IN–STREAM WETLAND DESIGN FOR NON–POINT SOURCE POLLUTION ABATEMENT

K. C. Stone, P. G. Hunt, J. M. Novak, M. H. Johnson

ABSTRACT. *Nonpoint source pollution (NPS) of rivers and streams is a major concern worldwide. Most methods for NPS mitigation focus on source reductions; few have been developed to mitigate NPS once nutrients have entered streams. One system that has been shown to be effective in reducing stream nitrogen is by using in–stream wetlands (ISW). The objective of this research was to determine if design approaches used in constructed wetlands could be applied to predict ISW effectiveness in treating NPS. The 3.3–ha ISW studied was located in a 425–ha watershed in eastern North Carolina. We analyzed the data from the ISW to calculate the first–order rate constants (K_{20} and dimensionless temperature coefficient, θ) for the k–C* model used in constructed wetland design. We found that our calculated rate constants were in close agreement with literature estimates with TN K_{20} = 19–20 m/y and θ= 1.0 – 1.03. NO_3–N rate constants were K_{20} = 38 – 54 m/y and θ = 1.07 – 1.13. The design equations used for constructed wetlands can be successfully used to predict the performance of the ISW's prior to their implementation.*

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

Nonpoint source (NPS) pollution of stream and rivers is a major concern throughout the United States and the world. Baker (1992) reported that NPS pollution was the major cause of impairment of U.S. surface waters. Additionally, he reported that the dominate source of NPS pollution was from agricultural activities and that nutrients, sediments, and pathogens were the main detrimental constituents. He suggested that nitrogen was of particular concern in surface waters because it can promote eutrophication of streams and estuaries. Eutrophication occurs when excessive nutrients are present in the water body resulting in algal blooms that reduce light and oxygen for aquatic life.

NPS pollution from agriculture may occur when nutrients are applied at rates greater than crops can utilize or when timing of nutrient applications occurs in close proximity to heavy rains. In the eastern Coastal Plain, nutrient leaching to ground water is a potential problem because of high rainfall, sandy textures, and low soil organic matter levels. Nutrients can also reach streams by overland flow or by lateral movement of shallow ground water (Novak et al., 2002). Nutrient leaching and runoff are particularly problematic in the eastern Coastal Plain because of the large amounts of swine and poultry waste being produced and applied to crops (Stone et al., 1995, 1998).

Natural landscape characteristics of eastern Coastal Plain watersheds, such as large wooded riparian zones and soils with high organic matter, typically have helped prevent elevated nutrient levels from reaching streams and shallow ground water (Gilliam, 1991). However, with the large influx of animal production and limited land for waste application, these natural features can become overloaded and their effectiveness negated.

Once elevated nutrients reach streams, it becomes difficult to mitigate their impact on the aquatic environment. Haggard et al. (2001) found very little nitrate reduction in a stream in Oklahoma and Northwest Arkansas. Jansson et al. (1994) found that a small stream in Sweden reduced nitrogen by less than 3%, however he found that using a retention pond in the stream reduced nitrogen up to 50%. In the U.S. eastern Coastal Plain, Hunt et al. (1999) found that an in–stream wetland could annually reduce the nitrogen levels in a contaminated stream by approximately 50%.

Wetlands have long been recognized as active systems for transforming and treating nutrients. Wetlands occur naturally in the environment in streams and riparian areas adjacent to streams. Wetlands have been constructed in upland environments for treating wastewater (Kadlec and Knight, 1996).

The utilization of a wetland to treat NPS pollution would provide an additional method for mitigation of the impact of excess nutrients in streams. The in–stream wetland described in this study was located at the outlet of an eastern Coastal Plain watershed located in North Carolina. The overall impacts and nutrient reductions of the in–stream wetland were reported by Hunt et al. (1999). The objective of this work was to determine if the functionality of an in–stream wetland could be described using design approaches from constructed wetland design.

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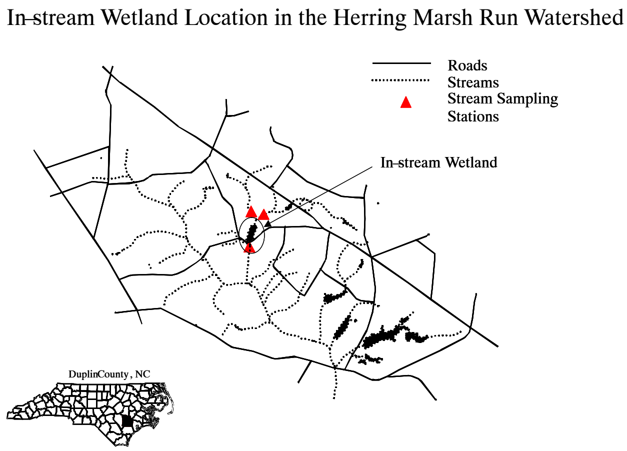

Figure 1. Herrings Marsh Run watershed located in Duplin County, N.C.

Methods

Site Description and Operation

In the Herrings Marsh Run watershed (fig. 1) in Duplin County, North Carolina, a 425–ha sub–watershed had been overloaded with nitrogen (N), and the stream draining this sub–watershed contained excessive N (Stone et al., 1995). At the stream exit, there was a small wetland landscape area. Hunt et al. (1999) hypothesized that enhancement and repair of a breached dam at the wetland area would create an in–stream wetland (ISW) that would improve stream water quality by lowering the nitrate–N concentration. Prior to replacing the breached dam, beavers began constructing their own dam at the ISW outlet. We reinforced the beaver dam to prevent the side walls from eroding and to direct water over the center of the dam. The ISW impounded approximately 3.3 ha, and it ranged in depth from about 0.2 to 2 m (fig. 2). Emergent aquatic weeds occupied approximately 40% of its surface area. The ISW perimeter was dominated by trees [swamp tupelo (*Nyssa biflora*), red maple (*Acer rubrum*), and black willow (*Salix nigra*)].

The soil series surrounding the ISW in upland locations is predominately an Autryville fine sand. The soil within the ISW is a Bibb loam. The soils in the Coastal Plain sediments are sandy to clayey unconsolidated marine and fluvial deposits (Daniels et al., 1999). The geomorphic surfaces in the study area are Pliocene to early Pleistocene ranging from ~5 to 0.5 million years old (Daniels et al., 1978).

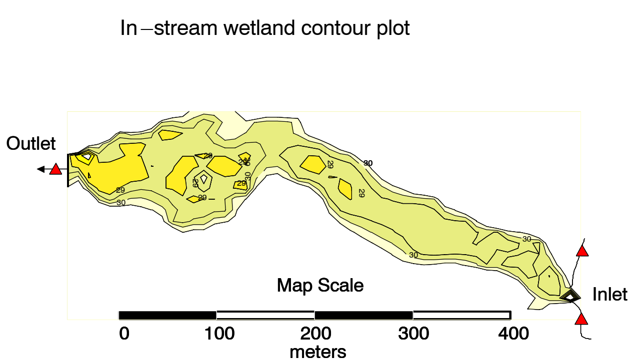

Figure 2. In–stream wetland contour plot (meters above sea level).

Water samples for chemical analyses as well as flow measurements had been obtained since 1990 as part of the USDA–Water Quality Demonstration Project on the entire Herrings Marsh Run. Water samples at both the ISW inlet and ISW outlet were collected using automated samplers at timed intervals. The U.S. Geological Survey measured the flow at the ISW outlet. In October 1993, an automated water sampler was installed at the ISW inlet. Flow at the inlet was manually measured using a current meter (Scientific Instruments Model 1205 Price–type current meter, Milwaukee, Wis.). In the winter of 1994, beavers raised the height of their dam, and the original inlet sampling station was flooded. Two new sampling stations were established approximately 100 m upstream on two small streams entering the expanded ISW. Nitrate and ammonia values were obtained at the two new inlet streams using weekly grab samples from April 1994 until March 1995, when automated stream samplers were installed on the two inlets. Concentration values from the two inlet streams were multiplied by their corresponding flows, during sample collection interval, and summed to calculate ISW nutrient loading. All stream samples collected with the automated samplers were collected at 4–h intervals and combined into 3.5–day composites. Mass loading rates were obtained by multiplying the nutrient concentration with the average flow rate for the samples.

Stream water samples were analyzed for Total Kjeldahl Nitrogen (TKN), Ammonia–Nitrogen (NH_4–N), and Nitrate–Nitrogen (NO_3–N) using EPA methods (U.S. EPA, 1983). All samples were analyzed using automated analyzers (Technicon Instruments Corp., Tarrytown, N.Y. and Bran Lubbe Corporation, Buffalo Grove, Ill.). Total Nitrogen (TN) was calculated as the sum of TKN and NO_3–N.

Background Data on ISW Performance

The in–stream wetland was very effective in reducing the TN and NO_3–N concentrations in the subwatershed (Hunt et al., 1999). The TN concentration reduction through the ISW was 56% with a mean inflow concentration of 7.9 mg/L and an outflow concentration of 3.6 mg/L (table 1 and fig. 3). The mean TN mass removal during the study period was approximately 2.9 kg/ha/d. The predominate form of nitrogen entering the ISW was NO_3–N. The NO_3–N concentration reduction through the wetland was 71% with inflow and outflow mean concentrations of 6.6 and 2.0 mg/L, respectively (fig. 4). The mean NO_3–N mass removal by the ISW was approximately 3.2 kg/ha/d. Hunt et al. (1999) reported that the large reduction of NO_3–N in the ISW was related to denitrification, microbial assimilation, and plant uptake, particularly during the warmer months. Conversely, the NH_4–N mean concentration increased through the ISW by ~40%, from an inflow concentration of 0.5 to 0.7 mg/L at the ISW outlet (fig. 5). The mean NH_4–N mass reductions by the ISW were approximately –0.24 kg/ha/d. Hunt et al. (1999) found that during the cooler months (December–March) NH_4–N was removed by the ISW and that during the warmer months, NH_4–N actually increased which resulted in the mean annual increase in NH_4–N concentration and mass loading from the ISW.

The mean flow from the ISW for the study period was 0.03 m^3/d, which corresponded to a hydraulic loading rate of 0.09 m/d and a residence time in the ISW of approximately 23 days (table 2).

Table 1. Mean TN, NO_3–N, and NH_4–N concentrations, and removals for the in–stream wetland.

	Inflow (mg/L)		Outflow (mg/L)		Removal (mg/L)		
	mean	std	mean	std	mean	std	% Reduction
TN	7.9	1.4	3.6	2.0	4.4	2.0	56
NO_3–N	6.6	1.3	2.0	1.9	4.7	2.1	71
NH_4–N	0.5	0.3	0.7	0.5	–0.2	0.6	–40

Regression Analysis

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

$$C_{out} = aC_{in}{}^{b}q^{c} \tag{1}$$

where

C_{out} and C_{in} = the outlet and inlet nutrient concentrations (mg/L), respectively,
q = hydraulic loading rate (m/d),
a,b,c = regression coefficients.

Equation 1 was transformed in order to perform the regression in the SAS system with the Proc Reg procedure and was analyzed as:

$$\ln(C_{out}) = \ln(a) + b\ \ln(C_{in}) + c\ \ln(q) \tag{2}$$

Design of surface flow wetlands for municipal and animal waste treatment was presented by 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 a 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 treatment wetlands is generally intermediate between plug flow and completely mixed, the use of the first–order model with plug flow assumptions provide a conservative design estimate (Knight et al., 2000).

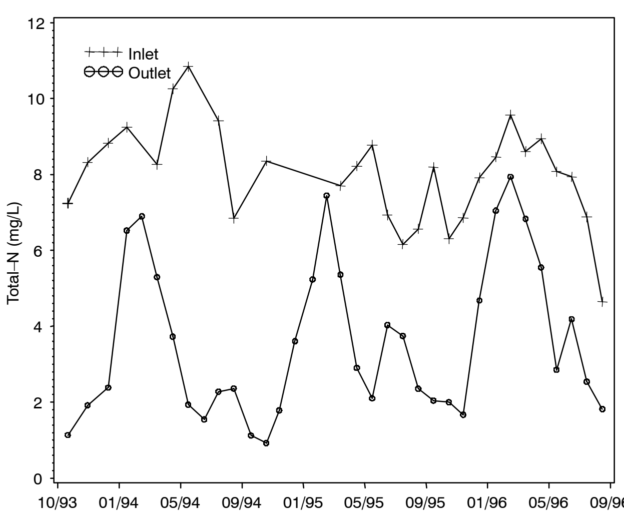

Figure 3. Mean monthly Total–N inflow and outflow for the in–stream wetland.

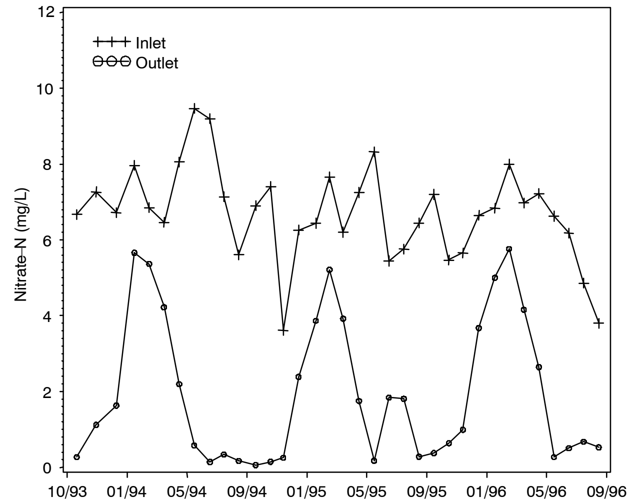

Figure 4. Mean monthly Nitrate–N inflow and outflow for the in–stream wetland.

Kadlec and Knight (1996) presented the area–based first–order plug flow design model as:

$$\left[\frac{C_{out} - C*}{C_{in} - C*}\right] = \exp\left(-\frac{K_T}{q}\right) \tag{3}$$

where

C_* = background concentration (mg/L),
K_T = rate constant adjusted for temperature (m/d).

$$K_T = K_{20}\theta^{(T-20)} \tag{4}$$

K_{20} = rate constant at 20°C (m/d),
θ = dimensionless temperature coefficient,
T = temperature (°C).

The hydraulic loading rate (q) is defined as

$$q = \frac{Q_{in}}{A} \tag{5}$$

where

Q_{in} = inflow (m^3/d), and
A = wetland surface area (m^2).

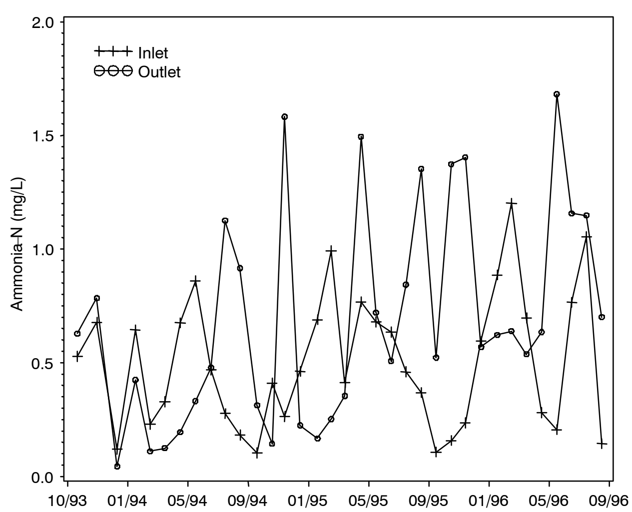

Figure 5. Mean monthly Ammonia–N inflow and outflow for the in–stream wetland.

Table 2. Mean flow, residence time, and hydraulic loading rate for the in–stream wetland.

Flow (m^3/d)		Residence Time (d)		Hydraulic Loading Rate (m/d)	
mean	std	mean	std	mean	std
0.03	0.02	23.42	36.06	0.09	0.06

The temperature–related rate constant for TN and NH_4–N from the wetland data was calculated rearranging equation 3 as:

$$K_T = \frac{Q}{A}\ln\left[\frac{C_{in} - C_*}{C_{out} - C_*}\right] \quad (6)$$

Equation 4 was then rearranged in order to calculate the K_{20} rate constant at 20°C and the dimensionless temperature coefficient.

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

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

In addition to solving for rate constants with regression analysis in SAS (1990), we used a spreadsheet function (Solver in Microsoft Excel) to simultaneously solve equations 3 and 4 for K_{20}, θ, and C_*. This simultaneous solution method minimizes the sum of squares between the measured and predicted outflow nutrient concentrations (R. H. Kadlec, 2000, personal communication).

Results

Regression Analysis

Regressions for outlet TN and NO_3–N concentration as a function of the inflow mass loading rate (inflow and inlet concentration) were calculated and shown in figures 6 and 7. Coefficients of determination for the two constituents were low < 0.5, which indicates that site–specific factors influencing the treatment effectiveness of the wetland were not included in the regressions. The coefficient of determination for the regression of TN outlet concentration as a function of inlet concentration and inflow was approximately 0.40. The NO_3–N removal through the ISW was greater than TN, but

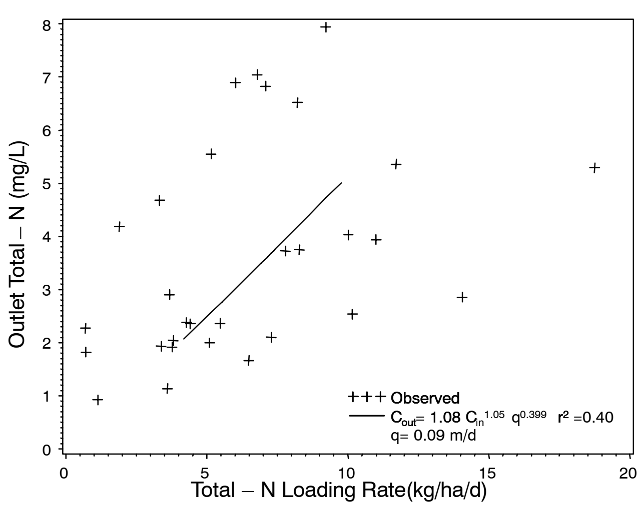

Figure 6. Relationship between Total–N mass loading and outlet concentration. Equations plotted with mean loading rate of q = 0.09 m/d for comparison.

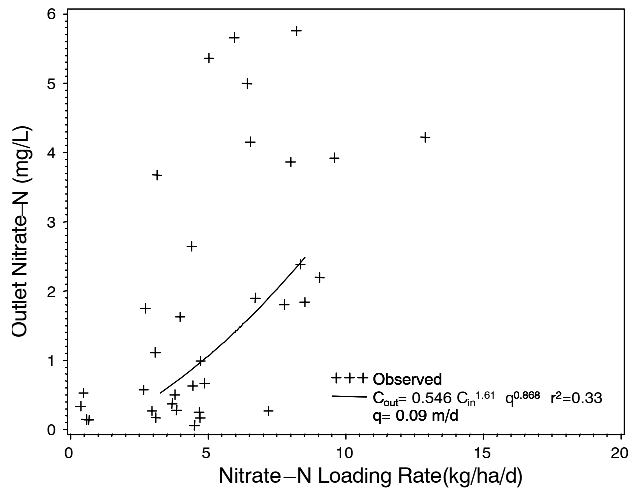

Figure 7. Relationship between Nitrate–N mass loading and outlet concentration. Equations plotted with mean loading rate of q = 0.09 m/d for comparison.

the regression for the NO_3–N had a coefficient of determination of 0.32. We also evaluated NH_4–N regression and found very poor results ($r^2 < 0.1$) using data from the entire study period; however, using only the cold season data, the NH_4–N correlation of determination was 0.8.

These regressions and corresponding coefficients of determinations (< 0.50) were similar to the results reported by Kadlec and Knight (1996). They reported regression results from 30 surface flow wetlands in the North American Wetland Treatment System Database (Knight, 1994). They also suggested two causes for the low coefficients of determination. One, the low regressions were probably caused by the lack of site–specific data not considered by the regression. Two, there is a strong sequential interrelation among the nitrogen species that needs to be included as precursor species as influences on outlet concentrations.

ISW Design Analysis

The ISW was evaluated to calculate first–order rate constants for TN and NO_3–N for the entire study period. We assumed the background concentrations of the constituents for the ISW to be zero ($C_* = 0$). Equation 8 was used to calculate K_T values, and we then regressed the K_T rate constants against the mean monthly temperatures to determine the K_{20} rate constants and θ values from equation 9. In table 3, K_{20} and θ values for the regression are shown along with their low coefficients of determination. The regressions have low coefficients of determination, indicating that there was a poor relationship between the K_T and mean monthly temperatures. The TN regression was not significant and produced a K_{20} value of 18.456 m/y along with θ = 1.035. An average K_T for TN was calculated as 21.5 m/y. An Excel Solver analysis of the system evaluating TN rate constants determined K_{20} and θ values of 18.92 and 1.0, respectively. These rate constants for TN are in close agreement with those constants from the literature. Kadlec and Knight (1996) reported a K_{20} = 22 m/y and and θ = 1.05 for surface water treatment wetlands. Stone et al. (2002) reported K_{20} = ~8 m/y for a swine waste treatment wetland and Knight et al. (2000) reported a K_{20} = 14 m/y for design of constructed wetlands for animal wastewater treatment systems.

Table 3. 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/y)	Slope	θ	r^2
Total–N	27	–2.985	0.051	18.456	0.034	1.035	0.065
Nitrate–N	34	–2.276	0.103	37.493	0.065	1.067	0.175
NH_4–N	14	–3.131	0.044	15.945	–0.002	0.998	0.00

The regression analysis for NO_3–N rate constants produced a higher coefficient of determination ($r^2 = 0.18$) and a significant model statistic. The low r^2 value would indicate that the influence of temperature on the model was minimal. The regression calculated K_{20} and θ values were 37.5 m/y and 1.07, respectively. The average K_T for NO_3–N was 44 m/y, while the Excel Solver simultaneous solution calculated the K_{20} and θ values of 54 m/y and 1.13, respectively. Kadlec and Knight (1996) reported NO_3–N rate constants and θ values of 35 m/y and 1.09, respectively. Using the θ = 1.09 value from Kadlec and Knight (1996), the Excel Solver calculated a K_{20} value of 44 m/y. The calculated NO_3–N rate constants were also similar to those reported by Reed et al. (1995).

A calculated NH_4–N rate constant for the entire study period was not possible because of the large number of points when the outlet concentration exceeded the inlet concentrations. Hunt et al. (1999) estimated that ammonia–N increases from the ISW inlet to outlet during the warmer months were likely related to detrital mineralization under low oxygen conditions that limited the conversion of ammonia to nitrate in the ISW. During the cooler months (December–March), we were able to calculate NH_4–N rate constants for the ISW. The regression calculations produced K_{20} and θ values of 15.9 m/y and 0.998, respectively. These values were in close agreement with Kadlec and Knight (1996) with K_{20} = 18 m/y and θ = 1.04. The Excel solver solution for the rate constants produced a K_{20} = 18.4 m/y and θ = 1.0. The average K_T value for NH_4–N was approximately 17 m/y. Knight et al. (2000) reported K_{20} values of 10 m/y for animal treatment wetlands that received much higher inlet concentrations than our ISW and natural wetlands would receive. Stone et al. (2002) reported K_{20} values and approximately 8 m/y for a swine lagoon wastewater treatment wetland.

Conclusions

1. A regression analysis of TN and NO_3–N inlet concentrations and flow against outlet concentrations for the ISW produced low coefficients of determination probably due to the lack of site–specific data not considered by the regression.
2. The rate constants for the first–order rate equation (K–C* model) developed by Kadlec and Knight (1996) were determined for TN, NO_3–N, and NH_4–N in the ISW. The calculated rate constants were generally similar to or slightly higher than those reported in the limited literature.
3. The design equations used for constructed wetlands in similar hydrologic settings could be reasonably utilized to predict the performance of the ISW's prior to their implementation.

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