The use of aeration to enhance ammonia nitrogen removal in constructed wetlands

T.S. Jamieson[1], G.W. Stratton[1], R. Gordon[2] and A. Madani[2]

[1]*Department of Environmental Sciences and* [2]*Department of Engineering, Nova Scotia Agricultural College, P.O. Box 550, Truro, Nova Scotia, Canada B2N 5E3*

Jamieson, T.S., Stratton, G.W., Gordon, R. and Madani, A. 2003. **The use of aeration to enhance ammonia nitrogen removal in constructed wetlands**. Canadian Biosystems Engineering/Le génie des biosystèmes au Canada **45**: 1.9-1.14. Nitrogen (N) resulting from manure run-off can impair receiving water bodies. Constructed wetlands are engineered systems built to utilize the treatment processes available in natural wetlands in a controlled and predictable manner. Nitrification is often limited in constructed wetlands treating agricultural wastewater due to the predominantly anaerobic conditions within the wetland. A greenhouse wetland model was constructed to assess the effects of aeration on ammonia-N removal. The system achieved a mean 50% ammonia-N removal efficiency prior to the introduction of an aquatic aeration system. After the commencement of continual aeration in the first cell of the system, ammonia-N removal efficiencies increased to a mean of 93%, parallelled by nitrate-N increases. This indicated that the addition of continual aeration has great potential to enhance nitrification in constructed wetlands receiving agricultural wastewater. **Keywords**: nitrification, constructed wetland, wastewater, aeration, ammonia.

L'azote contenu dans les lixiviats de fumier ou de lisier peut nuire aux cours d'eau qui les reçoivent. Des marais artificiels font appel aux processus de traitement qui surviennent dans des marais naturels d'une manière contrôlée et prévisible. Les processus de nitrification sont souvent limités dans les marais artificiels qui traitent les eaux usées agricoles en raison des conditions anaérobiques qui prévalent dans ces systèmes. Un modèle de marais sous serre a été construit pour évaluer les effets de l'aération sur l'extraction de l'azote ammoniacal. Le système a été capable d'extraire 50% l'azote ammoniacal avant l'introduction d'un système aquatique d'aération. Lorsque la première cellule du système a été aérée de façon continue, l'efficacité d'extraction de l'azote ammoniacal a augmenté à 93% en moyenne tandis que des augmentations de la teneur en nitrates étaient observées. Les résultats obtenus indiquent que l'aération continue des marais artificiels pésente un grand potentiel pour augmenter le processus de nitrification dans de tels systèmes de traitement pour les eaux usées agricoles. **Mots clés**: nitrification, marais artificiels, eaux usées, aération, ammoniaque.

INTRODUCTION

Pollutants from agricultural operations can be a significant contributor to the impairment of surface and groundwater quality (Knight et al. 2000). Manure containment facilities can overflow during periods of significant rainfall or become inadequate to handle the level of on-farm waste production, thereby contributing to nitrogen (N) loading to receiving water bodies. Eutrophication results when nutrient levels, namely phosphorus and N, exceed the natural ecological balance, negatively affecting aquatic ecosystems, causing algal blooms, growth of aquatic weeds, fish dieback, and a loss of biodiversity (Carpenter et al. 1998; Smith et al. 1999). Constructed wetland treatment systems are engineered systems built to utilize the treatment processes available in natural wetlands in a controlled and predictable manner (Benham and Mote 1999). Due to the natural processes taking place in wetland systems, little or no fossil fuel derived energy inputs and chemicals are required to treat wastes, making wetlands more economical waste treatment systems than conventional biological sewage treatment (Kadlec and Knight 1996; Peterson 1998).

Nitrogen entering constructed wetlands is present in particulate and dissolved organic and inorganic forms, the relative proportions of which depend on the type of waste and pretreatment (Reddy and D'Angelo 1997). The primary forms of inorganic N entering constructed wetlands are: ammonium/ammonia (NH_4^+/NH_3) and nitrate (NO_3^-) (Gale et al. 1993; Kadlec and Knight 1996). The sum of NH_4^+ and NH_3 will be referred to as NH_3-N. Organic N is present in wetlands in the form of amino acids, urea, uric acid, amines, purine, and pyrimidines (Stevenson 1986). Particulate forms are removed through settling and burial within the sediment layer (Reddy and D'Angelo 1997). Within wetlands, N is treated via two main pathways: (i) storage and (ii) removal through the N cycle (Hseih and Coultas 1989). Storage is achieved by assimilation into the biomass (ie: plant and microbial uptake) or adsorption to the substrate (ie: soil). This is only a temporary solution, because the wetland has a finite storage capacity, and the stored N can be remineralized back into solution or undergo desorption.

The more permanent removal of N in constructed wetlands is dependent on the N cycle. As part of the cycle, the various forms of N are converted into gaseous components that are expelled into the atmosphere as nitrogen gas (N_2) or nitrous oxide (N_2O). Key processes in the N cycle include ammonification, nitrification, and denitrification. Nitrification-denitrification reactions are the dominant removal mechanisms in constructed wetlands (Benham and Mote 1999). Nitrification is the biological formation of nitrite-N (NO_2^--N) or NO_3^--N (Alexander 1977) from NH_4^+. Nitrification occurs in aerobic regions of the water column, soil-water interface, and root zone (Reddy and D'Angelo 1997). Dissolved oxygen levels < 1-2 mg/L in water substantially reduces nitrification (Hammer and Knight 1994; Lee et al. 1999).

Denitrification is the biological process of reducing NO_3^--N or NO_2^--N, into N_2, N_2O, or nitric oxide (NO) (Kadlec and Knight 1996). Denitrification is a significant mechanism in

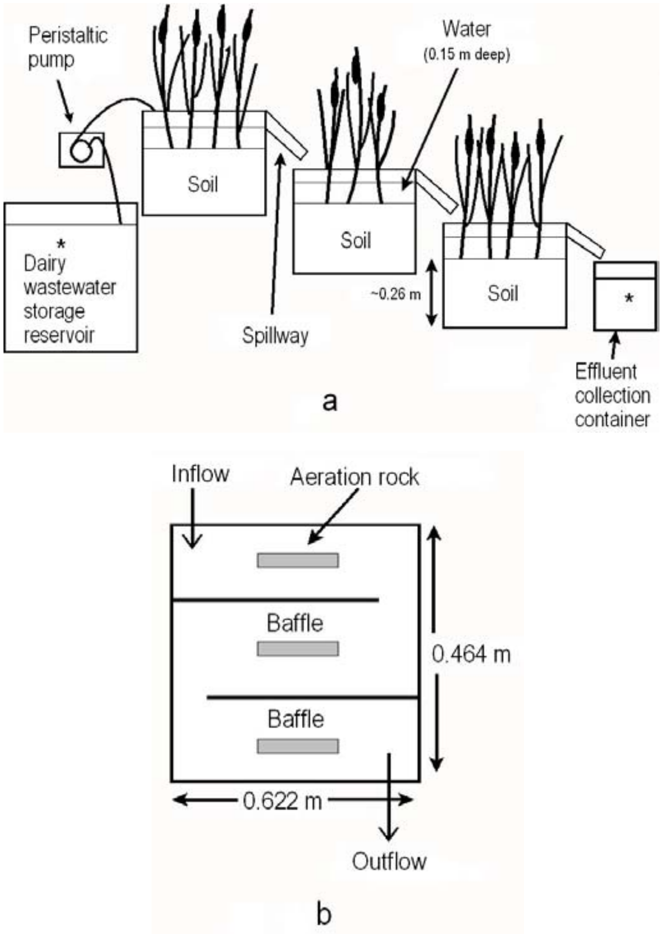

Fig. 1. Greenhouse scale constructed wetland model using three wetland cells in series; (a) side view of the system; (b) overhead view of a wetland cell; * wastewater sampling locations.

treatment of wetlands for the permanent removal of N from wastewater (Hammer and Knight 1994). However, denitrification cannot occur if NO_3^--N is not in adequate supply. In many wetlands, the nitrification rate is much slower than the denitrification rate, so the first process affects the latter (Verhoeven and Meuleman 1999). The NO_3^--N supply limiting the subsequent denitrification process has often been identified as a problematic issue (Hseih and Coultas 1989; Busnardo et al. 1992; Newman et al. 2000).

Nitrification is often limited in surface flow constructed wetlands treating agricultural wastewater derived from livestock (Sievers 1997; Rochon et al. 1999; Newman et al. 2000; Sartoris et al. 2000) due to the predominantly anaerobic conditions. Biochemical oxygen demand (BOD) levels are often very high in this wastewater and nitrification appears to be limited by oxygen availability (Cronk 1996). Currently, in Nova Scotia, primary treatment of agricultural wastewater prior to discharge to a treatment wetland usually consists of an anaerobic settling lagoon. In many of these wetlands, anaerobic conditions prevail, which are not conducive for nitrification reactions. As NH_3-N is one of the principal forms of N in livestock wastewater, and due to its potential role in water quality degradation, reducing NH_3-N concentrations drives the design process for many wetland treatment systems (Kadlec and Knight 1996). Thus, unsuitable conditions for nitrification can seriously limit the treatment potential of these systems. The use of supplemental aeration may enhance nitrification activity, due to the addition of dissolved oxygen into the wastewater which would induce a more aerobic environment for this reaction. Cottingham et al. (1999) found that aerating laboratory scale subsurface flow constructed wetlands promoted increased rates of nitrification. Surface flow, or free water surface constructed wetlands, however, are used for treating livestock wastewater in Nova Scotia due to their ability to handle relatively high solids content. Subsurface flow wetlands are generally not recommended for agricultural wastewater treatment with substantial solids content (NRCS 1991).

The primary objective of this study was to investigate how aeration affects the NH_3-N removal efficiency of constructed wetlands receiving dairy farm wastewater through the use of a small scale, pilot surface flow wetland.

MATERIALS and METHODS

Greenhouse wetland description

A surface flow, vegetated, wetland system was simulated in a greenhouse using three plastic containers measuring 0.622 m long by 0.464 m wide by 0.489 m deep (Fig. 1). The three containers operated in series, providing a total surface area of 0.866 m^2. Laboratory-scale wetland experiments have been employed in various wetland research studies (van Oostrom and Russell 1994; Benham and Mote 1999; Lee et al. 1999; Wood et al. 1999) to minimize uncontrollable factors, such as environmental conditions. A mineral wetland soil (6% organic matter, pH 6.5), to a depth of approximately 0.265 m, and cattail (*Typha sp.*) shoots were collected from a local cattail stand to reproduce natural wetland conditions. The plants, which were grown under flooded conditions with well water, were allowed to establish themselves within the containers and adapt to the greenhouse conditions for approximately one year prior to experimentation (thirty-five to forty 1.0 - 1.5 m shoots per cell).

Design requirements were based on recommendations outlined by the Natural Resource Conservation Service of the United States Department of Agriculture (NRCS 1991). These design guidelines assume that the vegetation consists of wastewater tolerant, emergent, hydrophytic plants, such as cattails. Nutrient uptake by vegetation also is assumed not to be a major consideration in nutrient removal, but the roots and stems serve as a medium for microbial growth (NRCS 1991). Throughout the present study, the cattails were at the mature phase of their growth cycle. Theoretical residence time for the wetland was calculated based on NRCS (1991) as:

$$t = AD\frac{P}{Q} \tag{1}$$

where:

t = hydraulic residence time (d),
A = surface area of constructed wetland (0.866 m^2),
D = flow depth in constructed wetland (0.15 m),
Q = loading rate (m^3/d), and
P = porosity, ratio of volume of constructed wetland occupied by water, to volume of wetland occupied by plant roots, soil, and water.

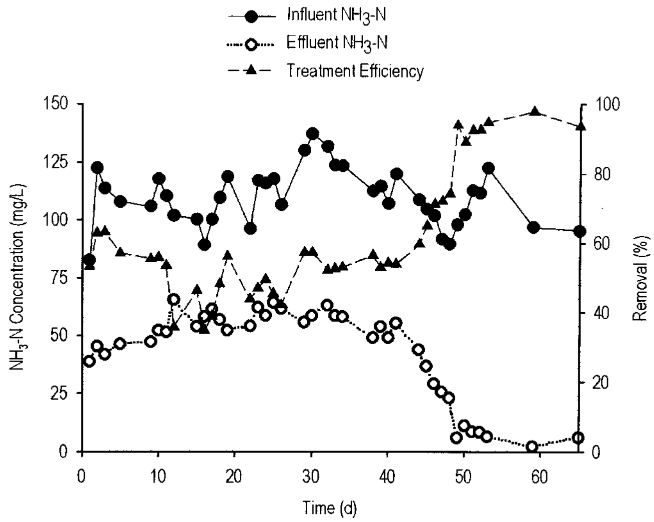

Fig. 2. Influent and effluent NH_3-N and associated treatment efficiencies, over time, for the greenhouse scale wetland model.

The NRCS assumes that cattails have a porosity value of 0.95. Wastewater was fed into the wetland by a peristaltic pump (Masterflex Pump Controller, Cole Parmer Instrument Co., Chicago, IL) to achieve a constant Q. The most appropriate Q that could be achieved by the peristaltic pump was 11.5 L/d, giving the treatment system a mean theoretical residence time of approximately 11 days.

The NRCS (1991) recommends a high length to width ratio (3:1 to 4:1) to achieve efficient surface area utilization. Baffles were installed within each wetland cell to minimize channelization of water flow, thereby attempting to make use of the entire surface area of the wetland. Since the greenhouse is a protected environment, variations in environmental conditions (ie: temperature) were minimized. To maintain a constant water depth, a 250-mL Erlenmeyer flask filled with distilled water was inverted in each wetland cell with the mouth of the flask level with the water level in each wetland cell. Water lost through evapotranspiration was replaced by the water within the flasks. The water level in each flask was monitored on a daily basis and replaced if empty.

After the one year stabilization period, well water was allowed to flow through the system at test rates for one week. Wastewater was then loaded into the wetland for an additional week prior to monitoring. Wastewater was collected from a dairy farm's settling lagoon which received runoff from a solid manure storage pad, as well as milk-house wash-water. The mean NH_3-N concentration was approximately 300 mg/L with a BOD_5 concentration of approximately 735 mg/L (Jamieson 2001). Prior to addition to the wetland, the wastewater was diluted with well water and effluent (see below) to maintain an influent NH_3-N concentration of 100 mg/L. The wetland was operated as a closed system, where the effluent water was collected, analyzed, and mixed with well water to dilute the dairy farm wastewater, as required, to maintain a consistent influent NH_3-N concentration. This was done to reduce the amounts of wastewater required and effluent to be disposed of.

Monitoring

Inflow and outflow samples (Fig. 1) were collected in the morning (Monday through Friday) of wetland operation and analysed for NH_3-N and NO_3^--N concentration. The inflow wastewater was grab sampled from the influent wastewater reservoir (Fig. 1) prior to any wastewater additions for that particular day. Outflow samples were obtained by grab sampling a composite from the accumulated effluent since the previous sampling. Ammonia-N was immediately analysed each sampling day using the phenate method (APHA 1998). Nitrate-N was analysed by ion chromatography (Dionex DX500, Dionex Corporation, Sunnyvale, CA). Samples were frozen until NO_3^--N analyses were conducted.

The wetland model was operated for 24 days without additional aeration to provide a background efficiency characterization for the removal of NH_3-N and NO_3^--N. Aeration was then commenced and monitored for 41 days. Preliminary analysis of NH_3-N removal efficiencies indicated that there was a lag time of about two weeks after the introduction of aeration when the removal rates were similar to the non-aerated phase of the experiment. A subsequent increase in NH_3-N removal was noted at this time, necessitating the longer monitoring period for the aeration phase of the experiment when compared to the non-aeration phase (41 versus 24 days).

Aeration was provided by an aquarium pump located near the inlet point of the wetland (Optima Aquarium Pump, Rolf C. Hagen Corp., Mansfield, MA). The aeration pump was equipped with three aeration rocks which were suspended at mid water depth in the first cell (Fig. 1). The pump was set to its maximum aeration capacity (5.5 L/min). Dissolved oxygen concentrations (Field Probe 5739, YSI Inc., Yellow Springs, OH) and pH (Accumet combination electrode, Corning Model 7 meter: Fisher Scientific, Nepean, ON) also were monitored at each sampling time in the wastewater storage reservoir and the middle of each of the three wetland cells. Daily treatment efficiency for the removal of NH_3-N (% Removal) was calculated as:

$$\% \, Removal = \frac{RC - Co}{RC} \times 100 \tag{2}$$

where:

RC = concentration of NH_3-N in storage reservoir (mg/L), and

Co = concentration of NH_3-N in wetland outflow (mg/L).

RESULTS and DISCUSSION

Results from NH_3-N analysis of inflow and outflow samples, along with the associated treatment efficiencies, are presented in Fig. 2. Nitrate-N concentrations in the influent and effluent wastewater are shown in Fig. 3. Mean NH_3-N and NO_3^--N influent and effluent concentrations and removal efficiencies also are presented in Table 1. Ammonia-N removal (Fig. 2) was relatively consistent during the non-aerated period (Day 1 - 24) with a mean removal efficiency of 50.5%. After the introduction of the aeration system on Day 24, there appeared to be a lag phase until approximately Day 49, when a maximum mean treatment efficiency of 93.3% was achieved. During the lag phase, the treatment efficiency was similar to the non-aerated phase of the experiment until approximately Day 40, when the

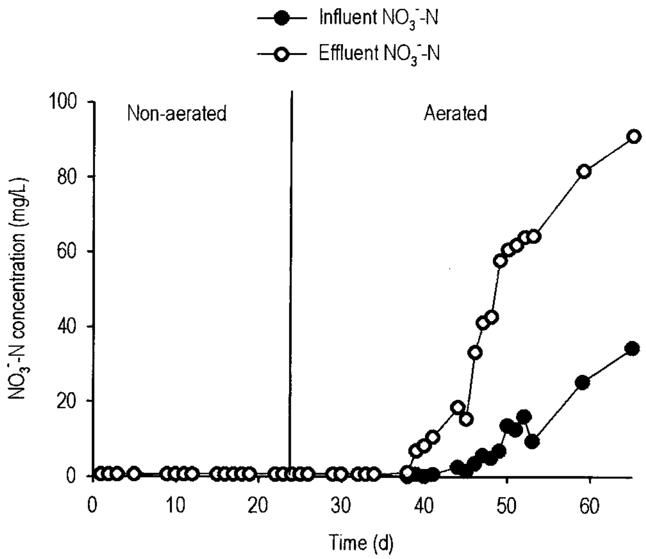

Fig. 3. Influent and effluent NO_3-N results, over time, for the greenhouse scale wetland model.

efficiency started to increase until the maximum was reached near Day 49. Mean influent and effluent NO_3^--N concentrations (Fig. 3) were < 1 mg/L during the non-aerated phase and were consistent until approximately day 38. Afterwards, influent and effluent NO_3^--N concentrations increased steadily until the end of the experiment. Influent NO_3^--N concentrations increased as a result of effluent recirculation.

Mean dissolved oxygen (DO) concentrations and pH levels for the wastewater storage reservoir and each wetland cell are shown in Table 2. The mean water temperature in the wetland was 20°C. Prior to the introduction of aeration, the mean DO concentration throughout the wetland model was < 1 mg/L. After aeration was introduced in cell 1, DO increased to 6.3 mg/L in cell 1. The DO concentration decreased as the wastewater progressed to cell 2 and 3, but remained > 1.5 mg/L. The pH levels were relatively consistent throughout the experiment, ranging from 7.9 to 8.5.

Nitrate-N concentrations (Fig. 3) increased concurrently with NH_3-N removal (Fig. 2), indicating the improvement of NH_3-N removal was mostly due to increases in nitrification activity. The time required for the growth of additional nitrifying bacteria could have been responsible for the lag phase observed. Prior to aeration, NH_3-N removal was most likely due to mechanisms other than nitrification, since mean DO concentrations were <1 mg/L throughout the wetland and nitrification is substantially limited at DO concentrations <2 mg/L (Hammer and Knight 1994; Lee et al. 1999). Ammonia-N removal during the non-aerated phase would be due to volatilization, biological immobilization, or attachment to soil substrates. The mean DO concentration after the commence-ment of aeration in cell 1 of the wetland was 6.3 mg/L, which decreased to 1.8 mg/L in cell 2, and 2.0 mg/L in cell 3. The enhanced availability of oxygen provided by aeration would have facilitated nitrification within the wetland. Volatilization of NH_3-N may also have increased after the commencement of aeration due to physical agitation, but the noted increase in NO_3^--N concentrations between the inlet and outlet of the wetland (Fig. 3) supports the hypothesis that increased nitrification activity was largely responsible.

Table 2. Mean dissolved oxygen (DO) and pH results for the greenhouse wetland experiment prior to and following aeration.

Sampling location	Mean DO (mg/L)	Mean pH
Prior to aeration (1 to 24 d)		
Wastewater storage reservoir	0.4 (0.16)*	8.1 (0.09)
Cell 1	0.4 (0.15)	8.0 (0.12)
Cell 2	0.4 (0.20)	8.0 (0.17)
Cell 3	0.8 (0.41)	7.9 (0.20)
After commencement of aeration (25 to 65 d)		
Wastewater storage reservoir	0.5 (0.39)	8.2 (0.10)
Cell 1	6.3 (0.80)	8.5 (0.09)
Cell 2	1.8 (0.68)	8.3 (0.08)
Cell 3	2.0 (0.92)	8.0 (0.11)

* Values in parentheses are standard deviations.

Cottingham et al. (1999) investigated the use of aeration to promote nitrification activity in a laboratory scale wetland, but used subsurface flow gravel bed models and compared planted and unplanted conditions. The models received primary domestic sewage. Prior to aeration, the mean NH_4^+-N concentration decreased by 5 and 18% in their unplanted and planted models, respectively. After aeration was introduced, the NH_4^+-N removal rate increased to 38 and 68% for the unplanted and planted models, respectively. Similar to the present study, the improvement in NH_4^+-N removal was parallelled by an increase in NO_3^--N and NO_2^--N concentrations, and Cottingham et al. (1999) concluded that the improvement in NH_4^+-N removal was due to increased nitrification activity. Reddy and Graetz (1981) also found that aerating soil and water columns increased nitrification and mineralization.

Table 1. Mean influent and effluent NH_3-N and NO_3^--N concentrations, and NH_3-N removal efficiency results, for the greenhouse scale constructed wetland experiments prior to and following aeration.

	Time (d)		
	1 to 24 (Non-aerated)	25 to 48 (Lag phase)	49 to 65 (Aerated)
Mean influent NH_3-N (mg/L)	107 (11.23)*	114 (13.66)	106 (10.14)
Mean effluent NH_3-N (mg/L)	54 (7.53)	49 (13.54)	7.0 (2.79)
Mean NH_3-N removal (%)	51 (2.17)	57 (9.12)	93 (2.61)
Mean influent NO_3^-N (mg/L)	0.6 (0.002)	1.4 (1.73)	16.8 (9.68)
Mean effluent NO_3^-N (mg/L)	0.6 (0.001)	11.3 (14.92)	68.7 (12.53)

* Values in parentheses are standard deviations.

Nitrate-N appeared to accumulate within the wetland system after aeration was introduced. The increase in influent NO_3^--N concentration was a result of the wetland being operated as a closed system. The effluent from the system had been collected and used to dilute the full strength dairy wastewater in order to provide consistent influent NH_3-N concentrations. Since NH_3-N was being nitrified to NO_3^--N within the system, effluent NO_3^--N increased as a result. The accumulation of NO_3^--N within the wetland indicates that after NH_3-N was nitrified, subsequent denitrification was limited. Possible factors that could limit denitrification include inadequate residence time for denitrification to remove NO_3^--N, the presence of DO, or lack of available carbon (C) within the system. Denitrification activity is reduced if available C supplies are low (Gersberg et al. 1983; Hammer and Knight 1994; Wood et al. 1999) and proceeds only when the oxygen supply is inadequate for microbial demand (Hammer and Knight 1994). However, limited denitrification activity has been observed in the presence of DO (Phipps and Crumpton 1994).

In constructed wetlands, after NO_3^--N is formed under aerobic conditions, it diffuses down into the anaerobic portion of the soil, where it is denitrified (Patrick and Reddy 1976; Nichols 1983). Cottingham et al. (1999) also experienced NO_3^--N accumulation within an aerated wetland model, and suggested that the addition of a C-source in the final section of the model may have improved denitrification activity. In the present study, C availability may have been inadequate to support high levels of denitrification due to the lack of an established litter layer in this relatively young wetland, a problem also experienced by Spieles and Mitsch (2000). As wetlands age, organic matter accumulates in the litter layer provided by decaying plant matter. Craft (1997) estimates that 5 to 10 years of constructed wetland development may be necessary for the accumulation of organic matter to become sufficient to support maximum denitrification. If, on the other hand, the influent wastewater itself was an adequate source of C, the lack of denitrification may be attributed to the short hydraulic retention time of the system.

CONCLUSIONS

The introduction of aeration to a pilot scale constructed wetland model improved the mean NH_3-N removal efficiency from 50.5 to 93.3%, following a 2 week lag phase. Increased removal was primarily attributed to increased nitrification. The accumulation of NO_3^--N within the wetland provides further evidence of nitrification activity. Denitrification of NO_3^--N may have been limited in the system, as demonstrated by the accumulation of effluent NO_3^--N. This may have been caused by an inadequate retention time for complete NO_3^--N removal to occur, or the amount of available C as a substrate may have been insufficient to support denitrifying bacteria.

This study was intended as a preliminary investigation into the potential benefit of providing additional aeration to constructed wetlands treating livestock wastewater. The results of this study indicate that providing aeration has the potential to improve constructed wetland treatment efficiency, as long as subsequent conditions are conducive to denitrification. In the present study, wetland conditions did not allow for sufficient NO_3^--N removal. When designing a wetland treatment system, the required residence time for both nitrification and denitrification reactions to occur should be taken into account, which would require wetlands to be evaluated on a case specific basis. Increasing treatment efficiency also decreases the land requirement needed for wetland construction. If aeration can be provided in a cost effective manner, this has the additional benefit of decreased construction and operation costs to the agricultural producer.

REFERENCES

Alexander, M. 1977. *Introduction to Soil Microbiology* 2nd edition. New York, NY: John Wiley & Sons, Inc.

APHA. 1998. *Standard Methods for the Examination of Water and Wastewater*, 20th edition. Washington, DC: American Public Health Association.

Benham, B.L. and C.R. Mote. 1999. Investigating dairy lagoon treatability in a laboratory-scale constructed wetlands system. *Transactions of the ASAE 42*: 495-502.

Busnardo, M.J., R.M. Gersberg, R. Langis, T.L. Sinicrope and J.B. Zedler. 1992. Nitrogen and phosphorus removal by wetland mesocosms subjected to different hydroperiods. *Ecological Engineering* 1: 287-307.

Carpenter, S.R., N.F. Caraco, D.L. Correll, R.W. Howarth, A.N. Sharpley and V.H. Smith. 1998. Nonpoint pollution of surface waters with phosphorus and nitrogen. *Ecological Applications* 8: 559-568.

Cottingham, P.D., T.H. Davies and B.T. Hart. 1999. Aeration to promote nitrification in constructed wetlands. *Environmental Technology* 20: 69-75.

Craft, C.B. 1997. Dynamics of nitrogen and phosphorus retention during wetland ecosystem succession. *Wetland Ecological Management* 4: 177-187.

Cronk, J.K. 1996. Constructed wetlands to treat wastewater from dairy and swine operations: A review. *Agriculture, Ecosystems, and Environment* 58: 97-114.

Gale, P.M., I. Devai, K.R. Reddy and D.A. Graetz. 1993. Denitrification potential of soils from constructed and natural wetlands. *Ecological Engineering* 2: 119-130.

Gersberg, R.M., B.V. Elkins and C.R. Goldman. 1983. Nitrogen removal in artificial wetlands. *Water Research* 17: 1009-1014

Hammer, D.A. and R.L. Knight. 1994. Designing constructed wetlands for nitrogen removal. *Water Science and Technology* 29: 15-27.

Hseih, Y.P. and C.L. Coultas. 1989. Nitrogen removal from freshwater wetlands: Nitrification - denitrification coupling potentials. In *Constructed Wetlands for Wastewater Treatment,* ed. D.A. Hammer, 493-500. Chelsea, MI: Lewis Publishers, Inc.

Jamieson, T.S. 2001. Nitrogen and phosphorus removal in a constructed wetland receiving agricultural wastewater. Unpublished M.Sc. thesis. Truro, NS: Department of Environmental Sciences, Nova Scotia Agricultural College (in cooperation with Dalhousie University, Halifax, NS).

Kadlec, R.H. and R.L. Knight. 1996. *Treatment Wetlands.* Boca Raton, FL: Lewis Publishers Inc.

Knight, R.L., V.W.E. Payne Jr., R.E. Borer, R.A. Clarke Jr. and J.H. Pries. 2000. Constructed wetlands for livestock wastewater treatment. *Ecological Engineering* 15: 41-55.

Lee, M.A., J.S. Stansbury and T.C. Zhang.1999. The effect of low temperatures on ammonia removal in laboratory-scale constructed wetland. *Water Environment Research* 71: 340-347.

Newman, J.M., J.C. Clausen and J.A. Neafsey. 2000. Seasonal performance of a wetland constructed to process dairy wastewater in Connecticut. *Ecological Engineering* 14: 181-198.

Nichols, D.S. 1983. Capacity of natural wetlands to remove nutrients from wastewater. *Journal of the Water Pollution Control Federation* 55: 495-505.

NRCS. 1991. *Technical Requirements for Agricultural Wastewater Treatment*. National Bulletin No. 210-1-17. Washington, DC: United States Department of Agriculture, Natural Resource Conservation Service.

Patrick, W.H. and K.R. Reddy. 1976. Nitrification-denitrification reactions in flooded soils and water bottoms: Dependence on oxygen supply and ammonium diffusion. *Journal of Environmental Quality* 5: 469-472.

Peterson, H.G. 1998. Use of constructed wetlands to process agricultural wastewater. *Canadian Journal of Plant Science* 78: 199-210.

Phipps, R.G. and W.G. Crumpton. 1994. Factors affecting nitrogen loss in experimental wetlands with different hydrologic loads. *Ecological Engineering* 3: 399-408.

Reddy, K.R. and E.M. D'Angelo. 1997. Biogeochemical indicators to evaluate pollutant removal efficiency in constructed wetlands. *Water Science and Technology* 35: 1-10.

Reddy, K.R., and D.A. Graetz. 1981. Use of shallow reservoir and flooded organic soil systems for wastewater treatment: Nitrogen and phosphorus transformations. *Journal of Environmental Quality* 10: 113-119.

Rochon, J.P., R. Gordon, A. Madani, V. Rodd and L. Cochrane. 1999. Seasonal influences on constructed wetlands for treatment of agricultural wastewater in Nova Scotia, Canada. ASAE Paper No. 992205. St. Joseph, MI: ASAE.

Sartoris, J.J., J.S. Thullen, L.B. Barber and D.E. Salas. 2000. Investigation of nitrogen transformations in a southern California constructed treatment wetland. *Ecological Engineering* 14: 49-65.

Sievers, D.M. 1997. Performance of four constructed wetlands treating anaerobic swine lagoon effluents. *Transactions of the ASAE* 40: 769-775.

Smith, V.H., G.D. Tilman and J.C. Nekola. 1999. Eutrophication: Impacts of excess nutrients on freshwater, marine, and terrestrial ecosystems. *Environmental Pollution* 100: 179-196.

Spieles, D.J. and W.J. Mitsch. 2000. The effects of season and hydrologic and chemical loading on nitrate retention in constructed wetlands: A comparison of low- and high-nutrient riverine systems. *Ecological Engineering* 14: 77-91.

Stevenson, F.J. 1986. *Cycles of Soil: Carbon, Nitrogen, Phosphorus, Sulfur, Micronutrients.* Toronto, ON: John Wiley & Sons, Inc.

van Oostrom, A.J. and J.M. Russell. 1994. Denitrification in constructed wetlands receiving high concentrations of nitrate. *Water Science and Technology* 29: 7-14.

Verhoeven, J.T.A. and A.F.M. Meuleman. 1999. Wetlands for wastewater treatment: Opportunities and limitations. *Ecological Engineering* 12: 5-12.

Wood, S.L., E.F. Wheeler, R.D. Berghage and R.E. Graves. 1999. Temperature effects on wastewater nitrate removal in laboratory-scale constructed wetlands. *Transactions of the ASAE* 42: 185-190.