WOOD–BASED FILTER FOR NITRATE REMOVAL IN SEPTIC SYSTEMS

W. D. Robertson, G. I. Ford, P. S. Lombardo

ABSTRACT. *The recognition that septic systems can generate groundwater plumes with nitrate concentrations exceeding the drinking water limit has led to a need for improved nitrogen removal in septic systems. Long–term (3 to 5 year) monitoring results are presented for four full–scale, on–site wastewater treatment systems using a novel porous media filter (Nitrex filter) for enhanced nitrogen removal. The filter removes nitrogen by denitrification of pretreated, nitrified, septic tank effluent using a slowly soluble carbon source (wood byproduct material) incorporated into the filter. Results are presented for a house (sewage flow ~1 $m^3\ d^{-1}$), a trailer park (7 $m^3\ d^{-1}$), a communal residence (18 $m^3\ d^{-1}$), and an inn (73 $m^3\ d^{-1}$). In each case, the septic tank effluent was pretreated using a sand filter, then flowed passively through the denitrification filter, and finally was dispersed in a conventional tile bed. Influent (sand filter) NO_3–N concentrations, averaging 14.2 to 37.7 mg L^{-1}, were significantly attenuated in the denitrification filters at each site ($p < 0.05$) by amounts ranging between 87% and 98%. Reaction rates were temperature dependent, ranging from 7 to >10 mg N $L^{-1}\ d^{-1}$, and showed no sign of deteriorating with system age at any of the sites. Results support previous mass balance calculations and pilot–scale field trials, suggesting that such filters have the potential to operate for years without the need for media replenishment. These filters offer a practical solution to nitrate control in small to medium sized on–site wastewater treatment systems where simplicity of operation and low maintenance are desirable.*

Keywords. *NO_3, On–site, Sewage, Treatment.*

Nitrogen in domestic wastewater typically occurs at concentrations of 40 to 80 mg L^{-1}, predominantly as ammonium (NH_4) and organic N (Siegrist et al., 1976; Canter and Knox, 1985; Converse and Converse, 1998). Oxidation reactions associated with on–site treatment in septic systems can convert this nitrogen to nitrate (NO_3), leading to concentrations in groundwater plumes that can exceed the drinking water limit for nitrate (10 mg L^{-1} NO_3–N; Walker et al., 1973; Andreoli et al., 1979; Robertson et al., 1991). Increased nitrogen loading from septic systems has also been implicated as a cause of water quality deterioration in coastal estuaries (Valiela et al., 2000). These concerns have prompted some jurisdictions (e.g., Ontario and Massachusetts) to adopt regulations limiting nitrogen loading from septic systems in some sensitive locations.

Several relatively simple, low–maintenance methods have been developed to achieve enhanced nitrogen removal in septic systems. These include: the Ruck system (Laak, 1981; Lamb et al., 1991) which uses dedicated household plumbing to separate toilet water ("blackwater") containing most of the nitrogen, which is nitrified and then subsequently denitrified using graywater as a carbon source; peat systems (e.g., Brooks et al., 1984) which attenuate N by assimilation into fungal biomass; and various recirculation schemes (e.g., Hines et al., 1978; Sandy et al., 1987) in which the effluent is oxidized using a sand filter or some other aeration technique and then a portion is returned to the septic tank where denitrification occurs. These methods routinely achieve nitrogen removal of 50% to 80% but have difficulty consistently producing treated effluent with total N below 10 mg L^{-1} (NO_3–N drinking water limit). Meeting this criterion generally requires nitrogen removal in excess of 80%.

We have previously reported nitrogen removal rates in excess of 80% in laboratory and pilot–scale field trials of reactive porous media barriers incorporating a variety of slowly soluble carbonaceous solids (compost, straw, leaf mulch, and wood byproducts) (Vogan, 1993; Blowes et al., 1994; Carmichael, 1994; Robertson and Cherry, 1995; Robertson et al., 2000) to promote heterotrophic denitrification, i.e. (Delwiche, 1981):

$$4NO_3^- + 5CH_2O \rightarrow 2N_2 + 5CO_2 + 3H_2O + 4OH^- \quad (1)$$

These field trials demonstrated a variety of barrier configurations including reactive layers installed below septic system infiltration beds, a reactive wall installed in the path of a groundwater plume, and containerized modules treating farm field drainage water. Several of these installations were monitored for extended periods (6 to 7 years) and indicated that barriers utilizing wood byproduct material (Nitrex^{TM} filters, Office of Research Technology Transfer and Licensing, University of Waterloo, Ontario, Canada) could potentially operate for many years without the need for carbon replenishment because of the slowly soluble nature of

Article was submitted for review in July 2003; approved for publication by the Soil & Water of ASAE in November 2004.

The authors are **William D. Robertson,** Research Associate Professor, Department of Earth Sciences, University of Waterloo, Waterloo, Ontario, Canada; **Greg I. Ford, ASAE Member Engineer,** President, Septech Environmental, Ltd., London, Ontario, Canada; and **Pio S. Lombardo, ASAE Member Engineer,** President, Lombardo Associates, Inc., Newton, Massachusetts. **Corresponding author:** W. D. Robertson, Department of Earth Sciences, University of Waterloo, Waterloo, Ontario, Canada N2L 3G1; phone: 519–888–4567, ext. 6800; fax: 519–746–7484; e–mail: wroberts@sciborg.uwaterloo.ca.

  ISSN 0001–2351

the cellulose and hemicellulose compounds contained in the wood media (Robertson et al., 2000). Nitrate reaction rates, assuming zero–order kinetics, were in the range 0.7 to 32 mg N L^{-1} d^{-1} at temperatures of 3°C to 20°C and NO_3–N concentrations of ~2 to 200 mg L^{-1}. Importantly, rates did not appear to deteriorate over the monitoring period. Although these reaction rates are modest compared to those achieved in some sewage treatment processes, for example those using liquid carbon amendments such as methanol (Sikora et al., 1978; USEPA, 1993; Kapoor and Viraraghaven, 1997; Koch and Siegrist, 1997), they are fast enough to allow relatively complete removal of typical sewage nitrogen levels using conveniently sized filters. In addition, these filters can be deployed in a manner that is passive and essentially maintenance free, which makes them attractive for use with smaller wastewater treatment systems such as septic systems. For this use, however, the longevity of the reactive media is crucial, as a need for frequent replenishment would generally make this technology less cost competitive.

Although the earlier field trials suggested that the wood–based media had the potential for considerable longevity, ultimate media life span could not be predicted with certainty because of the possibility of carbon consumption from competing reactions, such as sulfate reduction, and because of uncertainty in the fraction of the carbon that was in a form suitable for use by the denitrifying bacteria. Furthermore, these were not full–scale systems, and flow rates were relatively modest (6 to 2000 L d^{-1}). This article presents monitoring results at four sites where Nitrex filters have been in full–scale operation treating domestic sewage for periods of 3 to 5 years. These sites were chosen because they represent a variety of sewage sources and flow rates, and because they were the first four full–scale systems installed, they have the longest available monitoring records. Our objective was to demonstrate the longevity of the reactive media under full–scale operating conditions. In addition to nitrogen treatment, other wastewater parameters including biological oxygen demand (BOD), total suspended solids (TSS), soluble phosphorus (PO_4–P), and bacteria were assessed, and operation and maintenance experience is discussed.

Site Descriptions

The study sites are four septic systems installed during 1997–1999 at a house, a trailer park, a roadside inn, and a communal residence in southern Ontario. The house is occupied by two persons and has standard fixtures such as clothes laundering and automatic dishwashing facilities. Water usage was not measured directly but was estimated at ~1 m^3 d^{-1} based on typical household rates (MOE, 1982). The trailer park operates seasonally (May to October) and has wastewater generated by ~100 mobile home units. Sewage flow during the summer months averaged 7 m^3 d^{-1}, based on pumping records (Levenick, 2001). The low per–unit water usage rate (70 L d^{-1}) presumably reflects the intermittent occupancy of the trailer units (higher on weekends) and the fact that some water use activities such as showering and clothes laundering were probably done less frequently than is normal. The roadside inn has 71 motel units, a restaurant, and banquet facilities and is used throughout the year. Wastewater flow averaged 73 m^3 d^{-1}, based on pumping records. The communal residence consists of 35 townhouse units occupied by senior citizens. Wastewater flow averaged 18 m^3 d^{-1}, based on pumping records. The septic systems at all of the sites have similar components consisting of standard septic tanks followed by pump chambers from which the effluent is pumped to secondary treatment units consisting of either single–pass or recirculating sand filters (table 1). These are of standard design (e.g., Converse and Converse, 1998) and receive average hydraulic loading rates of 2 to 5 cm d^{-1} (0.5 to 1.2 usgpd ft^{-2}) for the single–pass sand filters and 19 cm d^{-1} (4.6 usgpd ft^{-2}) for the recirculating sand filter at the inn.

The sand filter effluent drains by gravity flow into the Nitrex filters, which are subsurface modules housed in either a concrete tank (house), or in lined excavations for the sites with larger flows (inn, trailer park, communal residence). The filters contain a reactive media consisting of particulate wood byproduct material (bark, sawdust, and woodchips) in the 0.5 to 50 mm dia. size range. This media has been selected because of its low cost, high permeability, high C:N ratio, and because its reactivity is such that the media maintains its physical properties and carbon availability for a number of years after installation. Effluent migrates through the media under saturated flow conditions so that anaerobic conditions necessary for denitrification can develop. The filters are sized to achieve hydraulic retention times on the order of 1 to 10 days, depending on factors such as wastewater composition, temperature, and treatment criteria. For example, a Nitrex filter treating sand filter effluent with NO_3–N concentration of 40 mg L^{-1} and a temperature of 10°C, would be sized to achieve ~3 days retention if the treatment goal was to lower NO_3–N to the drinking water limit of 10 mg L^{-1}. In each case, effluent from the Nitrex filter is collected in a pump chamber and pumped to a conventional tile bed. Figure 1 is a schematic of a typical Nitrex filter showing flow–through characteristics, while table 1 summarizes the design parameters of the important septic system components at each of the four sites. The only exception to this treatment sequence is the trailer park, where twin subsurface–flow constructed wetland cells were installed

Table 1. Septic system characteristics at the four study sites.

Site	Startup	Wastewater Source	Pretreatment Type	Pretreatment Area (m^2)	Pretreatment Loading Rate (cm d^{-1})	Nitrex Filter Size (m^3)	Flow Rate (m^3 d^{-1})
House	October 1997	STE[a]	Single–pass SF[b]	21	5	9	~1
Trailer park	July 1998	STE	Single–pass SF	310	2	108	7
Roadside inn	April 1999	STE	Recirculating SF	390	19	360	73
Communal residence	March 1999	STE	Single–pass SF	470	4	120	18

[a] STE = septic tank effluent.
[b] SF = sand filter.

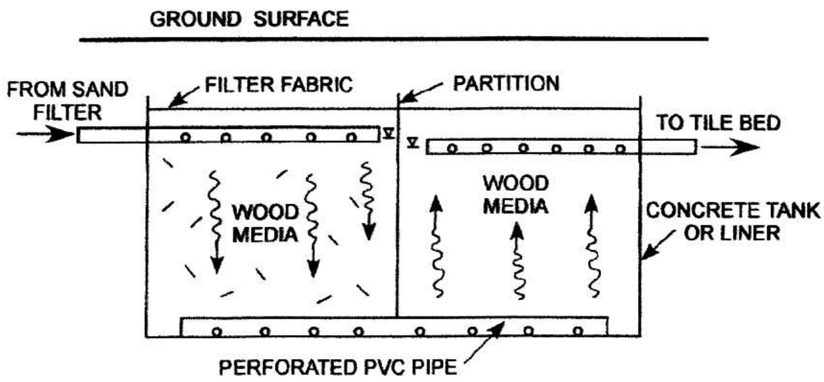

Figure 1. Schematic view of the Nitrex filter design and flow–through characteristics.

between the sand filter and the Nitrex filter. These are each 84 m^2 in surface area and were planted with reeds (*Phragmites austrialis*). Nitrex filter influent at this site is thus reed bed effluent, rather than sand filter effluent, although detailed monitoring in the first two years of operation (Levenick, 2001) indicated that the reed beds had only a minor influence on nitrogen concentrations (~20% removal).

Methods

Monitoring generally occurred monthly to quarterly at each site and focused on the Nitrex filter influent and effluent characteristics. Samples were retrieved using a peristaltic pump from access ports located in the inlet and outlet pipes. Samples for inorganic nitrogen analyses (NO_3 and NH_4) were generally filtered (0.45 μm) prior to atmospheric exposure and were collected untreated in 20 mL polyethylene containers. Samples for BOD and TSS were collected unfiltered in 500 mL glass bottles and were generally delivered to the laboratory for analysis within 8 h of collection. Samples for bacteria analyses were collected unfiltered in 250 mL plastic bottles containing sodium thiosulfate preservative and were also delivered to the laboratory within 8 h of collection. Samples for total Kjeldahl nitrogen (TKN) were collected unfiltered and untreated in 20 mL polyethylene containers. Samples for phosphate analyses were generally collected filtered and untreated in 20 mL polyethylene bottles.

Laboratory analyses of NO_3 and NH_4 were completed colorimetrically, using a Braun and Luebbe model 800 autoanalyzer. TKN analyses, which quantify the reduced N species present (organic N + NH_4), were completed after sulfuric acid digestion (APHA, 1992). Although TKN values include NH_4, ammonium was also analyzed separately, using sample splits, for greater precision. BOD analyses used a standard five–day incubation (APHA, 1992). *Escherichia coli* (*E. coli*) and total coliform bacteria were enumerated after membrane filtration and 24 h incubation of the filtrate on agar plate cultures.

A standard t–test, assuming normal distribution, was used to assess the significance of year–to–year changes in mean nitrate concentrations and the significance of differences in influent and effluent composition.

Denitrification rates were calculated using equations 2 and 3, which assume zero–order kinetics:

$$r = (C_i - C_o)/t \tag{2}$$

$$t = Vn/Q \tag{3}$$

where

r = reaction rate (mg N L^{-1} d^{-1})
C_i = influent NO_3–N concentration (mg L^{-1})
C_o = effluent NO_3–N concentration (mg L^{-1})
t = hydraulic retention time (days)
V = volume of filter (L)
n = effective porosity of filter media
Q = volumetric hydraulic loading rate (L d^{-1}).

Results and Discussion

Nitrogen

Figure 2 compares Nitrex filter influent and effluent NO_3 concentrations over 3 to 5 years of operation at the four sites, while table 2 summarizes yearly mean NO_3 and NH_4 concentrations. Influent NO_3–N concentrations ranged from an average of 14.2 mg L^{-1} at the inn to 37.7 mg L^{-1} at the trailer park, while average effluent values ranged from 0.7 to 1.9 mg L^{-1}. Nitrate concentrations in the treated effluent were significantly lower ($p < 0.05$) at each site and in each year of operation at each site, except for those years when monitoring occurred infrequently and influent nitrate concentrations were variable (inn year 4, trailer park years 1 and 4, and house years 4 and 5). Overall nitrate attenuation, considering all sites and weighted by number of samples (109 total), was 96% (table 3) and ranged from 87% at the inn to 98% at the trailer park (table 2). Overall attenuation, on a yearly basis, ranged from 97% in year 3 to 93% in year 5 (table 3). NO_3–N concentrations in the treated effluent, considering all sites, averaged 0.8 mg L^{-1} in year 5, which was lower than any other year except year 3 (table 3). Again considering all sites, nitrate concentrations in the treated effluent in years 2, 3, 4, and 5 were not significantly different ($p > 0.05$) from that of year 1. Thus, there was no indication of declining nitrate treatment with system age at any of the sites. Similar amounts of nitrate removal (>90%) were observed in two Nitrex filters treating household wastewater in Oregon (ODEQ, 2004).

Figure 3 shows that occasional higher NO_3–N spikes (up to 9.3 mg L^{-1}) occurred during winter operation at the inn when colder temperatures decreased denitrification rates. For the four elevated NO_3 values shown in figure 3, which

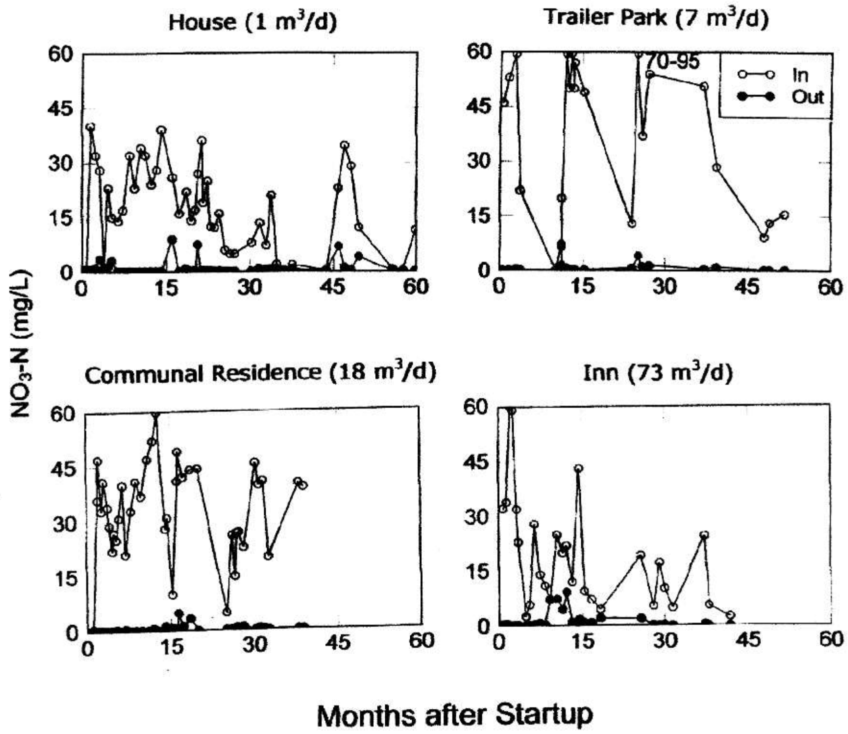

Figure 2. Comparison of Nitrex filter influent (In) and effluent (Out) NO_3 concentrations at the four sites during 3 to 5 years of operation. Influent is sand filter effluent except at the trailer park site where it is reed bed effluent.

Table 2. Site–by–site yearly mean NO_3 and NH_4 concentrations in the sand filter (In) and Nitrex filter (Out) effluents and percent removal (%).

	House				Trailer Park				Roadside Inn				Communal Residence			
Year	*n*[a]	In	Out	%	*n*	In	Out	%	*n*	In	Out	%	*n*	In	Out	%
NO_3–N (mg L^{-1})																
1[b]	7	23.7	0.54	98	4	24.0	0.8	95	7	15.9	3.0	81	9	36.2	0.3	99
2	14	22.7	1.4	94	6	52.4	0.3	99	6	16.4	2.7	84	9	39.5	1.5	96
3	11	8.5	0.46	95	4	50.4	1.9	96	5	11.5	0.6	95	10	27.6	0.4	99
4	4	15.0	2.2	85	2	18.9	0.6	97	3	10.9	0.2	98	2	39.5	0.2	99
5	5	10.7	1.1	90	2	14.4	0.1	99	—	—	—	—	—	—	—	—
All	41	16.9	1.0	94	18	37.7	0.8	98	21	14.2	1.9	87	30	34.8	0.7	98
NH_4–N (mg L^{-1})																
1	7	0.56	0.37	34	4	0.25	0.70	−280	6	1.1	0.7	36	8	5.4	1.8	66
2	12	0.76	0.32	58	5	1.2	4.4	−270	6	2.5	2.7	−8	9	2.1	1.2	43
3	11	4.7	0.66	86	4	1.9	0.45	76	5	6.0	1.5	75	9	9.0	9.0	0
4	4	5.3	3.0	43	2	16.7	3.2	81	3	2.2	0.9	59	2	2.5	0.1	96
5	5	6.0	4.9	18	2	22.4	26.2	−17	—	—	—	—	—	—	—	—
All	39	3.0	1.2	60	17	5.2	5.3	−2	20	3.0	1.6	47	28	5.3	3.8	28

[a] *n* = number of samples.
[b] Data from first five months of operation (startup period) are excluded from all tables, but are included in the figures.

Table 3. All–site year–by–year mean NO_3, NH_4, and inorganic (NH_4+NO_3) N concentrations in the Nitrex filter influent (In) and effluent (Out) and percent removal (%).

		All Four Sites (mg L^{-1})								
		NO_3–N			NH_4–N			Inorganic N		
Year	*n*	In	Out	%	In	Out	%	In	Out	%
1	26	25.9	1.1	96	2.1	0.9	57	28.0	2.0	93
2	35	31.0	1.5	95	1.5	1.6	−07	32.5	3.1	90
3	30	21.0	0.7	97	5.9	3.4	42	26.9	4.1	85
4	11	19.1	1.0	95	6.0	2.5	58	25.1	3.5	86
5	7	11.9	0.8	93	10.7	11.0	−03	22.9	11.8	49
All	109	24.9	1.1	96	4.0	2.7	32	28.9	3.8	87

were for temperatures of 5 °C to 10 °C (months 9 to 12, January to April 2000), the average denitrification rate was 7 mg N L^{-1} d^{-1}. However, winter nitrate spikes did not occur at all the sites. For example, at the communal residence during months 8 to 12 of operation (November 1999 to March 2000), effluent temperatures ranged from 3 °C to 6 °C, but NO_3–N still remained <1 mg L^{-1} in the treated effluent (fig. 2). This indicated that denitrification remained active at these lower temperatures, which is consistent with previous studies indicating that septic system biodegradation reactions can remain active even during the winter months in cold climates (Viraraghaven, 1977). Better winter nitrate treatment at the

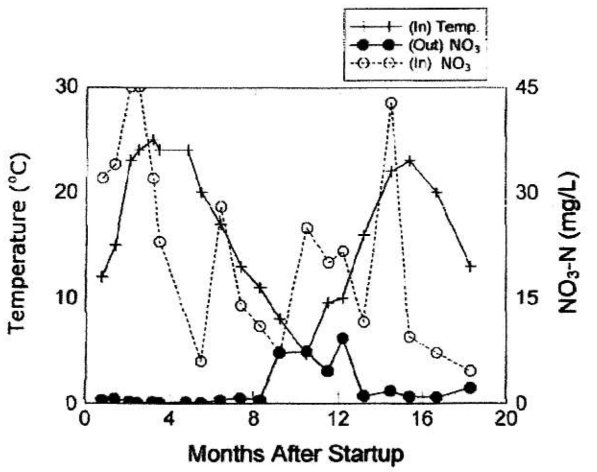

Figure 3. NO_3 treatment during winter operation at the inn. Months 8 to 12 represent December 1999 to April 2000.

communal residence was presumably a result of the lower hydraulic loading rate (table 1), which resulted in a longer retention time in the Nitrex filter.

Table 4 presents a more detailed suite of parameters for the communal residence site, including TKN, soluble PO_4, Cl, SO_4, BOD, TSS, pH, total coliform bacteria, and *E. coli*, which allows organic nitrogen (TKN − NH_4–N) and total nitrogen (TKN + NO_3–N) to also be determined. The communal residence data indicate that there was possibly significant ($p = 0.2$) organic N in the septic tank effluent (mean TKN − NH_4–N = 7.0 mg L^{-1}) but that there was less than 3 mg L^{-1} of organic N in both the sand filter and Nitrex filter effluent, representing <5% of the initial TN in the septic tank effluent. Presumably organic N was mineralized to NH_4 and then oxidized to NO_3 in the sand filter. Although relatively complete conversion to NO_3 was indicated in the sand filter effluent (34.8 mg NO_3–N L^{-1} vs. 5.2 mg NH_4–N L^{-1}), the difference in TN between the septic tank and sand filter effluent (60.7 vs. 34.6 mg L^{-1}) indicated that nitrogen loss of 43% occurred during sand filter treatment. This may be the result of NH_3 volatilization or denitrification within microsites and is typical of nitrogen treatment in sand filters (Converse and Converse, 1998). In another study where two Nitrex filters treated household wastewater in Oregon, similarly low organic N was observed in both the sand filter and Nitrex filter effluent (ODEQ, 2004). At these two sites, TKN in the sand filter and Nitrex filter effluent was significantly higher than NH_4–N ($p < 0.05$) but only by a small amount, averaging 0.95 to 1.5 mg L^{-1}, again indicating that organic N in both the sand filter and Nitrex filter effluent represented <5% of the septic tank TN. Thus, although organic nitrogen was not measured at all of our sites, we feel that the fate of most of the nitrogen was well established based on NO_3 and NH_4 analyses alone. However, when treatment to very low levels of TN is desirable (<5 mg L^{-1}), more comprehensive analyses, including TKN, should normally be undertaken during monitoring.

For most sampling episodes, precise reaction rates could not be established because NO_3–N concentrations were depleted to low levels in the treated effluent (<1 mg L^{-1}); thus rates became nitrate-limited. The highest rates can be inferred for the communal residence and the trailer park sites

Table 4. Means and standard deviations of the septic tank effluent (STE), sand filter effluent (In), and the Nitrex filter effluent (Out) during four years of operation (March 1999 to November 2002) at the communal residence.

		Communal Residence			
	Unit	n[a]	STE	In	Out
TN[b]	mg L^{-1}	3 to 4	60.8 ±2.6[c]	34.6 ±8.5	<4.6
TKN	mg L^{-1}	3 to 4	60.7 ±3	<3.7	<4.4
NH_4–N	mg L^{-1}	6 to 29	49.5 ±9.3	5.2 ±6.1	3.7 ±6.0
NO_3–N	mg L^{-1}	6 to 31	<0.1	34.8 ±11	0.7 ±1.0
PO_4–P	mg L^{-1}	4 to 5	7.1 ±2.0	4.5 ±1.2	3.2 ±0.3
Cl	mg L^{-1}	5 to 6	227 ±40	218 ±60	216 ±55
SO_4	mg L^{-1}	2	—	68 ±5	7.9 ±0.9
BOD	mg L^{-1}	6 to 25	207 ±27	<4 ±2	73 ±61
TSS	mg L^{-1}	2 to 5	70 ±36	7 ±10	9 ±14
pH	—	5 to 6	7.6 ±0.4	7.0 ±0.4	6.4 ±0.2
E. coli	CFU/100 mL	5 to 8	>12,000	560 ±520	320 ±790
Total coliform	CFU/100 mL	4 to 8	>15,000	>9900 ±8500	1400 ±1800

[a] n = number of samples; excludes results from first five months of operation, Cl (In and Out) from 18 April 2001, and pH (Out) from 16 November 2000.
[b] TN = total nitrogen = TKN + NO_3–N, calculated using only sampling events when both TKN and NO_3 were determined.
[c] Standard deviation.

(>10 mg N L^{-1} d^{-1}), where influent NO_3–N values were relatively high (34.8 to 37.7 mg L^{-1} average, table 2). These higher rates pertain to effluent temperatures of 15°C to 21°C at the trailer park, but also occurred at temperatures as low as 3°C at the communal residence.

Modest NH_4 concentrations were occasionally present in the sand filter effluent, reflecting periodic incomplete oxidation of the sewage nitrogen. Average NH_4–N concentrations in the sand filter effluent ranged from 2.9 mg L^{-1} at the inn to 5.3 mg L^{-1} at the communal residence, whereas NH_4–N in the treated effluent averaged 1.2 mg L^{-1} (house) to 5.3 mg L^{-1} (trailer park, table 2). Although at three of the four sites, modest 1 to 2 mg L^{-1} average reductions in NH_4–N were noted in the treated effluent (table 2), at only one site (house) was the difference significant ($p < 0.05$). Thus, these data indicate that Nitrex filters have little or no effect on NH_4 concentrations. At the house and trailer park sites, increasing NH_4 concentrations in the sand filter effluent during years 3 to 5 indicated deteriorating sand filter performance with system age at these sites.

BOD

Nitrex filter effluent typically has high BOD (>100 mg L^{-1}) during startup and for up to several months thereafter, as the soluble organic constituents (tannic acids, etc.) are leached from the reactive media. However, these soluble constituents comprise only 1% to 2% of the wood mass (Browning, 1963), and subsequently BOD generally stabilizes at much lower values (10 to 40 mg L^{-1}, fig. 4) depending on factors such as hydraulic retention time, temperature, and wastewater characteristics. Although BOD values, particularly during startup, exceed values considered acceptable for tertiary treated effluent (10 mg L^{-1}), they remain generally equivalent to or lower than values in normal septic tank effluent (120 to 350 mg L^{-1}; Canter and Knox, 1985). The onset of sulfate-reducing conditions, which may occur if effluent retention time extends beyond when nitrate is fully depleted, also appears to increase BOD. At the communal

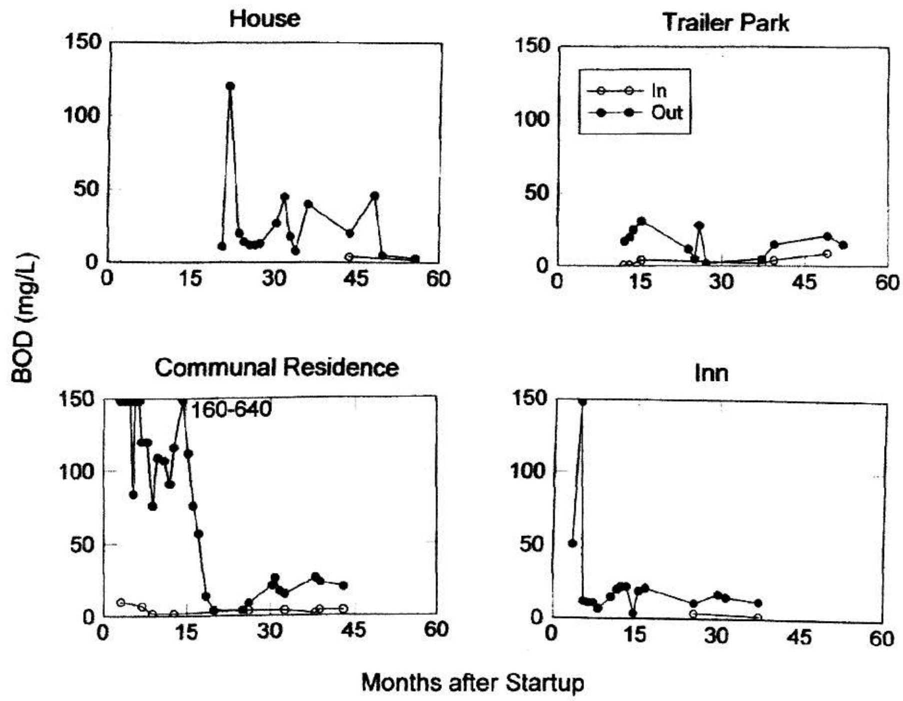

Figure 4. Comparison of biological oxygen demand (BOD) in the Nitrex filter influent (In) and effluent (Out) at the four sites.

residence, depleted SO_4 concentration in the Nitrex filter effluent (7.9 vs. 68 mg L^{-1} in the influent, table 4) suggests that sulfate reduction was active. Sulfide oxygen demand may thus contribute to the BOD at this site. Sulfate reduction appears to be more active during the summer, when higher temperatures lead to increased reaction rates (Robertson et al., 2000) and nitrate is depleted faster. This is reflected in occasionally higher BOD values during the summer months, for example in month 15 at the communal residence (June, 2000), when BOD increased temporarily to 220 mg L^{-1} (fig. 4) as the effluent temperature increased from 6°C to 13°C in the preceding six weeks. However, any sulfide that is present in the treated effluent is presumably reoxidized to SO_4 when the effluent is dispersed in the tile beds.

Bacteria

Figure 5 compares *E. coli* concentrations in the Nitrex filter influent and effluent at the four sites, while table 4 summarizes total coliform and *E. coli* concentrations observed at the communal residence. The majority (79%) of the 24 Nitrex filter effluent samples tested had no detectable *E. coli* (<10 colony forming units (CFU)/100 mL), indicating that this bacteria is attenuated in the filter. However, several effluent samples had elevated *E. coli* counts of up to 4200 CFU/100 mL (month 44 at the inn, fig. 5). It should be noted that most of the *E. coli* from the communal residence septic tank was attenuated in the sand filter (table 4); thus, concentrations entering the Nitrex filters were relatively low, in the range of 20 to 2000 CFU/100 mL (fig. 5).

Operation and Maintenance

In September 2001 (month 29 of operation), the upper surface of two of the three Nitrex cells at the communal residence were uncovered using a backhoe to determine the cause of an apparent blockage that was impeding drainage from the sand filters. Visual inspection revealed that the reactive media, in a ~15 cm thick zone immediately underlying the perforated PVC pipes that load the Nitrex filters, had been degraded to the extent that permeability was substantially reduced. The degraded material was excavated and replaced with coarse–grained wood particle media, after which normal function of the sand filters and Nitrex filters resumed (fig. 2). At the other sites where a coarser wood media was selectively placed adjacent to the influent pipes at the time of construction, similar backup problems did not occur. At these sites, no maintenance was required other than routine servicing of the pumps and associated filters and purging of the pressure distribution systems in the sand filters.

Conclusions and Implications

These installations have demonstrated the ability of passive wood media filters to achieve consistently high nitrate removal in septic systems during full–scale, all–season operation. Nitrate removal averaged 96%, which is superior to most other passive treatment methods we are aware of, and resulted in average annual NO_3–N concentrations in the treated effluent of ≤3 mg L^{-1} at all four sites. Total inorganic nitrogen removal was slightly less, averaging from 80% to 89% at the four sites, as a result of occasional incomplete nitrification of NH_4 during pretreatment in the sand filters and because Nitrex filters do not treat NH_4. Nonetheless, inorganic nitrogen (NO_3 + NH_4) averaged ≤6.1 mg N L^{-1} in the treated effluent at each of the sites (table 2), which would be adequate when the treatment goal is to lower N to below the NO_3–N drinking water limit of 10 mg L^{-1}. Reaction rates observed during full–scale

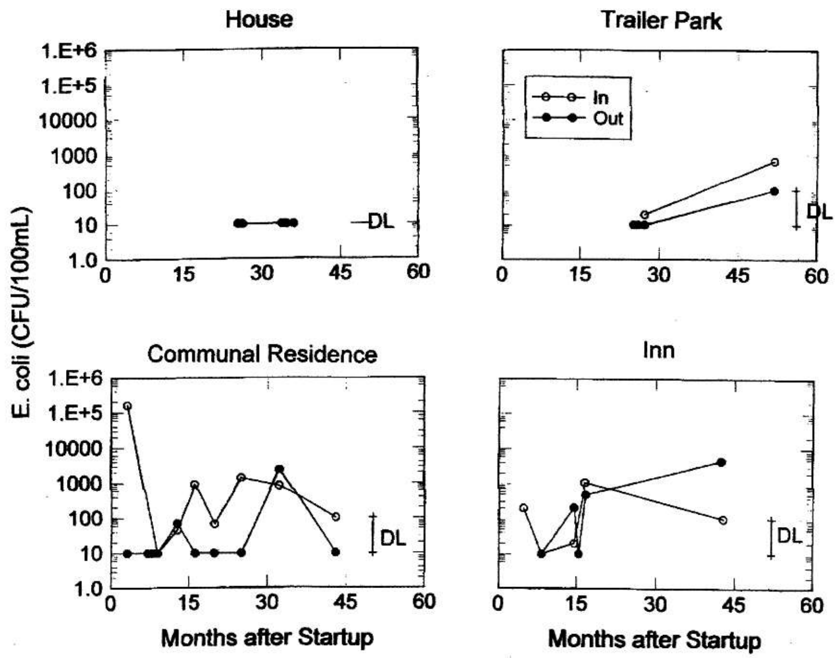

Figure 5. Comparison of *E. coli* concentrations in the Nitrex filter influent (In) and effluent (Out) at the four sites. DL is detection limit range.

operation at these sites (7 to >10 mg N L^{-1} d^{-1}) were consistent with other field trials using similar wood–based media (Schipper and Vojvodic–Vukovic, 1998; Robertson et al., 2000). Treatment consistency with system age indicates that the media has the potential to remain viable for at least five years during full–scale operation, which also supports the results of previous mass balance calculations using equation 1 and long–term pilot–scale monitoring (Robertson et al., 2000).

The primary negative side effect of these filters is the elevated BOD levels that are generated during the first several months of operation as the soluble organic constituents are leached from the reactive media. However, even during startup, BOD values (several hundred mg L^{-1}) were not substantially higher than normal septic tank effluent, and TSS values were much lower (e.g., 9 vs. 70 mg L^{-1}, table 4); thus, temporarily elevated BOD values should not be of concern when final dispersal occurs to a normally sized tile bed. However, in cases were high initial BOD concentrations are considered unacceptable, it may be feasible to preleach the reactive media, prior to installation, without disrupting long–term treatment effectiveness. During longer–term operation, BOD normally stabilizes at much lower levels (10 to 40 mg L^{-1}) depending on retention time and temperature. Current filter designs now incorporate compartmentalization features to better control retention times and hence final BOD values. Monitoring evidence indicates that under field conditions, NO_3–N removal to <10 mg L^{-1} can be achieved on a regular basis, while maintaining BOD <20 mg L^{-1}. Maintenance associated with Nitrex filter operation at these sites was surprisingly modest considering that these were the first four full–scale systems installed. However, we would recommend that systems be installed with devices for measuring hydraulic backpressure as a means of monitoring for permeability deterioration in the wood particle media.

Nitrex filters have little or no effect on NH_4 concentrations; thus, successful effluent pretreatment in a well maintained sand filter, or other aeration device capable of relatively complete nitrification of wastewater nitrogen, is the key to the successful use of these filters.

REFERENCES

APHA. 1992. *Standard Methods for the Examination of Water and Wastewater.* 18th ed. Washington, D.C.: American Public Health Association.

Andreoli, A., N. Bartilucci, R. Forgione, and R. Reynolds. 1979. Nitrogen removal in a subsurface disposal system. *J. Water Pollution Control Federation* 51(4): 841–854.

Blowes, D. W., W. D. Robertson, C. J. Ptacek, and C. Merkley. 1994. Removal of agricultural nitrate from tile–drainage effluent water using in–line bioreactors. *J. Contaminant Hydrology* 15(3): 207–221.

Brooks, J. L., C. A. Rock, and R. A. Struchtemeyer. 1984. Use of peat for on–site wastewater treatment: II. Field studies. *J. Environ. Quality* 13(4): 524–529.

Browning, B. L. 1963. The composition and chemical reactions of wood. In *The Chemistry of Wood*, 57–102. B. L. Browning, ed. New York, N.Y.: Interscience Publishers.

Canter, L. W., and R. C. Knox. 1985. *Septic Tank System Effects on Groundwater Quality.* Chelsea, Mich.: Lewis Publishers.

Carmichael, P. A. 1994. Using wood chips as a source of organic carbon in denitrification: A column experiment and field study implementing the funnel and gate design. MSc thesis. Waterloo, Ontario: University of Waterloo, Department of Earth Sciences.

Converse, J. C., and M. M. Converse. 1998. Pump chamber effluent quality following aerobic units and sand filters servicing residences. In *Proc. 8th National Symposium on Individual and Small Community Sewage Systems*, 388–402. St. Joseph, Mich.: ASAE.

Delwiche, C. C. 1981. The nitrogen cycle and nitrous oxide. In *Denitrification, Nitrification, and Atmospheric Nitrous Oxide*, 1–17. C. C. Delwiche, ed. New York, N.Y.: Wiley Interscience.

Hines, M. W., E. R. Bennett, and J. A. Hoehne. 1978. Alternative systems for effluent treatment and disposal. In *Proc. 2nd National Home Sewage Treatment Symposium*, 137–148. St. Joseph, Mich.: ASAE.

Kapoor, A., and T. Viraraghavan. 1997. Nitrate removal from drinking water – Review. *J. Environ. Eng.* 123(4): 371–380.

Koch, G., and H. Siegrist. 1997. Denitrification with methanol in tertiary filtration at wastewater treatment plant Zürich–Werdholzli. *Water Science and Tech.* 36(1): 165–172.

Laak, R. 1981. Denitrification of blackwater with graywater. *J. Environ. Eng.* 107(3): 581–590.

Lamb, B. E., A. J. Gold, G. W. Loomis, and C. G. McKiel. 1991. Nitrogen removal for on–site sewage disposal: Field evaluation of buried sand filter/graywater systems. *Trans. ASAE* 34(3): 883–889.

Levenick, J. L. 2001. Geochemical and hydraulic controls affecting nutrient removal in a subsurface flow constructed wetland system. MSc thesis. Waterloo, Ontario: University of Waterloo, Department of Earth Sciences.

MOE. 1982. Manual of policy, procedures, and guidelines for private sewage disposal systems. Technical report. Toronto, Ontario: Ministry of the Environment.

ODEQ. 2004. LaPine demonstration project database. Oregon Department of Environmental Quality. Available at: www.deq.state.or.us/wq/onsite/lapine.html.

Robertson, W. D., and J. A. Cherry. 1995. In situ denitrification of septic–system nitrate using reactive porous media barriers: Field trials. *Ground Water* 33(1): 99–111.

Robertson, W. D., J. A. Cherry, and E. A. Sudicky. 1991. Ground–water contamination from two small septic systems on sand aquifers. *Ground Water* 29(1): 82–92.

Robertson, W. D., D. W. Blowes, C. J. Ptacek, and J. A. Cherry. 2000. Long–term performance of in situ reactive barriers for nitrate remediation. *Ground Water* 38(5): 689–695.

Sandy, A. T., W. A. Sack, and S. P. Dix. 1987. Enhanced nitrogen removal using a modified recirculating sand filter. In *On–Site Wastewater Treatment: Proc. 5th National Symposium on Individual and Small Community Sewage Systems*, 161–170. St. Joseph, Mich.: ASAE.

Schipper, L. A., and M. Vojvodic–Vukovic. 1998. Nitrate removal from groundwater using a denitrification wall amended with sawdust: Field trial. *J. Environ. Quality* 27(3): 664–668.

Siegrist, R., M. Witt, and W. C. Boyle. 1976. Characteristics of rural household wastewater. *J. Environ. Eng.* 102(3): 533–548.

Sikora, L. J., J. C. Converse, D. R. Keeney, and R. C. Chen. 1978. Field evaluation of a denitrification system. In *Proc. 2nd National Home Sewage Treatment Symposium*, 202–207. St Joseph, Mich.: ASAE.

USEPA. 1993. Nitrogen control manual. Technical report EPA/625/R–93/010, Washington, D.C.: U.S. EPA.

Valiela, I., M. Geist, J. McClelland, and G. Tomasky. 2000. Nitrogen loading from watersheds to estuaries: Verification of the Waquoit Bay nitrogen loading model. *Biogeochemistry* 49(3): 277–293.

Viraraghaven, T. 1977. Influence of temperature on the performance of septic tank systems. *Water, Air, and Soil Pollution* 7(1): 103–110.

Vogan, J. L. 1993. The use of emplaced denitrifying layers to promote nitrate removal from septic effluent. MSc thesis. Waterloo, Ontario: University of Waterloo, Department of Earth Sciences.

Walker, W. G., J. Bouma, D. R. Keeney, and P. G. Olcott. 1973. Nitrogen transformation during subsurface disposal of septic tank effluent in sands: II. Ground water quality. *J. Environ. Quality* 2(4): 521–525.