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Denitrification System for Onsite Wastewater Treatment

by

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WATER RESOURCES RESEARCH CENTER
UNIVERSITY OF MASSACHUSETTS AT AMHERST

Denitrification System
for
Onsite Wastewater Treatment

Completion Report
Project No. G1428

by

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for

Water Resources Research Center
University of Massachusetts
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ABSTRACT

Conventional, properly working septic systems release significant amounts of nitrate nitrogen ($\text{NO}_3\text{-N}$) to the groundwater often in concentrations exceeding 10 mg L^{-1} . This study evaluated the performance of a pilot-scale peat bed system removing nitrogen from septic tank effluent. Intermittent flow reactors were packed with sphagnum peat and their performance compared with uniformly packed medium washed sand reactors representing conventional septic systems. In one-half of the reactors a water table was maintained in the lower portion of the reactor. All treatments were triplicated.

Primary municipal wastewater was dosed from a 1000-gal septic tank to the reactors at a rate of 3 cm day^{-1} . Inorganic NH_4^+ , NO_3^- , TKN and COD were measured biweekly for 7 months. Removal of COD was 91%, 92%, 70% and 76% for the sand reactors with and without a water table and the peat reactors with and without a water table respectively. COD appeared to be elevated in the peat systems, probably due to organic exudates from the peat material.

No net removal of N was observed in the sand reactors with or without a water table, and the peat reactors without a water table. The average output of $\text{NO}_3\text{-N}$ was 24 mg L^{-1} , 26 mg L^{-1} and 19 mg L^{-1} respectively. Peat reactors with a water table showed 21% N removal, with an average of 9 mg L^{-1} $\text{NO}_3\text{-N}$ in the effluent. Sorption and temporary immobilization of N in the peat system may have provided some removal, however, denitrification was assessed to be active in the partially saturated peat reactors. Based on a comparison of the performance of both unsaturated and partially saturated peat reactors, it can be concluded that denitrification played an active role in nitrogen removal. The peat process appeared to be an efficient sewage treatment system having significant potential in reducing groundwater pollution once appropriate design factors are further evaluated.

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CHAPTER I

INTRODUCTION

Septic systems are the preferred treatment technology for domestic wastewater where central sewage treatment facilities are not available. In the United States onsite sewage disposal systems are used by one quarter of all households (Chen, 1988). In Massachusetts, approximately 27% of all housing units are not connected to central sewage treatment facilities (Veneman, 1982). Over 60 million gallons of wastewater, containing some 11 metric tons of nitrogen (N), must be treated and disposed of into the Massachusetts environment on a daily basis, without groundwater or surface water pollution, and without negative public health impacts. When properly designed, located and operated, onsite septic systems provide effective and low cost wastewater treatment.

Increasingly, homes are built on land less suited for onsite wastewater disposal. Lot sizes in some of these areas are quite small as well. With greater density of onsite sewage disposal systems, even where soils are suitable, there are increased reports of groundwater contamination (Chen, 1988; Prins and Lustig, 1988). Areas using onsite sewage disposal systems commonly use groundwater for drinking water. Historically, planning officials have attempted to abate drinking water contamination through density control. This approach, however, does not address pollution problems that occur in already developed areas or where zoning does not exist. Additionally, the ability of modern septic systems to remove nitrogen from wastewater is limited, even when properly designed, installed, and operating. Whereas ammonium and organic N can be converted to nitrate, there is virtually no nitrate removal occurring in the soil leaching area because conditions needed for denitrification do not exist. These conditions are anaerobiosis and the presence of metabolically available carbon (Lance, 1972). The zone immediately below the gravel distribution area is aerobic and may have insufficient carbon to provide energy for the denitrifying organisms. Once in the groundwater, nitrate remains chemically unchanged.

Nitrate levels exceeding the federal drinking water standards of 10 mg L^{-1} may cause methemoglobinemia, also known as "Blue Baby Syndrome" (EPA, 1975; Menzer and Nelson, 1986). Subsequently, there is an increasing demand for alternative disposal and treatment systems in environmentally sensitive areas. These innovative systems oftentimes are complicated to operate and expensive to install. Several studies have been conducted on the removal efficiencies of onsite sewage disposal systems. Only recently have workers recognized the limitations of current technology for nitrate removal.

Some experimental field and laboratory systems using peat have shown excellent treatment results (Brooks et al., 1984; Rock et al., 1984; Baker, 1986). The use of peat (*Sphagnum* spp.) for treatment of septic tank effluent (STE), particularly to enhance denitrification, is attractive because of its potentially low costs and beneficial treatment potential. Peat filtration systems may provide an alternative for treatment of septic

tank effluent under certain situations (Viraraghavan, 1986). This report describes the results of a study aimed at developing a pilot scale sewage treatment system enhancing nitrogen removal.

STUDY OBJECTIVES

The primary objective of this study was to evaluate the potential of peat for onsite treatment and disposal of domestic wastewater, with the specific purpose of reducing the amount of $\text{NO}_3\text{-N}$ released into the groundwater. Specific study objectives were:

1. to assess the suitability of horticultural peat as a wastewater treatment medium in a pilot-scale study,
2. to evaluate the nitrogen removal efficiency of peat, and
3. to estimate the potential impact of innovative peat systems on groundwater quality.

CHAPTER II

METHODS AND MATERIALS

Physical Plant Layout

The project was conducted at the pilot sewage treatment plant on the University of Massachusetts-Amherst campus. This facility is located adjacent to the Amherst municipal sewage treatment facility, which provides the pilot plant with untreated, primary and secondary treated wastewater via underground pipes. To house the treatment reactors, a 5.2 x 7.3 m plastic Quonset hut was constructed. A 3780-L dual baffle concrete septic tank and 2300-L concrete wet well were installed in excavations adjacent to the Quonset hut. A small wooden shed was placed at one end of the Quonset, to house pumps, timers and collection equipment as well as to provide access to the facility. The full length of the center of the Quonset and the wooden shed was excavated to a depth of 1.2 m allowing placement of collection vessels, a septic tank effluent (STE) storage container, and effluent pumps. A heavy duty 5 kW heater (W.W. Grainger, Stock No. 2E669) and shutter mounted exhaust fan (W.W. Grainger, Stock No. 2C708A) were installed to prevent freezing during the winter and minimize extreme temperatures during warm weather conditions.

Hydraulic System

Primary effluent from the Amherst municipal wastewater treatment facility was dosed to the septic tank twice daily at approximately 567 L per dose to the pilot plant at a rate of about 38 L min^{-1} using a MoynoTM progressive cavity positive displacement pump (Robbins Myers, Model No. 1L6) with a 1 HP gearmotor. Prior to each dosing, a 2.5-cm self-priming centrifugal pump (W.W. Grainger, Stock NO. 2P390) transferred approximately 200 L from the outlet baffle of the septic tank to a 227-L high density polyethylene (HDPE) STE storage tank (Nalgene, Model No. 14100). Twice daily, a 0.3 HP submersible effluent pump (Zoeller Co., Model No. M53) was operated for 60 seconds pressurizing a 3.8-cm SCH 40 PVC force main. The total volume of STE pumped per dose was 134 L. The force main was split at 1 m from the pump into two 3.8-cm SCH 40 PVC sub-mains. At 2 m and thereafter every 1.25 m, six 3.8-cm tee connectors were installed. At each tee, a 3.8-cm to 0.8-cm reducer followed by a 2-cm QICTM ball valve (Hayward, Model No. QV10075S) was located just before the treatment reactors. Distribution laterals were constructed in a "race track" configuration 91 x 30.5 cm (length x width) from 2-cm diameter SCH 40 PVC. Ten 0.8-cm orifices were spaced at 20-cm intervals along the length of the distribution lateral.

Effluent loading was 3 cm day^{-1} ($0.74 \text{ gal ft}^{-2} \text{ day}^{-1}$) with a discharge pressure of 2 cm and was equilibrated by adjusting the 2-cm ball valves in front of each reactor. Each reactor drained through a 2-cm outlet into 2-cm Tygon tubing. The tubing was raised to 38 cm on the reactors with an induced water table, and subsequently drained into a 57-L HDPE carboy. The free draining reactors drained directly into the 57-L HDPE carboys. At the

shoulder of each carboy, a 1.25-cm overflow fitting was attached with Tygon tubing to a central 7.6-cm wastewater drain pipe, which also carried overflow from the septic tank and STE storage tank. All wastewater eventually flowed into a 2300-L concrete wet well from which it was pumped back to the Amherst municipal treatment facility.

Treatment Reactor Design

Twelve experimental treatment reactors were constructed to evaluate four treatments systems in triplicate (Figure 1). Systems I and II were constructed with medium sand. System I was free draining, whereas in System II the lower 38 cm of the reactor was kept saturated. Systems III and IV were constructed with peat moss packed in two densified layers over 8 cm of medium sand. The upper peat layer was 40 cm deep with a bulk density of 0.10 g cm^{-3} . The lower peat layer was 30 cm deep and had a bulk density of 0.12 g cm^{-3} . System IV was saturated in the lower 38 cm, whereas system III was free draining (Figure 2).

The reactors (Figure 3) were constructed using 680-L HDPE tanks (Nalgene, Model No. 14100) with dimensions of 1.22 x 0.61 x 0.92 m (L x W x D). Air exchange piping was built using perforated 2.54-cm SCH 40 PVC pipe and installed at the 46 cm level of the reactors. In the reactors with peat, 0.6-cm galvanized hardware cloth was installed between the two peat layers and held in place by 2.54-cm SCH 40 PVC pipe attached to the air exchange piping.

The experimental reactors were fitted with inlets and outlets before peat or sand was added. An 8-cm deep layer of medium sand was placed in the bottom of each reactor. A sump around the bottom outlet was made using 1.28 cm washed crushed limestone.

Prior to filling the reactors with peat, the moss was placed in garbage bags. Subsamples from each bag were oven-dried at 105°C for 24 hours and the moisture content was determined. The peat with known dry weight then was added to the reactors. The peat was mixed and fluffed by hand to ensure homogeneity. The layer was compressed with the hardware cloth to the appropriate thickness and the air exchange piping was installed. The upper peat layer was added in the same manner until the upper level reached the 78 cm level. Ten centimeter wide strips of geotextile were placed under the distribution laterals to prevent channeling.

The sand reactors were constructed by adding sand instead of peat. Grab samples were taken during packing for determination of the moisture content. Upon determining the moisture content at 105°C after 24 hours, additional sand was added to provide a final bulk density of 1.4 g cm^{-3} . Ten centimeters of 1.28-cm washed, crushed limestone was added over the sand, and the distribution laterals were installed.

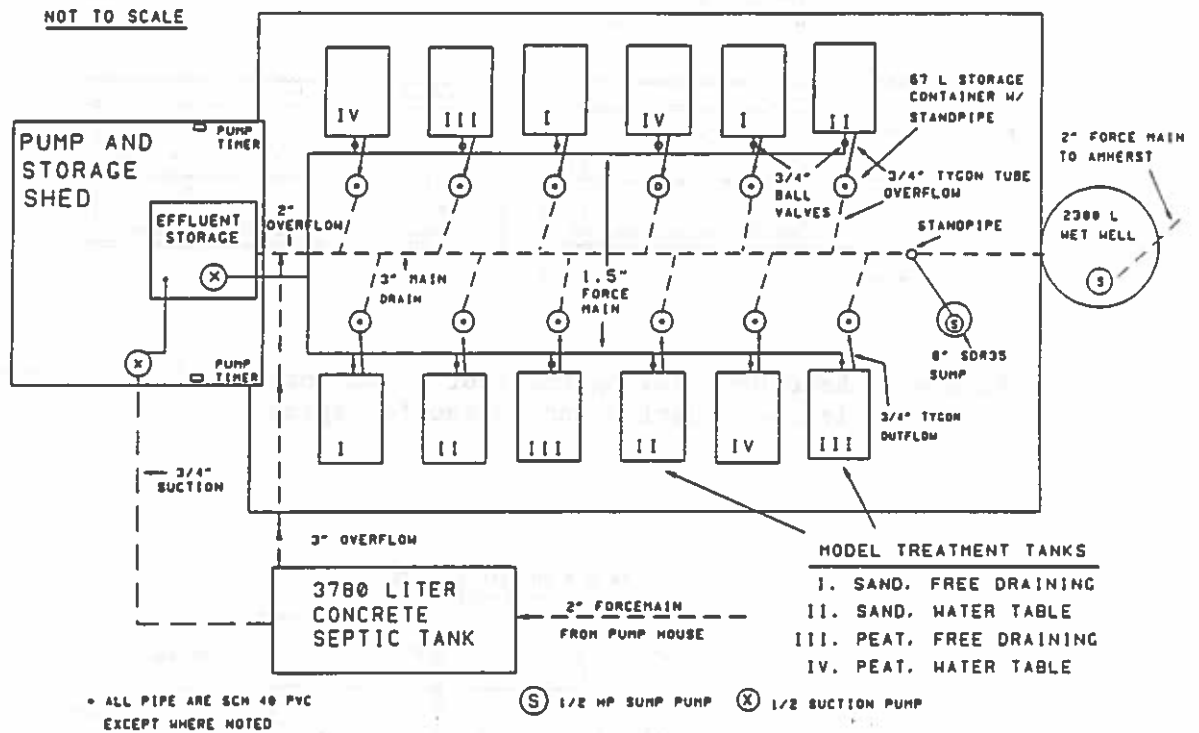


Figure 1. Plan view of the sewage distribution system between reactors.

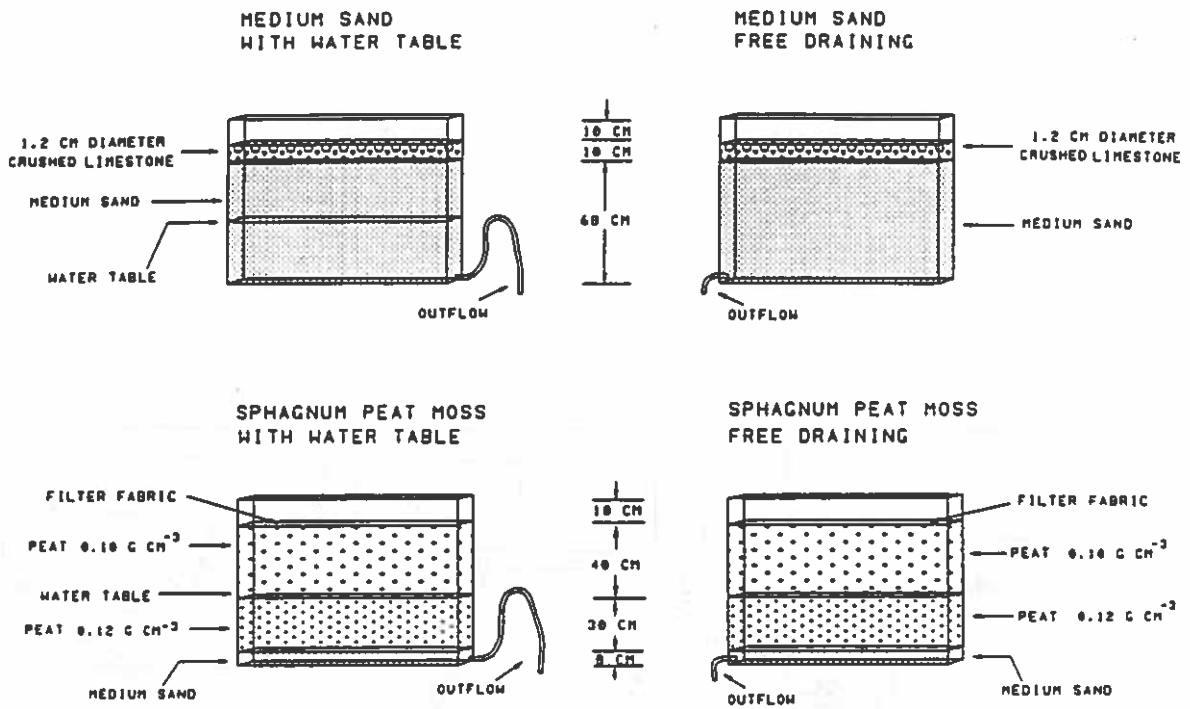


Figure 2. Reactors showing the four treatments resulting in differences in construction and hydraulic regime.

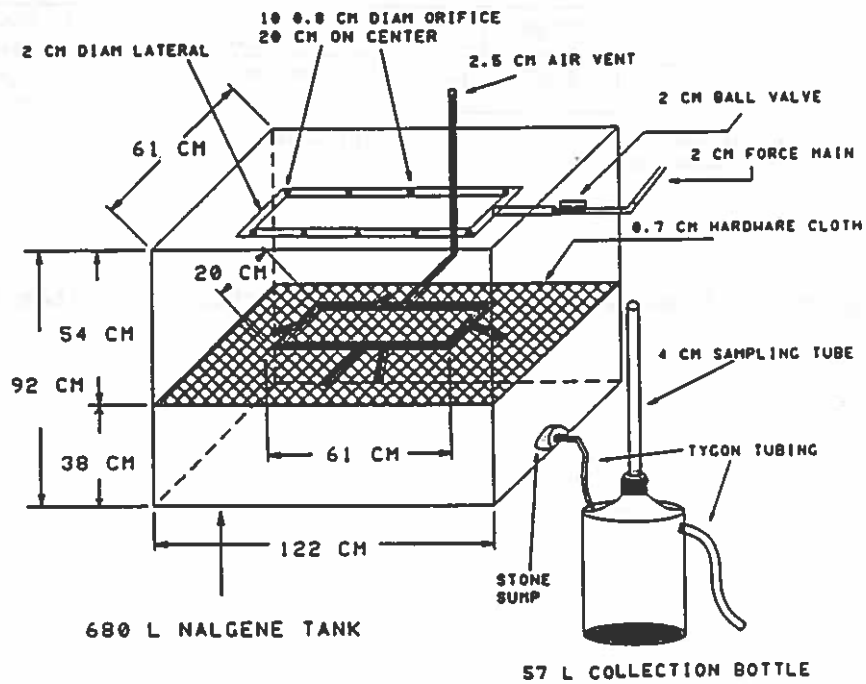


Figure 3. Schematic of a treatment reactor showing lateral, air duct, depth of materials and drain outlets.

Sampling

One day prior to each sampling date, the sampling carboys were emptied using a Teel self priming utility pump (W.W. Grainger, Model No. 1P579) with a 2-cm SCH 40 PVC evacuating tube. The following day, samples were collected using the same pump but with a shorter 2-cm sampling tube. Prior to collecting the actual sample, about 3 L of sample was flushed through the pump, including a 1 L rinse of the glass collection container. STE from the storage tank was collected on the same day after the treatment reactors had been sampled. Effluent samples for N and COD analysis were collected in glass containers and returned to the laboratory within 2 hours.

In the laboratory, a subsample was taken for measurement of pH, another portion was acidified to pH 1.5 - 2.0 with 36 N H_2SO_4 and stored at 4°C, and a 125-ml sample was frozen for future analysis. All samples were analyzed within the time period specified by the American Public Health Association (1985).

Analytical Methodology

Effluent pH was measured immediately after sampling, with a Fisher combination hydrogen ion electrode. Total Kjeldahl Nitrogen (TKN) was determined by semi-micro Kjeldahl digestion of a 25-ml aliquot in H_2SO_4 using K_2SO_4 - $CuSO_4$ catalyst by block digestion (American Public Health Association, 1985). After digestion, NH_3 -N was determined by semi-micro steam distillation in 4% boric acid, followed by acid titration with 0.02 M HCl (American Public Health Association, 1985). Total oxidized N (NO_x -N) was determined in the remaining sample with NH_3 -N distillation using Devarda's alloy reduction and semi-micro steam distillation in 4% boric acid, followed by acid titration with 0.02 M HCl (American Public Health Association, 1985). Chemical Oxygen Demand (COD) was determined using the H_2SO_4 - $K_2Cr_2O_7$ reactor digestion method (Hach Company, Loveland, Co.) followed by titration with 0.1 N $Fe(NH_4)(SO_4)_2 \cdot 6(H_2O)$ as titrant for excess oxidizing agent (American Public Health Association, 1985).

Chemical and Physical Properties of Sand and Peat

Soil texture, coefficient of uniformity and gradation coefficient were determined using standard dry sieving techniques on particles smaller than 2 mm with 0.07-, 0.1-, 0.25-, 0.5-, 1- and 2-mm sieves (Means and Parker, 1963). Acidity (pH) was measured in 0.01 M $CaCl_2$. Cation exchange was determined as the sum of extractable acidity and exchangeable bases using $BaCl_2$ at pH 8.1 and NH_4OAc at pH 7 as leaching agents respectively (Peech et al., 1962). Organic matter was determined on samples, initially dried at 104°C (Davies, 1974). Carbon and nitrogen analysis was performed according to the modified Pregal Dumas combustion technique (Ma and Rittner, 1979), using a Control Equipment Model No. 440 elemental analyzer. Fiber content of the peat was measured using a manual wet sieving procedure using 0.1-, 0.25-, 0.5-, 1- and 2-mm sieves respectively (Boelter, 1969).

Peat Hydraulic Conductivity

Constant head permeameters, 5 cm in diameter and 25 cm tall (similar to Soil Test, Model No. K600) were packed with peat at a known moisture content using a tremmie device. Saturated hydraulic conductivity (K_{sat}) was measured in the permeameters which were packed with peat at bulk densities of 0.125, 0.130, 0.135 and 0.140 g cm⁻³ respectively. All weights were on an oven-dry basis (104°C for 24 hours). Three permeameters for each treatment were placed randomly along the laboratory bench top. A Marriot device was used to feed all 12 permeameters through a central connection to two plastic tubing manifolds, each connected to six permeameters. The six permeameters were operated at the same time by closing a ball valve at the head of one of the two manifolds. Conductivity was measured twice daily for two weeks, followed by daily measurements for nine weeks and then once every three days for one week. Prior to each measurement, a minimum of one pore volume was allowed to pass through each permeameter, after which 3 successive 50-ml volumes were timed and averaged for that period. Darcy's law was used to calculate K_{sat} using the following equation:

$$K_{sat} = \frac{Q.L}{A.t.H} \quad (\text{Equation 1})$$

Where K_{sat} - saturated hydraulic conductivity in cm sec⁻¹; Q - volume of water passing through column in ml; t - length of time of measurement in seconds; A - cross sectional area of column in cm², L - length of column in cm, and H - length of column plus head of water in cm (Boelter, 1965).

Residency Time Study

A residency time study was conducted on reactors 1, 2, 7 and 8, representing peat with a water table, peat without a water table, sand without a water table, and sand with a water table respectively. Bromide (Br⁻) as KBr, was used as the tracer element. A 5% solution was added to the STE using a peristaltic pump at a rate of 0.44 ml sec⁻¹. After dilution with incoming nonbrominated STE, the Br⁻ concentration in the STE averaged 40 mg L⁻¹. The reactor outlets were equipped with ball valves to allow for sampling of effluent as it left the reactor. Twelve hours after initial loading and every twelve hours after that, a 20-ml sample was taken from the outlet of the reactors. A sample of the STE was also taken at the same time, using a peristaltic pump. Bromide was measured on 10-ml sample aliquots mixed with 10 ml of 5 M NaNO₃ using an Orion selective ion electrode (Model No. 943500). Percent bromide breakthrough was expressed as:

$$\% \text{ Breakthrough} = \frac{C_t}{C_o} \times 100\% \quad (\text{Equation 2})$$

Where C_t - concentration at time of sampling and C_o - concentration in STE going into the reactor. Concentrations were plotted and residency time was recorded at 100% breakthrough.

Acetylene Blocking

Denitrification was measured by N_2O evolution using the acetylene blocking technique (Tiedje et al., 1989). One core was taken from reactor 1 and 4 respectively, both representing the peat treatment with a water table. A hole was cut through the reactor sidewall and plugged with a SCH 40 threaded endcap to allow access. Twenty-four hours prior to sampling, approximately one liter of effluent was taken from each reactor 10 cm below the water level, and sparged with acetylene overnight. A peat core was removed horizontally from each reactor 10 cm below the water level using a 2.5-cm diameter hole saw welded to a stainless steel tube of the same diameter. The core samples were taken from the middle of the reactor after removing 20 cm of peat with a 2.5-cm screw auger. Upon core removal, approximately 750 mg of the peat sample was transferred to a 125-ml erlenmeyer flask, flooded with the acetylene saturated effluent and stoppered with a rubber septum. The flasks were immediately moved to the laboratory where they were sparged with N_2 for 5 minutes. Subsequently, 3 ml of acetylene was injected into the head space of the flask. The flasks were allowed to equilibrate to ambient conditions for 6 hours. After 6, 12, 16 and 21 hours a 3.5-ml volume of air was added and a 3.5-ml air sample was taken subsequently from the headspace. The sample volume was reduced to 3 ml and injected in a 3-ml vacutainer (Fisher, Model No. 02-683-77). Gas samples were analyzed for N_2O using gas chromatography with an electron capture detection unit.

Statistical Analysis

Repeated measures analysis of variance was used to analyze differences in water quality data between sand and peat treatments (Wilkinson, 1989). Two between factors, material and water status; and one factor, time, were analyzed using the following expected mean squares model, where α = peat or sand material, β = water status, σ = time, and δ = independent and normally distributed (IND).

$$Y_{ijkl} = \mu + \alpha_i + \beta_j + \alpha_i\beta_j + \sigma_k + \alpha_i\sigma_k + \beta_j\sigma_k + \alpha_i\beta_j\sigma_k + \delta_{ijkl} \quad (\text{Equation 3})$$

Multiple comparison tests were performed on STE and effluent after treatment through the reactors, using Tukey's Studentized Range (HSD) test on means over time for each treatment reactor (SAS Institute Inc., 1985). SAS actually performs the Tukey-Kramer modification when uneven cell counts exist, as was the case in this experiment.

Repeated measure analysis of variance was also used to analyze data from the peat hydraulic conductivity study (SAS Institute Inc., 1985). One between factor, density; and one within factor, time, were analyzed using the following expected mean squares model, where α = density, σ = time, and δ = IND.

$$Y_{ijk} = \mu + \alpha_i + \sigma_j + \alpha_i\sigma_j + \delta_{ijk} \quad (\text{Equation 4})$$

Multiple comparison tests were performed on the four density treatments using Tukey's Studentized Range (HSD) test on means over time for each treatment column (SAS Institute Inc., 1985).

CHAPTER III

RESULTS AND DISCUSSION

System Startup and Operation

The piping and pressure distribution system was checked using tap water during July and August of 1989. Dosing with STE began on September 8, 1989. During the initial three months, 12 chemical feed pumps were used to dose the reactors. These pumps were gravity fed with the desired dose volume, 11 liters, and run twice daily at 12 hour intervals. The chemical feed pumps were run in sequence using a manifold connected to all the pumps. Each pump had a separate force main to each reactor. In November, it was observed that reactors 11 and 12 were not receiving their required dose. Over the next two weeks, the feed system was observed closely, after which the overall system was shut down due to inconsistent dosing. The feed system described in the methods section (Chapter II) was installed during a two-week period and was fully operational on December 6, 1989. On one occasion, December 14, reactor twelve was not dosed in the morning due to ice accumulation in the force main. On another occasion, December 27, the outlet to reactor 11 froze causing effluent to backup into the reactor. The outlet was dismantled, the ice removed, and the reactor was allowed to drain completely. The dosing system operated flawlessly from that date on. It was observed that the volume of the peat in the upper layer of this reactor had decreased slightly.

In addition to changes in the reactor feed system, the system which dosed STE to the storage chamber was modified after repeated clogging occurred in the suction pump feeding the storage container. A stone filter was installed in the outlet baffle in the septic tank. The baffle was constructed using 1.3-cm crushed washed limestone enclosed in 0.7-cm mesh wiring. The stone box was designed so it could be removed and cleaned. It has not been necessary to clean the box to date.

After 12 months of operation no organic mat had formed at the stone sand interface of the sand reactors. In one of the peat reactors with a water table, a slime layer formed at the surface. In all the peat reactors live moss was observed growing. Two times during the study period, weed plants were removed from the surface of the reactors. Algae were observed growing on the sidewalls of all of the reactors. A 5-10 cm capillary rise above the induced water table level was observed through the side wall in the peat reactors with a water table. No capillary rise was observed in the other treatment reactors.

Sand and Peat Description

Based on grain size distribution (Figure 4), the sand used in the reactors was a well graded, medium sand having a uniformity coefficient (D_{60}/D_{10}) of 4.4 and a gradation coefficient of 0.8 (Means and Parker, 1963). The pH of the sand was 5.2 in 0.01 M CaCl_2 . The cation exchange capacity of the sand was $1.2 \text{ cmol}(+) \text{ kg}^{-1}$. Organic matter content was less

than 0.1% as determined by loss on ignition (Davies, 1974). Total carbon and total nitrogen in the sand were 0.15% and <0.05% respectively, based on the modified Pregal Dumas combustion technique.

Percent fiber in the peat ranged from 87% to 93%. Based on Boelter's (1969) characterization of peat, this material may be described as fibric peat. The pH of the peat was 3.3 in 0.01 M CaCl_2 . The cation exchange capacity was $172 \text{ cmol}(+) \text{ kg}^{-1}$ based on oven-dried weight. Organic matter content was 97.9% by loss on ignition (Davies, 1974). Total carbon and total nitrogen in the peat were 50.48% and 0.69% respectively.

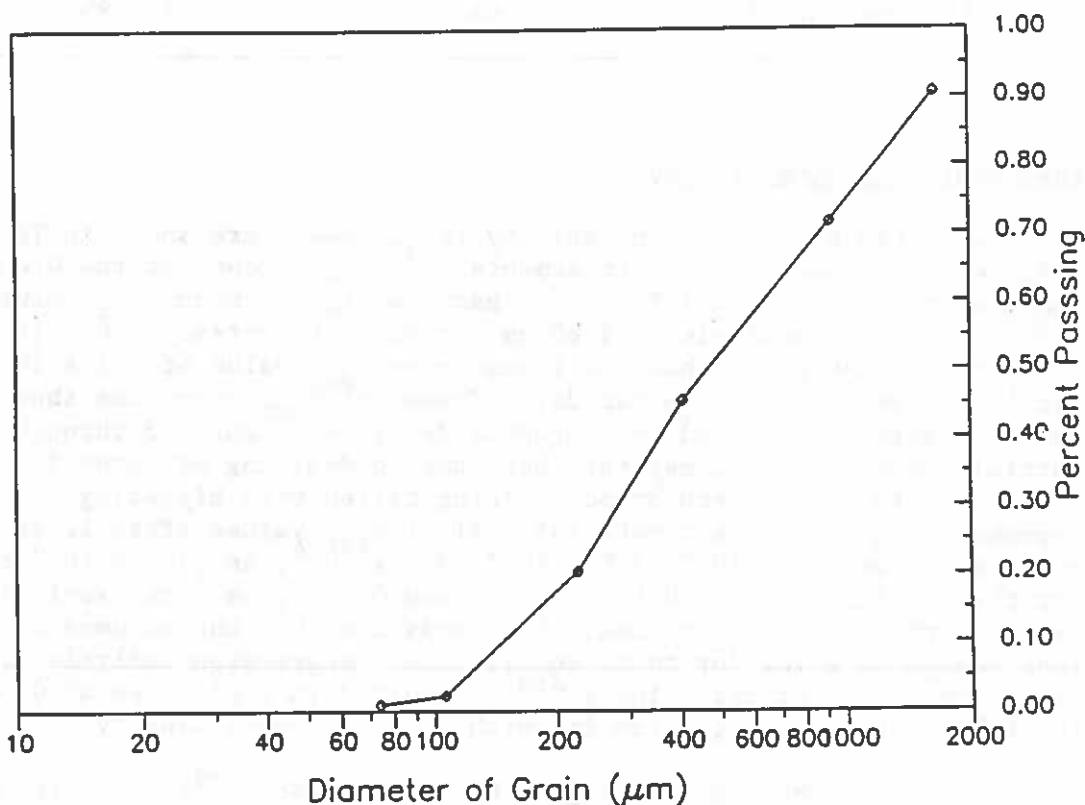


Figure 4. Grain size distribution curve of the sand used in the reactors.

Table 1. Description of sand and peat material used in treatment reactors.

Item	Sand	Peat
Description	Medium Sand	Fibric Peat (Canadian sphagnum)
Bulk density [g cm ⁻³]	1.4	0.10 - 0.12
pH	5.2	3.3
CEC [cmol(+) kg ⁻¹]	1.2	172
Organic matter [%]	0.1	97.9
Total carbon [%]	0.15	50.48
Total nitrogen [%]	<0.05	0.69
Uniformity coefficient	4.3	na
Gradation coefficient	0.8	na
Fiber content [%]	na	87 - 93

Peat Hydraulic Conductivity

Saturated hydraulic conductivity (K_{sat}) means are shown in Table 2. Differences between density treatments were significant at the 0.05 level. The lowest density of 0.125 g cm⁻³, had the highest mean K_{sat} value of 9.2×10^{-4} cm sec⁻¹, equivalent to 80 cm per day. Conversely, the highest density of 0.140 g cm⁻³ had the lowest mean K_{sat} value of 4.2×10^{-4} cm sec⁻¹, equivalent to 36 cm per day. Trends of K_{sat} over time showed an initial increase followed by a gradual decrease (Figures 5 through 8). Initial changes in K_{sat} may have been due to deairing of pores followed by a decrease in K_{sat} caused by pores being filled with migrating particles or decomposition of the peat material. Final K_{sat} values after 12 weeks of measurement were 2×10^{-4} , 3.0×10^{-4} , 3.5×10^{-4} , and 3.2×10^{-4} cm sec⁻¹, for the densities 0.125, 0.130, 0.135, and 0.140 g cm⁻³ respectively. Based on the final K_{sat} values, it is possible that the columns did not run long enough to allow for total equilibrium. Regression analysis of transformed K_{sat} values using $10^{K_{sat}}$, resulted in a R^2 value of 0.94 using the following linear equation in which bd is the bulk density:

$$\text{Mean } K_{sat} = \log [1.0113 - 0.07446(10^{bd})] \quad (\text{Equation 5})$$

Table 2. Mean K_{sat} and final K_{sat} of the peat used in the treatment reactors.

Bulk Density g cm^{-3}	Mean K_{sat} * $\text{cm sec}^{-1} \times 10^{-4}$	Mean K_{sat} cm day^{-1}	Final K_{sat} $\text{cm sec}^{-1} \times 10^{-4}$	Final K_{sat} cm day^{-1}
0.125	9.2 ^a	79.5	4.2 ^d	36.3
0.130	6.4 ^b	55.3	3.0 ^d	25.9
0.135	5.5 ^c	47.5	3.5 ^d	30.3
0.140	4.2 ^d	63.3	3.2 ^d	27.7

* numerical values with common letters within columns are not different at the 0.05 level of significance.

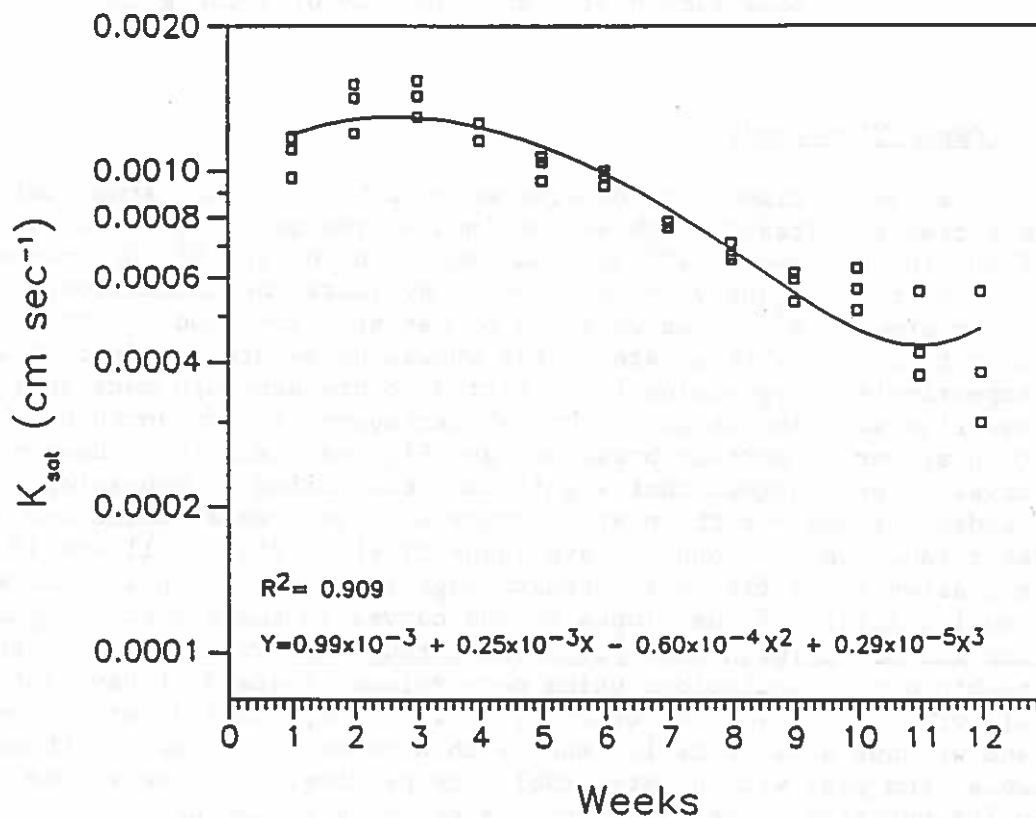


Figure 5. Change in hydraulic conductivity over time of sphagnum peat moss packed at a bulk density of 0.125 g cm^{-3} .

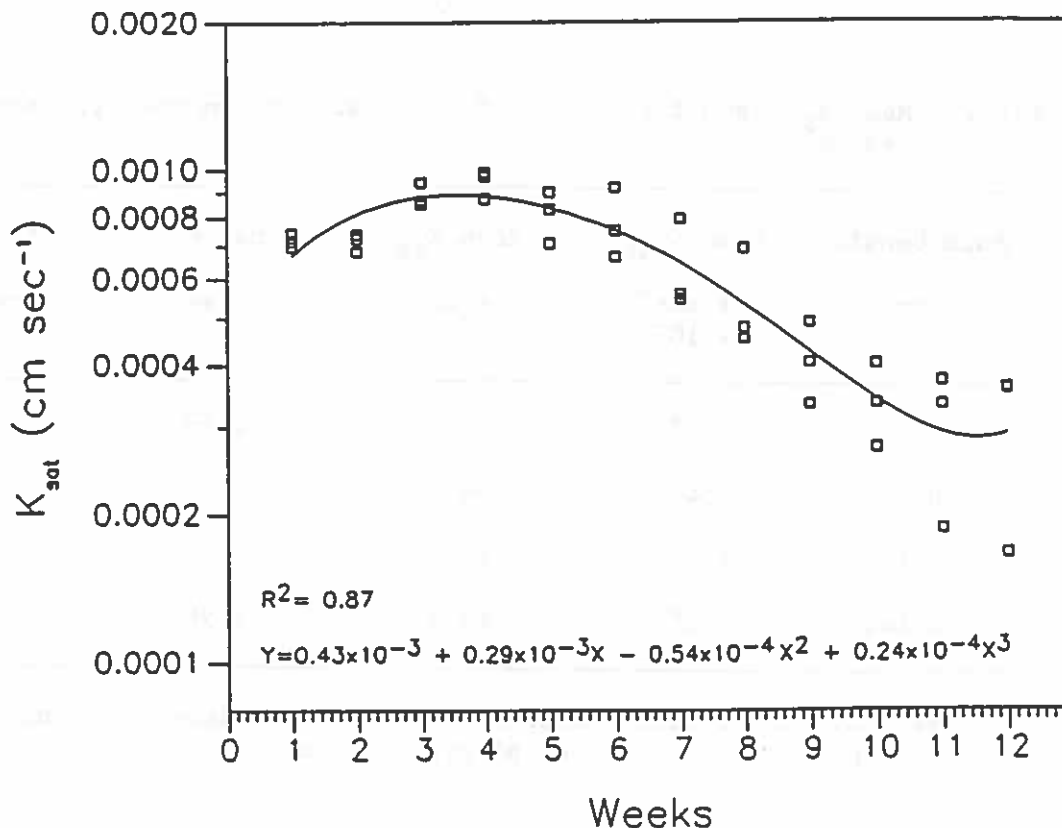


Figure 6. Change in hydraulic conductivity over time of sphagnum peat moss packed at a bulk density of 0.130 g cm^{-3} .

Residence Time study

Residence times were determined in a breakthrough study using bromide as a tracer. Breakthrough was defined as the point at which concentration of Br^- in the reactor effluent was equal to the initial Br^- content in the STE. Regression analysis was used to evaluate the breakthrough curves. In all treatments R^2 values were 97% or better. The sand treatments without a water table and with a water table showed breakthrough after 22 and 19 days respectively. Regression lines fitted to breakthrough data suggest that some flow may have occurred through macropores as evidenced by the initial steep slopes in percent breakthrough (Figures 9 and 10). However, the curves do not suggest that significant channeling or bypassing occurred. Residence times for the peat reactors without a water table and with a water table were 28 and 30 days respectively. Figures 11 and 12 show regression lines fitted to breakthrough data from the peat treatments. Visual analysis of the slopes of the curves in these plots suggests that flow was not through macropores but through the smaller micropores. Breakthrough calculations using pore volumes instead of days generate the following values for residency: 1.9, 1.7, 1.4, and 1.5 pore volumes for sand without a water table, sand with a water table, peat without a water table, and peat with a water table, respectively. These values were based on 50% porosity in the sand and 85% porosity in the peat. In sum, the results suggest that wastewater loaded to the reactors had a relatively long residency time and was distributed throughout the pores of the reactors. The peat reactors showed a greater proportion of micropores resulting in longer residency times.

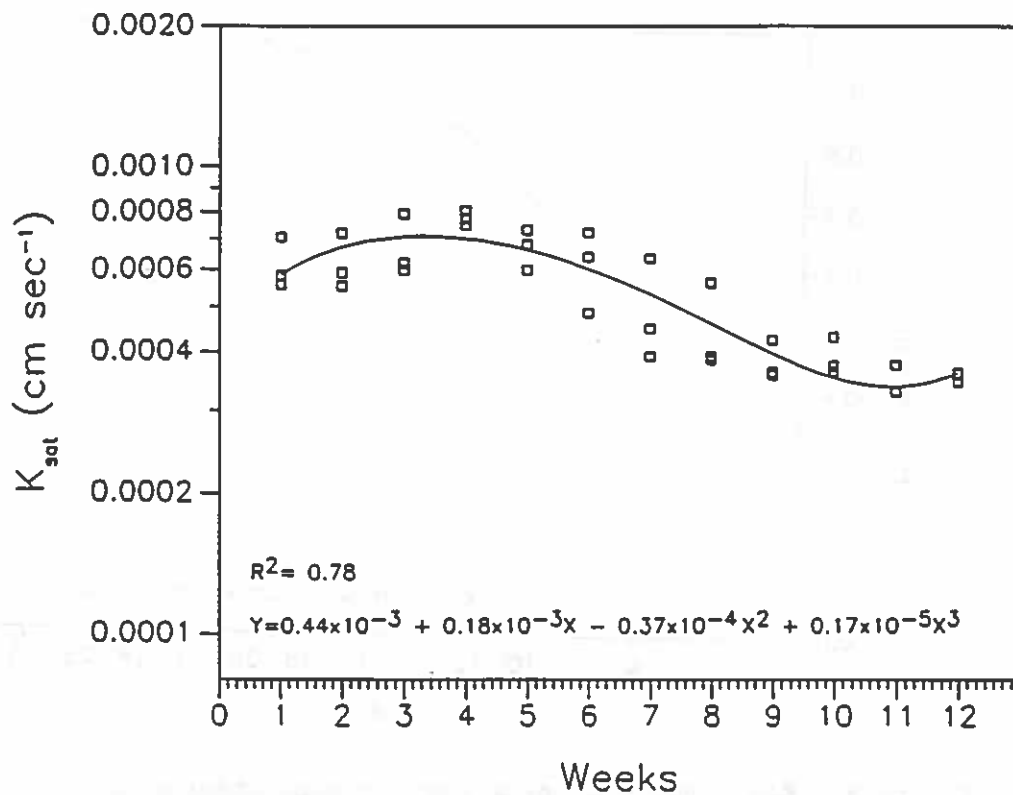


Figure 7. Change in hydraulic conductivity over time of sphagnum peat moss packed at a bulk density of 0.135 g cm^{-3} .

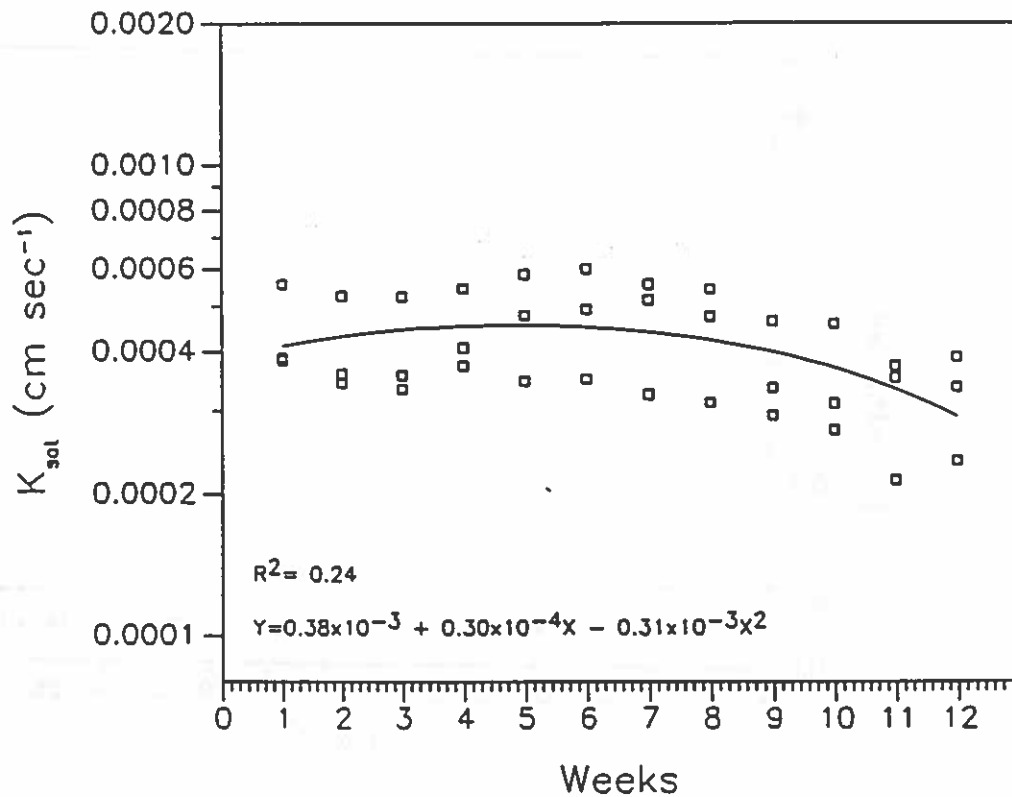


Figure 8. Change in hydraulic conductivity over time of sphagnum peat moss packed at a bulk density of 0.140 g cm^{-3} .

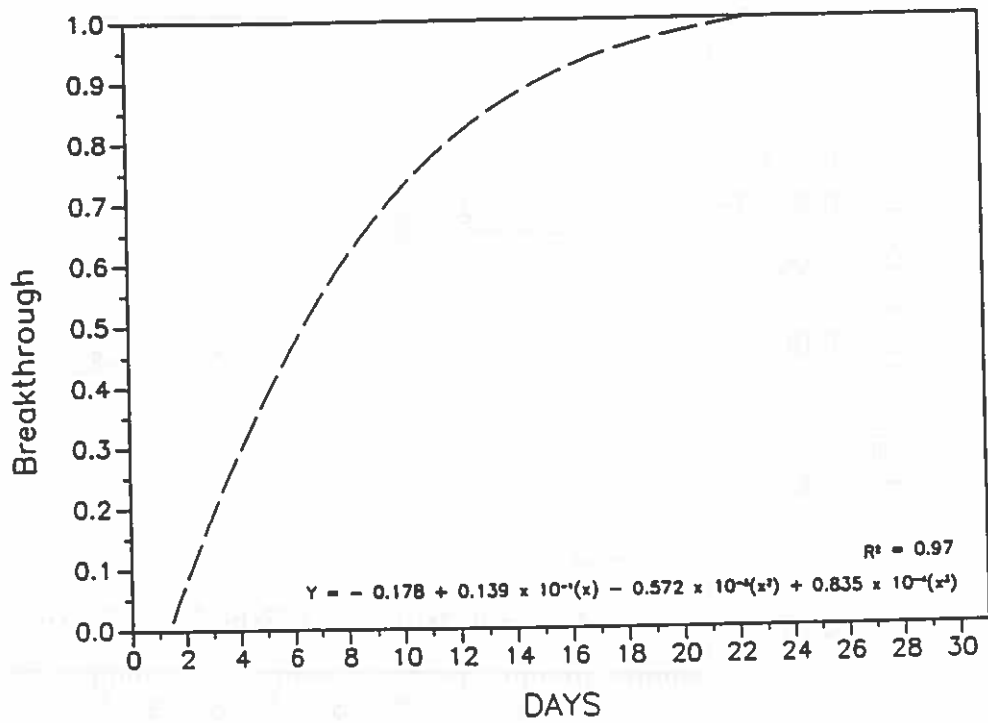


Figure 9. Residency time plot for the sand treatment without a water table.

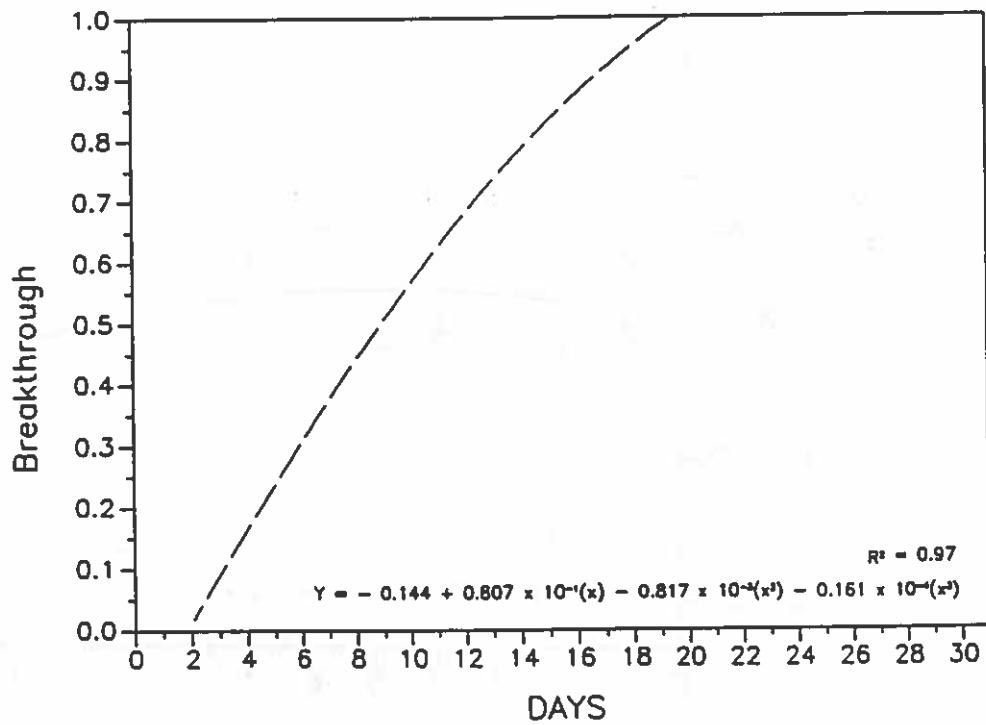


Figure 10. Residency time plot for the sand treatment with a water table.

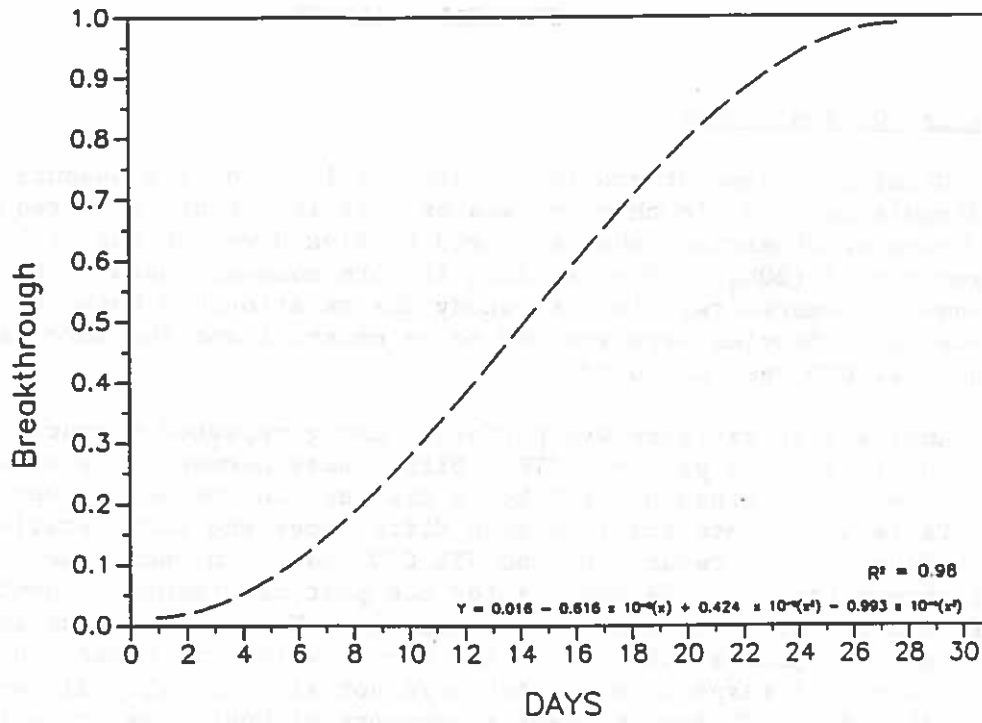


Figure 11. Residency time plot for the peat treatment without a water table.

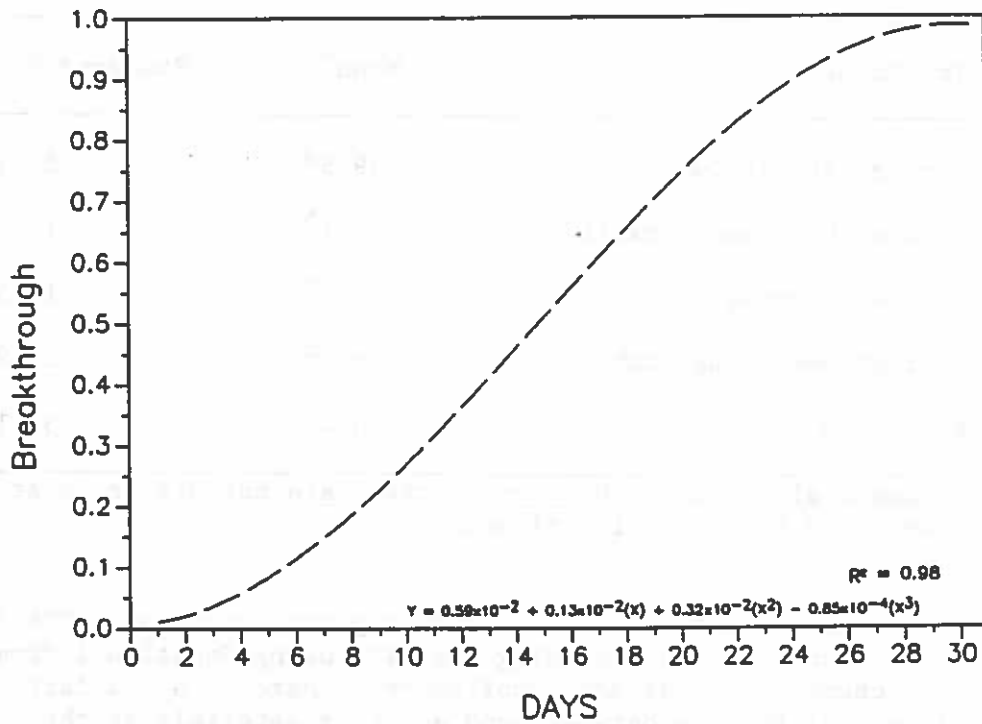


Figure 12. Residency time plot for the peat treatment with a water table.

CHEMICAL ANALYSIS

Chemical Oxygen Demand

Chemical oxygen demand (COD) commonly is used as a measure of oxidizable material in chemical wastes. It is a rapid test requiring only two hours of digestion, when compared to five days for the biological oxygen demand (BOD₅). Whereas BOD₅ is more commonly used in the analysis of domestic wastewater, in this study the relationship between COD and BOD₅ removal efficiencies were assumed to be constant and the more rapid and economical COD test was used.

Analysis of variance was performed using repeated measures design on the four treatments plus the STE. Differences between treated wastewater and STE was determined using Tukey's HSD test on the means over time (Table 3). Table 4 presents absolute mean differences and their statistical significance. Net reduction from STE COD, based on means, was 92% for the sand treatments, and 77% and 71% for the peat treatments without a water table and with a water table, respectively. Mean COD for the sand treatments without a water table and with a water table were 18.1 mg L⁻¹ and 20.4 mg L⁻¹ respectively which were not statistically different (p=0.05). Mean COD for the peat treatments without a water table and with a water table were 55.7 mg L⁻¹ and 70.3 mg L⁻¹ respectively which was not significant at the p=0.05 level. Peat treatments showed significantly (p=0.01) higher COD levels than the sand treatment.

Table 3. Mean values of COD (mg L⁻¹) for septic tank and treatment reactor effluent.

Treatment	Mean*	Standard Deviation
Septic tank effluent	239.5 ^a	56.1
Sand without water table	18.1 ^b	16.5
Sand with water table	20.4 ^b	17.5
Peat without water table	55.7 ^c	26.2
Peat with water table	70.4 ^c	38.1

* numerical values with common letters are not different at the 0.05 level of significance.

More information is revealed when repeated measures ANOVA is performed on the reactor effluent excluding the STE, using Equation 3 from the previous chapter. These data confirm that there is a statistically significant difference between sand and peat materials at the 0.01 level (Table 5). However, the difference between water status was statistically insignificant (p=0.67). In addition, a significant interaction between

Table 4. Differences in mean COD values (mg L^{-1}) for septic tank and treatment reactor effluent.

Treatment	----- Sand -----		----- Peat -----	
	without -- water table --	with -- water table --	without -- water table --	with -- water table --
Sand without water table				
Sand with water table		2.3		
Peat without water table	37.6*	35.4*		
Peat with water table	52.3*	50.0*	14.6	
STE	221.3*	219.1*	183.7*	169.1*

* indicates significant difference at 0.05 level.

material and water status was found. Within treatments, COD over time was significantly different at the 0.03 level, as well as an interaction between time, material and water status ($p=0.01$). Figure 13 indicates that the lines do not follow similar patterns. Significant interaction therefore exists based on visual plots and the statistical analysis.

Table 5. Analysis of variance for COD values related to selected treatment variables.

Source	SS	df	MS	F-ratio	p-value
<u>Between Effects</u>					
Material	25253.88	1	25253.88	68.27	<0.01
Water status	68.04	1	68.04	0.18	0.67
Material by water status	1745.52	1	1745.52	4.72	0.04
Error	7398.13	20	369.91		
<u>Within Effects</u>					
Time	7151.28	8	93.90	2.23	0.03
Time by material	4207.55	8	525.94	1.31	0.24
Time by water status	3798.31	8	474.79	1.19	0.31
Time by material and by water status	9799.35	8	1224.92	3.06	<0.01
Error	558.52	160	3.49		

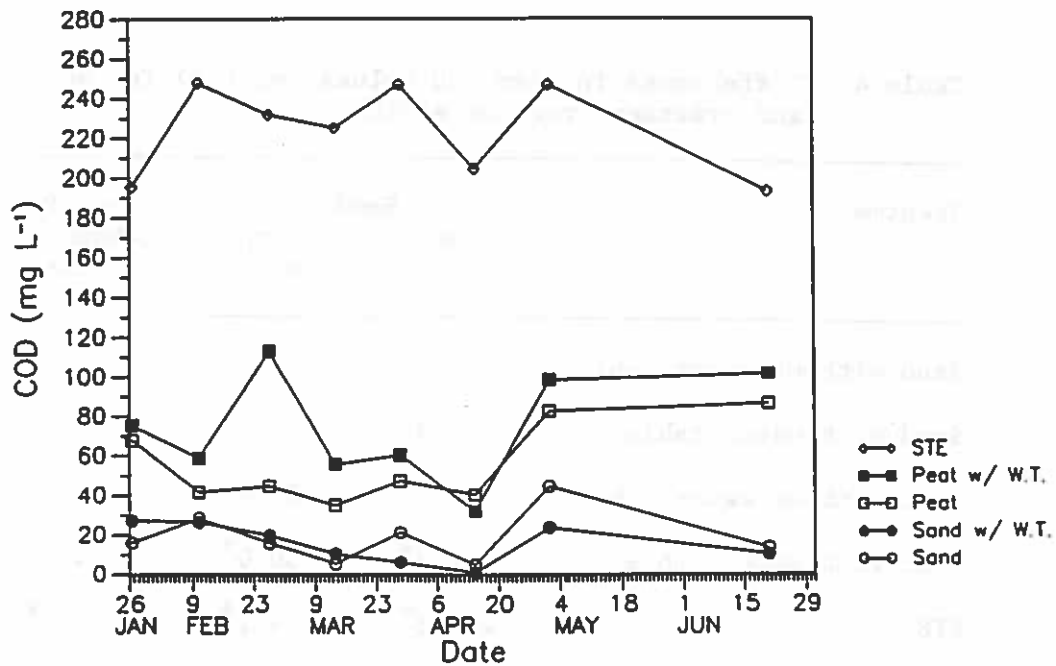


Figure 13. Mean COD values (mg L^{-1}) for septic and treatment reactor effluent over time.

Finally, null hypothesis testing that there are no significant differences between water status and material is confounded by the positive interaction between effects.

Based on means, the removal of COD by the sand system is better than that by the peat systems, and falls below the federal BOD_5 standard of 30 mg L^{-1} . Higher COD levels in the peat system may be explained by the significantly higher carbon content in the organic soils. In addition, solubilized organic compounds including aromatic molecules part of the peat structure, are not readily metabolized by organisms found in mineral soils. Lavigne (1989) reports that after 18 months of leaching sphagnum peat beds with tap water, background TOC levels still measured 50.7 mg L^{-1} . Therefore, it may be reasonable to assume that higher COD values in the peat systems, were related to the peat exudates and were not bioavailable. Hence, if BOD_5 were used as a measure of oxygen demand, the peat reactors may have shown the oxygen demand levels observed in the sand reactors.

Nitrogen

Ammonia nitrogen ($\text{NH}_3\text{-N}$) was measured in the STE and reactor effluent from February 10 through August 8, 1990. Analysis of variance on the results was performed using a repeated measures design on the four treatments plus STE. A significant difference in $\text{NH}_3\text{-N}$ was observed at the 0.0001 confidence level. Differences between treated wastewater and STE were evaluated using Tukey's HSD test on the means over time. Mean $\text{NH}_3\text{-N}$ in STE and reactor effluents during the study period are shown in Table 6. Mean $\text{NH}_3\text{-N}$ for the sand treatments without a water table and with a water table were 2.4 mg L^{-1} and 3.2 mg L^{-1} respectively, which was not statistically different at the 0.05 level. Mean $\text{NH}_3\text{-N}$ for the peat treatments without a water table and with a water table were 11.3 mg L^{-1}

Table 6. Mean $\text{NH}_3\text{-N}$ values (mg L^{-1}) for septic tank and treatment reactor effluent.

Treatment	Mean*	Standard Deviation
Septic tank effluent	23.5 ^a	5.3
Sand without water table	2.4 ^b	4.0
Sand with water table	3.2 ^b	3.7
Peat without water table	11.3 ^c	2.5
Peat with water table	10.9 ^c	4.5

* numerical values with common letters are not different at the 0.05 level of significance.

and 10.9 mg L^{-1} respectively which was not statistically significant at the 0.05 level. Mean STE $\text{NH}_3\text{-N}$ was 23.5 mg L^{-1} . Table 7 presents mean differences and their statistical significance. Net reduction from STE $\text{NH}_3\text{-N}$ based on means, was 89% and 86% for sand treatments without a water table and with a water table respectively; and 50% and 51% for the peat treatments without a water table and with a water table, respectively. Peat treatment effluent had higher $\text{NH}_3\text{-N}$ levels than sand treatment effluent, with differences significant at the <0.001 level. In addition, peat reactor effluent and sand reactor effluent were statistically different from STE at <0.001 level.

Table 8 presents results from an ANOVA performed on the reactor effluents excluding the STE, using Equation 3. These data confirm a statistically significant difference between sand and peat materials. The difference between water status within peat or sand material was not statistically significant. Within treatments, $\text{NH}_3\text{-N}$ over time was significantly different at the <0.01 level. In addition, significant interactions occurred between time and material and time, material and water status (Figure 14). An increase in $\text{NH}_3\text{-N}$ can be seen through the month of April. Peat reactor effluent had a corresponding increase delayed by two to three weeks. The sand reactor effluent was unaffected by this increase. Another increase in early August produced no immediate effect on reactor effluent.

Expected levels of $\text{NO}_3\text{-N}$ in reactor effluents were not achieved indicating insufficient nitrification in the aerated peat zone. The sand reactors reduced $\text{NH}_3\text{-N}$ levels to near acceptable limits. The most likely cause for incomplete oxidation is limited O_2 supply. It is also possible that high COD resulted in a decrease in available O_2 , thus limiting nitrification. Metcalf and Eddy (1972) suggested that nitrogenous oxygen demand (NOD) is not reduced until 6 to 10 days after BOD is exerted. Considering that the water level in the sand and peat reactors was identical, it is suggested that 40 cm of unsaturated material is sufficient to oxidize the $\text{NH}_3\text{-N}$ given a fixed retention time. Therefore, it can be

hypothesized that longer retention times may have resulted in increased nitrification. Implications of this hypothesis will be discussed later.

Table 7. Differences in mean $\text{NH}_3\text{-N}$ values (mg L^{-1}) for septic tank and treatment reactor effluent.

Treatment	----- Sand -----		----- Peat -----	
	without -- water table --	with water table --	without -- water table --	with water table --
Sand without water table				
Sand with water table		0.8		
Peat without water table	8.8*	8.1*		
Peat with water table	8.5*	7.7*	0.4	
STE	20.0*	19.2*	11.1*	11.4*

* indicates significant difference at 0.05 level.

Table 8. Analysis of variance for $\text{NH}_3\text{-N}$ values related to selected treatment variables.

Source	SS	dF	MS	F-ratio	p-value
<u>Between Effects</u>					
Material	2122.67	1	2122.67	95.47	<0.01
Water status	17.02	1	17.02	0.77	0.39
Material by water status	17.50	1	17.50	0.79	0.38
Error	442.58	20	22.13		
<u>Within Effects</u>					
Time	757.71	8	94.71	14.13	<0.01
Time by material	550.00	8	68.75	10.25	<0.01
Time by water status	38.25	8	4.78	0.71	0.68
Time by material and by water status	163.83	8	20.48	3.05	<0.01
Error	1072.67	160	6.70		

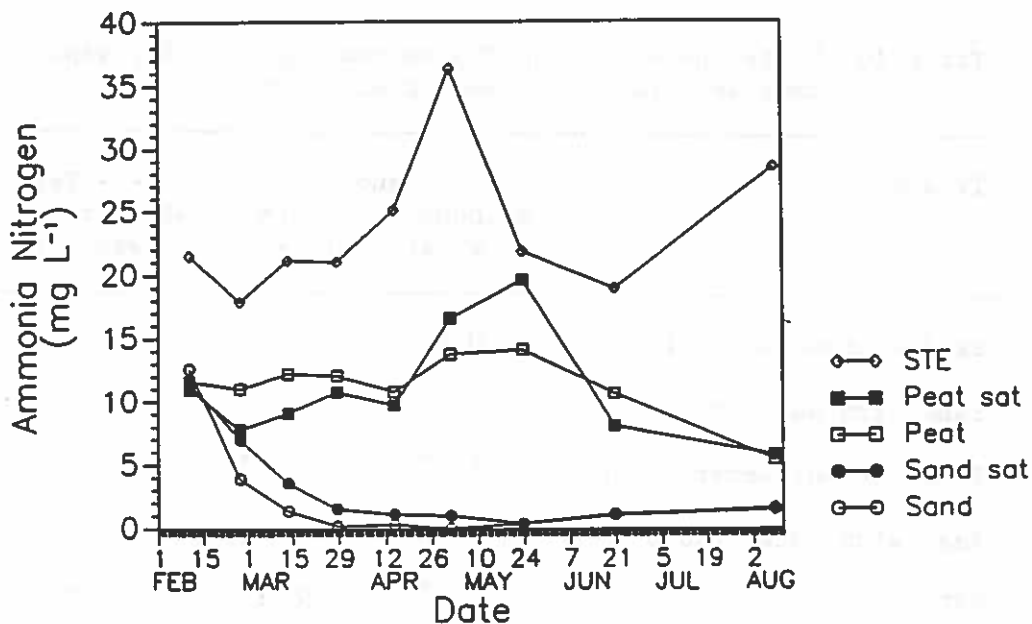


Figure 14. Mean NH₃-N values (mg L⁻¹) for septic tank and treatment reactor effluent over time.

TKN was measured in the STE and reactor effluent from February 10 through August 8. Analysis of variance using repeated measures design on the four treatments plus STE indicated a significant difference in TKN ($p=0.0001$). Differences between mean values over time of the treated wastewater and STE were determined using Tukey's HSD (Table 9). Mean STE TKN was 23.5 mg L⁻¹. Mean TKN for the sand treatments without a water table and with a water table were 2.2 mg L⁻¹ and 3.2 mg L⁻¹ respectively, which was not statistically different at the 0.05 level. Mean TKN for the peat treatments without a water table and with a water table were 10.2 mg L⁻¹ and 9.8 mg L⁻¹ respectively, which was not statistically significant at the 0.05 level. Table 10 presents absolute mean differences and their statistical significance. Net reduction from STE TKN based on means, was

Table 9. Mean TKN values (mg L⁻¹) for septic tank and treatment reactor effluent.

Treatment	Mean*	Standard Deviation
Septic tank effluent	23.5 ^a	3.2
Sand without water table	2.2 ^b	2.9
Sand with water table	3.2 ^b	3.7
Peat without water table	10.2 ^c	2.4
Peat with water table	9.8 ^c	2.5

* numerical values with common letters are not different at the 0.05 level of significance.

Table 10. Differences in mean TKN values (mg L^{-1}) for septic tank and treatment reactor effluent.

Treatment	----- Sand -----		----- Peat -----	
	without -- water table --	with water table --	without -- water table --	with water table --
Sand without water table	0.0			
Sand with water table	1.0			
Peat without water table	8.0*	7.0*		
Peat with water table	7.7*	6.7*	0.4	
STE	21.3*	20.3*	13.3*	13.6*

* indicates significant difference at 0.05 level.

91% and 86% for sand treatments without a water table and with a water table respectively, and 57% and 58% for the peat treatments without a water table and with a water table respectively. Peat treatment effluent had higher TKN levels than sand treatment effluent, with differences significant at the <0.001 level. In addition, peat reactor effluent was statistically different from STE at the <0.001 level.

Table 11 presents results from an ANOVA performed on the reactor effluents excluding the STE, using equation 3 from the previous chapter. These data confirm a statistically significant difference between sand and peat materials. However, no difference between water status within peat or sand material was statistically significant. Within treatments, TKN over time was significantly different at the $p < 0.01$ level. Significant interactions occurred between time and material and time, material and water status (Figure 15). Similar trends between TKN and $\text{NH}_3\text{-N}$ are evident from Figures 14 and 15. Repeated measures ANOVA was performed on means of $\text{NH}_3\text{-N}$ and TKN for all reactor effluents and STE were not significantly different at the 0.05 level. This suggests that the fraction of N in the organic form is not detectable with the analytical procedures used in this study and that factors affecting the change in $\text{NH}_3\text{-N}$ also correspond to changes in TKN. Another explanation for not finding any appreciable organic N may be due to treatment of the primary wastewater prior to reaching the septic tank. Aeration and subsequent oxidation of organic matter are likely to have occurred and reduced organic N to $\text{NH}_3\text{-N}$.

Nitrate and nitrite-N ($\text{NO}_x\text{-N}$) was measured in the STE and reactor effluent from February 10 through August 8. Analysis of variance using repeated measures design on the four treatments plus STE indicated a significant difference in $\text{NO}_x\text{-N}$ ($p=0.0001$). Differences between treated wastewater and STE were determined using Tukey's HSD test on the means over time. Mean $\text{NO}_x\text{-N}$ in STE and reactor effluents during the study period are shown in Table 13. Mean $\text{NO}_x\text{-N}$ for the sand treatments without a water table and with a water table were 26.6 mg L^{-1} and 24.4 mg L^{-1} respectively, and were not statistically different at the 0.05 level. Mean

Table 11. Analysis of variance for TKN values related to selected treatment variables.

Source	SS	dF	MS	F-ratio	p-value
<u>Between Effects</u>					
Material	1737.13	1	1737.13	122.17	<0.01
Water status	27.83	1	27.82	1.96	0.18
Material by water status	25.62	1	25.62	1.80	0.19
Error	284.38	20	14.22		
<u>Within Effects</u>					
Time	417.49	8	52.19	14.95	<0.01
Time by material	487.55	8	60.94	17.46	<0.01
Time by water status	41.00	8	5.12	1.47	0.17
Time by material and by water status	70.51	8	8.81	2.52	0.01
Error	558.52	160	3.49		

NO_x-N for the peat treatments without a water table and with a water table were 19.1 mg L⁻¹ and 9.3 mg L⁻¹. Net increase in reactor effluent NO_x-N as compared to STE NO_x-N, was 25.8 and 23.6 mg L⁻¹ for sand treatments without a water table and with a water table respectively, and 18.3 and 8.6 mg L⁻¹ for the peat treatments without a water table and with a water table

Table 12. Differences in mean NH₃-N values and TKN values for septic tank and treatment reactor effluent.

Treatment	TKN				STE
	Sand		Peat		
NH ₃ -N	without water table	with water table	without water table	with water table	
Sand without water table	0.0				
Sand with water table	1.0	0.0			
Peat without water table	9.1*	8.7*	1.1		
Peat with water table	8.7*	7.7*	0.4	1.1	
STE	20.2*	19.2*	12.2*	12.5*	1.1

* indicates significant difference at 0.05 level.

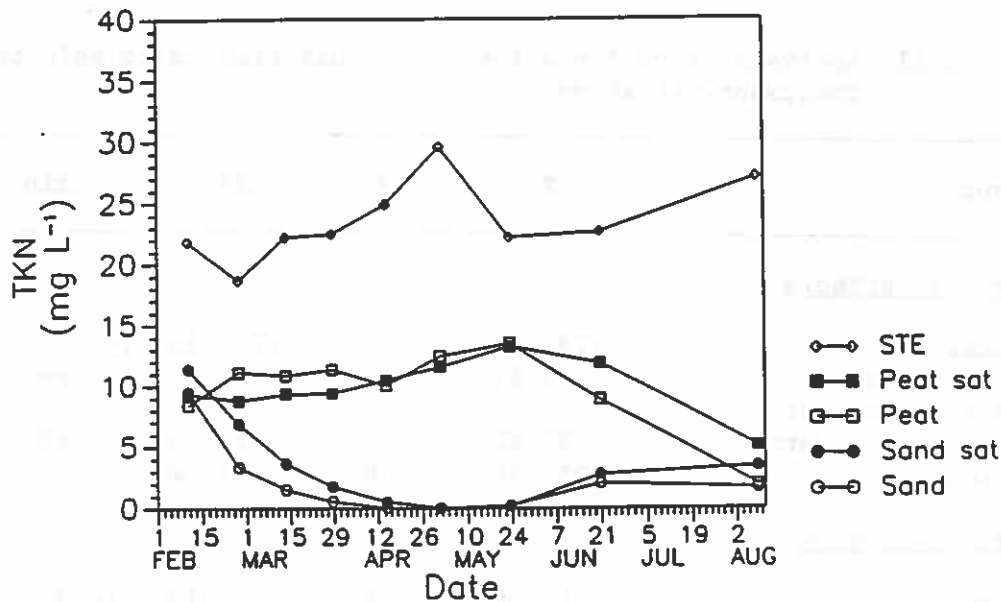


Figure 15. Mean TKN values (mg L^{-1}) for septic tank and treatment reactor effluent over time.

respectively (Table 14). Peat reactor effluents had lower $\text{NO}_x\text{-N}$ levels than sand treatment effluent, with differences significant at the $p < 0.001$ level. Peat reactor effluent and sand reactor effluent were statistically different from STE ($p < 0.001$).

Table 15 presents results from an ANOVA performed on the reactor effluents excluding the STE, using Equation 3 from the previous chapter. These data confirm a statistically significant difference between sand and peat materials. An interaction between material and water is evident. Within treatments, $\text{NO}_x\text{-N}$ over time was significantly different at the $p < 0.01$ level. In addition, significant interactions occurred between time

Table 13. Mean $\text{NO}_x\text{-N}$ values (mg L^{-1}) for septic tank and treatment reactor effluent.

Treatment	Mean*	Standard Deviation
Septic tank effluent	0.8 ^a	0.9
Sand without water table	26.6 ^b	6.3
Sand with water table	24.4 ^b	8.7
Peat without water table	19.1 ^c	8.4
Peat with water table	9.3 ^d	6.0

* numerical values with common letters are not different at the 0.05 level of significance.

Table 14. Differences in mean NO_x-N values (mg L⁻¹) for septic tank and treatment reactor effluent.

Treatment	----- Sand -----		----- Peat -----	
	without -- water table --	with water table --	without -- water table --	with water table --
Sand without water table				
Sand with water table		2.2		
Peat without water table	7.5*	5.3*		
Peat with water table	17.3*	15.1*	9.7	
STE	25.8*	23.6*	18.3*	8.6*

* indicates significant difference at 0.05 level.

and material; time and water status; and time, material and water status (Figure 16). An increase in NO_x-N can be seen until the end of May after which levels decreased. Increased NO_x-N in the effluent correspond chronologically with elevated NH₃-N in the STE, while NO_x-N in the STE remained low throughout the study period.

Table 15. Analysis of variance for NO_x-N values related to selected treatment variables.

Source	SS	dF	MS	F-ratio	p-value
<u>Between Effects</u>					
Material	1527.91	1	1527.91	19.92	<0.01
Water status	132.34	1	132.34	1.73	0.20
Material by water status	765.20	1	765.20	9.97	<0.01
Error	1531.31	20	76.72		
<u>Within Effects</u>					
Time	1872.53	8	234.07	9.62	<0.01
Time by material	923.42	8	115.43	4.75	<0.01
Time by water status	555.77	8	69.47	2.86	0.01
Time by material and by water status	523.14	8	65.39	2.69	0.01
Error	3891.86	160	24.32		

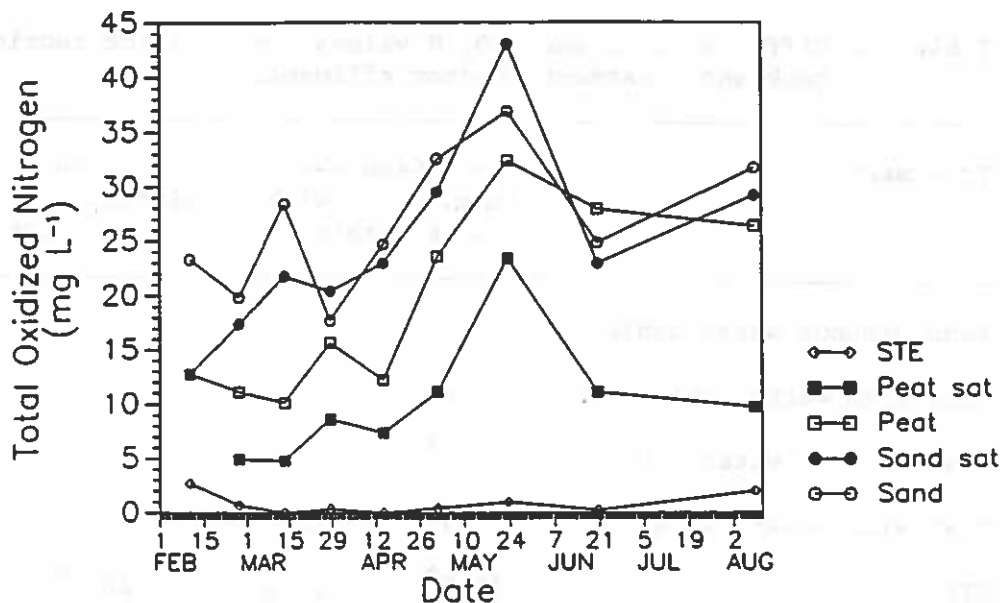


Figure 16. Mean NO_x-N values (mg L⁻¹) for septic tank and treatment reactor effluent over time.

Nitrogen Loss

Changes in the various forms of nitrogen in reactor effluents are best described by relating net reductions in total N. Net loss was calculated by the ratio of the difference between influent and effluent divided by influent. Influent and effluent N was derived as the sum of NO_x-N and TKN. NH₃-N was not considered since it was shown previously to be not significantly different from TKN. Average percent net loss of N was -19%, -15%, -22%, and 20% for the sand without a water table, sand with a water table, peat without a water table and peat with a water table respectively (Table 16). Negative loss values in both sand treatments and the peat treatment indicate relative gains in N (Figure 17). Net loss of N was observed in the peat treatments with a water table. Gains in N may be attributable to variation in STE not detected in the sampling periods. Mean nitrogen STE was 23.2 mg L⁻¹ with 12.9 mg L⁻¹ variance from 9 sampling periods. In addition, elevated levels of NO_x-N, reported on May 23, were on average 1.9 times of the mean NO_x-N over time for each reactor. On this date, NO_x-N from selected reactors differed up to 4.2 times their average NO_x-N over time. When the net N loss was calculated without the May 23 sampling date, average net N losses were -14%, -6%, -12% and 30% for sand without a water table, sand with a water table, peat without a water table and peat with a water table respectively. Whereas there are still net gains in N in both sand treatments and the peat treatment without a water table, their values are less negative and may be attributed to variation in STE N. Mineralization of retained N from prior application is another possible explanation. Significantly more N was released during the early spring. It is possible that changes in reactor temperature resulted in increased biological activity. In addition, COD values dropped in late April, possibly indicating increased net N mineralization. Another explanation is possible N₂ fixation by phototropic organisms.

Table 16. Mean reduction in nitrogen values for treatment reactor effluent

Treatment	Mean*	Standard Deviation
Sand without water table	-19% ^a	32%
Sand with water table	-15% ^a	33%
Peat without water table	-22% ^a	49%
Peat with water table	20% ^b	35%

* numerical values with common letters are not different at the 0.05 level of significance.

Acidity

Acidity was measured as pH in the reactor effluents and STE from October 4, 1989 through August 8, 1990. Mean pH values (Table 17) for the sand treatments without a water table and with a water table were identical at 6.8. Mean pH for the peat treatments without a water table and with a water table were 5.0 and 5.2 respectively. Differences in pH values between the peat treatments were not statistically significant at the 0.05

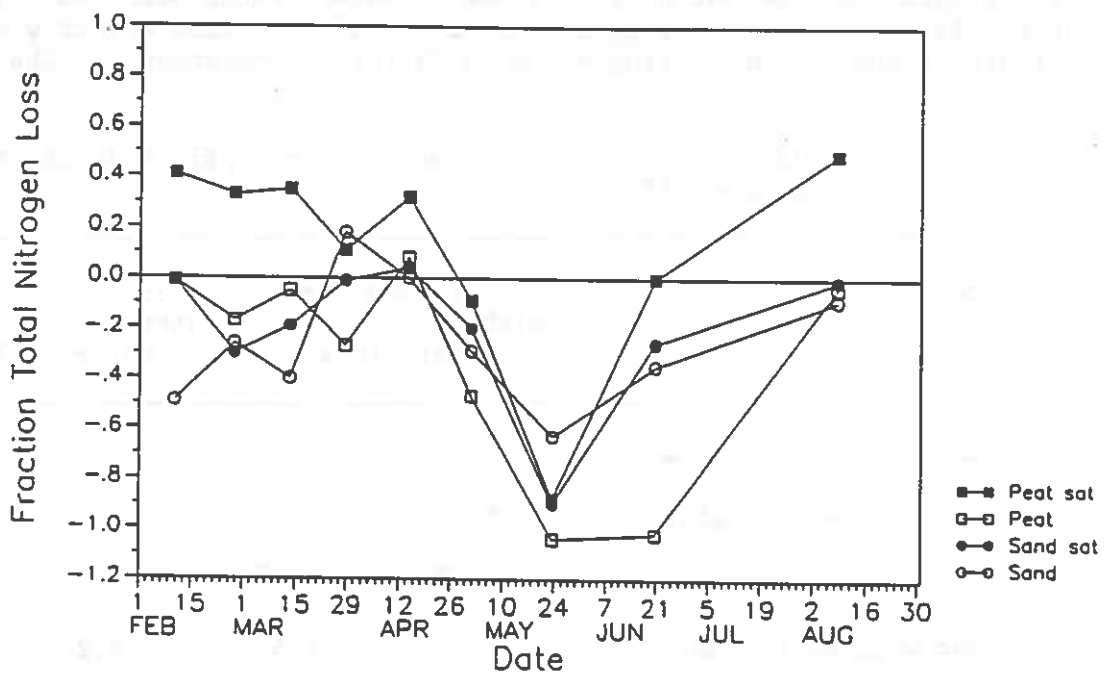


Figure 17. Net nitrogen loss in the various treatment reactors over time.

Table 17. Mean pH values for septic tank and treatment reactor effluent.

Treatment	Mean*	Standard Deviation
Septic tank effluent	6.8 ^a	0.2
Sand without water table	6.8 ^a	0.3
Sand with water table	6.8 ^a	0.2
Peat without water table	5.0 ^b	0.2
Peat with water table	5.2 ^b	0.3

* numerical values with common letters are not different at the 0.05 level of significance.

level. Mean STE pH was 6.8. Table 18 presents absolute mean differences and their statistical significance. The pH remained unchanged from the STE pH after passing through both sand treatments. However, STE pH was lowered after passing through the peat treatments. The difference in pH of both level peat treatment effluents were statistically significant at the 0.05 level.

Repeated measures ANOVA on the treatment reactor effluents, alone, using Equation 3 from the previous chapter were performed (Table 19). These data confirm that no effect due to water status occurred. This analysis does, however, suggest that a significant time effect was present. Figure 18 shows a decreasing trend in pH for all treatments. The observed

Table 18. Differences in mean pH values for septic tank and treatment reactor effluent.

Treatment	----- Sand -----		----- Peat -----	
	without -- water table --	with water table --	without -- water table --	with water table --
Sand without water table				
Sand with water table	0.0			
Peat without water table	1.7*	1.7*		
Peat with water table	1.5*	1.5*	0.2	
STE	0.0	0.0	1.8*	1.6*

* indicates significant difference at 0.05 level.

Table 19. Analysis of variance for pH values related to selected treatment variables.

Source	SS	dF	MS	F-ratio	p-value
Between Effects					
Material	76.56	1	76.65	127.69	<0.01
Water status	0.00	1	0.00	0.00	0.96
Material by water status	0.47	1	0.47	0.78	0.40
Error	4.80	8	0.60		
Within Effects					
Time	3.57	16	0.22	6.00	<0.01
Time by material	1.20	16	0.08	2.02	0.02
Time by water status	0.46	16	0.03	0.77	0.72
Time by material and by water status	0.71	16	0.04	1.19	0.28
Error	4.76	128	0.04		

decrease in effluent pH after passing through the peat reactors is consistent with the exchange properties of the peat material. The high extractable acidity in the peat is most likely the source of additional acidity found in the effluent from the peat reactors.

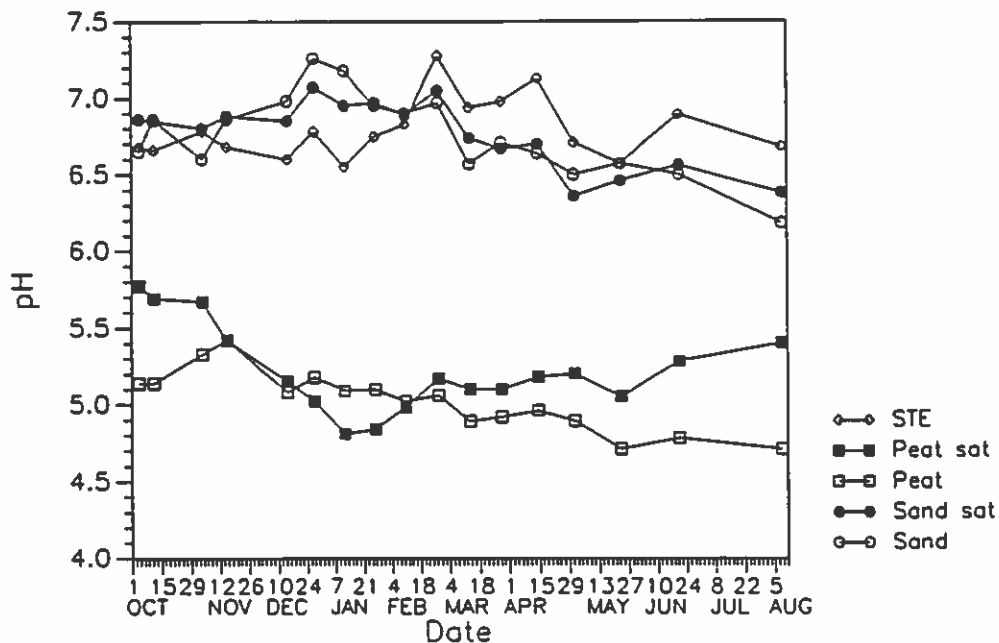


Figure 18. Mean pH values for septic tank and treatment reactor effluent over time.

Acetylene Blocking

The acetylene blocking technique was used on peat samples taken from two peat reactors with a water table. Assessment of denitrification activity was based on evolution of N_2O gas from the peat samples. The ratio of the highest N_2O-N concentration after 12 hours of incubation to the N_2O-N concentration at 6 hours was used to evaluate N_2O-N generation in the samples. Net increase in N_2O-N concentration ranged from 2 to 70 times, strongly suggesting that denitrification occurs in the reactors. The procedure used does not provide a quantitative assessment of the denitrification rate. Experiments using small columns perhaps provide a better means to evaluate denitrification quantitatively. Denitrification, however, appears to significantly reduce nitrogen output from the partially saturated peat reactors.

The research results indicate that denitrification occurs in the partially saturated peat reactors. This process appears somewhat limited by incomplete nitrification in the aerobic upper part of the reactor. Lower application rates or larger column length may provide enhanced conversion into nitrate and improved denitrification. Carbon does not appear to be limiting in the systems evaluated. When design changes are made to provide improved nitrification, the enhanced oxidation may lead to lower COD and BOD_5 values. If this proves to be the case, the experimental design may have to be altered through a by-pass of the oxidation section of the reactors allowing additions of raw sewage prior to the denitrification reactor.

CHAPTER IV

SUMMARY AND CONCLUSIONS

Septic tank systems are the preferred treatment method where central sewerage is unavailable. Over 30% of the households in the United States, 27% in Massachusetts, rely on such systems for the removal of organic carbon, organic N, $\text{NH}_3\text{-N}$ and pathogens. The soil below the leaching field in conventional septic systems provides for proper aeration needed to oxidize the pollutants resulting in their removal, transformation or die-off. However, conventional systems produce $\text{NO}_x\text{-N}$ in quantities exceeding federal drinking water standards (10 mg L^{-1}), because they lack the conditions required for the reduction of $\text{NO}_x\text{-N}$. This may result in pollution to ground- and surface waters particularly in areas with high septic system densities. For this reason it is important to develop and study low-cost systems which have both oxidizing and reducing components as part of the same system.

In this study, effluent from pilot-scale reactors filled with peat or sand, and either partly saturated or entirely nonsaturated was analyzed after application of septic tank effluent. Successful operation over a twelve-month period has proven that these pilot-scale wastewater treatment reactors show potential to provide adequate treatment of domestic wastewater.

The sand filters showed excellent oxidizing potential. A 92% reduction in COD, 86-89% reduction of $\text{NH}_3\text{-N}$ and 100% removal of organic-N was observed in these reactors. The sand filters produced more total-N than was put into the filters. Output of $\text{NO}_x\text{-N}$ from the sand filters and the unsaturated peat reactors exceeded the total input of N over the course of the study. In addition, both peat reactors exhibited fairly high effluent COD values. Most notable was the reduction in total N observed in the partly saturated peat reactors. Oxidized-N in the effluent from these reactors was 9.3 mg L^{-1} . While there was still $\text{NH}_3\text{-N}$ in the effluent, indicating incomplete nitrification in the aerated part of the reactor, a net reduction of up to 40% in oxidized-N was measured.

Whereas factors such as ionic sorption, and biological and chemical immobilization may account for some of the nitrogen reduction, denitrification was measured in samples of peat taken from several partly saturated peat reactors by evolution of N_2O after incubation in an acetylene environment. Denitrification is favored by anaerobic conditions, and the presence of both $\text{NO}_x\text{-N}$ and organic C. The relatively low N removal rates may result from incomplete nitrification prior to denitrification, limited bioavailability of carbon, short retention times, or excessive N loading.

The basic design of the treatment reactors provided a suitable structure for the placement of peat and sand and the collection of reactor effluent. Two reactors in the peat treatment with a water table accidentally became completely saturated with water, resulting in changes to the design bulk density for the upper layer. Otherwise, treatment

reactors performed flawlessly with respect to hydraulic loading.

No organic mat was observed at the stone sand interface in the sand filters after one year of operation. No organic mat formed in the peat reactors after one year. Peat reactors packed at a density of 0.10 to 0.12 g cm^{-3} can accept STE at a rate of 3 cm day^{-1} (0.74 $\text{gal ft}^{-2} \text{day}^{-1}$) for at least one year without hydraulic failure. Sand filters packed at 1.4 g cm^{-3} will accept STE at a rate of 3 cm day^{-1} (0.74 $\text{gal ft}^{-2} \text{day}^{-1}$) for more than one year without hydraulic failure.

Saturated hydraulic conductivity in the peat was measured over a period of 12 weeks. Mean K_{sat} was 9.2, 6.4, 5.5, and 4.2 $\times 10^{-4} \text{ cm sec}^{-1}$ for columns packed at 0.125, 0.130, 0.135, and 0.140 g cm^{-3} respectively. No significant differences in final K_{sat} were observed.

Net reduction from STE COD was 92% for the sand treatments, and 77% and 71% for the peat treatments without a water table and with a water table, respectively. No effect due to water status with respect to COD was statistically significant at the 0.05 level. Both peat treatments produced effluent COD in excess of 30 mg L^{-1} , which when compared to BOD_5 , may be in excess of federal drinking water standards.

Net reduction from STE $\text{NH}_3\text{-N}$ was 89% and 86% for the sand treatments without a water table and with a water table respectively; and 50% and 51% for the peat treatments without a water table and with a water table, respectively. No effect due to water status with respect to $\text{NH}_3\text{-N}$ was statistically significant at the 0.05 level. Incomplete nitrification may have resulted from limited O_2 in the reactors or excessive loading rates. Throughout the sampling period, $\text{NH}_3\text{-N}$ was detected in the effluents of all treatment.

Net reduction from STE TKN was 91% and 86% for the sand treatments without a water table and with a water table respectively; and 57% and 58% for the peat treatments without a water table and with a water table, respectively. No effect due to water status with respect to TKN was statistically significant at the 0.05 level. TKN used as a measure of organic nitrogen was not statistically different from $\text{NH}_3\text{-N}$ content. Low organic nitrogen in the STE may have resulted from pretreatment at the municipal wastewater facility prior to entering the pilot sewage treatment plant.

Net increase in $\text{NO}_x\text{-N}$ in reactor effluents as compared to STE $\text{NO}_x\text{-N}$ was 25.8 and 23.6 mg L^{-1} for sand treatments without a water table and with a water table respectively; and 18.3 and 8.6 mg L^{-1} for the peat treatments without a water table and with a water table respectively. No effect due to water status with respect to $\text{NO}_x\text{-N}$ was statistically significant at the 0.05 level. All treatments, except the peat with a water table, exceeded federal drinking water standards of 10 mg L^{-1} .

Net gain in N was 19%, 14%, and 21% for the sand treatments without a water table, with a water table, and the peat treatment without a water table, respectively. Net increases in N from these treatments may have resulted from mineralization of N retained from early applications. Net loss of N was 21% in the peat treatment with a water table. This loss in N may be due to denitrification of $\text{NO}_x\text{-N}$.

Future studies should quantify the chemical, physical and biological processes responsible for nitrogen reduction, and additional experiments should be conducted to quantify denitrification rates. The following modifications to the peat filtration reactors may be appropriate:

- * increase the $\text{NO}_x\text{-N}$ fraction in the peat reactor influent, possibly by loading STE to the sand filters prior to application to the peat filters,
- * increase depth of the saturated zone in the peat filters to assure anoxic conditions,
- * adjust loading rates to reduce total-N input if the design C/N ratio is not high enough, and
- * consider adding an organic carbon source such as raw STE or methanol if insufficient carbon is available internally.

CHAPTER V

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