CONSTRUCTED WETLAND DESIGN AND PERFORMANCE FOR SWINE LAGOON WASTEWATER TREATMENT

K. C. Stone, P. G. Hunt, A. A. Szögi, F. J. Humenik, J. M. Rice

ABSTRACT. *Although constructed wetlands have been identified as a potentially important component of animal wastewater treatment systems, their design requirements have been based mainly on municipal systems. The objective of this investigation was to examine various design approaches for constructed wetlands in relation to the performance of our constructed wetlands for swine wastewater treatment. The free water surface wetlands in Duplin County, North Carolina, investigated in this study were constructed in 1992 based on the Natural Resources Conservation Service (NRCS) presumptive design method. We used four wetland cells (3.6 m × 33.5 m) with two cells connected in series; the two series of cells were planted and predominated, respectively, by either bulrushes or cattails and were studied from 1993 to 1999. The wetlands were effective in treating nitrogen with mean total nitrogen and ammonia–N concentration reductions of approximately 85%; however, they were not effective in the treatment of phosphorus. Regression analyses of outflow concentration vs. inflow concentration and hydraulic loading rate for total N and ammonia–N were reasonably correlated ($r^2 \geq 0.66$ and $r^2 \geq 0.65$, respectively). Our calculated first–order plug–flow kinetics model rate constants (K_{20}) for total–N and ammonia–N (8.4 and 8.9, respectively) were slightly lower than those reported in the limited literature and currently recommended for use in constructed wetland design. Nonetheless, use of our calculated rate constants would result in about the same size constructed wetland for treating swine lagoon wastewater.*

Keywords: *Wetlands, Nitrogen, Ammonia, Phosphorus, Design.*

The rapid expansion of high–population animal production has resulted in greater amounts of concentrated animal waste to be utilized or disposed of in an efficient and environmentally friendly manner. This has resulted in an adaptation of some municipal wastewater treatment technologies while new treatment technologies are being developed. We evaluated the effectiveness of a constructed wetland that has been used to treat swine wastewater since 1993, and we calculated design parameters from these wetlands that can be used in future systems.

Constructed wetlands have been used for many years in municipal wastewater treatment. In the late 1980s, interest began to increase in using constructed wetlands for animal wastewater treatment. The technical requirements were based mainly on municipal systems and limited data on animal waste systems. The majority of constructed wetlands for treatment of animal waste have been installed since 1989 (Payne and Knight, 1997). Constructed wetlands were originally thought to be able to produce an effluent that could be discharged. However, concern for the environment and discharge regulations have precluded this approach. Constructed wetlands are used to reduce the nutrient loading of wastewater spray fields. This is an important concern where land for application is limited (Barker and Zublena, 1995).

The USDA Natural Resources Conservation Service (USDA, 1991) constructed wetland design guidelines for animal waste treatment were based on BOD_5 loading to the wetlands (presumptive method). These guidelines stated minimum levels of BOD_5 and ammonia–N exiting the wetland with a recommended residence time of at least 12 days. The NRCS cautioned that the design guidelines were preliminary, and that they would be modified as more information on using constructed wetlands for animal waste became available.

A more physically based approach was presented for municipal wastewater treatment wetlands by both Reed et al. (1995) and Kadlec and Knight (1996). Both design models are based on a first–order kinetics area–based uptake model. Reed et al. (1995) incorporated flow rate, wetland depth, wetland porosity, a temperature–based rate constant, and inflow and outflow concentrations. Their rate constant is a function of depth and porosity of the wetlands.

Kadlec and Knight (1996) refer to their model as the k–C* model. The model incorporates the hydraulic loading rate, concentrations into and out of the wetlands, and a temperature–based rate constant. They also include a background concentration parameter (C*). Their rate constant differs

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from Reed et al. (1995) in that it is independent of depth and porosity of the wetlands.

Payne and Knight (1997) compared both the Reed et al. (1995) and Kadlec and Knight (1996) design methods. They found that the Kadlec and Knight (1996) method typically required a greater surface area for the constructed wetland than the Reed et al. (1995) method. The main difference was based on the design depth of the wetland in the Reed et al. (1995) model. Payne and Knight (1997) suggested that if the Reed et al. (1995) model were to be used, an initial minimum depth should be used in order to maximize the surface area of the wetland.

The wetlands discussed in this article were constructed to treat swine lagoon effluent. They were installed in 1992 as part of a Water Quality Demonstration Project (WQDP) in the Cape Fear River Basin, Duplin County, North Carolina (Stone et al., 1995). Aspects of their performance have been discussed by Hunt et al. (1994, 1999, 2002), Szögi et al. (1995), and Poach et al. (2002). The objective of this article was to examine the various design approaches for constructed wetlands in relation to the performance of our NRCS presumptive method designed constructed wetlands.

METHODS

SITE DESCRIPTION AND OPERATION

In 1992, a free water surface (FWS) wetland system consisting of four 3.6 × 33.5 m wetlands cells was constructed in Duplin County, North Carolina, and studied from 1993–1999 (fig. 1). The wetland systems were designed by the Natural Resources Conservation Service (NRCS). The NRCS and Murphy Farms, Inc., constructed the wetlands as part of a USDA Water Quality Demonstration Project (Stone et al., 1995). A detailed discussion of the performance and component functions can be found in Hunt et al. (2002). The wetland cells were excavated, and the sidewalls as well as bottoms were lined with 0.3 m of clay and covered with 0.25 m of loamy sand topsoil. The cells were arranged into two parallel sets of two end–to–end connected cells (fig. 1). The lengthwise slope of the wetland cells was approximately 0.2%, and a water level of 0.15 m was maintained at the outlet of each cell. The four wetland cells were planted with native vegetation. Wetland system 1 (cells 1 and 2) contained rush (*Juncus effusus*) and bulrushes (*Scirpus americanus*, *Scirpus cyperinus*, and *Scirpus validus*), and wetland system 2 (cells 3 and 4) contained bur–reed (*Sparganium americanum*) and cattails (*Typha angustifolia* and *T. latifolia*).

The wetland cells began receiving swine lagoon effluent in 1993. Initially, to prevent possible damage to the wetland plants, the lagoon effluent was diluted with well water 10–fold to provide a target total nitrogen loading rate of 3 kg/ha/day. Later, after determining that the plants in the system could receive effluent with greater ammonia concentrations, the wetlands were loaded at target total nitrogen rates of 8 kg/ha/day in 1995, 15 kg/ha/day in 1996, and 25 kg/ha/day from 1997 to 1999 (Rice et al., 1999). The wetland cells were loaded using an automated system with float control valves in the dilution mixing tank.

Flow into and out of the wetland cells was measured using V–notch weirs and four PDS–350 ultrasonic open–channel flow meters (Control Electronics, Morgantown, Pa.). In 1997, a backup flow measurement system was added using tipping bucket samplers. Inlet manifolds and, later, buried troughs were installed at the wetland inlets to equally distribute flow across the width of the cells. An automated wastewater sampler was installed to measure wastewater inflow, and two samplers were installed to measure wastewater at the outlets of each wetland system. From 1993 to September 1997, the automated samplers combined twenty–one 4–hr samples into a 3.5–day composite. In September 1997, the automated samplers were programmed to collect twenty–one 8–hr samples into a 7–day composite. Sulfuric acid was added as a preservative to the sample bottles prior to sample collection.

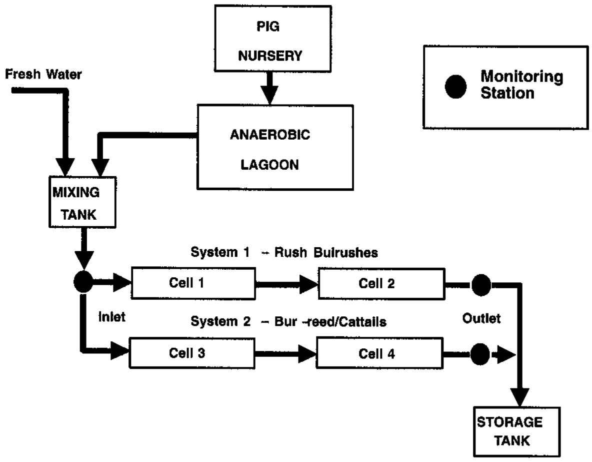

Figure 1. Schematic of constructed wetlands system.

Wastewater samples were analyzed for total Kjeldahl nitrogen (TKN), ammonia–nitrogen (NH_4–N), nitrate–nitrogen (NO_3–N), and total phosphorus (TP) using EPA methods (U.S. EPA, 1983). All samples were analyzed with automated analyzers (Technicon Instruments Corp., Tarrytown, N.Y., and Bran+Luebbe Corporation, Buffalo Grove, Ill.). Total nitrogen (TN) was calculated as the sum of TKN and NO_3–N.

STATISTICAL ANALYSES

Statistical analyses on the constructed wetland data were performed using the Statistical Analysis System (SAS) software (SAS, 1990). Nutrient concentration reductions were calculated as:

$$C_{REDUCTION} = C_{in} - C_{out} \tag{1}$$

where

C_{out} = outflow concentration (mg/L)

C_{in} = inflow concentration (mg/L)

Percentage nutrient concentration reductions were calculated as:

$$\%C_{REDUCTION} = \frac{C_{in} - C_{out}}{C_{in}} \times 100 \tag{2}$$

REGRESSION ANALYSIS

A regression analysis was performed to determine if significant relationships existed between inflow and outflow concentrations to the wetlands. The regression equation was modeled to predict outflow concentration as a function of inflow concentration and hydraulic loading rate and took the form of:

$$C_{out} = aC_{in}^{b} q^{c} \tag{3}$$

where

q = hydraulic loading rate (m/d)

a, b, and c = regression coefficients.

Equation 3 was transformed in order to perform the regression in the SAS system with the Proc Reg procedure and was analyzed as:

$$\ln(C_{out}) = \ln(a) + b\ln(C_{in}) + c\ln(q) \tag{4}$$

The regression models provide useful information on the overall performance of the wetlands, but they are typically considered valid only for the range of data used to model them. To determine how our regression analysis relates to other wetlands that treat animal wastewater, we compared our results to the regression models published by Knight et al. (2000). Their study included summary data from various constructed wetlands treating dairy, cattle, swine, poultry, catfish pond water, and runoff from cattle feeding operations. Their data, although extremely important and on a wide–ranging variety of systems, did not provide an extensive study nor quantification of wetlands performance and operation that are provided in this study. This comparison was useful because it combined the strength of both scales of study.

WETLAND DESIGN ANALYSIS

Design of surface flow wetlands for animal waste treatment was originally derived from municipal treatment wetlands (Kadlec and Knight, 1996). Surface flow treatment wetlands typically have nutrient concentration profiles that decrease exponentially with distance from the inlet (Knight et al., 2000). This exponential decrease in nutrient concentration through the wetland is generally modeled as a simple first–order reaction. The first–order reaction model is typically integrated with a plug flow assumption (Kadlec and Knight, 1996; Reed et al., 1995). Although the flow in constructed wetlands is generally intermediate between plug flow and completely mixed, the use of the first–order model with plug flow assumptions provides a conservative design estimate (Knight et al., 2000). Kadlec and Knight (1996) presented the area–based first–order plug flow design model as:

$$\ln\left[\frac{C_{out} - C^*}{C_{in} - C^*}\right] = -\frac{K_T}{q} \tag{5}$$

where

C^* = background concentration (mg/L)

K_T = rate constant adjusted for temperature (m/d):

$$K_T = K_{20}\theta^{(T-20)} \tag{6}$$

where

K_{20} = rate constant at 20° C (m/d)

θ = dimensionless temperature coefficient

T = temperature (° C).

The hydraulic loading rate (q) is defined as:

$$q = \frac{Q_{in}}{A} \tag{7}$$

where

Q_{in} = Inflow (m^3/d)

A = wetland surface area (m^2).

We rearranged equation 5 to solve for the temperature–related rate constant for TN and NH_4–N from the wetland data as:

$$K_T = \frac{Q}{A}\ln\left[\frac{C_{in} - C^*}{C_{out} - C^*}\right] \tag{8}$$

Equation 6 was then rearranged in order to calculate the K_{20} rate constant at 20° C and the dimensionless temperature coefficient:

$$\ln(K_T) = \ln(K_{20}) + (T - 20)\ln(\theta) \tag{9}$$

where $\ln(K_T)$ would be regressed against the temperature term $(T - 20)$.

We used this procedure for TN and NH_4–N. However, the TP rate constant is typically not considered a function of temperature (Reed et al., 1995). Therefore, we assumed that TP reduction was not a function of temperature, and we calculated the TP rate constant based on equation 8.

In addition to solving for rate constants (K_{20}, θ, and C^*) using regression analysis in SAS, we used the Solver spreadsheet function in Microsoft Excel 2000 to simultaneously solve equations 5 and 6 for K_{20}, θ, and C^*. This required an Excel spreadsheet to be constructed with columns of C_{in}, C_{out}, q, mean monthly temperature, initial estimates of K_{20}, C^*, θ, estimated C_{out}, and the sum square error (SSE) term for the difference between observed and estimated C_{out}. The Solver routine then minimized the total SSE term by changing the estimated K_{20}, C^*, and θ values. This simultaneous solution method minimizes the sum of squares between the measured and predicted outflow nutrient concentrations (R. H. Kadlec, 2000, personal communication). We used this Excel spreadsheet procedure for TN and NH_4–N, and TP. Additionally, we calculated K_{20} and θ using $C^* = 0$ to compare with our SAS regression results, and using C^* values estimated from Knight et al. (2000) to compare with their results.

RESULTS

During the study period, the system had a mean residence time of approximately 12 to 14 days with a hydraulic loading rate of ~0.011 m/day (table 1). The mean TN loading rate was ~14 kg/ha/day with the yearly loading rates ranging from ~5 kg/ha/day to ~30 kg/ha/day. The actual TN loading rates varied from target rates due to rainfall and occasional malfunctions in the dilution tank and in the pumping system delivering lagoon effluent to the wetland cells. Inflow TN concentrations increased from an initial year mean concentration of ~43 mg/L to ~250 mg/L for the greater loading rates (fig. 2). Corresponding outflow TN concentrations varied from ~3 mg/L to ~75 mg/L. Concentration reduction efficiencies ranged from 92% at the lower TN loading rate to 70% at the greater loading rates. The overall concentration reduction efficiency for the entire 1993–1999 operation of the wetlands was 84% (table 2). Mean monthly TN concentration reductions for the two wetland systems were consistently between 75% and 100% (fig. 3).

Much of the TN inflow into the wetlands consisted of ammonia–N (NH_4–N). The NH_4–N loading rate ranged from 5 to 27 kg/ha/day. Inflow NH_4–N concentrations had an initial yearly mean of ~35 mg/L and increased to ~225 mg/L at the higher loading rates (fig. 4). Outflow NH_4–N concentrations were initially ~2 mg/L and increased to ~58 mg/L at the higher loading rates. The overall concentration reduction for the entire study was ~86%. At the lower loading rates, the concentration reduction efficiencies were ~92–95%. Concentration reduction efficiencies at the higher loading rates were ~74% (fig. 5). Ammonia volatilization from the wetlands was not a significant removal pathway (<15% of TN loading rate, Poach et al., 2002).

The TP loading of the wetlands ranged from ~1 kg/ha/day initially to ~5 kg/ha/day at the higher TN loading rates. The TP concentration entering the wetlands at the lower loading rate was ~8 mg/L and ~55 mg/L at the higher loading rates (fig. 6). Initially, the wetlands were very effective at removing the TP, with outflow concentrations ranging from 1 to 3 mg/L. This provided an initial concentration reduction efficiency of 85%. At the higher loading rates, the outflow concentration increased to ~40 mg/L, with concentration reduction efficiencies ranging from 6% to 35% (fig. 7). It appears that the wetland system was not as effective in removal of TP as it was with nitrogen. Similar high–TP initial treatment levels and reductions over time were reported by Kadlec and Knight (1996) and Reed et al. (1995). To accomplish more efficient removal of P in the wetland systems, pre/post treatment will likely be required.

REGRESSION ANALYSIS

Coefficients of determination (r^2) for the regression of TN outlet concentration as a function of inlet concentration and flow were approximately 0.67 and 72 for the two wetland systems, respectively (fig. 8). For comparison, we plotted our calculated regression equation with the mean hydraulic loading rate of 0.011 m/day along with those from Knight et al. (2000) (fig. 8). The results from our two wetland systems were very similar to each other. Our regression results

Table 1. Means of flow, residence time, and hydraulic loading rate for the constructed wetland systems.

	Flow (m^3/day)		Nominal Residence Time (day)		Hydraulic Loading Rate (m/day)	
System	Mean	Std.Dev.	Mean	Std.Dev.	Mean	Std.Dev.
1	2.633	1.076	12.832	6.802	0.011	0.004
2	2.960	1.398	11.160	7.125	0.012	0.006

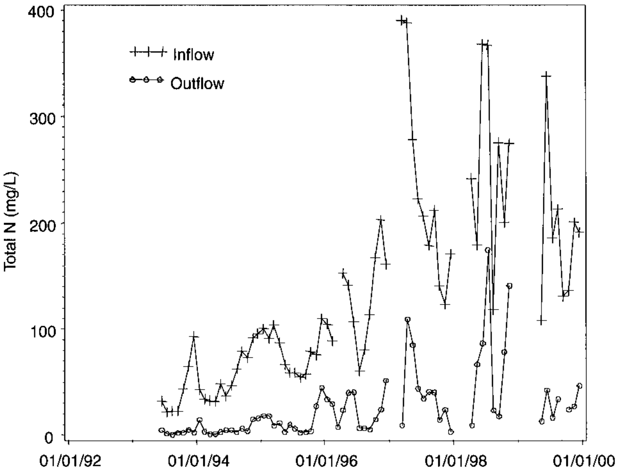

Figure 2. Total nitrogen inflow and outflow for constructed wetland system 1.

Table 2. Means of inflow, outflow, removal, and percent removal for the constructed wetland systems.

	Inflow (mg/L)		Outflow (mg/L)		Removal (mg/L)		% Reduction (mg/L)	
	Mean	Std. Dev.	Mean	Std. Dev.	Mean	Std. Dev.	Mean	Std. Dev.
System 1								
TN	134	95	26	33	109	76	84	12
NH_4-N	118	84	20	28	98	67	86	12
TP	30	21	22	15	7	13	25	32
System 2								
TN	134	95	27	36	107	72	84	13
NH_4-N	118	84	21	32	97	64	86	14
TP	30	21	20	16	10	13	38	31

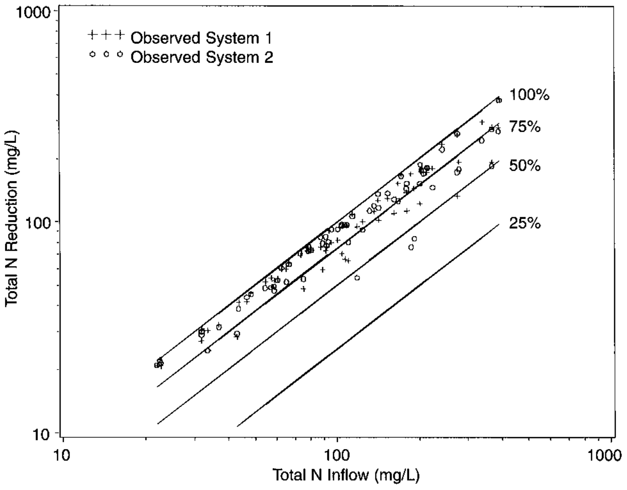

Figure 3. Total nitrogen concentration reduction for the two constructed wetland systems.

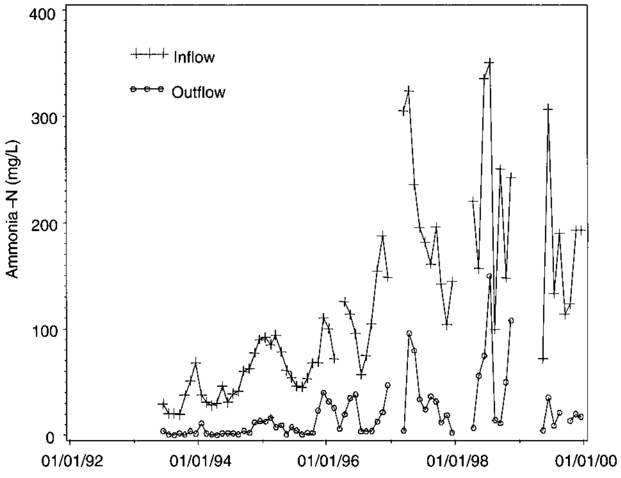

Figure 4. Ammonia–N inflow and outflow for constructed wetland system 1.

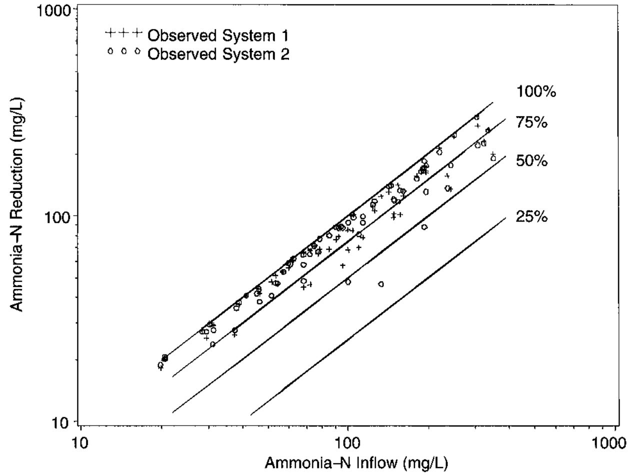

Figure 5. Ammonia–N concentration reduction for the two constructed wetland systems.

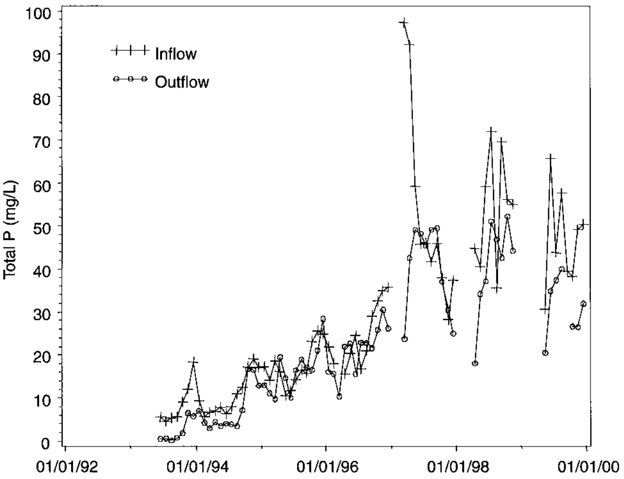

Figure 6. Total phosphorus inflow and outflow for constructed wetland system 1.

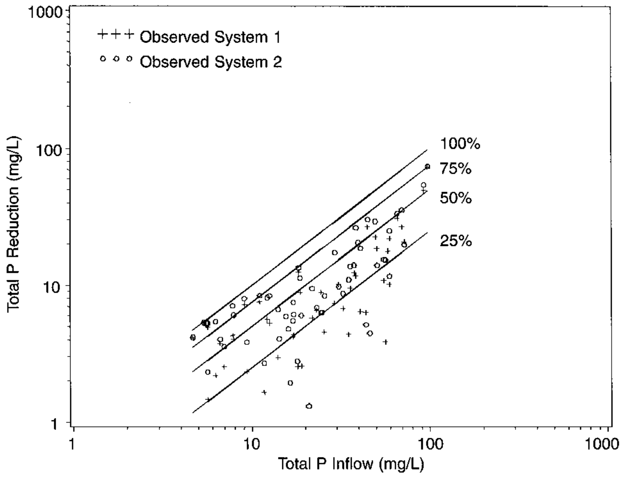

Figure 7. Total phosphorus concentration reduction for the two constructed wetland systems.

predicted more treatment over the range of loading rates than those of Knight et al. (2000).

The regression for NH_4–N had r^2 values of approximately 0.65 for both of the wetland systems (fig. 9). Our regression

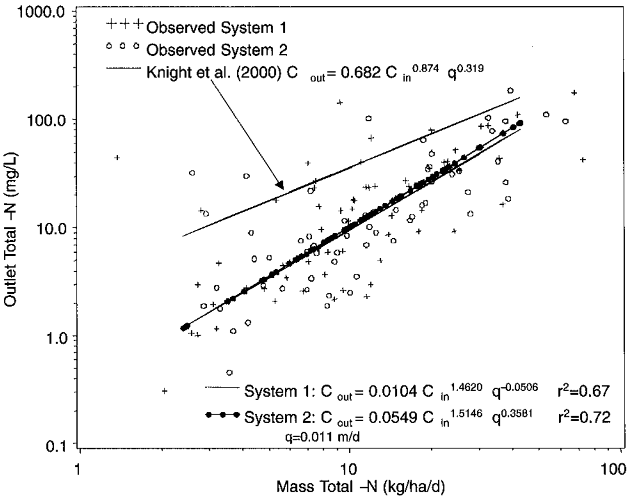

Figure 8. Relationship between total nitrogen mass loading and outlet concentration. Equations plotted with mean loading rate of q = 0.011 m/d for comparison.

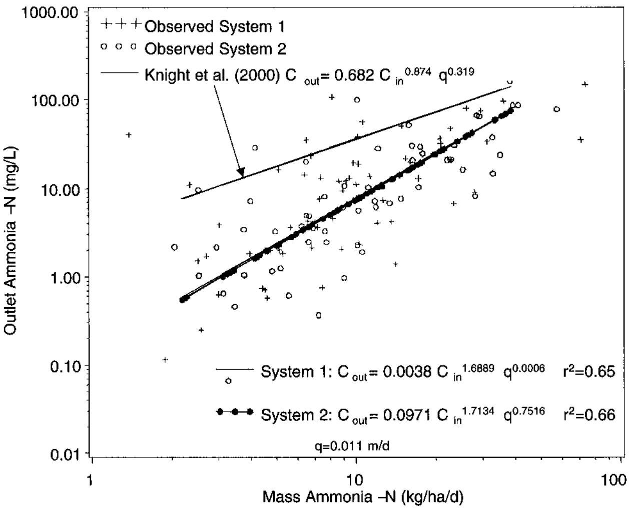

Figure 9. Relationship between ammonia–N mass loading and outlet concentration. Equations plotted with mean loading rate of q = 0.011 m/d for comparison.

results for NH_4–N were very similar to those for TN, with our two wetland systems predicting more NH_4–N treatment than Knight et al. (2000).

The TP regression analysis for the inlet concentration and hydraulic loading rate versus the outlet TP concentration had r^2 values of approximately 0.70 for both systems (fig. 10). For TP, our regression equations predicted more treatment than Knight et al. (2000) at the lower loading rates. At the higher loading rates, our regressions predicted less treatment, which is consistent with the substantial decrease in treatment efficiency during the study period as the loading rates were increased.

WETLAND DESIGN ANALYSIS

The wetland data for the entire study period were analyzed to calculate the rate constants of TN, NH_4–N, and TP for the two wetland systems. The temperature–based rate constants were calculated using equation 8 and then regressed against the temperature to determine the K_{20} rate constant and θ from equation 9. In table 3, K_{20} and θ are shown for TN and NH_4–N

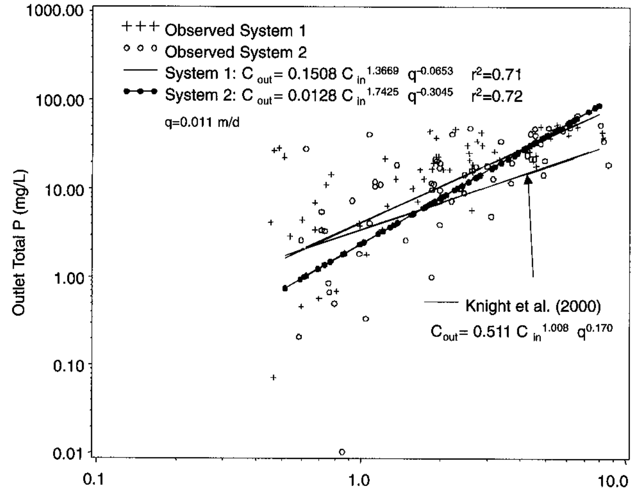

Figure 10. Relationship between total phosphorus mass loading and outlet concentration. Equations plotted with mean loading rate of q = 0.011 m/d for comparison.

Table 3. Regression parameters for the calculation of rate constants for the first–order area–based treatment design model.

	N	Intercept	K_{20} (m/d)	K_{20} (m/yr)	Slope	θ	r^2
TN							
System 1	62	–3.841	0.021	7.835	0.038	1.039	0.167
System 2	60	–3.740	0.024	8.668	0.023	1.023	0.082
NH_4–N							
System 1	62	–3.734	0.024	8.723	0.041	1.042	0.180
System 2	60	–3.615	0.027	9.824	0.022	1.022	0.071

Table 4. Summary of parameter values used in the first–order area–based uptake design model for sizing livestock wastewater treatment wetlands (Knight et al., 2000).

	Livestock Treatment Wetland Data			Kadlec and Knight (1996)		
	K_{20} (m/d)	C^* (mg/L)	θ	K_{20} (m/d)	C^* (mg/L)	θ
TN	14	10.0	1.06	22	1.50	1.05
NH_4–N	10	3.0	1.05	18	0.00	1.04
TP	8	2.0	1.05	12	0.02	1.00

for the two wetland systems studied. There was little difference among the individual constituents across the two wetland systems.

These results compare favorably but are lower than those from Reed et al. (1995), Kadlec and Knight (1996), and Knight et al. (2000) (table 4). The NRCS field test method (Payne and Knight, 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 7.4 to 8.7 m/yr and NH_4–N K_{20} values of 8.2 to 9.8 m/yr. The TN and NH_4–N K_{20} values are similar for our system because most of the TN in our system was in the NH_4–N form. Our lower values for the rate constants compared with the NRCS field test method were calculated assuming $C^* = 0$. Furthermore, 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 coefficients of determination, which suggests that the rate constants in our systems were not strongly related to temperature. This may have occurred due to the location of our systems, where the mean monthly air temperatures ranged from 4.2° C to 28.7° C.

Additionally, we simultaneously solved equations 5 and 6 for K_{20}, θ, and C^* using Solver in Microsoft Excel (table 5). These rate constants (K_{20}) and θ values are in similar agreement with our previous calculations. The calculated C^* values were in general agreement with those in literature. Our calculated K_{20} values are below those in the literature, indicating that our system did not perform as efficiently as those in the literature (Kadlec and Knight, 1996; Knight et. al., 2000; Payne and Knight, 1997; USDA, 1991; and table 4).

The rate constants for TP were calculated based on equation 8. The K_{20} values for TP ranged from 1.04 to 1.79 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 lower end of their range of values. The data from this project had a much higher loading rate for TP than many of those reported in the references. In addition, after the first year, the efficiency of the wetlands for phosphorus treatment declined dramatically. This suggests

Table 5. Rate constant (K_{20}), dimensionless temperature coefficient (θ), and background concentrations (C^*) calculated simultaneously using the Excel solver routine to minimize sum of squares between observed and predicted outflow concentrations.

	K_{20}, θ, and C^* Calculated			C^* Assumed[a]			$C^* = 0$	
	K_{20} (m/yr)	θ	C^* (mg/L)	K_{20} (m/yr)	θ	C^* (mg/L)	K_{20} (m/yr)	θ
TN								
System 1	8.85	1.02	10.99	8.71	1.02	10	7.45	1.03
System 2	8.66	0.98	5.81	9.35	0.98	10	7.86	0.99
NH_4–N								
System 1	8.98	1.03	7.73	8.6	1.03	3	7.82	1.03
System 2	9.39	0.98	4.32	9.2	0.98	3	8.62	0.99
TP								
System 1	1.79	1	14.9	1.11	1	2	1.04	1
System 2[b]	–	–	–	–	–	–	1.39	–

[a] C* assumed from Knight et al. (2000).

[b] The Excel solver routine would not converge for a solution for TP in system 2. The TP mean rate constant is reported.

that an alternative method of phosphorus removal should be investigated.

To determine the implications of the calculated design parameters on the sizing of a constructed wetland to treat swine lagoon effluent, we used our calculated parameters and those from Knight et al. (2000) to predict wetland system size by rearranging equation 8 and solving for the area, A (table 6). We assumed the mean temperature of 18.5° C. The original wetland consisted of two wetland cells of 3.6 × 33.5 m each, for a total area of 241.2 m^2. Mean C_{in}, C_{out}, and Q_{in} values from tables 1 and 2 were used for input concentration and flow parameters. The K_{20}, θ, and C^* parameters were mean values for the two wetland systems from table 3 for the regression analysis in SAS and from table 5 for the Excel solver analysis. Using the estimated parameters from Knight et al. (2000), the calculated wetland size would be ~157 m^2 for TN and 203 m^2 for NH_4–N.

Using our calculated parameters, the wetland size ranged from 30% to 42% larger for TN (205 to 224 m^2), and from –2% to 11% larger for NH_4–N (198 to 225 m^2). Our calculated wetland size was ~10% smaller than the original constructed wetland. For NH_4–N, both our rate constants and those from Knight et al. (2000) were in close agreement, and the resulting wetland sizes were also in close agreement. However, for TN, our rate constants were less than those from Knight et al. (2000). One reason for the large difference could be that Knight et al. (2000) estimated the TN rate constant (K_{20} = 14 m/yr) from three sites (1 poultry, 2 swine) with widely ranging rate constants (K_{20} = 5 to 32 m/yr), while our TN rate constant (average K_{20} = 8.4) was near the lower end of their sampling range. Additionally, their calculated rate constants were from summaries of the early years of only three wetlands sites, while our results are from seven years of intensive monitoring at this experimental site. In addition, in our system, our TN and NH_4–N K_{20} values were similar because the TN consisted primarily of NH_4–N.

CONCLUSIONS

Constructed wetlands at a North Carolina swine farm were evaluated for treatment of swine lagoon effluent. Overall, these constructed wetlands were very effective in treating nitrogen from swine lagoon wastewater based on its initial presumptive design by NRCS. Mean total nitrogen concentration reduction efficiency was 84%. The mean NH_4–N concentration reduction efficiency was 86%.

The constructed wetlands removed only small amounts of phosphorus. At low loading rates, initial TP concentration reduction efficiency was ~88%. However, at the higher loading rates, concentration reduction efficiencies decreased. The overall mean TP concentration reduction efficiency was 25% and 38% for the two constructed wetland systems. To accomplish more efficient removal of P in the wetland systems, pre/post treatment may be required.

The calculated regression equations to predict outflow concentration from inflow concentration and hydraulic loading rate were in general agreement with those in the literature. These equations should be appropriate for use in estimating wetland treatment of nutrients within the observed range of nutrient loading rates.

Rate constants for the first–order rate equation (K–C^* model) developed by Kadlec and Knight (1996) were determined for nutrient treatment (TN, NH_4–N, and TP) in two constructed wetlands treating swine lagoon effluent in eastern North Carolina. The calculated rate constants were generally similar to or slightly lower than those reported in the limited literature. Use of our calculated rate constants and parameters would result in a slightly more conservative design. Based on our calculated rate constants, a newly constructed wetland with mean loading rates and concentrations similar to our system would result in a wetland slightly larger (~5%) based on NH_4–N compared to those based on the currently available guidelines. This is very important since TN and, in most cases, BOD are dominated by NH_4–N in swine lagoon wastewater.

Table 6. Calculated wetland areas using calculated parameters[a] and parameters from Knight et al. (2000).

	C_{in} (mg/L)	C_{out} (mg/L)	Q_{in} (m^3/day)	K_{20} (m/yr)	θ	C^* (mg/L)	Area (m^2)	Difference (%)	Mean Difference (%)
TN									
Knight et al. (2000)[b]	134	26	2.7	14.0	1.060	10.0	157.3	–	
Regression	134	26	2.7	8.3	1.031	0.0	205.0	30	
Excel	134	26	2.7	8.8	1.000	8.4	221.2	41	
Excel, C^* = 10	134	26	2.7	9.0	1.000	10.0	223.5	42	
Excel, C^* = 0	134	26	2.7	7.7	1.010	0.0	214.3	36	
									37%
NH_4–N									
Knight et al. (2000)[b]	118	20	2.7	10.0	1.050	3.0	202.7	–	
Regression	118	20	2.7	9.3	1.032	0.0	197.8	–2	
Excel	118	20	2.7	9.2	1.005	6.0	225.0	11	
Excel, C^* = 3	118	20	2.7	8.9	1.005	3.0	213.3	5	
Excel, C^* = 0	118	20	2.7	8.2	1.010	0.0	218.0	7	
									5%

[a] Mean C_{in}, C_{out}, and Q_{in} values from tables 1 and 2 were used for input concentration and flow parameters. The K_{20}, θ, and C^* parameters were mean values for the two wetland systems from table 3 for the regression analysis in SAS and from table 5 for the Excel solver analysis.

[b] Knight et al. (2000) rate constants for TN were calculated from three sites (1 poultry, 2 swine) and ranged from 5 to 32 m/yr. Rate constants for NH_4–N were calculated from five sites and ranged from –1 to 26 m/yr.

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