Denitrification in free water surface wetlands receiving carbon supplements

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Abstract Wetlands may be used as fixed film reactors for removal of NO_3-N in wastewater. Two 1 hectare free water surface wetlands were constructed for denitrification of nitrified effluent from intermittent sand filters in Connell, Washington. The wetlands were designed to remove NO_3-N from wastewater prior to land application. The design flow as 5300 m^3/d (1.4 mgd). Primary effluent from a potato processing facility was used as a carbon supplement for detnitrification. Addition of primary effluent (COD = 2800 mg/L) resulted in a COD:NO_3-N mass load ratio that ranged from 10 to 25. The total hydraulic retention time in the two wetlands varied from 1–2 days in the summer and winter. The NO_3-N load ranged from 10 to110 kg/ha d. The NO_3-N mass removal rate ranged from 50 to 99% of the influent load. During the five months of data presented, January to May 1998, average monthly effluent NO_3-N was 1.6 mg/L; monthly average influent NO_3-N was 20.5 mg/L. An average of >85% of the NO_3-N influent load was removed. The NO_3-N removal rate coefficient (K_{20} = 358 m/yr) was higher than that measured in wetlands without carbon supplements and was independent of temperature above 12°C.

Keywords Constructed wetlands; denitrification; potato processing wastewater; nitrogen removal

Introduction

Constructed wetlands for wastewater treatment are used around the United States for treating municipal and industrial wastewater. Over the last 20–30 years there has been world-wide research, development and implementation of wetlands for all aspects of wastewater treatment (Cooper, 1999; Kadlec and Knight, 1996; Reed *et al.*, 1995).

Wetlands are naturally effective "reactors" for denitrification since they are generally anoxic, contain large amounts of surface area, and have a naturally occurring population of denitrifying bacteria. The plant denitrus and stems that accumulate in the wetlands provide high amounts of surface area for denitrification. Denitrifying cateria also reside in the wetland soil and denitrify nitrate that diffuses into the soil. In general, the critical design constraints for denitrification in wetlands are the influent mass load, carbon avaialbility, and temperature (Metcalf and Eddy, 1991).

In the absence of dissolved oxygen, the aerobic heterotrophic bacteria utilize nitrates instead of oxygen in their metabolism. The bacterial populations reduce the nitrates to nitrites and then to nitrogen gas according to the following denitrification stoichiometric equation.

$$10NO_3 + C_{10}H_{19}O_3N \rightarrow 5Nv2 + 10CO_2 + 3H_2O + NH_3 + 10OH^- \quad (1)$$

The denitrifying bacteria utilize organic compounds as electron donors in denitrification. Equation 1 shows a "model" septic tank wastewater compound as the carbon source for denitrification. Denitrifying bacteria use a variety of carbon sources. The rate of denitrification will proceed faster with the simple organic carbon compounds (Lee, 1984). Methanol is a common but expensive source of carbon. Other sources that may be used are effluent from municipal or industrial primary clarifier and acetic acid. This paper discusses the use of effluent from a primary clarifier treating potato processing wastewater.

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Summary of constructed wetland/natural system design

A wastewater treatment system consisting of a series of integrated natural systems is located in the northwestern United States in the central portion of Washington State, in Connell, Washington. The integrated natural system consists of wetlands, intermittent sand filters, and a storage lagoon. The design wastewater flow is 5300 m^3/d (1.4 mgd) of primary clarifier effluent from a potato processing facility. The average influent water quality is shown in Table 1. The free water surface wetlands prior to the intermittent sand filters remove COD, TSS and some NH_4-N (Burgoon *et al.*, 1999). The intermittent sand filters nitrify the wastewater and further reduce COD and TSS. Following the sand filters are 2 free water surface wetlands designed to reduce the NO_3-N prior to storage and land application of treated wastewater. This paper will focus on the performance of the denitrifying wetlands receiving carbon supplementation via slipstream of primary effluent. The denitrifying wetlands consisted of two 1 hectare (2.4 acre) cells in series, designated DN1 and DN2 (Figure 1). The hydraulic retention time in the denitrification wetlands varied from 1 to 2 days in the summer (depth = 30 cm) and winter (depth = 45 cm).

The free water surface wetlands were constructed and planted with cattail *Typha latifolia* and bulrush *Scirpus* sp. in summer and fall 1995. Plants were acclimated to wastewater and then loading with full strength wastewater began in September of 1996. The data set presented was collected from January 25 through May 30, 1998. Samples were collected 1 to 3 times per week from the sand filter effluent, and from the effluent of each of the denitrification wetland cells. Flow meters monitored influent flow to the sand filters and the flow rate of the primary clarifier effluent used as the supplemental carbon source. Flow balances and calculation of removal rate coefficients for the wetlands do not account for evapotranspiration or precipitation.

Results and discussion

Carbon supplementation

The slipstream of primary effluent was mixed with the effluent from the intermittent sand filters prior to entering the wetlands. Monthly average concentrations of COD, TKN,

Table 1 Monthly average concentrations (mg/L) and characteristics of primary effluent

Month	T, °C	pH	COD	TKN	NH_4-N	TSS
January 98	34.5	5.5	3489	191	120	265
February	33.0	5.9	2950	169	109	Na
March	33.7	6.1	3125	191	112	Na
April	31.7	6.0	2104	104	56	Na
May 98	Na	Na	2488	152	95	Na

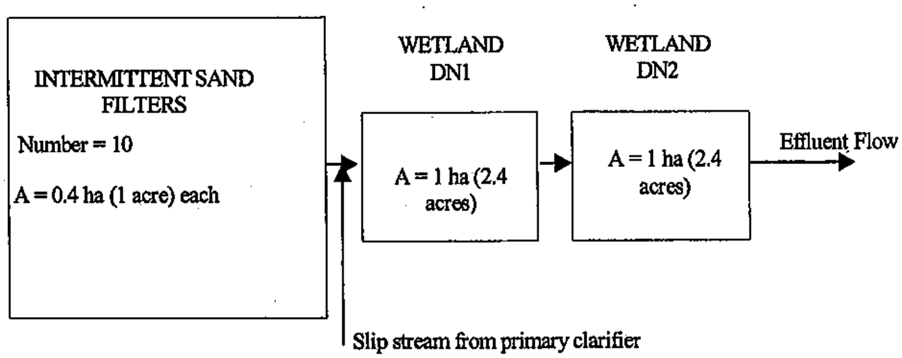

Figure 1 Basic plan view of intermittent sand filters and denitrification wetlands

NH_4-N, TSS in the slipstream are shown in Table 1. The hydraulic loads of nitrified sand filter effluent and slipstream are presented in Table 2.

The resulting COD and NO_3-N mass loads are also shown in Table 2. The COD mass load does not include COD in the nitrified effluent, since it was considered unusable by the denitrifying bacteria if it resisted degradation in the previous anaerobic and aerobic treatment processes. The COD/NO_3-N mass load ratio varied between 13 and 28. This was influenced by the influent hydraulic loads, concentration and, operator changes in the field.

Recommended COD/NO_3-N ratios for activated sludge systems may range from 2–10 depending on the carbon source (Lee, 1984; WPCF, 1983). The ratio used in the denitrifying wetlands was much higher than the ratio of 4–8 reported for subsurface flow wetlands (Gersberg *et al.*, 1984). The COD was clearly in excess since COD mass removal was always incomplete (Table 2); also the wetland effluent COD was always greater than the sand filter effluent COD (data not shown).

Water quality

Effluent concentrations of COD, NH_4-N, NO_3-N and organic nitrogen from the wetlands are shown in Figures 2–5. The influent concentration for all constituents is flow weighed based on flow into and concentration out of the intermittent sand filters, and flow and concentration of slipstream. Addition of the primary clarifier effluent to the sand filter effluent elevated the influent concentration of all constituents except NO_3-N. The warm primary effluent increased the influent temperature in the first wetland. The temperature approaches the original temperature of the sand filter effluent when it is discharged from the second wetland (Figure 6).

The majority of the COD was removed in wetland DN 1 (Figure 2). The average mass COD removal for DN1 was 1156 kg/ha d versus 245 kg/ha d for DN2. The lower rate of removal in the second wetland may be due to a less available form of COD for the denitrifying bacteria. The effluent COD from CN2 was always greater than the COD of the sand filter effluent. This implied that the wetlands always had excess carbon for denitrification. The effluent concentration decreased as the water warmed, implying a significant effect of temperature on COD removal (Figures 2 and 6).

The majority of the NO_3-N was also removed in wetland DN 1 (Figure 3). The lower rate of removal in DN2 was due to the lower loads and may be due to a less available form of COD for the denitrifying bacteria (Figure 3 and Table 2).

The two wetlands combined, received a maximum average NO_3-N load of 58 kg/ha d in April. The loads and removal rates to DN1 were significantly higher than the total wetland area. The average mass NO_3-N removal for DN1 was 65 kg/ha d versus 18 kg/ha d for DN2.

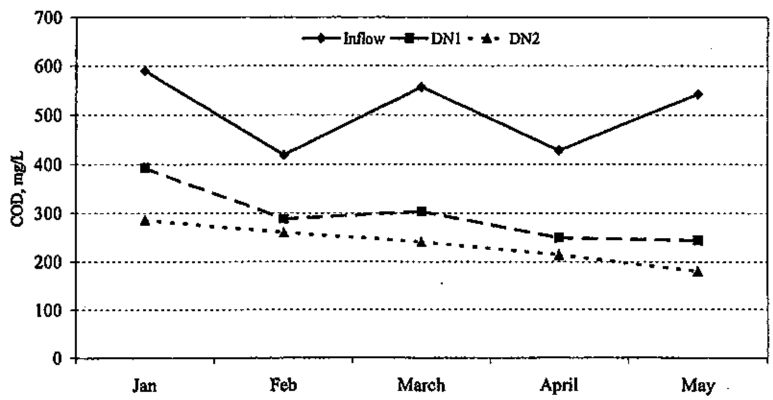

Figure 2 COD removal from denitrification wetlands

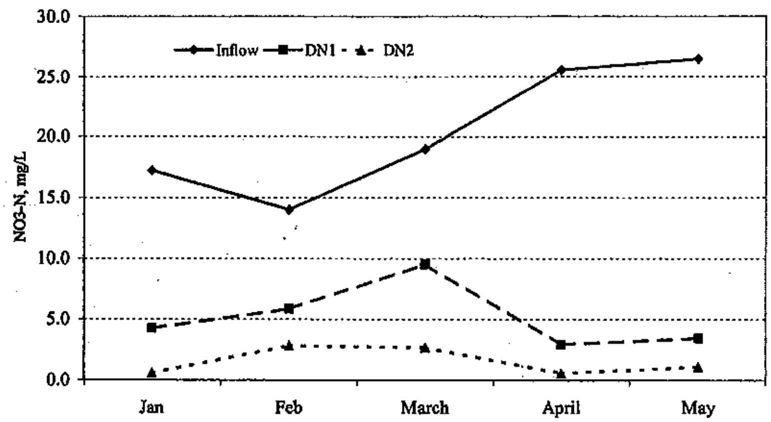

Figure 3 NO_3-N removal in denitrification wetlands

The weekly loads to DN1 were greater than 111 kg/ha d with a peak of 164. The peak removal rate for DN1 was 161 kg/ha d. Throughout April and May, an average of 89% of the NO_3-N load was removed in DN1. In comparison an average of 48% of the NO_3-N load to DN2 was removed during April and May.

The reduced nitrate removal in February and March, less than 61% removal (Table 2), can not be explained but may have been due to inconsistent loading of COD to the W4s. Averaging loads results in rates that appear similar but may actually have been applied inconsistently over the time period. It is not unusual to review the daily data and see very high daily loads followed by several days of no flow. Often this is beyond the control of the operator, such as when the processing plant shuts down, resulting in limited supply of carbon supplement. We assume that the performance in April and May was indicative of improved operations.

Denitrification is expected to increase pH due to alkalinity production during NO_3-N reduction. This was indicated by the increase in pH after passing through each denitrification wetland. The pH of the effluent from the sand filter, DN1, and DN2 averaged 6.9, 7.08, and 7.22, respectively. This pH increase may also have been due to minimeralization of the organic matter in the slipstream (Burgoon *et al.*, 1999).

Addition of the slipstream as a carbon supplement resulted in addition of organic and NH_4-N nitrogen. The influent concentrations are shown in Figures 4 and 5. The average increase, above the sand filter effluent, was 8 mg/L NH_4-N and 3 mg/L of organic nitrogen. Both forms of nitrogen were removed in the DN wetlands. During the first 4 months NH_4-N removal was significantly higher in DN1 than DN2. In the last month the removal was greater in DN2 due to increased mineralization of organic N in DN1 in May (Figures 4 and 5). The net addition of NH_4-N after treatment in the DN wetlands was 4 mg/L. The net

Table 2 Monthly average hydraulic and mass loads to the denitrification wetlands

1998	HLR, cm/d Sand filter eff.	slipstream	COD, kg/ha d load	removal	% COD removal	NO_3-N, load	kg/ha d removal	COD /NO_3-N	% NO_3-N removal
January	24.0	2.6	924	790	85%	41	38	23	93%
February	23.8	1.8	538	425	79%	32	20	17	61%
March	32.0	4.0	1207	813	67%	55	31	22	57%
April	23.5	3.3	720	461	64%	58	56	13	96%
May	20.5	3.5	894	785	88%	41	39	28	96%

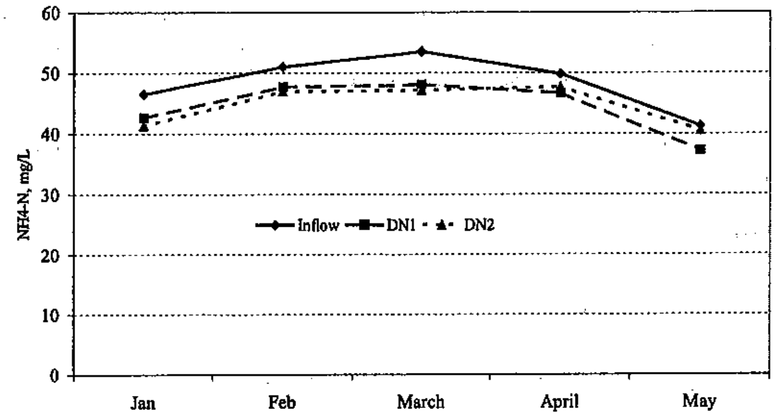

Figure 4 NH_4-N removal in denitrification wetlands

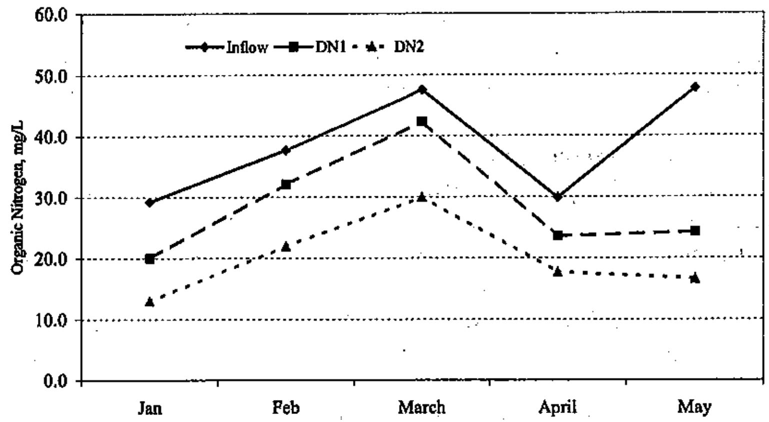

Figure 5 Removal of organic nitrogen in the denitrification wetlands

removal of NO_3-N averaged 22 mg/L. Therefore net N removal in the DN wetlands averaged 5.5 – NO_3-N removed per NH_4-N added in the carbon supplement.

Design models for denitrification wetlands

The first order area model, solved for effluent concentration (Kadlec and Knight, 1996) is shown below:

$$C_{out} = C^* + (C_{in} - C^*)exp(-kA/0.0365Q) \qquad (2)$$

C_{out} = expected effluent concentration
C_{in} = influent concentration
C^* = background concentration in wetland (lowest concentration achievable), 0.0 mg/L for NO_3-N
k = first order areal rate constant, m/yr
Q = flow rate, m^3/d

Equation 2 is used for estimation of removal rate coefficients for COD, organic nitrogen, and NO_3-N. Use of equation 2 for estimating removal coefficients for nitrogen species results in "apparent" removal rate coefficients. A sequential model accounting for mineralization of organic nitrogen and nitrification of NH_4-N (Kadlec and Knight, 1996) is used to calculate the NH_4-N removal rate coefficient. The sequential removal model estimates a k_{20} = 63 versus an "apparent" k_{20} = 1 m/yr. The sequential rate coefficient is a better estimate of NH_4-N dynamics because it accounts for the internal production and loss of NH_4-N.

The effect of temperature is compensated for by use of the van't Hoff-Arrnhenius relationship (eqn. 3).

$$k_T = k_{20}\Theta^{(T-20)} \tag{3}$$

k_{20} = areal removal rate constant at 20°C
Θ = temperature correction efficient
k_T = areal removal rate coefficient for design water temperature

Apparent removal rate coefficients for both DN1 and DN2 combined were calculated using equation 2 and 3. It is important to note that the denitrification process was not influenced by temperature during April and May when the temperature was above 12°C. Denitrification did appear to be inhibited in the colder weather. The effect of temperature was most pronounced in DN1, where NO_3-N and COD loads were the greatest.

Comparison to wetland treatment systems

Models currently available for denitrification in wetlands assume that denitrification will be fueled by the organic matter available from decaying organic matter generated in the wetland. The annual production of above ground biomass regenerates the carbon source each year. The net amount of carbon available from plants limits the total nitrate that can be removed in the wetland. The data base for free water surface wetlands in Kadlec and Knight (1996) is valid only for nitrate loads less than about 0.3 g NO_3-N/m^2 d. Based on a conser-

Table 3 Summary of design coefficients for denitrification wetlands

Parameter	k_{20}, m/yr	C*, m/yr	Theta
COD	150	100	1.04
Org N	159	10	1.05
NH_4-N	63	0.25	1.03
NO_3-N			
< 12°C	381	0.0	1.01
> 12°C	358	0.0	1.00

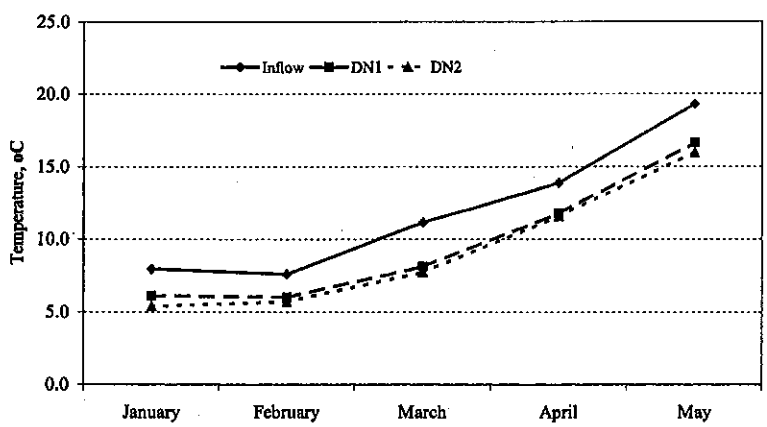

Figure 6 Temperature of water in the denitrification wetlands

vative estimate for primary productivity in a wetland (Reed *et al.*, 1995; Wetzel, 1983) and a limiting C/N ratio of 5 required for denitrification (Ingersoll and Baker, 1998), approximately 0.3 g/m^2d of NO_3-N can be denitrified with available plant biomass.

Although there is very little information available on supplementation of wetlands with organic matter to enhance denitrification, research has shown that denitrification rates are significantly different in wetlands with and without supplemental carbon (Kadlec *et al.*, 1997; Kadlec and Knight, 1996; Gersberg *et al.*, 1984). Carbon may be supplemented by providing ground plant matters (Gersberg *et al.*, 1984; Bachand, 1996), methanol or primary sewage effluent (Gersberg *et al.*, 1984). Kadlec and Knight (1996) calculated apparent removal rate coefficients for mulched subsurface flow wetlands of 74 m/yr, and 215 m/yr for substrate flow wetlands supplemented with methanol.

Gersberg *et al.* (1984) achieved greater than 90% denitrification with a COD/NO_3-N mass loading ratio of 6.8. Lower ratios will improve net nitrogen removal and will depend on metabolic efficiency of the denitrifying bacteria using the carbon source. Other studies with industrial waste stream as carbon supplement for activated sludge have shown greater than 95% NO_3-N reduction with ratio less than 10 (Water Pollution Control Federation, 1983). Comparison to other systems implies that carbon supplementation to the DN wetlands was in excess.

This full-scale system has clearly shown that wetlands can be used effectively for removal of NO_3-N. When supplemented with exogenous carbon the removal rates are an order of magnitude higher than those seen in natural wetlands.

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