SWINE WASTEWATER TREATMENT IN CONSTRUCTED WETLANDS

HUNT, P.G., POACH, M.E.*, REDDY, G.B.**, STONE, K.C.*, VANOTTI, M.B.*, and HUMENIK, F.J.****

**USDA-ARS, 2611 W. Lucas St., Florence, SC 29501, USA*
***Department of Natural Resources, Carver Hall, NCA&T State University, Greensboro, NC 27411, USA*
****CALS Animal Waste Management Program, NC State University, Box 7927, Raleigh, NC 27695, USA*

ABSTRACT

One of the passive technologies being used for animal wastewater treatment is constructed wetlands. We have investigated swine lagoon wastewater treatment in both continuous marsh and marsh-pond-marsh (MPM) type constructed wetlands for their nitrogen treatment efficiency, ammonia volatilization, denitrification, and treatment modeling. Continuous marsh systems were better than the MPM systems particularly if planted to bulrush. Ammonia volatilization was generally <10% of the applied N in the marsh sections. However, the pond sections of the MPM systems had high levels of ammonia volatilization when loading rates exceeded 15 kg N ha^{-1} day^{-1}. Denitrification was particularly high in the floating sludge layer of the wetland surface especially when the wastewater was partially nitrified. Treatment efficiency was reasonably predicted by current modeling techniques used for municipal wastewater treatment in constructed wetlands.

INTRODUCTION

Wetlands have been used successfully for advanced treatment of municipal and residential wastewaters in the USA and around the world for over three decades (Kadlec and Knight, 1996). However, their function and reliability for animal wastewater treatment are less documented (Hunt and Poach, 2001). Generally, the focus of animal wastewater treatment in constructed wetlands is to remove nutrients and, thereby, decrease the land necessary to receive, transform, and assimilate the nutrients in the wastewater (Knight et al., 2000; Hunt et al., 2002). To gain a better understanding of wetland wastewater treatment, research was conducted on two types of constructed wetlands, continuous marsh and marsh-pond-marsh (MPM). The major points to be discussed in this paper are 1) treatment performance, 2) potential ammonia volatilization, 3) denitrification, and 4) wetland design.

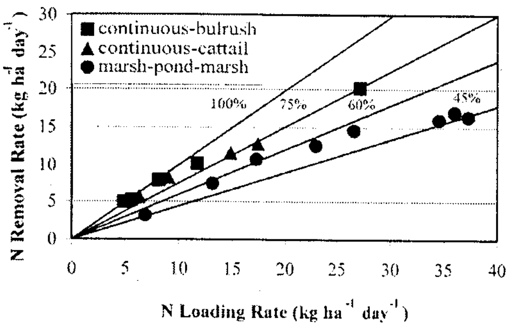
TREATMENT PERFORMANCE

Research in North Carolina has found that constructed wetlands were very effective in the removal of nitrogen from the swine wastewater. At nitrogen loading rates from 5 to 37 kg N $ha^{-1}day^{-1}$, the wetlands removed >45% of the nitrogen from the wastewater (Figure 1). Continuous marshes were more efficient at nitrogen removal than were MPM systems. Pathogens were also reduced by wetland treatment (Hill et al., 1999). Neither type of wetland system was effective in removing phosphorus (Hunt et al., 2002). Removal of P will likely be by a precipitating agent. However, nitrogen treatment can potentially be enhanced and the remainder of this paper will focus on nitrogen.

AMMONIA VOLATILIZATION

Ammonia (NH_3) volatilization has been implicated as a significant nitrogen (N) removal mechanism operating in treatment wetlands when wastewater contains greater than 20 mg L^{-1} ammonia (Kadlec and Knight, 1996). Animal wastewater generally has ammonia concentrations greater than 20 mg L^{-1}, high oxygen demand, and very little nitrogen in the nitrate form (Knight et al., 2000). The contribution of NH_3 volatilization to nitrogen removal was determined using a special open-ended enclosure with forced airflow on 1- x 4-m plots in continuous marsh and MPM systems. Poach et al. (2002) give a detailed description of the method used to measure NH_3 volatilization.

1Fig. 1. Nitrogen removal efficiency of constructed wetlands receiving swine wastewater.



Ammonia-N volatilization from the marshes of both systems tended to account for <10% of the N applied (Table 1). Therefore, nitrification-denitrification is more likely the major nitrogen removal process in these marsh wetlands. The pond sections of the MPM systems also exhibited minor ammonia volatilization at the lowest N loads, but at N loads higher than 15 kg N ha^{-1} day^{-1}, NH_3 volatilization accounted for about 50% of nitrogen removed by the systems. The high NH_3-N volatilization exhibited by these pond sections resulted from their higher pH values and higher wind-exposed surface area compared to the marshes. These results indicate that at high N loads continuous marshes may be better suited for treating swine wastewater than MPM systems. Also, while NH_3 volatilization was a minor nitrogen removal mechanism in the marshes, its contribution to nitrogen loss should not be ignored. Pre-wetland nitrification of swine wastewater can reduce ammonia volatilization. Ammonia-N volatilization averaged only 0.2 kg N ha^{-1} day^{-1} for the continuous wetlands when they were loaded with wastewater that was partially nitrified.

Table 1. Ammonia volatilization from constructed wetland surfaces. (± 1 sd. in parentheses).

Wetland	Nitrogen Applied (kg N ha^{-1} day^{-1})						
Surface	6	13	18	23	26	34	37
	kg N ha^{-1} day^{-1}						
Marsh	0.4 (0.1)	2.2 (1.4)	1.7 (0.8)†	2.2 (0.4)	1.3 (0.9)	-	2.6 (1.3)
Pond	0.5 (1.2)	0.3 (0.8)	-	5.1 (5.3)	16.2 (3.4)	22.9 (2.2)	15.8 (3.2)

† Average ammonia-N volatilized from the continuous bulrush and cattail systems. Data from Poach et al. (2002) and subsequent study of nitrogen treatment by the continuous marshes.

DENITRIFICATION

Microbial denitrification has often been considered one of the primary loss mechanisms of nitrogen in wetlands. Significant denitrification is consistent with wetland function (Hunt and Lee, 1975; Kadlec and Knight, 1996), but the magnitude of denitrification is ofttimes questioned. We measured denitrification over a four-year period in the bulrush/rush and cattail/burreed continuous marsh wetland treatment systems of Figure 1. Intact soil cores and disturbed soil samples were analyzed for denitrification rate and denitrification enzyme activity (DEA), respectively, via the acetylene inhibition method.

Denitrification rates varied with operating parameters, particularly the total accumulated N load and water depth. In the control DEA treatment, which measured denitrification without amendments, bulrush wetlands had two-fold higher DEA means (P∃0.001) than did cattail wetlands, 0.516 and 0.210 $\mu g\ N\ g^{-1}\ hr^{-1}$, respectively. Both plant systems were limited by low quantities of nitrate-N (nitrate-N concentration of the wastewater was generally <0.5 mg L^{-1}). Partial pre-wetland nitrification greatly increased denitrification in the intact cores; in the first meter of the cattail wetland, the rate increased to >16 kg N ha^{-1} day^{-1}. However, the nitrate was quickly consumed, and beyond first 3 meters denitrification rates were not different than when the wastewater contained no nitrate.

The previous denitrification values were from the soil layer. However, denitrification in wetlands is associated with microbes in the soil, detritus layer, and the floating sludge. Denitrification in the floating sludge layer (Table 2) was much (>20-fold) higher on a unit weight basis than in the soil and/or detritus layers. The rates of denitrification in the floating sludge were similar to those we obtained with polyvinyl alcohol-immobilized-denitrifying sludge pellets. These data indicate the very high potential for wetland removal of nitrate-N. However, at some level of nitrate loading, available carbon for microbial respiration would be the limiting factor, and carbon would need to be added to the wetland.

Table 2. Denitrification enzyme activities from wetland surface components.

Wetland System	Soil Amendment	DEA (mg N gm $Sample^{-1}$ h^{-1})		
		Soil Layer	Detritus Layer	Floating Sludge
Bulrush/rush	None	0.053	0.016	2.190
	Nitrate-N Added	0.093	0.277	4.752
Cattail/burreed	None	0.014	0.035	0.258
	Nitrate-N Added	0.054	0.158	5.116

WETLAND DESIGN PARAMETERS

Technical requirements for wetland design have been based mainly on municipal systems and limited data on animal waste systems. Design guidelines for sizing wetlands are typically based on a first-order kinetics area-based uptake model (Reed et al., 1995; Kadlec and Knight, 1996). At the RAMIRAN 2000 conference, Hunt et al. (2000) presented preliminary design parameter estimates for the first-order area-based uptake model for the surface flow constructed wetland in Duplin County, NC. In Stone et al. (2002), we used the Kadlec and Knight (1996) k-C* model to calculate rate constants for nitrogen treatment in the continuous wetlands (Table 3). This model incorporates the hydraulic loading rate, concentrations into and out of the wetlands, and a temperature-based rate constant. The calculated rate constants were generally similar to or slightly lower than those reported in the limited literature. Many of the literature-cited rate constants were from summary data from various wetlands. Our data provide detailed estimates for both bulrush and cattail constructed wetlands. Based on our calculated rate constants, newly constructed wetlands with similar mean loading rates and concentrations would be slightly larger (~5%) than wetlands designed with the currently available guidelines.

Table 3. Design parameters of constructed wetlands used for swine wastewater treatment.

	Calculated†			C* Assumed ‡			C* = 0	
	K20 ($m\ yr^{-1}$)	θ	C* ($mg\ L^{-1}$)	K20 ($m\ yr^{-1}$)	θ	C* ($mg\ L^{-1}$)	K20 ($m\ yr^{-1}$)	θ
TN Bulrush/rush	8.85	1.02	10.99	8.71	1.02	10	7.45	1.03
TN Cattail/burreed	8.66	0.98	5.81	9.35	0.98	10	7.86	0.99
NH_4-N Bulrush/rush	8.98	1.03	7.73	8.60	1.03	3	7.82	1.03
NH_4-N Cattail/burreed	9.39	0.98	4.32	9.20	0.98	3	8.62	0.99

† Rate constant (K20), dimensionless temperature coefficient (θ), and background concentrations (C*) calculated simultaneously using the Excel solver routine. ‡ C* assumed from Knight et al. (2000).

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