TREATMENT OF SWINE WASTEWATER IN WETLANDS WITH NATURAL AND AGRONOMIC PLANTS

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ABSTRACT

In the USA, we assessed constructed wetlands (surface flow) with both natural and agronomic plants for treatment of anaerobic-lagoon swine wastewater. We used both continuous marsh and marsh-pond-marsh systems. The plant communities grown in the wetlands were rush-bulrush, bur-reed-cattail, soybean grown in saturated-soil culture, and flooded rice. Nitrogen loading rates ranged from 2 to 35 kg ha^{-1} day^{-1}. The N removal rates were remarkably constant for all of the wetland types up to a loading of 25 kg ha^{-1} day^{-1} [y = 0.86 (N loading rate) - 0.3, R^2 = 0.98]. At the lower loading rates, plant and soil accumulation constituted a significant portion (~ 30%) of the total amount applied, but at the higher loading rates, microbial transformations were likely the more dominant treatment factors. Denitrification enzyme assays (DEA) indicated that nitrate was the limiting factor and that denitrification could be increased by protocols to promote oxygen incorporation or pre-wetland nitrification of wastewater. Ammonia volatilization was determined to be low indicating that nitrate or other denitrification intermediates were present albeit difficult to measure. The wetland data were analyzed to calculate the rate constants of TN and TP for our wetland systems. Our calculated K_T and C_* values were, respectively, lower and similar to those in literature. We consider constructed wetlands a viable alternative to or augmentation to current anaerobic lagoons and land application. They are likely to be most effective when used in a total waste management system.

INTRODUCTION

In the USA there is enormous public sentiment against traditional animal waste management via anaerobic lagoons and land application of wastewater. Much of this concern is because the large number of swine in concentrated areas promotes heavy nutrient loading to available land. Resolution of the problem via traditional municipal wastewater treatment is not economically or managerially feasible. Thus, passive treatment alternatives with very high loading potential are desired. One such alternative is constructed wetland treatment (Hunt et al., 1995; Szogi et al., 2000). The objective of our research was to establish the treatment mechanisms and loading rates for treatment of swine wastewater in constructed wetlands.

MATERIALS AND METHODS

The continuous marsh wetland cells were 4 by 33.5 m. They were arranged in two cell sequences with one sequence planted to rush-bulrushes and the other planted to bur-reed-cattails. The third pair of wetlands contained saturated soybean and flooded rice. A nitrogen-loading rate of 3 kg ha^{-1} day^{-1} was used during 1993 (the first year of operation). In subsequent years, the loading rate for the soybean and rice remained < 10 kg ha^{-1} day^{-1}, but the rate was increased to 25 kg ha^{-1} day^{-1} in the cells containing natural wetland plants. The wetlands were operated at depths ≈ 0.2 m. The marsh-pond-marsh (M-P-M) wetlands contained a combination of cattails and bulrushes. The M-P-M sections were 10 by 10, 10 by 20, and 10 by 10 m, respectively. The marsh portions were ≈ 0.2 m deep, and the pond section was ≈ 1 m deep. They were operated in duplicates (4 wetlands) at loading rates of 15 and 35 kg ha^{-1} day^{-1}. Measurements of flow into and out of the cells as well as samples for analyses were obtained by automated methods. Denitrification enzyme assays were done on disturbed soil samples by the acetylene blockage method (Tiedje, 1982). The ammonia volatilization measurements were made using a 1- by 4- by 2.5-m open chamber in which inflow and outflow ammonia masses were determined.

RESULTS AND DISCUSSION

Treatment effectiveness: The annual mass N removal vs. N load was remarkably consistent for loading rates of 2 to 25 kg ha^{-1} day^{-1} for all wetland systems (Fig. 1). This was very encouraging data because it represented treatment over a six-year period in multiple locations with very different wetland systems. The rate of treatment in these wetland systems far exceeded the capacity of agronomic systems to assimilate nitrogen. After wetlands have dramatically reduced total amount of N, much less cropland will be required to accept the lower nitrogen load. Moreover, the timing of the application can be more easily maintained in balance with weather patterns and crop needs. For instance, greater than 5 Mg N ha^{-1} would be removed each year by the wetlands with 25 kg N ha^{-1} day^{-1}, 70% removal, and 250 days of application. The wetland would thereby substitute for > 30 ha of cropland loaded at 150 kg N ha^{-1} yr^{-1}. Reduction of load or land is the primary concern in the USA because wastewater may not be directly discharged to streams. It must be applied to land. In addition to lowering the total amount of N in the effluent, wetlands also lowered the N concentration. Effluent [N] was moderately correlated to N load to the wetlands after a natural log conversion of the data (R^2 = 0.41-0.44). However, in contrast to N treatment, these wetlands have not been as effective in removing P, particularly at high loading levels. Thus, P must be removed via auxiliary treatment before or after wetland treatment - we are investigating several promising possibilities. In addition to the treatment functionality of wetlands, their operational passivity and moderate cost make them a good choice for many animal producers.

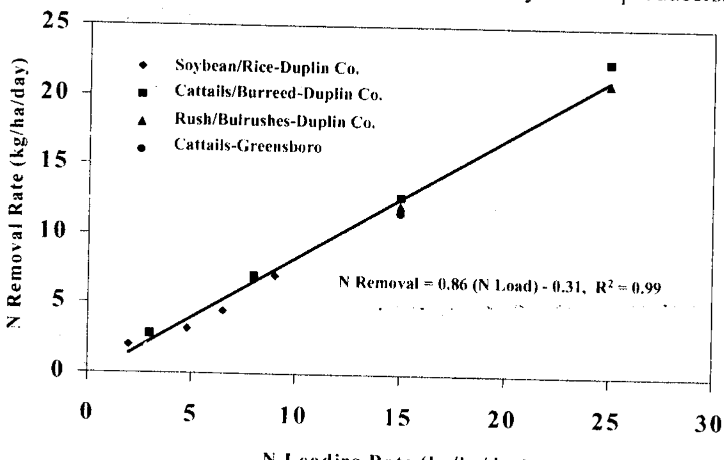

Fig 1. Nitrogen removal efficiency of different constructed wetlands.

Treatment processes: Although significant amounts of N and P accumulated in the plants and soil, it was small (<10%) relative to the total N applied. The plants were not harvested; they were allowed to accumulate on the wetland surface. Thereby, they were able to serve as carbon sources and reaction sites for microbial reactions. The assumed major mechanism for N removal was microbial denitrification. The redox conditions of the wetland soils were consistent with this assumption. They were highly reducing, generally 100 to -200 mV. Denitrification potential was highest in the shallower portion of the wetlands (Fig. 2). This may have been due to more effective interfacing of nitrifying and denitrifying microenvironments. The effect of depth was consistent for the control and the treatment with additional nitrate. However, the addition of nitrate essentially doubled the DEA values. This information was consistent with data we obtained from microcosms that indicated much higher treatment potential with nitrified effluent. We will obtain more definitive data on the value of pre-wetland nitrification this year from an experiment with untreated vs. nitrified anaerobic lagoon wastewater as the effluent into paired wetlands.

Ammonia loss: The other possible mechanism for loss of large quantities of N is ammonia volatilization. This loss had been assumed to be minimal because most of the ammonia was in the nonvolatile ammonium form in the wetlands with a water pH < 8. Nonetheless, the

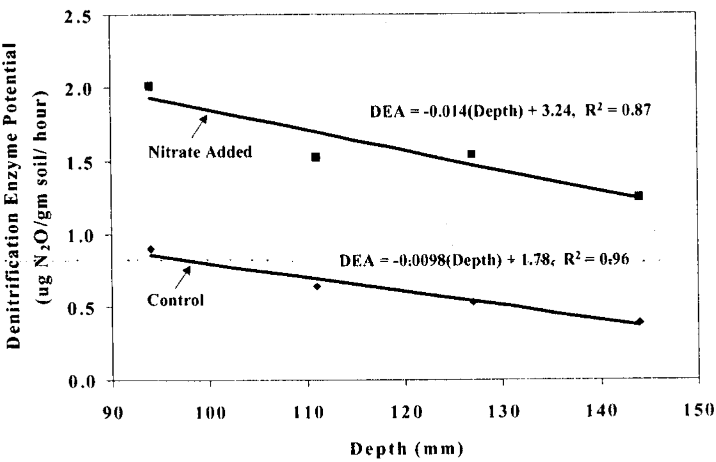

Fig 2. Mean DEA for continuous marsh wetlands over four years.

very high rates of N removal caused concern about ammonia volatilization. When we considered the options for measurement of ammonia volatilization, we determined that an open chamber device would be best. The chamber was 2.5 m high and 1 x 4 m in area. Synchronized inlet and outlet fans controlled air speed through the chamber, and ammonia was captured in acid taps. In control tests, we determined that the chamber was capable of recovering ~ 98% of the ammonia volatilized from a standard. Ammonia volatilization from the wetlands was found to occur, but it was only at levels < 10% of the loading rate (Table 1). Additionally, when pre-wetland nitrification was used to increase the rate of denitrification, ammonia volatilization was minuscule.

Our N removal results suggest that 1) denitrification is very active and 2) nitrate or other intermediates necessary for denitrification are active in the wetlands, even though they are not present in easily measurable quantities. These findings are consistent with those of Harper et

Table 1. Initial values of ammonia volatilized from continuous marsh constructed wetlands loaded with swine wastewater.

N Loading	N Volatilization	Loss to Volatilization
---------kg N ha^{-1} day^{-1}---------		%
43 (7)*	3 (2)	7

*Standard Deviation

al. (2000); they found lower than expected ammonia volatilization but higher than expected denitrification from swine wastewater anaerobic lagoons. Thus, the consistent N removal and simplicity of wetlands make them a suitable technology for swine wastewater treatment. This is particularly true where alternatives to traditional methods of swine wastewater treatment are needed because of insufficient land or regulator pressure.

Design: The design of wetlands for animal waste treatment was originally derived from municipal treatment wetlands (Kadlec and Knight, 1996). Constructed wetland design has typically been approached as a first-order rate equation based on plug flow assumptions. The most popular approach in design has been presented by Kadlec and Knight (1996) as

(equation 1)

$$\left[\frac{C_{out} - C_*}{C_{in} - C_*}\right] = \exp(-\frac{K_T}{q})$$

where C_{out} = the outflow concentration (mg/L),
C_{in} = the inflow concentration (mg/L),
C_* = the background concentration (mg/L),
q = the hydraulic loading rate (m/d),
K_T = the rate constant adjusted for temperature (m/d).

(equation 2)

$$K_T = K_{20}\theta^{(T-20)}$$

$_{20}$= the rate constant at 20EC (m/d),
2 = dimensionless temperature coefficient,
T = the temperature (EC).

Equation (2) was then rearranged in order to calculate the K_{20} rate constant at 20EC and the dimensionless temperature coefficient.

(equation 3)

$$\log(K_T) = \log(K_{20}) + \log(\theta)(T - 20)$$

Total phosphorus rate constant is not considered a function of temperature and was calculated from equation (1).

The wetland data were analyzed to calculate the rate constants of TN and TP for the two wetland systems (rushes and cattails). The temperature-based rate constants were calculated using equation (2). The rate constants were calculated and then regressed against the temperature to determine the K_{20} rate constant and 2 from equation (3). In Table 2, K_{20} and 2 are shown for TN for the two wetland systems studied. There was little difference among the individual constitutes across the wetland systems. These results compare favorably but are lower than those from Kadlec and Knight (1996) and Reed et al. (1995). The NRCS field test method (Payne Engineering, 1997) suggests using a K_{20} of 14 m/yr for TN and 10 m/yr for NH_4-N. We calculated TN K_{20} values of 6.5-7.5 m/yr. Our lower values for the rate constants were calculated assuming $C_*=0$. Also, using a lower K_{20} value would result in a more conservative prediction for treatment in the wetland systems. In our regression analysis, we had very low correlation coefficients, which suggest that the rate constants in our systems were not related to temperature. Calculation of the rate constants without the influence of temperature is shown in Table 3. Additionally, we simultaneously solved equation (1) for both K_T and C_* (Table 3). These mean rate constants and C_* values are in similar agreement with the previous regression results. Our calculated K_T values are below those in literature, and the C_* values are in close agreement with those in literature. In our analysis, we assumed minimal ammonia volatilization based on the work previously discussed.

The rate constants for TP were calculated based on equation (1). The K_T values for TP ranged from 1.25 to 2.1 m/yr for the two wetland systems studied. These rate constant values were much lower than those reported in Kadlec and Knight (1996) and Reed et al. (1995). Their values from the analyzed data bases ranged from 2 to 24 m/yr with a mean of 12 m/yr, and Reed et al. (1995) suggested a value of 10 m/yr. Our data were on the low range of their values. The data from this project had a much higher loading rate of TP than many of those reported in the references. Also, after the first year, the efficiency of the wetlands for phosphorus treatment declined dramatically. This suggests that an alternative method of phosphorus removal should be investigated.

Table 2. Regression parameters for the calculation of rate constants for the first-order area-based uptake design model.

	N	Intercept	K_{20} (m/d)	K_{20} (m/yr)	Slope	è	r^2
TN System 1	62	-3.994	0.018	6.726	0.031	1.031	0.133
TN System 2	63	-3.906	0.020	7.343	0.029	1.030	0.079

Table 3. Calculated rate constants and C_* parameters for the first-order area-based uptake design model.

	K_T (m/yr)	C_* (mg/L)	Mean K_T (m/yr)
TN System 1	7.5	9.5	6.4
TN System 2	8.2	8.7	7.0
TP System 1	-	-	1.25
TP System 2	-	-	2.1

Mean K_T calculated assuming $C_*=0$.

CONCLUSIONS

1. Constructed wetlands with natural wetland plants were capable of effectively treating 25 kg N ha^{-1} day^{-1}. However, phosphorus removal was not as effective, and would need some other treatment augmentation.
2. Denitrification was the apparent main mechanism of nitrogen loss.
3. Ammonia volatilization was low, generally less than 15%.
4. Design criteria were generally found to be appropriate, but some adaptation and refinements are needed.
5. Constructed wetlands can be an effective treatment method for animal producers, particularly when used as part of a total waste management system.

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