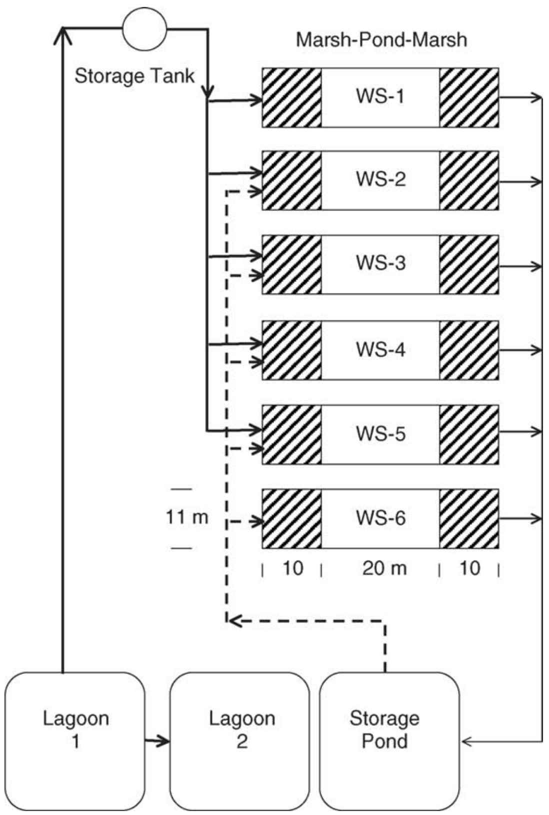

Fig. 1. Schematic of the marsh–pond–marsh constructed wetland design showing the sources and flow paths for swine wastewater.

Wastewater flows to each WS were controlled by ball valves. Effluent for each WS was discharged back to the storage pond. The inflows and outflows of each WS were measured with tipping buckets wired to an electronic totalizer (cycle counter). Periodically, mechanical flow measurements were verified manually. The operating depths of wastewater were maintained at 15 cm in the marsh sections and 75 cm in the pond sections.

Whereas the m–p–m wetlands were used mainly for N removal, wastewater was applied to them based on N. Wastewater from each source was applied at ratios intended to produce six different N loads but maintain similar hydraulic loads (Table 1). The initial N concentrations of the two sources were used to determine the ratios necessary to produce N loads between 5 and 50 kg N ha^{-1} day^{-1}. Over the duration of the study, the N concentration of L_1 ranged from 138 to 248 mg L^{-1} with an average of 175 mg L^{-1} while the N concentration of the storage pond ranged from 17 to 96 mg L^{-1} with an average of 32 mg L^{-1} (Table 2). Because N concentrations of the sources changed throughout the study, influent ratios were adjusted accordingly on a weekly basis to reduce the variability in N load that each WS received. These adjustments caused hydraulic loads to vary over the course of the study from 7.1 to 12.6 m^3 day^{-1}. These loads produced theoretical retention times that ranged from 10 to 18 days.

Discrete wastewater samples were collected from the two inlet sources (L_1 and the storage pond) and from the outlet of each WS using autosamplers (ISCO

Table 1
Percent of total wastewater flow derived from two on-site wastewater sources for each of six marsh–pond–marsh wetlands

Wetland	Wastewater source	
System	Primary waste lagoon (%)	Storage pond (%)
1	100	0
2	70	30
3	92	8
4	28	72
5	41	59
6	0	100

Table 2
Characteristics of wastewater contained in two on-site wastewater sources used to load six marsh–pond–marsh wetlands

Wastewater constituent	Primary waste lagoon ($mg\,L^{-1}$)	Storage pond ($mg\,L^{-1}$)
Total suspended solids	363	160
Chemical oxygen demand	808	313
Total nitrogen	175	32
Total Kjeldahl nitrogen	174	30
Total phosphorus	73	38

3700, Lincoln, NE).[1] For each location, the samplers combined daily samples into weekly composites. Concentrated hydrochloric acid was added to sampling bottles prior to sample collection to lower the pH below 2.5. At the end of each weekly sampling period, samples were transferred to the laboratory for analysis and stored at 4 °C.

2.3. Data analyses

Wastewater samples were analyzed for nitrate/nitrite-N (353.1), total Kjeldahl-N (351.2), and total phosphorus (P, 365.4) using EPA methods (Kopp and McKee, 1983). These analyses were performed with a TrAAcs 800 Auto-Analyzer (Bran + Luebbe, Buffalo Grove, IL). Total N was the sum of total Kjeldahl-N and nitrate/nitrite-N. Total suspended solids (TSS) were determined as follows: a 20 mL sub-sample of wastewater from each sample was filtered through a pre-dried glass fiber filter, which was subsequently dried at 105 °C to constant weight. Chemical oxygen demand (COD) was determined using the closed reflux, colorimetric method (5220; APHA, 1998).

Rates of constituent mass load and discharge were calculated to determine mass removal. For these systems, mass removal is a more informative parameter than concentration reduction because regulations prevent the direct discharge of wastewater from confined animal operations. The rates of constituent mass load and discharge were determined weekly for each WS using the following equation:

$$R_L = \frac{[\{(C_L/10^6) \times Q_L\} + \{(C_S/10^6) \times Q_S\}]}{\mathrm{Aw}} \quad (1)$$

$$R_D = \frac{[(C_0/10^6) \times Q_0]}{\mathrm{Aw}} \quad (2)$$

where $R_{L/D}$ is the rate of constituent mass load/discharge ($kg\,ha^{-1}\,day^{-1}$), C_L the constituent concentration in lagoon wastewater ($mg\,L^{-1}$), C_S the constituent concentration in storage pond wastewater ($mg\,L^{-1}$), C_0 the constituent concentration at WS outlet ($mg\,L^{-1}$), Q_L the average daily wastewater inflow from lagoon source ($L\,day^{-1}$), Q_S the average daily wastewater inflow from storage pond ($L\,day^{-1}$), Q_0 the average daily wastewater outflow ($L\,day^{-1}$), and Aw the wetland area (ha).

Rates of constituent mass load and discharge were averaged across each month of operation. For each wastewater constituent, monthly treatment efficiencies (% mass removal) for each WS were determined using the following equation:

$$\mathrm{Eff} = \left[\frac{R_L - R_D}{R_L}\right] \times 100 \quad (3)$$

where Eff is the treatment efficiency (%), R_L the average rate of constituent mass load ($kg\,ha^{-1}\,day^{-1}$), and R_D the average rate of constituent mass discharge ($kg\,ha^{-1}\,day^{-1}$).

Monthly values for each WS were averaged to determine the system's overall treatment efficiency for each wastewater constituent.

2.4. Statistical analysis

For each wastewater constituent, analysis of covariance was used to test if treatment efficiency was significantly affected by experimental period. Analysis of covariance was used because wastewater constituents were loaded to each WS at different rates. For the analysis, treatment efficiency was the dependent variable; loading rate was the independent variable; and experimental period (winter or summer) was the covariate. Significant covariate effects were only valid if the analysis indicated no significant interaction between loading rate and experimental period. The analysis was conducted using the GLM Procedure of the SAS system (SAS, 1990).

[1] Mention of trade name, proprietary product, or vendor is for information only and does not constitute a guarantee or warranty of the product by U.S. Department of Agriculture and does not imply its approval to the exclusion of other products or vendors that may also be suitable.

Previous research on four of our wetland systems indicated that treatment performance was affected by air temperature (Reddy et al., 2001). Multiple regression analysis was used to investigate the effect on wetland treatment efficiency of monthly average air temperature and monthly rainfall accumulation. The analysis was conducted using the Regression Procedure of the SAS system (SAS, 1990). Air temperature and rainfall data were collected by an on-site weather station.

3. Results

3.1. TSS

When TSS was loaded to the wetlands at rates ranging from 17 to 116 kg ha^{-1} day^{-1}, monthly rates of TSS removal ranged from 0.2 to 55 kg ha^{-1} day^{-1} (Fig. 2). These rates translated to monthly TSS removal efficiencies ranging from 1 to 70%. Wetland systems 1 to 6 (WS-1 to -6) removed wastewater TSS by an average of 36, 35, 39, 49, 51, and 45%, respectively, with an average treatment efficiency of 43% (Table 3). The effect of experimental period on treatment efficiency could not be performed because analysis of covariance indicated a significant interaction ($P = 0.02$) between loading rate and experimental period.

3.2. COD

When COD was loaded to the wetlands at rates ranging from 34 to 291 kg ha^{-1} day^{-1}, monthly COD removal rates ranged from 0.5 to 149 kg ha^{-1} day^{-1} (Fig. 3). These rates translated to monthly COD removal efficiencies ranging from 1 to 61%. WS-1 to -6 removed wastewater COD by an average of 43, 37, 43, 44, 50, and 30%, respectively, with an average treatment efficiency of 41% (Table 3). Average COD removal efficiency was significantly higher ($P = 0.01$) during the summer experimental period (46%) compared to the winter (37%).

3.3. Total N

When total N was loaded to the wetlands at rates ranging from 2 and 51 kg ha^{-1} day^{-1}, monthly total N removal rates ranged from 0.6 to 21 kg ha^{-1} day^{-1} (Fig. 4). These rates translated to monthly total N

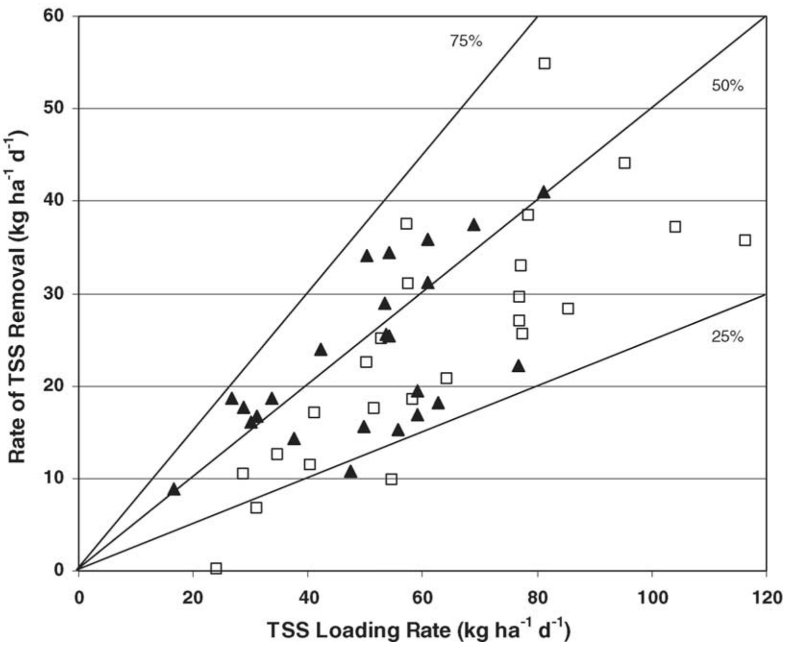

Fig. 2. Monthly rates of total suspended solid (TSS) removal as a function of TSS loading rate for six marsh–pond–marsh wetlands treating swine wastewater during two experimental periods. Open squares, November 2000 to February 2001; closed triangles, May–August 2001.

Table 3
Average rate of constituent mass load into and discharge out of and resulting wetland treatment efficiency (% removal) of six marsh–pond–marsh systems treating swine wastewater

Wetland system	Total suspended solids ($kg\,ha^{-1}\,day^{-1}$)			Chemical oxygen demand ($kg\,ha^{-1}\,day^{-1}$)			Total nitrogen ($kg\,ha^{-1}\,day^{-1}$)			Total phosphorus ($kg\,ha^{-1}\,day^{-1}$)		
	In	Out	Removal (%)	In	Out	Removal (%)	In	Out	Removal (%)	In	Out	Removal (%)
1	82 ± 21	53 ± 15	36 ± 10	190 ± 73	109 ± 44	43 ± 8	40 ± 10	25 ± 11	41 ± 15	16 ± 3	12 ± 3	26 ± 12
2	58 ± 12	38 ± 10	35 ± 13	126 ± 30	80 ± 25	37 ± 13	27 ± 4	17 ± 7	37 ± 17	12 ± 2	10 ± 2	16 ± 12
3	69 ± 9	42 ± 6	39 ± 8	159 ± 44	89 ± 21	43 ± 8	33 ± 4	21 ± 7	39 ± 13	14 ± 2	11 ± 2	20 ± 11
4	44 ± 16	21 ± 5	49 ± 11	93 ± 29	54 ± 22	44 ± 7	15 ± 5	8 ± 5	49 ± 20	10 ± 3	7 ± 3	31 ± 10
5	55 ± 11	26 ± 7	51 ± 12	118 ± 19	60 ± 12	50 ± 6	21 ± 3	11 ± 4	51 ± 13	12 ± 2	9 ± 2	26 ± 8
6	30 ± 12	16 ± 7	45 ± 24	61 ± 16	43 ± 16	30 ± 19	7 ± 4	4 ± 3	44 ± 15	8 ± 3	7 ± 3	13 ± 12
Treatment average			43 ± 7			41 ± 7			44 ± 6			22 ± 7

removal efficiencies ranging from 10 to 75%. The majority of efficiencies below 40% occurred during the winter experimental period, while the majority of efficiencies above 40% occurred during the summer experimental period. WS-1 to -6 removed total wastewater N by an average of 41, 37, 39, 49, 51, and 44%, respectively, with an average treatment efficiency of 44% (Table 3). The average total N removal efficiency was significantly higher ($P < 0.0001$) during the summer experimental period (53%) compared to the winter (34%).

3.4. Total P

When total P was loaded to the wetlands at rates ranging from 3 to 22 $kg\,ha^{-1}\,day^{-1}$, monthly total P removal rates ranged from −0.3 to 7.5 $kg\,ha^{-1}\,day^{-1}$ (Fig. 5). These rates translated to monthly total P removal efficiencies ranging from −9 to 46%. Negative removal rates and efficiencies indicate that wastewater P was increased rather than removed by the respective wetland system. WS-1 to -6 removed total wastewater P by an average of 26, 16, 20, 31, 26, and 13%, respectively, with an average treatment efficiency of 22% (Table 3). The average total P removal efficiency was similar ($P = 0.11$) during both experimental periods.

4. Discussion

Each WS removed by varying degrees wastewater TSS, COD, total N, and total P. Removal of each wastewater constituent was the result of a complex combination of physical and biological processes. Filtration and sedimentation are primarily responsible for TSS removal; biochemical and physical conversions of organic compounds and ammonia are primarily responsible for COD removal (Kadlec and Knight, 1996). While TSS and COD are removed by m–p–m wetlands, they can also be increased in the pond sections through the growth of floating plants and algae (Cathcart et al., 1994). Therefore, for TSS and COD, the difference in production and removal determines the overall system performance.

Wetlands remove wastewater N and P by precipitation, soil adsorption, plant uptake with organic matter accumulation, and microbial immobilization (Kadlec and Knight, 1996). Wastewater N is also removed by

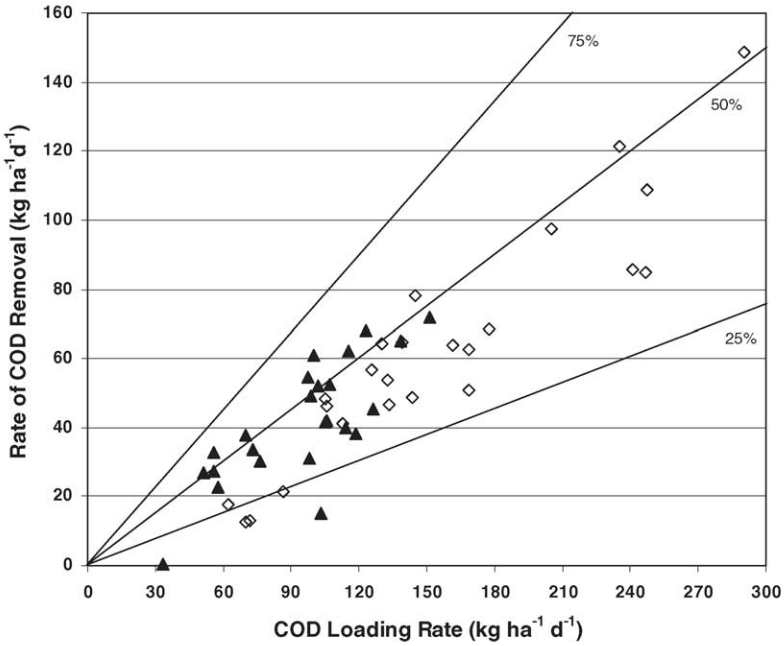

Fig. 3. Monthly rates of chemical oxygen demand (COD) removal as a function of COD loading rate for six marsh–pond–marsh wetlands treating swine wastewater during two experimental periods. Open squares, November 2000 to February 2001; closed triangles, May–August 2001.

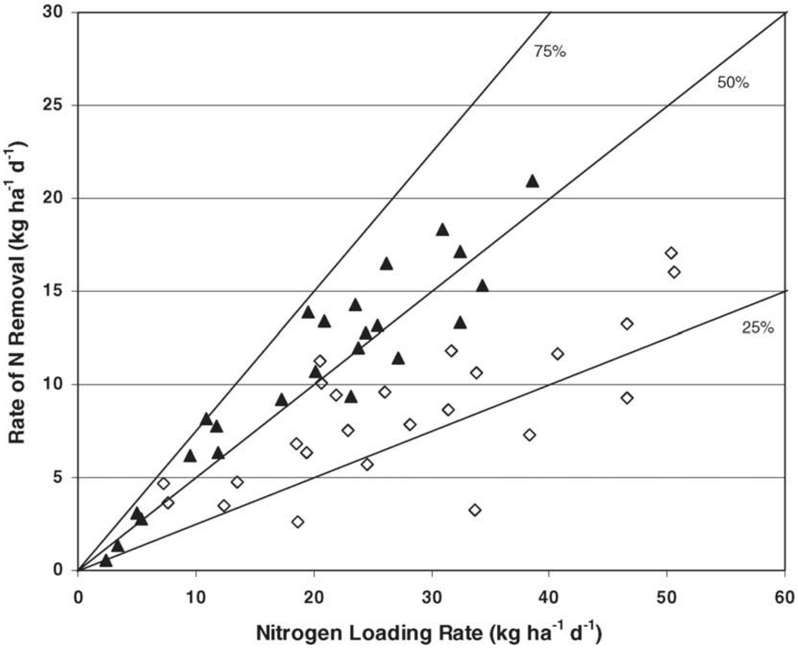

Fig. 4. Monthly rates of nitrogen (N) removal as a function of N loading rate for six marsh–pond–marsh wetlands treating swine wastewater during two experimental periods. Open squares, November 2000 to February 2001; closed triangles, May–August 2001.

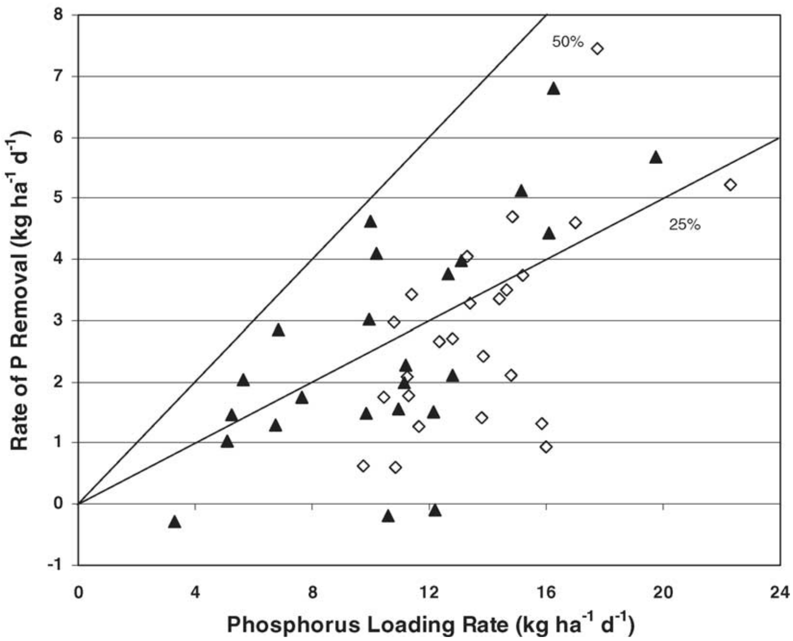

Fig. 5. Monthly rates of phosphorus (P) removal as a function of P loading rate for six marsh–pond–marsh wetlands treating swine wastewater during two experimental periods. Open squares, November 2000 to February 2001; closed triangles, May–August 2001.

the processes of ammonia volatilization and coupled nitrification/denitrification, which convert wastewater N to gaseous forms of N. Conversion of wastewater N to gaseous N is the major mechanism for N removal by constructed wetlands treating swine wastewater because the removal of N by plant uptake and soil accumulation accounted for less than 10% of the N load (Hunt et al., 2002; Poach et al., 2004; Reddy et al., 2001). Both nitrification/denitrification and ammonia volatilization were important for N removal by m–p–m systems. For m–p–m systems that received N loads greater than 15 kg ha^{-1} day^{-1} during July and August, greater than 50% of the wastewater N removal resulted from ammonia volatilization (Poach et al., 2004). The dominance of ammonia volatilization was surprising because for continuous-marsh systems treating swine wastewater it accounted for less than 16% of the N load (Poach et al., 2002).

Compared to the summer experimental period, the significantly lower average COD and total N treatment efficiencies exhibited by the wetlands during the winter experimental period likely resulted from lower air temperatures. For example, total N treatment efficiencies lower than 40% tended to occur in months that experienced air temperatures below 0 °C. The effect of air temperature on total N treatment was also reported by Reddy et al. (2001) for four of the m–p–m systems. For the present study, monthly average air temperature and monthly rainfall accumulation accounted for 32% of the variation in the square of COD treatment efficiency and accounted for 58% of the variation in the total N treatment efficiency (Table 4). Air temperature and rainfall accumulation also had a significant effect on TSS ($R^2 = 0.36$) and total P ($R^2 = 0.37$) treatment efficiencies.

Treatment efficiencies for all constituents decreased with decreasing air temperatures and with increasing rainfall amount (Table 4). Cold temperature likely reduced treatment efficiencies by lowering the plant and microbial activity that effect constituent removal. High rainfall likely reduced treatment efficiencies by shortening the hydraulic residence time of the systems.

Average treatment efficiencies for each WS tended to be lower than those reported for other m–p–m wetlands used to treat swine wastewater. With areas of 396 m^2 and an average wastewater retention time of 12 days, two m–p–m wetlands in Mississippi had average treatment efficiencies of 69, 54, 71, and 44%

Table 4
Relationship between treatment efficiency and monthly average air temperature and monthly rainfall accumulation for each wastewater constituent as determine by multiple regression analysis

Wastewater constituent	Regression equation[a]	R^2
Total suspended solids	Eff = 1.70 temp. − 0.35 rain + 44.69	0.36
Chemical oxygen demand	Eff^2 = 96.39 temp. − 22.04 rain + 2127[b]	0.32
Total nitrogen	Eff = 2.06 temp. − 0.28 rain + 39.54	0.58
Total phosphorus	Eff = 1.32 temp. − 0.36 rain + 29.83	0.37

[a] Eff = monthly treatment efficiency, temp = monthly average air temperature in °C, and rain = monthly rainfall accumulation in mm.
[b] Treatment efficiency for COD was squared before analysis to meet the assumption of equal variances.

for TSS, COD, ammonia-N, and total P, respectively (Cathcart et al., 1994). Cathcart et al. (1994) indicated that their efficiencies should be considered upper limits because their sampling procedure may have underestimated effluent flows. While tending higher than the average treatment efficiencies presented here, efficiencies reported by Cathcart et al. (1994) are still in the range of treatment efficiencies exhibited by the systems in the present study. Because of the intensive nature of the sampling protocol, the efficiencies recorded by the present study may give a more realistic representation of m–p–m wetland treatment efficiency.

Results indicate that m–p–m wetlands can be used to reduce the land area required for wastewater application when application rates are based on N. One hectare of m–p–m wetland with an N removal efficiency of 53% and a P removal efficiency of 22% will remove ~5.3 Mg of wastewater N and ~0.9 Mg of wastewater P when operated for 250 days at an N loading rate of 40 kg ha^{-1} day^{-1} and a P loading rate of 16 kg ha^{-1} day^{-1}. When wastewater application rates are based on N, each hectare of m–p–m wetland will replace either ~13 ha of forage that removes 400 kg N ha^{-1} $year^{-1}$ or ~35 ha of a row crop that removes 150 kg N ha^{-1} $year^{-1}$ (yearly N removal based on data from Johnston and Usherwood, 2002). Based on estimates of P removal, one hectare of m–p–m wetland could remove more P than a similar area of forage or row crop. However, our m–p–m wetlands are expected to eventually lose their ability to remove wastewater P and may actually begin to export P. Research has shown that over time wetland P removal sites become saturated (Kadlec and Knight, 1996). Therefore, the conclusion that a m–p–m system can remove more P than a similar area of forage or row crop should be viewed with caution.

While the m–p–m systems of the present study treated more N than an equal area of land used for wastewater application, they did not appear superior in their ability to treat swine wastewater when compared with previously studied continuous-marsh systems. The m–p–m systems tended to be less efficient at N removal than a continuous-marsh system in Alabama. With a theoretical retention time of 6 days and an N load of 25 kg ha^{-1} day^{-1}, the first 400 m^2 of their continuous-marsh system removed 70% of swine wastewater N during the summer (McCaskey et al., 1994). While this efficiency is within the range of efficiencies exhibited by the m–p–m systems, the average summer efficiency exhibited by the m–p–m systems was less than 70%.

The m–p–m systems also exhibited lower treatment efficiencies than those recorded by a long-term, intensive study of swine wastewater treatment by continuous-marsh systems. Over a 4-year period and at N loads between 3 and 40 kg ha^{-1} day^{-1}, two continuous-marsh systems in North Carolina with theoretical retention times of ~13 days removed 50–90% of wastewater N (Hunt et al., 2002). For these systems, the majority of monthly treatment efficiencies were greater than 75% while none of the m–p–m systems produced monthly efficiencies greater than 75%. The continuous-marsh systems had shallower wastewater depths than the m–p–m systems, and decreasing wastewater depth was found to increase denitrification rates in the continuous-marsh systems (Hunt et al., 2003). However, the effect of wastewater depth on denitrification is not sufficient to explain why the m–p–m systems were not as efficient as the continuous-marsh systems at removing N. It is not sufficient for the following two reasons: (1) in a few m–p–m systems, lower denitrification rates would likely have been counterbalanced by higher rates of ammonia volatilization (Poach et al., 2004) and (2) the m–p–m systems were not as efficient as the continuous-marsh systems of McCaskey et al. (1994), even though the average wastewater depths in the marshes of both systems were similar.

Removal of N by animal waste treatment wetlands is nitrate limited (Hammer, 1994). The better treatment performance of the continuous-marsh systems suggests that they were more effective at stimulating nitrification than the m–p–m systems of the present study. This inference is surprising because the pond sections of other m–p–m systems were shown to increase wastewater oxygen concentrations (Cathcart et al., 1994). In fact, the pond sections of four m–p–m systems used in the present study were shown by previous research to be more oxidized than adjacent marsh sections (Reddy et al., 2001). Reduced rates of nitrification in spite of the potential for greater oxidized conditions might have resulted from the oxygen demand of the wastewater.

Because heterotrophic organisms can outcompete nitrifying bacteria for oxygen (Hanaki et al., 1990), the biochemical oxygen demand (BOD) of wastewater can inhibit nitrification. To insure that BOD is reduced to levels that would not inhibit nitrification in the pond section, Hammer (1992) recommended that the first marsh of the m–p–m system receive a mass load of 5-day BOD (BOD_5) less than 100 kg ha^{-1} day^{-1}. In the present study, this recommended mass load of BOD_5 was likely exceeded for the first marsh section of each m–p–m system. The first marshes received mass COD loads between 136 and 1164 kg ha^{-1} day^{-1}. These values translate to mass BOD_5 loads of 54–466 kg ha^{-1} day^{-1} at a BOD_5:COD conversion of 0.4:1 (Hunt et al., 2002). Therefore, it is possible that nitrification was inhibited in the pond sections of the m–p–m systems because the first marsh did not sufficiently reduce wastewater BOD_5.

While nitrification potential of the pond section may be realized by increasing the length of the first marsh section, such a modification may not mitigate the other drawbacks of using a pond section for animal wastewater treatment such as ammonia volatilization and TSS and COD production. Poach et al. (2004) reported that ammonia volatilization was exacerbated by the presence of a pond section when m–p–m wetlands were loaded with swine wastewater at rates >15 kg N ha^{-1} day^{-1}. They concluded that the excess ammonia volatilization was the result of the wastewater having greater exposure to the wind in the pond section and having a higher pH in the pond section as a result of algal photosynthesis. In two m–p–m systems in Mississippi, the production of algae in the pond section was implicated for the increased wastewater oxygen demand (Cathcart et al., 1994). The production of floating plants and algae by the pond section also increases wastewater TSS. The wastewater treatment ability of m–p–m systems needs further critical examination to determine the optimal wetland design for animal wastewater treatment.

5. Conclusions

Each WS removed by varying degrees TSS, COD, total N, and total P from swine wastewater. The m–p–m wetlands removed an average 35–51% of wastewater TSS, 30–50% of wastewater COD, 37–51% of total wastewater N, and 13–26% of total wastewater P. Overall COD and total N treatment efficiencies were significantly lower during the winter experimental period compared to the summer. The significant differences likely resulted from the winter period experiencing lower air temperatures. Treatment efficiencies for all constituents decreased with decreasing air temperatures and with increasing rainfall amount.

Total N treatment efficiencies indicated that m–p–m wetlands could be used to reduce the land area required for wastewater application when application rates are based on N. However, total N treatment efficiencies exhibited by our m–p–m wetlands tended to be lower than those reported for continuous-marsh systems. The comparison suggests that the addition of a pond section to the design of treatment wetlands does not automatically stimulate increased nitrification of swine wastewater. For the present wetland design, the mass BOD_5 load may have suppressed wastewater nitrification in the pond section. Therefore, these systems should be modified for heavy swine wastewater loads. To determine the optimal wetland design for animal wastewater treatment, the wastewater treatment ability of m–p–m systems needs further critical examination.

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