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Ammonia Volatilization from Marsh–Pond–Marsh Constructed Wetlands Treating Swine Wastewater

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ABSTRACT

Ammonia (NH_3) volatilization is an undesirable mechanism for the removal of nitrogen (N) from wastewater treatment wetlands. To minimize the potential for NH_3 volatilization, it is important to determine how wetland design affects NH_3 volatilization. The objective of this research was to determine how the presence of a pond section affects NH_3 volatilization from constructed wetlands treating wastewater from a confined swine operation. Wastewater was added at different N loads to six constructed wetlands of the marsh–pond–marsh design that were located in Greensboro, North Carolina, USA. A large enclosure was used to measure NH_3 volatilization from the marsh and pond sections of each wetland in July and August of 2001. Ammonia volatilized from marsh and pond sections at rates ranging from 5 to 102 mg NH_3–N m^{-2} h^{-1}. Pond sections exhibited a significantly greater increase in the rate of NH_3 volatilization ($p < 0.0001$) than did either marsh section as N load increased. At N loads greater than 15 kg ha^{-1} d^{-1}, NH_3 volatilization accounted for 23 to 36% of the N load. Furthermore, NH_3 volatilization was the dominant (54–79%) N removal mechanism at N loads greater than 15 kg ha^{-1} d^{-1}. Without the pond sections, NH_3 volatilization would have been a minor contributor (less than 12%) to the N balance of these wetlands. To minimize NH_3 volatilization, continuous marsh systems should be preferred over marsh–pond–marsh systems for the treatment of wastewater from confined animal operations.

Constructed wetlands remove N from wastewater by sedimentation, adsorption, organic matter accumulation, nitrification–denitrification, microbial assimilation, and NH_3 volatilization (Brix, 1993; Johnston, 1991). Of these mechanisms, NH_3 volatilization is the least desirable because NH_3 gas is an atmospheric pollutant that can adversely affect terrestrial and aquatic environments through dry and wet deposition (Asman, 1994). This pollution potential has generated concerns that NH_3 volatilization may govern nitrogen loss from wetlands treating wastewater from confined animal operations because the wastewater ammoniacal N concentration is greater than 20 mg L^{-1} (Payne and Knight, 1997). To be an effective waste management tool, constructed wetland systems should be designed to minimize NH_3 volatilization.

Two wetland designs used in the treatment of animal wastewater are continuous marsh and marsh–pond–marsh. Research on continuous marsh systems verified that NH_3 volatilization did occur when they received swine wastewater, but the volatilization was a minor contributor to the N budget of the wetlands (Poach et al., 2002, 2003). Ammonia volatilization generally accounted for less than 20% of the N removed by these wetlands even though they received wastewater with N concentrations as high as 300 mg L^{-1}.

Because a marsh–pond–marsh system is a continuous marsh system bisected by a pond section, the marsh sections should exhibit rates of NH_3 volatilization similar to a continuous marsh. Therefore, based on results from the continuous marsh, NH_3 volatilization from the marsh sections is expected to be a minor component of the N budget of marsh–pond–marsh systems. However, the presence of the pond section prevents conclusions about the magnitude of NH_3 volatilization for the complete system.

The pond section was added to the design of treatment wetlands with the intent of enhancing nitrification (Hammer, 1994; Reaves, 1996). Research on continuous marsh wetlands treating swine wastewater indicated that NH_3 volatilization was reduced when the wastewater was nitrified before wetland application (Poach et al., 2003). If the pond section enhances nitrification then it may also reduce NH_3 volatilization, but research on marsh–pond–marsh systems receiving swine wastewater do not support the contention that the pond section enhances nitrification of the wastewater. Marsh–pond–marsh systems did not improve N removal compared with continuous systems as would be expected if the

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pond section enhanced nitrification of the animal wastewater (Moore and Niswander, 1997). Therefore, pond sections may not reduce NH_3 volatilization.

Research on anaerobic lagoons containing swine wastewater have shown that NH_3 volatilization is affected by wind blowing across the lagoon surface (Harper et al., 2000). The pond section is similar to a waste lagoon and, compared with the marsh it replaces, has a greater surface area exposed to the wind. Therefore, the pond section could enhance NH_3 volatilization from marsh–pond–marsh wetlands compared with wetlands without a pond section.

This research was part of a larger project investigating the ability of marsh–pond–marsh constructed wetlands to treat wastewater from a confined swine operation. The objective of this research was to use a steady-state enclosure to quantify NH_3 volatilization from these marsh–pond–marsh wetlands. Specific objectives were to determine (i) the contribution of NH_3 volatilization from the marsh sections to the overall N removal of marsh–pond–marsh systems and (ii) the effect of the pond section on the NH_3 volatilization potential of constructed wetlands.

MATERIALS AND METHODS

Site Description

The experiment was conducted using six marsh–pond–marsh wetlands at the swine facility (130–250 sows) of the North Carolina A&T State University farm in Greensboro, NC. The wetland cells (11 × 40 m) were constructed in 1995. Each cell consisted of an 11- × 10-m marsh at both the influent and effluent ends and a 11- × 20-m pond section separating the marshes (Fig. 1). 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.

Experimental Design

Two on-site sources of wastewater were used to provide each wetland cell with a different N load while ensuring each cell received the same hydraulic load. The first source was the primary lagoon 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 been receiving the outflow from the constructed wetlands since their initial operation in 1997 (Reddy et al., 2001). Wastewater from the primary lagoon was transferred by a submersible pump to an 8000-L storage tank and discharged into the wetland cells by gravity. A shallow-well pump was used to transfer wastewater from the storage pond to the wetland cells. Wastewater flows to each wetland cell were controlled by ball valves. Effluent from each wetland cell was discharged back to the storage pond. Flows to and from each wetland cell were measured with tipping buckets wired to an electronic cycle counter.

From September 2000 to September 2001, wastewater from each source was applied at different ratios to each wetland cell to produce six different N loads. The initial N concentrations of the two sources were used to determine the ratios necessary to target N loads between 5 and 50 kg N ha^{-1} d^{-1}. Because N concentrations of the sources changed throughout the study period, influent ratios were adjusted accordingly on a weekly basis to reduce the variability in N load that each wetland received. All cells received the same hydraulic loading rate, but the daily hydraulic load varied from 7.1 to 12.6 m^3 d^{-1} throughout the study period because of variations in the nutrient concentration of the primary lagoon. The operating depths of the marsh and pond sections were 15 and 75 cm, respectively.

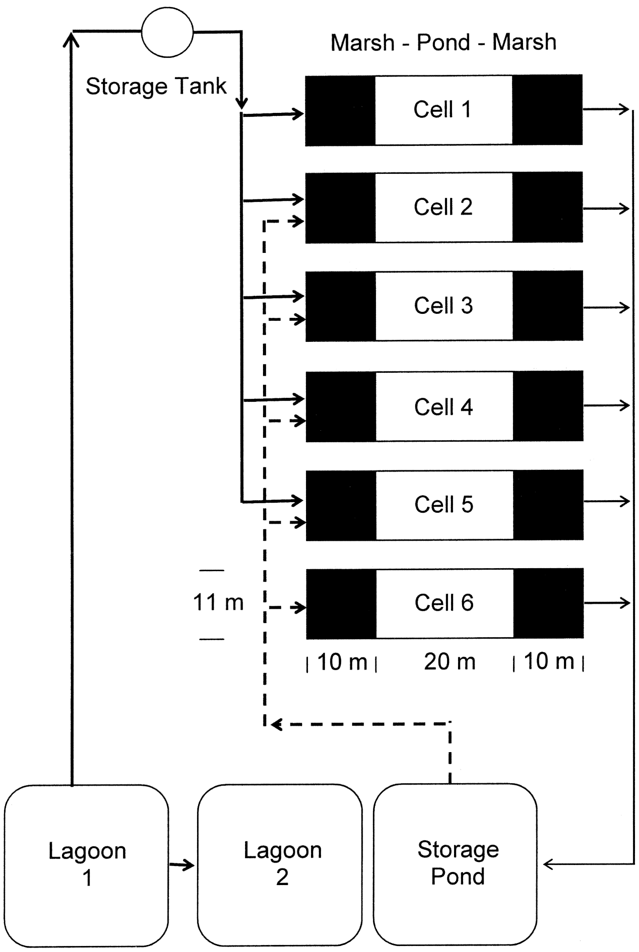

Fig. 1. Schematic of the marsh–pond–marsh constructed wetland design showing the sources and flow paths for swine wastewater.

Wastewater samples were collected from the two inlet sources (primary lagoon and the storage pond) and from all six of the wetland cell outlets using autosamplers (Model 3700; Isco, Lincoln, NE). The samplers combined daily samples into weekly composites. Concentrated hydrochloric acid was added to each sampling bottle to lower the pH below 2.5. At the end of a weekly sampling period, samples were transferred to the laboratory for analysis and stored at 4°C.

During a field campaign in July and one in August 2001, a special open-ended enclosure was used to measure NH_3 volatilization from each section of a wetland cell at a plot located near the middle of the section. This constituted 18 tests during each field campaign. The enclosure method was used because it was the best method for such experimental areas. The enclosure was similar to that described by Poach et al. (2002) except an extension was attached to the inflow end of the enclosure to allow it to span the width of the wetland cells (Fig. 2). At the beginning of a test, the enclosure was set over a plot with the sides rolled up. The enclosure was set so that the bottom was just below the water surface in pond sections and just below the sediment surface in marsh sections. Two gas-washing bottles were mounted at the inlet

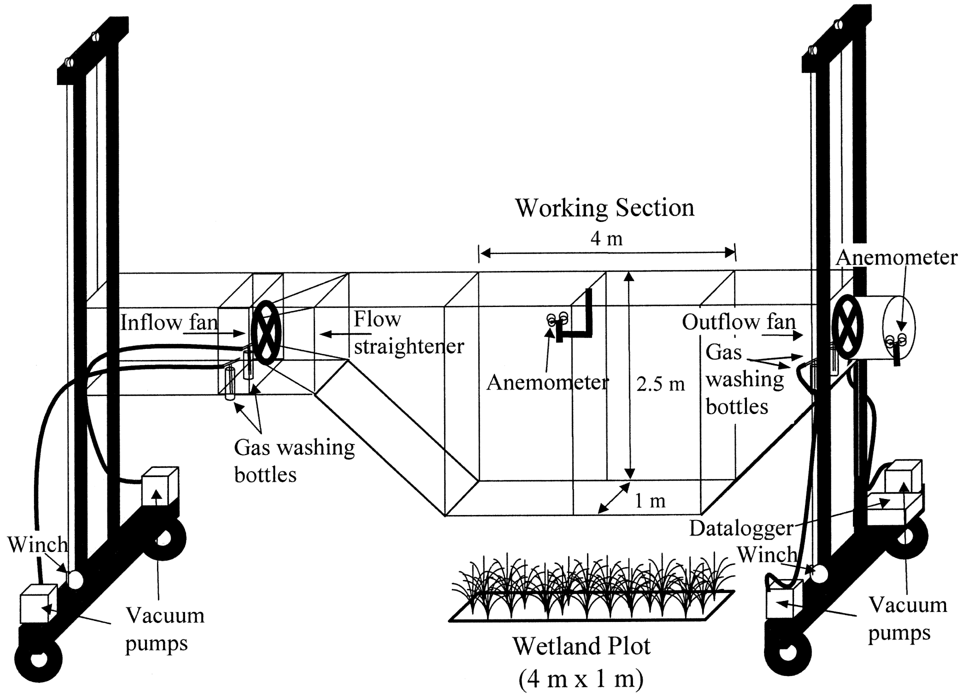

Fig. 2. Diagram of enclosure used to measure NH_3 volatilization showing dimensions and component placement.

and two at the outlet of the enclosure, and they were attached to vacuum pumps. The plastic sides were then lowered to the bottom of the enclosure and locked into place. Two variable-speed fans mounted at each end of the enclosure were turned on and their speeds were adjusted to equilibrate pressure inside the enclosure as indicated by the plastic sides remaining slack. Vacuum pumps were then turned on to begin NH_3 sampling through the gas-washing bottles. The gas-washing bottles contained an 80-mL solution of acid (0.2 M H_2SO_4) to extract NH_3 from the sampled air. The duration of each test was two hours.

During each test, environmental conditions were measured and recorded (Table 1). The air speed generated by the fans was measured with two anemometers, one located at a 2-m height at the center of the enclosure and one located after the outlet fan (Fig. 2). The data from the outflow anemometer were used to determine airflow during field tests as described by Poach et al. (2002). Wastewater temperatures were measured using a thermocouple attached to the enclosure. Due to improper placement of the thermocouple, wastewater temperature was not measured during a few of the tests. Wind speed and temperature were recorded continually with a data-

Table 1. Wastewater parameters and plot air speed for NH_3 volatilization tests conducted in July and August 2001 on six marsh–pond–marsh constructed wetland systems in Greensboro, NC that received swine wastewater.

		July 2001				August 2001			
			Plot wastewater				Plot wastewater		
Wetland cell	Wetland section	Airspeed†	Temperature	pH	$NH_{3,4}$–N‡	Airspeed†	Temperature	pH	$NH_{3,4}$–N‡
		m s^{-1}	°C		mg L^{-1}	m s^{-1}	°C		mg L^{-1}
1	Marsh 1	1.1	ND§	7.7	162	0.6	24.5	7.0	131
	Pond	0.9	ND	7.9	59	1.0	26.9	7.4	62
	Marsh 2	1.3	24.7	7.4	60	0.9	20.2	7.1	50
2	Marsh 1	1.0	24.7	7.5	122	0.9	22.3	7.0	93
	Pond	1.1	ND	7.2	33	0.7	ND	8.0	56
	Marsh 2	0.6	25.2	7.5	47	0.3	20.6	7.0	45
3	Marsh 1	0.2	ND	7.4	153	0.8	22.8	6.7	88
	Pond	1.1	ND	7.5	57	1.4	27.6	7.9	74
	Marsh 2	1.1	21.2	7.3	53	1.0	22.7	6.6	71
4	Marsh 1	1.2	23.3	6.6	42	1.3	24.7	7.1	58
	Pond	1.1	ND	7.0	23	1.5	24.2	7.1	21
	Marsh 2	0.2	21.1	6.2	21	0.5	23.5	7.3	21
5	Marsh 1	1.0	22.5	7.7	46	0.6	23.4	6.5	97
	Pond	1.4	24.0	7.7	29	1.1	26.3	7.1	36
	Marsh 2	0.8	23.4	6.9	43	0.5	23.4	6.8	38
6	Marsh 1	0.7	23.5	7.2	4	0.8	26.5	6.6	10
	Pond	1.3	22.5	6.9	5	1.3	26.4	6.9	4
	Marsh 2	1.5	22.0	6.6	6	0.8	23.5	6.6	4

† Airspeed measured by an anemometer located at the center of the enclosure, 2 m above the plot surface.
‡ Ammoniacal nitrogen (NH_3–N + NH_4–N) concentration.
§ Not determined.

logger (Model CR23; Campbell Scientific, Logan, UT). Wastewater samples and pH readings were collected from an area contiguous to the study location during each test (Table 1).

Data Analyses

Gas-wash-bottle samples were treated as if they were digested samples and were analyzed for ammoniacal N using USEPA Method 351.2 (Kopp and McKee, 1983). Also using USEPA methods, wastewater samples were analyzed for ammoniacal N (351.2), nitrate and nitrite N (353.1), and total Kjeldahl N (351.2). Samples were analyzed with a TrAAcs 800 Auto-Analyzer (Bran + Luebbe, Buffalo Grove, IL). Total N was the sum of total Kjeldahl N and nitrate and nitrite N.

Hourly rates of NH_3 volatilization in mg NH_3–N m^{-2} h^{-1} were determined from the difference in NH_3–N collected by the inlet and outlet gas-washing bottles over a 2-h period after adjusting for the air sampling ratio (Eq. [2] in Poach et al., 2002). The contribution of NH_3 volatilization to the N budget of each wetland cell was estimated by averaging NH_3 volatilization across each cell, extrapolating these averages to daily rates, and comparing the result with the nitrogen loading and removal rates for that cell. Total N removal was determined by the difference in the monthly average mass N load between the inlet and outlet of each wetland cell. The extrapolation of daytime hourly rates to daily rates may have overestimated NH_3 volatilization because volatilization tends to exhibit a diurnal pattern where NH_3 volatilization is lower during the night (Bussink et al., 1996).

Statistical Analysis

For each NH_3 volatilization test, significant difference between mean NH_3–N captured by inlet and outlet bottles was determined using a Student's *t* test. Individual *t* tests were made more powerful by pooling standard deviations for all tests within a section (marsh vs. pond) to estimate the sampling variance. A difference that was not significant indicated that NH_3 volatilization was below the detection limit of the enclosure.

The influence of environmental factors and wastewater characteristics on NH_3 volatilization was investigated with the regression procedure of the SAS system (SAS Institute, 1990). To determine if wetland section affected NH_3 volatilization, NH_3 volatilization was plotted versus N load for each wetland section (marsh or pond) and the slopes of the resulting regression lines were compared with the GLM procedure of the SAS system (SAS Institute, 1990). This analysis was repeated for the regressions of NH_3 volatilization versus the ammoniacal N of the plot wastewater.

RESULTS AND DISCUSSION

Ammonia volatilized from marsh and pond sections of the wetlands during July and August as indicated by significant differences in NH_3–N collected at the enclosure outlet and inlet (Tables 2 and 3). Ten tests in July and seven tests in August had differences that were significant at a 90% confidence level. Differences ranged from −16 to 163 μg NH_3–N in July and from −2 to 132 μg NH_3–N in August. Positive differences indicate NH_3 volatilization while negative differences indicate NH_3 deposition. Ammonia deposition probably occurred because the NH_3 in the air entering the enclosure was higher than the NH_3 compensation point of the plot (Farquhar et al., 1980).

Rates of NH_3 volatilization associated with the differences that were statistically significant ranged from 5 to 102 mg NH_3–N m^{-2} h^{-1} (Tables 2 and 3). Only one test had a significant negative value, −14 mg NH_3–N m^{-2} h^{-1}. During this test, the inlet of the enclosure was drawing air from an area close to the pond section of the wetland system exhibiting the highest NH_3 volatiliza-

Table 2. Ammonia volatilization from six marsh–pond–marsh constructed wetland systems in Greensboro, NC that received swine wastewater at six different N loads during July 2001 as determined by the difference in NH_3–N captured at the inlet and outlet of a steady-state enclosure.

Wetland cell	Wetland section	N load	Air flow†	NH_3–N In	NH_3–N Out	NH_3–N Out – in	NH_3 volatilization‡ Per cell	NH_3 volatilization‡ Cell mean§
		kg ha^{-1} d^{-1}	L min^{-1}	μg	μg	μg	mg NH_3–N m^{-2} h^{-1}	mg NH_3–N m^{-2} h^{-1}
1	Marsh 1	30.4	51 327	31	45	14¶	15	
	Pond		22 242	43	206	163#	76	
	Marsh 2		40 265	22	34	12¶	10	44
2	Marsh 1	27.9	27 316	9	23	14¶	8	
	Pond		25 005	71	220	149#	78	
	Marsh 2		42 714	46	30	−16¶	−14††	41
3	Marsh 1	28.3	32 371	21	16	−5	−3	
	Pond		31 835	51	185	135#	89	
	Marsh 2		48 496	3	7	5	5	46
4	Marsh 1	12.3	50 019	6	19	13¶	13	
	Pond		31 456	13	18	6	4	
	Marsh 2		40 611	9	8	−2	−1	5
5	Marsh 1	15.5	33 225	11	25	14¶	10	
	Pond		38 848	3	49	46#	37	
	Marsh 2		47 290	5	12	7	7	23
6	Marsh 1	2.5	31 894	7	10	3	2	
	Pond		29 429	7	16	9	5	
	Marsh 2		38 655	7	11	5	4	4

† Determined from airspeed measured at outflow of enclosure.

‡ Volatilization = [(NH_3–N out − NH_3–N in) × (enclosure airflow/6 L min^{-1})/4 m^2/2 h] × (1 mg/1000 μg).

§ Negative values below detection limit of enclosure were considered to be zero in the determination of cell mean.

¶ Statistically different from zero ($LSD_{0.1}$ = 7 μg) indicating that volatilization was above detection limit.

Statistically different from zero ($LSD_{0.1}$ = 21 μg) indicating that volatilization was above detection limit.

†† Not included in cell mean.

Table 3. Ammonia volatilization from six marsh–pond–marsh constructed wetland systems in Greensboro, NC that received swine wastewater at six different nitrogen loads during August 2001 as determined by the difference in ammonia nitrogen captured at the inlet and outlet of a steady-state enclosure.

				NH_3–N			NH_3 volatilization‡	
Wetland cell	Wetland section	N load	Air flow†	In	Out	Out − in	Per cell	Cell mean§
		kg ha^{-1} d^{-1}	L min^{-1}	μg			mg NH_3–N m^{-2} h^{-1}	
1	Marsh 1	27.4	31 652	12	23	11¶	7	
	Pond		22 934	40	158	118#	56	
	Marsh 2		36 321	9	19	10¶	8	32
2	Marsh 1	26.7	35 403	7	10	3	3	
	Pond		21 001	70	202	132#	58	
	Marsh 2		31 778	19	18	−1	−1	29
3	Marsh 1	37.1	39 189	7	10	3	2	
	Pond		41 779	10	127	117#	102	
	Marsh 2		50 299	28	31	3	3	52
4	Marsh 1	12.7	29 985	5	13	8¶	5	
	Pond		31 518	19	18	−2	−1	
	Marsh 2		38 682	8	10	2	2	2
5	Marsh 1	14.7	30 625	5	22	16¶	11	
	Pond		29 432	6	15	9	6	
	Marsh 2		29 889	12	14	2	1	6
6	Marsh 1	4.1	30 208	5	7	2	1	
	Pond		33 391	7	5	−2	−2	
	Marsh 2		37 645	8	9	1	1	1

† Determined from airspeed measured at outflow of enclosure.

‡ Volatilization = [(NH_3–N out − NH_3–N in) × (enclosure airflow/6 L min^{-1})/4 m^2/2 h] × (1 mg/1000 μg).

§ Negative values below detection limit of enclosure were considered to be zero in the determination of cell mean.

¶ Statistically different from zero ($LSD_{0.1}$ = 7 μg) indicating that volatilization was above detection limit.

Statistically different from zero ($LSD_{0.1}$ = 21 μg) indicating that volatilization was above detection limit.

tion. As a result, this test had the highest background concentration of NH_3–N for tests conducted on marsh sections. Because the measurement procedure may have imposed an unrealistic background NH_3 concentration, this value was not used in subsequent analyses.

Results supported the hypothesis that the pond section could produce rates of NH_3 volatilization greater than marsh sections. As N load increased, pond sections exhibited a significantly greater increase in the rate of NH_3 volatilization ($p < 0.001$) than did the marsh sections (Fig. 3). Pond sections that received N loads greater than 15 kg ha^{-1} d^{-1} produced rates of NH_3 volatilization greater than 36 mg NH_3–N m^{-2} h^{-1}, while all marsh sections produced rates less than 16 mg NH_3–N m^{-2} h^{-1} (Tables 2 and 3). Different trends were also displayed by regressions of NH_3 volatilization versus the ammoniacal N concentration of plot wastewater. As the ammoniacal N concentration increased, pond sections exhibited a significantly greater increase in the rate of NH_3 volatilization ($p < 0.0001$) than did the marsh sections (Fig. 4).

When data within a section (marsh or pond) were

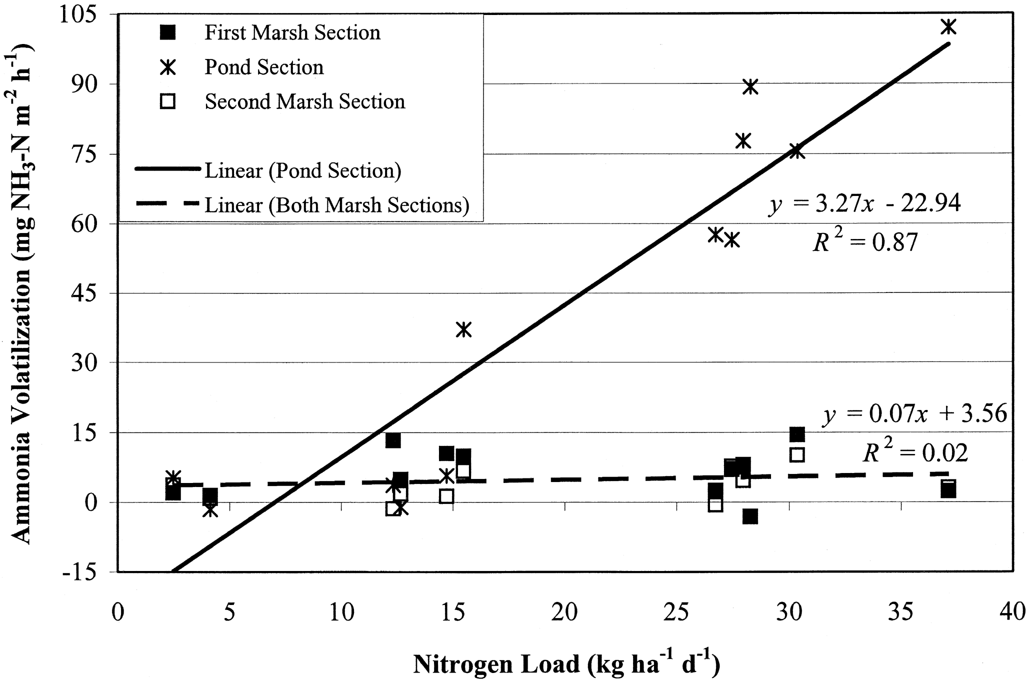

Fig. 3. Regression by wetland section (marsh or pond) of NH_3 volatilization versus monthly average N load.

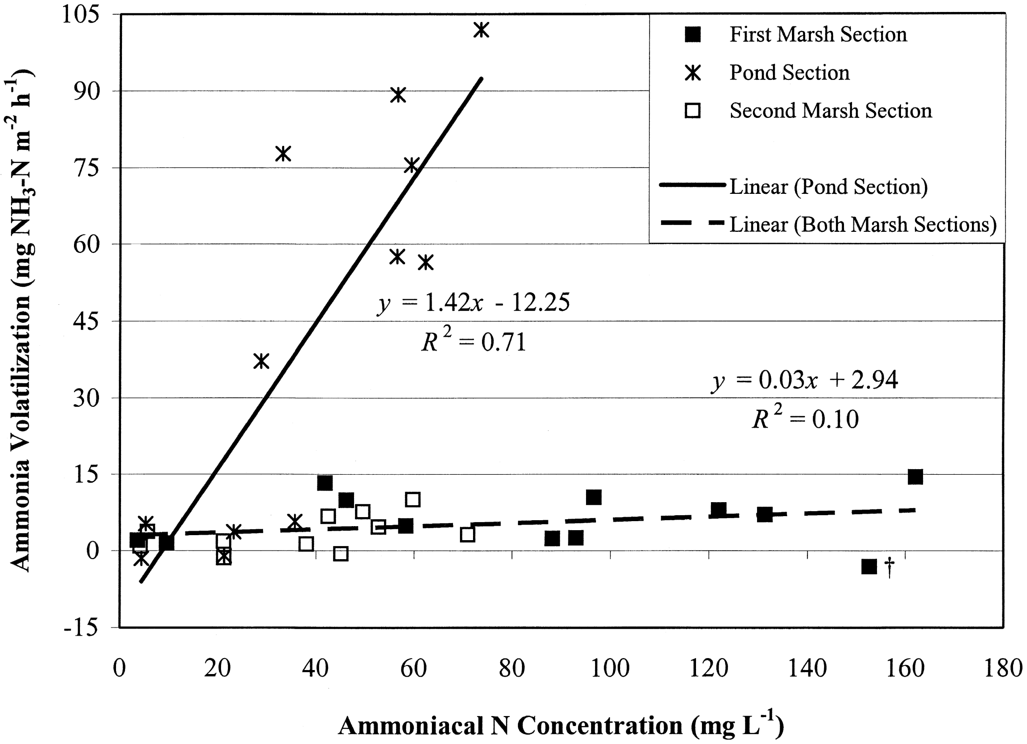

Fig. 4. Regression by wetland section (marsh or pond) of NH_3 volatilization versus ammoniacal N concentration of wastewater in the plot. † If a data point is excluded from regression analysis, the regression equation is $y = 0.06x + 1.92$ ($R^2 = 0.31$).

analyzed by regression, ammoniacal N concentration was a significant regressor ($p < 0.001$) that explained 71% of NH_3 volatilization from pond sections (Fig. 4). Ammoniacal N concentration and air speed measured over the plot were significant regressors ($p < 0.002$) that explained 49% of NH_3 volatilization from marsh sections. This relationship improved ($R^2 = 0.54$, $p < 0.001$) when the volatilization value of −3 mg NH_3–N m^{-2} h^{-1} was excluded from the analysis, an indication that this point may be an outlier. Regression also indicated that NH_3 volatilization was affected by the month in which the tests occurred, with NH_3 volatilization tending lower in August, but this was probably the result of lower ammoniacal N concentrations during August (Table 1).

When the full data set was analyzed by regression, wetland section (marsh versus pond) and the pH of plot wastewater were significant regressors ($p < 0.0001$) that explained 54% of the variation in NH_3 volatilization. Ammonia volatilization tended to increase as pH increased. This was expected because the percent of wastewater ammoniacal N present as the volatile form would have increased with an increase in pH (Kadlec and Knight, 1996). This partly explains why the pond sections exhibited higher NH_3 volatilization than marsh sections. The pond sections that received N loads greater than 15 kg ha^{-1} d^{-1} tended to have higher wastewater pH than their adjacent marshes (Table 1). The higher pH probably resulted from the presence of algae in these pond sections. As algae photosynthesize during the day, the consumption of carbon dioxide can raise the pH of their surroundings (Reddy, 1981). The pond sections of wetlands with lower N loads appeared to be dominated by duckweed (*Lemna minor* L.).

Research on NH_3 volatilization from manure storage lagoons indicated that NH_3 volatilization was affected by wind blowing across the lagoon along with wastewater pH, ammonia concentration, and temperature (Harper et al., 2000). This would indicate that the different NH_3 volatilization trends could have resulted from the pond sections having a larger wind-exposed surface area compared with marsh sections, but such a relationship was not supported by regression analysis. The lack of evidence for such a relationship was due mainly to the fact that pond sections exhibited rates of NH_3 volatilization similar to marsh sections at N loads below 15 kg ha^{-1} d^{-1} (Tables 2 and 3). It is possible that the duckweed covering the surface of those pond sections reduced the effect of wind on NH_3 volatilization. No reliable conclusions could be drawn about the effect of wastewater temperature because of the missing data points. Therefore, more research needs to be conducted to fully explain the different NH_3 volatilization trends displayed by pond and marsh sections.

During the study period, NH_3 volatilization was important to the N budget of these wetlands when N loads were greater than 15 kg ha^{-1} d^{-1}. At these loads, NH_3 volatilization removed 23 to 36% of the N loaded to the wetlands, and its contribution tended to increase as N load increased (Table 4). This NH_3 volatilization also accounted for 54 to 79% of the total N removed by these wetlands. These results indicate that NH_3 volatilization was the dominant N removal mechanism at N loads greater than 15 kg ha^{-1} d^{-1}. It should be noted

Table 4. Contribution of mean NH_3 volatilization to the nitrogen budget of six marsh–pond–marsh constructed wetlands in Greensboro, NC that received swine wastewater.

		Mean NH_3 volatilization			
N load in†	**N removed†**	**Marsh and pond†**	**Marsh only‡**	**Marsh and pond**	
kg ha^{-1} d^{-1}		kg NH_3–N ha^{-1} d^{-1}		% load	% removed
32.7	**14.8**	**11.8**	**0.6**	**36**	**79**
28.9	**16.0**	**9.1**	**2.4**	**31**	**57**
27.3	**14.3**	**8.4**	**0.6**	**31**	**59**
15.1	**6.4**	**3.4**	**1.7**	**23**	**54**
12.5	**7.3**	**0.8**	**1.2**	**7**	**11**
3.3	**1.1**	**0.6**	**0.5**	**17**	**51**

† Values are means for both sampling periods and are listed in order of decreasing N load.

‡ Ammonia volatilization expected if each wetland was continuous marsh instead of marsh–pond–marsh.

that these results only apply to the daytime hours during the summer. Ammonia volatilization can be expected to be lower at night and lower during the winter due to higher atmospheric stability and lower wastewater temperatures (Harper et al., 2000; Bussink et al., 1996). A drop in pH as a result of the cessation of photosynthesis in the pond sections would also lead to a reduction in NH_3 volatilization during the nighttime.

Even though the results only apply to the sampling period, they still indicate that NH_3 volatilization is a concern for animal wastewater treatment by marsh–pond–marsh systems, especially since, at N loads greater than 15 kg ha^{-1} d^{-1}, NH_3 volatilization was greater from these systems than that expected to occur if the wetlands were of the continuous marsh type. At these loads, the mean NH_3 volatilization values from the marsh–pond–marsh systems were 3.4 to 11.8 kg NH_3–N ha^{-1} d^{-1}, but if the wetlands were of the continuous marsh type then the mean NH_3 volatilization would have been 0.6 to 2.4 kg NH_3–N ha^{-1} d^{-1} (Table 4). The latter rates, which are similar to those reported by Poach et al. (2002), would represent a minor component (less than 12%) of the total N budget of these wetlands.

Results indicate that NH_3 volatilization by marsh–pond–marsh systems can be reduced by reducing the ammoniacal N concentration of the wastewater (Fig. 4). One means of reducing the ammoniacal N concentration of the wastewater is by diluting the wastewater with water, but that would incur the disadvantage of increasing the total volume of wastewater that needed treatment. The ammoniacal N concentration of wastewater can also be reduced by converting ammoniacal N to nitrate and nitrite N by the process of nitrification. Nitrification of swine wastewater before wetland application was shown to reduce NH_3 volatilization by continuous marsh systems (Poach et al., 2002). However, if the addition of pre-wetland nitrification is needed to improve swine wastewater treatment by marsh–pond–marsh systems, then the pond section becomes unnecessary because the pond section was added to the treatment wetland design specifically to enhance wastewater nitrification (Hammer, 1994; Reaves, 1996). Therefore, at N loads greater than 15 kg ha^{-1} d^{-1}, continuous marsh systems should be preferred over marsh–pond–marsh systems for the treatment of wastewater from confined animal operations.

CONCLUSIONS

Enclosure measurements indicated that NH_3 volatilized from marsh and pond sections of the wetlands during July and August of 2001. As N load increased, NH_3 volatilization increased at a significantly greater rate over pond sections compared with marsh sections. Pond sections that received N loads greater than 15 kg ha^{-1} d^{-1} produced rates of NH_3 volatilization greater than 36 mg NH_3–N m^{-2} h^{-1}, while all marsh sections produced rates less than 16 mg NH_3–N m^{-2} h^{-1}. The difference in NH_3 volatilization between pond and marsh sections was partially explained by wastewater pH. However, more research needs to be conducted to fully explain the different NH_3 volatilization trends displayed by pond and marsh sections.

During the study period, NH_3 volatilization was an important contributor to the N balance of marsh–pond–marsh systems when N loads were greater than 15 kg ha^{-1} d^{-1}. At these loads, NH_3 volatilization removed 23 to 36% of the N loaded to the wetlands, and it accounted for 54 to 79% of the total N removed by these wetlands. Marsh sections were minor contributors to the overall NH_3 volatilization of these wetlands, so the pond section exacerbated rather than ameliorated the NH_3 volatilization at N loads greater than 15 kg ha^{-1} d^{-1}. At these loads, continuous marsh systems should be preferred over marsh–pond–marsh systems for the treatment of wastewater from confined animal operations.

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