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Anoxic treatment wetlands for denitrification

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ABSTRACT

Anoxic subsurface flow (SSF) constructed wetlands were evaluated for denitrification using nitrified wastewater. The treatment wetlands utilized a readily available organic woodchip-media packing to create the anoxic conditions. After 2 years in operation, nitrate removal was found to be best described by first-order kinetics. Removal rate constants at 20 °C (k_{20}) were determined to be 1.41–1.30 d^{-1}, with temperature coefficients (θ) of 1.10 and 1.17, for planted and unplanted experimental woodchip-media SSF wetlands, respectively. First-order removal rate constants decreased as length of operation increased; however, a longer-term study is needed to establish the steady-state values. The hydraulic conductivity in the planted woodchip-media SSF wetlands, 0.13–0.15 m/s, was similar to that measured in an unplanted gravel-media SSF control system.

1. Introduction

Nitrate has been identified as a constituent of concern for many wastewater systems that disperse effluent to the soil because of potential impacts on groundwater. In some aquifers, nitrate concentrations above the drinking water limit have been found to extend more than 100 m from septic systems (Robertson et al., 1991). Elevated concentrations of nitrate in drinking water have been linked to methemoglobinemia in infants, a medical condition that interferes with the oxygen-carrying capacity of blood (U.S. EPA, 2002). Due to this health concern, the U.S. EPA and other regulatory agencies have set the maximum contaminant level for nitrate in drinking water at 10 mg N/L. Currently, there are limited options available for decentralized wastewater systems for the removal of nitrogen. The lack of cost-effective decentralized treatment options for nitrogen has resulted in the installation of capital intensive centralized collection and treatment systems in some communities. Therefore, an effective and inexpensive denitrification process for use in decentralized wastewater management applications is needed (Oakley et al., 2010).

1.1. Onsite wastewater systems

Onsite wastewater management for an individual home consists typically of a septic tank and effluent dispersal system. The septic tank provides primary treatment for the wastewater and acts as an anaerobic digester for the organic waste that settles out of the water. Effluent from the septic tank contains nitrogen that is primarily in the ammonium form. A commonly used effluent dispersal system uses perforated subsurface pipes to infiltrate septic tank effluent into the soil by gravity. In the soil, the septic tank effluent undergoes additional treatment as the wastewater is exposed to oxygen and soil bacteria, resulting in the conversion of ammonium to nitrate. The wastewater nitrate then percolates through the soil matrix and may accumulate in groundwater aquifers and contaminate surface waters (Kellogg et al., 2010; U.S. Geological Survey, 2004).

1.2. Nitrate removal from wastewater

In conventional activated sludge type wastewater treatment plants, a small amount of nitrogen is removed through the production and wasting of biomass. High levels of nitrogen removal require the application of specialized biological nutrient removal processes. Conventional biological nutrient removal processes convert the organic and ammonia nitrogen to nitrate in an aerobic environment (nitrification) and then reduce the nitrate to nitrogen gas in an anoxic environment (denitrifica-

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tion process involves the anoxic biological oxidation of organic substrates in wastewater using nitrate as the electron acceptor (Tchobanoglous et al., 2003).

In wastewater treatment plants designed for nitrogen removal, nitrification and denitrification are typically integrated processes that utilize anoxic zones either before or after aerobic treatment. In processes that utilize anoxic zones before aerobic treatment, nitrates and biomass are returned from aerobic treatment to the anoxic zone where influent organics are utilized as the carbon source in the denitrification reaction. A common pre-anoxic denitrification method is the Modified Ludzack-Ettinger process (MLE) that achieves nitrate removal through an internal recycle step (Tchobanoglous et al., 2003). However, processes such as the MLE are not well suited for decentralized wastewater systems with stringent nitrogen limits because the variability in the loading conditions experienced in these small systems can lead to unreliable performance. For example, a number of decentralized wastewater systems recirculate nitrified effluent to the septic tank for denitrification but can only achieve total nitrogen removal rates around 50–60% reliably (Oakley et al., 2010).

In processes that utilize anoxic zones after aerobic treatment, the influent wastewater carbon is oxidized in the aeration and nitrification process and is no longer available for denitrification. Therefore, an external carbon source must be added to supply energy to the nitrifying organisms (Tchobanoglous et al., 2003). Several proprietary post-anoxic denitrification methods have been developed to overcome this limitation, including the use of both liquid carbon feed systems and solid phase carbon filters (Oakley et al., 2010; Schipper et al., 2010). For decentralized wastewater systems, liquid carbon feed systems can pose problems because the chemical source needs to be replenished on a regular basis and there is difficulty in applying the correct chemical dose to wastewater with varying characteristics (Leverenz et al., 2007).

1.3. Nitrogen removal in constructed wetlands

Natural wetlands have been shown to be a simple and energy-efficient method of removing nutrients (i.e., phosphorous and nitrogen) from wastewater (Nichols, 1983). Nichols (1983) concluded that while natural wetlands are good at removing phosphorous, nitrogen removal was dependent on the organic content of the wetland soils. Artificial open water wetlands have also been shown to be effective for the removal of nitrogen from wastewater (Gersberg et al., 1983, 1984). These results are explained by plant assimilation, the presence of microscopic anoxic zones that occur in bacterial films, and, over time, the presence of decaying plant material that provide carbon for denitrifying bacteria. Nitrate disappearance in open water constructed wetlands has been modeled as a volume-based first-order reaction (Kadlec and Knight, 1996).

Another alternative treatment wetland technology is the subsurface flow (SSF) constructed wetland, which is well suited for onsite wastewater applications because they provide odor and vector control and mitigate public access issues (U.S. EPA, 1993). Artificial SSF wetlands are typically designed with an inert rock medium and can be either planted or unplanted, and are designed so that the water flows below the surface of the wetlands through the packed-bed porous medium. The rock medium provides a surface area for the growth of bacterial films but inhibits the carbon cycling from plant debris because the packing material impedes the plant debris from reaching the water. As a result, conventional subsurface wetlands are only marginally successful at removing nitrogen from wastewater and generally require a prenitrification step to enhance denitrification capacity, however, these systems remain carbon limited (U.S. EPA, 1999). The nitrogen removal that does occur in rock medium SSF wetlands is the result of plant assimilation and microbial denitrification that utilizes any remaining carbon source in the influent and from rhizosphere plant decay (Kadlec and Knight, 1996). Thus, an alternative carbon source is required to increase the denitrification performance, assuming that nitrification has already taken place. For example, Gersberg et al. (1983) demonstrated that the addition of carbon, in the form of methanol, stimulated bacterial denitrification and increased nitrate removal efficiencies to 95%. However, the use of liquid carbon feed systems in small wastewater systems are subject to the limitations noted in Section 1.2.

1.4. Nitrogen removal in anoxic filters

Based on previous research reported in the literature, it has been found that a variety of organic solids can be used simultaneously as media and as a carbon source to support the denitrification process. These include plant biomass (Gersberg et al., 1983), cotton burr and mulch compost (Su and Puls, 2007), wheat straw (Aslan and Turkman, 2003), sawdust (Robertson and Cherry, 1995; Schipper and Vojvodic-Vukovic, 1998), and woodchips (Healy et al., 2006; Robertson and Merkley, 2009). Schipper and Vojvodic-Vukovic (1998) demonstrated that porous groundwater treatment walls amended with sawdust were successful in removing nitrate from contaminated groundwater. Schipper et al. (2010), also employed woodchip-based denitrification bioreactors to reduce end-of-pipe losses from agricultural drainage systems. Robertson et al. (2005) demonstrated that the Nitrex filters, which utilize a proprietary nitrate reactive material, produced septic tank effluent nitrate removal rates of up to 96%, remaining effective for at least 5 years, but removal rates were diminished during the winter months. However, the use of a readily available organic medium in a subsurface flow constructed wetland as a method for denitrification of nitrified septic tank effluent has not been investigated.

1.5. Purpose of study

The purpose of this research was to evaluate the use of constructed subsurface flow wetlands filled with an organic woodchip-media for denitrification of wastewater. The specific objectives were to assess the effect that aquatic plants, temperature, length of operation, hydraulic performance properties, and nitrate concentration had on nitrate removal performance. The results were used to determine nitrate removal rates and temperature coefficients that can be used for the preliminary design of constructed wetlands using organic woodchip-media.

2. Materials and methods

The pilot facility used in this study consisted of a septic tank, a packed-bed nitrification system, and experimental subsurface flow wetland units. Details of the experimental system and operational parameters are presented below.

2.1. Pretreatment system

Wastewater used in the study was diverted from the influent to the University of California Davis Wastewater Treatment Plant (UCD WWTP). The septic tank was a conventional design with a nominal volume of 7.6 m^3 and retention time of about 2 d. The packed-bed nitrification system consisted of three parallel single-pass units that utilized a synthetic textile media (Orenco Systems, Inc., Sutherlin, OR) and employed natural ventilation

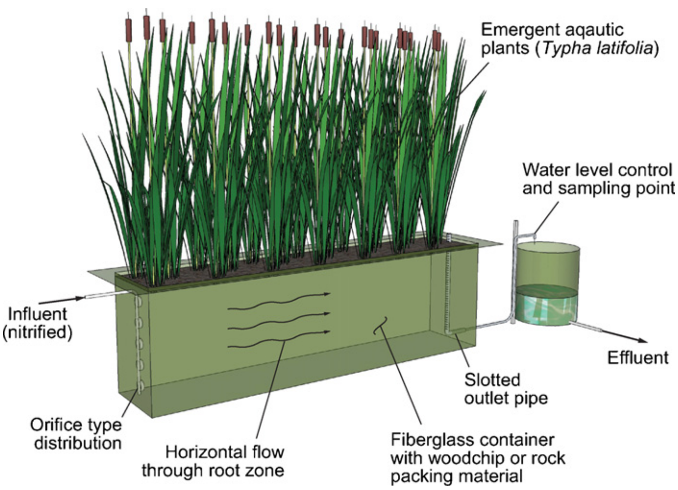

Fig. 1. Schematic of experimental constructed wetland.

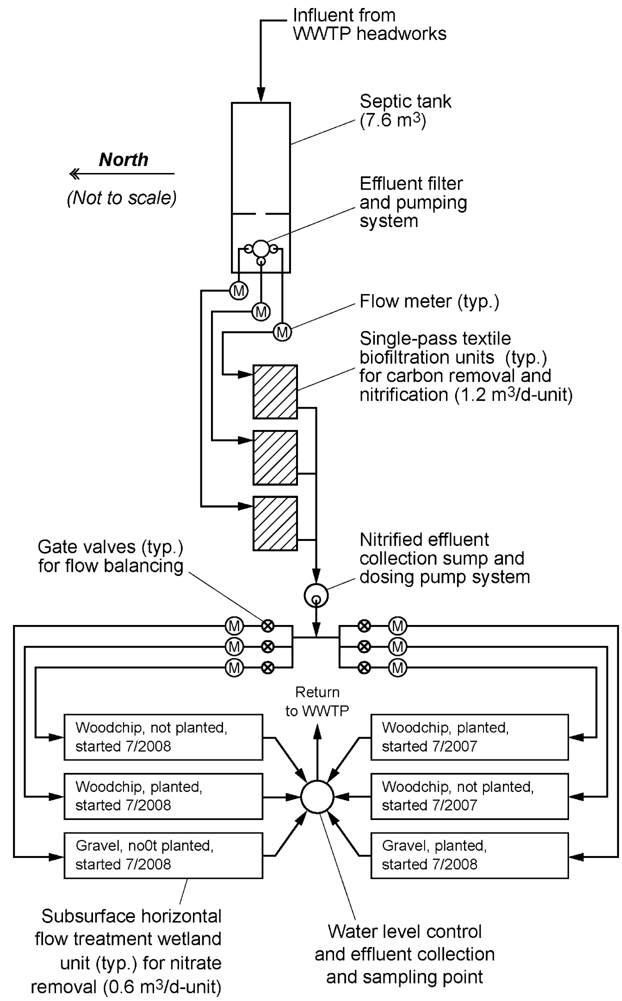

Fig. 2. Plan view of pilot testing system.

for oxygen transfer. To ensure nitrification occurred reliably in the packed-bed filters, the filters were operated at high dosing frequencies (96 dose/d) and relatively low organic loading rates (100 g BOD/m^2 d). The nitrified effluent was collected in a pump tank and evenly distributed to six subsurface wetlands using water meters and throttling gate valves. Additional details on the nitrification system may be found in Leverenz et al. (2001).

2.2. Experimental wetland units

Six different subsurface wetlands were used to study the effect that media type, time of operation, and aquatic plants (*Typha latifolia*) have on the removal of nitrate. The subsurface wetlands were housed in rectangular fiberglass tanks (3 m long, 1 m high and 0.6 m wide). The media depth was initially filled to the top of the basin (1 m) and the water depth was set at 0.15 m below the surface of the media. A vertically placed orifice type inlet structure for the wetlands was designed to allow the nitrified wastewater to be distributed evenly along the height of the tank, as shown in Fig. 1. To investigate the effect of medium type, four of the SSF wetlands units were filled with readily available recycled pallet woodchips (Waste Management, Inc., WMCR/K&M, Sacramento, CA) with particle lengths ranging from 13 to 152 mm, and an average thickness of 6.3 mm. Two additional SSF wetland units were filled with gravel classified as 19 mm clean crushed rock. To investigate the effect of time of operation, two of the woodchip filled SSF wetland units were placed in operation in July 2007 (not monitored) and the other four wetlands were put into operation in June 2008. To investigate the effect of the presence of aquatic plants, three of the wetland units (a woodchip wetland placed into operation in 2007, a woodchip wetland placed into operation in 2008, and a gravel wetland) were planted with *T. latifolia* at the time of startup and the remaining three wetland units were left unplanted. A diagram of the pilot system configuration is shown in Fig. 2. A summary of the experimental wetland unit design information is presented in Table 1. Each of the SSF wetland units received approximately 0.6 m^3/d of nitrified effluent, applied intermittently in equal doses every 15 min.

2.3. Sample collection and analysis

Regular influent and effluent grab samples were collected from each of the wetlands and were analyzed for temperature, nitrate, and nitrite. The temperature was measured in the field using a Myron L handheld meter. The latter parameters were measured using Ion Chromatography [DIONEX LC20 Chromatography Enclosure, DIONEX ION Pac AS14A 4X250 mm Analytical (ANION)]. Periodically, ammonium ion and total Kjeldahl nitrogen (TKN) were measured in accordance with Standard Methods for the Examination of Water and Wastewater (1998) to ensure that the wetland influent was completely nitrified. The (5-d biochemical oxygen demand) BOD_5 was also measured in accordance with Standard Methods (2000) to evaluate effluent water quality.

Influent and effluent grab samples were collected about once a week from each wetland. Grab samples were also collected periodically along the length of each wetland to determine nitrate removal profiles. Intermediate samples were obtained from sampling wells (PVC pipe sections with perforated ends) inserted in the media with

Table 1
Summary of wetland design information.

Startup date	Medium	Plant used[a]	Designation
7/2008	Rock	*Typha latifolia*	G, P, 08
7/2008	Rock	None	G, UP, 08
7/2007	Woodchips	*Typha latifolia*	W, P, 07
7/2007	Woodchips	None	W, UP, 07
7/2008	Woodchips	*Typha latifolia*	W, P, 08
7/2008	Woodchips	None	W, UP, 08

[a] Where plants are indicated, rhizomes were embedded at system startup.

the perforated section at mid-depth, and samples were withdrawn using a hand pump.

2.4. Porosity measurements

The porosity of the media contained in the unplanted woodchip SSF wetland units was measured by volumetric displacement to evaluate degradation of the woodchip-media over time. Media samples were obtained from 0.3 m below the water surface and at several locations along the length of the basin. The porosity values were compared to gravel and unused woodchips.

2.5. Hydraulic conductivity measurements

Hydraulic conductivities of SSF media were measured using a permeameter test procedure (Crites et al., 2006). The permeameter testing was conducted directly in the SSF wetland unit basins by measuring headloss across a section of the system during loading at a constant flow rate. Darcy's Law of laminar flow through porous media was then used to determine the hydraulic conductivity value.

During the test procedure, the influent wastewater supply pump was turned off and a perforated pipe was inserted next to the influent pipe. Potable water was distributed through the perforated pipe at a constant flow as determined from volumetric testing. Piezometers installed 0.2 m from the inlet and outlet on basin sides were monitored and the head difference was recorded after steady-state conditions were obtained. Following the measurements, the Reynolds number through porous media was determined to ensure laminar regime assumptions were accurate. The limit of the laminar regime within porous media holds when the associated Reynolds's numbers are less than 10 (Charbeneau, 2000).

2.6. Tracer study

Tracer testing was performed in May 2009 using sodium chloride (NaCl). The effluent electrical conductivity was measured using a handheld conductivity meter (Myron L Ultrameter). For purposes of the study, 7.5 L of NaCl solution at a concentration of 20 g/L was added to the influent feed to each wetland system. An effluent composite sample and grab sample were obtained every 4 h during the study, which lasted for a total of 100 h. After the 100 h testing period, the effluent conductivity values had been observed to return to the baseline conditions, indicating that the tracer had been flushed from the system.

3. Results and discussion

The experimental results are presented and discussed in this section, including performance characteristics of the pretreatment system, overall nitrate removal performance, nitrate removal profiles, nitrate removal rates, effluent biochemical oxygen demand, hydraulic characteristics of SSF wetlands, and effects of plants on the system operation.

3.1. Performance of pretreatment system

Packed-bed filters were used to pretreat the wastewater prior to treatment in the wetland systems. The effluent BOD_5 concentrations from the pretreatment system were consistently less than 2 mg/L throughout the study. Effluent grab samples from the pretreatment system were also analyzed for ammonium and organic nitrogen. Average warm season ammonium and organic nitrogen concentration were 0.2 and 0.8 mg/L, respectively. Average cool season ammonium and organic nitrogen concentration were 1.4 and 1.2 mg/L, respectively. The pretreatment system effluent nitrite concentrations were non-detectable throughout the study. Based on the ammonium nitrogen and nitrite data, near complete nitrification was occurring throughout the study.

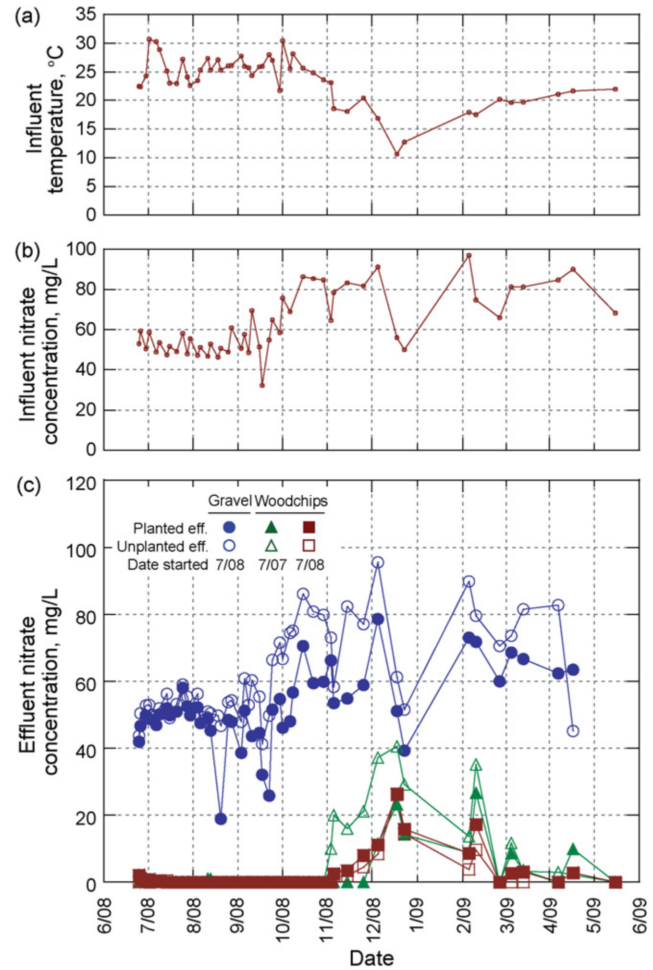

Fig. 3. Summary of SSF wetland performance (a) influent temperature, (b) influent nitrate concentration for all systems, and (c) effluent nitrate concentrations.

3.2. Nitrate removal performance

The influent temperature profile, shown in Fig. 3a, varied from 22 to 30 °C during the first 4 months of operation. In November, the influent temperature began to decrease reaching a low of 11 °C. The influent concentration of nitrate to the constructed wetlands is shown in Fig. 3b. For the first 4 months of operation, the influent concentrations averaged 53 mg/L, after which the influent concentration increased to an average of 82 mg/L when the student population increased at the start of the academic year.

The effluent concentration of nitrate from each wetland is presented in Fig. 3c. Nitrate removal in the unplanted gravel wetland (G, UP, 08) was negligible throughout the study. The nitrate concentration in the planted gravel (G, P, 08) wetland was reduced by an average value of 10 mg/L. On an area basis, this equates to a removal rate of 0.74 g N/m^2 d. Other researchers have observed values in the same range; for example, Lin et al. (2008) reported maximum nitrogen removal rates in SSF wetlands of 1.161 g N/m^2 d. While the observed nitrate reduction in the planted gravel SSF wetland is associated with plant growth, the specific removal mechanism has not been determined.

Reductions in the nitrate concentrations were observed in all of the woodchip wetlands throughout the study, with removals ranging from 60 to 100 mg/L. For the first 5 months of operation the woodchip wetlands removed an average of 99.7% of the influent nitrate, which ranged from 45 to 80 mg/L. However, beginning in November, the effluent nitrate concentration from the wetlands began to rise as the influent water temperature dropped. The reduced performance is attributed to decreased bacterial activity at lower temperatures (Sawyer et al., 1994). On an area basis, the nitrogen removal rate is estimated to be about 5.9 g N/m^2 d at temperatures above 15 °C, or 8 times higher than in the gravel-based SSF wetland system.

As shown in Fig. 3c, there was not a significant difference in the effluent nitrate concentrations between the 2008 planted and unplanted woodchip wetlands (W, P, 08 and W, UP, 08), which indicates that the availability of carbon from the woodchips was not rate limiting in these wetlands during this period. Similarly, for the first 4 months of operation there was no significant difference in the effluent concentrations between the planted and unplanted woodchip wetlands constructed in 2007 (W, P, 07 and W, UP, 07). However, in November when the temperatures began to decline, the unplanted woodchip wetland constructed in 2007 (W, UP, 07) exhibited higher effluent nitrate concentrations than the planted woodchip wetland constructed in 2007 (W, P, 07), with an average increase in concentration of 20 mg/L. The difference between the planted and unplanted systems is attributed to plant assimilation or synergistic effects between the plant roots and microbial community.

3.3. Nitrate profiles

Nitrate profile data collected at varying influent nitrate concentrations and temperatures are presented in Fig. 4. In each profile data set, nitrate removal in the unplanted gravel wetland (G, UP, 08) did not occur. Planting the gravel wetland (G, P, 08) consistently improved nitrate removal, but only slightly. This observation is consistent with the low overall nitrate removal for the planted and unplanted gravel wetlands (G, P, 08 and G, UP, 08) as shown in Fig. 3. The effect of temperature variation is evident when the profiles presented in Fig. 4a, b, and c are compared. The profile data reflects a decline in the nitrate removal rate with declining temperature. This temperature dependent removal relationship is consistent with lower bacterial activity that would be associated with lower temperatures.

3.4. Nitrate removal rates

The results of nitrate profile measurements, along with retention time in the wetland units as determined with a tracer study (see Table 2), were used to assess nitrate removal kinetics of the woodchip SSF wetlands. The profile data was best described with a first-order removal rate model (Tchobanoglous and Schroeder, 1985). A number of other researchers have described denitrification reactions in packed-beds as zero order (Robertson et al., 2000; Van Driel et al., 2006). However, it is proposed that while most field-scale systems are well approximated assuming zero order reaction kinetics, at low nitrate concentrations and at reduced temperatures, first-order kinetics may provide a better fit. Additional controlled studies are recommended to further characterize the nitrate removal kinetics.

The first-order removal constants, calculated for a temperature of 20 °C are summarized in Table 3. As shown in Table 3, the reaction rate decreases as the woodchip packing ages. In addition, the presence of plants resulted in a slight increase in the observed reaction rate, possibly due to combined effects of denitrification and plant uptake. The temperature coefficient, θ, was calculated to be 1.10 and 1.17 for the planted and unplanted systems, respectively (Benefield et al., 1982). The temperature coefficient can be used to calculate the reaction rate at temperatures ranging from 11 to 20 °C, as shown in the following equation:

$$k_T = k_{20}\theta^{(T-20)}$$

where k_{20} = removal rate constants at 20 °C; k_T = removal rate constant at temperature T; θ = temperature coefficient.

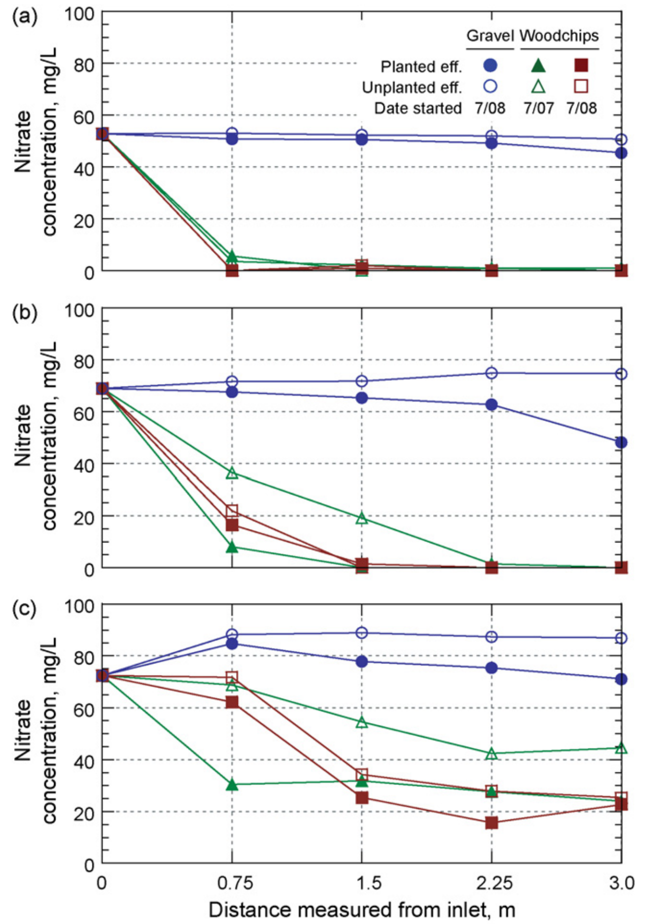

Fig. 4. Nitrate profile along the length of the wetland on (a) 8/13/08, 25 °C; (b) 2/26/09, 19 °C; and (c) 12/12/08, 11 °C.

Table 2
Characteristics of wetland systems.

Wetland unit	Retention time (d)[a]	Hydraulic conductivity (m/s)[b]	Media porosity[b]
Planted			
G, 08	1.0	0.34	–
W, 07	1.9	0.15	–
W, 08	1.8	0.13	–
Unplanted			
G, 08	2.2	0.14	0.37
W, 07	2.0	0.54	0.58
W, 08	1.2	0.36	0.59

[c]Unused woodchip porosity was 0.65.
[a] Measurements made in May 2009.
[b] Measurements made in August 2009 for unplanted systems only.

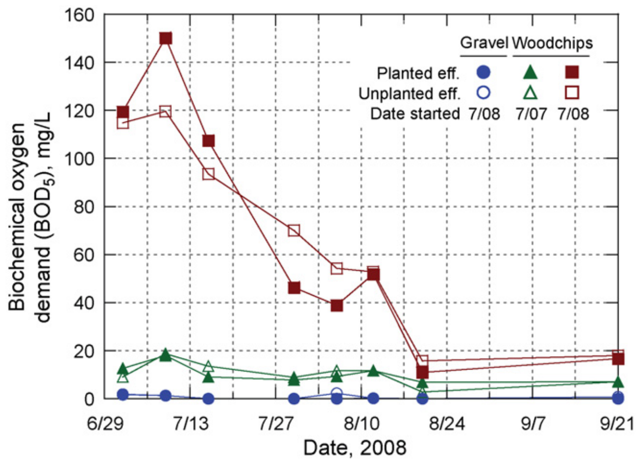

Fig. 5. Effluent BOD_5 concentration for each of the wetlands (influent BOD_5 was consistently less than 2 mg/L).

While a preliminary assessment of the impacts of temperature is presented in this paper, additional research is needed to evaluate the effects of temperature over a wider range. However, it is apparent that temperature effects should be taken into consideration for systems that must meet a regulatory limit. As shown in Table 3, the planted systems had a lower temperature coefficient than the corresponding unplanted systems. The smaller θ value is a result of being less sensitive to temperature fluctuations, particularly at low temperatures. It is therefore possible that the plants buffered the microbial community somewhat from the effects of temperature.

3.5. Biochemical oxygen demand

Effluent concentrations of biochemical oxygen demand (BOD_5) for each wetland are shown in Fig. 5. The influent BOD_5 concentration to all systems and the effluent BOD_5 concentrations of the planted and unplanted gravel wetlands (G, P, 08 and G, UP, 08) remained below 2 mg/L for the duration of the experiment. For the SSF woodchip wetlands constructed in 2008, the effluent BOD_5 concentrations were high (e.g., 120 mg/L) during the first month of operation, reflecting a significant release of carbon from the new woodchips. The effluent BOD for the systems started in 2007 were also high for the first few months after startup, however, quantitative measurements were not made at the time. The elevated effluent BOD_5 concentrations associated with the release of carbon was also observed by Robertson et al. (2005) for the Nitrex system. Following the first month of operation, the effluent BOD_5 concentration decreased to less than 20 mg/L. The effluent BOD_5 concentrations in both the planted and unplanted woodchip wetlands constructed in 2007 (W, P, 07 and W, UP, 07) increased from the influent concentration of 2 mg/L to effluent values ranging from 10 to 20 mg/L.

The high initial effluent BOD could be a problem in areas where there are strict effluent limitations that need to be observed. In these cases, the initial flow can be discharged to alternate location or treated in an aerobic process to remove the residual organic matter until satisfactory levels are attained. Another option would be to bypass and blend a portion of the nitrified influent with the high carbon effluent in a separate post-anoxic denitrification process. It should be noted that the effluent BOD is almost completely derived from the woodchips and not from wastewater.

Table 3
Summary of first-order reaction rate and temperature coefficients for woodchip wetlands.

Wetland unit	k_{20} (d^{-1})	θ^a
W, P, 07	1.41	1.10
W, P, 08	2.61	
W, UP, 07	1.30	1.17
W, UP, 08	2.28	

[a] Valid from 11 to 20 °C (Sawyer et al., 1994).

3.6. Wetland hydraulic characteristics

Hydraulic conductivity measurements were made in August 2009, approximately 25 months and 13 months after the startup of the systems initiated in July 2007 and July 2008, respectively. Porosity for the woodchip SSF wetland systems was also measured in August 2009, following the hydraulic conductivity testing. The characteristics of the gravel and woodchip SSF systems are presented in Table 2.

In the planted woodchip SSF systems, the hydraulic conductivity values were similar, 0.15 and 0.13 m/s for the 2007 and 2008 systems, respectively. The similar values could be an indication that after 1 year of service, the root growth in the planted systems had reached an equilibrium status. By comparison, the unplanted woodchip SSF systems had much higher conductivity values of 0.54 and 0.36 m/s for the 2007 and 2008 systems, respectively. It is expected that plant root growth is the cause of the reduced conductivity values in the planted systems, however, it is not clear why there is an increase in the conductivity value for the older unplanted woodchip SSF. One reason for the increase could be the degradation of small woodchip particles and/or the development of preferential flow paths. As reported in Table 2, there was little change in porosity between woodchip samples that were unused as compared to after use in the wetlands.

In the planted and unplanted gravel SSF wetlands, an increased conductivity of 0.34 m/s was measured in the planted system compared to 0.14 m/s measured in the unplanted system. While the growth of plants was expected to decrease the hydraulic conductivity, other researchers have reported a similar phenomenon (Grismer et al., 2001). It is proposed that the presence of plant roots may create preferential flow paths through the gravel bed where the smaller porosity inhibits flow. Alternatively, the growth of plant roots may expand the gravel bed and increase the effective porosity. However, these concepts remain to be tested in a controlled study.

3.7. Effects of plants

During the course of the study, plants were found to have several specific impacts in addition to the minor performance effects described in Sections 3.2 and 3.3. For example, it was noted that the unplanted systems were subject to media settling, which occurred mostly in the first year and equal to about 0.1 m of settlement. In contrast, due to root growth, the planted systems did not experience settlement and the woodchip-media was even slightly expanded. Plants in the woodchip SSF wetlands had robust growth on the inlet side (0–1.5 m) of the system and stunted growth on the outlet side (1.5–3.0 m) of the system. The stunted growth was correlated with the lack of nitrogen and resulted in significantly reduced growth, shorter plants, and yellowed vegetation color. On the outlet side of the wetland, plant growth only occurred near the edges of the basin, perhaps in response to preferential flow paths at the sidewalls. In this case, plants could be used as a visual indicator of nitrate progression through the anoxic reactor. An example of the variation in plant growth in the woodchip SSF compared to the gravel SSF is shown in Fig. 6. In the long-term, there is a possibility

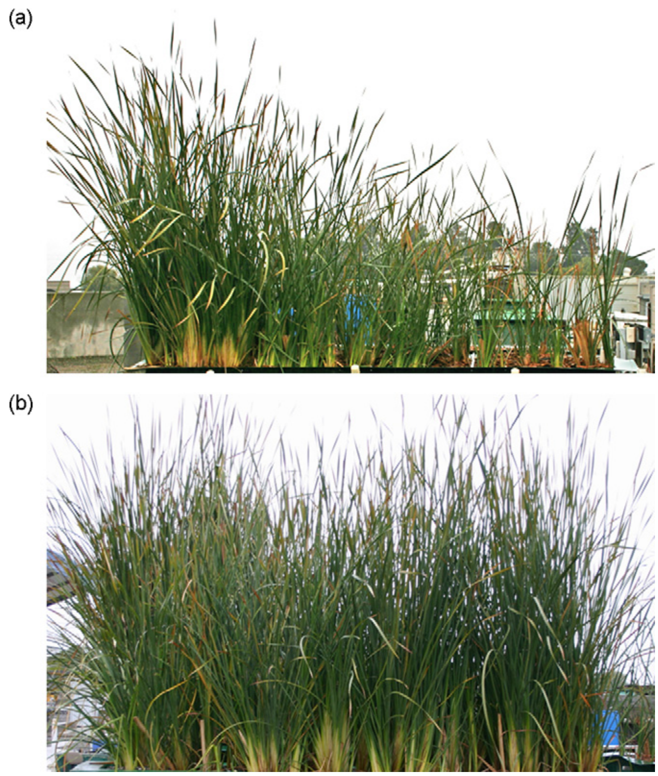

Fig. 6. Views of plant growth in (a) woodchip SSF wetland and (b) gravel SSF wetland. Inlet is on the left side and outlet is on the right side. Photographs taken on the same day, for systems of same age (8 months after startup), with identical loading.

that plants could contribute additional carbon to the system due to decay of plant material.

4. Findings

The purpose of this research was to evaluate the use of subsurface wetlands constructed with a readily obtained organic medium for the denitrification of wastewater. Nitrate removal performance and the effects of temperature, length of operation, and aquatic plants were assessed, as summarized below.

- Readily available woodchips were an effective source of the carbon for denitrification of nitrified septic tank effluent. Waste woodchips are available at a fraction of the cost compared to gravel and thus may be an economically viable alternative media in subsurface flow wetlands.
- The observed nitrate removal performance in subsurface flow wetlands constructed with woodchips can be described with first-order reaction rate kinetics with rate constants at 20 °C (k_{20}) that varied from 1.41 to 1.30 d^{-1} for planted and unplanted systems, respectively, after 2 year in operation. Corresponding temperature coefficients for planted and unplanted systems were 1.10 and 1.17, respectively. Additional research is needed to further characterize the nature of the reaction kinetics and establish the temperature effects over a wider range.
- Longer operation times for the woodchip wetlands resulted in lower first-order removal rate coefficients and temperature coefficients. However, steady-state was not reached and no estimate of the long-term removal rate can be determined.
- The presence of plants in the woodchip SSF systems resulted in the decrease of the hydraulic conductivity to the same range as measured in an unplanted gravel SSF system (0.14 m/s).
- Porosities of the woodchips did not change significantly over the course of the study.
- Plants were found to have several beneficial effects, including buffering against low temperature effects, prevention of woodchip-media settling, and visual indicator of nitrate removal.

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