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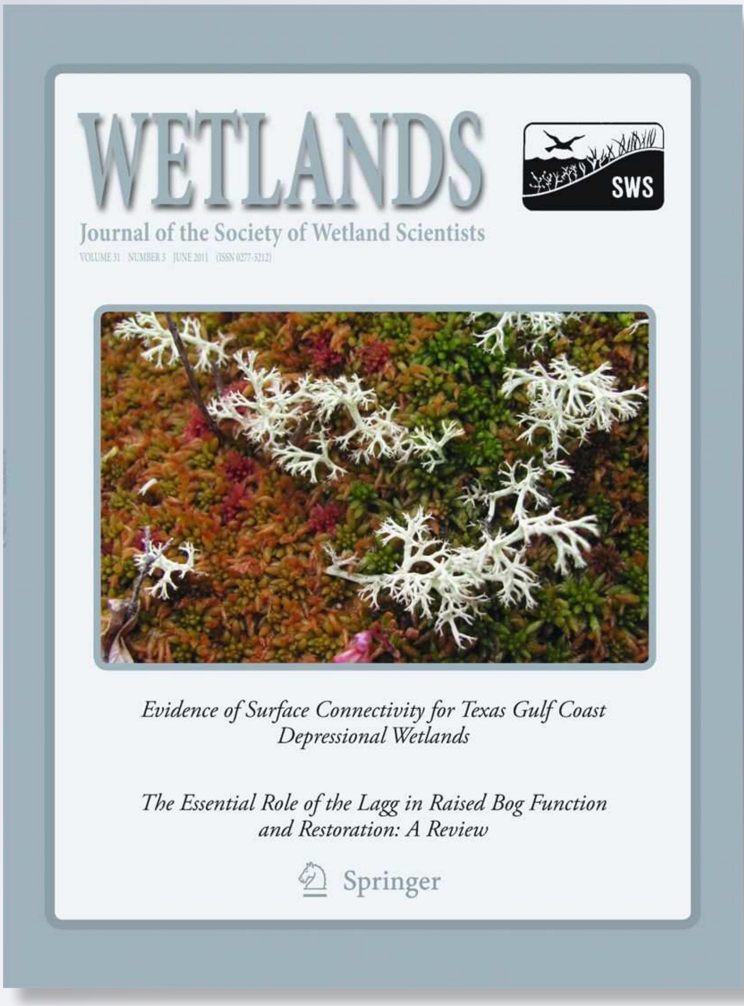

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ARTICLE

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Abstract A restored drained wetland should provide a comparable environment to remove nonpoint source nitrogen (N) from animal feeding operation (AFO) runoff as an equivalent constructed wetland. Our research addresses the N removal efficiency of the drained and restored Long Acres Wetland (LAW) to its modeled constructed wetland equivalents, and the relationship of the five denitrifying variables of bacteria population size, bacteria-pollutant contact, N Cycle continuity, kinetics, and bacteria-pollutant contact time to rate their relative influence on N removal. Although, we found restoration improved percent total nitrogen removed (%TNR) from 90.9% to 94.6%, analysis of variance (ANOVA) of the actual drained versus actual restored %TNR datasets indicated no statistical difference. However, ANOVA of the actual restored versus modeled restored %TNR datasets showed a significant statistical difference, indicating that the actual restored LAW was not operating at its theoretical optimal efficiency. Subsequent ANOVA of the denitrifying variables' pre- versus post-restoration datasets indicated that restoration had favorably influenced all denitrifying variables except bacteria-pollutant contact. Our results suggest lack of mixing was the limiting factor negating all other enhanced variables potential treatment contributions. Nevertheless, the actual drained LAW significantly removed N, indicating that drained wetlands maybe removing considerable N from polluted runoff.

Keywords Animal feeding operation · Best management practice · Denitrification · Treatment Wetlands · Wetland restoration

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Introduction

A potential water quality best management practice (BMP) is the utilization of hydraulically restored drained wetland basins to treat agricultural nonpoint source (NPS) nitrogen (N) polluted runoff via denitrification (Mitsch 1992; Schaafsma et al. 2000; Burns and Nguyen 2002; Stone et al. 2003; Szogi et al. 2003). Organic N, the dominant form of N in animal feeding operation (AFO) runoff (USEPA 1998; Gerke et al. 2001; Follett and Delgado 2002) is readily converted to NH_3, NH_4^+, and NO_3^- in downstream surface waters (Kadlec and Knight 1996) degrading water quality. A restored drained wetland basin should provide a chemical, biological, and physical environment, as an equivalent constructed wetland basin, for bacteria to denitrify agricultural NPS N pollutants into various benign N gases. Thus, removing N from polluted surface waters, and releasing it into the atmosphere, in a one-way nonreversible process (Mitsch 1992; Davidisson et al. 2000; Andreassi and Banerjee 2003; Poach et al. 2003).

Restoration of a drained wetland basin directly influences the hydraulic restoration variables of water depth, area, and volume (Fig. 1) (Reddy and Patrick 1984; Koreny et al.

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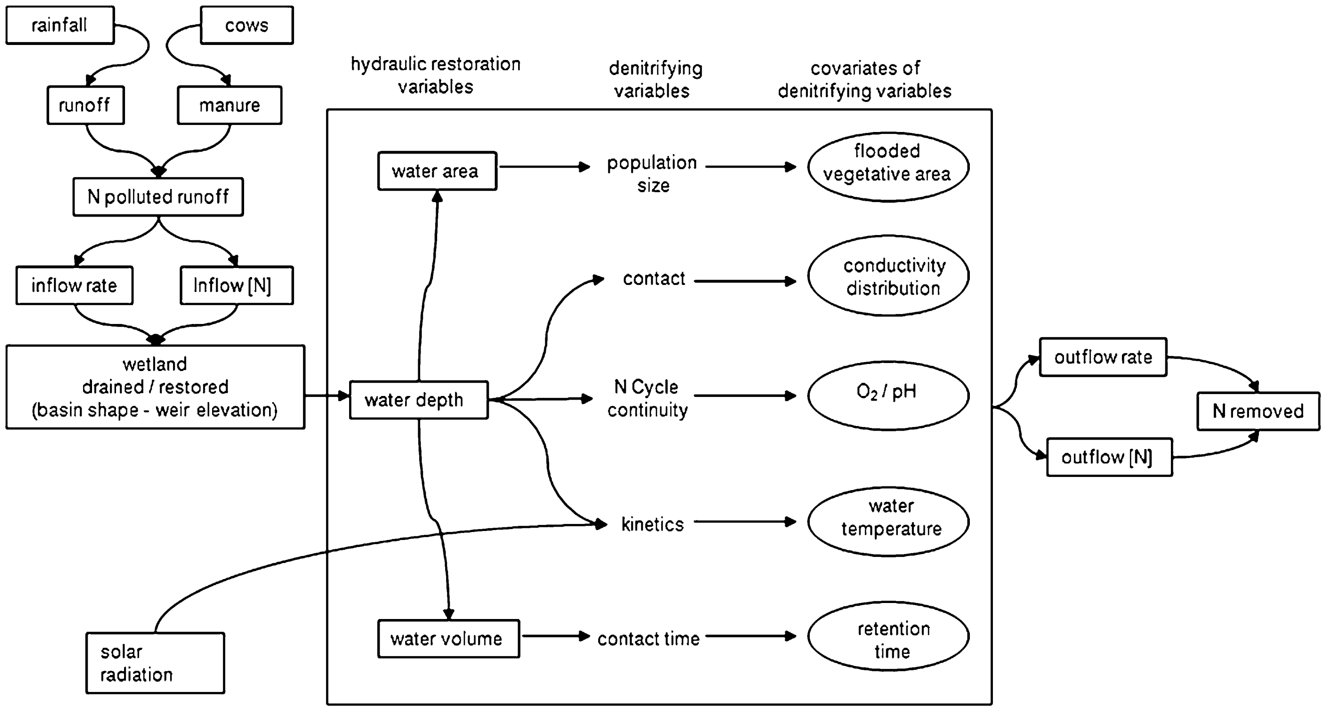


Fig. 1 Restored wetland treatment best management practice: the chemical, biological, and physical variables of a drained/restored wetland interact to remove nitrogen (N) from animal feedlot operation runoff

1999). Modifications to these variables in turn directly influences the five denitrifying variables of bacteria population size, bacteria-pollutant contact, N Cycle continuity, kinetics, and bacteria-pollutant contact time (Kadlec and Knight 1996; Bachand and Horne 2000; Bendoricchio et al. 2000; Tsihrintzis and Madiedo 2000). Thus, a restored drained wetland's N removal performance is dependent on the optimization of these variables. These five variables, in turn, can be quantified by their field measurable covariates of flooded vegetative area, conductivity spatial distribution, $[O_2]$ and pH, water temperature, and retention time respectively.

Bacteria Population Size—Flooded Vegetative Area

Vegetation plays an essential role in denitrification by creating suitable micro-environments for bacteria to metabolize N (Bavor et al. 1995; Bendoricchio et al. 2000; Luckeydoo et al. 2002). As much as 90% of the N removal in constructed wetlands can be attributed to these wetted vegetative surfaces (Moshiri 1993; Bendoricchio et al. 2000). Hence, bacteria population size is directly proportional to the flooded area of the wetland inhabited by vegetation (Hammer 1992; Kadlec and Knight 1996; Knight et al. 2000). In constructed wetlands the water depth and surface area, and vegetation are matched to site conditions to maximize wetted vegetative area.

Bacteria-Pollutant Contact—Conductivity Spatial Distribution

Bacteria must be in contact with the N pollutants to biotransform them (D'Amore et al. 2000; Werner and Kadlec 2000). Since most bacteria are sessile, attached to vegetation and litter, unrestrained advective flows are essential to transport N pollutants to them (Jordan et al. 2003). Dispersion in wetlands is typically measured by comparing the concentration of total dissolved solids (as measured by conductivity (μS/cm)) at spatially referenced surface sample sites. In constructed wetlands initial lateral spreading of inflow runoff, uniform water depth, and a graded bottom surface assures maximum dispersion.

N Cycle Continuity—Oxygen and Hydrogen Concentrations

N Cycle continuity refers to the sequencing of ammonification, followed by nitrification, and finally denitrification (Reddy and Patrick 1984; Mitsch and Gosselink 1993). Ammoniafying and nitrifying bacteria require aerobic conditions, while denitrifying bacteria require anaerobic conditions (Szogi et al. 2003). If one of these conditions is eliminated, the continuity of the N Cycle will be broken and desired N removal will cease (Reddy and Patrick 1984; Fedler et al. 2002). Constructed wetlands are typically stratified vertically into aerobic surface and anaerobic sediment zones (Burns and Nguyen 2002). This stratification, thus promotes the growth of both nitrifying and denitrifying bacteria in the same water column (Reddy and Patrick 1984; Mosier et al. 2002).

The pH can also directly influence N Cycle continuity by increasing [NH_3] promoting NH_3 removal from the wetland through atmospheric loss or volatilization, which in turn increases the ammonification rate (Tanner et al. 1999; Follett and Delgado 2002). In a wetland solution, NH_4^- enters into an equilibrium reaction with NH_3 as shown in the following equation $NH_4^- = NH_3 + H^+$, with the ionized form predominant and the un-ionized subject to gaseous loss. The fraction of solution in the NH_3 form increases by an order of magnitude for every unit of pH above 6.0 units, thus increasing the rate loss of NH_3 to the atmosphere (Tanner et al. 1999). In a constructed wetland, the optimal pH range is maintained at a circum-neutral pH by a variety of water management techniques and/or addition of various additives (Reddy and Patrick 1984).

Kinetics—Water Temperature

The bacteria mediated N Cycle processes of ammoniafication, nitrification, and denitrification have all been shown to be water temperature dependant (Jordan et al. 2003). In constructed wetlands water depth is maintained between 30 and 40 cm to simultaneously promote [O_2] stratification and maximize thermal transfer to the bottom sediments (Gambrell and Patrick 1978; Ibekwe et al. 2003). Thus, a typical constructed wetland strongly mimics the air temperature swings with only a slight delay in timing due to the thermal inertia of the surface soils and vegetation.

Bacteria-Pollutant Contact Time—Retention Time

The longer a bacteria cell is in contact with an N pollutant, the greater its probability to trap and transform it (Jordan et al. 2003). Retention time in constructed wetlands is calculated by dividing storage volume by inflow. Inflow runoff is a function of rainfall interacting with the wetland watershed variables of area, slope, vegetative cover, soils, and land use; while water storage is a function of the constructed wetland basin shape, vegetative porosity, and outlet water control structures. One method of calculating constructed wetland retention time is the hydraulic nominal detention time Eq. 1 (Novotny and Olem 1994; Bendoricchio et al. 2000):

$$t = (\varepsilon A h)/Q \tag{1}$$

where t is nominal detention time (d), ε is water volume fraction in the water column (m^3/m^3), A is wetland area (m^2), h is mean water depth (m), and Q is water inflow rate (m^3/d). In constructed wetlands, retention time is matched to the pollutant requiring the longest time to be metabolized. To meet this minimal time criteria, inflow may be regulated by construction of upstream detention basins and/or diversion of non-polluted runoff or the basin is excavated to meet the desired storage volume.

The objectives of our case study were: 1) evaluate restoration's influence on total nitrogen ((TN) N species of NH_4^+, NH_3, NO_2^-, and NO_3^-) removal by comparing actual drained versus actual restored percent total nitrogen removed ((%TNR) normalized total nitrogen [TN] datasets), 2) evaluate operational treatment efficiency by comparing actual drained and actual restored %TNR datasets to modeled drained and modeled-restored %TNR datasets, 3) evaluate restoration's influence on each denitrifying influencing variable by comparing the drained versus restored datasets of their associated covariates, and 4) evaluate each variable's influence on TN removal by comparing their associated covariate's drained and restored datasets to actual drained and actual restored %TNR datasets.

Methods

Study Site

The site of our research was the Long Acres Wetland (LAW). The LAW is located in the northeast corner of Chenango County, New York, which is within the Susquehanna River-Chesapeake Bay Watershed (Fig. 2). In the 1950s an open ditch was excavated from the Unadilla River into the LAW (i.e., wetland-2), to improve the drainage of adjacent cropland fields. The ditch lowered the wetland-2 basin's normal water level (NWL) elevation by ~0.45 m, which not only reduced its hydroperiod and retention time, but increased its hydraulic gradient and exposed its rough-hummocky bottom.

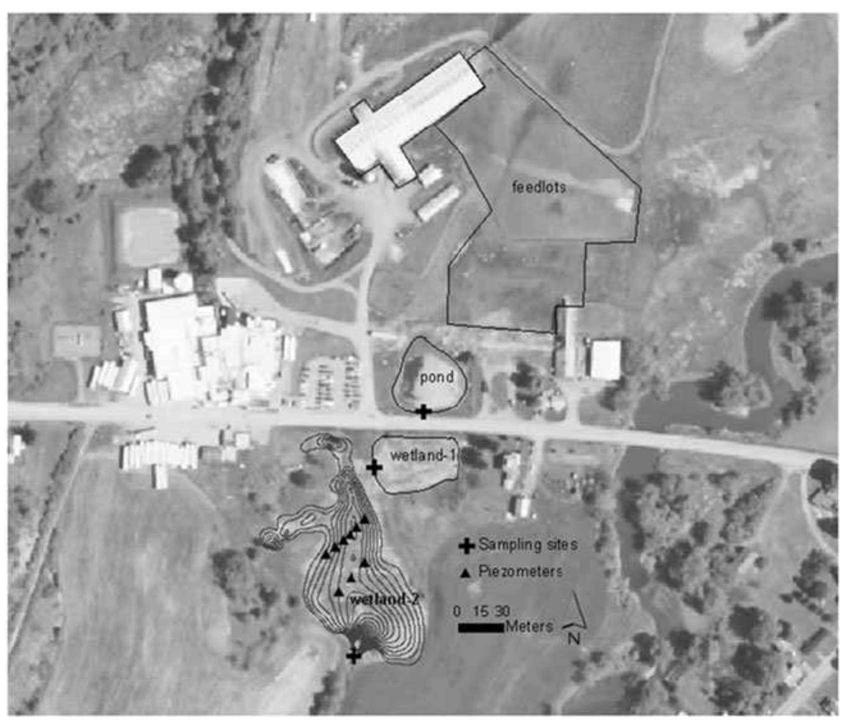
The LAW and its watershed are comprised of several distinct landscape features (Fig. 3). Its 2.14 hectare (ha) watershed is comprised of 33% buildings, 54% feedlots, and 13% heavy use areas. The soil is a compacted Howard gravelly silt loam (USDA SCS 1982) with slopes ranging from 6% to 8%. The mix of buildings, driveways, bunk silos, and feedlots—all virtually impervious surfaces, result in ~99% of the rainfall leaving as runoff ([USDA NRCS] 1990).

The pond, immediately downstream of the feedlots, has a surface area of 0.2 ha with depths ranging from ~0.9 to 1.5 m. Its NWL is controlled by a road culvert. Wetland-1 (0.12 ha), immediately downstream of the pond, is mapped a Saprist soil (USDA SCS 1982). Its NWL is controlled by a natural swale outlet. Its primary macro-vegetation (by

Fig. 2 Maps of study area: the maps show the locations of Chenango County, its major watersheds, and the Long Acres Wetland

area covered) is narrow leaved cattail (*Typha angustifolia)* 80%, reed canary grass (*Phalaris arundinacea*) 15%, and stick tights (*Bidens cernua*) 5%. Other minor macro-species included water smartweed (*Polygonum amphibium)*, joe-pye weed (*Eupatorium maculatum*), blue flag iris (*Iris versicolor*), black willow (*Salix nigra*), and duckweed (*Lemna minor*). The primary micro-flora is green algae (*Chlorella vulgaris*). Although receiving animal waste polluted runoff for nearly 100 years, it appears to be relatively unaffected. Wetland-2 (0.8 ha), immediately downstream of wetland-1 is also mapped Sparist. However, for most of the growing season very little surface water is

Fig. 3 Map of Long Acres Wetland



evident with the emergent vegetation a monoculture of reed canary grass.

Data Collection

Field measurements of surface water elevation (meters above mean sea level), conductivity (μS/cm), pH (units), and temperature (°C) were taken ~ bi-weekly from 21 March 2005 to 21 December 2005 (sample year 2005 (SY2005)-drained condition) and from 21 March 2006 to 21 December 2006 (sample year 2006 (SY2006)-restored condition) at three sample sites and nine nested piezometers (Fig. 3), using permanently installed stream gauges and portable HACH® Conductivity, OAKLON® pH, and HACH® Thermometer instrumentation, respectively. At the same time, surface water grab samples were taken at the three sample sites, which were analyzed at the State University of New York Binghamton's Center for Integrated Watershed Studies lab for [NH_3], [NH_4^+], [NO_2^-], and [NO_3^-] (mg/L) using a Lachat Autoanalyzer®. Flooded vegetative area (ha) was calculated by ArcGIS® Spatial Analyst, using the water surface elevation measurements combined with a digital elevation map (DEM) of the LAW (LAW-DEM), overlaid on global positioning system (GPS) mapped SY2005 and SY2006 end-of-growing-season vegetative spatial inventories. Bi-weekly retention time (d) was calculated by dividing the pond, and wetland-1 and −2 basins' storage volume by the bi-weekly watershed runoff inflow. Bi-weekly runoff was calculated from the National Weather Service's Broome County Airport Station's daily rainfall database inputted into the USDA NRCS Engineering Field Manual Chapter-2 (EFM-2) Small Watersheds Runoff Program (USDA NRCS 1990). While wetland basin water storage volume was calculated by ArcGIS® Spatial Analyst with inputs of water surface elevations and the LAW-DEM.

CW-656 Modeled Datasets

The LAW's SY2005 and SY2006 basin measurements of runoff inflow, flooded vegetative area, water storage volume, and water temperature were inputted into the USDA NRCS Constructed Wetland Standard 656 (CW-656) designing Eq. 2 (USDA NRCS 2000, 2002), to calculate coinciding bi-weekly wetland-2 outlet modeled pre- and post-restoration [TN] datasets ([TN] is equivalent to the CW-656 model's wetland effluent concentration *Ce* (mg/L):

$$A = (Qa/Kt)\ln[(Ce - C*)/(Ci - C*)](356/tcw) \tag{2}$$

where A is wetted surface area of the wetland (m^3), Qa is annual flow into the wetland (m^3/year), Kt is $K_{20}\Theta^{T-20}$ (rate constant adjusted for temperature: K_{20} is 14 for TN, Θ is 1.06 for TN, T is average operating temperature (°C)), Ci is wetland influent concentration (mg/L), Ce is wetland effluent concentration (mg/L), $C*$ is background concentration (mg/L) assumed to be 10 for TN, 365 is days of the year (d), and tcw is days that the wetland will be in operation (the length of growing season).

Statistical Analysis

To compensate for possible variations in SY2005 and SY2006 herd management and/or environmental conditions influencing the feedlots' runoff [TN], all actual and modeled pre- and post-restoration [TN] datasets were normalized to percent total nitrogen removed (%TNR). %TNR was calculated by subtracting each sample site's bi-weekly [TN] from the pond outlet's [TN], then dividing that difference by the pond outlet's [TN], and finally multiplying by 100. Conversion of [TN] datasets to %TNR datasets assumes that any external wetland variables influencing AFO runoff [TN] are minimized, so that the pre- and post-restoration TN removal statistical comparisons are valid.

Conductivity spatial distribution was assessed by comparing the SY2005 bi-weekly piezometer field surface water conductivity measurements' sample variance (σ^2) dataset to the SY2006 bi-weekly σ^2 dataset—the assumption being, the greater the σ^2 the less uniform the mixing.

The above datasets of [TN], %TNR, flooded vegetative area (ha), conductivity spatial distribution (σ^2), pH (units), water temperature (°C), and retention time (d) were then analyzed for 1) restoration's influence on these variables and 2) relationships between %TNR and flooded vegetative area (ha), conductivity distribution (σ^2), pH (units), water temperature (°C), and retention time (d) using Microsoft Excel® data analysis functions of summary statistics, analysis of variance (ANOVA: single factor ($F/Fcrit$)), temporal correlation (r), and simple linear regression (r^2).

Results

Actual and CW-656 Modeled TN Removal

Actual and CW-656 modeled pre- (SY2005) and post-restoration (SY2006) TN removal statistics are graphically presented in Fig. 4. The first column lists the summarized bi-weekly sampled actual or modeled CW-656 SY2005 and SY2006 pond and wetland-2 outlets' [TN] datasets as mean [TN] ($\mu = xx$ mg/L). The second column lists the ANOVA with P-value statistics comparing these [TN] datasets. The third column lists the summarized SY2005 and SY2006 bi-weekly calculated %TNR between the pond and wetland-2 outlets as mean %TNR ($\mu = xx\%$). The fourth column lists the ANOVA with P-value statistics comparing same year

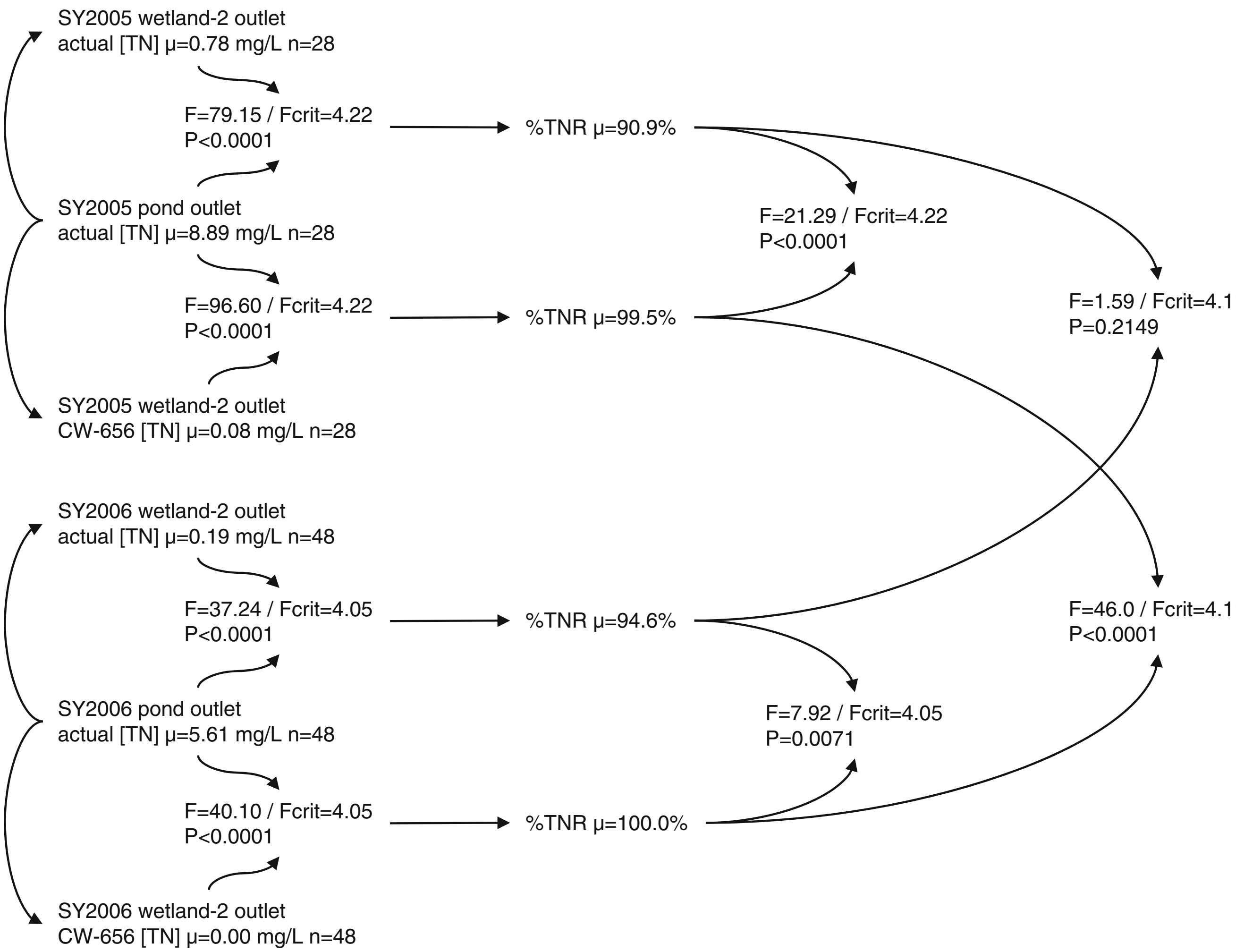


Fig. 4 Pre- (SY2005) and Post-Restoration (SY2006) Statistical Analysis Flow Diagram: First, actual and CW-656 modeled SY2005 and SY2006 wetland-2 outlet [TN] datasets are compared to respective SY2005 and SY2006 actual pond outlet [TN] datasets by analysis of variance (ANOVA: single factor (*F/Fcrit*)) with P-values. These [TN] datasets are then normalized to actual and CW-656% total nitrogen removed (%TNR) datasets and summarized as mean (μ) %TNR. Next, SY2005 and SY2006 actual %TNR datasets are compared to respective CW-656 %TNR datasets by ANOVA with P-values. Finally, the SY2005 actual %TNR dataset is compared to the SY2006 actual %TNR dataset, and the SY2005 CW-656 %TNR dataset is compared to the SY2006 CW-656 %TNR dataset by ANOVA with P-values

actual to CW-656 %TNR datasets. While the fifth column lists the ANOVA with P-value statistics comparing SY2005 actual to SY2006 actual and SY2005 CW-656 to SY2006 CW-656 %TNR datasets.

Referring to Fig. 4, ANOVA of the actual SY2005 pond outlet [TN] versus the actual SY2005 wetland-2 outlet [TN] datasets (F=79.15/$Fcrit$=4.22, P=0.0001) and the actual SY2006 pond outlet [TN] versus the actual SY2006 wetland-2 outlet [TN] datasets (F=37.24/$Fcrit$=4.05, $P<$ 0.0001) each indicates significant reductions of runoff [TN] as it flowed from the pond outlet to the wetland-2 outlet. Likewise, ANOVA of the actual SY2005 pond outlet [TN] versus the CW-656 modeled SY2005 wetland-2 outlet [TN] datasets (F=96.60/$Fcrit$=4.22, P=0.0001) and the actual SY2006 pond outlet [TN] versus the CW-656 modeled SY2006 wetland-2 outlet [TN] datasets (F=40.10/$Fcrit$= 4.05, $P<$0.001) also each indicates significant reductions of runoff [TN] as it theoretically flowed from the actual pond outlet to the modeled wetland-2 outlet.

However, ANOVA of the actual SY2005 %TNR versus the CW-656 modeled SY2005 %TNR datasets (F=21.29/$Fcrit$= 4.22, $P<$0.0001), and the actual SY2006 %TNR versus the modeled SY2006 %TNR datasets (F=7.92/$Fcrit$=4.05, P= 0.0071), both indicate significantly greater TN removal in the SY2005 and SY2006 modeled scenarios. Corroborating this observation is ANOVA of the actual SY2005 %TNR versus the actual SY2006 %TNR datasets (F=1.59/$Fcrit$= 4.11, P=0.2149), indicating no significant difference in TN removal between the actual drained and restored conditions. While ANOVA of the CW-656 modeled SY2005 %TNR versus the CW-656 modeled SY2006 %TNR datasets (F= 46.0/$Fcrit$=4.11, $P<$0.0001), indicates a significant difference. These findings point to the obvious conclusion that the actual restored wetland-2 basin was operating at a lower operational efficiency then the CW-656 modeled predictions.

Pre- Versus Post-Restoration Denitrifying Covariates and TN Removal

Restoration's impact, on each of the five denitrifying covariates is presented in a Table 1. The first column lists the five covariates, the second and third columns lists the means *(μ)* of the pre- (SY2005) and post-restoration (SY2006) covariate bi-weekly sampled or calculated datasets. The fourth column lists ANOVA statistics of the SY2005 versus SY2006 covariate datasets with their associated P-values in the fifth column. The pre- (SY2005) and post-restoration (SY2006) relationships between each of the five denitrifying covariates and % TNR is presented in Table 2. The first column lists the five covariates. The second and third columns list the SY2005 and SY2006 temporal correlation statistics of each covariate's wetland-2 basin's dataset to its corresponding SY2005 or SY2006 %TNR dataset. While the fourth column lists the simple linear regression analysis statistic of combined SY2005 and SY2006 wetland-2 basin covariate datasets compared to its corresponding combined SY2005 and SY2006 %TNR datasets.

Referring to Table 1, ANOVA of each covariate's sampled or calculated pre-(SY2005) versus post-restoration (SY2006) datasets, except conductivity spatial distribution *((F*=1.75/ *Fcrit*=4.15*, P*=0.1952*)*, were improved by restoration, with retention time showing the most significant increase followed by pH (still maintaining a circum-neutral range), flooded vegetative area, and water temperature. Consequently, the lack of mixing or non-uniform distribution of the TN polluted runoff is acknowledged as the limiting factor controlling operational efficiency. Table 2 tends to corroborate this finding with no statistically significant temporal correlation *(r)* between the SY2005 and SY2006 covariates' datasets versus %TNR datasets (range 0.48 to −0.12), or by simple linear regression *(r^2)* of combined SY2005 and SY2006 covariates' datasets versus combined SY2005 and SY2006 %TNR datasets (range 0.01 to −0.11).

Discussion

In our case study the drained and restored LAW's [TN] datasets were compared to determine restoration's influence on TN removal. These actual pre- and post-restoration datasets were then compared to modeled pre- and post-restoration datasets to quantify actual operational efficiency based on a theoretical benchmark. Next, pre- and post-restoration datasets of each denitrifying influencing variable's covariate were compared to estimate restorations influence on these associated covariates. And finally, actual pre- and post-restoration %TNR datasets were compared to pre- and post-restoration covariates' datasets to rank each of the five primary wetland denitrifying influencing variables' relationship to TN removal.

Our results indicate that both the actual pre- and post-restoration LAW significantly removed TN from AFO N polluted runoff as it flowed through the wetland. However, statistical comparison of the wetland outlet's actual pre- to actual post-restoration %TNR datasets indicates no statistical difference. Comparisons of actual post-restoration % TNR to modeled post-restoration %TNR predictions indicate that the actual post-restoration wetland was not operating at its theoretical optimal TN removal efficiency. Statistical analyses of the pre- and post-restoration denitrifying covariates datasets to actual pre- and post-restoration %TNR datasets, indicates a lack of contact (i.e., non-uniform distribution of N pollutants) as the limiting factor negating all other favorable increases in the other denitrifying influencing variables. In other words, without bacteria and N pollutant contact, the potential positive treatment performance gains of increased retention time, flooded vegetative area, water temperature, and maintenance of a circum-neutral pH were all negated.

Restored Versus Constructed Wetland

Conceptually, a restored drained wetland basin should provide the equivalent chemical, biological, and physical environment for bacteria to biotransform N pollutants as a comparable constructed wetland. However, the actual restored wetland-2 basin was not ideal (Fig. 3). Unlike the

Table 1 Wetland-2 Denitrification Covariates: The table lists the pre- (SY2005) n=24 and post-restoration (SY2006) n=48 denitrifying covariate datasets means *(μ)* and their SY2005 versus SY2006 analysis of variance (ANOVA: single factor (*F/Fcrit*)) comparison with *P-values*

Denitrification covariates	SY2005 *μ*	SY2006 *μ*	SY2005/SY2006 *(F/Fcrit)*	SY2005/SY2006 *P- value*
Flooded vegetative area (ha)	0.03	0.40	88.89/4.11	<0.0001
Conductivity distribution (σ^2)	11673	8302	1.75/4.15	0.1952
pH (units)	7.5	7.8	19.67/4.11	<0.0001
Water temperature (°C)	12.4	16.9	5.19/4.11	0.0286
Retention time (d)	1.2	14.7	460.90/4.11	<0.0001

Table 2 Wetland-2 Denitrifying Covariates Versus Percent Total Nitrogen Removed: The table lists the temporal correlations (r) of the pre- (SY2005) n=24 and post-restoration (SY2006) n=48 denitrifying covariates to percent total nitrogen removed (%TNR), and the simple linear regression (r^2) of combined SY2005 and SY2006 covariate datasets to combined SY2005 and SY2006 % TNR datasets

Denitrification covariates	SY2005 covariate/%TNR r	SY2006 covariate/%TNR r	SY2005–SY2006 covariate/%TNR r^2
Flooded vegetative area (ha)	0.22	0.20	0.01
Conductivity distribution (σ^2)	0.48	−0.12	0.00
pH (units)	−0.01	−0.02	−0.01
Water temperature (°C)	0.44	0.23	−0.11
Retention time (d)	0.06	0.08	−0.05

idealized CW-656 model, (which assumes uniform depth, mixing, and vegetative cover), the wetland-2 basin was created by a series of random events of nature with the last glacial epoch having the most profound impact, followed by anthropomorphic intervention—predominantly by agriculture. Its inlet is a narrow grassed waterway—located at its northeast corner ~25 m south and down slope of its most northerly extent, resulting in very little initial horizontal or lateral runoff distribution. The shape of the basin—a sinuous oval with ~1:5 width to length ratio, combined with a shallow shoreline—with irregular oxbow features, forms several stagnate pools further contributing to dampening of mixing. And finally, the basin substrate surface is very irregular with non-uniform hummocks of growing and decaying reed canary grass and cattail clumps—each with varying stand densities, forming both shallow and deep water, concentrating the flow into distinct channels.

However, the drained and restored LAW's average N removal efficiency of 90.9 and 94.6% respectively were each much better than the typical constructed wetland's 80% N removal rate (Baker 1992; Boesch et al. 2001; Driscoll et al. 2003; Ducks Unlimited 2003). This may be attributed to constructed wetlands significantly longer start-up period—taking months or years to build up organic sediments (soils) and establish hydrophytic vegetation. While drained wetlands intrinsically possess these essential variables (Carlsson et al. 2003; Ducks Unlimited 2003). Hence, restoring drained wetland basins, as an alternative BMP to constructed wetlands, may not only provide cost-environmental benefits associated with utilizing an existing resource, but also greater operational efficiency associated with a shorter start-up time coupled with the stability of a naturally evolved indigenous ecosystem (van der Valk and Jolly 1992).

Surface Flow Enhancement

In a constructed wetland polluted runoff flows evenly across the wetland cell, throughout its entire length, with limited stagnant pools and short-circuiting. However, unlike constructed wetlands, restored wetlands apparently do not uniformly distribute pollutants. Therefore, suggested remedial practices to improve mixing and spatial distribution begin with three techniques commonly used in municipal and industrial constructed wetlands (Kadlec and Knight 1996). First, large sophisticated constructed wetlands typically have a lateral inlet distribution structure, such as a header pipe, level lip spreader, or crushed stone trench to spread the polluted effluent over the inlet end of the basin. Second, water elevation controlling structures are installed at the outlet to modulate the surface water level reducing the hydraulic gradient and mitigating substrate grading imperfections. And third, internal lateral deflective flow structures such as earthen berms or pilings made of steel, wood, or plastic are installed perpendicular to the major surface flow currents, forcing surface water to the perimeter of the wetland, increasing uniformity of pollutant concentration. However, these structures are expensive to install and may require exceptional amounts of maintenance (beyond the capabilities of the typical farmer) due to heaving caused by freezing and thawing, breakage by ice and soil subsidence, and accelerated decay by microorganisms. Hence, an alternative to these routinely used practices may simply be to add small uniquely designed, strategically placed constructed wetland cells within the restored wetland basin. For example, cells at the inlet could spread the runoff; within the wetland perimeter, cells could add retention time and/or flooded vegetative area; or in our case study, cells constructed outside the perimeter (at explicit elevations) could redirect the surface flows improving mixing and eliminating concentrated flows.

Restored Wetland Application

During the last two centuries the USFWS (2000) estimates that over 50% of U.S. wetlands have been drained or destroyed. Following this national trend, Chenango County, NY has had over 40% of its wetlands cleared and drained for agricultural production, primarily because of their hydrological impact on adjacent productive agricultural lands (Johnson 2010). Thus, regions having both extensive agricultural areas and numerous drained wetlands potentially have the best opportunity for restored drained wetland BMP projects (Baker 1992). However, quantity does not necessarily equate to quality (i.e., biotransformation functional capacity) (ASWM 2000; Cedfeldt et al. 2000). Hence, predictors must be developed, which not only assess the physical location of a drained wetland to a pollution source, but also its effectiveness to actually perform. Suggested criterion for initial selection are: 1) wetland juxtaposition—what is the distance from the AFO (source of pollution) to the drained wetland, 2) feasibility—how much or to what extent has the wetland been degraded, which as a rule is directly related to its converted land use (i.e., farmstead, cropland, or pasture), 3) utility—relative elevation of the wetland to the source of pollution (critical in determining if simple relatively inexpensive gravity waste transfer systems are appropriate or if more complex and expensive mechanical pumping systems are required), 4) priority—obviously with limited resources (time, people, and money) not all drained wetland restoration treatment projects can be implemented simultaneously (in the case of AFO runoff pollution, typically the larger the herd size the greater the pollution threat), and 5) offsite impacts—if a drained wetland is to be restored, what are the implications or impacts on adjacent installed drainage practices or properties (Baker 1992; van der Valk and Jolly 1992).

Restored Wetland Policy

Although constructed wetlands are generally accepted at this time, the use of natural wetlands as treatment wetlands is not. The principal barrier impeding the use of natural wetlands for treating NPS pollution is imposed by "precautionary principle" federal legislation (Gopal 2003) associated with provisions of the 1972 Clean Water Act (CWA). These provisions, which specifically prohibit the discharges of pollutants into natural wetlands, are a result of the public's perceived associated environmental risks (Smardon and Hammer 1989). This anxiety is linked directly to the unanswered technical question of, do natural wetlands have an unlimited capacity to treat wastes, transform nutrients, and other substances without being impacted (Mainguy et al. 2000; Kiker et al. 2001; Gopal 2003).

However, at the same time this federal legislation is rather vague for the use of restored drained wetlands as treatment wetlands. According to Fields (1992), two important provisions of the CWA that actually promotes the use of wetlands to treat NPS pollution are Sections 303 and 319. Section 303 requires States to adopt water quality standards that include designating uses for wetlands and other waters, and to assign water quality criteria that will meet those uses. And, under Section 319 individual States are required to perform, under US Environmental Protection Agency oversight and approval, assessments of the status of NPS pollution and to develop management programs to control NPS pollution. These management programs will ultimately require the use of various BMPs to ensure compliance with the applicable State water quality standards. Consequently, Sections 303 and 319 may provide the possible mechanism for State NPS management programs to use restored drained wetlands to treat NPS pollution (Olson 1992).

Conclusion

Restoring drained wetlands to treat AFO NPS N polluted runoff is an emerging bio-engineering concept. Conceptually, a restored drained wetland should provide the equivalent chemical, biological, and physical environment for bacteria to biotransform AFO runoff N pollutants into N gases as a comparable constructed wetland. And like its constructed wetland equivalent, a hydraulically restored wetland's TN removal performance is dependent on the optimization of the denitrifying variables of bacteria population size, bacteria-pollutant contact, N Cycle continuity, kinetics, and bacteria-pollutant contact time. Which in turn, are quantified by their covariates of flooded vegetative area, conductivity spatial distribution, pH, water temperature, and retention time respectively.

Our research results indicate that both the pre- and post-restoration LAW significantly removed TN from the AFO N polluted runoff as it flowed through the wetland. However, statistical comparison of actual pre- to actual post-restoration wetland-2 outlet %TNR datasets indicated no statistical difference. Additional comparisons of actual post-restoration results to modeled post-restoration predictions indicate that the actual post-restoration wetland was not operating at its theoretical optimal TN removal efficiency. Statistical comparison of pre- to post-restoration denitrifying variables' covariate datasets to corresponding %TNR datasets revealed that bacteria-pollutant contact, as measured by its covariate conductivity spatial distribution, was the operational efficiency limiting factor. In other words, without bacteria and N pollutant contact, the positive treatment performance gains of increased retention time, flooded vegetative area, water temperature, and maintenance of a circum-neutral pH were all negated.

However, one unanticipated outcome of our case study was the 90.4% removal of TN from the runoff by the LAW in its drained condition. This evidence leads to the speculation that perhaps drained wetlands that receive polluted runoff from AFOs or other agricultural pollution sources, maybe functionally removing large amounts of TN from the runoff contributing to improving water quality without any manipulation or intervention required. Future research should therefore be directed to determine drained wetlands in situ N removal functional capacity.

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