NITRATE REMOVAL IN WETLAND MICROCOSMS

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Abstract—The effect of hydraulic loading rate (HLR; 5 to 20 cm day^{-1}) and carbon addition (1-6 g dried plant material per week) was determined at two temperatures (28°C and 35°C) in 18 flow-through sediment-water wetland microcosms that were fed nitrate-contaminated water (30 mg N L^{-1}). Nitrate removal efficiencies varied from 8% to >95%, decreasing with increasing HLRs and increasing with increasing carbon addition rates. The effect of HLR and carbon addition rate were integrated into the C:N ratio, which alone was highly predictive of nitrate removal efficiency. As nitrate removal efficiencies increased, dissolved organic carbon in the effluent also increased, as did chloroform formation potential. Nitrate-treatment wetlands may be a feasible method of remediating nitrate-contaminated groundwater. © 1998 Elsevier Science Ltd. All rights reserved

INTRODUCTION

Nitrate (NO$_3^-$) in drinking water is regulated by the U.S. Environmental Protection Agency (EPA) because it causes methanoglobanemia in infants at high concentrations. Nitrate contamination of groundwater is common in agricultural areas throughout the United States (Nielsen and Lee, 1987). In a statistical survey of drinking water quality, 2.4% of rural domestic wells and 1.2% of community groundwater supply wells exceeded the nitrate maximum contaminant level (MCL) of 10 mg NO$_3$-N L^{-1} (EPA, 1990). In Phoenix, Arizona, more than half (61%) of the 224 Salt River Project (SRP) wells exceeded the nitrate MCL in 1993 (SRP, 1993). Water from these wells cannot be used for municipal water supply unless it is diluted with low nitrate water to acceptable levels or treated to reduce nitrate levels.

Wetlands are efficient at removing nitrate from water (Gersberg *et al.*, 1983; Johnson, 1991; Kadlec and Alvord, 1989; Knight *et al.*, 1990, 1993; van Oostrom and Russel, 1994; Horne, 1995, and others). Nitrate removal in wetlands occurs through plant uptake and by denitrification. With high nitrate loading rates typical of treatment wetlands, denitrification is generally considered the dominant mechanism of nitrate loss. Wetlands have two environmental characteristics that promote denitrification: (1) the sediments are anoxic, a requisite

condition for denitrification (redox potential < 300 mV; Kadlec and Knight, 1996), and (2) plant growth provides a source of carbon fuel for denitrification. Given these characteristics, wetlands should be an excellent natural treatment system for waters contaminated mainly with nitrate, such as irrigation return flow and nitrified sewage effluent. Baker (1994) postulated that constructed wetlands may also provide a low-cost alternative for remediating nitrate-contaminated groundwater.

Typical designs for sewage treatment wetlands may not be applicable to wetlands designed to treat waters high in nitrate and low in organic matter for several reasons. First, wastewater treatment wetlands usually receive most of their nitrogen in the form of ammonium (NH$_4^+$) or organic nitrogen. These forms of N must be converted to NO$_3^-$ before denitrification can occur. The preliminary steps (mineralization; nitrification) would not be necessary for wetlands designed to treat waters in which most of the N was already in the form of nitrate. Second, much of the organic carbon needed for denitrification in wastewater wetlands is provided by the wastewater itself. In contrast, the dissolved organic carbon (DOC) concentration in groundwater is generally very low, and most of this DOC is likely to be recalcitrant (Thurman, 1985). Thus, virtually all of the carbon needed to fuel the denitrification in a nitrate treatment wetland would have to come from the plants growing in the wetland. It is therefore reasonable to postulate that carbon supply may be the limiting factor in a wetland designed for nitrate removal of low-DOC waters.

This paper describes a laboratory microcosm experiment designed to evaluate the feasibility of

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using constructed wetlands for treating nitrate-contaminated groundwater or other nitrate-contaminated, low DOC waters. The microcsoms were "fed" carbon in the form of chopped-up cattails and supplied with nitrate-contaminated water (30 mg N L^{-1}) at hydraulic loading rates (HLRs) of 5, 10, and 20 cm day^{-1}. The study shows that carbon supply appears to limit nitrate removal efficiencies and that adequate nitrate removal can be accomplished at relatively high HLRs. Elevated concentrations of dissolved organic carbon (DOC) in the effluent from nitrate treatment wetlands may be problematic from a standpoint of trihalomethane (THM) formation if the effluent from these wetlands were to be chlorinated.

METHODS

Eighteen flow-through wetland microcosms were used to simulate nitrate-treatment wetlands. The microcosms were made of Plexiglass (13 cm width × 20 cm length × 30 cm high) and provided with internal baffles to prevent short circuiting (Fig. 1). Sediment from the littoral wetland of a local lake was added to a depth of 10 cm. Outlets were placed 25 cm from the bottom of the microcosms to maintain water depths of 15 cm throughout the experiment.

Two Manostat 10-channel high speed cassette pumps were used to pump nitrate-rich source water to microcosms. The source water was tap water augmented with potassium nitrate (KNO_3) to a concentration of 30 mg NO_3-N L^{-1}. This concentration was chosen because nearly all (92%) of the 135 SRP wells that exceeded the nitrate drinking water MCL had NO_3-N concentrations between 10 mg L^{-1} and 30 mg L^{-1}. Source water was stored in a 200-L drum, which was refilled every week. The experiment was performed inside a Jam Scientific Corporation constant temperature room.

Experimental treatments were carbon addition (three rates) and HLR (three rates). Each treatment was duplicated, for a total of 18 microcosms. Carbon was added weekly in the form of dried, coarsely chopped cattails. Plant addition rates were determined based upon a review of productivity of emergent macrophytes (Westlake, reported in Wetzel, 1983). Temperate climates were found to yield 3,000–4,500 g DW (dry weight) m^{-2} yr^{-1} and tropical wetlands yielded 6,500–8,500 g DW m^{-2} yr^{-1}. Based upon this information and a consideration of the local climatic conditions, a value of 6,000 g DW m^{-2}

yr^{-1} was considered a reasonable estimate for a productive wetland in our region (Sonoran desert). A value of 2,000 g DW m^{-2} yr^{-1} was used to represent a low-production wetland, and a rate of 12,000 g DW m^{-2} yr^{-1} was used to bracket the upper limit for wetland plant productivity. These productivity values translated into additions of 1, 3, and 6 grams of dried plant material per microcosm per week. Based on published design guidelines (Kadlec and Knight, 1996; Reed et al., 1995) and a pilot study conducted in our lab by Nehrbass (1995), HLRs of 5 cm day^{-1}, 10 cm day^{-1}, and 20 cm day^{-1} were used, corresponding to hydraulic retention times of 5 days, 1.5 days, and 0.75 days, respectively. An "applied" C:N ratio for each microcosm was calculated as the ratio of plant carbon added: NO_3-N in the source water. Plant carbon content was assumed to be 50% of ash-free dry weight (Boyd and Hess, 1970). Applied C:N ratios ranged from 0.43 (1 g DW $week^{-1}$; HLR = 20 cm day^{-1}) to 10.3 (6 g DW $week^{-1}$; HLR = 5 cm day^{-1}. The flow rate from each microcosm was measured gravimetrically each week and adjusted as needed. Because of pump problems, steady flow rates to the microcosms were achieved only after six weeks. The experiment was run at 35°C for 17 weeks at which time the temperature was lowered to 28°C (room temperature) and run for an additional 12 weeks. The higher temperature represented an extreme that might be experienced in a desert climate; the lower temperature is close to the annual average temperature in Phoenix.

Microcosm effluent and source water samples were collected weekly and analyzed for nitrate concentrations using a HACH nitrate ion specific probe (model #44560). Net nitrate removal was calculated as the difference in nitrate concentrations between the microcosm effluent and source water. Ammonia in the effluent from all microcosms was measured once (week 5 in the 28°C phase) using a Hatch ammonia electrode. Anions in the effluent from all microcosms were measured once by ion chromatography at the end of the 35°C phase of the experiment (week 17).

Dissolved organic carbon (DOC) concentrations were measured with a Rosemount/Dohrman DC-180 Total Organic Carbon Analyzer on several occasions. Near the end of the 28°C phase of the experiment, we analyzed chloroform formation potential (CCl_3FP) in the effluent from each microcosm. Measured DOC values were used to determine chlorine dosing solution volumes required for CCl_3FP analysis. Chloroform was measured by Standard Methods method #5710-B. Briefly, samples were buffered to a pH of 7.0, chlorinated with an excess of free chlorine using a 5:1 ratio (wt:wt) of chlorine to DOC, and allowed to react at 25°C for seven days before being analyzed. Concentrations of ammonia in the effluent were below detection limits as determined by ion selective electrode (< 1 mg L^{-1}), so no adjustment in chlorine dose was needed to account for chloramine formation. Chloroform was extracted into methyl t-butyl ether (MTBE), which was injected into a Hewlett Packard 5890 Series II Gas Chromatograph. Source water measurements for anions revealed bromide concentrations below detection limits and preliminary runs indicated that chloroform was the only important trihalomethane (THM) formed.

Daily calibration was conducted for all analyses. For the electrode techniques, the measured slope of the calibration curve was compared with the theoretical slope before samples were analyzed. Over the course of the study, the average slope for the nitrate electrode was 57.2 ± 1.2 mV/decade, very close to the theoretical slope of 59 mV/decade.

At the end of the experiment, all plant material remaining in the microcosms was collected, dried at 105°C to determined dry weight and then ashed at 550°C to determine ash-free dry weight.

Inlet (metered pump) 30 mg NO_3-N L^{-1}

Internal baffle

Outlet (gravity)

Water

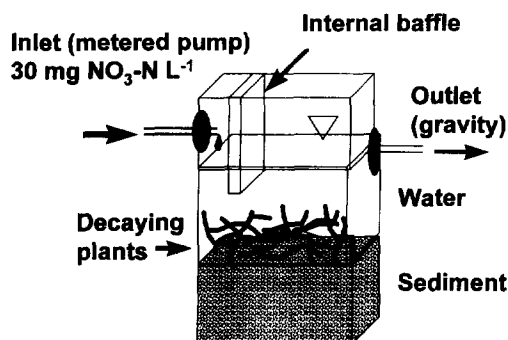**Decaying plants**

Sediment

Fig. 1. Schematic of the wetland microcosms. Sediment was collected from the littoral zone of a nearby lake. Carbon was supplied to the wetland in the form of dried, coarsely chopped cattail stems and leaves.
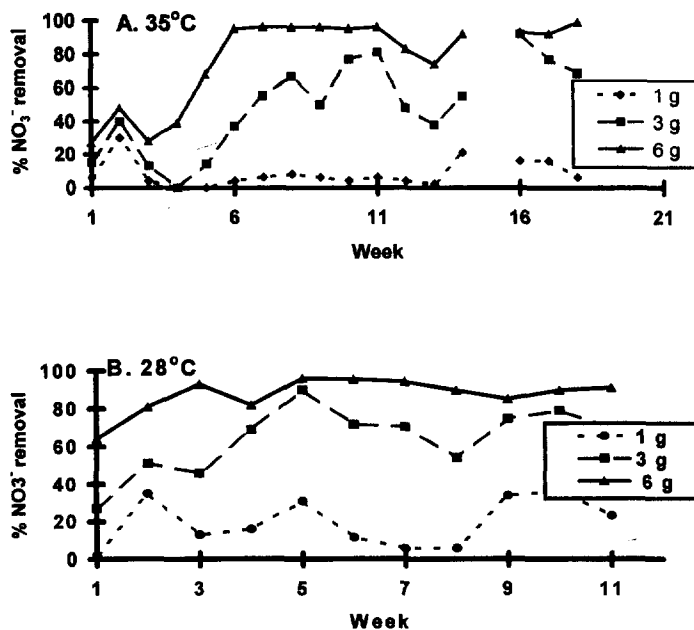
Fig. 2. Effluent nitrate concentrations in the HLR = 10 cm day^{-1} microcosms during (A) the 35°C phase of the experiment, and (B) the 28°C phase of the experiment. Carbon was supplied as chopped-up cattails at rates of 1 g DW week^{-1}, 3 g DW week^{-1}, and 6 g DW week^{-1}. Data points represent the mean of duplicate microcosms.

RESULTS

For the 35°C phase, steady-state nitrate removal was achieved by week 7 (Fig. 2A). Because the 28°C phase (actual average temperature = 27.8 ± 0.6°C) was started when the microcosms had a well-established biota, the new steady-state was achieved within only one week (Fig. 2B). Measured HLRs during steady state for the 35°C phase were (means and standard deviations) 5.5 ± 0.4, 13.0 ± 2.4, and 19.8 ± 4.6 cm day^{-1}, respectively, for the nominal 5, 10, and 20 cm day^{-1} treatments. Flowrates were less variable during the 28°C phase of the experiment because pump problems had been fixed: average measured HLRs during steady state were 5.3 ± 0.4, 11.3 ± 0.9, and 20.2 ± 1.5 cm day^{-1} for the nominal 5, 10, and 20 cm day^{-1} treatments. The average NO_3-N concentration in the source water during the 35°C phase was 29.2 ± 10.1 mg NO_3-N L^{-1}. The high variability resulted from mixing problems within the 55-gallon carboy. After this problem was resolved, the average influent NO_3^--N concentration during the 28°C phase was 28.4 ± 3.6 mg NO_3-N L^{-1}.

Nitrate removal efficiencies

For both phases of the experiment, net nitrate removal efficiencies (per cent of nitrate removed from the inflow) at each HLR increased with increasing carbon addition rates. At the highest plant addition rate (6 g DW day^{-1}), nitrate removal efficiencies were ≥90% at HLRs above 10 cm day^{-1}.

Conversely, at a given carbon addition rate, net removal efficiencies declined with increasing HLR (Fig. 3). At the lowest plant addition rate (1 g DW day^{-1}), nitrate removal efficiencies never exceeded 40%. On the average, the decrease in net nitrate removal was 25% less at 28°C than at 35°C, but the difference was not statistically significant.

In contrast, net areal removal rates (g NO_3-N m^{-2} 2 day^{-1}) increased with increasing HLR and plant addition rate at both temperatures (Table 1). Net areal nitrate removal rates during the 35°C phase were higher than those in the 28°C phase. Areal nitrate removal rates at the highest carbon addition rate were 5–10 times higher than those at the lowest carbon addition rate. At the highest carbon addition rate and HLR, nitrate removal exceeded 4 g N m^{-2} day^{-1} (40 kg N ha^{-1} day^{-1}), roughly 10 times the average rate of total N removal in free water surface treatment wetlands (Kadlec and Knight, 1996) and comparable with the highest rates observed in treatment wetlands (50 kg ha^{-1} day^{-1}; see Horne, 1995). Even at the highest denitrification rate observed in this experiment, sulfate loss in the microcosms was minimal, except at the lowest HLR and highest carbon addition rate, where an average of 29% of sulfate input was consumed. Reduction potentials of the other microcosms were apparently not low enough to allow sulfate reduction.

The dominant mechanism of net nitrate removal in the microcosms was almost certainly denitrification. Ammonia concentrations at week 8 in the 28°C phase were below the detection limit (1 mg
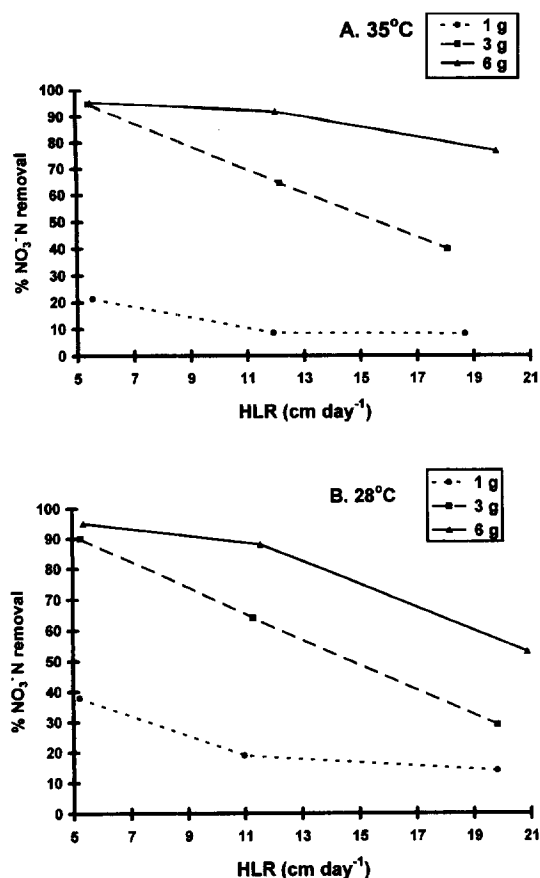
A. 35°C

B. 28°C

Fig. 3. Effect of hydraulic loading rate and carbon addition on nitrate removal efficiencies at steady-state in (A) the 35°C phase, and (B) the 28°C phase. The average standard deviations for nitrate removal efficiencies were 2.9% (at 35°C) and 2.1% (28°C).

L^{-1}) in the effluents from all 18 microcosms, indicating that ammonia was not an important endproduct of nitrate reduction. Under the conditions of the experiment, algae uptake and other removal mechanisms are unlikely to have been important. Although decaying plants contributed some soluble nitrogen during decomposition, this flux would have been small relative to the nitrate supplied by the inflow. Assuming that the cattails were 2% N on a dry weight basis (Johnson, 1991) and that ~50% of the N was never released (see discussion on residual fraction below), the maximum soluble nitrogen flux from the added plants would have been ~0.3 g m^{-2} day^{-1} (at the 6 g DW plant
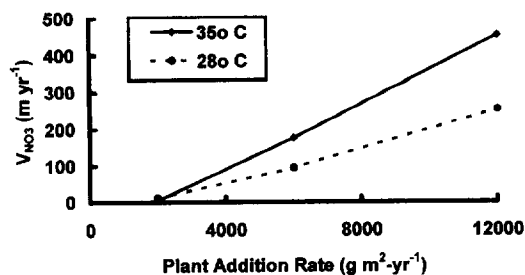

Fig. 4. Effect of carbon addition rates on V_{NO_3} at 28°C and 35°C.

addition rate), about 20% of the inflow nitrate loading at the lowest HLR (5 cm day^{-1}). At the lowest plant addition rate and highest HLR, solubilized plant N would have been < 1% of the inflow nitrate loading.

Denitrification reaction rate coefficients were determined from the equation:

$$[NO_3] = \frac{L_{NO_3}}{HLR + V_{NO_3}} \qquad (1)$$

where: L_{NO_3} = areal nitrate loading rate, mg m^{-2}-yr^{-1}; HLR = hydraulic loading rate, m yr^{-1}; V_{NO_3} = reaction rate constant, m yr^{-1}.

At both temperatures, V_{NO_3} varied linearly with carbon addition rate (Fig. 4). The effect of carbon addition on V_{NO_3} was determined by regression analysis:

At 35°C:

V_{NO_3}, m yr^{-1}

$= -183 + 0.05$ (plant addition rate, g DW m^{-2} yr^{-1})

$n = 18; \ r^2 = 0.66 \qquad (2)$

At 28°C:

V_{NO_3}, m yr^{-1}

$= -96 + 0.04$ (plant addition rate, g DW m^{-2} yr^{-1})

$n = 18; \ r^2 = 0.61 \qquad (3)$

To further examine the role of carbon supply, we integrated the effect of HLR (nitrate supply) and carbon addition rate into carbon:nitrogen (C:N) ratios. Figure 5 shows that as the C:N ratio increases, nitrate removal efficiency increases up to a C:N ratio around 5:1. At ratios >5:1, nitrate removal efficiencies were >90% and appeared to be

Table 1. Net areal nitrate removal rates (g N m^{-2} day^{-1}) in the microcosm experiment

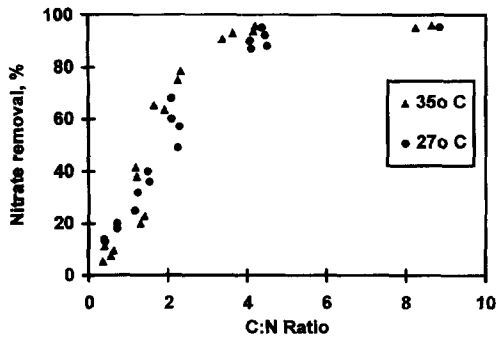| | Plant addition rate | | | | | |
| | 1 g DW week^{-1} | | 3 g DW week^{-1} | | 6 g DW week^{-1} | |
Hydraulic load	35°C	28°C	35°C	28°C	35°C	28°C
5 cm day^{-1}	0.34	0.10	1.52	0.35	1.58	0.49
10 cm day^{-1}	0.26	0.11	2.29	0.38	3.24	0.58
20 cm day^{-1}	0.43	0.14	2.14	0.28	4.43	0.54

Fig. 5. Effect of C:N ratio on nitrate removal efficiencies.

independent of further increases in the C : N ratio. At very low C : N ratios, very little denitrification occurred (Fig. 5).

We attempted to determine denitrification temperature coefficients (θ) from the equation:

$$\frac{V_1}{V_2} = \theta_{12}^{(T-T)} \tag{4}$$

The average value of θ (1.01) was not statistically different from 1.0 (no temperature effect), but the variability among treatments was sufficiently high (std. dev. for θ = 0.10) that we would not have detected a temperature effect of the magnitude generally seen for denitrification (θ = 1.05 to 1.13; Brezonik, 1994).

Fate of plant carbon in the microcosms

A partial mass balance was developed to track the fate of plant carbon added to the microcosms (Fig. 6). Residual plant carbon (residual C) was determined by harvesting and weighing the plant material remaining in the microcosms at the end of the experiment. Net DOC production was calculated by subtracting the inflow DOC mass from the outflow DOC mass. Total CO_2 production was computed as total carbon addition minus the sum of residual C plus net DOC flux. The amount of CO_2 evolved in the course of denitrification ("CO_2-N_2") was computed from measured nitrate loss rates and the equation (Stumm and Morgan, 1995):

$$1.25CH_2O + NO_3^- + H^+ \rightarrow 1.25CO_2 + \tfrac{1}{2}N_2 + 1.75H_2O \tag{5}$$
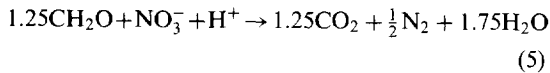
Finally, "other CO_2" was computed as total CO_2 minus CO_2-N_2. This term represents CO_2 generated by aerobic respiration and anaerobic processes other than denitrification.

Much of the plant material added to the microcosms remained as undecomposed material. The efficiency of decomposition increased with increasing carbon addition rates. At the 1 g DW week^{-1} plant addition rate, the residual fraction averaged 55% of added plant carbon, and total CO_2 evolution was 39% of added plant carbon, but at the 6 g DW week^{-1} plant addition rate, the values were

almost exactly reversed (38% residual; 55% CO_2). This is reflected as a decrease in percent residual with increasing C : N ratios (Fig. 6).

Denitrification accounted for a substantial fraction of the overall respiration in the system, averaging 37% of the total CO_2 produced (17% of added plant carbon). As would be expected, the relative importance of denitrification in total CO_2 production decreased as C : N ratios increased (from 13% to 68% of total CO_2 production), whereas other (non-denitrification) CO_2 production increased as C:N ratios increased (Fig. 7).

DOC in the effluent from the microcosms increased with increasing carbon loading and decreased with increasing HLRs (Fig. 8). Effluent DOC concentrations were much lower in the 28°C phase than in the 35°C phase of the experiment (cf. Figure 8A and B). The percentage of plant carbon added that appeared as effluent DOC did not vary
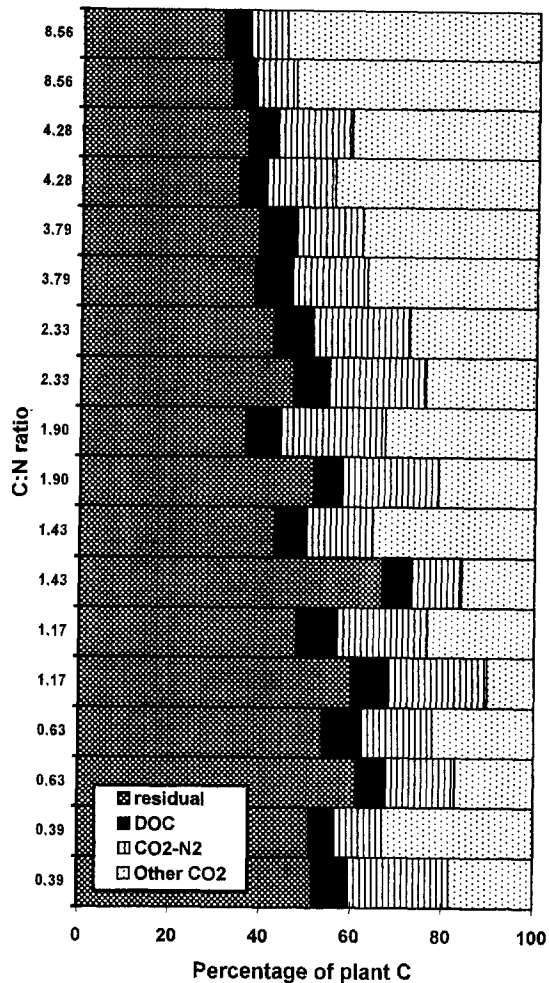

Fig. 6. Carbon mass balance for the microcosm experiment. "Residual C" is carbon remaining in plant tissue at the end of the experiment. "DOC" is the net DOC output from the microcosm, "CO_2-N_2" is carbon dioxide used in denitrification, as computed from equation (5), and "Other CO_2" is total CO_2 minus CO_2-N_2.
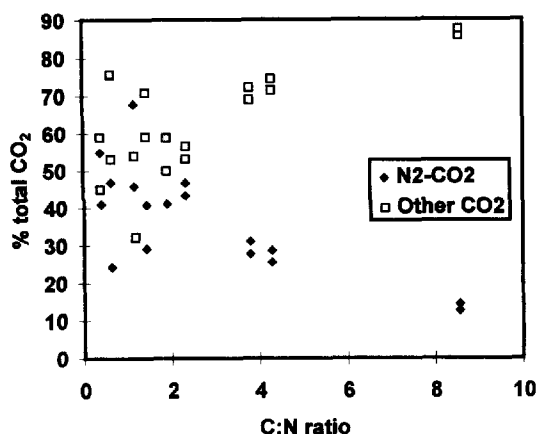
Fig. 7. Estimated fraction of total CO_2 production utilized by denitrification and other respiration processes in relation to the C : N ratio.

with carbon addition rates but increased somewhat with increasing HLRs (from 6.1% of plant carbon addition at 5 cm day^{-1} to 8.3% of plant carbon addition at 20 cm day^{-1}.

Interestingly, for a given carbon addition rate, effluent DOC changed far less in response to HLR than one would expect from simple dilution. For example, at the 3 g $week^{-1}$ plant addition rate in
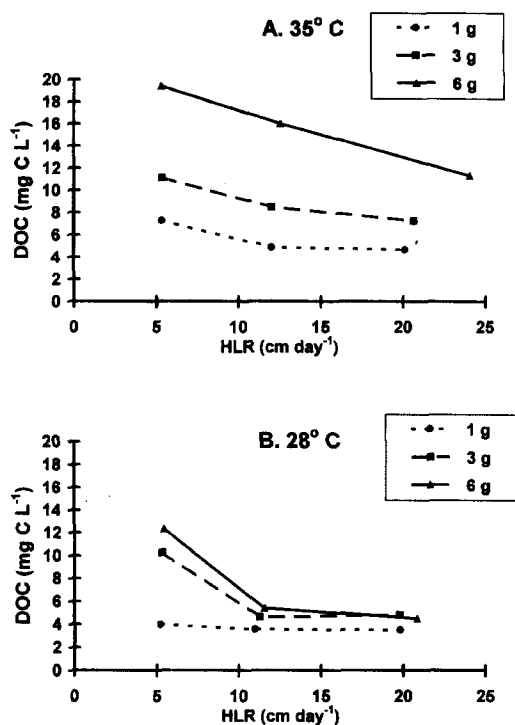

Fig. 8. Effect of HLR and carbon addition on DOC in the effluent from the microcosms (A) 35°C phase (at week 17) and (B) 28°C phase (at week 12) DOC in the source water was 3.7 mg L^{-1} during the 35°C phase and 2.8 mg L^{-1} during the 28°C phase. The average standard deviations for microcosm replicates was 1.4 mg L^{-1} (35°C, week 17) and 0.6 mg L^{-1} (28°C, week 12).

the 35°C phase (Fig. 8A), DOC declined from 12 mg L^{-1} at a HLR of 5 cm day^{-1} to 8 mg L^{-1} at a HLR of 20 cm day^{-1}, a decline of only ~30%. If DOC was being produced at the same rate in each of the microcosms with the same carbon addition rate and was conservative, one would expect a four-fold decline in DOC output as the HLR increased from 5 cm day^{-1} to 20 cm day^{-1}. We hypothesize that HLR had a smaller effect on DOC concentrations than would be expected from dilution alone because both production and decomposition of DOC were occurring simultaneously in the microcosms. Increasing the HLR (a) increased the dilution factor, which tended to lower effluent DOC, and (b) reduced the residence time and therefore the extent of DOC degradation, which tended to increase the effluent DOC. Effluent DOC represented a balance between these countervailing tendencies. The increase in percent carbon addition appearing as effluent DOC with increasing HLR supports this hypothesis.

Chloroform formation potential

We observed a roughly linear relationship between effluent DOC and $CHCl_3$ formation potential (Fig. 9). The lowest $CHCl_3$ concentrations in the effluent were comparable with concentrations in the source (tap) water, but at higher carbon loadings and lower HLRs, $CHCl_3$ concentrations increased to around 400 μg L^{-1} (Table 2). The average yield was 30 μg $CHCl_3$: mg DOC, quite similar to the average yield observed for a variety of surface waters (P. Westerhoff, per. comm.). These data suggest that effluent from nitrate treatment wetlands may have a significant capacity to produce THMs.

DISCUSSION

This study confirms the hypothesis of Gersberg et al. (1983) and others who have suggested that carbon limitation limits denitrification in wetlands receiving high nitrate loadings. Over a broad range of HLRs and carbon addition rates and under carefully controlled conditions, our study shows that the limiting C : N ratio is about 5 : 1. Net nitrate removal efficiencies increase as the C : N ratio increases to 5 : 1. At C : N ratios > 5 : 1, nitrate loss is nearly complete. A key finding of this paper is
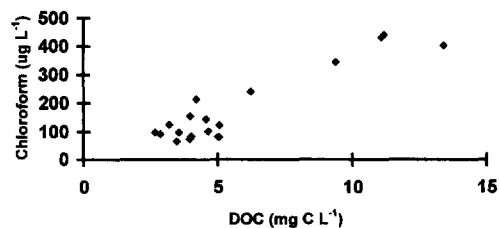

Fig. 9. Relationship between DOC and chloroform produced in the chloroform formation potential test.

Table 2. Average chloroform (CHCL$_3$) concentrations (μg L^{-1}) in the microcosms near the end of the experiment. The source water had a chloroform concentration of 95 μg L^{-1}

HLR	Plant addition rate		
	1 g DW week^{-1}	3 g DW week^{-1}	6 g DW week^{-1}
5 cm day^{-1}	81	388	422
10 cm day^{-1}	100	169	193
20 cm day^{-1}	82	92	188

that V$_{NO3}$ was directly dependent upon carbon addition rate. Current design criteria (e.g., Kadlec and Knight, 1996; Reed et al., 1995) do not recognize the effect of carbon on denitrification rate constants. Although process coefficients cannot be extrapolated directly from microcosms to natural ecosystems (see Perry et al., 1986), we hypothesize that V_{NO_3} in full scale wetlands also varies linearly with plant productivity (i.e., carbon supply).

The importance of carbon supply on denitrification efficiency also suggests that the performance of a nitrate-treatment wetland would be enhanced by increasing plant growth, and hence carbon supply. This could be done by optimizing the physical habitat for maximum plant productivity, fertilizing the plants with phosphorus (nitrogen would be oversupplied in any nitrate treatment wetland relative to plant requirements), or cutting the plants, leaving the residual in place.

Nitrate treatment wetlands may be feasible for treating nitrate-contaminated groundwater. The main limitation for a nitrate-treatment wetland intended to treat water for municipal use may be elevated concentrations of DOC in the wetland effluent. When reacted with chlorine, this DOC would react to form THMs, as we demonstrated. One approach to minimize the potential DOC problem would be to optimize the two treatment goals of low nitrate and low DOC. Because nitrate-contaminated groundwaters often have nitrate levels only a few times higher than the nitrate MCL, removal efficiencies may not have to be particularly high. As noted earlier, 92% of the SRP wells that exceed the nitrate MCL have nitrate concentrations < 30 mg L^{-1}, which means that treatment efficiencies would generally not have to exceed 70%. For a temperature of 28°C and a plant productivity rate of 6,000 g DW m^{-2} yr^{-1}, V_{NO_3} calculated from equation 3 would be 80 m yr^{-1}. Using this V_{NO_3} in equation 1 yields a HLR of 10 cm day^{-1} to achieve 70% nitrate removal. Figure 8B shows that at this HLR and productivity level (the 3 g/week plant addition rate), effluent DOC would be around 6 mg L^{-1}. This value represents a net increase of only ~3 mg L^{-1} compared to the inflow. Even so, concentrations of THM precursors may be too high to use wetland effluent in a conventional surface water treatment plant without some additional DOC removal (e.g., by enhanced flocculation).

One potential use for nitrate treatment wetlands would be to renovate aquifers (Baker, 1994). Used

in this way, groundwater would be pumped up to the wetland, treated in the wetland, and then recharged to the aquifer through percolation ponds (Fig. 10). For a loamy sand, long-term percolation rates ~100 m yr^{-1} would be expected (H. Bouwer, per. comm.), around three times higher than annual HLR of the hypothetical wetland discussed above (10 cm day^{-1} = 37 m yr^{-1}), indicating that area of the infiltration basins would generally be less than that of the wetland. For soils with finer texture, the infiltration basins may be as large or larger than the wetland. Infiltration through the soil would remove particulates, including pathogens, and would likely remove some of the DOC produced by the wetlands. A field test for a wetland-recharge system in which the inflow is nitrate-contaminated canal water is underway at Avondale, Arizona (Thompson and Bachman, 1994). The goal of this project is to obviate the need for a drinking water treatment plant.

CONCLUSIONS

Nitrate removal in the wetland microcosms varied in response to carbon addition and HLR. The data suggest that the factor controlling denitrification is the C:N ratio, which can be manipulated by varying either carbon loading or the HLR. Based on observations from 18 microcosms, the critical C:N ratio appears to be 5:1. The denitrification rate constant, V_{NO_3}, was directly proportional to carbon supply. DOC production from the microcosms was 5% to 8% of the plant carbon additions, and DOC concentrations were inversely related to nitrate removal efficiencies. The chloroform formation potential was directly related to DOC, with an average yield of 30 μg CHCl$_3$ per mg DOC.
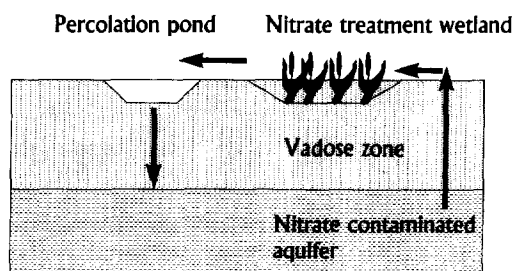

Fig. 10. Schematic of a nitrate treatment wetland and percolation basin to remediate a nitrate-contaminated aquifer.

Nitrate treatment wetlands could be used to remediate nitrate-contaminated aquifers. The wetland would remove nitrate but produce some DOC. Recharging the treated groundwater through percolation ponds would remove particulates and would probably remove much of the DOC produced in the wetland. Nitrate treatment wetlands would be most cost effective where the groundwater is shallow, land prices are low, and the climate is moderate.

REFERENCES

Baker L. A. (1994) Constructed Wetlands for Water Quality in the Arid West? *Proc. 1994 Conf. Universities' Council on Water Resources (UCOWR)*, Big Sky, MT, August, 1994.
Bouwer H. USDA-ARS, U.S. Water Conservation Laboratory, 4331 E. Broadway Rdg, Phoenix, Arizona, 85040, U.S.A.
Boyd C. E. and Hess L. W. (1970) Factors influencing shoot production and mineral nutrient levels in Typha latifolia. *Ecology* **51**, 296–300.
Brezonik P. L. (1994) *Chemical Kinetics and Process Dynamics in Aquatic Systems*. Lewis Publishers, CRC Press, Inc. Boca Raton, Fla. 754 pp.
EPA (1990) *National Pesticide Survey: Project Summary*. U.S. Environmental Protection Agency, Washington, D.C.
Gersberg R. M., Elkins B. V. and Goldman C. R. (1983) Nitrogen removal in artificial wetlands. *Water Res.* **17**, 1983, 1009–1014.
Horne A. J. (1995) Nitrogen removal from waste treatment pond or activated sludge plant effluents with free-surface wetlands. *Wat. Sci. Tech.* **33**, 341–351.
Johnson C. A. (1991) Sediment and nutrient retention by freshwater wetlands: effects on surface water quality. *Critical Reviews in Env. Control* **21**, 491–565.
Kadlec R. H. and Alvord H. (1989) Mechanisms of water quality improvement in wetland treatment systems. *Wetlands: Concerns and Successes*, American Water Works Association, 489–498.

Kadlec R. H. and Knight R. L. (1996) *Treatment Wetlands*. Lewis Publishers, CRC Press, Boca Raton, Fla. 893.
Knight R. L. (1990) Chapter 9: Wetland systems. In: *Natural Systems for Wastewater Treatment*, Manual of Practice FD-16, Water Pollution Control Federation, 211–259.
Knight R. L., Ruble R. W., Kadlec R. H. and Reed S. (1993) Wetlands for wastewater treatment: performance data base. Moshiri G. A. (ed.). *Constructed Wetlands for Water Quality Improvement*, Lewis Publishers, Ann Arbor.
Nehrbass P. (1995) Nitrogen removal using simulated wetlands. Undergraduate research project, Department of Civil and Environmental Engineering, ASU.
Nielsen E. G. and Lee L. K. (1987) *The Magnitude and Costs of Groundwater Contamination from Agricultural chemicals: A National Perspective*. AGES870318. National Resources Economics Division, Economic Research Service, U.S. Department of Agriculture, Washington, D.C.
Perry T. E., Baker L. A. and Brezonik P. L. (1986) Comparison of sulfate reduction rates measured in laboratory microcosms, field mesocosms, and *in situ* in Little Rock Lake, Wisconsin. *Lake and Reservoir Management*, Volume 2, edited by G. Redfield, J. F. Taggart, L. M. Moore, North American Management Soc, 309–312.
Reed S. C., Crites R. W. and Middlebrooks E. J. (1995) *Natural Systems for Waste Management and Treatment*. 2nd Ed. McGraw Hill. 433.
SRP (1993) *Annual Water Quality Report*. Salt River Project, Operational Support Division, Phoenix, Arizona.
Stumm W. and Morgan J. J. *Aquatic Chemistry*, 3rd edition. Wiley-Interscience, New York, 1022.
Thompson T. J. and Bachman S. B. (1994) Wetlands treatment and artificial recharge of Colorado River water in Avondale, Arizona. *Artificial Recharge of Groundwater*, *Proceedings*, ASCE Irrigation and Drainage Division, July 17–22, 1994, Walt Disney World Swan, Florida.
Thurman M. (1985) *Organic Geochemistry of Natural Waters*. Nijhoff/Junk, Boston, MA. 497.
van Oostrom A. J. and Russel J. M. (1994) Denitrification in constructed wastewater wetlands receiving high concentrations of nitrate. *Water Sci. Technol.* **29**, 7–14.
Westerhoff P. Department of Civil and Environmental Engineering, Arizona State University, Tempe, Arizona 85287-5306, U.S.A.
Wetzel R. G. (1983) *Limnology*. Saunders College Publishing, Harcourt Brace Jovanovich College Publishers, New York. 767.