

1991

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A SEPTIC TANK EFFLUENT TREATMENT SYSTEM
FOR ENHANCED NITRATE REMOVAL

A Thesis Presented

by

ERIC S. WINKLER

Submitted to the Graduate School of the
University of Massachusetts in partial fulfillment
of the requirements for the degree of

MASTER OF SCIENCE

February 1991

Department of Plant and Soil Sciences

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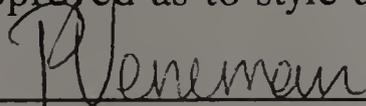
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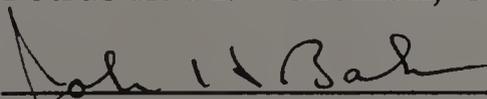
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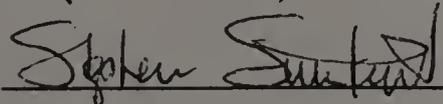
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DEDICATION

To my loving and patient Bonni.

ACKNOWLEDGEMENTS

Over two and one half years there have been many people who have lent a strong hand and bending ear that made this work possible. To those people I owe my thanks. I am fortunate to have worked under the guidance of Dr. Peter Veneman. His confidence in my abilities enabled me to strengthen my weaknesses and build on my strengths. This manuscript was edited by him and for this, I am grateful. I also owe special thanks to my committee members: Dr. Stephen Simkins, for the many hours of computer hacking and data analysis; and Dr. John Baker for his teaching and guidance in the chemistry used in this work. My appreciation to Dr. Ralph Baker and Dr. Ronald Lavigne for sharing their expertise of engineered, peat wastewater systems. I am indebted to Bill Donnelly, who for 12 months labored to build the facility. Thanks to Dr. Peter Groffman and Gay Hanson for use of their G.C.. My special appreciation to my friend Lloyd Grunvald of Preci Manufacturing, who provided equipment and frequent advice. Thanks to my parents Eileen and Martin for interest and support that went well beyond that which was asked, and to my daughter Raina who brought true joy into my life during the summer of 1990. Finally, I owe my deepest appreciation to my loving wife Bonni Alpert, who understood my passion for work. She has always been a kind and supportive friend to me.

This research was funded in part by the University of Massachusetts Water Resources Research Center and by a grant-in-aid of research from Sigma Xi, the Scientific Research Society.

ABSTRACT

A SEPTIC TANK EFFLUENT TREATMENT SYSTEM FOR ENHANCED NITRATE REMOVAL

FEBRUARY 1991

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Septic systems release significant amounts of NO_3^- -N to the ground water often in concentrations exceeding 10 mg L^{-1} . In Maine, several full scale systems have shown excellent treatment of BOD, suspended solids and pathogens. This study evaluated the performance of a peat bed system to remove nitrogen from septic tank effluent. Intermittent flow reactors were packed with sphagnum peat and their performance compared with uniformly packed medium washed sand reactors. In half the reactors a water table was maintained in the lower portion of the reactor. Primary municipal wastewater stored in a 1000 gal septic tank was dosed to the reactors at 3 cm day^{-1} . Inorganic NH_4^+ , NO_3^- , TKN and COD were measured for 7 months. No net removal of N was observed in the sand reactors with and without a water table and peat reactors without a water table, where average output of NO_3^- -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 9 mg L^{-1} NO_3^- -N in the effluent. 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 appears to be elevated in peat systems, probably due to organic exudates from the peat material.

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CHAPTER I

INTRODUCTION

Overview

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

There is increasing pressure to build homes on land less suitable for on-site wastewater disposal. Additionally, lot sizes in some of these areas are quite small. With increased density of sewage disposal systems, even where soils are suitable, there are increased reports of contamination to ground water (Chen, 1988; Prins and Lustig, 1988). Also, areas using on-site sewage disposal systems commonly use ground water for drinking water. Historically, planning officials have attempted to abate drinking water contamination through zoning density.

However, this cannot 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 little or no ability to remove nitrate from the percolating effluent. Subsequently, there is an increasing demand for alternative disposal and treatment systems where conventional effluent disposal fields do not provide adequate treatment. These innovative methods oftentimes are complicated to operate and very expensive to install.

Some experimental field systems and laboratory experiments 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, particularly to enhance denitrification, is attractive because its much lower costs and potential beneficial treatment potential. Peat filtration systems may be an alternative for treatment of septic tank effluent (STE) under certain situations (Viraraghavan, 1986).

Many studies have been conducted on the removal efficiencies of on-site sewage disposal systems. But, only recently, workers have recognized the limitations of current technology for nitrate removal. Questions are being asked by scientists and public officials. What are the conditions promoting conversion of nitrate to elemental nitrogen? Is it possible to design an on-site sewage disposal system that maintains these conditions for long term use?

Justification

There are two important reasons for research on septic systems. The first is that increased development on marginal lands and in sensitive ecosystems, results in higher concentrations of pollutants introduced into the ground water. Secondly, better monitoring of septic system soil absorption fields has shown that virtually no nitrate nitrogen removal occurs in the unsaturated zone below the clogging layer. The reason for this is simple, 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 not anaerobic and may not have sufficient carbon to provide energy for the denitrifying organisms. Nitrate entering ground water will not undergo further changes and may enter ground water or reemerge in surface water. Nitrate levels exceeding federal drinking water standards (10 mg L^{-1}) can cause methemoglobinemia, known as "Blue Baby Syndrome" (EPA, 1975; Menzer and Nelson, 1986). Menzer and Nelson (1986) report the production of nitrosamines from the reaction of nitrites, an intermediate in the conversion of ammonia-N to nitrate-N, with amides, ureas, carbamates and other nitrogenous compounds. They suggest that nitrosamines represent a significant class of carcinogenic and teratogenic compounds.

For these reasons a concerted effort must be made to develop innovative technologies that will enhance nitrate removal from septic tank effluent. The systems must be cost effective, simple in construction and operation, and provide

superior treatment to conventional systems, especially in regard to N-removal.

This study aims to develop on a pilot scale, an innovative system providing these beneficial attributes.

Objectives

The primary objective of this study is to evaluate the potential of peat for on-site treatment and disposal of domestic wastewater, with the specific purpose to reduce the amount of nitrate-nitrogen introduced into the ground water.

Specific objectives are:

1. To develop design specifications that provide enhanced denitrification in an innovative peat septic system,
2. To evaluate the suitability of horticultural peat as a wastewater treatment medium under pilot scale conditions, and
3. To model the efficiency of nitrogen removal from septic tank effluent in an innovative peat septic system.

CHAPTER II

REVIEW OF LITERATURE

Septic Systems

The design of septic systems has changed significantly during the last 30 years. The oldest systems existing today are single pit cesspools usually filled with stone, that receive, treat and dispose of wastewater. Modern septic systems have at least two components; a septic tank and an effluent subsurface disposal field or soil absorption field (SAF).

Role of the Septic Tank

The septic tank serves as a primary settling chamber. Raw wastewater flows through the septic tank, while solid matter settles out and anaerobic microbial treatment processes occur. A two-day residence time in the septic tank usually results in the removal of 65% of total suspended solids (TSS), 55% of 5-day biochemical oxygen demand (BOD₅) and 10% of organic N (Bouma, 1972). Some reduction of inorganic N may result from N complexed with solids, and anaerobic microbial assimilation. Virtually no phosphorus or pathogens are

removed in the primary treatment process. Most of the N in the STE is in the NH_3 form (Brandes, 1978).

Soil Absorption Field

The major purpose of this review is to address the removal mechanisms of nitrogen in peat filter disposal fields. Processes that affect other STE parameters will only be mentioned briefly. Further elaboration on nitrogen removal mechanisms in SAF's will be covered in the fourth section of this chapter.

Characteristics of STE have been discussed in depth (Baker, 1986; Brandes, 1978; Painter and Viney, 1959). After leaving the septic tank the STE is distributed into the SAF where additional treatment and final disposal is accomplished. The SAF serves a dual purpose. First, it provides a medium where physical, chemical and biological process can reduce the efficacy of STE. Second, it discharges the treated STE into the ground, where dilution may lessen pollution to groundwater and exposure to humans or animals is prevented.

Removal Mechanisms

Physical Processes

In most conventional septic systems within 2-3 months, a clogging layer develops at the stone and soil interface (Magdoff and Keeney, 1975). In the clogging layer TSS and pathogenic bacteria are filtered by sorption to surfaces and phenolic compounds in the peat. Hydraulic failure of SAFs have been observed due to reduction in hydraulic conductivity caused by excessive clogging. Clogging layer development, however, is important in the treatment and removal of organic constituents of STE (Walker et al., 1973). Additionally, reduced hydraulic conductivity increases effluent retention time and may cause anaerobiosis. In this zone further microbial reactions occur. Some of these, such as ammonification and urea hydrolysis, convert organic N to NH_3 and $(\text{NH}_4)_2\text{CO}_3$ (Laak, 1982; Sikora and Corey, 1976). Gaseous products will diffuse and are subject to volatilization.

In some alternative septic systems, using *Sphagnum* peat moss, effective filtration occurs while no clogging layer forms. In a column study representing *Sphagnum* peat filter disposal fields, Baker (1986) observed high reductions in TSS and pathogens, while no clogging layer developed.

Chemical Processes

Phosphorus removal is dependent upon chemical sorption by hydrous oxides of iron (Fe^{3+}) and aluminum (Al^{3+}), and precipitation of Fe, Al and Ca phosphates (Nichols and Boelter, 1982). Tilton and Kadlec (1979) report that P removal from secondary treated wastewater applied to peat soils results from sorption by fulvic and humic acids that form stable organo-metallic phosphates.

Ionic bonding to the soil surface may be responsible for reduction of up to 90% of viruses in STE applied to SAFs (Sproul, 1975). Farnham and Brown (1972) report excellent fecal coliform removal from STE applied to peat filters.

A limited fraction of NH_4^+ -N may be removed from STE through adsorption onto cation exchange sites in clay minerals and organic matter. Rock et al. (1984) conducted STE filtration studies with *Sphagnum* peat columns. They conclude that adsorption of NH_4^+ -N is likely to occur. However, when soil pH is less than 7, total NH_4^+ -N adsorption decreases due to a reduction in pH dependent cation exchange capacity. Fixation of NH_3 by organic matter may also remove N (Gilbert et al., 1979). However, N fixed in this manner may be subject to mineralization, and is not truly removed.

Additional losses of N may occur from the reaction of NO_3^- -N with organic matter. Chemodenitrification results from an accumulation of NO_2^- -N under low pH and reducing conditions. NO_2^- , an intermediate product of the nitrification process, may be denitrified without being oxidized to NO_3^- first (Stevenson, 1986). The significance of N losses due to chemodenitrification is poorly understood (Stevenson, 1986). However, evidence of an interaction between NO_2^- -N and

organic matter has been established. With respect to wastewater treatment, there was no mention of chemodenitrification in any of the literature reviewed for this paper. Concern regarding the health effects of NO_2^- accumulation and carcinogenic nitrosamines have not been well established (Stevenson, 1986).

Biological Processes

Significant fractions of pollutants are transformed and removed through microbial processes. Within the SAF, these processes are partitioned into different zones. Above the soil-stone interface, the clogging layer generally is anoxic. Directly below the clogging layer, the soil is aerobic. Below the aerobic zone, if a water table is present the soil may be anaerobic. Depending on the oxygen status, microbial activity will vary.

In a recent study, STE treated in peat provided complete removal of fecal coliform organisms (Mote, 1985). Brooks et al. (1984), conducted field studies using peat for on-site wastewater treatment, and reported >99.9% removal of fecal coliform organisms. Three possible factors responsible for pathogen removal are, physical detention, microbial die-off, and predation by other pathogens. Brooks (1988) reported remarkable bacterial die-off in peat filtration systems. Detention times of 1 or 2 days in acid peat soils resulted in die-off of enteric bacteria. Another potential contributor to bacterial removal is *Penicillium* that is bacteriocidal to *Escherichia coli* (Brooks, 1988).

Nitrogen occurs in several forms in domestic wastewater. Total Kjeldahl Nitrogen (TKN) content in domestic wastewater ranges from 10 - 40 mg L⁻¹.

Approximately 80% of the TKN is in the form of urea from urine (Laak, 1982). Anaerobic microbial activity accounts for the mineralization of organic-N, such as urea, proteins, and amino acids to ammonia (Laak, 1982). Within the septic tank and the anoxic zone of the SAF, urea may be hydrolysed by the urease enzyme to ammonium-carbonate following the general reaction shown below.



Within the organic mat anaerobic heterotrophic organisms remove organic carbon, $\text{NH}_4^+\text{-N}$, $\text{NO}_3^-\text{-N}$, and P by assimilation into microbial tissue. Significant reduction in carbon content occurs in this zone due to immobilization. Microbial immobilization of N also may occur. Several bacteria and some filamentous fungi can utilize both $\text{NH}_4^+\text{-N}$ and $\text{NO}_3^-\text{-N}$ (Alexander, 1977). Additional N may be removed below the mat through immobilization by aerobic organisms. Phosphorus removal is less dependent on microbial immobilization than on precipitation with Fe and Al.

Microbial respiration accounts for a significant reduction in carbon and total solids loading. Oxidation of organic carbon by heterotrophs results in significant COD reductions in columns representing mound type disposal fields (Magdoff and Keeney, 1975).

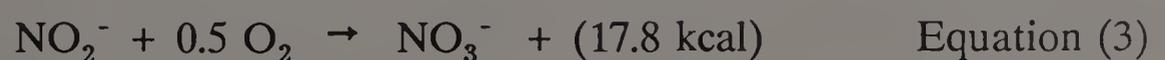
At high C:N ratios existing in the organic mat zone, organic N will be mineralized to $\text{NH}_3^-\text{-N}$. This biochemical transformation, called ammonification,

is mediated by both anaerobic and aerobic heterotrophic organisms. Large macromolecules are degraded to simpler N containing biochemicals. Other enzymes such as amino acid dehydrogenases, oxidases, reductases, amidohydrolases, and *N*-acetylglucosamine kinase release NH₃ from amino acids, nucleic acids and amino sugars (Stevenson, 1986). The process of ammonification is important for removal of N by nitrification and denitrification. Additionally, losses of NH₃-N will occur by volatilization when diffusion gradients exist (EPA, 1975).

In the aerobic zone below the organic mat conversion of NH₄⁺ to NO₃⁻ occurs. The nitrification process is carried out by obligate chemoautotrophic bacteria such as *Nitrosomonas* and *Nitrobacter* (Alexander, 1977). These organisms mediate two distinct reactions. The first oxidation reaction is carried out by *Nitrosococcus* and *Nitrosomonas* species. Equation 2 is the general form of the reaction.



The energy from this reaction allows these organisms to fix inorganic carbon for cell synthesis. The second reaction in the nitrification process mediated by *Nitrobacter* is shown in equation 3.



In this reaction, nitrite is converted to nitrate yielding a very small amount of energy for carbon fixation.

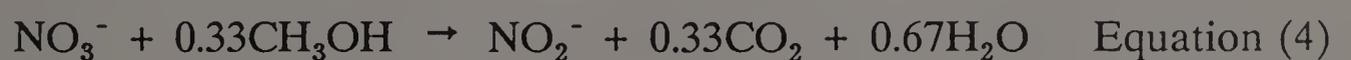
Several heterotrophs have been shown to oxidize NH_4^+ and organic N to NO_3^- and NO_2^- . They include bacteria, actinomycetes, fungi and algae. However, their contribution to nitrification in disposal fields has not been established (Stevenson, 1986). Nitrification is essential for removal of nitrogen in a septic system. The nitrogen must be in the nitrate form before N can be removed in significant quantities through denitrification. Nitrates do not readily form in the septic tank and aerobic conditions are essential in the SAF to promote significant denitrification.

Denitrification is the conversion of NO_3^- to N_2O or N_2 . There are several intermediates which have been studied by many workers. In denitrification, the organisms utilize NO_3^- as an alternate electron acceptor. Most denitrifiers are facultative anaerobes. There are 33 reported genera of bacteria with the capacity to reduce NO_3^- -N to gaseous N_2 (Stevenson, 1986). Some of the more important heterotrophic organisms are: *Alcaligenes*, *Agrobacterium*, *Bacillus*, and *Pseudomonas*. Several chemoautotrophs can also reduce NO_3^- -N, however, they generally are less important. Some of these species can initiate denitrification with nitrite, or only reduce nitrate to nitrite, or reduce nitric oxide to dinitrogen (Payne, 1981). In the presence of O_2 these organisms will function equally well as aerobes (Payne, 1981), and sufficient populations of denitrifying bacteria should exist within SAFs.

The problem facing design and operation of soil based wastewater treatment systems, particularly those aimed at nitrate reduction, is to attain the optimum environmental conditions required for denitrification. The following conditions are needed for denitrification: nitrogen in the nitrate form, anaerobic conditions, and presence of a carbon source for the denitrifying bacteria (Lance and Whisler, 1976).

The NO_3^- reduction process can be mediated via two biochemical pathways. Nitrate may be removed by microbes which assimilate N into biomass. Assimilatory nitrate reduction involves the reduction of NO_3^- to NH_4^+ . Many of the denitrifiers are capable of assimilating N in this manner. There are two different nitrate reductase enzymes that can reduce NO_3^- to either N_2 or NH_4^+ . Denitrifiers are unique in having two reductase enzymes, enabling them to obtain N for cell growth from NO_3^- in aerobic or anaerobic conditions, while utilizing O_2 or NO_3^- for energy (Payne, 1973). However, a study on peat soils which were either aerated or nonaerated reported that assimilatory nitrate reduction is not important in NH_4^+ rich systems (Avnimelech, 1971).

Denitrification has been described by many workers. The following independent reduction reactions simplify the general pathway (Andreoli et al., 1979).





The reaction shown in equation 5 does not include intermediates that are known to accumulate. They include nitrous oxide (N_2O) and nitric oxide (NO). The former is often detected when measuring denitrification in soils. The latter appears only rarely, and its role as an intermediate is being studied (Payne, 1981). The significance of intermediates arrives only when measuring independent steps of denitrification. In the SAF any one of the intermediates including N_2 will volatilize. Certain organisms may be inhibited by higher concentrations of these gases. Therefore, their study is important to fully optimizing the denitrification process.

Environmental Factors Affecting Nitrogen Removal

The importance of environmental conditions, substrate constituents, and biological populations, are reasons for many studies on denitrification in septic system SAFs. While there are too many documents to review on denitrification in wastewater treatment systems, there are too few that study the conditions promoting denitrification in peat wastewater systems. The following discussion attempts to clarify N removal in peat moss treatment systems.

Temperature

Temperature has been evaluated in many studies that focus on microbial kinetics and specifically on denitrification of STE. The effect of temperature on biochemical processes is known to follow an exponential rate function (Focht, 1974). From Focht's work on denitrification kinetics, he concludes that Q_{10} factors are more like 1.4 to 1.6 for rates of NO_3^- and N_2O reduction.

Denitrification is strongly affected by temperature. Mesophilic denitrifiers will operate from 2°C to 65°C (Alexander, 1977). Optimal temperatures for denitrification range from 24°C to 36°C . Denitrification in sludges at temperatures similar to those found in SAFs, were studied by Halmo and Eimhjellen (1981). They suggest that denitrification occurs at temperature ranges from 0°C to 17°C at rapid rates, and that psychrophilic bacteria play a more substantial role in denitrification. Avnimelech (1971) however, reports optimum temperatures for denitrification in peat soils at 60°F to 65°F .

Acidity

Denitrifying bacteria are sensitive to pH. A typical pH value for *Sphagnum* spp. peat moss is 5.0 (Walmsley, 1977). Alexander (1977) reports that denitrifying populations have been identified at pH values as low as 4.7. The optimal range in pH is from 5.0 to 8.0. In studies made on *Sphagnum* spp. peat moss filter beds, Brooks (1988) found that the pH in the beds tend to be 1.0 pH units lower than the STE. She observed much higher N removal due to fungal immobilization,

rather than bacterial denitrification. Her work suggests that in peat filter beds pH will significantly affect denitrifying populations.

Aeration

Oxygen status is likely to have the greatest effect on denitrification. The moisture characteristic and specific surface area also have an interrelated effect on the aeration state. Denitrifying bacteria prefer O_2 to NO_3^- as an electron acceptor (Klawijk, 1985). Nitrate reduction starts when the aeration status is near zero. However, denitrification has been observed in a sandy loam soil with 2% O_2 content (Parkin and Tiedje, 1984). An explanation for nitrate reduction in poorly aerated soils, is that anoxic microsites occur within aggregates or the soil matrix. The presence of these microsites will be affected by soil density and aggregate stability.

Walker et al. (1973) report an anoxic region occurs above the clogging layer in several sand disposal beds studied. In all of the systems examined, nitrates were absent from this zone. Below the clogging layer an aerobic zone existed where complete transformation of NH_4^+ to NO_3^- occurred within several hours. They concluded that complete removal of N by denitrification did not occur in the deep well-drained sands, because the NO_3^- did not pass through an anaerobic zone.

Redox potential (E_h) has been suggested as a measure of denitrification efficiency (Sikora and Keeney, 1976). Focht (1974), found when E_h ranged between 330 mv to 350 mv NO_3^- disappeared.

Moisture

Soil moisture tension is closely related to the O_2 status in the soil. Several studies have been conducted on the rates of denitrification as affected by moisture status. Pilot and Patrick (1976), measured nitrate reduction in three alluvial soils. Soil samples were air dried and mixed with 200 ppm NO_3^- and ground clover as a carbon source. Samples were compacted and moistened to 5%. Samples were placed on a pressure plate and different tensions were exerted. Nitrate reduction was observed at moisture tensions from 20 to 40 cm. They also noted that finer textured soils required higher tensions to inhibit NO_3^- reduction. Finally, they observed that a decrease in E_h corresponded with an increase in NO_3^- reduction.

Meek et al. (1970) studied the effects of an induced water table on NO_3^- -N concentrations in soil columns. They found that the disappearance of NO_3^- -N was associated with a decrease in E_h , and in O_2 levels in the soil solution and soil atmosphere. They observed denitrification without a water table when the E_h was below 100 mv.

Lance and Whisler (1972) examined the removal of N from secondary treated sewage effluent in soil columns intermittently flooded for 2 days and alternated with 5 day dry periods. Effluent that passed through the columns contained 67% less nitrogen than the incoming effluent. However, a spike of high nitrates appeared immediately after each flooding event. They suggest that even longer flooding periods (2-3 weeks) with about the same length of dry periods show even better N removal.

Baker (1986) used an artificial water table to enhance NO_3^- -N removal from STE using columns representing *Sphagnum* peat moss filter beds. He concluded that there were no significant differences between columns with and without a water table. However, significant decreases in NO_3^- -N were observed in columns having high bulk densities.

Organic Carbon

Magdoff and Keeney (1975) demonstrated removal of N and C from STE using columns representing fill-type disposal fields. They observed a decrease in denitrification caused by a reduction in readily available carbon. Smith et al. (1972) demonstrated 90% removal of NO_3^- -N from municipal waste water mixed with methanol dosed to columns filled with coarse sand. Lance and Whisler (1976) compared the effects of methanol and dextrose on denitrification. They found that adding dextrose at 150 mg L^{-1} , increased N removal from 30% to 90%. They also suggested that methanol was not effective in promoting denitrification. Smith et al. (1972) studied methods to improve denitrification in municipal wastewater. They found that methanol/nitrate ratios of 2.50/1.0 produced 90% reduction of NO_3^- -N in domestic wastewater passed through coarse sand columns.

Gilbert et al. (1979) observed increases in denitrifying bacteria during pulse additions of carbon enriched secondary sewage effluent to soil columns. Their results indicate that organic carbon enrichments can increase NO_3^- -N removal capacity.

A passive denitrification system was studied by Laak (1982). In his study, graywater was brought together with nitrified STE in anaerobic rock filled tank. NO_3^- -N levels were below 7 mg L^{-1} after treatment. Laak concludes that graywater may be a suitable alternative to methanol enhanced denitrification systems. Baker (1986) reported in peat systems higher levels of total organic carbon in the effluent than in the influent. This suggests that carbon may not be limiting in *Sphagnum spp.* peat moss filter beds. This also suggests that N mineralization may have occurred within the system.

In a recent study, Seech and Beauchamp (1988) measured denitrification rates in different sized aggregates of a silt loam soil. They reported 12 times higher denitrification rates in 0.25 mm aggregates than in 10 - 20 mm aggregates. They suggest that carbon supply was limiting denitrifier activity.

Additional Losses

Other significant losses of N occur within septic systems. Some of these include leaching and plant uptake. The most probable loss of NO_3^- -N is to the ground water because the NO_3^- ion is highly mobile it is easily leached. Pollution from NO_3^- -N in groundwater may be lessened by dilution. However, this approach is no longer acceptable under most circumstances.

Nitrate uptake by plants grown on the surface area of fields treated with municipal wastewater has been studied by Farnham (1974), Tilton and Kadlec (1979), and Brooks et al. (1984). They reported that effective nitrate removal can be accomplished through uptake by plants grown on the surface of the disposal area. Additionally, there are accepted procedures for land application of wastewaters to agricultural lands (EPA, 1981). Successful N removal for these practices requires harvest of the plants and their removal from the site. Farnham (1974) reports that grass crops grown on the surface of the peat filter were effective in removing nitrates. However, the system studied was used only seasonally. Therefore, one cannot assume that, for a continuously used system in temperate climates, plant uptake would be sufficient for total N removal. Despite reports of acceptable nitrate removal, the level of N removed by denitrification was not determined in the cases cited.

CHAPTER III

METHODS AND MATERIALS

Physical Plant Layout

The project was conducted at the pilot sewage treatment plant on the University of Massachusetts-Amherst campus. This location is adjacent to the Amherst Municipal Sewage Treatment Facility, which provides the pilot plant via underground pipes direct access to untreated, primary and secondary treated wastewater (Figure 1). To house the treatment reactors, a 5.2 m by 7.3 m plastic Quonset hut was constructed (Figure 2). 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 the end of the Quonset, to house pumps, timers and collection equipment as well as to provide access to the quonset. 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 STE storage container, and effluent pumps. A heavy duty 5 KW heater (W.W. Granger, Inc.; stock no. 2E669) and shutter mounted exhaust fan (W.W. Granger, Inc.; stock no. 2C708A) were installed to prevent freezing and minimize temperature changes.

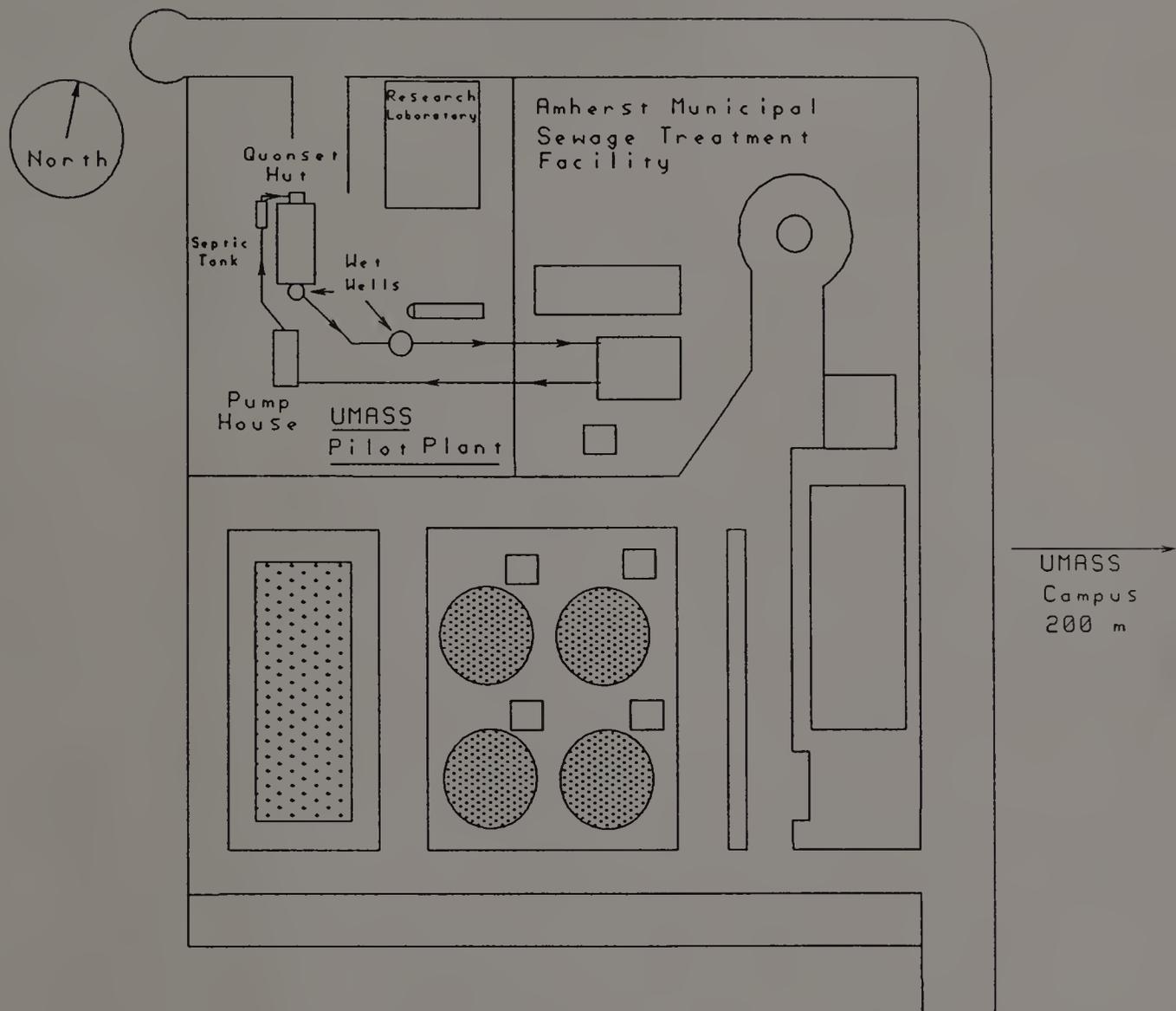


Figure 1. Plan view of University of Massachusetts Pilot Plant and Amherst Municipal Sewage Treatment Facility.

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 per minute using a MoynoTM progressive cavity positive displacement pump (Robbins Myers; Model no. 1L6) with a 1 HP gearmotor. Prior to each dosing, a 2.54 cm Self-Priming Centrifugal pump (W.W. Granger, Inc.; 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 cm x 30.5 cm (length x width) from 2 cm diameter SCH 40 PVC. Ten 0.8 cm orifices were spaced 20 cm along the length of the distribution lateral. Effluent loading was 3 cm day⁻¹ (0.74 ft² gal⁻¹ 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 was 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 then drained into a 57 L high density polyethylene (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 Sewage Treatment Facility.

Treatment Reactor Design

Twelve model treatment reactors were constructed, to allow the evaluation of 4 different treatments in triplicate (Figure 3). System II was constructed using medium sand with the lower 38 cm of the reactor kept saturated. System I also was constructed with medium sand, however, it was free draining. System 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 and had 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.

The reactors (Figure 4) were constructed using 680 L high density polyethylene tanks (Nalgene, Model no. 14100) with dimensions 1.22 m x 0.61 m

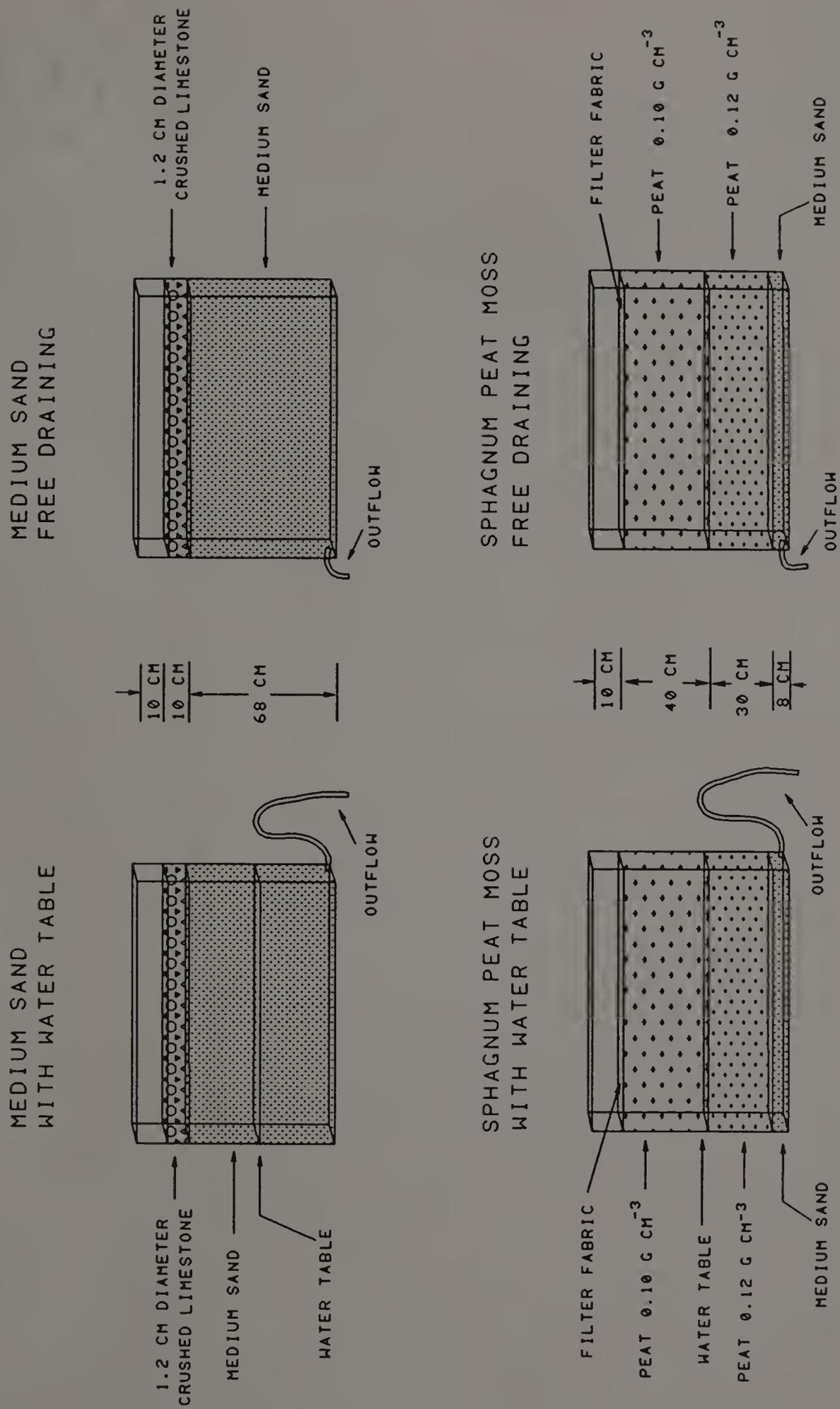


Figure 3. Reactors showing four different treatment, sand and peat and water status.

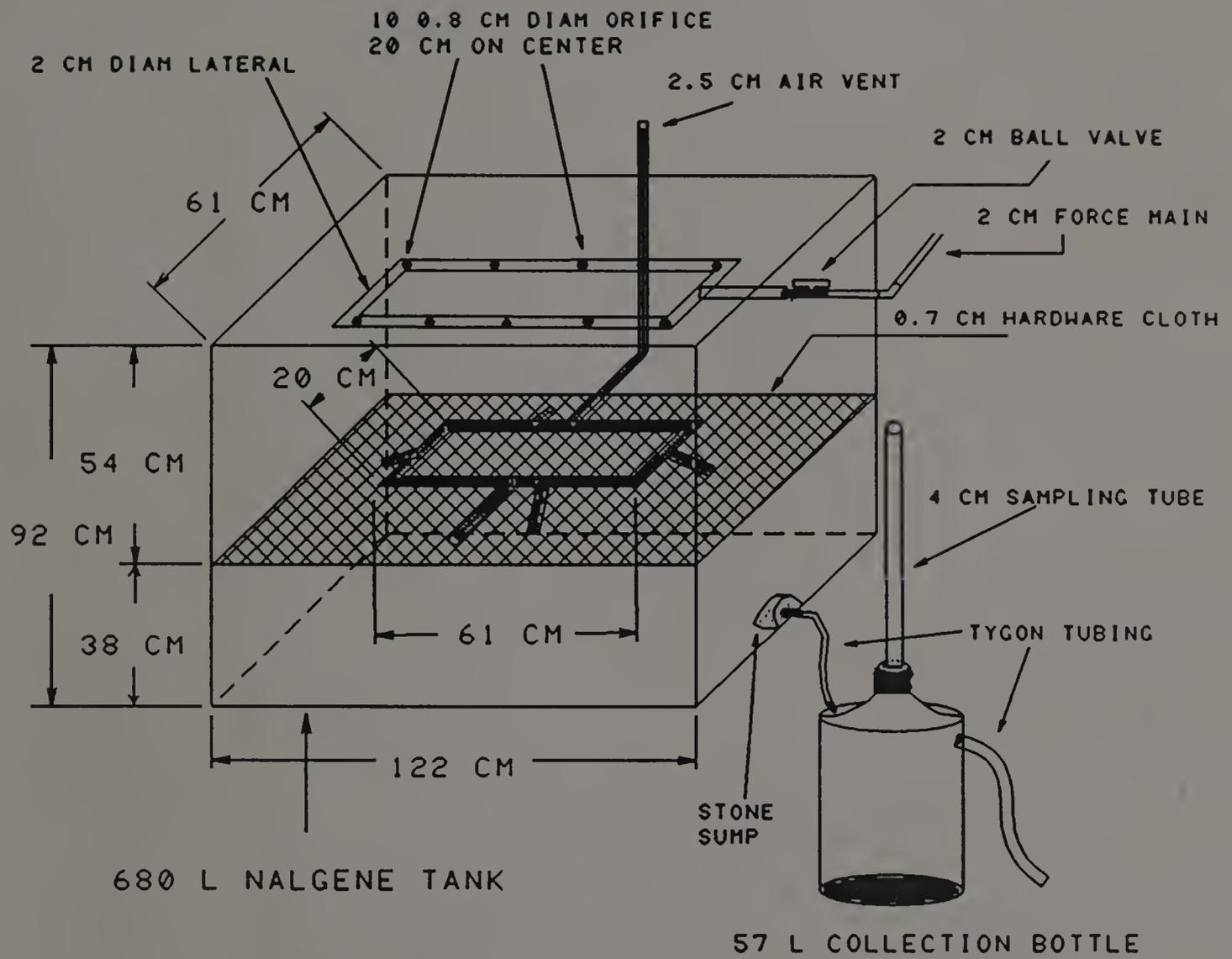


Figure 4. Schematic of individual treatment reactor, showing: lateral, air duct, material depths, and outlet drain.

x 0.92 m (L x W x D). Air exchange piping was built using perforated 2.54 cm SCH 40 PVC 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 densified layers and held in place by 2.54 cm SCH 40 PVC pipe attached to the air exchange piping. **

Treatment Reactor Construction

The polyethylene tanks were fitted with all the inlets and outlets before peat or sand was added. A layer of medium sand 8 cm deep 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 peat reactors, the peat moss was placed in garbage bags. Samples from each bag were oven dried to 104°C for 24 hrs and moisture content was determined. The bags of peat, with known dry weight were added to the reactors. Between adding bags the peat was mixed and fluffed by hand. The densified layer was compressed with the hardware cloth and the air exchange piping was installed. The upper layer was added in the same manner compressed to the 78 cm level. Strips of geotextile, 10 cm wide 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 dry weight determination. Upon determining moisture content at 105°C after 24 hrs, additional sand was added to provide a final bulk density of

1.4 g cm⁻³. Ten centimeter of 1.28 cm washed crushed limestone was added over the sand and the distribution laterals were installed.

Sampling

One day prior to each sampling date, the carboys were emptied using a Teel self priming utility pump (W.W. Granger, 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 sample, 3 L of sample was passed 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 sampled for the determination of the various forms of N and COD were collected in glass containers and returned to the lab within 2 hours. Several aliquots were split from the main sample for measuring pH and a 125 ml sample was frozen for future analysis. The remainder of the sample was acidified to pH 1.5 to 2.0 with 36N H₂SO₄, and stored at 4°C. These 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). Ammonium-nitrogen was determined directly from a 25 ml sample aliquot 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 on the remaining sample after 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 - $\text{K}_2\text{Cr}_2\text{O}_7$ reactor digestion method (Hach Company, Loveland, CO.) followed by titration with 0.1 N $\text{Fe}(\text{NH}_4)(\text{SO}_4)_2 \times 6(\text{H}_2\text{O})$ as titrant for excess oxidizing agent (American Public Health Association, 1985).

Chemical and Physical Properties of Sand and Peat

Sand texture, coefficient of uniformity and coefficient of gradation 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 for 24 hours, using the loss on ignition method at 450°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 (Boelter, 1969).

Peat Hydraulic Conductivity

Constant head permeameters, 5 cm in diameter and 25 cm tall (similar to Model no. K600, Soil Test Inc.), 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 cm³ respectively. All weights are on oven dry basis (104°C for 24 hours). Three permeameters for each treatment were placed randomly along the lab 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 passed 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:

$$K_{sat} = \frac{Q \Delta L}{A t \Delta H} \quad \text{Equation (7)}$$

Where K_{sat} = saturated hydraulic conductivity in cm sec^{-1} , Q = volume of water in ml passing through column, t = length of time of measurement in seconds, A = cross sectional area of column in cm^2 , ΔL = length of column in cm, and ΔH = 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 0.44 ml sec^{-1} . After dilution with incoming unbrominated STE, the concentration of Br^- in the STE averaged 40 mg L^{-1} . The reactor outlets were installed with ball valves to allow for sampling of effluent as it leaves 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 Br⁻ breakthrough was expressed as:

$$\% \text{ Breakthrough} = \frac{C_t}{C_o} \times 100 \quad \text{Equation (8)}$$

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

Acetylene Blocking

Denitrification was measured by N₂O evolution using the acetylene blocking technique (Tiedje et al., 1989). One core was taken from reactor 1 and 4 respectively, representing the peat treatment with a water table. The reactor sidewall was opened and plugged with a SCH 40 threaded endcap to allow access. Twenty four hours prior to taking a core, a liter of effluent was taken from the reactor 10 cm below the water level, and sparged with acetylene until added to the peat core. Each core was removed from the 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 after removing 20 cm of peat with a 2.5-cm screw auger. Upon core removal, approximately 750 mg of peat was

transferred to a 125 ml erlenmeyer flask, combined with the acetylene saturated effluent and stoppered with a rubber septum. The flasks were immediately moved to the laboratory where they were sparged with N₂ for 5 minutes. Three ml of acetylene was injected into the head space of the flask. After 6 hours the flasks were allowed to equilibrate to atmospheric pressure. After 6, 12, 16 and 21 hours a 3.5-ml volume of air was added to the head space after which a 3.5-ml sample was taken from the headspace. The sample volume was then reduced to 3-ml and injected in a 3-ml vacutainer (Fisher, Model no. 02-683-77). Gas samples were analyzed for N₂O 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 within factor, time, were analyzed using the following expected mean squares model, where α = peat or sand material, β = water status, τ = time and ϵ ~ independent and normally distributed (IND).

$$Y_{ijkl} = \mu + \alpha_i + \beta_j + \alpha_i\beta_j + \tau_k + \alpha_i\tau_k + \beta_j\tau_k + \alpha_i\beta_j\tau_k + \epsilon_{ijkl} \quad \text{Equation (9)}$$

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 did here.

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 and τ = time and $\epsilon \sim \text{IND}$.

$$Y_{ijk} = \mu + \alpha_i + \tau_j + \alpha_i\tau_j + \epsilon_{ijk} \quad \text{Equation (10)}$$

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 IV

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 ran 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 system was shut down due to inconsistent dosing. The feed system described in the methods section 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. It was observed that the bulk of the peat in the upper layer

of this reactor had decreased. The dosing system operated flawlessly from that date on.

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 that fed the storage container. A stone filter was installed in the outlet baffle in the septic tank. The baffle was constructed using 1.28 cm crushed washed limestone enclosed in 0.65 cm wire mesh box. The stone box was designed so it could be removed and cleaned. It has not been cleaned 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 to 10 cm capillary rise above the induced water table level, in the peat reactors with a water table was observed through the side wall of the reactor. 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 coefficient of gradation equal to 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 cmol Kg⁻¹. Organic matter was less than 0.1% as determined by loss on ignition method (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 a fibric peat. The pH of the peat was 3.3 in 0.01 M CaCl₂. The cation exchange capacity was 172 cmol Kg⁻¹ on 105°C oven-dried weight. Organic matter was 97.9% as determined by loss on ignition method (Davies, 1974). Total carbon and total nitrogen in the peat were 50.48% and 0.69% respectively.

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

Item	Sand	Peat
Description	Medium Sand	Fibric Peat (Canadian Sphagnum)
Bulk Density	1.4 g cm ⁻³	0.10 - 0.12 g cm ⁻³
pH	5.2	3.3
CEC	1.2 cmol ⁺ Kg ⁻¹	172 cmol ⁺ Kg ⁻¹
Organic Matter	0.1%	97.9%
Total Carbon	0.15%	50.48%
Total Nitrogen	<0.05%	0.69%
Uniformity Coefficient	4.3	na
Coefficient of Gradation	0.8	na
Fiber Content	na	87% - 93%

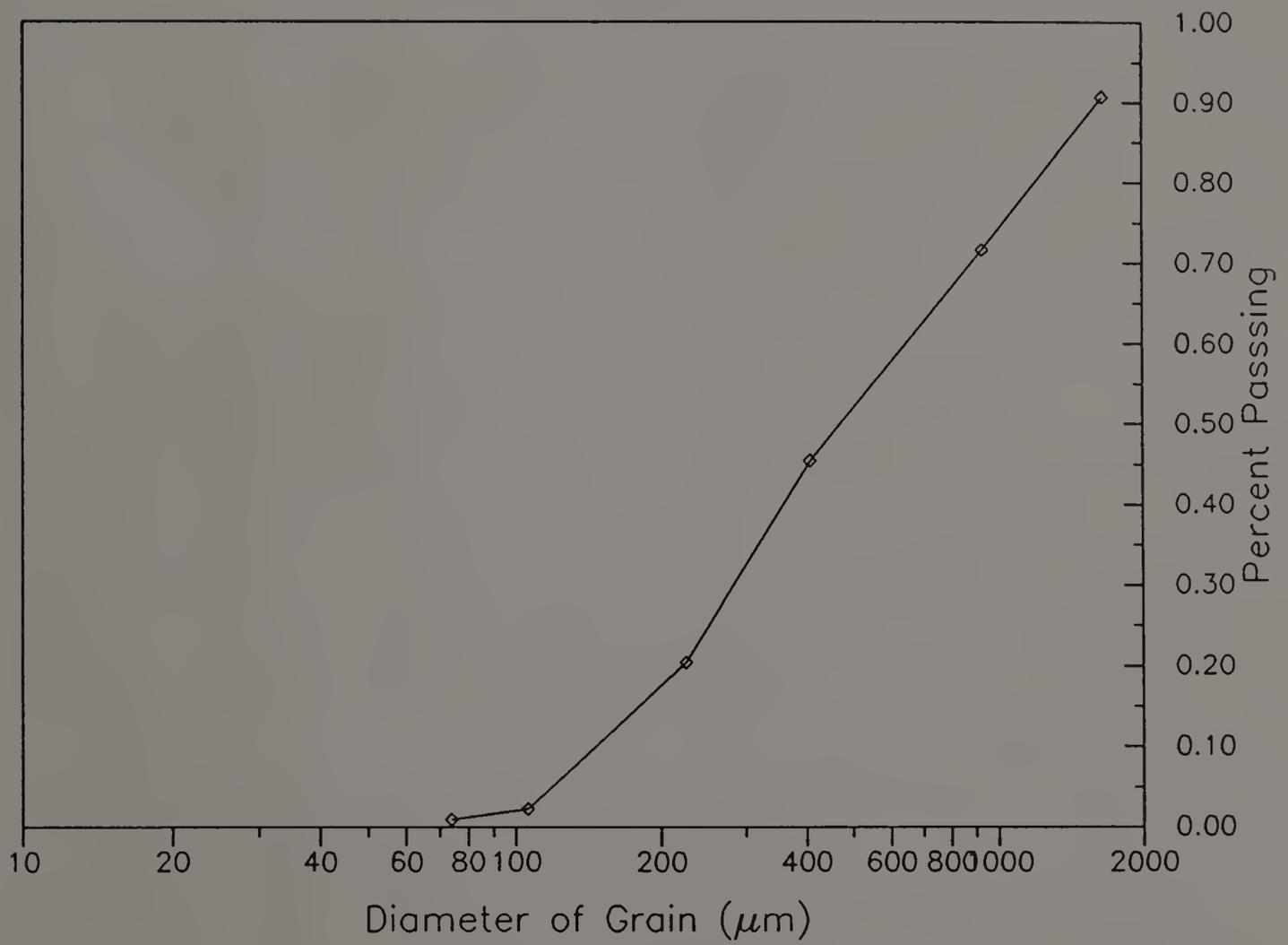


Figure 5. Grain size distribution of the sand used in the reactors.

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 ρ_b , 0.125 g cm⁻³, had the highest mean K_{sat} , 9.2×10^{-4} cm sec⁻¹ equivalent to 80 cm per day. The highest ρ_b , 0.140 g cm⁻³ exhibited 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 an initial deairing of pores after which 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 4.2×10^{-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 had not run long enough to allow for total equilibrium. Regression analysis of transformed K_{sat} values using $10^{K_{\text{sat}}}$, resulted in R^2 of 94.2, using the linear equation:

$$\text{Mean } K_{\text{sat}} = \text{Log} (1.0113 - 0.07446 \times \rho_b) \quad \text{Equation (11)}$$

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

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

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

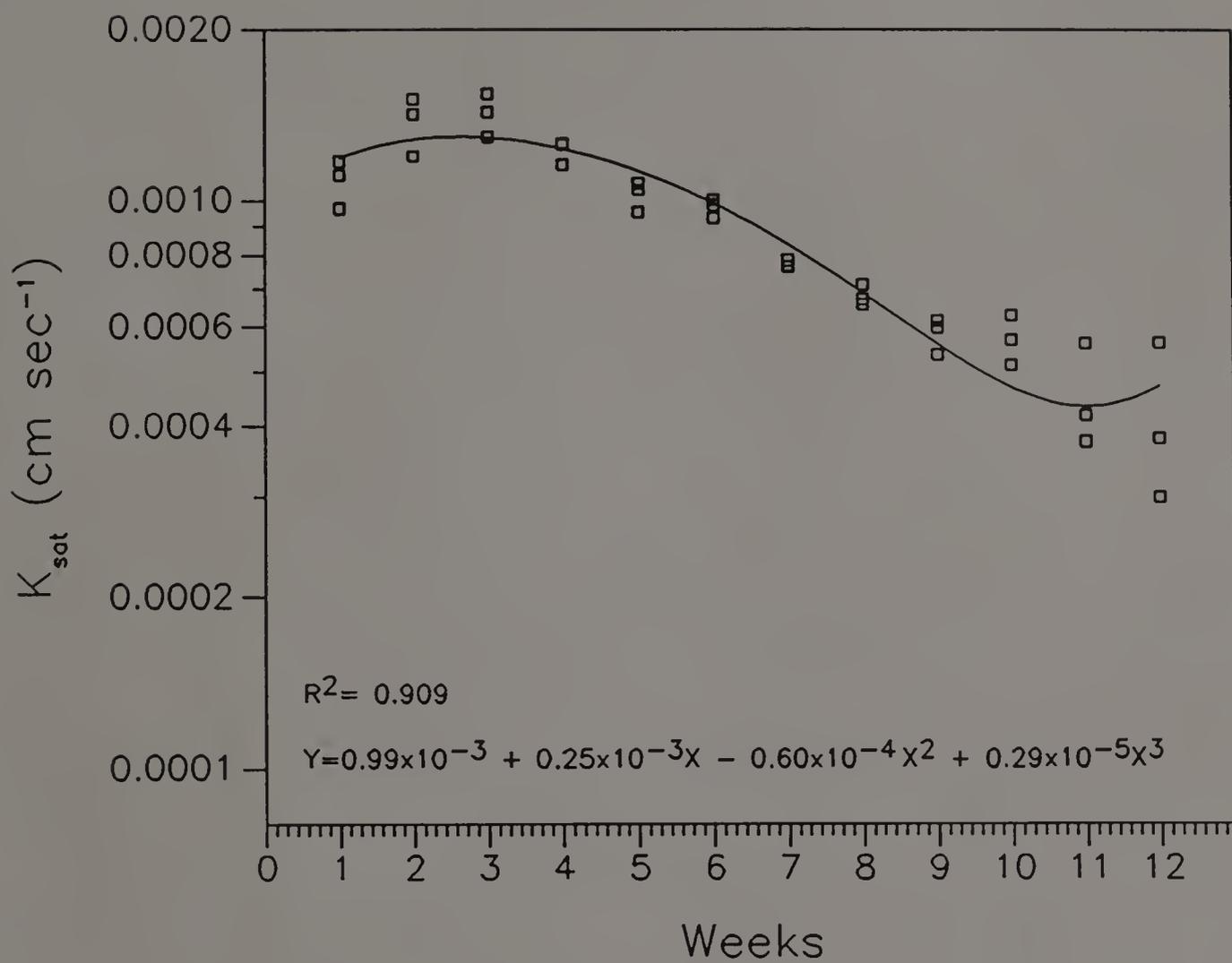


Figure 6. Change in hydraulic conductivity of sphagnum peat moss with bulk density 0.125 g cm^{-3} .

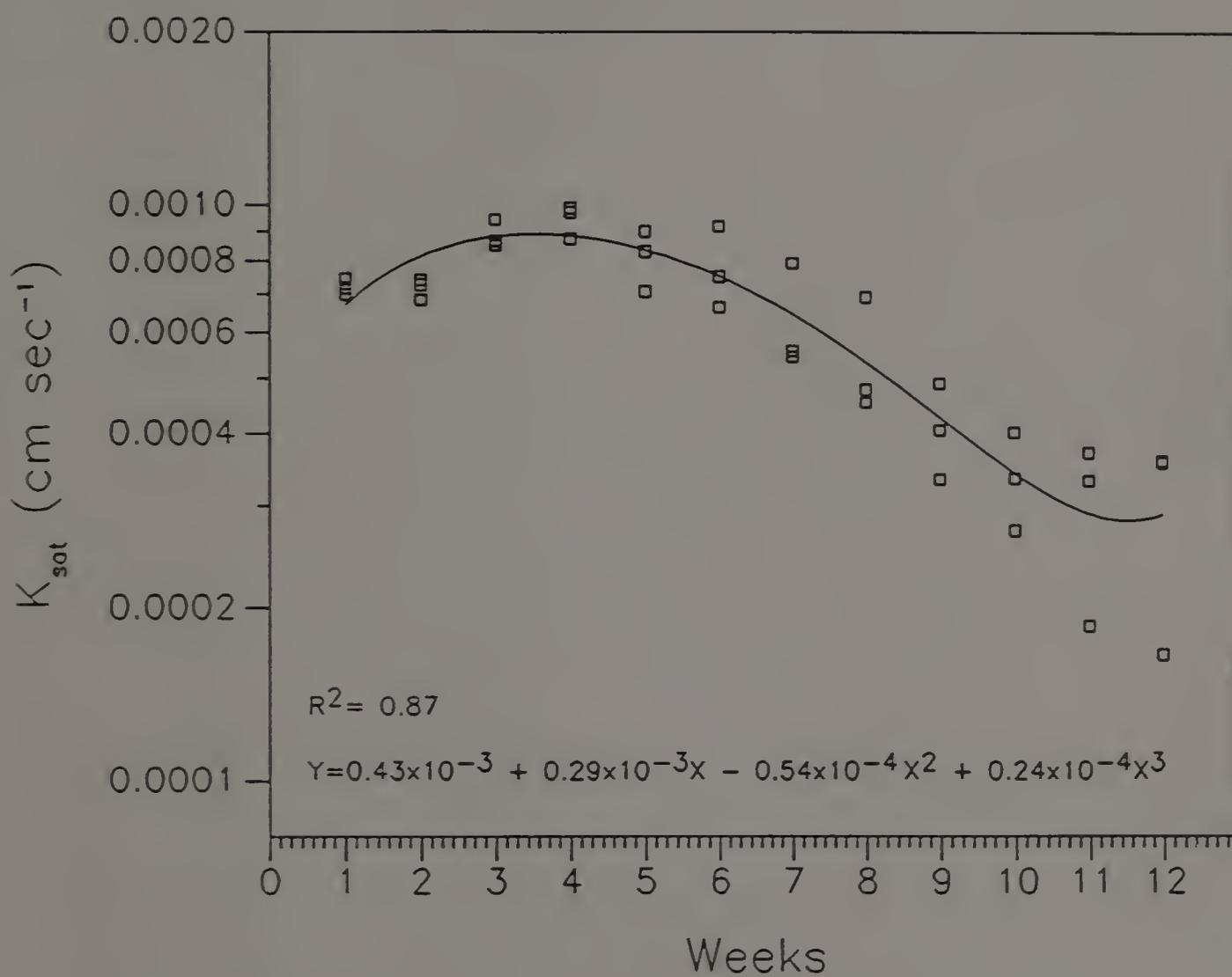


Figure 7. Change in hydraulic conductivity of sphagnum peat moss with bulk density 0.130 g cm^{-3} .

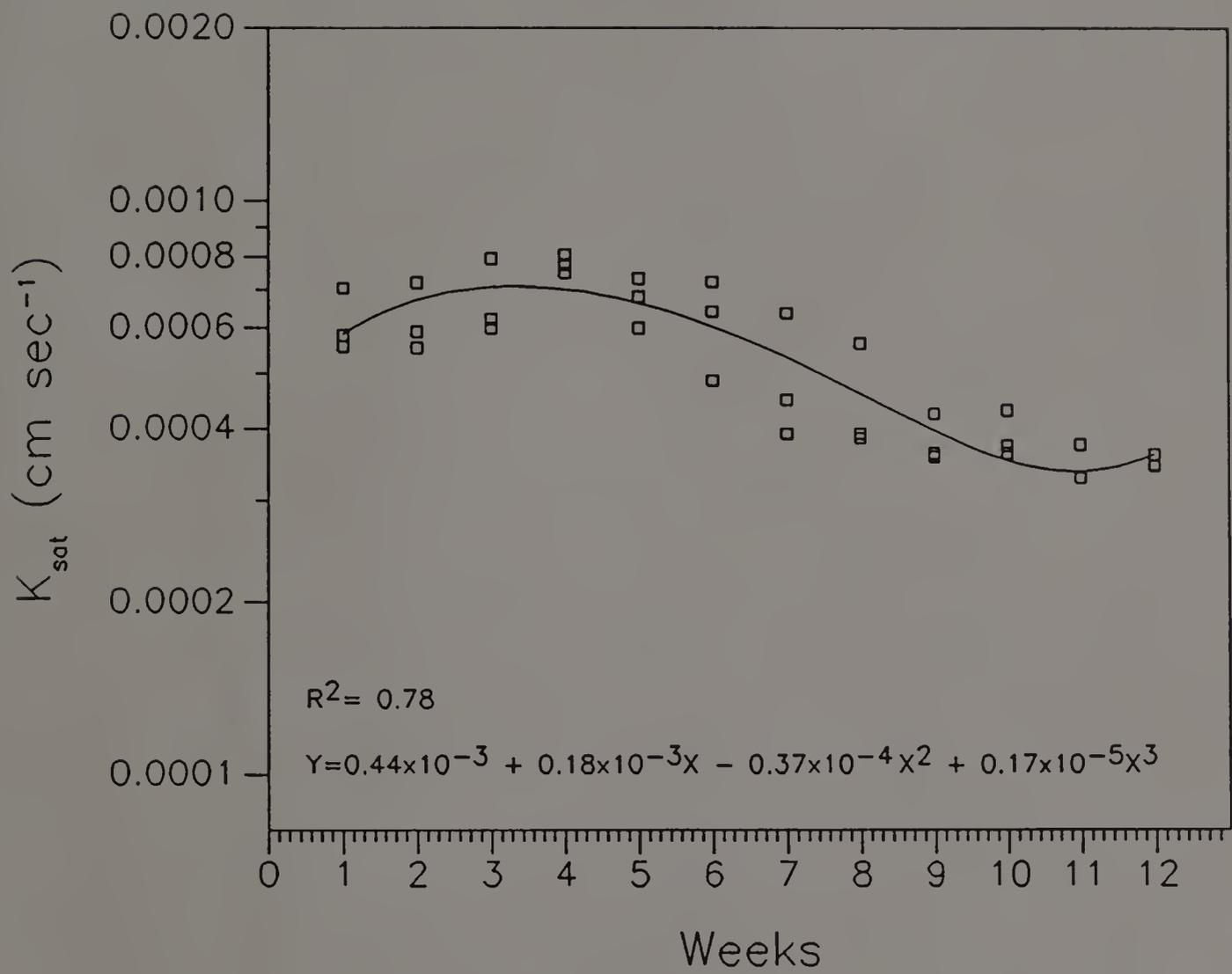


Figure 8. Change in hydraulic conductivity of sphagnum peat moss with bulk density 0.135 g cm^{-3} .

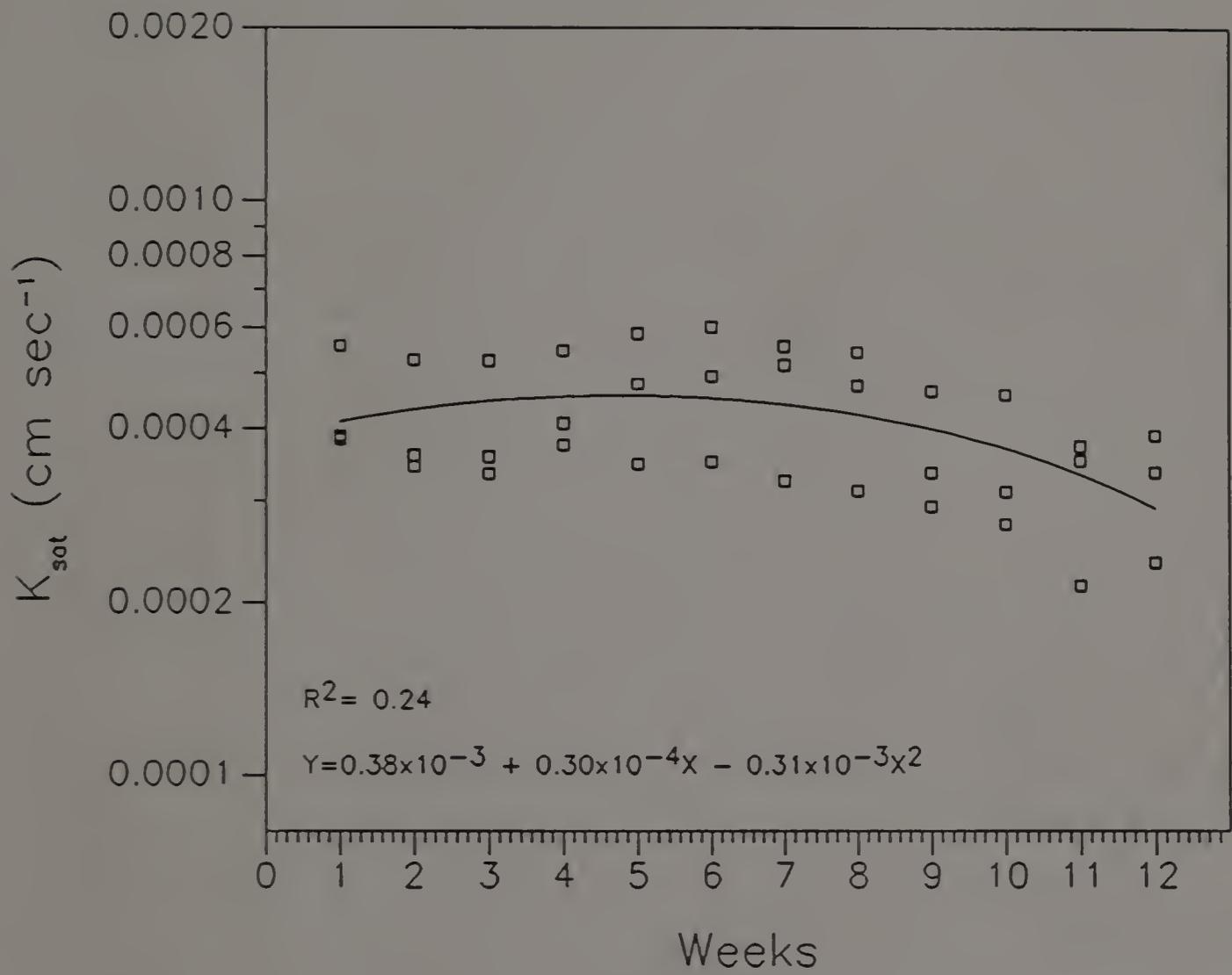


Figure 9. Change in hydraulic conductivity of sphagnum peat moss with bulk density 0.140 g cm^{-3} .

Residency Time Study

Residency times were determined with bromide solutions. Breakthrough was defined as the point at which concentration of Br^- in the reactor effluent was equal to the Br^- in the STE loading the reactors. Regression analysis was used to evaluate the breakthrough curves. Raw data are shown in Appendix B. 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 flow may have occurred through macropores as evidenced by the initial steep slopes in percent breakthrough (Figures 10 and 11). However, curves do not suggest that channeling or bypassing occurred. Residency times for the peat reactors without a water table and with a water table were 28 and 30 days respectively. Figures 12 and 13 show regression lines fitted to breakthrough data from peat treatments. Visual analysis of the slopes of the curves in these plots suggest that flow was not through macropores but through 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.

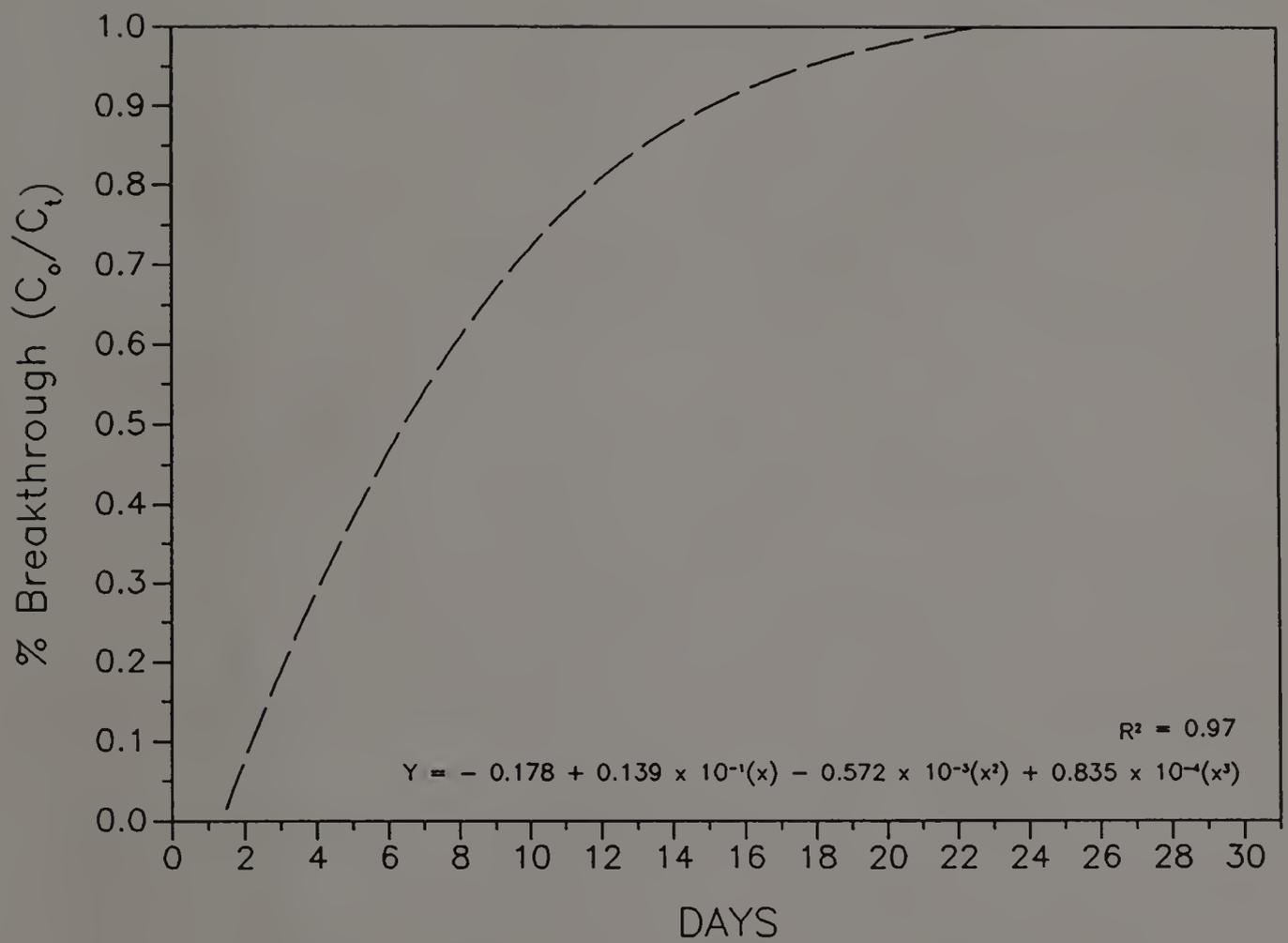


Figure 10. Residency time plot for sand treatment without a water table.

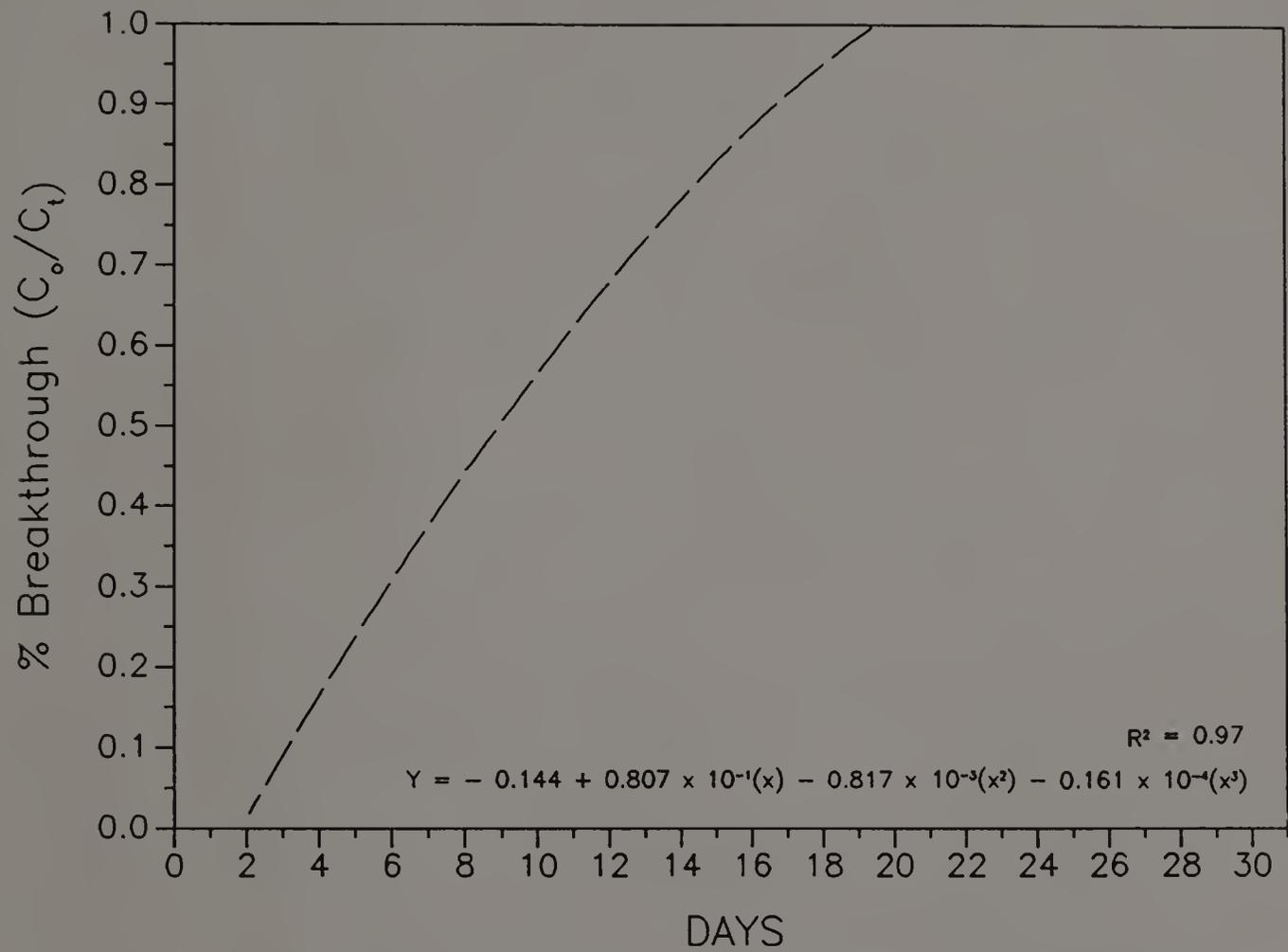


Figure 11. Residency time plot for sand treatment with a water table.

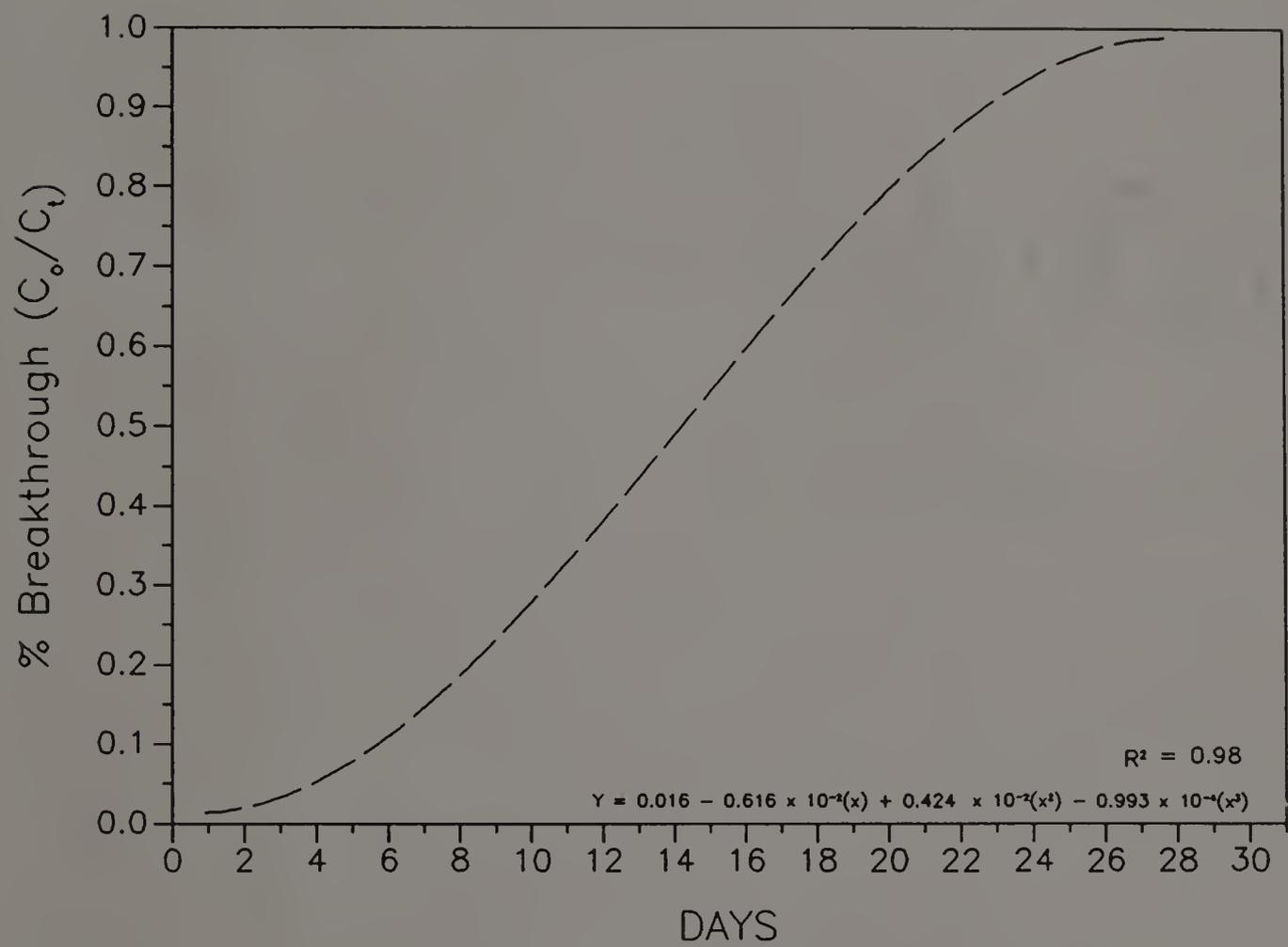


Figure 12. Residency time plot for peat treatment without a water table.

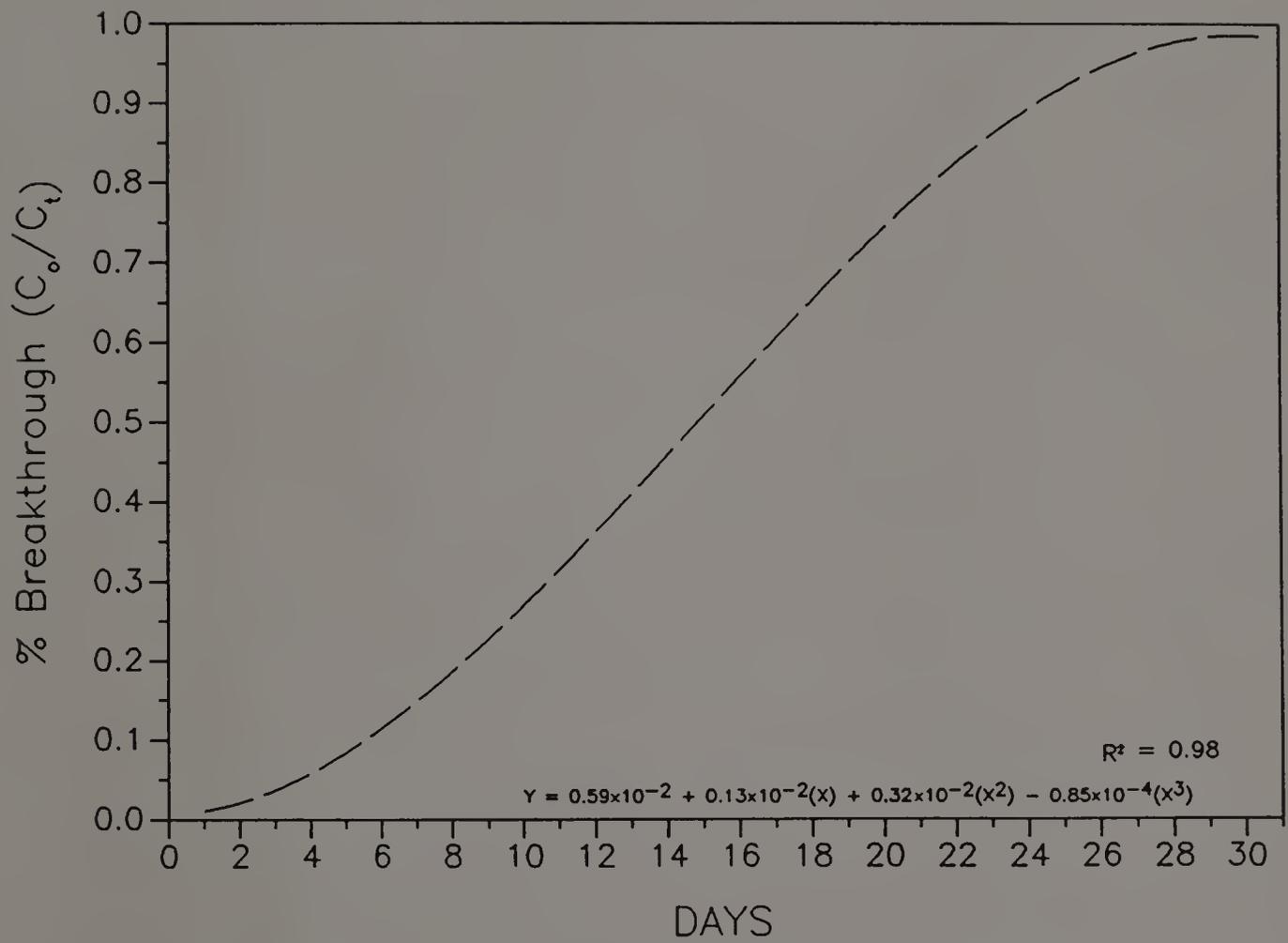


Figure 13. Residency time plot for peat treatment with a water table.

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 (BOD_5). Whereas BOD_5 is more commonly used in the analysis of domestic wastewater, in this study the relationship between COD and BOD_5 removal efficiencies were assumed to be consistent and the more rapid 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 were not significant different at the $p=0.05$ level. Peat treatments showed

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.

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

TREATMENT		Sand		Peat	
		w/out water table	with water table	w/out water table	with water Table
Sand	w/out water table				
	with water table	2.3			
Peat	w/out water table	37.6 *	35.4 *		
	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.

significantly higher COD levels than sand treatment, with differences significant at the 0.01 level.

More information is revealed when repeated measures ANOVA is performed on the reactor effluent excluding the STE, using Equation 9 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 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 10 indicates that the lines do not follow similar patterns. Significant interaction therefore exists based on visual plots and the statistical analysis. 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 standards 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 Canadian sphagnum peat beds with tap water, background TOC levels measured 50.7 mg L^{-1} . Therefore, it may be reasonable to assume that higher COD values in the peat systems, are related to

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 by Water Status	9799.35	8	1224.92	3.06	<0.01
Error	558.52	160	3.49		

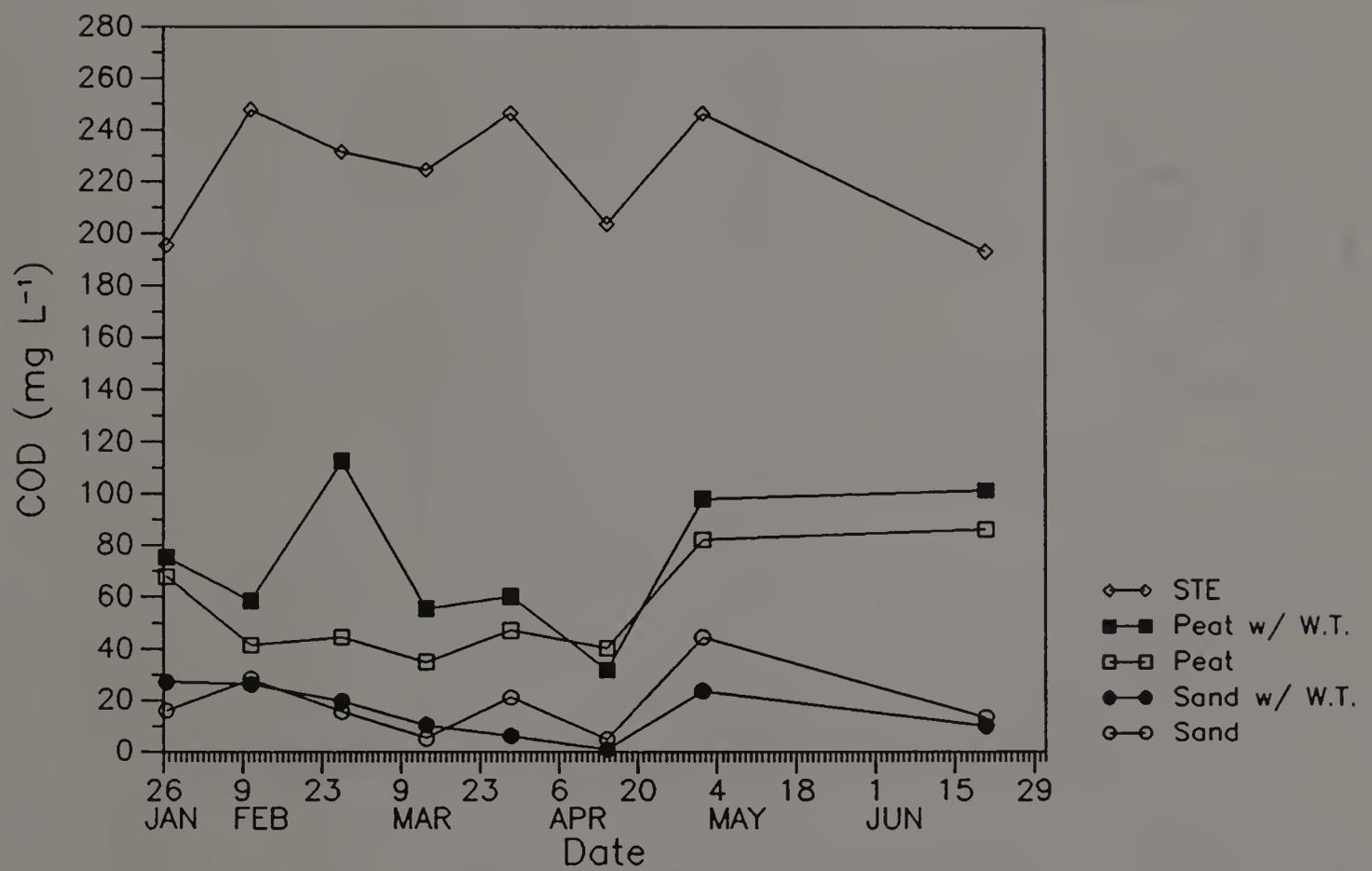


Figure 14. Mean values of COD (mg L^{-1}) for septic tank and treatment reactor effluent over time.

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 (NH_3 -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 NH_3 -N was observed at the 0.0001 confidence level. Differences between treated wastewater and STE were determined using Tukey's HSD test on their means over time. Mean NH_3 -N in STE and reactor effluents during the study period are shown in Table 6. Mean NH_3 -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, and were not statistically different at the 0.05 level. Mean NH_3 -N for the peat treatments without a water table and with a water table were 11.3 mg L^{-1} and 10.9 mg L^{-1} respectively which was not statistically significant at the 0.05 level. Mean STE NH_3 -N was 23.5 mg L^{-1} . Table 7 presents absolute mean differences and their statistical significance. Net reduction from STE NH_3 -N, based on means, was 89% and 86% for sand treatments without a water table and with a water table respectively; 50% and 51% for the peat treatments without a water table and with a water table, respectively. Peat treatment effluent had higher NH_3 -N levels than sand treatment effluent, with differences significant at the <0.001 level. In addition, peat reactor effluent and

Table 6. Mean values of $\text{NH}_3\text{-N}$ (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.

Table 7. Differences in mean NH₃-N values (mg L⁻¹) for septic tank and treatment reactor effluent.

TREATMENT		Sand		Peat	
		w/out water table	with water table	w/out water table	with water Table
Sand	w/out water table				
	with water table	0.8			
Peat	w/out water table	8.8 *	8.1 *		
	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

sand 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 9 from the previous chapter. These data confirm a statistically significant difference between sand and peat materials. The difference between water status within peat or sand material was statistically not 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 15). 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 $\text{NH}_3\text{-N}$ in reactor effluents were not achieved indicating insufficient nitrification in the aereated 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 $\text{NH}_3\text{-N}$ given a fixed retention time. Therefore, it

Table 8. Analysis of Variance for NH₃-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 by Water Status	163.83	8	20.48	3.05	<0.01
Error	1072.67	160	6.70		

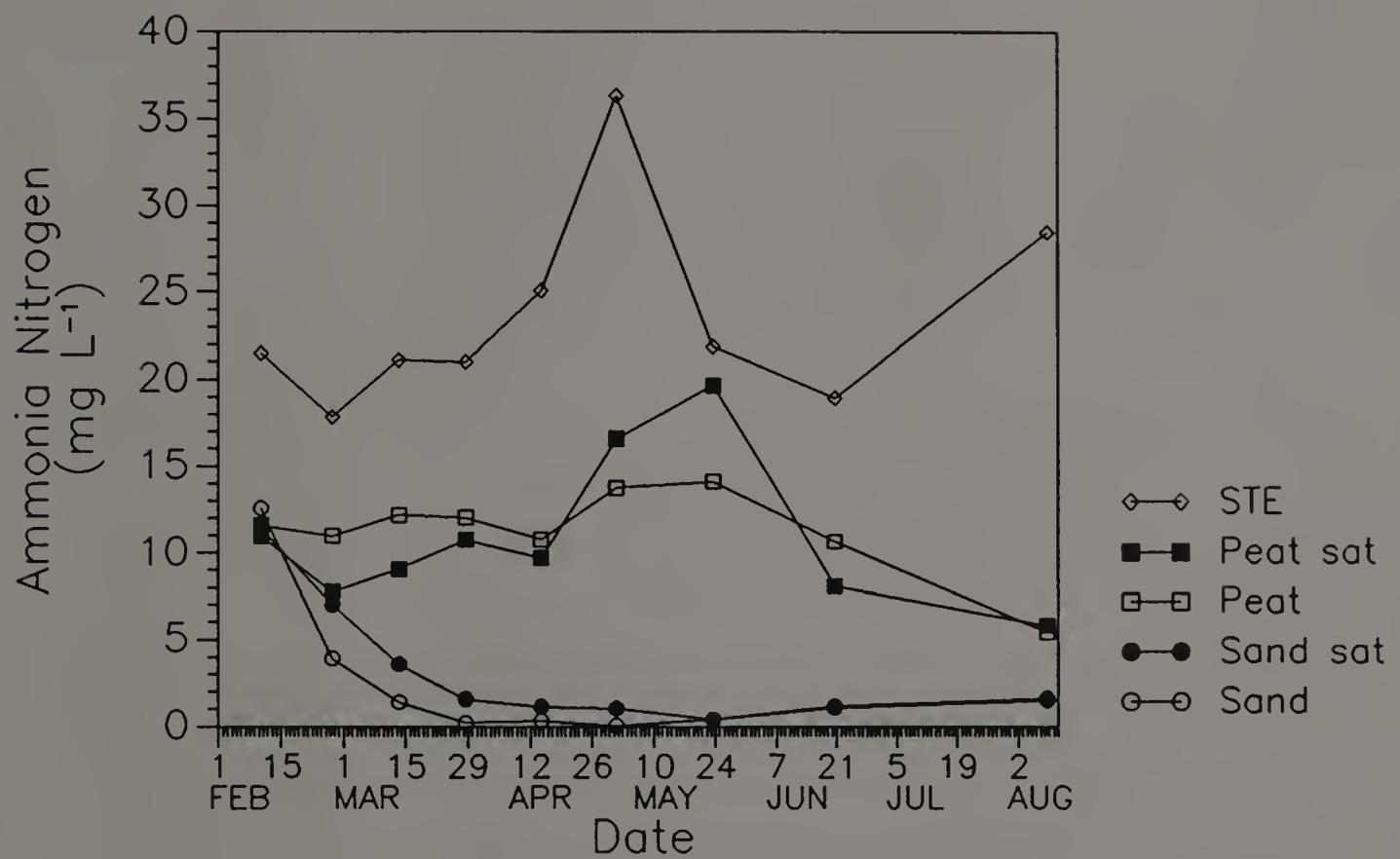


Figure 15. Mean values of $\text{NH}_3\text{-N}$ (mg L^{-1}) for septic tank and treatment reactor effluent over time.

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

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 waste water 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 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 and sand reactor effluent were 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 9 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

Table 9. Mean values of TKN (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	
		w/out water table	with water table	w/out water table	with water Table
Sand	w/out water table	.			
	with water table	1.0			
Peat	w/out water table	8.0 *	7.0 *		
	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

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	27.83	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 by Water Status	70.51	8	8.81	2.52	0.01
Error	558.52	160	3.49		

statistically significant. Within treatments, TKN over time was significantly different at the <0.01 level. Significant interactions occurred between time and material and time, material and water status (Figure 16). Similar trends between TKN and $\text{NH}_3\text{-N}$ are evident from Figures 15 and 16. Repeated measure ANOVA was performed on means of $\text{NH}_3\text{-N}$ and TKN for reactor effluents and STE. Tukey's HSD contrasts on means was performed and absolute mean differences are shown in Table 12. Contrasts between $\text{NH}_3\text{-N}$ 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 $\text{NH}_3\text{-N}$.

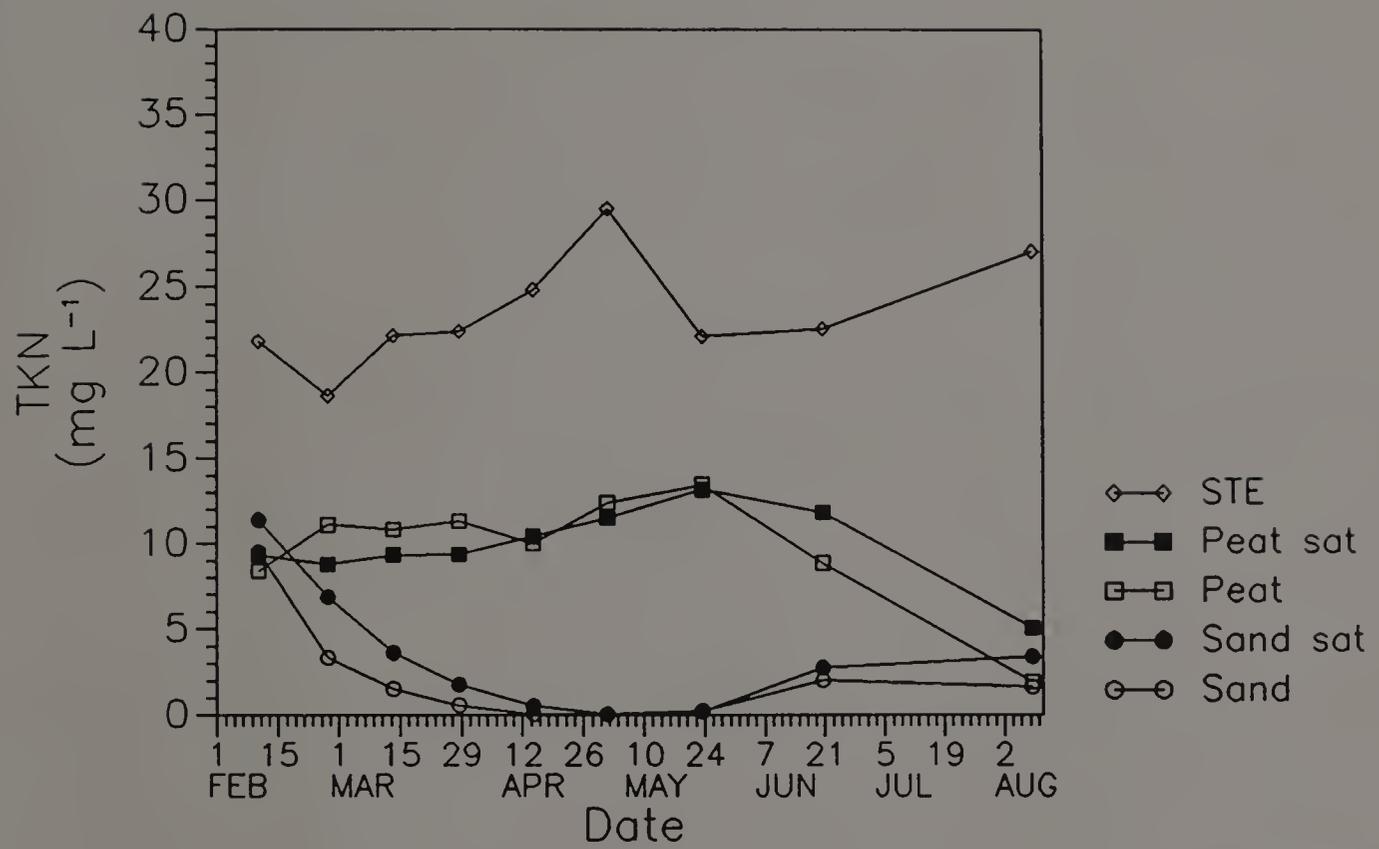


Figure 16. Mean values of TKN (mg L^{-1}) for septic tank and treatment reactor effluent over time.

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

TREATMENT		TKN				
		Sand		Peat		STE
NH ₃ -N		w/out water table	with water table	w/out water table	with water Table	
Sand	w/out water table	0.0				
	with water table	1.0	0.0			
Peat	w/out water table	9.1 *	8.7 *	1.1		
	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

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 was determined using Tukey's HSD test on their 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 $\text{NO}_x\text{-N}$ for the peat treatments without a water table and with a water table were 19.07 mg L^{-1} and 9.3 mg L^{-1} respectively. Differences in $\text{NO}_x\text{-N}$ levels between the peat treatments were statistically significant at the 0.05 level. Mean STE $\text{NO}_x\text{-N}$ was 0.8 mg L^{-1} . Net increase in reactor effluent $\text{NO}_x\text{-N}$ as compared to STE $\text{NO}_x\text{-N}$, was 25.8 and 23.6 for sand treatments without a water table and with a water table respectively, and 18.3 and 8.6 for the peat treatments without a water table and with a water table respectively (Table 14). Peat reactor effluents had lower $\text{NO}_x\text{-N}$ levels than sand treatment effluent, with differences significant at <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 9 from the previous chapter. These data confirm a statistically significant difference between sand and peat materials. However, no difference with respect to water status was statistically significant.

Table 13. Mean values of NO_x-N (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	
		w/out water table	with water table	w/out water table	with water Table
Sand	w/out water table				
	with water table	2.2			
Peat	w/out water table	7.5 *	5.3 *		
	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

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 by Water Status	523.14	8	65.39	2.69	0.01
Error	3891.86	160	24.32		

An interaction between material and water status is evident. Within treatments, $\text{NO}_x\text{-N}$ over time was significantly different at the <0.01 level. In addition, significant interactions occurred between time and material, time and water status and time, material and water status (Figure 17). An increase in $\text{NO}_x\text{-N}$ can be seen until the end of May after which levels decreased. Increased $\text{NO}_x\text{-N}$ in the effluent correspond chronologically with elevated $\text{NH}_3\text{-N}$ in the STE, while $\text{NO}_x\text{-N}$ in the STE remained low throughout the study period.

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 $\text{NO}_x\text{-N}$ and TKN. $\text{NH}_3\text{-N}$ was not considered since it was shown previously to be not significantly different from TKN. Average percent net loss of N were -19%, -15%, -22% and 20% for 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 are gains in N (Figure 18). 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 STE was 23.2 mg L^{-1} with 12.9 mg L^{-1} variance from 9 sampling periods. In addition, elevated levels of $\text{NO}_x\text{-N}$, reported on May 23, were on average 1.9 times mean $\text{NO}_x\text{-N}$ over time for each reactor. On this date, $\text{NO}_x\text{-N}$ from selected reactors

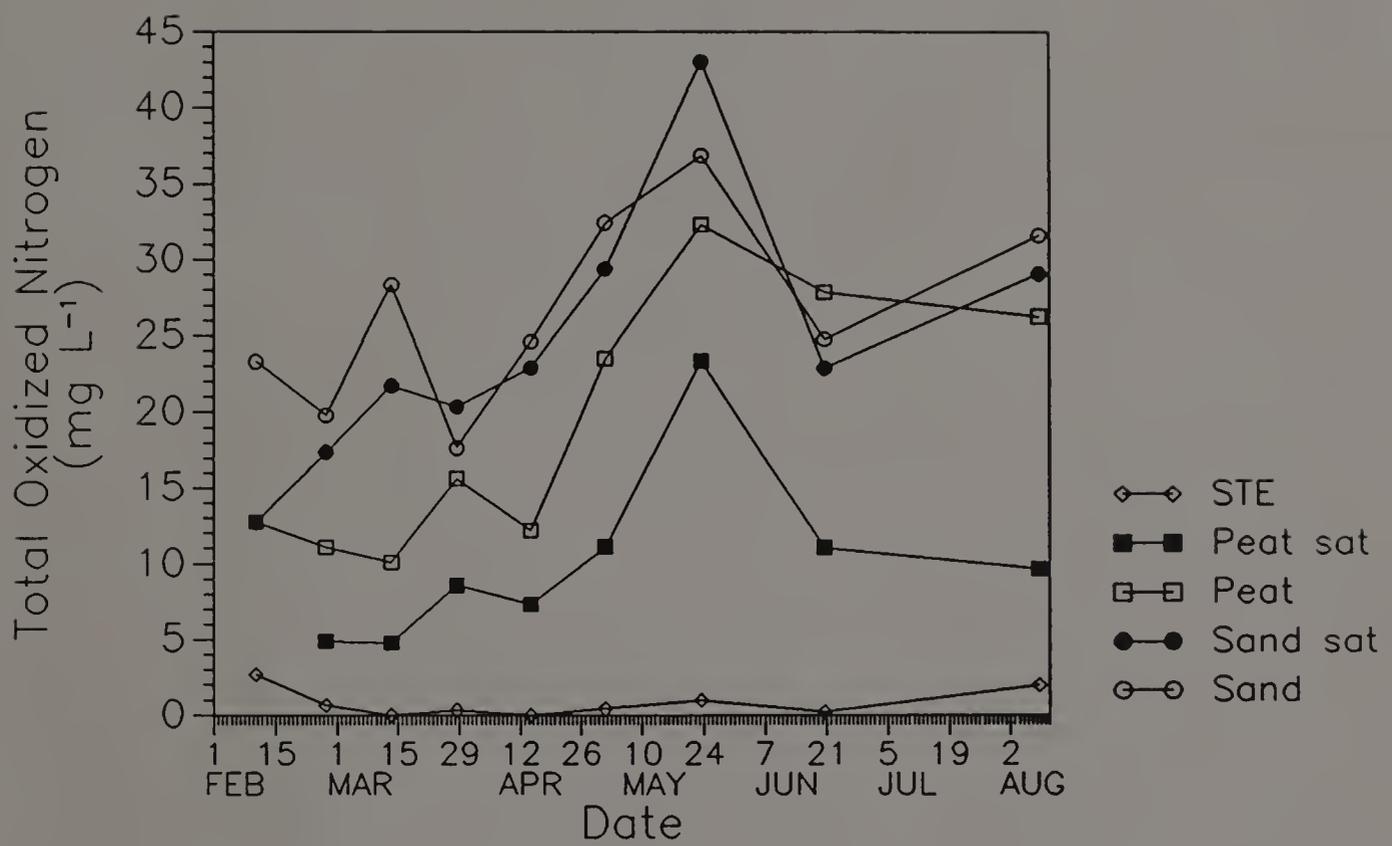


Figure 17. Mean values of $\text{NO}_x\text{-N}$ (mg L^{-1}) for septic tank and treatment reactor effluent over time.

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.

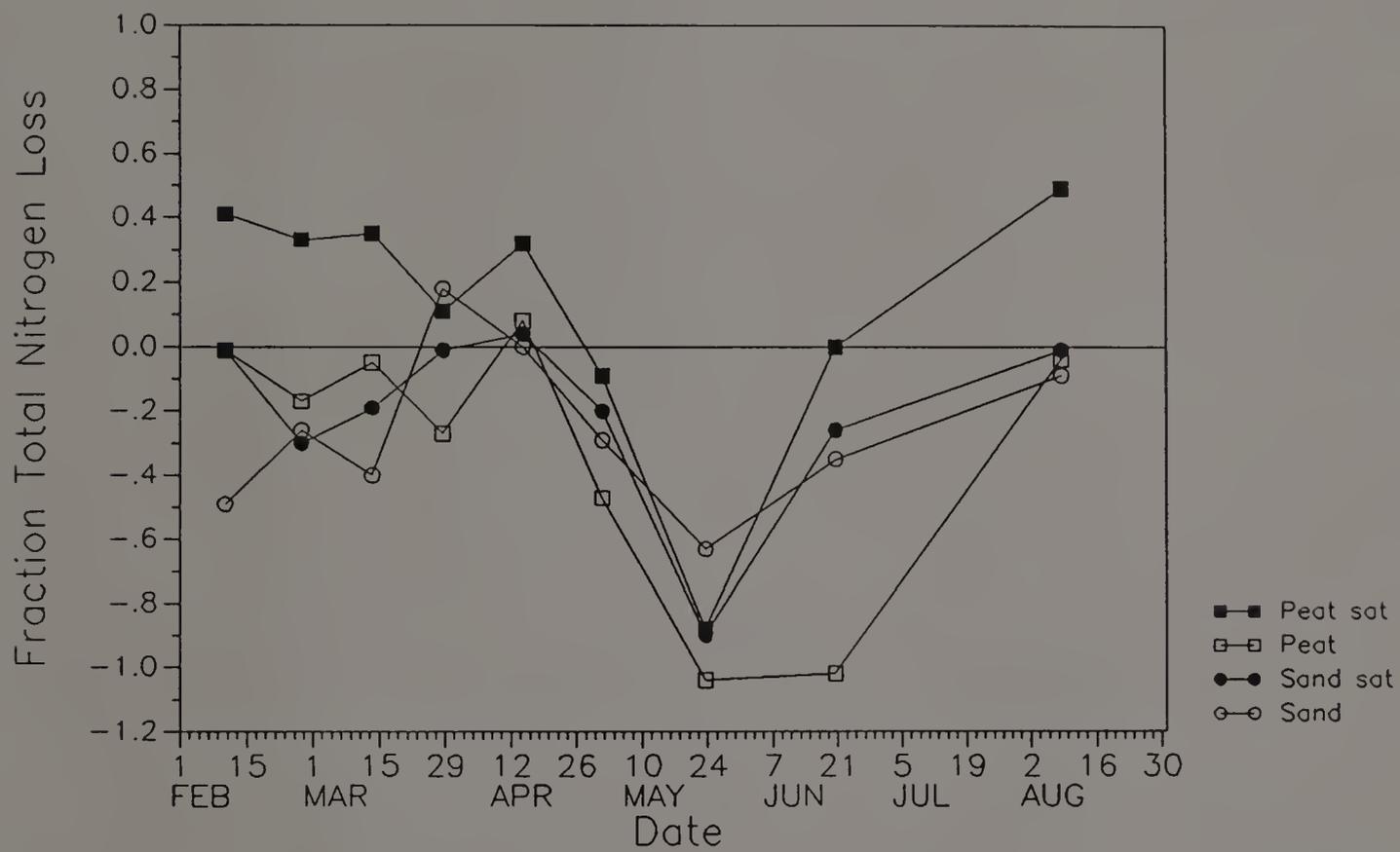


Figure 18. Net nitrogen loss in the various treatment reactors over time.

differed up to 4.2 times their average $\text{NO}_x\text{-N}$ 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 includes N_2 fixation by cyanobacter or other phototrophic organisms.

Acidity

Acidity 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 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 peat treatment effluents were statistically significant at the 0.05 level.

Repeated measures ANOVA on the treatment reactor effluents, alone, using Equation 9 from the previous chapter was 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 19 shows a decreasing trend in pH for all treatments. The observed 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.

Table 17. Mean values of pH 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.

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

TREATMENT		Sand		Peat	
		w/out water table	with water table	w/out water table	with water Table
Sand	w/out water table				
	with water table	0.0			
Peat	w/out water table	1.7 *	1.7 *		
	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
<u>Between Effects</u>					
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		
<u>Within Effects</u>					
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 by Water Status	0.71	16	0.04	1.19	0.28
Error	4.76	128	0.04		

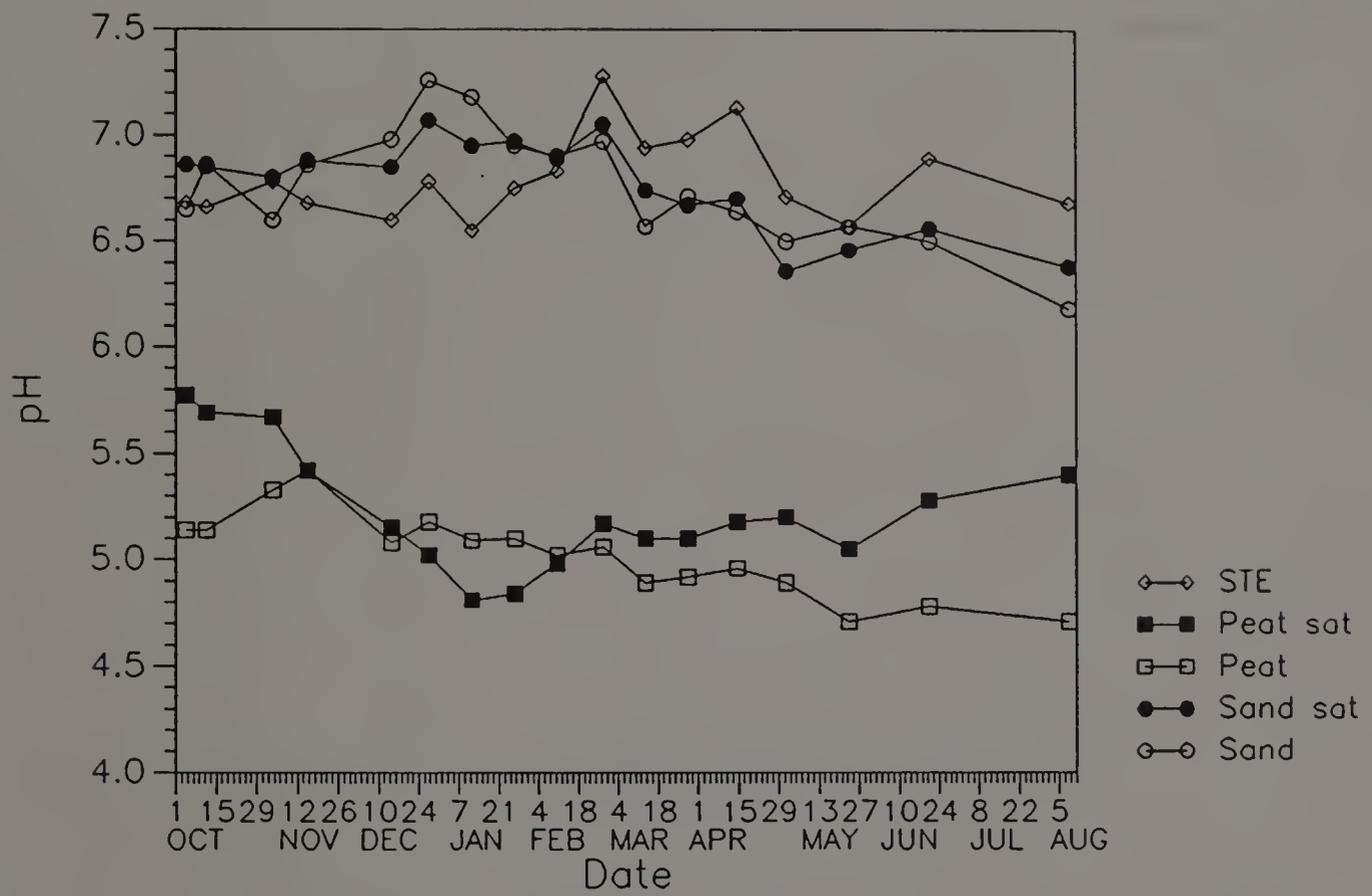


Figure 19. Mean pH values for septic tank and treatment reactor effluent over time.

Acetylene Blocking

Acetylene blocking technique was used on peat taken from two reactors from the peat treatment with a water table. The presence of denitrification 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 determine ΔN_2O-N . Net increase in N_2O-N concentration ranged from 2 to 70 times.

CHAPTER V

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, the conventional systems produce $\text{NO}_x\text{-N}$ in quantities exceeding federal drinking water standards (10 mg L^{-1}), because they lack reducing conditions required for reduction of $\text{NO}_x\text{-N}$. This may result in pollution to groundwater 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 are capable of both oxidizing and reducing conditions within 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 Nitrate/Nitrite-N from the sand filters and the unsaturated peat reactors exceeded the total input of N over the course of the 8-month study. In addition, both peat reactors produced high COD in their effluents. Most notable is the reduction in total N observed in the 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, a net reduction of up to 40% was measured.

Whereas several factors may be responsible for this reduction in total-N, most probable is the denitrification process. This process is favored by anaerobic conditions, and the presence of both $\text{NO}_x\text{-N}$ and organic C. Ion sorption also may account for some of the nitrogen reduction. The low nitrogen removal rates may be explained by limited bioavailable carbon, limited $\text{NO}_x\text{-N}$, short retention times, and 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. Peat reactors packed at a density of 0.12 g cm^{-3} and 0.10 g cm^{-3} can accept STE at a rate of 3 cm day^{-1} ($0.74 \text{ ft}^2 \text{ gal}^{-1} \text{ day}^{-1}$) for 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} ($0.74 \text{ ft}^2 \text{ gal}^{-1} \text{ day}^{-1}$) for one year without hydraulic failure.

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.

Saturated hydraulic conductivity in peat used in the treatment reactors was measured over 12 weeks. Mean K_{sat} was 9.2, 6.4, 5.5, and 4.2×10^{-4} cm sec⁻¹ for peat packed at 0.125, 0.130, 0.135, and 0.140 g cm⁻³ respectively. No significant difference in final K_{sat} was 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 BOD₅, may be in excess of federal drinking water standards.

Net reduction from STE NH₃-N was 89% and 86% for the sand treatments without a water table and with a water table respectively; 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 NH₃-N was statistically significant at the 0.05 level. Incomplete nitrification may have resulted from limited O₂ in the reactors or excessive loading rates. Throughout the sampling period, NH₃-N from all treatments was detected.

Net reduction from STE TKN was 91% and 86% for the sand treatments without a water table and with a water table respectively; 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 been the result of pretreatment oxidation 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 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 $\text{NO}_x\text{-N}$.

Denitrification measured in samples of peat taken from two reactors with a water table, was observed by the evolution of N_2O after incubation under an acetylene environment.

In future studies, the following modifications to the peat filtration reactors may be appropriate:

- increase the $\text{NO}_x\text{-N}$ fraction in the peat reactor influent by loading STE to sand filters prior to loading to peat filters,
- increase depth of saturated zone in 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.

In addition, further research should quantitatively measure the chemical, physical and biological processes responsible for nitrogen reduction in these peat filters. Additional experiments, used to measure denitrification should be conducted. These include: acetylene blocking with N_2O detection, N_2 detection using ^{15}N or by voiding the chamber of N_2 with a noble gas mixture followed by gas chromatography using thermal conductivity detection for N_2 in head space gas.

APPENDIX A

EFFLUENT QUALITY DATA

Chemical Oxygen Demand Mean values of COD (mg L⁻¹) for septic tank effluent and treatment reactor effluent.

Date:

	1/26	2/10	2/26	3/13	3/28	4/14	5/1	5/23	6/20
STE	195.4	247.7	231.5	223.9	228.2	203.7	248.7	382.8	193.1
I	16.0	28.2	15.4	5.1	22.0	4.6	44.4	14.0	13.2
II	27.1	26.2	19.6	10.3	43.0	0.8	22.6	23.9	9.8
III	67.8	50.6	40.3	34.8	41.0	40.1	82.1	58.8	86.1
IV	75.2	58.4	112.5	55.3	28.0	31.6	98.0	72.9	101.1

STE = Septic Tank Effluent, I = Sand free draining, II = Sand with water table, III = Peat free draining, IV = Peat with a water table.

Ammonia-N Mean values of NH₃-N (mg L⁻¹) for septic tank effluent and treatment reactor effluent.

Date:

	2/10	2/26	3/13	3/28	4/14	5/1	5/23	6/20	8/8
STE	21.5	18.2	21.3	21.4	25.1	35.0	21.9	18.9	28.4
I	12.6	3.9	1.4	0.2	0.4	0.4	0.4	1.1	1.5
II	11.5	7.0	3.6	1.6	1.1	1.0	0.4	1.2	1.6
III	11.6	11.0	12.2	12.0	10.8	13.7	14.1	10.6	5.4
IV	11.0	7.8	9.1	10.7	9.7	16.6	19.7	8.1	5.8

STE = Septic Tank Effluent, I = Sand free draining, II = Sand with water table, III = Peat free draining, IV = Peat with a water table.

Total Kjeldahl Nitrogen

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

Date:

	2/10	2/26	3/13	3/28	4/14	5/1	5/23	6/20
STE	21.8	18.7	22.2	22.4	24.8	29.5	22.1	22.6
I	9.5	3.3	1.5	0.6	0.1	0.3	0.3	2.0
II	11.4	6.9	3.6	1.8	0.5	0.2	0.2	2.8
III	8.4	11.2	10.9	11.3	10.0	12.4	13.5	8.9
IV	9.3	8.8	9.3	9.4	10.4	11.6	13.2	11.8

STE = Septic Tank Effluent, I = Sand free draining, II = Sand with water table, III = Peat free draining, IV = Peat with a water table.

Nitrate/Nitrite-N

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

Date:

	2/10	2/26	3/13	3/28	4/14	5/1	5/23	6/20	8/8
STE	2.5	0.6	0.0	0.3	0.0	0.4	1.0	0.2	2.0
I	23.3	19.8	28.3	17.7	24.6	32.5	36.9	24.8	31.6
II	12.7	17.4	21.7	20.4	22.9	29.4	43.1	22.9	29.1
III	12.7	11.1	10.1	15.6	12.2	23.5	32.3	27.9	26.2
IV	3.3	4.9	4.8	8.6	7.3	11.1	23.3	11.1	9.7

STE = Septic Tank Effluent, I = Sand free draining, II = Sand with water table, III = Peat free draining, IV = Peat with a water table.

Acidity Mean values of pH for septic tank effluent and treatment reactor effluent.

Date:	1989						1990			
	10/4	10/11	11/3	11/15	12/14	12/27	1/11	1/26	2/10	2/26
STE	6.7	6.7	6.8	6.7	6.6	6.8	6.6	6.8	6.8	7.3
I	6.7	6.9	6.6	6.9	7.0	7.3	7.2	7.0	6.9	7.0
II	6.9	6.9	6.8	6.9	6.9	7.1	7.0	7.0	6.9	7.1
III	5.1	5.1	5.3	5.4	5.1	5.2	5.1	5.1	5.0	5.1
IV	5.8	5.7	5.7	5.4	5.2	5.0	4.8	4.8	5.0	5.2

Date:	1990						
	3/13	3/28	4/14	5/1	5/23	6/20	8/8
STE	7.0	7.0	7.1	6.7	6.6	6.9	6.7
I	6.6	6.7	6.6	6.5	6.6	6.5	6.2
II	6.7	6.7	6.7	6.4	6.5	6.6	6.4
III	4.9	4.9	5.0	4.9	4.7	4.8	4.7
IV	5.1	5.1	5.2	5.2	5.1	5.3	5.4

STE = Septic Tank Effluent, I = Sand free draining, II = Sand with water table, III = Peat free draining, IV = Peat with a water table.

Bromide Study Raw Data - Bromide Concentrations in septic tank effluent and selected reactors.

DAYS	SEPTIC TANK EFFLUENT	REACTOR #			
		1 SAND w/out water	2 SAND w/water	7 PEAT w/out water	8 PEAT w/ water
(Br ⁻ in mg L ⁻¹)					
0.0	29.0	-	-	-	-
0.9	45.5	0.8	1.0	0.4	0.4
1.4	41.9	0.7	1.0	0.4	0.4
1.9	43.0	0.8	0.9	0.4	0.3
2.4	35.1	1.0	0.8	2.8	0.3
2.9	53.0	1.2	1.0	5.3	0.3
3.4	48.8	1.5	1.2	8.8	0.6
3.9	43.0	1.7	1.1	10.6	1.8
4.5	39.6	2.0	1.7	13.5	5.6
4.9	27.3	2.3	2.3	15.2	7.0
5.6	32.6	2.9	3.3	23.6	11.1
5.9	37.3	5.5	6.2	19.2	18.5
6.4	45.9	7.1	7.4	22.7	20.1
6.9	50.3	8.4	8.0	22.7	19.2
7.5	61.9	9.9	8.7	24.7	19.2
10.9	25.6	11.8	11.2	25.8	25.8
12.4	34.9	17.7	17.0	27.5	26.4
14.4	19.5	14.0	14.0	28.9	30.6
16.5	27.3	15.7	15.7	28.9	28.9
17.7	34.5	17.7	17.8	22.0	20.9
18.9	25.7	20.7	20.7	24.3	25.6
20.4	22.8	20.7	21.8	25.6	25.6
22.4	25.7	20.7	23.0	25.6	27.0
24.6	22.8	21.8	20.7	25.6	27.0
26.4	22.8	20.7	20.7	23.0	25.6
28.5	22.7	21.8	23.6	22.7	24.5
31.4	16.4	16.4	18.1	-	-

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