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DESIGN CONSIDERATIONS AND APPLICATIONS FOR WETLAND TREATMENT OF HIGH-NITRATE WATERS

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ABSTRACT

Nitrate contamination is a serious problem worldwide. By providing an ample supply of carbon and an anaerobic environment, wetlands are an valuable low technology for treating nitrate-contaminated waters with low organic carbon concentrations. Denitrification is apparently limited by the C:N ratio, with ratios > 5:1 resulting in >90% nitrate removal efficiencies. The denitrification rate constant, V_{NO3}, varies in direct proportion to carbon supply. Several novel or emerging applications of wetlands include renovation of nitrate-contaminated aquifers (a pump-and-treat strategy), denitrification of nitrified sewage effluents, and treatment of irrigation return flows. Treatment of dual sources is also discussed. In arid regions with limited supplies of high quality water, nitrate treatment wetlands may play a significant role in the development of water resources.

KEYWORDS

Constructed wetlands, denitrification, nitrate, recharge, irrigation, groundwater.

INTRODUCTION

Constructed wetlands have gained wide acceptance as a practical means for treating wastewater, particularly for small towns in located in rural areas where land is inexpensive. In the United States, most treatment wetlands receive secondary effluent (most often from lagoons) that is fairly poorly treated (average 5-day BOD = 39 mg/L; Knight et al., 1992). This paper examines the potential for using wetlands to treat water that is contaminated with nitrate but otherwise of high quality. These applications are important because nitrate is a widespread pollutant that causes eutrophication and poses a direct human health risk. Nitrogen is one of the two nutrients that most commonly cause eutrophication and is generally considered the limiting nutrient in estuaries and oceans (Thomann and Mueller, 1987). Nitrate also causes methemoglobinemia in human infants at elevated concentrations and is therefore regulated in drinking water [the U.S. Maximum Concentration Limit (MCL) is 10 mg NO_3-N/L]. Worldwide, about one third of the total nitrogen loading from the worlds rivers is of anthropogenic origin (Meybeck, 1982); sources include agricultural drainage waters, sewage, urban runoff, and precipitation. In the United States, the *median* total inorganic nitrogen concentrations in streams draining watersheds that were >90% agricultural was 5 mg/L (Omernik, 1977). Nationwide, the trend in nitrate concentrations in rivers has been generally upward (Smith, 1987).

Nitrate migrates easily into aquifers because it is highly mobile in soils. Nitrate contamination of groundwater is most common in regions with heavily fertilized agriculture, shallow groundwater, and porous soils (Nielsen and Lee, 1987). In the United States, about 3 million people are served by community supply wells that exceed the nitrate MCL (EPA, 1990). Groundwater nitrate contamination is widespread and apparently becoming worse in several other parts of the world (Nash, 1993)

The “best available technologies” for removing nitrate from drinking water are reverse osmosis and ion exchange (Goodrich et al., 1991). Treating nitrate-contaminated water by these methods is expensive and produces brines that require proper disposal. In most areas with well-developed municipal water systems, the main strategy for dealing with nitrate-contaminated aquifers has been to find alternative sources. In a few cities (such as Phoenix, Arizona), high-nitrate water is blended with low-nitrate surface waters in canals that transport municipal source water. Avoidance and blending strategies are likely to be inadequate long-term solutions. Municipal wastewater can be treated by nitrification/denitrification to reduce loadings of nitrogen to surface waters, but this too is expensive. Thus, the nitrate problem is one of the most pervasive and difficult issues facing environmental engineers.

Treating nitrate-contaminated waters using wetlands is an attractive alternative that is only now beginning to be exploited (Gersberg et al., 1983; Baker, 1994; Horne, 1995; Ingersoll and Baker, 1998). Although there is a considerable mystique associated with wetlands, there is little doubt that they are efficient at removing nitrate from water (Gersberg et al., 1983; Kadlec and Alford, 1989; Horne, 1995, and others). Nitrate removal in wetlands occurs through plant uptake and denitrification. With high nitrate loading rates denitrification is 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 carbon “fuel” for denitrification. These characteristics make wetlands an excellent treatment system for nitrate-contaminated waters. In the rest of this paper, I will first examine design considerations for nitrate treatment wetlands and then examine the potential for several novel and emerging applications of this technology.

DESIGN CONSIDERATIONS FOR NITRATE-TREATMENT WETLANDS

There are no published design criteria for wetland treatment systems intended specifically to treat nitrate-contaminated water that is otherwise uncontaminated. Consideration of the basic biogeochemical processes supported by limited experimental work shows that the optimal design of a wetland intended to remove nitrate from otherwise uncontaminated water would be considerably different from a wetland designed to polish low-quality sewage effluent.

The first consideration is the nature of the source water to be treated. Most wastewater treatment wetlands receiving effluents from lagoons or other secondary treatment systems have very high levels of reduced nitrogen (ammonia and/or organic nitrogen), typically in the range of 15-20 mg N/L. Nitrate concentrations are typically low (< 5 mg/L) unless the system is specifically designed for nitrification. In contrast, most of the nitrogen entering a wetland receiving nitrate-contaminated groundwater, nitrified wastewater, or irrigation return flow would be in the form of nitrate. For a non-nitrified wastewater effluent the sequence of nitrogen processes is: mineralization of particulate organic N (for lagoon effluents with algae), nitrification of ammonium, and denitrification (Figure 1). For a wetland receiving nitrogen primarily in the form of nitrate, the only important process is denitrification. Thus, to achieve a given nitrogen removal efficiency, a wetland receiving nitrogen in the form of nitrate could have a much higher hydraulic loading rate (HLR) than a wetland receiving an equivalent concentration of nitrogen in the form of reduced species, all other factors being equal.

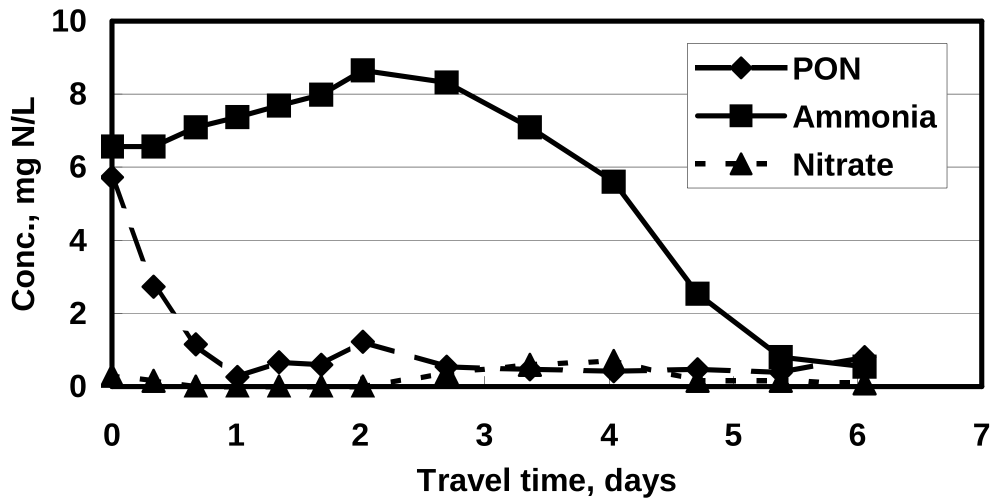

Figure 1. Nitrogen transformations in a wetland (Kingman, Arizona) receiving lagoon effluent during October, 1996. Inflow N was primarily in the form of particulate organic nitrogen (PON) and ammonium. PON was quickly mineralized to ammonium. Ammonium was slowly nitrified; the resulting nitrate was then removed efficiently by denitrification. On this sampling date, nitrate never exceeded 1 mg NO_3-N/L (Gerke et al., in press).

The second consideration is carbon supply. For a treatment wetland receiving poorly-treated secondary effluent, some of the carbon required for denitrification is contained in the effluent. In contrast, nitrate-contaminated groundwater would normally have almost no labile carbon to fuel denitrification: 100% of the carbon required for denitrification would have to come from the wetland itself.

Maximum potential rate of denitrification

An upper limit on the potential rate of denitrification can be estimated as follows: an upper limit on the annual productivity of a tropical wetland is probably on the order of 8,500 g dry weight/m^2-yr (Westlake, in Wetzel, 1983). Assuming a carbon content of 40% (Boyd and Hess, 1970) and a refractory fraction of 11-50% (Jewell, 1971) leaves ~1,700-3,000 g C/m^2-yr that is potentially available for denitrification. From the stoichiometry (Stumm and Morgan, 1995):

$$1.25CH_2O + NO_3^- + H^+ \rightarrow 1.25CO_2 + \tfrac{1}{2} N_2 + 1.75H_2O \qquad (1)$$

it follows that denitrification of one mole of nitrate requires 1.25 moles of carbon. This translates into a maximum theoretical rate of 1,600-2,800 g N/m^2-yr, or 44-77 kg N/ha-day. This potential rate is probably never realized, because most wetlands have lower productivity and some plant material is always “lost” by aerobic respiration. Net nitrate removal rates of 28-50 kg/ha-day have been observed in experimental wetlands receiving nitrified effluent (Gersberg et al., 1983; Horne, 1995); we measured a maximum rate of 40 kg/ha-day in wetland microcosms (Ingersoll and Baker, 1998). In contrast *total* N removal in wetlands designed to treat sewage effluent averages around 4 kg/ha-day (Kadlec and Knight, 1996). The lower N removal rates in wetlands receiving effluent reflects that fact that these wetlands must convert organic N or ammonium to nitrate before denitrification can occur.

Effect of C:N ratio

Treating high-nitrate, low BOD waters requires degradable carbon supplied by wetland plants (Gersberg et al., 1983; Reed et al., 1995). In a microcosm study with carefully controlled additions of carbon (chopped up cattails) and nitrate (hydraulic loading rates of 5, 10, and 20 cm/day, all with 30 mg NO_3-N/L) we found that the optimum C:N ratio was 5:1 (wt:wt) and that it made no difference whether this ratio was controlled by altering the C loading (by changing the plant addition rate) or the N loading (by altering HLR)(Figure 2).

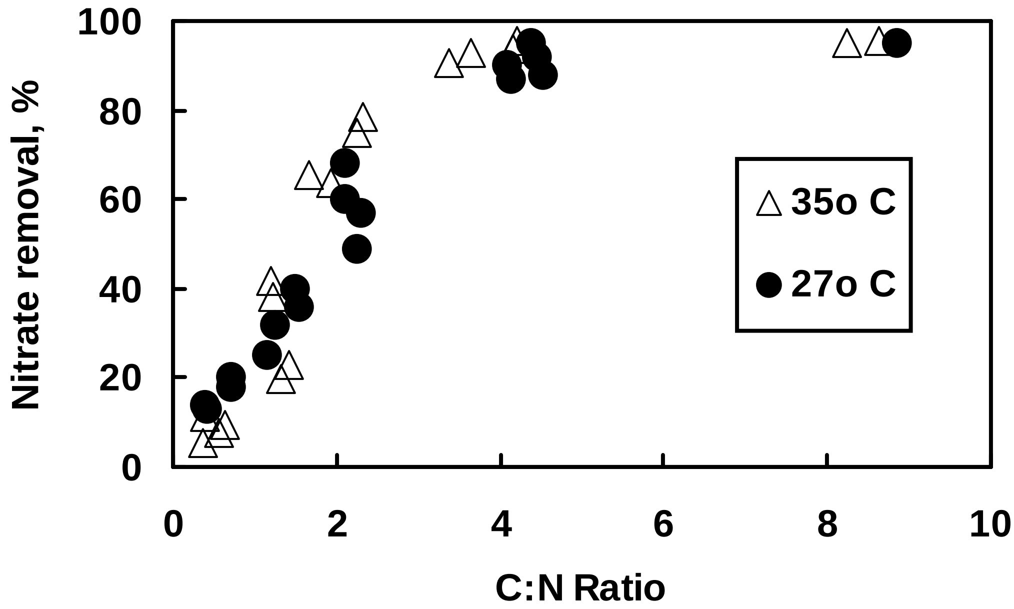

Figure 2. Nitrate removal efficiency as a function of C:N ratio in wetland microcosms (Ingersoll and Baker, 1998). Most of the carbon was supplied by added plants; most of the nitrogen was supplied by the feed water.

In this experiment, the denitrification loss rate, ($V_{NO3,}$ m/yr) was directly proportional to carbon addition rate in accordance with the equation:

$$V_{NO3}\text{, m/yr} = -43.0 + 0.0243*(\text{plant addition rate, g DW/m}^2\text{-yr}) \qquad (2)$$

(for 28°C; n= 18; $r^2 = 0.61$; $P < 0.001$)

Production of Dissolved Organic Carbon

Because of their capacity to remove nitrate, constructed wetlands have become integrated into several effluent recharge projects in the southwestern United States. These projects are intended to augment municipal water supplies for the future. One concern regarding treatment wetlands used in this manner is that they may add dissolved organic carbon (DOC) to the effluent. DOC serves as a precursor for disinfection by-products (DBPs) when the water is eventually chlorinated. Certain DBPs (trihalomethanes and haloacetic acids) are carcinogenic and are therefore regulated. In our microcosm study, we found that DOC accounted for 6-8% of added plant carbon. Effluent DOC values remained < 5 mg/L until nitrate removal exceeded 80% and then rose quickly, up to ~ 10-20 mg/L at C:N ratios > 10:1. When chlorinated, DOC from these microcosms yielded 35 μg chloroform/mg DOC, a ratio comparable with many surface waters (Paul Westerhoff, Department of Civil and Environmental Engineering, Arizona State University). Gray et al. (1996) reported that DOC increased by an average of only 18% in mesocosms at the Prado wetlands under conditions in which nitrate removal averaged 44% (year 4). DOC in the Prado wetland itself increases from ~4 mg/L to 8 mg/L (A.J. Horne, University of California at Berkeley, per. comm.) At conditions that result in moderate nitrate removal rates (50-80%), elevation of DOC levels in the effluent are expected to be fairly small. If the effluent from the wetland were recharged, much of the DOC would be lost in the vadose zone. In an ongoing lab experiment in which we are recharging lagoon effluent (15-20 mg/L DOC) through a 1-m long soil column, the DOC in the column effluent has remained < 5 mg/L for several months (Westerhoff et al., in prep.). Many other studies also show that the DOC of water passing through the vadose zone is greatly reduced (Bouwer et al., 1980; Bouwer and Rice, 1984; Kopchinsky et al., 1996)

Predicting nitrate removal rates

Our knowledge of wetland biogeochemistry can be used to develop preliminary concepts regarding the design of wetlands designed primarily for removal of nitrate. A key factor is the C:N ratio, which can be controlled by either plant productivity or nitrate loading rate. From a practical standpoint, the nitrate concentration of the source water is a "given"; controlling nitrate loading therefore means controlling the HLR. Figure 3 shows computed values of net nitrate retention as a function of plant productivity (g/m^2-yr) and HLR (cm/day). These calculations show that for a wetland with a plant productivity of 6,000 g DW/m^2-yr, 80% nitrate retention could be accomplished with a HLR of about 22 cm/day. It is important to emphasize that the V_{NO3} values used in these calculations were developed in laboratory microcosms. Further research is needed to refine these values using field-scale nitrate treatment wetlands.

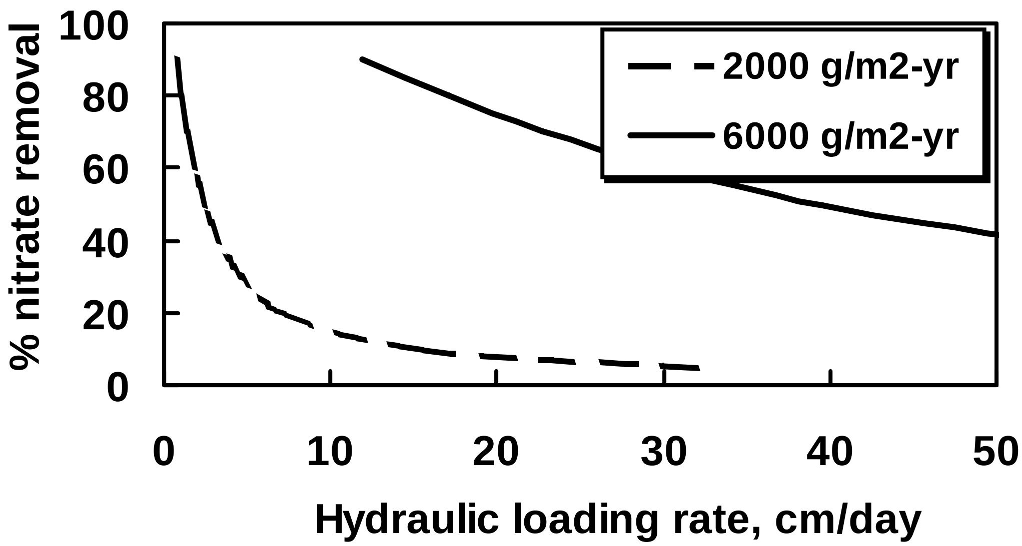

Figure 3. Computed net nitrate removal efficiency as a function of HLR at two levels of plant productivity (g/m^2-yr) using plug flow kinetics [C/C_{in} = exp-(V_{NO3}/HRT)]. V_{NO3} was estimated from equation 1 using data from the wetland microcosm experiment of Ingersoll and Baker (1998).

Where land area is limited (i.e., HLR must be high), steps to maximize plant productivity would be in order. This could be accomplished through fertilization with phosphorus (N is unlikely to be a limiting nutrient) or by mechanical harvesting (leaving the cuttings in place).

APPLICATIONS

Although wetlands have been widely used for treatment of wastewater effluent (Kadlec and Knight, 1996) and stormwater, their application to the task of treating low-BOD, high nitrate waters is relatively new. Several potential applications are discussed below.

Nitrate removal from nitrified effluents

Wetlands are being used to denitrify nitrified effluent at several locations, notably the Prado wetland in Orange County, California (Horne, 1995), the Tres Rios wetland (Phoenix, Arizona), and the San Joaquin Marsh (Irvine, California). At the first two sites, a goal is to make the effluent suitable for recharge; at San Joaquin, the goal is to reduce nutrient inputs to an estuary (A.J. Horne, per. comm.).

Pump-and-treat groundwater remediation

Baker (1994) proposed that nitrate treatment wetlands could be used to renovate nitrate-contaminated aquifers. 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. Additional treatment, particularly removal of DOC and pathogens, would occur in the vadose zone (Kopchynski et al., 1996; Bouwer, 1993). For loamy sands, long-term infiltration rates would be around 100 m/yr (~30 cm/day). For the hypothetical wetland described above (80% N removal at a HLR of 22 cm/day), the infiltration basin would therefore be of roughly the same size as the wetland itself.

Treatment of irrigation return flows

Irrigation return flows often contain high nitrate levels which contribute to water quality problems in receiving waters. Wetlands would be an ideal method to remove nitrate from irrigation return flows. They are easy enough to construct that farmers could build them using ordinary farm equipment and they require very little maintenance. In this application, the use of a sedimentation basin would be useful to extend the life of the wetland. The flow through the wetland could be shut off during the non-growing season.

On a larger scale, the City of Avondale, Arizona is constructing a wetland-recharge system in lieu of a conventional water treatment plant. The wetland is needed because the source water is contaminated with nitrate from agricultural return flow, urban runoff, and nitrate-contaminated groundwater that is pumped into the source water canal (Thompson et al., 1997). This unique project consists of several series of small lakes with wetland islands. After traveling through five lake-wetland cells, the water will be recharged through surface infiltration basins and stored in an aquifer for later use as the city grows.

Treatment of nitrate-contaminated industrial water

In the Phoenix area, much of the groundwater exceeds the 10 mg/L nitrate MCL. Several “peaking” power plants use this water for cooling water and then dispose of it to agricultural fields. As the area becomes urbanized, these fields are disappearing. An alternative is to discharge the water to canals, but this may not be permissible if the water is to be used for municipal water supply downstream. Wetlands may be a practical solution. Although the cooling water requirements of these plants changes rapidly, they already have the capability to store water in ponds. The flow buffering provided by the ponds means that treatment wetlands could be sized to handle a more-or-less constant flow.

Dual purpose wetlands

In arid climates, the use of stormwater wetlands is constrained by low rainfall and high evaporation rates. A hydrologic feasibility analysis conducted by Marin (1994) concluded that a stormwater wetland in Phoenix receiving only surface runoff (runoff coefficient = 0.2; 10 year historical record of measured precipitation and evaporation) would not remain wet throughout the year. Even with a watershed:wetland ratio of 200:1, such a wetland would remain wet only 90% of the time and would have an outflow only 10% of the time. Dual purpose wetlands could be used to treat both urban runoff *and* nitrate-contaminated groundwater. During dry periods, the wetland would receive nitrate-contaminated groundwater and during precipitation events the wetland would detain and treat urban runoff. The effluent from the wetland could be recharged to groundwater or discharged to a surface water.

CONCLUSIONS

Wetlands are well-suited for nitrate removal, providing both a carbon source and an anaerobic environment for denitrification. Potential nitrate removal rates of 40-50 kg ha-day are about 10 times higher than rates observed in wetlands receiving non-nitrified effluent. Denitrification efficiency appears to depend on the C:N ratio, with peak efficiencies occurring at C:N ratios > 5:1. Applications of nitrate-treatment wetlands include renovation of nitrate-contaminated aquifers, treatment of nitrified wastewater effluent, treatment of irrigation return flows, and treatment of groundwater used for cooling and other industrial purposes. Dual purpose wetlands also have considerable potential that has not been exploited.

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