

**UNIVERSITY OF ALBERTA**

**SOIL TREATMENT PROCESS FOR PRE-TREATED LIQUID SWINE MANURE**

By

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of the requirements of the degree of Master of Science in  
Environmental Engineering

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*I dedicate this thesis to my wife Li Chang, for providing constant support,  
encouragement, perspective, and tenacity.*

## ABSTRACT

This thesis presents a study on the treatment of high-strength liquid swine manure using a soil filter system, which was a part of the swine manure treatment research carried out at the University of Alberta. The manure used in this research was pre-treated through a physical/chemical treatment process and stored in an in-ground tank for half a year. Preliminary experiment of the soil systems took place outside in three manure application rates of 12, 25, and 50 mm d<sup>-1</sup> every day to find an optimal application rate. Then 17 mm d<sup>-1</sup> manure application rate was chosen with the application frequency of twice a week inside which was operated inside. Detailed analyses were carried out for (1) the carbonaceous contents and nutrients in the manure and leachate, including five day biochemical oxygen demand (BOD<sub>5</sub>), chemical oxygen demand (COD), total Kjeldahl nitrogen (TKN), ammonia nitrogen (NH<sub>4</sub><sup>+</sup>-N), nitrate plus nitrite nitrogen (NO<sub>3</sub><sup>-</sup>-N+NO<sub>2</sub><sup>-</sup>-N), total phosphorus (TP), and total dissolved solid (TDS); (2) the variation and movement of key nutrients in the soil profiles, including TKN, NH<sub>4</sub><sup>+</sup>-N, NO<sub>3</sub><sup>-</sup>-N+NO<sub>2</sub><sup>-</sup>-N, and TP, as well as other parameters such as electrical conductivity (EC), pH, and sodium adsorption ratio (SAR). The results showed that the soil systems have a good ability for the removal of organic matter and inorganic nutrients. Within the two month operation period, the average reductions in BOD<sub>5</sub>, TKN, and TP were 94, 97, and 92%, respectively. A certain amount of nitrogen loss was unaccounted for in the soil system, suggesting that some gaseous states of nitrogen were generated during and after the manure application periods. The leachate quality became worse over the course of the application periods, but was suitable for bermudagrass development.

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# 1 INTRODUCTION

## 1.1 Overview

The output of Canadian hogs (domestic slaughter plus live exports) set a record of 25.6 million head in 2004, representing a 3.4% increase from the previous year. Canada's hog slaughter rose about 2%, led by a gain of 3.1% in the West and 1.7% in the East (CMC, 2005). Similarly, Alberta's pork industry has maintained steady growth over the last 25 years. Between 1976 and 2001, the market hog production more than tripled, increasing from 1.1 million head to over 3.6 million head, and the hog inventory doubled, increasing from 0.79 million head to 1.82 million head (Agriculture, Food and Rural Department (AAFRD) 2003). Accordingly, the increase in the size of the industry results in an increase in swine manure output. On average, a pig produces two tons of liquid manure each year (Larson, 1991). Due to its rich nutrient component, manure can be utilized for crop growth and for the enhancement of soil properties (Sutton *et al.* 1984), which benefits both pork producers and agriculture. For example, the application of liquid swine manure (LSM) to forage lands can increase forage production, extend the length of the grazing season, and improve forage quality (Bittman *et al.*, 1999). Therefore, as a practical method, the application of manure is permitted to on arable land, and cultivated or non-cultivated land in Canada (Alberta government, 2004).

Land application of LSM is usually carried out under beneficial management practices (BMPs) in order to best utilize the nutrients in the manure. However, as the pig industry expands and becomes more intensive, BMPs are becoming more and more difficult. This is particularly true when pigs are produced in large numbers under confined conditions such as in intensive hog barns. One reason behind the increasing

difficulties is that manure is perceived to be a potential source of water and air pollution, contributing to environmental and health issues such as the spread of pathogens, leaking of nitrate, over-fertilization of soil, and pollution or eutrophication of water bodies, as well as the emission of malodor and greenhouse gases such as N<sub>2</sub>O. Therefore, health and environmental concerns over manure application are growing. The other reason is that intensive hog production systems have resulted in large volumes of manure being produced within limited geographical areas with insufficient land area to utilize the manure efficiently. In response to these growing concerns, hog producers have to adopt some effective practices and technologies that not only utilize the nutrients in manure, but are also aimed at reducing environmental risks.

## **1.2 Current condition of the issue**

Many efforts have been made in the past 20 years, and are being made at present, to improve manure management skills or to develop effective treatment processes for manure disposal. These technologies include two-stage anaerobic digestion (Metcalf and Eddy, 2003), sequencing batch reactors (Tilche *et al.*, 1999; Edgerton *et al.*, 2000; Ra *et al.* 2000), biofiltration (Buelna *et al.*, 1993; Hill *et al.*, 2002), and pellet technology (Vanotti and Hunt, 2000) etc. However, most of these technologies only focus on either the effects of pollutant removal, or nutrient utilization efficiency. One problem for pig producers that has been ignored is that these technical processes are not easy to carry out on a farm. A popular method used on farms is the lagoon, due to its ease of management. However, like the fresh swine manure, the lagoon effluent, generally cannot be used up through land application under confined feeding operation (CFO) conditions. It must be diluted before land application due to its high concentrations of salts, nitrogen and

phosphorus, or be normally applied only two or three times a year, which results in large amounts of excess manure, especially in intensive hog barn. This means that the problem of excessive manure cannot be solved on site. Simple pretreatment of liquid manure will be very helpful for utilizing larger amounts of manure on limited areas, despite the fact that the manure will still have a high concentration of nutrients, as well as other pollutants such as biochemical oxygen demand (BOD). Therefore, finding an easier and more effective way to treat or utilize the high intensity, and large quantities of LSM appears to be very important.

This project involves the soil column approach, which is an imitation of the nutrient and organic processing area (NOPA) system, to treat LSM and utilize the nutrients in it, because, as the most common material on any farm, natural soils have been shown to be an effective medium for removing pollutants and depositing excess nutrients. The leachate from the columns was also analyzed and applied to bermudagrass development.

### **1.3 Goals and objectives of the research project**

This research constituted the last section of a larger swine manure project in Environmental Engineering at the University of Alberta and was carried out to achieve the following objectives:

- Testing the soil's ability to treat liquid swine manure containing high concentrations of organic substrate and nutrients.
- Investigating the nutrient distribution and conversion in manured soil after manure application and after a period of rest or recovery from manure application.
- Estimating the potential of the leachate for direct crop irrigation.

## 2 LITERATURE REVIEW

### 2.1 Manure characteristics and its management practice

#### 2.1.1 Quantities and characteristics of swine manure

Livestock manure includes both the feces and urine of the animals involved. According to EPA (2001), 31 to 38 kg (70 to 80 lbs) of manure is produced daily per 450 kg (1000 lbs) swine weight. Barker *et al.* (1998) showed that the mean manure production for a finishing pig was 5.05 kg per day. However, the liquid manure produced by swine facilities may vary significantly. Swine facilities were estimated by ASAE to produce as much as 8.4% of the total swine body weight in manure daily. Table 2-1 summarizes the yield of swine manure during different stages of growth (Miner *et al.*, 2000).

**Table 2-1 Manure quantities from different stages of growth <sup>a</sup>**

Swine	Animal size (kg)	Manure produced (kg/day)	Moisture (%)
Nursery pig	16	1.0	91
Growing pig	30	1.9	91
Finishing pig	68	4.4	91
Gestating sow	125	4.0	75

<sup>a</sup> Adapted from Miner *et al.*, 2000

The characteristics of manure produced in animal operations depend primarily on the characteristics of the feed provided to the animals (Loehr, 1977). Housing and animal management are other factors that influence the quantity and quality of the manure produced on farms (EPA, 2003). Swine manure can be over 90% liquid, depending on the quantity of water spilled during drinking. Manure containing less than 4% solids is termed liquid manure and that containing 4 to 10 % solids is termed slurry (MWPS,

2004). ASAE (2003) published some data which provides further details on manure properties; these are shown in Table 2-2.

**Table 2-2 Fresh manure characteristics per 1000 kg live animal per day <sup>a</sup>**

<b>Parameter</b>	<b>Amount</b>	<b>Unit (wet basis)</b>
Total manure	84	kg
Urine	39	kg
Density	990	kg/m <sup>3</sup>
Total solids	11	kg
Volatile solids	8.5	kg
BOD <sub>5</sub>	3.1	kg
COD	8.4	kg
pH	7.5	not applicable
TKN	0.52	kg
Ammonia nitrogen	0.29	kg
TP	0.18	kg
Orthophosphorus	0.12	kg
Calcium	0.33	kg
Magnesium	0.070	kg
Sodium	0.067	kg

<sup>a</sup> Adapted from ASAE, 2003

### **2.1.2 Animal feeding operations (AFOs) and swine manure management**

USDA and USEPA (1998) explained that AFOs were agricultural enterprises where animals are kept and raised in confined situations. AFOs congregate animals, feed, manure and urine, dead animals, and production operations within a small land area. The main issue concerning AFOs in this project is swine manure management. Currently, the swine manure has two outlets — individual livestock farm utilization, or off farm utilization (turning manure into a marketable product), including composting and manure palletize (Richard, 1991). Favorable economics of the handling and application of manure are very important to encourage manure treatment and application on farms. In fact, manure disposal had a negative economic impact in the past because the disposal costs often exceeded the nutrient value (McKenna and Clark, 1970). Huijsmans *et al.*



(2004) showed that the costs of manure application by broadcast spreading were lower than those incurred by trailing hose, trailing foot, or shallow injector on arable land (about 2 € m<sup>-3</sup>). Freeze and Sommerfeldt (1985) indicated that manure use might be economical for crop production when the transport distance is less than 18 km.

In regards to management method or technology, the most widely used strategy is solid removal followed by land application. Land application systems are well developed, but always need continued work to improve efficiency and effectiveness. In terms of pathogen removal, primary treatment by anaerobic lagoon was thought to be the current best management practice (BMP) for swine manure in the USA (Hill and Sobsey, 1998). The authors also pointed out that, of the secondary treatment processes, constructed wetlands achieved the best indicator microbe reductions, ranging from 1.1-2.5log. Covered lagoons, like anaerobic digesters, can significantly reduce odors and the release of unwanted gases. Anaerobic digesters can also be used to significantly reduce carbonaceous pollution (Ra *et al.*, 1997). Naturally aerobic lagoons, as another technique, can reduce nitrogen in liquid, but are impractical because of large size requirements. In addition, the wetland treatment of manure liquids has received some research attention because it has been shown to offer some nutrient reduction advantages when designed properly. But, the feasibility of constructed wetlands depends on waste characteristics and climate (Cronk, 1996); therefore, continued research is required to adapt wetland systems to different types of livestock operations. Chemical amendment is yet another management method, but some aspects of this practice remain questionable. Both feed additives and manure additives have been tested by a number of researchers and have

achieved, at best, only moderate success. Much research is needed before chemical additives will be major contributors to manure control solutions.

### **2.1.3 Benefits of manure management**

The biggest benefit of manure management and land application is that it increases crop yield. Generally, increasing the LSM application rate increases crop yield (Xie and MacKenzie, 1986), although the magnitude of these increases are dependent on the actual application rates, soil types, manure characteristics, and crop growing conditions. Mooleki (2004) found that the annual application of cattle manure resulted in a linear increase in grain yield corresponding to the increase in the application rates, but the manure had no effect on the nitrogen concentration in the grain. Liquid pig manure can effectively increase ryegrass biomass by 164% during a 20 week growth period, with a nitrogen recovery of 5.3% in the ryegrass (Yang, 2004). It can also increase alfalfa dry matter (DM) yield 37% on Xerofluent soil (Lloveras, 2004). The effects of different manure for crops were analyzed by Salazar *et al.* (2005) and the conclusion was that agronomic management was more important than manure type in influencing nitrogen losses, where soil cultivation appeared to be a key factor when comparing maize and grass systems.

Chase *et al.* (1991) evaluated the effects of various liquid manure application rates in Iowa, and found greater economic returns from land treated with manure applied at rates comparable to those used for commercial fertilizer. They concluded that, with increasing fertilizer costs, the profitability of LSM as a nutrient source would increase. Improved grass production, as mentioned above, can reduce the amount of imported feed required, thus providing additional economic and environmental benefits (Sullivan *et al.* 2000).

## **2.2 Manure treatment technologies**

Many methods have been used for swine manure treatment. The manure disposal processes can be divided into four categories: physical/chemical treatment process, biological treatment process, soil treatment process, and land application. The soil treatment process is actually a combination of the two former processes.

### **2.2.1 Physical/chemical treatment processes**

In agricultural waste treatment, the physical/chemical processes that may possibly be applied would include chemical precipitation, incineration, sedimentation, and flotation (Loher, 1974).

Solid/liquid separation is a widely used method for manure disposal. Solid/liquid separation could assist in nutrient management (Zhang and Westerman, 1997). The reason for separating coarse solids from flushed manure is mainly to prevent damage to the distribution nozzles during irrigation and to reduce organic loadings, thereby extending the service life of the manure holding area (Powers *et al.*, 1995). Sedimentation, screening, centrifugation, and filtration are the most common methods for solid/liquid separation and are used as primary separation unit operations. Jett *et al.* (1975) tested the removal of solids from swine liquid manure using a settling basin and found that solids removal as a function of time indicated that 1% total solids (TS) in liquid swine manure would give the highest solids removal efficiency. Zhu (2003) reported 60 to 70% suspended solids removal from liquid swine manure containing 5 to 6% total solids after 24 hour preliminary sedimentation. Screening separators included a stationary screen, a vibrating screen, and rotating screen separators. Piccinni and Cortellini (1987) found that vibrating and rotating screens performed better than

stationary screens for pig slurries containing 1 to 7% total solids. Hegg *et al.* (1981) also indicated that the vibrating screen separator accomplished a significant removal of dry matter from dairy and beef slurries. Holmberg *et al.* (1983) reported TS recovery in the range of 11 to 23% through a vibrating wet sieve shaker and an eight mesh screen. He also mentioned total kjeldahl nitrogen removal in the range of 7.9 to 8.9% and phosphorus removal of 3 to 23%. However, the animal slurries separated by the vibrating screen separator contained most of the inorganic material that was proved to be inhibitory to anaerobic digestion; therefore, the vibrating screen separator-shaker was not desirable to use in conjunction with an anaerobic digester system to meet the influent requirements (Piccinni and Cortellini, 1987). In addition, the end product from these separators still contained high moisture content (85 – 95%) and needed further dewatering before composting or being put to any other end use (Zhang and Westerman, 1997). After testing the effects of a centrisieve, a decanter centrifuge, a vacuum filter, and a vibroscreen, Glerum *et al.* (1971) found that the centrisieve was the best in terms of the strength of results, capacity, and initial expense for separating the liquid and solid parts of pig slurries.

Several research efforts have focused on the use of chemical treatment processes to treat animal manure. However, since the effectiveness of the chemical processes is largely depend on the specific situation, close attention has to be paid in interpreting the results (Zhang and Lei, 1998). Hanna *et al.* (1985) investigated the effects of eight coagulants (aluminum sulfate, ferrous sulfate, calcium hydroxide, ferric chloride, magnesium chloride, chitosan, lignosulfate, and an organic polymer) on 1% total solids flushed swine manure on the bench and field scales. They found an 8 to 13% reduction in

volatile solids concentration, except in the cases of magnesium chloride and lignosulfate, which removed only 2% of volatile solids. However, with the help of polymers, ferric chloride was found, in another study, to reduce volatile solids 60 to 70% in swine and cattle manure (Sieves *et al.*, 1994). Gao *et al.* (1993) reported that the addition of polymers increased solids removal efficiency by 20%, but was ineffective in removing phosphates. In this study, lime and alum showed the best performance in phosphorus removal. But Miner *et al.* (1983) found that alum and/or polymer did not change the phosphorus and nitrogen content, although they reduced BOD and suspended solids. Powers *et al.* (1995) found that different combinations of chemicals could improve the treatment results for total solids, TKN, and TP. The highest removal efficiencies were achieved by using of a combination of CaO and  $\text{Fe}_2(\text{SO}_4)_3$ . Zhang and Lei (1998) indicated that the use of a metal salt ( $\text{FeCl}_3$ ) together with a polymer (cationic polyacrylamide) considerably enhanced the removal of phosphorus from manure and would potentially reduce the amount of polymer required. Other researchers reported that polymers could effectively removed solids and organic forms of phosphorus from swine liquid manure, but that they have proven ineffective in removing soluble phosphorus (Vanotti *et al.*, 2003; Szogi *et al.*, 2003). Bromley *et al.* (2002) reported a 95% removal of total suspended solids and total phosphorus, as well as a 40% removal of TKN via 24 hour settling and coagulation and flocculation with alum, followed by settling in a sludge blanket clarifier, and then filtration through glass bead media filters.

### **2.2.2 Biological treatment processes**

Biological treatment processes are methods of transforming biodegradable constituents into acceptable end products and removing nutrients such as nitrogen and

phosphorus. In general, biological treatment processes can be classified into five categories: aerobic processes, anoxic processes, anaerobic processes, combined aerobic, anoxic, and anaerobic processes, and pond processes (Metcalf and Eddy, 2003). The most widely used methods in manure treatment include lagoon, biofiltration, sequencing bench reactor (SBR), anaerobic digestion, and soil systems (soil filters, constructed wetlands, and land application).

### **2.2.2.1 Lagoons**

Pader (1986) indicated that lagoons designed for manure treatment could reduce organic content and nitrogen by more than 50%. In the study carried out by Hermanson *et al.* (1980), liquid dairy manure was flushed into and stored in a surface-aerated lagoon for up to six weeks before irrigation disposal. The mean total nitrogen (TN) recovery of 70% was attributed to good sludge removal. Volatile solids destruction was 32%.

Anaerobic lagoons are generally preferred to aerobic lagoons. Almost all livestock lagoons are anaerobic (Pfoest *et al.*, 2000) due to the tremendous area required for aerobic lagoons to treat livestock manure. Anaerobic lagoons have been used as an integral part of many swine production systems to provide practical treatment and storage of swine manure (Humenik *et al.*, 1980). A single-stage anaerobic lagoon treating dairy manure was studied by Safley and Westerman (1992) for approximately four years. Compared to influent in their study, chemical oxygen demand (COD), TS, volatile solids (VS), and volatile fatty acid (VFA) reductions exceeded 80%.

Lagoon profile studies were undertaken by Ginnivan (1983) in perspex columns which were either unaerated (A), or aerated to a depth of 8 (B), 20 (C) or 40 cm (D). The dissolved oxygen level in the aerated columns was maintained at approximately 70% of

saturation and all columns were loaded at the rate of 50–55 g BOD<sub>5</sub> m<sup>-3</sup> day<sup>-1</sup>. The results of the study indicated that all treatments removed at least 77 and 90% of the incoming TS and BOD<sub>5</sub>, respectively. The treatment efficiencies of the unaerated and shallowly aerated systems (8 and 20 cm) were similar and higher than that of treatment D. Nitrate-nitrogen levels in the aerated treatments were less than 23 mg L<sup>-1</sup>. The average percentage reductions of faecal *coliform* (FC) and faecal *streptococci* (FS), which were the least for treatment D, were greater than 99% for FC and greater than 96% for FS. The absolute numbers, however, remained high in all treatments.

Despite the advantages, lagoons would become more odorous when overloaded, due to sludge buildups, additional inputs, and cold weather (Ritter, 1989). Therefore, studies have been done in the past and are being carried out at present, to determine and reduce the odor problems associated with anaerobic lagoons (Chen *et al.*, 2003; Schmidt *et al.*, 1999; Smith and Watts, 1994).

#### **2.2.2.2 Anaerobic digestion**

Anaerobic digestion is a biodegradation process which converts organic matter (OM) in waste to a biogas such as methane. Anaerobic digestion includes four stages: hydrolysis, acidification, acetogenesis, and methanogenesis. Ra *et al.* (2000) indicated that anaerobic digestion could be used to reduce carbonaceous pollution and odors in swine manure. It may play an important role in the removal of nitrogen and phosphorus (Metcalf and Eddy, 2003). However, Bernet *et al.* (1996) indicated that, in order to achieve nitrogen removal, anaerobic digestion had to be combined with a denitrification process.

Anaerobic digestion efficiency depends on the availability of nutrients and their composition. After review, Gronauer and Nesor (2003) indicated that the pH should be between 6.8 and 7.2; the reductions in BOD and COD could reach 81% and 83%, respectively; and the DM in the manure could be reduced from 7.6 to 4.3.

### **2.2.2.3 Sequencing batch reactors**

In general, sequencing batch reactors (SBR) are fill-and-draw activated sludge treatment systems. The advantage of this technology is that there is no need for a return activated sludge system because settling and aeration occur in the same tank.

Fernandes *et al.* (1991) used a one stage SBR to treat screened liquid swine manure and the results showed that the removal efficiencies achieved for ammonia nitrogen, TKN, COD, and total suspended solids (TSS) were 99%, 93%, 97%, and 97%, respectively. The nitrification and denitrification effects were tested by Bortone *et al.* (1992). They found that the nitrogen and phosphorus removal efficiencies were 88 to 90%, and 95%, respectively. After comparing the intermittent aeration process (IAP) with the non-limited aeration process (NLAP) in a one stage SBR, Osada *et al.* (1991) indicated that both of the two methods could achieve high removal efficiencies for BOD and total organic carbon. They noted that the ratio of nitrogen to BOD influenced the treatment results of the two methods. When the ratio of nitrogen to BOD was 0.18, total nitrogen removal by the IAP was 96.9%, which was much higher than the 59% achieved by the NLAP. Total phosphorus removal through the IAP was 80%, as compared to 48% through the NLAP. In addition, pH adjustment could effectively increase the removal efficiency of BOD, nitrogen, and phosphorus.



Ra *et al.* (1998) employed a two-stage aerobic-anoxic SBR for manure treatment and 90% removal efficiencies for BOD<sub>5</sub>, COD, ammonia nitrogen, and TP were reached. Cheng *et al.* (2001) reported that his SBR system achieved removal efficiencies for BOD<sub>5</sub>, total organic carbon (TOC), and TSS of 97.5%, 93.4%, and 97.3%, respectively. They also indicated that the removal efficiencies of nitrate nitrogen and PO<sub>4</sub><sup>3-</sup> were greatly influenced by the carbon content of the wastewater. The low nitrate content in the effluent suggested that the second stage anoxic reactor might not be necessary if the system were to be operated under a high organic loading situation with adequate aeration.

The effects of the anaerobic two-stage SBR (ASBR) on manure treatment were also studied by Zhang *et al.* (2000). Two systems, a mesophilic (35°C)-mesophilic (35°C) system, and a thermophilic (55°C)-mesophilic (35°C) system, were evaluated at a system hydraulic retention time (HRT) of six days and at four volatile solid (VS) loading rates (1, 2, 3, 4 g L<sup>-1</sup> day<sup>-1</sup>). The thermophilic-mesophilic system was found to perform much better in treating dairy and swine manure, with 6 to 15% more VS removal than the mesophilic-mesophilic system. Both systems were effective in reducing the generation of odorous sulfur gases during storage. The thermophilic-mesophilic ASBR system was more advantageous than the mesophilic-mesophilic ASBR system for treating animal manure because of its better capability for destroying fecal bacteria in animal manure. However, the higher energy requirement for heating the reactors in the thermophilic system needs to be considered.

An up-flow anaerobic sludge blank process was reported to be suitable for the pre-treatment of the liquid fraction of various types of manure. The maximum organic loading rate was approximately 12 g COD L<sup>-1</sup> day<sup>-1</sup> for hen or pig manure and 6 for cattle

manure with 1 day hydraulic retention time. The total COD reduction was about 75% for hen or pig manure and 42% for cattle manure. Further treatment was suggested to satisfy the discharge standard (Kalyuzhnyi *et al.*, 1999).

#### **2.2.2.4 Biofilters**

A biofilter is a fixed film biological reactor where the microbial growth takes place on a fixed media, such as stone or plastic. Normally, the liquid is fed through the top of the biofilter where the distributor evenly distributes the wastewater over the media within the biofilter. As the liquid percolates down over the media, the biological population growing on the media surface absorbs the soluble and particulate biodegradable organic material and converts it into either cellular matter or respiration products (CO<sub>2</sub> and H<sub>2</sub>O). The cellular matter periodically sheds or sloughs from the media and passes out of the biofilters.

Using graded-sand and a granular-activated-carbon packed filter, Ng and Chin (1988) investigated the effects on swine wastewater treatment. To compare performance, the system was operated at eight different hydraulic retention times. The results showed that the removal efficiencies for COD and VSS ranged from 27 to 90%, and from 15 to 93%, respectively. The carbon filter was reported to be marginally better than the sand filter in terms of COD and VSS removal.

Hill *et al.* (2002) evaluated the reduction of enteric microbial indicators in swine wastewater through an aerobic up-flow fixed media biofilter system equipped with a low pressure ultraviolet collimated beam apparatus. The results demonstrated that the aerobic biofilter could be an effective method for flushed swine waste treatment in terms of enteric microbe reduction. Compared to that obtained from the lagoon, the effluent of this

system had lower quantities of fecal *coliform* and other enteric microbes, as well as a much shorter hydraulic retention time. Another advantage of the aerobic up-flow biofilter was mentioned by Williams (2001), who indicated that the system significantly reduced the emission of odour associated with the flushed swine manure.

Westerman *et al.* (2000) developed a pilot plant with two up-flow aerated biofilters connected in series to treat up to  $8 \text{ m}^3 \text{ day}^{-1}$  of supernatant from settled flushed swine wastes. When operated under warm weather conditions (average temperature of  $27^\circ\text{C}$ ), the system removed about 88% of BOD, 75% of COD, and 82% of TSS with a loading rate of  $5.7 \text{ kg COD m}^{-3} \text{ day}^{-1}$  of biofilter media. The TKN, ammonia nitrogen, and TN reductions averaged 84%, 94%, and 61%, respectively, with a significant portion of the ammonia nitrogen being converted to nitrite plus nitrate nitrogen. When operated at lower temperatures (average of  $10^\circ\text{C}$ ) from December through March, COD, TKN, ammonia nitrogen, and TN removals averaged 56%, 49%, 52%, and 29%, respectively. The results of the mass balance average for the 12 months indicated that about 30% of the influent volume, 35% of TN, and 60% of total phosphorus are removed with the biofilter backwash, suggesting that management and utilization of the backwash are important factors in implementing this type of system on farms. The lost nitrogen (about 24%) was attributed to ammonia volatilization, or possibly denitrification within the biofilm.

### **2.2.3 Soil system**

#### **2.2.3.1 Overview of waste treatment through soil processes**

The land, or soil, is a gigantic bio-digestion system which can digest animal and plant waste such that it becomes part of the soil. It can also act as, and has already been used as, a natural reactor for the disposal of manmade waste. As pointed out by USEPA

(1982), land treatment implies that the land or soil was used as a medium to treat hazardous waste. A land treatment facility is defined as the portion of a given facility within which hazardous waste is applied onto, or incorporated into, the soil surface. Waste can be treated or utilized on either cultivated land or non-cultivated land. Waste disposal utilization programs may be grouped into many categories, such as agricultural, municipal, industrial, and institutional (Fuller and Warrick, 1985). For liquid waste, the basic processes include slow rate (SR), rapid infiltration (RI), and overland flow (OF) (Crites *et al.*, 2000), which normally reflect the rate of water movement and the flow path employed within the processes. Different waste applications, application methods, and application durations may result in different effluent qualities and different component variations within the soil profile, as well as different vegetation responses.

For example, the land application of biosolids caused an increase in both the size and activity of the soil microbial biomass related to the degree of stabilization of the composting mixture (Sanchez-Monedero *et al.*, 2004). However, the application of biosolids poses the potential risks of accumulation of heavy metals and organic substances (Bright, 2003). When biosolids were applied together with fly ash, the mixtures reduced metal leaching and were beneficial for biomass production, without contributing significantly to metal uptake or leaching (Sajwan *et al.* 2003).

Land irrigation using meat processing wastewater at a loading rate of 600 kg N yr<sup>-1</sup> leads to a considerable increase in plant production and low concentrations of nitrogen and phosphorus in leachate (Luo *et al.* 2004). The authors also indicated that the TN in the soil did not show any significant increase, whereas phosphorus accumulation was

observed in the surface soil. The loss of cations ( $\text{Na}^+$ ,  $\text{K}^+$ ,  $\text{Ca}^{2+}$  and  $\text{Mg}^{2+}$ ) to leaching was reported to have increased.

In a soil column study, in which dredged sediment from a polluted river was applied to sandy loam soil for two and a half months, including a 15-day un-watered period, Chen *et al.* (2003) indicated that the un-watered period could increase the downward movement of phosphorus.

A willow vegetation soil filter has recently been studied in Sweden for the treatment of wastewater (Aronsson and Bergstrom, 2001). Under nitrogen ion rates of 110-224 kg  $\text{ha}^{-1} \text{yr}^{-1}$  and an irrigation load of 6 mm  $\text{d}^{-1}$  during the growing season,  $\text{NO}_3\text{-N}$  leaching loads were shown to be very high the first year after plant establishment (on average 341 kg N  $\text{ha}^{-1}$  from clay and 140 kg N  $\text{ha}^{-1}$  from sand lysimeters). However,  $\text{NO}_3\text{-N}$  leaching loads were found to have decreased to the point of being low or negligible during the second (43 kg N  $\text{ha}^{-1}$  from clay and 17 kg N  $\text{ha}^{-1}$  from sand lysimeters) and third years (3 kg N  $\text{ha}^{-1}$  from clay and less than 1 kg N  $\text{ha}^{-1}$  from sand lysimeters). Plant harvest and irrigation rate in this research did not significantly affect  $\text{NO}_3\text{-N}$  leaching loads, whereas soil type and N application rate strongly influenced the leaching loads.

In addition to reducing liquid and solid waste, soil can also be used for air purification. Kikuchi (2000) found that a malodorous gas containing 600 mg  $\text{L}^{-1}$  ammonia could be removed by soil filter at an efficiency of up to 98.5%.

In summary, because of the natural characteristics of soil, it can be used to treat many different wastes, including animal waste. This makes it a very attractive option for waste treatment and necessitates further and more detailed research.

### 2.2.3.2 Soil filters

Very limited information regarding animal manure disposal through soil was found in this literature review. However, the effects of a soil matrix on manure disposal can be demonstrated by analyzing the quality of effluents from soil filters, lysimeters, and wetlands.

A soil treatment process termed barriered landscape wastewater renovation system (BLWRS) was developed in the USA. It consists of a mound of soil underlain by an impermeable barrier and drainage system, which establishes an aerobic zone (unsaturated) in the top of the BLWRS and an anaerobic zone (saturated) next to the impermeable barrier (Ritter and Eastburn, 1978). Using the BLWRS system, Zelechowska and Rybinski (1985) researched the process occurring in the unsaturated zone while purifying pig slurry. The medium filled in the filter was not dust or loam fractions, but rather sand only, and the average daily application rate varied from 3.9 mm to 5.6 mm, resulting in the following average loading rates: 0.00108 – 0.00362 kg m<sup>-2</sup> organic nitrogen; 0.00353 – 0.00661 kg m<sup>-2</sup> TKN; 0.00358 – 0.00666 kg m<sup>-2</sup> TN; and 0.01892 – 0.05443 kg m<sup>-2</sup> COD. The pig slurry applied to the filter was pre-treated through mechanical and coagulation methods, or a two hour settling period. The results showed that the purification effects for organic nitrogen, TKN, TN, and COD were 86.4-89.5%, 86.3-90.8%, 67.4-75.3%, and 94.0-95.5%, respectively. The authors also reported that the processes of ammonification, nitrification, and denitrification of pig slurry in the unsaturated zone followed the first order pseudo kinetics:  $C_m = C_0 e^{-k \cdot m}$ , where  $m$  was expressed by the filter's thickness instead of time.

Based on the idea of BLWRS, a system called “Solepur” was developed in France. The Solepur process is a soil filter system used to treat pig slurry. This process involves three operations: (1) a managed field covered with ryegrass to which the pig slurry is applied, (2) a storage-pump-reactor system for denitrification, and (3) a non-managed field for completing treatment. By supplying raw pig slurry to the first operation with a normal load of  $986 \text{ m}^3 \text{ ha}^{-1} \text{ yr}^{-1}$ , Martinez (1997) reported that the removal effects were 99.9% for COD, 99.9% for phosphorus, and approximately 90% for nitrogen. The leachate leaving the first operation contained nitrate concentrations as high as  $1500 \text{ mg L}^{-1}$  and was treated in the second operation using anaerobic digestion. After five years of application, the Solepur soil treatment system maintained its capacity to remove organic matter and nitrogen from pig slurry (Chadwick *et al.*, 1998). It was also found that this system appeared to promote the emission of a gaseous state of nitrogen.

Lam *et al.* (1993) assessed the feasibility of the soakaway soil filter system (3 m high) for treating piggery waste. They reported that the purification efficiency was higher in the volcanic soils than in the coarser granitic soils. Most of the contaminants were removed in the top 1 m zone of the columns. The authors suggested that choosing an appropriate loading rate was critical, and that when the correct loading rate was chosen, the purification efficiencies could reach more than 99% for BOD<sub>5</sub>, TP, TN, and NH<sub>4</sub>-N, and 95% for nitrate.

Similar systems have been developed for the disposal of animal waste. The reed bed system is one of these approaches to treating liquid wastes including manure, municipal wastewater, and landfill leachate. Garcia *et al.*, (2004) reported that hydraulic loading rate and water depth were determining factors in the performance of the system. Beds

with a water depth of 0.27 m removed more COD (70–80%), BOD<sub>5</sub> (70–85%), ammonia (40–50%), and dissolved phosphorus (10–22%) than beds with a depth of 0.5 m (60–65% for COD, 50–60% for BOD<sub>5</sub>, 25–30% for ammonia, and 2–10% for dissolved phosphorus). The manure treatment abilities of this system are not significantly affected by different arrangements of bed media (progressively-sized and anti-sized), but the anti-sized system demonstrated a clear advantage in its ability to slow down the clogging of bed media, thus avoiding impairment of long-term functioning and promoting the sustainability of the beds (Zhao *et al.*, 2004).

NH<sub>4</sub><sup>+</sup>-N in the manure was removed by a two-stage process: adsorption onto the reed bed media, followed by nitrification into nitrite-nitrogen and nitrate-nitrogen. The average removal rate in a three hour treatment was 44% when the initial NH<sub>4</sub><sup>+</sup>-N level was around 150 mg L<sup>-1</sup> (Connolly *et al.*, 2004).

Kowalik and Obarska-Pempkowiak (1985) indicated that soil air-filled porosity ( $n_g$ ) could be used to determine the amount of liquid manure to be applied, based on the maximum BOD loading. They suggested that the permissible amounts of organic pollutants, such as BOD<sub>5</sub>, in wastewater are 0.2 kg m<sup>-3</sup> for  $n_g = 0.10 \text{ m}^3 \text{ m}^{-3}$ , BOD<sub>5</sub> = 0.9 kg m<sup>-3</sup> for  $n_g = 0.20 \text{ m}^3 \text{ m}^{-3}$ , and BOD<sub>5</sub> = 2.75 kg m<sup>-3</sup> for  $n_g = 0.30 \text{ m}^3 \text{ m}^{-3}$ . However, their conclusion was based on some assumptions, such as a uniform distribution of air-filled porosity in the upper part of the soil, oxygen respiration of soil being similar to that of pig slurry, an aerobic zone present in the upper 50 cm of the soil profile, and irrigation every five days.

Clogging appears to be a problem in waste treatment through soil processes. The main cause of clogging is attributed to the removal of suspended solids (SS) inside the



bed matrix. Another reason is the increasing biomass (Zhao *et al.*, 2004). Koerner and Koerner (1992) investigated the permeabilities of soil filters, as well as geotextiles, by using six different types of leachate. In all cases, permeabilities decreased over time due to a combination of sediment clogging and/or biological clogging. The authors evaluated the remediation methods after approximately six months when a steady state permeability value appeared to have been reached and found that water backflush was the most effective method of reinstating high flow rates, followed by backflushing with leachate and nitrogen gas.

### **2.2.3.3 Constructed wetland**

As another soil treatment process, wetlands were considered to be one of the best management practices for treating animal wastewater from dairy and swine operations, but the pretreatment of wastewater was important (Cronk, 1996) and the feasibility of constructed wetlands varies with waste characteristics and climate. As Fraser *et al.* (2004) noted: vegetated microcosms are more effective than unvegetated microcosms for reducing concentrations of total nitrogen and total phosphorus; cooler temperature has a negative effect on nutrient removal; different species have different potentials for the reduction of nitrogen and phosphorus. They did not find any evidence to indicate plant mixtures were more effective than monocultures for nitrogen and phosphorus reduction.

Longer hydraulic retention times are usually needed in wetland processes. When Lee *et al.* (2004) employed a subsurface flow constructed wetland (SSFCW) to treat swine waste, the pretreated swine effluent was passed through the system at three hydraulic retention times (HRT): 8.5 days (Phase I), 4.3 days (Phase II), and 14.7 days (Phase III). The averaged reduction efficiencies in the three phases were: SS 96–99%, COD 77–84%,

TP 47–59%, and TN 10–24%. While physical mechanisms were dominant in removing pollutants, the contributions of microbial mechanisms increased with the duration of wetland use, achieving 48% COD removal and 16% TN removal in the last phase. The authors noticed that the plants (water hyacinth) made only a minimal contribution to the removal of nutrients, suggesting further land application treatment was necessary for nutrient assimilation. After analyzing over 1300 operational data records, Knight *et al.* (2000) reached a similar conclusion regarding the average concentration reduction efficiencies of: BOD<sub>5</sub> 65%, TSS 53%, NH<sub>4</sub>-N 48%, TN 42%, and TP 42%. Removals are a function of inlet concentrations and hydraulic loading rates.

However, better results were reported by Schaafsma *et al.* (2000) when they used a wetland to treat dairy farm wastewater. Relative to initial concentrations, TN was reduced 98%, ammonia 56%, TP 96%, ortho-phosphate 84%, SS 96%, and BOD 97%. The only exception was nitrate plus nitrite, which increased by 82%. Nitrification in the vegetated filter strip accounted for the increase in nitrate/nitrite. Like Lee *et al.* (2004), Schaafsma *et al.* (2000) also suggested further removal, for example, through the addition of another anaerobic wetland cell downstream from the system or through the recirculation of wastewater through the wetland cells to promote denitrification and the uptake of nutrients by plants.

Weather conditions influence the treatment results of the wetland system. Newman *et al.* (2000) found that mass retention in their experiment was significantly greater ( $P < 0.05$ ) during the summer than during the winter for all variables except FC, although the overall percentages of mass retention were 94, 85, 68, 60, and 53% for TSS, BOD<sub>5</sub>, TP, nitrate and nitrite nitrogen, and TKN, respectively. Moreover, denitrification rates had

shown denitrification to be a minor removal mechanism (<1%) for nitrogen in this wetland. Settling and increased storage were considered to be the largest removal mechanisms.

It was found to be easier to remove TN and nitrate and nitrite nitrogen from partially nitrified swine manure prior to wetland treatment. Moreover, partial nitrification reduced ammonia volatilization because it lowered ammonia nitrogen in the wastewater (Poach *et al.*, 2003). For wastewater containing high amounts of nitrate, three parameters, namely seasonal temperature fluctuations, hydrology, and nitrate loading, account for a significant percentage of the seasonal variability in nitrate retention (Speiles and Mithcs, 2000).

Tanner *et al.* (1998) researched organic matter (OM) accumulation in a gravel-bed constructed wetland by applying dairy wastewater with different hydraulic loading rates (21, 26, 46 and 72 mm d<sup>-1</sup>) over a five year period. The mean accumulations of OM in the wetlands ranged between 6.8 and 14.9 kg m<sup>-2</sup>, increasing with wastewater loading rate. The results showed that the annual rates of accumulation during the first two years were 1.2 to 2-fold higher than those in the subsequent three years. OM accumulation was associated with decreasing retention time, but not directly, which suggests that other factors, such as bulk density, contribute to OM accumulation.

On the whole, based on the present data, soil has a very good ability to remove carbonaceous pollutants. By itself, or combined with plants, it is also able to reduce nutrients. However, this ability is variable. Further treatment with denitrification processes was recommended because denitrification seems to be a minor removal

mechanism in present studies. Clogging usually appeared to be a typical problem in soil treatment systems.

#### **2.2.4 Land application**

In order to utilize the nutrients in animal manure, many management practices focus on land application. Compared to soil filter and wetland processes, which only focus on pollutant or nutrient removal, land application of manure might be the most economic method for manure management. The reason for this is that, during land application, not only are pollutants reduced, but the nutrients in manure are utilized as well. However, the extent of land application is restricted by environmental issues.

##### **2.2.4.1 Irrigation standards and regulations**

Despite the advantages of manure for crop growth, manure application was limited by legislation in some countries, due to environmental issues (Jongbloed and Lenis, 1998). In Alberta, spreading manure on arable land, cultivated, and non-cultivated land is permitted by Agriculture Operation Practice Act and Regulations (Alberta Government, 2004). However, the Code of Practice for Responsible Livestock Development and Manure Management (2000) also indicated that manure, when applied in appropriate locations at rates that should be in balance with crop uptake, poses a minimal risk to the environment. The code also states that high salinity applications must not result in an increase in the soil EC value of more than  $1 \text{ dS m}^{-1}$ . Moreover, the odor nuisance associated with the spreading of manure on land can be minimized through proper timing, siting, method of incorporation, and frequency of application. The nitrate-nitrogen limitation, for the irrigated medium and fine textured soils, is  $270 \text{ kg ha}^{-1}$  ( $240 \text{ lb ac}^{-1}$ ) when manure is applied on forage and direct seeded crops (Alberta Government, 2004).

#### 2.2.4.2 Application methods

Application methods are very important factors, influencing nutrient loss and the spread of pollutants from manure. Now, the methods of manure application include sub-surface injection, surface injection, broadcast with incorporation, broadcast, trailing shoes, plowing down, and sprinklers etc.

Ammonia volatilization is a major concern with different manure application methods. A dilemma here is that high ammonia volatilization is helpful to high intensity nitrogen removal in manure, but it is not benefit in terms of agronomy or air quality. Chadwick *et al.* (2001) reported that surface and injected application methods, respectively, resulted in average cumulative ammonia emissions of 15% and 11% of manure nitrogen added eight days after application. There was some evidence that the shallow injection of slurries resulted in the effective reduction of ammonia emissions, when compared to conventional surface application (Weslien *et al.*, 1998; Frost, 1994), and the injection of slurry into the soil showed an increased utilization of slurry nitrogen (Rubaek *et al.*, 1996). Hoff *et al.* (1981) demonstrated 0-2.5% ammonia nitrogen loss with the injection application of liquid swine manure, as compared to 10-16% from surface broadcast. Similar result was reported by Lorimor *et al.* (1997), who indicated a 5% loss of nitrogen when slurry was injected into the soil. The highest ammonia nitrogen loss from surface application was reported by Vanderholm (1975), whose value was 30-90%. He also reported 5 and 15 % losses of ammonia from applied manure with plowing down and disking. Wu *et al.* (2003) showed that the loss of ammonia nitrogen from droplet volatilization was only a few percentages, which was considered insignificant compared to common soil surface volatilization losses of 20 to 50% at the end of one week after application. However, a different result was reported by Safley *et al.* (1992)

and Sharpe and Harper (1997), who gave a range of from 13 to 37.7% for ammonia-N losses during sprinkler irrigation. In Lorimor's (1997) research, the value was as high as 40%.

The application method seems not to influence nitrate leaching very much. In the study performed by Thomsen (2005), farmyard manure (FYM) was either mixed into the soil (7–15 cm soil depth), giving an intimate contact with the soil, or was placed in a layer at a depth within the soil of 15 cm. Reference lysimeters with soil till, but without FYM application, were included. The results showed that the application technique had only a minor influence on nitrate leaching and then only after autumn application. The extra leaching losses due to FYM accumulated over three winter periods averaged 21% of nitrogen for FYM applied in September and 13% of nitrogen for FYM applied in December or March.

Shirani *et al.* (2002) think the effect of tillage systems on yield and soil physical properties was not significant, although moldboard plowing increased the depth of root penetration significantly. Minimum tillage and crop rotations improved the physical soil status and prevented soil crusting from developing, along with its negative effects on infiltration (Thierfelder *et al.*, 2005). However, the subsoil tillage to a depth of 30 cm had multiple effects, including overcoming a natural or tillage-induced dense layer and increasing volumetric soil water content and crop nitrogen uptake (Motavalli *et al.*, 2003). The results of this study suggested that deep tillage and applications of organic amendments are management tools that may overcome restrictions in both nitrogen and soil water availability due to subsoil compaction in sandy-textured soils. In conventional tillage and no tillage soils, the stability of soil aggregates against the dissolution and

dispersive actions of water decreased by 13 and 16%, respectively, following the application of fresh manure (Pare *et al.*, 1999).

#### **2.2.4.3 Application time**

From an agronomic perspective, the best time to apply manure is before the early stages of crop growth. The longer the time between manure application and the stage at which the crop can use the nutrients, the higher the risk of nutrient losses (AAFRD, 2002). Gangbazo *et al.* (1995) stated that hog manure application in the fall resulted in greater N loss and water contamination due to winter and spring runoff. In addition, manure application was not desirable during the fall and winter, in the absence of crop growth and when soils were frozen, because of the high risk of nitrate leaching (Choudhary *et al.* 1996).

The time for farmyard manure application is decisive for both nitrate leaching losses and for crop N utilization (Thomsen, 2005). Thomsen's lysimeter study estimated that 6–10% of the nitrogen applied in manure was leached in the first winter when manure had been applied in September. Nitrate leaching after application in December and in March was of the same magnitude as for unmanured lysimeters.

#### **2.2.4.4 Application rates**

From a waste removal perspective, application rate is usually determined by hydraulic loading and substrate loading. With respect to agronomy, generally, manure application rate is based on the crop demand and manure component. However, it is not easy to determine due to the variation of manure characteristics. From 1996 to 2000, Mooleki *et al.* (2002) investigated soil and crop responses to applications of swine manure at different rates (100, 200, and 400 kg TN ha<sup>-1</sup> yr<sup>-1</sup>), frequencies, and application methods

in the Black soil zone in east-central Saskatchewan. The crops used in the study were Argentine canola, hard red spring wheat, and hulless barley. Annual application at low to medium rates (100-200 kg N ha<sup>-1</sup>), in this study, were concluded to provide better utilization of nitrogen (25-45%) and pose less danger of environmental contamination than single or repeated high application rates. Kowalenko and Bittman (2000) found that the influence of the nitrogen application rate was greater than that of the timing of nitrogen availability during the growing season.

Excessive amounts of manure may result in the accumulation of nutrients and salts to levels that adversely affect crop growth. The problems were reported by Sawyer *et al.* (1991) in terms of pH, soluble salts, and other soluble ions such as phosphorus, potassium, and zinc. Over the course of his four-year study, Johnson (2004) found that increasing annual rates of nitrogen and phosphorus in dairy wastewater and poultry litter increased phosphorus removal in forage. At the highest nitrogen rate (1000 kg N ha<sup>-1</sup>) of each nutrient source, less than 13% of the applied phosphorus was recovered in forage. The highest nitrogen rate delivered eight times more phosphorus in wastewater, or 15 times more phosphorus in poultry litter, than was removed in forage harvests during an average year.

So far, nitrogen is still used as the determination of to determine nutrient application rate. However, the main problem at present seems to be phosphorous, as crops cannot remove all of it in only one season, whereas the issue of nitrogen can be solved by matching crop requirements (ECOMatters Inc, 2002). Moreover, due to the differences in nitrogen and phosphorus ratios in manure and in crops (4:1 in manure versus 8:1 in crops), manure application should be based on phosphorus application instead of a crop's



nitrogen requirements (Beegle, 2000). It is likely that the application of manure needs adjustment in order to give a better ratio of nitrogen and phosphorus, and to meet phosphorus requirements.

#### **2.2.4.5 Environmental concerns**

Soil properties may be characterized by many attributes. Wang *et al.* (2003) assessed the soil quality of land irrigated over a long period of time with reclaimed wastewater and indicated that the soil quality might be evaluated by comparing the total porosity (or drainable porosity), pH, electrical conductivity, magnesium, phosphorus, and zinc of soils in the control and the treated fields. The results demonstrated that, except for the total porosity and Mg, the other soil parameters were not significantly different between the control and treated fields. While the soils of both fields supported successful crop production, irrigation with reclaimed wastewater appeared to increase soil compaction slightly and reduce the soil's capacity for holding nutrient elements, such as magnesium.

In fact, there was very limited information available on the effect of swine manure on soil physical properties. However, the effects of swine manure may be similar to those reported for other manures. Recent research stated that long-term cattle manure application on natural fields increased the soil's water retention capacity and decreased differences in water retention between erosion levels. Soil bulk density was also decreased by 10%, whereas hydraulic conductivity was doubled in the top 7.6 cm of soil (Arriaga and Lowery, 2003). Shirani *et al.* (2002) reported that, although the hydraulic conductivity was improved, both on the row and inter-row positions, the decrease of soil bulk density was significant only on the row tracks. Prior research also shows that manure application can improve soil aggregation and improve the structure and water

holding capacity of soil due to increased organic matter (Weil and Kroontje, 1979; Unger and Stewart, 1974).

Although, the climate seems to contribute to changes in soil properties, a temperate climate was reported to cause no significant changes in soil physical properties due to manure application (Sommerfeldt *et al.*, 1986).

Because of the LSM application, the chemical composition of the soil was variable and highly influenced by factors such as soil texture, rate, time, and method of manure application, amount of precipitation, crops grown, and time of sampling (Choudhary, 1996). An assessment in 1970 in Alberta indicated that an animal manure application of 700 kg ha<sup>-1</sup> annually for 40 years did not cause an undesirable buildup of nitrogen, phosphorus, or soluble salts in the soil (Sommerfeldt *et al.* 1973). The overall conclusion from the feedlot study was that levels of nitrogen and phosphorus were increased in the soil, but that there was little evidence of extensive downward movement of nutrients, except in the case of one feedlot that was located in a slight depression. Another experiment was carried on for five years in eastern Canada. Cattle manure was applied at rates of up to 900 kg ha<sup>-1</sup> annually on sandy clay loam soil and about 250-300 kg ha<sup>-1</sup> of mineral nitrogen (nitrate, ammonium, and nitrite) was found in the 0-12 cm depth (Culley *et al.*, 1981).

However, different results were observed in this research field. Nitrate nitrogen (NO<sub>3</sub>-N), Mehlich I extractable phosphorus, and Na were found to have accumulated in subsoil from manure effluent application and the level of accumulation increased with increasing manure application rate (King *et al.* 1985). This conclusion was proven again by Liu *et al.* (1997) when they applied swine lagoon effluent at the rates of 560, 1120, and 2240 kg

$\text{N ha}^{-1} \text{ yr}^{-1}$  on loamy sand soils covered with bermudagrass and ryegrass. They observed that the application of swine lagoon effluent resulted in a buildup of  $\text{NO}_3\text{-N}$  in the lower depths of soil profiles (120 cm), especially for the high nitrogen loading rate, where concentrations reached approximately  $30 \text{ mg kg}^{-1}$ . The application of effluent also resulted in significant phosphorus buildup, but did not increase soil total carbon. Compared to inorganic fertilizer, a heavy application of manure increased  $\text{NO}_3\text{-N}$ , available phosphorus, and exchangeable potassium and sodium (Evans *et al.*, 1977).

Similar results were reported by Eghball *et al.* (2004), who indicated that the residual effects of manure application significantly increased soil  $\text{NO}_3\text{-N}$ , and plant-available phosphorus concentrations. But  $\text{NH}_4\text{-N}$  in soil was noticed to undergo no significant change through manure applications. Nitrification and plant uptake of slurry nitrogen were found to be significantly greater in the sandy loam than in the clay soil during the growing season, but the residual effect of slurry nitrogen on the nutrition of subsequent crops was similar for both soils (Chantigny, 2004).

Contrary to the results reported by Liu *et al.* (1997) for total carbon concentration, (Peacock *et al.*, 2001) reported a significant increase after five years of dairy manure application at a rate of  $504 \text{ kg N ha}^{-1} \text{ year}^{-1}$ . However, in his study the nitrogen concentrations in the soils were also reported as increasing, especially in the top soils. Normally, high levels of soil organic carbon concentration were considered to improve soil quality and consequently, crop productivity (Unger, 1997), but the concentrations, and their variations were reported to have decreased with depth (Cannavo *et al.*, 2004).

Due to the nitrogen based application rate, the phosphorus concentration was usually built-up because the manure had lower nitrogen to phosphorus ratio than the crops

needed; thus, when a sufficient amount of nitrogen was supplied to the crops, more phosphorus than necessary was applied. This may have resulted in the downward movement of phosphorus (Choudhary, 1996). Moreover, phosphorus application caused a greater increase in soil phosphorus when applied to soil that was initially high in phosphorus (Pote, 2002). Furthermore, when the accumulation reached soil capacity, phosphorus would be lost from the soil profile. McDowell and Sharply (2001) conducted an experiment to examine the loss of phosphorus in subsurface flow from three cultivated soils of varying soil phosphorus concentration. Phosphorus measurements were made before manure application, three weeks after sowing the soil to grass, and one year after manure application. The results showed that the concentrations of soil phosphorus significantly increased down the soil profile. The explanation offered for this was that the soil had reached phosphorus saturation prior to the manure application. Moreover, the results suggested that, despite the establishment of fast growing grass, phosphorus concentrations could not be mitigated in the short-term (one year), due to the large contribution of phosphorus to subsurface pathways.

High salt content is one of the main livestock manure characteristics and has an impact on soil properties. The impact of 25 annual cattle feedlot manure applications on soil salinity and soluble salt content had been investigated in the semi-arid region of southern Alberta. Cattle manure had been applied in the experiment at rates of 0, 30, 60, and 90 Mg ha<sup>-1</sup> per year under non-irrigated conditions, and at 0, 60, 120, and 180 Mg ha<sup>-1</sup> per year under irrigated conditions each fall since 1973. The results indicated that soil EC values increased with the cumulative amount of manure used over the years and that the increases were greater under non-irrigated than irrigated conditions. Based on

this experiment, Hao and Chang (2003) reported that the average soil EC (0–150 cm) increased by  $0.1108 \text{ dS m}^{-1}$  under non-irrigated conditions for every ton of salt applied. Significant increases in soluble salts and EC in soil were also reported by Liu *et al.* (1998) and Eghball *et al.* (2004). Hao and Chang also indicated that the soluble  $\text{Na}^+$ ,  $\text{K}^+$ ,  $\text{Mg}^{2+}$ ,  $\text{Cl}^-$ ,  $\text{HCO}_3^-$ , and SAR all increased, and that the increases were greater under non-irrigated than irrigated conditions. Under both conditions,  $\text{Ca}^{2+}$  was found to have decreased in surface soil (0–15 cm) but increased at depths below 30 cm.  $\text{K}^+$  became the dominant cation in manured surface soil. The increases in EC and soluble ions were lower under irrigated conditions due to greater downward movement which might convey salts into groundwater. Soil pH was normally found to have decreased (Liu *et al.* 1997) after manure application, but increased results were also observed (Eghball *et al.*, 2004).

Apart from physical and chemical properties, bacteria activity in soil after manure application was also detected by researchers. Peacock *et al.* (2001) reported a significant increase in biomass activity in soil after manure application. He explained that the manured soils likely had higher levels of soluble organic carbon than the unmanured soils, and were therefore able to support higher levels of microbial activity. The enhancement of carbon in soil and the lower nutrient mineralization might result in a larger, and potentially more robust, microbial community. Using the potential nitrification activity (PNA) method, Ceccherini *et al.* (1998) investigated the effects of fertilization with swine manure on ammonia oxidizing activity. It was found that additions of swine manure to soil plots increased both the potential ammonia oxidizing activity and the ammonia oxidizer sequences. Stimulation of the soil ammonia oxidizing activity was mainly due to the manure's effects on the indigenous bacterial population. In

addition, microbial activity was strongly influenced by weather; seasonal variations in microbial activities increased with depth (Cannavo *et al.*, 2004)

### **Impact on surface and ground water quality**

Nutrients in surface and ground water can affect human and aquatic organisms that rely on water for consumption and habitat. The primary parameters in this issue are nitrogen and phosphorus. Nitrogen is of concern with regards to water quality because of the solubility of its oxidized form, nitrate. Nitrate is not only a plant nutrient with implications for promoting excess plant growth, but it also presents a health concern in drinking water because it is toxic to human beings and aquatic organisms. Phosphorus is an essential nutrient for animals and for algal growth; therefore, is often identified as the control nutrient in preventing excess algae from developing (Miner *et al.* 2000). The oversupply of nutrients will result in excessive vegetative growth in the receiving water; the subsequent water quality degradation is defined as eutrophication. The main pathways by which nutrients affect surface and ground water quality are run-off and leaching.

In 1997, the Canada-Alberta Environmentally Sustainable Agriculture Agreement (CAESA) organized a study on nitrate leaching in crop rotation in the Alberta area. They found that the addition of nitrogen through commercial fertilizer, manure, or legumes, tended to increase the nitrate-N content in the root zone, but no one crop system seemed to promote excess nitrate-N accumulation and potential leaching. In this regard, there appears to be no difference between the influences of tillage and conventional tillage systems. It is the weather and cover cropping, rather than the nutrient source, that affects nitrate leaching (Ball-Coelho *et al.*, 2004).

However, heavy land application of manure may increase the risk of water body pollution. Increasing the application rate of manure could increase the NO<sub>3</sub>-N in the soil profile up to 122 cm depth (Sutton *et al.* 1984). The authors concluded that the leaching of soil nutrients to lower portions of the soil profile (62-122 cm) might be of greater concern when manure was applied by injection rather than broadcast on the soil surface, because there was less chance for slurry to evaporate or be lost through runoff.

Under subtropical grassland vegetation, increasing the rate of swine manure application from 335 to 1340 kg N ha<sup>-1</sup> significantly increased leaching of NO<sub>3</sub>-N, P, K, and Mg (King *et al.*, 1985, 1990), and NO<sub>3</sub>-N and phosphorus concentrations in rainfall runoff (Westerman *et al.*, 1985). Excessive manure or fertilizer application in combination with high precipitation could cause the leaching of nitrate to exceed tolerable values in fresh water (e.g. 50 mg nitrate/L in Netherlands) (Jongbloed and Lenis, 1998). Dauden and Quilez (2004) reported that nitrate leaching in the mineral treatment was found to be equal or higher than in the pig slurry treatments, indicating that slurry nitrogen does not contribute more than mineral fertilizer to nitrate leaching when applied at agronomic rates. The risk of nitrate leaching in corn crops fertilized with mineral N or slurry was weather and soil dependent.

Burton *et al.* (1994) demonstrated that the increased carbon associated with manure application might increase the extent of denitrification in the soil and could reduce nitrate contamination in groundwater.

However, the application of cattle slurries to the silty clay loam soil had little effect on NO<sub>3</sub><sup>-</sup>-N losses through surface water flow; instead, it increased the loss of solids and NH<sub>4</sub><sup>+</sup>-N by that same pathway (Smith *et al.*, 2001a). The researchers concluded that

rainfall events occurring immediately after manure application were particularly likely to be associated with nutrient run-off losses. Losses via subsurface flow (30 cm interflow) were consistently much lower than those occurring via surface water movement. Increasing slurry application rate and, in particular, slurry solids loading, increased solids and  $\text{NH}_4^+$ -N losses via surface run-off.

A study, which investigated the effect of using three natural organic (dairy and swine compost and a biosolid) and two synthetic organic nutrient sources ( $200 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ ) on Kentucky bluegrass and perennial ryegrass in sandy loam soil, indicated that the nutrient concentrations and losses in both runoff and leachate were highest for the 20 week period after turfgrass seeding. The loss of  $\text{NO}_3$ -N and  $\text{NH}_4^+$ -N declined significantly once turfgrass cover was established. In contrast,  $\text{PO}_4$ -P levels increased during the second year (Easton and Petrovic, 2004).

Phosphorus losses to runoff water depended less on soil phosphorus concentrations and more on the application rate and method, and rainfall intensity if rainfall occurred soon after manure application (Sauer *et al.*, 2000; Sharpley and Tunney, 2000). Novak *et al.* (2000) assessed both phosphorus accumulation in soil and movement to ground water after 10 years of typical high rate swine manure application. Loading rates were  $334 \text{ kg N ha}^{-1} \text{ yr}^{-1}$  and  $83 \text{ kg P ha}^{-1} \text{ yr}^{-1}$ . Phosphorous was found to accumulate in the soil profile and subsequently move into the shallow ground water beneath the spray field.

In the study conducted by Zheng *et al.*(2004), similar results were demonstrated. It was shown that the elevated soil phosphorus likely constituted a risk of phosphorus transfer from soil to surface waters after the application of dairy liquid manure as well as mineral fertilizer, although the application was positive in regards to crop production.



The authors also found that organic phosphorus pools were more important than inorganic phosphorus pools in the phosphorus transformations taking place in the soil they used (silty clay). Soil carbon contributes to soil phosphorus dynamics through its role as an energy source for microbes which affect biological reactions. The loss of phosphorus to runoff was also observed by Smith *et al.* (2001b), who indicated that the application of cattle manure and, especially slurry, to silty clay loam soil increased both particulate and soluble phosphorus loss in surface water flow. Losses via subsurface flow (30 cm interflow) were consistently much lower than those incurred via surface water. Increased application of slurry solids increased all forms of phosphorus loss via surface run-off. Flow-weighted mean phosphorus leaching losses on clay loam plots averaged 39 times higher than those on loamy plots. Furthermore, the losses varied among application seasons on the clay loam soil, with the highest losses generally measured for early fall applications (van Es *et al.* 2004). Other reports highlighted the risk of fast losses of phosphorus, through surface and subsurface pathways, when the surface application of manure or fertilizer to grassland soil was followed by intermittent rain (Preedy *et al.*, 2001). It was recommended that incorporating manure, recent extensive tillage, and little or no surface residue was a best management practice for minimizing surface water pollution (Tabara, 2003).

Grant *et al.* (2004) developed a model to predict phosphorus loss from the land application of hog and cattle manure. The results showed that a positive interaction exists between annual rainfall and application rate for minimizing phosphorus losses. An application rate greater than  $30 \text{ Mg ha}^{-1} \text{ yr}^{-1}$  would cause TP concentrations in the water leaving the site to rise above acceptable limits. The interaction between rainfall and

application rate suggests that phosphorus losses from manure application at any site should be assessed under the upper range of likely rainfall intensities.

### **Impact on air quality**

Ammonia and nitrous oxide are considered to be the two main gases generated during the land application of manure. Some studies have shown that the scope of ammonia N loss ranged from 10% to 99% during storage and application periods (Moal *et al.*, 1995; Vanderholm, 1975). The application method, characteristics of the manure, and weather and field conditions are considered to be important factors influencing the nitrogen volatilization from the liquid pig manure applied to, and incorporated into, arable land (Huijsmans *et al.*, 2003). It was reported in their study that the mean total volatilization (expressed as a percentage of ammonia nitrogen) was 68% for surface spreading, 17% for surface incorporation, and 2% for deep placement. Higher ammonia content in the applied manure, application rate, and temperature resulted in higher volatilization. Wind speed had a substantial effect on the volatilization rate only when the manure was surface applied or surface incorporated.

In New Zealand, measurement results for daily ammonia loss from free-draining silt loam soils covered with perennial ryegrass and white pastures demonstrated that, in most cases, over 80% of the total losses occurred within four days of urea application. Loss of nitrogen by denitrification was low (7-14 kg N ha<sup>-1</sup>) and largely confined to the winter period (Ledgard *et al.*, 1996).

Chadwick *et al.* (1998) noticed that after the October and June manure applications in their Solepur soil treatment system, between 6 and 31% of the total N in the slurry was lost through NH<sub>3</sub> volatilization. Emissions of N<sub>2</sub>O were very high following the slurry

application in October, when a total of 272 kg N ha<sup>-1</sup> was emitted as N<sub>2</sub>O in the 111 days following application. In contrast, N<sub>2</sub>O emissions totaled 1.1 kg N ha<sup>-1</sup> eight days after slurry application to dry soil in June, representing 23 and 0.17% of the total nitrogen applied, respectively. The October application of slurry resulted in large losses of N<sub>2</sub>, which represented 12% of the total nitrogen applied in the slurry.

Chantigny *et al.* (2004) evaluated ammonia volatilization following the application of anaerobically stored and anaerobically digested pig slurry to a bare loamy soil. Their results indicated that 35% of slurry-added NH<sub>4</sub><sup>+</sup>-N was lost as NH<sub>3</sub>-N for both slurries after two days, and the net soil NH<sub>4</sub><sup>+</sup>-N disappearance accounted for about 60% of the slurry-added NH<sub>4</sub><sup>+</sup>-N for both slurries after nine days. The authors assumed that biological processes, such as immobilization and nitrification, would play a significant role in slurry NH<sub>4</sub><sup>+</sup> disappearance.

Nitrous oxide emitted by soils can be produced either by denitrification in anoxic conditions or by nitrification in the presence of O<sub>2</sub>. Khalil *et al.* (2004) reported that oxygen pressure appeared to strongly influence both nitrification and denitrification rates, as well as N<sub>2</sub>O emissions. They found that nitrification was the main source of N<sub>2</sub>O at O<sub>2</sub> concentrations greater than 0.35 kPa. The amounts of N<sub>2</sub>O-N emitted by nitrification were linearly related to the amounts of nitrogen nitrified, but the slope of the regression was highly dependent on O<sub>2</sub> concentration - it varied from 0.16 to 1.48% when O<sub>2</sub> concentration was reduced from 20.4 to 0.76 kPa. Emissions of N<sub>2</sub>O by nitrification may be quite significant, then, if nitrification occurs at a reduced O<sub>2</sub> concentration.

Under aerobic soil conditions, Muller *et al.* (2004) found that the process of NO<sub>3</sub><sup>-</sup> reduction was the predominant N<sub>2</sub>O producing mechanism. These conclusions differ

from those drawn by researchers who developed theoretical relationships between soil moisture and other factors, and N<sub>2</sub>O production. Complex relationships were considered to exist between physical and biological soil properties, which supported transport processes and microbial activities in the soil profile. Even during the summer when low N<sub>2</sub>O emissions were observed, applied nitrogen fertilizers might have been partially transported below the main rooting zone by rainfall, and there have stimulated the build-up of large N<sub>2</sub>O concentrations in deeper soil layers. Still, in this experiment, the denitrification potential was found to be relatively low. Only simple carbohydrates could be readily utilized by the resident microorganisms (Murray *et al.*, 2004), possibly due to a low carbon to nitrogen ratio (only 0.2 in the subsoil and 5.9 in the topsoil).

Denitrification, or transformation into organic nitrogen compounds, was determined by the soil moisture (Ruckauf *et al.*, 2004). In their study, up to 80% of the <sup>15</sup>N nitrate added was transformed into organic nitrogen compounds under dry soil moisture conditions (water filled pore space: 31%); this transformation process was not affected by plant growth. Under reflooded conditions (water filled pore space: 100%), the total gaseous nitrogen losses were highest (77–95% of the <sup>15</sup>N nitrate added) and the transformation of <sup>15</sup>N nitrate into organic N compounds was very low (1.8% of <sup>15</sup>N nitrate added). The N<sub>2</sub> emissions were found to exceed the N<sub>2</sub>O emissions by a factor of 10–20 in planted soil, and by as much as 30 times in unplanted soil. Different plants were observed to have different influences on N<sub>2</sub>O emissions. Due to uptake, plant growth reduces nitrogen losses by 20-25% under almost all plant-growth soil conditions.

## **Pathogen transportation in the soil**

Pathogens found in swine manure might persist in the soil and be transported by runoff into surface water, consequently contaminating the water. Manure application can result in higher pathogen populations in the soil; however, *Salmonella*, fecal *coli*, and male-specific *coliphages* died-off quickly in the runoff mixing zone and survival time did not relate to the manure application rate (Gessel *et al.*, 2004). In the same report, Somatic *coliphages* was pointed out as being the most persistent pathogen in surface soil (up to 143 days) and presenting a higher risk of off-site transport by runoff than bacterial pathogens. In addition, their persistence in the surface soil was correlated with the manure application rate. Moreover, the survival of fecal bacteria in liquid manure was affected by the physical and chemical conditions of the soil before manure application, as well as by the conditions after mixing soil and manure (Unca and Goss, 2003).

However, *Salmonella*, as well as *E. coli* O157, and *Campylobacter* were observed surviving in the soil for up to one month after application to both the sandy arable and clay loam grassland soils, whereas *Listeria* commonly survived for more than one month (Nicholson *et al.*, 2004). These pathogens survived for up to three months in stored slurries and dirty water, but lived for less than one month in solid manure heaps with a temperature over 55 °C (Nicholson *et al.*, 2004).

### **2.3 Research needs**

Based on the above literature review, the following research needs have to be addressed:

- Although many studies have been conducted in the field of animal manure treatment and application, using natural soil as a biofilter or bioreactor to treat LSM

has not been extensively described. The changes in soil properties which occur in the soil filters during and after intensive manure application need to be investigated in detail.

- After manure application, another problem faced is how to recover the manured soil. It is economically reasonable to find natural recovery methods for soil recovery, such as natural rest or water flushing.
- For land application, the amount of manure was usually determined from an agronomy perspective. It is necessary to research what kind of influence on crops or soils would result from manure application based on hydraulic loading and carried out more frequently.
- The quality of leachate should be investigated to determine if it is good enough for crop development or other reuse objectives. Research is also needed to investigate the reuse or recovery potential of the manured soil in the filters by cultivating crops on it.

### **3 MATERIALS AND METHODS**

#### **3.1 Project overview**

The full-scale swine manure treatment project has been carried out for three years (from 2002 to 2004). The goal of the project is to investigate the following swine manure treatment system: LSM was settled for 24 hours and then subjected to physical/chemical treatment, primarily to reduce its phosphorus and solids content. Effluent from the physical/chemical process would be sent to a nitrogen and organic processing area (NOPA) system for further treatment. The infiltrate from the NOPA would be collected for possible reuses including crop irrigation, barn watering, and animal feeding etc.

Prior to 2004, some research (using physical/chemical methods to treat swine manure) had been achieved which provided a great deal of information for the third year of research. In 2004, a total of three systems were studied to treat liquid swine manure, including a struvite formation system, an up-flow blanket clarifier and filter system, and a soil column system (imitating NOPA). The last one constitutes the main content discussed in this thesis.

#### **3.2 NOPA description and experiment design**

##### **3.2.1 NOPA system**

NOPA is a planned manure treatment system that would consist of a storage/treatment pond followed by an area consisting of non-vegetated land hydraulically isolated from the surrounding area, but linked to another pond and land area by an under-drain system (See Appendix D).

Incoming pre-treated LSM would be stored in the pond until the following spring. During this period, some organic content would be reduced due to anaerobic conditions, as well as some suspended solids removal occurring through sedimentation. Some nitrogen would also be emitted out of the pond. In the spring, the liquid manure in the pond would be sprayed over the land area for further treatment. At this time, the soil would work as a bioreactor and biofilter intended to reduce much of the organic matter in the manure via interception and bio-degradation within the soil. Nitrogen content would be reduced via interception, nitrification, and denitrification within the soil, or by plant uptake if plants could survive in the soil. The leachate can be directly reapplied over land for crop development. The manured soil can also be used as fertilizer.

### **3.2.2 Experiment design**

This soil application experiment was designed and soil columns were employed to simulate the soil in NOPA processing.

This experiment included three phases (See Table 3-1). In Phase I, three manure application rates (treatments), 12, 25, and 50 mm d<sup>-1</sup>, respectively, were carried out to investigate the variations of hydraulic conditions in soil, the leachate quality, and the changes in soil properties with manure application. An important issue in this phase was to provide an optimal application rate for further study. Each treatment had one control (apply tap water) and two replicates (apply pre-treated manure); thus, a total of nine soil columns were employed. Kentucky bluegrass sod covered soil at the beginning of this run, but died due to its poor salt or nitrogen tolerance ability, and was removed. All of these columns were run for an eight-week period, except the columns with the application rate of 25 mm d<sup>-1</sup>, which were continuously irrigated with manure after eight weeks, but



at a new application rate of 17 mm day<sup>-1</sup> for two weeks, followed by 6mm day<sup>-1</sup> for another two weeks.

In Phase II, the three columns with the 12 mm d<sup>-1</sup> application rate rested for one and a half months without any irrigation, and the three columns with the 25 mm d<sup>-1</sup> application rate were recovered by tap water for more than two months. The objective of this phase was to evaluate the recovery ability of the soil properties after the discontinuation of the manure application.

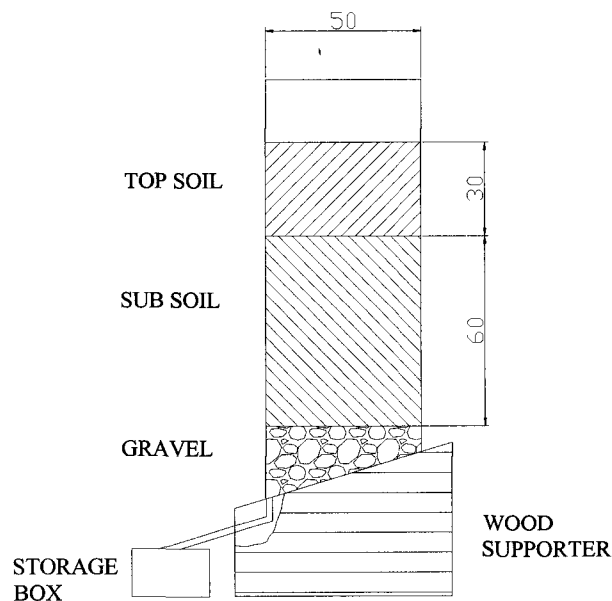
In Phase III, a suggested application rate (17 mm d<sup>-1</sup>) based on the results of Section I was adopted and applied to four columns filled with fresh bare soils. Four columns were used as replicates to accept the aged, partially treated swine manure. The other soil column was used as a control to accept tap water. Also, the top soils in the columns were tilled before each application. The objective of the final phase was to investigate the manure treatment results via soil and the soil response after manure application, as well as the nitrogen conversion within the soil column systems. Moreover, the mixture of soil leachate was reused in another soil column to develop bermudagrass and the crop response was observed.

### **3.3 Experiment system establishment and sampling**

#### **3.3.1 Experiment system establishment**

Study sites were located on the University of Alberta farm, where the Swine Research Facility could provide swine manure for research. The experiments for Phases I and II were run in natural conditions (outside). Phase III was run inside under greenhouse conditions due to the approach of winter, as well as the requirements of crop development. Pretreated manure was pumped out of the storage tank, delivered to the

experiment sites, and stored in several 50 gallon barrels for experimental use. The soil columns were made up of PVC pipes with a diameter of 50 cm. These stood on wooden supporters with a bottom slope of around 0.3. The bottom ends of the PVC pipes were closed and a hole was drilled and connected to a plastic tube for leachate collection. The bottoms of the columns were filled with gravel, then 60 cm of subsoil, and 30 cm of topsoil (See Figure 3-1).

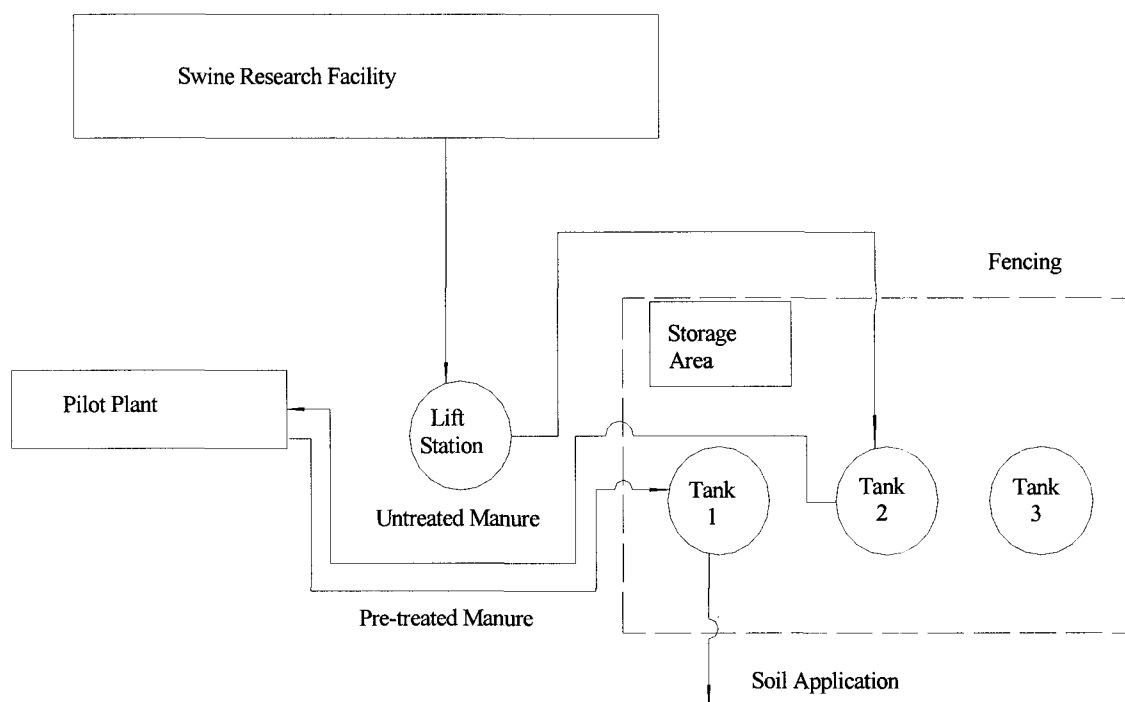


**Figure 3-1 Soil column profile (Unit: cm)**

### 3.3.2 Manure pretreatment

The liquid manure used in this experiment was pretreated in a pilot plant set up in a trailer unit adjacent to the manure storage tanks located beside the Swine Research Facility on the University of Alberta farm. The treatment process at this pilot plant employed the physical/chemical method, using a customized sludge blanket clarifier with rapid and slow mixing, a patented Martin filter unit, two axial-flow auger pumps, three PVC storage tanks, and PVC pipes and hoses for connections. As shown in Figure 3-2,

the initial manure coming out of the research facility was stored first in a tank (Tank 2), and then pumped into the pilot plant for treatment. The treated manure was then stored in another tank (Tank 1), from which the manure was pumped out and delivered to the soil application experiment site for use.



**Figure 3-2 Schematic of pilot plant and manure facility**

### 3.3.3 Soil packing and sampling

The soils used in this experiment were natural soils prepared by a roter from the University of Alberta farm and then packed into PVC columns with a 15 kg tamper. Due to the different appearance of natural soil at different depths (top soil was much darker than sub soil), the soils were filled into the columns without mixing. Sub soils were still used as sub soils in the columns and filled to a depth of 60 cm, followed by top soils filled a depth of 30 cm.

Prior to filling the columns with soil, initial soil samples were collected randomly across the prepared soil piles and then combined. Two or three different sub samples were then sent to the soil analysis lab for initial testing of soil properties. During the manure application periods, top and sub soils were sampled in each column at two depths (0 - 30cm and 30 – 90cm) every two weeks in Phase I. After each sampling, the holes created were refilled with the original soil and tamped. In Phase III, soil samples were collected only at the beginning and the end of the experiment. Soil samples were collected from each column at five different depths (0-15cm, 15-30cm, 30-50cm, 50-70cm, and 70-90cm). In Phase II, the rested samples were only taken at the beginning and the end of testing period at two depths (0-30cm and 30-90cm). The deeper soil samples (30-90cm) were used in an attempt to investigate the nutrient budget and to track the vertical movement of nutrients in the soil profiles. The collected soil samples were stored in plastic bags, kept in a cooler, and sent directly to the Natural Resources Analytical Laboratory of the department of Renewable Resources at the University of Alberta for analysis. The following analyses were performed: Total Kjeldahl Nitrogen (TKN), ammonium nitrogen ( $\text{NH}_4^+\text{-N}$ ), nitrate and nitrite nitrogen ( $\text{NO}_3^-\text{-N} + \text{NO}_2^-\text{-N}$ ), organic matter (OM), electrical conductivity (EC), pH, total phosphorus (TP), moisture content, and sodium adsorption ratio (SAR). In Phases I and II, the temperature and the precipitation were also monitored.

#### **3.3.4 Manure and leachate sampling**

Pretreated manure was randomly sampled from the different storage barrels at different times throughout the application period. Leachate was collected through the under-drainage collection system every day and grab samples were taken every one or

two weeks during Phase I. In Phase III, for the purpose of nitrogen balance, weekly-combined samples were taken each week for analysis. The collected samples were bottled and stored in a cooler and immediately sent to the environmental engineering lab for analysis. If samples could not be analyzed that day, they were stored at 4 °C. Before sampling, the total volume of the liquid was measured and recorded. Liquids were tested for five day biochemical oxygen demand (BOD<sub>5</sub>), chemical oxygen demand (COD), total dissolved organic carbon (TOC), TKN, NH<sub>4</sub>-N, nitrate and nitrite nitrogen, TP, EC, pH, total solids (TS), and total dissolved solids (TDS).

### **3.3.5 Grass development**

Bermudagrass was seeded in one column filled with the original soil at the beginning of October, 2004. This column was situated in a greenhouse with an environmental temperature of 24.5 to 31°C. The bermudagrass was then allowed to develop for about two months. Tap water was used during this period. From November 30, 2004 until the end of the following January leachate was applied to irrigate the grass. The grass was cut every two weeks for more nutrient uptake and the residual grass was maintained at a height of around 2 cm. Grass performance was observed and the dry weight (DW) of the ripened grass was recorded for calculation of the yield.

### **3.4 Statistical analysis**

Data were analyzed using two sample t-test to evaluate soil content variations before and after manure application, as well as the difference in soil characteristics during different experimental phases. Analysis of variance (ANOVA) was used to evaluate the leachate quality variations in different weeks and the soil properties in different layers, in

order to compare the different layers' responses to manure application. All differences were considered significant at  $p < 0.05$ , unless otherwise noted.

### **3.5 Research scope of this thesis**

This thesis focuses on the field shown below (See Table 3-1):

- soil columns with a  $12 \text{ mm d}^{-1}$  manure application rate (Column 1, 2, and 3 in Phase I)
- soil columns at rest without any application (Column 1, 2, and 3 in Phase II)
- soil columns with a  $17 \text{ mm d}^{-1}$  manure application rate (Phase III)
- grass irrigated with the leachate from the soil columns with a  $17 \text{ mm d}^{-1}$  manure application rate (Phase III)

**Figure 3-3 Experiment content, schedule, and three stages**

Column Number	Time					
	08-Jun-04	03-Aug-04	14-Sep-04	05-Oct-04	30-Nov-04	30-Jan-05
Col. 1	12 mm/d	Rest without application		Grass development	Irrigated with leachate	
Col. 2	12 mm/d	Rest without application		17 mm/d, control		
Col. 3	12 mm/d, control	Rest without application		17 mm/d		
	Phase I		Phase II		Phase III	
	08-Jun-04	16-Aug-04	31-Aug-04	14-Sep-04	23-Nov-04	
Col. 4	25 mm/d, control	17mm/d	6 mm/d	Tap water recovery		
Col. 5	25 mm/d	17mm/d	6 mm/d			
Col. 6	25 mm/d	17mm/d	6 mm/d			
	Phase I			Phase II		
	08-Jun-04	03-Aug-04		05-Oct-04	30-Nov-04	
Col. 7	50 mm/d, control			17 mm/d		
Col. 8	50 mm/d			17 mm/d		
Col. 9	50 mm/d			17 mm/d		
	Phase I			Phase III		

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### 3.6 Analysis methods for liquid samples

All quality parameters for the LSM and leachate were analyzed in the Environmental Engineering lab, University of Alberta, according to the procedures described in the Standard Methods (APHA, 1995).

#### 3.6.1 Five day biological oxygen demand (BOD<sub>5</sub>)

BOD is usually defined as the amount of oxygen required by microorganisms when stabilizing decomposable organic matter under aerobic conditions. BOD<sub>5</sub> is used to indicate the amount of biodegradable organic matter in liquids within five days and is determined by measuring the amount of dissolved molecular oxygen (DO) utilized by the biochemical degradation of organic material during a five-day incubation period. Sometimes it also contains the oxygen used to oxidize inorganic material such as sulfides and ferrous iron (AHAP, 1995). The BOD<sub>5</sub> test is performed more often than the longer BOD tests (such as BOD<sub>7</sub> and BOD<sub>20</sub>) because it takes less time for completion and it avoids the nitrification process that normally occurs after five to seven days. The method used was Standard Method 5210B (APHA, 1995) and the apparatus used was an air incubator and a DO meter (YSI Incorporated Medel 50B).

According to Standard Methods, DO uptake after five days has to be at least 2 mg/L and the residual DO has to be at least 1 mg/L. Otherwise, the BOD<sub>5</sub> test should be repeated.

BOD values were calculated using the following equation:

$$BOD = \frac{(DO_0 - DO_5) - (DO_{s0} - DO_{s5})}{P}$$



Where:  $DO_0$  = 0 day DO of samples, mg/L;

$DO_5$  = five-day DO of samples, mg/L;

$DO_{S0}$  = 0-day DO of seed check (blank) samples, mg/L;

$DO_{S5}$  = five-day DO of seed check (blank) samples, mg/L;

P = decimal volumetric fraction of sample used.

### 3.6.2 COD

The COD test is used to determine the oxygen equivalent of the organic materials in a sample that is susceptible to oxidation by a strong chemical oxidant (APHA, 1995). The COD value should be higher than the BOD value because it not only contains the biodegradable organic substrate, but also the non-biodegradable organic substrate in the liquid. Two methods are commonly used to test COD; these are the open reflux and closed reflux methods (5220 D in Standard Methods), respectively. The latter was employed in this project. The apparatus included a HACH COD reactor and a Spectrophotometer (Pharmacia Biotech Novaspec II at 600 nm)

### 3.6.3 TKN and ammonia nitrogen

Nitrogen exists in various forms in the environment, including organic nitrogen and inorganic nitrogen. The most common forms of nitrogen, and their corresponding oxidation states in wastewater and in the water/soil environment, are ammonia ( $NH_3$ ), ammonium ( $NH_4^+$ ), nitrogen gas ( $N_2$ ), nitrite ion ( $NO_2^-$ ), and nitrate ion ( $NO_3^-$ ). Total nitrogen (TN) comprises all of them. TKN analysis is used to determine the sum of ammonia nitrogen and organic nitrogen. Therefore, TN includes TKN, nitrate nitrogen, and nitrite nitrogen.

A volumetric procedure was used to measure ammonia nitrogen because of the accuracy desired in terms of the range of nitrogen concentration in the samples. In the TKN test, samples were digested first and then distilled to convert all organic nitrogen into ammonia nitrogen. The ammonia nitrogen was then titrated by hydrochloric acid through an auto titration apparatus. Ammonia analysis only subjects a sample to the digestion and titration processes. The organic nitrogen can be calculated by taking the difference between TKN and ammonia nitrogen, as mentioned above. The method used is 4500-N in Standard Methods. The apparatus included a digestion block (Tecator Kjeldahl 2020), a distillation apparatus (Tecator 1020), and a titration apparatus (Mettler Telodo DL 50 autotitrator).

$$\text{TKN or ammonia nitrogen} = \frac{(T - B) \times N \times 14.007 \times 1000}{\text{sample volume}}$$

Where: T = Volume of titrant used for sample, mL

B = Volume of titrant used for blank, mL

N = Normality of titrant to four decimal places

#### **3.6.4 Nitrate and nitrite**

Normally, the concentrations of nitrate and nitrite in surface and groundwater are low. For this reason, sensitive methods are needed for their measurement. In this project, nitrate and nitrite were analyzed by ion chromatography (IC). The Standard Methods used were 4500-NO<sub>2</sub><sup>-</sup>C and 4500-NO<sub>3</sub><sup>-</sup>C. Although conventional colorimetry is available for determining individual anions, only this method provides a single instrumental technique that may be used for rapid, sequential measurement.

### **3.6.5 Total phosphorus**

Analysis of total phosphorus (TP) requires the conversion of all phosphorus forms to the dissolved orthophosphate form. This conversion is achieved through sample digestion. The colorimetric method was followed to analyze the dissolved orthophosphate. In this project, the 4500- P E, ascorbic Acid Method (APHA, 1995) was employed to determine the total phosphorus concentration levels in the manure and leachate samples. The apparatus included a Spectrophotometer (Ultraspec 2000 Pharmacia Biotech).

### **3.6.6 Total solids, total suspended solids, and total dissolved solids**

Total Solids (TS) refers to the matter that remains as residue in the crucible upon evaporation and drying at 103 to 105 °C. TS in a liquid sample consists of total dissolved solids (TDS) and total suspended solids (TSS). TDS is material in the water that will pass through a filter with a 2.0 µm or smaller nominal average pore size, and TSS is the material retained by the filter. In this project, TS and TDS were determined experimentally, whereas TSS was calculated from the difference between TS and TDS. The method used was Standard Method 2540 (APHA, 1995).

### **3.6.7 TOC**

In this project, TOC refers to all covalently bonded carbon atoms in the dissolved materials. Total dissolved inorganic carbon (TIC) refers to dissolved carbon dioxide, bicarbonate, and carbonates in solution. Total dissolved carbon (TC) is the combination of both. The combustion infrared method (APHA, 1995) was employed in this project. Total organic carbon can be oxidized to carbon dioxide in a high temperature furnace

(680 °C) that is continuously flushed with pure oxygen. The resulting mixture of oxygen and carbon dioxide is then passed through the detector. To determine TIC, the sample has to be acidified so that inorganic carbon can be purged from the sample, as carbon dioxide and then can be measured directly. Carbon dioxide from either TC or TIC is passed through a nondispersive infrared detector (NDIR) by which TC or TIC is measured directly. TOC is determined by subtracting TIC from TC. All samples were analyzed in triplicate. The method used was Standard Method 5310 B (APHA, 1995). The main apparatus was a Dohrmann Carbon Analyzer (DC-80)

### **3.6.8 pH, EC**

pH is used to express the hydrogen-ion activity and is very important in every phase of environmental engineering practice. pH is represented by

$$pH = -\log\{H^+\} \text{ or } pH = -\log[H^+]$$

The electrical conductivity (EC) is a measure of the ability of a solution to conduct an electrical current. The measured EC value is usually used as a surrogate measure of TDS concentration because the electrical current is transported by the ions in solution. But, the relationship between them is not solid in different wastewaters. The EC of water is one of the most important parameters used to determine suitability for agriculture irrigation. The apparatus used included a pH meter (Fisher Scientific AR 20), a conductivity meter (YSI model 34), and a conductivity cell (YSI 3417).

### **3.7 Analysis method for soil**

Most soil samples were analyzed by the Natural Resources Analytical Laboratory of the Department of Renewable Resources at the University of Alberta. For detailed

methods and procedures of soil analysis, please refer to Maynard and Kalra (1993), Page *et al.* (1982), Kalra and Maynard (1991), and Kalra (1994). The analysis methods are described briefly as follows.

### **3.7.1 Organic matter**

#### **3.7.1.1 General consideration**

Organic matter in soil can be oxidized when heated at 375 °C and emitted from the soil. Therefore, the amount of organic matter can be estimated by weight loss. Generally this method is referred to as loss-on-ignition (LOI). This procedure is subject to error, as weight loss may include carbon from carbonates and water and hydroxyl groups from clay. Combustion of inert carbon compounds, as well as some volatilization of substances other than organic material, can also be sources of error. Incomplete oxidation of carbonaceous materials in soil may happen at 375 °C.

#### **3.7.1.2 Apparatus**

Muffle furnace: 375±5 °C

Desiccator

Porcelain Crucibles

Analytical balance: APX-200

#### **3.7.1.3 Procedure**

Crucibles had to be heated for one hour at 375 °C, then cooled in the open to about 150 °C. Before weighting, they had to be cooled to room temperature in a desiccator. Five grams of oven dried soil (105 °C) was put into each crucible. The crucibles were then placed in a muffle furnace. A temperature of 375±5 °C was maintained overnight and

then the crucibles were removed and cooled to room temperature in a desiccator. The following equation was used to calculate the organic matter.

$$\text{Loss on ignition (\%)} = \frac{\text{weight of oven-dried sample (g)} - \text{weight of sample after ignition (g)}}{\text{weight of oven-dried sample (g)}} \times 100$$

### **3.7.2 Moisture content, bulk density, and porosity**

#### **3.7.2.1 General consideration**

Soil bulk density refers to the dry weight of soil per unit volume and is normally expressed in  $\text{g cm}^{-3}$ . A very compacted soil would have a bulk density of 1.4 to 1.6  $\text{g cm}^{-3}$ . An open friable soil with high organic matter content would have a bulk density less than 1.0  $\text{g cm}^{-3}$ . Soil porosity is the volume of open space or pore volume, which is independent of the size of the pores. The porosity is important for growing things in soil, as it defines the volume of water that can be held in a given volume of soil.

#### **3.7.2.2 Apparatus**

Dry oven: 103-105 °C

Sampling Ring

Analytical Balance

#### **3.7.2.3 Procedure**

The first step was to take a sample from a soil column using a ring core. The sample was then weighted. After that, a subsample was extracted from the sampling soil to determine the moisture content and the dry soil weight. The subsample was taken and placed in a tin cup, which was preweighted. The tin cup containing the subsample was

placed into the oven at 105 °C overnight, and then cooled to room temperature and weighed. The calculation equations are shown below.

$$\text{Soil water content (g/g)} = \frac{\text{weight of moist soil} - \text{weight of oven dry soil}}{\text{weight of oven dry soil}}$$

$$\text{Soil bulk density (g/cm}^3\text{)} = \frac{\text{oven dry weight of soil}}{\text{volume of soil}}$$

$$\text{Soil porosity (\%)} = 1 - \left( \frac{\text{soil bulk density}}{2.65} \right)$$

### **3.7.3 Soil sodium adsorption ratio**

#### **3.7.3.1 General consideration**

Soils with an accumulation of exchangeable sodium are often characterized by poor tilth and low permeability, making them unfavorable for plant growth. The Sodium Adsorption Ratio (SAR) characterizes salt-affected soils by providing information on the comparative concentrations of Na<sup>+</sup>, Ca<sup>2+</sup>, and Mg<sup>2+</sup> in soil solutions. When the SAR rises above 12 to 15, serious physical soil problems arise and plants have difficulty absorbing water (Munshower, 1994).

#### **3.7.3.2 Apparatus**

Atomic Absorption Spectrometry (AA): Spectr AA-880

#### **3.7.3.3 Procedure**

The first step was to make the soil into a saturated paste by mixing it with a certain amount of DI water. Then the paste was put into the funnel of a vacuum filter to extract the soil solution. After that, the soil solution was passed through AA to analyze the concentrations of Na<sup>+</sup>, Ca<sup>2+</sup>, and Mg<sup>2+</sup>. SAR can be calculated by the following equation.

$$SAR = \frac{Na^+}{\sqrt{\frac{1}{2}(Ca^{2+} + Mg^{2+})}}$$

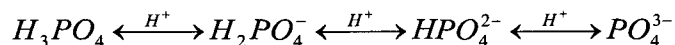
Where the concentrations of  $Na^+$ ,  $Ca^{2+}$ , and  $Mg^{2+}$  are expressed in millequivalents per liter (meq/L)

### 3.7.4 TKN and TP

#### 3.7.4.1 General consideration

Nitrogen is one of the most important nutrients in soil and is circulated in nature. Most of the nitrogen in soil is present in the form of complex organic molecules. The inorganic nitrogen added to the soil can be converted to organic nitrogen by microorganisms or by plant uptake. However, organic nitrogen in soil is not understood very clearly (Stevenson and Cole, 1999). TKN is normally used to indicate the total nitrogen in soil. However, this method does not recover  $NO_3^-$ -N and  $NO_2^-$ -N; therefore, it must be modified to include these forms of nitrogen if the soil under analysis contains significant amounts of  $NO_3^-$ -N and  $NO_2^-$ -N (Nelson and Sommers, 1982).

Phosphorus (P) is another important nutrient in soil. In the pedosphere, P is found largely in its oxidized state as orthophosphate, mostly as complexes with Ca, Fe, Al, and silicate minerals (Stevenson and Cole, 1999). The form of the phosphate ion in solution varies with pH. In dilute solutions, phosphoric acid dissociates as follows:



From 15 to 80% of the P in soils occurs in organic forms, the exact amount being dependent upon the nature of the soil and its composition (Stevenson and Cole, 1999). The total phosphorus (TP) is the sum of inorganic and organic phosphorus. In this project, TP



was determined using the Kjeldahl phosphorus method, which is based on the colorimetric method.

#### **3.7.4.2 Apparatus**

Technicon AutoAnalyzer II

Digestion apparatus: Technicon BD-20 Heating Unit

Soil grinder

Sieve: 100 mesh

Reciprocating shaker: 160 strokes per minute

#### **3.7.4.3 Procedure**

The procedure employed followed the Industrial Method for the Technicon AutoAnalyzer II. Around 0.2 g of very fine air-dried soil was first weighted and then put into a 250 mL digestion tube. 10 mL of concentrated sulfuric acid was then added, along with one Kjeltab. The digestion tube was placed on an electrically heated aluminum block and digested for four hours at a temperature of 360 °C. The color intensity which developed was measured at 660 nm on a Technicon AutoAnalyzer. A similar process was performed for TP analysis, which is also based on the colorimetric method in which a blue colour is formed by the reaction of orthophosphate, the molybdate ion, and the antimony ion, followed by reduction with ascorbic acid at an acidic pH. The colour intensity was also read at 660 nm on a Technicon Auto Analyzer

### **3.7.5 Ammonia, nitrate, and nitrite**

#### **3.7.5.1 General consideration**

Normally there is little nitrogen present in soils in the immediately usable nitrate or ammonium forms. However, the organic forms can be converted into ammonium and nitrate by soil microorganisms, a process called mineralization. The amount of nitrogen available to plant roots depends on the rate of mineralization, and this in turn depends on all those environmental factors that affect the activity of the microorganisms - the amount of carbon, temperature, and so on.

#### **3.7.5.2 Apparatus**

Reciprocating shaker: 160 strokes per minute

Technicon AutoAnalyzer II

#### **3.7.5.3 Procedure**

Ten grams of air dried soil was mixed with 50 mL 2M KCl solution and shaken for 30 minutes. The mixture was filtered using a filter. Ammonium, nitrate, and nitrite were extracted out of the soil and then analyzed using a Technicon Auto Analyzer.

### **3.7.6 EC and pH**

#### **3.7.6.1 Apparatus**

pH/EC meter: Fisher AR20

Reciprocating shaker: 160 strokes per minute

#### **3.7.6.2 Procedure**

A soil sample of approximately 15 mg was put into a bottle and mixed with 30 mL DI water. The bottle with the mixture was shaken for 30 minutes and then passed through a

filter. After that, the filtrate was analyzed for EC and pH using different electrodes on the meter. Before analysis, the pH meter was calibrated with a commercial buffer at pH 4 and 7. A temperature probe provided temperature compensation and the conductivity data could be converted to a saturated paste equivalent.

## 4 RESULTS AND DISCUSSIONS

### 4.1 Manure and leachate quantity and quality

#### 4.1.1 Manure and leachate quantity

The application rates of manure in Phases I and III were  $12 \text{ mm d}^{-1}$  and  $17 \text{ mm d}^{-1}$ , respectively. However, due to the change in operation conditions, the actual amounts applied were lower than expected, especially in Phase I. After the eight week operation period, the total application amounts for Phases I and III were 96970 and 181390 mL, respectively. In Phase III, because of water clogging, column 3 had a lower total application amount of 128850 mL. Taking column 1 in Phase I and columns 3 and 8 in Phase III as examples, the cumulative volumes of applied manure and leachate are showed in Figures 4-1, 4-2, and 4-3. Values for the other columns are shown in Appendix A.

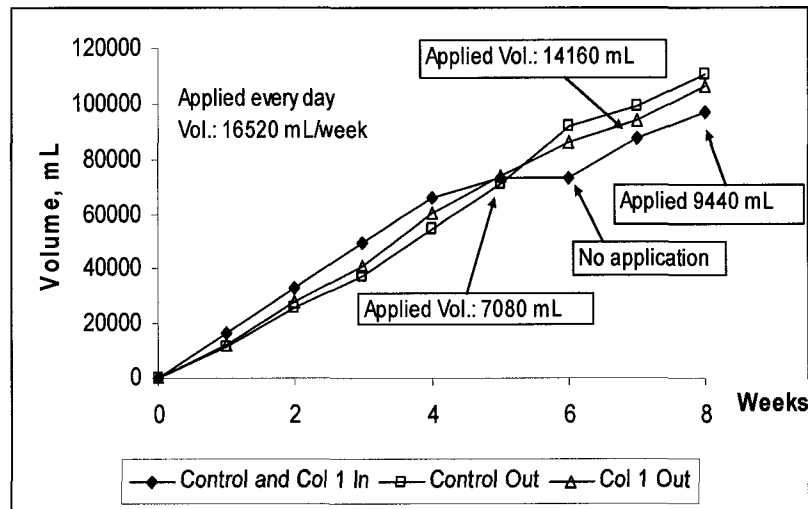


Figure 4-1 Cumulative volumes of manure and leachate in Phase I (Col. 1)

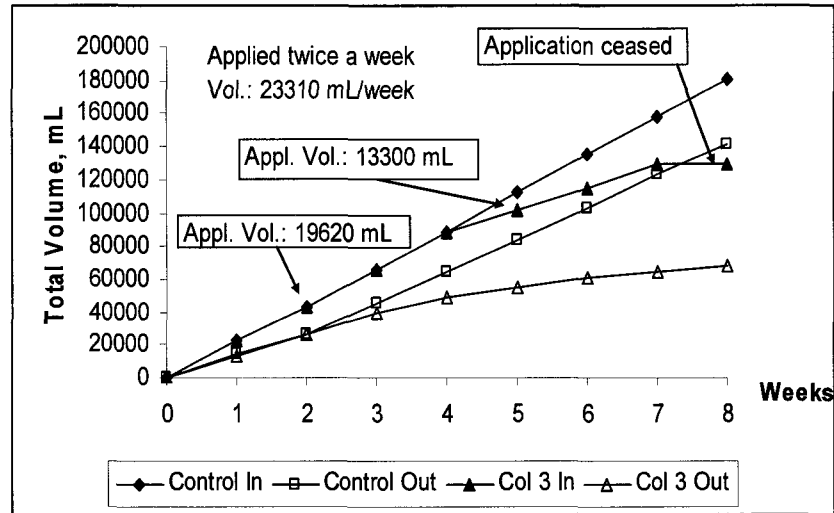


Figure 4-2 Cumulative volumes of manure and leachate in Phase III (Col. 3)

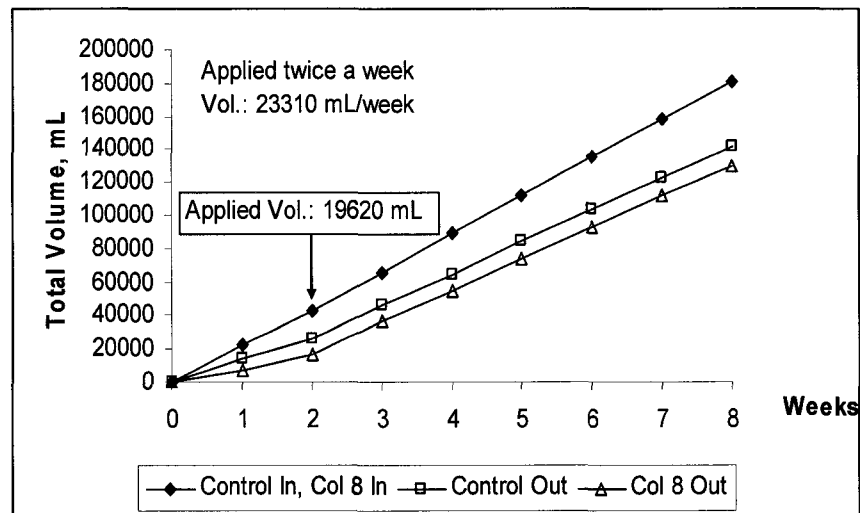


Figure 4-3 Cumulative volumes of manure and leachate in Phase III (Col. 8)

The reduction in manure application during Phase I was mainly due to the heavy rain occurring after the fourth week, which resulted in a large amount of rain water accumulating on the soil surface. The manure application was even paused during the sixth week. Figure 4-1 illustrates that, at the end of the experiment, the amount of leachate was higher than the amount of manure applied. The excess leachate was contributed by precipitation. In Phase III, the volumes of manure applied and liquid

leached out increased constantly throughout the eight week period (Figure 4-3). This was a result of the static indoor operating conditions. Moreover, clogging was not observed in Columns 7, 8, and 9, therefore consequently manure application was not paused. Column 3 clogged after the fourth week, resulting in reduced manure application in the subsequent experiment. In addition, column 3 leaked during the experimental period, resulting in a certain amount of leachate not accounted for.

Water balances were conducted in Phase III. There was no run-off in the system, due to the use of soil columns in this experiment. Therefore, the analysis was based on the following equation:

$$(\text{In} - \text{Out}) - \text{Net Loss} = \text{Storage} = \text{Final soil} - \text{Initial soil}$$

The water balance results are tabulated in Table 4-1.

**Table 4-1 Water balance in the soil column system in Phase III**

	Control	Col.3	Col. 7	Col. 8	Col. 9
Water in (mL)	181390	128850	181390	181390	181390
Water accumulation in soil (mL)	10670	15870	38730	41040	43090
Water out (leachate, mL)	141800	68190	138140	130210	131980
Net loss (mL)	28920	44790	4520	10140	6320
Average net loss (mm/d)	2.6	4.1	0.4	0.9	0.6

The net loss of water was attributed to evaporation. Columns 7, 8, and 9 showed similar amounts of evaporation. Compared to the application rate of 17 mm day<sup>-1</sup>, the amount which evaporated accounted for less than 6%. The control column showed a relatively higher amount of evaporation, which accounted for 15% of the water applied. Due to seepage, the water loss in column 3 was very high.

#### 4.1.2 Characteristics of pre-treated swine manure

The manure used in this project was pre-treated by a physical/chemical pilot plant and had been stored in an in-ground steel tank for half a year. Before land application, the manure was transferred into several 200 L (50 gallon) plastic barrels located at the experimental site, and was then applied to the soil columns. Due to the limited storage capacity of the barrels, two batches of manure were pumped out from the steel tank before Phase I (June 7, 2004; batch one) and Phase III (Sept. 27, 2004; batch two), respectively. The duration between applications of the two batches was around four months. Grab samples from each of the batches were collected for data analysis and the results are tabulated in Tables 4-2 and 4-3.

**Table 4-2 Pre-treated swine characteristics in batch one**

Parameters	Number of samples	Maximum	Minimum	Mean	Standard Deviation
EC, dS/m, 25°C	6	12.13	9.86	10.96	0.97
pH	6	8.42	7.91	8.16	0.18
TS, mg/L	6	5195	4193	4527	396
TDS, mg/L	6	5042	3937	4527	445
TP, mg/L	6	14	10	12	1.43
TKN, mg/L	6	1547	1424	1496	53
NH <sub>4</sub> -N, mg/L	5	1439	1357	1405	36
NO <sub>2</sub> -N, mg/L	2	0.3	0	0.15	0.21
NO <sub>3</sub> -N, mg/L	2	0.4	0.3	0.35	0.07
BOD <sub>5</sub> , mg/L	5	4314	3124	3870	561
COD, mg/L	5	8807	7258	8118	556
TOC, mg/L	2	1653	1569	1611	60

**Table 4-3 Pre-treated swine manure characteristics in batch two**

Parameters	Number of samples	Maximum	Minimum	Mean	Standard Deviation
EC, dS/m, 25°C	7	12.02	11.24	11.75	0.27
pH	7	8.1	7.87	7.98	0.09
TS, mg/L	6	4918	3918	4192	378
TDS, mg/L	6	4803	3778	4050	390
TP, mg/L	7	17.51	12.25	14.60	1.98
TKN, mg/L	6	1651	1361	1533	103
NH <sub>4</sub> -N, mg/L	6	1572	1328	1493	88
NO <sub>2</sub> -N, mg/L	4	6.18	0	1.54	3.09
NO <sub>3</sub> -N, mg/L	4	6.22	0.6	2.50	2.54
BOD <sub>5</sub> , mg/L	7	4440	3610	3866	335
COD, mg/L	6	6464	5683	6037	273
TOC, mg/L	5	1965	1166	1568	336

Statistical analysis showed that there were no significant differences between most of the parameters for the two batches. The only parameter that showed a significant difference between the two batches was the pH value ( $p = 0.025$ ). However, the difference on average was only 0.18 pH units, which is negligible.

Obviously, the LSM exhibited relatively stable chemical properties after physical/chemical pre-treatment. It was a liquid with high salt content, high TKN and ammonia nitrogen, low nitrate and nitrite nitrogen, and low TP concentrations. It was also an intensely organic liquid with BOD and COD concentrations of greater than 3000 and 5000 mg L<sup>-1</sup>, respectively. The physical/chemical pre-treatment process effectively removed TP removal, but was not very effective for nitrogen and organic substrate removal. The neutral or slightly alkaline condition ensured that the liquid contained high concentrations of ammonia nitrogen, which accounted for more than 90% of the TKN. The ratio of BOD<sub>5</sub> to COD was 0.6, indicating that this liquid could be easily biologically degraded (Metcalf and Eddy, 2003).



### 4.1.3 Leachate

Leachate showed an obvious difference in appearance from the pre-treated manure after soil column filtration. Figures 4-4 (Phase III; left bottle is control, then column 3, 7, 8, and 9, and right bottle is pre-treated manure) and 4-5 (Phase I, column 1) compare the appearance of leachate from different columns. In Phase III, leachate from columns 7, 8, and 9 showed a similar appearance, being a little darker than that obtained from column 3. In Phase I, all the leachate looked very clear and was similar to the leachate from column 3 in Phase III.



**Figure 4-4 Leachate appearance in Phase III**



**Figure 4-5 Leachate appearance from column 1 in Phase I**

#### 4.1.3.1 Carbonaceous substrates

The hydraulic loading for Phase I was  $12 \text{ mm d}^{-1}$ , representing the organic loading of  $0.046 \text{ kg BOD m}^2 \text{ d}^{-1}$ . The variations in the concentrations of  $\text{BOD}_5$ , COD, and TOC in the leachates are shown in Figures 4-6, 4-7, and 4-8.

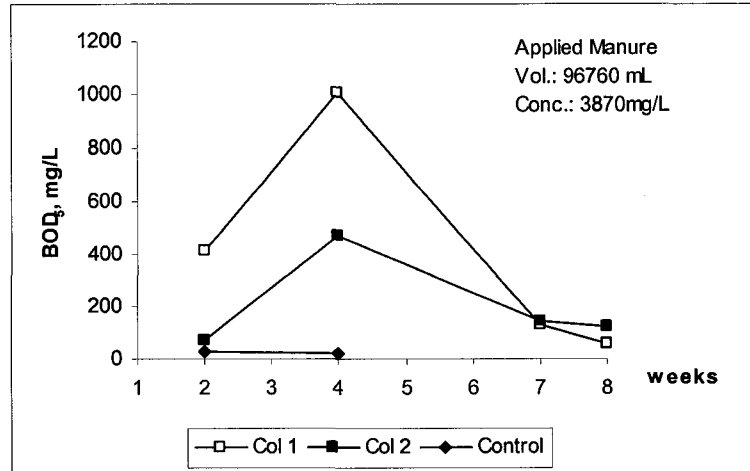


Figure 4-6  $\text{BOD}_5$  concentration in leachate from Phase I

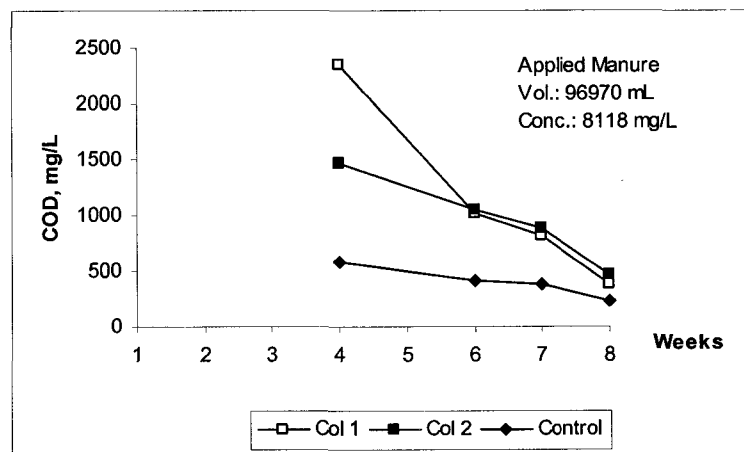
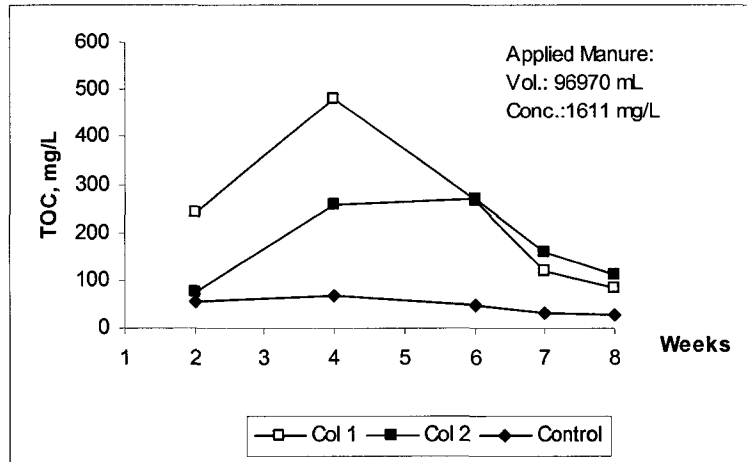
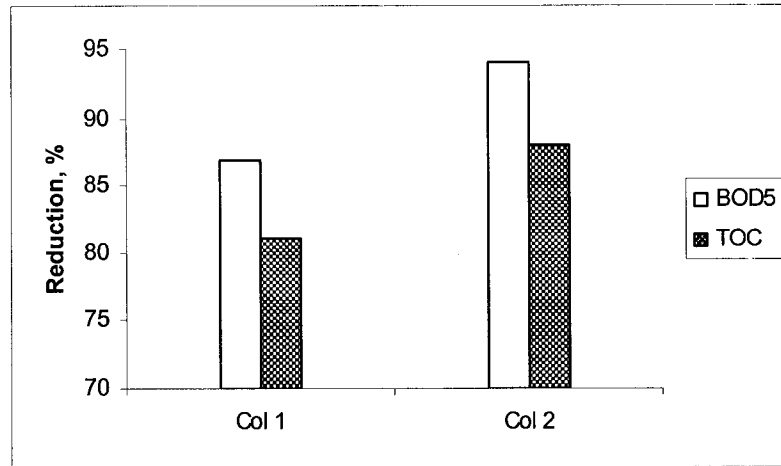


Figure 4-7 COD concentration in leachate from Phase I



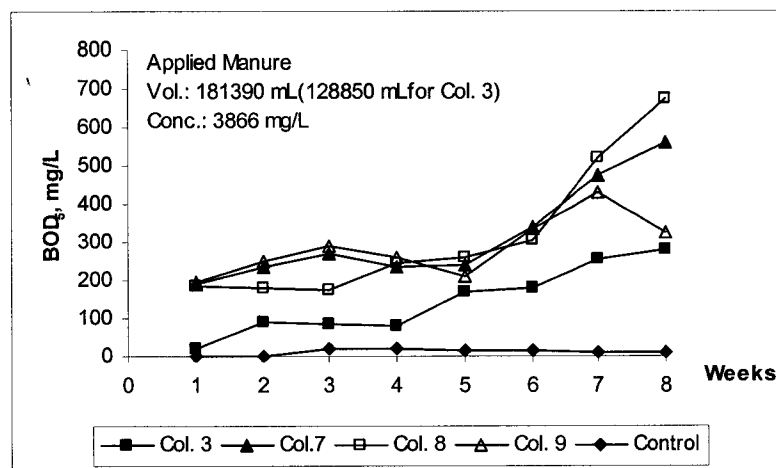
**Figure 4-8 TOC concentration in leachate from Phase I**

It is illustrated that the concentration of BOD<sub>5</sub> and TOC increased during the first few weeks and reached a summit during the fourth week. After that, both of the concentrations decreased until the end of Phase I. The BOD<sub>5</sub> and TOC mass reductions were about 90 and 85 percent, respectively, over the course of the eight week period (See Figure 4-9). Due to the lack of data, the variation of COD concentration in the leachate is only illustrated from the fourth week to the end of Phase I. COD had the highest value in the fourth week and then declined continuously to the end of the run. Based on the expressions of the BOD and TOC curves, it was reasonable to assume that the COD values were lower before the fourth week, meaning that the values for the fourth week were also the summit values in the Phase I experimental period.

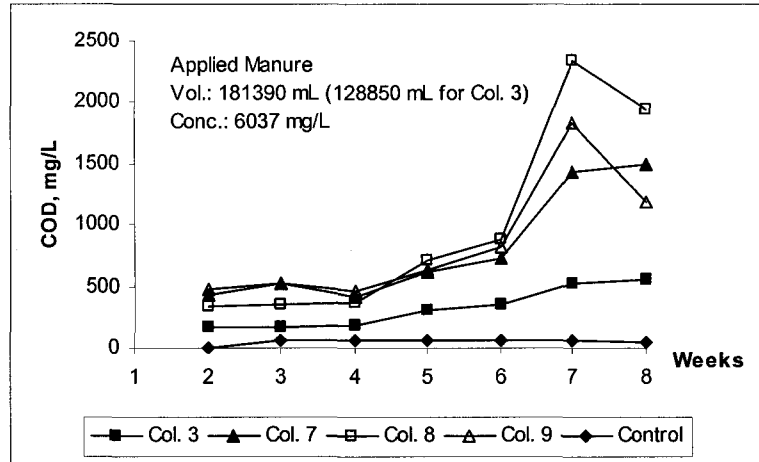


**Figure 4-9 Mass reduction of BOD and TOC in Phase I**

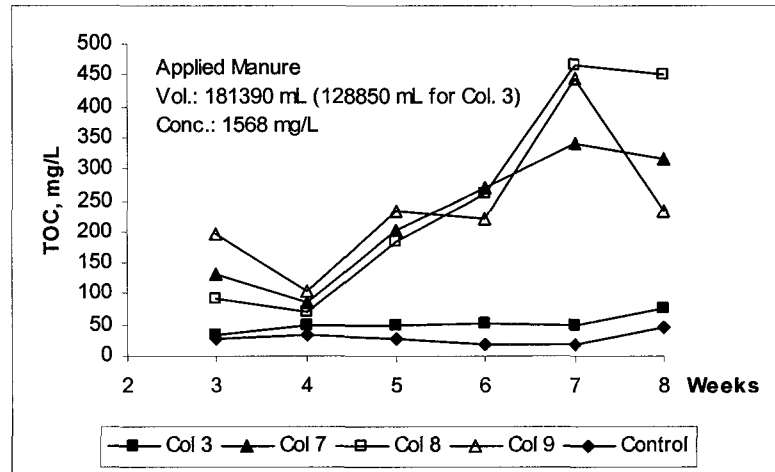
As mentioned before, the hydraulic loading for Phase III was 17 mm d<sup>-1</sup>, representing the organic loading of 0.065 kg BOD m<sup>-2</sup> d<sup>-1</sup>. The BOD<sub>5</sub>, COD, and TOC concentrations of leachate are illustrated in Figures 4-10, 4-11, and 4-12, respectively.



**Figure 4-10 BOD<sub>5</sub> concentration in leachate from Phase III**



**Figure 4-11 COD concentration in leachate from Phase III**



**Figure 4-12 TOC concentration in leachate from Phase III**

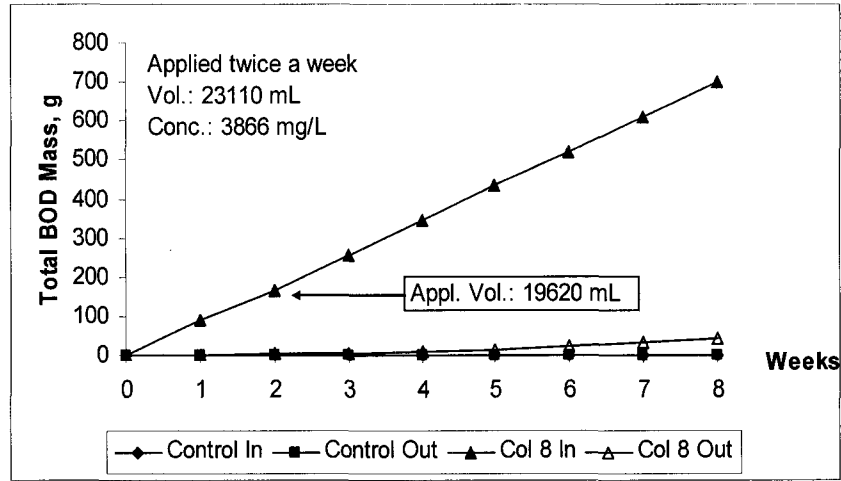
Although  $BOD_5$  and COD concentrations in the leachate increased significantly from the beginning to the end of the experiment, both concentrations remained much lower than that of pre-treated manure throughout the whole experiment. Figure 4-10 shows that from the first week (October 05, 2004) through to the sixth week (Nov. 16, 2004), the  $BOD_5$  in the leachates from columns 7, 8, and 9 was very similar. The highest  $BOD_5$  concentration during the six-week period in the different soil columns was  $340 \text{ mg L}^{-1}$ , which accounted for 8.8% of the pre-treated manure concentration. After the sixth week, a quite clear increase in  $BOD_5$  appeared, with the highest values of  $676 \text{ mg L}^{-1}$ , which

accounted for 17% of the pre-treated manure concentration. This trend of rapid increases weakened during the last week of the experiment. However, column 3, although it showed the same continuously increasing BOD<sub>5</sub> curve as the other columns, had a better treatment result relative to the other three. The highest BOD<sub>5</sub> concentration in column 3 leachate was 279 mg L<sup>-1</sup>, which was only 7.2% of the pre-treated manure concentration.

Figure 4-11 shows that the COD values for columns 7, 8, and 9 exhibited no obvious variation within the first four weeks (from Oct. 05 to Nov. 02, 2004). After that, COD increased continuously and reached the highest point at the end of the seventh week. The highest COD concentration was 2344 mg L<sup>-1</sup> among replicates that accounted for 38% of the pre-treated manure COD concentration. Similarly, the tendency of increasing COD concentration weakened during the last week of the experiment. Still, column 3 had the best COD removal result with the highest effluent COD concentration of 557 mg L<sup>-1</sup>, less than 10% of the COD concentration of the pre-treated manure.

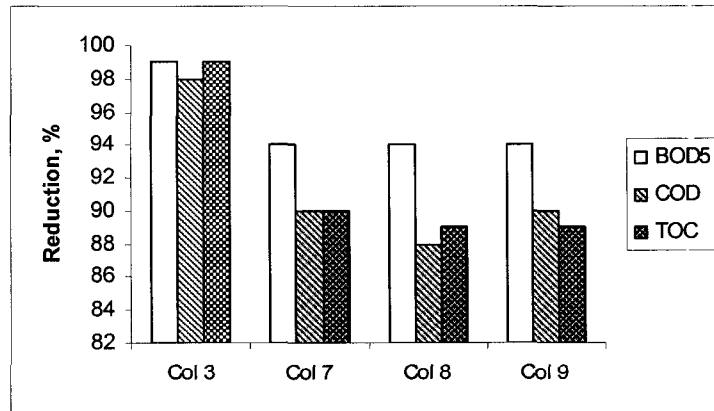
Due to problems with testing instruments, TOC analysis started two weeks after the beginning of manure application (Oct. 26, 2004). On the whole, the TOC concentration increased with the application time, but the values fluctuated significantly. Column 3 TOC remained at a very low level, with a final concentration of 75 mg/L. The highest final TOC concentration belonged to column 8, which reached 452 mg/L.

The cumulative mass of BOD in the manure and leachate is illustrated in Figure 4-13 (Taking Column 8 as an example, other values are shown in Appendix A). Compared to manure, the cumulative BOD mass in the leachate was quite low, indicating that most BOD in the manure was intercepted or removed by the soil. COD and TOC also showed similar results.



**Figure 4-13 Cumulative BOD mass for manure and leachate in Phase III (Col. 8)**

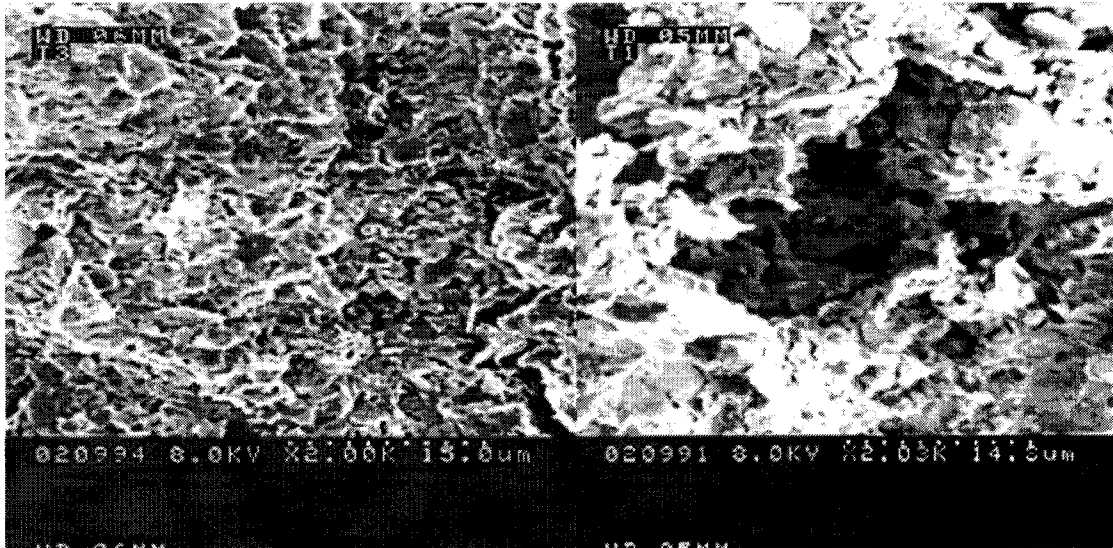
The total reduction in BOD, COD, and TOC mass during Phase III (eight week period) is shown in Figure 4-14; these results indicated that soils have a very good ability for pollutant removal. The values from columns 7, 8, and 9 are very consistent. The reductions in COD and TOC were around 89%, whereas the reduction of BOD<sub>5</sub> reached 94%. Column 3 had a better treatment result than did the other columns, with BOD<sub>5</sub>, COD, and TOC reductions of 99, 98, and 99%, respectively.



**Figure 4-14 Reduction efficiency of carbonaceous substrate for manure in Phase III**

The overall removal efficiencies of BOD<sub>5</sub>, COD, and TOC mass in Phases I and III were fairly close, which might suggest similar microorganism populations and activities

during the experimental period. The variation in soil structure is clearly shown in the SEM images of Phase I soils (Figure 4-15). The tighter structure of the soil after manure application resulted mainly from TSS accumulation and microorganism development, suggesting the microbe's contribution to substrate removal in soils.



**Figure 4-15 SEM image comparison of manured and unmanured soils**

Notes: the left image is the top soil image from column 1 after manure application in Phase I; the right image is the top soil image from the control column after tap water application in Phase I

Summit values for substrate concentrations in the leachate appeared in both Phase I and Phase III, possibly indicating that a certain incubation time was necessary for microorganisms to adapt to the manure characteristics, and then to effectively degrade the OM in the pre-treated manure. However, the summit values appeared at different times in Phase I (week 4) and Phase III (week 7). This difference might be due to the different operating conditions. The experiment in Phase III was carried out in a greenhouse that provided relatively consistent and warmer operating circumstances. Microorganisms developed very well, and faster than they did in Phase I, therefore attaining better treatment results in a relatively shorter time. Phase I was carried out in the field and so



was strongly affected by weather conditions. Due to the slower development of microorganisms, the effluent quality worsened quickly. The improvement in leachate quality was possibly due to the heavy precipitation after the fourth week, which resulted in the reduction of manure application loading. The second reason for the better leachate quality may be the further development of microorganisms in the soil with the passage of time.

In addition, microbe activity is not the only mechanism for pollutant removal in a soil system (Zhao *et al.*, 2004; Connolly *et al.*, 2004). For example, interception by soil particles, physical absorption, or adsorption can all contribute to substrate removal, especially during the beginning stages of the experiment. Pollutant substances might first be deposited or settle on the surface of soil particles and accumulate there due to continuous pollutant feeding. Simultaneously, part of the substrate is decomposed by microbes under aerobic conditions. Therefore, the decrease in substrate in the leachate was a combined result of microbe activities and physical actions in the soil matrix. When the accumulated substrate exceeded the soil's capacity, the non-decomposed substrate would leach out of the system and result in worse effluent quality. This might explain why the substrate concentrations in the leachate remained at the same level for about six weeks in during Phase III, and then increased at the end of the run. If the manure characteristics and experiment operation conditions are held constant, it is suggested that the leachate quality will remain at another level, which should also be relatively constant, but worse than before. All the reductions in substrate during this period are due to the activities of microorganisms. A longer experiment duration is suggested to test this hypothesis in the future.

Although the total removal percentages of BOD<sub>5</sub> and COD were very high (> 90% in mass) over the eight weeks of operation time, the highest values of BOD<sub>5</sub> and COD in the leachate, for example, in Phase III, reached up to 676 mg L<sup>-1</sup> and 2344 mg L<sup>-1</sup>. These were very high values and indicate that the leachate is not suitable for flushing water or direct discharge. Therefore, with regard to carbonaceous substance removal or utilization, longer soil columns or further land application is necessary.

#### 4.1.3.2 TKN and ammonia nitrogen

In Phase I, TKN and ammonia loadings were 0.025 and 0.024 kg N m<sup>-2</sup> d<sup>-1</sup>, respectively. Figures 4-16 and 4-17 show an increase-decrease trend for both TKN and NH<sub>4</sub>-N concentrations in the leachate; these variations were quite significant.

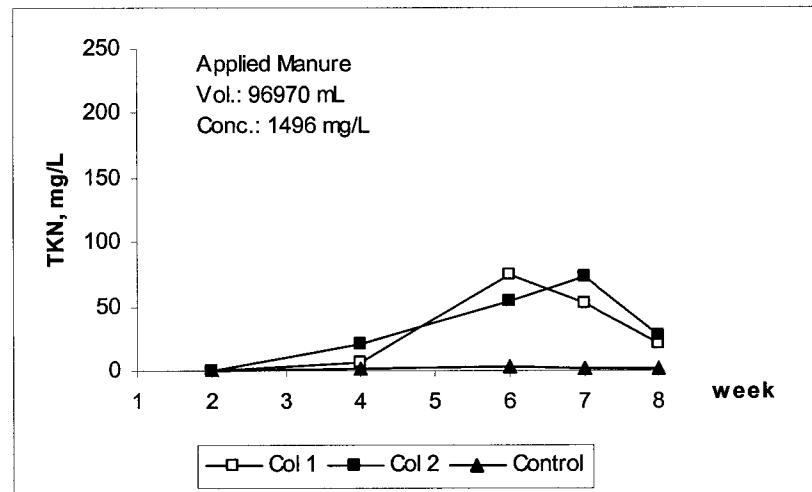
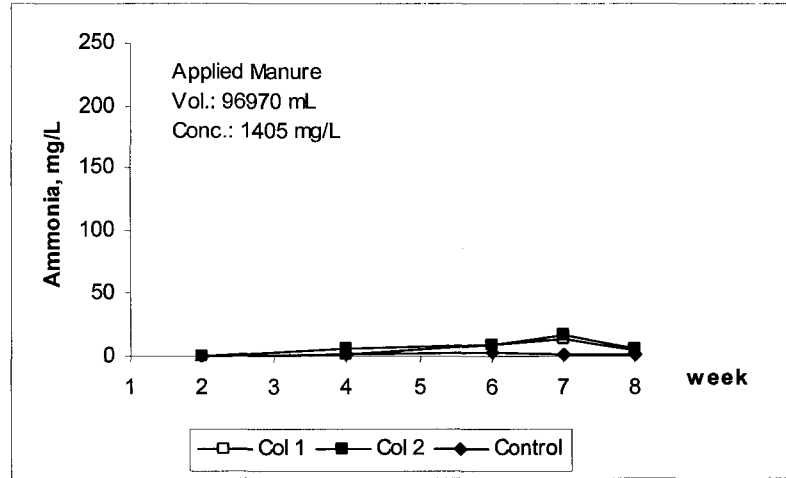
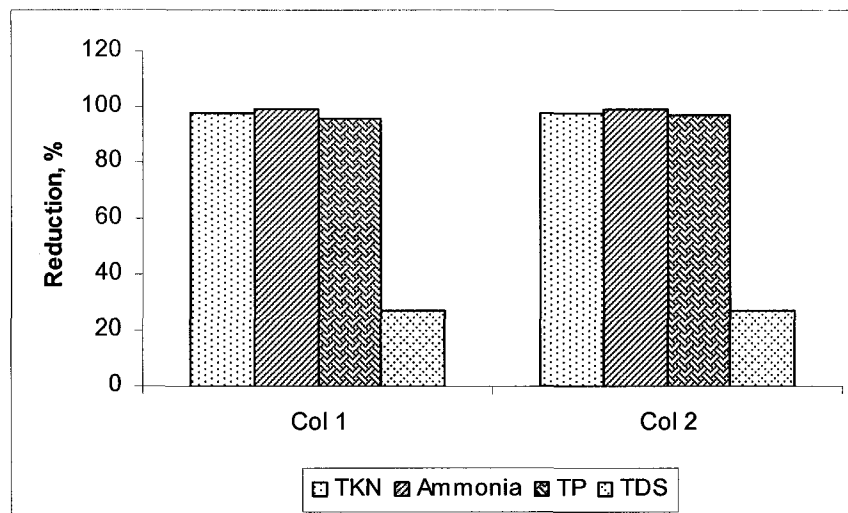


Figure 4-16 TKN concentration in the leachate from Phase I



**Figure 4-17 NH<sub>4</sub><sup>+</sup>-N concentration in the leachate from Phase I**

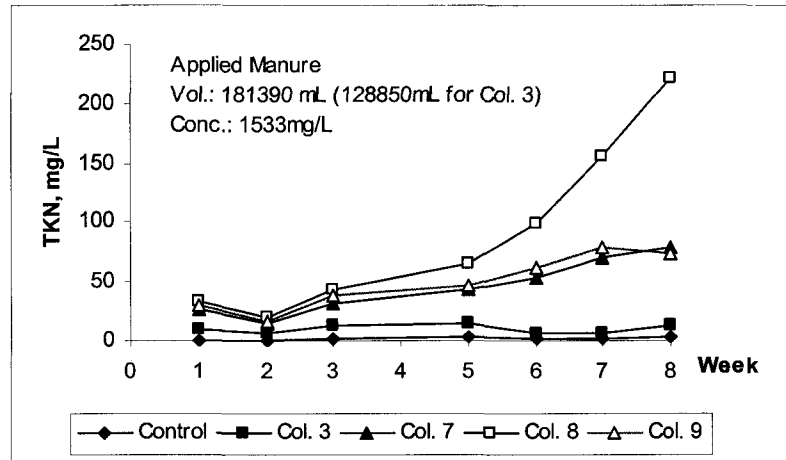
TKN concentrations reached summit values of 75 mg L<sup>-1</sup> in the sixth week (July 20, 2004) in column 1, and 73 mg L<sup>-1</sup> in the seventh week (July 27, 2004) in column 2, accounting for 4% of the TKN concentration in the pre-treated LSM. The final TKN concentrations in the effluents from columns 1 and 2 were 20 and 27 mg L<sup>-1</sup>, respectively. The NH<sub>4</sub><sup>+</sup>-N concentrations attained summit values of 14 and 17 mg L<sup>-1</sup>, respectively, in column 1 and column 2 in the seventh week. These values account for less than 1% of the original pre-treated LSM concentration. The final concentrations of NH<sub>4</sub><sup>+</sup>-N in columns 1 and 2 were only 5 and 7 mg L<sup>-1</sup>, respectively. The total mass reductions of TKN and NH<sub>4</sub><sup>+</sup>-N in Phase I were over 98% and 99% (See Figure 4-18) in both column 1 and column 2, indicating a very good nitrogen removal effect.



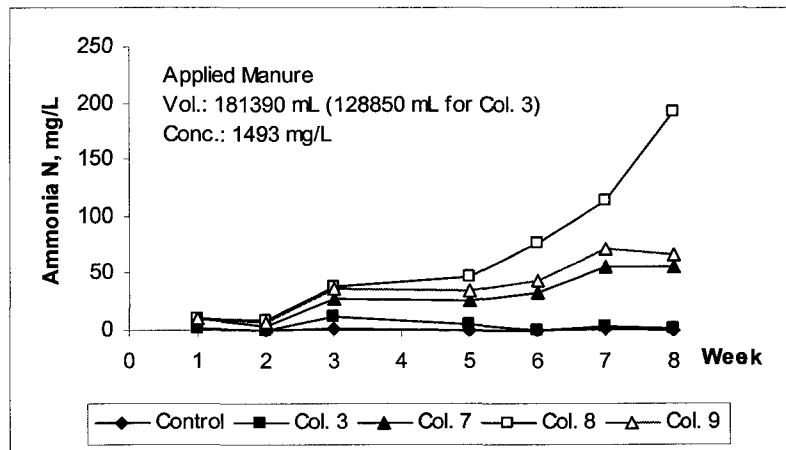
**Figure 4-18 Reduction efficiency of TKN,  $\text{NH}_4^+\text{-N}$ , TP, and TDS in Phase I**

In Phase III, the TKN and ammonia nitrogen loadings were  $0.026$  and  $0.025 \text{ kg N m}^{-2} \text{ d}^{-1}$ . Statistical analysis indicated that, from the beginning to the end of the experiment, TKN and  $\text{NH}_4^+\text{-N}$  concentrations in the leachate did not change significantly. However, Figures 4-19 and 4-20 illustrate an obvious increasing trend in these parameters for the leachate from column 8. Compared to column 8, the oscillation of TKN in the leachates from columns 7 and 9 were smooth. The highest TKN concentration in the effluents from columns 7 and 9 was  $79.60 \text{ mg L}^{-1}$ , twice the TKN measured on the first sampling day, but still only accounting for 5% of the pre-treated manure concentration. The dramatic increase in TKN in column 8 started during the fifth week and reached  $221 \text{ mg L}^{-1}$  by the end of the experiment, a value which was 5.6 times that obtained on the first sampling day and which accounted for 14% of the TKN in the pre-treated manure. The concentration of  $\text{NH}_4^+\text{-N}$  in each column showed a variation trend identical to that of TKN. The highest  $\text{NH}_4^+\text{-N}$  concentration was  $192 \text{ mg L}^{-1}$ , and was also from column 8. Neither TKN nor ammonia in the leachate from column 3 showed any increase with time.

The highest concentrations of TKN and ammonia nitrogen were 14 and 11 mg L<sup>-1</sup>, which were markedly lower than the values from the other columns.

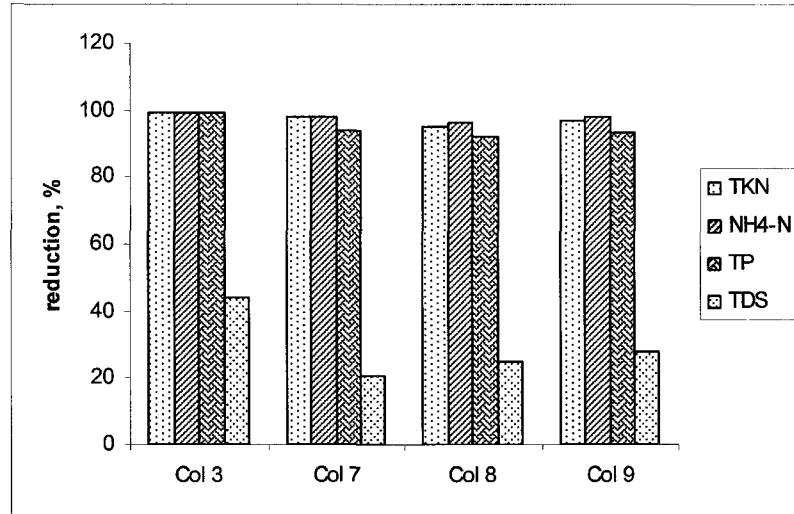


**Figure 4-19 TKN concentration in leachate from Phase III**



**Figure 4-20 NH<sub>4</sub><sup>+</sup>-N concentration in leachate from Phase III**

The total mass reductions in TKN and NH<sub>4</sub><sup>+</sup>-N during the eight week period are illustrated in Figure 4-21. The average reductions of TKN and NH<sub>4</sub><sup>+</sup>-N were 97 and 98%, respectively. Even column 8, which had the worst leachate quality in terms of nitrogen, achieved 95 and 96% reduction levels for TKN and NH<sub>4</sub><sup>+</sup>-N.



**Figure 4-21 Reduction efficiency of TKN, NH<sub>4</sub><sup>+</sup>-N, TP, and TDS in Phase III**

The fluctuations of the TKN and NH<sub>4</sub><sup>+</sup>-N concentrations in Phase I were attributed to the heavy precipitation after the fourth week, which promoted higher leaching of TKN and ammonia by washout. This conclusion was also supported by the facts that EC values in the leachate were highest during this period, and the soil ammonia content dropped significantly during the sixth week.

In Phase III, column 8 had much higher concentrations of both TKN and NH<sub>4</sub><sup>+</sup>-N at the end of the run. This was possibly due to the additional macropores and preferential flow channels formed in this column while the soil was being filled in, which resulted in more nitrogen content leaching.

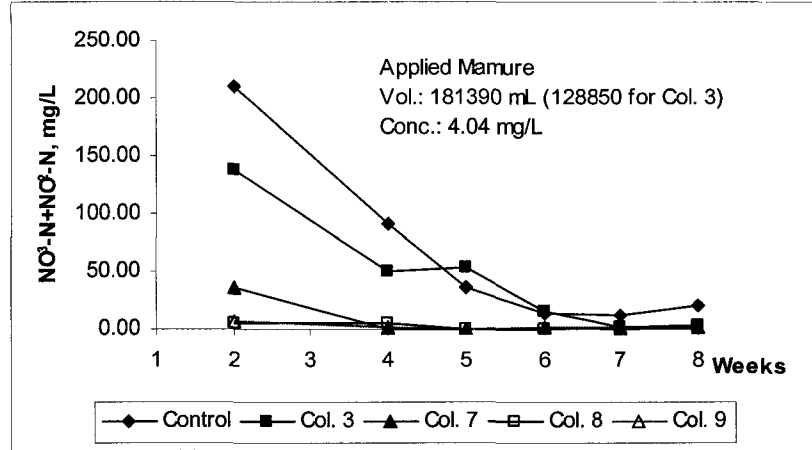
With respect to the removal effect of TKN and NH<sub>4</sub>-N, the soils in Phase I seemed to have a better ability than those in Phase III, because the summit values (75 and 17 mg L<sup>-1</sup> for TKN and NH<sub>4</sub>-N, respectively) in Phase I represented only the normal level in Phase III. However, the lower TKN and ammonia application loading in Phase I provided less nitrogen to the soil system; therefore, it is reasonable for the leachate to have lower nitrogen content than it did in Phase III. The second reason for the different performance

for nitrogen removal was that the columns in Phase I had worse filtration ability, allowing more particles and ion-groups to be deposited and intercepted in the soil profiles. The two reasons mentioned above could also explain why column 3 in Phase III had lower nitrogen content in its leachate than did the others. Column 3 was clogged since the fifth week, indicating that its filtration ability was comparatively poor. This clogging also resulted in the reduction of manure application (See Figure 4-2). In addition, the higher TKN concentration in the initial soil 2 (See Table 4-5) may also contribute to the higher concentrations in the leachates of columns 7, 8, and 9.

#### **4.1.3.3 Nitrate and nitrite nitrogen**

The concentration of  $\text{NO}_3^-$ -N plus  $\text{NO}_2^-$ -N was only detectable in the second week (June 22, 2004) in Phase I. In another words, the concentration of  $\text{NO}_3^-$ -N plus  $\text{NO}_2^-$ -N for other sampling dates below the method detection limit. This indicates that a very small amount of  $\text{NO}_3^-$ -N plus  $\text{NO}_2^-$ -N was leaching out of the system.

In Phase III, the  $\text{NO}_3^-$ -N plus  $\text{NO}_2^-$ -N concentration decreased as the experiment time continued (Figure 4-22). By the end of this experiment, the highest concentration among the replicates was  $2.98 \text{ mg L}^{-1}$ . Most  $\text{NO}_3^-$ -N plus  $\text{NO}_2^-$ -N concentrations in the leachates from the replicates remained at a low level, except for that of column 3, which had a high  $\text{NO}_3^-$ -N plus  $\text{NO}_2^-$ -N concentration at the beginning of the run, followed by a gradual decrease with time until the end of the eighth week, at which point the concentration had dropped to  $1.41 \text{ mg L}^{-1}$ , putting it at the same level as the other columns.



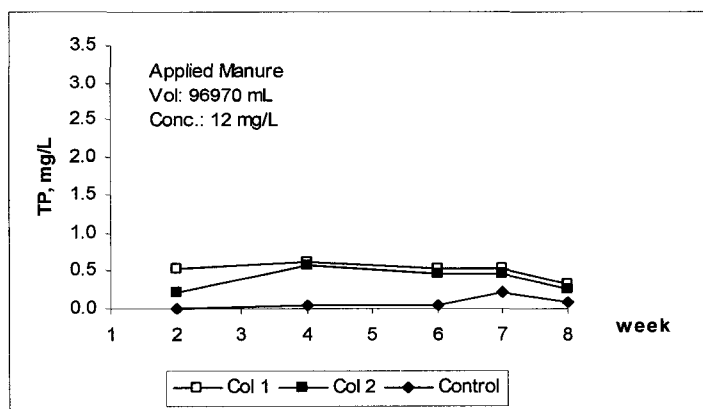
**Figure 4-22 Nitrate plus nitrite nitrogen concentration in the leachate from Phase III**

This variation in  $\text{NO}_3^-$ -N plus  $\text{NO}_2^-$ -N concentration in the leachate was due to variations in soil conditions. Before the manure application, the soils in the columns contained a certain amount of oxygen, which was stored during the process of soil digging, delivering, and filling. When manure application commenced, the soil system was under aerobic conditions that promoted the nitrification process and generated nitrate and nitrite. Due to the negative charge of the two groups, they were repulsed by the soil clay particles (also negative charged) and easily leached out of the soil systems. The  $\text{NO}_3^-$ -N plus  $\text{NO}_2^-$ -N in the leachate also included the amount of  $\text{NO}_3^-$ -N plus  $\text{NO}_2^-$ -N in the original soil. Because of the tillage before the manure application, the topsoil maintained the aerobic condition, contrary to the subsoil which, due to oxygen consumption, gradually assumed an anaerobic condition. The anaerobic conditions promoted the denitrification that converted nitrate or nitrite generated in the top soil into  $\text{N}_2\text{O}$  or  $\text{N}_2$ , which was emitted out instead of leaching out of the system.



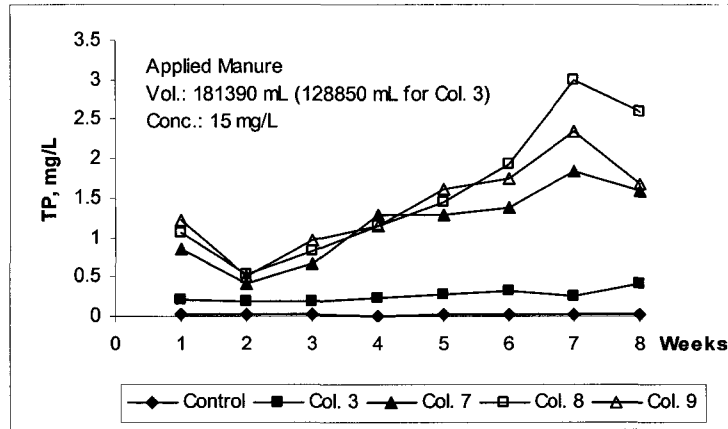
#### 4.1.3.4 Total phosphorus

The phosphorus loading in Phase I was  $0.21 \text{ g P m}^{-2} \text{ d}^{-1}$ . Within the eight week manure application period, the TP concentrations in the leachate were all much lower than the initial value, and the differences between concentrations among the different sampling dates was statistically insignificant (See Figure 4-23). The total average reduction in mass during the eight weeks was over 96% (See Figure 4-18).



**Figure 4-23 TP concentration in leachate from Phase I**

In Phase III, the phosphorus loading was  $0.25 \text{ g P m}^{-2} \text{ d}^{-1}$ . TP concentrations in the leachate for different sampling days are shown in Figure 4-24, which illustrates an increasing trend, especially in columns 7, 8, and 9. By October 23, 2004 (seventh week), the highest TP concentration was observed in the leachate from column 8. The value was  $2.99 \text{ mg L}^{-1}$ , which accounted for 20% of the TP concentration in the pre-treated LSM. TP concentration in columns 7, 8, and 9 showed a tendency to decrease during the last week of the run.

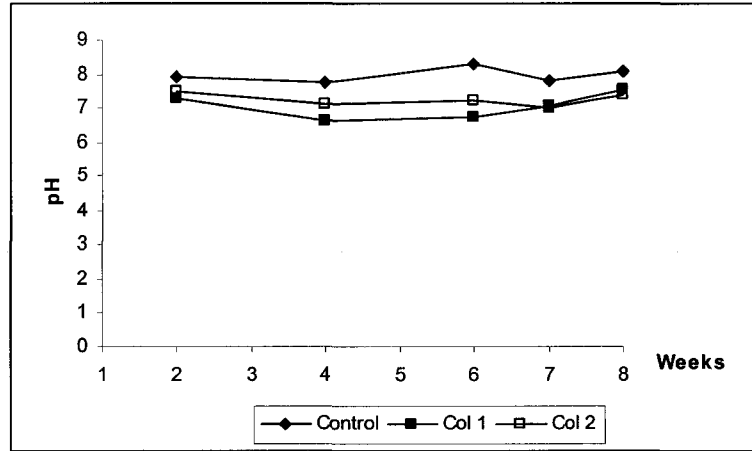


**Figure 4-24 TP concentration in the leachate from Phase III**

The reductions in TP in Phase III is presented in Figure 4-21, which indicated a very good total phosphorus mass removal through the soil profiles (over 92%). However, the continuous increase in effluent concentration in Phase III might suggest that the phosphorus in the soils was reaching saturation. Like nitrogen content, the TP content (2.99 mg L<sup>-1</sup> being the highest value) in the leachate can be taken up by crops through further land application.

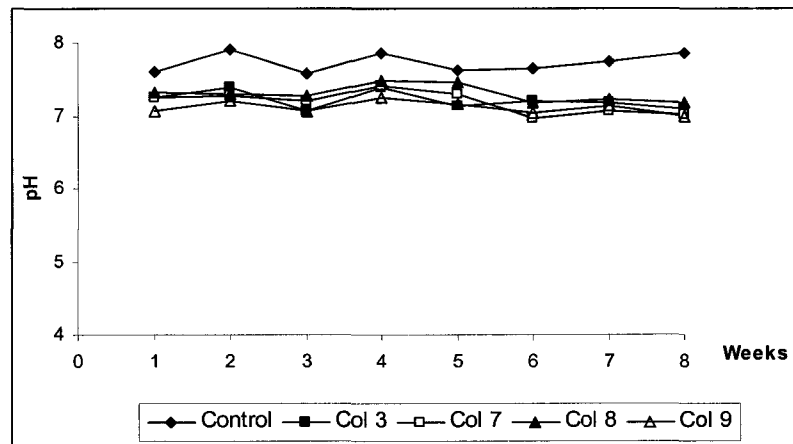
#### 4.1.3.5 pH

The pH of the leachate in Phase I had a constant value, which varied in the range of 6.64 to 7.49 (See Figure 4-25). Compared to those obtained from the pre-treated LSM, the pH values in the leachate decreased.



**Figure 4-25 pH value in leachate from Phase I**

Similar to Phase I, from the beginning through to the end of the eight week experimental period, the leachate in Phase III remained at a constant pH value across all the columns (See Figure 4-26). The pH range of the column effluent was 6.97 to 7.48, indicating the neutral condition of the effluent. Statistically, there was no significant change in pH values in the leachate throughout the eight week period, but there was a significant decrease when compared to the applied pre-treated manure.



**Figure 4-26 pH value in leachate from Phase III**

The pH value being lower in the leachate than in the pretreated manure indicates that some alkalinity was neutralized, which shows that nitrification occurred in the soils.

Normally, the nitrification reaction is shown as follows:



Hydrogen ions were generated during this process.

#### 4.1.3.6 EC and TDS

In Phase I, EC values in the liquid dropped dramatically from 11.65 in the manure to less than 3.5 in the leachate after soil filtration. But EC values in the leachate oscillated during the eight week manure application period, and there were significant differences among the results obtained on different sampling dates (Figure 4-27), although they were all much lower than the initial values. The EC values in the leachate from Phase I were lower than those in Phase III.

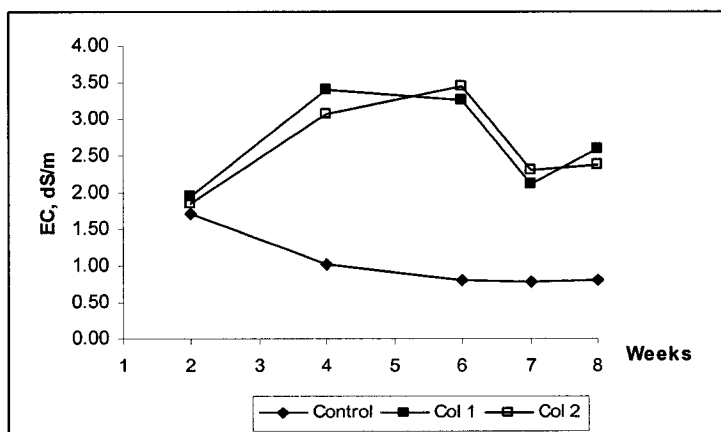
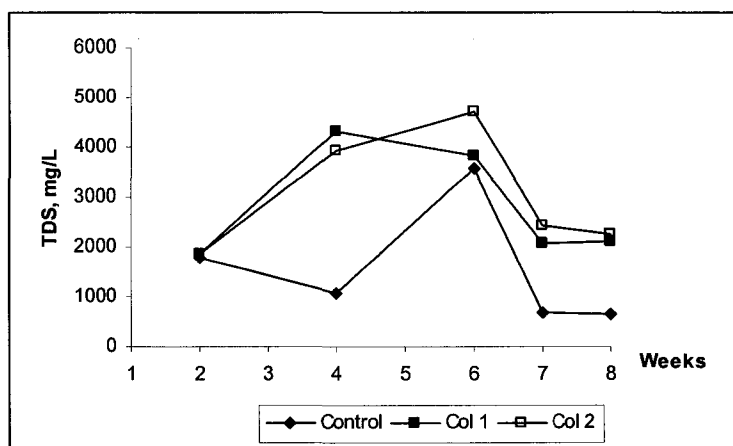


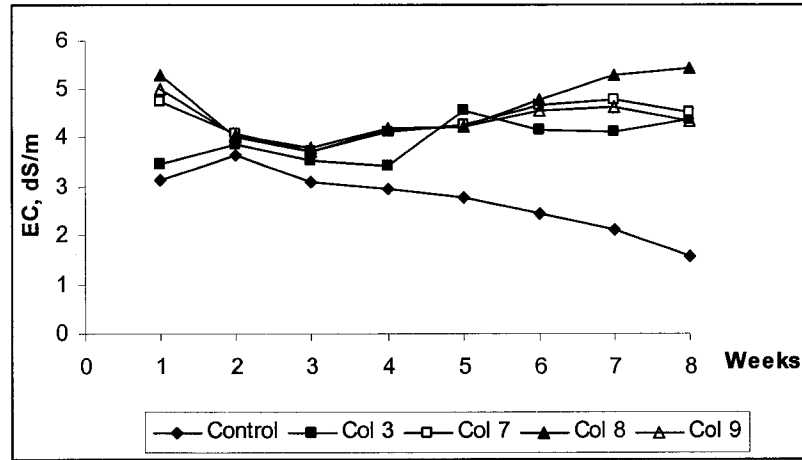
Figure 4-27 EC values in the leachate from Phase I

TDS variation with time is shown in Figure 4-28, which illustrates a similar trend as EC in columns 1 and 2, implying that there is a correlation between them. The mass reduction in TDS was around 25% (See Figure 4-18).



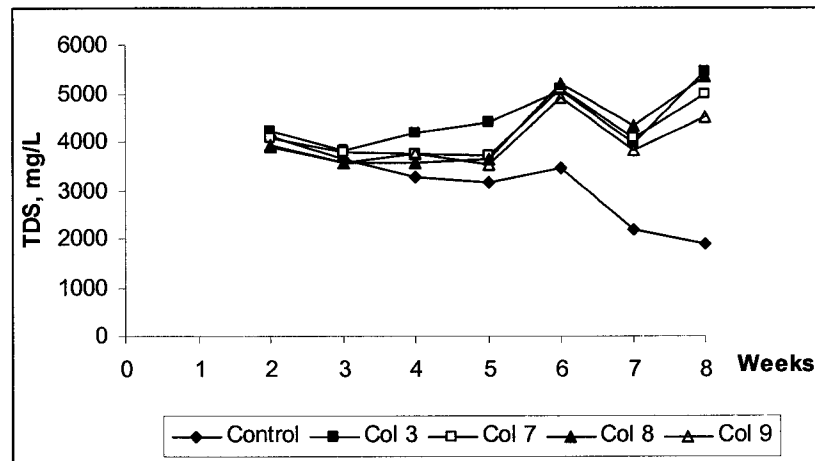
**Figure 4-28 TDS concentration in the leachate from Phase I**

In Phase III, EC values in the leachate were also much lower than those obtained from the pre-treated manure, dropping from 11.75 to 3 - 6 dS m<sup>-1</sup>. This decrease was significant and the reduction was about 66%, indicating the soils' effective salt interception abilities. Figure 4-29 shows the EC variation in the leachates from different columns in Phase III. Within the eight weeks of application, no significant variation in EC values was observed in the leachate; the highest EC was 5.26 dS m<sup>-1</sup> and the lowest EC was 3.41 dS m<sup>-1</sup>.



**Figure 4-29 EC value in the leachate from Phase III**

Figure 4-30 illustrates the variation in TDS with time. As was the case with EC, TDS values in the leachate showed no significant changes between the beginning and the end of the experiment. Contrary to EC, the TDS in the liquid exhibited no significant change before entering and after exiting the soil profiles.

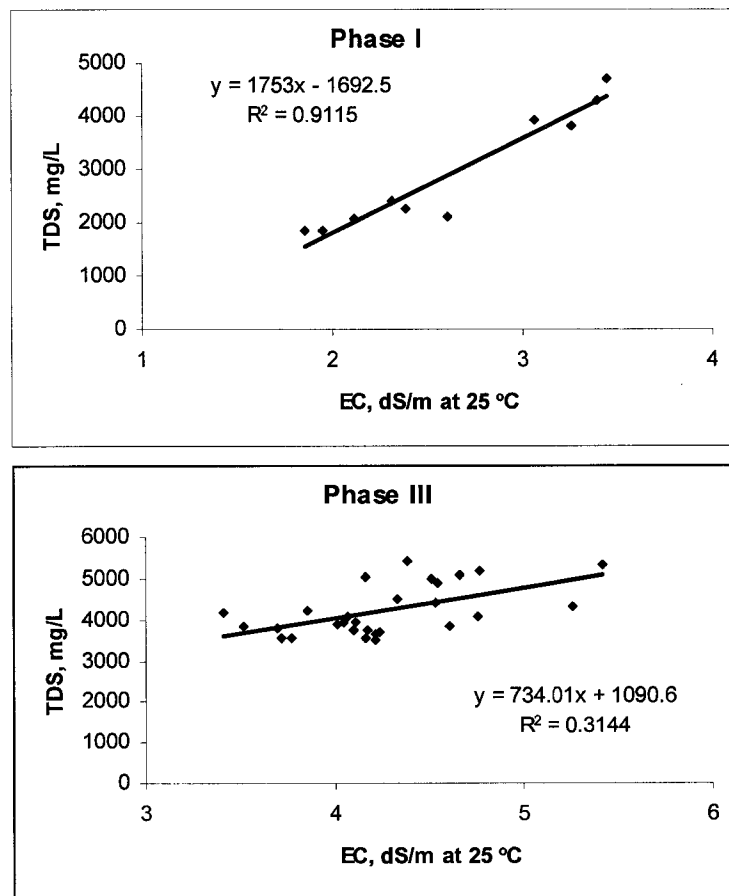


**Figure 4-30 TDS concentration in the leachate from Phase III**

TDS mass reduction is illustrated in Figure 4-21. Most of the reduction efficiencies were between 20 to 30%, except for column 3, which had a reduction of 44%.

The electrical conductivity of water is a measure of the ability of a solution to conduct an electrical current. Because the electrical current is transported by the ions in

the solution, the conductivity increases as the concentration of ions increases. Normally, a fixed correlation between EC and TDS can be established through experiment. Therefore, the measured EC value can be used as a surrogate measure of TDS. However, the above relationship is not suitable for every solution, especially for high intensity wastewater. In this experiment, the relationship of EC and TDS in the leachate was investigated and the results are illustrated in Figure 4-31.



**Figure 4-31 Relationship between TDS and EC values**

In Phase I, EC correlated very well with TDS in the leachate ( $R^2 = 0.91$ , See Figure 4-34). This correlation can also be observed from the shape of the EC and TDS curves in Figures 4-27 and 4-28. However, this well-correlated relationship between EC and TDS was not observed during Phase III ( $R^2 = 0.31$ ). Actually, the TDS concentrations in the

manure in this period did not change significantly before entering and after exiting the soil profiles, whereas the EC values changed dramatically. Clay colloid might be the reason for the non-correlation between TDS and EC values in Phase III. By definition, the solids contained in the filtrate that passes through a filter with a nominal pore size of 2.0  $\mu\text{m}$  or less are classified as dissolved (APHA, 1995). The diameter range of clay colloid in soils ranges from 0.002 to 2  $\mu\text{m}$ , which means that the clay colloid from soil profiles can stay in the filtrate of the leachate that pass through the filters, and consequently, can then contribute to the TDS result. In another words, part of the TDS in the leachate was not from the manure, but from the soils. Another possible reason was the organic substances formed during the manure application period. Those organic substances might contribute to the TDS value but not contribute to EC value. In addition, *E. coli*, having a size of 0.6-1.2  $\mu\text{m}$  (Metcalf and Eddy, 2003), might also the other reason. But the possibility was quite small. The difference in performance between Phases I and III might be due to the different degree of soil profile uniformity in the columns and different chemical conversions in the soil matrix. That is, more channels existed in the soil profile in Phase III, providing a pathway out of the system for the clay colloid.

## **4.2 Soil response**

### **4.2.1 Characteristics of initial soil**

Soils used in this project were taken directly from the University of Alberta farm. However, due to changes in the experimental design, the amount of soils taken for Phases I and II was not sufficient for Phase III, and extra soils had to be obtained. This resulted



in possible differences between the two soils which were filled in columns 1, 2, 3 (named soil 1) and columns 7, 8, 9 (named soil 2), respectively, in Phase III.

The particle and chemical contents of the two soils were analyzed and the average results are tabulated in Tables 4-4 and 4-5, respectively. Table 4-4 shows that the top soils of both soils 1 and soil 2 were clay loam, and that the sub soils of soil 1 and soil 2 were loam and clay loam, respectively. In terms of particle content, statistical analysis indicated that the two soils were the same ( $p > 0.05$ ), except for the clays in the sub soils ( $p = 0.01$ ). With respect to the chemical content, there were no significant differences in the TKN, TP, and pH of the two top soils (Table 4-6).

**Table 4-4 Particle content in soils used in Phase III**

Soil		Clay	Silt	Sand
Soil 1	Top	32.31	36.28	31.42
	Sub	11.43	40.41	48.18
Soil 2	Top	37.00	37.43	25.58
	Sub	29.28	31.94	38.79

**Table 4-5 Chemical content in soils**

Soil	pH	EC ds/m @25°C	TKN g/kg soil	TP g/kg soil	NH <sub>4</sub> -N mg/kg soil	NO <sub>3</sub> -N mg/kg soil	NO <sub>2</sub> -N mg/kg soil	Moisture %	SAR meq/L	
<b>Soil in Phase I</b>										
	Top	8.25	0.86	4.2	1.7	1.89	21.4	0.26	25.41	0.91
	Sub	8.27	0.90	1.0	0.6	1.98	34.73	0.29	34.10	0.82
<b>Soil in Phase III</b>										
Soil 1	Top	7.57	1.70	4.3	1.4	0.47	94.29	0.21	29.66	1.18
	Sub	7.6	1.04	1.5	0.5	0.04	4.14	0.00	16.02	0.53
Soil 2	Top	7.38	0.55	5.1	0.8	1.65	21.96	0.29	24.19	0.78
	Sub	7.49	1.50	1.7	0.5	1.69	8.54	0.40	10.84	0.43

**Table 4-6 p-value of chemical parameters to compare soils 1 and 2**

Soil	pH	EC	SAR	TKN	TP	Nitrate N
Top	0.28	0.01	0.04	0.13	0.05	0.004
Sub	0.16	0.27	0.04	0.08	0.01	0.02

Particle size contents and statistical analysis indicates that the main difference between the two soils was the clay content in the subsoil, which was 11.43% in soil 1 and 29.28% in soil 2. Based on this, soil 1 should have had better filtration ability (Lam *et al.*, 1993). However, in reality, the columns filled soil 2 expressed better filtration ability. The reason for this might be the initial soil moisture content. It is an easily observable fact that wetter soil is easier to compress. Soil 2 had a lower initial moisture content (topsoil 24.19%; subsoil 10.84) than soil 1 (topsoil 29.66%; subsoil 16.02), and therefore was not compressed as tightly as was soil 1. Some macropores and preferential flow channels were formed during the soil fill process, which would influence the subsequent experimental results.

The main chemical differences between the two soils were nitrate in both the top and sub soils, phosphorus in the subsoil, and EC in the top soil. Nitrate might affect the quality of the leachate and denitrification in the soil profiles. The extent of phosphorus saturation in the soil might result in different phosphorus leaching. No report showed the significant influence of EC on substrate and nutrient removal.

## **4.2.2 Soil variation after manure application**

### **4.2.2.1 Nitrogen**

#### **4.2.2.1.1 Nitrogen budget**

Nitrogen balances were conducted in Phase III to help understand nitrogen variation or conversion in the soil column system. The balance was based on the following equation.

$$(\text{In} - \text{Out}) - \text{Unaccounted amount} = \text{Storage} = \text{Final soil} - \text{Initial soil}$$

The balance results of different columns are shown in Table 4-7. The details of the nitrogen balance calculation are shown in Appendix B.

**Table 4-7 Nitrogen balance in soil column system in Phase III (Unit: g)**

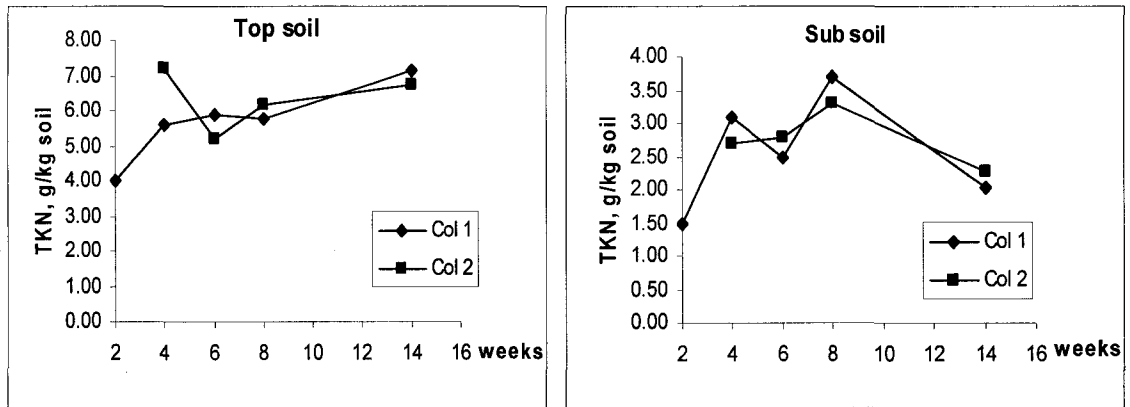
	Control	Col 3	Col 7	Col 8	Col 9
Final Soil	374	537	606	652	798
Initial Soil	427	427	531	531	531
Storage	-53	110	74	121	267
Manure In	0	198	279	279	279
Leachate Out	11	6	8	13	7
Unaccounted	42	82	197	145	5

Table 4-7 shows that nitrogen was lost from the soil column system during the manure application period, indicating that ammonia volatilization or denitrification occurred in the soils. This will be discussed in detail in the following sections.

#### **4.2.2.1.2 TKN and ammonia nitrogen**

TKN variations in Phases I and II (fourteen weeks) are illustrated in Figure 4-32. All the TKN values in columns 1 and 2 during Phase I (eight weeks) were greater than those in the initial soils (See Table 4-5), indicating that nitrogen accumulated in soils after the manure application. An increasing tendency is illustrated with oscillated curves through the Phase I period in both the top and sub soils. This oscillation is a consequence of the heavy rain that resulted in the reduction of manure application loading.

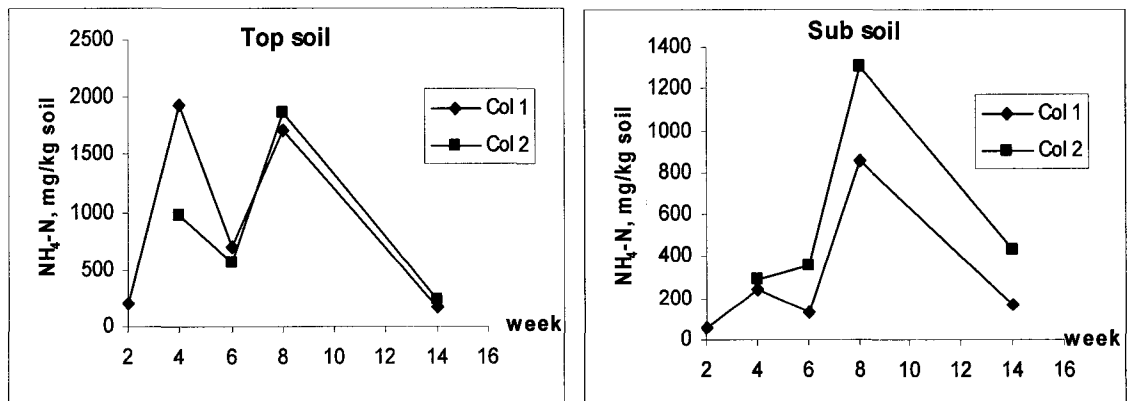
TKN values in the top soils were observed increasing slightly during the rest period (Phase II), but this increase was insignificant ( $p=0.08$ ). In the sub soils, on the contrary, TKN values decreased significantly during the rest period.



**Figure 4-32 TKN content variation with time in Phase I and Phase II**

A similar trend was observed for ammonia variation in the soil and is shown in Figure 4-33. Ammonia nitrogen values oscillated in both the top and sub soils throughout Phase I; the absolute values were all much greater than those from the initial soils, indicating the accumulation of ammonia nitrogen in the soil after manure application. This also implied that the TKN variation was largely dependent on the ammonia increase resulting mainly from the pre-treated swine manure.

Ammonia content declined significantly in both top and sub soils after the six week rest period (Phase II, See Figure 4-33), demonstrating the conversion of nitrogen forms.



**Figure 4-33 NH<sub>4</sub>-N variation with time in Phase I and Phase II**

Figure 4-34 illustrates TKN distribution in the soil matrices of four columns after eight weeks of manure application in Phase III. The top soils in all replicates showed a very significant increase in TKN concentration, especially at a depth of 0 – 15 cm. The highest TKN content in this layer, after manure application, was 7.8 g kg<sup>-1</sup> dry soil in column 9, an increase of 53% over the initial soils. However, the increase in TKN found in the top soils did not occur in the sub soils. Actually, there was almost no change in sub soil TKN content. In fact, some of the sub soils exhibited lower values. Statistical analysis indicated that a significant change in TKN content only happened in layers 1 and 2 in the top soils.

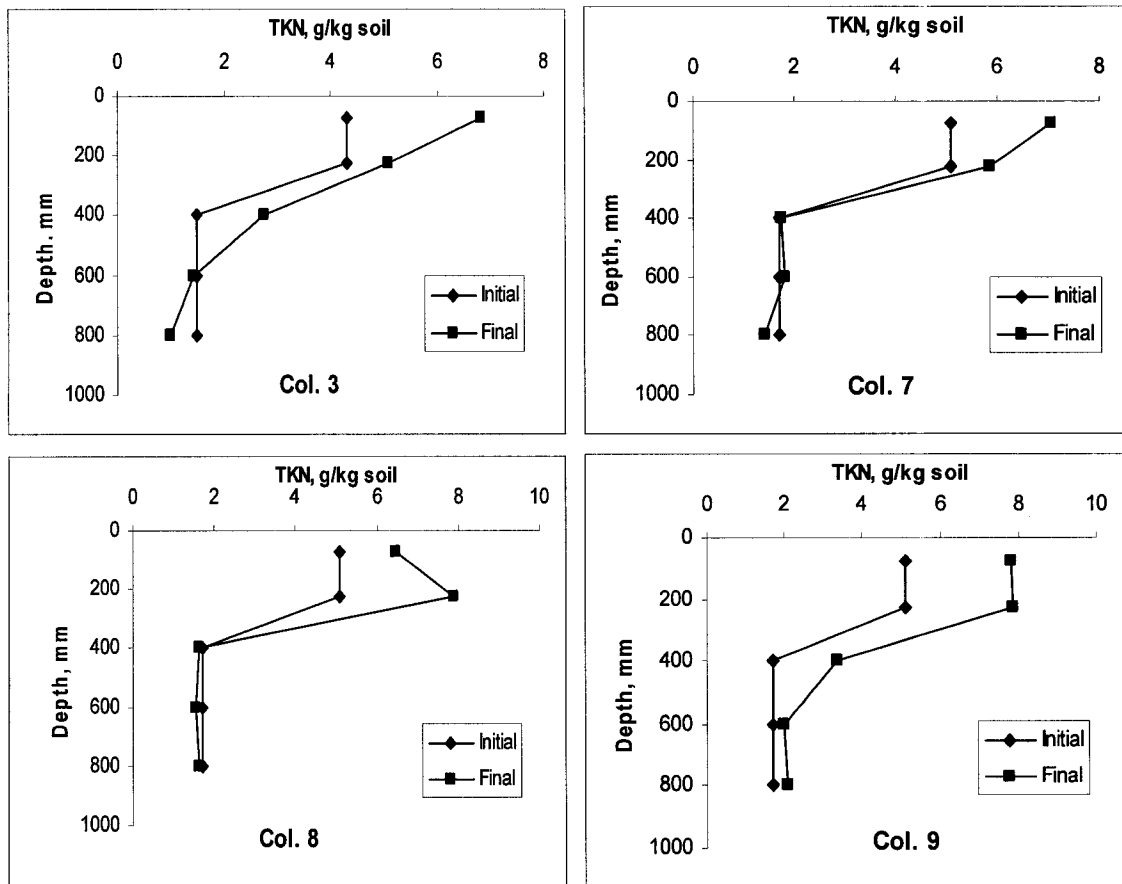


Figure 4-34 TKN contents in manured soils of Phase III

The ammonia nitrogen content in the soils demonstrated a variation trend similar to that of TKN, as is illustrated in Figure 4-35. Compared to the initial soils, the increase in ammonia N in the top soils was dramatic - from almost nothing to over 1000 mg kg<sup>-1</sup> dry soil. The highest value reached was over 2200 mg kg<sup>-1</sup> soil. An increase in ammonia N also occurred in the sub soils, but not to the same extent as in the top soils. Statistical analysis showed the significant changes in ammonia content in layers 1, 2, and 4.

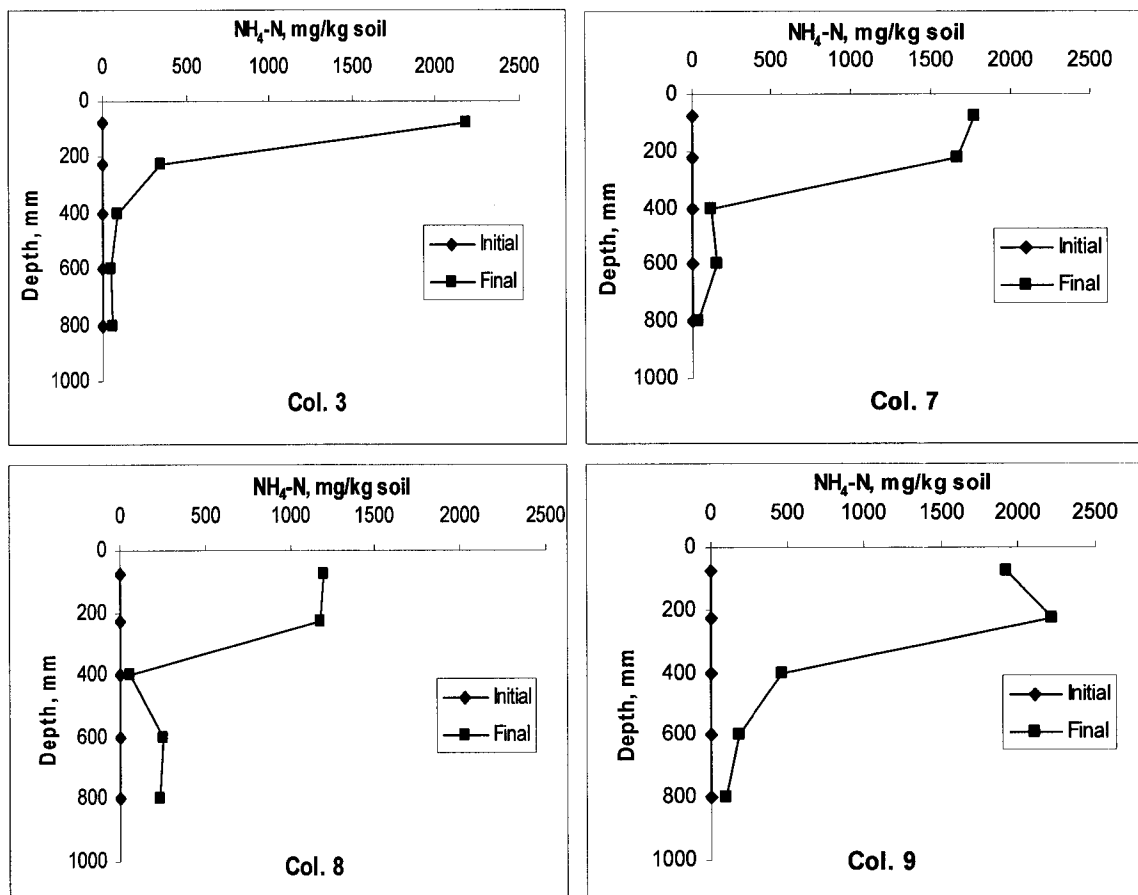
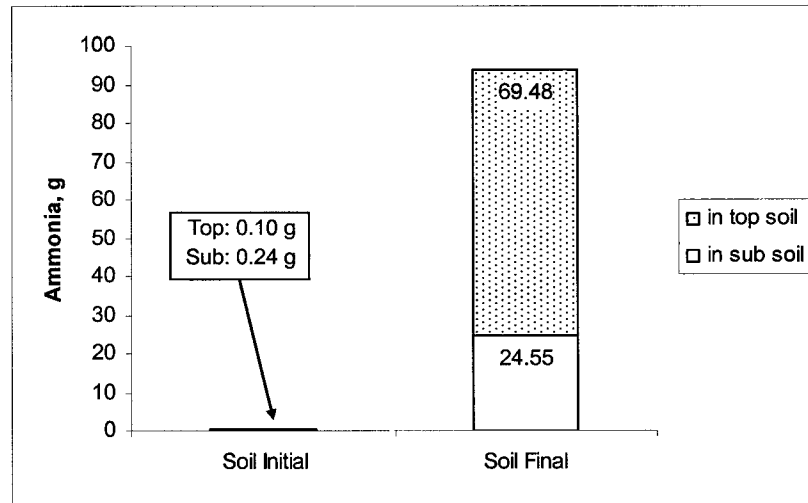


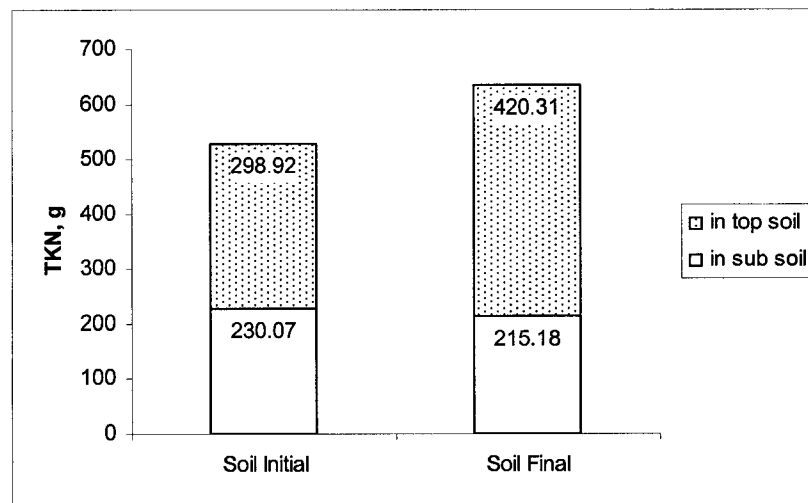
Figure 4-35  $\text{NH}_4\text{-N}$  content distribution in manured soil from Phase III

The significant increases in TKN and ammonia in the top soils were mainly due to the accumulation of nutrients coming from the manure application, indicating that soil interception and adsorption were some of the mechanisms by which TKN and  $\text{NH}_4\text{-N}$  were removed from the manure.

Although manure application also increased  $\text{NH}_4^+\text{-N}$  in the subsoil in Phase III, this increase was very limited compared to that occurring in the top soil. Taking column 8 as an example (See Figure 4-36), as it had the highest increase in ammonia nitrogen in the subsoil among the four replicates, the ammonia increase in the top soil was approximately three times that observed in the subsoil. The TKN content in the subsoil was even lower than that of the initial soil in column 8 (See Figure 4-37).



**Figure 4-36 Ammonia variation in soil (Col. 8)**



**Figure 4-37 TKN variation in soil (Col. 8)**

The first possible reason was that, during Phase III, the top soil intercepted or adsorbed most of the nitrogen in the manure; the second reason was that the subsoil had worse adsorption ability or had macropores or preferential channels in the soil profiles which resulted in TKN and ammonia leaching. Macropores and preferential flow channels can impact the leaching rate of solutes in agricultural soils (White, 1985). This phenomenon was in agreement with the leachate quality, which was better in Phase I than in Phase III. The increase in ammonia and decrease in TKN in the subsoil means that a conversion of the form of nitrogen took place. In other words, part of the ammonia was converted into  $\text{NO}_3$ ,  $\text{NO}_2$ , or gaseous nitrogen through nitrification and denitrification.

#### 4.2.2.1.3 Nitrate and nitrite nitrogen

After eight weeks, 12 mm  $\text{d}^{-1}$  pre-treated manure application did not significantly change the nitrate plus nitrite nitrogen contents in either the top soils or the subsoils (See Figure 4-38). In Phase II, a dramatic increase in nitrate plus nitrite nitrogen was observed in the top soils and the values reached up to 458 and 369  $\text{mg kg}^{-1}$  in columns 1 and 2, respectively, which were much higher than the 22  $\text{mg kg}^{-1}$  value measured in the initial soils.

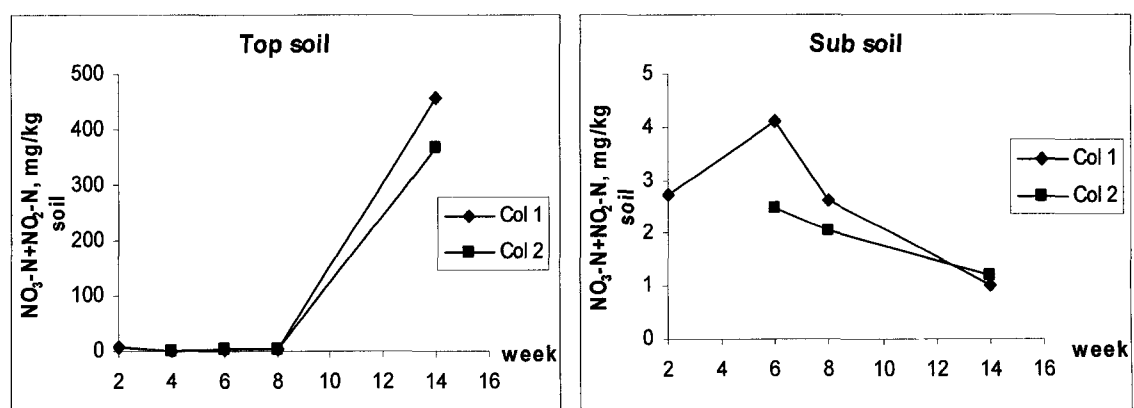


Figure 4-38  $\text{NO}_3\text{-N} + \text{NO}_2\text{-N}$  content variation with time in Phase I and Phase II



In Phase III, the variation trends of nitrate plus nitrite nitrogen were quite dissimilar. In columns 8 and 9, nitrate and nitrite nitrogen concentration was greater at the 0-30 cm depth with eight weeks of manure application, but the values in the two columns were quite different (See Figure 4-39). Higher nitrate plus nitrite were also observed in the sub soils of column 8. In column 7, nitrate plus nitrite was only greater at a depth of 0-15 cm. Column 3, as a special case, had lower final values at all depths.

There was a significant decline in nitrate plus nitrite content from 0-15 cm to 15-30 cm soil depth, indicating a soil condition change in the top soils. However, no significant change was found among the three layers in the sub soils.

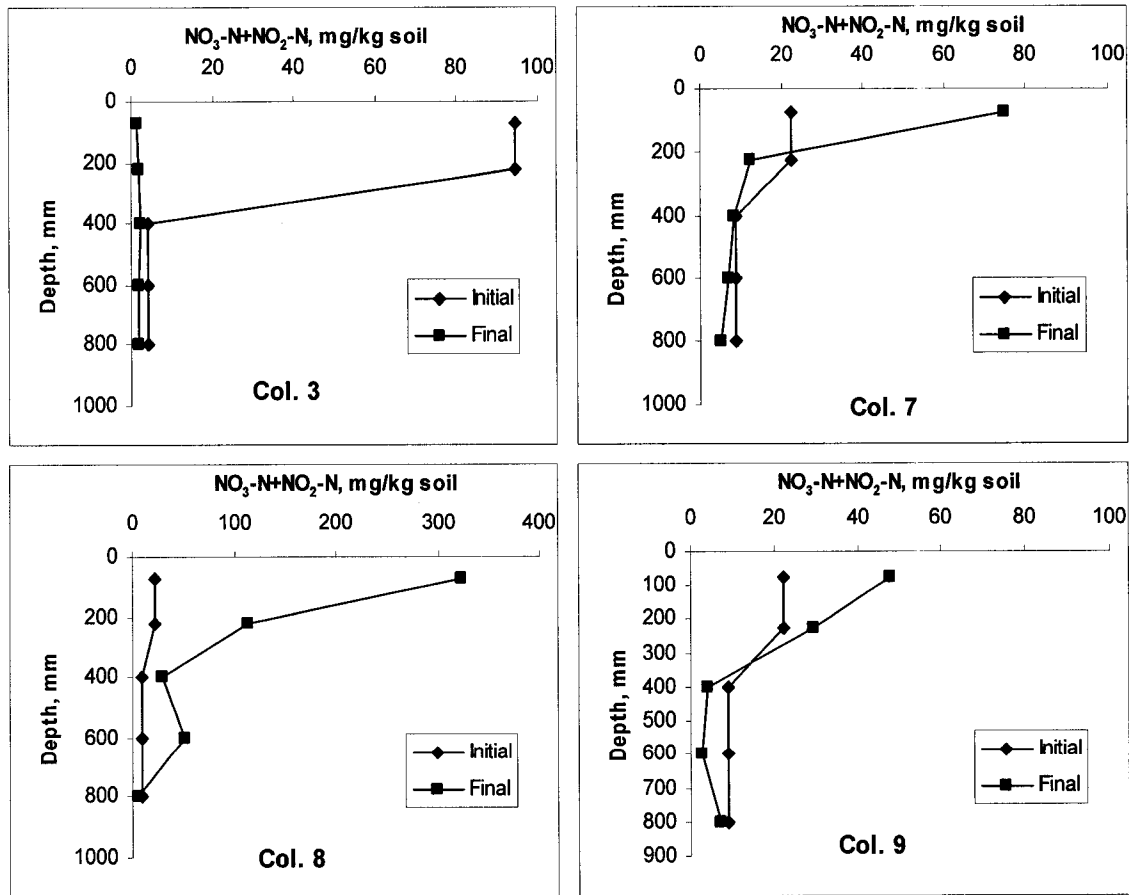
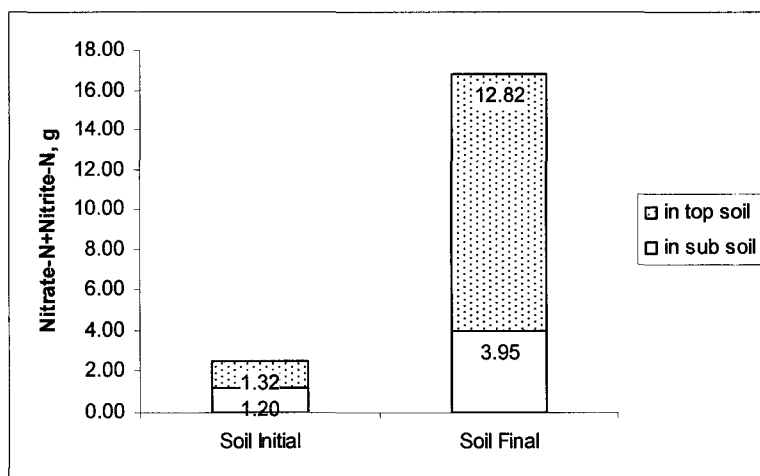


Figure 4-39  $\text{NO}_3\text{-N}+\text{NO}_2\text{-N}$  content distribution in soil from Phase III

#### 4.2.2.1.4 Discussion of nitrogen conversion in the soil system

Normally, except for volatilization, ammonium can be adsorbed by soil particles or taken up by plants without being transformed. However, in most cases, it is converted through nitrification to nitrate soon after its formation or addition as fertilizer. The distribution of nitrate plus nitrite nitrogen along soil depth demonstrated the manure's impact on the soil profile (Figure 4-39). Compared to the initial soil, the soils in columns 7, 8, and 9 (Phase III) had higher nitrate plus nitrite nitrogen content in the top soils, especially within the top 15 cm, indicating that the nitrification process occurred there. In contrast, column 3 had a lower nitrate plus nitrite nitrogen content than the initial top soil, as in Phase I, meaning nitrification was not the predominant process in the removal of ammonia from these columns. Reviewing the whole experimental process, it was reasonable to consider that the reason for the decrease in nitrate plus nitrite nitrogen in those columns was water clogging. Column 3 of Phase III showed a lower permeability than the others columns from the beginning of the run. Therefore, it had to be manured at a lower rate after the fifth week. However, even though it was tilled, there was still water cover on the soil surface. During Phase I, precipitation began on July 09, 2004 and lasted until July 21, which resulted in a large amount of water accumulating on the soil surface. The often predominant anaerobic conditions in the flooded soils during this period meant that little ammonium was nitrified. A build-up of ammonium in the soil resulted (Eriksson *et al.*, 2003). However, due to the accumulation of water, the manure application was suspended for some days in Phase I of the experiment. Lack of new nitrogen sources and downward migration through the soils resulted in a decline of TKN and  $\text{NH}_4\text{-N}$  concentrations in the soil and an increase in the leachate.

In the sub soils in Phase III, nitrate plus nitrite content in most of the columns were lower than the initial concentrations (See example in Figure 4-40) after manure application. Leaching was one of the reasons provided to account for this loss, but not the primary one. The main reason might have been the denitrification, which, in soil, can reduce nitrate to nitrite and then to  $N_2O$  or  $N_2$ , thus decreasing the environmental damages of nitrate leaching.



**Figure 4-40 Nitrate and nitrite nitrogen variation in soil (Col. 8)**

Based on nitrogen balance (See Table 4-7), it was true that a certain amount of nitrogen in the manure remained unaccounted for after the eight week application of pre-treated manure. This phenomenon was observed also in the resting soil in Phase II (See balance in Appendix B). These unaccounted for quantities might be gaseous nitrogen emitted out of the soil system. However, it was difficult to determine what kinds of gases were emitted because no gaseous phase was detected in this experiment. It was assumed that ammonia volatilization, nitrification, and denitrification accounted for the loss, and that the gaseous forms might include ammonia,  $N_2$ , and  $N_2O$  etc.

The application method, characteristics of the manure, weather, and field conditions are considered to be important factors in influencing the volatilization of nitrogen from liquid pig manure applied and incorporated on arable land (Huijsmans *et al.*, 2003). There was a large emission range, from 10% to 99%, during the storage and application periods (Moal *et al.*, 1995; Vanderholm, 1975). N<sub>2</sub>O emissions from cultivated land would be increased through the application of an organic amendment (Yang *et al.* 2004). Moreover, N<sub>2</sub>O can be produced either by denitrification in anoxic conditions, or by nitrification in the presence of oxygen. Prieme and Christensen (2001) reported that in the German grassland soil, denitrification was responsible for nearly all N<sub>2</sub>O emission following wetting, whereas at the German ploughed site and the two Swedish sites, nitrification was a major source of N<sub>2</sub>O after wetting. Emissions of N<sub>2</sub>O by nitrification may be quite significant if the nitrification occurs at a reduced oxygen concentration (Khalil *et al.*, 2004). Wolf and Russow (2000) found that under water-unsaturated conditions the microbial oxidation of NH<sub>4</sub><sup>+</sup> to NO<sub>3</sub><sup>-</sup> predominated, but that a reduction in NO<sub>3</sub><sup>-</sup> also occurred. The emission of NO exceeded the emission of N<sub>2</sub>O by a factor of up to 20 at the beginning of the experiments. Under water-saturated conditions, denitrification was the dominant process and N<sub>2</sub>O emission was greater than under unsaturated conditions. N<sub>2</sub> could be detected under water-saturated conditions. Simek *et al.* (2002) reported that at pH values above 7, N<sub>2</sub> is a much more important denitrification product than N<sub>2</sub>O.

Aerobic autotrophic bacteria are normally considered to be responsible for nitrification in soils. It has been shown that nitrifying bacteria may be responsible for N<sub>2</sub>O peaks shortly after wetting (Dendooven *et al.*, 1994). Hadas *et al.* (1996) suggested

that extensive emissions of  $N_2O$  immediately following the application of organic amendments might be the result of the rapid growth of zymogenous bacteria in response to the addition of the high-energy organics present in the amendments. Fungi might also contribute to the nitrification and denitrification in this soil column process. After researching acidic brown earth (pH 6.3), Laughlin and Stevens (2002) reported that fungi were responsible for most of the  $N_2O$  production, and that all of the  $N_2O$  was derived from  $NO_3$  reduction. They found that the flux of  $N_2O$  was always greater than that of  $N_2$ , the average mole fraction of  $N_2O$  being 0.7. They also indicated that denitrification was not the predominant process for  $N_2$  generation, although the  $NO_3$  pool was still the primary N source because about 92% of the  $N_2$  was derived from codenitrification ( $NO_2$  combined with another N atom from a naturally abundant source).

Apart from bacteria, many other factors affect nitrification in soil, including temperature, moisture, pH, and concentration of the substrates, as well as  $NH_3$ . Soil oxygen concentration, which is regulated by soil water content as well as soil texture and microbial respiration, is the main controller of this process. Because there was no clogging in columns 7, 8, and 9 in Phase III, it is reasonable to believe that there was considerable oxygen supplied through intermittent manure application (twice a week) and transported into the soil system via diffusion through the open surface layer. Moreover, top soil tillage before and after application in this period promoted oxygen diffusion, establishing good aerobic conditions. Normally, oxygen was used first for the decomposition of carbonaceous substrates that remove BOD and COD, and was then used for oxidizing ammonium to  $NO_2$  and  $NO_3$ . However, when there is only enough oxygen for substrate decomposition, the nitrification rate will be very low. The reason for

this is that the substrate-consuming heterotrophic bacteria have higher yield coefficients and faster growth rates than the autotrophic nitrifiers, and therefore are more competitive than nitrifying bacteria. Thus, significant nitrification occurs only after the BOD concentration has been appreciably reduced. Some researchers mentioned that significant nitrification cannot take place until the  $BOD_5$  is reduced to  $200 \text{ mg L}^{-1}$  or even lower (Zhao *et al.*, 2004; Sun *et al.*, 1998). This could explain why nitrification was not the predominant process in the columns of Phase I and in column 3 of Phase III, in which water clogging resulted in a lack of oxygen.

In addition, high operation temperature ( $24.5 \text{ }^\circ\text{C}$  to  $31.5 \text{ }^\circ\text{C}$  in Phase) is helpful for the nitrification rate. Fdz-Polance *et al.* (1995) indicated that, under conditions of abundant oxygen, an increase in temperature of  $1 \text{ }^\circ\text{C}$ , with  $10\text{-}29 \text{ }^\circ\text{C}$  internal, brought about an increase of 2% in the nitrification rate.

For denitrification, anaerobic conditions are extremely important. The oxygen consumption rate depends on the amount of easily degradable organic carbon compounds available and on the interplay of water and carbon in the soil with reduced anoxic conditions, which regulate not only the total amount of denitrification, but also the ratio of  $N_2O$  to  $N_2$  produced (Mosier *et al.*, 2001). Other necessary conditions for denitrification are waterlogged soils, carbon sources (from organic matter or plant residues) for use by anaerobic microbes, and a nitrogen source as either  $NO_3$  or  $NO_2$ . The denitrification rate is greatly accelerated by higher temperatures (Oldham, 2003).

There were many conditions in this experiment which met the denitrification requirements. The first was anaerobic conditions in the subsoil. The second was abundant organic substances in the soil (OM =2%-10% in initial dry soil) and in the manure

(Soluble organic carbon = 1568 mg/L), which provided the electron donors for oxidation reductions using nitrate generated in the top soil. The third necessary condition was the high temperature (24 to 31.5 °C in the greenhouse). In addition, neutral or slightly alkaline conditions (soil pH was around 8 in this experiment) support denitrifying bacteria, which are sensitive to low pH. Through reviewing the research on the interactions between pH and denitrification in soil from the 1950s to 2002, Simek and Cooper (2002) found that total gaseous emissions to the atmosphere (N<sub>2</sub>O, NO, and N<sub>2</sub>) have repeatedly been shown to be less in acidic than in neutral or slightly alkaline soil. However, the relationship between soil pH and the potential for denitrification, as determined by various incubation methods, remains unclear, the results being influenced both by the original conditions in the soil samples and by unknown changes during incubation. In fact, Simek *et al.* (2002) suggested that expressions like *optimum pH for denitrification* should be avoided, as different denitrification characteristics obtained using different methodologies can be quite differently related to soil reactions.

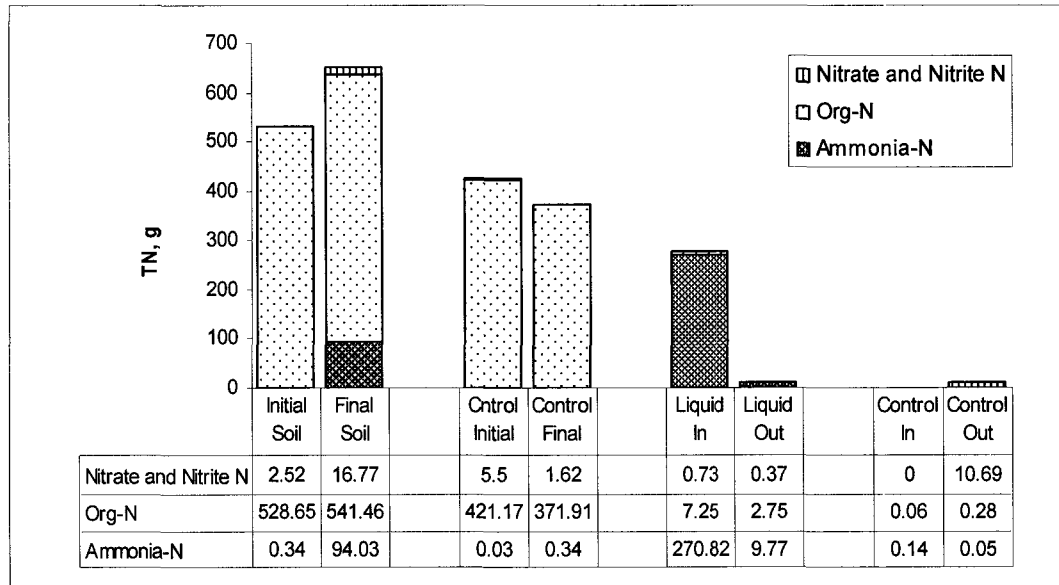
Soil moisture content also affects the denitrification rate. Shelton *et al.* (2000) reported that 60% water filled pore space (WFPS) was the threshold for denitrification in soil, and showed an apparent linear relationship between denitrification and soil WFPS. Similar results were reported by Abbasi *et al.* (2000). In their experiment, the rate and amount of N loss, and fluxes of N<sub>2</sub>O from both NO<sub>3</sub><sup>-</sup> and NH<sub>4</sub><sup>+</sup> sources, were greater in soils at 84% WFPS than in soils with 71% and 63% WFPS. Taking column 7 as an example, the subsoil WFPS in this experiment was estimated as 68%, which supported denitrification in the sub soils.

Furthermore, soil structure heterogeneity in the form of macropores and preferential flow channels (a previously mentioned problem in this experiment) can complicate efforts to quantify the physical and biological characteristics of wetland systems. Casey *et al.* (2004) reported that the presence of macropores could greatly elevate denitrification levels and suggested that heterogeneity in hydrology and soil structure can be accompanied by localized microbial activity.

Finally, due to the high amount of ammonium supplied to the soil system, it was necessary to ensure that this practice did not inhibit the nitrification potential of the soil. The increased nitrate plus nitrite content in the top soil in Phase III partially demonstrated that nitrifiers, or at least some of the nitrifiers, were still active at the end of experiment, in spite of the high ammonium loading. When the manured soils in Phase I rested for six weeks (Phase II), they showed a dramatic increase in nitrate plus nitrite content in the top soils (See Figure 4-38), demonstrating that soil rest after manure application promoted the process. It also implied that continuous nitrogen or ammonia application inhibits the nitrification process. It has been reported (Prosser and Cox, 1982) that, in soils with high pH and  $\text{NH}_4^+$  concentration, nitrobacters ( $\text{NO}_2^-$  oxidizers) may be selectively inhibited, resulting in the accumulation of  $\text{NO}_2^-$ . Through researching biofilters, Fdz-Polanco *et al.* (1994) indicated that the threshold for specific free-ammonia concentration inhibiting nitrobacter activity was 1 mg  $\text{NH}_3\text{-N/mg VAS}$  (volatile attached solid), because above this value, the nitrobacter's activity (utilization rate of substrate per unit of microorganism mass ( $\mu_{\text{max}}/Y$ )) was under 10 mg  $\text{NO}_2^-\text{-N/gVAS}\cdot\text{h}$ , while the activity increased exponentially to 50 mg  $\text{NO}_2^-\text{-N/gVAS}\cdot\text{h}$  when the value of the specific free ammonia concentration was lower than 1 mg  $\text{NH}_3\text{-N/mgVAS}$ .

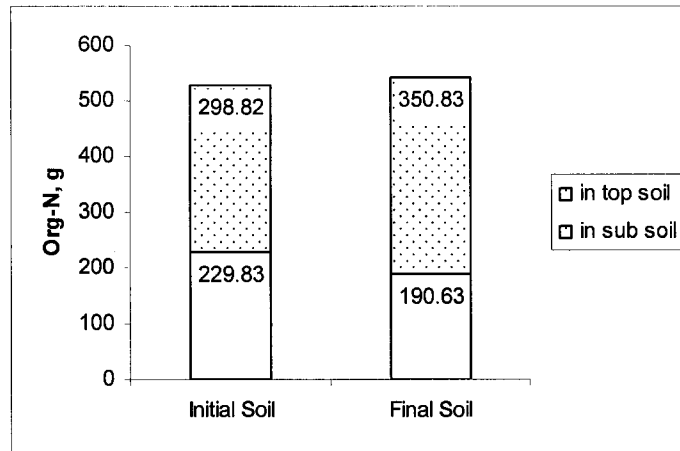


Including the variation in nitrate plus nitrite nitrogen, the total nitrogen change in the manure and soil system is showed in Figure 4-40 (Taking column 8 as an example, others are shown in Appendix A).



**Figure 4-41 Nitrogen transfer and conversion in manure-applied soil column system (Col. 8)**

It is clearly indicated that about half of the nitrogen in the manure stayed in the soil matrix, showing the soil's excellent nitrogen interception ability. Ammonia, which constituted the main nitrogen content in the manure, was the primary contributor to the increase in soil total nitrogen. Nitrate plus nitrite nitrogen was also generated during the application period, but the increase was not particularly notable. This might be due to the emission of  $N_2O$  or  $N_2$ . The increase in organic nitrogen in manured soil is also shown, and is illustrated more clearly in Figures 4-42 and 4-43.



**Figure 4-42 Organic nitrogen variation in soil (Col. 8)**

After manure application, some of the ammonia and nitrate was rapidly utilized by the heterotrophic microbes and converted into microbial components; that is, the inorganic nitrogen was immobilized (Stevenson and Cole, 1999). In other words, the decay of organic residues in the soils was accompanied by a conversion of C and N into microbial cells. Thierfelder *et al.* (2004) demonstrated that a significant proportion of the ammonium-N in pig manure was immobilized during the microbial decomposition of organic matter in the manure soon after its application to soil. This immobilized ammonium-N will be released slowly over many years. In addition to the biological transformations, nitrogen entering the soil system was also subjected to various chemical transformations. For example, some nitrogen was incorporated into stable humus forms (Jansson and Persson, 1982).  $\text{NH}_3$  in soil can react with soil organic matter to become “fixed”. It was found that from 20 to 40% of the nitrogen applied as fertilizer remained behind in the soil in organic forms after the first growing season (Stevenson and Cole, 1999). However, nitrogen immobilization was reported to have decreased when the rate of nitrification increased with temperature, suggesting that the nitrification and immobilization processes compete for the added  $\text{NH}_4^+$ -N (Sierra, 2002).

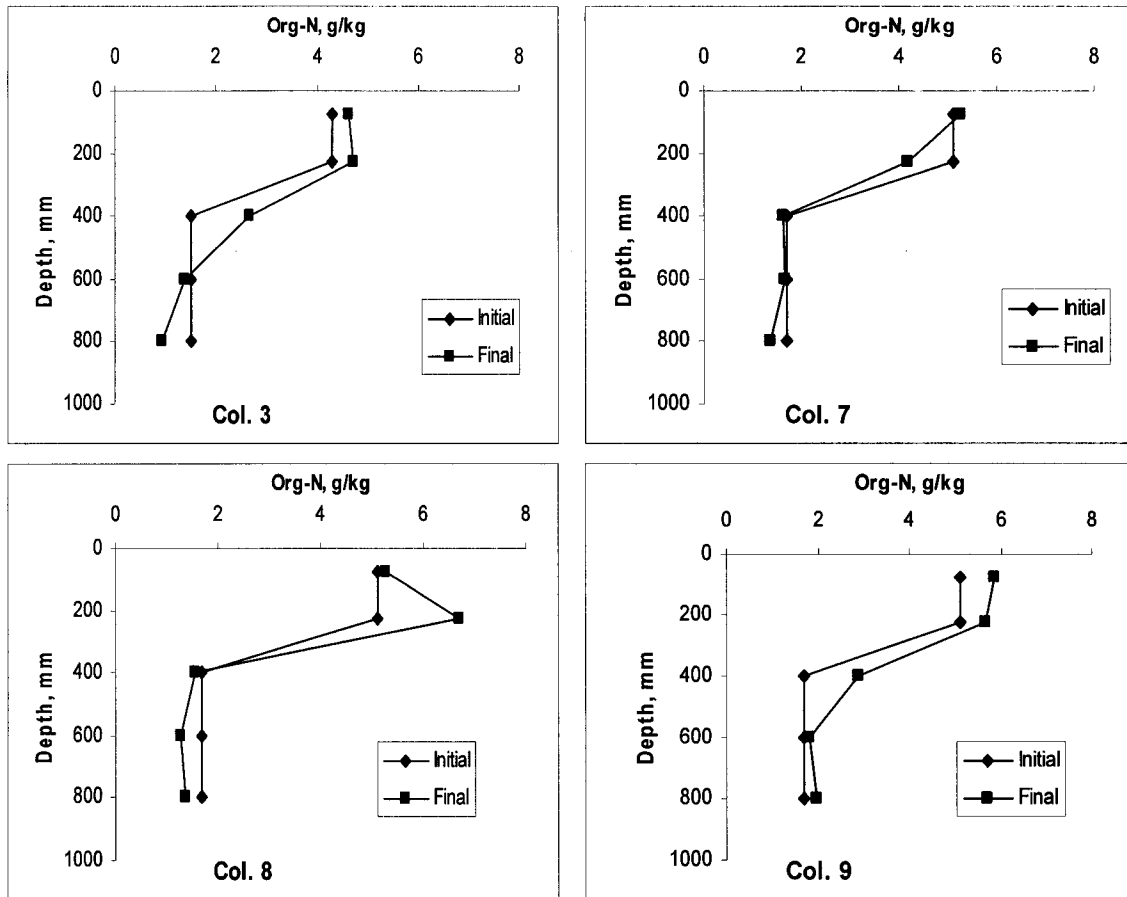
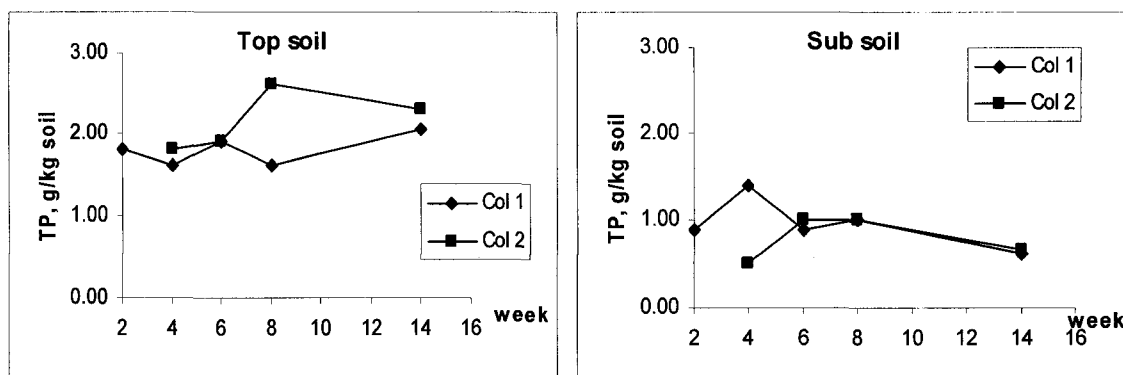


Figure 4-43 Organic nitrogen distribution in the soil profile during Phase III

#### 4.2.2.2 Total phosphorus

The budget of phosphorus in Phase III (See Appendix C) indicated high phosphorus accumulation in the soils, which cannot be explained. Therefore, only the Phase I results from the 12 mm d<sup>-1</sup> manure application are shown here. Oscillating results can be observed in Figure 4-44, and no agreement could be found between the replicates. But, the final concentration was higher than that of the initial soil. After rest, the average content increased in the top soils and decreased in the sub soils (See Figure 4-44).

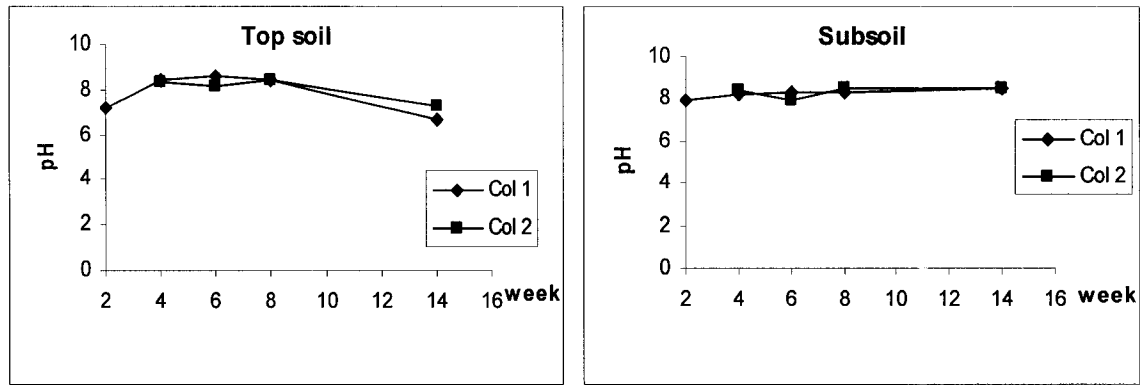


**Figure 4-44 TP variation in soil from Phase I and Phase II**

The normal range of phosphorus in soils is on the order of 500 to 800 mg kg<sup>-1</sup> dry weight (Stevenson and Cole, 1999). Therefore, TP in the initial top soils was present at a very high level (1400 mg kg<sup>-1</sup> dry weight soil), implying a limited capacity for excess P accumulation. But based on the leachate quality, the TP in the LSM was still reduced as it traveled through the soil profiles. During the rest period (Phase II), the decrease in TP content in the sub soils and the increase in TP content in the top soils might be due to soluble-P migration driven by evaporation in the soil.

#### 4.2.2.3 pH

pH values remained at the same level through Phase I in both the top soil and the subsoil (See Figure 4-45). No significant change was found between the beginning and the end of the eight week period. At the end of this period, the average pH value for columns 1 and 2 was 8.39 in the topsoil and 8.41 in the subsoil. These values were slightly higher than those of the initial soils, which were 8.25 and 8.27, respectively. The subsequent six-week soil rest period (Phase II) showed a slight decline in the pH value in the top soil, but not in subsoil.



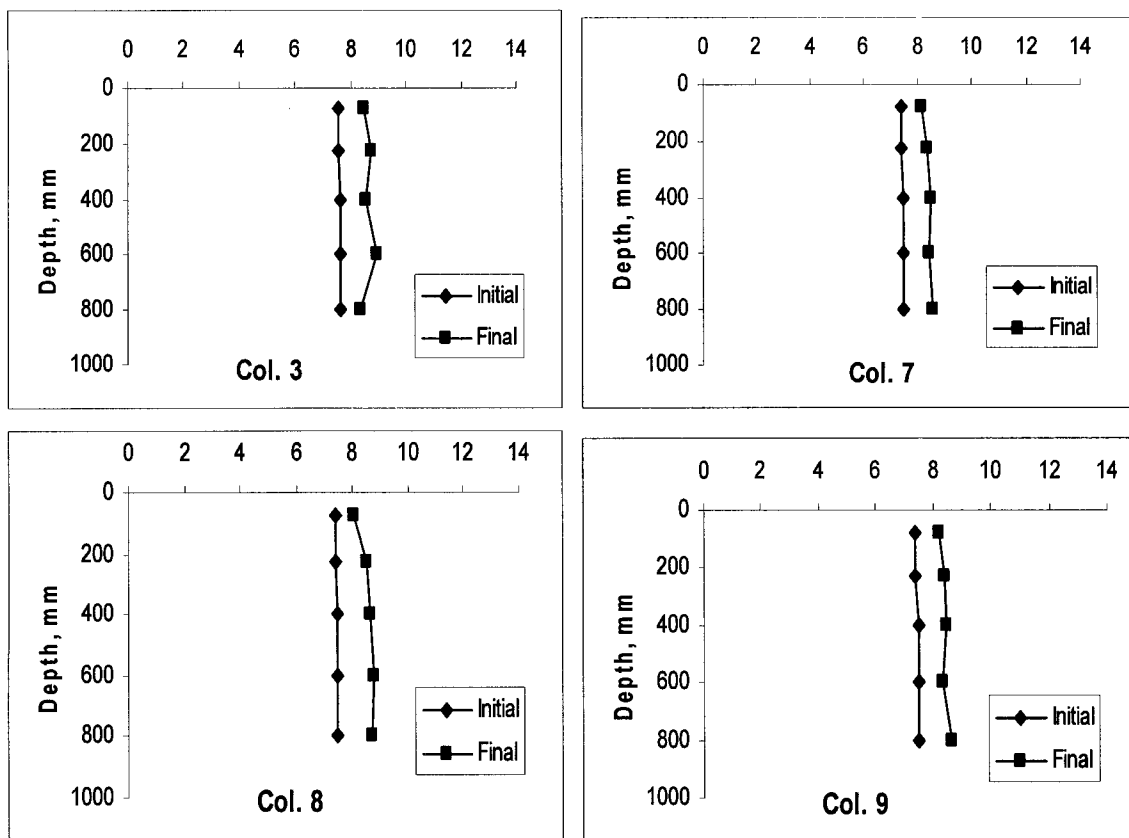
**Figure 4-45 pH variation with time in the manured soil profiles of Phase I and II**

Compared to the initial soil, LSM application did not significantly affect the pH in any of the soil profiles in Phase III, but the final values were all higher than the initial values (See Figure 4-46). Statistical analysis indicated that there was no significant difference between the two layers in the top soil. Similarly, the pH values did not differ significantly among the three layers in the subsoil.

The pH increased in the soils during both Phase I and Phase III. Manure can influence soil pH in any direction. Eghball (2002) found that the application of N-based manure significantly increased pH in surface soil (0-15cm). Liu *et al.* (1997) and Chang *et al.* (1991) reported decreases in pH after manure application.

In this experiment, the pH increase was derived mainly from the continuous addition of ammonia. In Phase III, the pH value was lower in the first layer than in the second layer because of the nitrification that provided  $H^+$  to neutralize some of the alkalinity. Soil pH values in Phase I showed that the continuous feeding of manure did not result in a continuous increase in soil pH, meaning that a relatively steady relationship exists between manure and soils. Soil rest showed a distinct decrease in top soil pH, indicating that the nitrification process was promoted. Subsoil pH, on the other hand, remained at the same level as before the six week rest period. The reason for this was that more

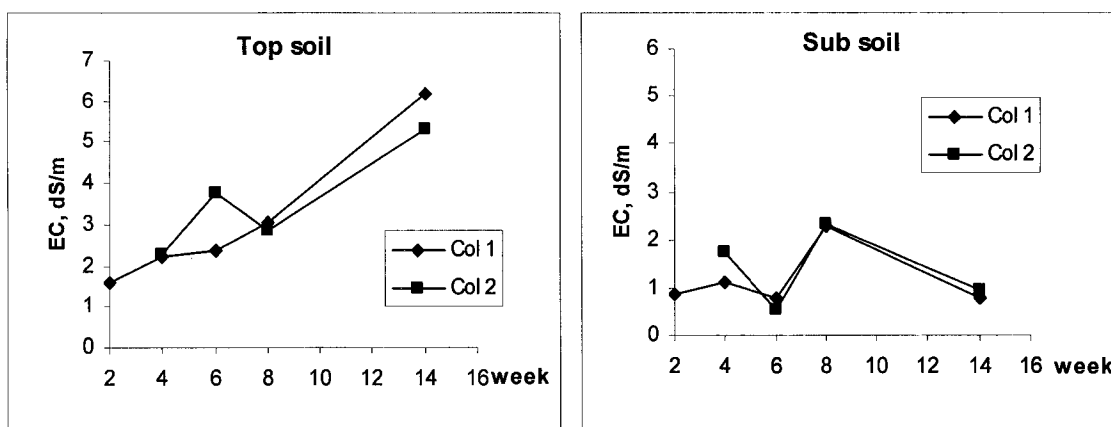
oxygen could diffuse into the top soil when no water was added, but this diffusion was limited and did not reach to the deeper soil profile.



**Figure 4-46 pH variation before and after manure application in Phase III**

#### 4.2.2.4 Electrical conductivity

In Phase I, EC values in the topsoil increased with continued manure application (See Figure 4-47). This increasing trend continued throughout the soil rest period. The EC values also increased in the subsoil throughout the eight week manure application, but with some oscillation of values. During the six-week rest, EC values dropped significantly in the sub soils. Conversely, EC values increased in the top soils during this period.



**Figure 4-47 EC variation with time in manured soil profiles during Phase I and Phase II**

A similar tendency in EC variation appeared in Phase III among the soil columns. The eight-week manure application significantly increased EC in the top soils, but had an insignificant effect on EC in the sub soils (See Figure 4-48). The average increases in soil columns 7, 8, and 9 on layers 1 and 2 were 4.28 and 3.99 dS m<sup>-1</sup>, respectively. The highest EC value in top soils after manure application was 4.75 dS m<sup>-1</sup>. Statistical analysis indicated that the difference between the EC increases in the top two layers was insignificant, as was that occurring among the three sub soil layers. Column 3 was slightly different than the others, in that the EC increased in all five soil layers, but the EC increase in layer 1 was 4.02 dS m<sup>-1</sup>, which was almost same as in the other columns.

Due to the different application rates, it was reasonable that the observed increase was higher in Phase III than in Phase I. The top soils seemed to intercept most of the ions in the manure, as they showed greater EC value increases than did the subsoils. For example, the average value of increase in the top soils in Phase I was 2.1 dS m<sup>-1</sup>, while the average increase in the sub soils was only 1.41dS m<sup>-1</sup>. However, the sub soils in Phase III did not show any increase in EC value, unlike the sub soils in Phase I. One possible reason for this was that the soils had worse ion “catching” ability, which may

have been due to channels in the soils through which ions entering the sub soils were directly flushed out of the soil system.

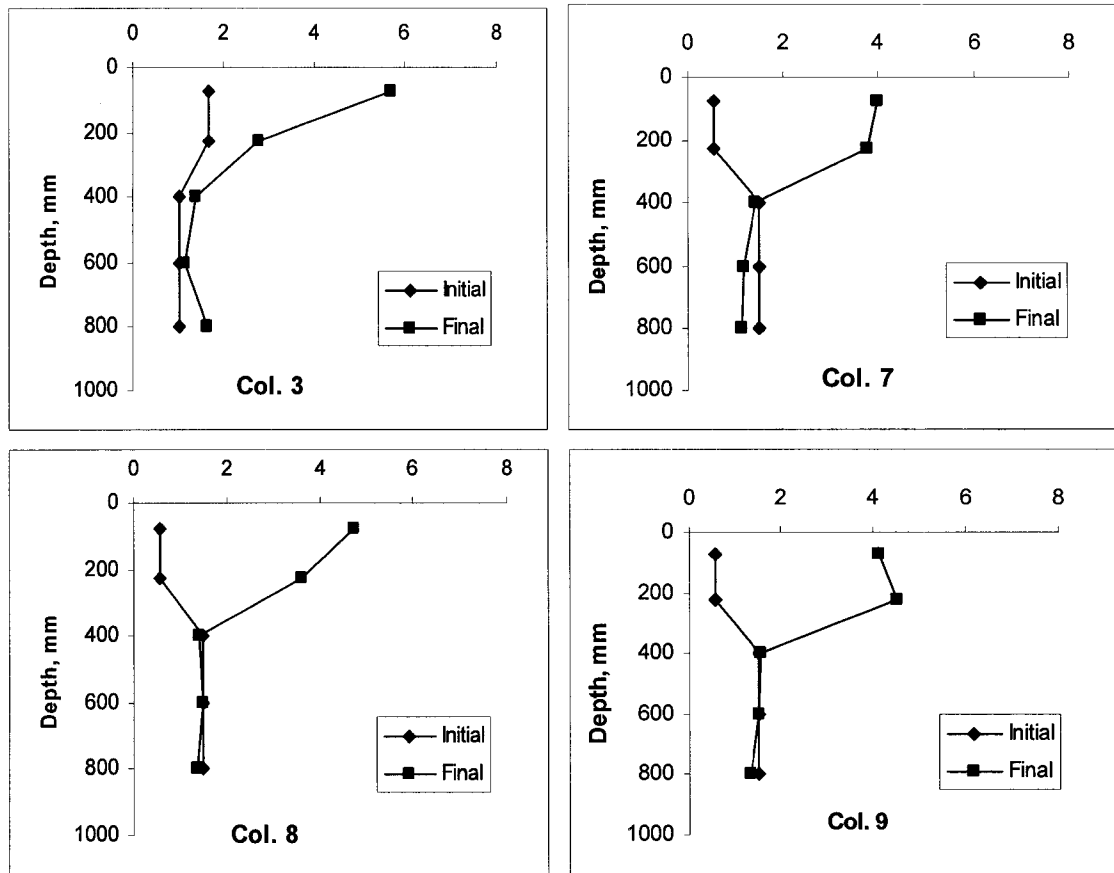
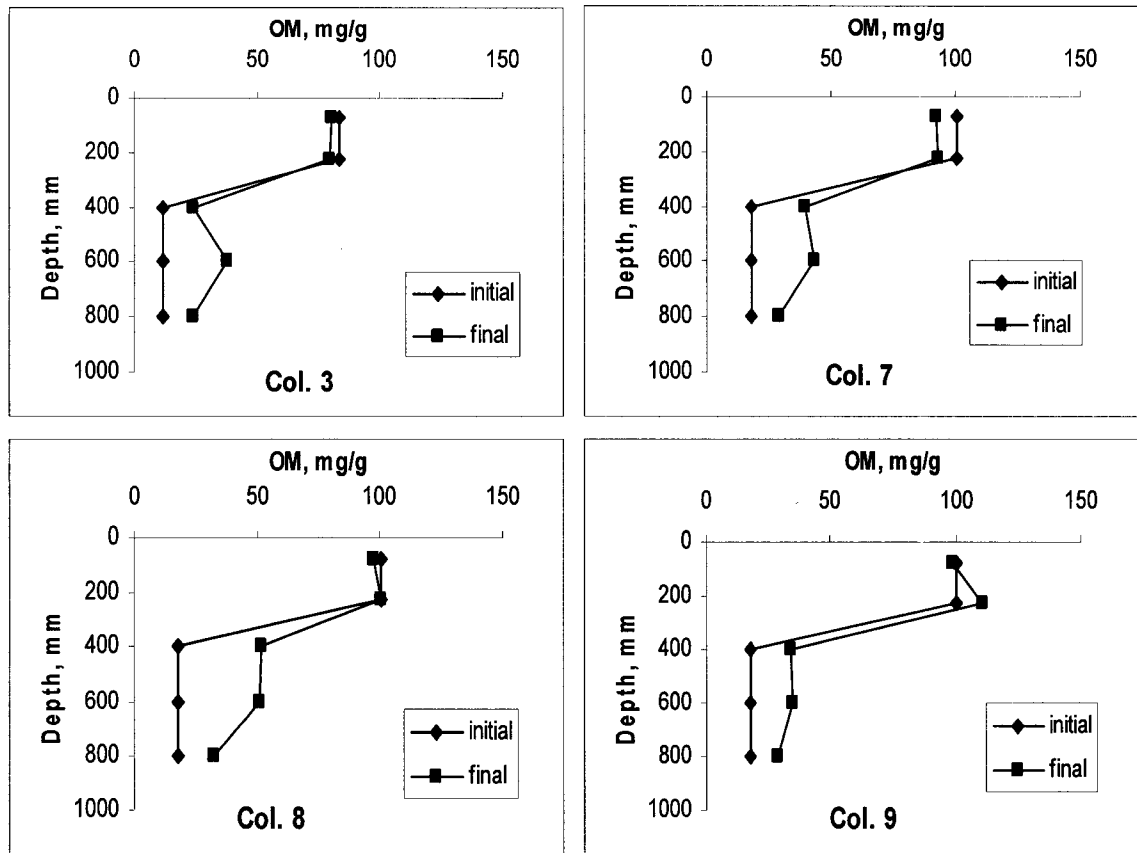


Figure 4-48 EC variation before and after manure application in Phase III (dS/m)

#### 4.2.2.5 Organic matter

Organic matter (OM) is different from most other parameters in that it exhibited no significant change in the top soil after eight weeks of manure application (See Figure 4-49). Some OM values in layer 1 were even lower than those of the initial soils. Conversely, a significant increase was observed in the sub soils. The changes in OM in the soil profiles are illustrated in Figure 4-36. OM was not analyzed in Phases I and II.





**Figure 4-49 OM variation before and after manure application during Phase III**

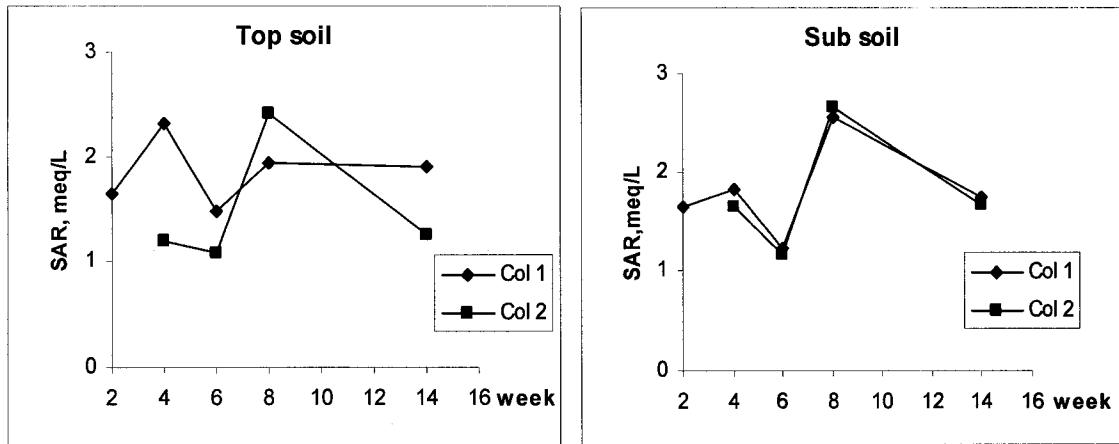
The results of this experiment were different from those obtained by Liu *et al.* (1997), who reported that total carbon was highest in the surface soil (0-20 cm) after swine lagoon effluent application and decreased with soil depth. Moreover, no significant differences were observed to a soil depth of 40 cm in Liu's experiment. In this experiment, on the contrary, no significant change in OM in the top soils was observed, demonstrating that the organic substrate added was effectively decomposed by microbes after the manure application. In order to reproduce and function properly, a microbe must have sources of energy, carbon for the synthesis of new cellular material, and inorganic elements such as nitrogen, phosphorus, sulfur, potassium, calcium, and magnesium (Metcalf and Eddy, 2003). The high concentration of BOD and TOC in the pre-treated manure could provide a large amount of carbon for heterotrophic microbe development.

In addition, carbon dioxide can be used as another carbon source for autotrophic microbe development, although autotrophic microbes need more energy for synthesis than heterotrophs do. Due to the greenhouse operating conditions, light was limited, which suggests that the energy needed for cell synthesis was obtained mainly from chemical reactions (by chemoheterotrophs or chemoautotrophs).

A significant increase in OM in the sub soils means that OM was less effectively decomposed in the subsoil. Under anaerobic conditions in the subsoil, anaerobic fermentation should be processed if OM decomposition were to be observed and normally, CO<sub>2</sub> and CH<sub>4</sub> would be generated during this process. However, there were no gases monitored in this experiment, so it was difficult to determine if this process actually occurred. Furthermore, anaerobic processes usually need a longer start-up time to develop the necessary biomass inventory. The only possible clue was the increase in pH values in the soil, which might be an indication of the processing of the anaerobic treatment.

#### **4.2.2.6 SAR**

Compared to the initial soils, SAR values for both the top and sub soils in Phase I were all greater, but the values oscillated on different sampling days within a range of 1 to 2.65, with no significant differences among them on different sampling days (See Figure 4-50). Statistical analysis also indicated that the six-week rest had no significant effect on SAR in the topsoil ( $p=0.27$ ), but had a significant effect in the subsoil ( $p<0.01$ ). However, the actual values changed by less than 1 unit.



**Figure 4-50 SAR variation with time in manured soil profiles during Phase I and Phase II**

Figure 4-51 shows how SAR values increased in the soil profiles after manure application in Phase III. These increases occurred in all the soil layers and the increasing trend was almost the same. Statistical analysis indicated that the increases were all significant. However, although the SAR increase was significant, the actual increases in the values were all less than 1. It was also found by statistical analysis that there were no significant differences among the different soil layers after manure application.

In sodic soil, physicochemical reactions cause the slaking of aggregates and the swelling and dispersion of clay minerals, leading to reduced permeability and poor till. Current terminology classifies a soil as sodic at an SAR of 13 (Soil Science Society of America, 1997), which is much higher than that of the manured soils in this experiment which suggests that short term LSM application would not affect the soil structure very much. Reductions in filtration rate were directly related to increases in soil SAR (Wienhold and Trooien, 1998); therefore, the water log during Phase I was not due to a change in the soil's physical structure, but rather to SS accumulation on the surface soil, or to the biomass produced during organic substrate oxidation.

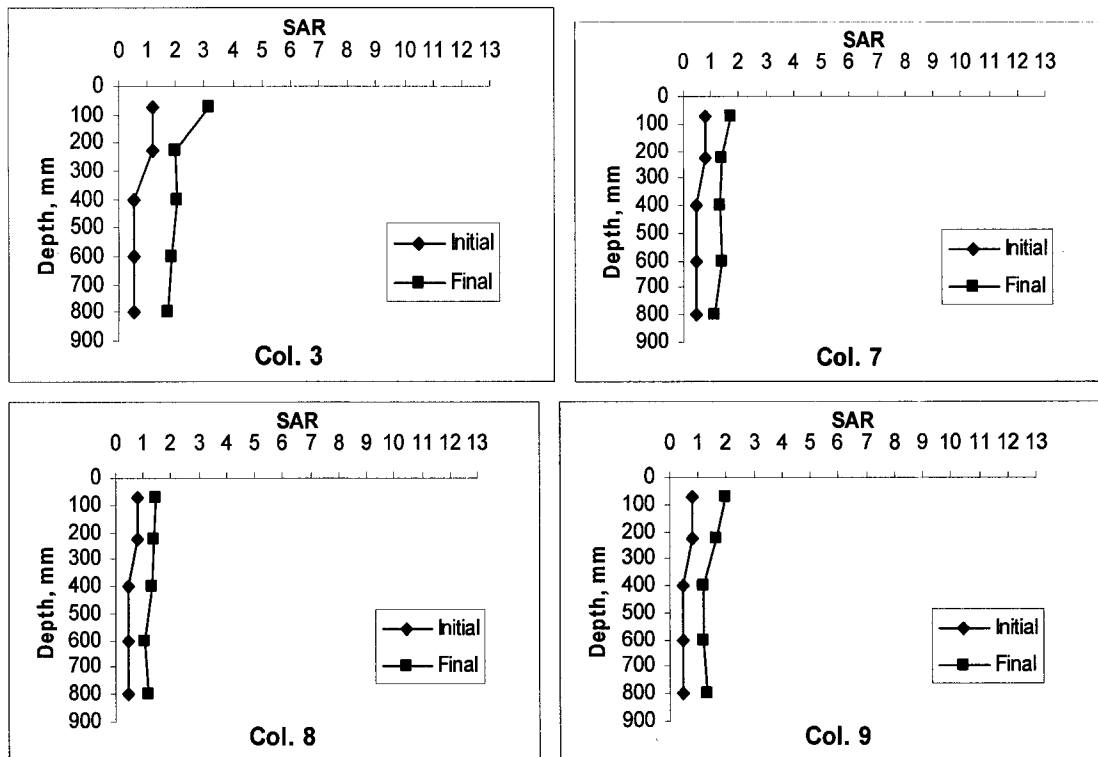


Figure 4-51 SAR variation before and after manure application during Phase III

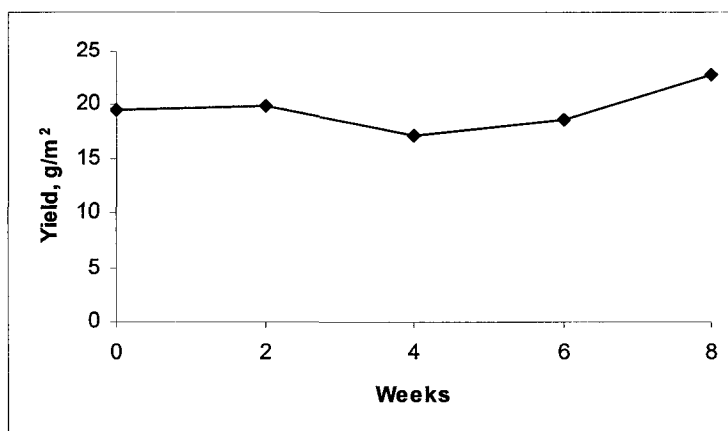
### 4.3 Grass response

In Phase III, the Bermudagrass in column 1 developed very well through leachate irrigation (Figure 4-39).



Figure 4-52 Grass growth through leachate irrigation

The dry weight (DW) yield from each two week period is shown in Figure 4-40. The value for November 30, 2004 was the two week yield of grass irrigated with tap water prior to the leachate irrigation. Compared to this, the yield of leachate-irrigated grass dropped slightly during the second week of application, but rebounded and exceeded the yield of grass irrigated with tap water by the end of the experiment. In total, the DW yield was about 700 kg ha<sup>-1</sup>.



**Figure 4-53 Variation of DW yield with time**

The comparison between bermudagrass and other crops is tabulated in Table 4-8, which shows that the good performance of bermudagrass can be attributed to its excellent nutrient uptake and salt tolerance abilities.

**Table 4-8 Nutrient uptake rate and salt tolerance of different grasses**

Forage Crops	Nitrogen <sup>a</sup> g/m <sup>2</sup> -y	Phosphorus <sup>a</sup> g/m <sup>2</sup> -y	Salt Tolerance <sup>b</sup> dS/m (rating)
Tall Fescue	15-32	3	3.9 (MT)
Kentucky Blue Grass	20-27	4.5	2 (MS)
Alfalfa	23-54	2-3.5	2 (MS)
Reed Canary Grass	33-45	4-4.5	(MS)
Bermuda Grass	40-67	3-4.5	6.9 (T)

Source: a- EPA .1981. Process Design Manual, Land Treatment of Municipal Wastewater  
 b- L.E. Francois and E.V. Mass. 1994. Crop response and management on salt-affected soils in *Handbook of plant and crop stress*.  
 T- tolerance; MT- moderate tolerance; MS – moderate sensitive

The leachate application experiment was run for eight weeks. Surprisingly, it was found that the grass still appeared to be healthy two months after irrigation ceased, indicating the grass' strong ability to survive in harsh living conditions.

The EC value of the leachate mixture was  $4.24 \text{ dS m}^{-1}$ , which was considered to fall within the severe degree of restriction on irrigation use (Agriculture and Agri-Food Canada, 2003). Bermudagrass was classified as a crop with high salt tolerance, and therefore performed very well during the period of leachate utilization. The fresh green color of the grass leaves suggested that the N level of  $86.65 \text{ mg L}^{-1}$  ( $2700 \text{ kgN ha}^{-1} \text{ yr}^{-1}$  counted for 180 days) in the leachate was very suitable for bermudagrass growth. However, it was much higher than the standard fertilizer N recommendation for bermudagrass hay in Alabama, which could have resulted in as much as  $448 \text{ kg N ha}^{-1} \text{ y}^{-1}$  when hay was harvested four times annually (Liu *et al.*, 1997). Based on the 8:1 N/P ratio for grass growth, the concentration of P in the leachate was quite low ( $1.49 \text{ mg L}^{-1}$ ) and would not influence grass growth very much.

High temperature ( $24.5$  to  $31^{\circ}\text{C}$ ) and no irrigation for almost two months did not kill the grass, but rather demonstrated bermudagrass' capacity for drought resistance. According to the Phase II results, the soil EC would increase after irrigation was stopped by evaporation. This suggests that salt content would no longer be a problem for bermudagrass growth under leachate utilization.

## 5 CONCLUSIONS AND RECOMMENDATIONS

In this study, the effectiveness of soil as a natural filter/bioreactor for treating swine manure was evaluated. Variations in soil properties due to manure application were also investigated both during and after manure application. 12 mm d<sup>-1</sup> and 17 mm d<sup>-1</sup> manure application rates were carried out for eight weeks. The corresponding loading values for BOD<sub>5</sub>, N, and P were 464, 179, and 1.5 kg ha<sup>-1</sup>d<sup>-1</sup> for the 12 mm d<sup>-1</sup> manure application and 657, 260, and 2.48 kg ha<sup>-1</sup>d<sup>-1</sup> for the 17 mm d<sup>-1</sup> manure application. In order to observe the crop's response, the leachate was utilized for bermudagrass growth and the final yield of the grass after two months of leachate application was evaluated. The following conclusions are drawn from this project:

- With regard to the removal of BOD<sub>5</sub>, COD, TKN, NH<sub>4</sub><sup>+</sup>-N, and TP, soils were found to be very effective for swine manure treatment within two months. The average mass removals of BOD<sub>5</sub>, COD, TKN, NH<sub>4</sub>-N, and TP with the 17 mm d<sup>-1</sup> application rate were 95, 91, 97, 98, and 90 %, respectively. However, the leachate quality became worse with application time and the deviation among the replicate columns was large.
- Soils were found to effectively decrease EC, but did not effectively reduce TDS. No correlation between EC and TDS could be established in the leachate.
- A certain amount of nitrogen-nutrient (42% of manure nitrogen) was unaccounted for, likely due to the emission of gaseous nitrogen during manure application.
- The leachate can be utilized for bermudagrass cultivation and had no negative influence on the grass' growth.

- Climate conditions, for example precipitation, influence the manure application rate and treatment results by changing the soil conditions.
- Till helps the nitrification process in soils.
- Soil rest effectively promotes nitrification and denitrification in manured soil.

According to the experimental process and results, the following recommendations need to be considered for future investigations:

- Nitrogen's gaseous states should be tested to help understand the transformations of nitrogen, to investigate nitrification and denitrification production, and to determine a condition whereby more nitrogen gas is produced than greenhouse gases.
- Oxygen content in soils should be analyzed in order to understand nitrogen conversion.
- The experimental period should be prolonged to more than two months in order to investigate the hypothesis that microbe incubation time resulted in the summit value of substrate in the leachate.
- Soils with quite different physical and chemical characteristics should be compared in terms of organic substrate and nutrient removal to investigate whether or not the soil filter method is universally suitable for manure treatment.
- The manured soils can be considered as nutrient sources and spread on other land for crop development.
- The second stage leachate (the leachate from the soil column covered by grass and irrigated using the leachate from the manured soil column) should be



monitored to investigate its potential for reuse objectives, such as water for washing barns.

The following recommendations are made for engineering design and operation:

- For swine manure with high nitrogen concentration and high slat content, bare soil should be used for first stage infiltration and the soil type can be loam or clay loam. The leachate can be used for crop development.
- It is not necessary to choose different soil types for top and sub soils in a soil filtration system. Using soils containing lower nutrients and organic matter would be better for system operation.
- Even when using leachate to irrigate crops, the crops should be salt tolerant and have good nutrient consumption abilities.
- Operation time is from May to October in Alberta.
- BOD loading less than  $0.065 \text{ kg BOD m}^{-2} \text{ d}^{-1}$  and TKN loading less than  $0.026 \text{ kg N m}^{-2} \text{ d}^{-1}$  will present no problem for the system.
- A hydraulic loading rate of  $16 \text{ mm d}^{-1}$  (in Phase III, the evaporation rate inside was about  $1 \text{ mm d}^{-1}$ ) is appropriated, but average daily evaporation must be added and average daily precipitation must be subtracted.

For example, if  $17 \text{ mm d}^{-1}$  (same as that in the experiment) is chosen to be the hydraulic loading rate, then during the 180 day operating period, the total applied manure will be 3.06 m. The average number of hogs on a farm in Alberta was 757 head in 2001 (AAFRD, 2003), and one pig can produce two tons of manure per year (Larson, 1991). Therefore, the annual manure application is  $1514 \text{ m}^3$ , and the land area needed for this amount of manure application is about  $500 \text{ m}^2$ .

In addition, the sludge in the storage pond and the manured soil can be spread directly on farm land as fertilizers.

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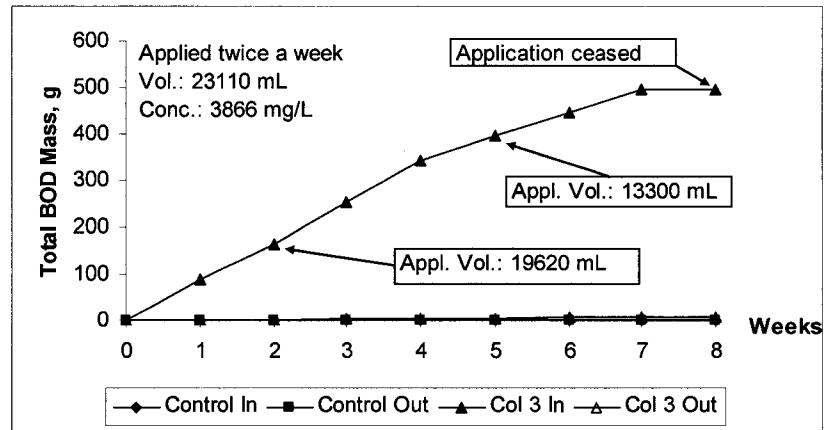
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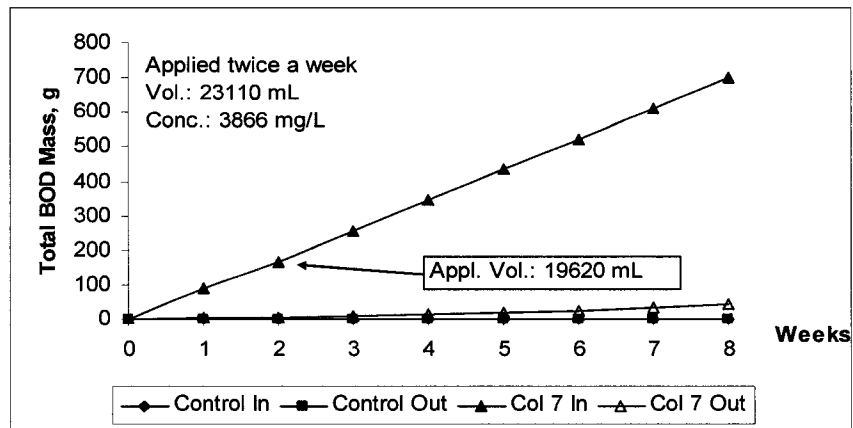
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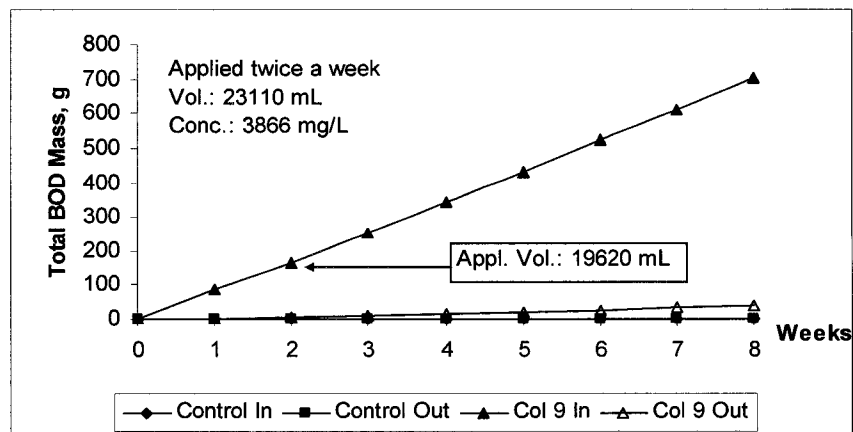
## APPENDIX A



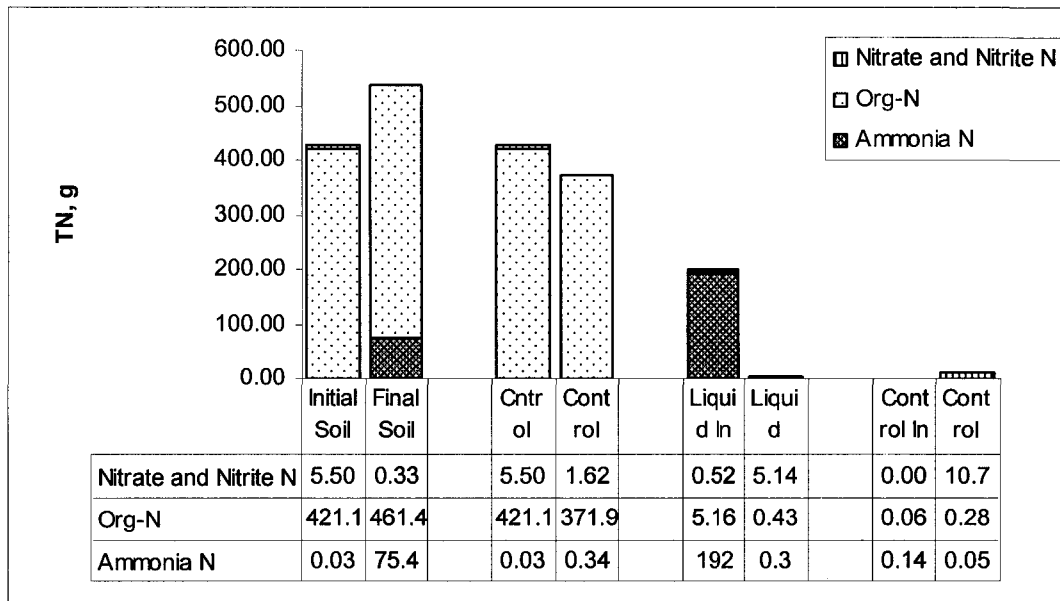
**Figure A 1 Cumulative BOD mass in manure and leachate in Phase III (Col. 3)**



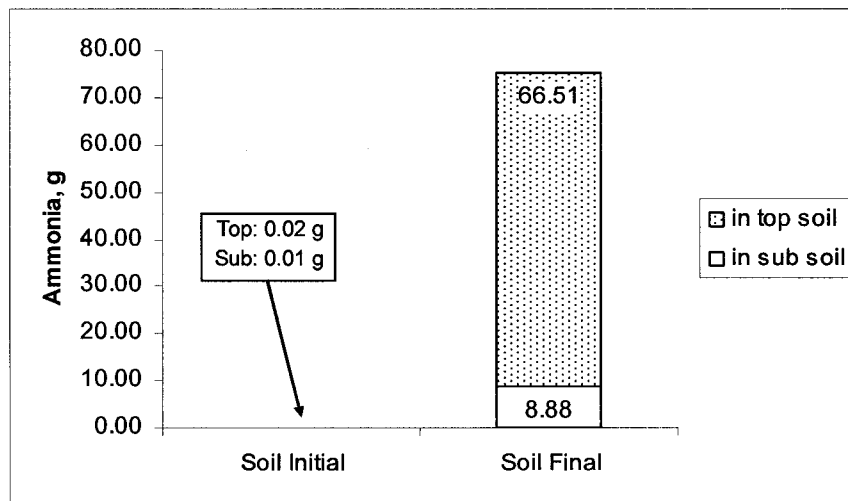
**Figure A 2 Cumulative BOD mass in manure and leachate in Phase III (Col. 7)**



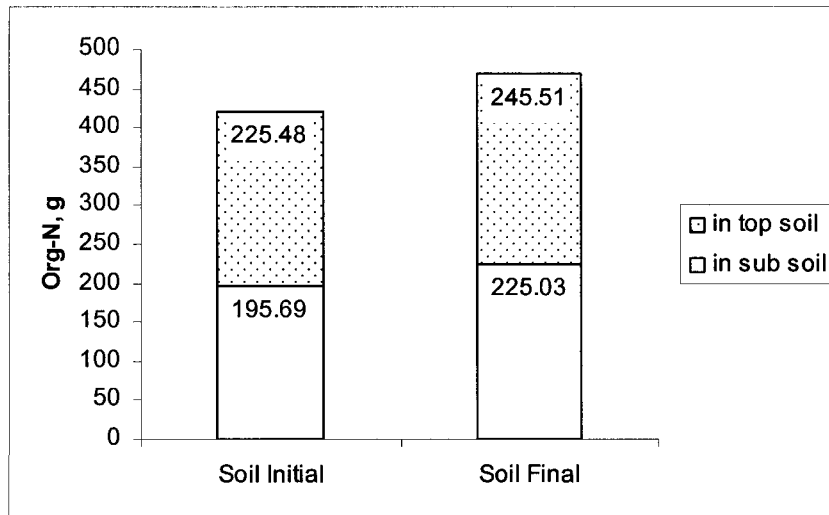
**Figure A 3 Cumulative BOD mass in manure and leachate in Phase III (Col. 9)**



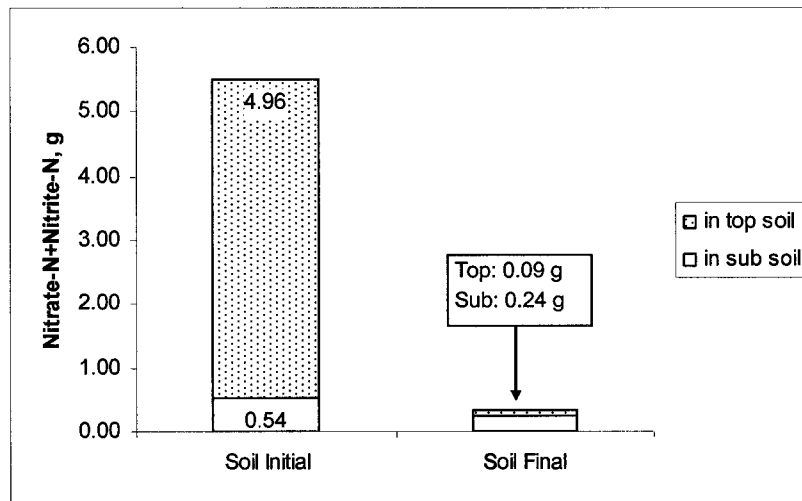
**Figure A 4 Nitrogen transfer and conversion in manure-applied soil column system (Col. 3)**



**Figure A 5 Ammonia variation in soil (Col. 3)**

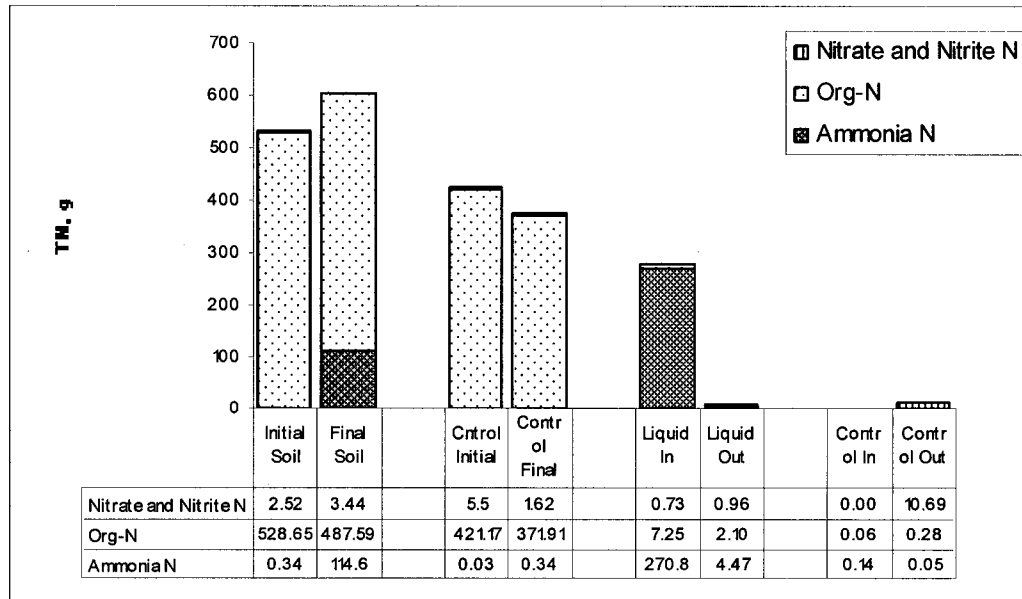


**Figure A 6 Organic nitrogen variation in soil (Col. 3)**

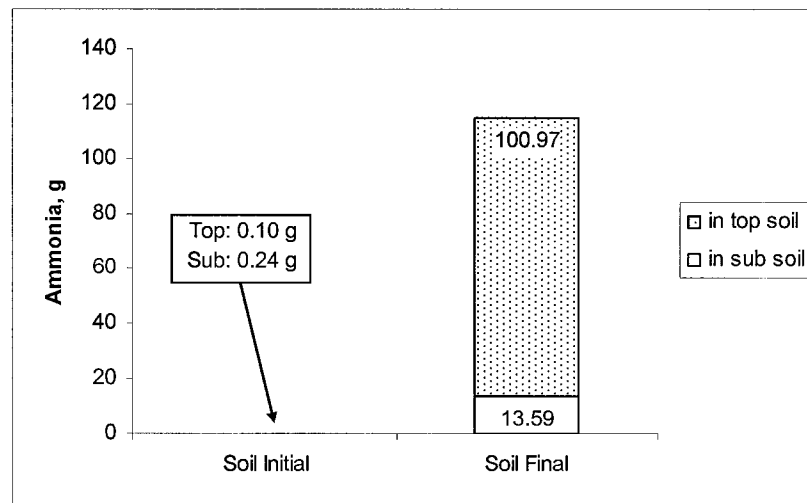


**Figure A 7 Nitrate plus nitrite nitrogen variation in soil (Col. 3)**

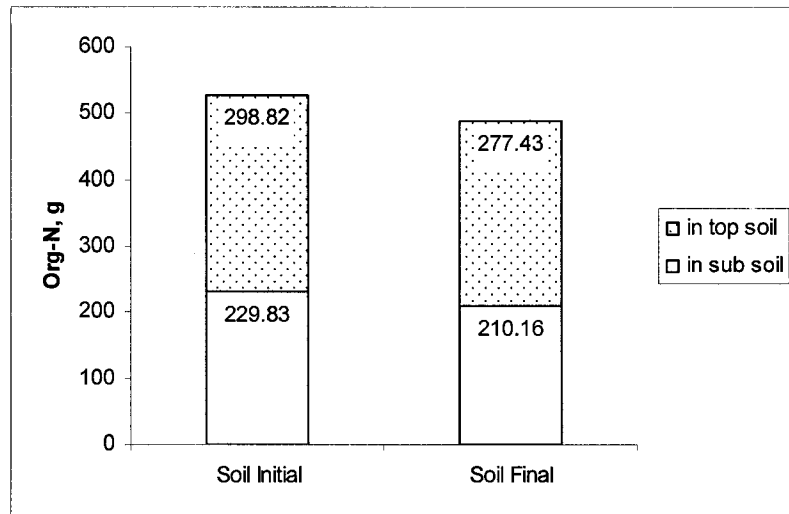




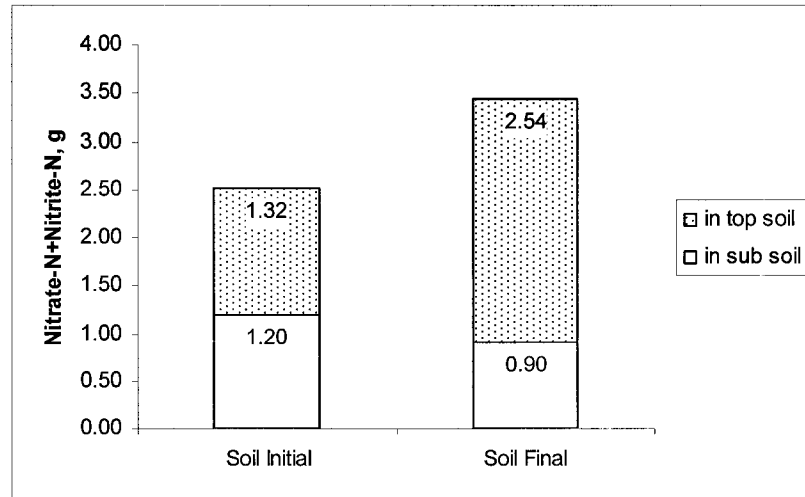
**Figure A 8 Nitrogen transfer and conversion in manure applied soil column system (Col. 7)**



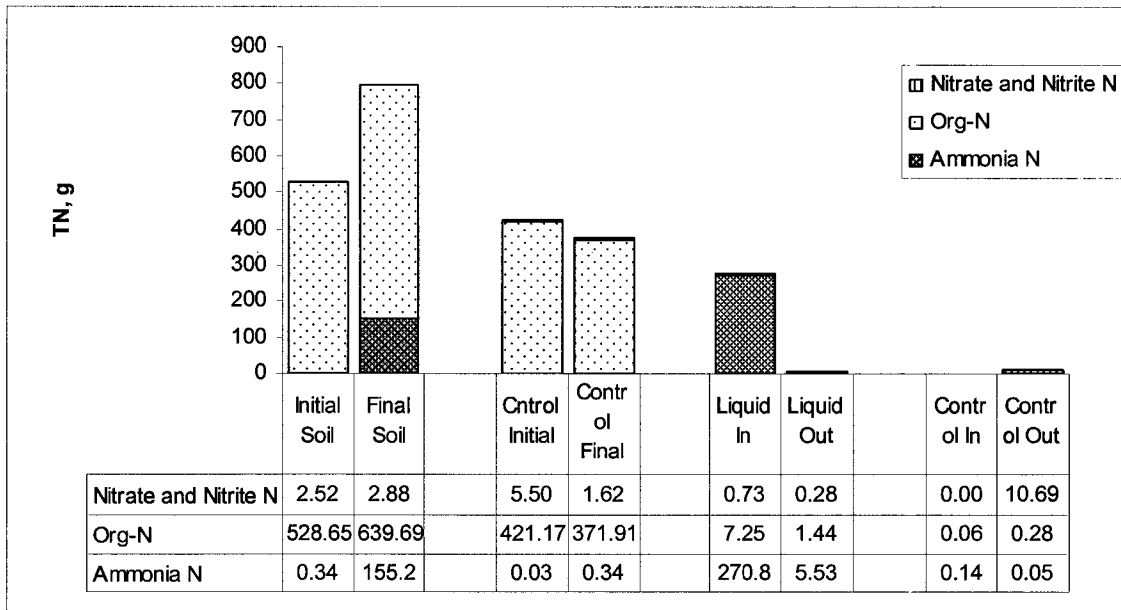
**Figure A 9 Ammonia variation in soil (Col. 7)**



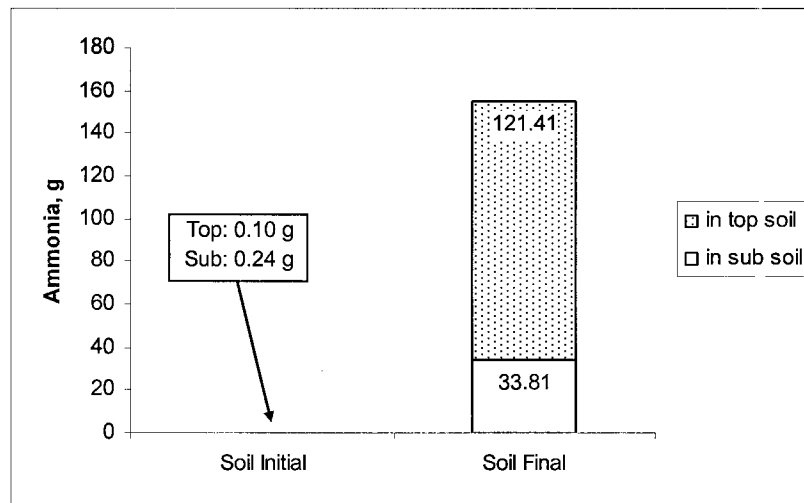
**Figure A 10 Organic nitrogen variation in soil (Col. 7)**



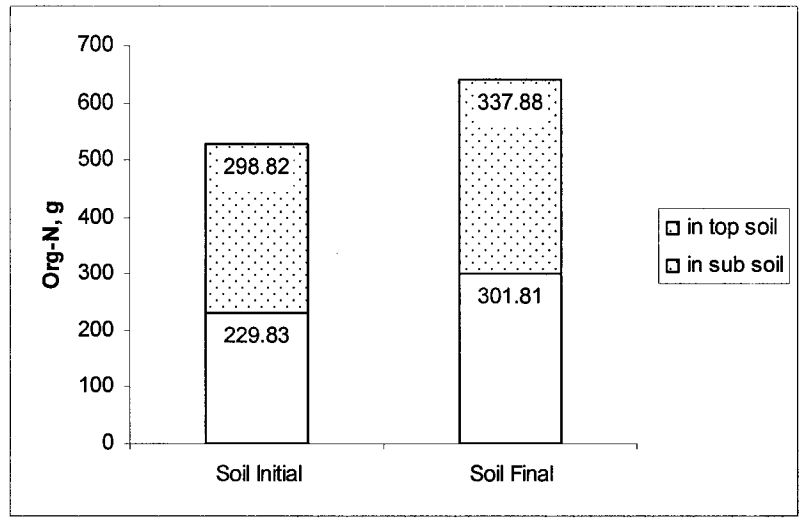
**Figure A 11 Nitrate plus nitrite nitrogen variation in soil (Col. 7)**



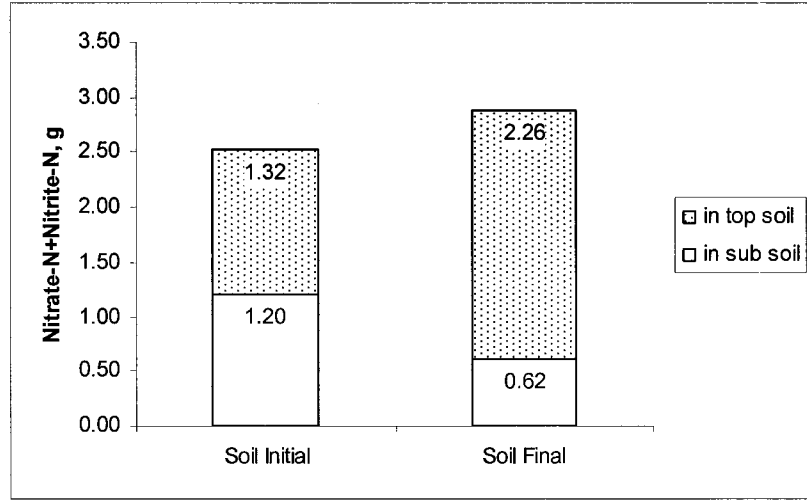
**Figure A 12 Nitrogen transfer and conversion in manure applied soil column system (Col. 9)**



**Figure A 13 Ammonia variation in soil (Col. 9)**



**Figure A 14 Organic nitrogen variation in soil (Col. 9)**



**Figure A 15 Nitrate plus nitrite nitrogen variation in soil (Col. 9)**



**Table B 2 Nitrogen balance of Col. 3 in Phase III**

<b>Manure in</b>								<b>Unit: g</b>			
TN	198.05										
NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N									
0.52	5.16	192.37									
				<b>Final Soil</b>				<b>Difference in Soil</b>			
TN	115.23			TN	179.12			TN	63.89		
NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N		NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N		NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	
2.48	112.74	0.01		0.04	121.72	57.36		-2.44	8.98	57.35	
TN	115.23			TN	132.99			TN	17.76		
NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N		NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N		NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	
2.48	112.74	0.01		0.05	123.79	9.15		-2.43	11.05	9.14	
TN	65.41			TN	119.66			TN	54.25		
NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N		NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N		NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	
0.18	65.23	0.002		0.09	115.51	4.06		-0.09	50.28	4.06	
TN	65.41			TN	62.68			TN	-2.73		
NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N		NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N		NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	
0.18	65.23	0.002		0.07	60.45	2.16		-0.11	-4.78	2.16	
TN	65.41			TN	42.69			TN	-22.72		
NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N		NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N		NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	
0.18	65.23	0.002		0.08	39.95	2.66		-0.1	-25.28	2.66	
<b>Initial Soil</b>								<b>Total difference in soil</b>			
				TN	5.87			TN	110.44		
				NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N		NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	
				5.14	0.43	0.3		-5.17	40.25	75.36	
				<b>Leachate Out</b>				<b>Difference in liquid</b>			
								TN	-192.2		
								NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	
								4.62	-4.73	192.07	
								<b>Total Balance</b>			
								TN	-81.74		
								NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	
								-0.55	35.52	116.71	

**Table B 3 Nitrogen balance of Col. 7 in Phase III**

Manure in			Final Soil			Difference in Soil		
TN	278.8		TN	208.82		TN	58.7	
NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N
0.73	7.25	270.82	2.18	154.65	51.99	1.52	5.24	51.94
↓ ↓ ↓ ↓ ↓								
Initial Soil								
TN	150.12		TN	172.12		TN	22.00	
NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N
0.66	149.41	0.05	0.36	122.78	48.98	-0.3	-26.63	48.93
TN	150.12		TN	78.42		TN	1.33	
NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N
0.66	149.41	0.05	0.38	72.76	5.28	-0.02	-3.85	5.2
TN	77.09		TN	82.4		TN	5.31	
NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N
0.4	76.61	0.08	0.3	75.27	6.83	-0.1	-1.34	6.75
TN	77.09		TN	63.83		TN	-13.26	
NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N
0.4	76.61	0.08	0.22	62.13	1.48	-0.18	-14.48	1.4
↓ ↓ ↓ ↓ ↓			Leachate Out			Total difference in soil		
			TN	7.53		TN	74.08	
			NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N
			0.96	2.1	4.47	0.92	-41.06	114.22
						Difference in liquid		
						TN	-271.27	
						NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N
						0.23	-5.15	-266.35
						Total Balance		
						TN	-197.19	
						NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N
						1.15	-46.21	-152.13

**Table B 4 Nitrogen balance of Col. 8 in Phase III**

Manure in							Unit: g		
TN	278.8								
NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N							
0.73	7.25	270.82							
			<b>Final Soil</b>			<b>Difference in Soil</b>			
TN	150.12		TN	199.13		TN	49.01		
NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	
0.66	149.41	0.05	9.49	154.59	35.05	8.83	5.18	35	
TN	150.12		TN	234		TN	83.88		
NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	
0.66	149.41	0.05	3.33	196.24	34.43	2.67	46.83	34.38	
TN	77.09		TN	74.85		TN	-2.24		
NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	
0.4	76.61	0.08	1.32	70.72	2.81	0.92	-5.89	2.73	
TN	77.09		TN	71.38		TN	-5.71		
NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	
0.4	76.61	0.08	2.36	57.79	11.23	1.96	-18.82	11.15	
TN	77.09		TN	72.9		TN	-4.19		
NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	
0.4	76.61	0.08	0.27	62.12	10.51	-0.13	-14.49	10.43	
<b>Initial Soil</b>						<b>Total difference in soil</b>			
			TN	12.89		TN	120.75		
			NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	
			0.37	2.75	9.77	14.25	12.81	93.69	
			<b>Leachate Out</b>			<b>Difference in liquid</b>			
						TN	-265.9		
						NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	
						-0.36	-4.5	-261.05	
						<b>Total Balance</b>			
						TN	-145.2		
						NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	
						13.89	8.31	-167.36	



**Table B 5 Nitrogen balance of Col. 9 in Phase III**

**Unit: g**

<b>Manure in</b>								
TN	278.8							
NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N						
0.73	7.25	270.82						
↓	↓	↓						
			<b>Final Soil</b>			<b>Difference in Soil</b>		
TN	150.12		TN	230.02		TN	79.9	
NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N
0.66	149.41	0.05	1.4	172.15	56.47	0.74	22.74	56.42
TN	150.12		TN	231.53		TN	81.41	
NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N
0.66	149.41	0.05	0.86	165.73	64.94	0.2	16.32	64.89
TN	77.09		TN	152.21		TN	75.12	
NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N
0.4	76.61	0.08	0.19	131.13	20.89	-0.21	54.52	20.81
TN	77.09		TN	90.34		TN	13.25	
NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N
0.4	76.61	0.08	0.12	81.96	8.26	-0.28	5.35	8.18
TN	77.09		TN	93.69		TN	16.6	
NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N
0.4	76.61	0.08	0.31	88.72	4.66	-0.09	12.11	4.58
<b>Initial Soil</b>						<b>Total difference in soil</b>		
			↓	↓	↓	TN	266.28	
			↓	↓	↓	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N
			↓	↓	↓	0.36	111.04	154.88
			<b>Leachate Out</b>			<b>Difference in liquid</b>		
			TN	7.25		TN	-271.6	
			N&N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N
			0.28	1.44	5.53	-0.45	-5.81	-265.29
						<b>Total Balance</b>		
						TN	-5.27	
						NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N
						-0.09	105.23	-110.41

**Table B 6 Nitrogen balance of Col 1 in Phase II**

Column 1 Unit: g

Manure in					
TN	0				
NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N			
0	0	0			

Initial Soil			Final Soil			Difference in Soil		
TN	302.15		TN	400.2		TN	98.05	
NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N
0.11	211.72	90.32	24.08	367.56	8.56	23.97	155.84	-81.76
TN	417.3		TN	228.28		TN	-189.02	
NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N
0.3	320.78	96.22	0.11	208.72	19.45	-0.19	-112.1	-76.77

Leachate out			Total difference in soil		
TN	0		TN	-90.97	
NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N
0	0	0	23.78	43.78	-158.53

**Table B 7 Nitrogen balance of Col 2 in Phase II**

Column 2 Unit: g

Manure in					
TN	0				
NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N			
0	0	0			

Initial Soil			Final Soil			Difference in Soil		
TN	326.85		TN	373.46		TN	46.61	
NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N
0.12	228.45	98.28	19.4	341.4	12.66	19.28	112.95	-85.62
TN	373.4		TN	256.41		TN	-116.99	
NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N
0.23	226.35	146.82	0.14	207.58	48.69	-0.09	-18.77	-98.13

Leachate out			Total difference in soil		
TN	0		TN	-70.38	
NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N	NO <sub>3</sub> +NO <sub>2</sub> -N	Org-N	NH <sub>4</sub> -N
0	0	0	19.19	94.18	-183.75

## APPENDIX C

**Table C 1 Tap water quality used for control columns**

Parameters	Unit	test 1	test 2	average
Total KN	mg/L	1.26	0.97	1.12
NO <sub>2</sub> -N	mg/L	0.00	0.00	0.00
NO <sub>3</sub> -N	mg/L	0.00	0.00	0.00
NH <sub>4</sub> -N	mg/L	0.74	0.83	0.79
Total P	mg/L	0.02	0.01	0.02
pH		8.28	8.12	8.20
EC	dS/m @25°C	0.36	0.24	0.30

**Table C 2 Leachate quality of different columns on different sampling dates**

Col 2	Unit	12-Oct	19-Oct	26-Oct	02-Nov	09-Nov	16-Nov	23-Nov	30-Nov
BOD <sub>5</sub>	mg/L	0.6	0.7	15	20	9	14	10	11
COD	mg/L		1	61	68	61	57	60	49
TDS	mg/L		4105	3630	3267	3155	3468	2170	1877
TSS	mg/L		135	38	30	47	0	12	23
Total KN	mg/L	0.37	0.26	1.39		4.04	2.22	2.22	2.81
NO <sub>2</sub> -N	mg/L		0		0.61	0.46	0.40	0.23	0.02
NO <sub>3</sub> -N	mg/L		210		90.09	36.04	14.12	12.27	20.44
NH <sub>4</sub> -N	mg/L		0.13	1.14		0.00	0.00	0.99	0.15
Total P	mg/L	0.03	0.02	0.03	0.01	0.02	0.02	0.02	0.03
TOC	mg/L			27	34	27	18	20	46
pH		7.6	7.91	7.58	7.85	7.63	7.64	7.74	7.85
EC	mS/cm @25°C	3.11	3.64	3.08	2.95	2.77	2.43	2.11	1.55

Col 3	Unit	12-Oct	19-Oct	26-Oct	02-Nov	09-Nov	16-Nov	23-Nov	30-Nov
BOD <sub>5</sub>	mg/L	21	92	87	82	170	182	256	279
COD	mg/L		170	170	185	305	355	526	557
TDS	mg/L		4225	3817	4175	4408	5053	3947	5447
TSS	mg/L		190	103	125	150	20	148	53
Total KN	mg/L	9.54	6.98	13.58		14.31	6.67	7.52	13.86
NO <sub>2</sub> -N	mg/L		13.54		5.54	7.21	3.44	0.31	0.38
NO <sub>3</sub> -N	mg/L		123.67		43.83	46.20	11.28	1.66	1.03
NH <sub>4</sub> -N	mg/L	2.53	0.68	11.5		4.97	0.86	4.23	1.61
Total P	mg/L	0.21	0.18	0.18	0.24	0.27	0.32	0.26	0.42
TOC	mg/L			33	49	48	52	50	75
pH		7.24	7.38	7.06	7.38	7.13	7.21	7.17	7.08
EC	mS/cm @25°C	3.45	3.86	3.52	3.41	4.53	4.16	4.11	4.38

Col 7	Unit	12-Oct	19-Oct	26-Oct	02-Nov	09-Nov	16-Nov	23-Nov	30-Nov
BOD <sub>5</sub>	mg/L	191	234	239	236	241	340	473	562
COD	mg/L		426	519	419	611	732	1435	1497
TDS	mg/L		4057	3772	3758	3693	5095	4087	4992
TSS	mg/L		150	177	135	133	205	243	113
Total KN	mg/L	26.61	15.72	32.18		43.08	54.13	70.29	79.60
NO <sub>2</sub> -N	mg/L		7.75		0.20	0.69	0.63	0.28	0.71
NO <sub>3</sub> -N	mg/L		28.45		0.25	0.17	0.29	0.04	0.28
NH <sub>4</sub> -N	mg/L	10.26	3.26	28.73		26.94	33.89	55.77	56.13
Total P	mg/L	0.85	0.41	0.67	1.28	1.28	1.38	1.83	1.59
TOC	mg/L			133	86	204	270	341	316
pH		7.24	7.27	7.2	7.41	7.29	6.97	7.07	7.02
EC	mS/cm @25°C	4.73	4.07	3.7	4.1	4.24	4.66	4.75	4.51

Col 8	Unit	12-Oct	19-Oct	26-Oct	02-Nov	09-Nov	16-Nov	23-Nov	30-Nov
BOD <sub>5</sub>	mg/L	184	181	177	246	262	307	522	677
COD	mg/L		340	348	369	710	874	2344	1949
TDS	mg/L		3887	3552	3575	3627	5182	4315	5352
TSS	mg/L		145	40	108	150	195	147.5	168
Total KN	mg/L	33.47	19.47	43.01		64.90	99.39	155.31	221.64
NO <sub>2</sub> -N	mg/L		2.31		4.79	0.14	0.48	0.81	2.77
NO <sub>3</sub> -N	mg/L		2.65		0.25	0.21	0.15	0.06	0.21
NH <sub>4</sub> -N	mg/L	10.74	8.52	38.42		47.01	77.61	114.09	192.42
Total P	mg/L	1.06	0.54	0.82	1.15	1.45	1.92	2.99	2.60
TOC	mg/L			93	70	184	262	465	452
pH		7.31	7.29	7.27	7.48	7.45	7.18	7.22	7.18
EC	mS/cm @25°C	5.26	4.01	3.72	4.16	4.21	4.76	5.26	5.42

Col 9	Unit	12-Oct	19-Oct	26-Oct	02-Nov	09-Nov	16-Nov	23-Nov	30-Nov
BOD <sub>5</sub>	mg/L	197	240	291	259	209	336	429	324
COD	mg/L		483	526	469	632	817	1840	1192
TDS	mg/L		3945	3565	3740	3528	4900	3835	4517
TSS	mg/L		145	80	55	108	218	153	63
Total KN	mg/L	30.33	17.35	38.27		47.37	62.69	78.67	74.01
NO <sub>2</sub> -N	mg/L		0.9		1.62	0.48	0.92	0.87	0.26
NO <sub>3</sub> -N	mg/L		6.21		0.14	0.24	0.23	0.26	1.07
NH <sub>4</sub> -N	mg/L	10.24	6.37	36.3		34.41	43.96	71.93	66.12
Total P	mg/L	1.23	0.5	0.96	1.16	1.61	1.75	2.35	1.68
TOC	mg/L			198	105	232	222	445	233
pH		7.07	7.2	7.05	7.24	7.15	7.03	7.13	6.98
EC	mS/cm @25°C	4.99	4.05	3.77	4.17	4.21	4.54	4.61	4.33

**Table C 3 Mixed leachate properties in Phase III for crop irrigation**

Parameters	Unit	Average Value
BOD <sub>5</sub>	mg/L	225
COD	mg/L	967
TDS	mg/L	4312
TSS	mg/L	175
TKN	mg/L	86.85
NO <sub>2</sub> -N	mg/L	0.33
NO <sub>3</sub> -N	mg/L	0.19
NH <sub>4</sub> -N	mg/L	63.69
Total P	mg/L	1.49
TOC	mg/L	202
pH		7.53
EC	mS/cm @25°C	4.24
Cl	mg/L	570
SAR		2.21

**Table C 4 Soil properties before and after manure application in Phase III**

Col 2	Top soil	Initial			Sub soil			
		Initial	Final		Initial	Final		
			layer 1	layer 2		layer 3	layer 4	layer5
Moisture	%	29.66	34.85	39.86	16.02	33.27	27.04	27.53
Total KN	%	0.43	0.463	0.504	0.15	0.134	0.066	0.073
NO <sub>2</sub> -N	ug/g soil	0.21	0.52	0.30	0.00	0.26	0.22	0.37
NO <sub>3</sub> -N	ug/g soil	94.29	12.51	11.43	4.14	7.35	6.43	7.73
NH <sub>4</sub> -N	ug/g soil	0.47	3.22	1.10	0.04	1.42	0.72	3.08
Total P	%	0.14	0.159	0.171	0.05	0.077	0.056	0.054
pH		7.57	8.76	8.82	7.6	8.43	7.95	8.63
EC in soil	mS/cm	1.7	0.664	0.662	1.04	0.473	0.481	0.51
SAR		1.18	0.68	0.78	0.53	0.7	0.74	0.7
OM	%	10.02	8.72	8.97	1.78	4.89	2.24	1.74

Col 3	Top soil	Initial			Sub soil			
		Initial	Final		Initial	Final		
			layer 1	layer 2		layer 3	layer 4	layer5
Moisture	%	29.66	51.37	46.31	16.02	25.85	31.63	28.47
Total KN	%	0.43	0.683	0.507	0.15	0.275	0.144	0.098
NO <sub>2</sub> -N	ug/g soil	0.21	0.00	0.30	0.00	0.66	0.33	0.27
NO <sub>3</sub> -N	ug/g soil	94.29	1.36	1.56	4.14	1.55	1.32	1.49
NH <sub>4</sub> -N	ug/g soil	0.47	2187.6	348.79	0.04	93.28	49.61	61.07
Total P	%	0.14	0.174	0.158	0.05	0.105	0.066	0.06
pH		7.57	8.51	8.73	7.6	8.57	8.91	8.34
EC in soil	mS/cm	1.7	5.718	2.807	1.04	1.39	1.153	1.653
SAR		1.18	3.18	1.98	0.53	2.06	1.86	1.69
OM	%	10.02	8.07	7.98	1.78	2.37	3.8	2.39

<b>Col 7</b>		<b>Top soil</b>			<b>Sub soil</b>			
		Initial	Final		Initial	Final		
			layer 1	layer 2		layer 3	layer 4	layer5
Moisture	%	24.19	46.86	45.19	10.84	32.13	32.14	21.41
Total KN	%	0.51	0.705	0.586	0.17	0.173	0.182	0.141
NO <sub>2</sub> -N	ug/g soil	0.29	24.57	1.58	0.40	4.56	1.56	1.25
NO <sub>3</sub> -N	ug/g soil	21.96	49.87	10.63	8.54	3.92	5.16	3.63
NH <sub>4</sub> -N	ug/g soil	1.65	1773.4	1671.1	1.69	116.99	151.37	32.72
Total P	%	0.08	0.162	0.148	0.05	0.295	0.116	0.067
pH		7.38	8.13	8.33	7.49	8.51	8.41	8.54
EC in soil	mS/cm	0.55	3.989	3.807	1.5	1.414	1.199	1.15
SAR		0.78	1.7	1.38	0.43	1.34	1.37	1.14
OM	%	8.34	9.22	9.33	1.17	4.01	4.32	2.93

<b>Col 8</b>		<b>Top soil</b>			<b>Sub soil</b>			
		Initial	Final		Initial	Final		
			layer 1	layer 2		layer 3	layer 4	layer5
Moisture	%	24.19	42.34	43.2	10.84	32.58	32.57	29.47
Total KN	%	0.51	0.647	0.787	0.17	0.163	0.153	0.161
NO <sub>2</sub> -N	ug/g soil	0.29	67.89	9.19	0.4	8.25	8.21	1.33
NO <sub>3</sub> -N	ug/g soil	21.96	255.86	104.53	8.54	21.16	44.05	4.65
NH <sub>4</sub> -N	ug/g soil	1.65	1195.9	1174.6	1.69	62.27	249.02	232.99
Total P	%	0.08	0.191	0.214	0.05	0.099	0.068	0.069
pH		7.38	8.04	8.52	7.49	8.66	8.76	8.67
EC in soil	mS/cm	0.55	4.754	3.625	1.5	1.398	1.498	1.356
SAR		0.78	1.45	1.39	0.43	1.31	1.08	1.2
OM	%	8.34	9.76	10.09	1.17	5.19	5.07	3.26

<b>Col 9</b>		<b>Top soil</b>			<b>Sub soil</b>			
		Initial	Final		Initial	Final		
			layer 1	layer 2		layer 3	layer 4	layer5
Moisture	%	24.19	50.18	49.92	10.84	32.68	29.67	28.31
Total KN	%	0.51	0.78	0.787	0.17	0.337	0.2	0.207
NO <sub>2</sub> -N	ug/g soil	0.29	22.04	2.49	0.4	0.6	0.95	3.15
NO <sub>3</sub> -N	ug/g soil	21.96	25.67	26.99	8.54	3.56	1.71	3.8
NH <sub>4</sub> -N	ug/g soil	1.65	1926.6	2215.5	1.69	463.11	183.2	103.23
Total P	%	0.08	0.214	0.213	0.05	0.1	0.075	0.082
pH		7.38	8.21	8.38	7.49	8.44	8.34	8.64
EC in soil	mS/cm	0.55	4.11	4.539	1.5	1.567	1.54	1.337
SAR		0.78	1.98	1.63	0.43	1.21	1.21	1.34
OM	%	8.34	9.88	11.1	1.17	3.42	3.48	2.88

**Table C 5 Soil properties variation in Phase II**

Col. 1				
Parameter	Top soil		Sub soil	
	Aug. 03	Sep. 14	Aug. 03	Sep. 14
pH	8.40	6.66	8.34	8.48
EC, ds/m	3.064	6.167	2.293	0.789
moisture content, %	47.71	36.13	36.95	22.73
NO <sub>3</sub> -N, ug/g	2.00	457.60	2.63	0.25
NO <sub>2</sub> -N, ug/g	0.00	0.72	0.00	0.73
NH <sub>4</sub> -N, ug/g	1719.45	162.95	856.07	173.04
TN %	0.575	0.716	0.371	0.203
TP %	0.164	0.204	0.099	0.061
Na, meq/L	3.49	6.49	4.73	3.40
Ca, meq/L	4.34	2.30	4.46	4.99
Mg, meq/L	2.08	21.05	2.40	2.58
SAR	1.95	1.90	2.55	1.75

Col 2				
Parameter	Top soil		Sub soil	
	Aug. 03	Sep. 14	Aug. 03	Sep. 14
pH	8.38	7.27	8.48	8.50
EC, ds/m	2.852	5.324	2.329	0.966
moisture content, %	44.59	30.29	29.25	25.42
NO <sub>3</sub> -N, ug/g	2.34	367.71	2.05	0.58
NO <sub>2</sub> -N, ug/g	0.00	1.59	0.00	0.61
NH <sub>4</sub> -N, ug/g	1871.02	240.93	1306.22	433.17
TN %	0.622	0.674	0.332	0.228
TP %	0.257	0.23	0.103	0.067
Na, meq/L	4.64	7.20	4.21	2.93
Ca, meq/L	4.72	44.85	3.04	4.16
Mg, meq/L	2.64	22.27	2.00	2.05
SAR	2.42	1.24	2.65	1.66

Control				
Parameter	Top soil		Sub soil	
	Aug. 03	Sep. 14	Aug. 03	Sep. 14
pH	8.26	7.98	8.30	8.27
EC, ds/m	1.396	0.569	0.717	0.380
moisture content, %	40.94	34.71	27.87	29.60
NO <sub>3</sub> -N, ug/g	28.62	22.76	6.40	0.65
NO <sub>2</sub> -N, ug/g	0.75	0.12	0.33	0.00
NH <sub>4</sub> -N, ug/g	33.00	0.00	2.77	0.00
TN %	0.509	0.731	0.192	0.216
TP %	0.243	0.218	0.086	0.061
Na, meq/L	1.52	0.87	1.35	0.96
Ca, meq/L	3.16	2.89	2.83	2.50
Mg, meq/L	1.72	1.40	1.21	1.00
SAR	0.97	0.60	0.95	0.73

**Table C 6 TP balance in Phase III**

	Col 2	Col 3	Col 7	Col 8	Col 9
Final Soil	168	187	297	213	228
Initial Soil	139	139	115	115	115
Storage	29	49	182	98	113
Manure In	0	2	3	3	3
Leachate Out	0	0	0	0	0
Unaccounted for	29	47	180	96	111



**APPENDIX D**

**NOPA system schematic drawing**

